BALTIC SEA ENVIRONMENT PROCEEDINGS

No. 64 B

THIRD PERIODIC ASSESSMENT OF THE STATE OF THE MARINE ENVIRONMENT OF THE BALTIC SEA, 1989-93;

Background document

HELSINKI COMMISSION
Baltic Marine Environment Protection Commission
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PREFACE

Within the framework of the Baltic Marine Environment Protection Commission - Helsinki Commission (HELCOM) - marine environment monitoring data have been collected since 1979 based on the jointly agreed guidelines for the Baltic Monitoring Programme (BMP) which were published in the Baltic Sea Environment Proceedings Nos. 27 A-D. The monitoring data provided by all Contracting Parties to the Helsinki Convention are stored and processed in the HELCOM data bank established by the Commission on a consultant basis.

In 1981, an Assessment of the Effects of Pollution on the Natural Resources of the Baltic Sea was published by the Commission (Baltic Sea Environment Proceedings Nos. 5 A and B). Since then, periodic assessments of the state of the marine environment of the Baltic Sea have been published regularly by the Commission with five years intervals. Each publication has a comprehensive scientific background part and more general conclusions drawn on the basis of the scientific overview. The previous periodic assessments covered the periods 1980-85 (Baltic Sea Environment Proceedings Nos. 17 A and B) and 1984-88 (Baltic Sea Environment Proceedings Nos. 35 A and B). In addition to the periodic assessments, an assessment of the state of the coastal waters of the Baltic Sea was published in 1993 (Baltic Sea Environment Proceedings No. 54).

The present assessment report for the period 1989-93, as the previous ones, has been prepared by the experts nominated by the Contracting Parties. A Steering Group for the Coordination of the Third Periodic Assessment (EC BETA) was appointed in 1993 for the overall coordination as agreed by the third meeting of the Environment Committee and endorsed by the Commission. Taking into account the experience from the previous assessments, the Steering Group agreed that for certain disciplines a sub-regional approach should be applied. For some items, such as meteorological, hydrological and hydrographic conditions, trace elements and organic contaminants, the whole Baltic Sea was agreed to be covered. For that purpose appropriate expert groups were established. Sub-regional expert groups were established for the Gulf of Bothnia, the Gulf of Finland, the Gulf of Riga, the Baltic Proper, and the Kattegat and Belt Sea. To ensure that the outcome of the sub-regional groups would directly fit together for an overall assessment, the Steering Group established four expert groups on different disciplines, i.e., (1) on statistics and data treatment, (2) on hydrochemistry, i.e., oxygen and hydrogen sulphide and nutrients, (3) on pelagic biology, and (4) on benthic biology. These discipline groups met first and gave their advice and detailed instructions for the sub-regional groups, e.g., on the handling and presentation of the different types of data.

Some of the chapters in this assessment were elaborated by special working groups or *ad hoc* expert groups, *inter alia*, Chapter 10.3 (Microbial food web) by the *ad hoc* Working Group on Microbiology, Chapter 5.4 (Artificial radionuclides) by the Group of Experts on Monitoring of Radioactive Substances in the Baltic Sea, and Chapter 7 (Nature conservation and biodiversity) by the Working Group on Nature Conservation and Biodiversity. Chapter 3 (Contaminant input), Chapter 8.2 (Sanitary conditions in coastal waters) as well as Chapter 8.3 (Dumping of chemical munition) have been compiled by the HELCOM Secretariat on the basis of recent reports published within the framework of HELCOM or based on other recent information provided by the Contracting Parties. Several of these aspects have not been included in the previous periodic assessments. The International Council for the Exploration of the Sea (ICES) has coordinated the elaboration of Chapter 6 (Baltic fish stocks and diseases).

The first meeting of the Steering Group was chaired by *Lars Edler*, Sweden, the second, third and fourth meetings by *Lars Rahm*, Sweden and the two last meetings by *Lutz Brügmann*, Sweden. The Environment Secretary of the Commission, *Eeva-Liisa Poutanen*, acted as Secretary for EC BETA. This assessment presents results emerging from the monitoring activities supported by national data, scientific publications and other relevant information, evaluated by experts and written by the authors of each chapter. A complete list of persons involved in the work is annexed to this publication.

In general, the Third Periodic Assessment of the State of the Marine Environment of the Baltic Sea, 1989-93, has been prepared in accordance with the plan as originally defined. The report represents a consensus opinion on the present state of the Baltic Sea in physical, chemical and biological terms. It is hoped that this assessment provides a sound scientific basis for further decisions on remedial action.

On behalf of the Helsinki Commission, sincere gratitude is expressed to the Chairmen of EC BETA, to the Conveners of the expert groups and discipline groups, to all scientists who have contributed to the work, to *Graham Topping* for linguistic corrections and to IRH konsultointi/Ecocommunication Finland Ltd for the final layout of the publication. Great appreciation is given to the Swedish Environmental Protection Agency in Stockholm for their support to the Chairmen and the editorial work.

CONTENTS

INTRODUCTION

1

The Baltic Sea is one of the major brackish water basins of the world. The history of this unique aquatic system, which is a product of the last glacial period covers about the same time scale as the development of the human beings since the stone age. The Baltic Sea has always been of great importance for the people living around it, providing a common bond as well as routes of navigation between the current nine bordering countries. Its fisheries represent a valuable part of each country's livelihood and its waters are also a recreational resource of increasing value. Otherwise, for the almost 85 million people living in the drainage area which is shared by 15 countries with mostly high-developed industry and agriculture, the Baltic Sea is exploited also as a huge natural wastewater treatment plant which is expected to cope with discharges of different origin and composition. The Baltic Sea ecosystem is able to assimilate some anthropogenic inputs, but recycled and more persistent compounds may, however, build up in the system with impacts on its functioning and stability.

The very specific hydrographic, chemical and physical conditions, and the geological history of the Baltic Sea explain why it hosts quite unusual aquatic biota. Both marine and freshwater organisms live side by side with a number of living relicts. Although the number of species is relatively small, some occur in abundance but most may be necessary to keep the ecosystem in balance. This is also true for sea birds and marine mammals who are endangered, *inter alia*, by virtue of their high level in the food web, and this influences their uptake of harmful lipophilic organohalogens through bioaccumulation.

The multitude and beauty of coastal landscapes around the Baltic Sea Area represents a heritage for the present society. It deserves protection against the anthropogenic pressure caused by activities in the drainage area, at the shoreline and on sea.

Public opinion about the state of the Baltic Sea is strongly influenced by the media who from time to time pick up certain dramatic events and episodes, frequently judging them in a negative ("...The Baltic Sea is dying...") and only occasionally positive ("...Most of the Baltic Sea problems are solved...") manner. Without providing at the same time necessary background information about the possible causes and the scale of those events, and how they fit into more general trends and tendencies, a misleading picture may be generated which tends to persist in the minds of the general public.

In the framework of the Convention on the Protection of the Marine Environment of the Baltic Sea Area (Helsinki, 1974, revised 1992), the state of the Baltic Sea is regularly assessed in about 5-year intervals. Hundreds of experts from a multitude of disciplines participate in this assessment process. The outcome is a basically scientific background document covering most of the topics related to the state of the Baltic Sea. The respective situation, trends and tendencies are highlighted. The multidisciplinary approach attempts to secure a balanced view on the different problems. As typical for any science, a compilation of today's results does not provide 'final answers' even when the Baltic Sea belongs to one of the longest and most intensively studied sea areas of the world. The assessments are understood to be a time limited consensus which has been reached between scientists participating in long-term studies.

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METEOROLOGY, HYDROLOGY AND HYDROGRAPHY

S. Bergström[1], W. Matthäus

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[1] See ANNEX 11.3 for addresses of authors

2.1 INTRODUCTION

The environmental conditions of the Baltic Sea are strongly dependent on the meteorological, hydrological and hydrographic processes and their interaction. These processes influence the temperature and ice conditions, the regional inflow of fresh water from the rivers, the atmospheric deposition of pollutants on the sea surface, the exchange of water between sub-basins and with the Kattegat, the transport and mixing of water within the water bodies of the Baltic Sea and thus all biological life in the system.

It has been estimated [29] that the Baltic Sea, excluding Belt Sea and Kattegat, has an annual fresh water surplus of 476 km^3, i.e., 436 km^3 river run-off, 224 km^3 precipitation and 184 km^3 evaporation. Since the annual inflow of saline water into the Baltic Sea is calculated to be 471 km^3 this results in an overall outflow of 947 km^3 yr^{-1}.

The climatic, hydrological and hydrographic conditions of the Baltic Sea system have been subject to two previous assessments covering the period 1980-85 [392] and 1984-88 [43]. The conditions of the assessment period 1989-93 are summarised in the following paragraphs. The presentation is based on the selection of stations shown in Figure 2.1.

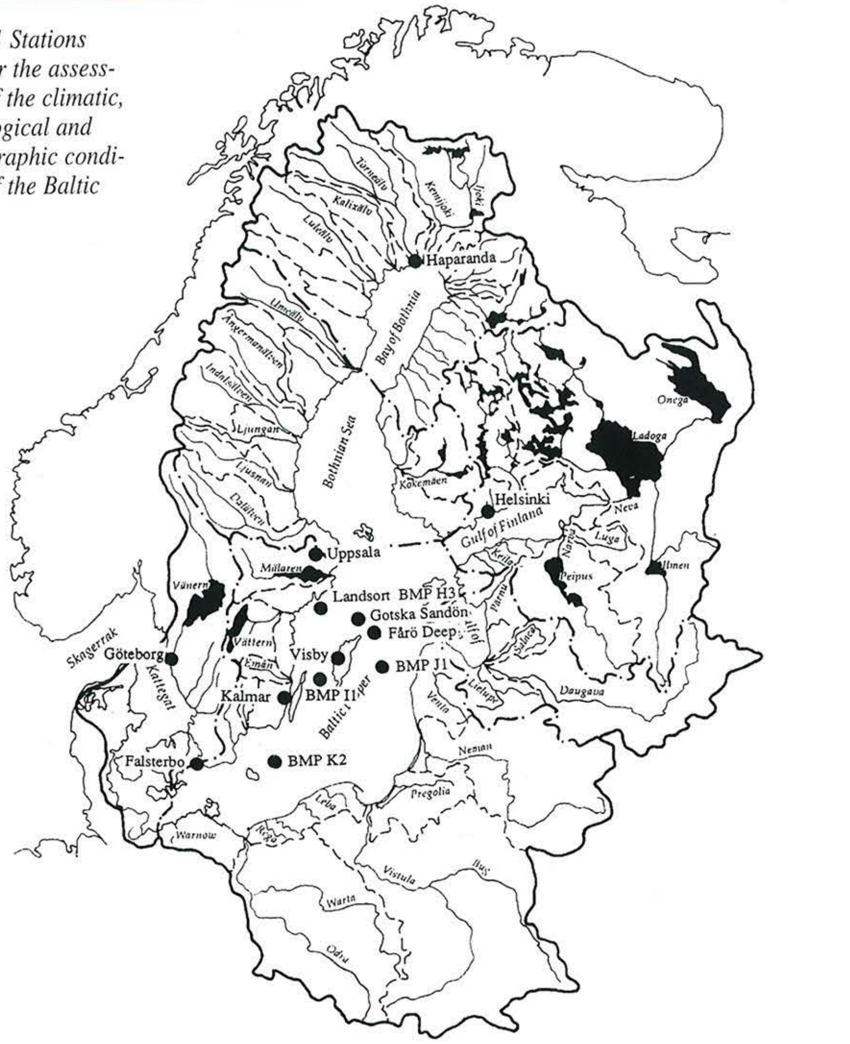


Fig. 2.1 Stations used for the assessment of the climatic, hydrological and hydrographic conditions of the Baltic Sea

2.2 CLIMATE

The Baltic Sea is located within the west-wind zone where cyclones coming from the west or south-west dominate the weather scene. Periodically, cyclones from a more southerly direction enter the region. The temperature climate of the region is to a large extent coupled to the latitude of the main cyclone tracks. Also the cloud cover plays an important role for the air temperatures, especially in winter.

In winter, most of the precipitation is frontal, especially inland. In summer, around half of the precipitation can be characterised as convective and is commonly more inland than over the water. Winds are closely related to the cyclones and the pressure gradients around these wind systems. Winds of storm force, i.e., at least 25 m s^{-1}, are almost exclusively connected to deep cyclones that have formed west of Scandinavia and occur mainly from September to March.

It should be noted that the large water body of the Baltic Sea has a strong impact on the local climate of the area. In particular this influences the air temperatures, precipitation, cloud cover, radiation and winds, and in the coastal areas this can lead to pronounced gradients. Therefore, coastal observations are not always representative for the open sea.

2.2.1 Air temperature

Records of mean annual air temperatures from the late nineteenth to the twentieth century fo[r] the stations Haparanda, Helsinki, Gotsk[a] Sandön and Falsterbo are shown in Figure 2.2. These records show an upward trend to a maximum during the warm period of the 1930s and then a decrease in the north and variable conditions in the south until the recent warm period, which began in the late 1980s. Although, these general features are observed at all stations they are more pronounced in the north. More detailed descriptions of the temperature climate of the area are available i[n] several other papers [*9, 45, 131, 150, 204, 46[0], 461, 495*].

Figure 2.2 shows clearly that the mean annu[al] temperature over the Baltic Sea was muc[h] above normal for the assessment period 198[9]-93. Compared to the whole period 1880-9[3], the mean annual anomaly for these five yea[rs] and these four stations was +1.2 °C. Th[e] anomalies are mainly concentrated in the winter and spring. The summers of the assessment period show the following average characteristics: 1989 was close to normal, 1990 was fai[r]ly warm in the whole region, 1991 was close t[o] normal, 1992 was very warm in the south and cooler in the north, and 1993 was cool in the whole region. Every summer season shows, however, contrasting periods. For example, June 1991 was very cold. The July of the same year was very warm while the August showed temperatures slightly above normal.

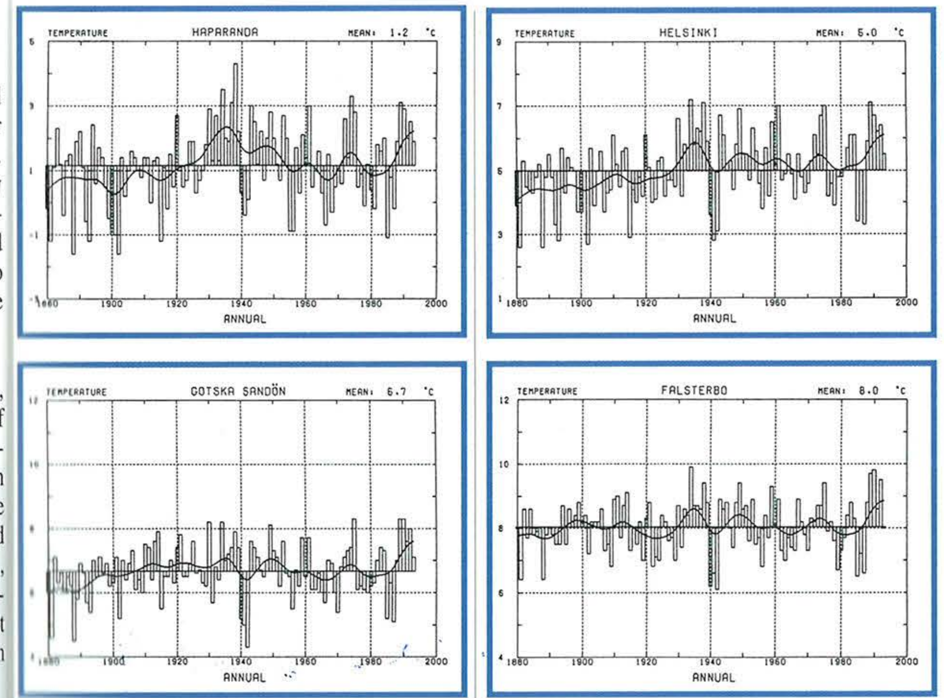


Fig. 2.2 Records of mean annual air temperatures from the 20th century for four stations at the coast of the Baltic Sea

(values in °C; the smoothed curve is obtained by *Gaussian* filtering technique corresponding to a ten year moving average)

2.2.2 Precipitation

Records of annual precipitation for the late nineteenth to the twentieth century from the four stations are shown in Figure 2.3. Although four stations are a very small number for the analysis of areal precipitation over such a large area as the Baltic Sea, Figure 2.3 indicates slightly wetter conditions than normal in the north during the assessment period and more variable conditions in the south. The mild winters have resulted in an unusual precipitation pattern over the whole basin, with extreme winter precipitation in the Scandinavian mountains. This is, however, not reflected in the records from these coastal stations. When analysing Figure 2.3, it should be noted, that the earlier data might suffer from homogeneity problems as the instrumentation and routine procedures for measurements were not entirely consistent for the whole period. Reference is made to [*45,131,150,205,495*], among others, for further details about the precipitation climate of the area.

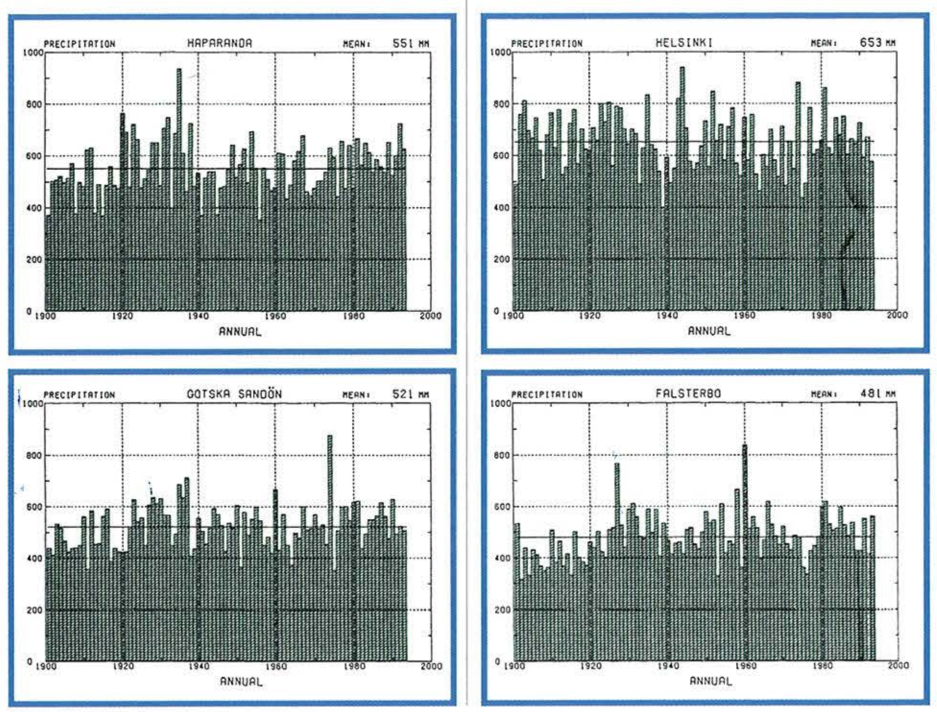


Fig. 2.3 Records of annual precipitation for the twentieth century for four stations at the coast of the Baltic Sea

2.2.3 Radiation, cloudiness and ozone

The Baltic Sea has a strong local impact on the radiation and cloud cover in its drainage basin. This is particularly pronounced in spring and in summer when the cloud cover is much lower over the cool sea than over the warmer land masses. Figure 2.4 shows, as an example, an analysis of the total cloud cover for 1993 based on satellite information and a cloud classification algorithm [*322,323*]. The picture is based on 1,327 scenes with NOAA AVHRR data.

Seasonally separated global radiation measurements in Visby for the period 1958-93 are shown in Figure 2.5. They show that at least for the Baltic Proper, the annual mean global radiation during the assessment period was very close to the long-term average.

The total ozone, i.e., the integrated amount of ozone in a vertical column throughout the atmosphere, has decreased during the last decades especially at high latitudes and in spring-time [*717*]. There are large regional differences in the depletion and also large natural variations during a year and from day to day. The total ozone is also influenced significantly by major volcanic activities.

In Figure 2.6, the monthly deviations in total ozone at Norrköping 1988-95 from a reference period of monthly means at Uppsala 1951-66 are plotted as an update from [*300*]. The time of the volcanic eruption of Mount Pinatubo on the Philippines in 1991 is indicated by an arrow. As can be seen, the total ozone fluctuated around the mean of the reference data but the values for the last year of the assessment period were highly influenced by the eruption.

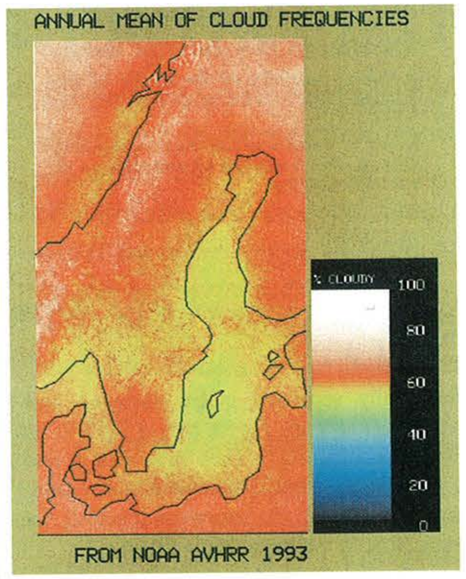


Fig. 2.4 Analysis of the total cloud cover for 1993 based on NOAA AVHRR data and a cloud classification algorithm [322]

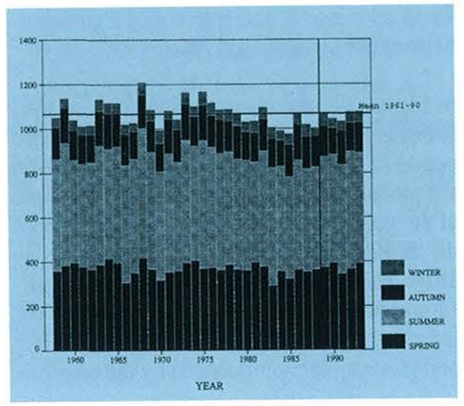


Fig. 2.5 Seasonally separated global radiation measurements in Visby for the period 1958-93

(values in kWh m^{-2}; winter is defined as December-February, spring as March-May, summer as June-August and autumn as September-November)

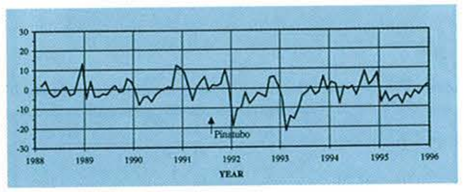


Fig. 2.6 Monthly deviations (%) in total ozone at Norrköping 1988-95 from monthly means at Uppsala for the reference period 1951-66

(Data from SMHI and the Swedish Environmental Protection Agency, updated from [300]; note the strong effect of the volcanic eruption of *Mount Pinatubo* in 1991)

2.2.4 Wind

The long-term trends in the wind climate can be analysed from geostrophic winds from air pressure data. Pressure data are far more exact and homogeneous than wind speed data which suffer from homogeneity problems and short records. In Figure 2.7, the zonal (westerly) and the meridional (southerly) geostrophic wind components, derived from a triangle Göteborg-Kalmar-Uppsala in Sweden, are shown. The results are believed to be representative of, at least, the southern half of the Baltic Sea.

The zonal wind has apparently recovered from lower values to much higher values in recent years. This is coupled to the mild winters (Chapter 2.2.1, Fig. 2.9) and coincides with observations from the North Sea and the North Atlantic [574]. Zonal winds were also generally strong during the first three decades of this century. The meridional component has generally low values during the latest decade with dominately negative values.

To illustrate more extreme wind conditions, daily observations from Landsort are shown in Figure 2.8. These observations indicate a short-term increase in storm frequencies during the assessment period, which is not so evident when longer periods are analysed.

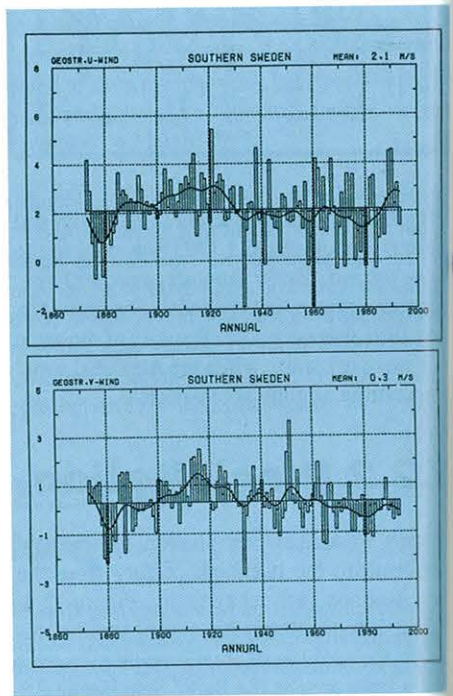


Fig. 2.7 Analysis of annual means of the zonal (westerly winds positive, u) and the meridional (southerly winds positive, v) geostrophic wind components derived from air pressure data in a triangle Göteborg-Kalmar-Uppsala in southern Sweden

(values in m s^{-1}; smoothed curve obtained by *Gaussian* filtering technique corresponding to a ten year moving average)

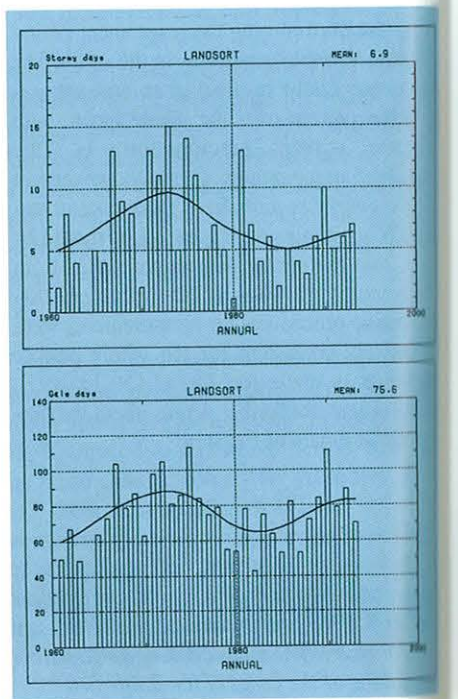


Fig. 2.8 Extreme wind conditions analysed from daily observations at Landsort

(stormy days have at least one observation above 21 m s^{-1} and gale days have at least one observation above 14 m s^{-1}; smoothed curve obtained by *Gaussian* filtering technique corresponding to a ten year moving average; data for 1964 missing)

2.3 ICE CONDITIONS

The ice conditions of the Baltic Sea are strongly related to the severity of the winter season. A summary of the maximum extent of ice coverage, reported in detail elsewhere [608], is shown in Figure 2.9. Each reading corresponds to the situation prevailing at the beginning of the winter.

As can be seen from Figure 2.9, the ice coverage was very modest during 1989-93. The record reveals that those conditions are unusual for such a long period of time.

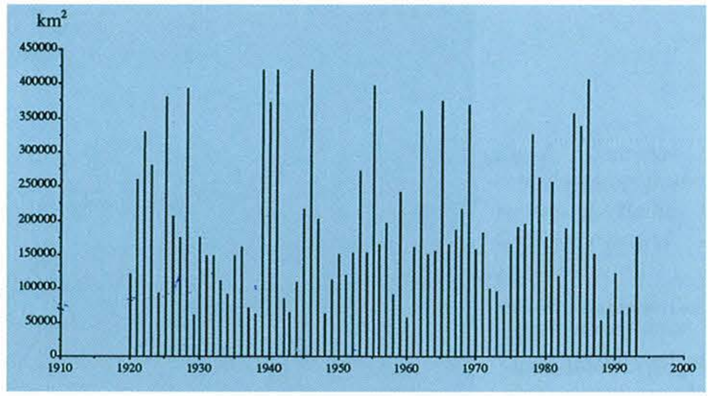


Fig. 2.9 Time series of maximum ice coverage in the Baltic Sea for the period 1920-93

(data from [608] with supplements for 1993-94 from the Finnish Institute of Marine Research; note that the dates refer to the beginning of the winter)

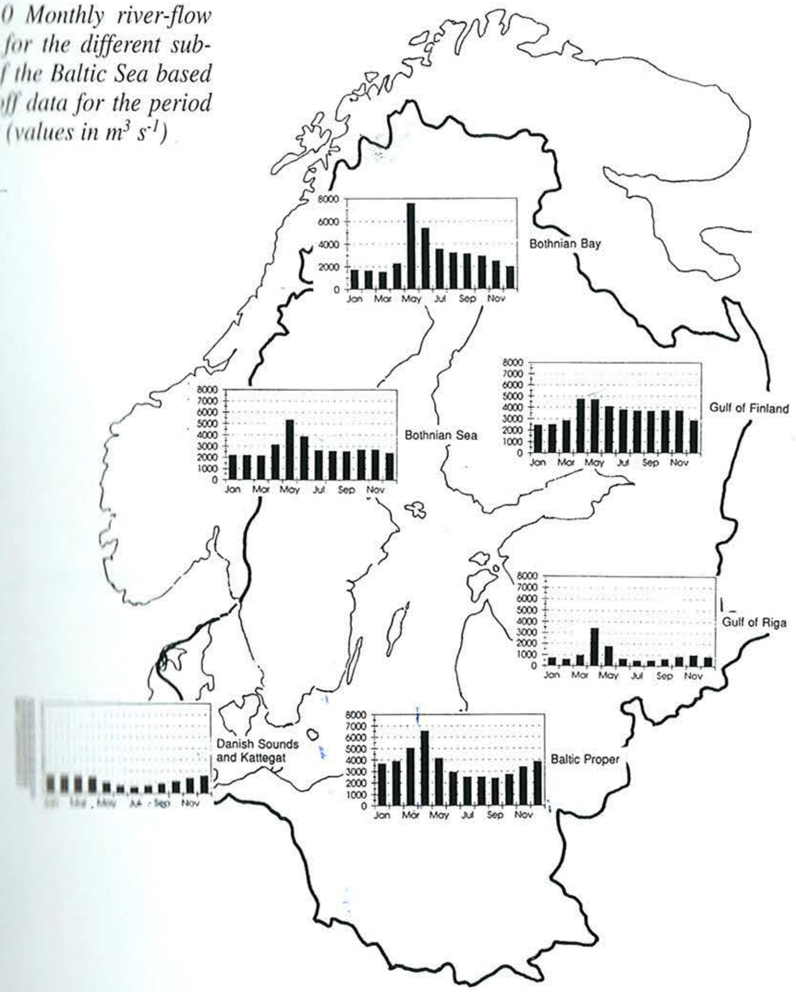


Fig. 2.10 Monthly river-flow regimes for the different sub-basins of the Baltic Sea based on run-off data for the period 1950-90 (values in m^3 s^{-1})

2.4 HYDROLOGY

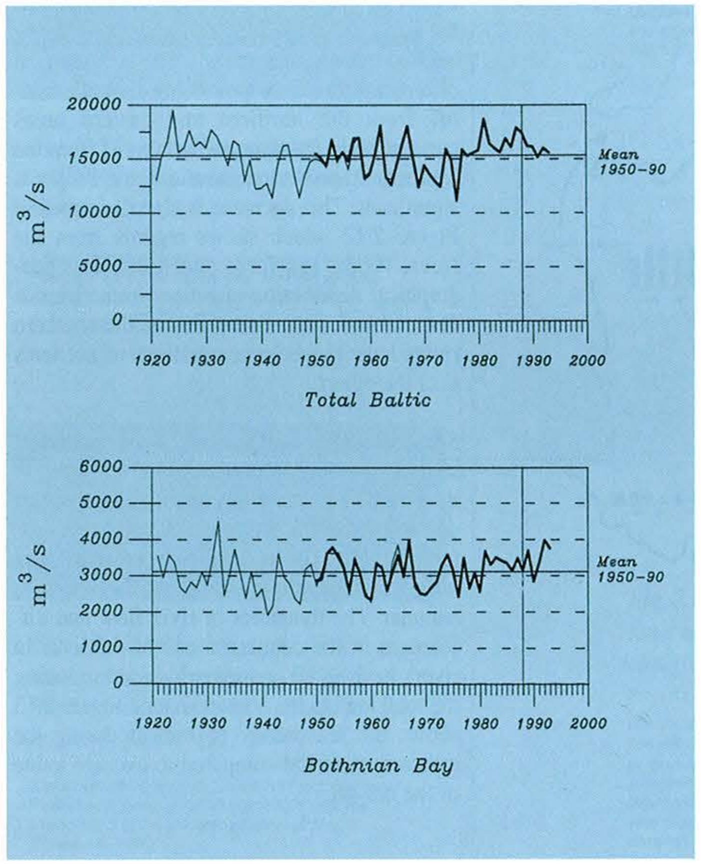
Within the Swedish research programme "Large-scale Environmental Effects and Ecological Processes in the Baltic Sea", a database on inflow from the drainage basin to the Baltic Sea has been established [70,71]. This database and previous work [462] cover the period 1920-90 and form the background to the assessment of the period 1989-93. The data for the period 1991-93 have been supplemented with data from the hydrological agencies of other countries around the Baltic Sea. However, some data are still missing in the database for the assessment period. These gaps are for the period 1991-93 for the rivers Pregolia and Luga, and for 1993 for river Narva. The gaps have been filled in by comparison with records from neighbouring stations and are not considered serious for the final result.

Figure 2.10 provides an overview of the river-flow regimes to the different sub-basins of the Baltic Sea for the period 1950-90. Figure 2.11 shows the annual river flow to the total sea area and its sub-basins for the period 1921-93, with the start of the assessment period indicated.

Throughout the assessment period, there is a progressive decline in the total amount of inflow to the Baltic Sea (Fig. 2.11). Towards the end of that period, the values are partly back to the long-term mean. The geographical differences are very pronounced. While run-off from the northern and eastern areas remains high, the decrease in run-off from the southern drainage areas to the Baltic Proper is significant. This decrease is also illustrated in Figure 2.12 which shows records from the Neva, Vistula and Torne plus Kalix. This geographical distribution of inflow from rivers is favourable for the Baltic Sea as the southern rivers have higher concentrations of nutrients than the others.

Seasonal inflow is of interest in environmental studies, as the concentrations of nutrients in the rivers may vary considerably over the year due to biological activities and other reasons (cf. [100,651]). In lake-rich systems, the nitrate concentrations are particularly low in summer. The dynamics of river flow and differences in the concentrations of nutrients in rivers have to be considered when estimating the total load to the coastal waters. Figure 2.13 shows the seasonality of run-off during the assessment period compared to average value of the run-off.

It has been pointed out [*70*] that the pronounced increase in the annual run-off during the late 1980s was dominated by increasing winter flows. The influence from increased flows in spring and autumn was found only minor. The increase in winter run-off was a common feature for all basins, with the possible exception of the Gulf of Finland which showed a more irregular pattern. Figure 2.13 shows that winter run-off during the assessment period was higher than the long-term mean, while the other seasons showed values more close to the long-term means. The high winter flows were related to the mild winters in the northern parts of the Baltic Sea. However, most of the northern rivers drain forested areas and thus have relatively low concentrations of nutrients.



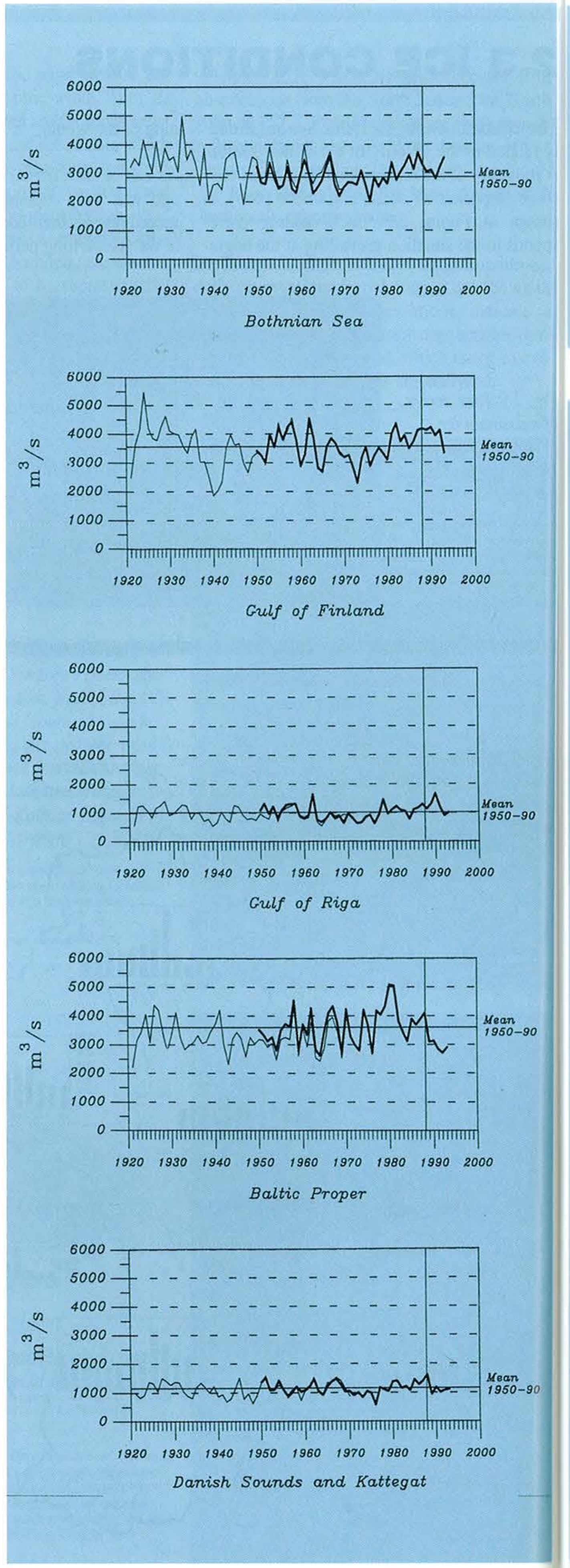


Fig. 2.11 Total annual contribution of fresh water to the entire Baltic Sea and its sub-basins for the period 1921-93

(thin lines represent data from [*462*], thick lines the updated database of [*70*]; start of the assessment period is indicated by a vertical line)

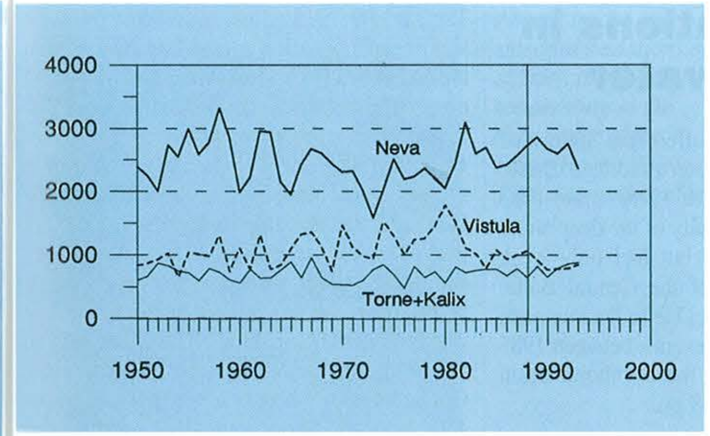


Fig. 2.12 Records of annual run-off for the rivers Neva, Vistula and Torne plus Kalix for the period 1950-93

(values in $m^3 s^{-1}$; start of the assessment period is indicated by a vertical line)

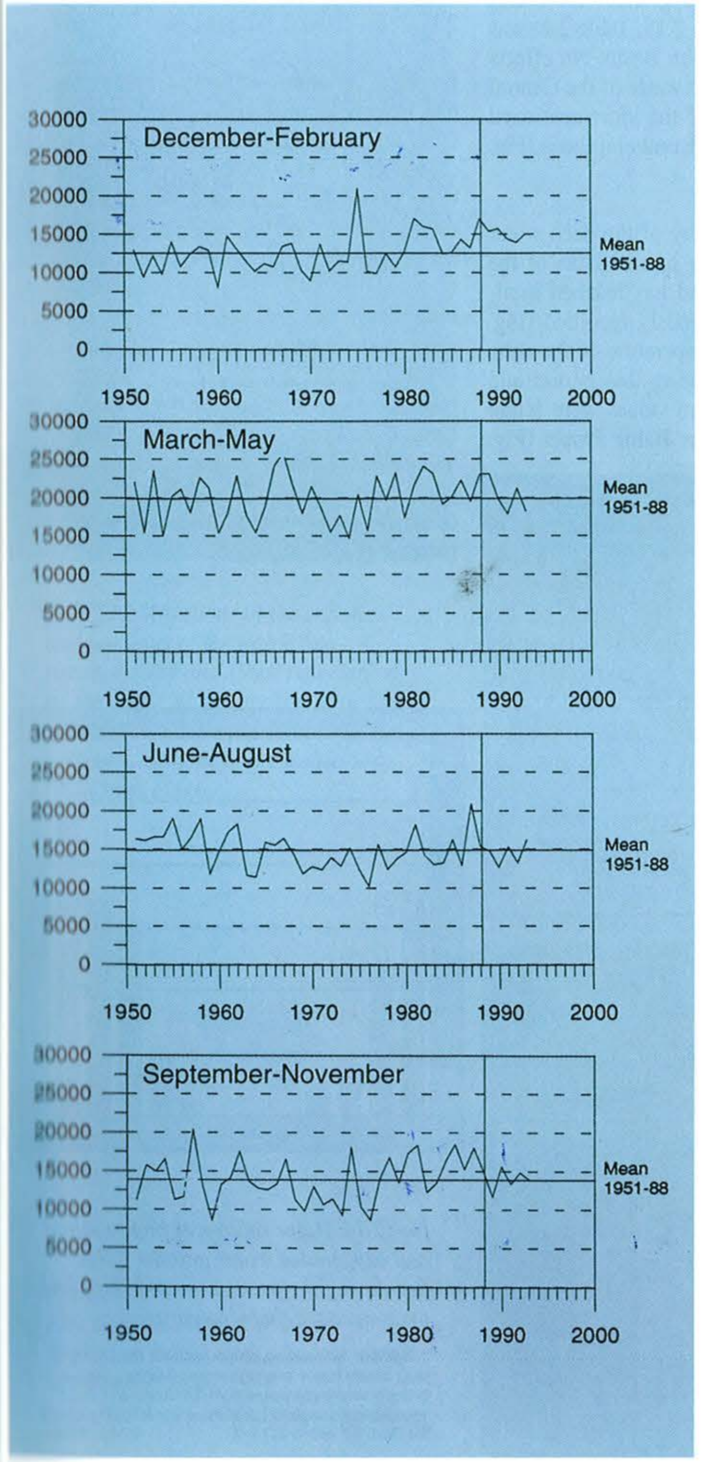


Fig. 2.13 Seasonal contribution of fresh water to the Baltic Sea for the period 1951-93

(start of the assessment period is indicated by a vertical line)

2.5 HYDROGRAPHY OF THE BALTIC PROPER

2.5.1 Introduction

The hydrographic conditions are characterised by continuation of the stagnation period in the deep water of the Eastern Gotland Basin during most of the assessment period. The last effective inflows of highly saline and oxygenated water occurred in 1975/76 and the last 'event' was recorded in early 1983 (Fig. 2.14). Throughout the assessment period, the salinity of the Baltic Sea was observed to decrease in all water layers within the system.

The year 1990 indicated a change in the general water exchange pattern between the North Sea and the Baltic Sea. Small inflows to the Baltic Sea occurred in early 1990 and during the turn of both 1990 to 1991 and 1991 to 1992 (Fig. 2.15). At the end of the assessment period, a major inflow caused a renewal of the deep water of the Central Baltic Proper. The stagnation period was finally terminated by inflows of minor magnitude in December 1993 and March 1994 but these did not achieve the magnitude of the inflow during the major 'events'.

2.5.2 Salt-water inflow 1993

In January 1993, after 16 years of stagnation, a major inflow occurred in the Eastern Gotland Basin [*201,449*]. During a three-week period of strong westerly winds, a total of about 310 km^3 of water, 150 km^3 of which was highly saline (>17 psu) and oxygenated, entered the Baltic Sea. Compared to the major inflows identified during the present century (Fig. 2.14), the recent inflow was rather unique with respect to both the short duration of the total process (21 days) and the size of the inflow which resulted in a large increase in the volume of the Baltic Sea (70 cm above mean water level) (Fig. 2.15). Taking into consideration the duration and mean salinity of the inflowing water at the Darß Sill, the 'event' could be regarded as an only moderate one. However, due to the largest salt transport ever observed across the Drogden Sill during major inflows, the 1993 'event' must be characterised as a very strong one.

Most of the highly saline water, entering the Baltic Sea in January 1993, flowed through the Bornholm Channel and replaced the bottom water in the Bornholm Basin, increasing the salinity to 20 psu (Figs. 2.16 D-F, 2.17). Between October 1992 and March 1993, the oxygen concentration was raised from 1 to 7.5 cm^3 dm^{-3} (Fig. 2.18). The stagnant bottom water of the Bornholm Basin was lifted above, the Slupsk Sill (60 m) by the inflow of saline water and entered intermittently the Eastern Gotland Basin (Figs. 2.16 E-H).

2.5.3 Variations above the halocline

The predominating signal in the water layer above the halocline is the annual cycle of the water temperature. The variations of the air temperature (cf. Chapter 2.2.1) and radiation (cf. Chapter 2.2.3) are reflected in the surface water layer. The mild winters during the assessment period (Fig. 2.9) caused positive water temperature anomalies between 0.5 and 2 K in the 'winter-water' layer. This was partly observed in spring, particularly in 1990 and 1993. In general, the water temperatures in summer were mostly close to the long-term mean, except for the extremely warm summer of 1992 which resulted in positive water temperature anomalies of 1.5-2 K. In 1992, the summer thermocline reached greater depths than usual, and in the open Baltic Proper positive anomalies between 5 and 8 K were measured at depths of 20-30 m.

The intermediate water layer stores the water temperatures of the preceding winter season from spring to late fall. During the assessment period, the mild winters resulted in positive temperature anomalies between 1 and 2 K for the intermediate layer.

2.5.4 Variations in the deep water

The Bornholm Basin buffers the inflow of saline water into the Eastern Gotland Basin. The major inflows between 1975/76 and 1983 strongly affected the salinity of the deep water of the Bornholm Basin, but had only weak effects on the salinity of the Central Baltic Proper (Figs. 2.17, 2.18). Due to the complete absence of major inflow events, between 1983 and 1990, the salinity in the Bornholm Basin dropped from 17.2 to 13.8 psu.

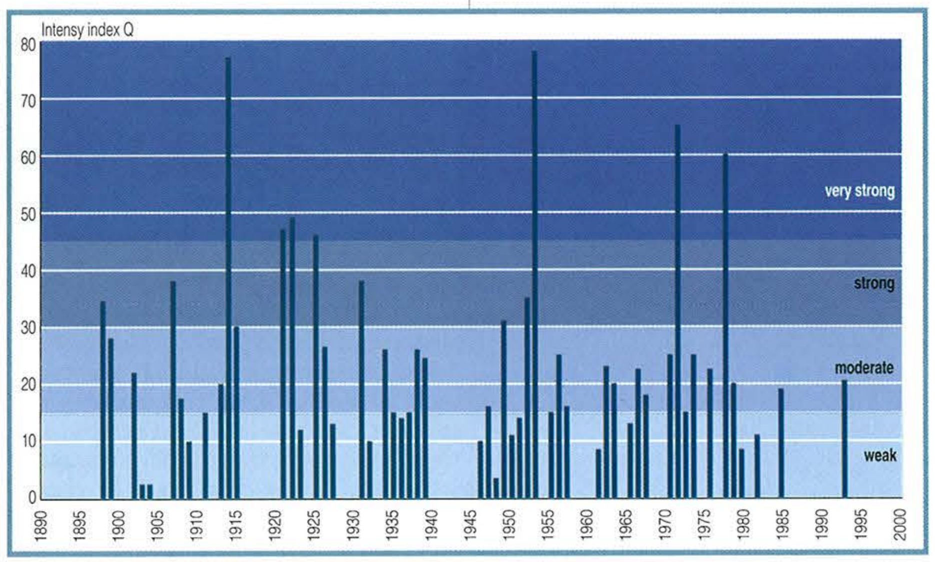
With the intensification of smaller inflows, an increase of the salinity (Fig. 2.17, Table 2.1) and oxygen content (Fig. 2.18, Table 2.2) was observed for the Bornholm Basin. No effects could be seen in the deep water of the Central Baltic Proper because of the aforementioned buffering capacity of the Bornholm Basin (Fig. 2.16 B-D).

Since 1975/76, the salinity of the deep water has decreased drastically in all basins of the Central Baltic Proper and has reached minimum values never previously recorded (Fig. 2.17, Table 2.1). The temperature of the deep water also decreased during this period and during 1989/90, minimum values were found in the northern part of the Baltic Proper (Fig. 2.19, Table 2.3).

During the assessment period, a decrease in the oxygen concentration and a drastic increase in the concentration of hydrogen sulphide were recorded in the waters of the Gotland and Fårö Deeps (Fig. 2.18). These are some of the highest H_2S concentrations ever measured in the Baltic Sea (Table 2.2). In contrast, the decreasing salinity and water column stability during this period produced caused an increase in oxygen concentrations in the Western Gotland Basin (Landsort and Karlsö Deeps; Fig. 2.18), particularly since 1989.

The January 1993 inflow interrupted the most significant stagnation period ever observed in the Baltic Sea [446]. Following the inflow, the deep waters of the Gotland Deep between 200 m and the bottom were completely oxygenated in mid May 1993 (Fig. 2.18). Changes in salinity were, however, only moderate in the Gotland Deep reaching its highest value (11.8 psu) near the bottom (Fig. 2.17). During May and June, H_2S remained in the intermediate layer above the oxygenated bottom water. It was only in the beginning of July, that oxygen was found in the whole water column of the Gotland Deep, whereas it was present below 150 m depth throughout 1993 in the Fårö Deep. Weak effects of the inflow could be traced in the Landsort and Karlsö Deeps (Figs. 2.17-2.19, Table 2.1-2.3).

Extreme variations of hydrographic parameters have been observed in the deep water of the Central Baltic Proper [446,481]. In the course of the last 16 years, the salinity and temperature has decreased. In addition, the temperatures at the beginning of the stagnation period were the highest ever measured near the bottom (Fig. 2.19; cf. [166]), and the salinity observed at the end was the lowest recorded since the beginning of regular measurements in the Baltic Sea (Fig. 2.17). The hydrogen sulphide concentrations in the deep water of the Eastern Gotland Basin reached the highest value ever measured, and, in contrast, the oxygen concentrations in the near-bottom layers of the Western Gotland Basin were observed to increase (Fig. 2.18).



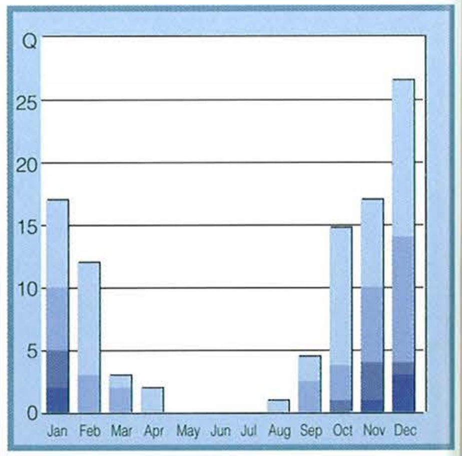


Fig. 2.14 Major inflows of highly saline and oxygenated water into the Baltic Sea during the present century (left) and their seasonal distribution (above).

(black bar: assessment period, updated from [448]; Q shall qualify major 'events' according to their relative intensity and is calculated from the duration of the 'event' and the salinity (S) of the water inflowing across the Darß Sill with S ≥17 psu)

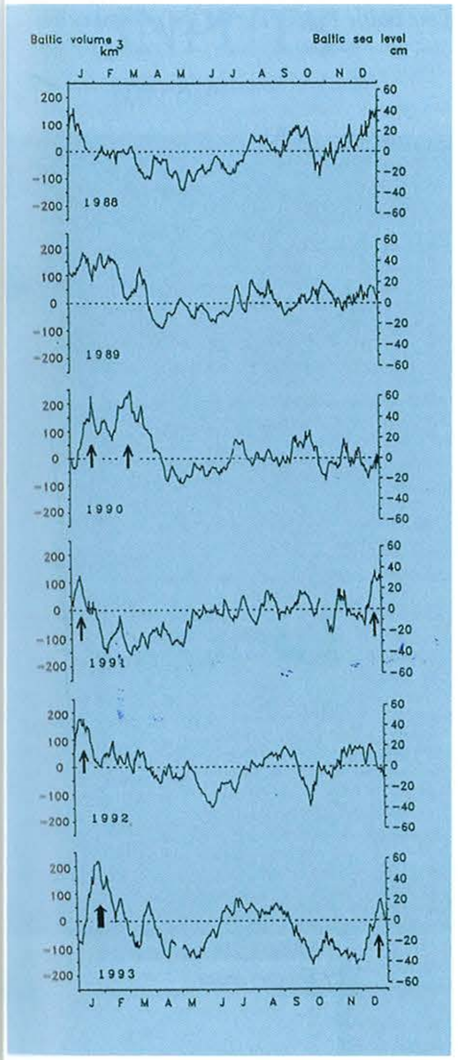


Fig. 2.15 Variations of the volume and sea level of the Baltic Sea between 1988 and 1993, represented by the sea level at Landsort which is considered to be representative for volume variations of the Baltic Sea (data from SMHI)

Annual means and standard deviations of salinity, oxygen and temperature in the deep water of the Baltic Proper 1989 – 1993 (minimum values underlined)

Table 2.1 Salinity (psu)

Station	Depth/m	1989	1990	1991	1992	1993
K2	80	14.38 ± 0.28	13.88 ± 0.37	14.19 ± 0.25	15.26 ± 0.45	17.66 ± 0.38
J1	200	11.56 ± 0.07	11.46 ± 0.09	11.26 ± 0.07	11.04 ± 0.04	11.16 ± 0.13
Fårö Deep	150	10.75 ± 0.05	10.48 ± 0.08	10.32 ± 0.16	10.18 ± 0.05	10.33 ± 0.14
H3	400	9.81 ± 0.09	9.51 ± 0.09	9.18 ± 0.05	8.99 ± 0.08	9.04 ± 0.15
I1	100	9.05 ± 0.22	8.65 ± 0.22	8.34 ± 0.20	8.22 ± 0.11	7.95 ± 0.16

Table 2.2 Oxygen concentration (cm^3 dm^{-3})

Station	Depth/m	1989	1990	1991	1992	1993
K2	80	0.58 ± 0.87	0.54 ± 0.73	2.26 ± 1.06	2.90 ± 1.38	4.81 ± 1.70
J1	200	-3.00 ± 0.87	-3.10 ± 0.91	-4.54 ± 1.62	-4.63 ± 1.74	-0.61 ± 2.39
Fårö Deep	150	-0.99 ± 0.40	-1.26 ± 0.64	-1.56 ± 0.66	-0.74 ± 0.27	-0.73 ± 0.59
H3	400	0.37 ± 0.57	0.47 ± 0.38	0.87 ± 0.80	1.17 ± 0.34	1.58 ± 0.19
I1	100	1.12 ± 0.54	1.69 ± 0.33	1.95 ± 0.34	2.33 ± 0.40	3.79 ± 0.61

Table 2.3 Temperature (°C)

Station	Depth/m	1989	1990	1991	1992	1993
K2	80	6.92 ± 0.51	6.98 ± 0.60	5.90 ± 0.97	6.22 ± 0.35	4.26 ± 0.18
J1	200	5.07 ± 0.07	4.98 ± 0.05	4.96 ± 0.04	5.03 ± 0.02	5.01 ± 0.13
Fårö Deep	150	4.57 ± 0.08	4.61 ± 0.20	4.97 ± 0.14	5.24 ± 0.04	5.21 ± 0.12
H3	400	4.00 ± 0.03	3.96 ± 0.18	4.41 ± 0.21	4.86 ± 0.09	4.83 ± 0.07
I1	100	3.73 ± 0.14	3.91 ± 0.28	4.27 ± 0.10	4.44 ± 0.09	4.15 ± 0.25

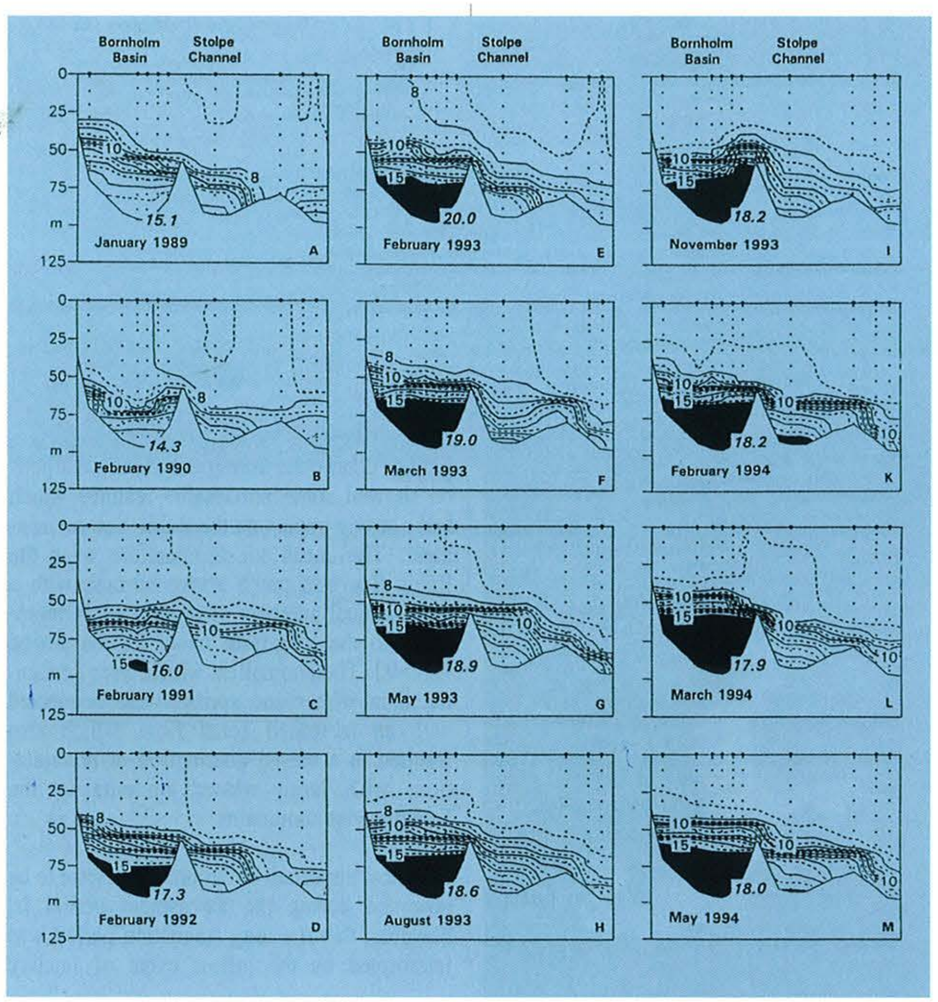


Fig. 2.16 Salinity distribution in the Bornholm Basin and adjacent areas between 1989 and 1994

(black areas: S ≥15 psu; figures in italics: maximum salinity; [447])

Long-term variations of salinity, oxygen, hydrogen sulphide and temperature in the deep water of the Baltic Proper during the present century (black bars: assessment period)

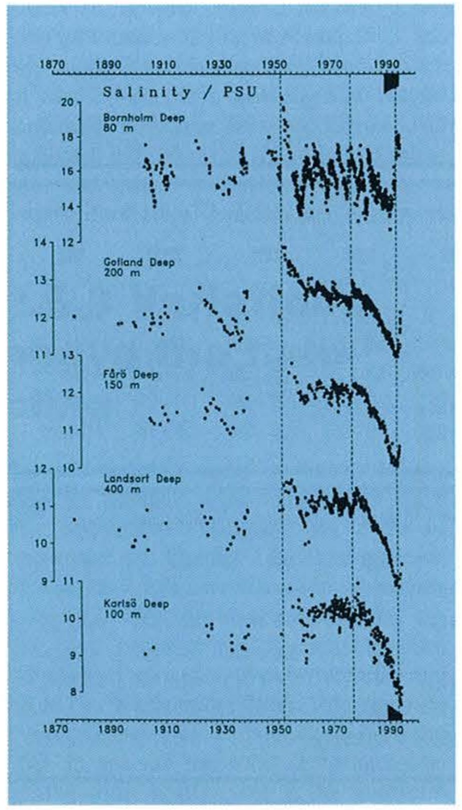


Fig. 2.17 Salinity

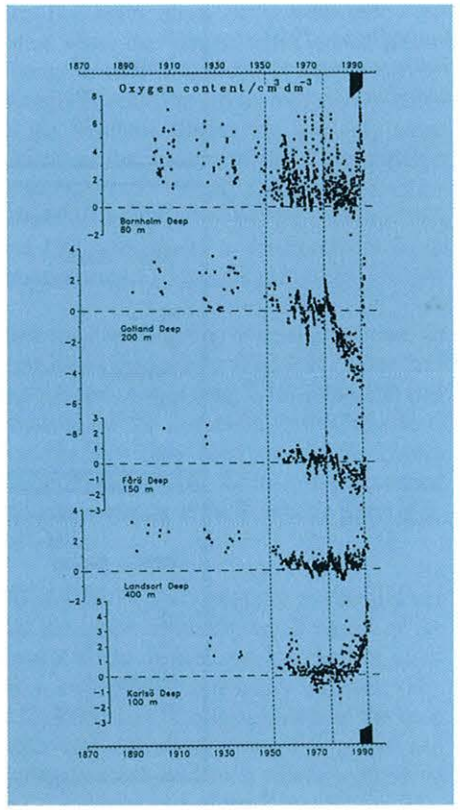


Fig. 2.18 Oxygen and hydrogen sulphide

(H_2S expressed as negative oxygen equivalents; black bars: assessment period)

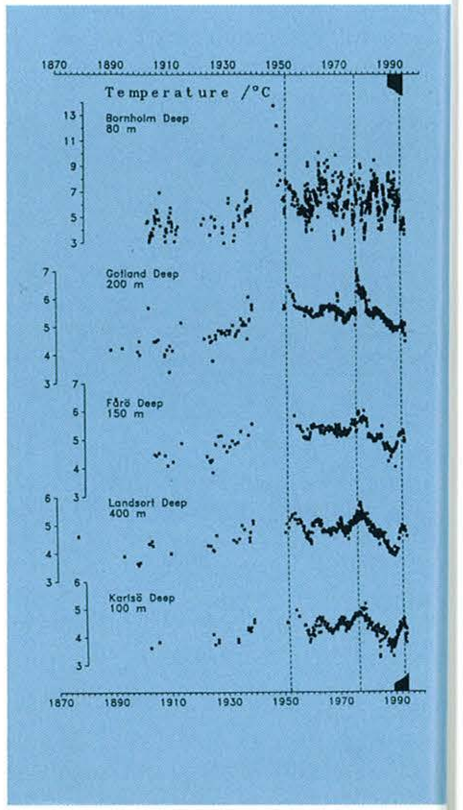


Fig. 2.19 Temperature

The stagnation period was finally terminated by inflows of lower magnitude in December 1993 and in March 1994 [*450*]. This water, however, proceeded directly into the Eastern Gotland Basin since the buffering capacity of the Bornholm Basin was still exhausted due to the major 'event' in January 1993 (Fig. 2.16). During spring 1994, oxygen concentrations of about 3-3.8 cm^3 dm^{-3} were measured between 170 m and the bottom of the Gotland Deep. These were the highest concentrations which have been observed at this station since the 1930s (Fig. 2.18). For the first time since 1977, the whole Baltic Proper was free of hydrogen sulphide.

2.6 SUMMARY

The meteorological, hydrological and hydrographic conditions during the assessment period showed some remarkable features which had a strong impact on the Baltic Sea environment. The mean air temperature over the Baltic Sea was much above normal, with a mean annual anomaly of around +1 °C compared to the long-term mean for the period 1880-93. The anomalies, which were concentrated in winter and spring, were connected with an increased zonal flow, which also resulted in a biased distribution of precipitation, with large winter amounts in the Scandinavian mountains.

The most important environmental factor to be observed during the assessment period is, however, that the long stagnation period was interrupted by the inflow event of January 1993. It is also important to note that there were regional differences in river run-off; the high inflows to the Baltic Proper dropped while inflows remained high from rivers draining into the northern and eastern basins of the Baltic Sea. One consequence of the changes in river inflow could be that the input of nutrients from non-point sources probably dropped during this period. Therefore, taken together, the meteorological, hydrological and hydrographic conditions of the assessment period must be considered as favourable for the environmental state of the Baltic Sea.

INPUTS

HELCOM[1]

3

STEPHEN F. LINTNER

HELCOM

[1] ANNEX 11.3 for addresses of authors

3.1 INTRODUCTION

Contaminants enter the Baltic Sea via rivers, through the atmosphere and by direct discharges from land, discharges from shipping, aquaculture and offshore installations. Within the HELCOM framework, the load of some contaminants entering the Baltic Sea via rivers and from different point sources located at the coast as well as through atmospheric deposition has been estimated. The results of these estimates are highlighted in this chapter.

3.2 RUN-OFF FROM LAND

3.2.1 Introduction

The Baltic Sea drainage area comprises 1,733,850 km² with approximately 92,200 km² located outside the territories of the Contracting Parties, i.e., in Belarus, Ukraine, Czech Republic, Slovak Republic and Norway. The quantities of contaminants entering the Baltic Sea via rivers as well as by direct discharges from land, varies from year to year, and the input is influenced by both natural and anthropogenic factors.

The assessments of contaminant run-off from land by HELCOM, known as *The Baltic Sea Pollution Load Compilations (PLC)*, started in the mid 1980s. The results of *PLC-1* were published in early 1987 [*214*] and represented a first attempt by HELCOM to compile very heterogenous data, very often preliminary or based on very rough background information that had been submitted to the Commission on various occasions. In 1989, the HELCOM adopted Recommendation 10/4 concerning a comprehensive study on pollutants discharged to the Convention Area from the territories of the riparian countries. This project, *PLC-2*, was implemented in 1990-92 in accordance with a unified methodology laid down in HELCOM Guidelines. The objective of this study was to assess, as accurately as possible, the major riverborne and direct inputs of selected contaminants in 1990. The report of *PLC-2* was published in 1993 [*225*].

In 1994, the third stage of the project (*PLC-3*) was initiated by HELCOM (HELCOM Recommendation 15/2) with the aim of collecting and assessing the data from the year 1995 taking fully into account the methodological experiences from *PLC-2* [*238*]. The report of *PLC-3* is expected to be ready for publishing in the second half of 1997.

3.2.2 Pollution Load Compilation (*PLC-2*)

This section deals with inputs to the coastal waters via rivers and from municipalities and industries discharging directly into the sea. The load data presented here originate from the report of *PLC-2* as well as from later information reported to HELCOM in relation to the follow-up of the 1988 Ministerial Declaration.

Although, *PLC-2* was more complete and precise than *PLC-1*, many uncertainties remained due to incomplete data sets, inadequate or non-comparable measurement methods and also the fact that *PLC-2* included only the main pollution sources. The information about small rivers (Q <5 m^3 s^{-1}), small settlements (<10,000 Population Equivalents - PE), the diffuse loading from the coastal zone between the measured rivers, by-passes and overflows were not included in *PLC-2*. The results of *PLC-2* should, therefore, be still used with caution in order to avoid misinterpretation.

Table 3.1 summarizes information on riverine and direct discharges in 1990 on a sub-regional basis according to the division of the Baltic Sea used in the Periodic Assessments. It should be noted that the quality of data varies with each of the parameters. The data sets for nutrients are more or less complete and thus a meaningful summary review could be produced. In contrast, information on heavy metals, due to incomplete data, is only given for general information. Data on organohalogens were not available as this was not required by *PLC-2*.

3.2.2.1 Nutrient input

According to the Report of *PLC-2*, the inputs of total phosphorus and total nitrogen in 1990 were estimated as more than 45,825 tons and 661,855 tons, respectively. After *PLC-2*, the information on nutrient inputs have been collected within HELCOM on several occasions, i.e. for the preparation of the 1993 Interim Status Report on the Implementation of the 1988 Ministerial Declaration and during the assessment of national measures to reduce nutrient inputs [*244*]. The data reported by the Contracting Parties to HELCOM 15 (1994) in relation to the Interim Status Report on Implementation of the 1988 Ministerial Declaration [*240*] are only available on a country-wide breakdown. However, this information is useful as it provides a rough comparison of the years 1990 and 1992 with regard to inputs of nutrients.

Area	Drainage area (countries)	BOD_7	P	N	Hg	Cd	Cu	Zn	Pb
BB	277 (FI-SE-NO)	100	2.3	38	0.05	1.3	94	570	>14
BS	219 (FI-SE)	150	2.3	47	>0.01	>1.7	>230	>1,100	>42
AS	9 (FI)	8.7	0.8	9.9	>0.01	0.03	2.9	16	1.1
GUF	413 (EE-FI-LV-RU)	290	12	140	>28	>18	>430	>280	>400
GUR	132 (EE-LV-LT-RU-BY)	140	3.4	85	n.i.	n.i.	>39	>130	n.i.
BAP	569 (DK-EE-DE-LV-LT-PL-RU-SE-BY-UA-CZ-SL)	>610	>18*	210	>22	>35	>440	>2,700	>830
BS & Kat	115 (DK,DE, SE)	>110	7.4	130	>0.14	>0.7	>43	>240	>9.2
Total	**1,734**	**>1,400**	**>46***	**660**	**>50****	**>57****	**>1,300****	**>5,000****	**>1,300****

* Data incomplete (riverine tot-P from Lithuania and Russia to BAP missing in *PLC-2*)
** Data incomplete for heavy metals
- loads and totals are rounded to two significant numbers

Abbreviations:
BB - Bothnian Bay, BS - Bothnian Sea, AS - Archipelago Sea, GUF - Gulf of Finland, GUR - Gulf of Riga, BAP - Baltic Proper, BS & Kat - Belt Sea & Kattegat & Sound; DK - Denmark, EE - Estonia, FI - Finland, DE - Germany, LV - Latvia, LT - Lithuania, PL - Poland, RU - Russia, SE - Sweden, BY - Belarus, UA - Ukraine, CZ - Czech Republic, SL - Slovak Republic, NO - Norway; n.i - =no information.

Table 3.1 Pollution load via rivers and by direct discharges from coastal municipalities and industries, 1990 [225] (metals in t; BOD_7, N and P -as total- in kt; drainage area in 10^3 km^2)

Table 3.2 Nutrient discharges into the Baltic Sea -in kt- from the territories of the Contracting Parties in 1990 [225] and 1992 [240]

	P		N	
	1990	1992	1990	1992
Denmark	5.3	3.9	83	70
Estonia	2.8	1.6	59	51
Finland	3.4	4.7	72	85
Germany	1.2	1.6	14	16
Latvia	3.2	1.8	94	89
Lithuania	1.7*	1.6	19	20
Poland	15	12	120	140
Russia	9.5**	6.5	81	32
Sweden	4.0	4.3	119	134
Total	**46**	**38**	**660**	**640**

* data for riverine tot-P from Lithuania are missing in *PLC-2*; for calculations the 1987 figure was used
** data for riverine tot-P from Russia are incomplete in *PLC-2* (no information was available for BAP)
totals are rounded to two significant numbers

Table 3.2 gives a general overview on loads of tot-P and tot-N introduced to the Baltic Sea from the territories of the Contracting Parties in 1990 and 1992. However, a comparison of the load values from both years should be done with caution as the differences between them do not necessarily reflect reductions or increases in loads since they are more likely due to differences in fresh-water run-off. In the Interim Status Report on Implementation of the 1988 Ministerial Declaration it was revealed that none of the Contracting Parties had achieved the overall reduction target of 50 %. Therefore, in 1994, the Commission decided that the national measures to reduce nutrients should be critically assessed and reported to HELCOM 17 in 1996 [*244*].

According to the conclusions from the assessment of national programmes on nutrient reduction there are substantial differences between the Contracting Parties with regard to the implementation state of the goals of the Ministerial Declaration 1988. The progress made by Contracting Parties on this issue is as follows:

– In **Denmark,** the 50 % reduction requirement is expected by the implementation of the National Action Plan on the Aquatic Environment (1987-95). Denmark has in practice met the target with regard to discharges from both municipal waste-water treatment plants and industry. When it was realised that the target for agriculture could not be met within the agreed time frame additional measures were adopted by the Action Plan for Sustainable Agriculture (1991-2000).

– **Estonia** has partly met the 50 % reduction goal but further measures are needed. Additional measures are expected to be listed in the national environmental protection programme which is under preparation.

– **Finland** had a Water Protection Programme until 1995. The targets set in the programme were, however, insufficient to meet the 50 % reduction target. A Water Protection Programme to 2005 is under finalization.

– **Germany** has partly met the 50 % reduction goal but further measures are needed. These measures are listed in a national nutrient-reduction programme but the details were not available at the time this chapter was prepared.

– **Latvia** has met the 50 % reduction goal mainly due to economic recession in the country and decrease of the industrial and agricultural production. Achievements in municipal waste-water treatment are considered to be sufficient and nutrient load from this source has decreased by more than 50 %. The Latvian Government has adopted an Environmental Policy Plan which aims to keep pollution at a low level.

– **Lithuania** has a National Environmental Strategy under finalization and adoption. This strategy should be sufficient to meet the 50 % reduction target by the year 2000 with regard to discharges from municipal waste-water treatment plants. Additional measures with regard to discharges from industry and agriculture will allow the 50 % reduction target to be met in 1997/98.

– In **Poland**, the goals of the Ministerial Declaration 1988 on 50 % reduction of nutrient load by 1995 have not been fully achieved. In May 1991, a governmental paper entitled National Environmental Policy was approved. The document also defines tasks in order to meet the requirements of the Ministerial Declaration. According to this document, short-term priorities until 1995, mid-term priorities until the year 2000, and long-term priorities until 2015 have been set up. Due to the economical situation and remnants of the former structure, Poland could not meet the 50 % reduction target within the short-term priorities. It is assumed, however, that the 50 % reduction target will be met within the mid-term priorities of the National Environmental Policy programme. By 1993, 28 % reduction of nitrogen and 21 % reduction of phosphorus in the riverine load discharged into the sea had been achieved. It is assumed that the river water quality will improve significantly within the long-term priorities.

– In the **Russian Federation**, the requirements of the Ministerial Declaration 1988 on 50 % reduction of nutrient load by 1995 are implemented in accordance with the Federal Programme for Ecological Situation Improvement in the Russian Part of the Baltic Sea Area elaborated on the basis of five regional programmes by the Government Decree. Programme measures apply to all sectors of activity, especially to municipalities, agriculture and industry, and are implemented by all possible national sources of financing, e.g., federal and local budgets, commercial sources, different funds. The timetable of the Programme implementation is by 2000.

– Since 1989, **Sweden** had a national action programme with the aim of reaching the overall 50 % reduction target for nutrients. Despite this, Sweden has not been able to fulfil the target. There is a comprehensive work going on elaborating an additional programme to implement the necessary measures to meet the target.

Accoring to the conclusions from the sector-wide assessment, **agriculture** is the largest anthropogenic source of nutrient inputs to the Baltic Sea. The 50 % target has been achieved in some countries in transition due to decreased use of fertilisers and decreased agricultural production. This was mostly caused by structural changes and financial difficulties. It is possible that in event of an economic recovery in these countries the agricultural load might increase in spite of the existing protection programmes. In the EU countries, the 50 % target has not been achieved and there is a need for additional measures. For nitrogen, these measures will largely be taken in the framework of the implementation of the Nitrates Directive (91/676/EEC). Generally, the 50 % goal is planned to be reached in 2000-2005 in most of those countries.

Nutrient inputs from **aquaculture** are locally quite considerable and many marine fish farms are situated in areas considered as "sensitive". Those Contracting Parties which consider this sector to be of importance as a source of nutrients, i.e., Denmark, Finland and Sweden, have achieved a reduction of specific discharges per unit production. However, the decline in specific discharges has been offset by a rise in fish production.

Industry is one of the major sources of nutrient inputs, mainly P-discharges, to the Baltic Sea. Almost all Contracting Parties have been able to implement the 50 % reduction target for P by 1995. Some Contracting Parties have problems with N-reduction from industrial sources. There might be some eventual increase of nutrient discharges from industry in the countries in transition in the future following the economic recovery.

As regards to **municipal** loading of phosphorus, the level of 50 % reduction between 1988 and 1995 was achieved by most of the

Contracting Parties. The achieved level of nitrogen reduction varied from 10 to 50 %. Due to economic problems, the countries in transition are still far from achieving their goal. This means that the municipal nutrient load to the Baltic Sea has not substantially decreased. However, the purification efficiency is expected to increase considerably according to existing national programmes in most of those countries.

NO_x emissions from the **transport** sector have not significantly decreased by any of the Contracting Parties. In the countries in transition, the emissions increased because of increasing transport with vehicles many of which do not have advanced catalytic converters. In the EU countries, the reduction of emissions using catalysts was counteracted by increased activities in the transport sector, including aviation, off-road vehicles etc., but the predicted or expected values for the year 2000 show a clear reduction. In the EU countries, the transport sector is meanwhile responsible for 60-70 % of all NO_x emissions. In the countries in transition, the increase in emissions from the transport sector to levels similar to those in the present EU countries is expected to be reached by about 2010. If this happens in the Baltic Sea Area, this means that transport will also be one of the main NO_x emission sources in future. This was taken into account by the HELCOM Recommendation on reduction of emissions from traffic in the Baltic Sea Area.

The change of NOx emissions from **power-production** plants varies from country to country. Some countries met the reduction goal by 1995. In the EU countries, a rather stable reduction is observed. The difficult economic situation in the countries in transition has led to a significant decrease in NO_x emissions. This might, however, only be a temporary trend. For both the transport sector and power production there is a specific problem: NOx emissions into the atmosphere are transported across the boundaries of the drainage area of the Baltic Sea and this has to be taken into account in calculating emissions from and the reduction required by each country.

In the assessment of national programmes on nutrient reduction, undertaken jointly by the Technological and Environment Committees of HELCOM in 1995, special emphasis was given to evaluate loads and measures in the main "problem areas" of eutrophication identified by the first coastal assessment [226]. In Table 3.3 and Figure 3.1, information is presented on major sources of nutrient inputs and loads in "problem areas" for different years during 1987-95.

Table 3.3 Major sources of nutrient loads to problem areas of eutrophication for different years [226,244] (cf. Fig. 3.1)

	Countries problem areas	Sources	P kt yr^{-1}		N kt yr^{-1}	
BS & AS	FI Turku Archipelago and adjacent coastal areas	- municipal waste water (several municipalities >10,000 PE) - industry (11 pulp & paper mills) - fish farming, esp. in AS - agriculture	1.0 (1987)	1.0 (1992-95)	12 (1987)	12 (1993-95)
GUF	EE Narva & Tallinn Bights	- Narva River - municipal waste water	1.8 (1990)	0.8 (1994)	38 (1990)	20 (1994)
	FI Outer Helsinki & Kotka regions	- municipal waste water (Helsinki region, other municipalities) - industry (29 pulp & paper mills) - agriculture	0.5 (1987)	0.5 (1993-95)	9.8 (1987)	10 (1993-95)
	RU Neva Bight and adjacent coastal areas	- municipal waste water - industry (pulp & paper mills) - agriculture	5.9 (1987)	3.3 (1994)	31 (1987)	24 (1994)
GUR	EE Pärnu Bight	- Pärnu River - Pärnu municipal waste water	0.4 (1990)	0.1 (1994)	8.4 (1990)	3.5 (1994)
	LV Outer Riga region	- rivers Daugava & Lielupe - Riga region (municipal waste water, industry & agriculture)	2.3 (1987)	1.2 (1993)	80 (1987)	46 (1993)
BAP	DE Bodden	- agriculture - municipal waste water	2.1 (1990)	1.6 (1995)	38 (1990)	35 (1995)
	LT Curonian Lagoon	- Nemunas River - agriculture - municipal waste water - industry (3 plants, discharging directly to the sea)	1.8 (1987-90)	1.6 (1992)	40 (1987-90)	20 (1992)
	PL Oder Estuary, Pomeranian Bight, Gdansk Bight and Vistula Lagoon	- municipal waste water - industry - agriculture	18 (1988)	14 (1993)	230 (1988)	170 (1993)
	RU Curonian Lagoon and Vistula Lagoon	- municipal waste water - industry (pulp & paper mills) - agriculture	0.6 (1987)	0.3 (1994)	2.9 (1987)	2.3 (1994)
	SE Hanö Bight and Stockholm Archipelago	- agriculture - municipal waste water	0.2 (1987)	0.1 (1992-94)	7.6 (1987)	6.3 (1992-94)
BS & Kat	DK Southern Sound, Outer Arhus region, Great Belt	- major input (>80 %) via rivers from agriculture	1.2 (1989)	0.79 (1993)	19 (1989)	41 (1993)
	DE Flensburg Fjord, Kiel Bight, Lübeck Bight and Mecklenburg Bight	- agriculture - municipal waste water	1.4 (1990)	1.1 (1995)	25 (1990)	23 (1995)
	SE Southern Sound, Laholm Bight	- agriculture - municipal waste water	0.28 (1987)	0.15 (1992)	12 (1987)	9.7 (1992)

- totals are rounded to two significant numbers
- for abbreviations see Table 3.1

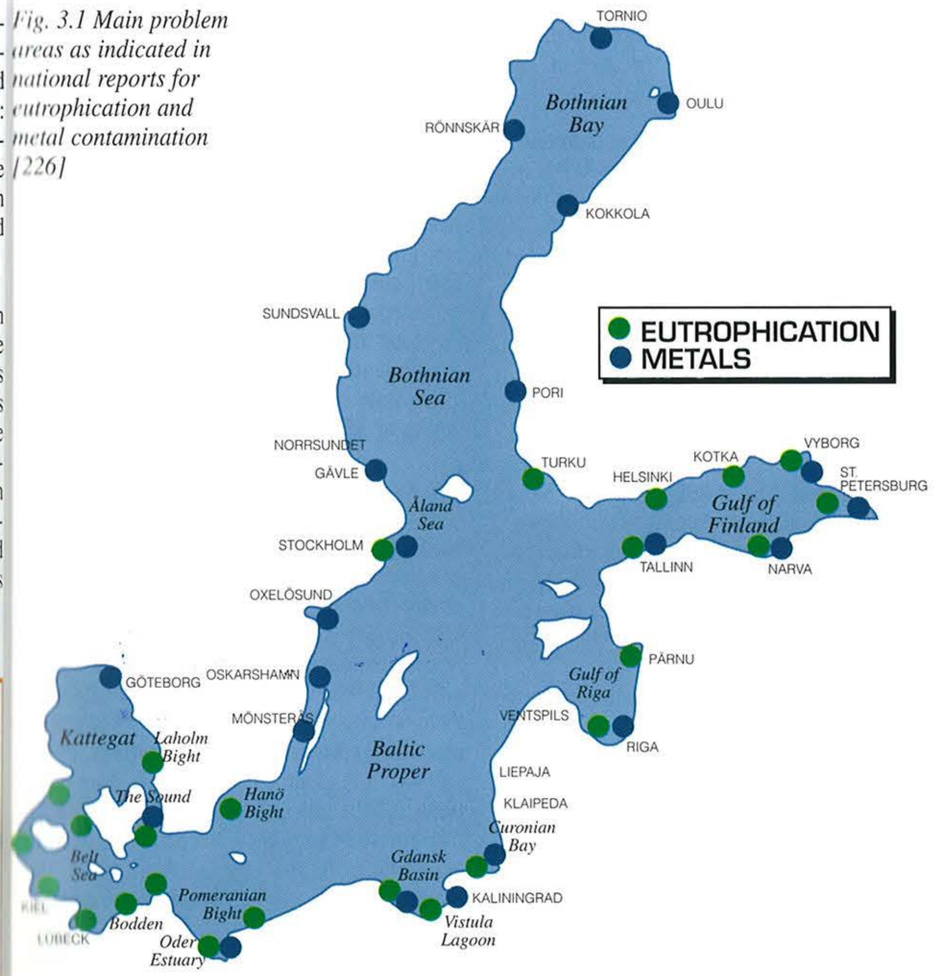


Fig. 3.1 Main problem areas as indicated in national reports for eutrophication and metal contamination [226]

3.3 ATMOSPHERIC INPUT

3.3.1 Introduction

The estimates given on the atmospheric inputs to the Baltic Sea and referred to in this document are based on studies of the Group of Experts on Airborne Pollution of the Baltic Sea Area (EGAP), especially on an EGAP report which gives the best possible estimate of the airborne pollution load to the Baltic Sea for the five-year period 1986-90 [222]. Also other detailed information, e.g., on the areal distribution of the deposition of oxidised and reduced nitrogen to the Baltic Sea (gridded values), and annual statistics on HELCOM/EGAP data for 1986-90 are available in that publication. A new report for 1991-95 is under preparation, but unfortunately the results were not available for this assessment. The atmospheric load of nitrogen has been shown to constitute a considerable fraction of the total load. No information on the load of organic nitrogen to the Baltic Sea is available since this parameter is not included in the monitoring programme.

Phosphorus is analysed by some countries in rain water samples, but only in very few cases do the concentrations exceed the detection limit. Therefore, phosphorus measurements are not included in the EGAP programme. The load of phosphorus from air is estimated to be very small and has been ignored in these estimations.

Air pollution released from land-based sources is often transported long distances before it is deposited either by dry or **wet deposition**. Wet deposition takes place by uptake of gases and particles by cloud droplets, followed by precipitation (rain-out) or by scavenging of gases and particles by falling raindrops or snowflakes (wash-out). The rate of wet deposition over the Baltic can be expected to show a seasonal dependence because precipitation intensity varies during the year. Wet deposition is also a function of the size distribution and chemical composition of the rain or cloud droplets, the diffusion constant and *Henry*'s constant of the gases involved. For particles, the scavenging is highly dependent on, e.g., their origin and size.

Dry deposition is influenced by a multitude of physical and chemical factors. The significance of these factors varies depending on the physicochemical characteristics of the pollutant, the meteorological conditions of the atmosphere and the properties of the surface of deposition. Dry deposition processes over the ocean differ from those over land because of the unique character of the surface and the temperature difference between the air and the ocean. This is of special importance for the Baltic Sea, which is partly covered by ice almost every winter.

The winds over the Baltic Sea are often quite variable, but on an annual basis the area is dominated by west-southwesterly winds. The Baltic Sea, as also the North Sea, is significantly affected by atmospheric long-range transport of man-made emissions of pollutants. Therefore, the results from the monitoring programme of EGAP which is limited to the Baltic Sea cannot alone give a complete picture of the state of air pollution and its origin from remote sources. The knowledge is necessary if control strategies for the reduction of atmospheric deposition fluxes of pollutants to the Baltic Sea are to be developed. The only way of delineating the atmospheric transport pathways, and hence the emitter-receptor relationship, is through numerical modelling. With modern computers it is possible to carry out calculations that cover not only a whole continent such as Europe but also include the complex physical and chemical processes that control the transport, transformation and deposition of air pollutants.

3.3.2 Monitoring programme

Estimating the atmospheric pollution load on the sea surface for the open sea is difficult because data on airborne pollution concentrations and meteorological data such as precipitation are scarce or lacking. To compensate for this it is necessary to resort to methods of approximation. The EGAP monitoring network consists of 26 land-based measuring sites situated in various types of rural areas to avoid the influence of local industrial sources. The main purpose of the monitoring network is to produce data that can be used for estimating the deposition of harmful substances to the Baltic Sea. That purpose can only be fulfilled if data reported by the Contracting Parties are of high quality.

The monitoring programme is based on measurements of "routine-minimum-requirement parameters", of other parameters on an "experimental (voluntary) basis", and on a quality assurance procedure. The HELCOM/EGAP monitoring programme calls for determination of the concentrations of NH_4 and NO_3 in precipitation as a minimum requirement. From 1990 onwards, measurements of Pb, Cd, Cu and Zn in precipitation have also been requested on a routine basis, from at least one station in each country. The measurement of ambient air concentrations of pollutants is not a mandatory part of the programme of EGAP and therefore, constitutes a voluntary contribution by Contracting Parties.

The quality assurance protocol used by Contracting Parties for the monitored parameters is similar to the EMEP Quality Assurance Plan. Accordingly, EGAP has initiated and participated in a number of intercalibration and intercomparison exercises [*224*]. For airborne pollution load estimations, the Baltic Sea is divided into five sub-basins, i.e., the Gulf of Bothnia (*A1*), Gulf of Finland (*A2*), Northern and Central Baltic Proper (*A3*), Southern Baltic Proper (*A4*) and Belt Sea, Kattegat and Sound (*A5*), as shown in Figure 3.2.

3.3.3 Nitrogen deposition

Intercalibrations have shown that ammonium and nitrate are accurately determined but that minor deviations in measurements of input load may occur because of the different designs of the precipitation samplers. Precipitation concentrations from stations selected as representative of the various sub-basins are used to calculate basin-average concentrations. The results for ammonium and nitrate are used to calculate the concentration of total nitrogen, defined here as the sum of the two values. The average total nitrogen concentrations for the period 1986-90 was 1.34 mg dm^{-3}, and no distinct changes for tot-N over this period have been observed. Geographically, however, there is a change in concentration from north to south as shown in Figure 3.3.

Estimates on the deposition of pollutants in the open sea have to be based on extrapolation of data obtained from sampling locations on either the coast or on islands. However, only wet deposition can be estimated in this way since only measurements of precipitation concentrations form part of the mandatory monitoring programme. Although, the data on contaminant concentrations in ambient air, which is collected on a voluntary basis may be used for estimating the dry deposition, the parametrization of dry deposition velocities is difficult and leads to considerable uncertainties in the flux calculation. In any event, it should be noted that the dry deposition component is expected to constitute only the minor part of the total deposition. For example, a recent experimental estimate for nitrogen deposition to the Baltic Sea yielded an overall dry contribution of 15 %.

The wet deposition is estimated on the basis of weighted mean concentrations measured at each of the sampling stations. To calculate the sub-basin depositions, it is necessary to first obtain the annual deposition flux, i.e., the area specific deposition. For this purpose two different methods are used. The first, the *experimental method* relies exclusively on measured data on concentrations and the precipitation recorded at the various coastal stations. In this method, it is assumed that both the concentrations and the precipitation are also representative for the open sea. Since this is a crude approximation, the results obtained by this method must be viewed with some caution.

The second, the *hybrid method*, relies on measurements of contaminants in rainwater and includes both observed and calculated amounts of precipitation. Model calculations are considered more reliable for estimating precipitation over the open sea than extrapolation of actual coastal measurements. The hybrid fluxes are 10-20 % higher than the experimental ones. However, for both types of fluxes there is a clear and consistent tendency for fluxes to decrease from about 1,100 kg N km^{-2} yr^{-1} in the southern parts of the Baltic Sea, which are closer to the European air pollution sources, to around 650 kg N km^{-2} yr^{-1} in the north.

On the basis of the deposition fluxes the average wet deposition for 1986-90 to the Baltic Sea has been estimated by the experimental and hybrid methods to be 314 and 330 kT N per year, respectively. The values for deposition in 1986 compare quite well with earlier estimates but the more recent estimates are lower in the north by about 20 % and higher in the two southern basins by about 10 %. The results show that the wet nitrogen deposition to the Baltic Sea is divided almost equally between oxidised and reduced compounds and that there is no discernible temporal trend during the latter half of the 1980s.

There is a third method for estimating the atmospheric deposition of contaminants which relies exclusively on *model calculations*.

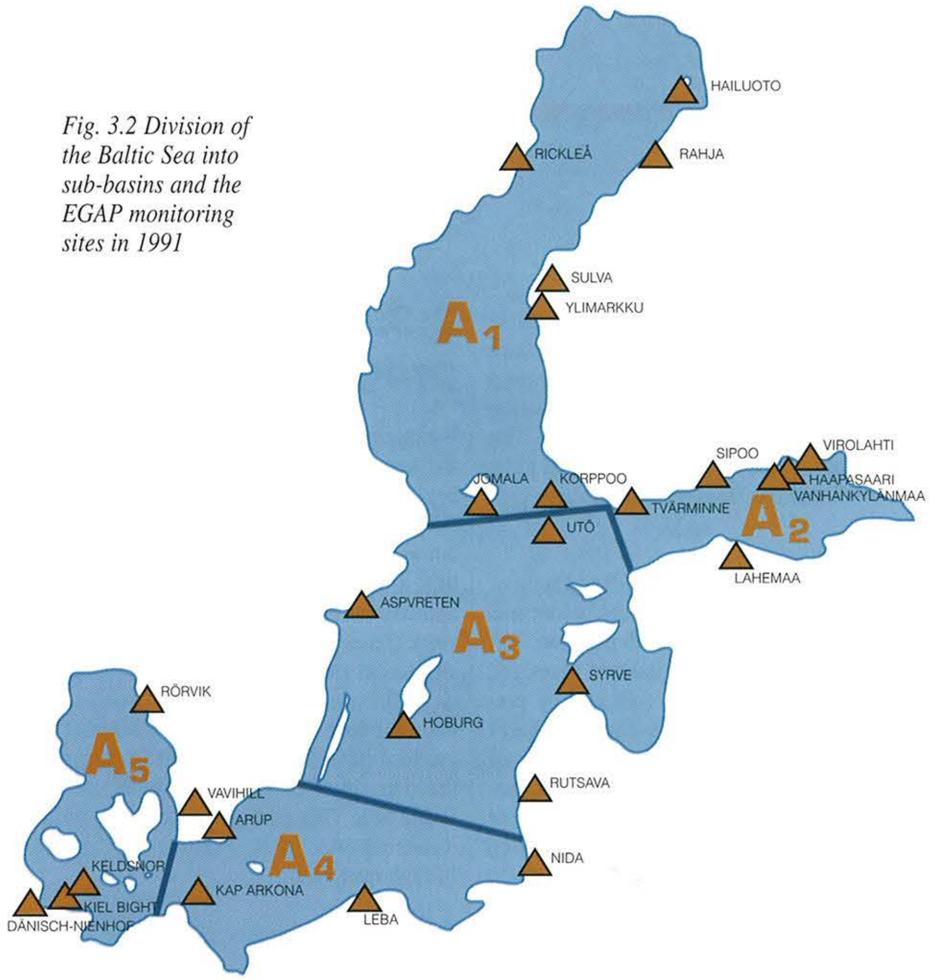


Fig. 3.2 Division of the Baltic Sea into sub-basins and the EGAP monitoring sites in 1991

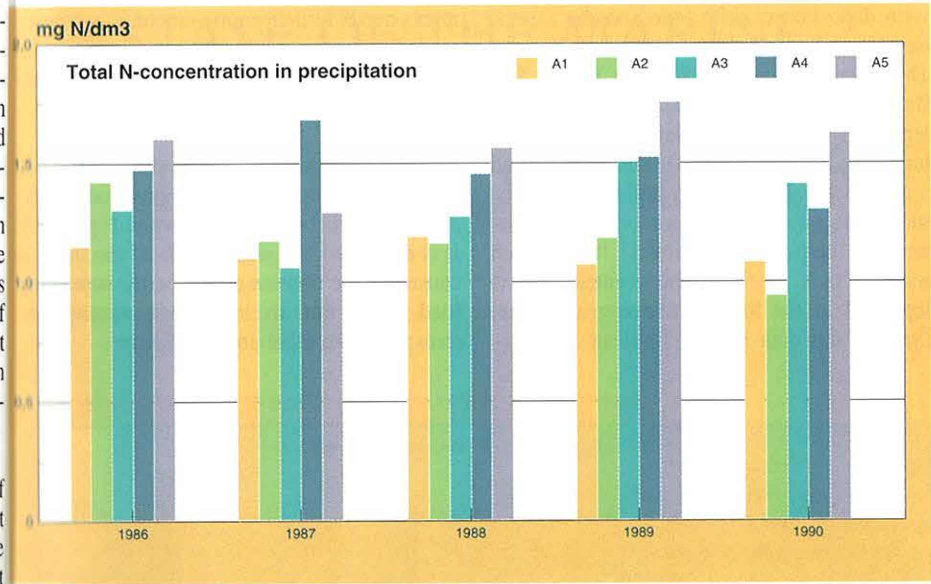


Fig. 3.3 Annual mean tot-N concentrations in the Gulf of Bothnia (A1), Gulf of Finland (A2), Northern and Central Baltic Proper (A3), Southern Baltic Proper (A4) and in Kattegat, Belt Sea and Sound (A5), 1986-90 [222]

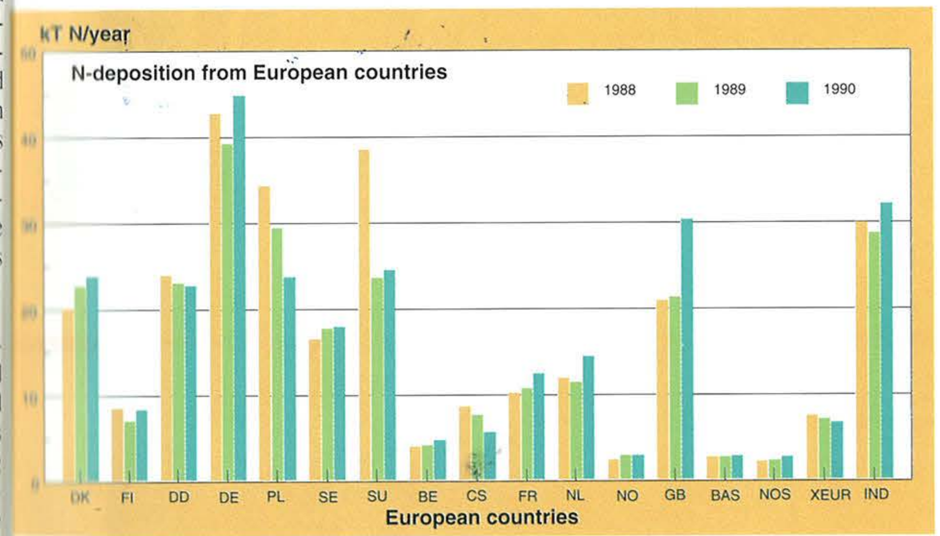


Fig. 3.4 Model results on the total (wet + dry) deposition D_T^{M-W} of nitrogen compounds to the Baltic Sea (kT N yr^{-1}), from EMEP/MSC-W [280]

(DK - Denmark, FI - Finland, DD - German Democratic Republic, DE - Federal Republic of Germany, PL - Poland, SE - Sweden, SU - Soviet Union, BE - Belgium, CS - Czechoslovakia; FR - France, NL - The Netherlands, NO - Norway, GB - United Kingdom, BAS - Baltic Sea, NOS - North Sea, XEUR - Minor contributors with <1,000 t yr^{-1}, IND - deposition of indeterminate origins)

These calculations include the deposition based on fields for both precipitation and concentration. They can be considered quite reliable if good emission data are available. The model results also include dry deposition calculated as the product of airborne concentrations and deposition velocities. The *Lagrangian* trajectory model of EMEP's MSC-W was one of the first to be developed for operational use. It is an one-layer model where concentrations are calculated as averages over the well-mixed boundary layer. The model is currently being used on a routine basis for the calculations of transboundary pollution fluxes and deposition of sulphur and nitrogen compounds over Europe. The results show that the concentration and deposition fields are predicted reasonably well.

For routine calculations of horizontal transport of air pollution the MSC-E of EMEP has developed and tested a *hybrid Lagrangian-Eulerian model* which allows complex atmospheric processes to be included.

The EMEP models calculate concentrations and depositions of nitrogen compounds. Monthly averages are given in 36 emission and deposition domains representing countries and sea areas. The models are able to keep track of the domain in which the pollution was emitted, which makes it possible to allocate the deposition on the whole Baltic Sea from relevant emitter countries. The reliability of model results is in general dependent on the quality of the input data. The better the model - the higher the requirement for good and reliable data. It is difficult to give an overall range of uncertainty for the results, but since the uncertainty of the input data over sea areas is larger than for land, the same must be true for the model results. In addition, the present models tend to overestimate depositions in Central Europe relative to measured depositions and to underestimate them in the very cold or frozen areas such as in Scandinavia. In addition, it is worth noting that the emission databases used for model calculations are still in a very preliminary stage.

For nitrogen deposition to the open Baltic Sea, it was found that 65 % of the main contribution came from the Baltic Sea countries, probably because of their proximity (Fig. 3.4). Other prominent contributors are Great Britain, France and the Netherlands, which are all upwind of the predominant westerly winds in the Baltic Sea and which are also among the major European emitters. The former Czechoslovakia is also another major contributor.

Based on the model calculations the total dry and wet deposition of nitrogen to the Baltic Sea was found to be between 260 and 285 kT N yr^{-1} during 1988-90. Thus, a reasonable estimate for the total deposition of nitrogen to the Baltic Sea in the latter half of the 1980s is 300 ± 30 kt N yr^{-1}. The nitrogen deposited is composed of reduced and oxidised forms of nitrogen in the ratio 40:60. The values for oxidised nitrogen deposited to the various sub-basins of the Baltic Sea have been estimated by MSC-E and the results are given in Table 3.4.

Table 3.4 Annual total (wet + dry) deposition D_T^{M-E} of oxidised nitrogen to the sub-basins of the Baltic Sea, 1988 and 1989 (values in kt; model results from MSC-E [222])

	Sub-basin	1988	1989
A1	Gulf of Bothnia	28.0	23.7
A2	Gulf of Finland	13.0	13.6
A3	Baltic Proper, Northern and Central	55.5	49.9
A4	Baltic Proper, Southern	45.8	40.2
A5	Kattegat, Belt Sea and the Sound	20.8	19.2
A0	**Baltic Sea, total**	**163.1**	**146.6**

3.3.4 Metal deposition

As shown by the results from trace-metal intercomparisons, the reliability of the data for all trace elements except lead is still very low. Therefore, only the total deposition of lead to the Baltic Sea was estimated in the last airborne pollution-load report [222]. Furthermore, model calculations for the period 1980-85 have been restricted to lead because the emission databases for other metals are still in a very preliminary stage. The data and any derived estimates should be viewed with some caution.

The wet/bulk[2] deposition of lead is estimated as for the nitrogen compounds, i.e., on the basis of the precipitation weighted mean concentrations measured at each of the sampling stations. The annual mean concentration of lead in precipitation was found to fall in the range 4.8 to 9.1 µg dm^{-3}. The average sub-basin depositions of Pb based on the *experimental method* (Dwb^X) and *hybrid estimates* (Dwb^H) are shown for 1986-90 in Table 3.5. The experimental results suggest that the deposition of lead to the Baltic Sea decreased during the latter half of the 1980s. Such a trend is not, however, to be seen in the hybrid estimates. Because of the predominantly low emission heights (automobiles) it has been assumed that 15 % of the emitted lead is deposited in the local emission-grid square. Even so the data quality for lead measurements needs further improvement, it is consid- ered that a reasonable estimate for the tota deposition of lead to the Baltic Sea in the lat- ter half of the 1980s was 1,300 ± 250 t yr^{-1}.

The annual deposition of lead to the Baltic Se in 1984-85 arising from emissions from differ- ent European countries is shown in Figure 3.5 It is estimated that about 70 % of the input i caused by the riparian countries, the remainin load being due to long-range atmospheri transport from other areas in Europe.

Table 3.5 Annual mean deposition of Pb for the period 1986-90, in t yr^{-1} [222]*

	Sub-basin	Dwb^X	Dwb^H
A1	Gulf of Bothnia	280	429
A2	Gulf of Finland	112	203
A3	Baltic Proper, Northern and Central	269	261
A4	Baltic Proper, Southern	218	275
A5	Kattegat, Belt Sea and the Sound	180	250
A0	**Baltic Sea, total**	**1,059**	**1,419**

Dwb^X -experimental method
Dwb^H - hybrid estimate
* For 1986, the average values for 1987-90 have been used

[2]The term bulk deposition refers to precipitation samplers which are also open during dry periods.

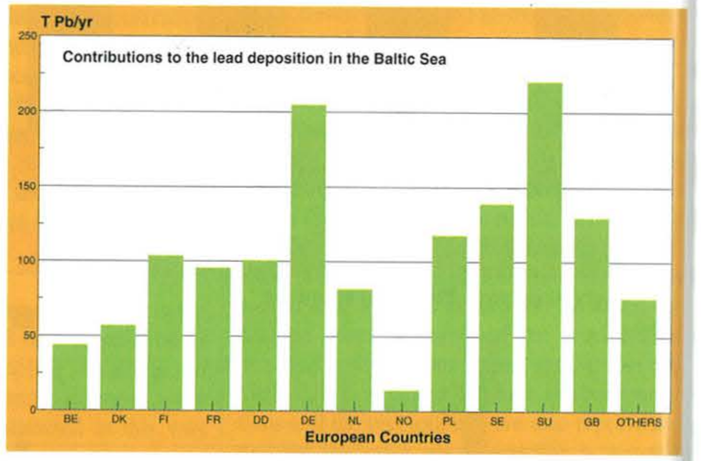


Fig. 3.5 Contributions of some European countries to the lead deposition in the Baltic Sea, 1984-85 (t Pb yr^{-1}; for abbreviations see Fig. 3.4)

3.4 SUMMARY

The results of the most recent pollution load estimates via run-off from land (*PLC-2*) should be used with caution since the information about small rivers (Q <5 m^3 s^{-1}), small settlements (<10,000 PE), the diffuse loading from the coastal zone between the measured rivers, by-passes and overflows was not considered. For the future, monthly load data on an annual basis are urgently needed. The *PLC-2* inputs of total phosphorus and total nitrogen in 1990 were estimated as >46 kt and >660 kt, respectively.

Regarding airborne load, a reasonable estimate for the total deposition of nitrogen to the Baltic Sea in the latter half of the 1980s is 300 ± 30 kt yr^{-1}. No distinct changes for the airborne deposition of total nitrogen over 1986-90 have been observed. However, geographically there is a change in concentrations from north to south. The nitrogen deposited is composed of both reduced and oxidised forms which are divided in the ratio 40:60. For nitrogen deposition it was found that the main contribution of 65 % came from the Baltic Sea countries. Other prominent contributors wer Great Britain, France and the Netherland which are all upwind of the predominant wes erly winds in the Baltic Sea area and which ar also among the major European emitter Czechoslovakia was another major contribu tor. For the Baltic Proper and the Gulf o Bothnia, the airborne load of nitrogen repre sents about 40 % and 30 % of the total inpu respectively. For the Gulf of Finland and fo the Belt Sea, Kattegat and Sound area, the ai borne component accounts for about 20 % o the total input.

Due to the incomplete data sets for input dat no comparison for trace metals can be mad between the loads from run-off and the atmos phere. A reasonable estimate for the total lea deposition to the Baltic Sea in the latter half o the 1980s seems to be 1,300 ± 250 t yr^{-1}. Abou 70 % of the input is estimated to be caused b the riparian countries, the remaining part i due to long-range atmospheric transport fro other areas in Europe.

STATE OF THE MARINE ENVIRONMENT OF THE BALTIC SEA REGIONS

4

HELCOM

4.1 GULF OF BOTHNIA

(Bothnian Bay, Bothnian Sea, Åland Sea, Archipelago Sea)

J. Wikner[1] (Convener)

4.1.1 Introduction

The Gulf of Bothnia is a brackish-water body with two major basins and two larger archipelago areas in the northern and south-eastern part. The northern basin has an average depth of 43 m and the southern basin of 68 m, separated by a shallow sill of 20 m, the Northern Quark. The maximum depths in the Bothnian Bay and in the Bothnian Sea are 147 and 230 m, respectively. A counter-clockwise rotation of long-term mean currents occurs in the basins and in the Gulf as a whole.

		Bothnian Bay	Bothnian Sea
Surface area[c]	km^2	36,260	79,256
Drainage area[c]	km^2	277,000[a]	228,000[b]
Depth (av, max.)[c]	m	43, 147	68, 230
Fresh-water inflow[c]	$km^3\ yr^{-1}$	105	87.6
Volume[c]	km^3	1,500	4,889
Residence time[d]	yrs	5.3	2.8

[a] Sweden 47 % and Finland 53 %; including the Quark
[b] Sweden 79 % and Finland 12 %; including the Åland Sea and the Archipelago Sea (9,000 km^2)
[c] [220]
[d] [8]

Table 4.1.1 Characteristics of the two major sub-basins of the Gulf of Bothnia

Inflow of fresh water, arising from numerous rivers and precipitation, forms the dominating estuarine character of the Bothnian Bay, while saline water enters from the surface of the Baltic Proper, promoting more marine conditions in the Bothnian Sea. The drainage area of the rivers includes latitudes in the very vicinity of the Polar Circle. Erosion and abrasion takes place and the rivers transport the material with cyclic pulses caused by the seasonal variations of the arctic climate, including humic substances derived from large peat-land areas and strongly podzolised soils. Surface salinity is around 3.5 psu in the northern basin approaching fresh water conditions in the river mouths. In the Bothnian Sea, the surface salinity increases to around 6.5 psu.

The upper 60 m water layer turns over twice a year as a consequence of spring warming and autumn cooling. This seasonal turnover of the whole water body promotes high oxygen concentrations at the sediment-water interface. The sediment surface is thus strongly oxidised throughout the whole Gulf of Bothnia, and in the open sea anoxic conditions have never been observed.

Before the end of the assessment period, a group of 10 large industries were considered as important sources of polluting substances to the Gulf by the Helsinki Commission [243]. Surrounding land areas are further exploited by forestry and water-power regulating activities. From the Finnish part of the drainage area, agriculture is the largest single source of anthropogenic nutrients [545].

This report is an assessment of the environmental state and its development in the Gulf of Bothnia based on the BMP monitoring data available. Additional national data have been included when possible. The station network utilised in the monitoring programme is shown in Figure 4.1.1 with the station names according to the nomenclature of the BMP. National stations included in the assessment are also indicated.

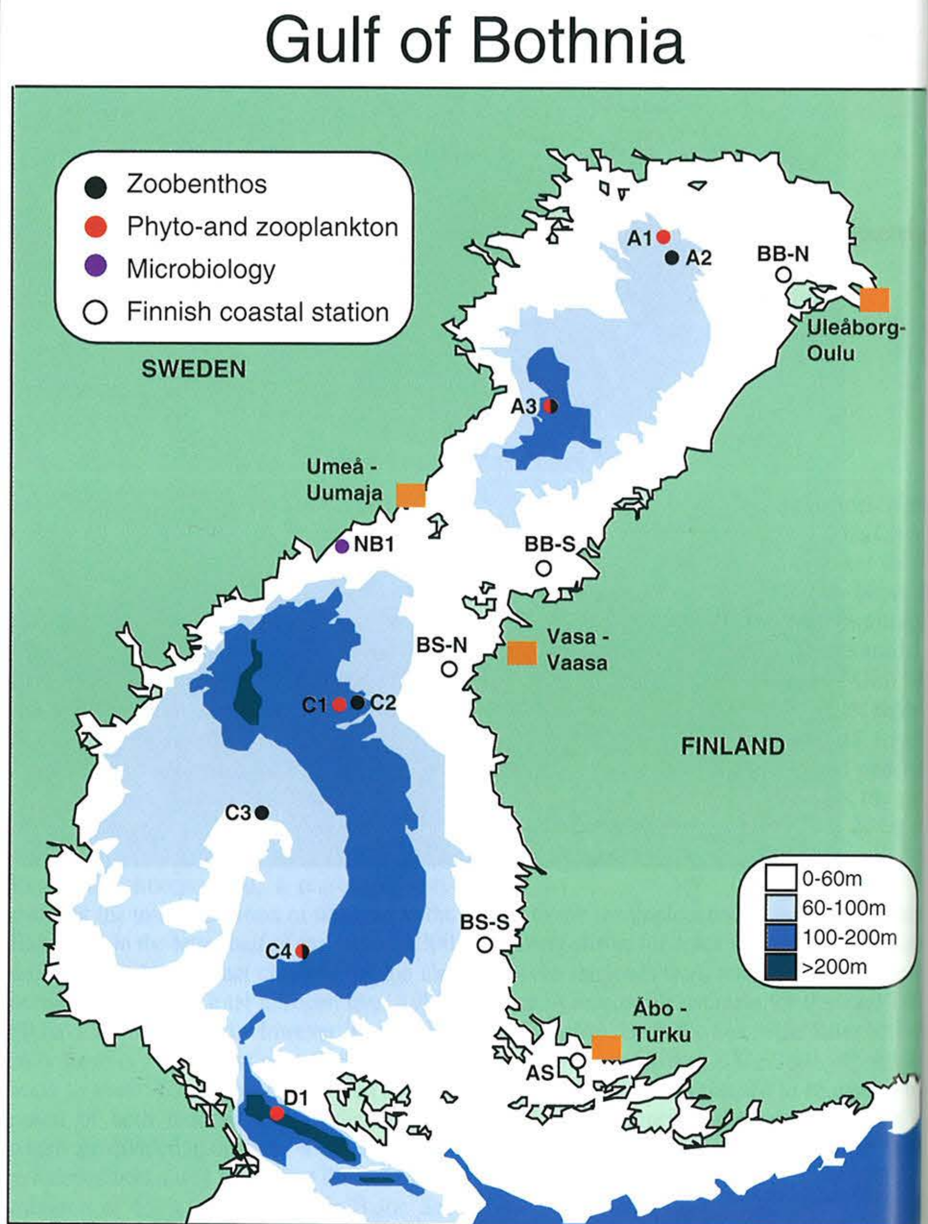


Fig. 4.1.1 The Gulf of Bothnia and its sub-basin's bottom topography (Stations included in the assessment are indicated using the BMP nomenclature, except coastal stations having national codes.)

[1] see ANNEX 11.3 for addresses of authors

4.1.2 Meteorological, hydrological and hydrographic conditions

P. Alenius[1], *A. Omstedt*

The assessment period was characterised by mild air temperatures and extremely mild ice conditions in the whole Baltic Sea. However, because of its northern location, the Bothnian Bay also freezes in mild winters. The deep water temperature in the Gulf of Bothnia is quite variable with no significant trends during the assessment period. The decreasing trend in salinity was observed to continue in both basins. The water level showed normal variability during the assessment period.

The circulation of the Gulf of Bothnia is basically determined by winds, buoyancy forces and earth rotation. This leads to long-term counter-clockwise residual circulation. There also exist large-scale eddies in the northern Bothnian Sea. Observations have shown that the netflow is from north-west to south-east in the northern Bothnian Sea [200].

Temperature - The temperature of the upper layer down to about 60 m varies considerably during the year. This layer is subject to turnover in spring and autumn. The upper layer is cold in winter but in summer the 10-20 m thick surface layer is warm, whereas the temperature of the deep water remains more stable throughout the year. For the purpose of the assessment, the 80 m depth was chosen to be representative of the deep water. The data show no significant trend in the deep water temperature either in the Bothnian Bay or Bothnian Sea (Figs. 4.1.2, 4.1.3). The sea-surface temperature data are much more scattered and also without any clear trend , probably due to the larger influence of river discharge and vertical mixing.

Salinity - The Gulf of Bothnia receives saline water to its deep layers primarily from the Baltic Proper surface layers. The deep water salinity is a slowly changing variable. In the Bothnian Sea, the salinity at 95-115 m showed a clearly decreasing trend of 0.07 psu per year during the assessment period (linear regression: p=0.072 and r^2=0.79; Fig. 4.1.4). This trend seemed to be independent of the season. For the Bothnian Bay, a decreasing tendency was not found statistically significant (p=0.44, r^2=0.64; Fig. 4.1.5; cf. [440]). The amount of data were, however, too small to identify the causes of the trend observed for the Bothnian Sea. Most probably, it was a result of the overall freshening of the whole Baltic Sea during the long period without effective saline water inflows following the 1976 major inflow event. The present values are below the long-term mean values [194]. In summer 1996, the 1993 saline water inflow into the Baltic Sea had not yet affected the deep water salinity in the Gulf of Bothnia.

[1] see ANNEX 11.3 for addresses of authors

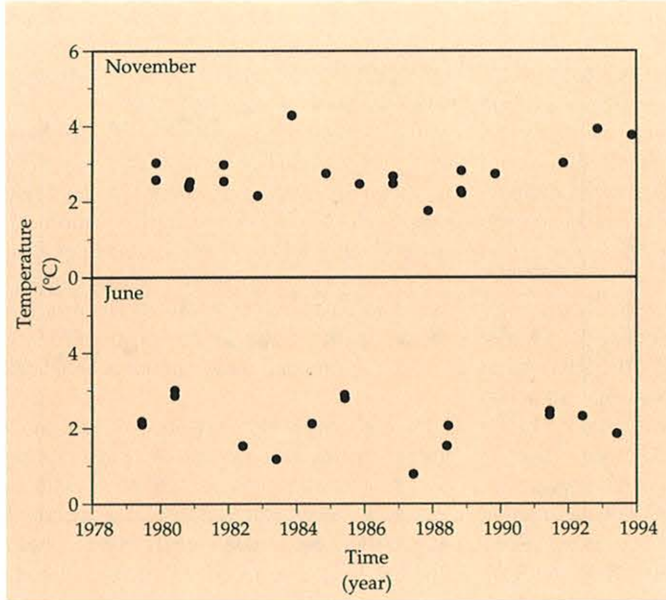


Fig. 4.1.2 Deep-water temperatures (80 m) at station C4 (=SR5) Bothnian Sea

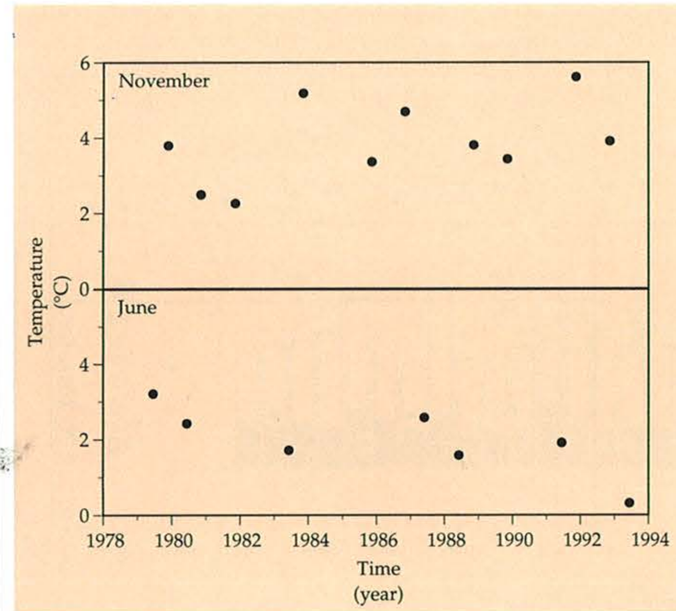


Fig. 4.1.3 Deep-water temperatures (80 m) at station A1 (=F2) Bothnian Bay

4.1.3 Hydrochemistry

P. Sandén[1], *H. Pitkänen*, *K. Eloheimo*

4.1.3.1 Nutrient variability

The Bothnian Bay has low phosphorus concentrations and comparatively high concentrations of nitrogen, while in the Bothnian Sea the phosphorus concentrations are considerably higher and the nitrogen concentrations, primarily inorganic species, are lower. This nutrient distribution favours a limitation of the lightsaturated planktonic production by the available phosphorus in the Bothnian Bay, while phosphorus and nitrogen might alternate as limiting substances in the Bothnian Sea.

[1] see ANNEX 11.3 for addresses of authors

Fig. 4.1.4
Deep-water salinity (80 m) at station C4 (=SR5) Bothnian Sea

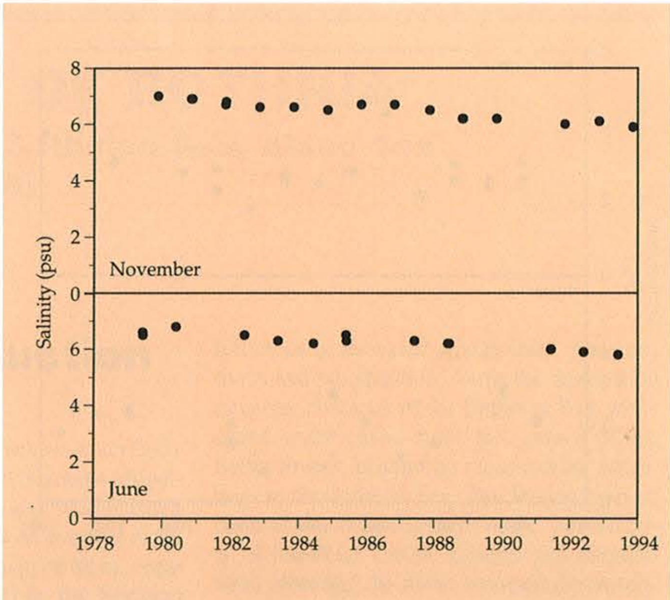
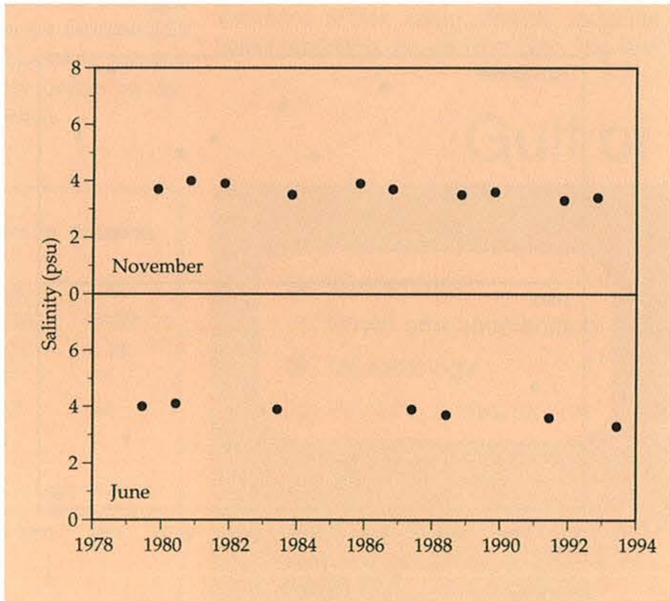


Fig. 4.1.5
Deep-water salinity (80 m) at station A1 (=F2) Bothnian Bay



Analyses of long-term changes in chemical variables during the period 1970-93 showed several significant trends. The concentrations of inorganic nitrogen have increased throughout the Gulf. Similar trends were also found for total nitrogen, which in part is due to the increase in inorganic nitrogen. Inorganic phosphorus concentrations have decreased in the Bothnian Bay, in both surface and bottom waters. Total phosphorus has increased in the surface waters of the Northern Quark, Bothnian Sea and Åland Sea. Silicate concentrations have generally decreased in the Gulf. The long-term changes presented here agree with previous published studies [*589,592*]. Trend estimates on the riverine load to these basins show significant increases in total nitrogen and silicate loads, while the changes in phosphorus loads are not significant. These changes are also in accordance with previously published studies [*558,635*]. Changes in riverine loads can, therefore, to some extent explain the observed trends in the Gulf. The increasing loads were mainly due to higher run-off during the 1980s compared with the 1970s. The increased input of nitrogen from the Finnish drainage area was also due to an increased use of fertilisers in the 1970s and 1980s [*545*].

Although there is only little measured evidence for increased primary production or phytoplankton biomass, several trends observed in hydrochemical variables during 1970-93 can be interpreted as an indication of increasing primary or bacterial production. The decrease of phosphate concentrations in the Bothnian Bay points in that direction, as does the increase in total phosphorus in large parts of the Gulf. Decreases in silicate and ammonium concentrations could also be caused by changes in the plankton production in the Gulf, although direct evidence is lacking. The decrease in concentrations is particularly hard to explain by changes in the load sit uation, because of the higher run-off in th later part of the period. However, reduction o coastal point sources during the 1970s an 1980s have contributed to the decrease o coastal phosphorus levels [*545*].

The increase in nitrogen concentrations coul be attributed to increased riverine load, as bot concentration and flow were higher during th 1980s compared to the 1970s [*545,558,635* Increase in nitrogen concentrations in th Baltic Proper [*589*], and increasing concentra tions in the atmospheric deposition [*183* might also have contributed. Internal cyclin of nutrient could be changing as well, leadin to trends in the nutrient concentrations. Th origin of the changes in observed concentra tions can presently not be clearly determined The complex interaction between differen loads and processes within basins and la effects occurring between the different factor contribute to the difficulty of defining singl key factors.

The limited data restrain the analysis of th hydrochemistry in the open Gulf. A muc denser sampling programme is needed for proper spatial analysis. Analysis of season variation and the ability to consider climati variation from year to year is also hampere by the low sampling frequency at the open-se stations. With a better harmonisation of th sampling programme, it should be possible t improve the data material for both types o analysis without incurring unreasonabl Within national programmes, several intensiv stations are used in coastal waters, and th increases the ability to assess the state of thes regions.

4.1.3.2 Nutrient load

The major part of variations in the riverin load of nutrient inputs is due to variations i run-off. The average loads of nitrogen over th whole period 1970-93 were 44,000 t yr^{-1} to th Bothnian Bay and 48,000 t yr^{-1} to the Bothni Sea. Corresponding data for total phosphoru are 2,800 and 2,300 t yr^{-1}, respectively. Figu 4.1.6 presents yearly loads of total nitrog and total phosphorus for the Bothnian Sea an Bothnian Bay [*634*].

Trend tests on the riverine load data suggest significant increase in nitrogen loads, whi the increase in phosphorus is not significan The tendency towards increasing loads of mo chemical variables during the investigat period was primarily a result of the higher ru off during the 1980s compared with that in th 1970s. The use of fertilisers may have co tributed to this trend on the Finnish side [*54* Regarding the Archipelago Sea, the input o nitrogen was also estimated to have increase

while the phosphorus load did not show a clear change.

The present analysis is limited to the riverine load and does not include the atmospheric load, point sources at the coast and imports from adjacent basins. Available literature suggests an increase in atmospheric deposition of nitrogen [*183*]. The transport between basins might also increase the load on the Gulf as the concentration of both nitrogen and phosphorus increased in the Baltic Proper [*589*].

Estimates on the origin of nutrients include uncertainties. It was suggested [*545*] that roughly 50 % of the total phosphorus input and 40 % of the total nitrogen input entering the Gulf of Bothnia from Finland has an anthropogenic origin. Agriculture, identified as the largest single source, was considered responsible for 24 % of the total nitrogen and for 28 % of the total phosphorus input to the Gulf of Bothnia from the Finnish drainage area. The contribution from the agriculture is almost 50 % of the bioavailable nitrogen input [*545*].

4.1.3.3 Spatial nutrient distribution, 1989-93

Spatial representations of the average surface (0-10 m) nutrient concentrations are given in contour plots (Figs. 4.1.7, 4.1.8). These graphs are based on data from stations with observations during 1989-93. The present analysis focuses on the second (April-June) and fourth quarters (October-December), defined as 'spring' and 'autumn', respectively. A division of seasons, based purely on climatic variation from year to year, cannot be justified because of problems with the scheduling of expeditions. The two quarters chosen represent one season with a high primary production and one season dominated by remineralization. However, four near-coast intensive stations provided some complementary information for a whole seasonal cycle (Figs. 4.1.9, 4.1.10).

Differences between basins - As previously pointed out [*729*], Figures 4.1.7 and 4.1.8 show that the major differences in the Gulf are found between the Bothnian Sea and the Bothnian Bay. Variations within these sub-basins are usually smaller. The different nitrogen fractions, as well as silicate, are higher in the Bothnian Bay, possibly due to the higher loading of fresh water per volume. Only small spatial variations in the concentration of total nitrogen can be observed during the two seasons. The main part of this variation can be attributed to the inorganic fractions.

Compared to nitrogen, phosphate and total phosphorus show the opposite pattern during autumn, with higher concentrations in the Bothnian Sea. Very low phosphate concentrations are found throughout the Gulf during spring, without any notable spatial pattern. Taken together, the general pattern for nitrogen and phosphorus results in a rapid change in concentrations in the Northern Quark area, separating the two main sub-basins. The origin of the low phosphate is not yet clearly determined, but precipitation with iron has been suggested as one possible mechanism [*682*].

The oxygen content of water is supersaturated during spring in most parts of the Gulf, but the highest values, around 115 % saturation, were found in the central part of the Bothnian Sea. Autumn-oxygen-saturation values, on the other hand, are <100 % (94-97 %) and exhibit only small variations throughout the Gulf.

Compared to the individual nutrients, the pattern for the inorganic nitrogen-to-phosphorus ratios (DIN:NIP) becomes more pronounced, while the silicate-to-inorganic-nitrogen ratio (DSi:DIN) shows uniform pattern. According to basic ecological theories, the nutrient ratios have a potential to indicate the growth-limiting nutrient. In the Bothnian Sea, the DIN:DIP values are around the *Redfield* uptake ratio of 16:1 indicating balance between the two nutrients, while phosphorus is the limiting nutrient in the Bothnian Bay. Throughout the Gulf no instances of DSi:DIN ratios of ≤1:1, the uptake ratio for diatoms, can be observed. Silicate-limiting conditions are therefore unlikely.

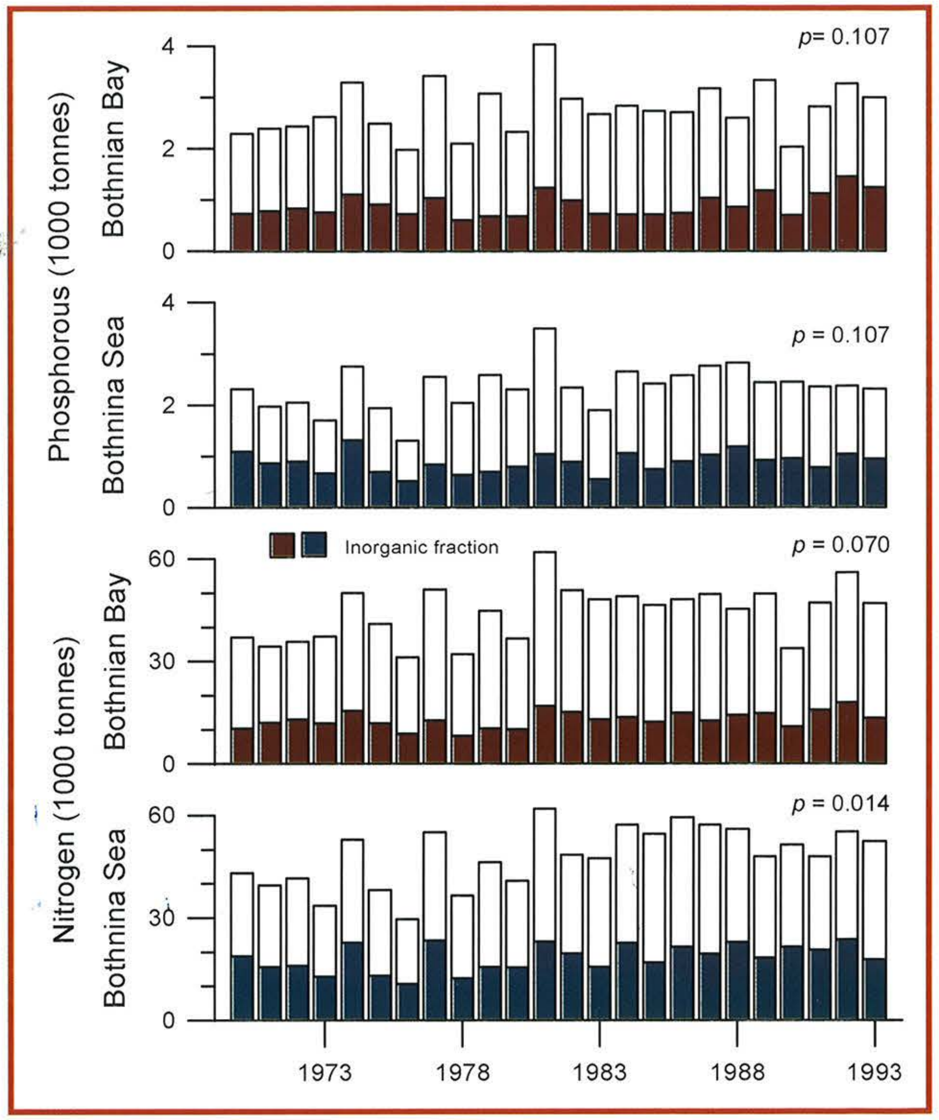
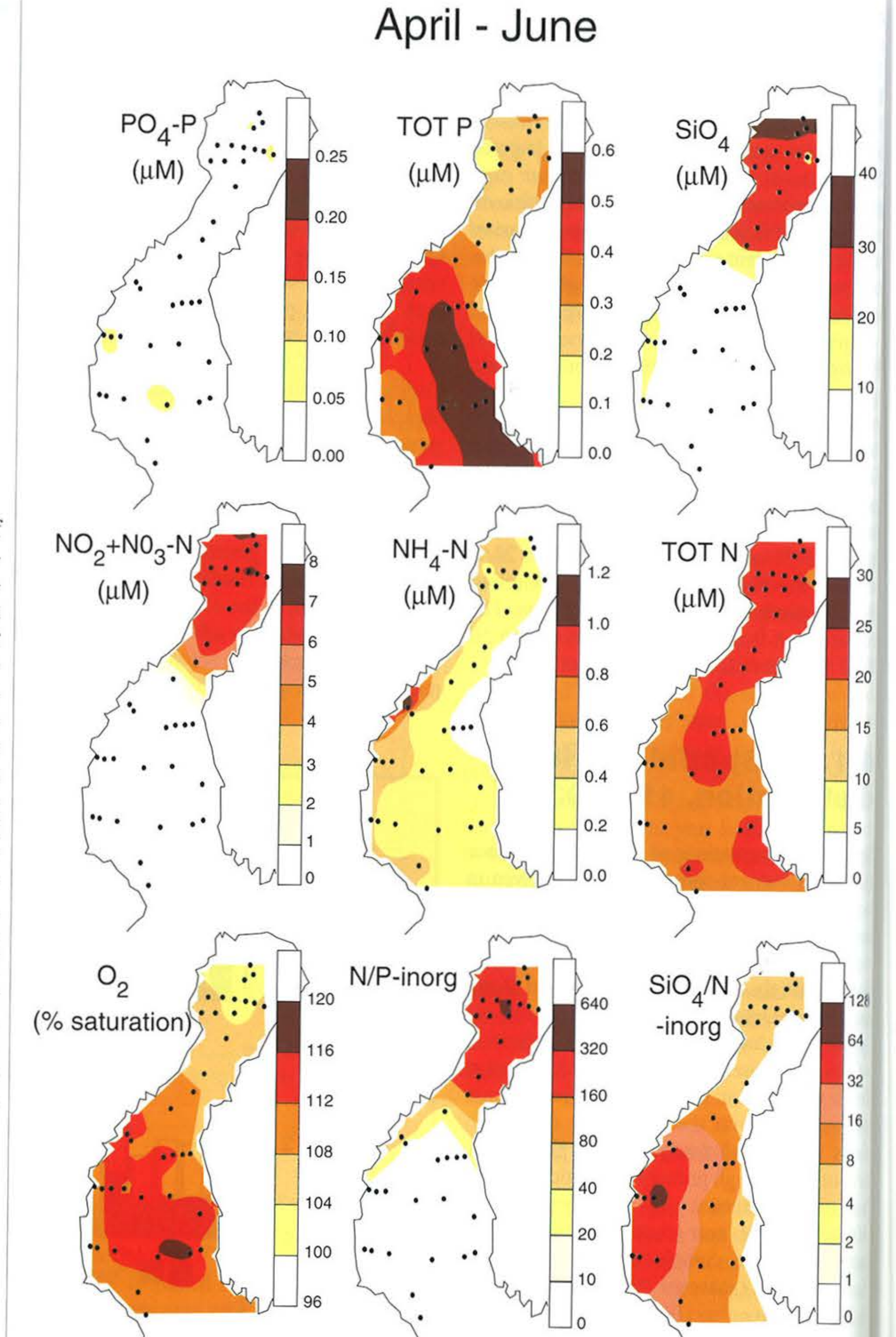


Fig. 4.1.6 Annual riverine loads of total phosphorus and total nitrogen to the Bothnian Bay and Bothnian Sea, 1970-93 [634]
(Results of a non-parametric trend test on the data are presented as p-values in the graph.)

Fig. 4.1.7 Median concentrations of nutrients , nutrient ratios and oxygen saturation in the 0-10 m layer, spring 1989-93

(Values are given in mmol m^{-3} (=µM). The nutrient ratios are presented in a logarithmic scale. Stations are indicated with dots. Depending on the variables, the number of stations differed between 30 and 37. First, the median concentrations were calculated for each sampling occasion. These values were then used to calculate seasonal medians for each station and year. The median concentration over the years was then calculated for each season. Based on these data, a *kriging interpolation* with a linear variogram was made, using the *Surfer* programme (*Golden Software®*). As no variogram analysis or estimate of the best variogram model was done, the graphs should only be viewed as a visual presentation of the data rather than a spatial analysis.)



Previous studies on the spatial distribution of nutrients within the entire Baltic Sea gave similar results [*591,729*]. Using data from 1980-89, statistical testing on differences between the different basins was made. The earlier findings correspond to the general pattern presented here (Figs. 4.1.7, 4.1.8). The differences between the sub-basins are most probably an effect of the higher loading of fresh water, rich in allochthonous organic carbon, to the Bothnian Bay. Here, the annual phytoplankton growth is pronouncedly hampered due to nutrient and/or light deficiency [*7,20,373*], while the bacterial growth is sustained, compared to the Bothnian Sea, through the supply of allochthonous organic carbon [*373*]. The relative successful competition of bacterioplankton in this basin is due to the riverine loading of organic carbon. Therefore, the different, effective uptake kinetics shown by bacterioplankton may therefore contribute to the observed hydrochemical situation in the Bothnian Bay. The food web, dominated by small effectively suspended plankton organisms and sparsely occurring diatoms, also results in an extremely low sedimentation rate of carbon, nitrogen and phosphorus in the open Bothnian Bay [*705*]. The low vertical stability further promotes an efficient supply of oxygen also to the deep water. Combined with the low sedimentation rate, this fosters a well-oxygenated bottom environment.

Nutrient variability within the basins - The results of the present study agree with a previous detailed analysis of transects in the Bothnian Sea [*591*]. The concentrations of several elements are lower in the central basin as compared with those at stations closer to the coast (Figs. 4.1.7, 4.1.8). Furthermore, the concentration of inorganic phosphorus seems to be higher on the eastern side of the basin, whereas ammonium and silicate tend to be higher on the western side. During spring, total phosphorus deviates from this pattern, showing the highest concentrations in the central part of the Bothnian Sea.

Several processes contribute to produce these patterns. The general circulation in the Gulf is one important factor, producing a northward transport along the Finnish coast and a southward transport along the Swedish coast. This brings water with higher phosphorus concentrations from the Baltic Proper northwards, and waters with higher silicate and inorganic nitrogen concentrations move southwards. The higher concentrations closer to the coast can also be a result of the riverine input and direct emissions from communities and industries located along the coast. Nutrient-rich water from the deeper layers could also be an important source, as upward transport mechanisms are more efficient in shallower areas.

Nutrient variability in coastal waters - The Finnish coastal waters are exposed to pronounced riverine discharge. Therefore, they generally show 2-4 times higher concentrations of total nitrogen and phosphorus compared to waters from corresponding offshore areas (Figs. 4.1.9, 4.1.10; [*328,547*]). The width of the zone, affected by the coastal effect, varies from several tens of kilometers in the Archipelago Sea, Northern Quark and northeastern Bothnian Bay to practically zero at open and less contaminated parts of the coast.

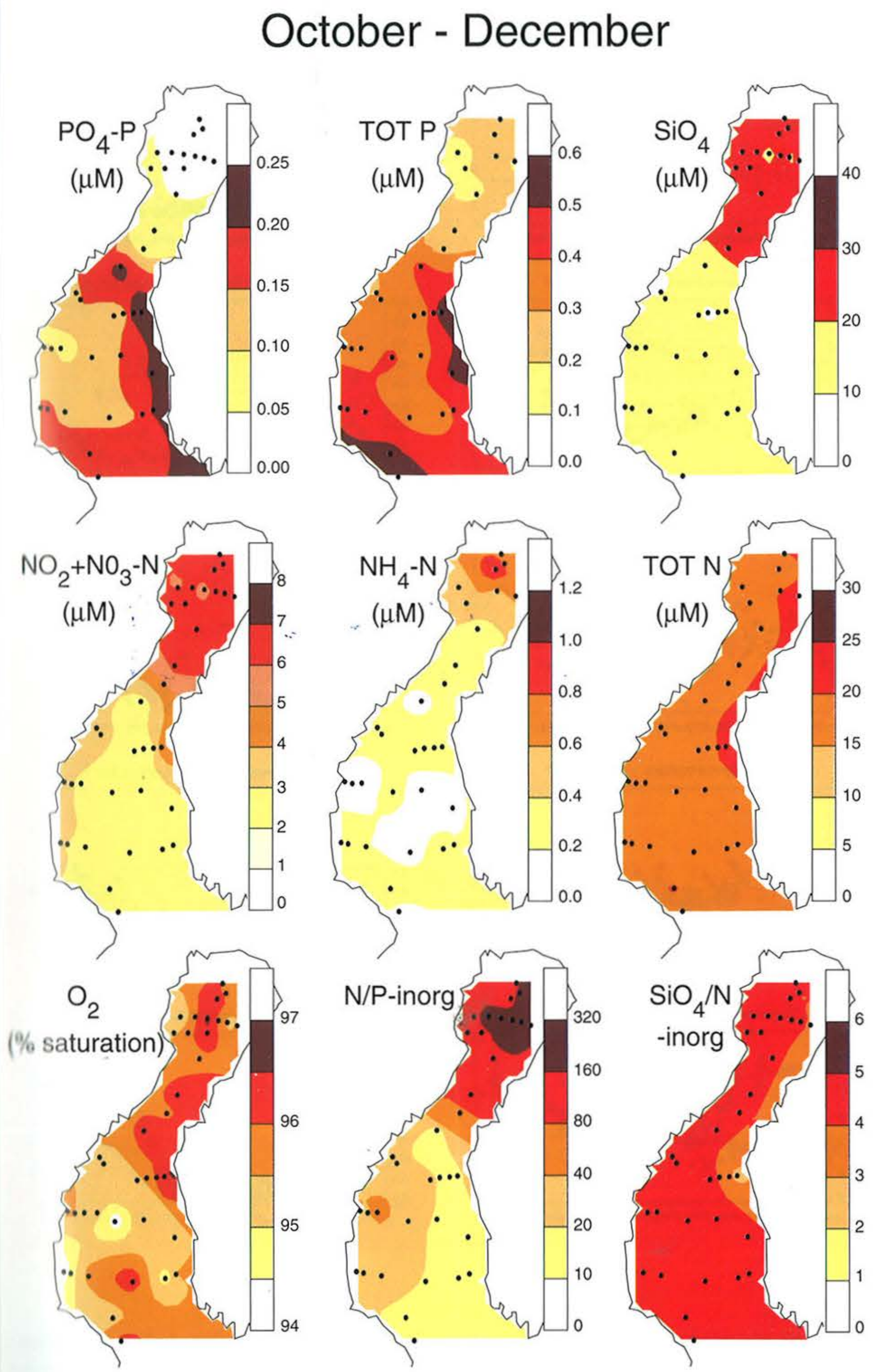


Fig. 4.1.8 Median concentrations of nutrients , nutrient ratios and oxygen saturation in the 0-10 m layer, autumn 1989-93 (cf. Fig. 4.1.7 for further details)

The area representing the most extensive coastal effect is the southernmost part of the Gulf, the Archipelago Sea, where during winter the average concentration of total nitrogen varies between 25 and 50 mmol m^{-3} and total phosphorus between 0.8 and 1.5 mmol m^{-3} (Fig. 4.1.9; [*301*]). The inner archipelago is characterised by loading from intensive agriculture and municipalities. In the outer archipelago, load from fish farming has been assessed as substantial and the most prominent external source of nutrients in summer [*301*]. The other Finnish coastal waters showing increased nutrient concentrations include areas such as those off Uusikaupunki, the Kokemäenjoki Estuary with adjacent waters, the Northern Quark Archipelago, the waters off Kokkola-Pietarsaari and the whole north-eastern Bothnian Bay. Municipal and industrial waste waters, agriculture and intensive forestry are the most likely primary sources for the nutrient increase [*317,328,545*], which has been reflected by an increase of the primary productivity and of chlorophyll *a* concentrations.

4.1.3.4 Seasonal variabilities in coastal waters

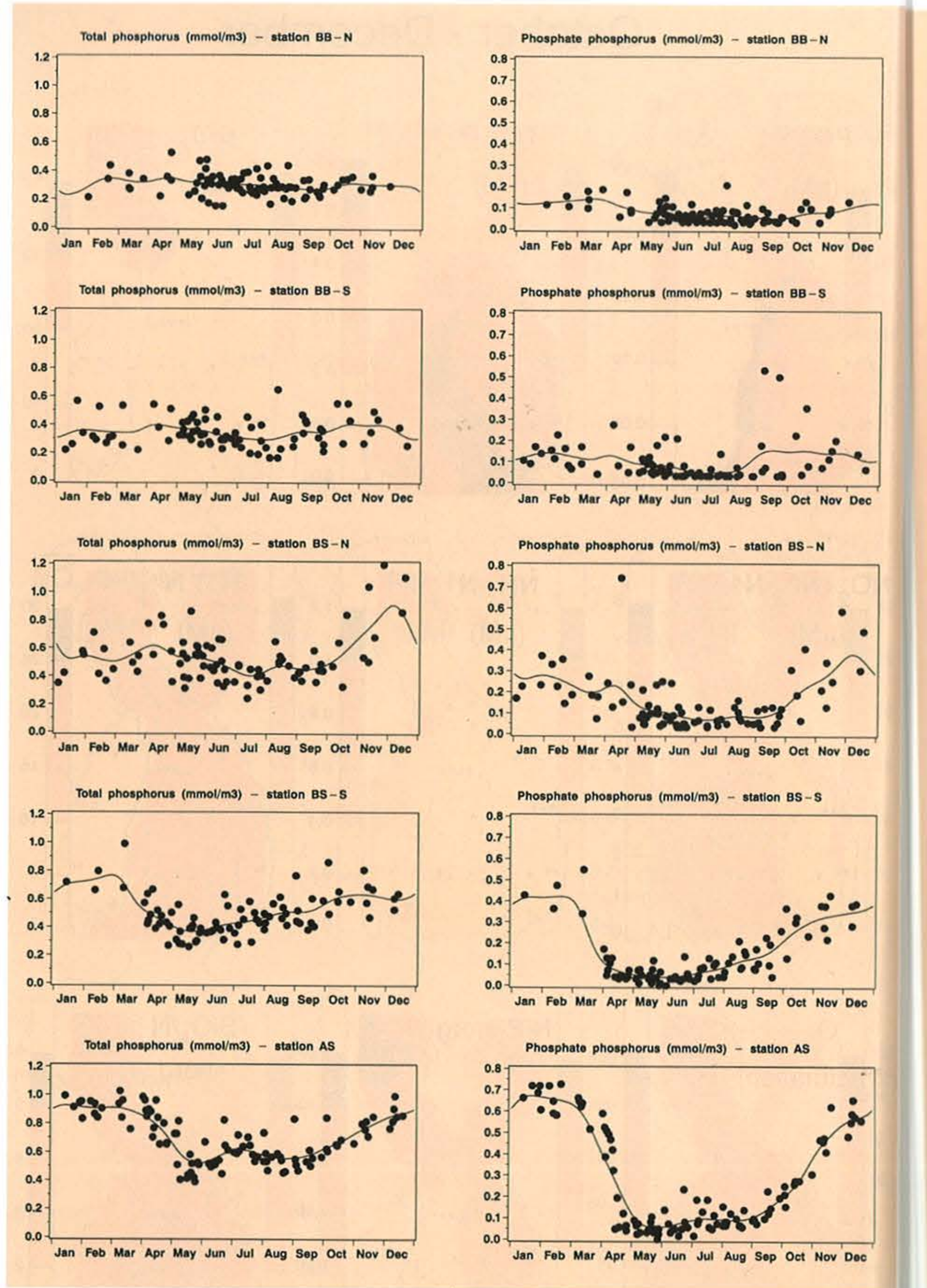
The description of the seasonal variation of nutrients at Finnish coastal stations is based on available literature and data from the Finnish intensive stations collected by the Regional Environment Centers and handled by the Finnish Environment Institute (Figs. 4.1.9, 4.1.10).

In the offshore regions of the Gulf of Bothnia, including most of the Finnish coastal zone, except the Archipelago Sea, the relatively low winter values for inorganic nitrogen and phosphorus suggest a low potential phytoplankton productivity [*7,317,394,728*]. In the open Finnish coastal waters of the southern Bothnian Sea, the winter values are somewhat higher (Figs. 4.1.9, 4.1.10). Only in the southernmost part of the Gulf, the Archipelago Sea, do higher winter values occur, which are similar to those in the eutrophic eastern Gulf of Finland, i.e., 20 mmol m^{-3} for nitrogen and 1.0 mmol m^{-3} for phosphorus [*544*].

According to the observed concentration patterns, phosphorus limits the primary production in summer both in the open and in the coastal Bothnian Bay. In early summer, the coastal water phosphate concentrations are occasionally close to the detection limit, while DIN seldom sinks below 2.5 mmol m^{-3} in this basin (Fig. 4.1.10; [*7,317*]). As the inorganic N/P ratios approach 100, this indicates that phosphorus predominates as the limiting nutrient. In the coastal waters of the Bothnian Sea, the lower summer concentrations of inorganic nitrogen, and consequently lower DIN:DIP ratios between 10 and 20, indicate a relatively balanced supply of nitrogen and phosphorus during the growing season (Figs. 4.1.9, 4.1.10). Caution should, however, be applied when interpreting hydrochemical data exclusively in terms of limiting nutrient, because in a regenerating summer food web the relative supply rate of the nutrients is the most important parameter.

The conditions in the Archipelago Sea clearly differ from those of the remaining part of the Gulf. Here both the general level and the annual variation of nitrogen and phosphorus con-

Fig. 4.1.9 Annual variation of the mean total and inorganic phosphorus concentrations –in mmol m⁻³– in the 0-10 m layer at five Finnish intensive stations, 1989-93



centrations, and the level of primary production, are much higher than in the open waters, i.e., up to 1.0 mmol m^{-3} P and 40 mmol m^{-3} N (Fig. 4.1.10; [*301*]). This is due to the average winter inflow of nutrient-rich surface water from the Northern Baltic Proper and from the Gulf of Finland. It is also due to land-based nutrient inputs as well as to the restricted vertical mixing caused by the complex geomorphology of this area.

Rivers contribute nutrients to coastal waters especially in April/May during the spring flood. This increases the nutrient concentrations in estuaries and in the adjacent waters, especially in the northeastern Bothnian Bay and in the inner Archipelago Sea. High riverine loading appears to increase the inorganic N/P ratio in coastal waters, compared with open waters in the Bothnian Sea and Archipelago Sea, promoting phosphorus as a limiting nutrient off the river mouths and in the enclosed basins of the inner archipelagos [*545*].

The largest seasonal variations in phosphate and inorganic nitrogen were found at a frequently sampled Swedish coastal station (*NB1*, 63°30'N, 19°48'E) (Fig. 4.1.11). Both variables approach zero during the summer. Other variables exhibited a less pronounced seasonal variation (Fig. 4.1.11).

The large variation in inorganic nitrogen and phosphorus is an obvious effect of the uptake by the osmotrophic phyto- and bacterioplankton. The increase in temperature and water column stability in spring leads to a pronounced increase in planktonic production. The production during summer is limited by the mineralisation rate of inorganic nutrients by micro- and macrozooplankton. The low seasonal variation in silicate indicates that this nutrient is available in amounts that clearly exceed the organisms' requirements. Seasonal variations in total nitrogen and phosphorus are largely a result of the variation in the inorganic fraction. The observation that total phosphorus does not decrease as much as expected from the decrease in phosphate concentration might be a result of the increase in dissolved organic phosphorus during summer due to riverine inflow to the estuary (Fig. 4.1.11).

Inorganic nitrogen-to-phosphorus ratios showed a weak seasonal variation, indicating balance between these nutrients at *NB1*. A much more pronounced variation was found for the ratio between dissolved silica and inorganic nitrogen.

4.1.3.5 Trends in the open sea

The data does not allow an evaluation of trends during 1989-93, since a period of at least ten years is needed to get reliable results from trend tests. This is due to the test statistics and the ability to separate short-term variations from changes in the system. The trend analysis was therefore made on two different periods, one covering the whole period of reliable data (1970-93) and the other covering the three HELCOM assessment periods from 1979 to 1993. Two different depths were considered, i.e., the trophogenic zone, here defined as the first 10 m, and the bottom sampling depth. Statistical results are presented in Tables 4.1.1 and 4.1.2. No differences in trends were discernable among seasons.

The development of total phosphorus, nitrate and silicate concentrations is shown for the surface water at station *C1* in Figure 4.1.12. Aggregated with data from two other stations, the trend analysis for the Bothnian Sea resulted in significant trends for all three variables at the 1 % level. Figure 4.1.12 illustrates the high variability of the data used in the analysis.

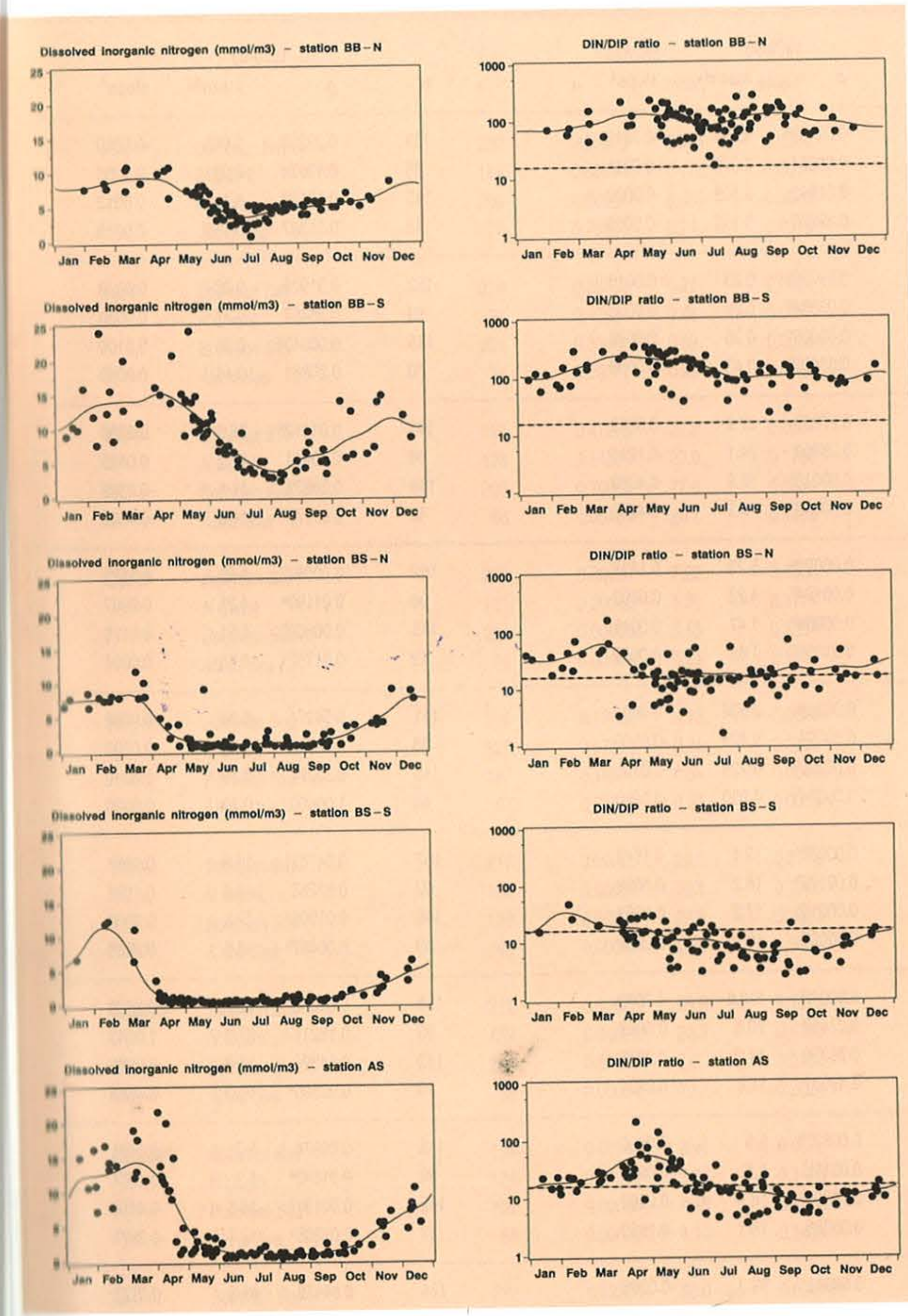


Fig. 4.1.10 Annual variation of the mean total dissolved inorganic nitrogen concentrations –in mmol m⁻³– and of the ratio between inorganic nitrogen and phosphorus in the 0-10 m layer at five Finnish intensive stations, 1989-93

Phosphorus - In general, trends of the order of 1-2 % yr^{-1} can be detected for phosphorus by applied statistics. Phosphate showed three significant trends, all of them with decreasing concentrations in the Bothnian Bay (Tables 4.1.1, 4.1.2). The results indicate that a change of 2 % yr^{-1} is needed for a significant trend over the whole period covered by reliable data, or 3 % yr^{-1} for the shorter period 1979-93. Significantly increasing concentrations of total phosphorus are seen during 1970-93 in the surface waters of all sub-basins but the Bothnian Bay.

The riverine load of nutrients to the Gulf has not changed systematically (phosphorus), or was nearly constant and even slightly increasing (nitrogen) (Fig. 4.1.6), and can therefore not explain the decrease of the concentrations in the Bothnian Bay. Both increasing phyto- or bacterioplankton production or increasing precipitation of metal (primarily iron) phosphates could also explain the observed decreasing trends. However, an increase of the potential primary production could not be found, and bacterioplankton and sedimentation series in offshore areas are too short to allow trend analyses. Since point sources at the coast contribute less than 10 % of the phosphorus load to this part of the Gulf [*225*], this makes changes in this source an unlikely cause of the decline in nutrient levels. The increase of total phosphorus in the surface waters could be due to decreased vertical stability caused by a decreasing salinity in the deep water. An increase of the riverine load in total phosphorus could also partly contribute to the observed changes.

Silicate - Silicate concentrations in the Bothnian Sea and Åland Sea were observed to decrease by about 3 % yr^{-1} in surface waters. At the Åland Sea station, decreasing concentrations were also found in the deep waters. The Bothnian Bay exhibited increasing concentrations during 1979-93, but there was no significant trend for 1970-93. Trends of the order of 1-2 % yr^{-1} can be detected. The river-

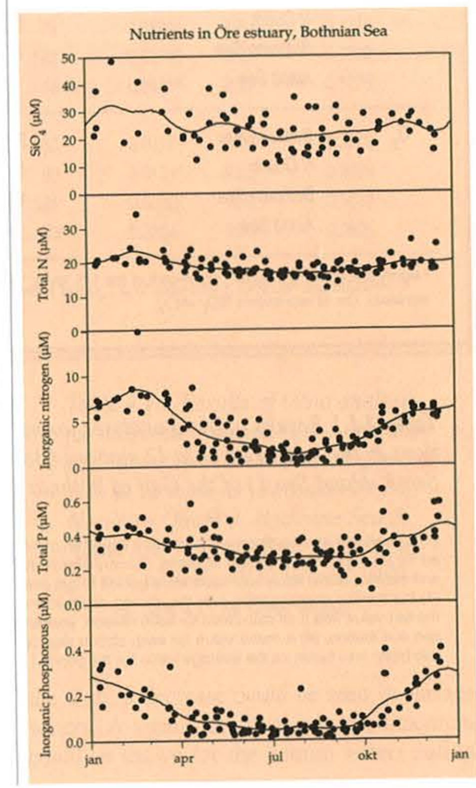


Fig. 4.1.11 Seasonal variation of nutrients –in mmol m⁻³– in the 0-10 m layer at station NB1 , Öre Estuary, Bothnian Sea, 1989-93

(Silicate data were available only from 1991 on. The smoothed curve was calculated by a *kernel smoother* , with the aid of a computer programme written by *Carsten T. Agger*.)

		1970-93 n	p	mean[d]	slope[d]	1979-93 n	p	mean[d]	slope[d]
PO_4-P	*Bothnian Bay*	230	0.02107[a]	0.054	-0.0013	169	0.23269	0.048	-0.0010
	N Quark	145	0.08074	0.079	-0.0011	95	0.43974	0.081	0.0000
	Bothnian Sea	204	0.51645	0.108	0.0000	145	0.18162	0.112	0.0013
	Åland Sea	70	0.59238	0.155	-0.0005	48	0.31387	0.159	-0.0013
Tot-P	*Bothnian Bay*	211	0.59735	0.23	0.0010	152	0.31926	0.23	0.0028
	N Quark	138	0.03884[a]	0.26	0.0035	89	0.38467	0.27	0.0020
	Bothnian Sea	205	0.00009[c]	0.35	0.0079	145	0.00313[b]	0.36	0.0100
	Åland Sea	75	0.00492[b]	0.43	0.0056	50	0.20861	0.46	0.0050
SiO_4	*Bothnian Bay*	204	0.27863	27.2	0.0929	148	0.01412[a]	26.9	0.3350
	N Quark	133	0.28860	24.1	-0.1215	88	0.70491	22.2	0.0625
	Bothnian Sea	201	0.00012[c]	12.9	-0.4063	146	0.08678	11.4	-0.2389
	Åland Sea	68	0.00006[c]	15.2	-0.4335	48	0.01173[a]	14.6	-0.4000
Ox.-N	*Bothnian Bay*	220	0.00003[c]	5.23	0.1215	162	0.00097[c]	5.59	0.1575
	N Quark	144	0.00124[b]	4.22	0.0700	96	0.01190[a]	4.21	0.0967
	Bothnian Sea	201	0.00888[b]	1.47	0.0048	145	0.00603[b]	1.51	0.0118
	Åland Sea	70	0.01050[a]	1.85	0.0052	49	0.01752[a]	1.88	0.0091
NH_4-N	*Bothnian Bay*	198	0.08346	0.388	-0.0094	141	0.54974	0.30	0.0008
	N Quark	130	0.09695	0.425	-0.0100	88	0.89612	0.36	0.0000
	Bothnian Sea	199	0.00226[b]	0.228	-0.0125	146	0.32227	0.19	-0.0010
	Åland Sea	69	0.00334[b]	0.230	-0.0200	48	1.00000	0.19	0.0000
Tot-N	*Bothnian Bay*	215	0.00082[c]	19.1	0.1542	157	0.04787[a]	19.6	0.1167
	N Quark	138	0.00109[b]	18.2	0.1688	92	0.07263	19.2	0.1192
	Bothnian Sea	203	0.00019[c]	17.2	0.2553	146	0.01096[a]	17.8	0.2000
	Åland Sea	73	0.00098c	18.2	0.2268	49	0.00487[b]	18.6	0.2625
DIN:DIP	*Bothnian Bay*	211	0.00049[c]	112.8	4.7570	154	0.02382[a]	123.5	5.2579
	N Quark 1	41	0.07450	70.8	0.8902	93	0.18211	67.7	1.0043
	Bothnian Sea	198	0.76756	14.0	-0.0375	142	0.64399	13.0	0.0563
	Åland Sea	69	0.49037	11.6	0.0824	48	0.02590[a]	10.6	0.4665
DSi:DIN	*Bothnian Bay*	200	0.00377[b]	5.9	-0.0490	145	0.09974	5.7	-0.0431
	N Quark	133	0.00111[b]	5.8	-0.0999	88	0.01648[a]	6.2	-0.1004
	Bothnian Sea	197	0.00309[b]	18.4	-0.2709	142	0.00131[b]	18.6	-0.4158
	Åland Sea	67	0.05086	18.1	-0.1953	47	0.03266[a]	17.7	-0.2971
O_2	*Bothnian Bay*	240	0.68911	99.3	-0.0255	174	0.84452	98.5	0.0128
	N Quark	145	0.34808	100.7	-0.0551	95	0.35606	101.1	-0.0876
	Bothnian Sea	199	0.31535	101.9	-0.0789	135	0.36410	101.4	-0.1426
	Åland Sea	84	0.24997	101.8	-0.0726	50	0.21171	100.9	-0.1570

[a] significant at the 5 % level; [b] significant at the 1 % level; [c] significant at the 0.1 % level; [d] units in mmol m^{-3} (=μM) for all parameters but O_2 (% of saturation), DIN:DIP and DSi:DIN (dimensionless); Ox.-N represents NO_2^-+NO_3^-

***Table 4.1.1** Results of trend analyses for nutrients, nutrient ratios and oxygen saturations in the 0-10 m layer at 12 stations (Bothnian Bay 5, Northern Quark 3, Bothnian Sea 3, Åland Sea 1) of the Gulf of Bothnia for two periods, 1970-93 and 1979-93*

(The trend test and the slope estimate were based on common methods [*251,252*]. The seasonal variation was accounted for by using four separate seasons, January-March, April-June, July-September and October-December. Seasons and stations within each sub-basin were pooled to get one estimate. The average values were calculated in five steps: (1) the median concentration over the chosen depth interval was calculated individually for all sample occasions, (2) a median value was then calculated for each season, year and station, (3) a median was taken over all years for each season and station, (4) a mean value for each station was calculated, and finally (5) the median over all stations in each sub-basin was taken as the average value for the period.)

ine load of silica increased during the period and the decreasing concentrations in the Gul[f] must therefore be related to other factors. Th[e] most probable cause is an increase in primar[y] production. Changes in the internal silic[a] cycling and in the exchange of silicate with th[e] Baltic Proper can, however, not be excluded.

Nitrogen - Total nitrogen concentration[s] increased throughout the Gulf, both at the sur[-]face and close to the bottom. This was eviden[t] for both tested periods, with only single non[-]significant results for the shorter period in th[e]

		1970-93 n	p	mean[d]	slope[d]	1979-93 n	p	mean[d]	slope[d]
PO_4-P	*Bothnian Bay*	230	0.00609[b]	0.095	-0.0025	169	0.0338[a]	0.110	-0.0033
	N Quark	145	0.50814	0.170	-0.0011	95	0.2164	0.190	-0.0043
	Bothnian Sea	206	0.06954	0.650	0.0076	146	0.4290	0.780	-0.0033
	Åland Sea	70	0.75917	0.740	0.0018	48	0.6226	0.925	-0.0046
Tot-P	*Bothnian Bay*	210	0.28755	0.21	-0.0021	151	0.7468	0.25	0.0000
	N Quark	136	0.46621	0.46	0.0027	87	0.5475	0.40	-0.0038
	Bothnian Sea	203	0.37583	0.83	0.0046	143	0.2752	1.14	-0.0100
	Åland Sea	74	0.13707	0.96	0.0089	49	0.5469	1.07	0.0080
SiO4	*Bothnian Bay*	197	0.47279	27.3	0.0667	142	0.1571	31.7	0.1667
	N Quark	128	0.14260	23.0	-0.1765	83	0.2062	23.7	-0.2278
	Bothnian Sea	200	0.70069	27.8	0.0454	144	0.1615	28.2	-0.3333
	Åland Sea	66	0.00750[b]	20.7	-0.4292	46	0.4861	17.9	-0.1833
Ox.-N	*Bothnian Bay*	219	0.00002[c]	7.22	0.1556	161	0.0058b	8.15	0.1267
	N Quark	141	0.00026[c]	5.30	0.1116	93	0.0433[a]	5.31	0.1029
	Bothnian Sea	203	0.00000c	5.55	0.2061	146	0.0098b	6.10	0.1200
	Åland Sea	69	0.00001[c]	5.21	0.1671	48	0.0236[a]	4.08	0.1010
NH_4-N	*Bothnian Bay*	192	0.19132	0.13	-0.0020	135	0.5901	0.30	0.0000
	N Quark	127	0.67925	0.31	0.0000	86	0.9222	0.56	0.0000
	Bothnian Sea	197	0.19639	0.31	0.0079	144	0.0232[a]	0.28	0.0215
	Åland Sea	67	0.27519	0.21	-0.0019	46	0.8632	0.41	0.0000
Tot-N	*Bothnian Bay*	211	0.00025[c]	20.3	0.2172	155	0.2826	19.9	0.0538
	N Quark	136	0.00059[c]	19.1	0.2300	91	0.1172	19.0	0.1000
	Bothnian Sea	199	0.00000[c]	22.6	0.3857	143	0.0223[a]	22.4	0.2000
	Åland Sea	69	0.00047[c]	22.5	0.3268	46	0.0180[a]	20.2	0.3000
DIN:DIP	*Bothnian Bay*	212	0.00025[c]	95.8	3.1679	154	0.0082[b]	87.3	3.2664
	N Quark	139	0.04020[a]	29.3	0.4399	91	0.0996	39.2	0.6411
	Bothnian Sea	203	0.00004[c]	8.6	0.2163	146	0.0019[b]	9.2	0.2354
	Åland Sea	69	0.00024[c]	7.1	0.2024	48	0.0257[a]	6.6	0.2032
DSi:DIN	*Bothnian Bay*	194	0.00042[c]	3.96	-0.0656	140	0.0467[a]	3.76	-0.0460
	N Quark	124	0.00014[c]	3.70	-0.1426	79	0.0104[a]	3.75	-0.1219
	Bothnian Sea	197	0.00000[c]	4.96	-0.1838	142	0.0019[b]	4.71	-0.1322
	Åland Sea	66	0.00000[c]	4.19	-0.2464	46	0.0131[a]	2.99	-0.1276
O_2	*Bothnian Bay*	232	0.73236	88.0	0.0341	165	0.0111[a]	87.9	0.3879
	N Quark	139	0.17435	93.5	0.1063	89	0.0174[a]	93.5	0.3659
	Bothnian Sea	183	0.81205	70.3	0.0372	121	0.0670	68.4	0.5539
	Åland Sea	77	0.10157	80.9	0.1838	43	0.5204	82.5	0.0905

[a] significant at the 5 % level; [b] significant at the 1 % level; [c] significant at the 0.1 % level; [d] units in mmol m^{-3} (=μM) for all parameters but O_2 (% of saturation), DIN:DIP and DSi:DIN (dimensionless); Ox.-N represents NO_2^-+NO_3^-

Bothnian Bay and in the Northern Quark. The order of the change was about 1 % yr^{-1} for the whole Gulf. A significant part of the increase was due to the increase in nitrate, especially in the deep water (Tables 4.1.1, 4.1.2).

Oxidised forms of nitrogen (nitrate and nitrite) increased throughout the Gulf of Bothnia both in the surface layer and close to the bottom. At the surface, the trend slope was steepest for the northern areas of the Gulf. The Bothnian Bay showed a slope >2 % yr^{-1} for both periods. In the southern areas, the slope value was considerably lower, i.e., only about 0.3 % yr^{-1}. Trends as low as 0.3 % yr^{-1} can be detected given, the characteristics of the present data set. For all sub-basins, the analysis in the surface waters indicated steeper slopes during 1979-93 compared with 1970-93. Bottom waters showed a reverse picture, with a steeper slope during the longer period and in the southern parts of the Gulf.

Trend analyses of the reduced form of nitrogen (ammonium) revealed three significant trends. For the 1970-93 period in the southern parts of the Gulf, a decrease could be seen in surface waters. A significant increase of ammonium could be shown for the bottom waters during

***Table 4.1.2** Results of trend analyses for nutrients, nutrient ratios and oxygen saturations in the bottom samples at 12 stations (Bothnian Bay 5, Northern Quark 3, Bothnian Sea 3, Åland Sea 1) of the Gulf of Bothnia for two periods, 1970-93 and 1979-93* (cf. Table 4.1.1 for further details)

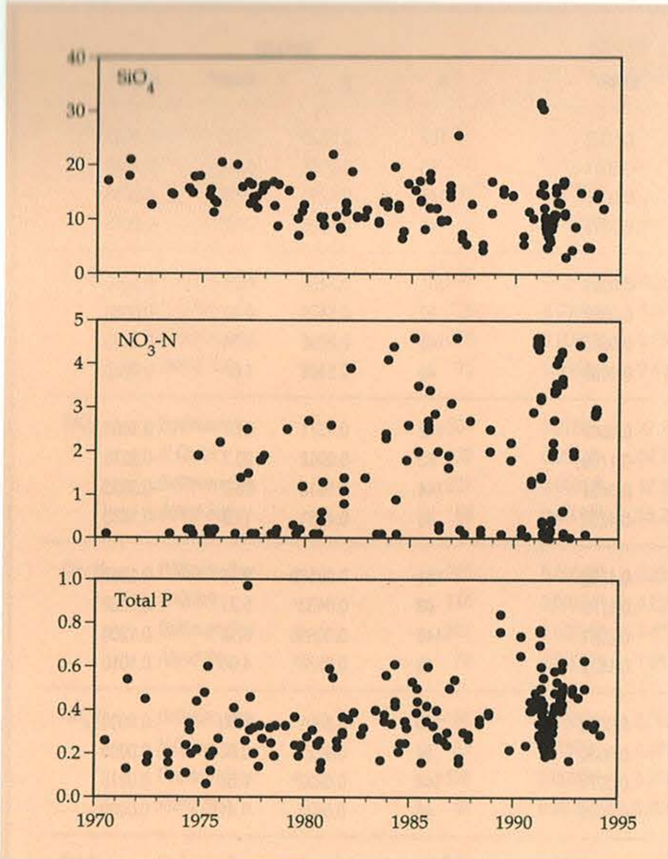


Fig. 4.1.12 Silicate, nitrate and total phosphorus –in mmol m^{-3}– in the 0-10 m layer at station C1 in the Bothnian Sea, 1970-94

the shorter period in the Bothnian Sea. Quite steep slopes, of the order of 5 % yr^{-1}, are needed to get significant trends in ammonium concentrations.

Most of the changes in nitrogen concentration can be explained by increasing loads from the surrounding rivers and from the atmosphere (Fig. 4.1.6; [*545,635*]). Increasing trends for nitrogen in the Baltic Proper and changes in the vertical stability of the water column in the Gulf of Bothnia may, however, also have contributed.

Oxygen saturation - Only limited changes in the oxygen saturation could be found in the time series. Increases in the saturation in the bottom waters of the Bothnian Bay and the Northern Quark were the only significant changes. These trends were <0.5 % yr^{-1} and were probably connected to the lower vertical stability with decreasing salinities.

Nutrient ratios - Analyses of trends in the inorganic nitrogen-to-phosphorus ratio revealed only a few significant trends in the surface water, whereas all but one test showed significant increases in the bottom waters. The latter is primarily due to the increase in nitrogen. For surface water, the observed trends were increases during both periods in the Bothnian Bay, and during the shorter period in the Åland Sea. The silicate-to-inorganic nitrogen ratio, on the other hand, decreased throughout the Gulf. Only two non-significant cases were observed. The change in the Si:N ratios was due to both the increase in nitrogen and, in most sub-areas, also due to the decrease of silica, both in the deep and in the surface water.

Differences between the assessment periods - The data set is too limited to make a thorough statistical evaluation on differences between the different assessment periods, 1979-83, 1984-88 and 1989-93. Given the present sampling frequency, sample variability and statistical methods, one cannot expect to find changes in a system by using trend analysis on such short data series. Apart from the long-term trends presented above, no major changes in the hydrochemistry could be seen between the periods.

4.1.3.6 Trends in the coastal zone

By the application of linear regression analysis, a significant increase in nitrogen and phosphorus for most parts of the Archipelago Sea was found for the period 1970-93 [*301*]. The increasing trends were explained by increased loading from agriculture, fish-farming and via the atmosphere during the 1970s and 1980s. In contrast, phosphorus decreased significantly in the innermost archipelago, possibly due to intensified purification of municipal waste waters in Turku and neighbouring towns.

In the open area between the Archipelago Se and the Åland Islands, the increase of nutrient was mainly due to a similar increase in th Baltic Proper [*589*]. In the Finnish coasta waters affected by the pulp and paper industry e.g., off Oulu in the southeastern Bothnia Bay, the loading of phosphorus had decrease by 100 t yr^{-1} at the beginning of the 1990 [*328*]. This caused lower phosphorus concen trations at some locations in these areas. Tim series of nutrients at the Swedish coast are to few and too short to allow a trend analysis.

4.1.4 Pelagic biology

H. Kuosa[1], *J. Kuparinen, J. Wikner*

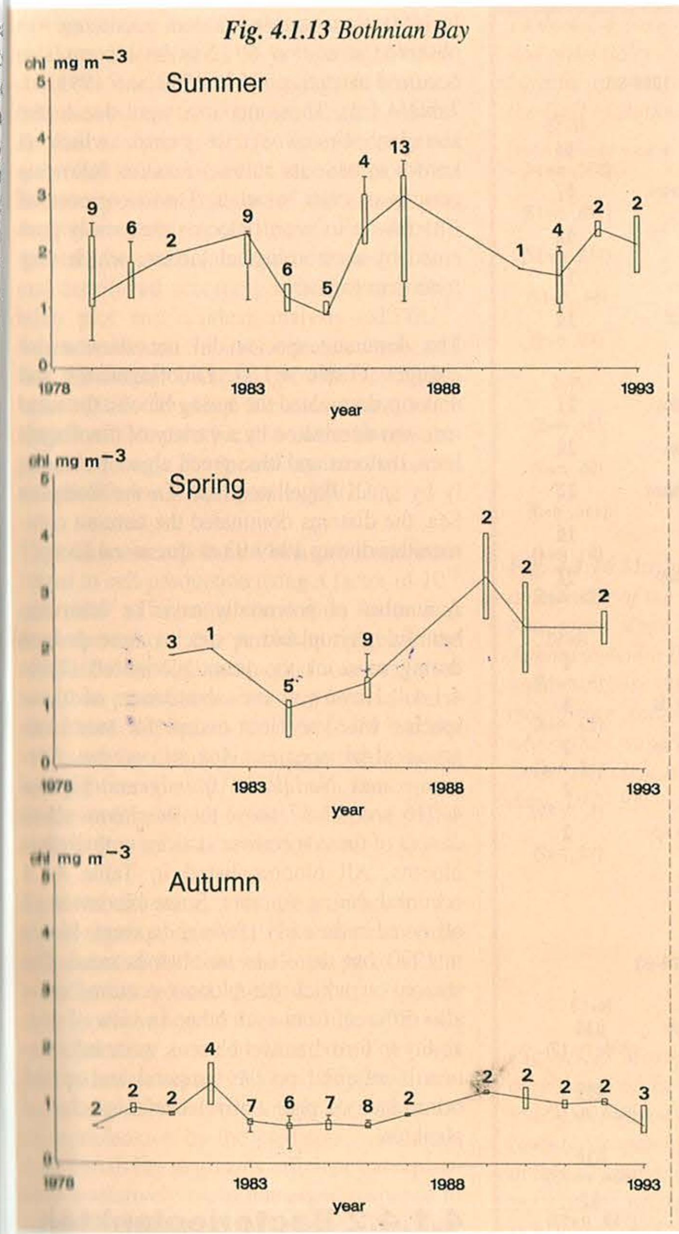
4.1.4.1 Chlorophyll a and phytoplankton

Chlorophyll a - Due to the low sampling fre quency and on the basis of statistical expertis the monitoring data were analysed for two sep arate geographical areas, the Bothnian Sea an Bothnian Bay, rather than for individual sta tions. Accordingly, the data from the monitor ing stations were pooled and analysed using non-parametric statistical test [*251,252*]. Th pooled seasonal chlorophyll *a* monitoring dat are presented in Figures 4.1.13 and 4.1.14.

Trends in chlorophyll *a* were tested statistica ly according to seasons. Using the monitorin data only in one case a significant trend wa found. This was a positive trend on summe values for the Bothnian Sea. The trend corr sponded to a doubling of chlorophyll *a* in 6 years. However, the uncertainty of this pos tive trend is considerable, and the 95 % conf dence levels give a doubling time between and 33 years. Thus, the doubling could occu with some certainty very slowly, i.e., withi ≥10 yrs, compared to the average trend. Th graph indicates that the chlorophyll *a* concen trations were high in the last 5 years, and th gave rise to the observed trend (Fig. 4.1.14 The regional and seasonal differences of th chlorophyll *a* concentrations at five BMP st tions, on a transect between the Norther Bothnian Bay and the Åland Sea, are shown Figure 4.1.15. From north to south, both th concentrations and the differences between th different seasons are observed to increase.

Potential primary production - The potenti primary productivity data had a much highe variability than chlorophyll *a*. Consequentl significant trends could not be demonstrated

Phytoplankton biomass and species compos tion - The algal biomass showed no trend Most of the material consisted of only on



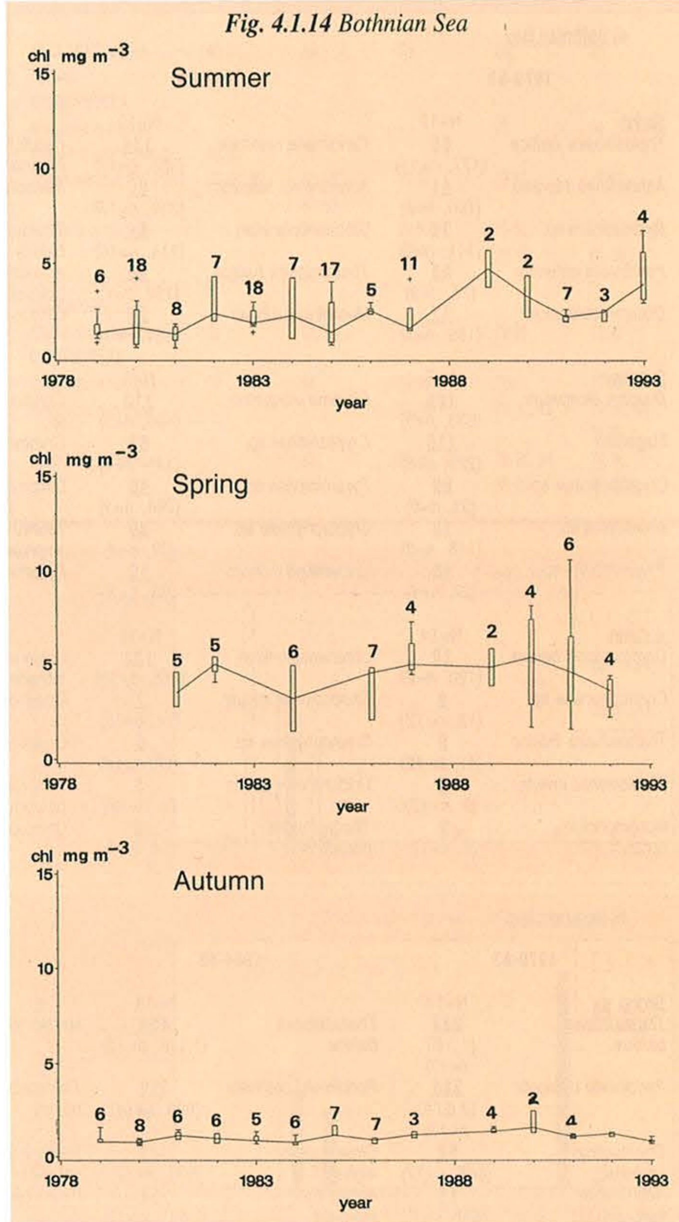
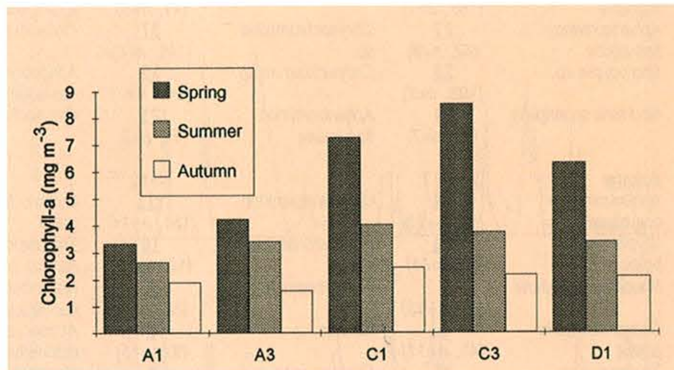


Fig. 4.1.13 and 4.1.14 Box-and-whisker plots of average chlorophyll a in the 0-10 m layer for three seasons, 1979-93

(Seasonal median values are connected with a solid line. The number of samples is indicated on the top of the plot. In case of missing years no number is given.)

Fig. 4.1.15 Seasonal average of chlorophyll a in the 0-10 m layer at five BMP stations on a transect Bothnian Bay - Åland Sea



sampling per season. The dominant species are presented in Table 4.1.3. Most of the data could be divided into three separate periods. The periods 1979-84 and 1989-93 were characterised by a low biomass. This was contrary to 1985-88, which exhibited a high biomass. This was seen at the stations *A1* and *D1* for spring and autumn, and at *A3* and *C1* for autumn. Summer samples were taken only occasionally. Therefore, these data were not suitable for a detailed analysis. The variability of plankton composition in spring was mostly caused by the biomass fluctuations of dinoflagellates and diatoms, which are typical components of common spring communities (Table 4.1.3). The variability in autumn was mostly due to the high variation of unidentified (ultra-)nanoplankton. The reason for the high biomass values during 1985-88 is unknown. A comparison with the chlorophyll *a* values showed that there was no chlorophyll maximum connected to the high phytoplankton biomass. This could indicate that a number of heterotrophic cells have been included in the ultrananoplankton. If defining phytoplankton as photosynthetic cells, the high 'phytoplankton' biomass values found must not necessarily indicate as well maxima in the biomass of photosynthetic cells.

a) Bothnian Bay

	1979-83		1984-88		1989-93
Spring	N=12		N=13		N=15
Thalassiosira baltica	65 (177, *n*=11)	*Peridiniella catenata*	134 (388, *n*=13)	*Peridiniella catenata*	84 (330, *n*=14)
Achnanthes taeniata	41 (160, *n*=8)	*Achnanthes taeniata*	90 (350, *n*=13)	*Mesodinium rubrum*	57 (175, *n*=13)
Gymnodinium sp.	34 (141, *n*=8)	Ultrananoplankton	84 (114, *n*=10)	*Thalassiosira baltica*	45 (112, *n*=12)
Peridiniella catenata	28 (77, *n*=9)	*Thalassiosira baltica*	40 (152, *n*=7)	*Achnanthes taeniata*	17 (54, *n*=12)
Diatoma elongatum	17 (169, *n*=3)	*Mesodinium rubrum*	24 (251, *n*=3)	*Gymnodinium* sp.	16 (69, *n*=9)
Summer	N=8		N=8		N=5
Diatoma elongatum	155 (833, *n*=5)	*Diatoma elongatum*	113 (496, *n*=3)	*Chrysochromulina* sp.	27 (34, *n*=5)
Flagellata	118 (220, *n*=8)	*Cryptomonas* sp.	68 (141, *n*=4)	*Cryptomonas* sp.	26 (95, *n*=5)
Cryptophyceae sp.	89 (93, *n*=8)	*Pyramimonas* sp.	40 (284, *n*=3)	*Diatoma elongatum*	23 (116, *n*=3)
Microcystis sp.	18 (118, *n*=3)	Cryptophyceae sp.	20 (32, *n*=4)	*Teleaulax amphioxeia*	16 (51, *n*=4)
Pyramimonas sp.	18 (29, *n*=4)	Unidentified monads	19 (68, *n*=3)	*Pyramimonas* spp.	12 (25, *n*=5)
Autumn	N=14		N=18		N=10
Cryptomonas baltica	10 (120, *n*=2)	Ultrananoplankton	122 (155, *n*=18)	Unidentified monads	4 (8, *n*=10)
Cryptophyceae sp.	9 (13, *n*=12)	*Rhodomonas minuta*	7 (24, *n*=18)	Cryptophyceae 'B'	3 (23, *n*=4)
Thalassiosira baltica	9 (41, *n*=12)	Cryptophyceae sp.	6 (17, *n*=14)	Cryptophyceae sp.	3 (18, *n*=5)
Rhodomonas minuta	6 (9, *n*=12)	Unidentified monads	5 (6, *n*=18)	*Monoraphidium contortum*	2 (6, *n*=10)
Monoraphidium contortum	2 (6, *n*=12)	*Thalassiosira pseudonana*	4 (14, *n*=8)	Cryptophyceae sp.	2 (12, *n*=2)

b) Bothnian Sea

	1979-83		1984-88		1989-93
Spring	N=17		N=14		N=13
Thalassiosira baltica	238 (1,187, *n*=17)	*Thalassiosira baltica*	454 (1,178, *n*=13)	*Mesodinium rubrum*	638 (936, *n*=13)
Peridiniella catenata	225 (3,071, *n*=17)	*Peridiniella catenata*	352 (609, *n*=14)	*Thalassiosira baltica*	406 (693, *n*=12)
Chaetoceros wighamii	52 (849, *n*=17)	*Chaetoceros wighamii*	189 (931, *n*=11)	*Peridiniella catenata*	316 (834, *n*=13)
Achnanthes taeniata	46 (547, *n*=7)	*Achnanthes taeniata*	158 (1,531, *n*=12)	*Pyramimonas* sp.	43 (113, *n*=13)
Ultrananoplankton	22 (289, *n*=15)	Ultrananoplankton	107 (161, *n*=11)	*Gymnodinium* sp.	27 (103, *n*=10)
Summer	N=9		N=8		N=8
Flagellata	80 (140, *n*=8)	*Protoceratium reticulatum*	43 (271, *n*=7)	*Chrysochromulina* sp.	178 (254, *n*=8)
Chaetoceros wighamii	32 (185, *n*=7)	*Cryptomonas* sp.	38 (47, *n*=6)	*Nodularia spumigena*	161 (1,094, *n*=7)
Aphanizomenon flos-aquae	27 (62, *n*=8)	*Chrysochromulina* sp.	37 (146, *n*=6)	*Cryptomonas* sp.	98 (332, *n*=7)
Microcystis sp.	25 (193, *n*=2)	*Chrysidiastrum* sp.	33 (149, *n*=2)	*Aphanizomenon flos-aquae*	79 (287, *n*=8)
Nodularia spumigena	21 (50, *n*=7)	*Aphanizomenon flos-aquae*	21 (71, *n*=8)	*Gymnodinium* sp.	26 (89, *n*=8)
Autumn	N=17		N=15		N=10
Aphanizomenon flos-aquae	14 (43, *n*=13)	Ultrananoplankton	118 (154, *n*=14)	*Coscinodiscus granii*	143 (643, *n*=3)
Cryptomonas baltica	14 (78, *n*=4)	*Rhodomonas minuta*	10 (18, *n*=15)	*Thalassiosira baltica*	16 (85, *n*=5)
Rhodomonas minuta	9 (13, *n*=13)	*Rhodomonas lens*	9 (56, *n*=3)	*Aphanizomenon flos-aquae*	9 (22, *n*=10)
Gomphosphaeria pusilla	8 (41, *n*=11)	Monads	8 (9, *n*=15)	*Actinocyclus octonarius*	6 (14, *n*=8)
Thalassiosira baltica	6 (22, *n*=12)	*Cryptomonas* sp.	5 (15, *n*=10)	*Mesodinium rubrum*	5 (10, *n*=10)

Table 4.1.3 Dominating phytoplankton species in the Gulf of Bothnia during three assessment periods

(arithmetical mean, in brackets maximum, values of species wet-weight biomass in mg m^{-3}; N - total number of samples, *n* - number of samples in which species were dominating)

In 1990, a pronounced diatom maximum was observed at station *C1*. Similar blooms also occurred at station *C4* in 1992 and 1993 (cf. Table 4.1.3). These maxima were due to the abundant *Coscinodiscus granii*, which is known to have its autumn maxima following periods of calm weather. The occurrence of *Coscinodiscus granii* blooms are mainly governed by meteorological factors, which vary from year to year.

The dominant species did not show major changes (Table 4.1.3). Dinoflagellates and diatoms dominated the spring bloom, the summer was dominated by a variety of dinoflagellates, diatoms and blue-green algae, but mostly by small flagellated taxa. In the Bothnian Sea, the diatoms dominated the autumn communities during 1989-93 as discussed above.

A number of potentially toxic or otherwise harmful phytoplankton species were present during most of the years monitored (Table 4.1.4). However, the abundance of these species was very low except for two blue-green algal species, *Aphanizomenon flos-aquae* and *Nodularia spumigena*. Figures 4.1.16 and 4.1.17 show the maximum abundances of these species at stations with distinct blooms. All blooms listed in Table 4.1.4 occurred during summer. Some blooms were observed in the early 1980s and a single bloom in 1990, but there was no obvious trend. The stations at which the blooms occurred were also different from each other. In view of their ability to form harmful blooms, more information is required on the temporal and spatial occurrance of potentially harmful species of plankton.

4.1.4.2 Bacterioplankton

The bacterioplankton community is expected to respond to eutrophication by increasing production and biomass [*120*]. Changes in climate variables such as temperature, UV-light and river discharge could also give rise to changes in bacterial values [*248,551*]. At the community level, the potential effects of environmental toxins on the bacterial activity are less clear.

Bacterial growth rates - No trends were found for the investigation period, 1984-94, on either seasonal or annual values. According to a *power* analysis of the annual data, the probability of detecting a true trend of 12 % yr^{-1} was determined to be 0.80 on a data set of 10 years. If there was any systematic change, it was therefore probably <12 % yr^{-1}. The average annual bacterioplankton-biomass production in the 0-20 m water layer during the 10-year period was 1.41 mol C m^{-2} yr^{-1}. This was 63 % of the average annual growth recorded for the Kiel Bight (cf. Chapter 4.4).

An increasing trend of the annual bacterioplankton growth of 21 % yr^{-1} during 1987-92 could be demonstrated on aggregated seasonal data according to a non-parametric *Mann-Kendall* analysis (*p*=0.018, independence not assumed). A similar result was obtained by a linear regression analysis of the annual production estimates (*p*=0.015, cf. Fig. 4.1.18). Annual integrated values were treated as normal distributed according to a normal probability plot and residual analysis (*SYSTAT*®, data not shown). Summer estimates indicated that the annual bacterial growth was higher before 1987. However, these data have to be treated with caution because of the lower sampling frequency. The increasing trend in bacterial growth was interrupted in 1992.

Thymidine incorporation was used and converted to cell production using a factor of 10^{18} cells mol^{-1}. The same conversion factor was used throughout the period to increase the comparability. Cell production was converted to carbon production according to an allometric volume dependent function [*618*] and the values of the cell volumes reported from the investigation site [*21*]. Cell abundance was determined by direct counts and converted to carbon biomass as described above.

Bacterial biomass - No clear trend in the annual average bacterial biomass could be demonstrated for the whole or parts of the investigation period. According to a *power* analysis of the annual averages, the probability of detecting a true trend of 5.0 % yr^{-1} was 0.80 on a data set of 11 years. The low inter-annual variation implies a tight control of the bacterioplankton by the predators, with moderate variability in grazing efficiency and plausibly a relatively rapid numerical response to variations in bacterial abundance. The latter agrees with the rapid growth kinetics of the small unicellular zooplankton which is defined as the major bacterivores in the northern parts of the Baltic Sea [*375,706*]. The average biomass in the 0-20 m water layer during the whole period was 2.78 mmol C m^{-3}, i.e., 17 % higher than the bacterial concentration recorded for the Kiel Bight (cf. Chapter 4.4).

Bacterial growth - According to a step-wise multiple regression analysis, the variation in the bacterial growth rate during 1988-94 showed a higher linear correlation to the specific growth rate (r=0.92, n=7) than the bacterial biomass. This indicated that variation in the quality of the bacterial growth environment, e.g., temperature or nutrient composition, caused the inter-annual variability rather than a variation in the bacterial biomass, arising from predation pressure. Caution with this interpretation should be used when an autocorrelation of 0.5 is expected for each variable, and a non-linear association between specific growth rate and the bacterial biomass is indi-

Table 4.1.4 Toxic and potentially harmful species in the Gulf of Bothnia

(Years of occurrence is shown as 19XX)

TAXA / BMP station	A1	A3	C1	C4	D1
Species					
CYANOPHYCEA					
Anabaena cylindrica	-	-	-	85	84
Anabaena lemmermannii	86, 92	79	-	-	-
Aphanizomenon flos-aquae	80, 81, 93	80, 81, 83, 93	79-93	79-93	79-92
Nodularia spumigena		79, 86	79,80-84, 86, 88-92	79-88, 90, 92	79, 80, 82-84, 86, 88
Planktothrix agardhi	83, 93	-	-	-	-
DIATOMOPHYCEAE					
Chaetoceros danicus	81	82, 91	79-93	79-93	79-92
DINOPHYCEAE					
Dinophysis acuminata	92	91	79-93	79-93	79-93
Dinophysis norvegica	-	-	82, 93	79, 82, 83	79, 82, 84, 91,92
Prorocentrum balticum	-	87	84, 85, 87	79, 80, 84, 85, 87	79, 80, 85, 87

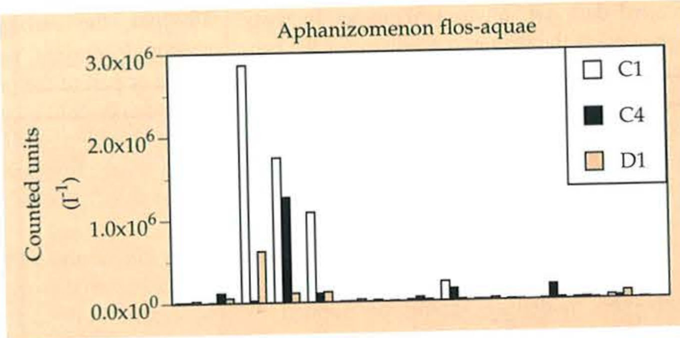


Fig. 4.1.16 Maximum abundance of the blue-green algae Aphanizomenon flos-aquae at those three monitoring stations where they were found sufficient abundant, 1979-93

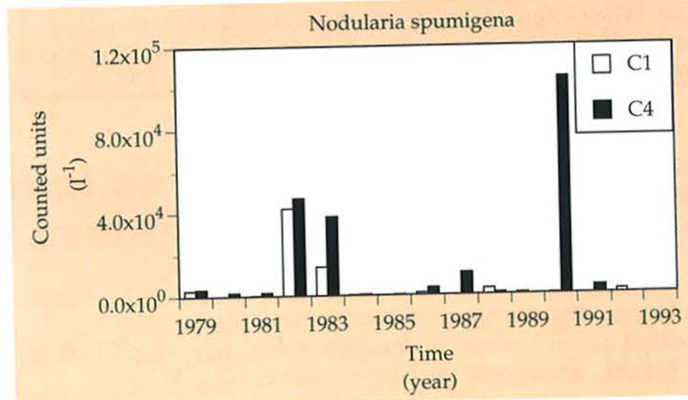


Fig. 4.1.17 Maximum abundance of the blue-green algae Nodularia spumigena at those two monitoring stations where they were found sufficient abundant, 1979-93

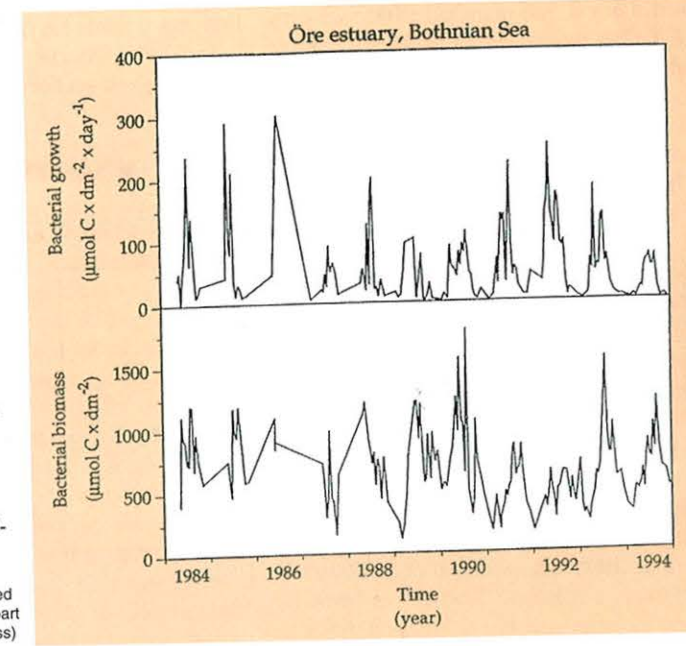


Fig. 4.1.18 Bacterioplankton in the 0-20 m layer of the Öre Estuary (NB1), northern Bothnian Sea, 1984-94

(n=5; upper part - integrated carbon production; lower part - integrated carbon biomass)

		Mean ± RSD (min.-max.)
Bacterial growth	mol C m^{-2} yr^{-1}	1.4 ± 38 % (0.8-2.7)
Bacterial biomass	mmol C m^{-3}	2.8 ± 19 % (2.1-3.7)
Bacterial production-to-biomass ratio	day^{-1}	0.077 ± 45 % (0.045-0.14)
Primary production-to-bacterial production ratio		2.4 ± 40 % (0.9-3.9)
Chllorophyll *a*-to-bacterial biomass ratio	(‡)	2.6 ± 45 % (1.4-5.0)

‡The ratio of chlorophyll *a* to bacterial biomass is given in µg chl. *a* µmol^{-1} C

Table 4.1.5 Basic statistics of annually integrated bacterial variables for the 0-20 m water layer of the north-eastern Bothnian Sea (NB1), 1984-94

(Bacterial biomass and specific growth rate are given as daily averages from data integrated over the year.)

cated. For the correlation analysis of annual values the shorter time period was chosen because of the more frequent and consistent sampling regime.

Among a set of plausible co-variables in the limited data set, phytoplankton productivity appeared as the strongest predictor of the bacterial specific growth rate (r^2=0.74, p=0.028). This required the exclusion of the 1990 value as an outlier, which was possibly caused by the lowest annual temperature and highest bacterial biomass recorded during that year. Collection of additional data over a longer period of time is needed to make this type of analysis more reliable. Non-linear statistical tools and modelling should be applied to achieve a better understanding of the interaction between forcing factors on the annual time scale. Based on the prevailing data and on the limited statistical analysis, it is not possible to reject the hypothesis that natural factors, more probably than anthropogenic influences, caused the period of increasing annual bacterial growth during 1987-92.

Bacterio-to-phytoplankton ratios - During 1990-94, the carbon dioxide fixation has shown decreasing values. However, for the whole or part of the period 1984-94, no significant trend could be demonstrated. During 1992-94, chlorophyll *a* also decreased. But again, for the whole or part of the period 1984-94, no significant trend was observed.

The ratio of the annual CO_2-fixation to the annual bacterial growth (*CF/BG*-ratio) showed no significant trend during the whole or part of the period 1984-94 (Fig. 4.1.19). The average ratio during the period was 2.4, with a minimum value of 0.93 in 1988 and a maximum of 3.9 in 1990.

No significant trend could be demonstrated for the period 1987-94 for the ratio between chlorophyll *a*, used as a proxy for the phytoplankton biomass, and the bacterioplankton biomass (Fig. 4.1.20). Although the ratio increased from 1987 to 1992 (p=0.019, linear regression), it decreased between 1992 and 1994. Variations in the phytoplankton biomass may be expected as a consequence of both varying growth rates and occurrence of predators feeding on phyto- and bacterioplankton. The interpretation of this variable is therefore ambiguous.

4.1.5 Benthic biology

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4.1.5.1 Soft-bottom macrofauna

General description of data treatment - To enhance the reliability of the present assessment, data from the HELCOM BMP and other national stations of Finnish and Swedish monitoring programmes in the Gulf of Bothnia were included. To allow for the fact that not all stations were sampled each year, and because the abundance and biomass differ between the stations, the results had to be presented in terms of index values. The year serving as base index was chosen to be the year when most stations were visited. The index for a specific year was based on 24 offshore stations in the Bothnian Sea (1986), and on 14 stations in the Bothnian Bay for the biomass index (1987), while 21 stations could be included for the abundance index (1965). Each index was calculated as:

$$Index(t) = \frac{\sum_{stn(t)=1}^{N} Ln\left(\frac{y(stn(t))}{y(stn(T))}\right)}{N}$$

where t is the observation year, T is the specific year for the base index, N is the number of stations in year t, and $y()$ denotes the data observed for the variable in question, i.e., abundance or biomass. A few years lacked samples. For those occasions time series analyses (*ARIMA*) were used to estimate the missing values by the *Kalmann*-filtering method [632].

Period 1989-93 - The species composition at the surveyed stations was dominated by *Monoporeia affinis* and *Saduria entomon* in both basins, while the amphipod *Pontoporeia femorata* as well as the marine polychaete *Harmothoe sarsi* occurred sparsely in the southern Bothnian Sea. *Macoma baltica* is a common littoral species in the Bothnian Sea,

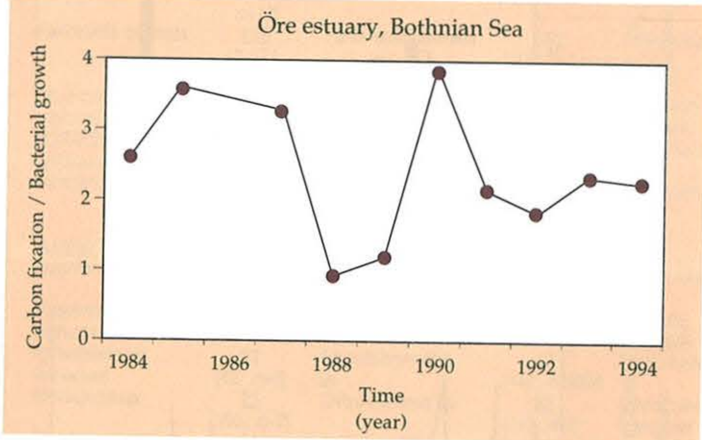


Fig. 4.1.19 Ratio between phytoplankton-carbon fixation and bacterial growth in the Öre Estuary (NB1), northern Bothnian Sea, 1984-94

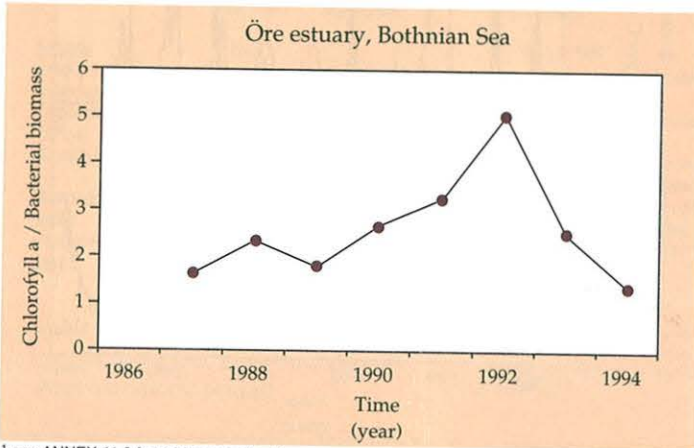


Fig. 4.1.20 Ratio between chlorophyll a and the bacterial biomass in the Öre Estuary (NB1), northern Bothnian Sea, 1987-94

[1] see ANNEX 11.3 for addresses of authors

but the lack of a station network covering these regions makes it difficult to evaluate this species properly. It should be noted that an analysis of any change during this short period is not meaningful. Since there were no drastic changes during this period, compared to the fluctuations observed in earlier time series, and because of the relatively short time span, trend analyses were not considered to be meaningful for the period 1989-93.

Data from the HELCOM BMP stations will serve as examples to elucidate the actual biomass of the macrofauna (Fig. 4.1.21, Tables 4.1.6 and 4.1.7). The densities of *M. affinis* were relatively high at the two stations in the Bothnian Bay, although their biomass was low due to the small individual size of this organism (Tables 4.1.6 and 4.1.7). In the Bothnian Sea, the densities were somewhat higher, except at station *D1* (*F64*). However, the biomass were much higher than in the Bothnian Bay. Numerically, the macrofauna was dominated at all stations by *M. affinis* and *P. femorata*, where it existed. When considering the biomass, *S. entomon* dominated at the stations *A3* (*BO3*) and *D1* (*F64*). The bivalve *M. baltica* was only present at station *SR1A*, being the main species in the group described as 'others' in Tables 4.1.6 and 4.1.7 for this station. The polychaete *H. sarsi* was found to be sparse in the southern Bothnian Sea. The time series of the total macrofauna dry weights from each of these stations showed that the stations have only a few characteristics in common. Otherwise, they are relatively independent.

Period 1989-93 vs. period 1984-88 - No significant differences could be found between the two assessment periods (Table 4.1.8). The total macrofauna abundance and biomass fluctuated in a more or less cyclic manner in the Bothnian Sea (Fig. 4.1.22). These huge short-term variations make it difficult to show changes when comparing two five-year periods. However, long-term evaluation is still possible. The fluctuations were less pronounced in the Bothnian Bay. The long-term increase in abundance and biomass in the Bothnian Bay ceased in 1986, and since then has levelled off (Fig. 4.1.22). At the HELCOM BMP stations (Fig. 4.1.21), there was a conspicuous drop in biomass during the period 1983-85 at most of the stations. One possible explanation for this drop may be the unusual large difference between the winter and the summer temperatures in the beginning of this

Table 4.1.6 Average density (ind. m^{-2}) of the most common macrofauna species in the Gulf of Bothnia, 1989-93

	BO 3	C VI	US 6B	SR 1A	SR 5	F 64
M. affinis	2,340.5	1,700.7	5,816.8	3,329.6	4,714.1	154.8
P. femorata	0.0	0.0	0.0	0.0	145.2	221.7
S. entomon	6.2	1.0	7.7	8.9	29.3	1.5
H. sarsi	0.0	0.0	0.0	0.0	21.3	7.4
Others	0.0	0.0	0.0	41.7	0.0	1.5
Total	**2,346.7**	**1,701.7**	**5,824.5**	**3,380.2**	**4,910.0**	**386.9**

Table 4.1.7 Average wet weight (g m^{-2}) of the most common macrofauna species in the Gulf of Bothnia, 1989-93

	BO 3	C VI	US 6B	SR 1A	SR 5	F 64
M. affinis	3.69	2.93	18.66	13.58	17.79	0.80
P. femorata	0.00	0.00	0.00	0.00	0.83	2.04
S. entomon	7.24	0.80	9.73	1.50	2.84	4.91
H. sarsi	0.00	0.00	0.00	0.00	0.09	0.01
Others	0.00	0.00	0.00	6.38	0.00	0.00
Total	**10.93**	**3.73**	**28.40**	**21.46**	**21.55**	**7.75**

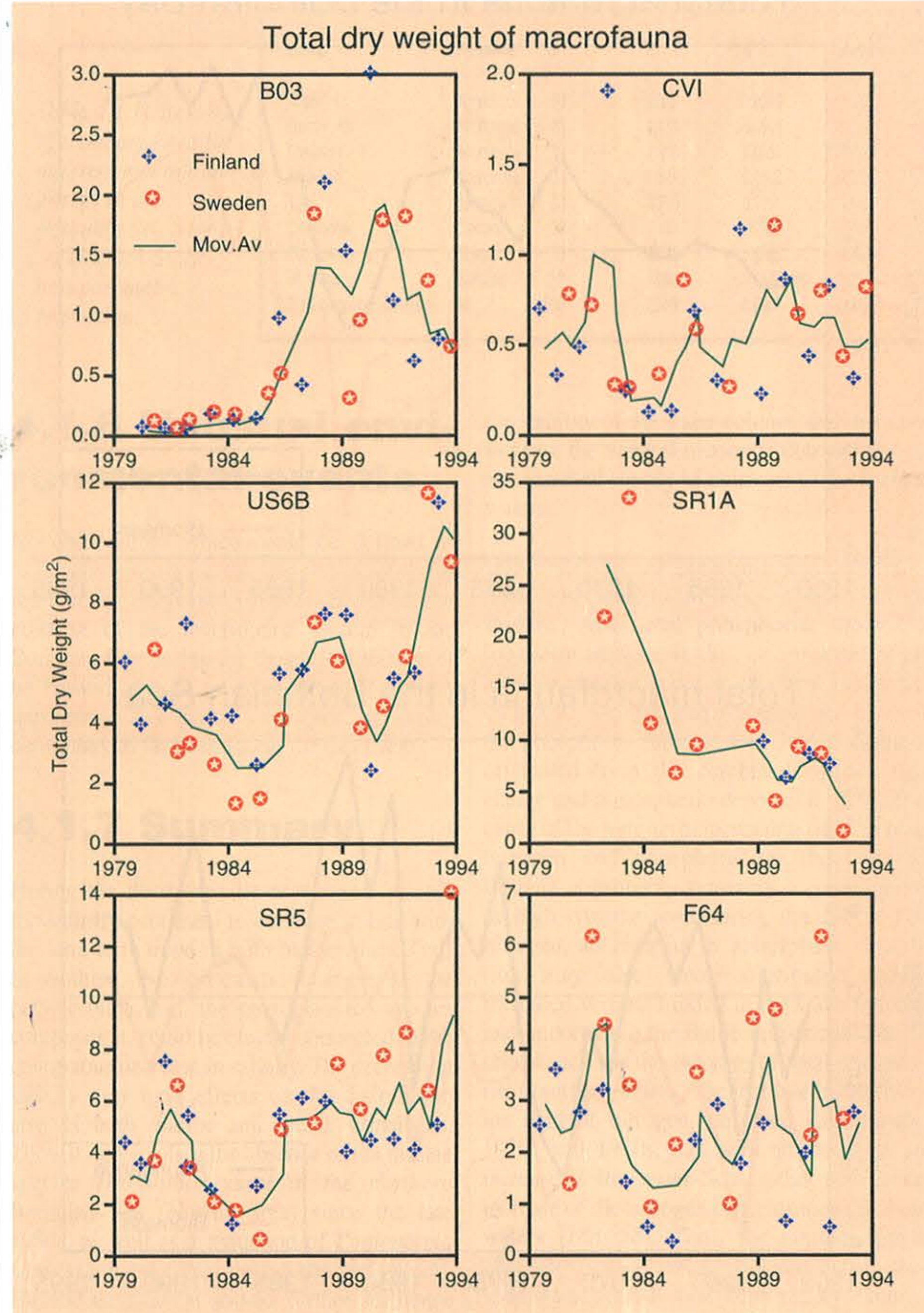


Fig. 4.1.21 Changes in the total macrofauna dryweight –ln g m^{-2}–at 6 HELCOM BMP stations in the Gulf of Bothnia, 1979-94

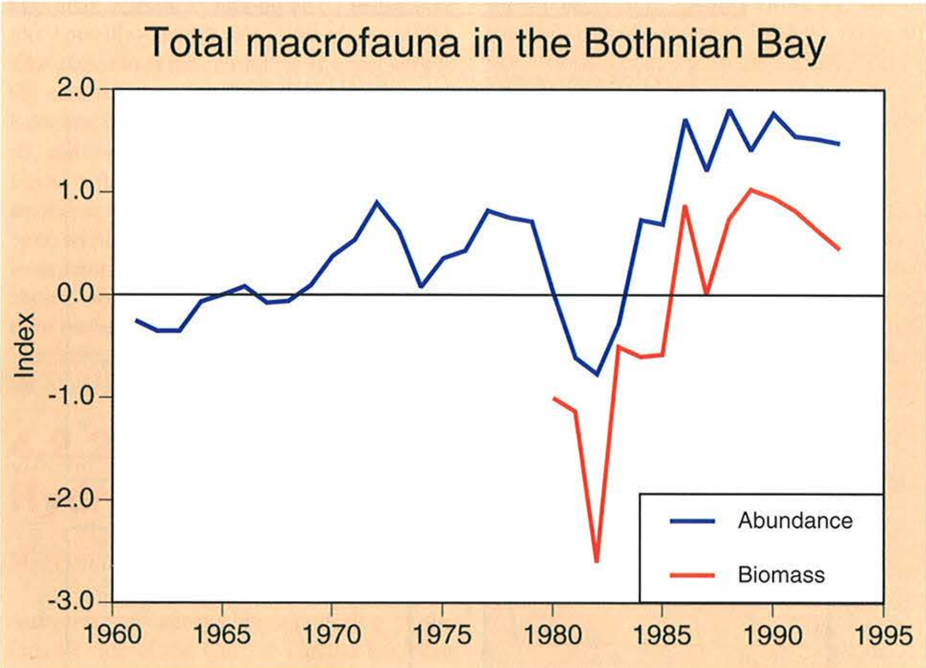
	Z	p
Bothnian Sea Total abundance	-1.7756	0.0758
Bothnian Sea Total biomass	-0.7311	0.4647
Bothnian Bay Total abundance	-0.7311	0.4647
Bothnian Bay Total biomass	-1.5667	0.1172

***Table 4.1.8** Results from Mann-Whitney U-tests for a comparison of the periods 1984-88 and 1989-93, (N=5)*

period (cf. Figs. 4.1.2 and 4.1.3), which could have imposed unfavourable energetic conditions for the macrofauna. Such seasonal differences are commonly found in the Bothnian Bay, where the macrofauna is also much poorer than in the Bothnian Sea. The long-term decrease in salinity may explain the absence of the marine species *Harmothoe sarsi* in the northern Bothnian Sea (Norrby area) since the late 1980s, as well as the reduction of *Pontoporeia femorata* at C4 (*SR5*) during the last decade.

Long-term evaluation 1961-93 - Concerning the total abundance and biomass of the macrofauna, there are increasing long-term trends both for the Bothnian Sea and the Bothnian Bay. The trend in the Bothnian Bay was interrupted in the early 1980s due to the pronounced reduction of the macrofauna community during the late 1970s. The macrofauna has recovered since then and has levelled off since the mid 1980s. This is especially evident for the total macrofauna abundance (Fig. 4.1.22). Also in the Bothnian Sea, there was a tendency towards levelling off during the 1980s, but not as pronounced as in the Bothnian Bay.

The asymptotic increasing trends are statistically significant in both basins (Tables 4.1.9 to 4.1.11). In the Bothnian Bay, this trend is found for all the main species. The picture in the Bothnian Sea is similar for the total abundance and biomass. The macrofauna group that contributes most to the increasing trend in the Bothnian Sea seems to be the predators, while the detritivores do not exhibit a significant trend. The interpretation of such a change in the benthic community is that the production of the detritivores has increased. However, the main part of their standing crop ends up in the predator populations. Earlier analyses of the benthic macrofauna have also shown increases between the 1960s and the 1990s [*13,96,97,372,399,504*]. Despite in one of those studies [*372*] many coastal stations were included, the same changes were found as for the open sea areas of the Gulf of Bothnia. The increase in the benthic fauna is most likely caused by enrichment. To this may be added the similar effects of toxic contaminants, as advanced in a complementary hypothesis [*399*].



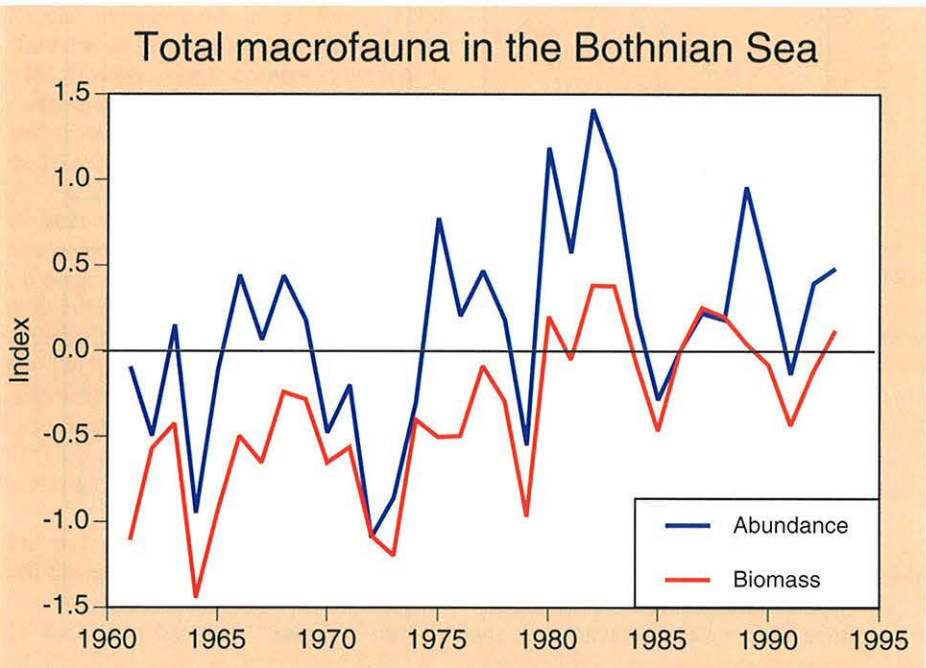


Fig. 4.1.22 Changes of the total macrofauna index in the Bothnian Bay and Bothnian Sea, 1961-93

(Base index for abundance and biomass is in 1986 for the Bothnian Sea. For the Bothnian Bay, the base index for abundance is in 1965, while for the biomass in 1987.)

4.1.5.2 Phytobenthic changes

Archipelago Sea - The decline of *Fucus vesiculosus* was the first alarming sign of eutrophication observed in the Archipelago Sea [*193*]. Aerial photography was used to study the distribution of *Fucus vesiculosus* in the Archipelago Sea in 1981-82. The decline of *F. vesiculosus* was most evident at moderately exposed localities in central and outer archipelago areas which were directly influenced by water from the Baltic Proper [*569*].

In 1993, the aerial mapping was repeated and the results demonstrated that only minor recovery had taken place in ten years [*436*]. During 1994, the continuous belt of *Fucus* only extended from 1 to 4 m depth, whereas individuals could be found down to 7-8 m [*439*].

Phytobenthos vegetation surveys in 1968/69 [*561*], 1981 [*438*] and 1991 revealed that the zone previously occupied by *F. vesiculosus* was now dominated by an intense growth of filamentous algae. At some places, *Fucus* was replaced by phanerogams, mainly by *Potamogeton* spp. and *Zannichellia palustris*. The change in bottom quality was explained by heavy sedimentation, for example due to increased primary production. In 1991, *Fucus* was still lacking, but the amount of filamentous algae had decreased.

In the 1990s, loosely lying algal mats have been recorded. These mats are formed since the opportunistic filamentous algae are capable of utilising nutrients, and they have been found at several places in the Archipelago Sea [*439*]. Algal mats have been formed mainly by filamentous brown and red algae like *Pilayella littoralis*, *Ectocarpus siliculosus* and *Ceramium tenuicorne*. The thickness as well as the species composition of these mats varied considerably. Under the thick algal mats, oxygen is consumed very quickly causing hypoxia and anoxia, and under the circumstances, sulphate is reduced and H_2S formed. The presence of H_2S has been recorded even in the surface water [*437*].

Åland Archipelago - In the Åland Archipelago, the phytal changes followed those of the Archipelago Sea and the Gulf of Finland. In the Åland area, loosely lying filamentous algae are commonly found on sandy and muddy bottoms, with an average of about 200 g and a maximum of 800 g dry-weight m^{-2}. *Pilayella littoralis* and *Ectocarpus siliculosus* represent about 60 %, and *Stichtyosiphon tortilis* and *Dictyosiphon foeniclaceus* 25 % of the biomass. Regularly, >50 % of the bottom area to a depth of 4-10 m was found to be covered by this type of algae, causing significant ecological effects for the zoobenthos and possibly also fish [*95,505-507*]. Decreases in the depths at which *Fucus vesiculosus* grows is also observed in the outer archipelago. The red algae species, found in the 1960s in the innermost part of the outer archipelago, and in the intermediate archipelago are currently covered by loosely lying algal mats. More clear changes have taken place in SW Åland [*444,445,525,570*]. Loosely lying algal mats are a very frequent phenomenon. Due to this fact, the macrophytic communities are presently poor in species, especially in the deep water. The growing depth of *F. vesiculosus* has further decreased [*570*].

***Table 4.1.9** Abundance*

Results of a non-parametric trend analysis [251,152] for different macrofauna groups in the Bothnian Bay

(*-BO3)* denotes that the station *BO3* was excluded from the tests.

Group	n	Z	p	Slope
Total	33	4.354	0.0000	0.059
Total (-BO3)	33	5.934	0.0000	0.057
M. affinis	33	3.951	0.0001	0.060
M. affinis (-BO3)	33	5.686	0.0000	0.058
M. affinis & S. entomon	33	5.841	0.0000	0.054
M. affinis & S .entomon (-BO3)	33	5.686	0.0000	0.059
Total indep. not assumed	198	5.670	0.0000	0.058

***Table 4.1.10** Biomass*

Group	n	Z	p	Slope
Total	14	2.518	0.0118	0.167
Total (-BO3)	14	1.533	0.1253	0.055
M. affinis	14	2.956	0.0031	0.192
M. affinis (-BO3)	14	1.314	0.1889	0.045
M. affinis & S. entomon	14	2.737	0.0062	0.200
M. affinis & S. entomon (-BO3)	14	1.095	0.2736	0.044
Total indep. not assumed	84	2.245	0.0248	0.115

***Table 4.1.11** Results of trend analyses for different macrofauna groups in the Bothnian Sea based on Whirsch-Slack non-parametric regression*

Group	Parameter	n	Z	p	Slope
Total	Abundance	33	2.309	0.0210	0.022
Detritivores	Abundance	33	1.751	0.0800	0.019
Predators	Abundance	33	2.959	0.0031	0.042
M. affinis	Abundance	33	1.844	0.0652	0.020
Total	Biomass	33	3.796	0.0001	0.030
Detritivores	Biomass	33	1.131	0.2580	0.014
Predators	Biomass	33	2.092	0.0365	0.030
M. affinis	Biomass	33	1.503	0.1328	0.018
Total indep. not assumed	All	330	2.729	0.0064	0.025

4.1.6 Unusual environmental events

No observations which could be defined as 'unusual environmental events' have been reported for the period 1989-93. However, the collapse of the macrofauna system in the Bothnian Bay during the the early 1980s must be viewed as such an event since it has not appeared at any station since then. No explanation has so far been found for this event.

4.1.7 Summary

During the third periodic assessment period, the **salinity** continued to decrease in line with the long-term trend in both basins of the Gulf of Bothnia. No systematic changes in the pelagic biota, e.g., the phytoplankton species composition, could be clearly connected to the continuous decrease in salinity. The decreasing salinity may have effects on the distribution area of both marine and limnic organisms. Thus, it may explain the absence of the marine species *Harmothoe sarsi* in the northern Bothnian Sea (Norrby area) since the late 1980s, as well as a reduction of *Pontoporeia femorata* at *C4 (SR5)* during the last decade. Decreasing salinity should further influence the stability of the water column, and this may increase the vertical mixing which can lead to an increased supply of nutrients to the surface waters.

Both total and inorganic **nitrogen** concentrations continued to increase in the Gulf of Bothnia. Also total **phosphorus** showed a long-term increase in the surface water of all areas except the Bothnian Bay. It was estimated that 92 % of the nitrogen input and 65 % of the phosphorus input to the Gulf of Bothnia originated from the combined riverine discharge and atmospheric deposition [*729*]. The cause of the long-term increasing trend in both nitrogen and phosphorus in the Gulf of Bothnia is probably primarily a consequence of high riverine load during the 1980s. For nitrogen, an increase in atmospheric deposition may also have contributed, while increased vertical mixing in the water column and import from the Baltic Proper could be of importance for the increase of total phosphorus in surface waters. The increase in the riverine load of nitrogen, observed between the 1970s and 1980s, has been attributed to an increase in the water flow, rather than to an increase of the nitrogen concentrations in river waters [*188,558*]. Thus, the nutrient trends observed in the Gulf of Bothnia during this assessment period, could be primarily a consequence of natural variation in climate. Locally, however, extensive agricultural activities can significantly affect the anthropogenic part of the riverine load, and this has been suggested to be of importance for the Finnish part of the drainage area to the Bothnian Sea.

The concentration of **oxygen** in the deep water showed no major changes in any of the sub-areas. Therefore, even if an increase in organic production in the open sea had occurred, it has not led to a decrease in the deep water oxygen concentration. This may partly be explained by a lower water stability and improved vertical mixing due to the long-term decrease in salinity. The data suggest that the oxygen supply to the deep water of the Gulf of Bothnia has been sufficent to meet the oxygen demand related to respiration of the sedimenting organic matter.

Until 1993, the data from the open sea did not reveal any pronounced increase of **pelagic organic production** in the Bothnian Bay or Bothnian Sea. The oxygen supply to the deep water appeared sufficient to sustain biologically acceptable conditions. Data from the coastal northwestern Bothnian Sea were in accordance with the assessment of the open sea. From the Archipelago Sea, however, elevated concentrations of nutrients and chlorophyll *a*, as well as changes in the phytobenthic community were reported. Some long-term changes in the open sea, such as a general increase in nitrogen and total phosphorus, may have an impact on the Gulf of Bothnia, if such increases are continued. The changes in nutrient concentrations are considered to be primarily caused by climatic variations rather than by anthropogenic inputs.

The decrease in the distribution of *Fucus vesiculosus* and the spreading of filamentous algae suggest further **eutrophication** of the Archipelago Sea. This is supported by increasing chlorophyll *a* and relatively high nitrogen and phosphorus concentrations. Fish farming is suggested as one important source of nutrients in this area.

A significant increase of **chlorophyll *a*** was observed for the period 1979-93 in the Bothnian Sea, primarily caused by higher values during the last 5 years of that period, and suggesting an increasing trophic state during summer. Neither phytoplankton biomass nor carbon fixation, however, showed a corresponding increase. Since this finding could be merely a reflection of the limited data set for these variables, one cannot exclude the possibility that the increase in chlorophyll *a* may lead to an increase in the pelagic organic production. A coastal station in the northwestern Bothnian Sea, intensively studied during 1984-94, however, lacked significant trends in both primary productivity and bacterial production. In the Bothnian Bay, no significant trends could be demonstrated for pelagic biological variables.

No clear systematic change in the species composition of either phytoplankton or benthic fauna has been reported, that could imply a change in the environmental state of the Gulf of Bothnia. However, several potentially **harmful phytoplankton species** were found to be present in the Gulf of Bothnia, and their continuous monitoring is therefore recommended. Otherwise, the phytoplankton species composition has remained roughly similar throughout the period 1979-93.

The **bacterioplankton growth** showed moderate inter-annual variability at a coastal station in the northern Bothnian Sea, with a period of continuous increase during 1987-92. The variability appeared to be largely explained by a variation in the specific growth rate, i.e., in the P/B ratio, and correlated well with the inter-annual variations in the phytoplankton productivity. However, no significant trend could be demonstrated over a 10-year period. Despite the variation in the bacterial growth, the **bacterial biomass** showed minor inter-annual variability without significant trends. This suggests an efficient and stable predator control of the bacterial community. No trend in the ratios between the phytoplankton CO_2-fixation and the bacterioplankton growth, or between chlorophyll *a* and the bacterial biomass, could be demonstrated. Thus, the analysis of the bacterioplankton variables did not suggest a systematic change in the environmental state, e.g., due to anthropogenic activities, in the northwestern Bothnian Sea. Generalisations over a larger sea area and a more reliable determination of covariates must be deferred until more data are available from off-shore stations.

An increase of the **benthic macrofauna** could be demonstrated both in the Bothnian Sea and Bothnian Bay since the beginning of the 1960s until the early 1980s. From the 1980s on, the macrofauna biomass has levelled off and no significant trend was indicated or could be shown statistically due to the large variance of data. The observed increases in the benthic macrofauna and in chlorophyll *a* do not match in time. Therefore, they do not support the hypothesis of an increase of the organic production in the Bothnian Sea. However, the increase in total macrofauna, which occurred earlier than the increase in chlorophyll, may reflect an increase in the availability of nutrients and in the pelagic organic production.

4.2 GULF OF FINLAND

M. Perttilä[1] (Convener), *O. Savchuk* (Co-Convener)

4.2.1 Hydrography

M. Perttilä, O. Savchuk

The Gulf of Finland is a direct extension of the Baltic Proper (Fig. 4.2.1). There is no threshold at the opening of the Gulf, and the maximum depth is some 100 m. The bottom depth decreases towards the east, being 60-80 m in the middle part, and 20-40 m at the eastern end of the Gulf, before the Neva Bight.

Main characteristics:
- overall length - 400 km (from the Hanko peninsula to St. Petersburg,
- largest width - 135 km (from Narva Bight to Vyborg),
- area - 29,600 km^2 (7 % of the total area of the Baltic Sea),
- average depth - 38 m (55 m for the Baltic Sea),
- deepest point - 123 m (459 m for the Baltic Sea),
- volume - 1,100 km^3 (5 % of the total volume of the Baltic Sea),
- drainage area - 421,000 km^2 (26 % of the total drainage area of the Baltic Sea),
- river inflow - 100-125 km^3 yr^{-1} (24-27 % of the total river inflow into the Baltic Sea),
- fresh water residence time 8-10 yrs (25-35 yrs for the whole Baltic Sea).

In spite of being the utmost eastern end of the system of Baltic Sea basins, the Gulf of Finland is, however, hydrographically still influenced by the inflows of saline waters from the Kattegat/Skagerrak. In addition, approximately one fourth of the total river water volume, received by the Baltic Sea, enters directly into the Gulf of Finland. Thus, the deep water hydrogen sulphide layer of the Baltic Proper may extend into the western parts of the Gulf of Finland [*16*]. On the other hand, the eastern end of the Gulf receives the fresh water of the river Neva. This is the largest river in the drainage area of the Baltic Sea, with an annual inflow of 78.6 ± 13.8 km^3 [*652*]. The combined effects of the large fresh water inflow in the eastern end and the water exchange with the Baltic Proper lead to strong salinity gradients in the Gulf of Finland (Fig. 4.2.2).

The bottom topography, forming an almost free entrance from the Baltic Proper into the Gulf of Finland, leads to a halocline in the western part of the Gulf, at the depth of 60-70 m. Here, the water below the halocline has its origin mainly in the Baltic Proper, and it mixes slowly with the less saline surface layer. As a result, the halocline becomes less distinct towards the east. However, in the eastern end of the Gulf, where the large fresh water input causes strong horizontal salinity gradients, the halocline becomes sharper again, climbing up to 10-30 m. In the easternmost shallow end with depths less than 10 m, vertical salinity gradients seldom occur because of the intensive wind-induced mixing. In addition, a strong thermocline develops in the summer at a depth of 10-30 m, further hampering the ver-

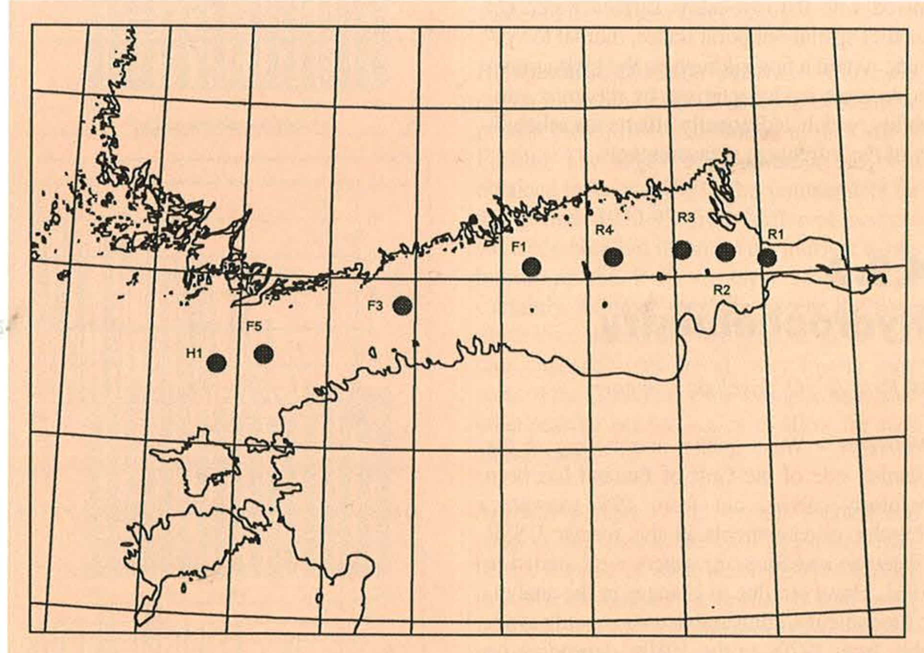


Fig. 4.2.1. The Gulf of Finland with the stations referred to in the text

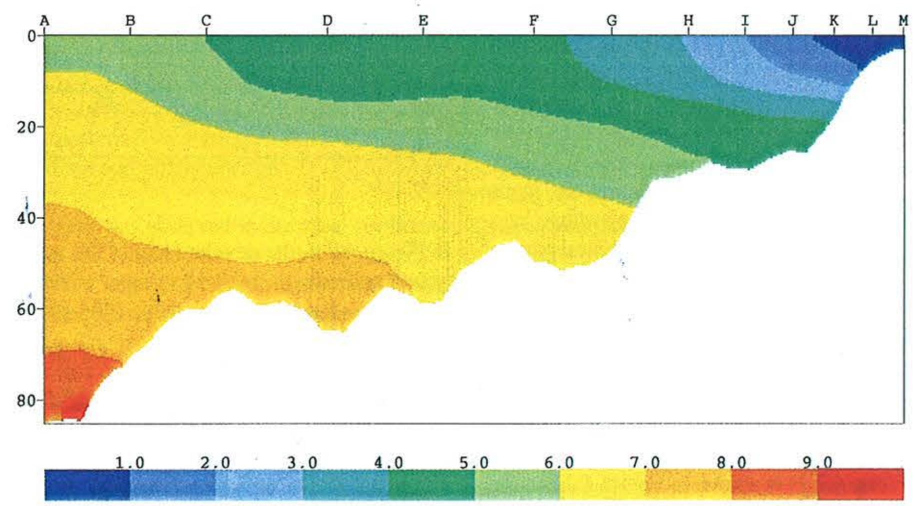


Fig. 4.2.2 The salinity distribution - values in psu- along the longitudinal axis of the Gulf of Finland obtained from the 3D hydrodynamically interpolated field [727]

[1] see ANNEX 11.3 for addresses of authors

tical mixing. The effect of the thermocline may be strong enough to cause a remarkably strong and sharp halocline at the same depth. Because of the restricted mixing through the density difference, the vertical profiles of the nutrients resemble those of salinity, producing a strong chemocline in summer. The autumnal cooling and consequent vertical mixing brings water from the deep, nutrient-rich layer up to the surface layer, resulting in strong biological production in the spring.

On a long-term, basin-wide scale, the overall water circulation in the Gulf of Finland is anticlockwise, causing an eastward transport along the southern coast, and a westward transport along the northern coast. In the general hydrodynamics of the entire Baltic Sea, the Gulf of Finland is considered to play an important role as a 'mixing box', because of the upwelling and mixing processes which take place along the coastal areas, especially in the easternmost parts. The relatively unstable halocline periodically becomes strongly eroded, and the saline water masses are frequently mixed with the low-saline surface water. On smaller spatial-temporal scales, diurnal to synoptic within a few kilometres, the hydrographic structure is characterised by the great variability, which additionally affects the reliability of the infrequent measurements.

4.2.2 Hydrochemistry

M. Perttilä[1], *O. Savchuk, I. Shpaer*

Nutrients - Water-quality monitoring at the Finnish side of the Gulf of Finland has been regularly carried out from 1962 onwards. Regular measurements in the former USSR (Estonian and Russian) waters were started in 1961. However, due to changes in the analytical techniques, comparable data are only available from 1970s to the 1980s, depending on the respective parameters.

Because of the strong chemocline, the permanent halocline, and the strong annual biological production, the seasonal variations in the 0-20 m layer have been considered to be too large for trend analyses of chemical parameters [*305*]. However, the observations in the deep water layer, i.e., below the average halocline depth, in the western and central parts of the Gulf can in most cases be used for trend evaluations without seasonal filtering [*539*].

The overall development of salinity, oxygen and the main nutrients up to 1995 in the surface and deep water layer of the BMP station F3 (*LL7*), lying in the middle part of the Gulf (cf. Fig. 4.2.1) is shown in Figure 4.2.3.

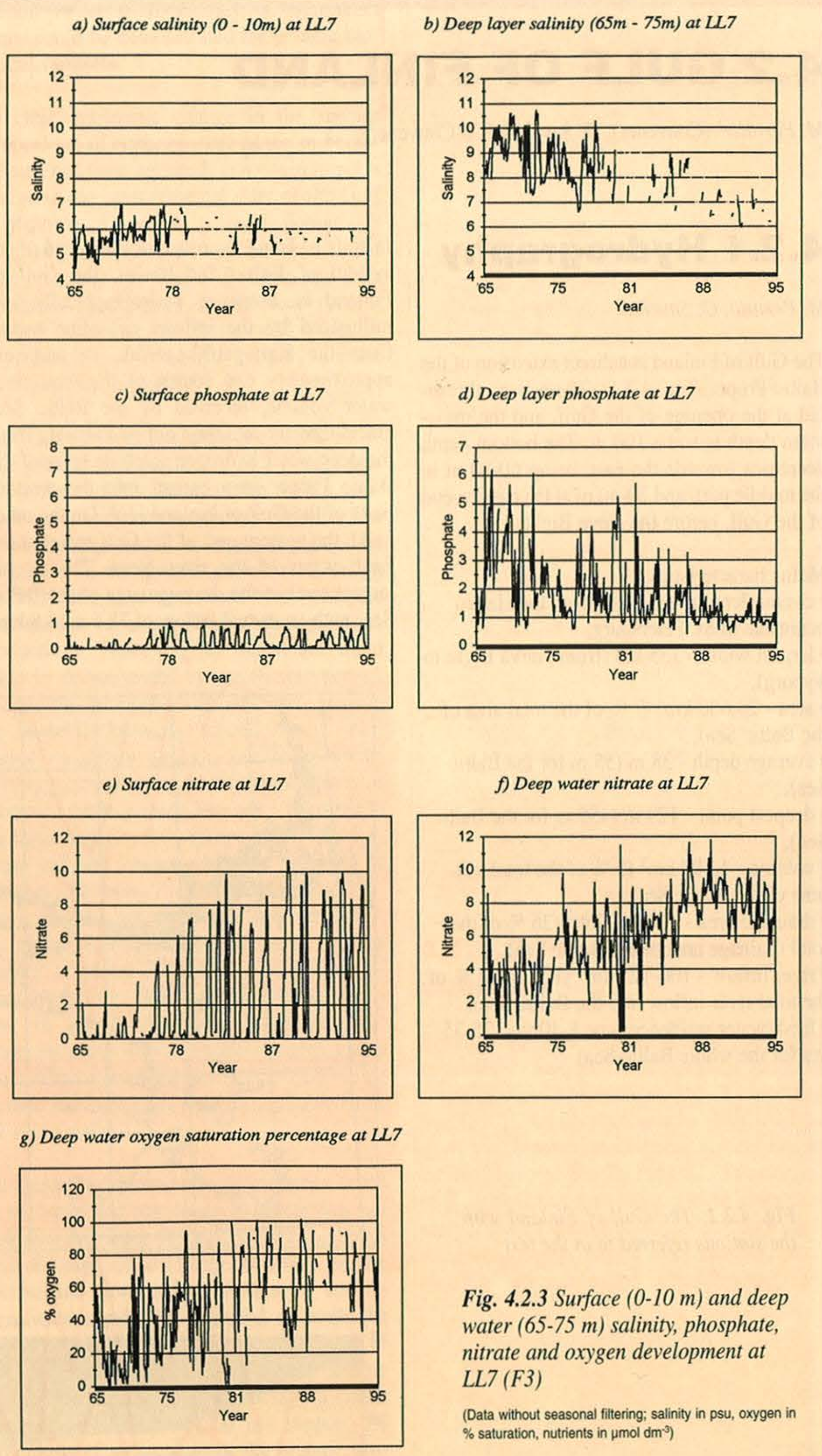


Fig. 4.2.3 Surface (0-10 m) and deep water (65-75 m) salinity, phosphate, nitrate and oxygen development at LL7 (F3)

(Data without seasonal filtering; salinity in psu, oxygen in % saturation, nutrients in μmol dm^{-3})

In Figure 4.2.4, the development of the salinity and nutrients in the 0-10 m layer at station *LL7* during the winter periods of 1984-88 and 1989-93 is presented.

For the eastern part, the long-term variations of salinity and oxygen in the bottom layer at the Russian stations *R1* and *R4* are shown in Figure 4.2.5. The development in the concentrations of phosphate at stations *R2* and *R4* during 1972-85 and 1986-92 is presented in Figure 4.2.6 (for positions, cf. Fig. 4.2.1).

In long-term time series (Fig. 4.2.5), the decrease in salinity can be observed in the near-bottom water layer. This development corresponds to that found for other Baltic Sea areas [*220*]. The decrease, which is due to the lack of major saline water inflow through the Belt Sea and the Danish Straits into the Baltic Sea, also reduces the intensity of the halocline. This has resulted in improved oxygen conditions (Fig. 4.2.5). During 1984-88 and 1989-93, no changes in oxygen could be detected in the surface layer.

[1] see ANNEX 11.3 for addresses of authors

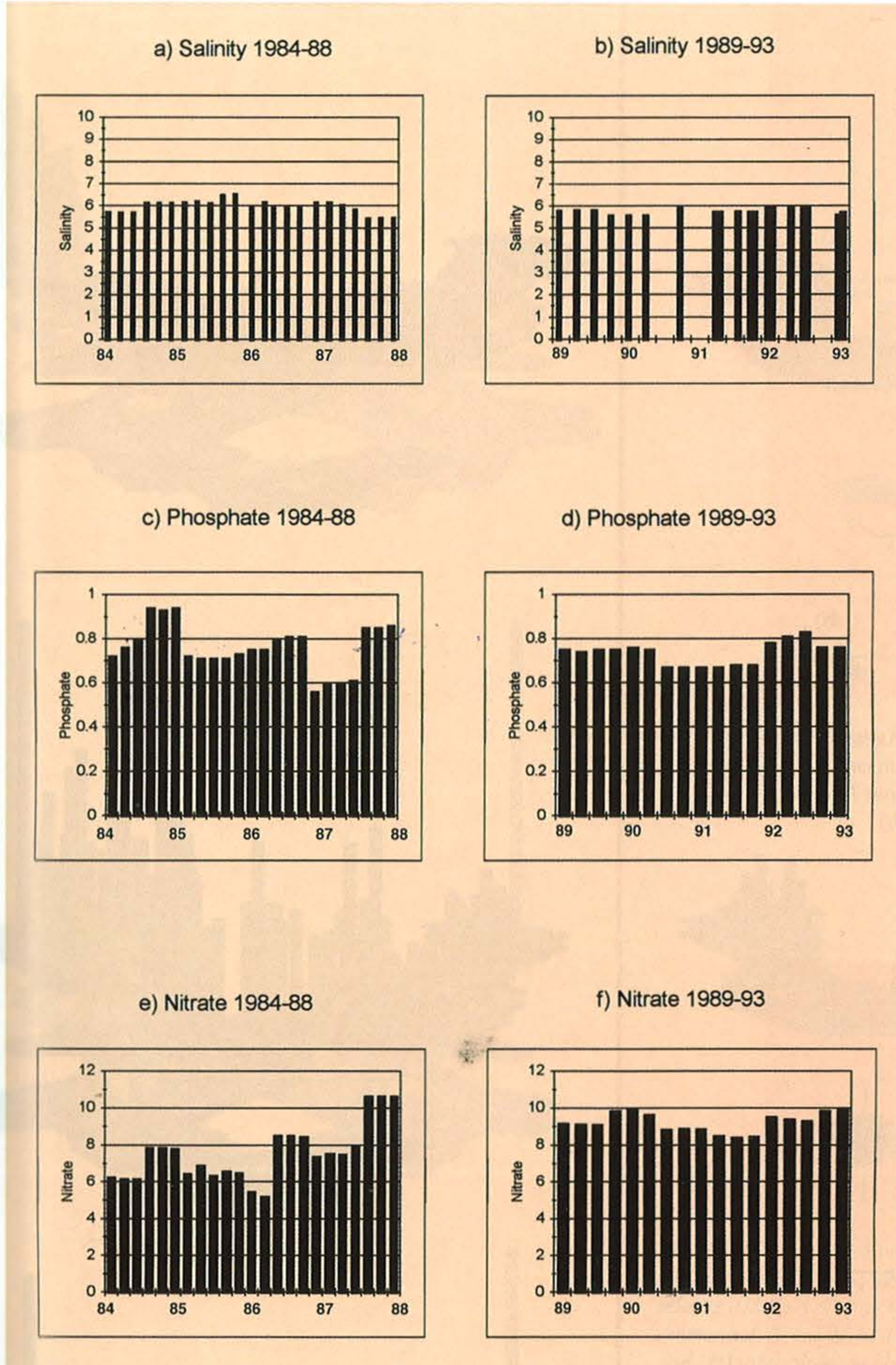


Fig. 4.2.4 Salinity, phosphate and nitrate in 0-10 m winter waters of station LL7, 1984-88 and 1989-93

(Data without seasonal filtering; salinity in psu, nutrients in μmol dm^{-3})

The long-term nitrate concentrations showed a significantly increasing trend. At station *LL7*, the increase of nitrate was 0.22 and that for total nitrogen 0.59 μmol dm^{-3} yr^{-1}. A similar development took place at other open-sea stations in the Gulf of Finland [*539*]. However, since 1990/91, the trend in the nitrogen concentrations has been levelling out. This is clearly seen when comparing the nitrate developments in Figure 4.2.4. During 1984-88, the nitrate concentrations in the surface layer increased, whereas they levelled out in 1989-93. The overall average nitrate concentrations in the bottom layer throughout the whole period did not practically change.

Phosphate was observed to decrease considerably in the bottom layer, and this is probably due to the improved oxygen conditions in the deep water layer caused by intensified vertical mixing. The bottom layer is presently oxidised down to a depth of about 80 m, resulting in a significant decrease in the re-mobilisation of phosphorus from sediments. A further analysis of data on oxygen and phosphorus in deep water shows that high phosphorus values have almost always been connected to low oxygen values in the bottom layer [*539*].

Horizontal distribution patterns - Currently, only a limited number of data are available to present the nutrient distribution in the Gulf of Finland. The largest areal coverage has been obtained in August 1991. The summer data for the period 1990-93 from different sources were combined to illustrate the nutrient distribution in the 0-10 m layer (Fig. 4.2.7). Certainly, summer may not appear the most illustrative period for that purpose as the surface concentrations are all very low in most parts of the Gulf. However, synoptic data from other seasons are too scarce to allow for such an analysis.

Nutrient budget - The overall sources and sinks of nutrients in the Gulf of Finland have been compiled in Table 4.2.1 [*539*]. Although this estimate ignores the spatial as well as the seasonal variability of the processes, it nevertheless indicates a rough balance between inputs and losses of nutrients.

The present apparent balance of the nutrient budget, as indicated by the levelled-out trends, is probably easily disturbed by a slight change in the significance of the water balance and sedimentation, and additionally by changes in the activity of the denitrification and nitrogen fixation processes and, for nitrogen, even by direct precipitation. Also, a decrease in rainfall would primarily affect the large riverine discharges and the direct nitrogen precipitation. Both the vague levelling-off or even decreasing trend in the dissolved and total deep water nitrogen and the phosphorus trend, which is decreasing rather than levelling out, emphasize the need for more precise information on these processes.

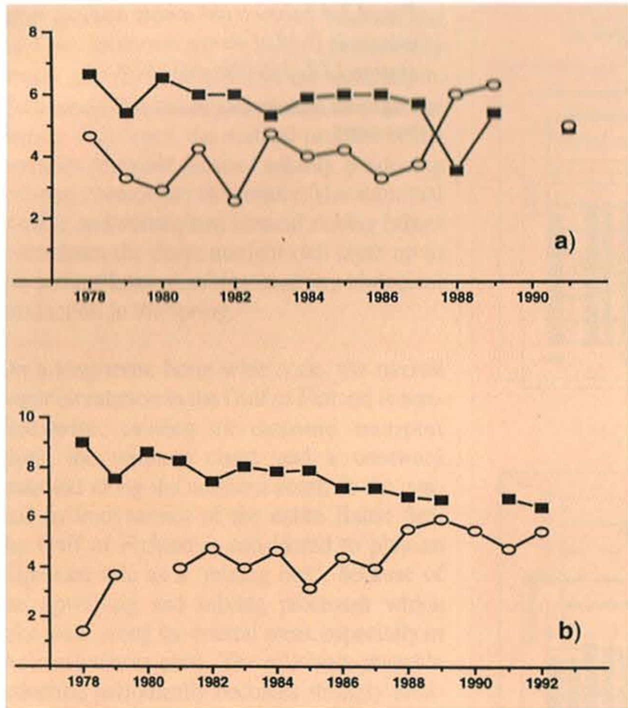


Fig. 4.2.5 *Inter-annual variations of summer (August) salinity (in psu, black rectangles) and oxygen (in cm^3 dm^{-3}, open ellipses) in the bottom layer at stations R2 (water depth 25-28 m) (a), and R4 (58-62 m) (b)*

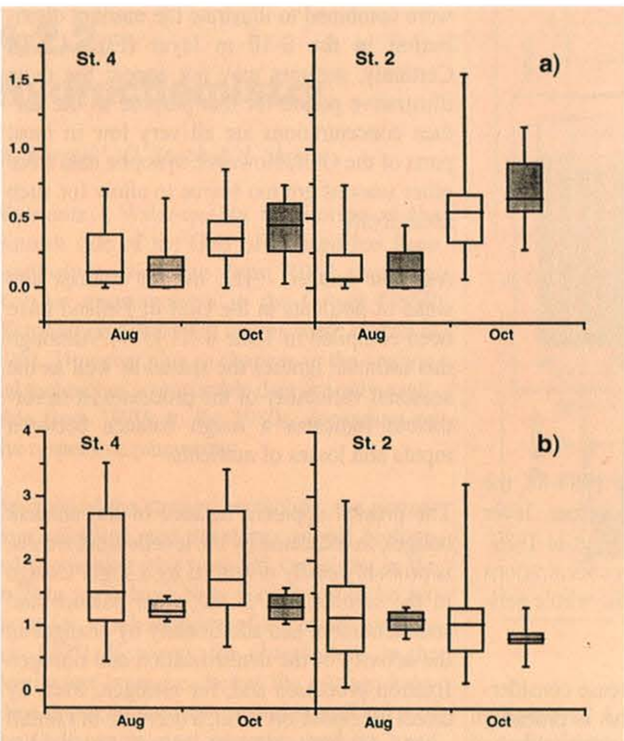


Fig. 4.2.6 *Box-and-whisker plots of phosphate concentrations (in μmol dm^{-3}) at stations R2 and R4 during 1975-85 (open boxes) and 1986-92 (black boxes), (a) surface layer, (b) bottom layer*

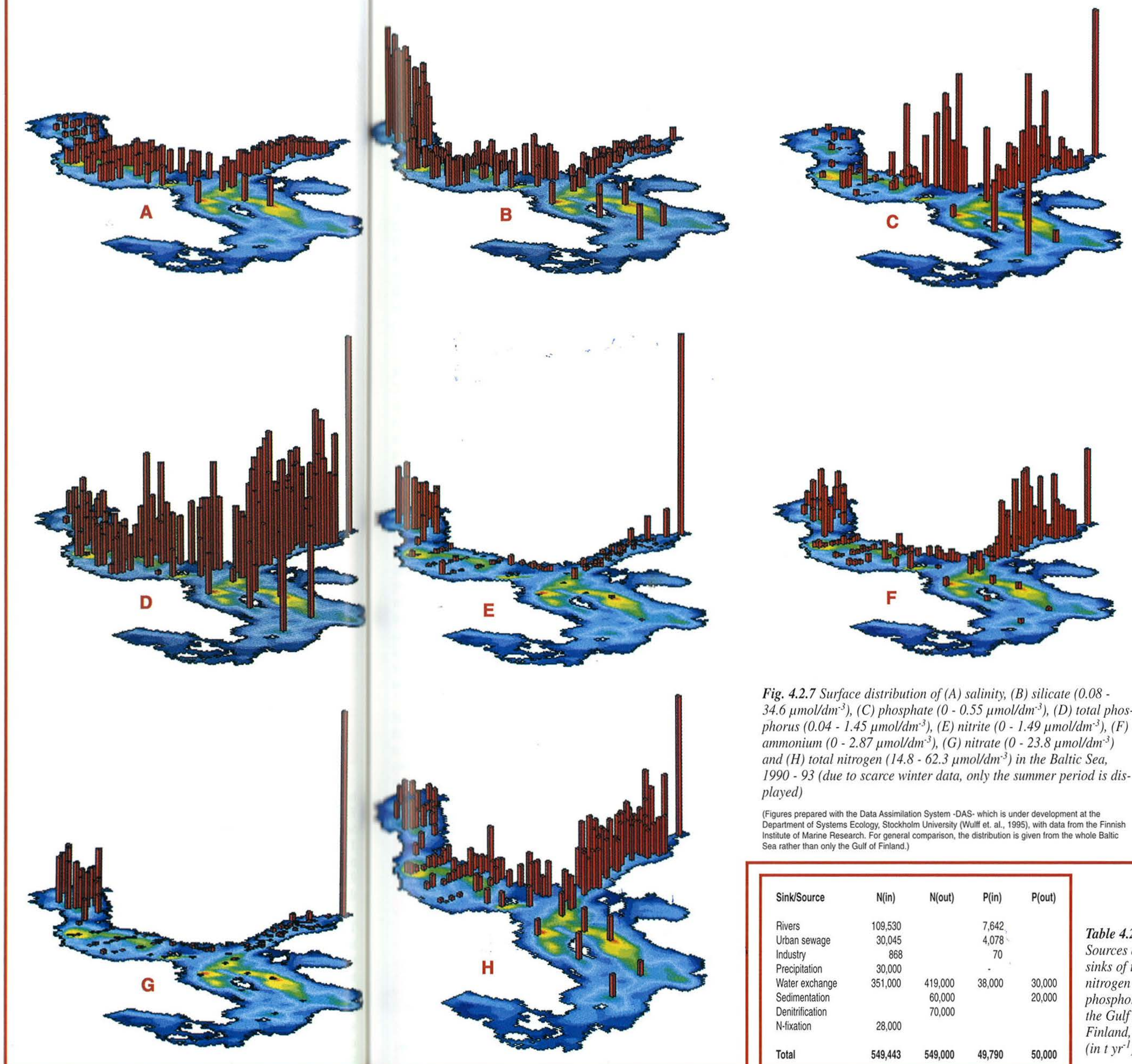


Fig. 4.2.7 *Surface distribution of (A) salinity, (B) silicate (0.08 - 34.6 μmol/dm^{-3}), (C) phosphate (0 - 0.55 μmol/dm^{-3}), (D) total phosphorus (0.04 - 1.45 μmol/dm^{-3}), (E) nitrite (0 - 1.49 μmol/dm^{-3}), (F) ammonium (0 - 2.87 μmol/dm^{-3}), (G) nitrate (0 - 23.8 μmol/dm^{-3}) and (H) total nitrogen (14.8 - 62.3 μmol/dm^{-3}) in the Baltic Sea, 1990 - 93 (due to scarce winter data, only the summer period is displayed)*

(Figures prepared with the Data Assimilation System -DAS- which is under development at the Department of Systems Ecology, Stockholm University (Wulff et. al., 1995), with data from the Finnish Institute of Marine Research. For general comparison, the distribution is given from the whole Baltic Sea rather than only the Gulf of Finland.)

Sink/Source	N(in)	N(out)	P(in)	P(out)
Rivers	109,530		7,642	
Urban sewage	30,045		4,078	
Industry	868		70	
Precipitation	30,000		-	
Water exchange	351,000	419,000	38,000	30,000
Sedimentation		60,000		20,000
Denitrification		70,000		
N-fixation	28,000			
Total	549,443	549,000	49,790	50,000

Table 4.2.1 *Sources and sinks of total nitrogen and phosphorus in the Gulf of Finland, 1990 (in t yr^{-1}) [539]*

4.2.3 Pelagic biology

H. Kuosa[1], S. Makarova, N. Silina

4.2.3.1 Phytoplankton

Chlorophyll a and primary productivity - The data used to describe the western and central parts of the Gulf originate from the stations *F1* (*LL3a*), *F3* (*LL7*) and *H1* (*LL12*). Due to the limited amount of monitoring data they were analysed as pooled data for the whole Gulf of Finland. The material was split into three seasons. The results are presented in Figure 4.2.10. No statistically significant trends were observed when the data were analysed with the non-parametric *Whirsch-test*. Primary productivity data showed even more variability, and it did not give additional information.

An increasing trend for chlorophyll *a* was clearly shown at a frequently sampled coastal station near the entrance to the Gulf of Finland [*190*]. From 1972 to 1988, an increase in spring values was observed from 1983 onwards. When the data from 1989-94 were compared to those from 1972-88, the average spring values in 1989-94 (9.7 mg m^{-3}) were closer to the period 1972-79 (7.9 mg m^{-3}) than to 1980-88 (19.6 mg m^{-3}). Thus, the increasing trend most probably has not continued. On the other hand, the average summer chlorophyll *a* concentration in 1989-94 (3.5 mg m^{-3}) was clearly higher than that of 1972-79 (2.1 mg m^{-3}) or 1980-88 (2.8 mg m^{-3}). This indicates increased phytoplankton biomass during summer months at this specific sea area during the last assessment period.

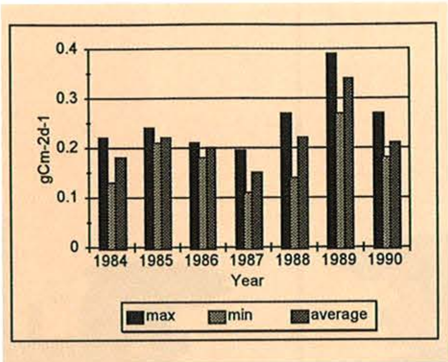


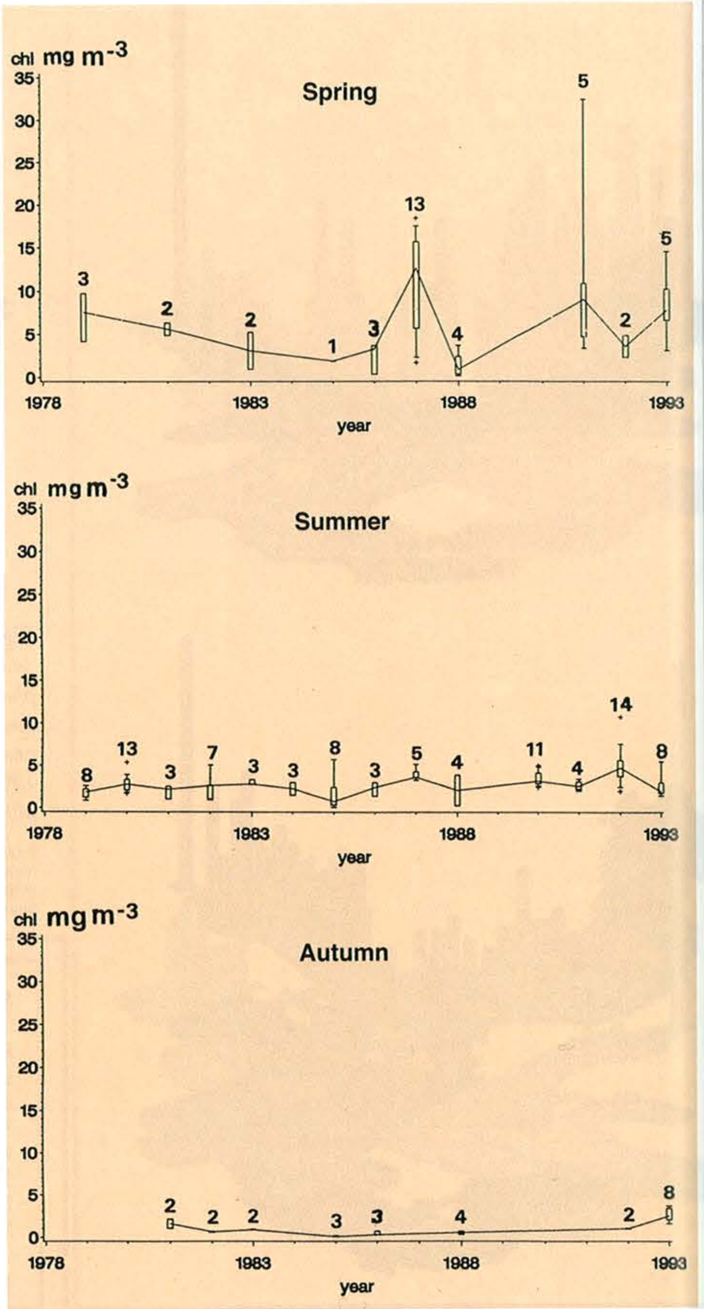
Fig. 4.2.9 Annual variations of the primary production rate in (in g C m^{-2} day^{-1}) the deep water region of the eastern part of the Gulf of Finland, 1984-90

(Seasonal averages for the region and deviations of local averages are presented)

For the deep-water region of the eastern part of the Gulf (*R1-R4*, cf. Fig. 4.2.1), data was collected and analysed 1-2 times a month during May-October by the State Hydrological Institute (St. Petersburg, Russia) in 1983-90. Subsequently, those surveys were carried out only 1-2 times during the growing season, but with a broader areal coverage. The annual maximum of chlorophyll *a* is built up during the spring bloom, which consists mainly of diatoms (Fig. 4.2.8). Both the averages and extremes were somewhat higher here in 1989-91 than in previous and subsequent years, and this was similar to the development at the station *F3*. The primary production rates display the same increase (Fig. 4.2.9). Compared to the whole period 1980-94, the maximum rates of 0.63-0.98 g C m^{-2} day^{-1} during 1988-94 were pronouncedly higher compared to 0.21-0.45 g C m^{-2} day^{-1} found in 1984-87. This increase coincided with the increase of phosphate concentrations in that region (cf. Fig. 4.2.6a).

Fig. 4.2.10 Seasonal occurrence of chlorophyll a (in mg m^{-3}) in the Gulf of Finland; 1979-93

(The numbers indicate the total number of samplings in each year during the respective season. In case of missing samples during a year, no number has been given.)



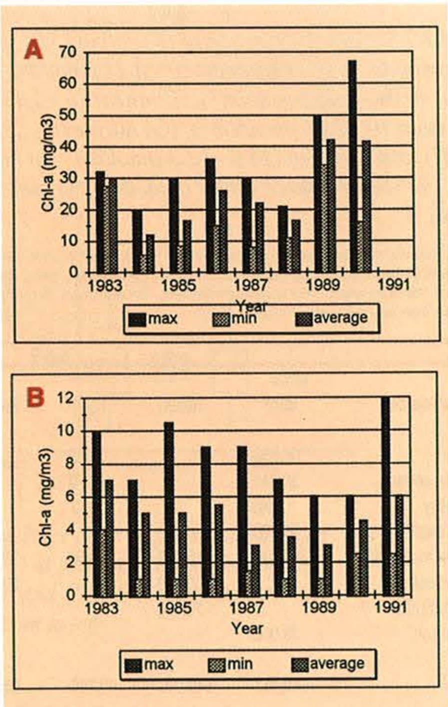


Fig. 4.2.8 Annual variations of chlorophyll a (in mg m^{-3}) in the deep-water region of the eastern part of the Gulf of Finland in May (A) and June-August (B), 1983-91

[1] see ANNEX 11.3 for addresses of authors

Phytoplankton biomass and composition - The occurrence of dominant phytoplankton species is summarised in Table 4.2.2. It reveals normal spring communities, i.e., diatoms and dinoflagellates, with some variability in dominance, but no signs of significant changes. Also the summer communities are as usual flagellate dominated, with a strong influence by blue-green algae. The occurrence of *Achnanthes taeniata* and *Coscinodiscus granii* in the middle period is probably not relevant, but more a sign of early and late sampling dates, respectively. In autumn, *Coscinodiscus granii* is consistently the dominant species, which is known to grow in autumn if calm weather prevails. As its occurrence is influenced by meteorological factors, its high autumn variability is not surprising, and the variability is probably not indicative of environmental changes.

In the deep water region of the eastern part of the Gulf of Finland, the spring phytoplankton consists mainly of the cold and brackish water species *Achnanthes taeniata*, *Chaetoceros wighamii* and *Thalassiosira baltica*, while *Gonyaulax catenata* develops at the end of May and in early June with a biomass of up to 85 % of the total. The observed spring biomass varied in the range of 1.3-12.8 g m^{-3}. The summer/autumn phytoplankton was dominated by *Aphanizomenon flos-aquae, Gomphosphaeria lacustris, Limnothrix planctonica, Monoraphidium contortum, Pyramimonas* spp., *Dinophysis acuminata* and *Actinocyclus octonarius*. During 1982-94, the summer/autumn biomass varied in the range of 0.05-2.5 g m^{-3}.

Table 4.2.2 *Dominating phytoplankton species in the Gulf of Finland during three assessment periods*

(arithmetical mean (maximum) values of species wet-weight biomass in mg m^{-3}; N - total number of samples, n - number of samples in which species were dominating)

1979-83		1984-88		1989-93	
Spring	N=3		N=11		N=5
Peridiniella catenata	1,072 (1,830, *n=3*)	*Peridiniella catenata*	1,537 (5,082, *n=8*)	*Achnanthes taeniata*	1,214 (2,657, *n=4*)
Thalassiosira baltica	299 (504, *n=2*)	*Achnanthes taeniata*	1,132 (5,353, *n=10*)	*Thalassiosira baltica*	526 (1,022, *n=5*)
Skeletonema costatum	254 (510, *n=3*)	*Skeletonema costatum*	308 (1,309, *n=10*)	*Skeletonema costatum*	293 (702, *n=5*)
Peridinium sp.	252 (568, *n=3*)	*Thalassiosira baltica*	121 (762, *n=6*)	*Peridiniella catenata*	244 (507, *n=4*)
Achnanthes taeniata	184 (283, *n=3*)	*Cryptomonas* sp.	80 (707, *n=5*)	*Cryptomonas* sp.	190 (830, *n=4*)
Summer	N=11		N=16		N=20
Cryptomonas sp.	96 (154, *n=9*)	*Coscinodiscus granii*	446 (2,800, *n=3*)	*Aphanizomenon flos-aquae*	184 (1,521, *n=18*)
Flagellata	85 (162, *n=9*)	*Cryptomonas* sp.	395 (3,348, *n=10*)	*Nodularia spumigena*	61 (447, *n=12*)
Aphanizomenon flos-aquae	52 (245, *n=9*)	Flagellata	162 (1,314, *n=12*)	*Cryptomonadales sp.*	47 (355, *n=11*)
Snowella lacustris	34 (376, *n=2*)	*Achnanthes taeniata*	82 (1,301, *n=2*)	*Dinophysis acuminata*	36 (172, *n=16*)
Dinophysis acuminata	22 (101, *n=9*)	*Dinophysis norvegica*	68 (940, *n=5*)	*Snowella lacustris*	23 (126, *n=20*)
Autumn	N=3		N=10		N=9
Coscinodiscus granii	240 (400, *n=2*)	*Coscinodiscus granii*	1,426 (5,600, *n=5*)	*Coscinodiscus granii*	632 (2,640, *n=7*)
Aphanizomenon flos-aquae	75 (140, *n=3*)	*Melosira varians*	948 (9,360, *n=3*)	*Snowella lacustris*	70 (201, *n=4*)
Peridinium sp.	88 (180, *n=3*)	*Dinophysis norvegica*	59 (202, *n=3*)	*Actinocyclus octonarius*	23 (198, *n=2*)
Rhodomonas minuta	19 (36, *n=3*)	*Cryptomonas* sp.	54 (186, *n=5*)	*Thalassiosira baltica*	22 (90, *n=6*)
Flagellata	10 (19, *n=2*)	*Dinophysis acuminata*	53 (492, *n=2*)	*Dinophysis acuminata*	22 (74, *n=4*)

BMP Station:	F1	F3	H1
Anabaena cylindrica	85	85, 87, 88	85, 87
Anabaena lemmermannii	79, 83, 85-87, 91, 92	82, 83, 85, 87-89, 91, 93	85, 86, 88, 91, 92
Anabaena spiralis	92	-	85
Aphanizomenon flos-aquae	79-83, 85-93	79-93	80-93
Coelosphaerium kuetzingianum	86, 88	93	88
Microcystis aeruginosa	-	93	93
Nodularia spumigena	79, 81, 85-87, 89, 92, 93	83-90, 92, 93	83, 85-89, 91-93
Planktothrix agardhii	88	88	88
Chaetoceros danicus	79, 81, 82, 86, 88, 93	79-93	80, 82-90, 92, 93
Dinophysis acuminata	79-83, 85-87, 89-93	79-93	80, 82, 83, 85, 86, 88, 89, 91-93
Dinophysis norvegica	79, 85-87	81, 82, 85-89, 92, 93	82, 83, 85-88, 90, 92, 93
Prorocentrum balticum	85	85, 87	85, 87
Prorocentrum minimum	-	-	89
Scrippriella trochoidea	-	-	83

Table 4.2.3 *Years of occurrence -as 19XX- of potentially harmful and/or toxic algal species in the Gulf of Finland*

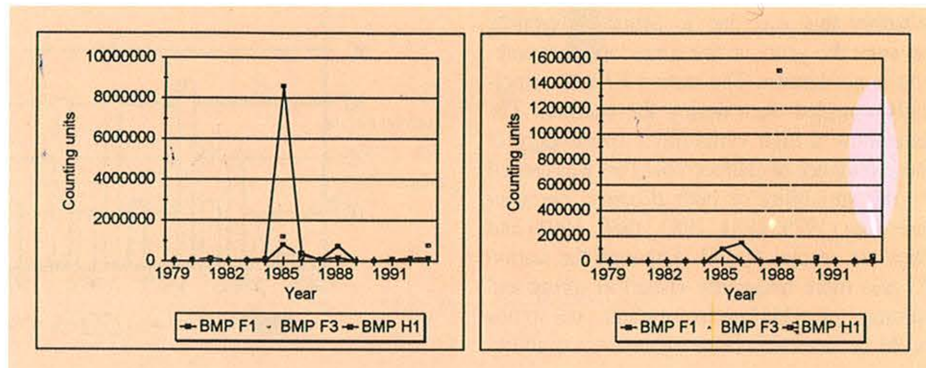


Fig. 4.2.11 Annual maximum abundances of bloom-forming blue-green algae at three stations in the Gulf of Finland, 1979-93
left - Aphanizomenon flos-aquae,
right - Nodularia spumigena

A number of potentially harmful or toxic species were encountered during the sampling programme (Table 4.2.3). Although blue-green algae and dinoflagellates were regular components of the phytoplankton communities, only two blue-green algal species were actually abundant. These were *Aphanizomenon flos-aquae* and *Nodularia spumigena* whose maximum summer abundances are shown in Figure 4.2.11. The occurrence of these two species was highly variable, both spatially and temporally. Although heavy blooms were observed in 1985, 1986, 1988 and 1993, no trends were found. If one ignores the very high value of *Nodularia spumigena* in 1988, both species did not show any difference between the stations. The data from the stations were significantly correlated indicating the similarity of the variability of the blue-green algal blooms.

Unattended monitoring of phytoplankton - Extensive unattended, i.e., automatically collected monitoring data on phytoplankton from the Gulf of Finland are available for 1993 [*559*]. In 1993, the blue-green algal species *Nodularia spumigena* formed large surface blooms in July and August in the western and central Gulf of Finland, especially in the open sea. It is noteworthy that routine monitoring samples contained only low abundances of *Nodularia spumigena* in that year, indicating the inability of the present monitoring programme to reveal actual bloom situations. Near the coast, the blooms were less intense compared to 1992, but local blooms were reported. In the Neva Estuary, heavy phytoplankton blooms prevailed during the whole growth season. In the eastern parts of the Gulf of Finland, two non-heterocytic blue-green algal taxa dominated, *Planktothrix agardhii* and *Oscillatoria* spp. The data on potentially toxic phytoplankton species from 1993 also revealed a great number of potentially harmful species. These consisted of a variety of blue-green algal taxa, some dinoflagellates and small flagellated species, confirming the information gathered from the monitoring data (Table 4.2.3).

4.2.3.2 Zooplankton

The basic interannual variability of the most common crustacean species was always high. Whether this was due to actual differences between the years or just a random phenomenon is not known. The station *F1* was principally sampled once during the summer. The data show a high variability, specifically in the abundance of cladocerans. This was caused by the variability of both *Bosmina coregoni maritima* (1979, 1981, 1983, 1986, 1991) and *Daphnia* species (1991). Although the station *F3* was more frequently visited in spring and autumn, in addition to summer, no trends could be detected. Once again the variability of cladocerans contributed to the summer variability. *Bosmina coregoni maritima* was abundant at *F3* in 1983, 1984, 1988 and 1992. However, *Daphnia* species were abundant only in 1990. This could be due to salinity differences between the stations, or due to the longer distance of the station *F3* to the eastern part of the Gulf of Finland, in which the basically fresh water *Daphnia* species are more abundant. *Acartia* and *Eurytemora* species were very abundant in 1992. The zooplankton biomass at the station *H1* appeared to be generally at a lower level than at the other two stations. However, this may also be a consequence of uneven sampling frequency and uncoordinated population development of zooplankton.

In the deep water region of the eastern part of the Gulf of Finland, the mesozooplankton community consists mainly of the brackish water species *Eurytemora affinis, Bosmina coregoni maritima, Limnocalanus macrurus, Acartia bifilosa* and *Synchaeta baltica*. In 1983-92, the seasonal averages of the zooplankton biomass, covering the range 0.25-0.83 g m^{-3} [*616*], were higher than those found in 1969-73 (0.30-0.47 g m^{-3}, [*625*]).

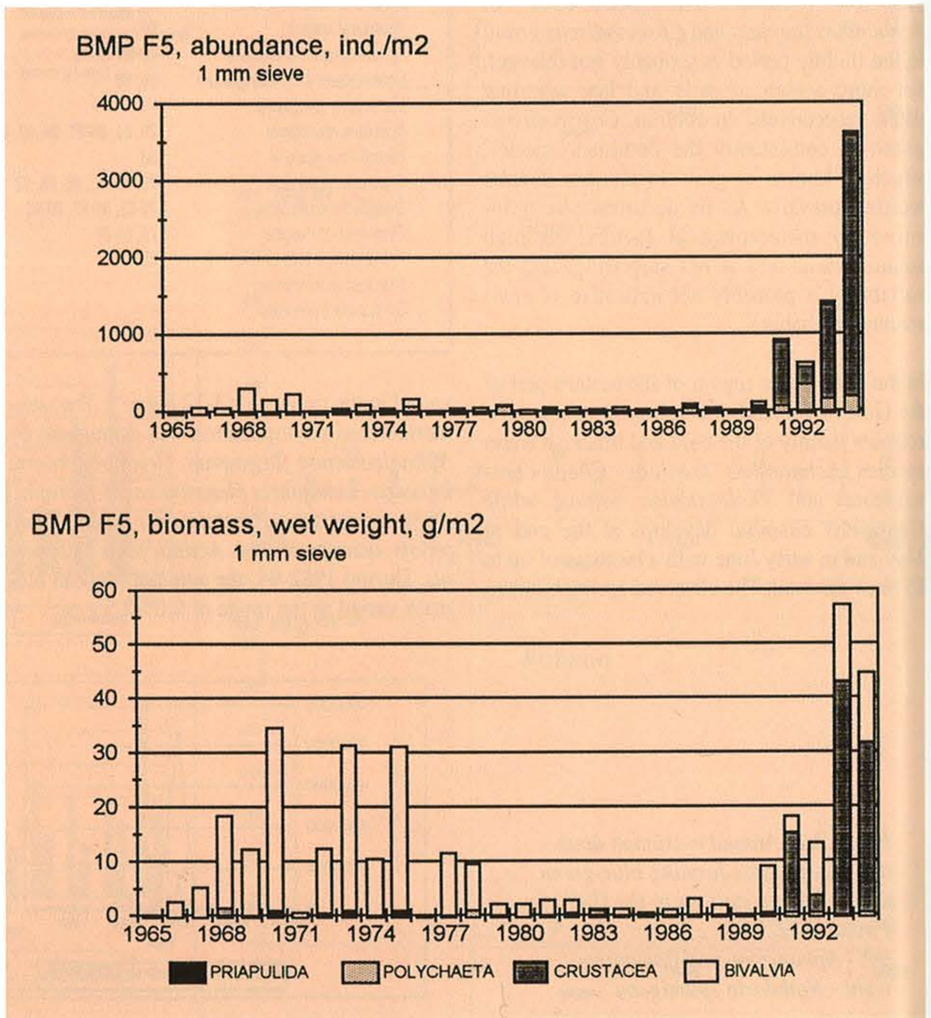
4.2.4 Macrozoobenthos

A.-B. Andersin[1], I. Kotta, A. Maksimov

The assessment of the state of the macrozoobenthos in the Gulf of Finland was based on samples collected from BMP stations *F2* and *F5* by Finland and Estonia. The data were supplemented with Estonian observations from the southern part of the Gulf, and with observations from the eastern part performed by the State Hydrological Insititute, St. Petersburg, Russia.

The assessment period 1989-93 differed strongly from the previous periods as regards benthic bottom fauna in the open sea areas of the Gulf of Finland. The recolonisation of the almost deserted bottoms started at the very beginning of the period. At the end of the period, all of the bottom sediment of the open sea area of the Gulf was inhabited by a dense benthic macrofauna community. In the eastern, somewhat shallower areas, no clear difference was observed beween the periods.

Fig. 4.2.12 *Fluctuations in macrozoobenthos abundance (ind.* m^{-2}*) and biomass (g wet weight* m^{-2}*, mesh size 1 mm) at station F5 (68 m) in the western Gulf of Finland*



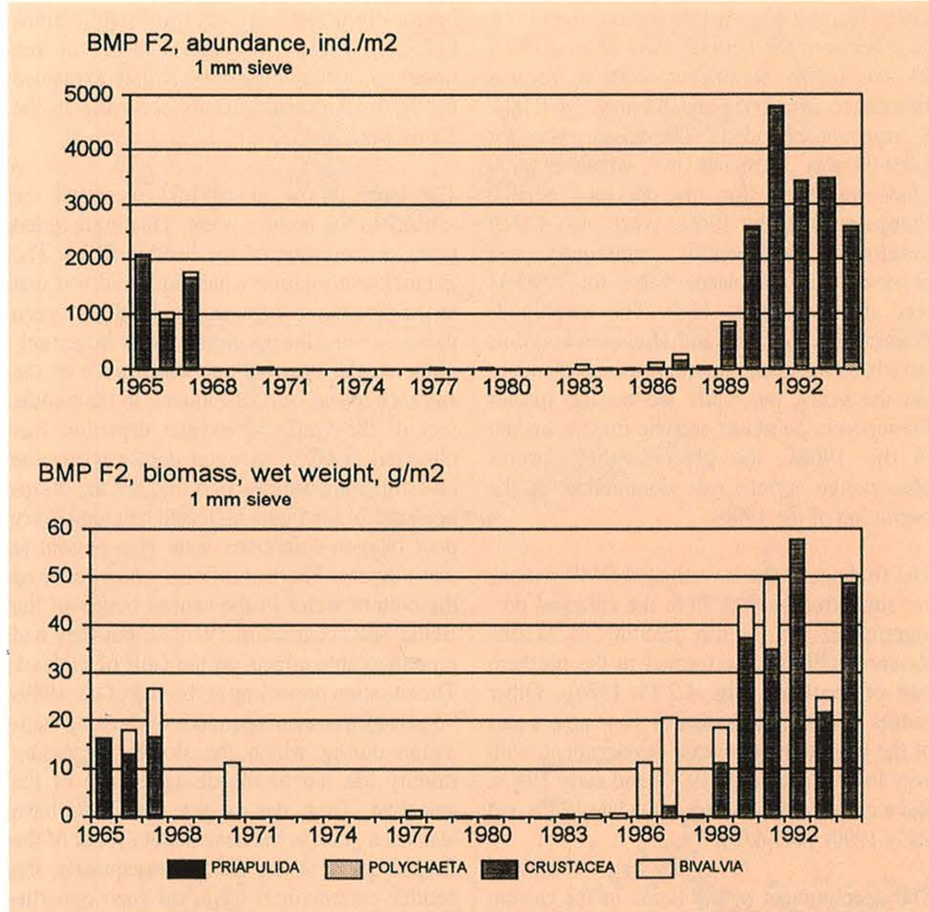
[1] see ANNEX 11.3 for addresses of authors

Station Period	Abundance ind. m^{-2}	Biomass (total) g m^{-2}	Biomass (excl. *Saduria*) g m^{-2}
F5			
1966-70	130 ± *144*	8.66 ± *8.46*	8.21 ± *7.69*
1979-83	39 ± *27*	2.15 ± *0.69*	1.96 ± *0.92*
1984-88	42 ± *26*	1.41 ± *0.98*	1.41 ± *0.98*
1990-93	613 ± *584*	16.22 ± *22.09*	14.46 ± *8.37*
F2			
1965-67	1,614 ± *539*	20.33 ± *5.57*	19.36 ± *3.91*
1979-83	22 ± *35*	0.18 ± *0.20*	0.14 ± *0.19*
1984-88	89 ± *107*	6.71 ± *9.08*	6.71 ± *9.08*
1989-93	3,025 ± *1,449*	40.76 ± *16.64*	23.23 ± *10.03*

Table 4.2.4 *Five-year-period statistics of macrozoobenthos abundance and biomass (formalin wet weight) at stations F2 and F 5 in the Gulf of Finland*

At station *F5* in the western part of the Gulf of Finland, both abundance and biomass values, excluding the occasionally occurring big isopod *Saduria entomon*, were significantly higher during 1989-93 compared to 1984-88 (t-test, *p*=0.04 and 0.01, respectively). The abundance values rose from <100 in 1990 to >3,000 ind. m^{-2} in 1994 (Fig. 4.2.12). The mean abundance value for the period 1990-93 was more than 10 times higher than that for 1984-88. Compared with the 1960s, a period with a rather well developed benthic community, the mean value for 1990-93 was still almost 5 times higher (Table 4.2.4).

During 1989-93, the abundance was strongly dominated by the amphipods *Pontoporeia femorata* and *Monoporeia affinis*, with *P. femorata* slightly more abundant, and with the polychaete *Harmothoe sarsi* which was slightly less abundant than the amphipods. During the previous impoverished period, small *Harmothoe sarsi*, specimens dominated, producing a difference in biomass which was even bigger than the difference in abundance (Fig. 4.2.13, Table 4.2.4). Compared with the 1960s, when the dominating species was the lamellibranch *Macoma baltica*, the mean biomass value of 1989-93, dominated now by amphipods, was only 1.3 times higher (Table 4.2.4).



At station *F2* in the central part of the Gulf, the total abundance increased from <1,000 in 1989 to about 3,000 ind. m^{-2} during 1990-93, with the maximum value of 4,790 recorded in

Fig. 4.2.13 *Fluctuations in macrozoobenthos abundance (ind.* m^{-2}*) and biomass at (g wet weight* m^{-2}*, mesh size 1 mm) station F2 (60 m) in the central Gulf of Finland*

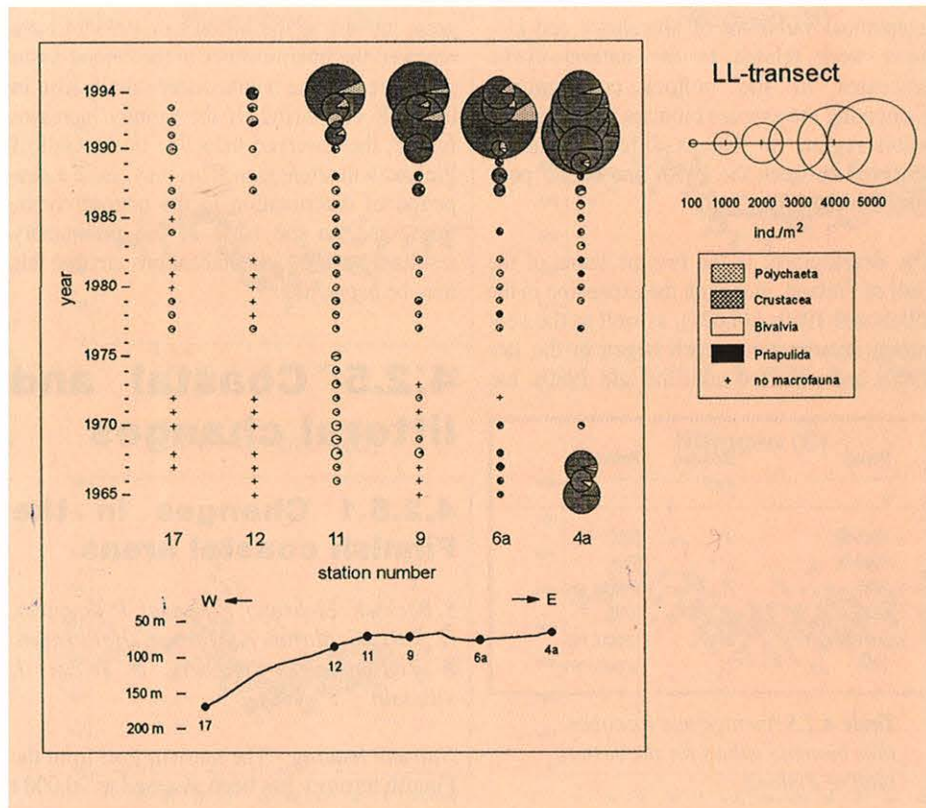


Fig. 4.2.14 *Development of the benthic macrofauna in the Gulf of Finland, 1965-93 [376]*

1991 (Fig. 4.2.13). At this station, the difference between the periods 1984-88 and 1989-93 was highly significant, both as regards abundance (p=0.005) and biomass (p=0.004, *S. entomon* excluded). The mean value for 1989-93 was 3,056 ind. m^{-2}, which is >100 times the mean for the previous period. Compared with the 1960s, when also a well developed macrobenthic community was observed, the abundance value for 1990-94 was about twice as high. The amphipods *Pontoporeia femorata* and *Monoporeia affinis* have been the most important species throughout the years, but while the marine species *Pontoporeia femorata* was the most abundant in the 1960s, the glacial relict species *Monoporeia affinis* was dominating at the beginning of the 1990s.

The findings at the investigated BMP stations are supported by data from the enlarged programme of the Finnish Institute of Marine Research (FIMR), performed in the northern part of the Gulf (Fig. 4.2.15, [*376*]). Other results from the deep areas of the southern part of the Gulf show the same development, with very low values in the 1970s and early 1980s, and a clear recolonisation in the late 1980s and early 1990s [*609,610*].

The development of the fauna in the eastern part of the Gulf has also been monitored by the State Hydrological Insititute in St. Petersburg. The impoverished species composition of the deep water region was observed to be greatly dominated by *Saduria entomon, Monoporeia affinis* and *Pontoporeia femorata*. The bivalve *Macoma baltica* and oligochaetes were found only at depths of less than 30 m. The large interannual variations of abundance and biomass were related to the natural cyclic processes in the bottom communities. Comparing the average biomass values for the whole region, no distinct difference can be observed between the 1990s and earlier periods (Table 4.2.5).

The development in the benthic fauna of the Gulf of Finland, including the expansion in the 1950s and 1960s [*18,621*], as well as the very strong deterioration which began in the late 1960s and persisted until the late 1980s, has been discussed in several publications [*14,15,16,376*]. The main causes for the observed fluctuation can be largely explained by the hydrographic events occurring in the Baltic Sea.

The huge inflow in 1951/52 increased the salinity in the bottom water. This initially led to an oceanisation of the benthic fauna. The gradual deterioration which followed was due to the prevailing stagnation and led to oxygen deficiency and the occurrence of hydrogen sulphide in the near-bottom water, even at the entrance to the Gulf of Finland. In the middle part of the Gulf, no oxygen depletion was observed at 60-80 m water depths where the investigations were carried out, but the disappearance of the fauna indicated that temporary poor oxygen conditions were also present at these depths. The last inflows which renewed the bottom water in the central basins of the Baltic Sea occurred in 1976/77, but they had no remarkable effects on the Gulf of Finland. The situation prevailing at the end of the 1989-93 period was a consequence of the long stagnation during which the slowly decreasing salinity has led to the disappearance of the halocline. Thus, the oxygen conditions have also been good in the near-bottom water of the deepest parts of the Gulf. Consequently, the benthic communities have had good opportunities to develop during that period. However, compared with the abundance and biomass values in the middle of 1960s, the values in the early 1990s were much higher. This may be an indicator of increasing food supply, i.e., eutrophication.

The further development of the fauna is of great interest as the inflows in 1993/94 have renewed the bottom water in the central basin and increased the bottomwater salinity also in the Gulf of Finland. If the salinity increases further, the observed halocline in the Gulf of Finland will strengthen. This may cause a new period of deterioration in the bottom conditions, and on the basis of the preliminary results from 1994, the indications are that this may be happening.

Period	Biomass g m^{-2}	Reference
1965-66	9	[285]
1969-72	27	[358]
1987	27	unpubl. res.
1990	28	[610]
1991-92	25	unpubl. res.
1995	21	unpubl. res.

Table 4.2.5 *Average macrozoobenthos biomass values for the eastern Gulf of Finland*

4.2.5 Coastal and littoral changes

4.2.5.1 Changes in the Finnish coastal areas

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Nutrient loading - The nutrient load from the Finnish territory has been assessed as 20,000 t yr^{-1} of total nitrogen and 890 t yr^{-1} of total phosphorus [*545*]. From the nitrogen inputs, 12,000 t yr^{-1} (65 %) are more or less immediately available for primary producers, while for phosphorus the corresponding figure is only 330 t yr^{-1} (45 %). Municipalities and agriculture, the primary sources of nitrogen, are respectively responsible for about 40 % and 50 % of the anthropogenic load. In the case of bioavailable phosphorus, the anthropogenic inputs from agriculture, municipalities and industry contribute about the same amount.

The level of phosphorus inputs to the Gulf of Finland has clearly decreased during the late 1980s and the early 1990s, mostly due to the extensive introduction of the activated sludge treatment method into the pulp and paper industry. But also phosphorus input from municipalities has decreased. On the other hand, in the nitrogen loading either from point or diffuse sources, no substantial changes have been observed.

Eutrophication - According to studies on nutrient and production dynamics of the Gulf of Finland [*333,546,648*], nitrogen is the primary limiting nutrient both in the western and eastern parts of the Gulf. Only the easternmost Gulf, the Neva Estuary, is either P-limited or equally N- and P-limited. This means that the reduction of only the P-inputs into the Gulf would not have led to significant decreases in productivity in coastal waters. In accordance with this, the monitoring data did not show any significant chlorophyll *a* trends for the outer coastal waters during 1979-93 (Fig. 4.2.15a,b, [*328*]). On the other hand, the development of eutrophication, which was evident in the open and coastal waters during the 1970s and 1980s [*189,539,544*], seems to have now stopped.

The only area where the degree of trophy has clearly decreased during the recent years is the archipelago zone off Helsinki. Here, since the late 1980s, the purified waste waters have been discharged at more offshore locations instead of previously to the inner bights. In the other areas, no clear changes have been taking place since the early 1980s (Fig. 4.2.15a,b).

The eastern archipelago off Kotka is the only area where summer chlorophyll *a* concentrations in extensive areas regularly exceed 5 mg m^{-3} [*328,548*]. In the near-coastal waters, this is caused by the nutrient inputs from local sources, such as pulp and paper industry, municipalities, agriculture and fish farming. In the outer coastal waters, however, the large nutrient load from the river Neva and from the St. Petersburg Region has led to increases in the nutrient concentrations and the degree of trophy (Fig. 4.2.16, [*545*]), and this phenomenon is strongest during the spring bloom. However, during summer, the Neva Estuary and the easternmost Gulf effectively retain, by sedimentation, and remove, by denitrification, the inputs of nutrients, and thus the effects on the Finnish coastal waters are quite small. High spring values of nutrients in the Virolahti area are due to outflow from the river Neva, while the increases in nutrient concentrations off the Helsinki and Loviisa areas arise from inputs from the rivers Vantaa and Kymi.

Most of the large pool of available nutrients is stored in the deeper water layers below the pycnocline [*548*]. However, because of the rather unstable physical stratification, these nutrient resources can reach the productive layer during storms and other mixing events. As a result, blue-green algal blooms and high chlorophyll *a* concentrations, >10 mg m^{-3}, are also common in the open coastal waters of the eastern Gulf, especially during the late summer season [*327*].

Littoral changes - Not only the pelagic but also the littoral ecosystem has been affected by the increased nutrient load into the Baltic Sea. Changes in the phytobenthos became evident in the late 1970s [*193,318,361*]. These changes in the coastal biotic communities became evident not only in locally polluted waters but also on a larger scale [*684*].

An extensive decline in *Fucus vesiculosus* occurred at the Finnish coast in the late 1970s. The amount of filamentous algae increased greatly and *F. vesiculosus* belts disappeared almost completely from the sheltered archipelago shores [*315*]. The decrease was rapid for both the occurrence and the average size of the plant. In the late 1980s, the recovery of *F. vesiculosus* occurred quite rapidly in the upper part of the littoral zone. [*316*]. This recovery has continued in the 1990s, and the *F. vesiculosus* belts are now vigorously, though limited in area, growing at depths of 0.5-1 m at sheltered shores and at 0.5-3 m at open shores of the Gulf of Finland. Single plants can now be found at maximum depths of 5-6 m [*52*]. Earlier, *Fucus vesiculosus* has been reported to grow at depths around 10 m in the Tvärminne archipelago [*557*].

[1] see ANNEX 11.3 for addresses of authors

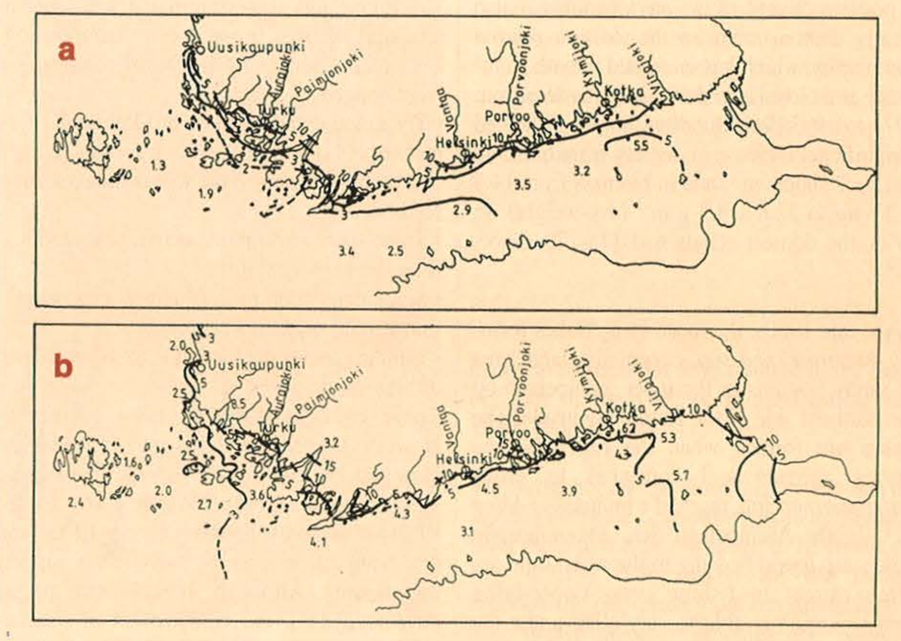


Fig. 4.2.15 *Chlorophyll a in the Gulf of Finland, 1979-83 (a) and 1989-93 (b) [328]*

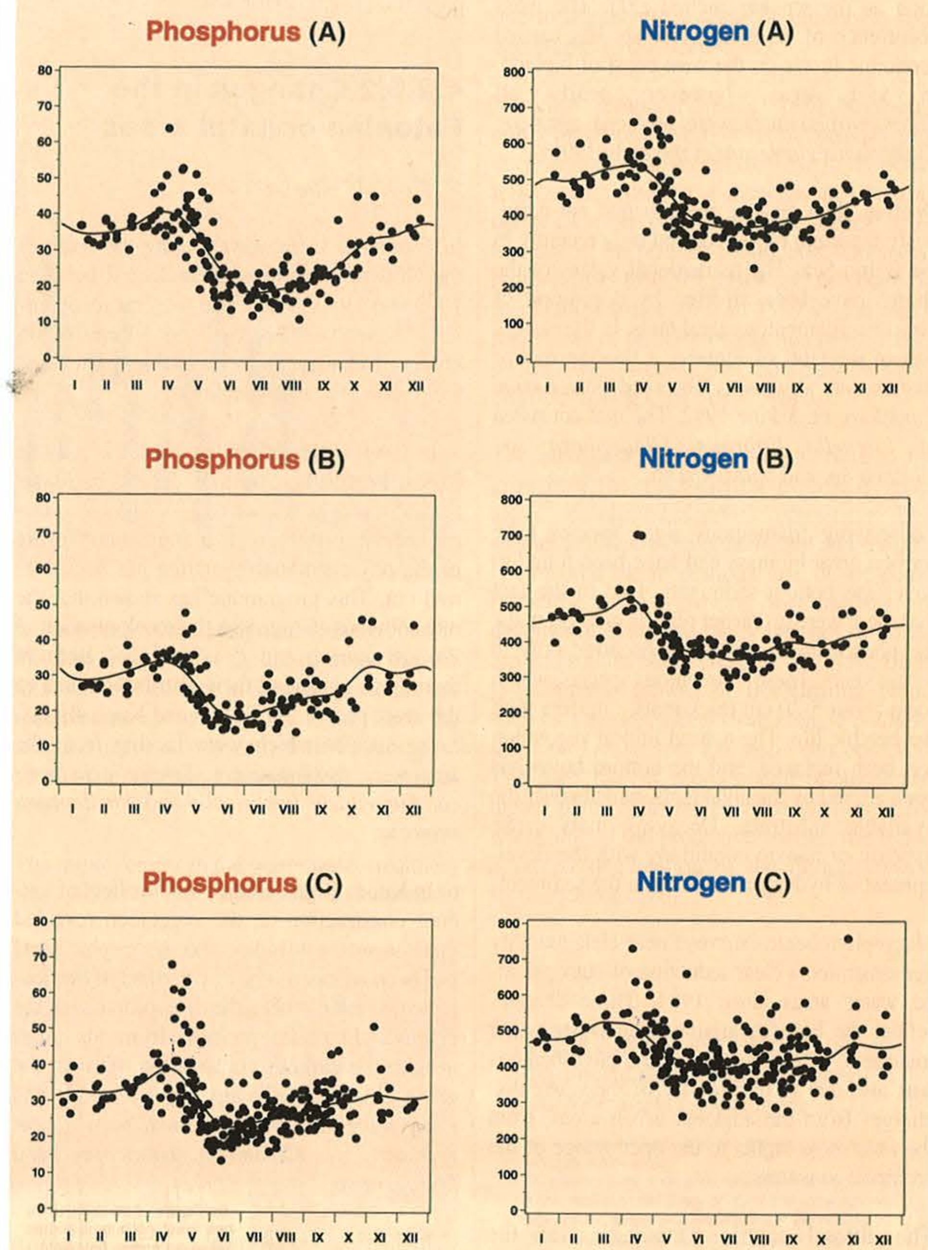


Fig. 4.2.16 *Seasonal variations of total phosphorus and total nitrogen (in µg dm⁻³) in the 0-10 m layer at three Finnish intensive stations, off Virolahti (A), Loviisa (B), and Helsinki (C), in the coastal waters of the Gulf of Finland, 1987-95*

(unpubl. data, Finnish Environment Institute)

A positive change of the phytobenthos is also clearly demonstrated on the *Zostera marina* community, which has increased in both abundance and biomass in the Tvärminne area from 1971 to 1993 [*98*]. Random sampling showed a significant increase in density from 6 ± 2 up to 17 ± 7 shoots m^{-2}, and in biomass from 14.3 ± 3.5 up to 22.8 ± 4.3 g m^{-2} (dry-weight). In 1993, the densest stands had 115-170 shoots m^{-2}.

In the late 1980s, the loose-lying flakes forming *Enteromorpha* spp. occupied a large area of sandy bottoms in the inner archipelago on the northern side of the Hanko peninsula. The flakes are formed when the *Enteromorpha* thallus spreads and continues to grow. *Enteromorpha* spp. reached a biomass of 440 g m^{-2}. In the Archipelago Sea, *Enteromorpha* flakes are found floating in the water, and are often caught in fishing nets. Loose-lying *Enteromorpha* is able to stay alive under the ice and is ready to exhibit vigorous growth as soon as the ice has melted [*51*]. The mass occurrence of *Enteromorpha* sp. has caused economic losses on the west coast of Finland. In vast areas, however, nearly all *Enteromorpha* mats were replaced again by *Cladophora glomerata* in the early 1990s.

Drifting algal mats, a phenomenon not previously reported, have occurred only recently in the Baltic Sea. The recreational values of the shores have been affected by deposition of decaying filamentous algal mats. In the eastern part of the Gulf of Finland, a floating mat of filamentous algae was observed near Kotka, Einonkari, on 3 June 1992. The mat consisted of *Pilayella littoralis*, *Cladophora* sp., *Ulothrix* sp. and *Spirogyra* sp.

Loose-lying filamentous algae species have formed great biomass and have been found to cover the bottom sediments. Due to the fact that there were no earlier reports on algal mats, the phenomenon must have occurred only in recent years. These filamentous species, which form about 5-20 cm thick mats, interfere with the benthic life. The natural littoral vegetation has been replaced, and the bottom fauna has been altered by the algal mats, which cover the available substrata. Decaying mats cause hypoxic or anoxic conditions with the development of hydrogen sulphide in the sediment.

Macrophytobentos surveys near Helsinki City demonstrated a clear reduction of eutrophicated water areas since 1984. These changes reflect the benefits arising from waste water treatment. The recovery of the phytobenthos was also due to the transfer of the waste discharges from the adjacent urban areas, from the innermost bights to the open fringe of the archipelago waters.

The littoral fauna communities of the Tvärminne area showed remarkable long-term temporal changes in the species composition. Four main changes in the faunal composition were noticed:

- The amount and biomass of filter feeders increased, both in relation to a certain area unit and in relation to the *Fucus* biomass they feed on.
- Some species of marine origin benefited from the higher salinity.
- Some deposition feeders and species using filamentous alga also benefited.
- Limnic species declined due to the presence of high saline water.

These changes occurred most noticeably between the early 1970s and early 1980s. However, in the middle of the 1980s, the species composition recovered partly to the situation that prevailed in the early 1970s, and observations made in the early 1990s support this finding. Although most limnic species have recovered, the composition of marine-origin species has not returned to the situation prevailing in the 1970s.

4.2.5.2 Changes in the Estonian coastal areas

H. Kukk[1], G. Martin

In Estonia, a large survey along the coast of the Gulf of Finland was conducted between 1970 and 1980. Since then, several local surveys have been carried out. Reports on observed changes were available from three areas along the Estonian coast:

a) In 1984, in the Tallinn Bight, at the coast of Viimsi Peninsula, a serious decline of *Fucus vesiculosus* was registered [*362*]. As a consequence, since 1991, regular annual monitoring of the phytobenthos vegetation has been carried out. This programme has shown that the most obvious change was the recolonisation of *Zostera marina* and *F. vesiculosus*. Both of them have expanded their distribution area in the inner part of the Tallinn and Kopli Bights. Some species which were lacking from the area, were also found, e.g., *Sphacellaria arctica*, *Furcellaria lumbricalis* and *Phyllophora truncata*.

b) In Kunda Bight, studies on the effect of harbour construction on the vegetation revealed that the water turbidity and the trophic level had been changed. The direct effect of the ecological stress has been the disappearance of the epiphyte *Elachista fucicola* from the areas near to the harbour construction. Also in the green algal belt, *Cladophora glomerata* and *Enteromorpha intestinalis* have been almost replaced by *Ceramium tenuicorne* and *Spirogyra* sp.

c) In the Hogland Island area, green and brown filamentous algae have increased. The distribution area of *F. vesiculosus* and other community-forming algae have remained unchanged [*364*].

Since 1995, similar surveys on macrophytes as those conducted in the vicinity of the City of Helsinki, have been carried out in the coastal waters of the Tallinn Bight. The anthropogenic effect on macrophytes is most pronounced near the harbour and the storm water outlets of the old town. Only temporary effects were recorded in the vicinity of the central waste water purification plant. The recreational areas of Kakumäe were intact and showed either no change or only slight signs of a trophic change.

4.2.5.3 Changes in the Russian coastal areas

I. Shpaer[1], O. Savchuk, N. Silina, S. Makarova, I. Telesh

The area under consideration includes the nearly fresh water Neva Bight with a predominant salinity of about 0.07 psu, and the adjacent shallow region where the salinity is very variable, ranging from 0.1-0.8 psu at the surface to 3-6.8 psu in the westernmost bottom layers. The monitoring started in 1968, but it is only from 1978 that the monitoring results are considered to be reliable. The State Hydrological Institute had made hydrobiological observations during 1982-95.

More than 80 % of the total nutrient and organic inputs from Russia to the Eastern Gulf of Finland are discharged to the Neva Bight, and consist of riverine inputs and effluents from the waste water treatment plants, whose outlets are situated close to the Neva Mouth. The overall high nutrient levels (Fig. 4.2.17), and a possible decrease of the water exchange due to the construction of the barrier, are probably the main factors responsible for the development of a special phytoplankton community in the shallow phosphate limited region of the outer Neva Bight, as compared to both the inner Neva Bight (Fig. 4.2.18) and the deep water region of the eastern part of the Gulf of Finland. In the outer bight, the summer blooms of blue-green algae, such as *Microcystis aeruginosa*, *Aphanizomenon flos-aque*, *Planktothrix agardhii*, with biomasses of 2-11 g m^{-3} are quite common [*548,616,617*].

During the 1980s, waste water treatment plants and the St. Petersburg flood protection barrier were built. The 1986-92 period (Fig. 4.2.18), therefore, represents the situation when the construction of the barrier had been nearly completed. The water gates and the 1 km wide opening south off Kotlin Island were still open, but the northern waste water treatment plant had been put into operation.

[1] see ANNEX 11.3 for addresses of authors

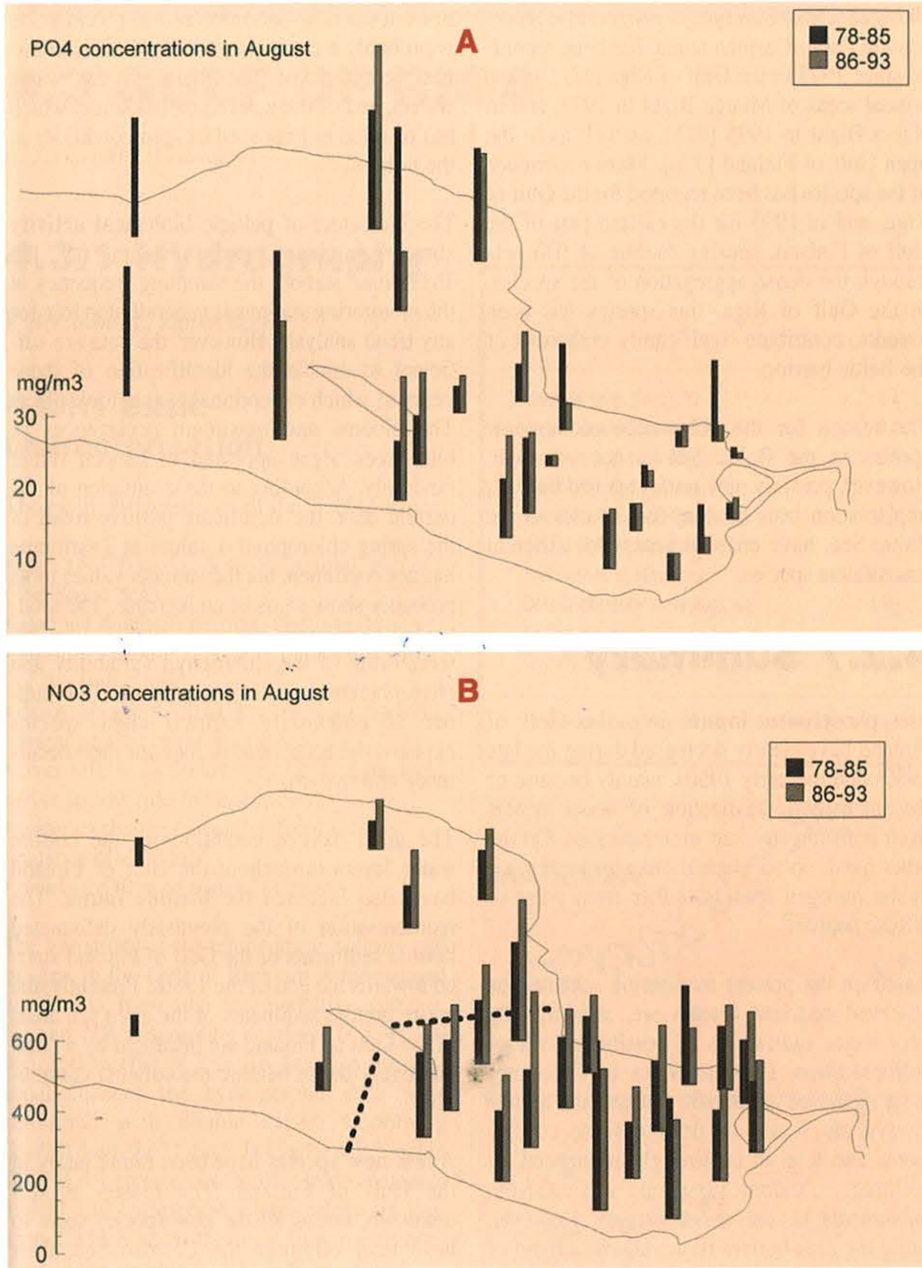


Fig. 4.2.17 Phosphate (A) and nitrate (B) distribution in August; in the outer Neva Bight and in the eastern shallow region of the Gulf of Finland, 1978-85 and 1985-93 (median values, in µg dm^{-3})

The purification of discharges to the Neva Bight resulted in a 30-70 % (depending on the season and sampling position) decrease of the labile organic matter, as BOD_5, and of the ammonium concentrations. In contrast, however, the nitrate concentrations were found to increase. Since the same increase has been observed also in the river Neva upstream of St. Petersburg, one possible reason for the increase in nitrate input from the drainage area might be the increased agricultural activities in this area. However, it should be noted that the total nitrogen content in the coastal waters has remained at about the same level, since the increase in nitrate concentrations was compensated by a decrease in ammonium concentrations.

The improvement in the waste water treatment was probably the reason for a substantial decrease of bacterial development in the eastern part of the Gulf of Finland during the 1980s (Fig. 4.2.19).

The influence of the flood protection barrier on the phosphate concentrations in the surface layer close to the barrier seems to be small (cf. Fig. 4.2.17). The reason for the strong decrease in phosphate concentrations in the deep water layer is probably the reduced import of phosphorus from the west, which is due to a general decrease of phosphate concentrations in the deep layers.

No clear temporal trends could be found, either for chlorophyll *a* or for the primary production rates. The recent increase in the water transparency due to a decrease in dredging operations has not resulted in the expected increase of the primary production, which currently ranges from 0.2 to 0.4 g C m^{-2} day^{-1}. In the shallow water area, in 1989-91 the maximum rates of primary production of 0.51-0.73 g C m^{-2} day^{-1} were higher than those in 1984-88 (0.25-0.47 g C m^{-2} day^{-1}, [*616*]).

The increasing eutrophication stimulated the growth of the macrophytes, especially green algae, mainly *Cladophora glomerata*. This species is already rather abundant and seriously impaired the recreational use of the coast. Particularly thick-growing macrophytes are developing along the sandy beaches of northern and southwestern coasts of the Neva Bight [*10*]. Intense growth of free-floating plants such as *Ceratophyllum demersum*, *Fontinalis* spp. and *Lemna* spp. has also been observed. The high rate of reproduction of algae and higher plants with increased stagnation of the water, due to the protection barrier, has led to a further degradation of the littoral area at some locations [*76*].

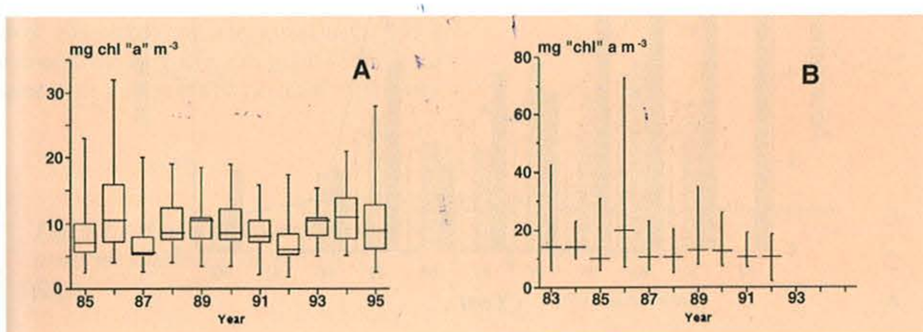


Fig. 4.2.18 Box-and-whisker plots on the annual chlorophyll a variations in the Neva Bight (values in mg m^{-3})

A) all measurements from the growth period in the shallow water area,
B) mean values and range for June-August
(The central boxes cover 50 % of the observations, the horizontal central lines represent the median and the vertical lines extend to minima and maxima, respectively [405])

4.2.6 Alien species

A. Laine[1]

In 1990, the polychaete *Marenzelleria viridis*, originating from North American estuaries, was found for the first time in the Finnish coastal waters, i.e., in the Hanko area. Since then it has been recorded in the entire Gulf of Finland and also in some areas of the Bothnian Sea. However, in contrast to the southern parts of the Baltic Sea, the abundance and biomass of the species has remained comparatively low, and the species has not been recorded in the open sea. As a selective deposit-feeder, *Marenzelleria* can be a competitor of the dominant soft bottom macrofauna and influence the structure of benthic communities. The species ability to live in deep burrows, down to about 40 cm in the sediment, may also reduce its predation by other invertebrates and fish.

In 1995, the bivalve *Dreissena polymorpha* was recorded in the eastern parts of the Gulf of Finland. The abundance of the species seems to be very low, but it has been found at most places investigated and in all parts of the archipelago. Judging from the large size of the specimens found, *Dreissena* must have been present for several years in the littoral zone of the Gulf of Finland. So far, the species has not been found to co-occur with *Mytilus*, and as a hard bottom filter-feeding bivalve it seems to have occupied an open niche in the Eastern Gulf of Finland. In fresh water areas, *Dreissena* has caused severe problems, e.g., in power plants by strong settlement in the cooling water systems.

The mysid shrimp *Hemimysis anomala*, originating from the Caspian area, has been found in the sublittoral zone of the Gulf of Finland since 1992. It is uncertain when the species was introduced to this area, since it lives in crevices of rocks and boulders and can only be observed by SCUBA-diving. The ecology of the species is poorly known. It has not been found at the southern coast of the Gulf.

The cladoceran *Cercopagis pengoi*, also representing Ponto-Caspian fauna, has been recorded since 1992 in the Gulf of Riga [*511*], in the coastal areas of Muuga Bight in 1992, and in Narva Bight in 1995 [*428*], as well as in the open Gulf of Finland [*376*]. Mass occurrence of the species has been reported for the Gulf of Riga, and in 1995 for the eastern part of the Gulf of Finland, causing fouling of fish nets through the dense aggregation of the species. In the Gulf of Riga, this species has been found to contribute significantly to the diet of the Baltic herring.

The reason for the occurrence of the new species in the Baltic Sea is not yet clear. However, possibly new routes opened by, e.g., implantation tests of new fish species in the Baltic Sea, have enabled access for different macrofauna species.

4.2.7 Summary

The phosphorus **inputs** into the Gulf of Finland have clearly decreased during the late 1980s and the early 1990s, mainly because of the intensified purification of waste waters from both forestry and municipalities. On the other hand, no substantial changes took place in the nitrogen loading, either from point or diffuse sources.

Based on the present monitoring scheme, the observed **nutrient trends** are, at best, only very vague indications of possible effects of eutrophication. Data, in terms of concentrations measured at specific stations, are almost always too scarce to draw definite conclusions, and tend to be strongly influenced by seemingly random variations arising from movements of the water masses. However, using the near-bottom observations, a trend of increasing nitrate concentrations in the Gulf of Finland can be detected, but this increase ended in 1990/91. The reasons for the increasing nitrate concentrations, and its subsequent levelling-off, are not known. For phosphate, the lack of the occurrence of high concentrations in the near-bottom waters in recent years is probably a consequence of both the intensified reduction of phosphates in the waste waters, and of the weakening halocline, which has resulted in improved oxygen conditions at the bottom.

The indicators of **pelagic biological activity** show no clear trends. Except for the Tvärminne station, the sampling frequency at the monitoring stations is generally too low for any trend analysis. However, the data are sufficient to enable the identification of those years in which exceptional events took place. The blooms and maximum occurrences of blue-green algae appeared to happen rather randomly. According to the evaluation of the current data, the significant positive trend of the spring chlorophyll *a* values at Tvärminne has not continued, but the summer values most probably show signs of an increase. The unattended monitoring is a promising tool for the monitoring of the chlorophyll variability and phytoplankton species composition. The number of potentially harmful algal species explains the acute need to monitor their occurrence and dynamics.

The good oxygen conditions in the bottom water layers throughout the Gulf of Finland have also favoured the **benthic fauna**. The recolonisation of the previously defaunated bottom sediments of the Gulf of Finland started towards the end of the 1980s. Presently, the entire bottom sediments of the open sea areas of the Gulf of Finland are inhabited by a comparatively dense benthic macrofauna community.

A few **new species** have been found lately in the Gulf of Finland. The reason for this is still unknown, but as all the new species seem to have their origin in the Caspian Sea, their occurrence seems to indicate that a new access has been opened for them by means of man-made activities.

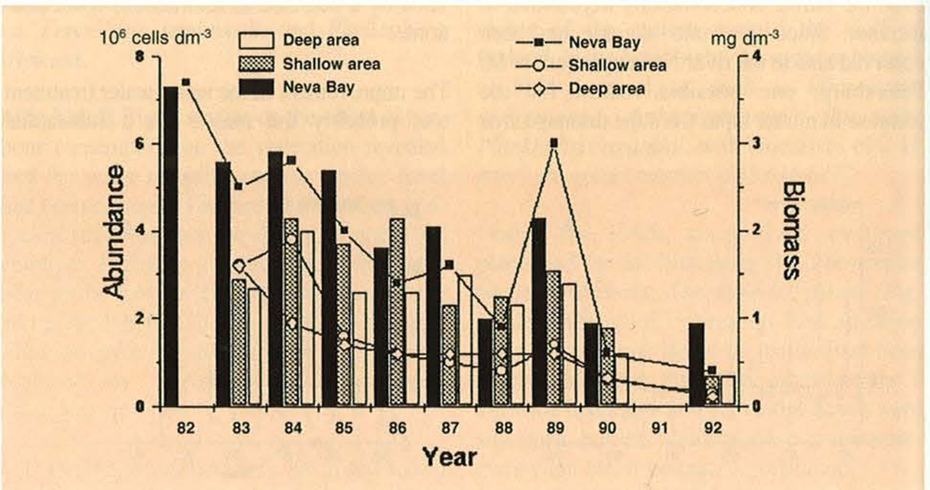


Fig. 4.2.19 Development of the abundance and biomass of bacteria at the eastern end of the Gulf of Finland, 1982-92

('Neva Bight' - Neva outlet to the Kotlin Island flood barrier, 'Shallow area' - Kotlin to the Gulf widening, 'Deep area' - from the Gulf widening to the Gogland/Suursaari islands)

[1] see ANNEX 11.3 for addresses of authors

4.3 GULF OF RIGA

A. Yurkovskis[1] (Convener)

4.3.1 Hydrography

V. Berzinsh[1], *E. Zaharchenko*[1]

4.3.1.1 Basic characterisation

The main characteristics of the Gulf of Riga:

- area 16,330 km^2 (3.9 % of the total area of the Baltic Sea)
- average depth 26 m, max. depth >60 m
- volume 424 km^3 (2.1 % of the total volume of the Baltic Sea)
- drainage area 134,000 km^2 (8 % of the total drainage area of the Baltic Sea)
- river inflow 18-56 km^3 yr^{-1} (7 % of the total river inflow into the Baltic Sea)
- water exchange calculated from salinity balance 60-200 km^3 yr^{-1}
- residence time of water 2-4 years

The variations of the temperature, salinity and oxygen in the Gulf of Riga are demonstrated using data from two representative stations, *G1* and *119*, at sampling depths of 0, 5, 10, 40 and ≥50 m (Fig. 4.3.1). To describe the seasonal changes the hydrographic data were smoothed with double kernel smoothing (k=400 days, λ=31 day). In Figure 4.3.2 and in Table 4.3.1 some characteristic hydrometeorological and hydrographic parameters are presented which show the most important variations in the periods which were compared.

The climatic conditions for the assessment period may be characterised as "Atlantic", i.e., increased air temperature, high wind velocity (especially in winter months) and increased river run-off. The mean air temperature in Riga was 1.6 °C higher than the long-term mean. The highest positive deviations (ΔT = +4.4 °C) were observed between December and March. The deviations were much lower in April and May (ΔT = +1.5 °C) and close to zero for summer and autumn. In general, the river run-off during the assessment period exceeded the long-term mean. However, at the same time a pronounced decrease of both the

Table 4.3.1 Characteristic changes of basic parameters between the present and previous assessment period at stations G1 and 119

Parameter	II) 1979-88 Mean	II) 1979-88 ± s	III) 1989-93 Mean	III) 1989-93 ± s	III) - II)
Yearly air temp. (Riga), *°C*	6.4		7.6		+1.2
Air temperature (Jan-Apr), *°C*	-1.8		2.7		+4.5
Water temp. (0-10 m; May), *°C*	5.6	3.06	7.8	2.43	+2.2
Water temp. (≥ 40 m; May), *°C*	0.6	0.44	2.2	0.83	+1.6
River run-off (Gulf of Riga), *km^3 yr^{-1}*	35.0		37.1		+2.1
Salinity (0-10 m; May), *psu*	5.34	0.53	5.20	0.22	-0.14
Salinity (≥ 40 m; May), *psu*	6.53	0.43	5.93	0.27	-0.60
Density difference* (May-Aug), $\Delta\sigma_t$	1.58		1.33		-0.25
Oxygen saturation (0-10 m; Aug), %	99.4	18.1	101.1	8.0	+1.7
Oxygen saturation (≥ 40 m; Aug), %	45.0	18.1	56.2	10.3	+11.2

* Difference between the upper (0-10 m) and the near-bottom layer (≥ 40 m)

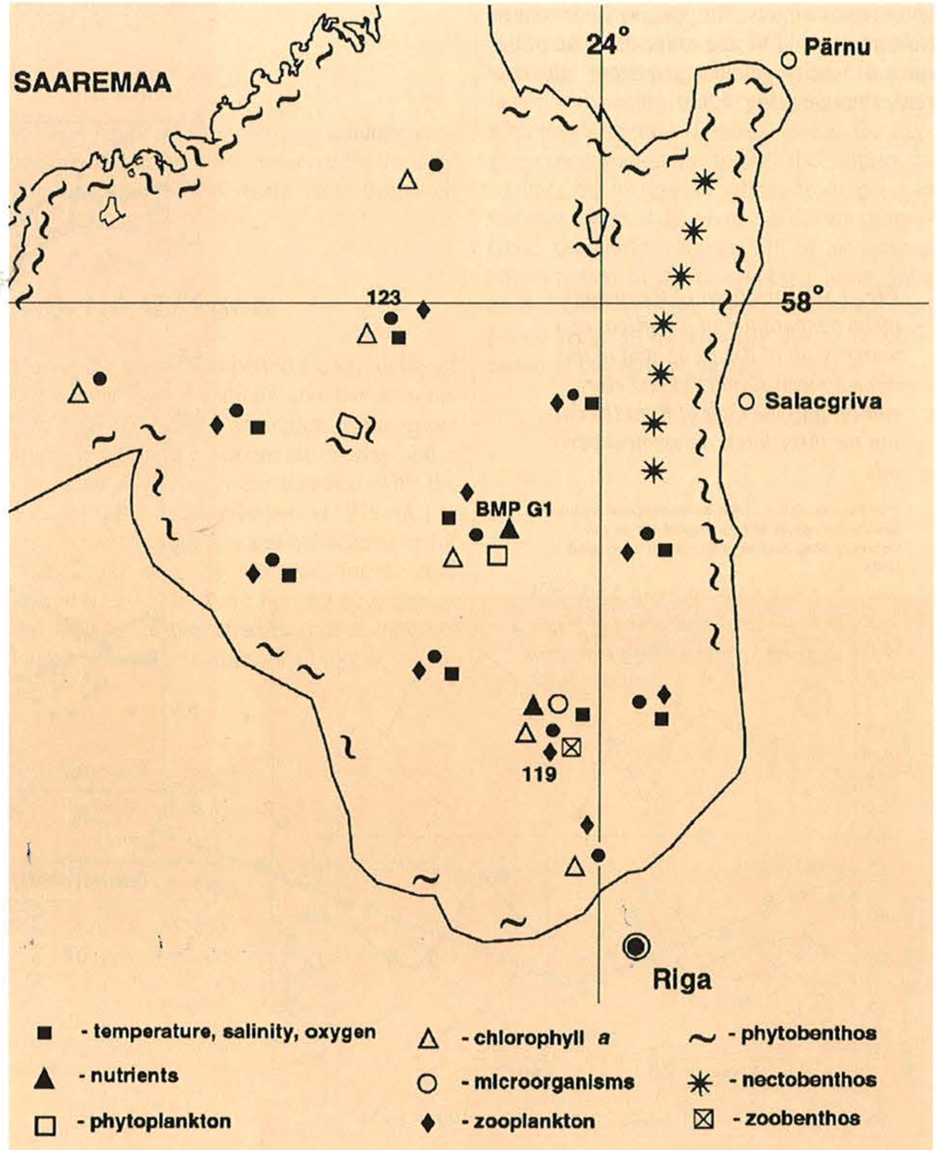


Fig. 4.3.1 Map of the stations and areas for monitoring in the Gulf of Riga

[1] see ANNEX 11.3 for addresses of authors

river run-off and air temperature was observed for the period 1989-93 (Fig. 4.3.2).

4.3.1.2 Temperature

The shallow depth of the Gulf of Riga results in complete vertical mixing during the winter and in a minor inertia of the water mass temperature. This is confirmed by a good coherence between the mean water and air temperatures, with a correlation coefficient of 0.92 for the yearly data of 1979-94 (Fig. 4.3.2 a). As a result of the higher air temperature during the assessment period, the water temperature was 0.7-0.9 °C higher and less variable than in the previous period.

For the assessment period, the seasonal course of temperature in the upper layer was close to the long-term mean. In contrast, in the near-bottom layer the temperature was significantly different from the mean value for 1979-88. The highest positive deviations (1.5-2.2 °C) were observed from March to May, but for the near-bottom layer only in the period March-July. This might be explained by the influence of the warm winters. The positive deviations in November could be due to the decrease of the vertical density gradient and more intensive vertical mixing (Fig. 4.3.3).

Fig. 4.3.3 Seasonal variations of the water temperature (°C) in the upper (0-10 m, a) and near-bottom layers (≥40 m, b) of the Gulf of Riga (stations G1 and 119) during the three assessment periods

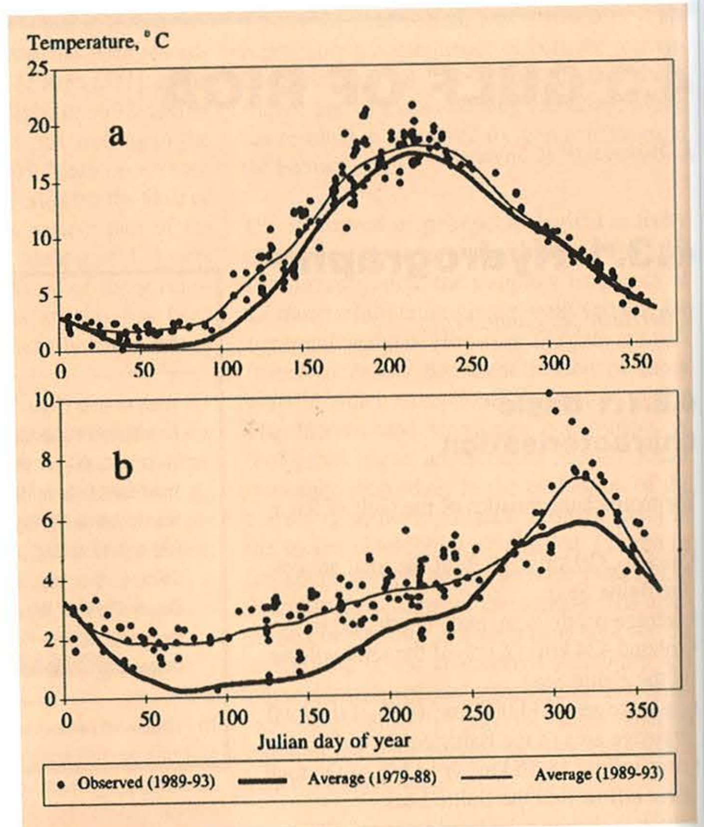
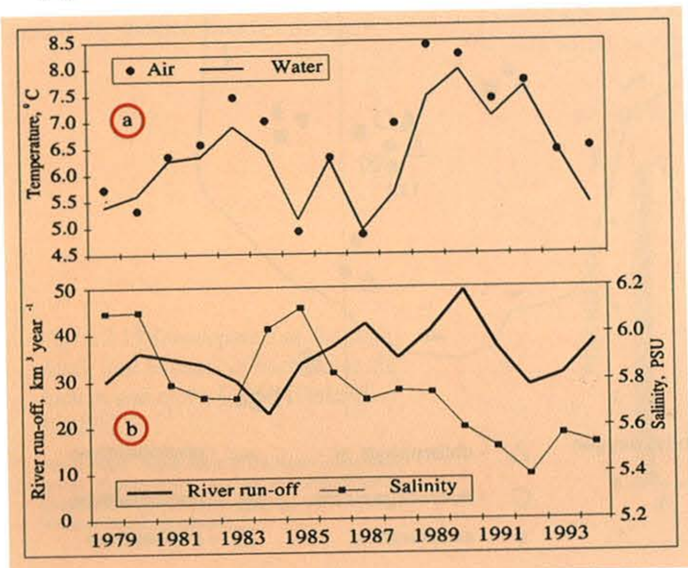


Fig. 4.3.2 Variations of the annual mean temperature of air (Riga) and water (Gulf of Riga - a), and of the annual mean salinity in and river run-off into the Gulf of Riga (b) during the three latest assessment periods

(For the calculation of the water temperature and salinity the results of hydrological surveys in February, May, August and October were used; [74])



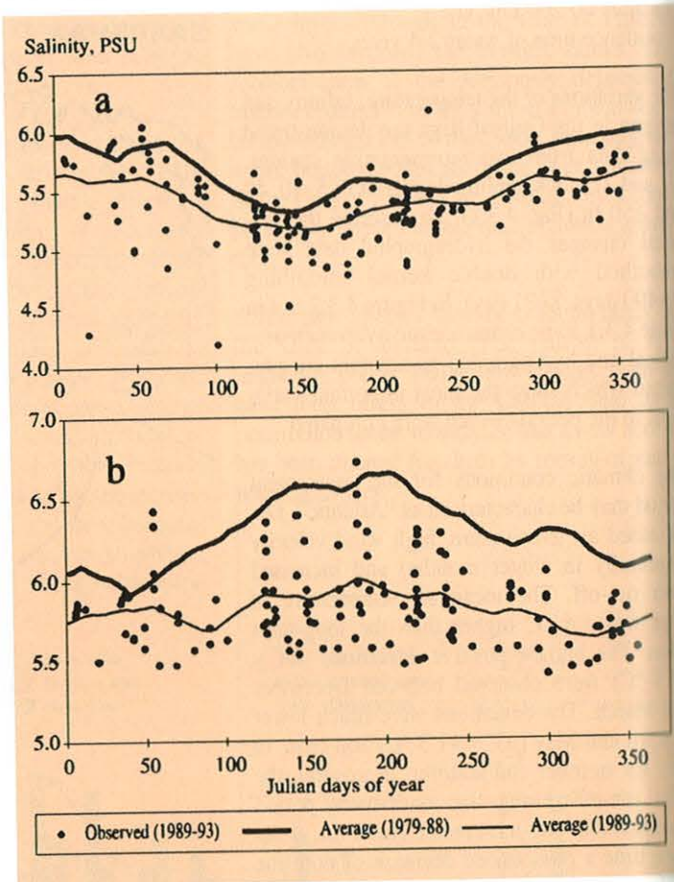


Fig. 4.3.4 Seasonal variations of the salinity (psu) in the upper (0-10 m, a) and near-bottom layers (≥40 m, b) of the Gulf of Riga (stations G1 and 119) during the three assessment periods

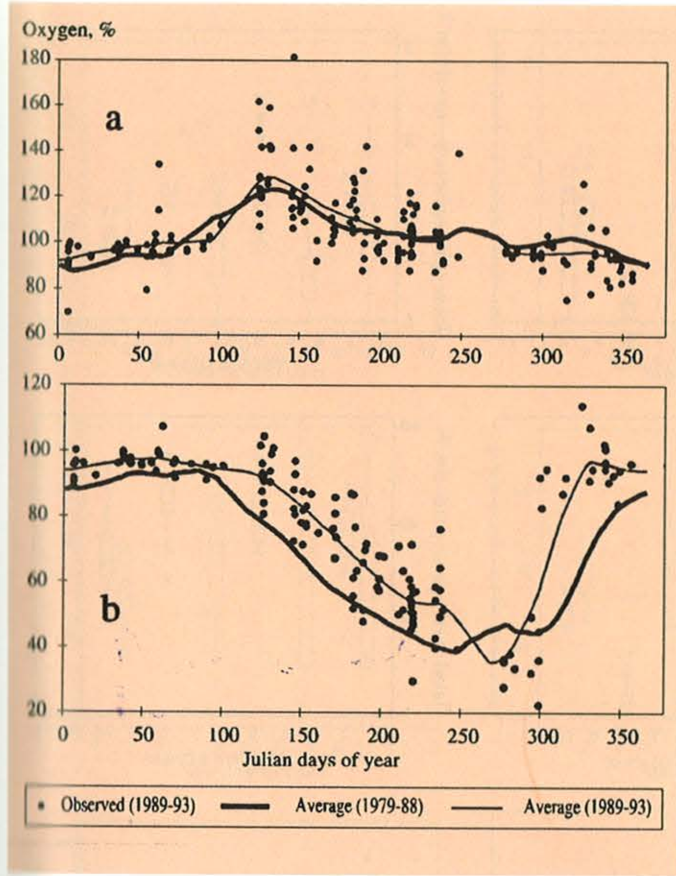


Fig. 4.3.5 Seasonal variations of the oxygen saturation (%) in the upper (0-10 m, a) and near-bottom layers (≥40 m, b) during the Gulf of Riga (stations G1 and 119) during the three assessment periods

4.3.1.3 Salinity

The decrease of the salinity in the Gulf of Riga began in 1978 and continued until 1992 when the lowest mean salinity in the Gulf for the previous 60-65 years was recorded (5.3 psu; Fig. 4.3.2 b). Earlier, such salinity was observed during 1928-31 [*46*]. In the assessment period, the salinity was 0.1-0.6 psu lower and less variable than in the whole 1979-88 period. However, the seasonal course in the upper layer, with maximum values in winter and minimum values in May, was normal (Fig. 4.3.4 a).

Usually, the water masses of the Gulf of Riga are divided from the waters of the Baltic Proper by a hydrological front. At the end of the 1970s, this front was located in the Irbe Strait [*73*]. In the assessment period, as a result of the high river run-off and the decreasing salinity, this front drifted to the west and there was a decrease in the input of more saline waters from the Baltic Proper.

Comparing the periods 1976-80 and 1988-93, there is clearly a westward shift of the hydrological front, e.g., a significant decrease in the horizontal and vertical salinity gradients. Thus, the salinity differences between the Irbe Strait and the Gulf during 1976-80 and 1988-93 were 0.67 and 0.53 psu. During these two periods, the 0-10 m layer differed from the near-bottom layer of the Irbe Strait (20-30 m) by 0.60 and 0.44 psu, respectively. Compared to these temporal variations, no summer maximum of the salinity was observed for the deep layer and the vertical stratification decreased (Fig. 4.3.4 b).

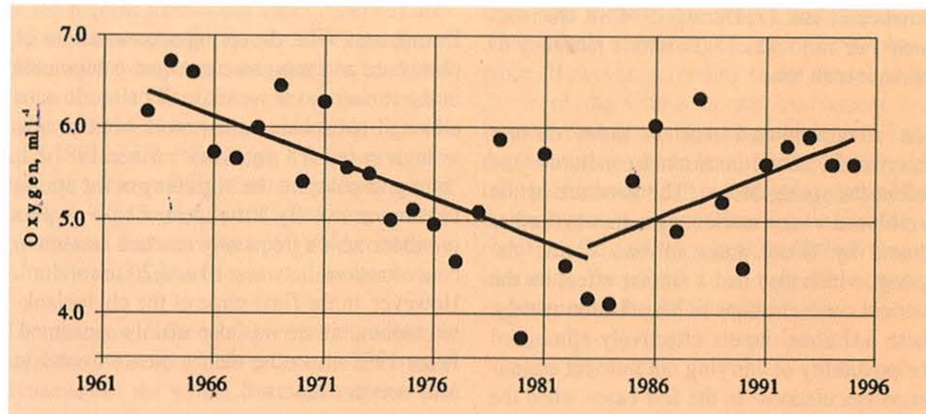
4.3.1.4 Oxygen

During the assessment period the variability of the oxygen concentrations and the seasonal course of the oxygen saturation in the upper water layer, with a maximum in May and a minimum in winter, were identical with the corresponding data for the period 1979-88. For the near-bottom layer, identical behaviour for both periods was only observed for the cold season (December-April; Fig. 4.3.5 a). During the period of vertical stratification of the water masses (May-September), the oxygen saturation in the near-bottom layer was 10-15 % higher than in 1979-88 due to a better vertical mixing (Fig. 4.3.5 b; Table 4.3.1).

The oxygen saturation minimum in late September and early October was for the period 1989-93 about 5-10 % lower than for 1979-88 (Fig. 4.3.5). However, this must not necessarily indicate a real trend because changes were made during the monitoring period regarding the sampling depths.

Until 1985, the oxygen concentrations of the deep water showed for the period 1964-93 a gradual decrease of the summer (August) values. Thereafter, the concentrations increased again (Fig. 4.3.6). The explanation for this pattern can be related to the statistically significant negative correlation (n=76, r=-0.69, p=0.01) between the oxygen concentration and the salinity in the near-bottom layer measured during 1979-93 always in summer (1-21 August). Consequently, the oxygen concentrations below the thermocline seem to be mainly governed by the stratification conditions which can be described by the vertical density gradient and the salinity level in the near-bottom layer. This is contrary to the conclusions from previous studies [*75,542*] which emphasized the anthropogenic impact as the main factor influencing the oxygen concentration whereas the natural variables, as e.g., the oxygen concentration in spring, the salinity in summer, the increase of salinity from spring to summer, were only given secondary importance. Concluding, the growth of the oxygen concentration in the near-bottom water layer of the Gulf of Riga during the assessment period seems to be primarily due to intensification of the vertical mixing.

Fig. 4.3.6 Long-term changes of oxygen concentrations (cm³ dm⁻³) in the deep layer (30-50 m) of the Gulf of Riga (August)



4.3.2 Hydrochemistry

A. Yurkovskis[1], M. Mazmachs

In the **upper layer**, according to the data from *G1* (Fig. 4.3.7), the seasonal cycle of phosphate in 1989-93 was characterised by concentrations of up to 1.1 µmol dm^{-3} in winter and by a sharp drop to <0.1 µmol dm^{-3} in spring. The high winter level of total phosphorus of up to 1.4 µmol dm^{-3} continued during spring, supported by flood water inflow. The summer minimum of total phosphorus was below 0.8 µmol dm^{-3}. The depletion of the 'nitrate' (nitrate+nitrite) winter pool in spring was partly masked by the riverine input. In spring, the concentrations were mainly in the range of 0.2-1.0 µmol dm^{-3}, whereas in winter they were ≤15 µmol dm^{-3}. Maximum levels of total nitrogen were observed in spring and summer. The annual minimum of silicate was found in spring with concentrations in the range of 0.6-1.6 µmol dm^{-3}.

In the **deep layer** (Fig.4.3.8), during 1989-93, concentrations of phosphate and total phosphorus were found to decrease in summer, despite regeneration of the nutrients taking place after the phytoplankton spring bloom. The seasonal patterns of nitrate, total nitrogen and silicate indicate their build-up in the deep water during spring and summer. The seasonal variations of total phosphorus and nitrogen in the deep layer were similar to their behaviour in the upper layer, and this reflects the rather good vertical exchange in summer. A comparison of the seasonal data for the three assessment periods indicated that there have been changes in the Gulf of Riga during 1979-93 in both the amount of nutrients involved in the annual cycles and in the internal structure of the nutrient cycles.

The structure of the nutrient winter pool was found to vary in the long-term. The gradual increase of the NO_3:PO_4 ratio from 19 in 1989 to 28 in 1991 was followed by a sudden drop to 13 in 1992-93. For the period 1989-93, the silicate-to-phosphate ratio was observed to be between 11 and 17. During 1974-88, the inorganic N/P ratio was 13-27 with a tendency to increase with time.

The aforementioned nutrient ratios do not describe the actual nutritional conditions just before the spring bloom. The structure of the pre-bloom winter nutrient system was further altered by flood water inflows during the spring, which also had a similar effect on the nutrient concentrations in May. Unfortunately, these additional inputs effectively eliminated the possibility of carrying out nutrient assimilation calculations. In the few cases when the

[1] see ANNEX 11.3 for addresses of authors

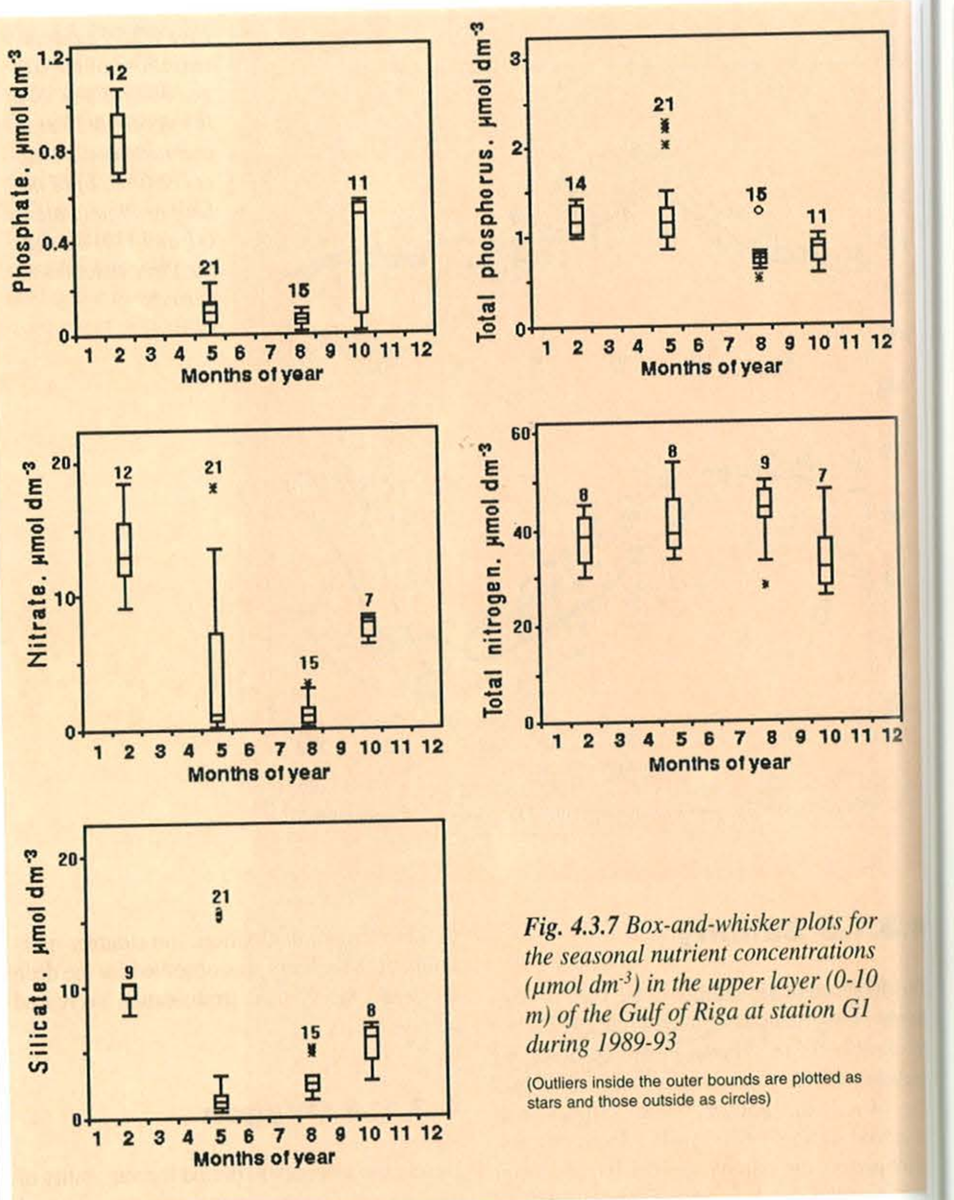


Fig. 4.3.7 Box-and-whisker plots for the seasonal nutrient concentrations (µmol dm^{-3}) in the upper layer (0-10 m) of the Gulf of Riga at station G1 during 1989-93

(Outliers inside the outer bounds are plotted as stars and those outside as circles)

phosphate values in the upper layer were above the detection limit, average Δ N : Δ P ratios of 14.3 for 1983 and 15.2 for 1993 were obtained. These values are close to the *Redfield* ratio for the assimilative nutrient uptake. In 1992, when the phosphate pool was more depleted, increased mean values for Δ N: Δ P (19.4) and Δ Si : Δ P (14.4) were found.

During the 1970s, the spring concentrations of phosphate and inorganic nitrogen compounds in the surface layer were usually close to zero although phosphate values were occasionally as high as 0.2-0.3 µmol dm^{-3}. Since 1981-82, 'zero phosphate' at the beginning of the spring bloom was usually accompanied by a surplus in nitrate which frequently reached maximum concentrations between 10 and 20 µmol dm^{-3}. However, in the final stage of the phytoplankton bloom, nitrate was also mainly consumed. From 1992 onwards, such a nitrate excess in May was not observed.

Beginning in 1991, the silicate spring concentrations began to approach zero. Although this phenomenon had been observed before, it was only seen in a few cases during 1985-86. Thus, during 1989-93, a transition from phosphorus limitation to a simultaneous shortage in phosphorus, nitrogen and likely silicon, took place during the phytoplankton spring bloom. In summer (August) 1988-92, nitrate concentrations of 1-3 µmol dm^{-3} co-existed with zero levels of phosphate in the upper layer. Silicate was not deficient in summer.

To evaluate the long-term changes, the data from winter (February) and summer (August) are used. The winter pool of nutrients represents a pre-spring situation under conditions of relatively low biological activity. The summer pool of nutrients in the deep waters consists of the spring excess of nutrients and regenerated nutrients. To characterise the winter pool of nutrients properly, instead of using only the

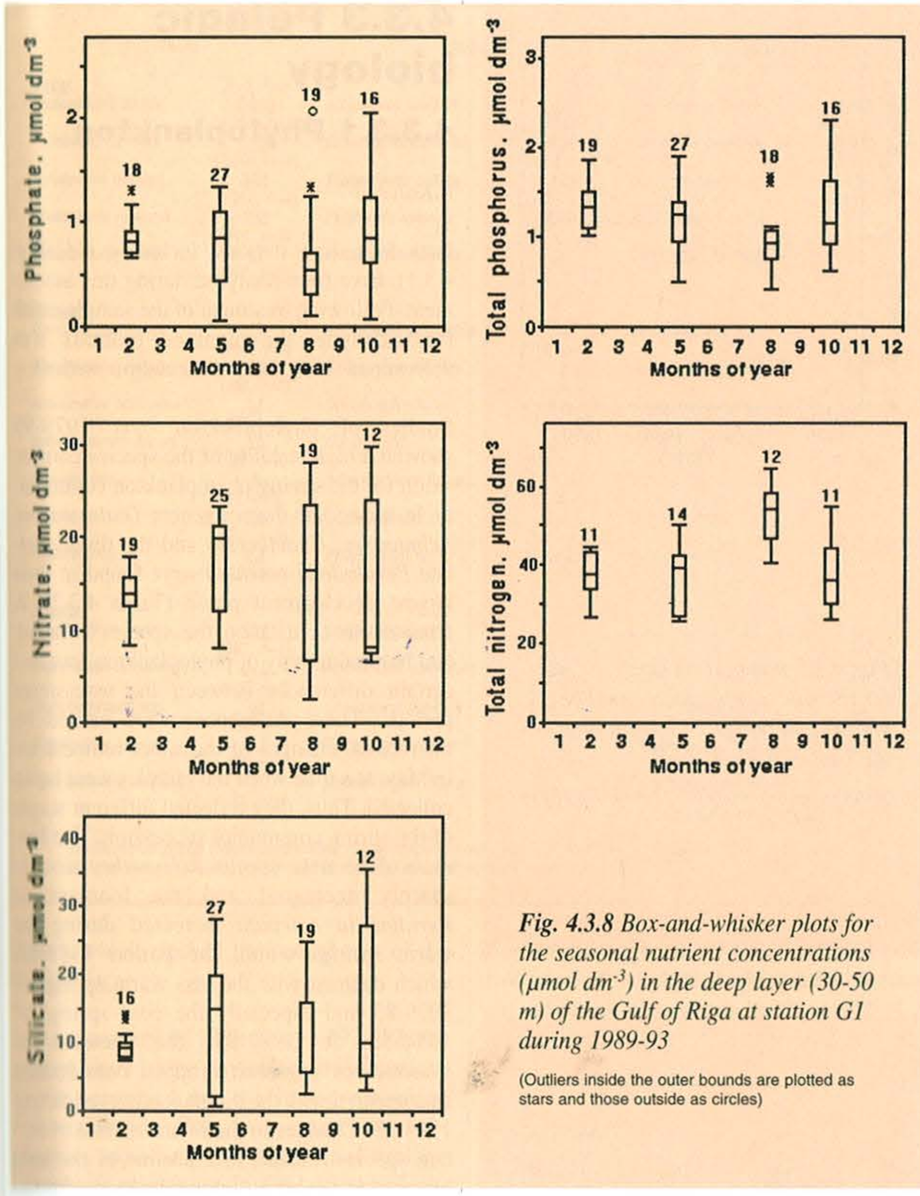


Fig. 4.3.8 Box-and-whisker plots for the seasonal nutrient concentrations (µmol dm^{-3}) in the deep layer (30-50 m) of the Gulf of Riga at station G1 during 1989-93

(Outliers inside the outer bounds are plotted as stars and those outside as circles)

upper layer (0-10 m), the layer 0-40 m was also considered. This should compensate for differences caused by an often observed inhomogeneous distribution of the concentrations. The winter pool of phosphorus showed a growth during 1974-93 with 0.016 and 0.022 µmol dm^{-3} yr^{-1} for phosphate (r^2=0.69; p=0.01) and total phosphorus (r^2=0.67; p=0.02), respectively (Fig. 4.3.9). The long-term increasing trend of the phosphate winter concentrations was found to accelerate during the assessment period, and even higher phosphate values found in 1995 support the exponential character of the curve shown in Figure 4.3.9. About 85 % of the variability of the phosphate values is due to the differences found between a few years. It appears that the upward trend of the phosphate winter pool was governed by anthropogenic sources during the 1970s and 1980s, when a steep rise took place in the economic activities over the drainage area.

Changes in the nitrate winter pool have been even more complicated (Fig.4.3.9). Despite large interannual variabilities, the period of 1974-91 might be statistically described as a time when there was a general growth of nitrate (r^2=0.69; p=0.008). The highest nitrate value in February 1991 could be due to intensive land-based loading in 1990, partly related to the highest fresh water input observed during the assessment period. The subsequent downward trend of nitrate seems to be realistic, as about 45 % of the winter nitrate dynamics over 1974-93 can be explained by the variability of the river discharge. The positive correlation between the nitrate and silicate winter levels (r^2=0.45; p=0.2; 1988-93) supports the conclusion that there were high contributions from the drainage area to the nitrate pool.

The changes in the total phosphorus summer level for the deep waters reflect the long-term dynamics of the winter data, and support the positive long-term trend (r^2=0.76; p=0.00001; Fig. 4.3.10). The increase of 0.018 µmol dm^{-3} yr^{-1} of the total phosphorus concentrations in summer in deep water reflects the rates for the increase of phosphate and total phosphorus in winter. The nitrate summer concentrations below the thermocline show an upward trend for 1975-91 (r^2=0.69; p=0.0002) and a decrease for 1989-93 (r^2=0.98; p=0.001; Fig. 4.3.10). Similar to the winter pattern, the maximum values in summer were also observed in 1989/90. The decrease in summer nitrate over the period 1989-93 was found to be 3.06 µmol dm^{-3} yr^{-1}. The fresh-water input explains 56 % of the nitrate dynamics for the period 1975-93. However, it should be noted that the use of fertilisers in the Latvian agriculture, which was gradually increasing during the late 1970s and 1980s, dropped sharply after 1989. These man-made impacts, coupled with changes in inputs from the drainage area, may therefore have significantly influenced the actual long-term course of the nitrate concentrations. However, the drastic growth of the phosphate and total phosphorus levels in the early 1990s which appeared in parallel with the nitrate drop seems to result from variations of the internal biogeochemical processes in the Gulf (e.g., phosphate release from sediments).

Special attention should be paid to silicate, since in the deep waters (Fig. 4.3.10) as well as in the upper layer, silicate summer concentrations decreased continuously over the period of measurements. There was a significant negative trend for the deep (r^2=0.81; p=0.0004; Fig. 4.3.10) and upper layers (r^2=0.62; p=0.01; 1984-93) with changes in silicate concentrations of 1.5 and 0.44 µmol dm^{-3} yr^{-1}, respectively. The negative correlation between silicate and total phosphorus in the deep (r^2=0.55; p=0.02) and surface waters (r^2=0.58; p=0.02) for 1984-93 suggests that the decrease in silicate followed the increase in phosphorus. The silicate loss was about 12.5 times that of the phosphorus increase in the upper layer. This meets almost the assimilative stoichiometry for both elements.

A nutrient budget model based on the data from 1989 [*731*] showed that the Gulf of Riga entirely exports the phosphorus it receives from land to the Baltic Proper, and that about 37 % of this phosphorus is exported as phosphate. However, according to the model, only 25 % of the Gulf's nitrate load enters the Baltic Proper. Estimates are possible as to how the decreasing nitrate levels in the Gulf could influence relevant parts of the nitrogen limited Baltic Proper. A comparison of the silicate levels of the Gulf of Riga and the Gotland Basin identifies the Gulf as a potential silicon importer over the whole year.

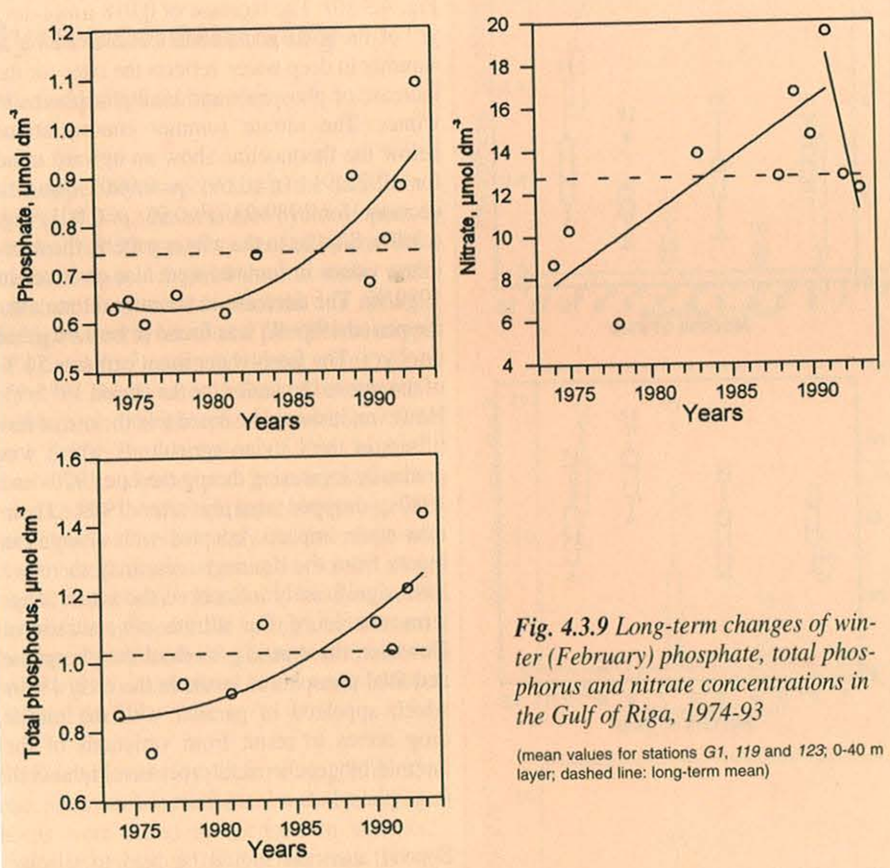


Fig. 4.3.9 Long-term changes of winter (February) phosphate, total phosphorus and nitrate concentrations in the Gulf of Riga, 1974-93

(mean values for stations *G1*, *119* and *123*; 0-40 m layer; dashed line: long-term mean)

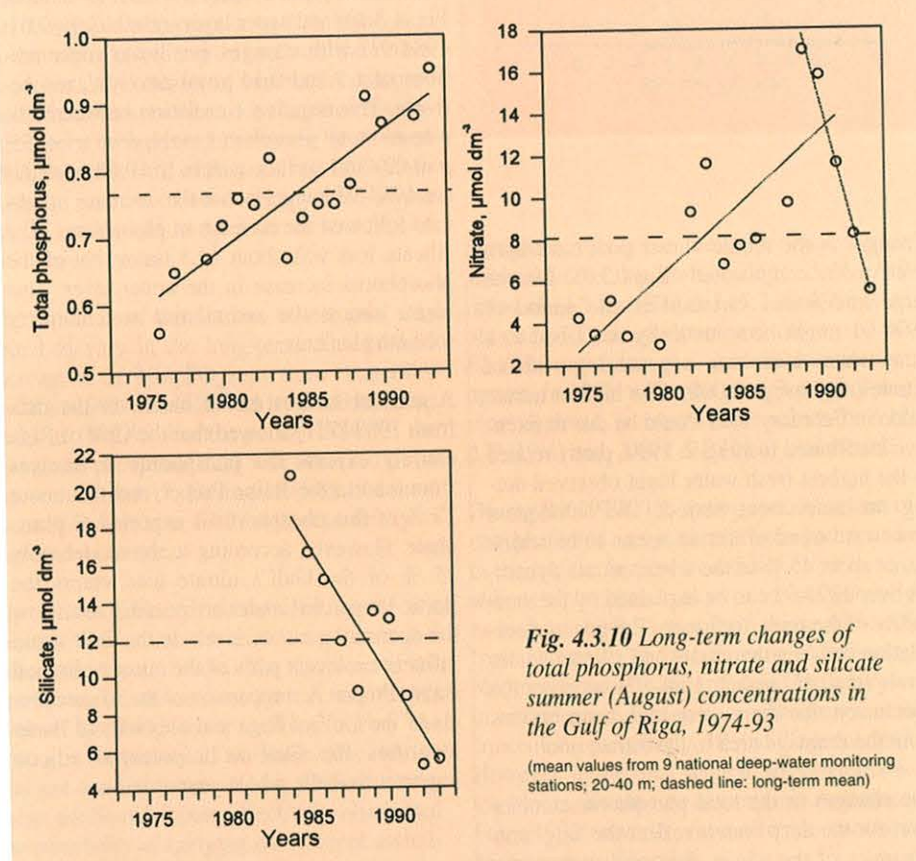


Fig. 4.3.10 Long-term changes of total phosphorus, nitrate and silicate summer (August) concentrations in the Gulf of Riga, 1974-93

(mean values from 9 national deep-water monitoring stations; 20-40 m; dashed line: long-term mean)

4.3.3 Pelagic biology

4.3.3.1 Phytoplankton

B.Kalveka[1]

Data from five stations, including *G1* (Fig. 4.3.1), have been analysed during this assessment. Following treatment of the samples with formaldehyde, the plankton biomass was determined using the sedimentation method.

Studies of phytoplankton over 1979-93 showed a high stability of the species composition for the **spring** phytoplankton community. In spring, the diatom genera *Thalassiosira*, *Achnanthes*, *Chaetoceros* and the dinoflagellate *Peridiniella catenata* were found in their largest development phase (Table 4.3.2). A comparison of data on the species structure and bioproductivity of phytoplankton revealed certain differences between the assessment periods. These differences were caused by interannual changes of the water temperature in May, at a time when the samples were being collected. Thus, they reflected different stages of the spring community succession. The biomass of the arctic species *Achnanthes taeniata* sharply decreased and the biomass of *Peridiniella catenata* increased during the warm springs within the period 1989-93, which contrast with the less warm springs of 1979-83 and especially the cool springs of 1984-88. In 1989-93, the biomass of *Chaetoceros wighamii* dropped considerably as compared with the biomass observed during 1984-88. Changes in the biomass ratio of certain species resulted in a decline of the total biomass of the spring phytoplankton community in 1989-93 which was lower than in 1979-83, and especially low as compared with 1984-88. During 1989-93, the proportion of Dinophyceae increased and that of Diatomophyceae decreased in phytoplankton, unlike 1984-88 (Fig. 4.3.11).

Characterising the total biomass in May for the three five-year periods it was noted that the phytoplankton was nearest to its initial spring bloom stages in 1984-88. However, even then shifts in time of the development of the microalgae were recorded. This happened for example in May 1984 when the peak of the bloom had already passed, as well as in the cold year of 1987 when the community was still at the initial stage of its succession. Long term observations showed that the greatest biomass of the diatoms *Achnanthes taeniata* (13,700 mg m^{-3}) and *Chaetoceros wighamii* (4,800 mg m^{-3}) was reached in 1985. Their intensive development was caused by a slow warming of the water and the low temperature of the upper layer, but it was also influenced by a nutrient input following the spring river freshet, which reached the central part of the Gulf of Riga.

In the **summers** of 1989-93 and 1979-83, the greatest share of the phytoplankton community was occupied by the blue-green algae *Aphanizomenon flos-aquae* and *Gomphosphaeria* spp. and by the dinoflagellate *Dinophysis acuminata*. In 1984-88, the plankton community was dominated by the diatom *Actinocyclus octonarius* and the dinoflagellate *Dinophysis acuminata* (Table 4.3.2, Fig. 4.3.11). The diatoms also dominated during 1984-88, but only because of their high values in 1988. This was not a typical phenomenon and was caused by unusual meteorological and hydrological conditions in the Gulf of Riga.

The analysis of long-term data sets showed that the summer phytoplankton in 1989-93 had higher total algal biomasses than those found in the previous two assessment periods (1979-88). This was caused by cyanobacteria, Dinophyceae and other groups, especially by the species *Gomphosphaeria* spp., *Aphanizomenon flos-aquae*, *Dinophysis acuminata*, *Oocystis* spp., by Cryptophyceae and *Pyramimonas* sp. (Table 4.3.2). For the summers of the period 1979-93, a reliable positive trend of the phytoplankton total biomass was observed. This could be the result of a growing degree of trophy in the Gulf of Riga.

[1] see ANNEX 11.3 for addresses of authors

Table 4.3.2 Dominating phytoplankton species in the Gulf of Riga during the three assessment periods

1979-83		1984-88		1989-93	
Spring	N=5, S=5		N=5, S=5		N=4, S=4
Thalassiosira baltica	2,958 (6,967, n=5)	*Achnanthes taeniata*	4,895 (13,700, n=5)	*Thalassiosira baltica*	2,508 (5,722, n=4)
Peridiniella catenata	1,160 (2,696, n=5)	*Chaetoceros wighamii*	1,912 (4,800, n=5)	*Peridiniella catenata*	2,078 (4,963, n=4)
Achnanthes taeniata	926 (1,708, n=4)	*Thalassiosira baltica*	1,525 (4,669, n=5)	*Chaetoceros wighamii*	429 (874, n=4)
Chaetoceros wighamii	338 (514, n=5)	*Peridiniella catenata*	760 (1,773, n=5)	*Melosira nummuloides*	59 (140, n=4)
Melosira nummuloides	87 (267, n=5)	*Melosira nummuloides*	71 (226, n=4)	*Achnanthes taeniata*	38 (139, n=2)
Summer					
Gomphosphaeria spp.	57 (152, n=5)	*Actinocyclus octonarius*	75 (316, n=4)	*Dinophysis acuminata*	114 (174, n=4)
Dinophysis acuminata	49 (168, n=5)	*Dinophysis acuminata*	43 (90, n=5)	*Actinocyclus octonarius*	85 (226, n=3)
Actinocyclus octonarius	37 (85, n=4)	*Gomphosphaeria* spp.	23 (74, n=3)	*Gomphosphaeria* spp.	63 (154, n=4)
Aphanizomenon flos-aq.	10 (36, n=5)	*Oocystis* spp.	7 (18, n=5)	*Oocystis* spp.	36 (72, n=4)
Oocystis spp.	5 (9, n=4)	*Aphanizomenon flos-aq.*	6 (13, n=4)	*Aphanizomenon flos-aq.*	27 (53, n=4)
Cryptophyceae	0.6 (2.7, n=3)	Cryptophyceae	0.4 (2.1, n=1)	Cryptophyceae	4.4 (13, n=4)
Pyramimonas sp.	0.5 (2.2, n=4)	*Pyramimonas* sp.	0.02 (0.1, n=1)	*Pyramimonas* sp.	2.3 (6, n=4)
Autumn					
Coscinodiscus granii	299 (660, n=4)	*Coscinodiscus granii*	1,369 (2,744, n=5)	*Coscinodiscus granii*	2,436 (5,479, n=4)
Actinocyclus octonarius	212 (476, n=5)	*Actinocyclus octonarius*	226 (738, n=5)	*Actinocyclus octonarius*	151 (226, n=4)
Gomphosphaeria spp.	67 (186, n=4)	*Chaetoceros danicum*	41 (158, n=3)	*Gomphosphaeria* spp.	78 (128, n=4)
Chaetoceros danicum	11 (23, n=4)	*Gomphosphaeria* spp.	25 (66, n=4)	*Chaetoceros danicum*	38 (83, n=4)
Aphanizomenon flos-aq.	7 (33, n=4)	*Aphanizomenon flos-aq.*	3 (10, n=3)	*Aphanizomenon flos-aq.*	2 (5, n=3)

(arithmetical **mean** (maximum) values of species wet-weight biomass in mg m^{-3}; N - total number of samples, S - number of stations, n - number of samples in which species were dominating)

The **autumn** peak of phytoplankton was formed by the diatoms *Coscinodiscus granii*, *Actinocyclus octonarius* and *Chaetoceros danicum*, and this was always less pronounced than the spring peak (Table 4.3.2, Fig. 4.3.11). For October, the total biomass of phytoplankton showed an increasing tendency during the three assessment periods. These changes were connected with a growing productivity of Bacillariophyceae, especially of the major species *Coscinodiscus granii*. In 1979-83, the share of *Coscinodiscus* at the total algal biomass was 47 %, and this increased to 81 % during 1984-88 and again to 88 % during 1989-93. The maximum value of the biomass of *Coscinodiscus granii* was observed in 1989, and this is considered to be due to the especially favourable hydrological and hydrochemical conditions for this species during that year.

In the Gulf of Riga, the following potentially toxic phytoplankton species were identified: *Aphanizomenon flos-aquae*, *Nodularia spumigena*, *Coelosphaerium kuetzingianum*, *Chaetoceros danicum*, *Dinophysis acuminata*, *Dinophysis acuta* and *Prorocentrum balticum*. The dominating species were *Aphanizomenon flos-aquae*, *Dinophysis acuminata* and *Chaetoceros danicum*. *Aphanizomenon flos-aquae* causes summer blooms of blue-green

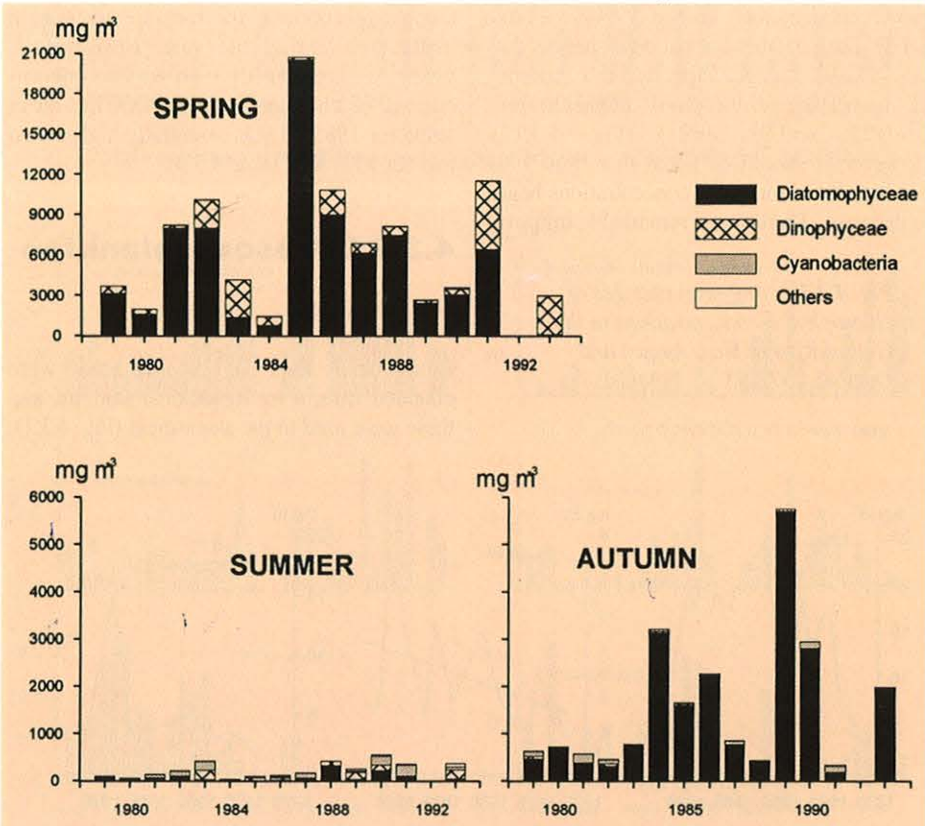


Fig. 4.3.11 Distribution of the mean biomass of different algal classes (in mg m^{-3}) in spring, summer and autumn at station G1, 1979-93

algae, and these are usually observed in July/August. The biomass of potentially toxic species (*Aphanizomenon flos-aquae*, *Dinophysis acuminata*) increased in 1989-93 by comparison with that found during 1979-88 (Table 4.3.2).

4.3.3.2 Chlorophyll *a*

B. Jansone[1]

Material obtained at six national stations was analysed for the assessment (Fig. 4.3.1). During **spring,** the yearly maximum for chlorophyll *a* was observed (Fig. 4.3.12.). Assessing the monitoring data, obtained at the beginning of May each year, showed that before 1985 there was a long-term growth of the pigment concentration in the Gulf of Riga. In the following years, this tendency was reversed (Fig. 4.3.12). The lowest values of the pigment for the period 1979-93 were obtained during 1989-93 (Table 4.3.3). This was caused by the fact that due to high water temperatures in these years the phytoplankton community was already on the final stage of its spring succession when the biomass started considerably to decrease. The curves for the long-term dynamics of chlorophyll agree well with analogous curves on temporal changes of the total phytoplankton biomass.

In **summer** (August), the chlorophyll concentrations were low and varied considerably from year to year (Fig. 4.3.12). On the basis of mean data for the three assessment periods, there was a slight tendency for increasing pigment concentrations during 1979-93 (Table 4.3.3). Data obtained at national stations during 1972-93 strongly supported this tendency of increasing chlorophyll concentrations (r^2=0.25; p=0.02; n=21) (Fig. 4.3.12). However, it should be noted that from 1992 onwards the chlorophyll concentrations began to decrease. There was a remarkable improvement in the statistical significance of the trend for 1972-91 (r^2=0.51; p=0.001; n=19) compared with that during 1972-93 which supported this observation. The decrease of the summer chlorophyll level for 1992-93 was probably due to both a decrease of the inorganic nitrogen pool and a change of the species structure of phytoplankton, i.e., the growing importance of blue-green algae. Together with chlorophyll *a*, phycobilin and phycocyanins are known to dominate in cells of this algae.

	Spring	Summer	Autumn
1979-83	17.8± 5.0 (12.7-24.6) *n*=24	2.8 ± 0.7 (1.9-3.7) *n*=30	4.4 ± 0.5 (4.1-5.0) *n*=18
1984-88	17.8 ± 1.8 (16.2-19.5) *n*=24	2.5 ± 1.1 (1.5-3.9) *n*=24	3.3 (3.3-3.3) *n*=6
1989-93	13.9 ± 6.8 (4.3-19.8) *n*=24	3.3 ± 1.6 (1.6-5.2) *n*=30	6.9 ± 7.2 (2.5-15.2) *n*=18

(mean values, standard deviation and range - in mg m^{-3} - for the 0-10 m layer)

***Table 4.3.3** Basic statistics on chlorophyll a at six national monitoring stations in the Gulf of Riga, 1979-93*

The **autumn** (October) data indicate that there was a tendency of chlorophyll concentrations to increase during 1972-89 (Fig. 4.3.12). The data for the subsequent years are not quite comparable because the measurements were made later during the year. Nevertheless, owing to exceptionally high autumn concentrations of chlorophyll *a* in 1989, the mean value for 1989-93 was essentially higher than that for 1979-88 (Table 4.3.3).

4.3.3.3 Mesozooplankton

E. Kostrichkina[1]

Generalized data on zooplankton were obtained from 6 to 10 national stations, and these were used in the assessment (Fig. 4.3.1).

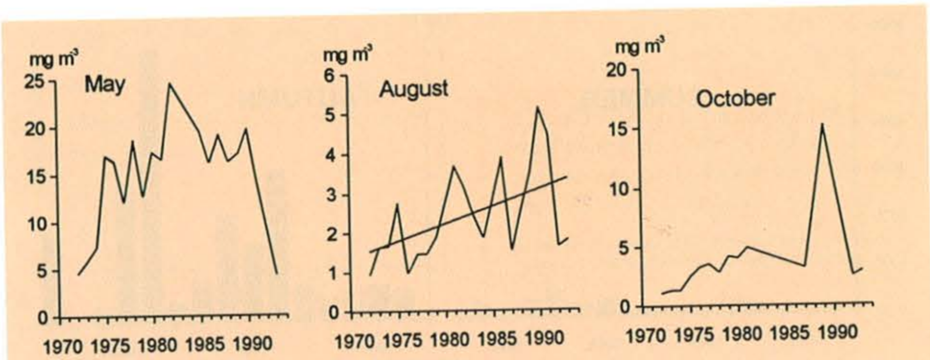
In **spring** (May) 1989-93, the species *Synchaeta* spp. (mainly *S. baltica*), *Acartia bifilosa* and *Eurytemora hirundoides* were found to dominate. Though varying considerably, by comparison with the previous five years, their abundance was found to increase during 1989-93, particularly for rotifers. This phenomenon was caused by higher water temperatures in 1989-93 as compared with 1979-88, leading in May to a later stage of the spring succession of the zooplankton community which was characterised by a significant increase in the abundance of the rotifers gen. *Synchaeta*. In warmer springs of 1989-93, the total abundance and biomass were larger than those found during the less warmer (1979-83) and the cool periods (1984-88) (Table 4.3.4). If one considers the whole series of observations during 1960-93, it is possible to produce a more accurate analysis of the fluctuations (periodicity) of the biomass and species composition of zooplankton (Fig. 4.3.13). It has been shown [*348*] that the long-term dynamics of the zooplankton abundance in spring (May) is closely connected with the severity of winter conditions and has approximately the same periodicity.

During 1989-93, the dominating zooplankton species in **summer** (August) were *Bosmina coregoni maritima*, *Eurytemora hirundoides* and *Acartia bifilosa*. The amount of *A. bifilosa* and especially *B. coregoni maritima* increased in these years as compared with 1979-88, whereas the abundance of *E. hirundoides* decreased. High increasing rates for *B. coregoni maritima* resulted in an increasing share of cladocerans and a decreasing share of copepods in the total abundance and biomass of zooplankton. The total zooplankton abundance observed in August 1989-93 was similar to that of the period 1979-83 (Table 4.3.4), but with slight variations each year, and higher than that during the period 1984-88. The highest total biomass was recorded in 1979-83 and the lowest in 1984-88, with intermediate values for the assessment period 1989-93 (Table 4.3.4).

The analysis of the data on the long-term dynamics of the total summer zooplankton biomass for 1960-93 showed that the growth of total biomass began in the early 1970s and continued until the beginning of the 1980s (Fig. 4.3.13). During 1960-83, the increase in biomass coincided with the growth in nutrient concentrations which determine the phytoplankton productivity (r^2=0.90, p=0.01) [*348*]. Statistically reliable connections with other environmental factors could not be detected. After 1983, the biomass of zooplankton varied considerably from year to year and did not show any clear tendency (Fig. 4.3.13). As a result, the positive trend of the total biomass calculated for the whole period 1960-93 became less significant (r^2=0.23, p=0.05).

***Fig. 4.3.12** Long-term changes of chlorophyll a concentrations in the Gulf of Riga in May, August and October, 1972-93*

(mean values for the 0-10 m layer, in mg m^{-3})



[1] see ANNEX 11.3 for addresses of authors

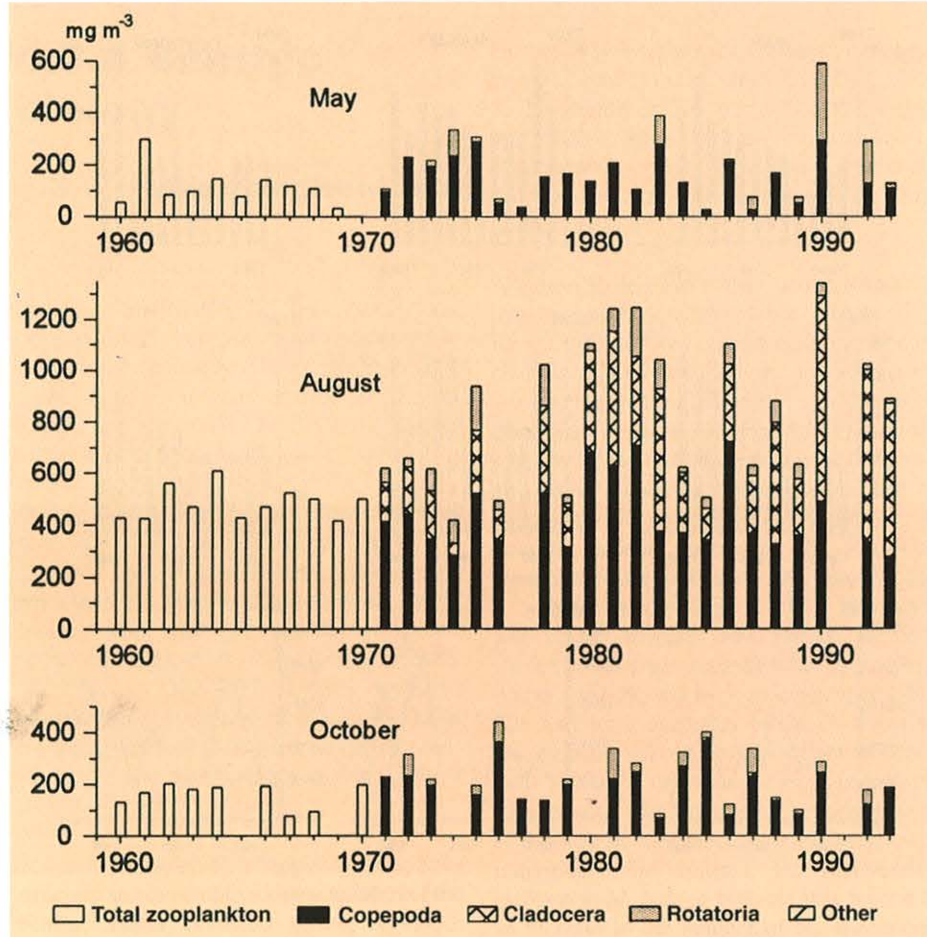
***Table 4.3.4** Long-term dynamics of zooplankton total abundances (10^3 ind. m^{-3}) and total wet weights of biomass (mg m^{-3}) in the 0-40 m layer of the Gulf of Riga in May, August and October, 1979-93*

		1979-83	1984-88	1989-93	1979-93
May	Abundance	11.7	12.8	36.2	19.1
	Biomass	201	124	272	194
	n	43	50	46	139
August	Abundance	97.2	62.0	96.2	84.2
	Biomass	1,034	752	975	916
	n	79	64	45	188
October	Abundance	23.5	32.3	22.6	26.6
	Biomass	232	269	190	233
	n	34	32	32	98

In **autumn** (October) 1989-93, *Acartia bifilosa*, *Eurytemora hirundoides* and *Synchaeta* spp. (mainly *S. baltica*) were dominant. In 1989-93, the abundance and biomass of the first two species were lower than those during 1984-88, but similar to those found during 1979-83. The abundance of *Synchaeta* spp. decreased in comparison to the previous assessment periods. However, there were no considerable changes in the species composition of the zooplankton community. In October 1989-93, the total abundance of zooplankton was close to that of 1979-83, but lower than that in 1984-88. The total biomass was less than in 1979-83 and especially low compared to 1984-88 (Table 4.3.4). For autumn, the long-term dynamics of the zooplankton biomass depends on temperature and food conditions [*348*]. During the last years these relationships were interrupted. The long-term dynamics (1960-93) of the zooplankton biomass revealed strong interannual fluctuations and a lack of a clear overall trend, i.e., there were periods of decrease (1973-79, 1986-93) and increase (1960-72, 1980-86) (Fig. 4.3.13).

It may be concluded that a noticeable effect of eutrophication on zooplankton was revealed in summer. Therefore, the analysis of the long-term dynamics of the number of the main species and of the trophic structure of the community is given for this season. Figure 4.3.14 shows that during 1960-93 the abundance of *Bosmina coregoni maritima* increased more efficiently than that of *Acartia bifilosa*. The abundance of *Eurytemora hirundoides* and *Synchaeta* spp. increased until the mid-1980s and *Keratella* spp. increased until the mid-1970s. Thereafter, the abundance of the three species decreased. After 1965, the abundance of *Limnocalanus grimaldii* showed a clear decreasing trend to almost the point of extinction.

***Fig. 4.3.13** Total wet-weight biomass of zooplankton (in mg m^{-3}) and the biomass of its main taxonomic groups in the Gulf of Riga for May, August and October, 1960-93*



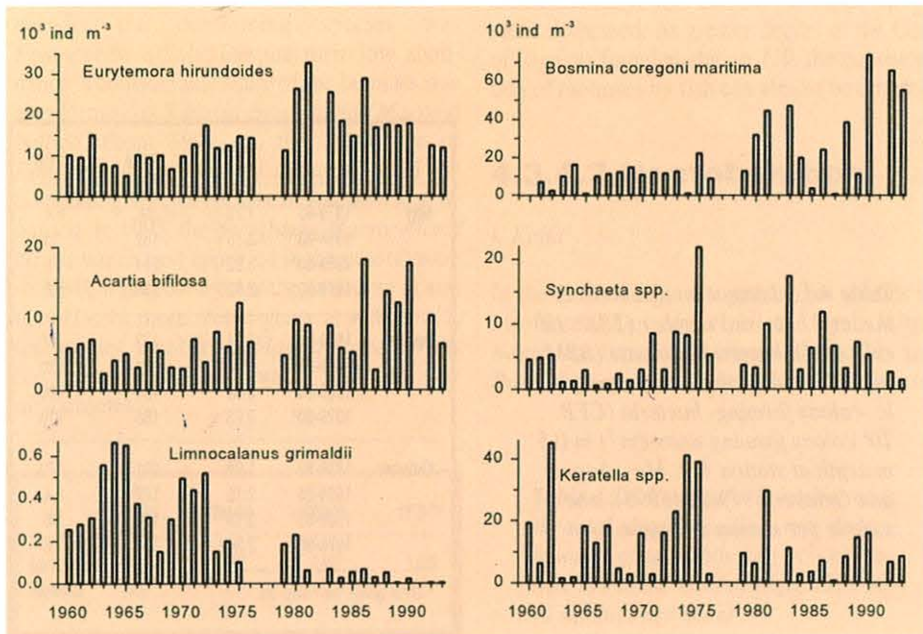


***Fig. 4.3.14** Abundance of the main summer (August) zooplankton species (in 10^3 ind. m^{-3}) in the Gulf of Riga, 1960-93*

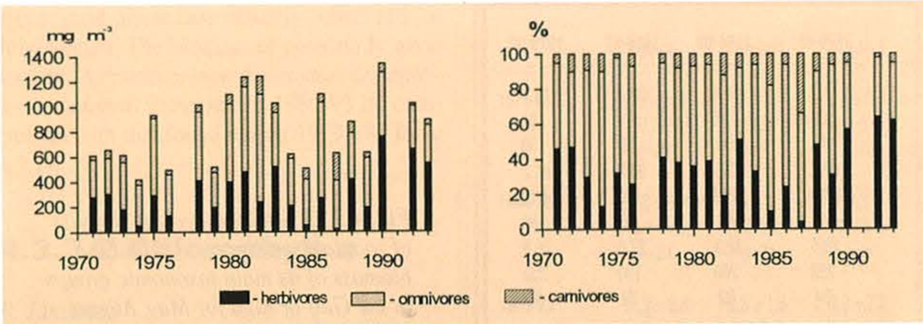


Fig. 4.3.15 Biomass of summer (August) zooplankton species (in mg m^{-3}) belonging to different trophic levels % in the Gulf of Riga, 1970-93

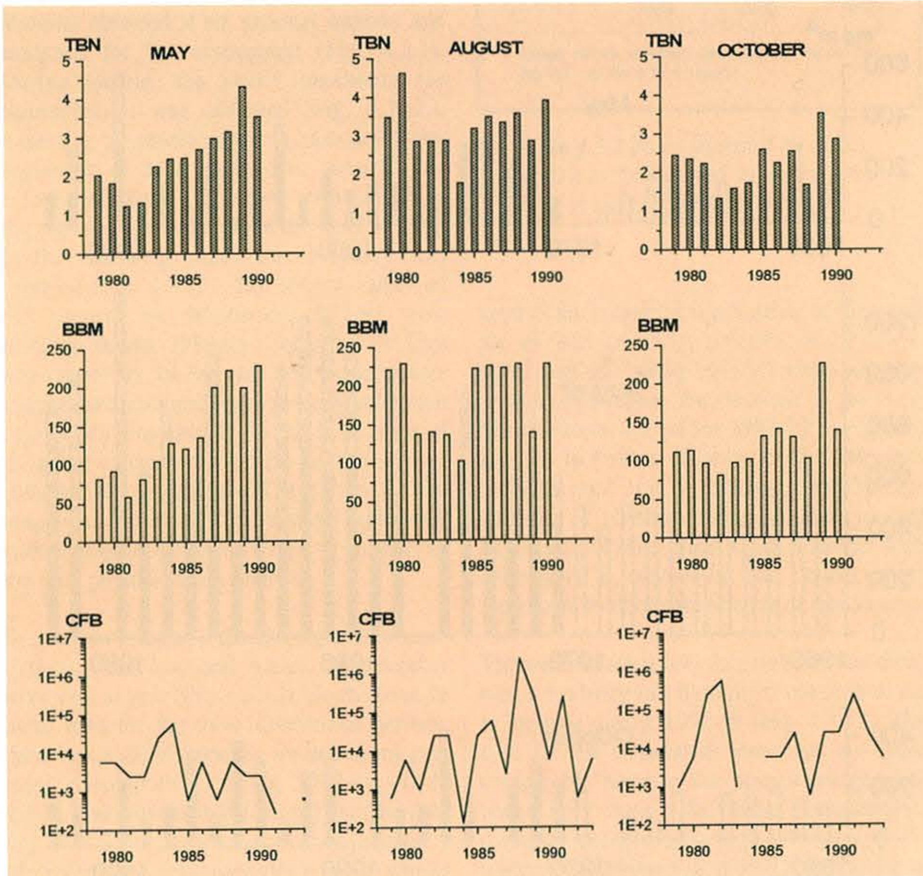


Fig. 4.3.16 Total bacterial number (TBN: 10^6 cells cm^{-3}), bacterial biomass (BBM: µg C dm^{-3}) and number of saprophytic -colony forming- bacteria (CFB: colony forming units cm^{-3}) in 0.5 m depth at station 119, May, August and October 1979-90 (1993)

Table 4.3.5 Long-term dynamics of the total bacterial number (TBN: 10^6 cells cm^{-3}), bacterial biomass (BBM: µg C dm^{-3}) and number of saprophytic -colony forming- bacteria (CFB: 10^3 colony forming units cm^{-3}) in 0.5 m depth at station 119, May, August and October 1979-90 (1993); one sample per season and year

		TBN	BBM	CFB
May	1979-83	1.73	84	8.4
	1984-88	2.75	162	15
	1989-90*	3.92	214	1.5
	1979-90*	2.52	138	8.6
August	1979-83	3.34	173	12
	1984-88	3.07	201	520
	1989-90*	3.38	184	100
	1979-90*	3.23	186	210
October	1979-83	1.98	101	170
	1984-88	2.15	122	9.4
	1989-90*	3.19	183	66
	1979-90*	2.26	123	88

* CFB given for 1989-93

In the 1980s and especially at the beginning of the 1990s, changes of the zooplankton species structure and the increase of herbivores (*Bosmina coregoni maritima*) within the total abundance and total biomass resulted in a change of the community's trophic structure. This was manifested by the growing role of herbivores in the community (Fig. 4.3.15).

4.3.3.4 Bacterioplankton

L. Daksha[1], *E. Kostrichkina*

Data obtained at station *119* (Fig. 4.3.1) have been assessed. According to chlorophyll *a* data, this station reliably reflects the seasonal biological processes proceeding in the offshore areas of the Gulf (r^2=0.51-0.52; p=0.05).

The seasonal dynamics of the total bacterial number and biomass are clearly shown by data gained during seasonal surveys (Table 4.3.5). In 83 % of the cases, the maximum values were observed in summer, the remaining maxima being found in spring.

On the basis of the data collected in May, the long-term dynamics of the total bacterial number and biomass showed a clear tendency to increase with time (r^2=0.77; p=0.002, and r^2=0.86; p=0.001, respectively; Fig. 4.3.16). Thus, the bacterial number was higher during 1989-90 than during the two previous assessment periods (Table 4.3.4). For August, there was no clear long-term trend either of the total bacterial number or biomass. During 1979-90, the total number of micro-organisms was approximately constant, with the exception of the years 1980, 1984 and 1990 (Fig. 4.3.16). In the warmest summers 1989-90, the total bacterial number was analogous to that of 1979-83, but slightly larger than during the colder summers 1984-88 (Table 4.3.5). Regarding the long-term dynamics of the bacterial biomass, two levels could be distinguished, higher values during 1979-80 and 1985-90, and lower values for 1981-84. Therefore, in 1989-90 the total bacterial biomass was higher than in 1979-83, but lower than in 1984-88.

In October, the long-term dynamics of the total bacterial number and biomass was more pronounced than in summer. In 1979-81, these parameters changed comparatively little, but showed a tendency of growth during 1982-90. Accordingly, their values in 1989-90 were higher than during the two previous five-year periods (Fig. 4.3.16, Table 4.3.5). Thus, since 1985, the data support an increase of the organic contaminant load to the Gulf of Riga in spring and to a less degree in autumn, and a stabilisation of the load in summer.

Surveys in 1979-93 on saprophytic bacteria, which are characteristic indicators of water contamination for labile organic substances, did not show any clear seasonal variations (Table 4.3.5). Maximum numbers were as follows: 31 % in summer and autumn, and 8 % in spring. In the remaining periods, there were no seasonal dynamics. The May values measured during 1979-93 did not show any tendency within the long-term dynamics of the saprophyte number. Until 1982, numbers were very similar, but grew sharply in 1983-84 to their maximum. Thereafter, though varying between years, the saprophyte number decreased (Fig. 4.3.16). In May 1989-93, the number of saprophytic bacteria was slightly lower than in 1979-83, but much lower than in 1984-88 (Table 4.3.5).

The saprophyte numbers in August were observed to increase with time. During 1979-91, there was a clear long-term tendency for the numbers to increase (r^2=0.41, p=0.025), except for the anomalously high number in 1988. This coincided with an increase of the phytoplankton productivity but indicated possibly also a growing contamination with labile organic compounds. During August 1989-93, the saprophyte number was an order of magnitude higher than that during 1979-83, but was lower than that during 1984-88. For October, the saprophyte numbers did not show a general tendency, though some comparatively short periods of either growth, stabilisation or decrease were observed (Fig. 4.3.16). However, during 1989-93, this number was higher than in 1984-88, and lower as compared with 1979-83.

[1] see ANNEX 11.3 for addresses of authors

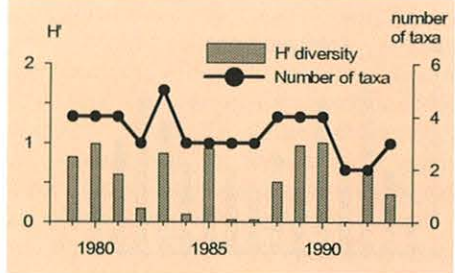


Fig. 4.3.17 Variations in the number of taxa and H' diversity at station 119 in the southern part of the Gulf of Riga, 1979-93

4.3.4 Benthic biology

4.3.4.1 Macrozoobenthos

M. Ceitlina[1]

Samples from station *119* (Fig. 4.3.1) have been assessed. This station is considered to indicate the processes relating to the benthic fauna of the deep water area of the Gulf of Riga, and in addition the zoobenthos dynamics for great parts of the Gulf.

During 1989-93, the species composition of zoobenthos was characterised by a low species diversity. Only 6 species were found and no more than 2-4 species occurred in any single sample. After 1990, the number of species decreased compared with that during 1979-88. The *Shannon* index (*H' diversity*) fluctuated over the three assessment periods. It was highest during 1979-83 (0.69), lowest during 1984-88 (0.32) and intermediate during 1989-93 (0.59) (Fig. 4.3.17).

The most considerable changes in the species structure and quantitative characteristics of the benthic fauna occurred during 1989-93. During 1989-90, as well as in the previous decade, the dominating species was *Pontoporeia affinis*. Despite their low abundance, a considerable share of the biomass was also formed by *Saduria entomon* and *Macoma baltica*. From 1991 on, the abundance of *Pontoporeia* spp. sharply decreased, and the dominating species became *S. entomon* and *M. baltica*. In 1993, the polychaete *Marenzelleria viridis* which first appeared in the coastal zone in 1990, was recorded for the first time at station *119*. In more recent years at station *119*, the species number of *Marenzelleria viridis* has been reduced and replaced by oligochaetes.

	1979-83	1984-88	1989-93	1979-93
Abundance	4,807	4,874	2,497	4,059
Biomass	19.3	14.7	19.3	17.7

Table 4.3.6 Long-term dynamics of the total abundance (ind. m^{-2}) and biomass of zoobenthos (g m^{-2}) at station 119 in the Gulf of Riga, 1979-93 (3 samples per year)

During 1989-93, the total biomass of zoobenthos was similar to that observed during 1979-83, but slightly higher compared with that during 1984-88. However, due to diminution of *Pontoporeia* spp., the total abundance decreased by almost 50 % compared with that during 1979-88 (Table 4.3.6).

A more detailed assessment of the long-term dynamics of the total abundance of zoobenthos for 1979-93 showed considerable interannual changes. Until 1989, there was a tendency for growth, but in the following years the abundance sharply decreased (Fig. 4.3.18). Besides a major decline of *Pontoporeia* spp., there was also a minor decrease in the *Macoma baltica* abundance. By contrast, the abundance of *Saduria entomon* increased. The long-term dynamics of the zoobenthos total biomass had a 'wavy' character (Fig. 4.3.18). The major part of the biomass was formed by both species of *Pontoporeia* until 1980, by *P. affinis* during 1981-90, and by *S. entomon* after 1990. The long-term dynamics of the *Pontoporeia* spp. biomass did not show any clear tendency until 1990, but decreased sharply in the following years. The biomass of *S. entomon* grew as a result of the increase in the number of individuals. In contrast, the incremental increase in *M. baltica* biomass was related to an increase in the number of big specimens. Simultaneously, the number of small specimens decreased. At greater depths of the Gulf of Riga, as found at station *119*, the consumption of molluscs by fish can almost be ignored.

4.3.4.2 Nectobenthos

I. Kotta[1]

In the Gulf of Riga altogether five species of Mysidacea, i.e., *Mysis relicta*, *Mysis mixta*, *Neomysis integer*, *Praunus flexuosus* and *Praunus inermis*, have been found. *M. relicta*,

[1] see ANNEX 11.3 for addresses of authors

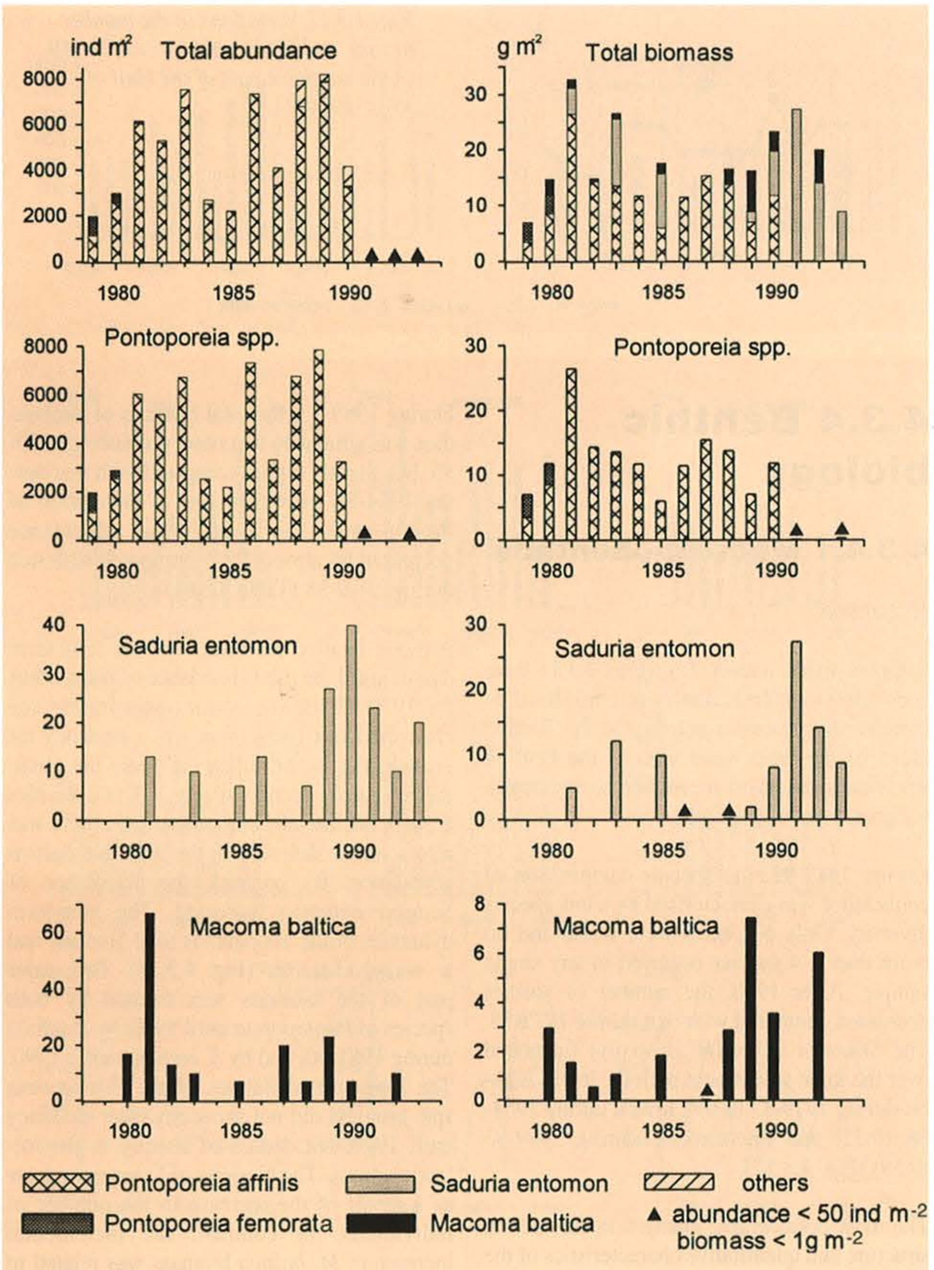


Fig. 4.3.18 Variations of the total abundance (ind. m^{-2}), total biomass (g m^{-2} formalin wet weight) and species composition at station 119 in the southern part of the Gulf of Riga, 1979-93

M. mixta and *N. integer* are the main necto-benthic species. *P. flexuosus* and *P. inermis* occur sparsely and in small numbers in shallow (5-10 m) coastal areas.

Using data of 1974-76 on the annual abundance and biomass dynamics of *M. relicta*, peaks can be discerned for spring (May) and summer (July) (Fig. 4.3.19). In all samples, both young and adult specimens have been found. From April to August, *M. relicta* females are present, bearing developed embryos. Consequently, the reproduction of *M. relicta* is confined to spring and summer and its abundance and biomass peaks are connected with the periods of intense reproduction. The offspring of *M. mixta* appear from March to June. The graphs for both abundance and biomass dynamics have a peak in June and June/July, respectively (Fig. 4.3.19). From June (July) to November, the abundance and biomass steadily diminish. For *N. integer*, the abundance and biomass regularly increase from June to September, i.e., during the period of its reproduction (Fig. 4.3.19). High total abundances of mysids (>40 ind. m^{-3}) have been found for the period between May and August. Despite an increase in July and October, the biomass of Mysidacea varies insignificantly from April to November (Fig. 4.3.19). A rather sharp decrease of the abundance and biomass of *M. relicta* in September, of *M. mixta* between July and September, and of *N. integer* in October/November is probably due to their consumption by fish [*347*].

The highest biomass values for *M. mixta* and *M. relicta* were found after the severe winters in 1975-76, 1979-80 and 1985-87 (1988) when the temperature in the bottom layers during spring was noticeably lower than after mild winters (Fig. 4.3.20). The biomass of *N. integer* was found to be related to the summer temperatures. In the Salacgriva area, a high biomass of this species occurred in periods with warm summers (1979-83, 1985-91), i.e., in years when the mean temperature of the coastal waters during June to August was clearly above the long-term mean (Fig. 4.3.20). However, for 1975 and 1987, the dynamics of *N. integer* could not be related to the summer temperature conditions. For the entire period of 1974-91, the biomass of Mysidacea showed maximum values during 1978-81 and 1985-91 (Fig. 4.3.20). Compared with the previous assessment periods, the total abundance of mysids in 1989-91 was higher than those during 1979-83 and 1984-88.

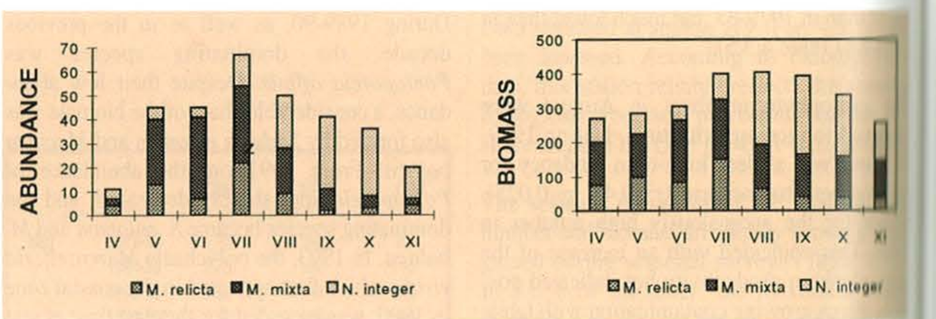


Fig. 4.3.19 Seasonal abundance (ind. m^{-3}) and biomass (mg m^{-3}) dynamics of Mysis relicta, Mysis mixta, Neomysis integer and Mysidacea abundance in the Salacgriva area of the Gulf of Riga, 1974-76

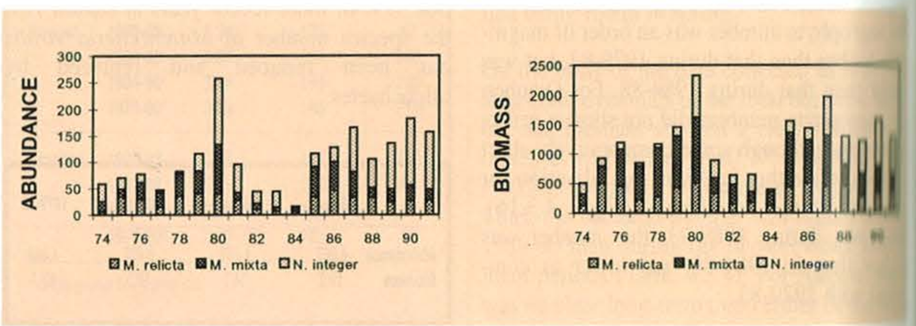


Fig. 4.3.20 Dynamics of Mysis relicta, Mysis mixta, Neomysis integer and Mysidacea abundance (ind. m^{-3}) and biomass (mg m^{-3}) in the Salacgriva area of the Gulf of Riga in August-September, 1974-91

4.3.4.3 Macrophytobenthos

H. Kukk[1]

In samples collected from the Gulf of Riga, 11 taxa of red algae, 9 species of brown algae, 15 species of green algae, 3 species of charophytes, and 13 species of phanerogams have been identified [*363*]. The dominant species are the red algae *Ceramium tenuicorne*, *Polysiphonia fucoidea* and *Furcellaria lumbricalis*, the brown algae *Pilayella littoralis*, *Ectocarpus siliculosus* and *Sphacelaria arctica* and *Fucus vesiculosus*, and the green algae *Cladophora glomerata* and *Spirogyra* sp. The charophytes were found only in small amounts. *Potamogeton filiformis*, *P. pectinatus*, *Zannichellia palustris* and *Schoenoplectus tabernaemontanii* are the most frequent phanerogams.

In the 1980s, in comparison with the 1960s and 1970s, the species composition of the bottom vegetation in the northern Gulf of Riga has considerably changed. These changes are most noticeable in the areas with high industrial and municipal loads. In the northern part of the Pärnu Bight and in the Kuressaare Bight, the number of bottom vegetation's taxa decreased by 11, i.e., the phyllophora *Truncata f. angustissima*, *Stictyosiphon tortilis*, *Cladophora rupestris*, *Enteromorpha ahlneriana*, *Zygnema* sp., *Tolypella nidifica*, *Chara canescens*, *Ch. aspera*, *Ch. tomentosa*, *Ch. connivens* and *Ruppia maritima* disappeared. At the same time, the areas with the green algae *Cladophora glomerata* and with the red algae *Ceramium tenuicorne* have considerably widened [*365*]. The brown algae *Ectocarpus siliculosus* and the phanerogam *Potamogeton pectinatus* were found to dominate in the inner part of Kuressaare Bight in 1976 [*657*]. In 1990, *E. siliculosus* was not found. *P. pectinatus* formed occasional scattered tufts. In the open part of the bight, *Myriophyllum spicatum* was dominant together with *C. glomerata* and *P. littoralis* growing as epiphytes on it. *Potamogeton perfoliatus* and *P. filiformis* were also identified. However, it should be stressed that this kind of damage in the bottom vegetation of the northern Gulf of Riga is localised and has only been recorded in areas with high inputs, i.e., in the immediate vicinity of pollution sources.

Compared to the northern parts, the changes are much greater in the southern Gulf of Riga where the sandy bottoms do not favour luxurious growth of the vegetation. In addition, this area is affected by large rivers, including the Gauja, Lielupe and Daugava. In the 1980s [*363*], in comparison with the 1920s [*624*] and early 1970s [*366*], the number of taxa of bottom vegetation has decreased. All chlorophytes disappeared as well as *Cladophora fracta*, *Anfeltia plicata*, *Polysiphonia violacea*, *Hildenbrandtia rubra*, *Callithamnion roseum*, *Dictiosiphon chordaria*, *Chorda filum* and *Elachista lubrica*. The habitats of *Furcellaria lumbricalis* and *Fucus vesiculosus* have also decreased. By contrast, the filamentous green algae increased intensively. The overgrowing of algae by epiphytes has increased as well.

4.3.5 New species

In summer 1991, the cladoceran *Cercopagis pengoi*, representative of Ponto-Caspian fauna, was found for the first time in the coastal zone of the southern part of the Gulf of Riga. In 1993, this species was found also in the central parts of the Gulf, predominantly in the upper layer (0-10 m) as single individuals. In the shallow north-eastern part of the Gulf and in the Pärnu Bight, this species was first recognised in 1992 [*511*]. In 1994, it was observed in the diet of herring.

The polychaete *Marenzelleria viridis*, a North-American estuaries species, was discovered for the first time in 1990 along the southern coasts of the Gulf. In 1993, this species was found to occur on all types of soft sediment in coastal and open areas with abundances ranging between zero (10) and 320 ind. m^{-2}

4.3.6 Summary

During 1989-93, the climatic conditions could be characterised as 'Atlantic'. The period was warm, i.e., the air **temperature** in Riga was 1.6 °C higher than the long-term mean. The river run-off exceeded the long-term mean by 17 %. However, within the assessment period, a continuous decrease of both the air temperature and the river run-off was observed. The mean water temperature during the assessment period was 0.7-0.9 °C higher than that during 1979-88, its seasonal course was normal and its variations were lower than those during 1979-88.

The **salinity** continued to decrease over 1989-93, a process that had begun already in 1978. The lowest salinity was observed in May/August 1992 (5.3 psu). Previously, such low values had only be measured during 1928-31. The variations in salinity across the Gulf were smaller than in 1979-88. The seasonal pattern of the salinity in the upper layer was normal. Due to a decrease in the inflow of more saline water, the summer-salinity maximum was almost absent, and this favoured the decrease of the vertical stratification. For the upper layer, the **oxygen** saturation and its seasonal pattern were identical with the patterns during 1979-88. This was true as well for the near-bottom layer for the cold part of the year. However, during the period of vertical stratification (May-August), the oxygen saturation in the near-bottom layer was on average 10-15 % higher than that during 1979-88. This could be explained by the decline of the vertical density gradient. During 1964-84, a gradual decrease in the oxygen content of the deep waters took place. This period was followed by an increase related to the intensity of the vertical mixing.

During 1979-93, the **nutrient** regime of the Gulf of Riga changed, resulting in trends in the seasonal concentrations. During 1989-93, the highest phosphorus concentrations were measured. The nitrate values dropped sharply and the negative trend of the silicate level continued. The long-term variability of the nitrate concentrations during 1974-93 showed a general increase until 1991. Thereafter, the concentrations decreased. This nitrate pattern correlated well with the use of fertilisers in the drainage area and with the input of fresh water. The continuous growth of the phosphorus level during 1974-93 resulted mainly from an increase in the anthropogenic load. The particularly high phosphorus values in the early 1990s, accompanied by a decrease in nitrate, seem to reflect the effects of internal biogeochemical processes within the Gulf. The continuous decline in the silicate concentrations since 1984 closely correlates with the growing phosphorus levels. It depicts the depletion of the silicate reserve under the conditions of a

[1] See ANNEX 11.3 for addresses of authors

progressing phosphorus input. In 1989-93, during the spring blooms of phytoplankton, a transition from phosphorus limitation to a simultaneous shortage of phosphorus, nitrogen and partly silicon took place. According to the data of 1989, the Gulf exports all the phosphorus, derived by run-off from land, to the Baltic Proper. In contrast, the Gulf could be considered as a potential silicon importer during the entire year.

The long-term dynamics of **plankton** during 1989-93 mainly reflected the changes in temperature and nutrient conditions. The increase in the water temperature shifted the time of the onset of the phytoplankton spring succession. Consequently, the temporal dynamics of the bacterioplankton and saprophytic bacteria were also changed. In addition, since 1991, the decreasing nitrate concentrations have caused a drop in the summer phytoplankton biomass. All these changes favoured blue-green algae blooms. The decrease in the summer phytoplankton levels during the early 1990s was followed by a declining number of the saprophytic bacteria.

Although a statistically significant relationship between the phyto and zooplankton dynamics during 1989-93 could not be found, the total biomass of **zooplankton** and the summer phytoplankton values were found to have decreased. The polyterm *Cladocera* spp. gained in importance, following the increasing water temperatures which simultaneously favoured herbivorial zooplankton. The continuing decrease in the salinity yielded an increasing number of fresh water species and their spread from the coastal areas to the open parts of the Gulf. However, their productivity and species structure did not change significantly.

The variability of the **phytoplankton** during 1979-93 was governed by all of the aforementioned factors. Based on the zooplankton data, the increase in the productivity of the phytoplankton started in the early 1970s and persisted until the mid-1980s. According to the chlorophyll *a* summer data, these long-term changes terminated in the late 1980s. By multiple regression analysis, it was possible to show the decisive contribution of the food supply to the long-term dynamics of the summer zooplankton until 1984. Subsequently, this connection was much weaker.

In the **benthic biology**, a brief decline in the oxygen content of the near-bottom layer to <1 $cm^3\ dm^{-3}$ in summer and autumn of 1979 and 1980 did not result in 'dead areas'. However, the species diversity and biomass of the zoobenthos was observed to decrease. Higher water temperatures during 1989-93 fostered the decrease in the abundance of arctic relict species (*Mysis mixta, M. relicta, Pontoporeia* spp.) and the increase of the boreal *Neomysis integer* (abundance) and *Macoma baltica* (biomass). The lower abundance of relatively small and the higher biomass of the larger organisms during 1989-93 resulted in both a reduction in the total zoobenthos abundance for this period as compared with that during 1979-88, and a biomass increase compared with that during 1984-88. The decrease in the *Pontoporeia* spp. abundance was considered to be produced by a growing stress in the predator-prey system by the omnivor *Saduria entomon*. Over shallow areas (<20 m), a maximum increase in the zoobenthos biomass took place during the 1970s. This also happened in deep waters during the first half of the 1980s [*172*], but in the late 1980s, the biomass was observed to decrease. The different abundance dynamics of the comparatively small *N. integer* and of the larger mysids produced only a slight increase in total mysid biomass during 1989-91 in comparison to that during 1984-88, but a pronounced increase as compared with that during 1974-83. However, the total zoobenthos biomass during 1989-91 was not higher than that found during 1985-88.

The impact of consumption on the abundance dynamics could be clearly seen in the nectobenthos. Though differing interannually due to variations of the water temperature, the nectobenthos abundance coincided with the augmentation of the phyto- and zooplankton productivity, i.e., it showed an indistinct upward trend. Limited information on the **phytobenthos** did not allow conclusions to be drawn on variations during 1979-93, particularly with regard to the present assessment period. With reference to published data, however, it could be concluded that the species diversity has decreased. Some species disappeared, and over the last 30-70 years, changes in the species structure of the associations have occurred. A spreading of the green algae *Cladophora glomerata* was noted.

4.4 BALTIC PROPER

G. Nausch[1] (Convener)

The Baltic Proper, which covers the area between the Darß Sill and the Gulfs of Bothnia, Finland and Riga, comprises the Arkona Sea, Bornholm Sea and Gotland Sea with its western and eastern sub-regions. The Gdansk Basin, belonging to the eastern subregion of the Gotland Sea, is dealt with separately (Chapter 4.4.3) as case study of a region greatly impacted from land-based nutrient sources. The position of the stations used in the following investigations are documented in Figure 4.4.1.

4.4.1 Hydrography

J. Elken, A. Trzosinska, E. Łysiak-Pastuszak, A. Omstedt, H.-U. Lass

The Baltic Proper is the largest subdivision of the Baltic Sea. Its surface area [*220*] is 211,069 km^2 (51 % of the whole Baltic Sea) and the volume is 13,045 km^3 (60 %). Rivers drain directly into the Baltic Proper with a flow of 100 $km^3\ yr^{-1}$, i.e., 21 % of the total run-off to the Baltic Sea. Topographic features subdivide the Baltic Proper into a number of deep basins. The volumes of those basins below the typical halocline are presented in Table 4.4.1. The volumes were calculated using a digitized Baltic Sea topography [*607*]. The residence times of the more saline water in the deep basins were calculated at the assumed inflow rate of 20,000 $m^3\ s^{-1}$.

Compared with the general overview given in Chapter 2, this sub-chapter presents more details on the hydrographic conditions in the Baltic Proper. The discussion is based on data from the HELCOM BMP stations *K2, L1, J1* and *H2*. Contour plots of hydrographic parameters (Figs. 4.4.2 to 4.4.4) have been constructed using optimal interpolation techniques. Oxygen trends and seasonality have been estimated in more detail for the Gdansk Basin. The variables studied cover salinity and temperature as well as oxygen conditions in connection with the horizontal and vertical water exchange.

4.4.1.1 Effects of saline water inflow and overflow

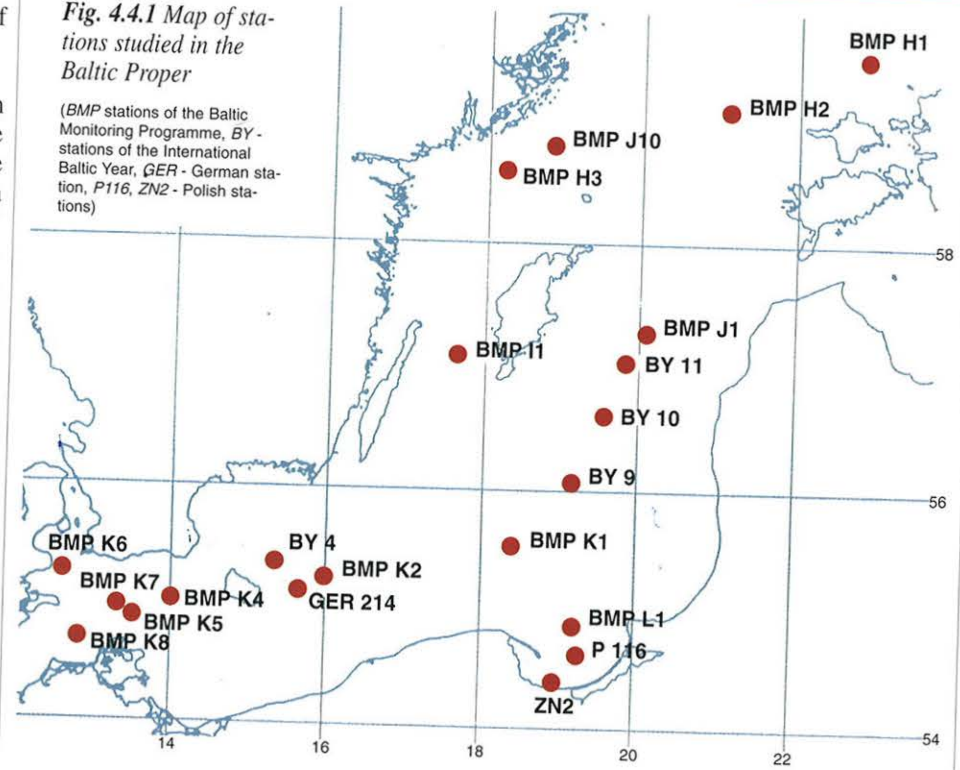
The most specific feature of the hydrographic regime of the Baltic Proper is the permanent saline stratification which is formed by intermittent inflow of saline water from the Belt Sea and the Sound (cf. Chapters 4.5.1 and 4.5.2), and by fresh water surplus from the adjacent basins (Gulfs of Bothnia, Finland and Riga) and draining rivers (Vistula, Odra, Nemunas, etc). Inflowing saline water is spread and mixed, by a cascade effect, into the deep sub-basins of the Baltic Proper. Overflow from one sub-basin to another is dependent on the halocline height above the sill depth of the connecting channel and the density/salinity difference between the basins [*522,637*], as well as on the direction and speed of dominating winds [*351*]. Actual conditions are controlled by a complex of transport and exchange processes. Observations of the deep flow into the Arkona Basin at the beginning of January 1993 were performed and analysed [*413*]. From CTD sections and ship-mounted ADCP records, a dense bottom pool including baroclinic geostrophic currents along the northern flank of the pool could be detected [*413*]. These observations are of great importance in understanding how the deep water enters the Baltic Sea.

Table 4.4.1 Volumes and residence times of water below the halocline in the basins of the Baltic Proper (calculated at specified boundaries, for a typical depth of the halocline and at an inflow rate of 20,000 $m^3\ s^{-1}$)

Basin	Boundaries	Core of halocline m	Volume km^3	λ months
Arkona	east of 12°00' E west of 14°20' E	35	36	0.7
Bornholm	east of 14°20' E west of 17°00' E	50	306	6
Gdansk	east of 17°00' E south of 55°50' N	60	426	8
Eastern Gotland	north of 55°50' N south of 57°50' N	70	1,195	23
Northern Gotland	east of 19°00' E west of 22°00' E north of 57°50' N	70	863	17
Western Gotland	west of 19°00' E west of 18°00' E	70	657	12

Fig. 4.4.1 Map of stations studied in the Baltic Proper

(*BMP* stations of the Baltic Monitoring Programme, *BY* - stations of the International Baltic Year, *GER* - German station, *P116, ZN2* - Polish stations)



[1] see ANNEX 11.3 for addresses of authors

Saline water passing the entrances to the Baltic Sea spreads in the **Arkona Basin** as a bottom layer of variable thickness. Between 1989 and 1992, winter inflows were evident in each January/February period, increasing the bottom salinity to 18-19 psu and lifting the 14 psu isohaline from the near-bottom position (about 43 m) in summer to depths between 35 and 38 m. Autumn inflows appearing in August/September lifted the 10 psu isohaline to depths above 25 m. The effects of more intense autumn and winter inflows were found in 1992. The major inflow in January 1993 (cf. Chapter 2.5.2; [*449*]) increased the bottom salinity to 23 psu and shifted the 10 psu isohaline up to a depth of 20 m. Since 1979, bottom salinities >20 psu were found only in winter 1980 and in autumn 1986.

After passing the Bornholm Channel, saline and dense water sinks in the **Bornholm Basin** to the bottom or to the level of neutral buoyancy, depending on the degree of former stratification in the basin, while lifting up the above layers of lower salinity. Saline stratification (Fig. 4.4.2a) in the period 1989-91 was characterised by small temporal changes indicating that salt-water overflow from the Arkona Basin took place into the different intermediate layers and was balanced by vertical mixing. The water in the bottom layer was renewed in the winter of 1991/92, when the bottom salinity increased from 14 to 16 psu, following more elevated isohaline depths upstream in the Arkona Basin. After the major inflow in January 1993, the bottom salinity increased to 19.5 psu, lifting the 16 psu isohaline from the bottom (about 90 m) to 70 m depth. From the previous period, bottom water renewal is evident at inflows during the winters of 1980 and 1983 and the autumn of 1986. The seasonal summer decrease of the halocline salinity had maximum amplitudes in 1986, 1987, 1988 and 1992.

The overflow of saline waters from the Bornholm Basin to the Gdansk and Gotland Basins is limited by the **Słupsk Sill** which has a depth of 60 m. In 1989-92, the layer above 60 m had salinities generally less than 10 psu (see Fig. 4.4.2a), except in 1992, when a maximum of 11 psu was observed. Such an overflow situation for low saline water was previously observed in 1981 and 1982. Since 1985, the overflowing water had salinities <11 psu which directly influenced the enhanced stagnation in the Gotland Deep. The major inflow in January 1993 increased the overflowing salinity range up to 12.3 psu. During winter 1993/94, the salinity at the Słupsk Sill depth exceeded 15 psu (cf. Fig. 2.16).

Part of the saline water flowing from the Bornholm Basin layer above 60 m is directed into the **Gdansk Basin,** which like the Bornholm Basin is a buffer zone for the saline water pathway into the Gotland Deep. In 1989-92, the salinity of the bottom water in the Gdansk Basin did not exceed 10.5 psu (Fig. 4.4.2b) which corresponds well to the low salinity in the Bornholm Basin at the Słupsk Sill depth. Strong storms during the winters of 1989 and 1990, coupled with the weakened halocline, caused temporary erosion of the halocline down to 80-90 m depth. After the 1993 inflow, the bottom salinity initially increased to 11.8 psu, and then increased to a maximum value of 13.6 psu in spring 1994.

Saline stratification in the **Gotland Deep** (Fig. 4.4.2c) exhibited decreasing salinity (i.e., a deepening of the isohaline) during the stagnation period until the major inflow in 1993. The bottom salinity decreased during 1980-93 from 12.7 psu to 11.3 psu while the 11 psu isohaline deepened from 90 m to 230 m. In general, the layer of maximum vertical stability controlling the cross-halocline mixing followed the depth course of isohalines 8.5-9 psu. At the same time, the halocline thickness defined as a layer with the *Brunt-Väisälä* frequency exceeding 0.02 s^{-1}, typically 20-30 m thick, did not change considerably. This is also evident from the small-scale vertical gradients derived from CTD profiles [*421*]. However, reduced vertical stability of the halocline was observed in 1990-91. Below the halocline, increased lateral transport of waters with 10-11 psu from the Bornholm and Gdansk Basins occurred during the period 1985-92, causing a decrease in the vertical stability of the intermediate waters. The long-term average of the transport rate of dense water from below the halocline was estimated to be 33,200 m^3 s^{-1} [*349*]. The deep waters were replaced by oxygenated saline waters in 1994 (cf. Chapter 4.5.2).

In the **Northern Baltic Proper** (Fig. 4.4.2d) the centre of the halocline generally followed the depth of the 8.5 psu isohaline, which deepened from its most shallow position in 1981 at 60 m depth to the extreme depths of 110-120 m observed in 1993. The effect of the 1983 inflow into the Bornholm Basin can be found in all downstream basins as elevated haloclines during 1983-86. The vertical maximum of the *Brunt-Väisälä* frequency, which is an indicator of the vertical stability, decreased from 0.025 s^{-1} in 1981 to 0.015-0.017 s^{-1} during 1990-92. Weakened vertical stability favoured winter time erosion of the halocline to greater depths and resulted in better aeration of deep layers. Below 150 m, salinity decreased from 11.5 psu in 1980 to 9 psu in 1992, which is a result of the decreased salinity in the Gotland Deep above the level of the Fårö Channel (115 m). During 1989-92, the maximum salinity of overflowing waters decreased in the Gotland Basin from 10.3 to 9.3 psu, which corresponds to the salinity of bottom waters of the Northern Deep. By the summer of 1994, bottom salinity had increased to 9.6 psu, which is well below the high values during 1979-81, which exceeded 11 psu.

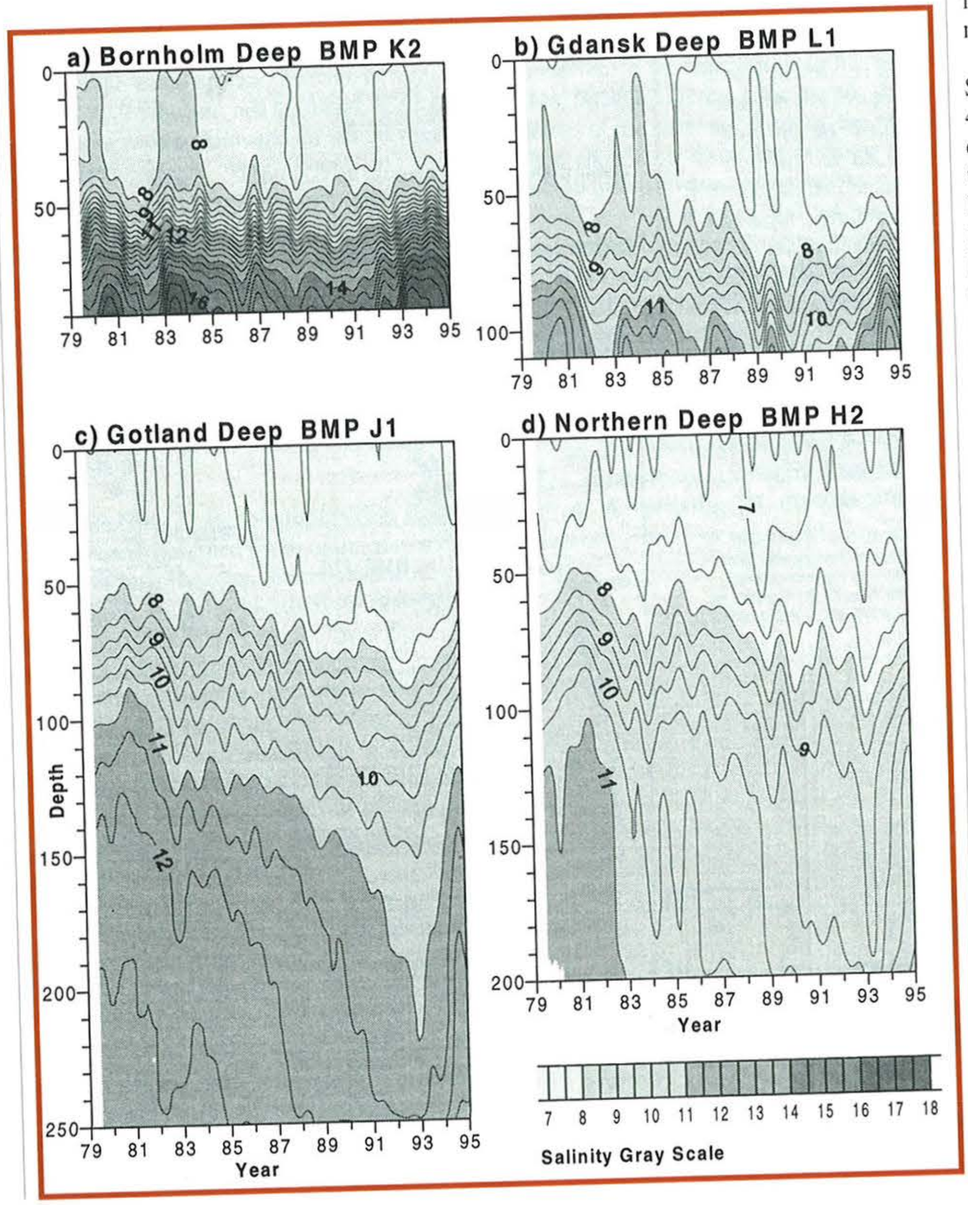
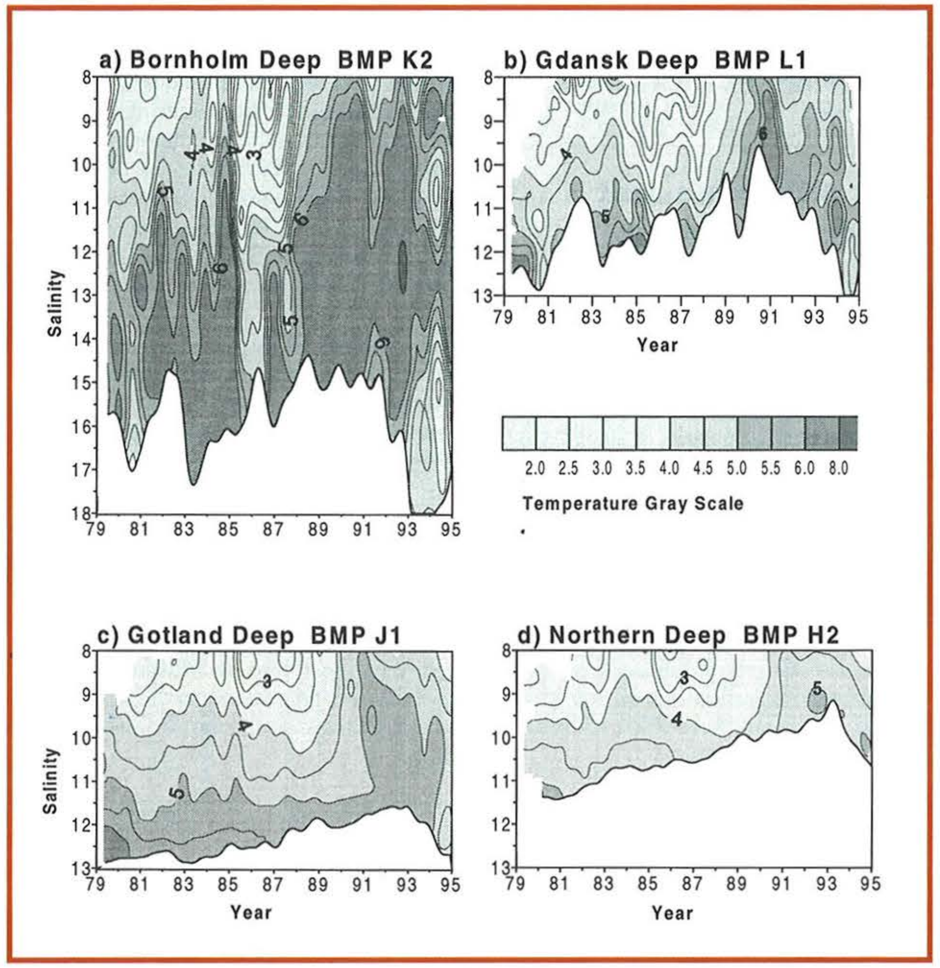


Fig. 4.4.2 Salinity as a function of time (contour interval 0.5 psu)

Fig. 4.4.3 Temperature as a function of time (contour interval 0.5 °C)



4.4.1.2 Changes in the temperature-salinity relation

The final phase of the stagnation period (1989-93) was characterised by an abnormal increase in the temperature of the intermediate layers. This warming is demonstrated in Figure 4.4.3 on the basis of the contour plots of temperature drawn in the salinity coordinate. During 1979-86, the Bornholm Basin water of 8-12 psu salinity (cf. Fig. 4.4.1 for the depth range of the isohalines), which was laterally transported through the Słupsk Furrow into the Gotland Basin, was characterised by temperatures of 5 °C (Fig. 4.4.3a). In 1987, the temperature started to increase, reaching a peak value of >7 °C in 1989-90 in waters with salinities between 10 and 13 psu. In the layer 8.5-10.5 psu of the buffering Gdansk Basin, the highest temperature, exceeding 6 °C (Fig. 4.4.3b), was found at the turn of 1990/91. This abnormally warm and dense water was transported further into the Gotland Basin and the Northern Baltic Proper, where the pattern of a typical increase of the temperature with the salinity was replaced in the winter of 1990/91 by homothermal conditions (Figs. 4.4.3c, 4.4.3d). The

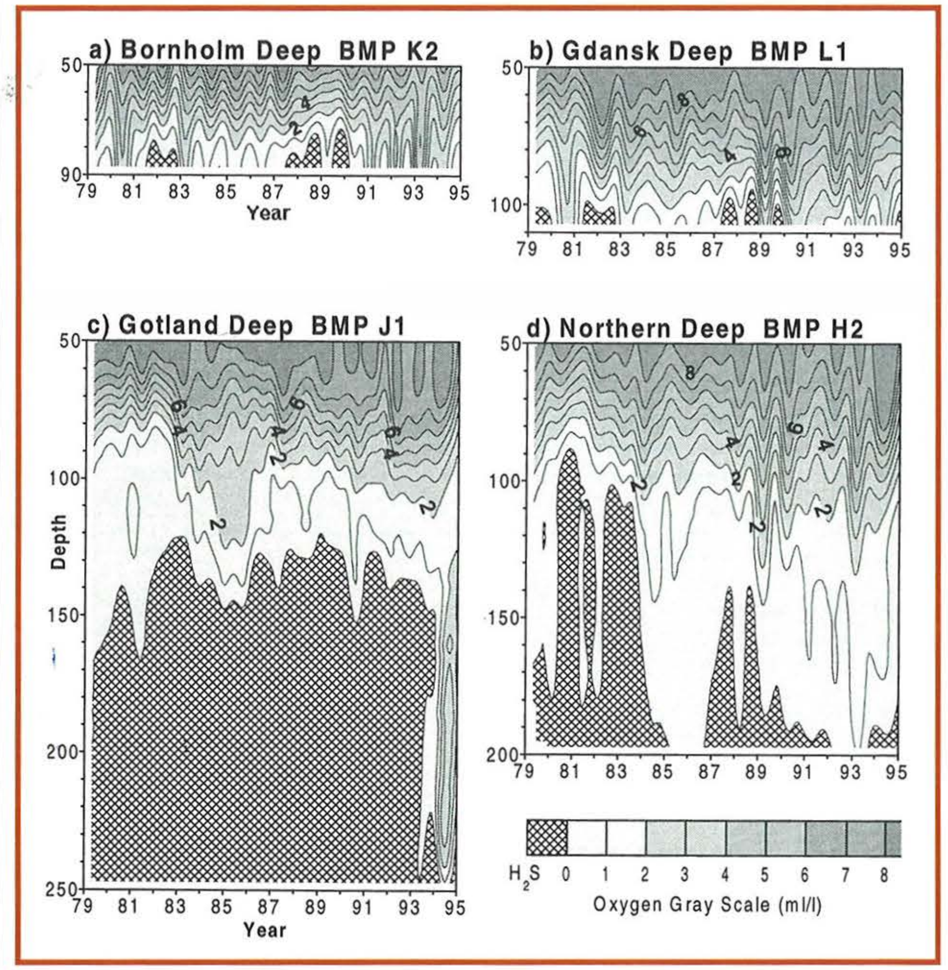


Fig. 4.4.4 Oxygen as a function of time (contour interval 1 cm^3 dm^{-3}, shaded area represents anoxic layers)

temperature at the isohaline 9.5 psu increased from a typical range of 3.6-4.2 °C to 5.4 °C in the Gotland Basin and to 5 °C in the Northern Baltic Proper.

4.4.1.3 Oxygen and hydrogen sulphide

Since the 1983 inflow event, oxygen depletion was observed below 50 m in the **Bornholm Basin**. up to 1990. The low-oxygen bottom water layer, with concentrations of <3 cm^3 dm^{-3}, increased in thickness, i.e., its upper boundary rose from a depth of about 75 m to 65 m during this period (Fig. 4.4.4). Anoxic conditions appeared below 80 m in 1988 and in 1989. Beginning in 1991, seasonal inflows created peak increases of the bottom-oxygen concentrations, reaching 4 cm^3 dm^{-3} in 1992 and more than 5 cm^3 dm^{-3} during the major inflow of 1993. Water from the depth range 50-60 m, overflowing to the Gdansk Basin, was characterised by concentrations of 6-8 cm^3 dm^{-3} during 1983-87, 4-6 cm^3 dm^{-3} during 1988-89 and 5-7 cm^3 dm^{-3} during 1990-94.

In the **Gdansk Basin**, the spring maximum of oxygen in the shallow coastal waters was generally observed in March/April, and approximately one month later in offshore waters. Due to regular supply of nutrients, water in the Vistula Estuary (Fig. 4.4.1, station *ZN2*) remained oversaturated with oxygen throughout the spring and summer, whereas in other areas the equilibrium with the atmosphere restored water conditions to normal saturation values in August or September (Fig. 4.4.5). The autumnal population of phytoplankton produced only a slight rise in oxygen concentration in the Gdansk Deep (*L1*), and this was detectable as late as November. In 1989-93,

Fig. 4.4.5 Seasonal development of water saturation with oxygen (%) in the Gdansk Basin, averaged for 1979-88 and 1989-93, respectively

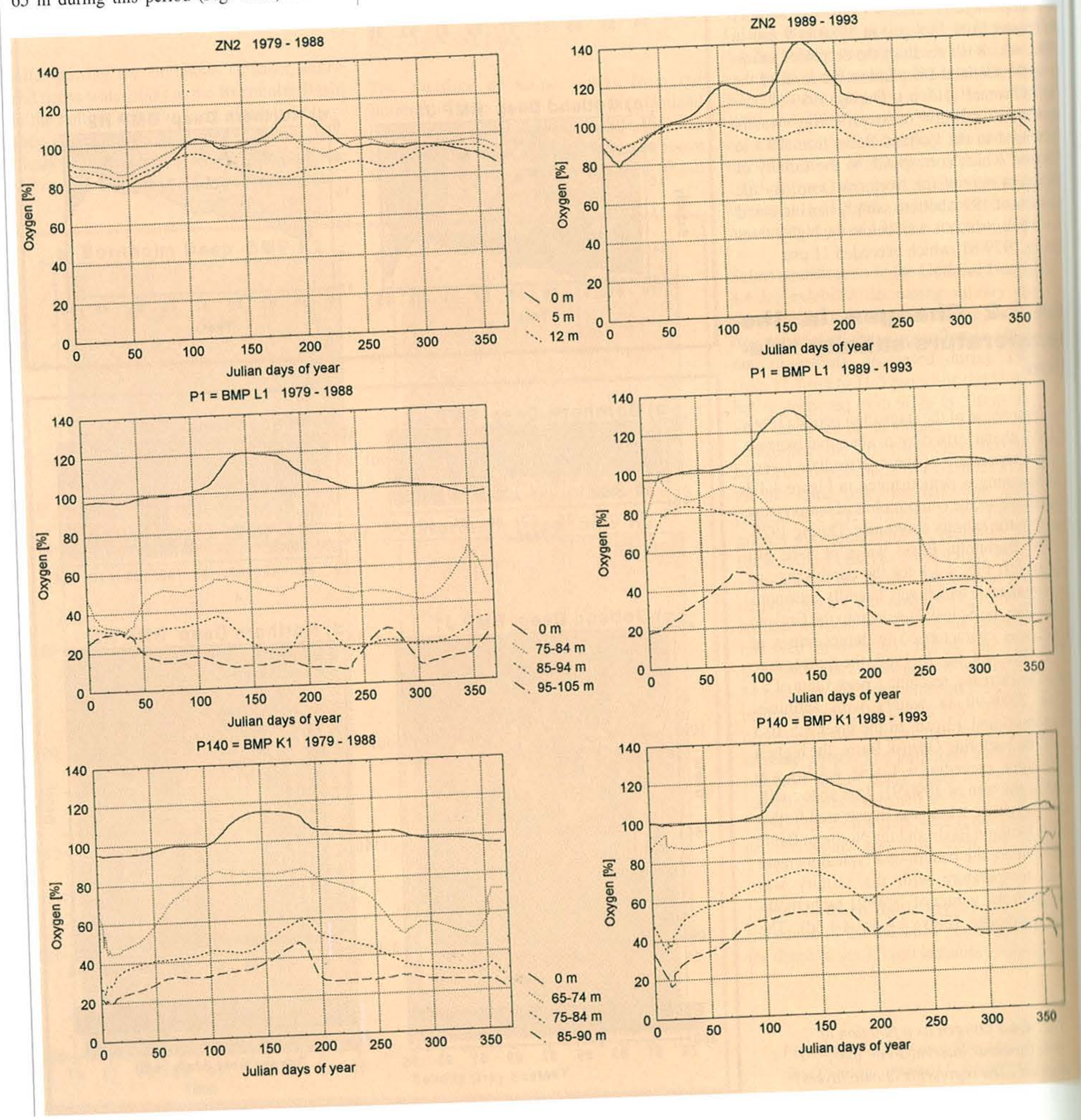


Table 4.4.2 Statistically significant oxygen trends (non-parametric test) in the Gdansk Basin, 1979-93 ($p <0.1$)

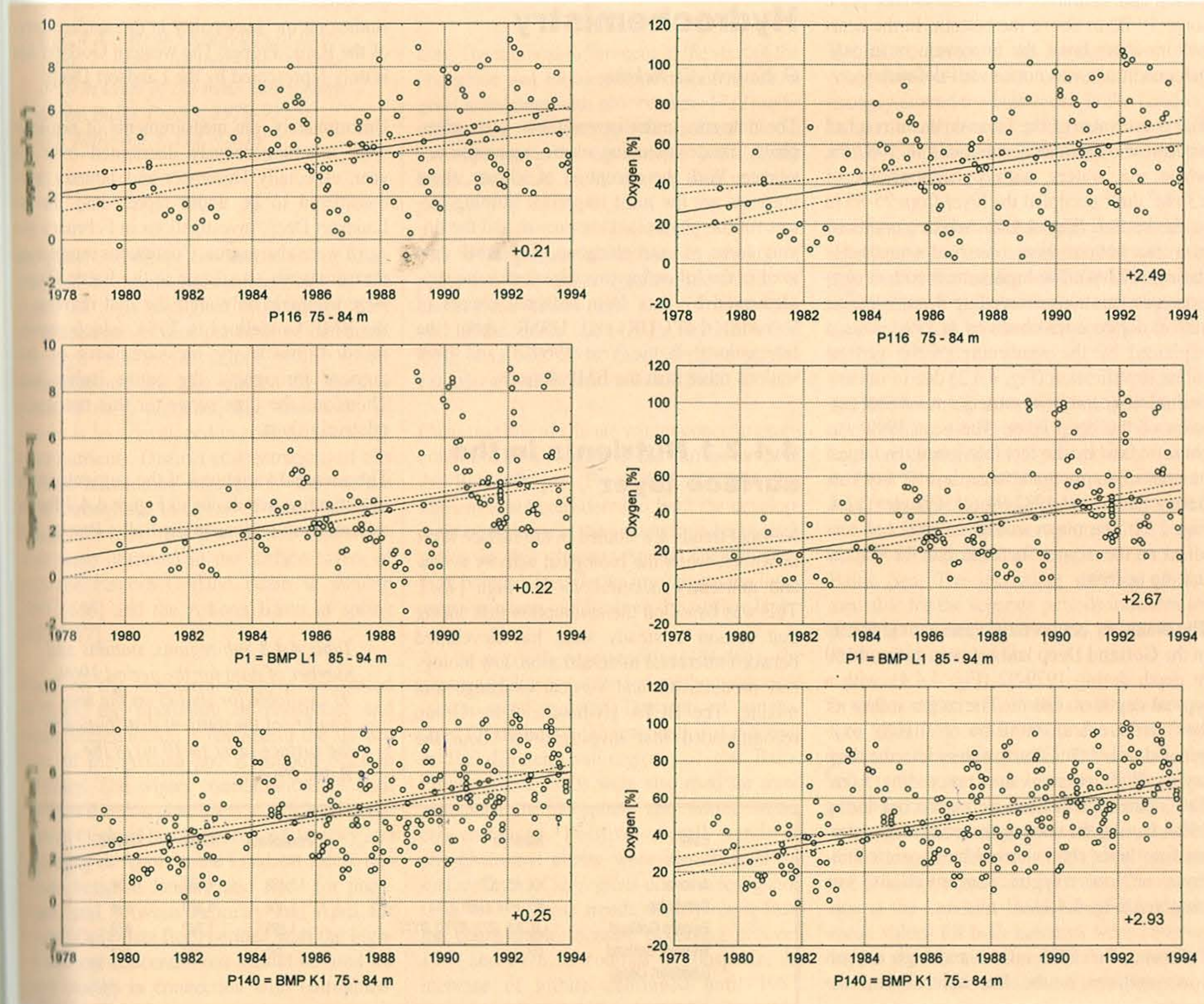
Basin / station	Depth (m)	O_2 concentration mean (cm^3 dm^{-3})	O_2 concentration slope (cm^3 dm^{-3} yr^{-1})	O_2 saturation mean (%)	O_2 saturation slope (% yr^{-1})
Gdansk Bight					
ZN2	0-2	7.82	0.11	101.7	1.45
	10-12	7.18	0.07	91.1	0.91
P116	65-74	6.37	0.16	73.2	2.12
	75-84	4.25	0.25	50.4	2.96
	85-90	2.95	0.13	36.3	1.90
Gdansk Deep					
L1 (P1)	65-74	6.77	0.18	77.9	2.05
	75-84	4.93	0.25	57.5	2.89
	85-94	3.35	0.24	40.7	3.03
	95-105	1.96	0.13	28.2	1.67
Northern Gdansk Basin					
K1 (P140)	65-74	6.79	0.12	77.2	1.66
	75-84	4.45	0.21	51.4	2.70
	85-90	3.13	0.13	37.5	1.51

the peak of oxygen saturation in the euphotic layer appeared earlier and was higher at all stations in the Gdansk Basin, compared with the previous 10-year period.

The effect of oxygen consumption, through the decomposition of allochthonic and autochthonic organic matter, which increased throughout the productive season, was detectable at the bottom of even relatively shallow stations. In contrast, the weakened density structure of the water column in the deep water zone and the temporary disappearance of the halocline during stormy weather [*125*] allowed the penetration of atmospheric oxygen into deeper water layers. These mechanisms continued to be active in 1993, even though the January

Fig. 4.4.6 Long-term trends of oxygen conditions in the deep waters of the Gdansk Bight (P116, P1=L1) and of the northern Gdansk Basin (P140=K1)

(Trend slopes in cm^3 dm^{-3} yr^{-1} and in % yr^{-1}, respectively)



inflow only affected the Gdansk Deep to a low degree [*434*].

In the near-bottom water layer of the Gdansk Basin, the oxygen deficiency reaches its maximum usually in late summer/early autumn. In 1989-93, this deficiency was, however, significantly reduced (Fig. 4.4.5). Hydrogen sulphide was last detected there in spring 1990 [*278*], in concentrations two orders of magnitude lower (0.3 μmol dm^{-3}) than those observed in previous years [*663*]. The area affected by hydrogen sulphide, which in 1989 stretched well into the Gdansk Bight, had decreased. From then on, until October 1994, the Gdansk Deep was free of hydrogen sulphide.

The surface and near-bottom oxygen concentrations increased in the Vistula Estuary. In other areas of the Gdansk Basin, this increase was only found to be significant in deep waters, i.e., below the depth of 65 m (Table 4.4.2, Fig. 4.4.6). Using the non-parametric *Mann-Kendall* test [*251,252,590,592*], the maximum rate of increase in oxygen concentration and saturation was found to take place some 10-20 m above the bottom. In the near-bottom water layer, the improvement in oxygen conditions was not so well-defined.

The deep water of the Gdansk Basin reached its lowest oxygen concentrations in 1987/88, while the waters with concentrations of <3 cm^3 dm^{-3} occupied the layer from 75-85 m to the bottom (Fig. 4.4.4). Anoxic conditions near the bottom were observed temporarily during 1987-90. The highest near-bottom oxygen concentrations, exceeding 4 cm^3 dm^{-3} at 105 m depth, were observed in 1990. This is explained by the weakening of the vertical saline stratification (Fig. 4.4.2) due to intensified mixing, and by a subsequent vertical aeration of the deep layer. The year 1990 was characterised by the fact that it had the largest number of stormy/gale force days of any year during the period 1982-94 (cf. Chapter 2.2.4, Fig. 2.7). The major inflow in 1993 had less effect on the oxygen increase than the vertical mixing in 1990.

The boundary between oxic and anoxic waters in the **Gotland Deep** laid between 110 and 160 m depth during 1979-92 (Fig. 4.4.4), with a typical depth of 130 m. The major inflow of 1993 produced an intrusion of slightly oxygenated (<1-2 cm^3 dm^{-3}) waters into the deep layers. High oxygen values, exceeding 3 cm^3 dm^{-3} in the bottom layer, were observed during 1994. During the deep water renewal, an intermediate layer characterised by anoxic conditions or low oxygen concentrations was observed (Fig. 4.4.11).

Following the 1983 inflow and high oxygen concentrations in the Bornholm Basin, the intermediate layers (100-130 m) of the Gotland Basin received greater supplies of oxygen during 1984-85. The layer of water with oxygen concentrations of 2 cm^3 dm^{-3}, which had the general tendency of deepening from 80 m in 1981 to 110 m in 1993, sank to >120 m during 1984-85. By the end of the stagnation period, the deepened and weaker halocline led to higher oxygen concentrations at the 70-90 m depths.

Anoxic conditions were found in the deep layers of the **Northern Baltic Proper** (Fig. 4.4.4) during 1980-84 and 1987-90. The presence of near-bottom oxygen in 1985-86 could be partly explained by the transport of waters from the Gotland Basin above 115 m depth, which had higher oxygen concentrations in 1984-85 (Fig. 4.4.4). After 1991, the decreased vertical stratification (Fig. 4.4.2) facilitated more intense vertical mixing, and since then anoxic conditions have not been found in the deep layers.

4.4.2 Hydrochemistry

G. Nausch[1], D. Nehring

The nutrients under investigation were phosphate, nitrate including nitrite, ammonia and silicate. With the exception of silicate, these nutrients are the most important limiting factors for the phytoplankton growth and the driving force of eutrophication. The BMP data used in the following investigations were supplemented by data from seasonal cruises of Sweden, the GDR and USSR, from the International Baltic Year 1969/70 and from stations other than the BMP stations.

4.4.2.1 Nutrients in the surface layer

Nutrient trends are studied in the surface layer in winter, when the biological activity is low and nutrient concentrations are high [*481*]. This was based on the assumption that during that season a steady state has developed between microbial mineralization, low biological productivity and vertical exchange and mixing. The BETA Hydro-chemistry Group recommended that investigations about the seasonality of nutrients should be aimed at the identification of their maxima or plateaus in the different regions of the Baltic Sea. Since the nutrient concentrations between the water surface and 10 m depth do not differ significantly all year-round, the mean values of this layer were used.

Within offshore areas of particular sub-regions of the Baltic Proper, the differences in nutrient concentrations are small, except during the spring outburst and the autumn bloom of the phytoplankton. These short periods are characterised by patchiness in the surface layer due to variations in the meteorological, hydrographic and biological conditions. Independently of that, the data of stations with long nutrient time series were summarized for particular sub-regions of the Baltic Proper and treated as uniform data sets (cf. [722]), with the aim of increasing the number of data available for investigations into the seasonality and trends in the surface layer.

Table 4.4.3 shows the sub-regions with stations summarized and the number of data for phosphate, nitrate and silicate available for studies on the seasonality in the surface layer of the Baltic Proper. The western Gotland Sea is only represented by the Landsort Deep.

Unfortunately, the measurements of nutrients were inhomogeneously distributed over the year, especially December and January were considered to be under represented. In the Landsort Deep, investigations in February and April were also scarce. Continuous monitoring measurements on silicate in the Baltic Proper were available only after the first revision of the BMP Guidelines in 1984, which recommended mandatory measurements of this nutrient throughout the entire Baltic Sea. Therefore, the time series for this nutrient is relatively short.

The seasonal variations of the nutrients under investigation are shown in Figure 4.4.7 for the different sub-regions in the Baltic Proper. The phosphate and nitrate concentrations are characterised by pronounced seasonality. In the period of high biological productivity, they often decrease to near the limit of detection. The high standard deviations of the winter concentrations are also caused by positive trends of these nutrients. The seasonality of silicate is less developed in comparison to the other nutrients. Distinct concentrations of this nutrient remain, on average, in the surface layer throughout the year. It must be mentioned, however, that low silicate concentrations were observed in the surface layers of both the Eastern Gotland Basin in autumn 1993 [*486*] and the Arkona Basin in spring 1995 [*487*].

Table 4.4.3 *Sub-regions, stations and number of data for the period 1969-93 available for studies on the seasonality of the nutrient distribution in the surface layer (0-10 m) of the Baltic Proper*

Basin	Stations	Phosphate	Nitrate	Silicate
Arkona	*K4, K5, K7*	1,488	1,396	434
Bornholm	*K2, BY4, GER 214*	1,408	1,379	595
Eastern Gotland	*J1, K1, BY9, BY10, BY11*	1,899	1,742	873
Western Gotland (Landsort Deep)	*H3*	295	273	133

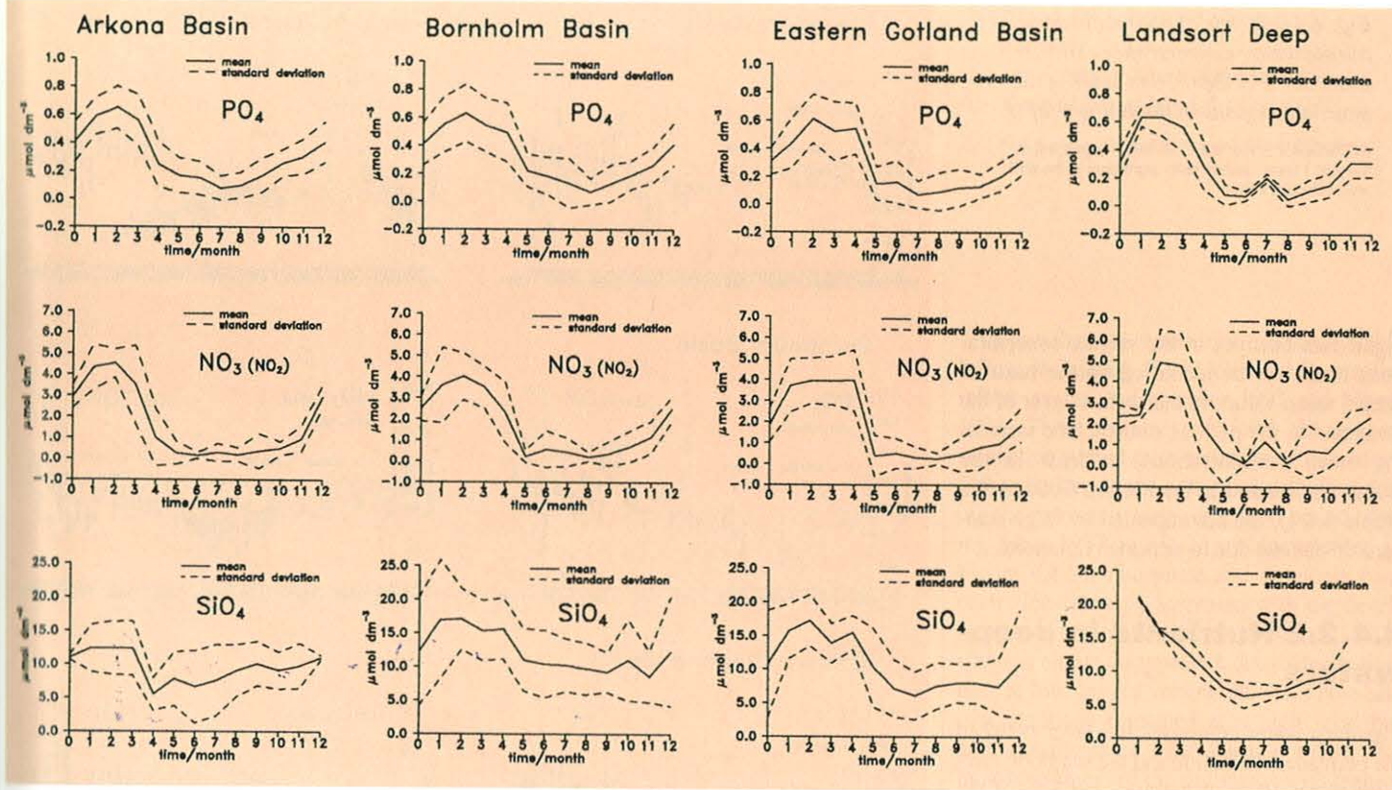


Fig. 4.4.7 Mean seasonality of inorganic nutrients (in μmol dm^{-3}) in the 0-10 m layer of the main sub-regions in the Baltic Proper, 1979-93 (cf. Table 4.4.3, Fig. 4.4.1)

Based on the long-term mean of the period 1979-93, the maximum of phosphate and nitrate accumulation is reached in the surface layer of the Arkona and Bornholm Seas in February. The winter concentrations of both nutrients develop plateaus at a high level in the Eastern Gotland Sea between mid-January and beginning of April. In the Landsort Deep, this occurs between January and April for phosphate, and between February and April for nitrate. Only data from periods with the highest nutrient concentrations should be used for trend studies in connection with eutrophication. The regional differences at the start of the phosphate and nitrate impoverishment are in good agreement with observations [*312*] indicating that the springphytoplankton bloom develops earlier in the western parts of the Baltic Sea than in its eastern and northern parts.

The investigations about the seasonality of phosphate and nitrate confirm the results of earlier trend studies in the Baltic Proper at the end of the 1970s (cf. [*477*]). The HELCOM BMP data sets allow more detailed studies extending over the entire Baltic Sea.

Clear maxima of silicate winter concentrations could not be identified in the surface layer of the Baltic Proper (Fig. 4.4.7). To date, this nutrient is not considered to limit the development of diatoms in the subareas under investigation in this chapter. Annual mean values may be used, in order to investigate the long-term variations of silicate in the surface layer.

As in previous assessments [*484,485*], the parametric linear regression method was used for the trend analysis. The stations summarized for studies on the seasonality of nutrients in the different sub-regions of the Baltic Proper (Table 4.4.3) were also used for trend studies of the phosphate and nitrate winter concentrations. Positive overall trends of phosphate and nitrate were identified in the surface layer of all regions under investigation (Fig. 4.4.8). These trends mainly result from the considerable increase occurring between 1969 and 1978. In contrast to phosphate, the increase of nitrate continued until 1983. Thereafter, except in the Landsort Deep, the concentrations of both nutrients fluctuated strongly around the high level. In the Landsort Deep, the increase of phosphate and nitrate winter concentrations continued during the recent assessment period.

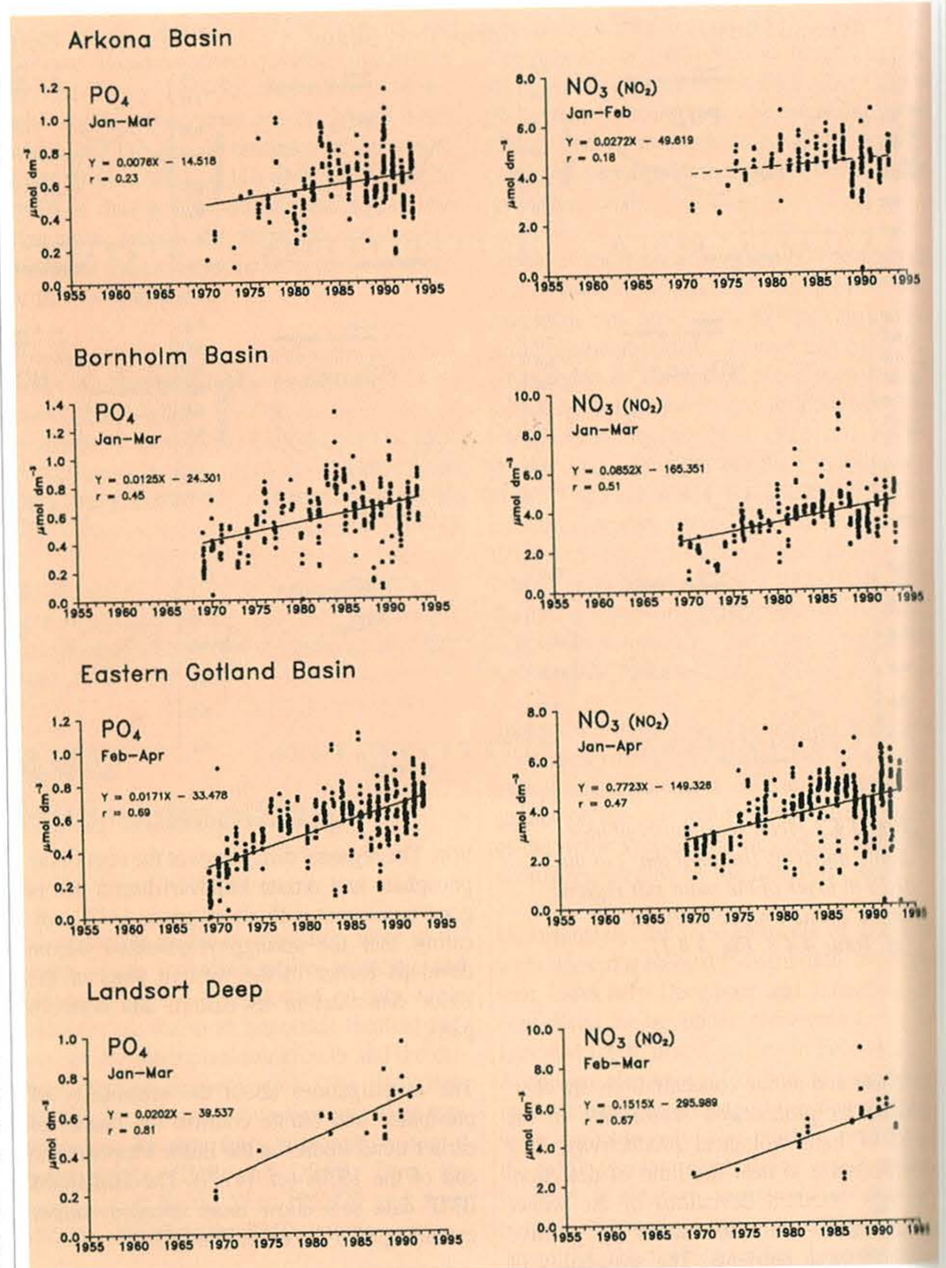
In contrast to the Kattegat and Belt Sea areas (cf. Chapter 4.5), no correlations between the inter annual fluctuations of the nutrient winter concentrations and the fresh water discharge by rivers could be identified for the offshore stations in the Baltic Proper.

In Table 4.4.4, the mean winter concentrations of phosphate, nitrate and silicate in the surface layer are averaged for 5-year periods starting in 1969. These periods are supplemented by data from a previous 11-year period consisting of a low number of observations at the beginning of monitoring studies on nutrients in the Baltic Sea. The increasing number of data available for the separate periods indicates the growing monitoring activities in the Baltic Proper in recent decades.

In agreement with the trend studies in the surface layer (Fig. 4.4.8), the mean winter concentrations, summarized in Table 4.4.4, also reflect the significant increase of the phosphate and nitrate pool at the beginning of the investigations. During recent decades, the concentrations remained at a high level but without a further significant increase. The exception is the Landsort Deep, where the highest mean values for both nutrients were observed in the present assessment period, exceeding those in the other areas under investigation.

Fig. 4.4.8 Trends of phosphate and nitrate winter concentrations (in µmol dm^{-3}) in the 0-10 m layer of the main sub-regions in the Baltic Proper

(cf. Table 4.4.3, Fig. 4.4.1; full lines - significant at the p<0.1 level, dashed line - significant at the p>5 level (Student's Test))



Significant changes in the silicate concentrations could not be identified on the basis of annual mean values in the surface layer of the areas and in the periods studied. The increasing mean concentrations, at least in the Bornholm Sea and in the Eastern Gotland Sea (Table 4.4.4), are accompanied by large standard deviations due to seasonal variations.

4.4.2.2 Nutrients in deep waters

The deep water comprises the layer between the permanent halocline and the sea floor, covering about 10 % of the area and 5 % of the volume of the Baltic Proper. Since the halocline strongly restricts vertical mixing, changes in the hydrochemical variables in the deep water are mainly caused by advective processes, vertical diffusion, microbial destruction of organic matter and the exchange with the sediments.

The Bornholm Basin is the most western basin in the Baltic Proper where stagnant conditions can persist for longer than one year. Its deep water is more often affected by advective processes as that of the Eastern Gotland Basin, which is only renewed after major inflow events. Water masses passing the Eastern Gotland Basin in the intermediate water layers may also renew the deep water of the northern and western basins of the Gotland Sea. With decreasing stability of the halocline, vertical mixing may cause oxygen and nutrient fluctuations in the deep water of the western and northern parts of the Baltic Proper.

Nitrate trends have been identified in the Gotland Deep and in the Landsort Deep area, at 100 m depth (Fig. 4.4.9) characterising the dynamically active deep layer. The average increase was stronger in the Landsort Deep. Phosphate concentrations increased significantly on average at this depth only in the Gotland Deep in the whole period under investigation, but were decreasing in the recent assessment period. This also applies to the Landsort Deep. With some exceptions, the ammonia concentrations were low (<1-1.5 µmol dm^{-3}) at the 100 m depth in both areas.

The long-term behaviour of the phosphate and nitrate concentrations is affected by fluctuations during shorter periods. These fluctuations reflect the intensity of advective processes, since hydrogen sulphide, affecting strongly the nutrient distributions, has not been observed in 100 m depth of the Gotland Deep and Landsort Deep until now, except in autumn 1982 in the Landsort Deep [*480*]. In spite of the temporarily limited data sets, clear reductions of silicate concentrations occurred at the 100 m depth and in the bottom layer of the Gotland and Landsort Deeps (Figs. 4.4.9, 4.4.10).

The distribution of nutrients in the near-bottom water layer is dominated by stagnation periods and advective water renewals. Very strong variations are observed, when the redox potential changes its sign, due to the replacement of hydrogen sulphide by oxygen and vice versa. Physicochemical and biogeochemical processes like removal and remobilisation of phosphate together with iron from the sediments, denitrification and nitrification, as well as accumulation of ammonia, strongly depending on the redox potential, are reflected in the nutrient concentrations. Silicate concentrations also increase during stagnation periods and decrease as a consequence of water renewals. The sequence of stagnation and anoxia, followed by water renewals, masks nutrient trends in the near-bottom water layer.

The bottom water of the Bornholm Deep exhibits strong variability in nutrients compared with other central basins (Fig. 4.4.10). The temporary improvement of the oxygen conditions, observed in the Bornholm Deep since the beginning of the 1990s, was accompanied by low phosphate, ammonia and silicate, and high nitrate concentrations.

The most important hydrographic event in the present assessment period was the bottom water renewal in the Eastern Gotland Basin

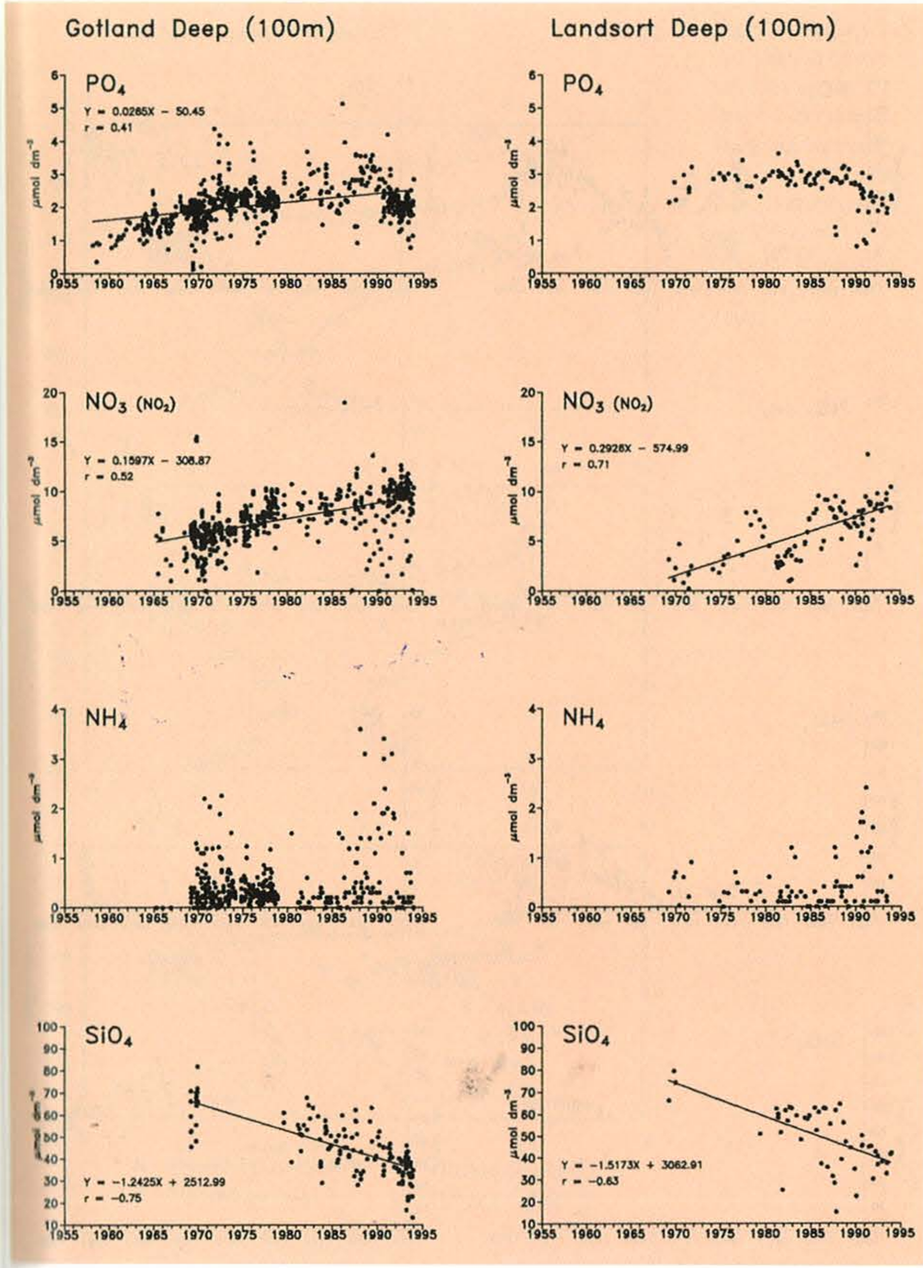


Fig. 4.4.9 Long-term changes of nutrients (in µmol dm^{-3}) in intermediate depths (100 m) of the Gotland Deep (J1) and Landsort Deep (H3)

(full lines - significant at the p<0.1 level (*Student's Test*))

This was the consequence of the major Baltic inflow in January 1993, followed by smaller inflow events in December 1993 and March 1994. These events terminated the longest stagnation period ever observed in this basin. The variations in the nutrient distribution during the extremely long stagnation period, lasting from 1977 to 1992, are documented in Figure 4.4.10. Phosphate and ammonia concentrations strongly increased with the development of anoxic conditions in the late 1970s, whereas nitrate disappeared, due to denitrification, at low oxygen concentrations. Phosphate concentrations remained at a high level but have not increased since the late 1980s. The impoverishment of the sediments with respect to remobilisable phosphate was discussed in this connection [*478*]. It is possible to observe relationships between the restricted phosphate pool in the bottom water and the decreasing concentration of this nutrient in 100 m depth (Fig. 4.4.9). The deep water renewals, observed in the Gotland Deep in 1993 and 1994, caused a sudden decrease of phosphate, ammonia and silicate, and an increase of nitrate. The reasons for this have already been discussed. Figure 4.4.11 shows the vertical structure of nutrient variables during water renewal in the Gotland Deep, indicating intermediate phosphate and silicate maxima. The close proximity of the nitrate minimum and maximum, in the range of low oxygen concentrations, is the result of nitrification and denitrification processes which depend on the redox potential.

In contrast to the Gotland Deep, but similar to the Bornholm Basin, the bottom water of the Landsort Deep was also renewed in the mid-1980s. This renewal was not only reflected by the hydrographic parameters but also by the distribution of nutrients (Fig. 4.4.10).

	Arkona Basin	Bornholm Basin	Eastern Gotland Basin	Landsort Deep
Nitrate	*Jan-Feb*	*Jan-Mar*	*Jan-Apr*	*Feb-Mar*
1958-68		0.69 ± 0.37, *(3)*	0.97 ± 0.36, *(5)*	
1969-73	2.68 ± 0.14, *(4)*	2.08 ± 0.73, *(34)*	2.44 ± 0.40, *(75)*	2.57, *(3)*
1974-78	3.93 ± 0.78, *(14)*	3.16 ± 0.56, *(46)*	3.81 ± 0.94, *(61)*	2.72, *(2)*
1979-83	4.61 ± 0.83, *(20)*	3.89 ± 0.94, *(53)*	4.15 ± 0.93, *(60)*	4.48 ± 0.53, *(6)*
1984-88	4.74 ± 0.46, *(35)*	4.37 ± 1.16, *(90)*	4.30 ± 0.99, *(132)*	3.71 ± 1.50, *(6)*
1989-93	4.17 ± 0.79, *(84)*	3.90 ± 0.83, *(89)*	4.23 ± 1.28, *(200)*	5.58 ± 0.71, *(12)*
Phosphate	*Jan-Mar*	*Jan-Mar*	*Feb-Mar*	*Jan-Mar*
1958-68		0.32 ± 0.15, *(22)*	0.27 ± 0.09, *(24)*	
1969-73	0.26 ± 0.07, *(5)*	0.35 ± 0.12, *(50)*	0.26 ± 0.14, *(95)*	0.20 , *(3)*
1974-78	0.49 ± 0.12, *(16)*	0.56 ± 0.14, *(46)*	0.54 ± 0.09, *(61)*	0.42 , *(2)*
1979-83	0.48 ± 0.11, *(50)*	0.58 ± 0.21, *(53)*	0.59 ± 0.16, *(60)*	0.58 ± 0.04, *(6)*
1984-88	0.67 ± 0.09, *(52)*	0.67 ± 0.19, *(88)*	0.60 ± 0.15, *(125)*	0.54 ± 0.05, *(6)*
1989-93	0.61 ± 0.17, *(155)*	0.64 ± 0.18, *(89)*	0.67 ± 0.11, *(182)*	0.72 ± 0.09, *(15)*
Silicate	*Jan-Dec*	*Jan-Dec*	*Jan-Dec*	*Jan-Dec*
1979-83	8.17 ± 4.66, *(122)*	9.98 ± 6.13, *(118)*	8.30 ± 5.14, *(158)*	10.62 ± 3.85, *(31)*
1984-88	8.65 ± 3.54, *(124)*	11.26 ± 5.53, *(240)*	9.01 ± 5.30, *(304)*	8.17 ± 3.59, *(51)*
1989-93	10.73 ± 4.04, *(188)*	12.09 ± 4.11, *(273)*	10.94 ± 4.94, *(411)*	10.93 ± 3.81, *(51)*

Table 4.4.4 Winter concentrations of phosphate and nitrate, and annual silicate mean values in the surface layer (0-10 m) of main sub-regions of the Baltic Proper, averaged for 5 and 11 years, respectively

(mean value, standard deviation; *(number of data)*; for stations cf. Table 4.4.3; values in µmol dm^{-3})

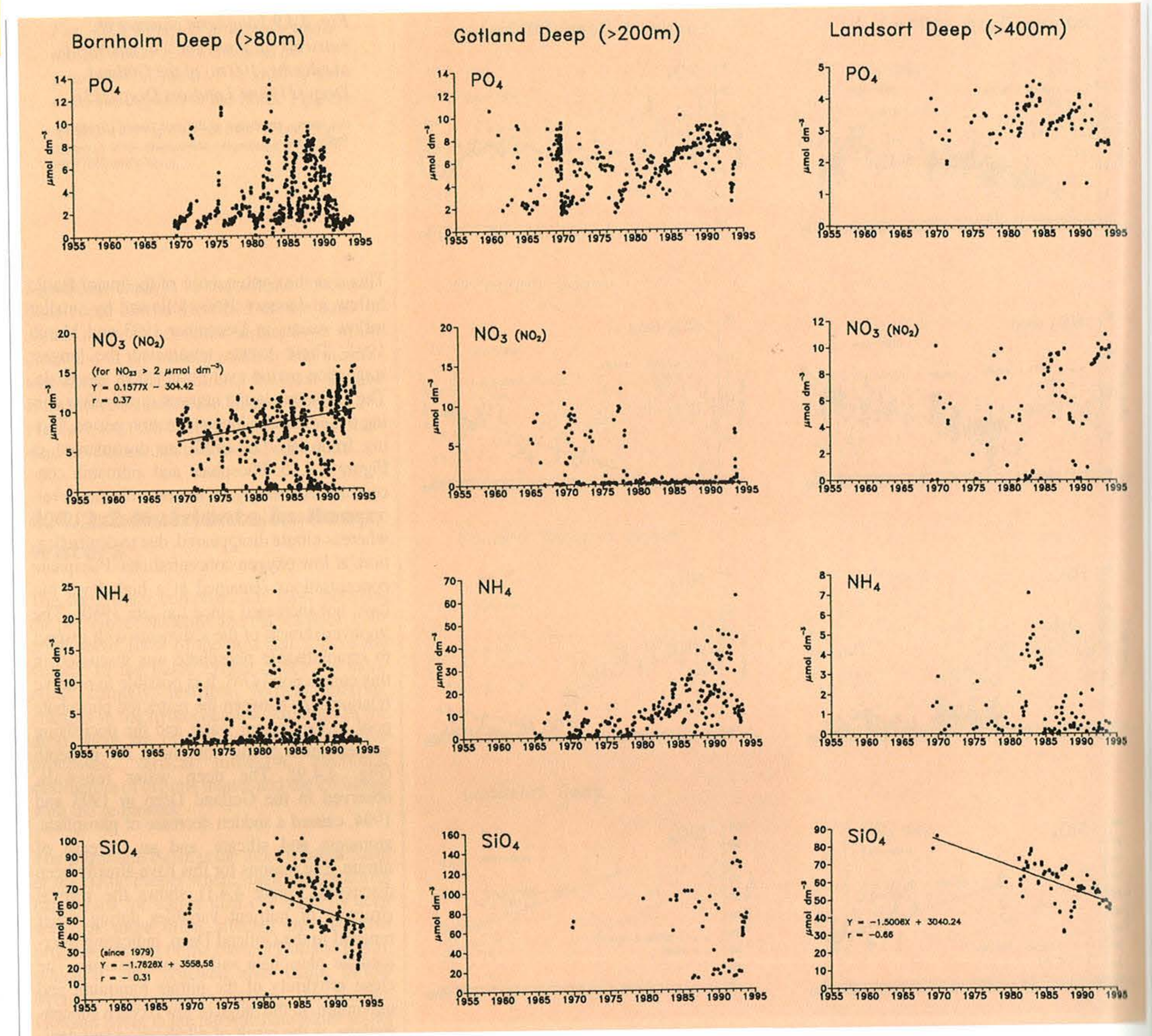


Fig. 4.4.10 *Long-term changes of nutrients (in µmol dm*$^{-3}$*) in the near-bottom layer of the Bornholm Deep (K2; >80 m), Gotland Deep (J1; >200 m) and Landsort Deep area (H3; >400 m)*

(full lines - significant at the $p<0.1$ level (*Student's Test*))

4.4.2.3 Nutrients in the Gdansk Basin

A. Trzosinska[1], *E. Lysiak-Pastuszak*

The Gdansk Basin covers, conventionally, the Gdansk Bight and the Gdansk Deep, as far as the Southern Gotland Basin. The main reason for distinguishing it as a separate assessment subject is the great impact of rivers, which flow directly into the Gdansk Bight, like the Vistula, the second largest river in the Baltic Sea drainage area, or indirectly, through the Vistula Lagoon. The Gdansk Deep serves as a sedimentary basin in this diversified hydrological system, which is under strong anthropogenic pressure.

Seasonal and long-term variations of the nutrient distribution in 1989-93 are discussed in comparison with those in the previous 10-year period. The specific features of the Gdansk Basin required spatial representation by four stations, two national Polish stations, i.e., *ZN2* in the Vistula Estuary and *P116* in the central part of the Gdansk Bight, and two BMP stations, *L1* in the Gdansk Deep and *K1* at the boundary with the Gotland Basin, for which comparable data sets were available (Fig. 4.4.1). The number of data from each of the four stations, which were used to compute seasonal variations and long-term trends in different water layers and seasons, ranged between 60 and 300 for the period 1979-93. The data covered the 15-year period quite evenly.

4.4.2.3.1 Seasonal variability

Seasonal fluctuations were computed by means of the *SEASONAL* software [*6*], in which gaps between measurements were filled with estimates, calculated for each day of the year as the weighted mean values. Spline functions were used for smoothing.

The Gdansk Basin is highly diversified as regards nutrient supply. This is illustrated in Figure 4.4.12, with examples on the seasonal development of nutrient concentrations in the Gdansk Bight, in an area, directly affected by discharges from land. Figure 4.4.13 provides examples of seasonal variations in the Gdansk Deep and in the northern part of the Gdansk Basin (*P140=K1*). In both figures, nutrient fluctuations during 1979-88 and 1989-93 are compared.

The annual load of total phosphorus, discharged by the Vistula River into the Gdansk Bight in 1972–93, ranged from 4,000 t in 1982 to 11,000 t in 1978 [*499*]. Using data covering the period 1985-89 [*148*], the mean phosphorus load discharged from the Vistula Lagoon was calculated at 1,200 t. On average, 73 % of the total phosphorus load is derived from the river run-off [*583*]. The spring high water in the Vistula lasts from January to May, with the maximum of the long-term flow rate in April [*126*]. In 1989–93, the largest phosphorus loads were carried down the Vistula between the end of March and the beginning of April [*278*]. These estimates of the phosphorus loads discharged into the Gdansk Bight are considered to be realistic. The calculation of their impact on the surface-water layer (0-20 m) in the Gulf yielded exactly the same range of phosphate and total phosphorus concentrations as observed in recent years (Fig. 4.4.12).

The spring development of phytoplankton in the Gdansk Bight begins in March, and in the northern part of the Gdansk Basin in April. The increased demand for nutrients coincides thus with their maximum discharge from land. This buffers the spring drop in nutrient concentrations to a degree dependent on the position of the station, but chiefly on the convergence of a number of climatic factors, determining the flow rate of rivers, the loads they carry, and the start of the growing season in any specific year. In 1979-88, the winter accumulation of **phosphate** reached a maximum in the first half of March, whereas in 1989-93 it had peaked in February (Figs. 4.4.12, 4.4.13). In its initial phase, which continued until around mid-March, the reduction in phosphate concentration occurred slowly, on average by 12 % off the Vistula mouth and by 20 % at more distant sites. In the deep water layers, the highest concentrations of phosphate were recorded in late summer early autumn, at the time when oxygen deficiency was the strongest. In the final phase of stagnation, only small quantities of phosphate were released from the bottom sediments, due to the improving oxygen conditions.

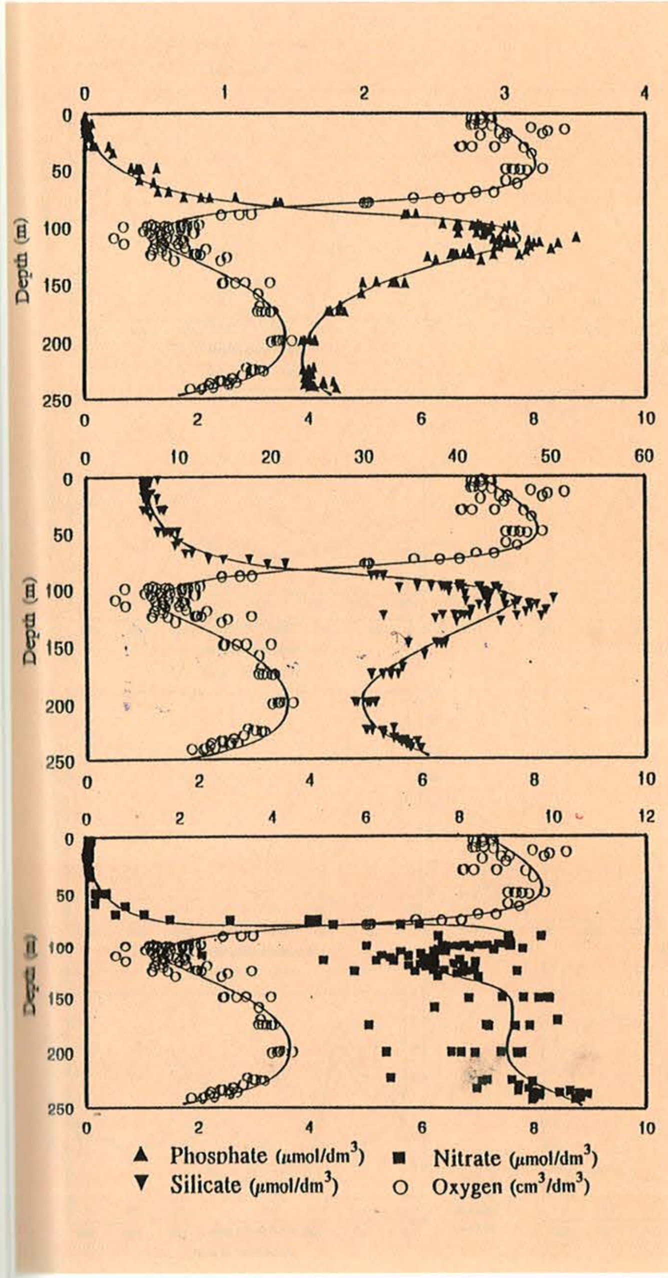


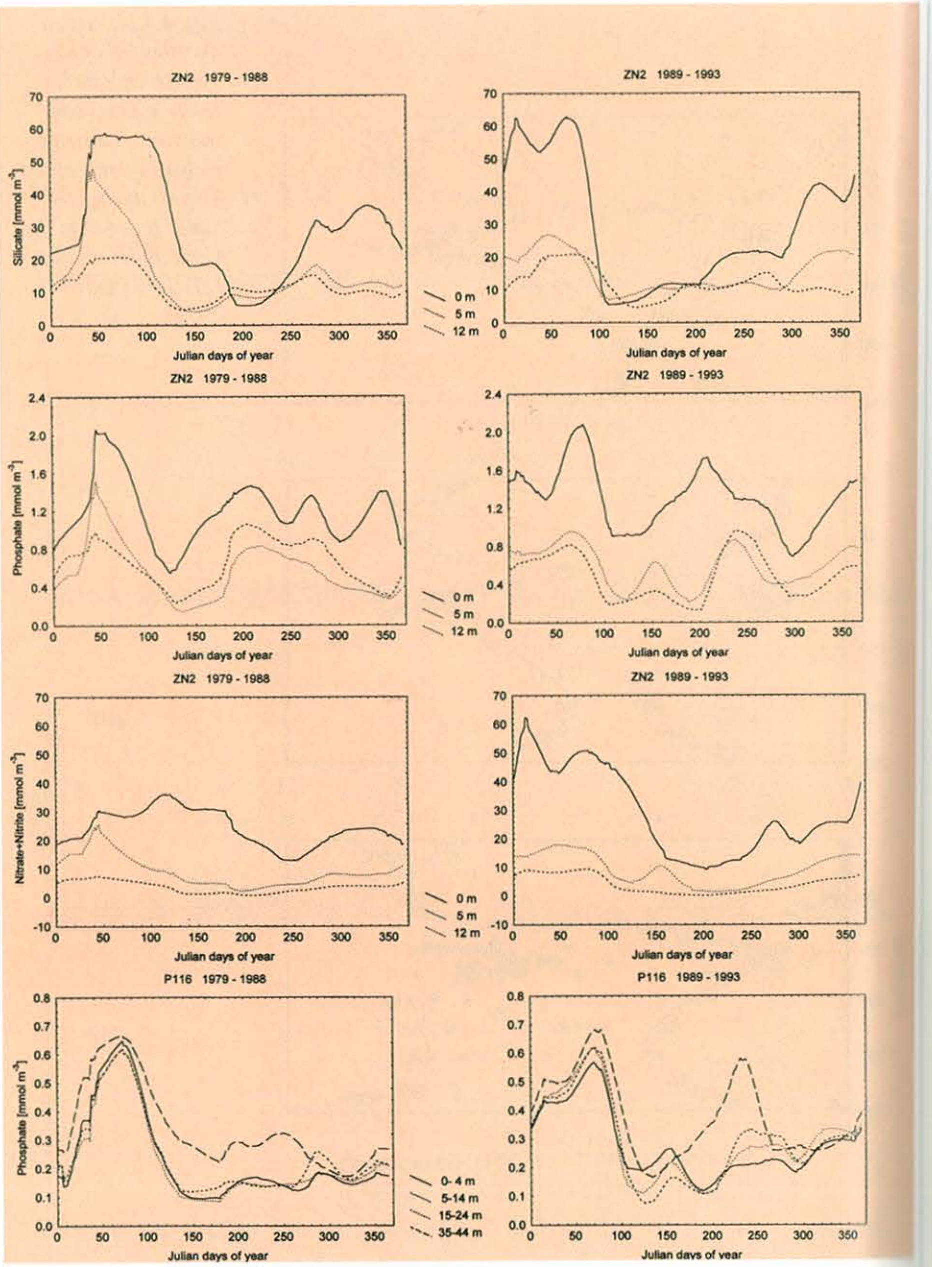
Fig. 4.4.11 *Vertical distribution of oxygen (in cm*^{3} *dm*$^{-3}$*; lower x-axes) and inorganic nutrients (in µmol dm*$^{-3}$*; upper x-axes) during the water renewal in the Gotland Deep area (J1; 17-27 July 1994)*

There were significant differences in the seasonal plot of **silicate** concentrations in 1989-93 compared with previous monitoring stages. Following its winter maximum, which, like the phosphate peak, had shifted in the surface water layer from March to February (Figs. 4.4.12, 4.4.13), silicate was gradually consumed until minimum concentrations were reached. This happened in the coastal zone of the Gdansk Bight in spring, but not before autumn in the deep areas. The second peak in autumn, characteristic for silicate in the Vistula Estuary, disappeared, as did the synchronic second minimum of concentrations in areas well away from the river mouth. The anomalies in the seasonal fluctuations of silicate, observed in the Gdansk Basin in 1989-93, can be partially put down to the low riverine flow rate and to changes in thermal conditions [*125*]. However, the substantial reduction in the annual amplitude of concentrations also suggests a decreasing demand for silicates due to changes in the phytoplankton species composition (cf. Chapter 4.4.3.1).

In 1989-93, winter **nitrate** concentrations did not change significantly in comparison with the previous decade [*660,661*]. The exception was the Gdansk Bight, particularly its southern part, where the winter nitrate concentrations and their annual amplitudes were 1.5 times higher than those previously recorded. The effects of the spring run-off of nitrogen compounds were evident throughout the coastal zone. However, by May or June, the nitrate content above the thermocline had decreased, in comparison with the winter levels, by 70-75 % in the coastal belt, and by 90 % elsewhere. In the entire area under consideration, except for the inner Gdansk Bight, summer nitrate concentrations fell to trace levels (0.1-0.5 µmol dm^{-3}). This produced conditions pro-

[1] see ANNEX 11.3 for addresses of authors

Fig. 4.4.12 Seasonal development of nutrient (in μmol dm^{-3}) concentrations in the Gdansk Bight, averaged for 1979-88 and 1989-93, respectively



moting the growth of cyanobacteria, which are able to make use of molecular nitrogen. Nitrate concentrations started to rise again in September or October. The replenishment of nitrate proceeded slowly until February, synchronously with that of the other nutrients in the upper water layers (Figs. 4.4.12, 4.4.13).

The situation in the Vistula Estuary, where between 1972 and 1993 about 55-165 kt of total nitrogen annually entered the Gdansk Bight [*499*], deserves special attention. The nitrogen load also fluctuated widely in 1989-93 (55-110 kt). Rivers carry to the sea more than 93 % of the total nitrogen load, two thirds of which is in the form of mineral compounds washed out from the drainage area during the high-water periods, usually in spring [*583*]. About 45 kt of total nitrogen enter per year the Baltic Sea from the Vistula Lagoon, including 6.5 kt in mineral forms [*148*]. In view of the relatively low flow rates of rivers in 1989-93, these loads accumulated in the coastal zone of the Gdansk Bight (Fig. 4.4.12).

The atmosphere is also an important source of assimilable nitrogen compounds. If the measurements taken at the meteorological station Łeba are believed to be representative for the Gdansk Bight, the mean flux of nitrate and ammonium nitrogen was 631 kg N km^{-2} yr^{-1} in the period 1989-93 [*278*]. Nutrient emissions from land-based sources fluctuate seasonally. The long-term maximum of mineral nitrogen outflow from rivers occurs in spring, while the airborne nitrate peak is found in summer [*721*]. Having been carried into the surface water of the Gdansk Bight, the top 20 m of which constitute a volume of 91.4 km^3 [*659*], these loads can generate concentration levels of the order of 40-70 μmol dm^{-3}. These figures were confirmed by monitoring results at station *ZN2*, where such amounts of nitrate and nitrite were recorded from January to April in 1989-93 (Fig. 4.4.12). Similarly, in the Gdansk Deep, a long distance from the Vistula Estuary, the winter nitrate concentration rose by 1 μmol dm^3 in comparison with the period 1979-88 (Fig. 4.4.13).

Due to the favourable oxygen conditions and low denitrification activity in deep waters, the nitrate concentrations were much higher in 1989-93 than those during the previous assessment periods. In the 1960s and 1970s, the ratios of assimilable nitrogen and phosphorus compounds (**N/P ratio**) in the water of the Baltic Sea were far lower than 16:1. However, it was observed that the elements were assimilated in proportions close to the *Redfield* ratio [*484*]. In the second half of the 1970s, there were clear signs that the N/P ratio was increasing in the upper water layers of the Gdansk Basin [*658*]. In fact, trophic conditions are unsteady throughout the Polish waters of the Baltic Sea, extremely so in the bights and along the coast [*662*].

Although the N/P ratio in water is taken into account when assessing the trophic level of the sea, the most important index to be used for the purpose is the ratio in which these elements are taken up by autotrophic organisms (dN/dP). The molar relationships between the concentration amplitudes of assimilable nitrogen and phosphorus compounds, calculated for the Polish sector of the Baltic Sea in 1989-93, were

$$dN_{in} = 20.59\ dPO_4 - 0.09 \qquad (R=0.97)$$
$$dNO_3 = 14.00\ dPO_4 + 0.66 \qquad (R=0.95)$$

The slope coefficients yield the average number of moles of nitrogen per mole of phosphorus, taken up by the spring species of phytoplankton in the open sea and in coastal zones and during the summer growing period in the open sea.

4.4.2.3.2 Long-term variations

Statistical trends in the long-term variations (1979-93) of nutrient concentrations and N/P ratios were investigated at four stations in the Gdansk Basin. Two computational methods were applied in parallel, the non-parametric

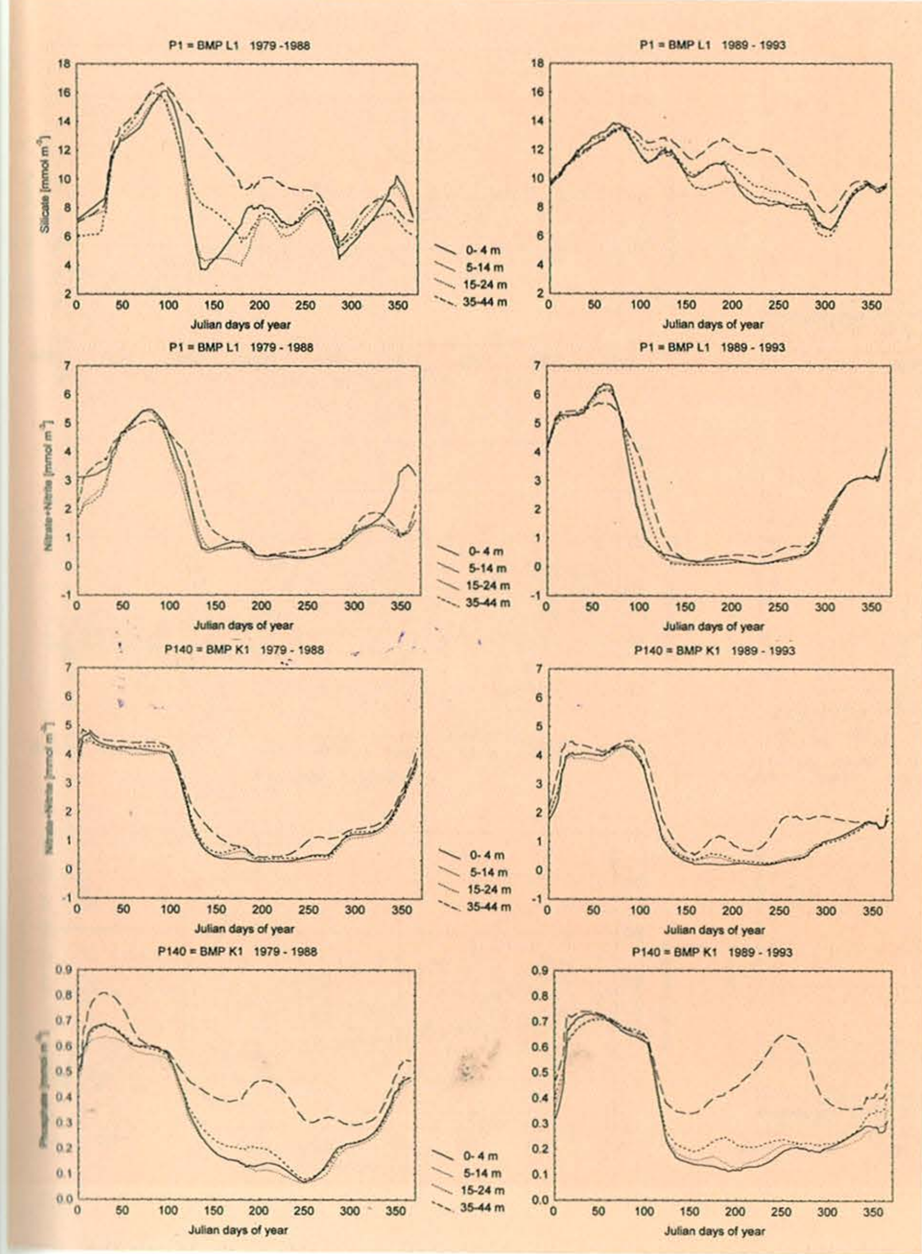


Fig. 4.4.13 Seasonal development of nutrient concentrations (in μmol dm^{-3}) in the Gdansk Deep area (P1=L1) and the northern Gdansk Basin (P140=K1), averaged for 1979-88 and 1989-93, respectively

Mann-Kendall test [*251,252,590,592*] and the least-square method, traditionally applied in the assessments of the Baltic Sea. The analysis encompassed 204 sets of data on the different water layers (at 10 m intervals) and seasons, one third of which yielded statistically significant changes. In 96 % of the cases, the results obtained by the two methods concurred as regards the significance level and slope coefficients. Tables 4.4.5 and 4.4.6 set out the results of the non-parametric test for those water layers, in which at least one of the components under consideration had changed significantly. Figures 4.4.14 and 4.4.15 give examples of trends determined by simple linear regression.

The analysis of long-term trends in nutrient variations showed that phosphate and silicate concentrations decreased significantly in deep waters (Table 4.4.5, Fig. 4.4.14), the current changes being to a high degree a continuation of the trends revealed earlier in the Gdansk Deep [*485*]. The most important differences are the disappearance of the negative trend in phosphate concentrations at 80 m depth, and the formation of weak, though statistically significant, positive year-round trends in the concentrations of silicate and phosphate in the upper water layers.

Table 4.4.5 Statistically significant year-round nutrient trends (non-parametric test) in the Gdansk Basin, 1979-93

($p < 0.1$; means in μmol dm^{-3}, slopes in μmol dm^{-3} yr^{-1}; N/P = $(NO_3+NO_2)/PO_4$)

Basin *station*	Depth *m*	NO_3+NO_2 mean	NO_3+NO_2 *slope*	PO_4 mean	PO_4 *slope*	N/P ratio mean	N/P ratio *slope*	Silicate mean	Silicate *slope*
Gdansk Bight									
ZN2	10-12	3.5		0.56		16.8	*0.69*	11.7	
P116	75-84	7.8		1.47	*-0.10*	8.1	*0.58*	29.5	*-2.0*
	85-90	7.9		2.13	*-0.16*	6.3	*0.48*	37.1	*-2.7*
Gdansk Deep									
L1 (P1)	0-4	1.9		0.27		8.9		9.7	*0.4*
	5-14	1.8		0.26	*0.01*	7.8		9.6	*0.3*
	15-24	1.9		0.29		8.5		10.2	*0.3*
	25-34	2.0		0.33	*0.01*	6.6		11.2	*0.4*
	35-44	2.1		0.28		7.0	*0.37*	11.8	*0.4*
	75-84	6.9		1.14		6.7	*0.24*	24.8	*-1.0*
	95-105	8.2		2.97	*-0.24*	4.5	*0.25*	47.1	*-1.5*
Northern Gdansk Basin									
K1 (P140)	0-4	1.6		0.30		4.8		10.5	*0.3*
	5-14	1.7		0.29		4.9		10.1	*0.3*
	15-24	1.7		0.32	*0.01*	4.9		10.5	*0.3*
	65-74	4.8	*-0.14*	0.98	*-0.04*	5.2		19.7	*-1.0*
	75-84	6.9		1.57	*-0.06*	5.0	*0.15*	30.3	*-1.5*
	85-90	7.8		1.95		4.6	*0.13*	38.8	*-1.0*

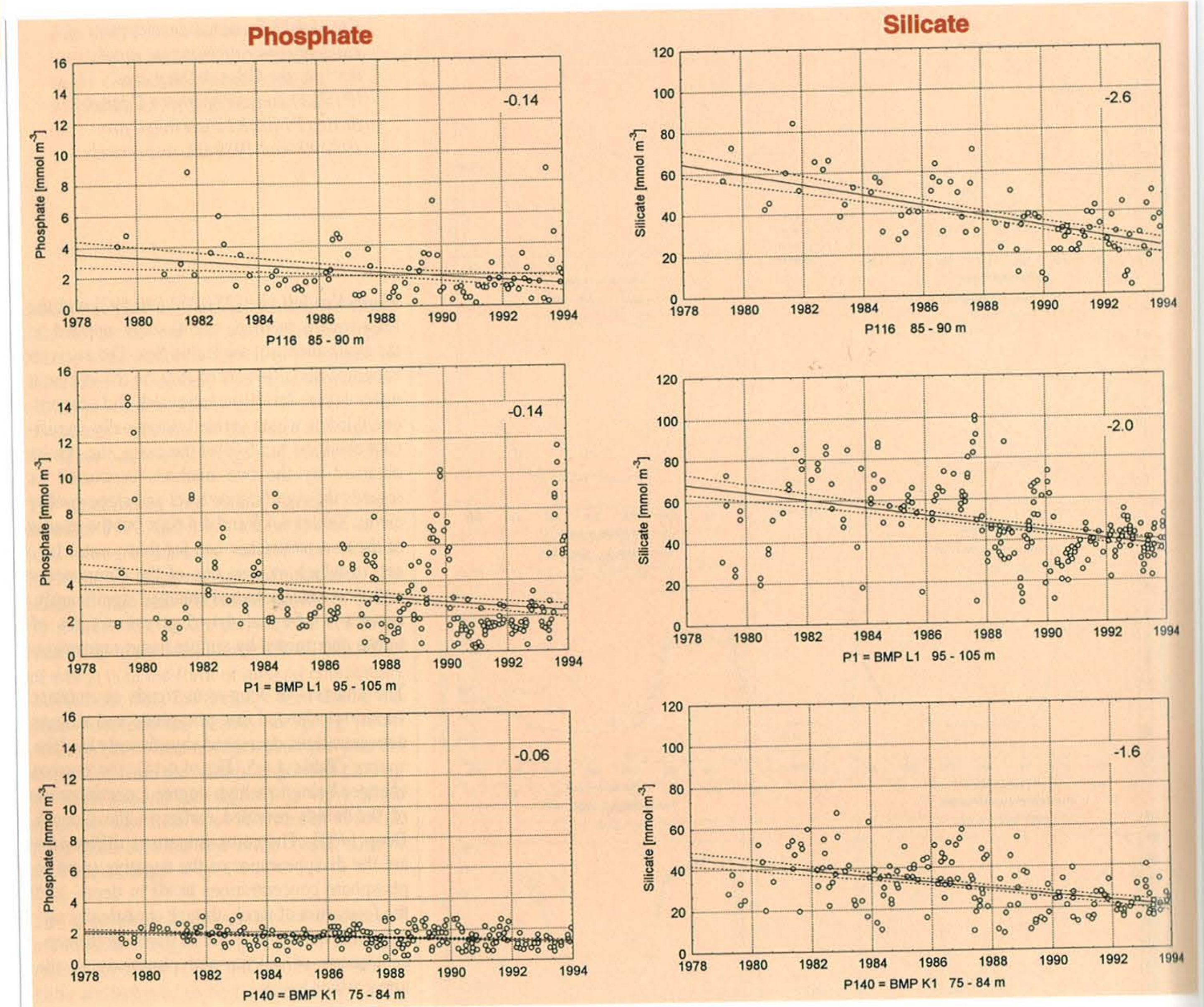


Fig. 4.4.14 Long-term trends of phosphate and silicate in the deep waters of the Gdansk Bight (P116; P1=L1) and the northern Gdansk Basin (P140=K1), 1979-93

(concentrations in μmol dm^{-3}: Trend slopes - in μmol dm^{-3} yr^{-1} - indicated)

As far as the long-term variations in nitrate (+ nitrite) are concerned, the non-parametric test revealed hardly any significant year-round trends (Table 4.4.5). However, there is an indirect evidence for the actual rise in their concentrations in deep waters, i.e., the positive trends in the N/P ratio. The molar proportion of the sum of nitrate and nitrite versus phosphate rose rapidly in the Vistula Estuary, but the rate of increase gradually diminished northward. This is confirmed by linear regression (Fig. 4.4.15), which seems to be a more powerful test in situations, when the parameter under scrutiny is subject to fluctuations over a broad range of values with interannual oscillations [*660*]. The slope coefficients indicate that, just as in the periods of favourable oxygen conditions during the years 1969-88 [*485*], nitrate accumulated in the near-bottom waters of the Gdansk Bight and the Gdansk Deep. The variations in nitrate concentrations in the deep water layers of the northernmost area of the Gdansk Basin (*K1*) were insignificant, but a concurrent fall in phosphate concentrations contributed to the significant increase in the N/P ratio.

Both the non-parametric test and the linear regression yielded entirely concurring results with respect to the development of nutrient accumulation in surface waters in the winter seasons (Table 4.4.6). The negative trend in phosphate concentrations in the surface layer of the Gdansk Deep in 1978-88 [*485*] was still evident throughout the Gdansk Bight, except for the Vistula Estuary, where the variations were statistically insignificant. In the northern Gdansk Basin, however, phosphate accumulated at a rate only slightly less than that found in the Gotland Deep for the period 1958-93 [*482*]. The strong negative trend in winter silicate concentrations in the Gdansk Basin disappeared almost completely. Similarly, the winter accumulation of nitrate in the upper water layers of the Gdansk Deep and the northern Gdansk Basin was retarded. By contrast, in the vicinity of the Vistula Estuary, nitrate accumulated at an unprecedented rate, which caused the N/P ratio to increase considerably. This was in good agreement with the long-term changes observed in the riverine flow rate, and in the amount of nitrogen and phosphorus carried by the Vistula in 1979-93.

Basin *station*	Depth *m*	NO_3+NO_2 mean	 *slope*	PO_4 mean	 *slope*	N/P ratio mean	 *slope*	Silicate mean	 *slope*
Gdansk Bight									
ZN2	0-2	40.5	*2.61*	1.59		38.8	*1.40*	56.4	
	10-12	7.9	*0.31*	0.79		26.6		16.7	
P116	0-4	7.2		0.55	*-0.02*	14.6		16.5	
	35-44	5.4	*0.23*	0.61		10.3	*0.37*	15.3	
	45-64	5.7		0.69		10.5	*0.40*	16.2	*-0.65*
Gdansk Deep									
L1 (P1)	35-44	5.0		0.58	*-0.02*	10.9	*0.46*	16.3	
Northern Gdansk Basin									
K1 (P140)	5-14	4.0		0.65	*0.012*	7.2		15.5	

Table 4.4.6 Statistically significant winter nutrient trends (non-parametric test) in the Gdansk Basin, 1979–93

(p <0.1; means in μmol dm^{-3}, slopes in μmol dm^{-3} yr^{-1}; N/P = $(NO_3+NO_2) / PO_4$)

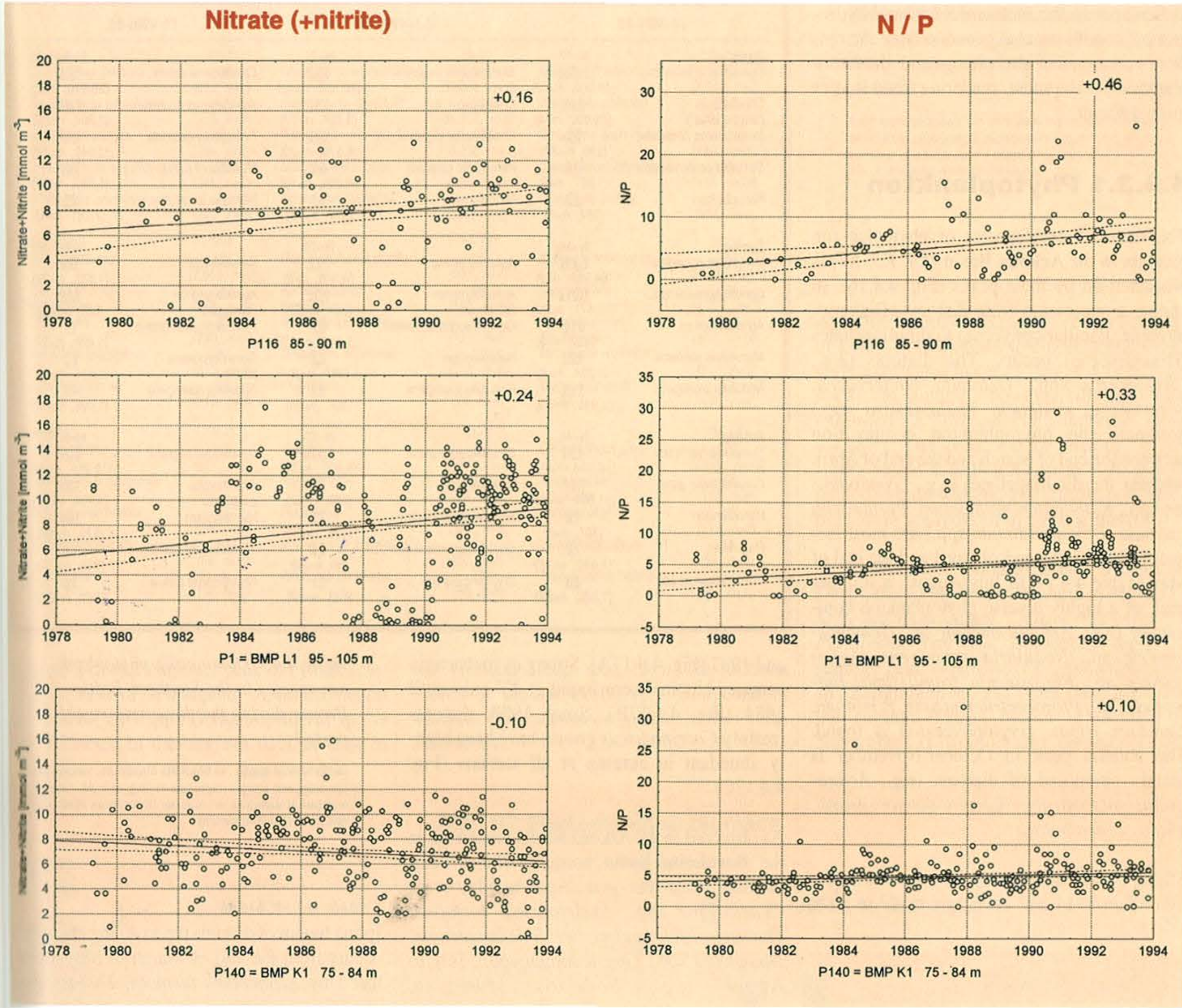


Fig. 4.4.15 Long-term trends of nitrate + nitrite and their ratio to phosphate (N/P) in the deep waters of the Gdansk Bight (P116; P1=L1) and the northern Gdansk Basin (P140=K1), 1979-93

(nitrate + nitrite concentrations in μmol dm^{-3}: Trend slopes - in μmol dm^{-3} yr^{-1} - indicated)

4.4.3 Pelagic biology

N. Wasmund[1], G. Breuel, L. Edler, H. Kuosa, R. Olsonen, H. Schultz, M. Pys-Wolska, L. Wrzolek

This chapter covers the composition of phytoplankton and its microscopically identified biomass, as well as chlorophyll *a* concentrations, in the surface layer (0-10 m) at BMP stations in the Baltic Proper (cf. Fig. 4.4.1). Additionally, the mesozooplankton is studied in the water column. *Whirsch* analyses of phytoplankton-biomass trends are based on chlorophyll data, rather than on the microscopically determined biomass as the latter comprises greater errors.

There is a general risk of missing short-term peaks due to inadequate sampling frequency, for instance during phytoplankton blooms.

[1] See ANNEX 11.3 for addresses of authors

Differences in the meteorological and hydrographic conditions also generate large interannual variations in the phyto- and mesozooplankton development, rendering trend studies more difficult.

4.4.3.1 Phytoplankton

The seasonal development of phytoplankton biomass in the **Arkona Basin** (*K4, K5, K7*) is characterised by three peaks (Fig. 4.4.16). In spring, a succession from diatoms (Diatomophyceae, Bacillariophyceae) to dinoflagellates (Dinophyceae) occurs. The diatoms (e.g., *Chaetoceros* spp., *Detonula confervacea*, *Skeletonema costatum*, *Thalassiosira* spp.) dominate the phytoplankton composition between the end of March and the end of April, whereas the dinoflagellates (e.g., *Gymnodinium* spp., *Katodinium rotundatum*, *Peridiniella catenata*, *Protoperidinium* spp.) are most frequent between the end of April and the end of May (Table 4.4.7). In July/August, a summer peak of a highly diverse phytoplankton community (e.g., *Aphanizomenon* sp., *Gomphosphaeria* sp., *Nodularia spumigena*, *Aphanothece* sp., *Rhizosolenia fragilissima*, *Eutreptiella* sp., *Prorocentrum micans*, *P. minium*, *Ceratium tripos*, cryptophyceans) is found. The autumn peak in October/November is mainly composed of diatoms (e.g., *Actinocyclus octonarius*, *Coscinodiscus granii*, *Thalassiosira baltica*).

Strong diatom spring blooms occurred at station *K5* in 1984 and 1993, and at *K7* in 1979 and 1987 (Fig. 4.4.17A). Strong cyanobacteria summer blooms were found at *K7* only until 1984 (Fig. 4.4.17B). Since 1988, diatoms (mainly *Coscinodiscus granii*) have been highly abundant in autumn at all stations (Fig. 4.4.17C).

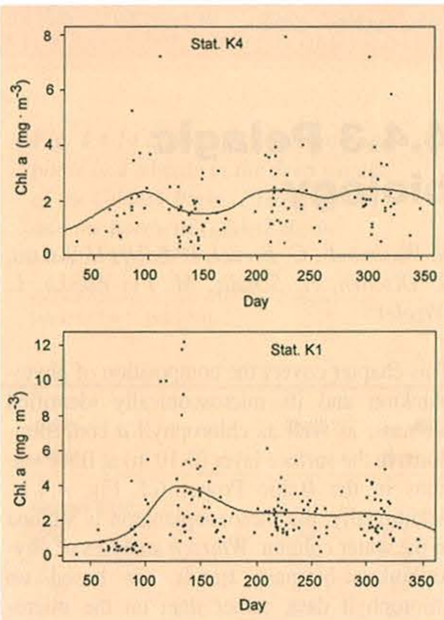


Fig. 4.4.16 Seasonal variations of the chlorophyll a concentrations (in mg m^{-3}) in the 0-10 m layer of the Arkona Basin (K4) and Gotland Basin (K1)

1979-83		1984-88		1989-93	
Spring	N=47		N=71		N=81
Chaetoceros holsaticus	1,026 (48,000, *n*=12)	*Skeletonema costatum*	532 (10,590, *n*=42)	*Chaetoceros debilis*	478 (28,036, *n*=2)
Chaetoceros ceratosporus	341 (16,000, *n*=4)	*Chaetoceros* sp.	117 (3,624, *n*=17)	*Skeletonema costatum*	184 (2,360, *n*=32)
Skeletonema costatum	53 (530, *n*=28)	*Chaetoceros wighamii*	106 (6,578, *n*=20)	*Peridiniella catenata*	146 (3,942, *n*=25)
Detonula confervacea	46 (1,997, *n*=5)	*Peridiniella catenata*	92 (4,490, *n*=13)	*Mesodinium rubrum*	141 (6,287, *n*=19)
Nanoplankton	43 (262, *n*=5)	Nanoplankton	54 (573, *n*=18)	*Nitzschia longissima*	91 (7,281, *n*=4)
Summer	N=56		N=60		N=98
Dinophysis norvegica	1,992 (64,599, *n*=9)	*Aphanothece* sp.	177 (9,346, *n*=6)	Nanoplankton	166 (1,535, *n*=22)
Gomphosphaeria spp.	1,010 (11,471, *n*=29)	*Aphanizomenon flos-aquae*	108 (604, *n*=53)	*Aphanizomenon flos-aquae*	124 (858, *n*=85)
Aphanothece sp.	518 (15,082, *n*=8)	*Gomphosphaeria pusilla*	67 (1,817, *n*=22)	*Prorocentrum minimum*	77 (1,498, *n*=37)
Microcystis reinboldii	420 (9,234, *n*=7)	Nanoplankton	65 (387, *n*=9)	*Gomphosphaeria pusilla*	72 (1,117, *n*=36)
Nodularia spumigena	198 (2,940, *n*=39)	*Nodularia spumigena*	54 (586, *n*=29)	*Nodularia spumigena*	48 (1,058, *n*=62)
Autumn	N=45		N=59		N=82
Gomphosphaeria spp.	498 (16,770, *n*=20)	*Coscinodiscus granii*	1,125 (8,400, *n*=34)	*Coscinodiscus granii*	830 (10,251, *n*=36)
Coscinodiscus granii	407 (11,850, *n*=13)	Nanoplankton	62 (180, *n*=22)	*Actinocyclus octonarius*	119 (1,704, *n*=32)
Nanoplankton	72 (387, *n*=8)	*Rhodomonas minuta*	20 (194, *n*=42)	Nanoplankton	108 (970, *n*=20)
Flagellates	72 (1,440, *n*=16)	*Gomphosphaeria pusilla*	12 (85, *n*=30)	*Thalassiosira baltica*	24 (344, *n*=34)
Rhodomonas minuta	53 (1,862, *n*=28)	*Cryptomonas* sp.	11 (123, *n*=23)	*Rhodomonas minuta*	16 (347, *n*=26)

***Table 4.4.7** Dominating phytoplankton species in the Southern Baltic Proper during the three assessment periods*

(arithmetical **mean**, in brackets maximum, values of species wet-weight biomass in mg m^{-3}; N - total number of samples, *n* - number of samples in which species were dominating)

As observed in the Arkona Basin, station *K2* in the **Bornholm Basin** normally shows three biomass peaks per year; April to May (e.g., *Chaetoceros* spp., *Skeletonema costatum*, *Thalassiosira levanderi*, *Peridiniella catenata*, *Dinophysis* spp., *Gymnodinium* spp.), July to August (e.g., *Nodularia spumigena*, *Aphanizomenon* sp., *Gomphosphaeria* sp., *Microcystis* spp., *Gymnodinium* sp.) and October to November (e.g., *Coscinodiscus granii*, *Actinocyclus octonarius*, *Thalassiosira baltica*, *Gymnodinium* sp.).

From 1979 to 1993, the start of the spring blooms seems to shift from May towards April, or even to the end of March. This might be due to mild winters in recent years (cf. [*604*]), and thus earlier stratification of the water column. It should, however, be stressed that the start or the peak of the blooms is hard to determine on the basis of the scattered data.

The proportion of diatoms (*Achnanthes taeniata* and, to a lesser degree, that of *Skeletonema costatum* and *Chaetoceros* spp.) has decreased, whereas dinoflagellates (e.g., *Peridiniella catenata*, *Dinophysis acuminata*, *Katodinium rotundatum*) have become more important in the Bornholm Basin in spring. In autumn, the diatoms are still the dominating group.

In the **Gdansk Basin** (*L1*, Gdansk Deep), the spring bloom of diatoms (up to 40 mg chl. *a* m^{-3}) occurs from the end of March to May, dominated by *Achnanthes taeniata*, *Thalassiosira* spp., *Chaetoceros* spp. and *Skeletonema costatum*, and followed by dinoflagellates in May to June (e.g., *Peridiniella catenata*, *Gymnodinium* spp.). In summer, i.e., mid of June to end of September, a diverse phytoplankton occurs, e.g., *Nodularia spumigena*, *Aphanizomenon* sp., *Prorocentrum* spp., *Heterocapsa triquetra*, cryptophyceans and other small flagellates, but without forming distinctive peaks (only about 3.5 mg chl. *a* m^{-3}). The relatively small autumn bloom (up to 5.5 mg chl. *a* m^{-3}) in October/November is mainly caused by the diatom *Coscinodiscus granii* [*99,476*].

In the southern part of the **Eastern Gotland Basin** (*K1*, Fig. 4.4.16), the spring bloom was found in the period April to May, but in the central part of the basin (*J1*, Gotland Deep) this only occurred in May. Diatoms, e.g. *Chaetoceros* spp., *Achnanthes taeniata* and *Skeletonema costatum*, and dinoflagellates, e.g., *Peridiniella catenata*, *Dinophysis* spp. and *Gymnodinium* spp., are the main groups (Table 4.4.8).

In contrast to the Bornholm and Arkona Basins, a distinct summer peak of cyanobacteria (*Nodularia spumigena*, *Aphanizomenon* sp., *Microcystis* spp., *Gomphosphaeria* sp.) is less reflected in the biomass data, at least in the Gotland Deep area (*J1*). The autumn bloom is almost exclusively composed of diatoms (*Coscinodiscus granii*, *Actinocyclus octonarius*).

1979-83		1984-88		1989-93	
Spring	N=7		N=10		N=17
Achnanthes taeniata	145 (635, *n*=5)	*Chaetoceros wighamii*	5,117 (51,005, *n*=3)	*Mesodinium rubrum*	271 (3,340, *n*=2)
Skeletonema costatum	63 (370, *n*=4)	*Skeletonema costatum*	3,524 (34,931, *n*=6)	*Peridiniella catenata*	212 (2,787, *n*=32)
Protoperidinium sp.	62 (189, *n*=6)	*Peridiniella catenata*	1,742 (15,235, *n*=7)	*Dinophysis acuminata*	21 (120, *n*=25)
Gymnodinium westificii	51 (346, *n*=2)	*Chaetoceros subtilis*	118 (930, *n*=2)	*Gymnodinium lohmannii*	16 (138, *n*=19)
Dinophysis acuminata	42 (240, *n*=4)	*Achnanthes taeniata*	42 (364, *n*=6)	*Dinophysis baltica*	15 (256, *n*=4)
Summer	N=9		N=11		N=16
Microcystis spp.	484 (2,565, *n*=2)	*Gomphosphaeria pusilla*	69 (637, *n*=4)	*Nodularia spumigena*	211 (2,149, *n*=11)
Aphanizomenon flos-aquae	143 (364, *n*=8)	*Aphanizomenon flos-aquae*	60 (158, *n*=9)	Nanoplankton	178 (1,227, *n*=3)
Flagellates	93 (233, *n*=6)	*Nodularia spumigena*	38 (157, *n*=6)	*Aphanizomenon flos-aquae*	111 (637, *n*=14)
Nodularia spumigena	92 (443, *n*=4)	*Dinophysis acuminata*	32 (138, *n*=6)	*Microcystis reinboldii*	79 (910, *n*=3)
Cryptomonas sp.	48 (110, *n*=4)	Flagellates	29 (78, *n*=8)	*Gomphosphaeria pusilla*	57 (496, *n*=6)
Autumn	N=6		N=9		N=12
Coscinodiscus granii	102 (574, *n*=2)	*Coscinodiscus granii*	366 (2,679, *n*=3)	*Coscinodiscus granii*	979 (3,840, *n*=12)
Actinocyclus octonarius	16 (36, *n*=4)	Nanoplankton	51 (79, *n*=8)	*Actinocyclus octonarius*	17 (163, *n*=6)
Rhodomonas minuta	15 (36, *n*=6)	Flagellates	21 (127, *n*=6)	Nanoplankton	16 (116, *n*=2)
Protoperidinium sp.	12 (40, *n*=2)	*Actinocyclus octonarius*	21 (96, *n*=3)	*Mesodinium rubrum*	15 (84, *n*=4)
Flagellates	9 (42, *n*=2)	*Rhodomonas minuta*	13 (50, *n*=8)	*Thalassiosira baltica*	14 (67, *n*=4)

Table 4.4.8 Dominating phytoplankton species in the Eastern Gotland Basin during the three assessment periods

(arithmetical **mean**, in brackets maximum, values of species wet-weight biomass in mg m^{-3}; N - total number of samples, n - number of samples in which species were dominating)

As in the Bornholm Basin, there is a tendency of a shift from diatoms to dinoflagellates (and cryptophyceans) in spring (Fig. 4.4.18A, Table 4.4.8). In autumn, however, diatoms have exhibited a pronounced increase (Fig. 4.4.18C). The reduction of diatoms in the spring bloom at station *K1* since 1990 may be related to mild winters. The silicate concentrations were not reduced in these winters.

In the **Western Gotland Basin** (*I1*, Karlsö Deep), the spring bloom is found in May (e.g., *Skeletonema costatum*, *Chaetoceros wighamii*, *Peridiniella catenata*, *Dinophysis* spp.), the summer peak in August (e.g., *Aphanizomenon* sp., *Eutreptiella* sp. and other flagellates) and the autumn diatom bloom in October to November (*Coscinodiscus granii*) (Table

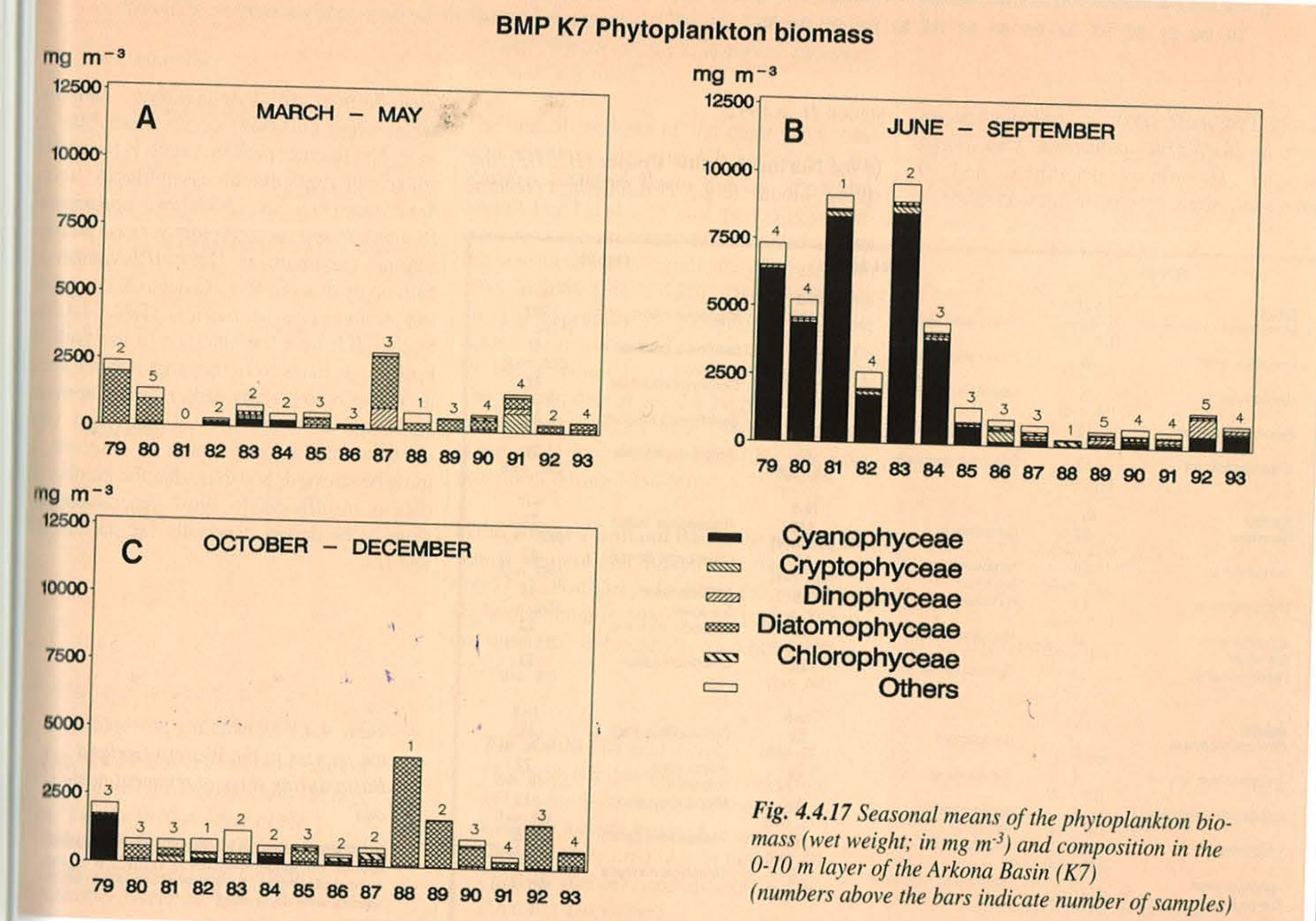


Fig. 4.4.17 Seasonal means of the phytoplankton biomass (wet weight; in mg m^{-3}) and composition in the 0-10 m layer of the Arkona Basin (K7) (numbers above the bars indicate number of samples)

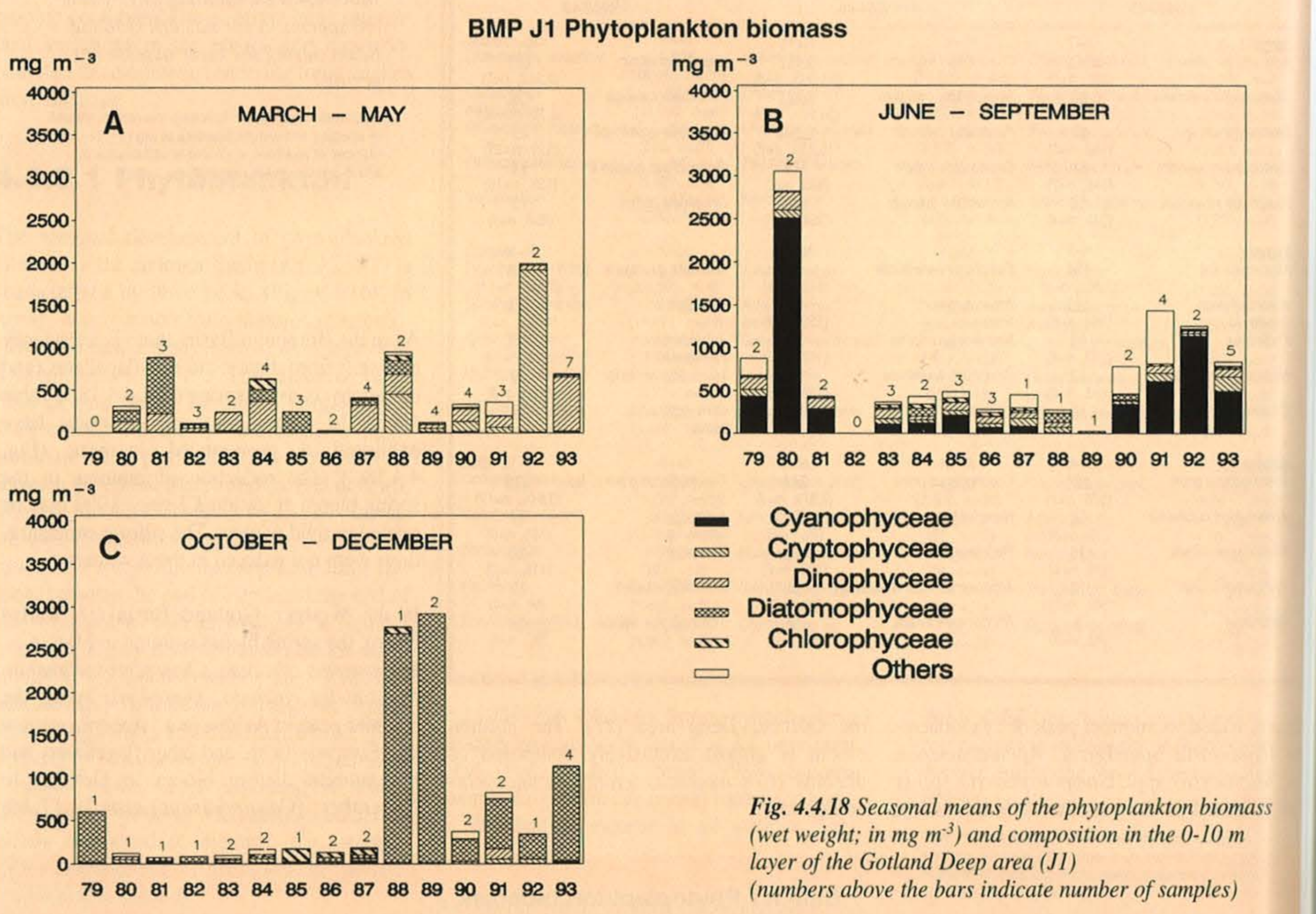


Fig. 4.4.18 Seasonal means of the phytoplankton biomass (wet weight; in mg m^{-3}) and composition in the 0-10 m layer of the Gotland Deep area (J1) (numbers above the bars indicate number of samples)

4.4.9). Potentially toxic or harmful species, such as *Nodularia spumigena*, *Chaetoceros danicus*, *Dinophysis acuminata* and *D. norvegica*, were present in high numbers at station *I1* in 1992.

In the **Northern Baltic Proper** (*H2*, *H3*), the spring bloom (e.g., *Peridiniella catenata*, *Glenodinium* sp., *Achnanthes taeniata*, *Skeletonema costatum*) occurs from May to June. The summer peak in August is formed by mixed phytoplankton assemblages (e.g. *Aphanizomenon* sp., *Nodularia spumigena*, *Dinophysis* spp. and cryptophyceans) and the autumn maximum in October/November is built up by diatoms (e.g., *Coscinodiscus granii* and *Actinocyclus octonarius*) (Table 4.4.10). Station *H1*, near the entrance to the Gulf of Finland, deviates from the other stations since its spring bloom starts earlier, i.e., diatoms in April and dinoflagellates in May, and very often no real autumn bloom is observed. It must be stressed, however, that the number of data is insufficient to allow general conclusions to be drawn, especially for stations *H1* and *H3*.

1979-83		1984-88		1989-93	
Spring	N=8		N=3		N=11
Skeletonema costatum	83 (470, n=6)	*Skeletonema costatum*	7,267 (21,620, n=2)	*Mesodinium rubrum*	249 (1,292, n=3)
Dinophysis acuminata	36 (70, n=7)	*Chaetoceros wighamii*	670 (1,378, n=2)	*Peridiniella catenata*	43 (255, n=5)
Gymnodinium sp.	16 (38, n=6)	*Eutreptiella* sp.	344 (1,378, n=2)	*Dinophysis acuminata*	39 (151, n=7)
Peridiniella catenata	15 (73, n=4)	*Peridiniella catenata*	39 (118, n=1)	*Skeletonema costatum*	34 (278, n=3)
Dinobryon balticum	13 (72, n=3)	*Dinophysis acuminata*	13 (23, n=2)	*Dinophysis norvegica*	26 (252, n=3)
Summer	N=5		N=9		N=7
Flagellates	82 (113, n=3)	*Eutreptiella* sp.	840 (728, n=6)	*Thalassiosira baltica*	71 (495, n=3)
Glenodinium sp.	60 (125, n=3)	*Aphanizomenon flos-aquae*	63 (196, n=6)	*Chaetoceros danicus*	57 (383, n=3)
Cryptomonas sp.	60 (106, n=3)	Nanoplankton	43 (112, n=3)	*Aphanizomenon flos-aquae*	48 (230, n=7)
Aphanizomenon flos-aquae	21 (78, n=3)	*Mesodinium rubrum*	28 (251, n=2)	*Chrysochromulina* sp.	38 (92, n=4)
Pyramimonas sp.	11 (23, n=4)	*Gymnodinium* sp.	26 (190, n=7)	*Mesodinium rubrum*	28 (78, n=4)
Autumn	N=7		N=8		N=8
Rhodomonas minuta	10 (30, n=6)	Nanoplankton	59 (115, n=5)	*Coscinodiscus granii*	355 (1,179, n=6)
Protoperidinium sp.	8 (32, n=2)	*Eutreptiella* sp.	26 (179, n=3)	*Actinocyclus octonarius*	22 (149, n=5)
Actinocyclus octonarius	8 (30, n=4)	*Aphanizomenon flos-aquae*	22 (115, n=6)	*Mesodinium rubrum*	17 (80, n=4)
Flagellates	6 (34, n=3)	*Mesodinium rubrum*	10 (45, n=2)	*Thalassiosira baltica*	10 (43, n=2)
Aphanizomenon flos-aquae	5 (15, n=4)	*Actinocyclus octonarius*	8 (30, n=2)	*Dinophysis norvegica*	7 (23, n=5)

Table 4.4.9 Dominating phytoplankton species in the Western Gotland Basin during three assessment periods

(arithmetical **mean**, in brackets maximum, values of species wet-weight biomass in mg m^{-3}; N - total number of samples, n - number of samples in which species were dominating)

Table 4.4.10 Dominating phytoplankton species in the Northern Baltic Proper during the three assessment periods

(arithmetical **mean**, in brackets maximum, values of species wet-weight biomass in mg m^{-3}; N - total number of samples, n - number of samples in which species were dominating)

1979-83		1984-88		1989-93	
Spring	N=25		N=22		N=21
Peridiniella catenata	287 (2,415, n=12)	*Glenodinium* sp.	420 (9,028, n=2)	*Mesodinium rubrum*	292 (3,054, n=6)
Chaetoceros wighamii	41 (1,010, n=4)	*Peridiniella catenata*	417 (2,634, n=15)	*Peridiniella catenata*	262 (1,111, n=16)
Achnanthes taeniata	36 (398, n=10)	*Dinophysis acuminata*	188 (3,600, n=6)	*Glenodinium* sp.	48 (661, n=4)
Protoperidinium sp.	32 (302, n=12)	*Achnanthes taeniata*	172 (1,274, n=14)	*Skeletonema costatum*	31 (424, n=6)
Thalassiosira levanderi	24 (585, n=2)	*Skeletonema costatum*	154 (851, n=20)	*Thalassiosira baltica*	28 (365, n=6)
Summer	N=16		N=24		N=17
Flagellates	126 (311, n=11)	*Aphanizomenon flos-aquae*	840 (688, n=19)	Cryptomonadales	84 (295, n=10)
Aphanizomenon flos-aquae	50 (160, n=13)	Cryptomonadales	81 (439, n=15)	*Mesodinium rubrum*	77 (406, n=9)
Peridiniella catenata	48 (531, n=3)	Nanoplankton	52 (251, n=6)	*Aphanizomenon flos-aquae*	77 (352, n=16)
Dinophysis acuminata	17 (110, n=13)	*Dinophysis norvegica*	44 (940, n=13)	*Nodularia spumigena*	33 (238, n=9)
Nodularia spumigena	16 (84, n=7)	*Dinophysis acuminata*	42 (360, n=15)	*Dinophysis norvegica*	23 (125, n=10)
Autumn	N=15		N=24		N=24
Coscinodiscus granii	29 (400, n=2)	*Coscinodiscus granii*	320 (3,060, n=5)	*Coscinodiscus granii*	715 (3,000, n=17)
Aphanizomenon flos-aquae	12 (67, n=9)	*Dinophysis norvegica*	40 (940, n=3)	*Actinocyclus octonarius*	82 (1,897, n=10)
Rhodomonas minuta	11 (34, n=12)	*Skeletonema costatum*	33 (766, n=6)	*Dinophysis norvegica*	32 (508, n=15)
Actinocyclus octonarius	10 (48, n=10)	*Aphanizomenon flos-aquae*	22 (115, n=18)	*Thalassiosira baltica*	14 (68, n=10)
Cryptomonas baltica	9 (103, n=2)	*Actinocyclus octonarius*	20 (371, n=6)	*Dinophysis baltica*	12 (267, n=2)

4.4.3.2 Chlorophyll *a*

The three annual peaks, observed in the phytoplankton biomass in the **Arkona Basin**, are also reflected by the chlorophyll *a* concentrations. Figure 4.4.16 shows the seasonal variations at station *K4* (Arkona Deep) for the entire period 1979-93. A significant increase in the chlorophyll concentrations was identified at station *K7*, but not at stations *K4* and *K5*, in this 15-year period. The *Wilcoxon two-sample test* shows that at the stations *K5* and K7, the autumn concentrations from 1988-93 were significantly higher than those from 1979-87.

For the **Bornholm Basin** (*K2*, Bornholm Deep), the chlorophyll concentrations in summer and autumn are considered separately in Figure 4.4.19. From 1979 to 1989, the decrease in summer concentrations of chlorophyll was highly significant, whereas the autumn concentrations showed the opposite tendency.

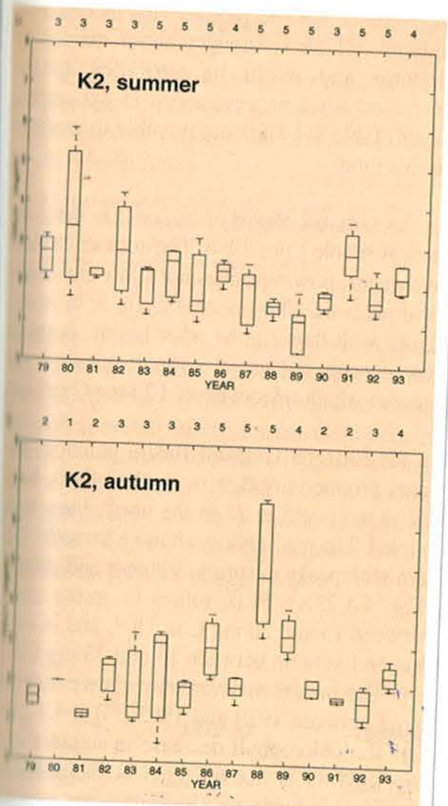


Fig. 4.4.19 Box-and-whisker plots of chlorophyll a concentrations (in mg m^{-3}) in the 0-10 m layer of the Bornholm Deep area (K2) in summer (June-September) and autumn (October-December)

Slightly increasing annual mean concentrations of chlorophyll were observed in the surface layer of the **Gdansk Basin** (*L1*, Gdansk Deep) in the period 1986-93. The insufficient number of data excludes, however, an examination of any trend.

The annual medians of the chlorophyll concentrations at stations *K1* and *J1* in the **Eastern Gotland Basin** did not show any overall trend if the 15-year period is considered. However, summer data revealed a clear but statistically non-significant decrease from 1984 to 1989 (Fig. 4.4.20). The two stations differed especially in autumn. The highest autumn values occurred at station *K1* in 1986-89, but at station *J1* in 1989-93 (Fig. 4.4.20). The autumn data showed a highly significant increase at station *K1* from 1980 to 1988, and at station *J1* from 1979 to 1993 (this was most prominent during 1984-90).

In the **Western Gotland Basin**, the peak of the annual mean of chlorophyll was found at station *I1* (Karlsö Deep) in 1985, followed by a slight minimum in 1989. From 1984 to 1990, the autumnal chlorophyll concentrations increased.

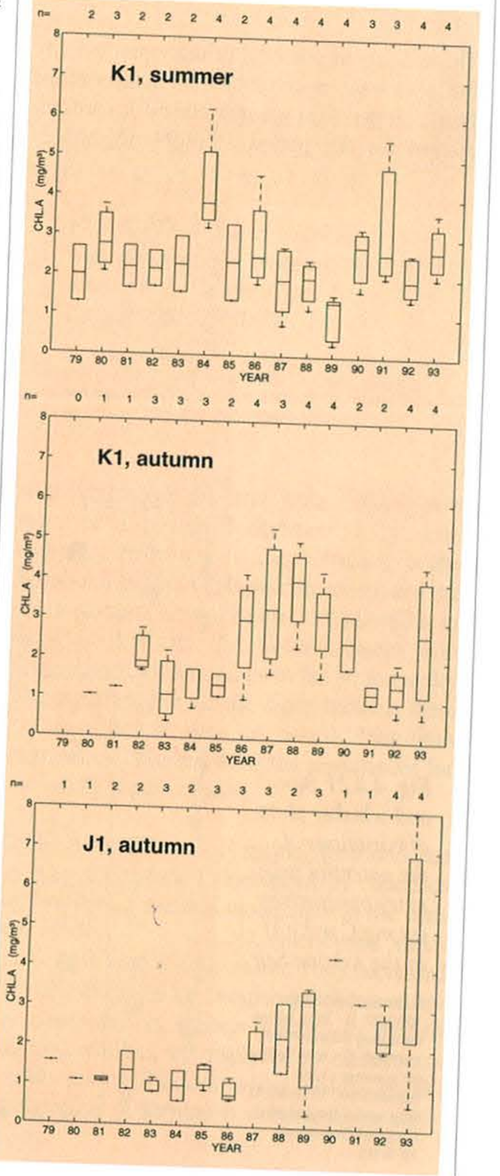


Fig. 4.4.20 Box-and-whisker plots of chlorophyll a concentrations (in mg m^{-3}) in the 0-10 m layer of the south-eastern Gotland Basin (K1) and Gotland Deep area (J1) in summer (June-September) and autumn (October-December)

	1979-83	1984-88	1989-93
Apr-May	2.19 ± 1.43 (0.22-7.61), *91*	3.42 ± 3.14 (0.37-17.85), *129*	2.97 ± 3.80 (0.26-27.31), *235*
Jun-Sep	2.40 ± 1.18 (0.35-7.99), *182*	2.20 ± 1.15 (0.10-7.00), *184*	2.42 ± 1.21 (0.08-7.01), *295*
Oct-Dec	1.85 ± 1.11 (0.30-5.00), *103*	2.28 ± 1.61 (0.10-7.49), *134*	2.51 ± 1.29 (0.00-7.99), *173*

Table 4.4.11 Chlorophyll a in the Baltic Proper during different seasons for the three assessment periods

(mean, standard deviation, range and *number of samples*; values in mg m^{-3})

The annual medians of chlorophyll concentrations in the **Northern Baltic Proper** revealed a significant increase at stations *H1* and *H2*, and an insignificant increase at station *H3*, from 1979 to 1993. At station *H1*, chlorophyll concentrations increased particularly in summer, whereas at station *H2* the increase took place in autumn. Unlike the primary production (cf. Table 4.4.12), the pooled chlorophyll data of the Baltic Proper revealed only a slight increase during the investigation period (Table 4.4.11).

4.4.3.3 Potential primary productivity

The analysis of potential primary productivity is based on the mean of the 0-20 m layer of the Baltic Proper. Average seasonal cycles are calculated for the period 1989-93. Results of potential primary productivity are treated for six different areas of the Baltic Proper. Interannual variation is shown for the entire BMP period 1979-93.

In the **Arkona Basin**, stations *K4*, *K5* and *K7* are located close together in the centre of the basin, whereas station *K6* is close to the entrance of the Sound. Despite this, the annual cycle of productivity during 1989-93 was very similar in the two areas, with obvious spring and autumn peaks (Fig. 4.4.21A). A summer peak of productivity can also be seen, but is less pronounced than the chlorophyll peak at this time. Rates in spring and autumn peaks may reach 35 and 30 mg C m^{-3} $h.^{-1}$, respectively. The summer peak is generally of the order of 10-20 mg C m^{-3} $h.^{-1}$, but peak values of 34 mg C m^{-3} $h.^{-1}$ have been measured at station *K6*.

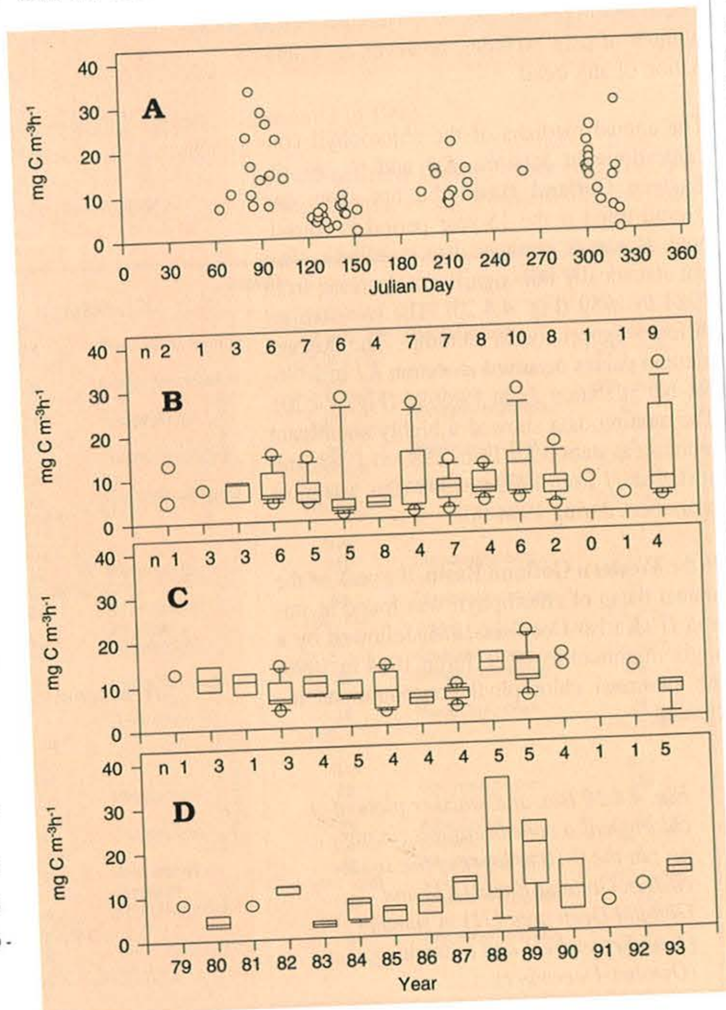


Fig. 4.4.21 Box-and-whisker plots of variations of the potential primary productivity (in mg C m^{-3} h^{-1}) in the Arkona Sea

A - seasonal variation, 1989-93; B - time series for spring (March-May), 1979-93; C - time series for summer (June-September), 1979-93; D - time series for autumn (October-December), 1979-93

The interannual variation in potential primary productivity in different seasons between 1979 and 1993 does not show a significant trend (Figs. 4.4.21B-D). An increasing tendency, however, can be seen in spring and autumn. This tendency is very weak. But if it continues it suggests a doubling of the potential primary productivity in about 20 years. A similar increase is also found in the autumnal chlorophyll data and is probably connected to the increased biomass of diatoms.

The annual cycle of potential primary productivity during 1989-93 in the **Bornholm Basin** at station *K2* resembles that of the Arkona Sea (Fig. 4.4.22A). The summer peak reaches about the same magnitude as the spring and autumn peaks with 20-25 mg C m^{-3} $h.^{-1}$. Compared to the Arkona Sea, maximum spring and autumn productivity is about 30 % lower, whereas the average summer productivity in the Bornholm Basin is about the same as in the Arkona Sea.

As for chlorophyll, there seems to be a weak increase in productivity in the autumn between 1979 and 1989. After 1989, there seems to be a drop in the autumn productivity. Because of very few data, however, this decrease may be false. An increase over the entire period can be seen in spring, whereas the summer productivity shows a slight decrease (Figs. 4.4.22B-D). Whether or not the productivity increases in autumn, which is brought about mainly by diatoms and might be attributed to the increased silica concentrations in the surface layer (Table 4.4.4), is not possible to evaluate at this time.

In the **Gdansk Basin** at station *L1*, there are data available since 1986. The number of measurements, however, does not allow any statistical analysis. The seasonal cycle is in accordance with those in the other basins, but with lower peak values in spring, summer and autumn which seldom exceed 20 mg C m^{-3} $h.^{-1}$.

In the **Eastern Gotland Basin**, potential primary productivity data from 11 stations, from *K1* in the south to *J1* in the north, have been pooled. The productivity shows a trimodal pattern with peaks in spring, summer and autumn (Fig. 4.4.23A). Peak values in spring range between 15 and 20 mg C m^{-3} $h.^{-1}$, and in summer and autumn between 15 and 25 mg C m^{-3} $h.^{-1}$. The interannual variation shows no clear trend between 1979 and 1993 (Figs. 4.4.23B-D). The chlorophyll decrease in summer during 1984-89 is not reflected in the potential primary productivity. The significant increase in chlorophyll in autumn is shown only as a weak tendency in the productivity, equivalent to a doubling of the productivity over the next 15 to 20 years.

Data from the Western Gotland Basin and the

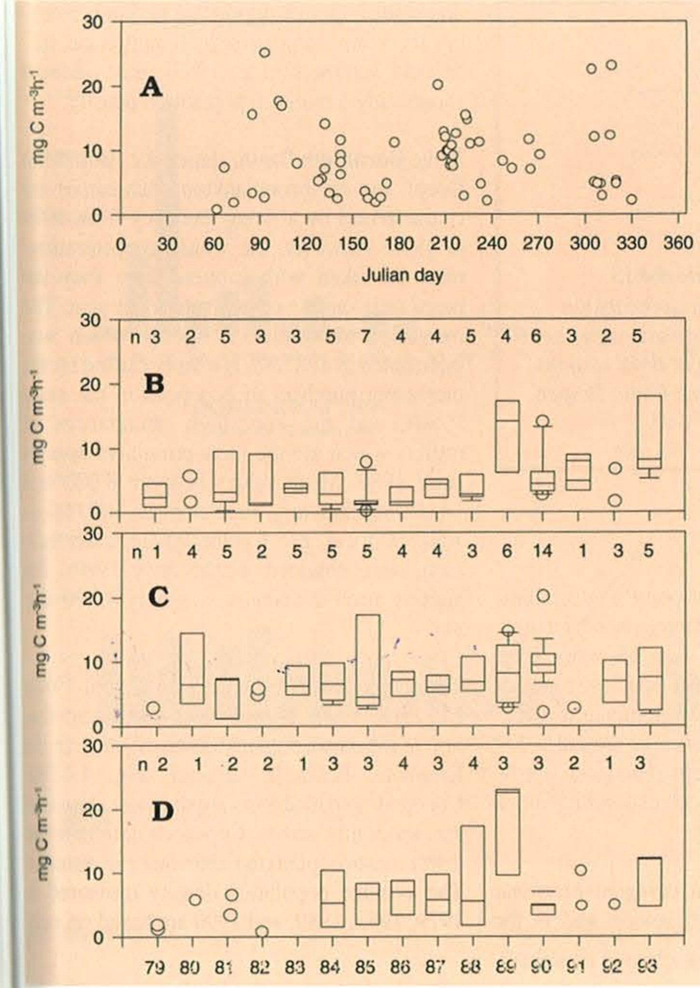


Fig. 4.4.22 Box-and-whisker plots of variations of the potential primary productivity (in mg C m^{-3} h^{-1}) in the Bornholm Basin

A - seasonal variation, 1989-93; B - time series for spring (March-May), 1979-93; C - time series for summer (June-September), 1979-93; D - time series for autumn (October-December), 1979-93

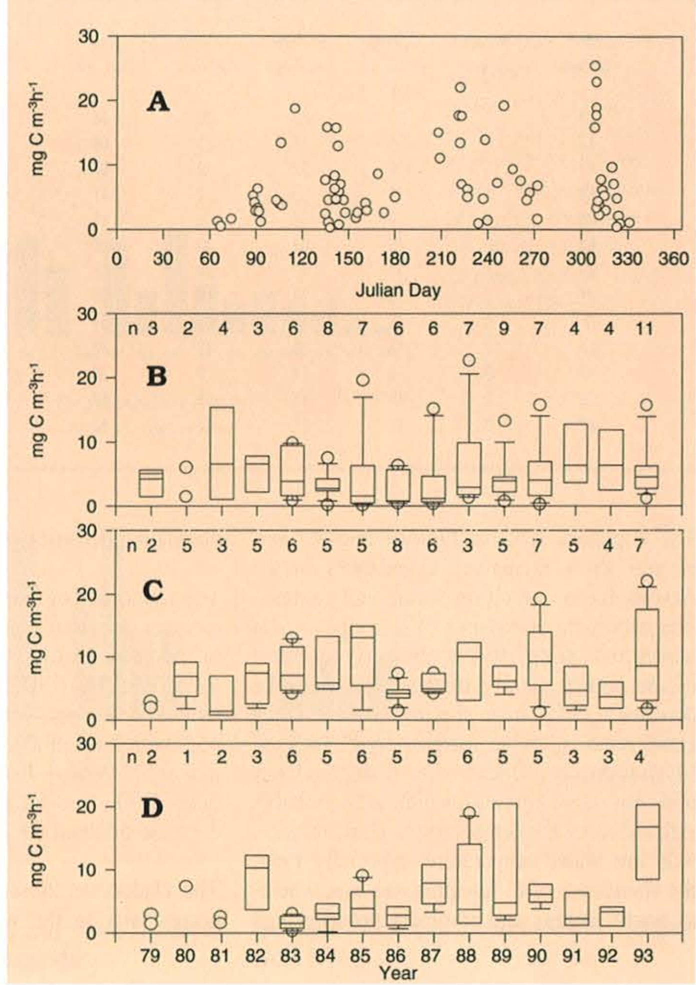


Fig. 4.4.23 Box-and-whisker plots of variations of the potential primary productivity (in mg C m^{-3} h^{-1}) in the Eastern Gotland Basin

A - seasonal variation, 1989-93; B - time series for spring (March-May), 1979-93; C - time series for summer (June-September), 1979-93; D - time series for autumn (October-December), 1979-93

Northern Baltic Proper are too few to show any trends. Pooling all measurements of potential primary productivity in the entire Baltic Proper into different assessment periods reveals an increase in productivity for spring, summer and autumn (Table 4.4.12). As the chlorophyll does not show the same increase (cf. Table 4.4.11), a strong increase of the assimilation number is suggested.

	1979-83	1984-88	1989-93
Apr-May	7.00 ± 4.50 (1.38-22.63), *123*	6.92 ± 6.94 (0.56-59.69), *172*	9.74 ± 7.76 (0.24-32.65), *142*
Jun-Sep	7.54 ± 4.54 (0.69-22.01), *186*	7.79 ± 4.42 (0.87-22.31), *253*	13.16 ± 18.21 (0.18-150.70), *197*
Oct-Dec	4.22± 3.41 (0.17-14.73), *121*	5.97 ± 6.20 (0.18-45.78), *181*	10.95 ± 7.61 (0.35-28.59), *122*

Table 4.4.12 Potential primary productivity in the Baltic Proper during different seasons for the three assessment periods

(mean, standard deviation, range and *number of samples*; values in mg C m^{-3} $h.^{-1}$)

4.4.3.4 Mesozooplankton

The sampling frequency of mesozooplankton varies with respect to stations and seasons in the period 1979-93 (Table 4.4.13). Whereas spring, summer and autumn are sufficiently covered at some BMP stations, observations in winter are generally too low at all stations in the Baltic Proper. The stations *K4*, *K5* and *K7* in the Arkona Basin have been pooled to increase the number of observations. In general, they all cover the same water depth (44-48 m) over a restricted distance of 20 n.m. Sampling frequency is also sufficient in the Bornholm Basin (*K2*, Bornholm Deep) and the south-eastern Gotland Basin (*K1*), as well as in the Gdansk Basin (*L1*, Gdansk Deep) with more than 70 observations in the period under investigation. For all the other stations, there are insufficient data to assess long-term changes. All figures show the abundances for the whole water column.

In the **Arkona Basin**, in spring, the abundance of mesozooplankton, dominated by copepods and rotifers, increased moderately in the period 1985-93 (Fig. 4.4.24). Interannual variations were high in this season. They decreased in summer, when the mesozooplankton, mainly copepods, on average reaches its highest density. Rotifers are displaced by cladocerans in this season. The decreasing mesozooplankton density in autumn is dominated by copepods.

BMP Station	Winter (Jan-Mar)	Spring (Apr-May)	Summer (Jun-Sep)	Autumn (Oct-Dec)	Year (Jan-Dec)
K5	7	23	22	16	68
K4	8	25	17	21	68
K7	14	19	32	22	87
K6	1	5	3	2	11
K8	3	7	5	6	21
K2	17	43	63	33	156
K1	12	31	40	29	112
J1	6	10	19	16	51
I1	0	15	22	14	51
L1	12	13	32	13	70
H1	8	9	14	8	39
H2	8	16	18	15	57
H3	3	11	21	17	52

Table 4.4.13 *Mesozooplankton observations per season at BMP stations of the Baltic Proper, 1979-93*

The copepods *Acartia bifilosa* and *A. longiremis* show increasing abundances in the Arkona Basin, mainly in spring and summer. The maximum abundance of *A. longiremis* in winter and spring 1984 is obviously an effect of the higher salinity in these seasons. The copepodites of *Acartia* species, assessed at the genus level, show an increase in all seasons. Much more pronounced is the rising tendency of *Centropages hamatus*, which were probably influenced by the temperature. Thus, in years with low water temperature, especially 1987, the abundance of *C. hamatus* was low, whereas warm springs and summers produce high population densities.

The abundance of the copepod *Pseudocalanus minutus elongatus* was characterised by a drastic decrease during the last 15 years (Fig. 4.4.25). In the 1970s, this species, together with *Acartia* spp., was the dominating copepod, with a mean abundance of around 5,000 ind. m^{-3} [*249,603*]. Its mean abundance is now only 1,000 ind. m^{-3}, corresponding to a decrease of about 80 %.

The cladoceran *Bosmina coregoni maritima* occurs only in the warm season and is the dominating species during hot summers reaching abundances of up to several million ind. m^{-2}. In cold summers (e.g., 1987), this cladocer shows only a minimal population density.

In the **Bornholm Basin**, station *K2* (Bornholm Deep), the mesozooplankton abundances are characterised by a rising tendency from 1979 to 1993. However, the winter concentrations must be taken with caution since they are based only on one or two values per year. The increasing abundance in spring, which was interrupted in 1987/88, is mainly caused by the increasing numbers of copepods in the early 1980s, and the very high abundances of rotifers which are the most prominent species since 1989. Mean values between 8,000 and 18,000 rotifers m^{-3}, equivalent to 700,000 to 1,600,000 ind. m^{-2} for the whole water column, were obtained. In the early 1980s, the highest mean abundance was only 4,000 ind. m^{-3}.

Undulations with maxima in the mid-1980s and early 1990s are the most significant feature in the mesozooplankton distribution in the Bornholm Basin in summer (Fig. 4.4.26). Copepods and cladocers are the most abundant species in this season. Copepods dominate the lower mesozooplankton abundance in autumn. The extreme population density measured in 1979, 1983, 1989, and 1990 are based on only

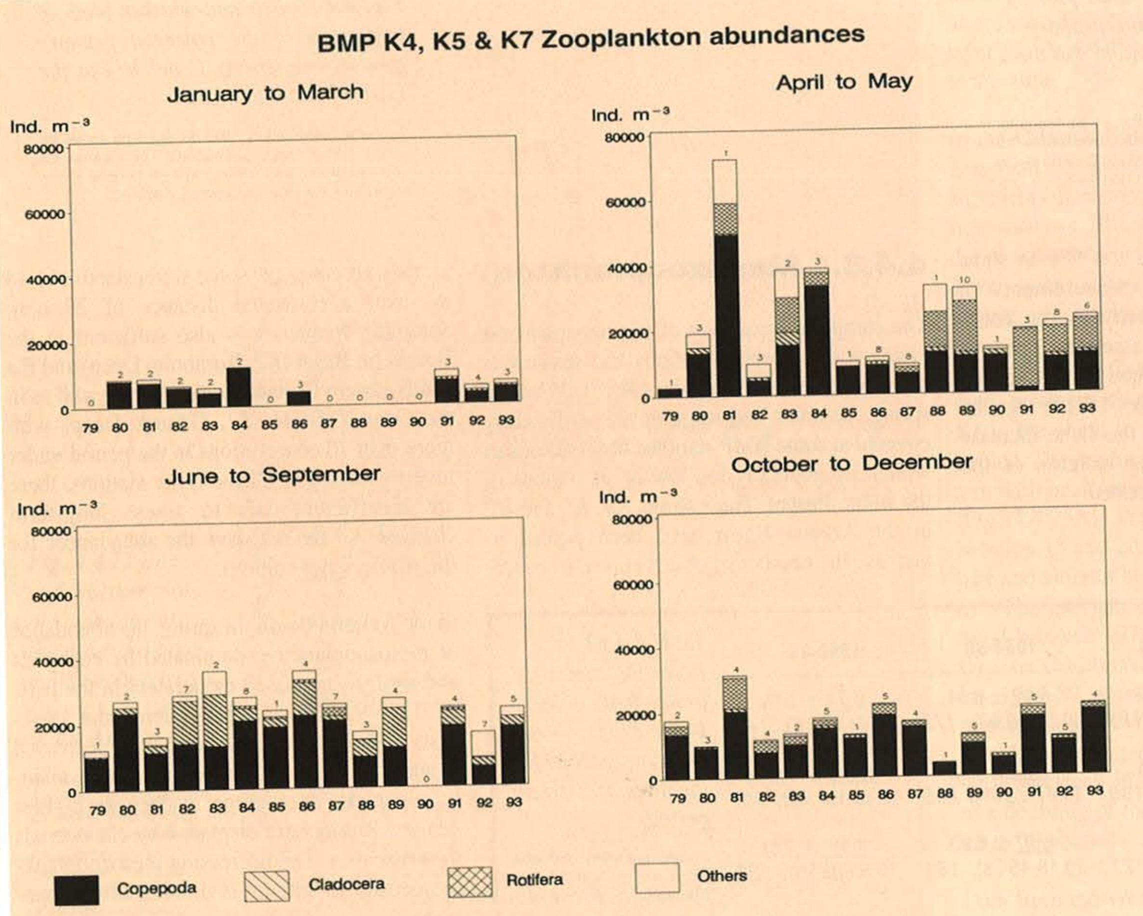


Fig. 4.4.24 *Total abundance of different groups of mesozooplankton (in ind. m^{-3}) in the Arkona Basin (K4, K5, K7, pooled)*

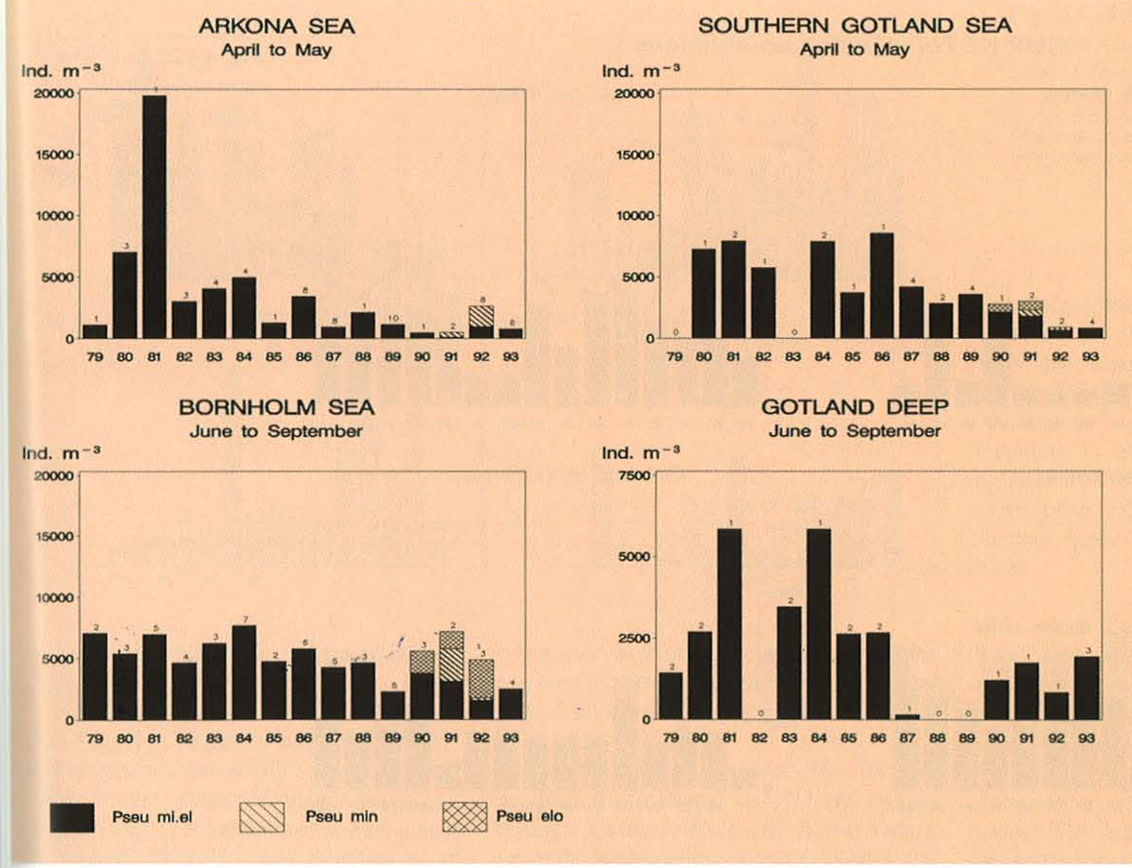


Fig. 4.4.25 *Abundance of Pseudocalanus minutus elongatus (in ind. m^{-3}) in different areas and seasons, Arkona Sea (K4, K5, K7, summarized), Bornholm Sea (K2), southern Gotland Sea (K1) and Gotland Deep area (J1)*

one observation.

At the species level, the copepods *Centropages hamatus* and *Temora longicornis* show a marked tendency to higher abundances from 1979 to 1993 in all seasons, especially in spring and summer, whereas the increase of *Acartia bifilosa* is only important in summer. Small changes are observed for *A. longiremis*. *Pseudocalanus minutus elongatus* shows a clear tendency of decreasing population density in all seasons. The cyclopoid copepod *Oithona similis* occurred in high abundances especially in the Bornholm Basin, following several deep water renewals (cf. Chapter 2) in the period 1989-93.

In the **Gdansk Basin** (*L1*, Gdansk Deep), the mesozooplankton community consists of about 20 taxa, including 9 copepods, 5 cladocera species and 2 rotifera genera. *Acartia* spp. (especially *Acartia bifilosa*), *Pseudocalanus minutus elongatus* and *Temora longicornis* dominated in 1989-93. Rotifers were codominant in the warm seasons, *Bosmina coregoni maritima* in summer and *Fritillaria borealis* in spring.

Considerable fluctuations in the total mesozooplankton abundance were observed to depend on water temperature [*718*]. The greatest variations of the zooplankton numbers occurred in spring. Highest mean abundances and the most diversified composition were found in summer (Fig. 4.4.27). In winter, only few copepod species and the Thaliacea *Fritillaria borealis* were present.

The mean abundance of mesozooplankton during 1989-93 was slightly higher than that during 1979-88. This phenomenon occurred with different intensity, and was caused by increasing numbers of rotifers (*Synchaeta* sp.) in all seasons, and nauplii Copepoda in winter, summer and autumn. Since the late 1980s, the increase of organisms near the border of micro- and mesozooplankton corresponds with the slightly higher water temperature and decrease of the salinity in the surface waters. A decrease in numbers of *Evadne nordmanni* occurred in summer, whereas the abundance of *Bosmina coregoni maritima* fluctuated considerably.

Changes in the copepod population structure have taken place in the period under investigation. The number of dominating *Pseudocalanus minutus elongatus* (Fig. 4.4.28) was found to slightly decrease from year to year, but with higher intensity from the end of the 1980s in all seasons. At the same time, the numbers of *Temora longicornis* in summer, *Acartia longiremis* and *A. bifilosa* in warm seasons, and *Centropages hamatus* during the whole year showed an increasing tendency. Currently, *Acartia* spp. and *Temora longicornis* have become the dominants in the area. The main reasons for these changes were probably the progressive decrease of salinity in the deeper water layers and, simultaneously, the slight increase in the surface water temperature [*718*].

No considerable changes in the population structure of copepods were found following the inflow in 1993, which strongly increased the deep water salinity in the Gdansk Basin. But an increase in frequency of halophilus *Oithona similis*, and a decrease in the number of *Acartia* genus, were observed.

Increasing numbers of copepods infected with parasitic protozoa and epibionts were recorded for the coastal zone of the Gdansk Bight [*343,345,710,713*], and in other near-shore regions.

Long-term changes in the **Eastern Gotland Basin** are only represented by data at one station, *K1*, located in its southern part. Excluding winters with zero or only one observation, the mesozooplankton abundances are, on average, increasing in the period 1979-93 in all seasons (Fig. 4.4.29). In agreement with the situation in the Arkona and Bornholm Basins, high population densities of rotifers are found in spring after 1989 and these exceed the abundance of copepods. The increase of the copepods in summer is mainly caused by *Centropagus hamatus*, *Acartia* spp. and *Temora longicornis*.

Pseudocalanus minutus elongatus is characterised by a steady decrease in abundance during winter and spring (Fig. 4.4.25) in the peri-

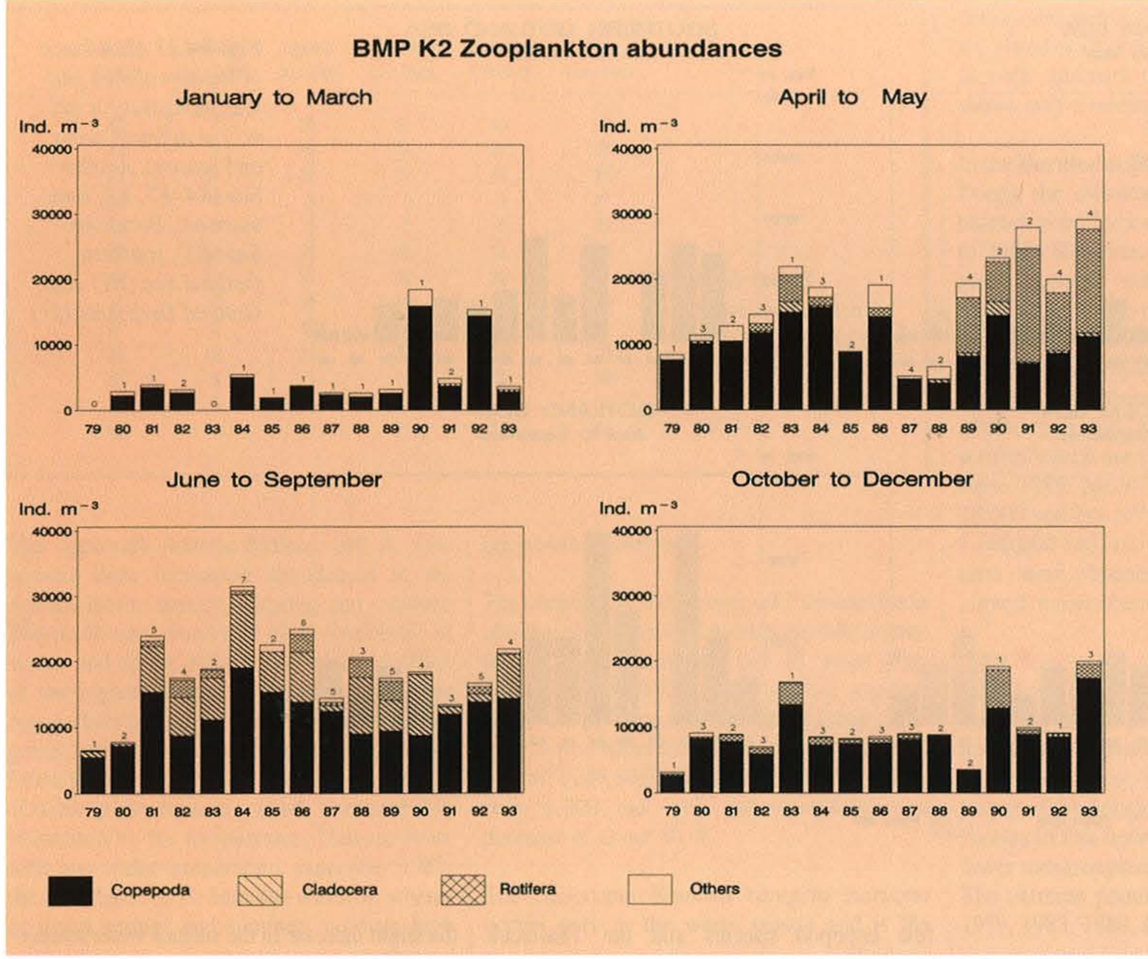


Fig. 4.4.26 Total abundance of different groups of mesozooplankton (in ind. m^{-3}) in the Bornholm Deep area (K2)

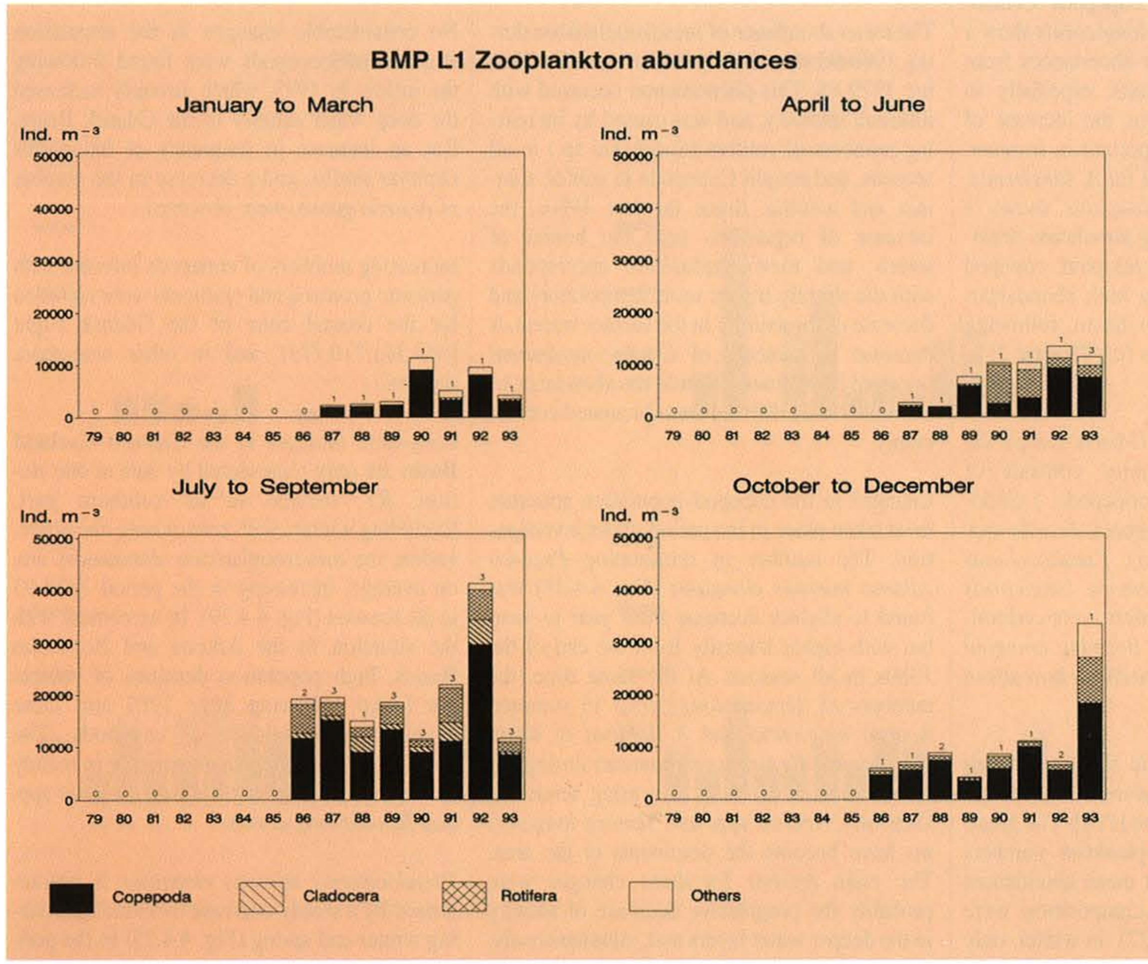


Fig. 4.4.27 Total abundance of different groups of mesozooplankton (in ind. m^{-3}) in the Gdansk Deep area (L1)

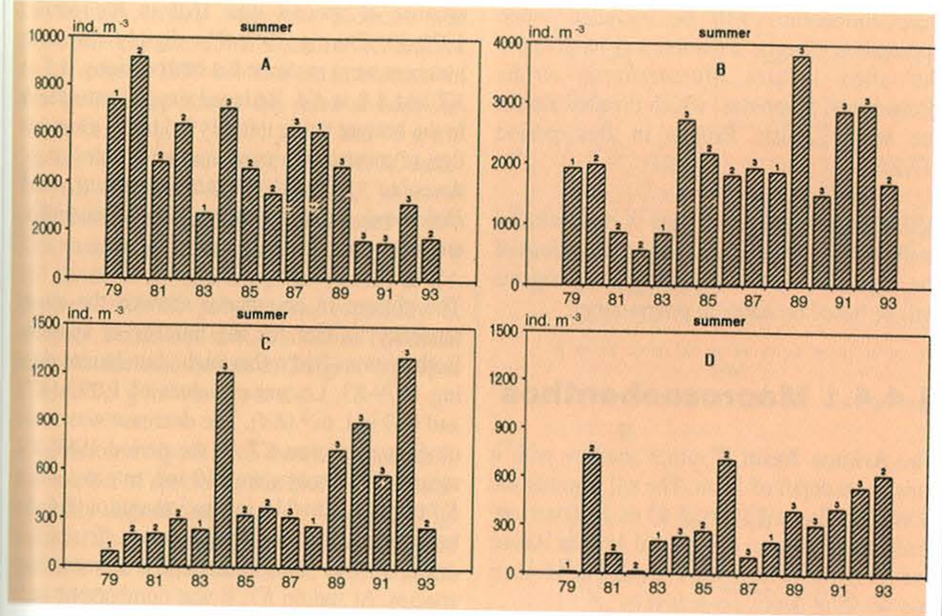


Fig. 4.4.28 Abundance of different copepod species (in ind. m^{-3}) in the Gdansk Deep area (L1) in summer

A - *Pseudocalanus minutus elongatus*, B - *Temora longicornis*, C - *Acartia longiremis*, and D - *Centropages hamatus*

od under investigation. In summer and autumn, the decrease is insignificant.

Since sampling frequency was too low, only general remarks are possible for the central station in the Eastern Gotland Basin (*J1*, Gotland Deep). During summer and autumn of the period 1979-93, the abundances of *Centropagus hamatus* appear to increase, whereas *Pseudocalanus minutus elongatus* decreased. The higher presence of rotifers in spring was observed from 1990 onwards, one year later as was the case in the southern parts of the Baltic Sea.

Only few remarks are possible for the **Northern Baltic Proper** (*H1*, *H2*, *H3*) and the **Western Gotland Basin** (*I1*, Karlsö Deep), due to the inadequate sampling frequency and the great interannual variations. As in the other sub-regions of the Baltic Proper, *Pseudocalanus minutus elongatus* showed a decreasing tendency in the period 1979-93 during all seasons. In the late 1970s/early 1980s, the mean abundance in spring was 4,000-5,000 ind. m^{-3}, whereas only about 1,000 ind. m^{-3} were found in the early 1990s. Beginning in 1990, the strong increase in rotifer concentrations, observed in the spring in the other regions, also occurred in the Northern Baltic Proper and Western Gotland Basin.

Alien species were observed in the Baltic Proper after the major salt water inflow in January 1993 and the subsequent inflow events. The planktonic turbellar *Alaurina composita* was found for the first time and in relatively large numbers in 1993 in the more haline deep water of the Arkona and Bornholm Basins. The horizontal distribution of this species is characterised by them spreading out to the east and north. Single specimen of *A. composita* were found in the Gotland Deep in February 1994. The alien species *Acartia*

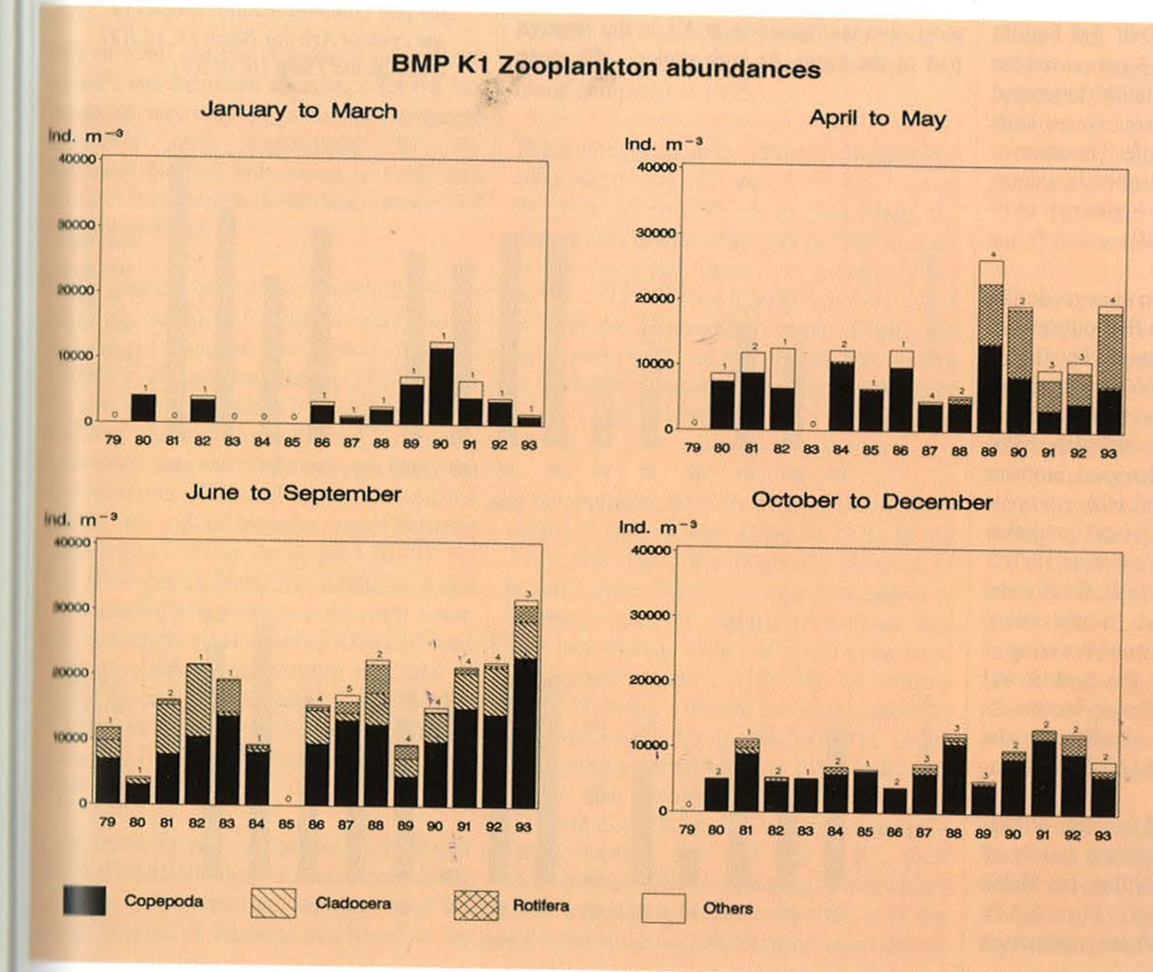


Fig. 4.4.29 Total abundance of different groups of mesozooplankton (in ind. m^{-3}) in the south-eastern Gotland Basin (K1)

tonsa [*106,284*] is now common in every summer and autumn in the Arkona Basin with increasing abundances. The propagation of this copepod to the central basins of the Baltic Sea is possibly supported by rising water temperatures in recent years.

4.4.3.5 Bacterioplankton

(An assessment of the regional distribution and temporal variations of bacterioplankton in the Baltic Proper is given in Chapter 10.3.)

4.4.4 Benthic biology

L.-E. Persson[1], A. Osowiecki, J. Warzocha, S. Olenin, S. Solovjova, A.-B. Andersin

The hydrographic conditions in the Baltic Proper have been already discussed in Chapter 2, and in more detail in Chapter 4.4.1. Therefore, only some additional information needs to be added here. The salinity in the bottom water of the Baltic Proper is varying due to inflows of salt water at irregular intervals. Its range in the Arkona Basin is 15-25 psu and in the Eastern Gotland Basin it is 10-14 psu. These salinities should provide the opportunity for a more marine community to persist in this area. However, unfavourable oxygen conditions, frequently occurring in the bottom water during at least 60 years, have caused the genesis of a disturbed community dominated by polychaetes. Even large areas are periodically devoid of macroscopic life. The absence of benthic macrofauna has resulted in a fourfold increase of areas with laminated sediments in the last 50 years [*294*].

Three more general long-term changes in the zoobenthic community of the Baltic Proper in later decades have been discussed. The first is the so-called 'oceanisation' [*401*] which was recognised in the southern part with the introduction of new marine species below the halocline. The second is the increased biomass above the halocline connected with eutrophication [*117,286*] and/or decreased predation pressure following the exploitation of flatfish in the 1920s and 1930s [*532,534*]. Finally, the most recent process called 'brackisation' [*602*], with decreasing salinity and lowering of the halocline, especially in the central and northern parts of the Baltic Proper, has resulted in improved oxygen conditions and recolonisation of macrofauna [*173*].

The following assessment of the state of benthos during 1989-93 will be based mainly on the representative stations within the Baltic Monitoring Programme (cf. Fig. 4.4.1). Moreover, data from national programmes on macrozoobenthos will be included when appropriate. Special attention will be given to the alien species *Marenzelleria viridis* (Polychaeta, Spionidae) which invaded almost the whole Baltic Proper in this period [*77,736*].

Although macrophytobenthos is not included in the BMP, some remarks on the conditions of the algal belts and submerged phanerogams will be based on national programmes.

4.4.4.1 Macrozoobenthos

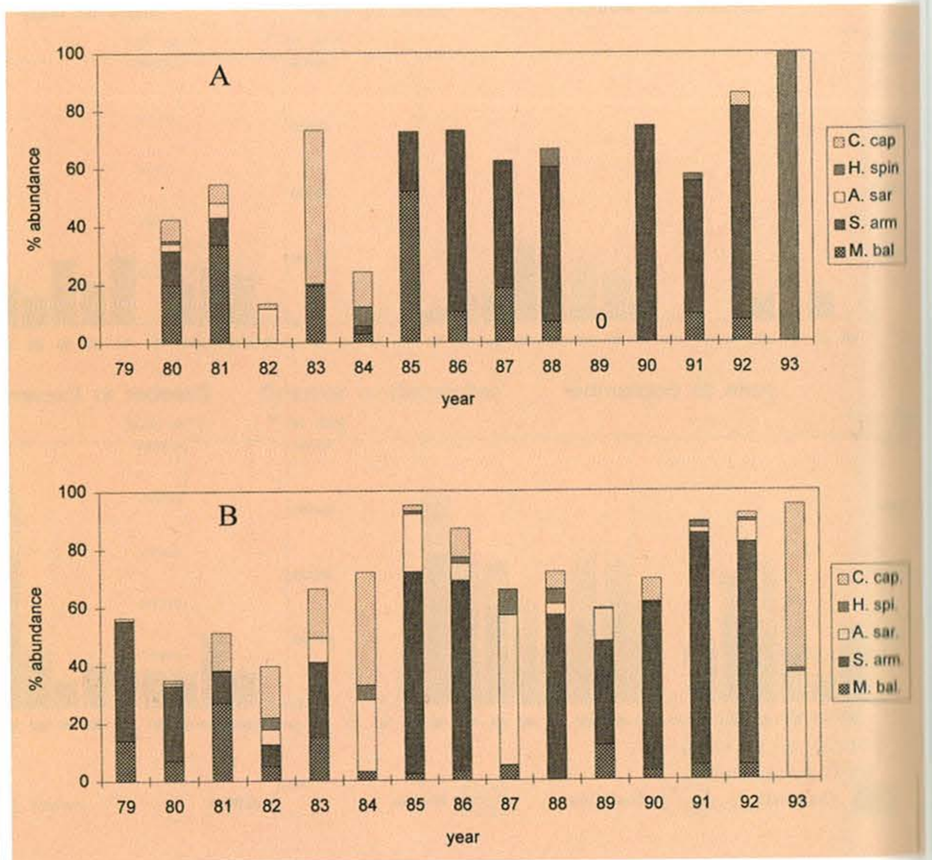
The **Arkona Basin** is rather shallow with a maximum depth of 53 m. The sill towards the Bornholm Basin is at about 43 m. The two representative stations, *K7* (central Arkona Basin) and *K4* (Arkona Deep), are situated at 44-46 m and 46-48 m depth, respectively.

The frequency of unfavourable oxygen conditions in the bottom water seems to have increased in the southern parts of the Baltic Sea in the 1980s [*15*], although hydrogen sulphide has not been measured in samples from this area according to common hydrographic casts. However, spots of *Beggiatoa* on top of the sediments have been reported in this area in June 1989 [*581*].

The number of species has decreased significantly at both representative stations, with the most obvious reduction at *K7* in the western part of the basin. At both stations, the mean number of species was 16.2 in the period 1979-83. During 1989-93, sligthly different numbers were measured at both stations, 3.2 at *K7* and 6.8 at *K4*. Reduced oxygen saturation in the bottom water initially led to the elimination of most of the more marine species (e.g., *Aricidea jeffreysi*, *Nepthys ciliata* and *Halcampa duodecimciratta*), but later influenced all species.

The change in abundance showed the same tendency as that for the number of species. Both stations had rather high abundances during 1979-83, i.e., mean values of 1,097 (*K7*) and 633 ind. m^{-2} (*K4*). The decrease was most obvious at station *K7*. In the period 1989-93, mean abundances were 119 ind. m^{-2} at station *K7* compared to 316 ind. m^{-2} at station *K4*. At both stations, the polychaete *Scoloplos armiger* was the numerically dominating species. At station *K7*, it was number one at 9 out of 20 samplings and at station *K4* at 18 out of 35 samplings. Its dominance was most pronounced in the present assessment period. At station *K7*, *Macoma baltica* ranked second in importance, dominating at 4 samplings, while *Capitella capitata* ranked second at station *K4*, dominating at 7 samplings. The variation in dominance values for five important species is shown in Figure 4.4.30.

Fig. 4.4.30 Variation in dominance for five common benthic species of the central Arkona Basin (A, at K7) and Arkona Deep (B, at K4)



1 see ANNEX 11.3 for addresses of authors

Excluding *Arctica islandica*, biomass was dominated by *M. baltica* at stations *K7* and *K4* in the period 1979-83. The mean biomass in this period was almost the same at both stations, i.e., 74 g m^{-2} (n=5) at *K7* and 75 g m^{-2} (n=13) at *K4*. In the period 1984-88, *M. baltica* still remained dominant at *K7*, whereas polychaetes and priapulids dominated at *K4*. The mean biomass declined drastically to 4.1 g m^{-2} (n=6) at *K7* and 5.4 g m^{-2} (n=12) at *K4*. During 1989-93, polychaetes, especially *S. armiger*, dominated at both stations. The mean biomass declined further to 2.2 g m^{-2} (n=9) at *K7* and 2.9 g m^{-2} (n=11) at *K4*.

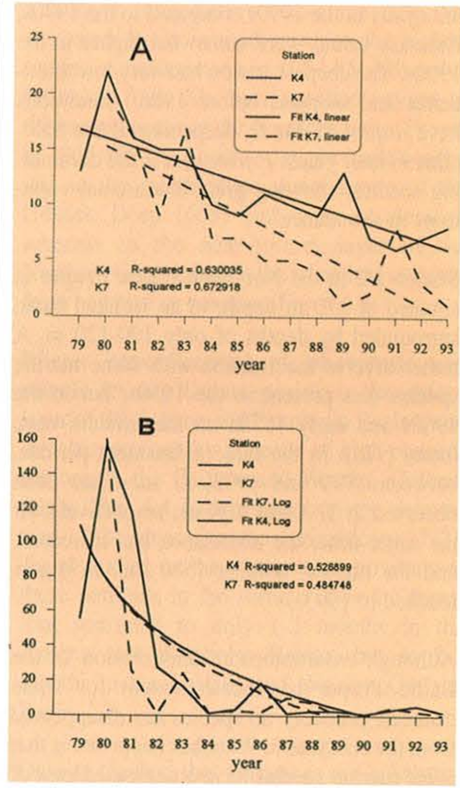
Regression analyses have shown the decline in the number of species and in biomass to be statistically significant at both stations (Fig. 4.4.31). The main change took place between the first and second assessment period, in the mid-1980s.

The representative station in the **Bornholm Basin** (*K2*, Bornholm Deep) is situated at about 90 m depth. This means that the deepest part at 105 m has not been studied. The bottom fauna of the Bornholm Basin below 70 m became impoverished in the early 1960s [*400*]. In the period 1979-93, station *K2* has been visited on 51 cruises. On 19 of them, no macrofauna were found. The maximum number of species (5) was found in 1980. During 1989-93, the number of species varied between zero and three.

Only in 1980 did the abundance exceed 100 ind. m^{-2}, and the mean abundance for the last five years was only 35, with the polychaete *Antinoella sarsi* dominating (87 %). Maximum biomass was found in 1980 with 6.4 g m^{-2}, compared with the mean value of 0.7 g m^{-2} for 1989-93.

In the southern part of the Bornholm Basin (60-70 m depth), 3-5 stations were visited yearly by Poland in the period 1978-93. According to historical data [*137,198*], this area was earlier inhabited by a diverse fauna, consisting of about 20 species with *S. armiger*, *Terebellides stroemi* and *Astarte* spp. being the most numerous. The total abundance reached 700 ind. m^{-2} and the biomass ranged between 3 and 40 g m^{-2}. Within the period 1980-93, the number of species found was limited to 6 and the abundance and biomass were much lower, with maximum values of only 250 ind. m^{-2} and 15 g m^{-2}, respectively. *Astarte* spp. and *T. stroemi* were missing. *M. baltica* dominated in biomass and *A. sarsi* and *S. armiger* in abundance. In 1993, no living macrofauna was found.

In the northern part of the Bornholm Basin with depths of 60-70 m, a community dominated by polychaetes in numbers and by *Astarte borealis* in biomass was found in the



middle of the 1970s [*195,533*]. In 1975, fifteen species were found in depths between 60 and 80 m. In 1990 and 1991, five stations were visited in depths between 64 and 71 m [*536*]. Only *A. sarsi* was found, usually at low levels of abundance and biomass. Empty shells of *A. borealis* were abundant. In shallower areas of this region, a 30-40 % increase in biomass was found compared to 1975.

The bottom fauna in the **Gdansk Basin** below 100 m depth was still rather diverse in the middle of the 1960s [*735*]. In the late 1960s, the macrofauna disappeared, and in later decades only *A. sarsi* was found to be present. In the period 1979-93, azoic conditions were found at 10 of the 23 sampling stations. During the last 5-year period, no macrofauna was present in 1989 and in autumn 1993. The abundance of *A. sarsi* was below 50 ind. m^{-2} in the whole period.

In the southern part of the Gdansk Basin (80 m depth), 5 species were found in 1978. In the 1980s, this region was frequently observed to be azoic. After improved oxygen conditions in the late 1980s, *M. baltica* recolonised this area, and biomass values of 20-35 g m^{-2} were subsequently found in 1992/93. In shallow areas (19-30 m), a normal zoobenthic community was found during 1987-93. The abundance mostly varied between 1,000 and 4,000 ind. m^{-2}, but maximum values of 20,000 ind. m^{-2} were also obtained. The biomass usually varied between 100 and 400 g m^{-2}, with bivalves (*M. baltica*) dominating. Studies from 1992 in depths between one and 60 m in the inner part of the Gdansk Bight revealed that no

Fig. 4.4.31 Regression analysis on the number of benthic species (A; in ind. m^{-2}), and on the benthic biomass (B; in g m^{-2}) of the central Arkona Basin (K7) and Arkona Deep (K4)

significant changes in the macrofauna community had taken place in comparison with data from the previous three decades.

Two areas in the southern part of the **Eastern Gotland Basin**, one at 82-86 m and one at 90 m depth (*K1*), have been sampled since 1980 and 1979, respectively. In the shallower area, a diverse community, numerically dominated by *S. armiger*, was found in 1980-81. Biomass values reaching 15 g m^{-2} were dominated by *M. baltica* and *S. entomon*. In the period 1982-88, macrofauna was absent in three of the seven years, and when present, the abundance and the biomass were low. This was probably an effect of deteriorated oxygen conditions in the bottom water. In 1989-93, recolonisation was observed, and high abundances (50-550 ind. m^{-2}) and biomasses (3-12 g m^{-2}) were found. Dominating species were *M. baltica* and *Pontoporeia femorata*.

At station *K1*, a macrofauna community was observed during all 37 cruises to this station over a 15-year period. The lowest number of species (1) was found in 1982 and the highest, (7) in 1985, 1987 and 1988 (Fig. 4.4.32). The most common species was *A. sarsi*, which was present at 36 of the 37 samplings. In the first three years, *S. armiger* and *A. sarsi* dominated in numbers, whereas in the last five years, *P. femorata* was most abundant. In this period, 84 % of the biomass was made up by *M. baltica* and *P. femorata*.

The eastern part of the Eastern Gotland Basin with depths between 70 and 140 m was sampled in a national programme between 1985 and 1989 [*514*]. Three zones were delineated, (*a*) the upper benthic zone down to 80-85 m with a rather diverse and numerous macrofauna community, whose structure was rather stable and can be classified as *M. baltica* community, (*b*) the transition zone down to about 120 m, inhabited by a poor and sparse macrofauna community with *H. sarsi* as the characteristic species, and (*c*) the benthic desert zone with no macrofauna, usually at depths below 120 m. By the end of the third assessment period, zone (*c*) reached the lower boundary of the primary halocline (80-85 m) in the northern part of the Eastern Gotland Basin.

On the eastern slope of the Eastern Gotland Basin, one station at 47 m depth (*J2*) has been visited during 21 cruises since 1981. In total, 15 species have been found (7-10 per cruise). Their abundance varied between 2,200 and

19,300 ind. m^{-2}. The biomass, between 36 and 422 g m^{-2}, was dominated by *M. baltica* (19-360 g m^{-2}). Numerically, *M. baltica* and *M. affinis* dominated. The latter showed a cyclical variation with maximum numbers in 1987-89.

On the western slope of the Eastern Gotland Basin, station *J10* at 46 m has been visited almost yearly since 1984. This station was also visited in a Swedish national programme between 1970 and 1973 [*195*]. Five to ten species were found, with *M. affinis* the numerically dominating species, accounting for over 95 % of the abundance. The total abundance decreased in the 1990s, and the 1993 value (939 ind. m^{-2}) was the lowest ever recorded. The biomass was much higher in the 1980s and 1990s compared to the 1970s. This was partly due to the higher mean weight of *M. affinis*, and partly to the occurrence of *S. entomon* in later years.

The representative station in the **Western Gotland Basin**, *I1* (Karlsö Deep) at 110 m, which had no macrofauna in 1965 [*115*], was recolonised by *A. sarsi* in 1985. Since then, this species has been found in low numbers at 11 out of 13 sampling sessions up to 1993. In this year, some hemipelagic species were also found together with *M. baltica*. This change is probably the result of improved oxygen conditions in the bottom water due to the mixing and deepening of the halocline ('brackisation') or horizontal exchange by inflow events.

In the northern part of the Baltic Proper, two representative stations were studied. The deepest station (*H3*, Landsort Deep, 453 m) was only sampled occasionally with the last visit in 1988. Macrofauna has been absent at this station since the mid 1960s [*115*].

In the Askö area, close to the Swedish coast, five stations at 22 to 80 m depth have been monitored since 1973 [*116*]. Two stations in the inner archipelago were subject to great variations in abundance and biomass without any obvious trend. Two stations in the outer archipelago showed decreased abundance of *M. affinis* in the 1990s compared to the 1970s. Biomass values were somewhat higher in the 1990s. The deepest station had very low abundance and biomass before 1986. Conditions have improved due to deepening of the halocline in later years. *P. femorata* is the dominating species, showing great interannual variations in abundance.

Station *H2* in the **Northern Baltic Proper** is situated at 170 m depth, in an isolated basin surrounded by depths of only 100-120 m. A rather diverse macrofauna with some marine species was present in the 1950s, but in the 1960s and early 1970s no macrofauna were found [*115*]. In the three assessment periods, between 1979 and 1993, *A. sarsi* has been observed at 1, 3 and 4 years, respectively. At the same time, the abundance has increased, and the highest number (580 ind. m^{-2}) was reached in 1993.

Although environmental deterioration in the Baltic Proper has been evident for some decades, probably no species has disappeared from the ecosystem. What has happened is that some marine species have a decreased area of distribution due to unfavourable oxygen conditions and reduced salinity below the halocline. Several species have disappeared from the sub-halocline areas of the south-eastern parts of the Baltic Proper and the Bornholm Basin, and are now only present in the Arkona Basin and in the south-western parts of the Baltic Proper.

4.4.4.2 Macrophytobenthos

The distribution and density of phytobenthic species in the Baltic Proper are less well known than those for the zoobenthic species. This lack of information is also a strong case for more knowledge on the long-term changes in the community, because only some areas have been monitored continuously.

In the Greifswald Bodden, Southern Baltic Proper, the rich red algae community, dominated by *Furcellaria fastigiata*, found in the 1930s down to 8 m depth, was totally absent in 1988 [*459*]. Today, the phytal zone is restricted to 4 m depth. The change was accompanied by a decrease in the *Secchi* depth from 2.5 to 0.2 m.

On the Słupsk Bank, in about 20 m depth, several species of red algae were found attached to stones in 1989 [*512*]. *Delesseria sanguinea*, reported in this area as early as 1929, was still present.

In the Puck Bight, in the western part of the Gdansk Bight, a diverse phytobenthos community was found in the late 1960s [*355*]. In the beginning of the 1970s, drastic changes took place. Species like *Fucus vesiculosus* and *Furcellaria fastigiata* disappeared, and the distribution of *Zostera marina* was considerably reduced. Today, filamentous brown algae of the family Ectocarpacae are dominating, and phytobenthic biomass has been reduced to 1/5 of that in the 1950s. The vigorous growth of filamentous species induces reducing conditions in the sediment, and this negatively influences the growth of vascular plants such as *Z. marina* [*355*]. In the beginning of this century, algal plants were found in depths of 20 m, whereas the current maximum depth is about 5 m.

Between 1978 and 1990, along the coasts of the Kaliningrad Region, Lithuania, Latvia and Estonia, the total stock of *Furcellaria lumbricallis* has decreased to 1/6 of the former level [*344,516*]. *F. vesiculosus* is almost totally absent from the same area, probably due to the decrease in water transparency, destructive wave effects and sand abrasion [*513*].

Along the Swedish coast, there exists a rather continuous *Fucus* community, although the depth distribution has decreased in later decades [*329*]. In some areas, the *Fucus* plants are subject to intense grazing from isopods [*656*]. In the Hanö Bight, the depth distribution of *F. vesiculosus* and *F. serratus* had decreased between 1975 and 1988, and the biomass was considerably reduced [*537*].

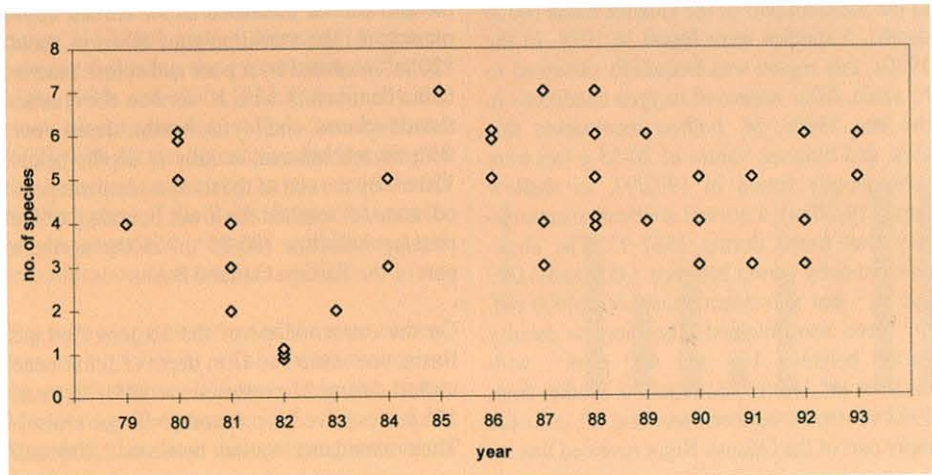


Fig. 4.4.32 Number of benthic species found in the south-eastern Gotland Basin (K1) on 37 sampling occasions

4.4.5 New species

More than 60 macrozoobenthos species have been introduced into the Baltic Sea during the past centuries, mainly due to shipping and aquaculture [*284,402,404*]. The latest invader, *Marenzelleria viridis* [*677*], a North American polychaete, has in ten years become abundant in many coastal areas, especially along the southern and eastern coasts of the Baltic Proper [*77,192,508,736*]. The highest abundance, about 10,000 ind. m^{-2}, is found in shallow basins with fresh-water inflows [*168*]. In the Gulf of Riga, it has been the most abundant species in recent years [*118*]. It has been found also at 50 m depth in the southern part of the Eastern Gotland Basin [*515*]. Along the Swedish coast, the new species is only found in the south, and usually up to now in low numbers [*536*].

So far it seems that the establishment of *M. viridis* in the Baltic Sea has had no negative effects on the existing community. Its metabolic strategy is the same as that of *Nereis diversicolor* in different salinity and oxygen regimes [*168*]. Ecologically, the two species differ by the ability of *M. viridis* to live deeper in the sediment [*597*], and to have a long planktonic stage, from early autumn to spring.

4.4.6 Summary

More detailed hydrographic investigations in the central basins of the Baltic Sea show decreasing **salinity** and reduced haline stratification, combined with the deepening of isohalines in the course of the stagnation period 1977-92, as well as the inverse processes after deep water renewals, which followed the major inflow in January 1993, and smaller inflow events before (Bornholm and Gdansk Deeps) and after (Gotland and Fårö Deeps) the major inflow (Fig. 4.4.2). The deep water exchange is regulated by the buffering capacity of the Bornholm and Gdansk Basins and the sill depth of the connecting channels.

Advective processes in the intermediate water layers caused abnormal warming-up. The highest **temperature,** above 7 °C, was observed in the Gotland Basin during 1989-90, and in the Gdansk Basin at the end of 1990 and the beginning of 1991, thereafter decreasing to the typical range of 3.6-5.4 °C.

The most striking differences in the **oxygen** conditions in the Gdansk Basin in 1989-93, as compared with the previous 10-year period, consist of much earlier and excessive saturation of the euphotic layer, and a significant reduction of oxygen deficiency in the deep water layers. The most unfavourable oxygen conditions occurred in the deep water of the Bornholm and Gdansk Basins in 1987-88, and were followed by improvement due to advective processes and vertical mixing. Anoxic conditions were terminated in the Gotland Deep in spring 1993. However, an intermediate layer of very low oxygen concentrations remained thoughout that year.

The layer immediately below the halocline was much better aerated, reaching the highest oxygen concentrations in 1984-85. Advective processes from this layer supplied oxygen to the bottom water of the Northern Baltic Proper in 1985-86. The positive oxygen trend continued in the intermediate water layers of the Gdansk Deep [*663*] and even accelerated, whereas in the near-bottom layer of the Northern Gdansk Basin, the trend had reversed from negative to positive (Fig. 4.4.6).

Winter concentrations of **phosphate** and **nitrate** develop distinct maxima in the surface layer of the Arkona and Bornholm Basins, and are characterised by plateaus of high concentrations in the Eastern and Western Gotland Basin (Fig. 4.4.7) depending on the delayed start of the phytoplankton springbloom [*312*]. Trend studies on the winter concentrations of these nutrients in the surface layer are therefore restricted to only 1-2 months in the Arkona and Bornholm Basins, but to 2-3 months in the Eastern and Western Gotland Basin.

Despite the fact that **silicate** is not a limiting nutrient in the Baltic Proper, its seasonal fluctuations are well developed. Since this nutrient also remains during the period of high biological productivity in considerable amounts in the euphotic layer of the Baltic Proper, year-round mean values may be used to increase the number of data which are available for investigations about the long-term variation of silicate in the surface layer.

Winter concentrations of phosphate and nitrate, being the most important nutrients available for the phytoplankton bloom in spring, are characterised by positive overall trends in the surface layer of all sub-regions of the Baltic Proper during 1958-93 and 1969-93 (Fig. 4.4.8). These trends mainly result from the strong increase in the 1970s and early 1980s (Table 4.4.4). In recent decades, the phosphate and nitrate concentrations fluctuated at a high level, whereas, on average, their long-term changes were insignificant.

Fertilisers consumed in the drainage areas are the most important source for eutrophication in shelf seas [*502*]. The synthetic phosphorus and nitrogen fertilisers, annually consumed in the drainage area of the Baltic Sea, were assessed together with the averaged phosphate- and nitrate-winter concentrations in the surface layer of the central parts of the Baltic Sea, e.g., the Bornholm Deep (cf. Fig. 10.1.2). Taking into account a delay of five to ten years, the correlations are obvious. However, the drastic reduction of the fertiliser consumption in the Baltic Sea drainage area, which began in 1989/90, is not yet significantly reflected by decreasing nutrient winter concentrations. However, this does not apply to the Arkona and Bornholm Seas where the first indications are found that the averaged phosphate and nitrate concentrations are decreasing in winter (Table 4.4.4, Fig. 10.1.2). These observations agree with the significantly decreasing nutrient load, for instance, in the Pomeranian Bight southern Arkona Sea, in recent times [*391*]. Chapter 4.3 describes similar results with respect to the nitrogen load in the Gulf of Riga. An exception for the Baltic Proper is the Landsort Deep, where more recently the highest phosphate and nitrate winter concentrations were observed (Fig. 4.4.8).

The annual mean values of the silicate concentrations behaved indifferent in the surface layer of the Baltic Proper during the recent assessment periods (Table 4.4.4). The high standard deviations of the mean values also include seasonal variations. It must be stressed, however, that the concentrations of this nutrient are decreasing occasionally to near the limit of detection. This occurred in recent years during diatom blooms in certain areas of the Baltic Proper [*486*].

The nitrate concentrations significantly increased in the intermediate water layers of the Bornholm Basin (Fig. 4.4.9), and later also of the Eastern and Western Gotland Basin (Fig. 4.4.10). In the bottom layer of the central basins, which occasionally influence the deep water of the Bornholm Basin, denitrification occurs at low oxygen concentrations, causing nitrate depletions.

In the central basins of the Baltic Sea, the silicate concentrations decreased significantly in recent decades at all depths below the halocline. This has already been discussed as the consequence of eutrophication [*485*]. Since this process also favours the development of diatoms, silicate fixed in the skeletons of these organisms is removed from water by deposition to the sediments.

The variation between oxic and anoxic conditions hides any phosphate trends in the bottom-water layer of the central basins of the Baltic Sea (Fig. 4.4.10). This is due to removal and regeneration of this nutrient depending on the redox potential and the iron cycle [*57*]. The mobilisation of phosphate stocks from the sediments seems to be exhausted during the course of long stagnation periods [*478*], limiting the phosphate accumulation, not only in the bottom water layer, but possibly also in intermediate depths (Fig. 4.4.9). Only in the Gotland Deep, significant phosphate trends could be identified in these depths.

Pronounced changes were observed in the seasonal development of the nutrient pool in the Gdansk Basin in 1989-93 as compared with the previous decade. The winter accumulation peaks of all nutrients had shifted from March towards February. In the most eutrophic coastal areas both phosphate and silicate winter pools were reduced to the lowest levels already in spring (Fig. 4.4.12). In offshore waters, minimum concentrations were reached in summer and late autumn, respectively (Fig. 4.4.13). The uptake of nitrogen was more synchronic. Nitrate was used up by May or June in all areas, except for the Vistula Estuary where its pool was sustained throughout the year.

Recent estimates for nitrogen and phosphorus loads, discharged from land-based sources, were reflected in the concentration levels of nitrate and phosphate in the Gdansk Bight. In the winter surface waters, the nitrate concentrations were much higher in 1989-93 than in 1979-88, which was not the case in the northernmost part of the Gdansk Basin. On the other hand, there was a well pronounced increase in the winter accumulation rate of phosphate in the Northern Gdansk Basin, whereas except for the Vistula Estuary their average concentrations in the Gdansk Bight decreased.

The negative trend in the winter accumulation of phosphate, found earlier in surface water of the Gdansk Deep, still existed in 1979-93, while the accumulation of phosphate in the Northern Gdansk Basin followed the overall positive trend. The strong negative trend in silicate (1971-88) ceased, probably due to the weakened demand for this nutrient (see below). The highly significant positive trend in nitrate concentrations of the winter surface waters of the Gdansk Deep, found for the period 1971-88, was no longer evident during 1979-93. However, in the vicinity of the Vistula Estuary, nitrate accumulated at an unprecedented rate, which caused the N/P ratio to increase considerably.

The impact of nutrients from rivers, with drastic surplus of nitrogen over phosphorus, is the most obvious reason for the advanced eutrophication of the Gdansk Basin. The increasing N/P ratio of assimilable nitrogen and phosphorus compounds could be traced in the isohaline winter waters of the Gdansk Bight as far as the Gdansk Deep (Table 4.4.6). The uptake ratio of nitrogen and phosphorus, calculated from the seasonal amplitudes of nutrients due to the spring and summer vegetation, yielded numbers oscillating around the *Redfield* ratio.

Owing to the favourable oxygen conditions, the concentrations of phosphate in the deep waters of the Gdansk Basin decreased considerably in 1989-93 (Figs. 4.4.6, 4.4.14). The same factor brought about a pronounced increase in nitrate concentrations. This coincidence of an effective phosphate sink and low denitrification activity has promoted a significant positive trend in the long-term changes of the N/P ratio throughout the Gdansk Basin.

Considering seasonal and long-term variations of the **phytoplankton** composition and biomass, the Baltic Proper is a relatively homogeneous area. An exception is station *H1* in the Northern Gotland Sea, which is possibly influenced by the Gulf of Finland. In the areas under investigation, a strong spring bloom develops in April/May, followed by a small summer bloom in July/ August, and an autumn bloom in October/November. From the Arkona Basin to the Northern Baltic Proper, there is a delay in the start of the spring bloom (cf. [*312*]). In recent years (mild winters), the differences in the timing of spring blooms seem to disappear in the Southern Baltic Proper.

The share of diatoms in the spring bloom has decreased, and as a result flagellates (mainly dinoflagellates) have benefited in the Southern Gotland and Bornholm Basins since 1989-90, possibly due to mild winters (cf. Chapter 2). On the other hand, the intensity of the autumn bloom seems to have increased since 1988, but this, however, is not reflected in the chlorophyll data. The phytoplankton biomass determined microscopically shows higher peaks than the related chlorophyll *a* concentrations, especially during the blooms.

For the period 1979-93, the annual mean concentrations of chlorophyll indicate a slight increase for most sub-areas of the Baltic Proper. This increase is only partly statistically significant. This agrees with the insignificant long-term changes of the winter concentrations of phosphate and nitrate in the surface layer during this period.

If only summer data are considered, a chlorophyll minimum occurred in 1989. However, at nearly the same time (1988-90), autumnal chlorophyll concentrations reached maximum values. Thus, summer and autumn values roughly show the opposite tendency, i.e., the summer chlorophyll concentrations mostly increased from 1989 to 1993, whereas the autumn values decreased.

In the entire Baltic Proper, three seasonal peaks of the phytoplankton development, biomass, chlorophyll *a* and the potential primary productivity, can be observed. Pooling all measurements for the whole Baltic Proper between 1979 and 1993, a slight increase for the three seasons is obvious.

The strong increase in numbers of rotifers in spring, beginning in the Baltic Proper in 1989, is the most important observation with respect to changes in the **mesozooplankton** composition in recent years. Rotifers have been discussed as indicator of eutrophication [*683*], but its development is also favoured by mild winters and springs. Another possible reason is the increase of smaller phytoplankton species, thus improving the food conditions for the rotifers.

The decrease in the abundance of *Pseudocalanus minutus elongatus* in all regions during 1979-93 has to be evaluated from both an abiotic and biotic viewpoint. Abiotic changes in the period under discussion are a decreasing salinity in the surface layer and in deep waters, and an increasing temperature, especially in the cold intermediate layer during the warm season (cf. Chapters 2 and 4.4.1). Biological changes reflect the food availability as well as the increasing grazing pressure by pelagic fish stocks and by the medusa *Aurelia aurita* [*60,600*]. The swimming behaviour differs between *Acartia* species, *Centropagus hamatus* and *Temora longicornis,* on one hand, and *Peusocalanus minutus elongatus* on the other hand. This may lead to growing stocks of the fast swimming or jumping group which can escape their enemies, and to a decrease of *P. minutus elongatus* which swims continuously with slow velocity [*124,643*] and is thus an easier prey.

Some **alien mesozooplankton** species entered the Baltic Proper or are spreading more to the east and north as a consequence of the inflow events of 1993/1994.

Due to the deterioration of the oxygen conditions, in the Arkona Basin, the **benthic macrofauna** shifted from a mollusc-dominated to a polychaete-dominated community in the early 1980s. At the same time, the species diversity, abundance and biomass decreased. The benthic macrofauna in the Bornholm and Gdansk Basins was found to be impoverished in the 1960s, and there were more or less azoic conditions below 90 m depth in 1993. In the 1980s, also depths between 60 and 80 m have been negatively influenced by unfavourable oxygen conditions. The polychaete *Antinoella sarsi* has become the most widespread organism. The azoic zone reached 120 m depth in the Eastern Gotland Basin, and areas between 80 m and 120 m were temporarily influenced by low oxygen concentrations. Some recolonisation took place during the recent assessment period due to a lowering of the halocline. The same effect, supported by oxygen supply due to advective processes, was found in the Western Gotland Basin where recolonisation began in 1985. In the central parts of the Northern Baltic Proper, the conditions in deeper areas have improved, resulting in an increased occurrence of *A. sarsi*.

The **macrophytobenthic community** seems to have suffered from eutrophication along most coasts, and this has resulted in shallower depths of settlement and increased abundances of filamentous algae, caused by reduced light penetration.

4.5 KATTEGAT AND BELT SEA (incl. Mecklenburg Bight, Kiel Bight, Fehmarn Belt, Great Belt, Little Belt and Sound)

G. Aertebjerg[1] (Convener)

4.5.1 Introduction

The Kattegat and Belt Sea constitute the transition area between the Baltic Proper and the North Sea (Fig. 4.5.1). The geographical Kattegat/Skagerrak border is also the delimitation of the Helsinki Convention Area, defined as the parallel of the Skaw in the Skagerrak at 57°44.43' N. The Kattegat surface area is 22,387 km² and the volume 421 km³. The narrow Little Belt and Great Belt connect the Kattegat to the southern Belt Sea, i.e., to the Kiel Bight, Fehmarn Belt and Mecklenburg Bight, which borders the Baltic Proper (Arkona Sea) at Darß with a sill depth of 18 m. The area of the Belt Sea, including the bights, is 18,273 km² and the volume is 262 km³. The narrow Sound connects the Kattegat and the Arkona Sea directly with a sill depth of only 8 m at Drogden. The area of the Sound, including the Køge Bight south of the sill, is 1,848 km² and the volume is 25 km³.

4.5.2 Hydrography

L. Rydberg[1]

Water exchange, volume and salt fluxes - The Kattegat and the Belt Sea form the outer part of the 'Baltic Estuary'. Although the areas have their own, well-defined hydrography, most oceanographic activities, measurements as well as theories, have been carried out with the Baltic Sea water exchange in mind. This is particularly so in the Belt Sea region where the instantaneous flux of water often exceeds 100,000 $m^3 s^{-1}$, thereby exceeding by an order of magnitude the mean outflow of fresh water from the Baltic Sea. The instantaneous flux is driven by the strongly variable sea level difference between the Kattegat and the Southwestern Baltic Proper, which in turn is mainly influenced by mesoscale winds and air pressure variations. This difference can be used to calculate the volume flux with a surprisingly high precision. On somewhat longer time scales, i.e., days to weeks, a record of the Kattegat sea level itself provides enough information to calculate the flux from relatively simple models ([*389,522,637,639*]; Fig. 4.5.2). Comparable results, used for example for the tuning of the models, may be obtained from records of the Baltic sea level, indicating the change in volume and thus the flux of water into or out from the Baltic Sea [*281*]. On time scales ≥14 days there is a high correlation between the Kattegat and the Baltic sea level [*587,638*], indicating that in- or outflows which continue for such a long time, are most effective for the water exchange (Fig. 4.5.3). The net supply of fresh water to the Baltic Sea, made up by river discharge, precipitation and evaporation, causes the mean sea level of the Southern Baltic Proper to be a few cm higher than the level in the Kattegat [*153*], thereby forcing the low salinity surface water through the Belt Sea. Ideally, because of higher salinities and densities in the Kattegat, there is at larger depths an inflow of Kattegat water. The instantaneous sea level differences, however, are much larger than the average. Thus, the ideal, average-built two-layer estuarine circulation is normally suppressed by one-layer flow in either direction.

The sea level variations alone, however, do not provide any further information, either on the quality of the water that is passing through, or of the complicated flow structure in the Belt Sea where two, three or four layers of different salinities may appear simultaneously. Thus, a record of the instantaneous flux itself is of limited value. To obtain the 'exchange of water', simultaneous measurements of the salinity, which indicates the origin of the water, are needed. Studies of the exchange of water between the Baltic Sea and the ocean have been carried out since the beginning of this century. *Knudsen* [*335*] assumed, on the basis of salinity measurements at the Darß Sill, that the heavier and more salty water entering the Baltic Sea had a mean salinity of 17.4 psu,

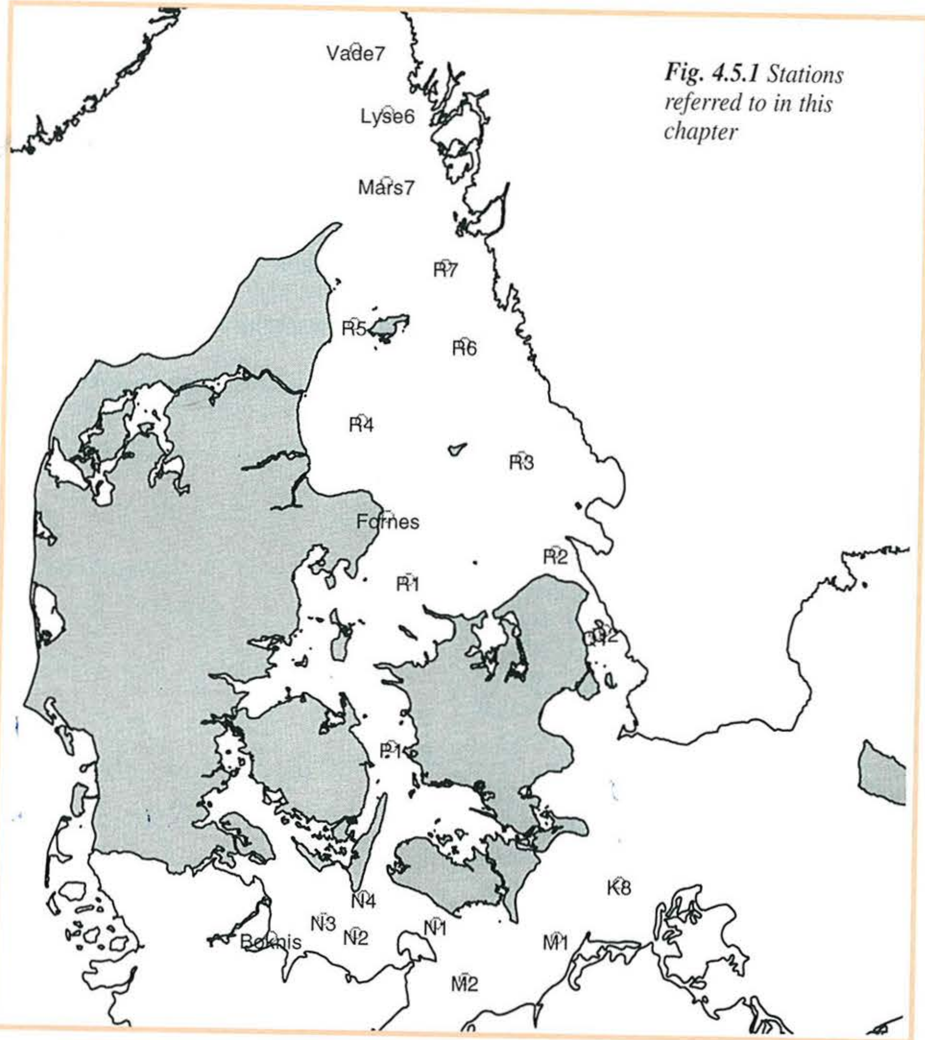


Fig. 4.5.1 Stations referred to in this chapter

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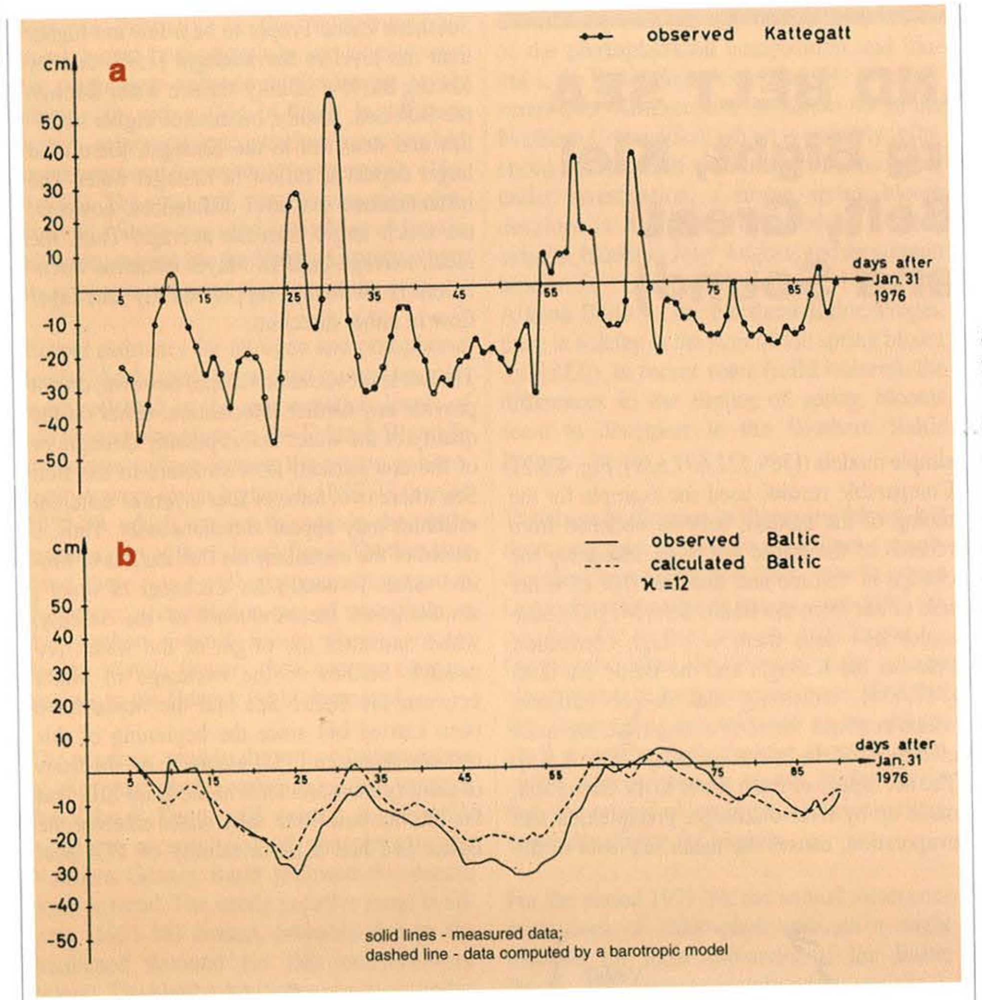


Fig. 4.5.2 The mean sea level in the Kattegat (a, daily average) and in the Baltic Sea (b) [637]

while the salinity of the outflowing Baltic Sea water was 8.7 psu. Using volume and salt balance, including an estimate of the net supply of fresh water to the Baltic Sea of 470 $km^3\ yr^{-1}$, *Knudsen* determined an average outflow of 940 $km^3\ yr^{-1}$ (30,000 $m^3\ s^{-1}$) and an inflow of 470 $km^3\ yr^{-1}$ (15,000 $m^3\ s^{-1}$). Attempts to improve the figures obtained by *Knudsen* have been partly focused on the fresh water budget, and partly on the relationship between salinity and volume fluxes. The long-term average river discharge has been determined more recently to 446 $km^3\ yr^{-1}$ or 14,100 $m^3\ s^{-1}$ [*71*]. Variations produced by rainfall and changing use of hydroelectric power are of further interest. By adding recent estimates of precipitation and evaporation over the Baltic Sea, the freshwater budget resulted in an average, total net fresh-water supply of 15,700 $m^3\ s^{-1}$ [*640*], i.e., not far from the original value used by *Knudsen*. Both precipitation and evaporation estimates can be further improved, however.

Because of the complicated hydrography within the Belt Sea region, much of the recent measurements and related water exchange theories deal with inflow of deep water inside, or outflow of surface water outside the region,i.e., either in the Arkona/Bornholm Sea area or in the northern Kattegat, where the water column is more or less persistently two-layered [*24,89,90,175,389,521,522,584,637,654,685, 686*]. From observations in the Bornholm Channel, an average inflow of deep water to the Baltic Sea of 12,000 $m^3\ s^{-1}$ ($10<S<18$) was determined [*686*]. An average outflow of Kattegat surface water of 55,000 $m^3\ s^{-1}$ ($20<S<30$) was observed [*24*]. These figures are still coarse and need to be improved.

Major inflows, large enough for a renewal of the bottom water ($11<S<13$) within the Baltic Sea, have been studied on the basis of long-term salinity measurements at the Darß Sill [*259,448*]. Although there was at times evidence for major inflows almost every year, the authors also found long periods without major inflows. The definitions used for a major inflow (e.g., S>17), however, imply that they represent only 10-20 % of the total inflow of deep water and salt as defined in [*686*]. The majority of inflows occurs with lower salinities, whereby a considerable part takes place through the Sound [*639*].

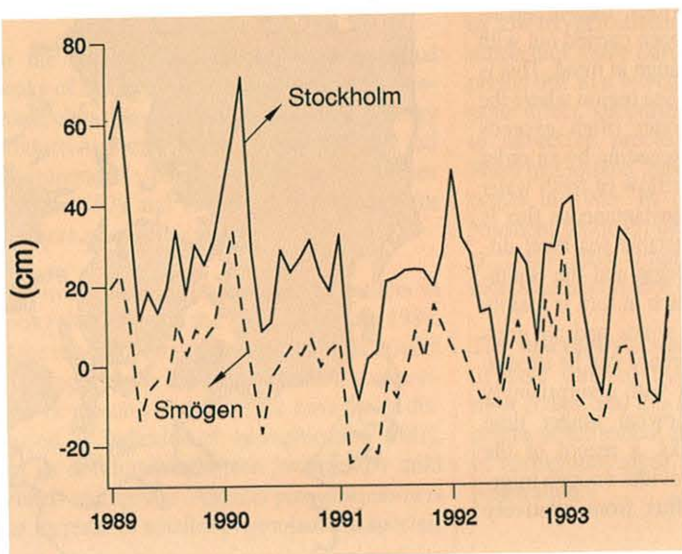


Fig. 4.5.3 Monthly mean sea level at Stockholm (representing the 'Baltic Sea') and at Smögen ('Skagerrak') [587]

A comprehensive study of sea level variations in the Baltic Sea, together with simultaneous current measurements, was carried out in the Belt Sea region [*281*]. The sea level data were used to calculate the volume changes of the Baltic Sea in relation to the observed volume fluxes in the Belt Sea. It was shown that the sea level at Landsort, averaged over a few days, is representative for the whole Baltic Sea [*281*]. On average, the instantaneous flux in the Belt Sea was divided between the Great Belt, the Sound and the Little Belt in the ratio 7:3:1. Recent measurements, related to the bridge constructions, indicated a ratio of 8:3:1.

Hydrography of the Belt Sea and the Kattegat - In the Kattegat, outflowing Baltic Sea water meets high salinity ocean water. The Kattegat surface water salinity varies from 12 to 30 psu with a strong north-south gradient. The vertical stratification is very strong. The halocline is normally situated at a depth of about 15 m. The deep water below has a salinity ranging from 32 to 34 psu. The horizontal variability, including seasonal and interannual variations, are further described elsewhere [*585,644*].

The nearly constant deep water salinity and the strong gradient in the surface water (Fig. 4.5.4) illustrate the Kattegat circulation, where the mixing is almost uni-directional upwards driven by wind [*90,637*]. The deep water flow into the Kattegat, therefore, varies with season. In winter, when the winds are stronger, the average flows increase by a factor of 2-3 compared to the summer period [*24*]. At the same time, the average salinity decreases in the deep water and increases at the surface. Although there is often a boundary current along the Swedish west coast, existing current measurements indicate that about 50 % of the surface water flow occurs on the Danish side of Laesö [*24*].

Because of the strong stratification, the tendency for two-layer circulation is rather high, but wind and air pressure driven uni-directional (barotropic) flow still dominates. The tidal circulation is weak. Maximum volume flows may exceed 200,000 $m^3\ s^{-1}$ in either direction. Such occasions are normally coupled with inflows to or outflows from the Baltic Sea.

In the Belt Sea, there are normally three different water masses, i.e., Baltic Sea surface water, Kattegat surface water and Kattegat deep water. In the deeper parts of the Sound, 'stagnant' basin water of higher salinity may appear [*24*]. Detailed studies of salinity, flow patterns and mixing indicate a complex structure. Although the flow is often thought of as two-layered, with inflowing Kattegat surface water underneath outgoing Baltic Sea surface water, the flow is uni-directional, either in or out, in at least 50 % of the cases [*281*]. Unlike the Kattegat, mixing in the Belt Sea is double-directed, with horizontal salinity gradients in both surface and deep water. The mixing/stratification conditions are illustrated by Figure 4.5.4, showing the salinities after a storm. The water masses in the Belt Sea are well-mixed, while in the Kattegat and in the Sound a strong halocline still exists.

Hydrography during 1989-93 - Regarding sea levels and salt water intrusions into the Baltic Sea during 1989-93, cf. Chapter 2.5. During this period, there was an extreme inflow situation in January 1993. A volume of 300 km^3 entered the Baltic Sea, of which 125-150 km^3 was high-salinity water. About 65 % of that water entered through the Sound. The mean sea level was more than 50 cm higher in the Kattegat than in the Southern Baltic Proper during that period [*620*]. Salinity measurements within the monitoring programmes did not show any remarkable deviations from 'normal values', although the average surface salinity during December to February was slightly higher within the Belt Sea region compared to the earlier periods of 1979-83 and 1984-88. Higher salinities indicate more Kattegat or Skagerrak water in the Belt Sea, but the difference was in this case insignificant. Such variations have to be considered on a seasonal or, preferably, on a monthly time scale. This is because the exchange of surface water in the Belt Sea/Kattegat region occurs within a few months or less, particularly during the winter period.

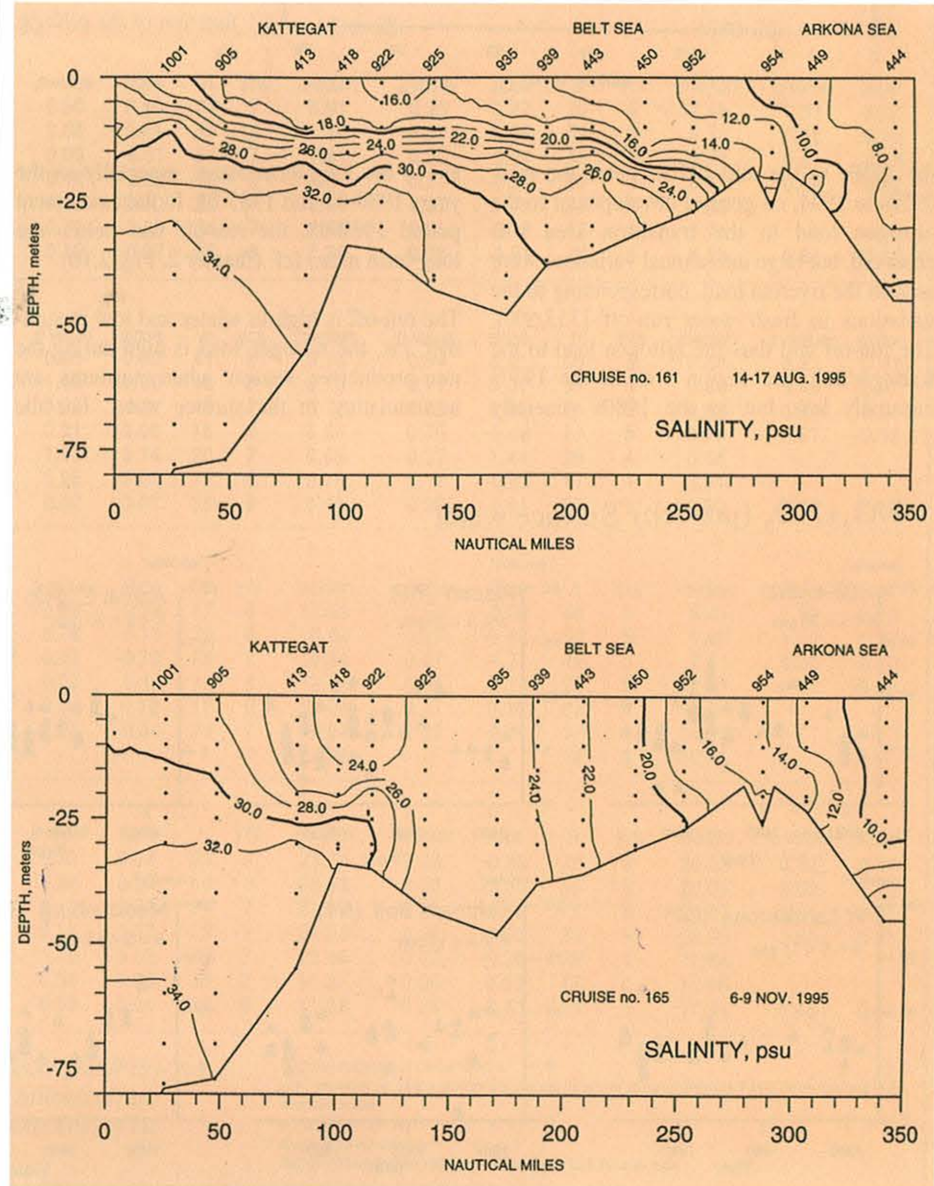


Fig. 4.5.4 Salinity distribution (psu) along a transect from the north-eastern Kattegat, through the Great Belt and Fehmarn Belt, to the Arkona Sea at two occasions, 14-17 August 1995 and 6-9 November 1995

4.5.3 Hydrochemistry

4.5.3.1 Nutrient load

G. Aertebjerg[1]

The nutrient load from sewage and the atmosphere is relatively evenly distributed over seasons and from year to year. Otherwise, the diffuse load of especially nitrogen, but also phosphorus, via rivers shows a pronounced seasonality and large variations from year to year, parallel to the variations in fresh water run-off.

The anthropogenic phosphorus input has increased until the 1970s, and since then decreased in parallel to the development of phosphorus removal from sewage in the drainage area, and in the later years was also influenced by the introduction of phosphate-free detergents. The nitrogen load from both land and atmosphere has almost doubled from

[1] see ANNEX 11.3 for addresses of authors

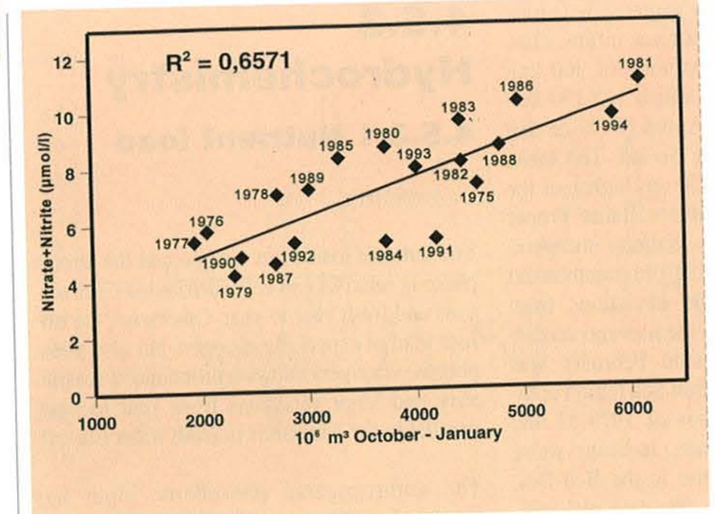


Fig. 4.5.5 Mean January/February concentrations of nitrate+nitrite in the 0-10 m layer at stations R3, R1 and P1 in the southern Kattegat and Great Belt as a function of the October-to-January fresh-water run-off from Denmark to the Kattegat and Belt Sea, 1975-94

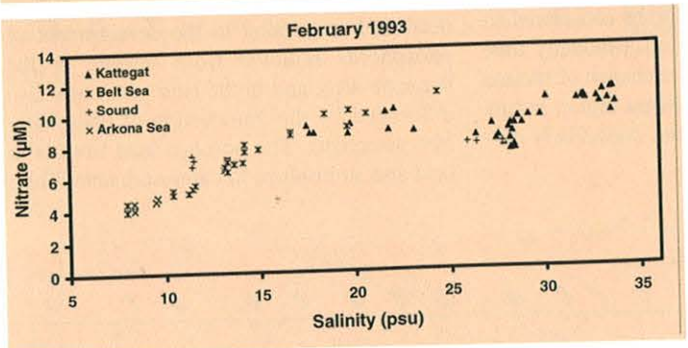


Fig. 4.5.6 Nitrate concentrations 8-18 February 1993 in the 0-10 m layer of the transition area between the northern Kattegat and the Arkona Sea as a function of the salinity

the 1950s to the mid-1970s. From the mid-1970s to 1994, no general development in the nitrogen load to the transition area was observed, but large interannual variations were seen in the riverine load, corresponding to the variations in fresh water run-off [353,606]. The run-off and thus the nitrogen load to the Kattegat/Belt Sea region was in the 1970s unusually low, but in the 1980s generally above the long-term mean, especially in the years 1980-81 and 1985-88. In the assessment period 1989-93, the run-off was below the long-term mean (cf. Chapter 2, Fig. 2.10).

The run-off is high in winter and low in summer, i.e., the nitrogen load is high during the non-productive season when nutrients are accumulating in the surface water. For the period 1975-94, a significant correlation has been shown between winter run-off from Denmark and winter concentrations of nitrate+nitrite in the surface water of the transition area, excluding the central Sound, where most of the nutrient load originates from sewage (Fig. 4.5.5). The winter surface concentrations changed significantly with $p<0.01$ by 7 % and 0.50 mol dm^{-3}, respectively, per 10 % change of the winter run-off.

The winter nitrate concentrations are higher in the transition area than in the Baltic Proper. If this was only due to mixing of Baltic Sea surface water (salinity 8 psu, nitrate 4-5 µmol dm^{-3}) and nitrate-rich Skagerrak/North Sea water (salinity 33.5 psu, nitrate 10-11 µmol dm^{-3}), the nitrate concentration would be a linear function of the salinity. The deviation seen in Figure 4.5.6 from the linear relationship is due to local nitrate load.

4.5.3.2 Nutrient variability in water

L. Andersson[1]

Seven BMP stations in the area have been selected for the analysis of nutrients, i.e., two in the northern (*R6, R7*) and two in the southern part of the Kattegat (*R1, R3*), one in the Sound (*Q2*) and two in the southern part of the Belt Sea area (*N1, M2*). The three inorganic nutrients, phosphate, nitrate+nitrite and silicate, mainly show the same pattern in their annual cycles in the surface water. During January and February, the concentrations are highest. At the end of February or in March, the spring bloom starts and the levels begin to

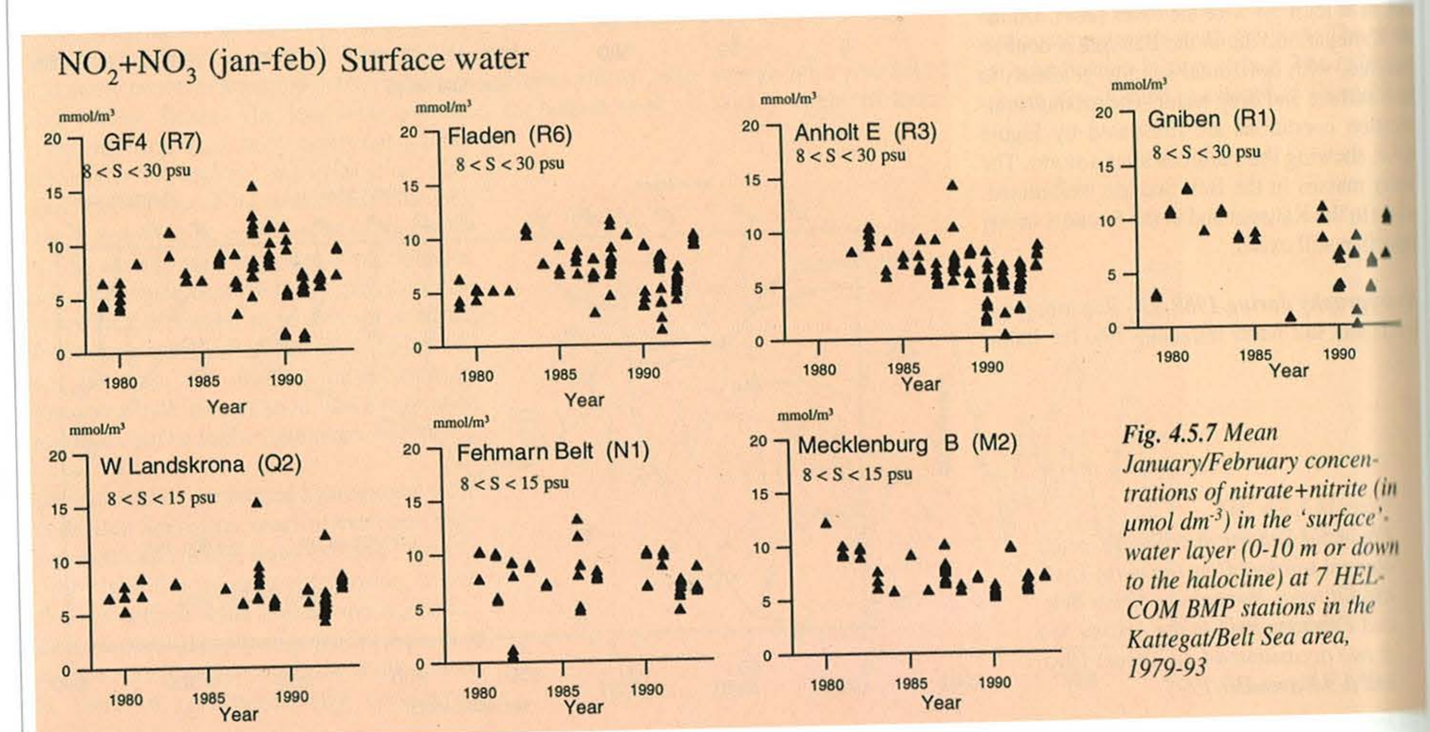


Fig. 4.5.7 Mean January/February concentrations of nitrate+nitrite (in µmol dm^{-3}) in the 'surface'-water layer (0-10 m or down to the halocline) at 7 HELCOM BMP stations in the Kattegat/Belt Sea area, 1979-93

[1] see ANNEX 11.3 for addresses of authors

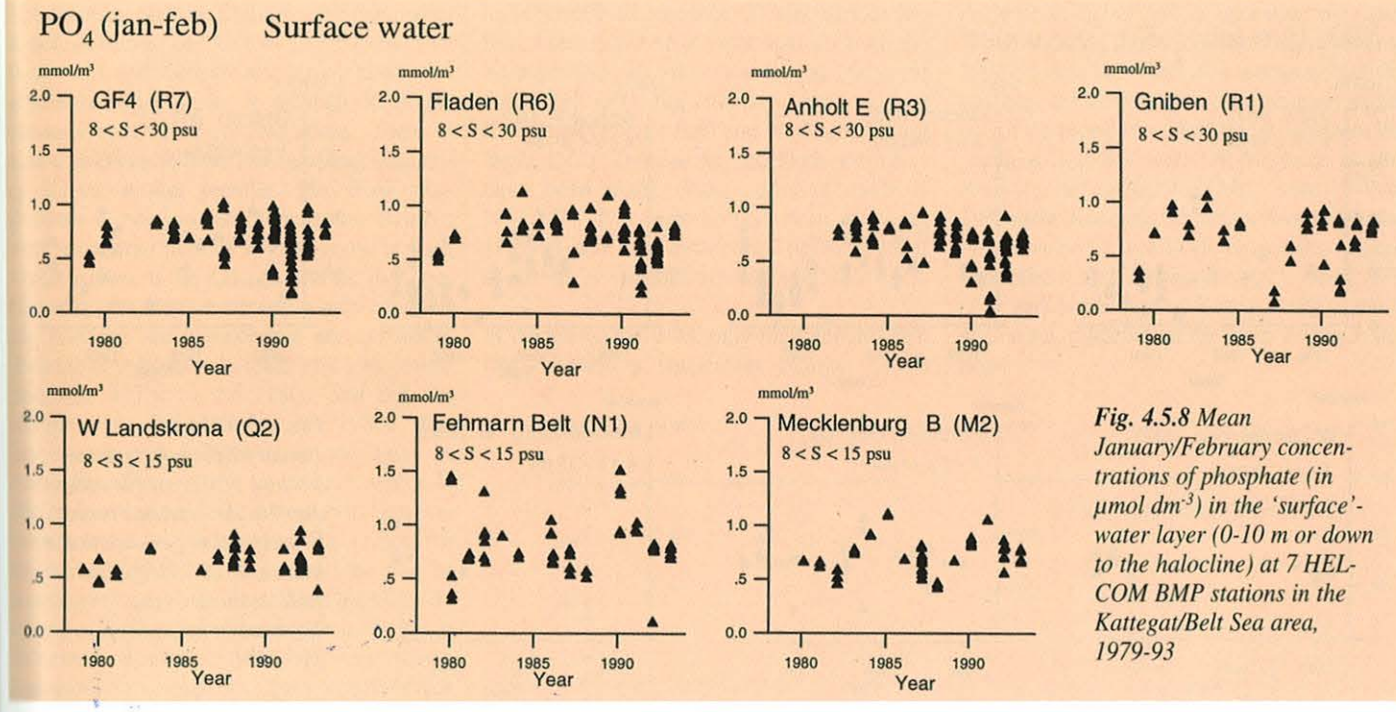


Fig. 4.5.8 Mean January/February concentrations of phosphate (in µmol dm^{-3}) in the 'surface'-water layer (0-10 m or down to the halocline) at 7 HELCOM BMP stations in the Kattegat/Belt Sea area, 1979-93

		1979–93					1979–88					1989–93				
NO_3+NO_2																
Stat. no.	*Stat. name*	*n*	yrs	median	*p*-value	slope	*n*	yrs	median	*p*-value	slope	*n*	yrs	median	*p*-value	slope
R7	*GF4*	*133*	14	7.18	0.58	0.06	*63*	9	6.91	0.35	0.27	*70*	5	7.48	0.81	-0.31
R6	*Fladen*	*109*	15	7.11	0.08	0.21	*50*	10	**7.26**	**0.02**	**0.24**	*59*	5	6.70	0.46	-0.39
R3	*Anholt E*	*127*	12	6.55	0.06	-0.31	*51*	7	7.81	0.55	-0.24	*76*	5	5.71	0.81	0.08
R1	*Gniben*	*46*	13	6.93	0.16	-0.32	*19*	8	8.01	0.27	-0.54	*27*	5	6.17	0.46	-0.22
Q2	*W Landskrona*	*54*	11	6.91	1.00	0.00	*25*	7	7.61	0.37	0.18	*29*	4	6.30	-	-
N1	*Fehmarn Belt*	*51*	11	7.58	0.76	-0.06	*34*	7	7.50	0.55	-0.18	*17*	4	7.74	-	-
M2	*Mecklenburg Bight*	*64*	14	7.15	0.19	-0.22	*35*	9	**7.78**	**0.05**	**-0.61**	*29*	5	6.39	0.81	0.23
PO_4																
Stat. no.	*Stat. name*	*n*	yrs	median	*p* value	slope	*n*	yrs	median	*p* value	slope	*n*	yrs	median	*p* value	slope
R7	*GF4*	107	13	0.69	0.95	0.00	41	8	0.74	0.54	0.01	66	5	0.66	0.81	0.01
R6	*Fladen*	95	13	0.71	1.00	0.00	40	8	0.76	0.71	0.01	55	5	0.68	0.46	-0.07
R3	*Anholt E*	117	12	0.68	0.00	-0.01	46	7	0.77	0.29	0.00	71	5	0.62	0.81	-0.01
R1	*Gniben*	45	14	0.68	0.91	0.00	18	9	0.64	0.75	-0.02	27	5	0.70	0.81	-0.02
Q2	*W Landskrona*	49	11	0.67	1.00	0.14	20	7	0.65	0.37	1.44	29	4	0.68	-	-
N1	*Fehmarn Belt*	53	12	0.81	0.95	0.00	36	8	0.76	0.17	-0.02	17	4	0.91	-	-
M2	*Mecklenburg Bight*	62	14	0.73	0.32	0.01	35	9	0.68	0.92	-0.01	27	5	0.80	1.00	-0.01
SiO_2																
Stat. no.	*Stat. name*	*n*	yrs	median	*p* value	slope	*n*	yrs	median	*p* value	slope	*n*	yrs	median	*p* value	slope
R7	*GF4*	*57*	11	7.91	0.09	-0.38	*17*	6	11.43	0.71	-0.29	*40*	5	6.41	1.00	-0.02
R6	*Fladen*	*79*	13	8.52	0.76	-0.13	*29*	8	10.66	0.27	0.43	*50*	5	7.28	1.00	0.32
R3	*Anholt E*	*75*	12	**8.10**	**0.03**	**-0.72**	*26*	7	10.54	0.07	-1.71	*49*	5	6.81	0.46	0.70
R1	*Gniben*	*38*	13	12.17	0.25	-0.40	*16*	8	12.94	0.54	-1.13	*22*	5	11.61	0.81	-0.55
Q2	*W Landskrona*	*37*	10	13.33	0.59	0.12	*10*	6	12.08	0.71	0.16	*27*	4	13.80	-	-
N1	*Fehmarn Belt*	*43*	11	15.32	0.88	-0.04	*26*	7	16.04	1.00	-0.04	*17*	4	14.21	-	-
M2	*Mecklenburg Bight*	*12*	3	15.85	-	-		-	no data	-	-	*12*	3	15.85	-	-
Salinity																
Stat. no.	*Stat. name*	*n*	yrs	median	*p* value	slope	*n*	yrs	median	*p* value	slope	*n*	yrs	median	*p* value	slope
R7	*GF4*	*133*	14	25.28	0.32	0.14	*63*	9	23.76	0.25	-0.26	*70*	5	26.65	0.22	-0.61
R6	*Fladen*	*109*	15	25.24	0.84	0.04	*50*	10	23.92	0.28	-0.35	*59*	5	26.36	0.09	-1.56
R3	*Anholt E*	*127*	12	23.68	0.19	0.39	*51*	7	21.92	0.76	-0.30	*76*	5	24.87	0.81	-0.41
R1	*Gniben*	*52*	14	21.46	0.06	0.22	*23*	9	24.03	0.92	-0.02	*29*	5	22.95	1.00	-0.25
Q2	*W Landskrona*	*54*	11	10.88	1.00	0.02	*25*	7	10.46	0.37	-0.25	*29*	4	11.24	-	-
N1	*Fehmarn Belt*	*55*	12	11.98	0.54	0.03	*38*	8	11.57	0.90	0.03	*17*	4	12.88	-	-
M2	*Mecklenburg Bight*	*64*	14	11.88	0.83	-0.04	*35*	9	11.86	0.35	-0.43	*29*	5	11.91	0.46	-0.89

Table 4.5.1 Results of trend analyses on 'winter'(Jan/Feb)-nutrient concentrations (in µmol dm^{-3}) and salinities (psu) in 'surface' water (0-10 m or down to the halocline) of the Kattegat, Sound and Belt Sea

n - total number of observations
yrs - number of years during the period in which measurements have been carried out
median value
p - significance level
slope in µmol dm^{-3} yr^{-1}
significant changes with $p \leq 0.05$ are marked in bold type

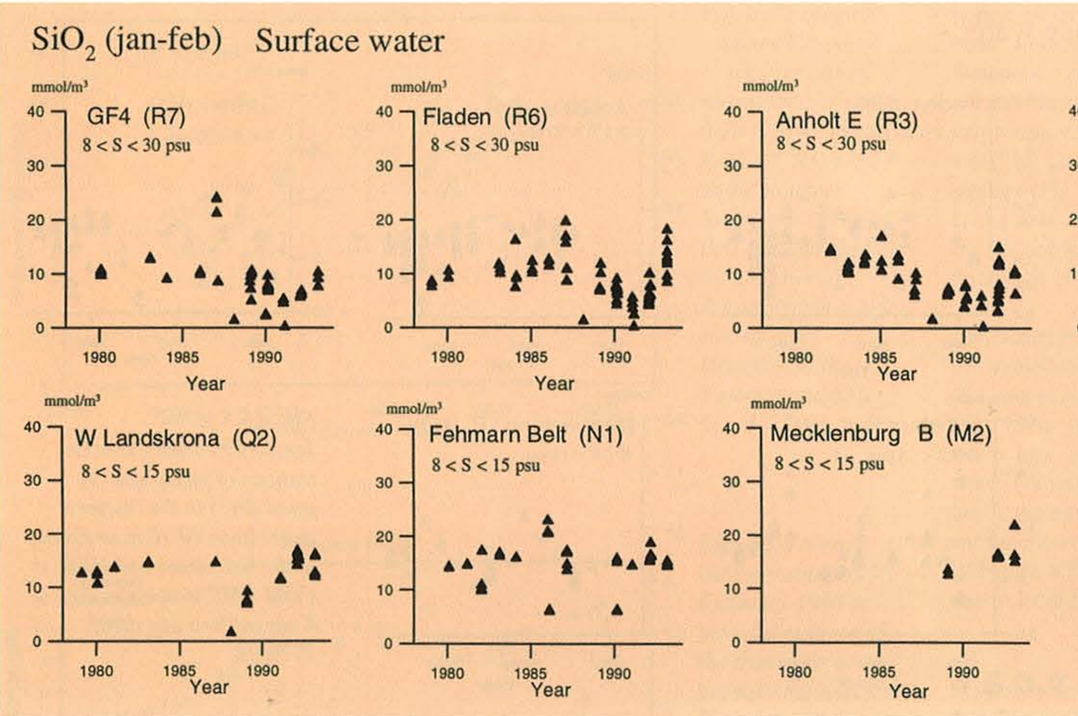


Fig. 4.5.9 Mean January/February concentrations of silicate (in µmol dm⁻³) in the 'surface'-water layer (0-10 m or down to the halocline) at 7 HELCOM BMP stations in the Kattegat/Belt Sea area, 1979-93

decrease. The decrease is rather fast, and in April, the nitrate+nitrite concentrations are close to the detection limit. The other nutrients normally have concentrations slightly above the detection limit. The concentrations remain low during the whole summer and early autumn. In September/October, the concentrations rise again and in December, the levels are about 50 to 70 % of the January/February values. In the Belt Sea area and in the Sound, silicate shows a slightly different pattern. After reaching its minimum in April, the concentrations start to rise, and continue to do so rather linearly until the winter concentration levels are reached. Even if the pattern is the same each year, the absolute levels might differ from year to year.

The main interest is focused on the conditions in the surface water during winter, when the concentrations are highest. Since the spring bloom normally occurs in March or late February, and because the levels in December are still far from maximum, the winter season is restricted to January and February. Analyses of the seasonal variation of nutrient concentrations in the surface waters at six stations in the Kattegat and Belt Sea area, and at two stations in the Baltic Proper, showed that the concentrations generally continued to increase during the winter until the onset of the spring bloom, and that no constant winter level was established [*202*]. However, in the transition area, the concentrations often do not differ much in January and February, as the concentrations depend on both remineralisation and on the varying load and dynamic water exchange. Therefore, the mean of January/February concentrations are used as 'winter values'. The winter values 1979-93 for the different nutrients have been plotted for each station (Figs. 4.5.7 - 4.5.9).

Using common methods [*251,242*], a trend test and slope estimate has been made. The analyses have been carried out for all four seasons, but only the winter results will be discussed here. The analyses have been done for the periods 1979-93, 1979-88 and 1989-93. The results are shown in Table 4.5.1. Very few of the detected changes are significant. Over the whole period 1979-93, only silicate at station *R3* (Anholt E) showed a significant decrease. For the period 1979-88, nitrate+nitrite increased significantly at station *R6* (Fladen) in the northern Kattegat, while significantly decreasing at station *M2* (Mecklenburg Bight). During 1989-93, no changes were significant. It must, however, be stressed that the data was too sparse at several stations. Over the assessment period, only five stations out of seven have been visited often enough during winter months to be considered for trend analyses. It is also to be noted that in the north-eastern Kattegat (*R3, R6, R7*), there was a significant decrease in silicate during spring and/or summer over 1979-93. These were the only significant changes detected for other parts of the year.

In another work concerning nutrient trends in this area [*22*], the results indicate that there was a significant increase during winter for phosphate and nitrate+nitrite, and a decrease during summer for silicate. In that work, however, all available data in the area have been included, also stations in the shallow coastal area, and two ten-year periods, 1971-80 and 1981-90, were compared. For recent years, a significant decrease in phosphorus concentrations in surface waters of Danish coastal areas and a decreasing tendency in the Belt Sea have been shown [*127*].

Station	Year	Slope 1970-95	Slope 1979-93	*P* 1970-95	*P* 1979-93
R7	1970-95	-0.19	0.03	0.039*	0.86
R3	1974-79 &				
	1982-95	-0.25	0.02	0.30	1.0
R1	1972-79 &				
	1981-95	-0.65	-0.04	0.090	0.85
Q2	1970-95	-0.51	-0.35	0.016*	0.41
P1	1974-95	-0.66	-0.13	0.015 *	0.87
N1	1979-95	-0.52	-1.47	0.40	0.039*
M2	1979-80 &				
	1982-94	0.69 (-1994)	0.09	0.45 (-1994)	0.96
Boknis Eck	1979-83 &				
	1986-90 &				
	1992-93		1.19		0.17

Table 4.5.2 Results of trend analyses for 1979-93 and 1970-95 on bottom water oxygen saturation from July to October in the Kattegat, Sound and Belt Sea

(Years are indicated for which data were available; *p* - significance level; slope in % saturation yr⁻¹; significant changes with $p \leq 0.05$ are marked with an asterix)

The Kattegat and the Belt Sea area have short residence times, one to two months, for surface water, and there are also strong horizontal gradients which make it difficult to detect changes in nutrient concentrations. There is also a problem with the low sampling frequency during winter months. The BMP programme is not capable of detecting changes over such short periods as five years, in such variable areas as the Kattegat and the Belt Sea. However, the aforementioned results concerning nitrogen and phosphorus are consistent with the development in load, which increased from the 1970s to the 1980s, and did not change very much during the 1980s. The assessment period 1989-93 is too short to detect the effects of the lower load, which is due to lower run-off and phosphorus removal from sewage and detergents. The reason for the decreasing silicate concentrations may be increased removal of silicate from the water by increased diatom growth and settlement due to increased 'eutrophication (nutrient load). Decreases in silicate have been observed not only in the Kattegat but also in the Gulf of Bothnia and Baltic Proper [*592*], in the southern North Sea and in other coastal areas of the world ocean [*121*]. The decrease in silicate most often reduces the DSi/DIN ratio, which might lead to silicate limitation of diatom growth, and thereby to changes in both species composition and food web dynamics [*121,558*].

4.5.3.3 Oxygen variability

G. Aertebjerg[1]

The bottom water oxygen concentration in the Kattegat and Belt Sea area is high during winter, with >80 % saturation, but starts to decrease after the spring bloom in March. Water exchange and mixing due to strong wind may temporarily stop the decrease during spring and summer, but a minimum is reached within the period July-October. The actual time of the oxygen minimum occurrence depends on the area and wind conditions. Re-oxygenation takes place within the period September-November, usually first in the Fehmarn Belt and latest in the south-eastern Kattegat, central Sound and southern Little Belt.

The mean oxygen concentrations in the bottom layer during the months of July to October have been analysed at eight stations from the Kattegat (*R1, R3, R7*), the Sound (*Q2*), the Great Belt (*P1*), the Fehmarn Belt (*N1*), the Kiel Bight (*Boknis Eck*) and the Mecklenburg Bight (*M2*). Trend tests and slope estimates have been made using common methods [*251,252*]. Two periods have been analysed, 1979-93 and long-term series up to 1995, if available. The results are shown in Table 4.5.2.

In the period 1979-93, only the Fehmarn Belt (*N1*) showed a significant change, i.e., a decrease of about 1.5 % saturation per year. However, the significance disappeared if 1994-95 data were included, showing that the analyses are very sensitive to the level at the end of the time series. Additional data from the southern Belt Sea were not available for the analyses, but inclusion of data from 1970 to 1995 at the Kattegat, Sound and Great Belt stations revealed a significant decreasing oxygen concentration in the Sound (*Q2*), Great Belt (*P1*) and north-eastern Kattegat (*K7*), and a decreasing tendency at the other Kattegat stations.

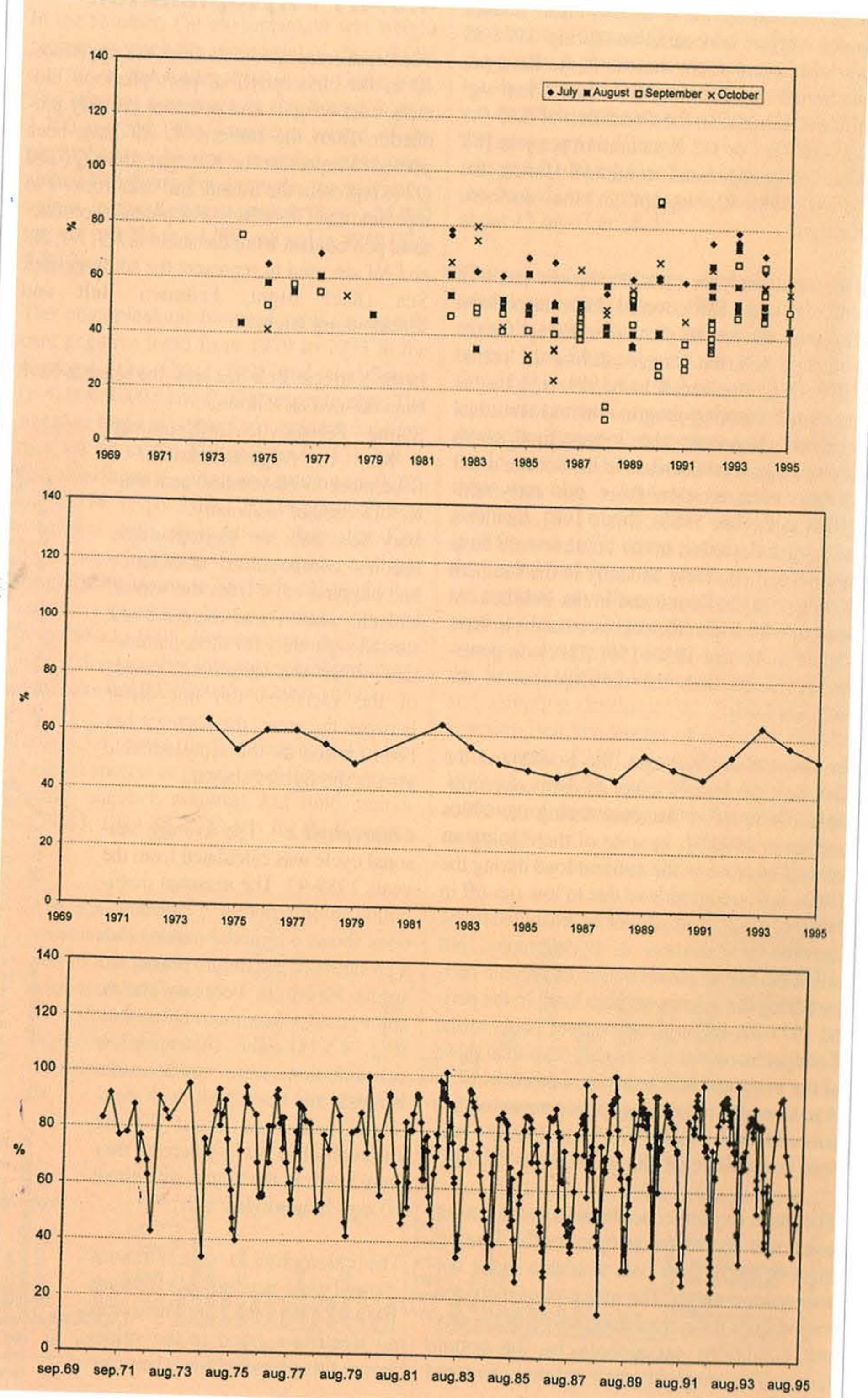


Fig. 4.5.10 Time series of oxygen saturation in bottom water at station R3 in the Kattegat

upper part - July-to-October data used in the trend analysis; mean saturation in the bottom layer >40 %,
middle part - July-to-October data, transformed to one value per year,
lower part - oxygen as recorded closest to the bottom per sampling series

[1] see ANNEX 11.3 for addresses of authors

In earlier investigations [50], a significant decrease of the mean sub-pycnocline oxygen concentration by 0.10-0.11 $cm^3\ dm^{-3}\ yr^{-1}$ in July-September at the station *Boknis Eck* in Kiel Bight was found for the period 1957-86. In Kiel Bight and Fehmarn Belt, increasing oxygen consumption trends close to the bottom of 0.15 $cm^3\ dm^{-3}\ yr^{-1}$ and 0.13 $cm^3\ dm^{-3}\ yr^{-1}$ (non-significant), respectively, were also estimated for the period 1976-90 [5,692]. From 1971 to the late 1980s, a significant decreasing trend of 0.05-0.1 $cm^3\ dm^{-3}\ yr^{-1}$ was estimated for the Kattegat deep water (>30 psu) during August-October [23,24]. A significant decreasing trend in the mean bottom water oxygen concentration during 1975-88 was also found at six stations in the Kattegat, the Sound and Fehmarn Belt, and a non-significant decrease in the Great Belt of 0.05-0.1 $cm^3\ dm^{-3}\ yr^{-1}$ or 1-2 % saturation per year [5]. The decreasing trend continued during the period 1989-92, except in the northern Kattegat.

The trend analyses mentioned were made in different ways and covered different periods. However, the results show that, except for the Fehmarn Belt, the changes during the period 1979-93 are too small to be detected by the present monitoring programme and statistical methods. However, the longer time series showed a general decrease in the bottom water oxygen concentration from the early/mid-1970s to the late 1980s. Since 1981, incidents of oxygen depletion in the bottom water have been observed nearly annually in the southern Kattegat, in the Sound and in the Belt Sea. At least in Kiel Bight, the decrease seems to have started in the late 1950s [50]. This is in general agreement with the development of the nitrogen load.

In the southern Kattegat, the Sound and the Belt Sea, the bottom water oxygen concentrations continued to decrease during the 1980s and up to 1990/91, in spite of there being no general increase in the nutrient load during the 1980s and a reduced load due to low run-off in 1989-92 [692]. This cannot be attributed to the interannual variation in hydrography, but might be due to accumulation of organic matter during the years with high load. In the period 1971-82, the organic nutrient pools in the Kattegat increased with higher rates than those of the inorganic winter pools of nutrients [23]. A tendency to increasing bottom water oxygen concentrations from 1990/91 to 1993/94 was seen in the Kattegat and in the Sound [127].

The mean oxygen concentration in the bottom water was chosen for analyses of the development of the oxygen pool available in the bottom water column. The mean oxygen concentration does not give a true picture of the oxygen conditions experienced by the bottom fauna. The single oxygen values, measured closest to the bottom during visits to the stations, is a better measure, even if they are measured 0.5-1.5 m above the sediment (Fig. 4.5.10).

4.5.4 Pelagic biology

J. Albjerg[1], G. Behrends, H. Giesenhagen, H.-P. Hansen, J. Wikner

4.5.4.1 Phytoplankton

The depth-integrated data from the uppermost 10 m for chlorophyll *a*, phytoplankton biomass (wet weight) and potential primary production from the stations *R1-R7* have been pooled to represent the Kattegat, from *Q1* and *Q2* to represent the Sound, and data from *P1* to represent the Great Belt. For potential production, pooled data from the stations *M1, N1, N3* and *N4* are used to represent the southern Belt Sea (Kiel Bight, Fehmarn Belt and Mecklenburg Bight).

In the Kattegat/Belt Sea area, the seasons have been defined as follows:
Spring - February to April, summer - May to August, autumn - September to November, and winter - December to January.
Very few data on phytoplankton species composition, abundance and biomass exist from the winter, and this season was therefore not treated separately for these parameters. When the variations/changes of the variables did not differ between the areas, the Kattegat has been chosen as the representative area in the figures shown.

Chlorophyll a - The average seasonal cycle was calculated from the years 1989-93. The seasonal distribution in chlorophyll *a* concentrations shows a bimodal pattern with a pronounced maximum during the spring bloom in February/March, and a smaller maximum in autumn (Fig. 4.5.11). The chlorophyll *a* concentrations were similar in the Kattegat and Great Belt, with maximum values of about 20 mg chlorophyll *a* m^{-3}, whereas they were lower in the Sound with about 10 mg chlorophyll *a* m^{-3}.

The chlorophyll *a* concentrations showed great interannual variations 1979-93 (Fig. 4.5.12). There was no significant trend in the values during this period, either when considering each season separately or when considering data from all seasons of each year together. Regarding the assessment periods 1979-83, 1984-88 and 1988-93, there may be a tendency to lower chlorophyll *a* concentrations in spring during the period 1988-93 compared to the earlier ones (Table 4.5.3). However, with only one or at the most a few samplings a month, the probability of sampling during the actual spring peak is low, and the measured concentration may thus not represent the actual peak. A tendency to higher concentrations in summer and autumn during the mid 1980s is seen.

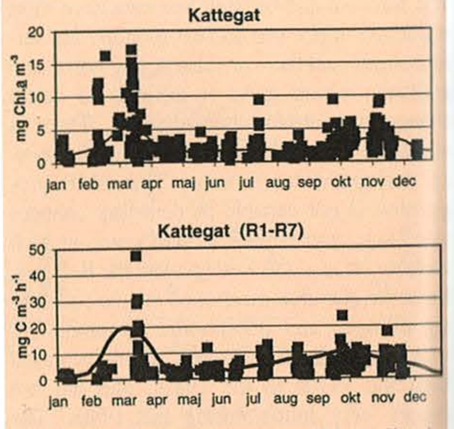
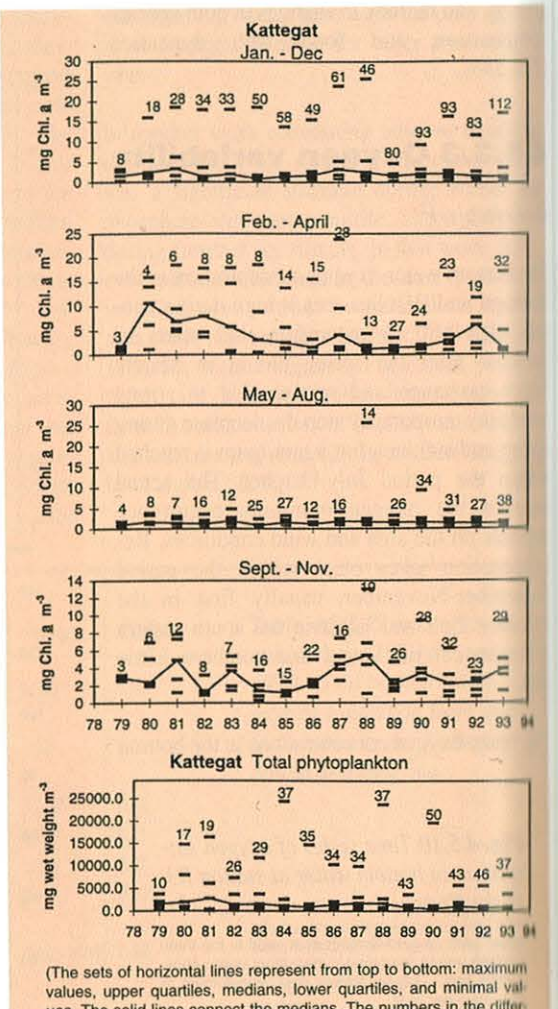


Fig. 4.5.11 Mean seasonal distribution of 0-10 m depth-integrated chlorophyll a concentrations and potential primary production values in the Kattegat, 1989-93

Fig. 4.5.12 Time series of mean 0-10 m depth-integrated chlorophyll a concentrations for different seasons, and time series of mean annual phytoplankton wet weight biomasses in the Kattegat, 1979-93



(The sets of horizontal lines represent from top to bottom: maximum values, upper quartiles, medians, lower quartiles, and minimal values. The solid lines connect the medians. The numbers in the different figures indicate the number of observations during the corresponding years.)

Phytoplankton biomass and species composition - When the phytoplankton biomass is expressed as wet weight, the importance of diatoms is exaggerated due to the large vacuoles in the diatom cells, contributing to the wet weight but not to the carbon biomass. Data from this region consisted of both wet weight and carbon biomass. In order to compare the phytoplankton levels with other areas in the Baltic Sea, wet weight has been used in all figures. Trend analyses were, however, performed on carbon biomass.

The seasonal distribution of the phytoplankton-carbon biomass is characterised by a spring bloom dominated by diatoms, a summer maximum with diatoms and dinoflagellates, contributing equally, and an autumn bloom, mostly consisting of dinoflagellates. Regarding wet weight, the diatoms also dominate the phytoplankton in the autumn (Fig. 4.5.13).

In the summer, the phytoplankon wet weight biomasses reach the same level of about 3,000 mg m^{-3} in the Kattegat as in the Great Belt. The summer maximum in the Sound is only 1,700 mg m^{-3}. The long-term record of the phytoplankton biomass revealed great interannual variations. For example, the maximum values in the Kattegat ranged from >24,000 mg m^{-3} in 1984 to <4,000 mg m^{-3} in 1989 (Fig. 4.5.12).

The phytoplankton biomass shows a significant negative trend from 1979 to 1993 in the Kattegat, the Great Belt and the Sound (p=0.008, 0.007 and 0.004, respectively). The negative trend is particularly evident for summer values in the Kattegat (p=0.013), but is also significant in spring and autumn (p=0.025 and 0.054, respectively; Fig. 4.5.14). This is consistent with observations from the Kiel Bight and Fehmarn Belt, where a slight insignificant decrease in both chlorophyll *a* and phytoplankton biomass was observed from 1986 to 1995. However, the decline was mainly due to rather low phytoplankton standing stocks in 1993, 1994 and 1995 [257].

Table 4.5.4 shows the five dominating phytoplankton species or groups in different seasons (spring, summer, autumn) and time periods (1979-83, 1984-88, 1989-93) for the Kattegat, the Sound and the Belt Sea. Dominance due to one or a few mass occurrences has been disregarded. In the Kattegat, this applies to incidents of *Gomphosphaeria* sp. and *Noctiluca scintillans* in the periods 1979-83 and 1989-93, respectively. In the Sound, *Coscinodiscus* sp. have been disregarded in the autumn plankton for the period 1984-88.

Table 4.5.3 Chlorophyll a concentrations (mg m^{-3}) and potential primary production (mg C m^{-3} h^{-1}) in spring, summer and autumn during the three assessment periods

Chlorophyll *a*

	Spring	Summer	Autumn
1979-83	6.1 ± 5.4 (0.3-18.5) *n=75*	1.9 ± 1.3 (0.2-10.6) *n=119*	3.2 ± 1.9 (0.2-9.2) *n=96*
1984-88	4.0 ± 4.8 (0.3-23.8) *n=182*	2.2 ± 2.5 (0.2-25.7) *n=223*	3.4 ± 2.3 (0.3-13.0) *n=207*
1989-93	3.0 ± 3.5 (0.0-18.3) *n=305*	2.1 ± 1.3 (0.0-11.8) *n=405*	3.1 ± 1.8 (0.0-11.5) *n=354*

Potential productivity

	Spring	Summer	Autumn
1979-83	16.8 ± 18.6 (0.6-84.6) *n=54*	8.0 ± 6.3 (0.6-52.4) *n=113*	8.5 ± 6.1 (0.9-28.5) *n=84*
1984-88	10.8 ± 13.1 (0.3-62.9) *n=123*	8.7 ± 9.4 (0.2-58.3) *n=166*	12.0 ± 8.6 (1.2-45.8) *n=147*
1989-93	7.7 ± 10.2 (0.2-51.3) *n=121*	5.8 ± 4.6 (0.8-32.9) *n=198*	8.9 ± 5.6 (0.9-34.4) *n=167*

(Data have been pooled from the Kattegat, Sound, Great Belt and southern Belt Sea; mean ± standard deviation, (range), *number of samples*)

Potential productivity - The average seasonal cycle of the potential production was calculated from the years 1989-93. As for chlorophyll *a*, the seasonal distribution of the potential productivity shows a bimodal pattern, with a pronounced maximum in March and a smaller maximum in autumn (Fig. 4.5.11). The level of the potential productivity was almost the same for the Kattegat, the Great Belt and the southern Belt Sea, with about 40-50 mg C m^{-3} h^{-1} in the spring, up to 10 mg C m^{-3} h^{-1} in the summer and 20-30 mg C m^{-3} h^{-1} in the autumn. In the Sound, the spring peak was lower with about 13 mg C m^{-3} h^{-1}.

The potential productivity showed great interannual variations from 1979 to 1993. For example, in the Kattegat, the maximum values ranged from 84 mg C m^{-3} h^{-1} in 1980 to 12 mg C m^{-3} h^{-1} in 1991. In the Kattegat, there was a significant negative trend in the potential productivity during this period (p=0.05; Fig. 4.5.15). In the Great Belt and in the Sound, there was also a tendency to lower values, although the trends were not significant (p=0.06 and 0.07, respectively). Analyses of the pooled data from the southern Belt Sea did not indicate any change in the potential productivity from 1979 to 1993. However, by separately analysing the potential production from the period 1985/86-95 for different stations and sampling depths in the Kiel Bight and Fehmarn Belt, tendencies of decreasing trends were visible [257]. Regarding the periods 1979-83, 1984-88 and 1989-93, there was a tendency of decreasing potential productivity in spring.

Fig. 4.5.13 Mean seasonal distribution of wet-weight biomasses (mg m^{-3}) for different groups of phytoplankton in the Kattegat, 1989-93

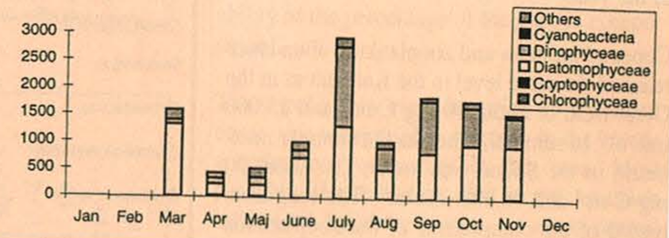
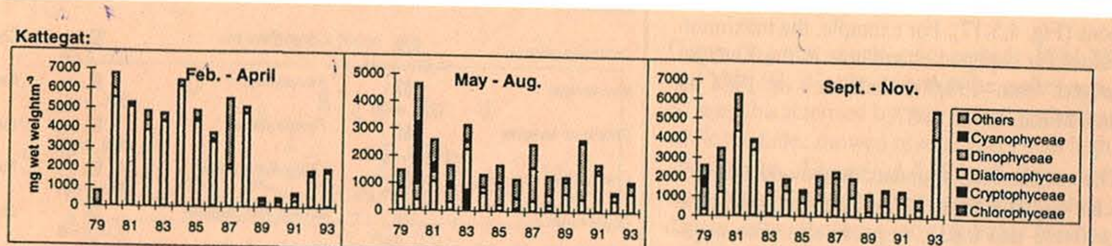


Fig. 4.5.14 Long-term variability of wet-weight biomass for different groups of phytoplankton and seasons in the Kattegat, 1979-93



[1] see ANNEX 11.3 for addresses of authors

4.5.4.2 Zooplankton

Depth-integrated zooplankton samples were collected until 1986 by a modified *WP-2* net with 100 µm mesh size [*670*] according to the HELCOM BMP Guidelines. After 1986, the Danish zooplankton samples were taken by a submersible pump (400 dm^3 min.$^{-1}$) fitted with a 100 µm net [*464*]. Sweden and Germany used the *WP-2* net throughout the whole period. Although the *WP-2* net may underestimate the zooplankton due to clogging of the net, this is not the case with the pump [*464*]. However, the appendicularian *Oikopleura dioica* is probably damaged by the pump [*464*], and thus underestimated in the Danish samples since 1986.

Kattegat, Sound and Great Belt **-** The zooplankton data from the stations *R1* to *R4* have been pooled to represent the Kattegat, *Q1* and *Q2* to represent the Sound and data from *P1* to represent the Great Belt. The seasons have been defined as follows:
Spring - February to March, summer - May to August, and autumn - September to November.
There was no zooplankton sampling in the winter from this area. Data exist from 1982 to 1993 for the Kattegat, and from 1980 to 1993 for the Sound and the Great Belt. The southern Belt Sea was treated separately, as the seasonal cycle seems different and the data material are more extensive.

The average year, constructed from all data for the period 1989-93, shows an unimodal seasonal distribution, peaking in June. Copepods dominate the zooplankton throughout the season. Their seasonal variation is more pronounced in terms of biomass than in terms of abundance (Fig. 4.5.16). The most important copepods in terms of biomass are *Pseudocalanus* spp. and *Centropages* spp. *Acartia* spp. also contribute significantly to the copepod biomass in the early season, thereafter *Oithona* spp. take over in the second half of the year.

Copepod biomass and zooplankton abundance reached the same level in the Kattegat as in the Great Belt, of about 50 mg C m^{-3} and 25,000 ind. m^{-3} in summer, whereas the summer maximum in the Sound was lower, i.e., about 30 mg C m^{-3} and 16,000 ind. m^{-3}. The long-term record of the components of the zooplankton community revealed great interannual variations (Fig. 4.5.17). For example, the maximum values for copepod abundance in the Kattegat ranged from 138,000 ind. m^{-3} in 1984 to 26,000 ind. m^{-3} in 1987.

The zooplankton abundances showed a significant negative trend in the Kattegat and Sound (p=0.033 and 0.011, respectively). The negative trend was most pronounced in the Sound

a) Kattegat

1979-83		1984-88		1989-93	
Spring	N=24, S=6		N=61, S=8		N=47, S=8
Skeletonema costatum	766 (4,658, *n*=23)	*Porosira glacialis*	880 (13,356, *n*=20)	*Thalassiosira* sp.	239 (3,624, *n*=17)
Thalassiosira sp.	716 (9,884, *n*=14)	*Detonula confervacea*	826 (9,237, *n*=21)	*Skeletonema costatum*	140 (1,289, *n*=24)
Detonula confervacea	678 (7,501, *n*=8)	*Thalassiosira polychorda*	160 (6,206, *n*=18)	*Coscinodiscus concinnus*	85 (1,236, *n*=6)
Porosira glacialis	586 (2,382, *n*=13)	*Skeletonema costatum*	143 (1,692, *n*=30)	*Rhizosolenia setigera*	39 (288, *n*=21)
Thalassiosira gravida?	559 (5,549, *n*=5)	*Chaetoceros wighamii*	132 (4,021, *n*=12)	*Detonula confervacea*	37 (1,074, *n*=11)
Summer	N=57, S=6		N=113, S=9		N=107, S=8
Nanoplankton	362 (1,051, *n*=33)	*Peridinium* sp.	124 (13,376, *n*=7)	*Rhizosolenia fragilissima*	280 (3,716, *n*=75)
Distephanus speculum	314 (8,301, *n*=14)	Nanoplankton	106 (432, *n*=45)	*Guinardia flaccida*	194 (5,153, *n*=54)
Rhizosolenia alata	132 (2,333, *n*=21)	*Chrysochromulina polylepis*	91 (4,310, *n*=16)	*Ceratium tripos*	121 (1,054, *n*=91)
Rhizosolenia fragilissima	117 (2,592, *n*=28)	*Ceratium tripos*	81 (1,818, *n*=39)	Flagellates	61 (258, *n*=74)
Ceratium tripos	104 (1,373, *n*=22)	*Rhizosolenia fragilissima*	60 (1,356, *n*=39)	*Rhizosolenia alata*	61 (1,152, *n*=45)
Autumn	N=36, S=6		N=76, S=8		N=64, S=7
Cerataulina pelagica	950 (12,979, *n*=17)	*Cerataulina pelagica*	150 (1,105, *n*=57)	*Guinardia flaccida*	463 (6,354, *n*=36)
Nanoplankton	244 (2,588, *n*=15)	*Guinardia flaccida*	110 (1,902, *n*=60)	*Ceratium tripos*	210 (1,137, *n*=60)
Gyrodinium aureolum	114 (1,225, *n*=10)	*Ceratium lineatum*	92 (1,170, *n*=46)	*Ceratium lineatum*	134 (1,173, *n*=59)
Cryptomonas sp.	112 (2,573, *n*=14)	Ultra-/ Nanoplankton	83 (3,504, *n*=50)	*Cerataulina pelagica*	86 (659, *n*=47)
Ceratium tripos	107 (766, *n*=24)	*Ceratium tripos*	77 (545, *n*=53)	*Thalassiosira eccentrica*	83 (656, *n*=17)

b) Sound

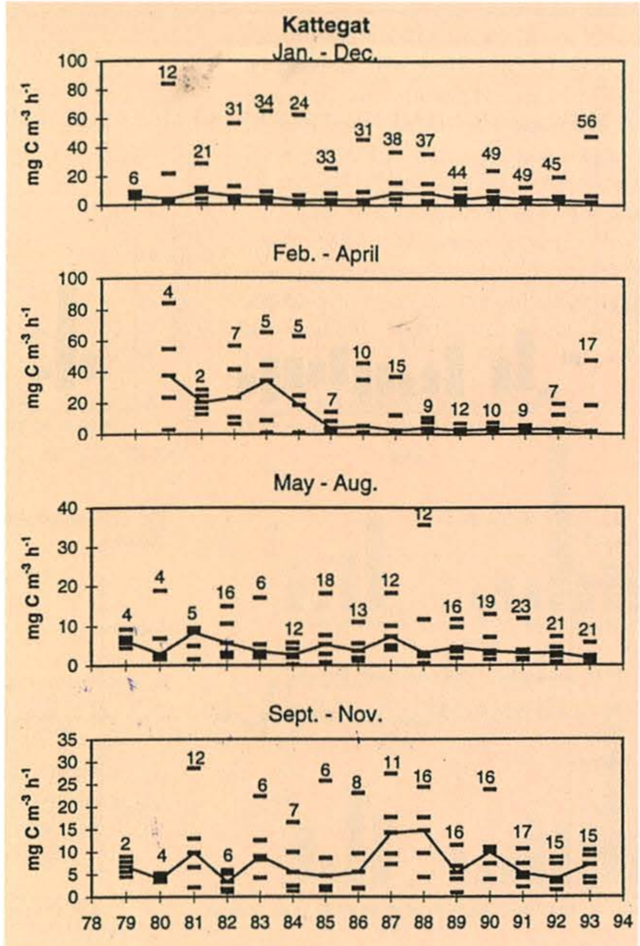
1979-83		1984-88		1989-93	
Spring	N=5, S=1		N=4, S=1		N=8, S=1
Detonula confervacea	286 (1,329, *n*=5)	*Thalassiosira levanderi*	255 (1,012, *n*=2)	*Thalassiosira* sp.	122 (556, *n*=3)
Nanoplankton	284 (355, *n*=5)	*Chaetoceros wighamii*	227 (811, *n*=4)	*Skeletonema costatum*	83 (505, *n*=4)
Achnanthes taeniata	146 (440, *n*=2)	*Chaetoceros holsaticus*	121 (350, *n*=3)	*Thalassiosira levanderi*	32 (252, *n*=2)
Chaetoceros debilis	112 (421, *n*=5)	Nanoplankton	94 (?, *n*=4)	*Eutreptiella* sp.	31 (221, *n*=3)
Skeletonema costatum	42 (135, *n*=5)	*Achnanthes taeniata*	55 (188, *n*=3)	*Detonula confervacea*	28 (213, *n*=4)
Summer	N=18, S=2		N=19, S=1		N=23, S=1
Gomphosphaeria sp.	1,366 (21,454, *n*=5)	Nanoplankton	352 (1,808, *n*=12)	*Rhizosolenia fragilissima*	263 (2,370, *n*=17)
Nanoplankton	746 (1,443, *n*=18)	*Rhizosolenia fragilissima*	137 (2,169, *n*=6)	Flagellates	99 (252, *n*=18)
Aphanothece sp.	248 (2,974, *n*=5)	*Ceratium tripos*	104 (1,436, *n*=8)	*Prorocentrum minimum*	83 (675, *n*=12)
Rhizosolenia fragilissima	177 (1,599, *n*=12)	*Cryptomonas* sp.	100 (259, *n*=13)	*Guinardia flaccida*	66 (307, *n*=11)
Cryptomonas sp.	137 (419, *n*=13)	*Ceratium furca*	46 (841, *n*=4)	*Ceratium tripos*	43 (162, *n*=14)
Autumn	N=8, S=1		N=10, S=2		N=12, S=1
Cerataulina pelagica	605 (4,813, *n*=4)	*Cryptomonas* sp.	91 (338, *n*=7)	*Cryptomonas* sp.	53 (158, *n*=6)
Nanoplankton	287 (324, *n*=8)	Nanoplankton	70 (256, *n*=5)	*Cerataulina pelagica*	47 (461, *n*=6)
Gyrodinium aureolum	189 (1,495, *n*=3)	*Porosira glacialis*	62 (486, *n*=2)	*Guinardia flaccida*	35 (226, *n*=4)
Coscinodiscus sp.	164 (1,020, *n*=2)	*Cerataulina pelagica*	60 (281, *n*=5)	Flagellates	34 (51, *n*=9)
Cryptomonas sp.	128 (227, *n*=6)	*Skeletonema costatum*	26 (234, *n*=4)	*Skeletonema costatum*	28 (152, *n*=8)

c) Belt Sea

1979-83		1984-88		1989-93	
Spring	N=41, S=9		N=71, S=12		N=58, S=10
Detonula confervacea	1287 (7,318, *n*=30)	*Thalassiosira decipiens*	764 (10,751, *n*=23)	*Chaetoceros teres*	3,543 (96,136, *n*=5)
Thalassiosira decipiens	423 (3,495, *n*=14)	*Skeletonema costatum*	532 (16,296, *n*=56)	*Skeletonema costatum*	355 (3,061, *n*=53)
Chaetoceros holsaticus	280 (11,016, *n*=15)	*Detonula confervacea*	191 (2,653, *n*=27)	*Thalassiosira baltica*	130 (1,240, *n*=12)
Melasira arctica	251 (10,230, *n*=6)	*Thalassiosira levanderi*	163 (3,777, *n*=13)	*Chaetoceros* sp.	98 (1,103, *n*=23)
Skeletonema costatum	232 (1,850, *n*=36)	*Porosira glacialis*	135 (5,770, *n*=7)	Flagellates	94 (884, *n*=40)
Summer	N=84, S=8		N=169, S=12		N=162, S=11
Rhodomonas minuta	5,235 (403,200, *n*=28)	Nanoplankton	671 (104,116, *n*=28)	*Rhizosolenia fragilissima*	470 (6,772, *n*=113)
Skeletonema costatum	2,779 (215,600, *n*=50)	*Ceratium tripos*	453 (16,949, *n*=94)	*Guinardia flaccida*	275 (3,804, *n*=69)
Gymnodinium simplex	1,486 (100,800, *n*=6)	*Rhizosolenia fragilissima*	189 (4,943, *n*=69)	*Ceratium tripos*	234 (1,956, *n*=121)
Rhizosolenia fragilissima	957 (45,081, *n*=30)	*Prorocentrum micans*	102 (1,004, *n*=81)	*Prorocentrum minimum*	111 (16,197, *n*=42)
Aphanizomenon flos-aquae	553 (45,840, *n*=17)	*Ceratium fusus*	56 (2,758, *n*=85)	Flagellates	109 (663, *n*=128)
Autumn	N=44, S=10		N=70, S=8		N=82, S=9
Cerataulina pelagica	431 (17,673, *n*=8)	*Ceratium tripos*	351 (4,855, *n*=53)	*Ceratium tripos*	490 (4,595, *n*=75)
Ceratium tripos	314 (1,682, *n*=33)	*Ceratium furca*	201 (6,922, *n*=29)	*Guinardia flaccida*	287 (8,248, *n*=49)
Ceratium furca	105 (1,434, *n*=28)	*Ceratium lineatum*	172 (3,657, *n*=39)	*Cerataulina pelagica*	253 (6,482, *n*=49)
Nanoplankton	100 (915, *n*=8)	*Prorocentrum micans*	118 (2,303, *n*=41)	*Ditylum brightwellii*	100 (4,800, *n*=33)
Prorocentrum micans	98 (1,184, *n*=40)	*Guinardia flaccida*	78 (724, *n*=26)	*Ceratium fusus*	94 (1,105, *n*=69)

ble 4.5.4 Dominating ytoplankton species in Kattegat, Sound and lt Sea during the three essment periods

hmetical **mean** (maximum) val- of species wet-weight biomass g m^{-3}; N - total number of sam- , S - number of stations, *n* - num- of samples in which species dominating)



. 4.5.15 Time series of mean 0-10 m depth- grated potential pri- y production (in mg C h.$^{-1}$) for different sea- in the Kattegat, 9-93

sets of horizontal lines repre- from top to bottom: maximum , upper quartiles, medians, quartiles, and minimal values. olid lines connect the medians. umbers in the different figures te the number of observations the corresponding years.)

in autumn (Fig. 4.5.18), and was due to a decline in copepod abundance and biomass. In terms of biomass, the negative trend was significant in the Kattegat, the Sound and in the Great Belt (p=0.006, 0.005 and 0.009, respectively).

Southern Belt Sea (Kiel Bight, Fehmarn Belt and Mecklenburg Bight) **-** For the total zooplankton analyses, the following stations were pooled:
Kiel Bight - *Boknis Eck, N3* and *N4*; Fehmarn Belt - *N1*; Mecklenburg Bight - *M2* and *M1*.
The detailed evaluations on species level were restricted to Kiel Bight data from 1985-93.
The data were splitted as follows:
January to March - winter (low abundance), April to June - spring (maximum), July to September - summer (minimum), and October to December - autumn (peak followed by minimum).

In the Kiel Bight, the annual mean abundance of the mesozooplankton community varied between 12,470 ind. m^{-3} (1986) and 73,561 ind. m^{-3} (1989), and in the Mecklenburg Bight between 7,615 ind. m^{-3} (1982) and 38,606 ind. m^{-3} (1989). The highest intra-annual variability was observed at the coastal station *Boknis Eck* (Figs. 4.5.19a,b,c). The seasonality of the total mesozooplankton community is shown in Figure 4.5.20.

The mesozooplankton community of the Kiel and Mecklenburg Bights was clearly dominated by copepods (Fig. 4.5.21). Regarding the abundances, in certain months other groups can occur in high densities. These are often either rotifers (April/May) or meroplankters, especially bivalve larvae (June). Among the copepods, *Oithona similis* is the most abundant species in the area. In spring and summer, *Centropages hamatus* and *Acartia* spp. can be more important in some years. The second rank is taken by *Pseudo-* and *Paracalanus* spp. in winter and autumn. *Temora longicornis* is a less abundant copepod species in this area. However, there is a strong interannual variability of the percentage of the species composition.

The water inflow event in spring 1993 did not cause any recognisable reaction in the mesozooplankton community in this area. Most probably, the transition area is not as much influenced by these events as more central parts of the Baltic Sea.

Comparing the monitoring periods 1984-88 and 1989-93, it is obvious, that the earlier period was characterised by very low abundances, while the latter showed several years with high abundances. Figures 4.5.19d,e,f show the deviations of the total mesozooplankton abundances from the long-term monthly median values. The differences between mesozoo-

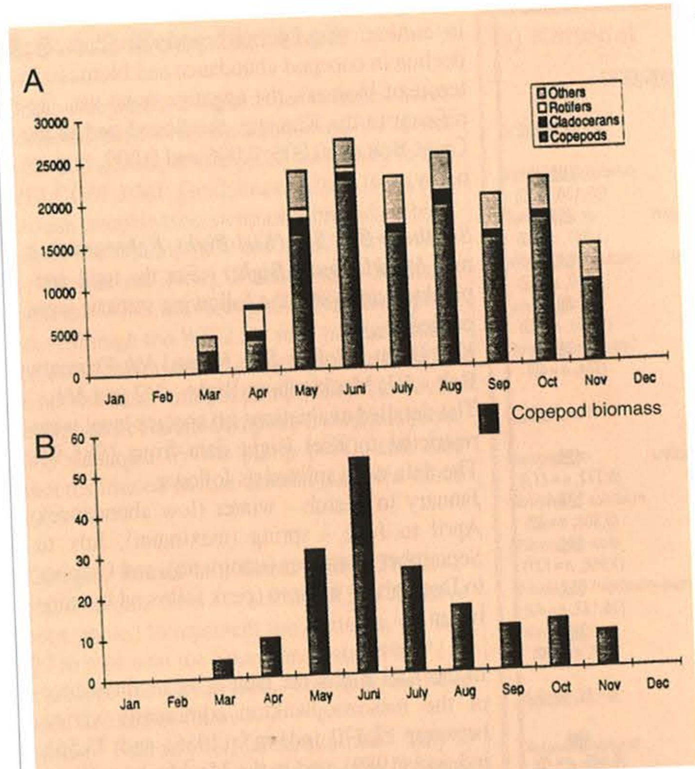


Fig. 4.5.16 Mean seasonal distribution of the total abundances of different zooplankton groups (ind. m⁻³; A), and copepod biomasses (mg C m⁻³; B) in the Kattegat, 1989-93

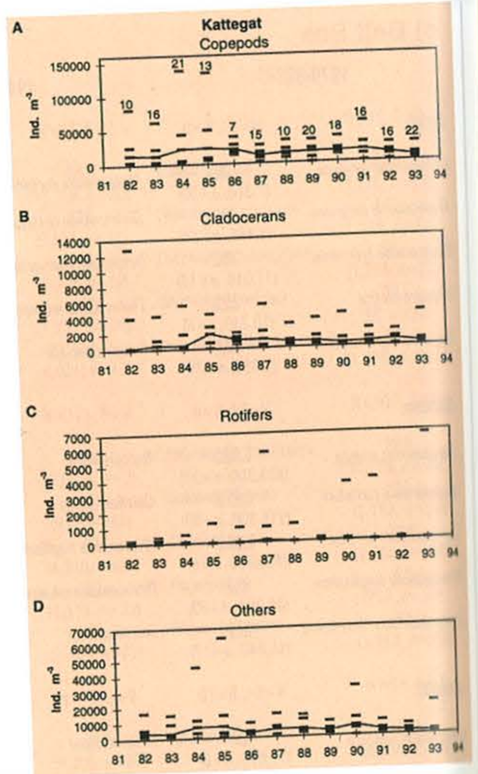


Fig. 4.5.17 Time series of zooplankton abundances (ind. m⁻³) in the Kattegat, 1982-93

(The sets of horizontal lines represent from top to bottom: maximum values, upper quartiles, medians, lower quartiles, and minimal values. The solid lines connect the medians. The numbers in the different figures indicate the number of observations during the corresponding years.)

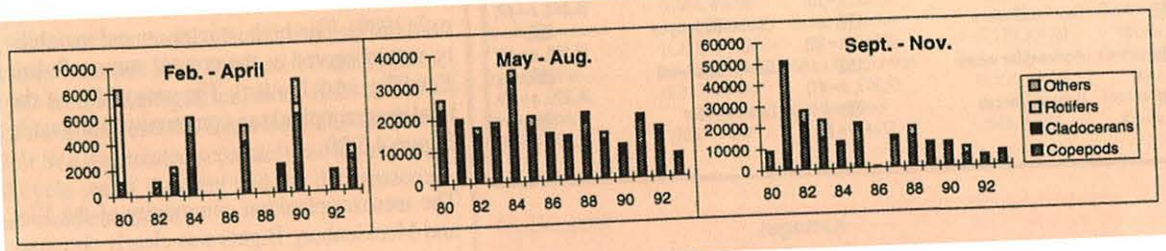
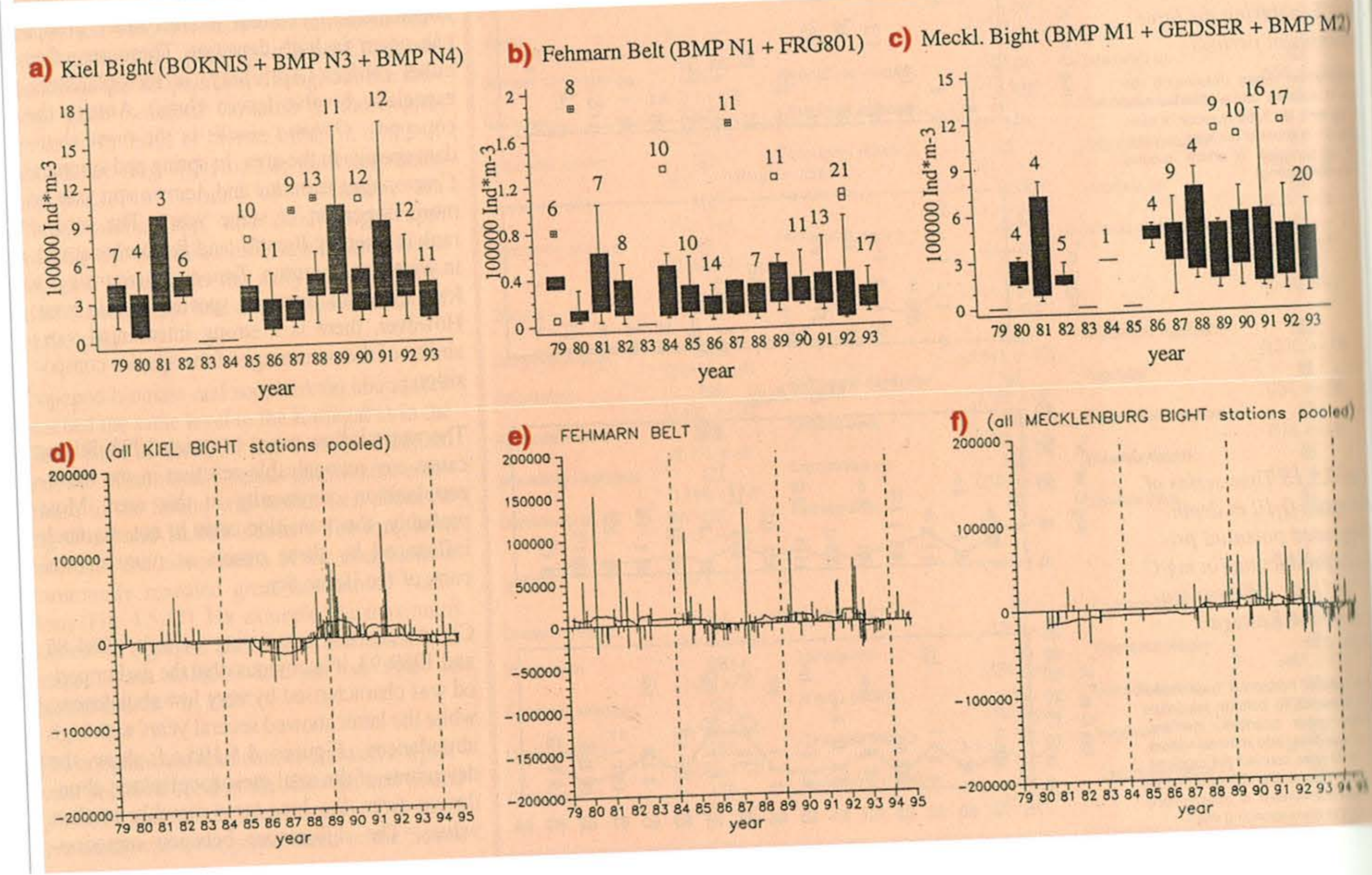


Fig. 4.5.18 Long-term variabilty of zooplankton abundances (ind. m⁻³) during different seasons in the Sound, 1980-93



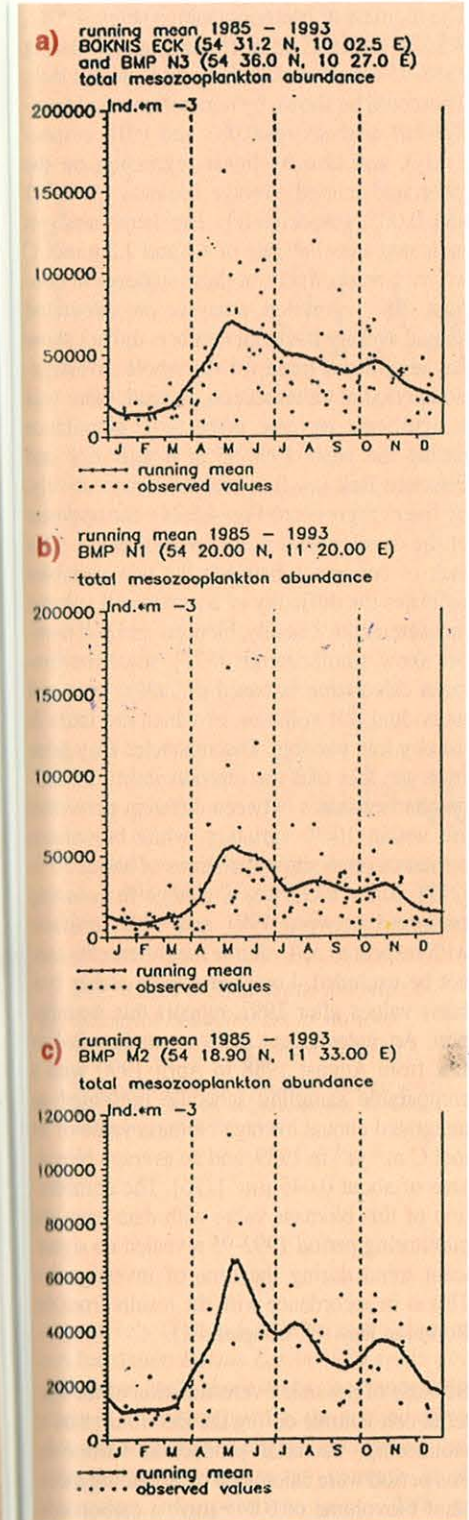


Fig. 4.5.20 Seasonal distribution of total mesozooplankton abundances (ind. m⁻³) in the Kiel Bight (a), Fehmarn Belt (b), and Mecklenburg Bight (c), 1985-93

(The vertical dashed lines separate the different seasons.)

Fig. 4.5.19 Total mesozooplankton abundances (10⁵, ind. m⁻³) in the Kiel Bight (a), Fehmarn Belt (b) and Mecklenburg Bight (c), and deviations from the long-term monthly median abundances in the Kiel Bight (d), Fehmarn Belt (e), and Mecklenburg Bight (f)

(The vertical dashed lines in *d*, *e* and *f* separate the different assessment periods.)

plankton-rich and -poor years are more pronounced at the Kiel Bight stations than in the Mecklenburg Bight, but such differences are not shown at the station Fehmarn Belt. This might be due to the fast currents at this station, which lead to an adjustment of the differences, while in the shallower and more stable coastal stations, differences can be more developed.

In spring and summer, there was an increase of mesozooplankton abundances from 1985/86 to 1989/91. Afterwards, the standing stocks decreased (Figs. 4.5.19, 4.5.20). This difference in abundances was caused mainly by the copepods *Oithona similis* and *Pseudocalanus minutus elongatus*, as well as by the meroplanktonic mollusc larvae, and in spring in the Mecklenburg Bight by rotifers. Both copepod species showed a minimum in 1986. Thereafter, the concentrations increased until 1992. In 1993, the abundances decreased again. The larvae of molluscs showed a low concentration during the period 1985-87. From 1988 onwards, the abundances doubled or even increased threefold.

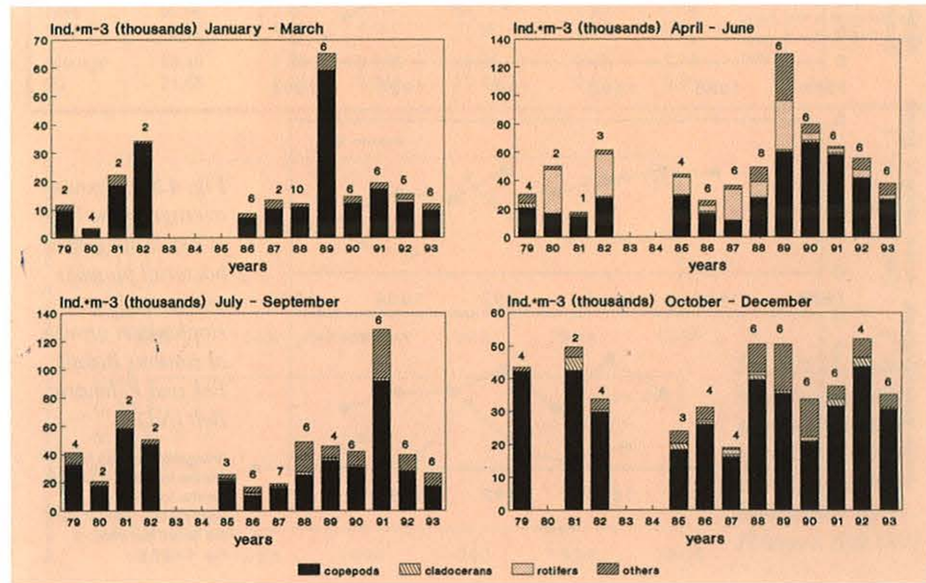
An important factor seems to be the winter temperature. The years 1985-87 were characterised by low winter temperatures. The stock of mesozooplankton was extremely low in these years. The warmer period during 1989-92 showed higher amounts of mesozooplankton. This was particularly true for *Oithona similis* and *Pseudocalanus minutus elongatus*, which both survive the winter as copepodide stages. However, the concentration of mesozooplankton was again lower in 1993, although this year was not very cold. Consequently, the statistical analyses did not show a significant correlation between temperature and mesozooplankton abundance.

An increase was observed in *Acartia tonsa*. This species is relatively new in the Baltic Sea, in that it migrated in the 1930s [*106*]. As it is a warm water species, it had favourable conditions during the last assessment period. Since 1990, the jellyfish, especially the *Aurelia aurita* populations, have been monitored in the Kiel Bight together with ichthyoplankton. To assess the influences of predation, the zooplankton data were evaluated, and showed a significant inverse relationship between jellyfish and mesozooplankton abundances [*60*]. The ichthyoplankton stocks seem also to be influenced by the jellyfish stocks, but it is not yet clear, whether this is a direct effect or through diminishing of the food source mesozooplankton.

4.5.4.3 Bacterioplankton in the Kiel Bight and Fehmarn Belt

Bacterioplankton has been monitored in the Kiel Bight, Fehmarn Belt and Mecklenburg Bight at monthly intervals at 4 stations, *Boknis Eck, N3, N1* and *M2*, which represent a gradient from the more eutrophic fjords to offshore areas [*177*]. Only data from *Boknis Eck* and Fehmarn Belt (*N1*) will be presented here to allow a direct comparison of integrated values at equal water depths. They are regarded as representative of the general temporal and spatial developments in the area.

Fig. 4.5.21 Long-term variability of the seasonal abundance of the main mezozooplankton groups in Kiel Bight, 1979-93



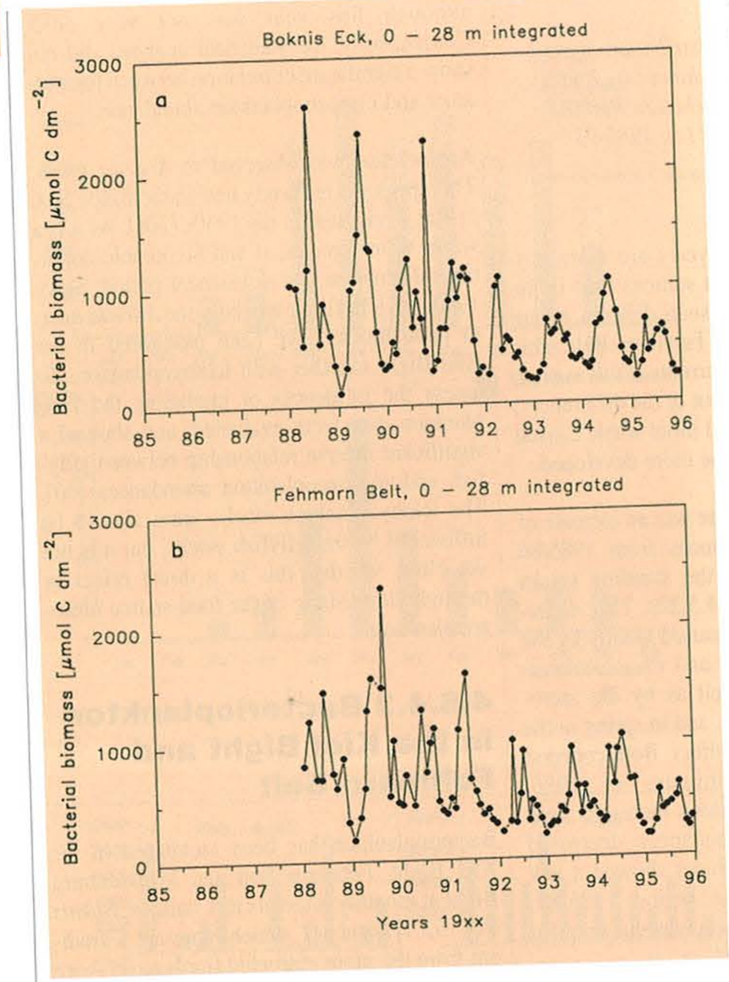


Fig. 4.5.22 0-28 m depth-integrated bacterial biomass (µmol C dm^{-2}) at stations Boknis Eck and Fehmarn Belt (N1)

(Integration based on 3 depths for 1988-90 and 5 depths for 1991-96, respectively, with 12 sample series per year. Cell numbers were determined by direct epi-fluorescence counts according to HELCOM Guidelines. Cell numbers were transformed to biomass by volume using a conversion factor of 0.35 pg C µm^{-3}.)

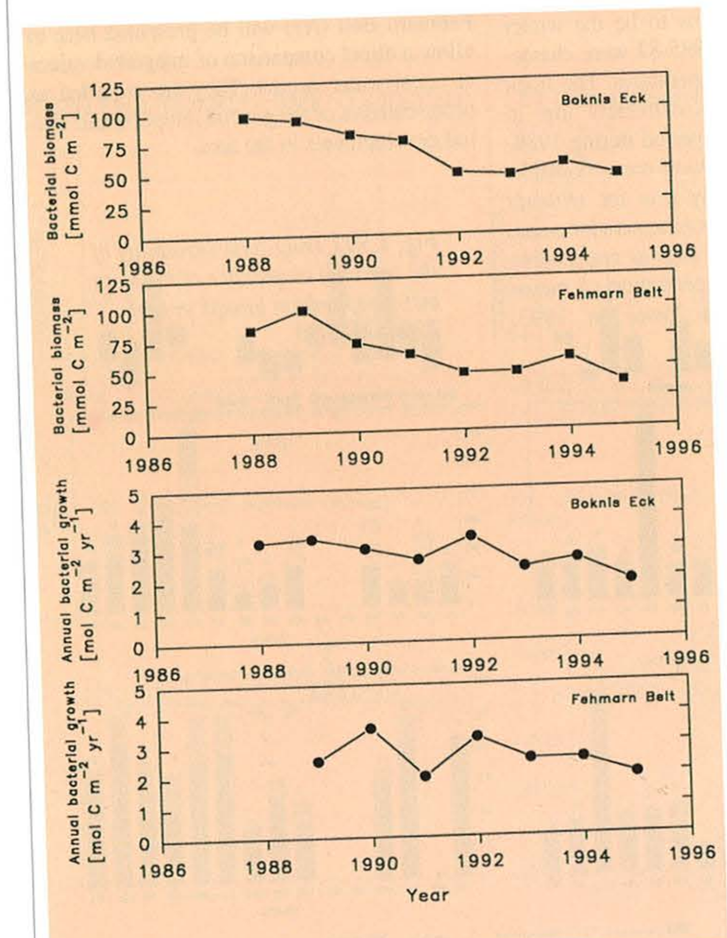


Fig. 4.5.23 Annual average of the 0-28 m depth-integrated bacterial biomass and annual bacterioplankton growth at stations Boknis Eck and Fehmarn Belt (N1)

(Integration based on 3 depths for 1988-90 and 5 depths for 1991-96, respectively, with 12 sample series per year; cf. Fig. 4.5.22.)

The biomass of bacterioplankton (Figs. 4.5.22, 4.5.23) declined in a linear manner during 1988-95 at both *Boknis Eck* and Fehmarn Belt. This could be shown by non-parametric *Mann-Kendall* analysis (p=0.003 and 0.01, respectively), and also by linear regression on the integrated annual average biomass (p=0.001 and 0.005, respectively). The latter analysis indicated a decline rate of 8.2 and 7.5 mmol C m^{-2} yr^{-1}, respectively, at these stations. In contrast, the regression analysis on integrated annual average bacterial numbers did not show any significant trend for the whole investigation period at either station. Instead, there was a significant increase in bacterial abundance during the years 1990-95 at *Boknis Eck* and Fehmarn Belt (p= 0.01 and 0.03, respectively, by linear regression; Fig. 4.5.24), the opposite of the development in bacterial biomass. This lack of coherence between the two variables indicates the difficulty of accurate cell volume measurements. Usually, biomass and cell number show similar trends *[371]*, since the biomass calculation is based on cell counts and individual cell volumes, of which the latter is usually less variable. Discrepancies may arise from the fact that the reproducibility of cell-number estimates between different personnel are within 10 % variation, while biovolume estimates often show variations of about 50 % *[277]*. Since there was a change in counting personnel between 1991 and 1992, artifacts with respect to cell volume measurements cannot be excluded. Lower and more stable biomass values after 1991 support this assumption. An independent study at station *Boknis Eck* from August 1988 to April 1990 with a comparable sampling schedule presented an integrated annual average biomass value of 5[illegible] mol C m^{-2} yr^{-1} in 1989, and an average biovolume of about 0.045 µm^3 *[176]*. The combination of this biomass value with data from the monitoring period 1992-95 revealed no significant trend during the time of investigation. This is in accordance with the results from the Bothnian Bay (cf. Chapter 4.1).

Because of possible overestimation of the bacterial cell volume during the first four years of monitoring, bacterial production values for this period were calculated by assuming a constant biovolume of 0.045 µm^3, a carbon conversion factor of 0.35 pg C µm^{-3}, and a conversion factor from thymidine incorporation to cell production of 1.1 10^{18} cells per mole of thymidine. The data from 1992-95 were calculated with the actual determined biovolumes.

No clear trend in bacterial growth (Figs. 4.5.23, 4.5.25) could be demonstrated at station Fehmarn Belt, whereas for station *Boknis Eck* a decline rate in growth of 0.18 mol C m^{-2} yr^{-1} (p=0.02) was obtained. The average annual growth of bacterioplankton during the whole period at *Boknis Eck* and Fehmarn Belt corresponded to 2.83 and 2.61 mol C m^{-2} yr^{-1}, respectively (Tab. 4.5.5). The production values for station *Boknis Eck* in 1989 are in good agreement with those from the independent study *[176]*, which justifies the applied correction and supports the slightly decreasing trend. Nonetheless, there was an increase in the thymidine incorporation rates (data not shown), which appeared in the same period when the cell numbers increased.

The specific bacterial growth rate, i.e., the P/B ratio, did not show any long-term development (Fig. 4.5.26). The average growth rate was 0.13±0.04 day^{-1} at both stations. But, the likely overestimation of bacterioplankton biomass during the first four years would imply an underestimation of the specific growth rate, and indicates a possible decline of this parameter for the whole investigation period and a stabilisation of the specific growth rate during the last four years. The corrected scenario

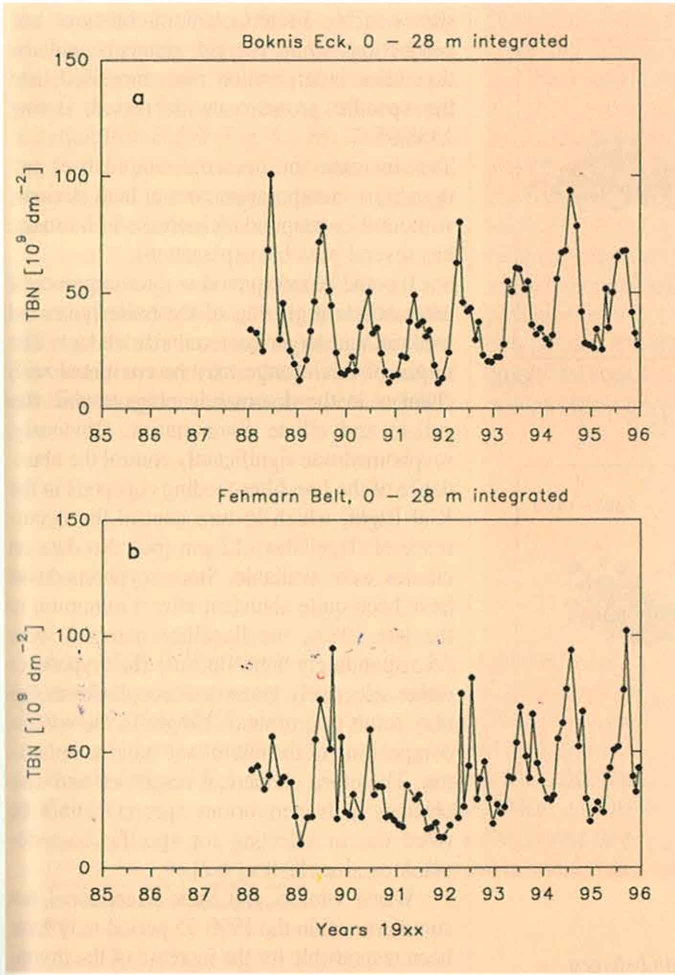


Fig. 4.5.24 0-28 m depth-integrated bacterial cell numbers at stations Boknis Eck and Fehmarn Belt (N1)

(Integration based on 3 depths for 1988-90 and 5 depths for 1991-96, respectively, with 12 sample series per year; cf. Fig. 4.5.22.)

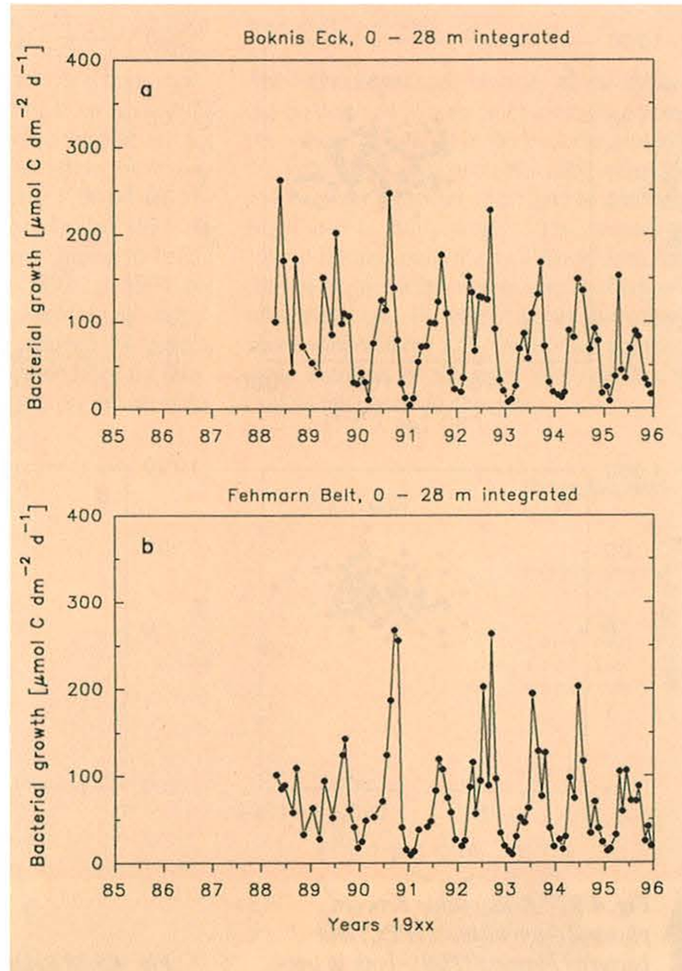


Fig. 4.5.25 0-28 m depth-integrated bacterial growth at stations Boknis Eck and Fehmarn Belt (N1)

(Integration based on 3 depths for 1988-90 and 5 depths for 1991-96, respectively, with 12 sample series per year; for further explanations see text.)

***Table 4.5.5** Integrated annual averages (0-28 m) of bacterioplankton biomass, production and specific growth rate (P/B) at the stations Boknis Eck and Fehmarn Belt (N1)*

Year	Boknis Eck Biomass (B) mmol C m^{-2}	Boknis Eck Production (P) mol C m^{-2} yr^{-1}	Boknis Eck P/B day^{-1}	Fehmarn Belt Biomass (B) mmol C m^{-2}	Fehmarn Belt Production (P) mol C m^{-2} yr^{-1}	Fehmarn Belt P/B day^{-1}
1988	97.67	3.27	0.144	84.24		0.101
1989	94.15	3.38	0.138	100.47	2.53	0.102
1990	82.11	3.05	0.110	72.44	3.58	0.150
1991	76.49	2.67	0.096	62.64	1.97	0.118
1992	49.29	3.40	0.219	46.85	3.24	0.206
1993	46.90	2.38	0.127	46.88	2.50	0.126
1994	55.84	2.63	0.116	58.11	2.50	0.105
1995	45.45	1.88	0.111	37.38	1.95	0.134
Average	68.49	2.83	0.130	63.63	2.61	0.13
SD	21.66	0.54	0.040	21.19	0.61	0.04

(Biomass and specific growth-rate values represent means of daily averages. Production values for 1988-91 were calculated with a fixed cell volume of 0.045 µm^3 - see text.)

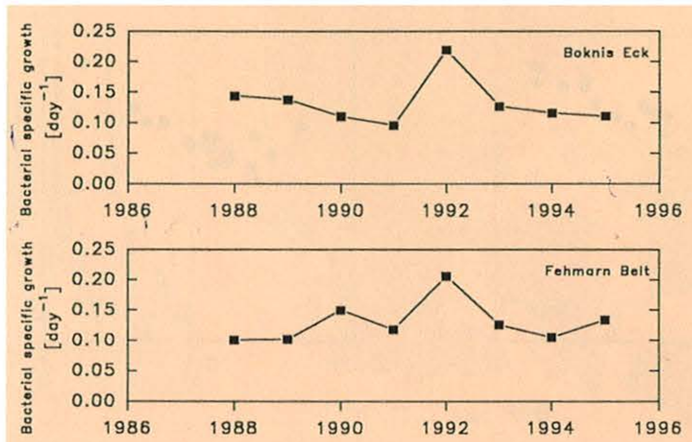


Fig. 4.5.26 Specific bacterial growth rate as the integrated daily average at the stations Boknis Eck and Fehmarn Belt (N1)

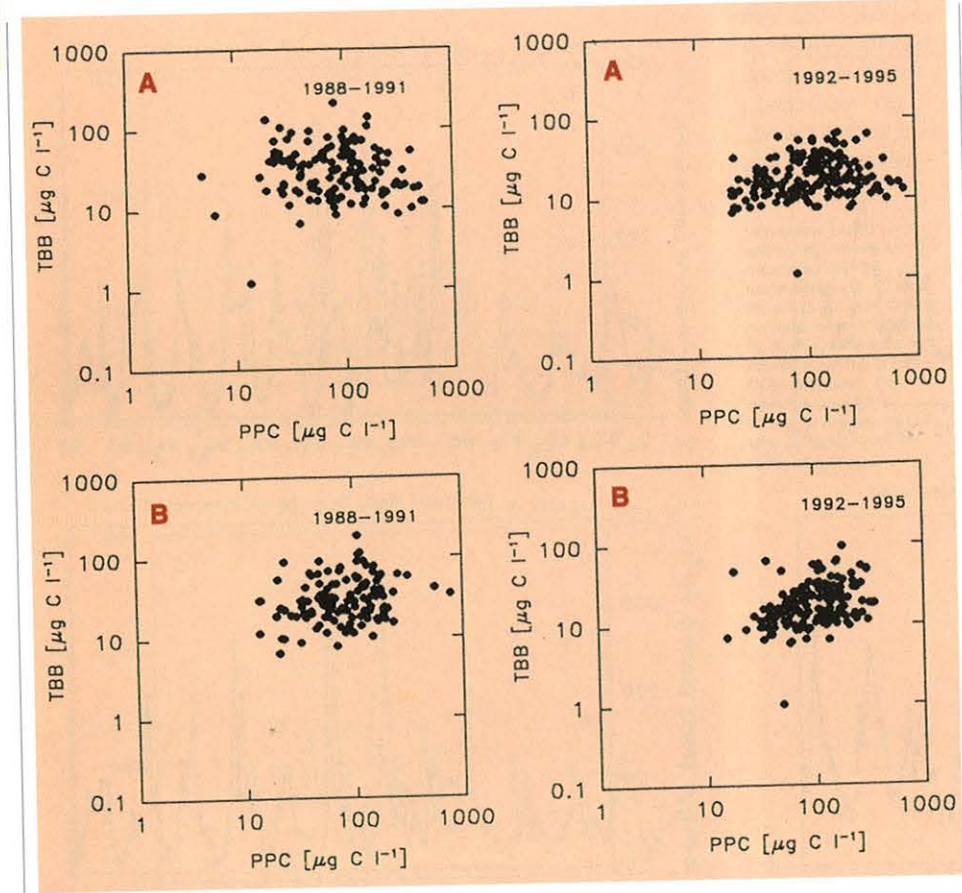
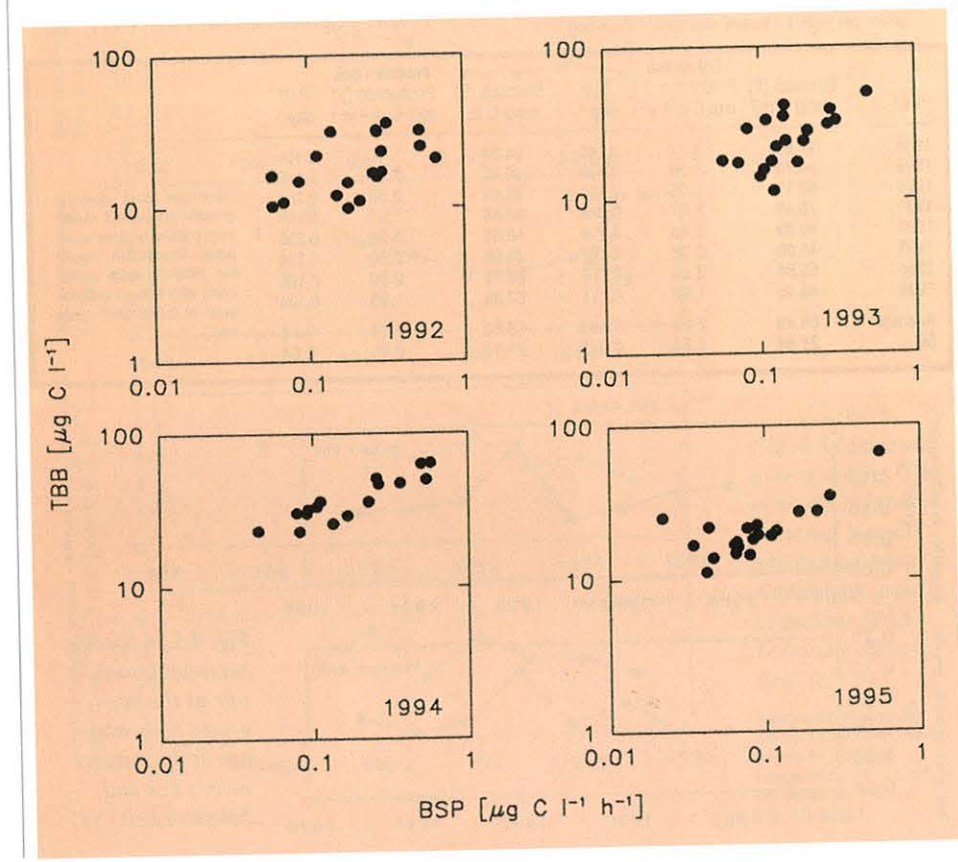


Fig. 4.5.27 Relationship between phytoplankton biomass (PPC) and bacterial biomass (TBB) –both in µg C dm⁻³– in the upper 20 m water column at the stations Boknis Eck (A) and Fehmarn Belt (N1) (B), 1988-91 and 1992-95

Fig. 4.5.28 Relationship between bacterial growth (BSP) and biomass (TBB) in the upper 20 m water column at Boknis Eck during May-September for 1992-95



shows stable bacterioplankton biomass and production, while the cell numbers and the thymidine incorporation rates increased, and the specific growth rate decreased, during 1988-95.

The increase in bacterial abundance and thymidine incorporation rates at both stations, without a corresponding increase in biomass, has several possible explanations:

- It could be interpreted as a consequence of more efficient grazing of the bacterivores on medium and larger bacterial size classes. The origin of this change may be combined with changes in the dominating phagotrophic flagellate and ciliate communities. Obviously, scyphomedusae significantly control the abundance of the fine filter-feeding copepods in the Kiel Bight, which in turn control the occurrence of flagellates <12 µm [*60*]. No data on ciliates were available. Since scyphomedusae have been quite abundant after a minimum in the late 1980s, the flagellate numbers were correspondingly high. Because the scyphomedusae selectively graze mesozooplankton, this may result in profound changes in the species composition of the micro- and nano-zooplankton. Therefore, numerical responses and also changes in bacterivorious species cannot be ruled out in selecting for specific bacterioplankton size classes [*302*].
- Warm winters and three exceptional hot summers within the 1990-95 period may have been responsible for the increase of the thymidine incorporation rate and of the bacterial abundance. The increase in accumulated temperature during the growth season may have enhanced the cell division rates, but the cells remained small due to other environmental factors, e.g., oligotrophic conditions during the growth season [*292*].
- The possible decrease in specific growth rates, without a significant decrease in growth at station Fehmarn Belt, and a small decrease of this variable at station *Boknis Eck* imply that the quality of the environment has become less favourable for bacteria during the past years, especially at station *Boknis Eck*. Higher water temperatures, which led to prolonged stagnant conditions during the growth seasons, may have increased not only bacterial activity but also competition between phytoplankton and bacteria for essential inorganic nutrients. This would also explain increasing rates of cell division producing only small cells.

Changes in micro-nutrient concentrations, e.g., by decreasing phosphorus loads, and composition, i.e., by increased N/P ratios, and changes in the grazing pressure may result in the occurrence of strong substrate control of the bacterioplankton without coupling between phytoplankton and bacteria. Although a linear relationship between phytoplankton carbon and bacterioplankton biomass at the two stations is lacking, both for the 1988-91 period with uncertain bacterial biomass data and for 1992-95 (Fig. 4.5.27), an increasing tendency towards bottom-up control of the bacterial community during the summer seasons of the last four years can be demonstrated at station *Boknis Eck* (Fig. 4.5.28). The linearity of values points towards bottom-up control, whereas data clouds may indicate a grazing-control of the bacterial community [*84*].

The uncoupling between phytoplankton and bacterioplankton parameters indicates that the observed substrate control at both stations is not closely connected to the primary producers. The exceptional warm summers in 1992, 1994 and 1995, and the low precipitation, especially during 1994/95, provide evidence that a possible decline in allochthonous inputs via precipitation and run-off from land 'dried out' the bacterial community. Support for this hypothesis may be derived from increasing glucose-turnover rates during 1990-94 and low values in 1995 (Fig. 4.5.29). During 1995, the summer temperatures were similar in height and duration compared to 1992 and 1994, but precipitation was almost absent. Long stagnation periods presumably changed the general nutritional state, and in turn modified the interactions between highly active organisms of the microbial food web.

Although pronounced changes in the system can be detected, which point towards increasing bottom-up control of the bacterioplankton, the lack of coupling between phytoplankton and bacteria identifies the Kiel/Mecklenburg Bight area as still eutrophic. The increase of colony-forming units during 1990-95 (data not shown) implies a favourable supply of organic substrates, and therefore leaves increasing water temperatures and competition for inorganic nutrients as the most likely factors to explain the observed changes.

Fig. 4.5.29 0-28 m depth-integrated daily average turnover time of ^{14}C-glucose (% day^{-1}) at stations Boknis Eck and Fehmarn Belt (N1)

(Integration based on 3 depths for 1988-90, and 5 depths for 1991-96, respectively, with 12 sample series per year; for further explanations see text)

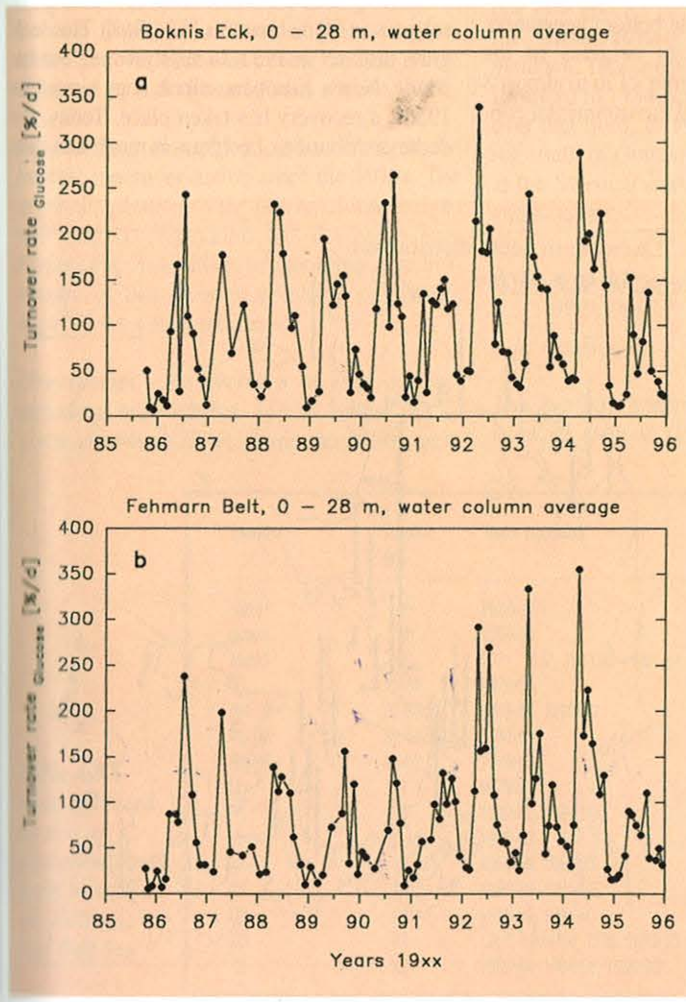
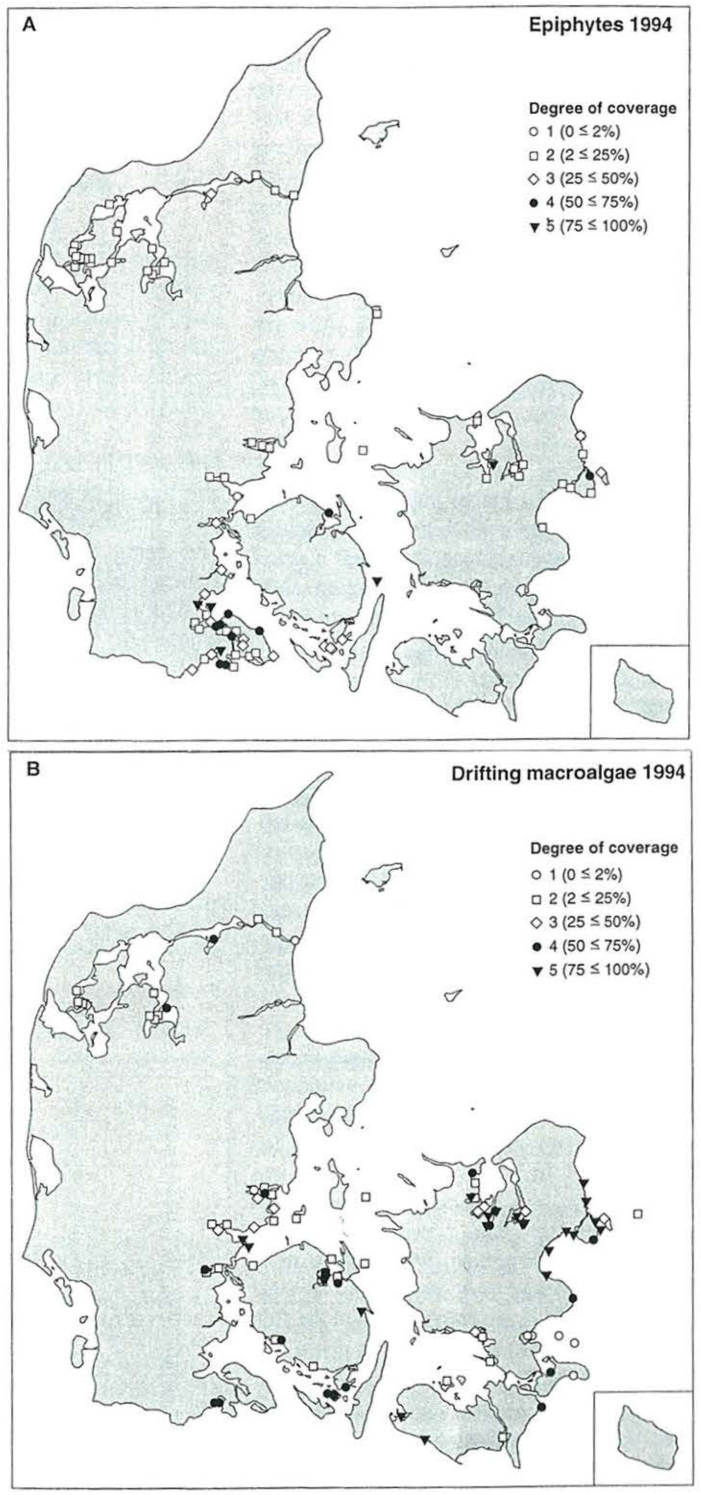


Fig. 4.5.30 Highest degree of coverage for epiphytic macrophytes (A) and drifting macrophytes (B) on transects in the Danish parts of the Kattegat and Belt Sea areas, 1994 [127]



BALTIC FISH STOCKS AND DISEASES

6

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6.1 FISH STOCKS

6.1.1 Introduction

The important commercially exploited fish species in the Baltic Sea in terms of abundance are the demersal species cod and flounder, the pelagic species herring and sprat, the anadromous species salmon and the catadromous species eel. In addition, a number of other species, including several flatfish species, and a number of fresh-water species are also exploited. The Kattegat harbours many more marine species, including the commercially important shellfish species *Nephrops norvegicus*, Norway lobster. Figure 6.1 shows the ICES Sub-divisions of the Baltic Sea that are used for fish stock assessments.

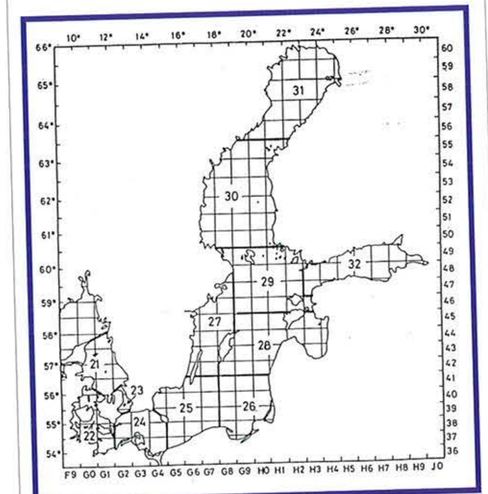


Fig. 6.1 ICES Fishing area designations in the Baltic Sea

In the first part of this chapter, the most important species of commercial fish are treated individually, and then an account is given of the trophic relationship between them. The information presented here is mainly based on the work of a number of working groups established by the International Council for the Exploration of the Sea (ICES) and the ICES Advisory Committee for Fishery Management (ACFM). The text has been prepared by *Bengt Sjöstrand*[1], Sweden, and *Henrik Sparholt*, ICES, and has been reviewed by the ACFM.

The total commercial catch by species for the period 1973-93 reveals that cod, herring and sprat, on average, constitute over 90 % of the total catch per year.

6.1.2 Demersal fish stocks

Unusually strong year classes in 1976, 1979 and 1980 formed the basis for a dramatic increase in the Baltic cod stock (Sub-divisions 25-32), and hence an expanding fishery. The stock started to decline around 1984, and its decline continued until 1992/93. Fleet capacity and fishing effort have not been reduced at the same rate, which has caused reallocation to other stocks and contributed to generally high exploitation rates on demersal stocks.

Cod reproduction is dependent on suitable environmental conditions and requires certain minimum levels of salinity and oxygen concentrations for successful fertilisation and survival of eggs [*501,687,696,699*]. The unusually long period with hardly any influx of North Sea water from 1983-91 coincided with low recruitment for cod. The recent improvement in salinity and oxygen conditions provide a possibility for improved recruitment but does not ensure it. A slight improvement in recruitment has been observed since 1992.

Because demersal species of fish and shellfish live in, on, or near the bottom of the sea, they are susceptible to near-bottom low oxygen concentrations. In the central and southern Kattegat, reduced near-bottom oxygen concentrations in the autumn have been observed for the past twenty years. Figures 6.2 and 6.3 show seasonal variations in the monthly mean concentrations of oxygen near the bottom. The data [*4,56*] have been updated by Swedish and Danish observations for 1991-94 [*Bagge, pers. comm*]. The oxygen deficit in late summer/early autumn reached a maximum in 1988, with values <1 $cm^3\ dm^{-3}$. In 1989-92, values <2 $cm^3\ dm^{-3}$ were noted, but in 1993-94, the values were >2 $cm^3\ dm^{-3}$.

6.1.2.1 Cod

Cod is distributed over the entire Baltic Sea except the Bothnian Bay. The abundance of cod is also rather limited in the Bothnian Sea and the Gulf of Finland, except in periods when the cod stock is large, and when cod seem to extend their distribution.

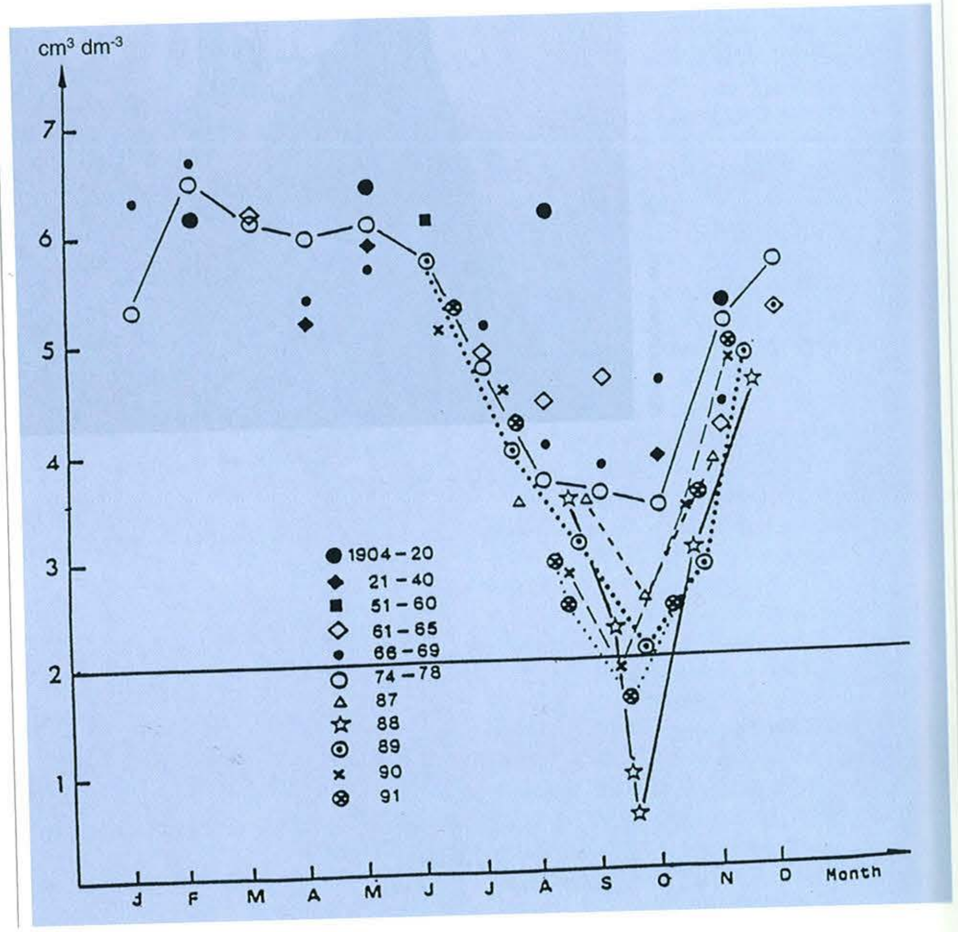


Fig. 6.2 Seasonal variations in near-bottom oxygen concentrations in the central and southern Kattegat, 1904 – 91 [4,56].

[1] see ANNEX 11.3 for addresses of authors

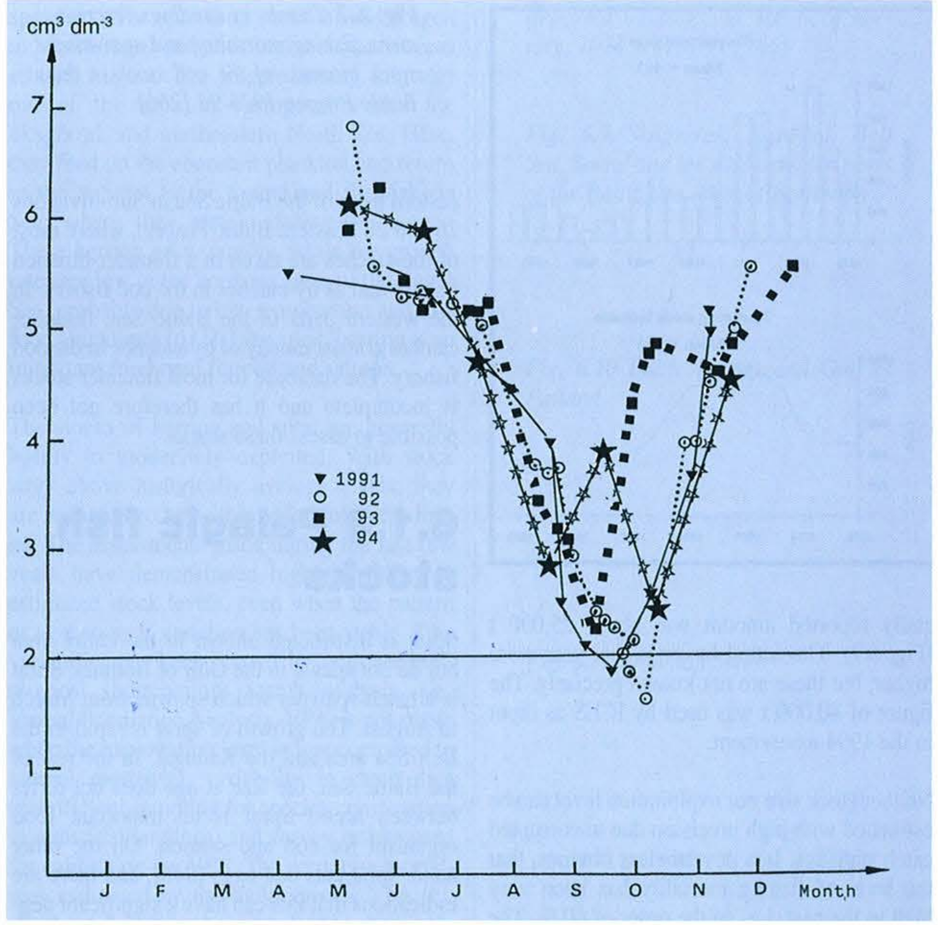
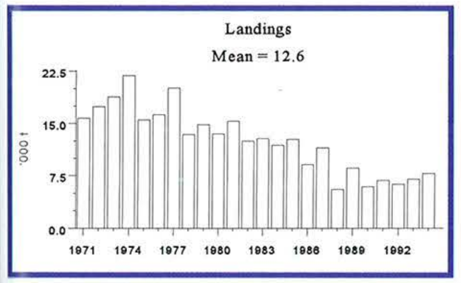


Fig. 6.3 Seasonal variations in near-bottom oxygen concentrations in the central and southern Kattegat, 1991-94 [Bagge, pers. comm.]

The cod stocks in the Baltic Sea are treated by ICES as two separate stocks. One stock occurs to the east of 15°E and forms spawning aggregations here. The other stock occurs to the west of 15° E. This stock mixes to some extent with the former stock, and with the Kattegat stock, especially during the juvenile stage and outside the spawning season. The cod in the Kattegat are also treated as a separate stock.

Some cod are sexually mature at an age of two years, and almost all cod of age five and older are mature. In the Baltic Sea, cod spawns over a period of several months, with peak spawning varying by year. In some years, the peak spawning is in May, but in recent years it has been in August. Spawning occurs in the deep parts of the sea area. Cod, however, avoid the deepest parts due to low oxygen concentrations. During spawning, cod concentrate in certain rather limited areas, and the fishing intensity can then be very high, with hundreds of fishing vessels fishing within a radius of only a few nautical miles. Cod eggs are small and a female can produce several millions of eggs per year depending on its size. The survival rate of cod eggs and larvae is, therefore, very important for the size of cod year classes. There is, however, also growing evidence from historical timeseries data for a spawning stocksize effect on year-class size. Although the cod stocks in the Baltic Sea have varied by more than an order of magnitude within the last 10-15 years, there have been only weak indications that the growth of cod has been affected, thus implying a very limited density-dependent effect. In general, the growth of cod is highest in the Kattegat and lowest in the Central Baltic Proper. This is probably due to lower temperatures and lower food supply in the water layers where cod is distributed in the Central Baltic Proper than in the Kattegat and the western parts of the Baltic Sea [*629*].

Fig. 6.4 Reported landings -in 10^3 t- of cod in the Kattegat, 1971-94 [266]



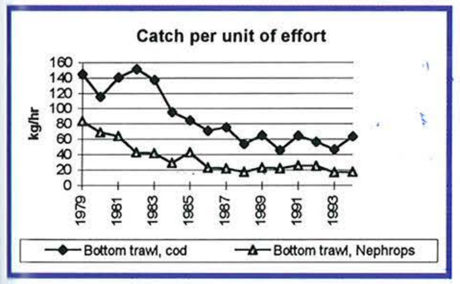


Fig. 6.5 Catch per unit of effort of cod in the Kattegat, 1979-94

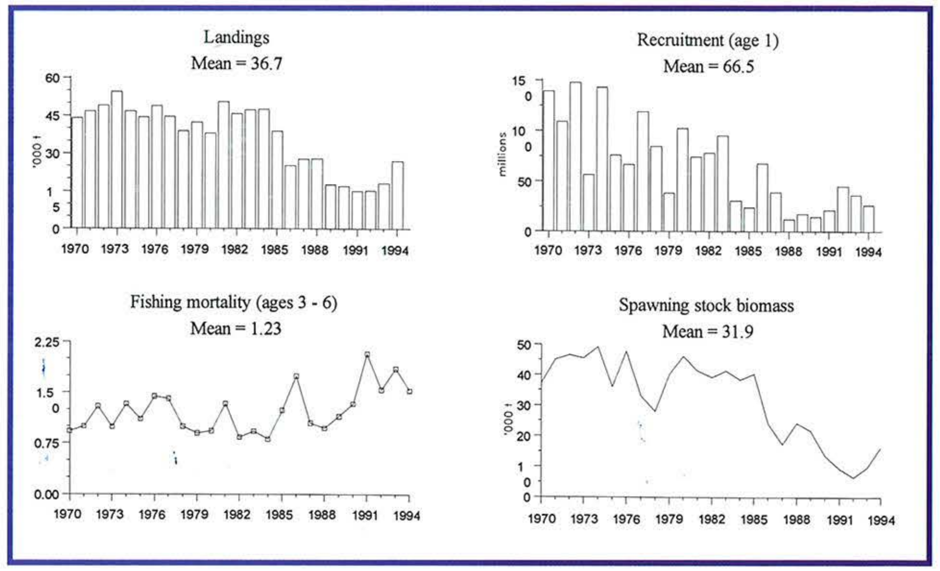


Fig. 6.6 Trends in landings, recruitment, fishing mortality and spawning stock biomass of the cod stock in the Belt Sea and in the southwestern parts of the Baltic Sea, west of Bornholm, 1970-94 [266]

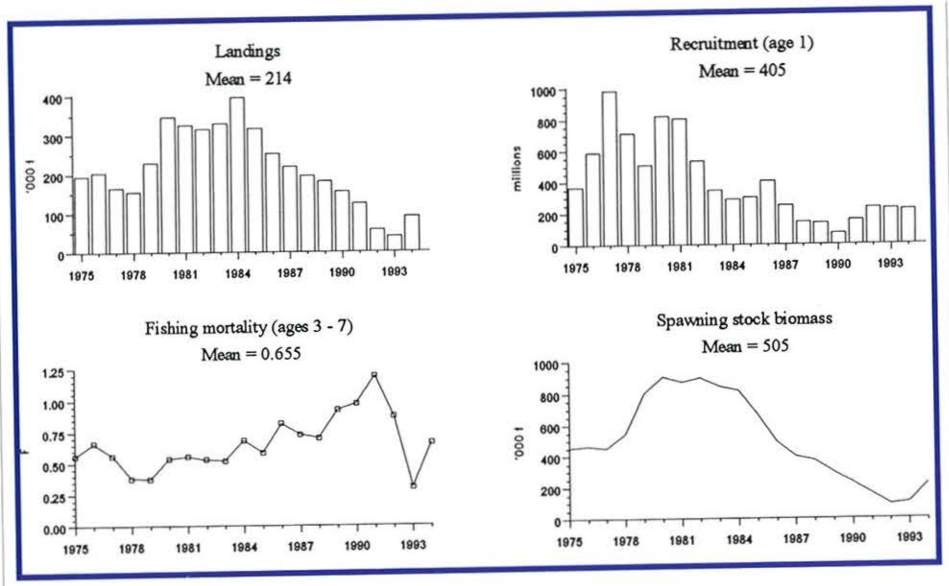


Fig. 6.7 Trends in landings, recruitment, fishing mortality and spawning stock biomass of the cod stock in the Baltic Proper, 1975-93 [266]

Cod in the Kattegat - Reported landings varied between 10,000 and 20,000 t over the period 1971-94, but since then have been below 10,000 t (Fig. 6.4). The actual level is uncertain for the last few years due to underreporting of landings.

Available data have not permitted any reliable assessment of the stock size to be conducted for the last few years. The general picture, however, is one of a heavily exploited stock. Data on catch per unit effort from two Swedish fisheries indicate a substantial drop in cod abundance during the last 15 years (Fig. 6.5). The fishing mortality rate is roughly estimated to be around 1.0, corresponding to a yearly removal of 63 % of the individuals.

The recruitment of young fish has apparently been low in most years during the period 1987-93. Only the year classes in 1989 and 1991 were more pronounced. The 1994 year class is indicated to be large.

Cod in Sub-divisions 22, 24 - The cod stock in the Belt Sea and in the southwestern parts of the Baltic Sea, west of Bornholm, yielded between 40,000 and 50,000 t per year during 1965-85, but the landings have since decreased to a level of around 15,000 t yr^{-1} (Fig. 6.6). The level of exploitation is estimated to be very high, i.e., about 55-75 % yearly mortality, and the adult cod stock has declined to around 30 % of its pre-1985 level. The survival of young fish has shown a decreasing trend during the whole period 1970-91, whereas the year classes 1991, 1992 and 1994 are relatively large and break this trend.

Cod in Sub-divisions 25-32 - There has been a change in spawning time for the cod stock in the Baltic Proper from peak spawning in May/June to peak spawning in August. Landings decreased from 1984, when they were almost 400,000 t, to 1993, when the officially reported amount was about 25,000 t (Fig. 6.7). The actual landings that year were higher, but these are not known precisely. The figure of 40,000 t was used by ICES as input in the 1994 assessment.

Neither stock size nor exploitation level can be estimated with high precision due to corrupted catch statistics. It is nevertheless obvious, that the level of fishing mortality has been very high in the past, i.e., in the order of 60 %. The biomass of adult fish has decreased sharply since 1984 and reached a minimum in 1992 of about 10 % of the 1984 level. The steep downward trend in recruitment was halted by the 1990 year class. There are indications that the 1991-94 year classes are slightly higher than the very poor 1987-89 year classes.

6.1.2.2 Flounder

The total landings of flounder have remained stable for the last 20 years. During recent years, the catch has decreased slightly in the eastern parts of the Baltic Sea in Sub-divisions 26 and 28 (Eastern Baltic Proper), where most of the catches are taken in a flounder-directed fishery and as by-catches in the cod fishery. In the western parts of the Baltic Sea, flounder catches consist mostly of by-catches in the cod fishery. The database for most flounder stocks is incomplete and it has therefore not been possible to assess these stocks.

6.1.3 Pelagic fish stocks

Sprat is distributed almost in the entire area but do not spawn in the Gulf of Bothnia. Sprat is a batch spawner which spawns from March to August. The growth of sprat is rapid in the Belt Sea area and the Kattegat. In the rest of the Baltic Sea, the size at age does not differ between areas. Sprat is an important food organism for cod and salmon. On the other hand, sprat eats cod eggs [346], and there are indications that this can have a significant negative impact on the year-class strength of cod recruiting to the fishery [630].

Herring is distributed all over the Baltic Sea, even in the almost fresh-water containing Bothnian Bay. A few decades ago, there were significant amounts of autumn-spawning herring, but now almost all herring are spring spawners. The spawning starts around Rügen in March/April and ends in the northern areas around August. The Rügen herring migrates out of the area in spring, and into the Skagerrak and northeastern North Sea. Here, they feed on the abundant plankton and return in the autumn to the Sound and the Arkona Sea, where they remain during the winter. These herring grow rapidly, while the growth rates are low in the northern areas of the Baltic Sea, probably due to low temperature and low food abundance [631]. Like sprat, herring is an important food item for cod and salmon.

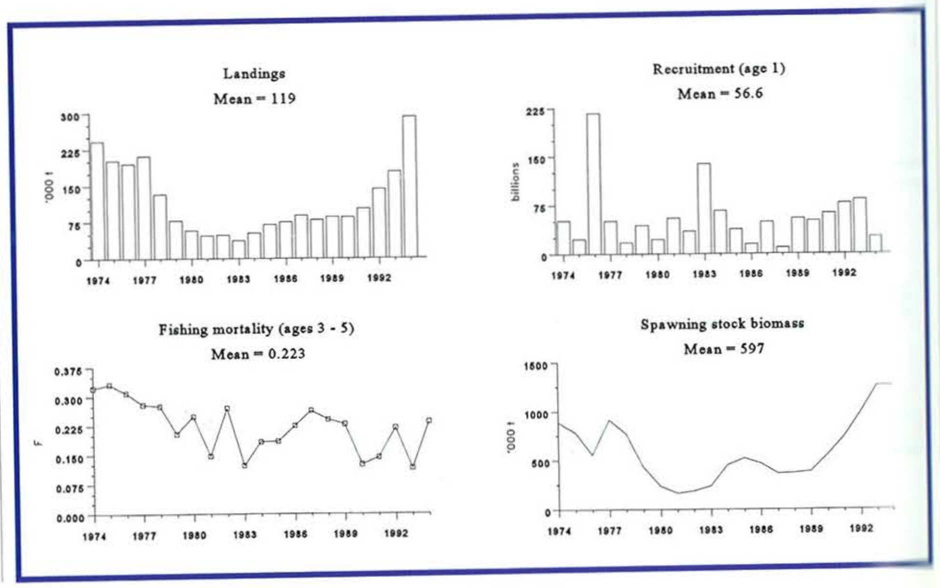


Fig. 6.8 Trends in landings, recruitment, fishing mortality and spawning stock biomass of the sprat stock in the Baltic Sea, 1974-94 [266]

The stocks of herring and sprat are generally lightly to moderately exploited. With stock sizes above historically average levels; they are regarded to be within safe biological limits. The assessments made during the last few years have demonstrated high variability in estimated stock levels, even when the pattern of year-to-year variation has been stable. This variability has been caused by both methodological shortcomings (catch analysis, i.e., Virtual Population Analysis -VPA, is not stable when the exploitation level is low compared to natural mortality), variability in catch data (insufficient sampling for species composition in industrial landings) and survey results used for calibrating the VPA. The predation mortalities estimated by the multi-species VPA (cf. chapter 6.1.7) have been applied to the single species assessments.

6.1.3.1 Sprat

Annual landings decreased from high values in the 1970s (about 200,000 t) to <50,000 t in the beginning of the 1980s (Fig. 6.8). Recently, a sprat-directed industrial fishery has increased the catches to about 300,000 t in 1994. A much decreased mortality from predation by cod and a period of low catches has contributed to the increase in stock size during the last five-year period. Estimates of stock level have varied pronouncedly, but the general pattern in stock dynamics has been rather stable.

6.1.3.2 Herring

Herring in Sub-divisions 22-24 and in Kattegat and Skagerrak - This spring-spawning stock spawns at several places in the Kattegat and the southwestern parts of the Baltic Sea, with its major spawning sites around the island of Rügen. Annual landings of spring-spawning herring have increased from about 100,000 t in the late 1970s to around 200,000 t (Fig. 6.9). The total landings, including the North Sea autumn-spawning herring caught in Kattegat and Skagerrak, have varied around 300,000 t. This stock is probably exploited at a higher level than other pelagic stocks in the Baltic Sea, but difficulties in identifying spring and autumn spawners during the non-spawning time and uncertain catch statistics introduce large uncertainties in the assessment of this stock.

Reported landings -in 10^3 t- of herring, 1974 (1973)-94 [266]

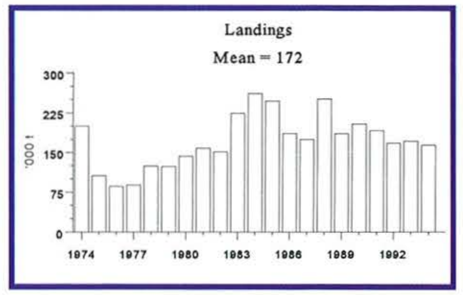


Fig. 6.9 Skagerrak, Kattegat, Belt Sea, Sound and the southwestern parts of the Baltic Sea, west of Bornholm

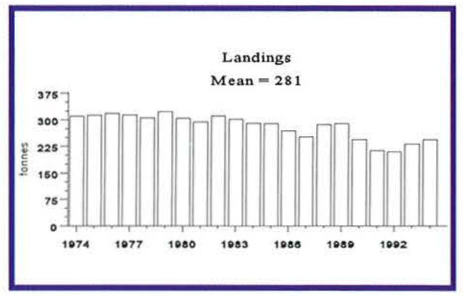


Fig. 6.10 Baltic Proper and Gulf of Finland

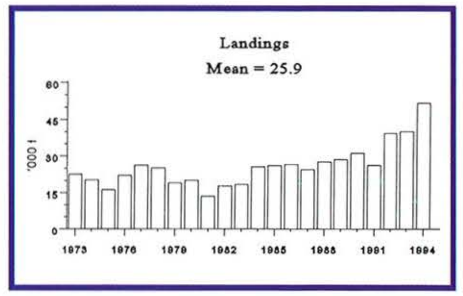


Fig. 6.11 Bothnian Sea

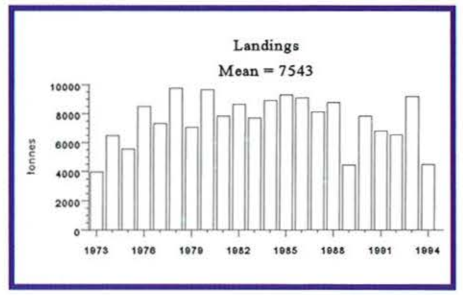


Fig. 6.12 Bothnian Bay

Herring in Sub-divisions 25-29 (Baltic Proper) and 32 (Gulf of Finland) - The herring in the Baltic Proper and Gulf of Finland, although assessed as a single stock, are of a rather heterogeneous nature, with a number of different spawning populations along the coasts. After spawning, they are mixed in the open sea. Pronounced differences in size at age are found between herring from the southern and northeastern parts of the area. Herring in the Gulf of Riga constitute a sub-stock, which in most years is fairly isolated from the rest of the herring in this area.

Annual landings fluctuated around 300,000 t from the beginning of the 1970s to 1985, but have since decreased to around 200,000-250,000 t (Fig. 6.10), mainly due to a declining market for herring. Fishing mortality is estimated to be at a comparatively low level, and the stock size is increasing.

Herring in Sub-division 30 (Bothnian Sea) - Annual landings have slowly increased during the last decade, from about 20,000 t to 40,000 t (Fig. 6.11). The low level of exploitation, about 20-25 %, exerted on this stock, is at the same level as the natural mortality. The stock size is increasing. Recruitment has been at a high level as a result of a sequence of above-average year classes.

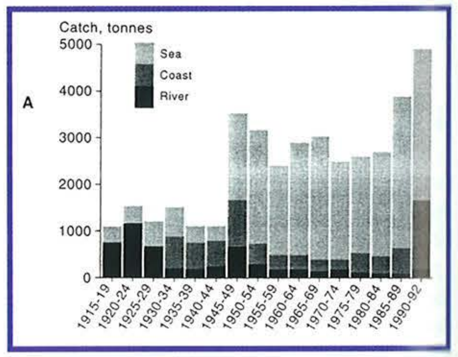
Herring in Sub-division 31 (Bothnian Bay) - Landings have never exceeded 10,000 t in this area (Fig. 6.12). Stock size and exploitation rates are difficult to assess by catch analysis, when the fishing causes only a minor proportion of the total mortality.

Fig. 6.13 The drainage area of the Baltic Sea, showing rivers and reaches of rivers supporting salmon runs in the past (thin lines) and present (thick lines) [324]

6.1.4 Salmon

Naturally reproducing salmon stocks exist in about thirty rivers in the Baltic Sea Area (Fig. 6.13). Many rivers have been dammed, and spawning and nursery areas have disappeared. To compensate, hatcheries have been built on these rivers and reared stocks are released. Normally, these fish feed in the sea and migrate back to the rivers as spawners, where they are taken and used for broodstocks. The fish are reared in the hatchery to the smolt stage and released. However, in Finland, hatchery-reared stocks are kept in hatcheries for their entire life span and are used as broodstock. The broodstock is genetically strengthened by using some spawning fish returning from the sea. This method may also be applied in other countries, mainly Sweden, due to the current problem with the *M-74* syndrome. Figure 6.14 shows the production of reared smolts by year from 1950-92. In the most recent years, the production has been around 5-6 10^6 smolts. This compares to about 0.4-0.6 10^6 smolts of wild salmon stocks, i.e., the wild salmon constitutes only about 10 % of the total

Fig. 6.15 Total annual catches as five-year means- ; in the Baltic Sea, 1915-92 [324]



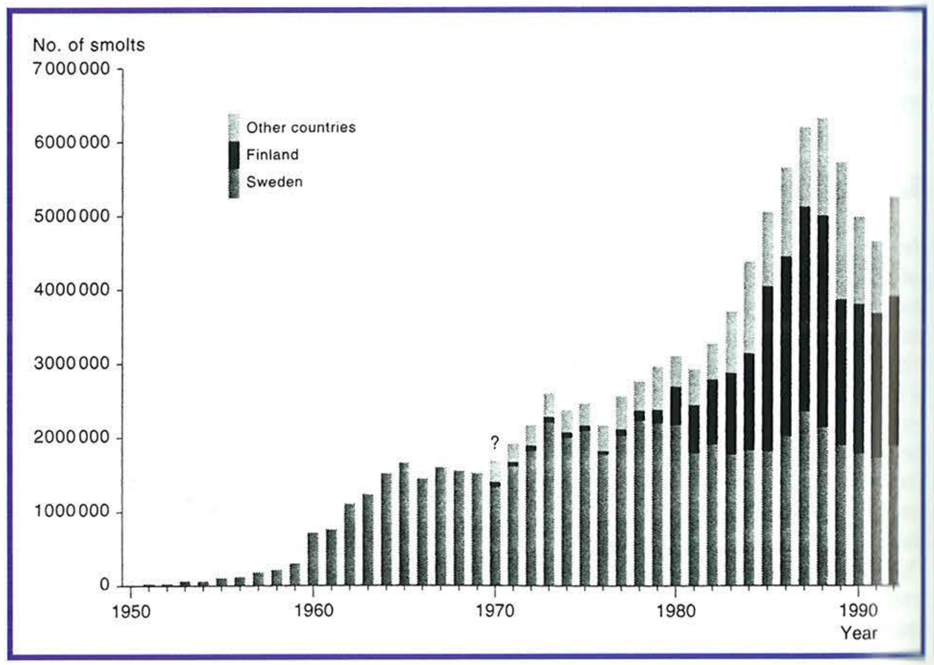


Fig. 6.14 Releases of reared salmon smolts to the Baltic Sea, 1950-92 [324]
(In 1970, the release of smolts from 'other countries' was of uncertain magnitude)

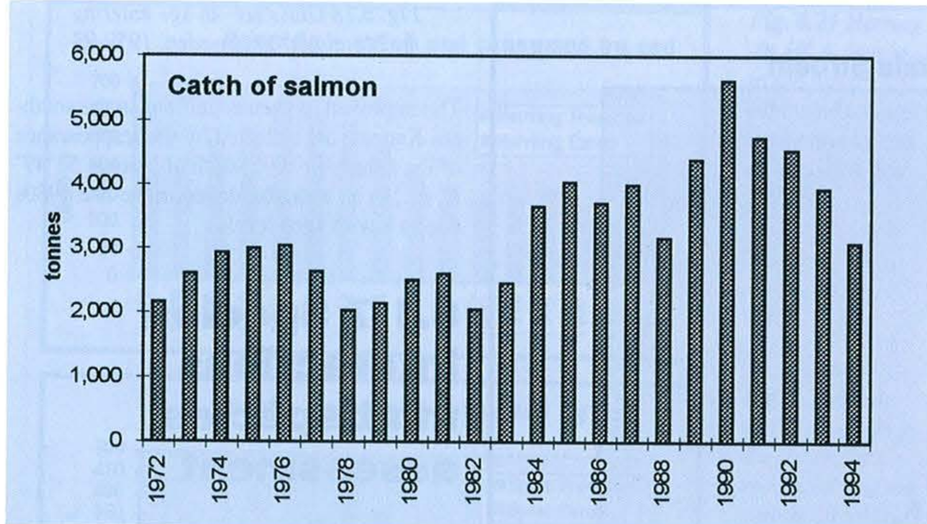


Fig. 6.16 Total annual catches of reared and wild salmon –in t– in the Baltic Sea, 1972-94

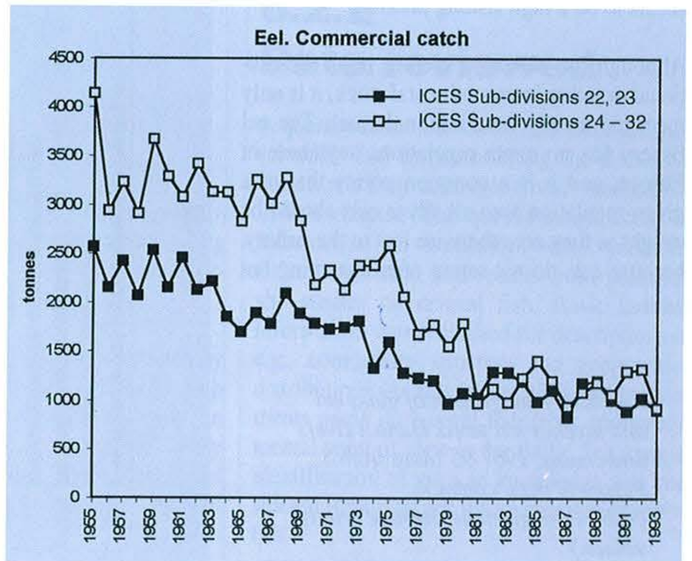
amount of salmon in the Baltic Sea. After being stable at 400,000 individuals, and even showing signs of increasing in 1993 and 1994, the production of wild smolt dropped in 1995, and is expected to drop further in 1996 due to the *M-74* syndrome.

While feeding in the sea, salmon are caught by drift nets and long lines while, during the spawning run, they are caught along the coast mainly in trap nets. In the river mouths, set-gill nets and trap nets are used, and there is a traditional recreational angling fishery in the rivers and a trolling fishery, i.e., a fishery from small boats dragging artificial fish baits, has developed during the last decade in coastal areas. The offshore fishery and most of the coastal fisheries exploit both wild and reared salmon. Reared salmon are mixed with wild salmon, except during the homing migration when reared salmon approach their release sites near river mouths. Only then is it possible to exploit reared salmon without catching wild salmon at the same time. Total annual catches of reared and wild salmon are given in Figures 6.15 and 6.16. It can be seen, that the annual catch has been fairly stable at around 3,000 t per year since 1945. There has been a tendency to an increase in recent years, which is due to a larger smolt production at the end of the 1980s and in the beginning of the 1990s, and to a high growth rate observed in salmon in the most recent years.

Approximately 20 % of the smolts entering the Baltic Sea survive to the minimal landing size for the fishery. Fishing mortality on salmon reaching this size is extremely high. Of the salmon belonging to the stocks in the rivers discharging to the Gulf of Bothnia, <0.5 % manage to return to the rivers as adults for spawning. In the Baltic Proper, the fishing mortality is lower and 1-2 % are able to return to spawn. The stocks are clearly both growth-overfished, i.e., the salmon are caught at too small a size, and recruitment-overfished, i.e., too few juvenile salmon are produced due to too few adult salmon managing to return to the rivers and spawn. Only about 10-20 % of the current production capacity in the rivers are at present utilised. The reduction in wild smolt production in 1995 and 1996 will result in a significant further reduction in returns of adult wild salmon beginning in 1997.

The wild salmon stocks in the Gulf of Bothnia are considered to be outside safe biological limits. At present, only 12 of the original 44 wild salmon stocks in the rivers discharging into the Gulf of Bothnia remain in existence. In the Gulf of Finland, only 6 stocks remain and in the Baltic Proper, 12 stocks remain. The estimated production of wild smolts in the Gulf of Bothnia decreased from 455,000 in 1994 to 204,000 in 1995 and 84,000 in 1996. Production in the Gulf of Finland and in the Baltic Proper has also declined.

Fig. 6.17 Officially reported commercial catches of eel –in t– in the Baltic Sea (ICES Fisheries Statistics)
(The time series goes further back for some countries and indicates that, at least back to 1930, the annual catch was at or above the 1955



The wild salmon stocks in the Baltic Proper are in a less severe state than those in the Gulfs of Bothnia and Finland. The sharp reduction in wild parr production since 1994 indicates that the gradual increase in the stocks in recent years will not continue, and the stocks are likely to decline. The smallest wild stocks are at risk of extinction, but the larger stocks and those in the Baltic Proper are not so sensitive to temporal variations in the size of the spawning stock. The survival of these small wild stocks has probably been possible because they are not exploited in coastal or river fisheries. The combination of decreased spawning stock and low survival of fry may result in total extinction of new salmon generations in the rivers currently supporting naturally reproducing stocks.

6.1.5 Eel

Eels enter the Baltic Sea as glass eels coming from the Sargasso Sea. Some eels remain in the Baltic Sea and the rest migrate up the rivers and brooks, and live there or in connected lakes. After a number of years, as so-called yellow eels, they develop into silver eels and start their migration back to the Sargasso Sea for spawning. This migration takes place in August to October.

The commercial catch of eels has decreased steadily since 1955, which is the first year for which reliable catch data are available (Fig. 6.17). There is no estimate of the stock size available and a certain number of landings are not reported. Indications from catch per unit effort from selected fisheries suggest, however, that the stock of silver eels has decreased in accordance with the decrease in the catches [253]. Higher fishing intensity and better fishing gears are likely to be part of the explanation for the decrease in stock size of silver eels. In addition, there are indications that the

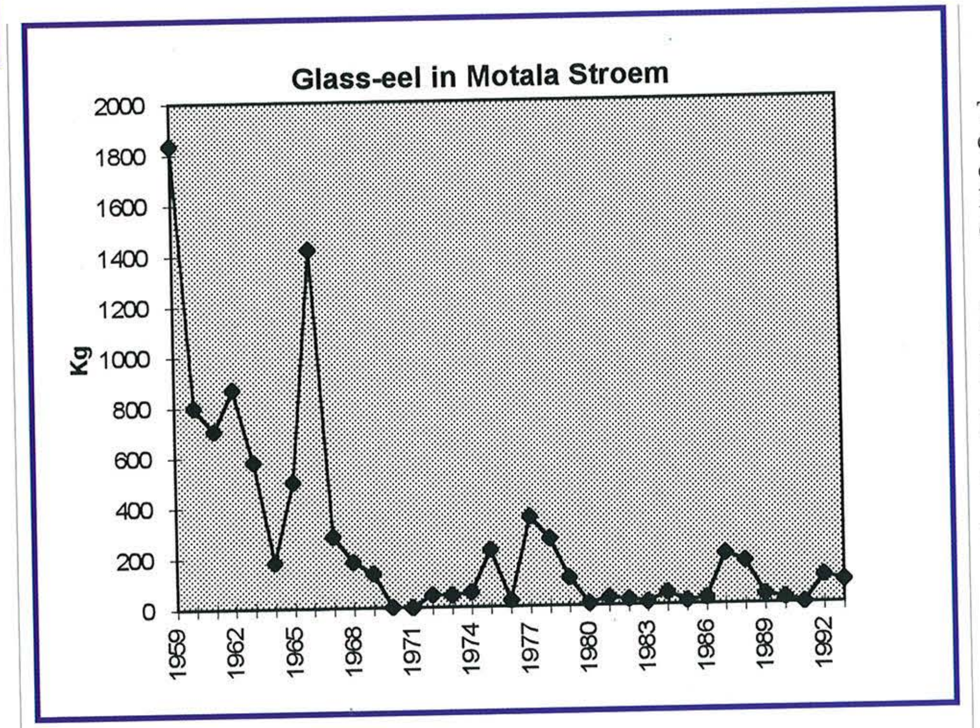


Fig. 6.18 Glass eel –in kg– entering the Motala Ström, Sweden, 1959-93

immigration of glass eels has decreased over the above-mentioned time period. Figure 6.18 shows the records of glass eels entering the Motala Ström in Sweden, and Figure 6.19 shows the immigration of glass eels and juvenile eels in six Danish rivers and canals. It is only in a few rivers that the number of glass eels entering has been recorded on a regular basis. There are, however, quite a few rivers where this has been recorded from time to time, and the general impression is the same downward trend. This is not only the situation for the Baltic Sea, but appears to be general for all of Europe. Whether the decrease in annual production of glass eels is due to a decrease in spawning stock size, or to changes in currents in the Atlantic Ocean transporting the eel larvae to Europe, is not known. It is, however, striking that other eel species in other parts of the world, e.g., the North American and Japanese eel, have shown similar downward trends, and the only likely common factor seems to be a high fishing pressure.

Although the eel stock clearly is an international or rather intercontinental stock, it is only managed locally on a national basis. The eel fishery has no quota regulations anywhere in Europe, and it is a common policy that in a given regulation area all silver eels should be caught as they are otherwise lost to the fishery, because eels do not return after spawning but die. Thus, it is not unlikely that eels are recruitment-overfished.

6.1.6 Norway lobster in the Kattegat

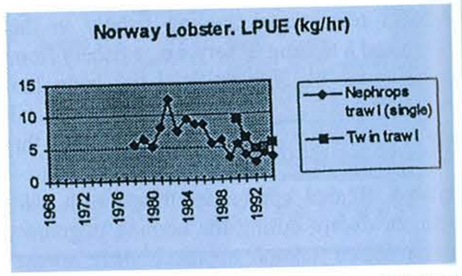
Annual landings of Norway lobster decreased from 2,000 t in 1984 to <1,000 t in 1993 and 1994. No proper assessment of stock sizes has been presented. The time series of landings per unit of effort in two Swedish fisheries, however, indicate that the stock has undergone a pronounced decline since 1984 (Fig. 6.20).

The improved oxygen conditions in the southern Kattegat are reflected by the reappearance of the fishery in 1992 south of latitude 56°49' N, i.e., in an area abandoned in the late 1980s due to low oxygen levels.

6.1.7 Species interactions - multispecies assessment

Interactions between the major fish species in the Baltic Sea, i.e., cod, herring and sprat, are significant and important for the fishery and the stock dynamics. Models have been developed to quantify the effects of predation on stock sizes. Multi-species VPA [*246,552*] has been used to estimate total number predated and predation mortality. Data on food consumption and consumption rations of predators are needed in addition to data needed in the single-species VPA. Large numbers of cod

Fig. 6.20 Catch per unit of effort of Norway lobster (kg lobster per trawl hour) in the Kattegat, 1978-94



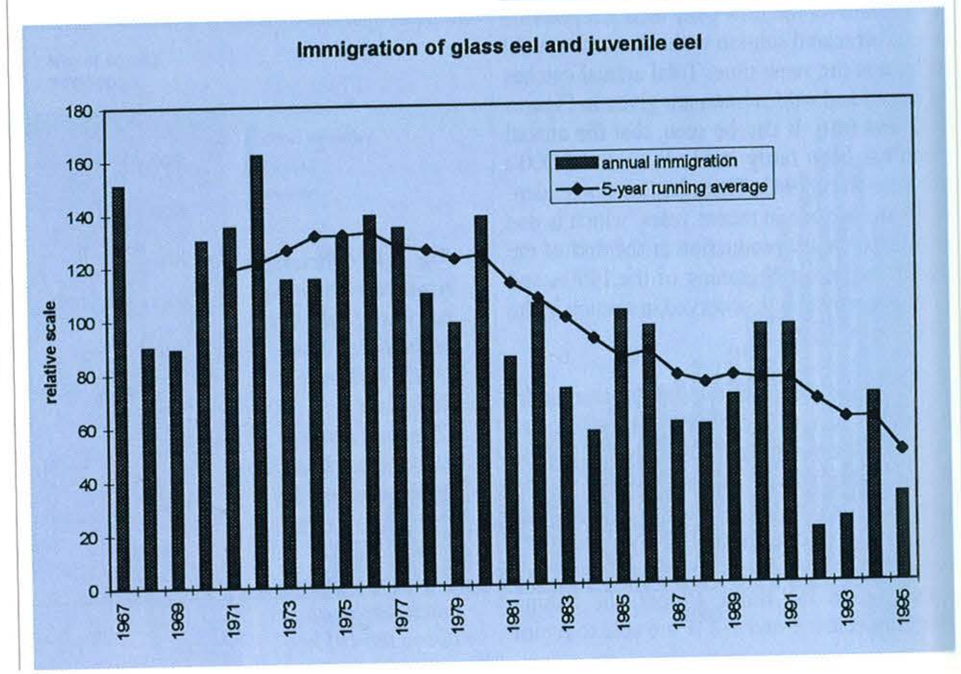


Fig. 6.19 Immigration of glass eel and juvenile eel to six Danish rivers and canals, 1967-95 (data from S. Petersen, pers. comm.)
(Values are given as average relative values.)

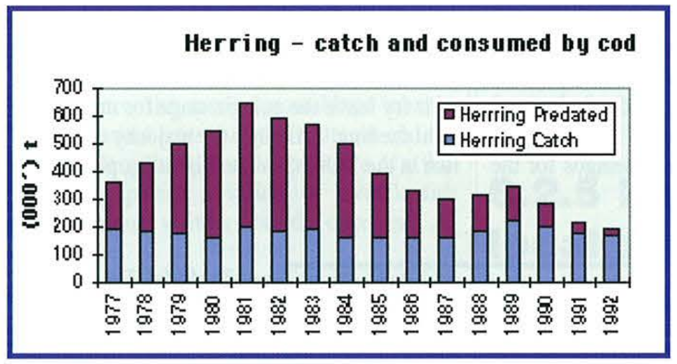


Fig. 6.21 Herring -in 10^3 t- caught in the fishery and estimated amount consumed by cod, in the Baltic Sea, 1977-92

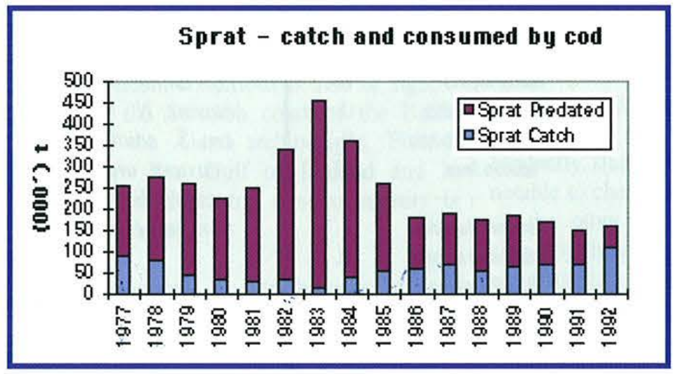


Fig. 6.22 Sprat - in 10^3 t- caught in the fishery and estimated amount consumed by cod, in the Baltic Sea, 1977-92

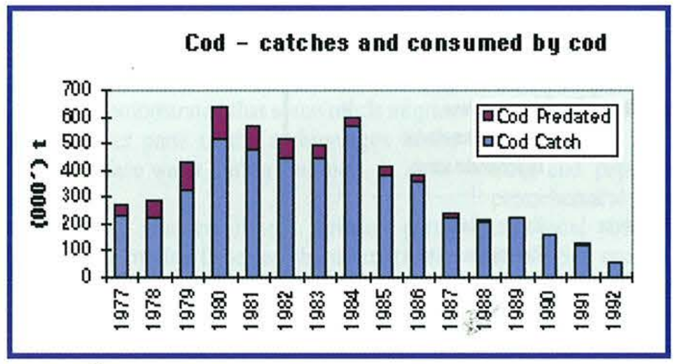


Fig. 6.23 Cod -in 10^3 t- caught in the fishery and estimated amount consumed by cod, in the Baltic Sea, 1977-92

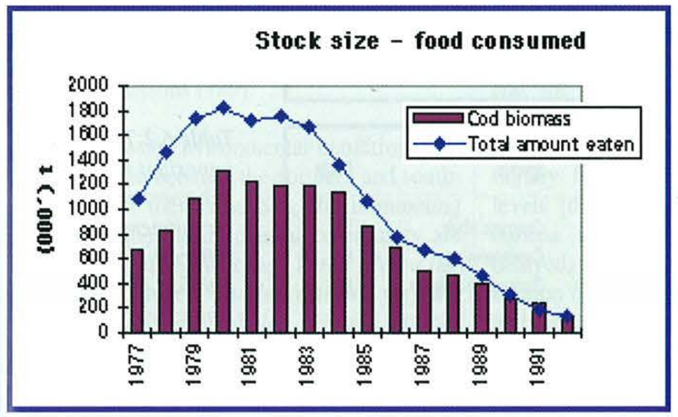


Fig. 6.24 Total biomass of cod -in 10^3 t- compared with the total amount of fish consumed by cod, in the Baltic Sea, 1977-92

stomach analyses have provided information on food composition [*263*]. Based on experiments on digestion rates of cod and other predatory fish, consumption rates have been estimated [*264*].

Results from multi-species assessments have been presented [*265,267*] and summarized including a comprehensive overview [*629*].

Cod becomes more piscivorous with increasing size and fish dominates the diet in specimens >40 cm in length. Yearly food consumption is estimated to increase from about 0,2 kg for a one-year-old cod to 8 kg for a nine-year-old cod.

In Figures 6.21 to 6.23, the quantities eaten by cod of each prey species are compared with the catches. The high predation mortality on ≤1 year old cod is an important regulatory mechanism for cod stocks. The total amount eaten by cod is shown together with cod biomass in Figure 6.24.

6.2 COASTAL FISH

6.2.1 Introduction

A first assessment of the state of the coastal waters of the Baltic Sea was officially available in 1993 [226]. The report included an introduction to coastal fishing and the impacts of pollution on fish stocks. Although the available monitoring data were scattered and not very homogeneous, the assessment demonstrated that coastal fish are exposed to a variety of pollution impacts and that the fish-community responses may often be quite dramatic. Sustainable use of fish resources was considered to be of major interest in the future management of the coastal zone, as expressed in a HELCOM interim report [*221*]. Five of the six overall goals are more or less related to the well-being of coastal fish. Consequently, fish studies at different levels of organization, including biological effects studies, have been suggested for future monitoring under the COMBINE (Cooperative Monitoring in the Baltic Marine Environment) umbrella. To provide a background for planning, it was decided to include a chapter on coastal fish in the present assessment.

This section has been prepared by *Olof Sandström*[1], Sweden, *Wladyslaw Borowski*, Poland, *Erik Hoffmann*, Denmark, *Holger Hovgård*, Denmark, *Mart Kangur*, Estonia, *Hannu Lehtonen*, Finland, *Else Nielsen*, Denmark, *Otto Rechlin*, Germany, *Rimantas Repecka*, Lithuania, *Krzysztof Skora*, Poland, and *Maris Vitinsh*, Latvia. The text has been reviewed by the ACFM.

6.2.2 Data reviewed

A questionnaire was circulated among fishery scientists and institutes along the Baltic Sea coasts, and their contributions are summarized in this chapter. The questionnaire asked for information on monitoring data, including fishery statistics, and results from pollution-site studies on coastal fish. Basic faunistic information was requested for descriptions of, e.g., community structure and geographical distributions. As this is one of the first assessments made on coastal fish from an environmental point of view in the Baltic Sea Area, an identification of gaps in knowledge was considered important for future monitoring activities.

[1] see ANNEX 11.3 for addresses of authors

6.2.3 Characteristics of the coastal waters as fish habitats

Morphometric characteristics strongly influence the properties of coastal fish habitats [*320,671*]. Five dominating coastal types have been distinguished in the Baltic Sea, i.e., archipelagos, fjords, open low coasts, klint coasts and lagoon or Bodden type systems [*226*]. Archipelagos dominate in the north and west, and open low coasts in the south and east, including the Kattegat. Open coasts, with deeper water sensitive to upwelling, are found along the western shoreline of the Bothnian Sea. The coastal lagoon is a specific phenomenon of the Baltic Sea, occurring in the southern and southeastern regions. Considering the specific requirements of coastal fish, three major classes of habitats can be identified, archipelagos, open coasts and coastal lagoons.

Several species, limnic as well as marine fish, need access to sheltered, warm and hydrographically stable conditions for spawning and early fry development. Archipelagos and coastal lagoons are consequently very important for the sustained production of coastal as well as open sea fish. Small coastal streams and rivers contribute to the supply of young fish in the coastal zone. In restricted areas, e.g., in the Bothnian Bay, archipelagos and lagoons can be the main habitats available for the spawning and early life stages of spring-spawning species as the surrounding coastal waters are too cold during the incubation and larval periods [*155,396*]. The generally high biological production in the coastal areas presents good feeding conditions for adult fish living there.

6.2.4 Coastal fish-species assemblages

Salinity is a major factor structuring the fish communities, but water temperature may also be of great importance. Most North Atlantic fish can be classified as cold-water adapted, and very few of them stay in shallow coastal waters during all developmental stages. The limnic species are generally more stationary, as their higher temperature optima allow them to stay in coastal waters close to their recruitment habitats throughout the year.

The fish-species assemblage in the Kattegat, Sound, along the coast of Skåne, in the Danish straits and along most parts of the German, Polish and Lithuanian coasts is dominated by marine species, and it is not easy to distinguish a specific coastal fish community. Herring (*Clupea harengus*) and garfish (*Belone belone*), among others, approach the coastal waters in spring for spawning, and their offspring stay in bights and archipelagos for the first part of their development. Fish with pelagic eggs and larvae, such as flatfish, also utilise the coastal zone as a nursery area when their fry leave the pelagic stage for more demersal feeding. Although the majority of marine fish in the Baltic Sea are migratory, pipefishes, wrasses, some gobies and the viviparous blenny (*Zoarces viviparus*) are examples of species with more sedentary behaviour. As a consequence, the species assemblage inhabiting the coastal waters of the Kattegat and the southwestern Baltic Sea, is a mixture of a few stationary species, a variety of juvenile fish and occasional visitors from the open sea.

A very different fish community, dominated by stationary warm-water adapted limnic fish, has developed in the archipelagos and coastal lagoons of the Baltic Sea. Salinity may be a dominating factor structuring these communities, but access to sheltered and high-temperature recruitment conditions is also of significance. At the Swedish coast of the Baltic Proper, in the Åland archipelagos, Finnish Archipelago Sea, Gulf of Finland and in Estonian coastal waters, this community is widely distributed.

During the summer, local fresh water fish populations may also utilise the open coasts close to suitable recruitment areas in the southern and southeastern parts of the Baltic Sea. Perch, pike perch, roach and bream are reported as generally occurring in most Polish, Lithuanian and Latvian coastal waters, and Swedish studies have demonstrated that some roach migrate to the outer parts of the archipelagos in the warm surface water during summer.

Cold-water adapted limnic species partly replace the marine fishes as characteristic elements in the open sea of the Gulf of Bothnia. The sea-spawning whitefish (*Coregonus lavaretus*), vendace (*Coregonus albula*) and smelt (*Osmerus eperlanus*) avoid the warm coastal waters in summer, but they migrate shorewards for spawning and feeding during the colder seasons [*489*].

Although basic environmental conditions differ very much between the northern and southern parts of the Baltic Sea, the dominating species in the limnic coastal community are the same all along the coast. Perch (*Perca fluviatilis*) and roach (*Rutilus rutilus*), together with ruffe (*Gymnocephalus cernua*), are the backbones of the fish community in the northern Råneå Archipelago as well as in the southern Curonian Lagoon. However, pike perch (*Stizostedion lucioperca*) and a variety of cyprinids, such as silver bream (*Blicca bjoerkna*), are very rare in the north, but may be of considerable importance in the eutrophic southern coastal lagoons.

Spawning and nursery areas are often scattered and restricted in size compared to the available feeding grounds for older fish. Even small-scale human influence, when directed to a fish-recruitment area, may have negative impact on stocks inhabiting surrounding waters. Consequently, a proper management of the Baltic Sea coastal areas must also consider the importance of small inlets, 'glo lakes' and coastal streams to secure coastal fish production and a high biological diversity.

Area	Habitat	Species	Origin
Gulf of Bothnia & N Baltic Proper (Lister-Viborg)		*Rutilus rutilus*	L
		Perca fluviatilis	L
		Gymnocephalus cernua	L
		Zoarces viviparus	MS
		Coregonus lavaretus	LN
		Myoxocephalus quadric.	M
		Platichthys flesus	MS
E and S Baltic Proper (Viborg-Rügen)	Coastal lagoons, shallow bights	*Pomatoschistus minutus*	M
		Gasterosteus aculeatus	ML
		Blicca bjoerkna	L
		Abramis brama	L
		Gymnocephalus cernua	L
		Perca fluviatilis	L
		Rutilus rutilus	L
		Stizostedon lucioperca	L
	Open coasts	*Zoarces viviparus*	M
		Pomatoschistus minutus	M
		Myoxocephalus scorpio	M
W Baltic Proper & Kattegat (Rügen-Lister)	Bights and archipelagos	*Zoarces viviparus*	M
		Pomatoschistus minutus	M
		Gobius niger	M
		Pholis gunellus	MW
		Pungitius pungitius	M
	Open coasts	*Myoxocephalus scorpio*	M
		Zoarces viviparus	M

Table 6.1 *The most important stationary fish species in different coastal habitats of the Baltic Sea (L - limnic, M - marine; N - northern, S - southern, E - eastern, and W - western distribution, respectively)*

Area	Habitat	Species	Origin
Gulf of Bothnia & N Baltic Proper (Lister-Viborg)		*Coregonus albula*	LN
		Clupea harengus	M
		Gasterosteus aculeatus	ML
E and S Baltic Proper (Viborg-Rügen)	Coastal lagoons, shallow bights	*Belone belone*	M
		Clupea harengus	M
		Vimba vimba	L
	Open coasts	*Platichthys flesus*	M
		Clupea harengus	M
		Gadus morhua	M
		Ammodytes tobianus	M
W Baltic Proper & Kattegat (Rügen-Lister)	Bights and archipelagos	*Platichthys flesus*	M
		Limanda limanda	M
		Clupea harengus	M
		Belone belone	M
		Ammodytes tobianus	M
	Open coasts	*Platichthys flesus*	M

Table 6.2 *The most important migratory fish species with recruitment areas in different coastal habitats of the Baltic Sea (L - limnic, M - marine; N - northern, S - southern, E - eastern, and W - western distribution, respectively)*

6.2.5 Impact of local and large-scale pollution

Migratory habits and reproduction biology strongly determine the reaction of fish to local pollution. Population effects of toxic effluents are generally related to chronic physiological damage to the reproductive system or to more acute toxic effects on early life stages. Evidently, stationary species are primarily vulnerable to chemical pollution. Eutrophication, on the other hand, mainly affects fish by changing the general qualities of their recruitment habitats. Consequently, migratory fish, with spawning or nursery areas in coastal waters, may be as sensitive to eutrophication as sedentary species.

6.2.5.1 Influence of local pollution

Pulp and paper mills, metal industries and petrochemical industries are the most important local sources of chemical pollution in Baltic Sea coastal areas. A more diffuse and complex contamination may be expected close to larger cities or other densely populated areas. Industrial site studies during recent years have mainly been made at pulp mills. In Sweden, considerable research has documented the effects of chlorine-bleached effluents and the restoration of receiving waters as bleaching techniques have been changed. During the 1980s, effects were seen on stationary fish from sub-cellular to population levels [*628*]. In perch, physiological disturbances appeared, sexual maturation was delayed, gonad growth was retarded, the production of surviving fry was strongly reduced and the abundance of adult fish was depressed. These responses appeared at several mills. The general impact on reproduction was verified in later Canadian studies [*473*]. Follow-up studies demonstrate that mill improvements may result in significant recoveries of the stationary coastal fish community [*593*]. These observations, however, are not consistent as there are recent indications of impaired reproduction also at mills substituting chlorine by other chemicals [*594*].

Reproductive damage in localised areas was also documented in burbot (*Lota lota*) in the northern Bothnian Bay [*556*], and in roach (*Rutilus rutilus*) in the eastern Archipelago Sea [*702*]. Although the response patterns may suggest a chemical exposure, supporting evidence from, e.g., chemical analyses, was lacking. Gonad damage seen in roach closely resembles observations made on the same species at power plants discharging cooling water to the environment [*427*].

Long-term studies on effects from land drainage were made in the lower parts of the Finnish river Kyrönjoki [*261,334*]. Extensive ditching programmes led to large-scale leaching of acid substances and metals, mainly aluminium, and a very low pH during the spring-flood season. In the beginning of the 1980s, recruitment of most species of fish was seriously damaged. Repeated observations of very low abundance of fry of spring-spawning species, such as the cyprinids and perch and the autumn- and winter-spawning whitefish (*Coregonus lavaretus*) and burbot, were correlated with pH levels <5 during the early life stages. In the beginning of the 1990s, recruitment was significantly improved. This was, however, mainly an effect of the recent mild winters, causing pH minima to occur much earlier in spring [*260*]. Similar negative influences on fish evidently occur in other rivers, e.g., the Malax and Petalax rivers, in this district of Finland [*260*].

In the area off Pori, complex effluents, including waste from a titanium dioxide plant, evidently caused serious harm to the coastal fish community. Catches of certain species, such as whitefish, by local fishermen decreased strongly up to the beginning of the 1980s [*397*]. It is not known, whether the situation has improved or not, as follow-up studies are lacking. Although the occurrence of local toxic contamination should be suspected in the Eastern Gulf of Finland and in coastal areas of northeastern Estonia, there is no information on effects on fish.

In Latvia and Lithuania, a long-term monitoring programme was started in the Daugava Estuary, downstream from the large city of Riga, and in the Nemunas Estuary in the Curonian Lagoon. No significant toxic responses have so far been detected, and analyses of organic contaminants did not detect any pronouncedly higher levels in stationary fish compared to the unpolluted Estonian Moon Sound reference area [*595*]. The Daugava and Nemunas estuaries are both eutrophic, and it has been suggested that the high level of primary production counteracts the contamination of fish.

Potential sites for local toxic pollution also occur along the Kaliningrad, Polish, German and Danish coasts. However, no studies have been made that permit interpretations of the occurrence of toxic influences on fish.

6.2.5.2 Influence of local eutrophication

Local eutrophication is a problem mainly in the southern parts of the Baltic Sea, but similar influences may be seen at pulp mills in the Gulf of Bothnia [*206,490*]. Fish communities are disrupted, resulting in a strong dominance by cyprinids and sometimes also ruffe (*Gymnocephalus cernua*). Recent mill improvements seem to allow at least partial restoration of the communities [*593*].

Previous studies on the fish community inhabiting the coastal waters off Helsinki [*41*] documented significant impacts of eutrophication. Cyprinids and ruffe predominated strongly in areas exposed to sewage effluents. Improved treatment facilities have resulted in clearly visible improvements, and the fish community has been restored to a more normal species distribution.

The eastern part of the Gulf of Finland is considered to be threatened by a wide variety of anthropogenic impacts [*357*]. The fish community in the Koporskaya Bight, receiving heated effluents from a nuclear power plant, has been changed towards more warm-water adapted fish such as pike perch, perch, cyprinids and sticklebacks, and the diversity of the fish fauna has been reduced [*582*]. Today, three-spined and nine-spined sticklebacks constitute more than 80 % of the total fish biomass. The combination of a high nutrient load and excessive heat was considered to produce a rapid eutrophication of the area. Similar changes in fish communities have been seen at Swedish nuclear power plants, although they have not been as dramatic [*488*].

Some species, such as the salmonids and coregonids, are very sensitive to local pollution. Among those species spawning near the coast, whitefish has suffered very much from eutrophication. Catches in Estonia have decreased from about 200 t in the 1950s to less than 10 t in 1993-94 [*319*].

The Latvian and Lithuanian monitoring programmes have documented fish communities clearly influenced by eutrophication [*595*]. In the Curonian Lagoon, the abundance of cyprinids, such as roach, silver bream and bream, is extremely high compared to most other coastal areas. The long-term development, which was described in the HELCOM coastal assessment [*226*], indicates a progressive eutrophication, reducing the stocks of many commercial species. A similar development was documented in Puck Bight [*623*]. Stationary as well as migratory fish evidently lost their recruitment areas due to changes in the benthic vegetation, more frequent oxygen deficits and the damming of some streams entering the bight. Whitefish disappeared totally. They were, however, reintroduced in 1993 and are now present again. Pike, formerly common in this area, suffered very much from the installation of dams, which have stopped their spawning migrations in the most important rivers. At the end of the 1980s, the fish community was dominated by one single species, the three-spined stickleback. A period of improved conditions for some species has been noted in these Polish coastal waters as an effect of increased sewage treatment. Since 1993, garfish has commonly been observed again on its former spawning grounds. However, at the same time, pipefish (*Sygnathus typhle*) has completely disappeared.

Eutrophication is considered to be the main local pollution problem in Danish coastal waters. Long-term monitoring in, e.g., Århus Bight, has documented poor oxygen conditions and a decline of the benthic fauna, down to a very low biomass at the end of the 1980s. The recruitment areas for flatfish species such as plaice evidently were destroyed, as the numbers caught in trawl samplings were very low in 1991. Counter-measures have resulted in pronounced reductions in nutrient discharges during recent years. Fish recruitment habitats seemed to be restored as young flatfish catches condiderably increased during 1992 and 1993.

Plaice recruitment in the Kattegat and Belt Sea areas is strongly correlated with hydrographic conditions, mainly due to a varying influence by the Baltic Proper [*55*]. However, frequently occurring oxygen deficits and long-term changes in benthic substrates are also considered to be important in localised areas. Negative trends in young plaice abundance have been documented in, e.g., the eutrophic Køge Bight. Flounder, however, seem to be favoured by the changed substrate conditions accompanying eutrophication [*59*].

6.2.5.3 Influence of basin-wide toxic contamination

Contaminant monitoring has documented a continuous decline in the concentrations of well-known toxic chemicals such as PCBs and DDTs [*80*]. Although there may be unknown new members of the family of hazardous substances accumulating in the Baltic Sea, the general contamination situation appears to have improved during the last 10-20 years.

Toxic effects on coastal fish have so far been documented only at localised polluted sites. The risk of a general increase in the load of hazardous substances to the Baltic Sea, also affecting coastal areas, is, however, still a real problem. A toxic influence on the reproduction of the open-sea species salmon and cod, called the *M-74* syndrome in salmon, has been suggested, although consistent evidence is still lacking [*65*].

6.2.5.4 Influence of basin-wide eutrophication

Large-scale eutrophication is a slow process, and effects on coastal fish communities will take time to be clearly manifested. Data for retrospective comparisons, however, are very sparse. One long-term data series is available from annual fish monitoring, which was started in 1962 in a Swedish coastal reference area, the Kvädöffjärden Bight, on the coast of the Baltic Proper. *Secchi* depth recordings, made during the test fishing, have documented a decrease in visibility from about 8 m in 1962 to about 4 m at the beginning of the 1980s. Since then, *Secchi* depths have changed very little. The observations correlate well with changes in plant-nutrient concentrations in the open sea areas and, consequently, represent the general development in this part of the Baltic Sea, more than local land-based influences. Some responses have been seen in the fish community, similar to observations made in inland waters undergoing eutrophication. Generally, fish biomass increased mainly as an effect of higher cyprinid abundance. This development was most rapid during the 1970s and the early 1980s.

6.2.6 Fish in monitoring

Coastal fish populations are not monitored as extensively as open sea commercial stocks. However, fish monitoring with an environmental objective was introduced some years ago in a joint Swedish, Finnish, Estonian, Latvian and Lithuanian programme [*595*]. Coastal reference areas are monitored as well as sites with different environmental impacts. The complex pollution from the Daugava and Nemunas rivers is studied, and pulp mill and nuclear power plant programmes utilise the reference data for comparisons.

Integrated monitoring of toxic responses to contaminants has been developed for sentinel species of fish, using biomarkers on low levels of organization coupled to ecological variables at the organism and population levels. This monitoring strategy has recently been introduced in guidelines for pulp-mill-effluent control in Canada and Sweden. Perch and viviparous blenny are suggested as suitable sentinels for conditions in the Baltic Sea, and may also be recommended for the monitoring of contaminants.

Monitoring data can also be provided as a by-product of fisheries statistics, and samples are often collected from commercial catches. Useful examples, demonstrating community responses to eutrophication, can be found in the Gulf of Riga, Curonian Lagoon and in the Puck Bight. Data from commercial fishing, however, are very sensitive to a number of factors, including changes in fisheries regulations, new techniques, unreliable catch reports, etc. More scientific monitoring systems are thus needed, especially if trends are to be detected with an acceptable level of precision.

6.2.7 Conclusions

This review verifies the conclusions made in the HELCOM coastal assessment [*226*], that there are evident impacts on coastal fish mainly from local or regional pollution. Fish monitoring, however, has still not been introduced at more than a few polluted sites, and fish are subordinate in joint monitoring within HELCOM. The recent development towards integrated systems for contaminants monitoring, however, is internationally recommended, and is now being discussed in the planning of the new HELCOM coastal monitoring programme. Fish are the most suitable organisms for such monitoring, and the data provided for this review show that suitable species occur for contaminants monitoring and biological effects studies in all coastal systems of the Baltic Sea of future interest for pollution control.

6.3 DISEASES AND PARASITES OF BALTIC FISH

6.3.1 Introduction

The information in this sub-chapter is based on the 1995 report of the ICES Working Group on Pathology and Diseases of Marine Organisms (WGPDMO), the results of the BMB/ICES Symposium "Diseases and parasites of flounder (*Platichthys flesus*) in the Baltic Sea", unpublished results from the BMB/ICES seagoing workshop "Fish diseases and parasites in the Baltic Sea" and information from the literature. This information was compiled by *T. Lang*[1], Germany, and the text has been reviewed by the ACFM and the Advisory Committee on the Marine Environment (ACME).

6.3.2 Available data

This chapter will provide a brief description of the prevalence and geographical distribution of selected fish diseases and parasites in the Baltic Sea, based mainly on data from the past twenty years. Information is provided for the major Baltic Sea fish species on which most research and monitoring activities have been conducted, namely, flounder (*Platichthys flesus*), cod (*Gadus morhua*), herring (*Clupea harengus*) and, recently, Atlantic salmon (*Salmo salar*). However, other species, affected by significant diseases and/or parasites, are also included in this chapter.

In addition to the grossly visible fish diseases/parasites that have been recommended by ICES for fish diseases surveys [*268*] and widely used for biological effects-monitoring purposes, other diseases/parasites that have previously received considerable attention due to their possible link to marine contamination and/or their suspected impact on mortality and fish-stock size are considered here. A summary of the diseases and parasites is given in Table 6.3.

The most current data on the prevalence and spatial distribution of diseases/parasites of flounder, cod and herring are those obtained during the BMB/ICES Workshop "Fish diseases and parasites in the Baltic Sea" which was held from 25 November to 8 December 1994 on board the rv "*Walther Herwig III*". Practical work was conducted on a transect from the Mecklenburg Bight to the western Gulf of Finland, representing the largest area in the Baltic Sea ever studied for this purpose in a narrow time-window and using identical methods. Although the results of the workshop have not yet been published, some of them have been included here [*34*].

Additional information on diseases/parasites of the Baltic flounder was presented at the BMB/ICES symposium "Diseases and parasites of the flounder (*Platichthys flesus*) in the Baltic Sea", held on 27-29 October 1994 at Åbo Academy University, Turku, Finland [*35*]. In the following sections, information is given for individual Baltic fish species on their major diseases and parasites, including a compilation of selected key publications.

6.3.2.1 Flounder (*Platichthys flesus*)

Information on larger-scale geographical distributions of externally visible diseases of Baltic flounder used for monitoring purposes, including data on temporal trends, has been provided by different authors [*141,142, 384,680*]. Other published data cover more restricted areas, including the Kattegat and the Skagerrak, and certain time periods [*147,304, 432,469,470,568,647,701,703*].

A current review has been provided on the unique parasitic fauna of the Baltic flounder, which, due to the strong salinity gradient from the western to the eastern/northern parts of the Baltic Sea, consists of both marine and freshwater species [*162*]. Other publications were based on more restricted areas [*431,467,568, 667,679*]. Of these publications, only a few [*467,667,568*] contain information on temporal trends.

The viral lymphocystis disease is by far the most prevalent externally visible disease of Baltic flounder. There is general consensus about the aetiology of lymphocystis, the infectious agent of which is a virus classified as belonging to the family of Iridoviridae [*381*].

The prevalences recorded in November/December 1994 during the BMB/ICES Workshop, according to single sampling sites, were in the range of 5-38 % in flounder ≥20 cm, with a decreasing trend from the western to the

[1] see ANNEX 11.3 for addresses of authors

eastern stations [*34*]. The high prevalences, detected at the westernmost stations, were well in accordance with data from 1993 recorded in the same area [*384*]. A clear increase in the prevalence of lymphocystis has occurred in the southwestern parts of the Baltic Sea, i.e., southwest and southeast of Bornholm, during 1986-93, the reasons for which have not yet been elucidated [*384*].

There is evidence that the prevalence of lymphocystis in Baltic flounder is influenced by both sex and size/age of the fish (males and intermediate size groups are more frequently affected than females and small/large size groups, respectively) [*384*], and by seasonal effects [*304,680*].

The prevalence of the bacterial skin-ulcer disease of flounder ranged from 0-12 % at the different workshop stations, with some indication

Host species	Disease / parasite	Geography	max. prev. (%)	Ref. no.	Aetiology
Flounder	lymphocystis	whole BS, W>E	38	[34]	Iridovirus
(Platichthys flesus)	skin ulcer disease	whole BS, E>W	12	[34]	bacterial infection
	fin rot/erosion	whole BS	<1	[34]	bacterial infection
	skeletal deformities	whole BS	<1	[34]	multiple
	liver nodules >2mm	whole BS, E>W	3	[34]	org. contaminants (?)
Cod	skin ulcer disease	whole BS, W>E	4	[34]	bacterial
(Gadus morhua)	skeletal deformities	whole BS, Polish coast	4	[34]	multiple
	X-cell disease	western BS	1	[34]	parasite (?)
	Cryptocotyle lingua	western BS	20	[34]	digenean parasite
	Lernaeocera branchialis	western BS	3	[34]	parasitic copepod
	yolk parasite	western BS	30	[530]	unknown prot. endoparasite
	M-74 (?)				
Herring	*Anisakis sp.*	western BS	40	[385]	parasitic nematode
(Clupea harengus)	*Ichthyophonus sp.*	western BS	5	[380]	parasitic fungus
	lymphocystis	whole BS, E>W	4	[34]	Iridovirus
	Eimeria sardinae	whole BS	80	[669]	sporozoan parasite
Atlantic salmon	*M-74* syndrome	whole BS	90	[270]	vitamin deficiency (?)
(Salmo salar)	in yolk-sac larvae				
Four-horned sculpin	skeletal deformities	Gulf of Bothnia	40	[61]	industrial discharge (heavy
(Myox. quadricornis)					metals, arsenic)
Dab	lymphocystis	Kattegat	14	[457]	Iridovirus, O_2-deficiency
(Limanda limanda)	epidermal papilloma	Kattegat	4	[34]	Adenovirus (?), O_2-deficiency
	skin ulcer disease	Kattegat	1	[34]	bacterial, O_2-deficiency
Turbot	yolk parasite	western BS	40	[530]	unknown prot. endoparasite
(Scophth. maximus)					
4-bearded rockling	skin ulcer disease	Polish waters	>50	[384]	bacterial (?)
(Enchelyopus cimbrius)					
Smelt	jaw erosion syndrome	Finnish west coast	12	[704]	industrial, urban pollution (?)
(Osmerus eperlanus)					
Eel	*Anguillicola crassus*	western BS, offshore	16	[199]	parasitic nematode
(Anguilla anguilla)		western BS, coastal	59	[563]	
		Swedish coast	60	[254]	thermal discharge (?)
Perch	fin erosion	Swedish east coast	>60	[417]	pulp mill effluents
(Perca fluviatilis)	gill cover deformities	Swedish east coast	>20	[417]	pulp mill effluents
	Diplostomum spp.	Swedish east coast	77–100	[255]	digenean parasite, thermal effects
Ruffe	fin erosion	Swedish east coast	>60	[417]	pulp mill effluents
(Gymnoceph. cernua)					
Pike	skeletal deformities	Swedish east coast	>30	[418]	pulp mill effuents
(Esox lucius)					
Roach	reproductive disorders	Finnish west coast	60	[704]	myxosporidean parasite (?)
(Rutilus rutilus)					

***Table 6.3** Grossly visible diseases/parasites of Baltic Sea fish species, with information on their geographical distribution, maximum prevalence and aetiology (BS - Baltic Sea, W - western part, E - eastern part)*

of increasing prevalences from the western to the eastern sampling sites [*34*]. Various bacteria considered to be involved in the aetiology of the ulcer disease of Baltic flounder have been isolated [*132,354,700*]. However, particularly 'atypical' *Aeromonas salmonicida* was consistently isolated from diseased Baltic Sea flounder [*700*], and it is therefore likely, that this bacterium is the main cause of the skin-ulcer disease. Acute as well as healed skin ulcers are more prevalent in male than in female flounder [*34,701*]. Seasonal changes in the prevalence were reported [*304,701*]. No temporal trends in the prevalence of acute/healing skin ulcers in flounder from the southwestern parts of the Baltic Sea, i.e., southwest and southeast of Bornholm, were found in the period 1986-93 [*384*].

Studies on the occurrence of neoplastic liver lesions in Baltic flounder have only recently started [*94,114,384*]. During the BMB/ICES Workshop in November/December 1994, the prevalences of liver nodules >2 mm in diameter (which, according to ICES standard methodologies for fish disease surveys [*268*], is recorded as a macroscopic indicator of the occurrence of neoplastic liver lesions) were found to be generally low (0-3 %), with a tendency for slightly increased prevalences at the easternmost stations [*34*]. These data are in accordance with a mean prevalence of 3 % in flounder of the same size group (≥20 cm) from Finnish coastal waters [*703*]. However, for flounder ≥30 cm, a prevalence of 10.1 % was calculated, which is very high compared to other Baltic Sea areas [*384*].

Histological confirmation of the liver nodules collected during the BMB/ICES Workshop showed, that tumours were rare and that the other nodules consisted either of unspecific storage cell changes or putative pre-neoplastic lesions, mainly clear cell foci or eosinophilic foci [*34*]. Other externally visible flounder diseases, such as acute/healed fin rot/erosion and skeletal deformities, were only recorded at low prevalences (<1 %), without revealing any spatial trends [*34*].

6.3.2.2 Cod (*Gadus morhua*)

Information published during the past two decades on prevalences and spatial/temporal trends of externally visible diseases and parasites of Baltic cod, including in the Kattegat and Skagerrak, are available from many sources [e.g.,*119,139,141-143,147,288,289, 378,383,386,422,688*]. Parasitological data, mainly on parasitic helminths, were also presented [*110,160,161,182,377,468,474,655, 673,688*].

The most prevalent externally visible disease of Baltic cod is the bacterial skin-ulcer disease (ulcus-syndrome) [*288*]. Although there is evidence that the conspicuous disease signs (open, red ulcers affecting the skin and, occasionally, even the underlying musculature) are caused by bacterial infections, the primary cause may be different. For example, Danish studies on ulcerated cod occurring at high prevalences (>40 %) around Bornholm in 1982 revealed, that these lesions were most likely primarily due to mechanical damage of the skin of specimens which escaped from fishing nets. Thereafter, the open lesions were probably invaded by pathogenic bacteria, thus leading to the occurrence of the typical skin ulcers. However, there is also evidence from the literature that pollution might have been involved in the aetiology of the disease in cod from the Danish Belt Sea in the late 1970s [*119,288*]. Prevalences in cod ≥20 cm total length, recorded in November/December 1994 during the BMB/ICES Workshop, were in the range of 0.0-4.2 % in relation to single sampling sites, with the highest prevalences in Polish waters [*34*]. These results are well in accordance with data from the southwestern part of the Baltic Sea for the period 1980-88 [*139*] and for December 1993 [*141*], regarding both the range of prevalences recorded and the identification of regions with the highest prevalences. However, considerably higher prevalences have been found at certain sites, such as 22 % reported for 1977 from polluted sites in the Danish Belt Sea [*289*], 14.1 % for the Gdansk Bight in 1983 [*688*] and 15.8 % for the Polish fishery zone in 1986 [*147*]. From the literature, there is no clear indication of consistent temporal trends in the disease. However, it appears that the prevalences generally were higher in the 1980s compared to the 1990s.

The second most common externally visible diseases of Baltic cod are skeletal deformities, such as vertebral compression, lordosis/scoliosis and deformation of the head skeleton, which have been known for a long time to occur occasionally in certain areas in the Baltic Sea at very high prevalences.

Factors involved in the aetiology of these lesions are manifold. Unfavourable hydrographic conditions during early ontogenesis, malnutrition, parasitic infestation, effects of heavy metals and differences in migratory behaviour between healthy and deformed fish are discussed as possible causes of elevated prevalences in wild fish [*62,63,140,471,472, 383*]. Significantly higher cadmium contents were detected in organs of cod with skeletal deformities compared to healthy cod [*383*]. However, these data did not permit any conclusions about the cause-effect relationship between higher cadmium residues and the occurrence of skeletal deformities.

During the BMB/ICES Workshop, the prevalence of externally visible skeletal deformities recorded in cod ≤20 cm was 0.0-4.0 %, with the highest prevalence in the Gdansk Bight and the lowest prevalence at the western- and easternmost stations [*34*]. These prevalences are in accordance with other published data for cod in the southwestern parts of the Baltic Sea [*139,141,142,383*]. Prevalences of <1 % were reported in cod from the Polish fishery zone [*422*]. However, from historic data, there is information, that skeletal deformities in Baltic cod may occur at much higher prevalences, i.e., >50 % in the Sound [*429*] and ≤75 % in the Gulf of Riga [*72*]. From the literature, there is no information on consistent trends as regards spatial and temporal distribution patterns.

The third grossly visible disease, recommended for monitoring purposes in Baltic cod, is the *X-cell* disease (pseudobranchial pseudotumours) causing proliferations in the pseudobranches and adjacent tissue. The epidemiology and histopathology of this disease have been described [*143,689*] and additional data were presented [*139,141*]. Although the disease has been recorded in a large variety of fish species during the past ten years, and its histopathology has been described thoroughly, its aetiology has so far not been resolved conclusively. However, there are indications that the disease is caused by proliferative parasitic protozoans, possibly Amoebae [*690*].

During the BMB/ICES Workshop in November/December 1994, the *X-cell* disease was only recorded at the two westernmost stations, with a prevalence of <1 % in cod ≥20 cm [*34*]. These data again correspond to previously published information, indicating that the disease is very rare and restricted to the southwestern parts of the Baltic Sea.

The infestation of Baltic cod with conspicuous externally visible parasites, the copepod *Lernaeocera branchialis* in the gill chamber and encysted metacercariae of the digenean *Cryptocotyle lingua* in the skin, is restricted to the southwestern parts of the Baltic Sea [*34,378*]. During the BMB/ICES Workshop, maximum prevalences in relation to sampling sites were 20.0 % and 2.9 % for *Cryptocotyle lingua* and *Lernaeocera branchialis,* respectively. Regarding the spatial distribution of cod infested by *Lernaeocera branchialis,* the workshop results confirm findings based on data from 1983-88 [*378*]. For cod from the southwestern Baltic Sea (ICES Sub-division 22), a prevalence of 4.9 % was calculated, with some annual fluctuations in the prevalence, without, however, revealing a temporal trend [*378*].

New diseases of Baltic cod causing some concern are the occurrence of a yet unknown protistan endoparasite in the yolk of cod embryos and larvae [531], and certain indications of a possible occurrence of a *M-74*-like syndrome, similar to the one known to have a deleterious impact on reproduction of the Baltic stock of Atlantic salmon (*Salmo salar*) [270].

6.3.2.3 Herring (*Clupea harengus*)

The diseases and parasites of Baltic herring, that have been the subject of most scientific activities during the past twenty years, are the infestation by larvae of the parasitic nematode *Anisakis* sp. and, since 1991, the epizootic caused by the parasitic fungus *Ichthyophonus* sp. Data on the prevalence and spatial distribution of *Anisakis* sp. in Baltic Sea herring have been provided, for example, by several authors [181,359,360,385,426,567,664,691]. From these data, there is evidence that the infestation mainly affects the western spring-spawning herring stock which enters the Baltic Sea for spawning after leaving its feeding grounds in the Kattegat, Skagerrak and the North Sea. This migration to the Baltic Sea usually starts in late autumn and, having spawned, the herring leave the Baltic Sea in spring. Since the major spawning areas are located in the southwestern parts of the Baltic Sea, infested herring can be mainly found west of Bornholm. However, at least some of the western spring-spawners migrate further east and have been recorded off the Polish coast [181] and the Latvian coast [664].

The infestation with larval *Anisakis* sp. is restricted to this particular herring stock, and specimens, belonging to the 'real' Baltic herring stocks, which stay in the Baltic Sea throughout their whole life, are infested only very sporadically. Therefore, it has been concluded that the infestation of herring takes place on feeding grounds outside the Baltic Sea by feeding on infested intermediate hosts of the parasite, probably euphausids.

Mean prevalences in herring in the size range 20-27 cm total length, sampled west of Bornholm (ICES Sub-divisions 22 and 24) in December 1987 and 1988, were in the range of 30-45 %, whereas prevalences in areas east of Bornholm (ICES Sub-divisions 25 and 26) were between 0.3 and 0.7 % [385]. The same authors demonstrated a clear positive relationship between the length of the herring and both the prevalence and intensity (number of nematode larvae per infested herring) of the infestation in samples from the Kiel fishery market covering the period 1973-88. The smallest infested herring were 19 cm, a steep increase occurred in the size group 22-29 cm, and herring larger than 30 cm were infested to 100 %. Based on the same data set, an analysis of temporal trends in the prevalence was conducted. However, the results did not reveal a clear upward or downward trend. Parasitic nematode species in Baltic herring other than *Anisakis* sp. have been also reported [161,169,622,668,673].

The *Ichthyophonus* sp. epizootic in Baltic Sea herring was noted for the first time in summer 1991, when mass mortalities occurred at the Swedish west coast, causing great concern by the public, the fishery industry, and the scientific community as well. This concern was mainly due to the well-known effects of epizootics in the 1950s that affected herring stocks in North American Atlantic coastal waters. These epizootics caused a drastic stock reduction, followed by considerable economic losses for the fishing industry [619]. From experimental studies, carried out following the first epizootics, summarized in [619], there was evidence that the infestation was lethal for herring, in contrast to many other fish species, within a relatively short period after infestation, thus explaining the massive stock reduction recorded.

Due to the fear in the Scandinavian countries, that the European herring stocks may experience a similar decline, studies were started nearly immediately after the observation of mortalities in 1991 in order to obtain baseline data on the spatial distribution of the epizootic and the prevalences in herring stocks in the Baltic Sea as well as in the North Sea. For the southwestern Baltic Sea, it was demonstrated very quickly that herring infested by *Ichthyophonus* sp. belonged to the western spring-spawning stock, the same stock which is characterised by high prevalences of *Anisakis* sp. larvae [380]. In areas, dominated by herring of the 'real' Baltic stocks, the infestation was at a low level, for example, since 1991 in herring of the Gulf of Finland and the Gulf of Riga [668].

Mean prevalences of grossly visible disease signs in the heart of herring ≥20 cm total length, recorded in December 1991 west of Bornholm, were 2.3 % (ICES Sub-division 22) and 5.3 % (ICES Sub-division 24). However, the prevalences were considerably higher in single catches, revealing a patchy distribution of infested herring. In contrast, no infested herring were recorded east of Bornholm (ICES Sub-divisions 25 and 26) [380]. In the following year, the prevalences had dropped to 0 % (ICES Sub-division 22) and 0.5 % (ICES Sub-division 24). At present, there is a general consensus that the *Ichthyophonus* epizootic in Baltic Sea herring is over, and that the prevalences have approached natural background levels [270]. There is evidence [468,668], that other Baltic fish species (sprat, cod) are infested at low prevalences. However, there is no indication of increased mortalities in these species.

Other significant diseases of Baltic herring include lymphocystis [28,34,147], and a bacterial infection by ***Pseudomonas anguilliseptica***, associated with haemorrhagic eye lesions, possibly involved in the transmission of the infection, which has been known for some time to occur in cultured Baltic salmonids between trout mariculture facilities in Finnish waters [423]. Additionally, the increased prevalence of the infestation of herring and sprat from the Gulf of Finland and the Gulf of Riga with the sporozoan parasite *Eimeria sardinae* [669] has been observed.

6.3.2.4 Atlantic salmon (*Salmo salar*)

Apart from the *Ichthyophonus* epizootic in herring, the most spectacular disease occurring in Baltic Sea fish species is the *M-74* syndrome of Atlantic salmon, which continues to cause high mortalities (80-90 %, [270]) in yolk-sac fry obtained from wild salmon and artificially reared salmon for re-stocking purposes within Swedish and Finnish compensatory stock-enhancement programmes. The syndrome has been known since 1974, but massive mortalities (>50 %) have only occurred since 1992 [269].

Whilst in previous years direct effects of environmental contaminants on the reproductive success of adult wild salmon were discussed as possible causes of the *M-74* syndrome, current information indicates that a vitamin deficiency (vitamin B_1, thiamine), due to the feeding of the adults on clupeids with high thiaminase contents, might be the main causal factor [270]. For example, experimental studies have shown, that the addition of thiamine to the water reduces both the behavioural *M-74* characteristics and the mortality of salmon fry significantly. However, although this therapy seems to be promising for the enhancement programme, the wild Baltic salmon stock will probably still continue to suffer from *M-74* and natural reproduction will be seriously endangered.

6.3.2.5 Other diseases in other fish species

Some other significant diseases or parasites studied in different Baltic Sea fish species during previous years are

- externally visible diseases of dab (*Limanda limanda*) in the Kattegat and Skagerrak, and their association with oxygen deficiencies [457],
- high prevalences (>50 %) of skin ulcer disease in four-bearded rockling (*Enchelyopus cimbrius*) in deep-water regions off the Polish coast,

- the infestation of Baltic cod and turbot (*Scophthalmus maximus*) embryos and yolk-sac larvae with a yet unknown protistan endoparasite [530],
- elevated prevalences of skeletal deformities in four-horned sculpin (*Myoxocephalus quadricornis*) from polluted Swedish coastal waters [61,64],
- parasitic helminths in *Salmo trutta* [109,111],
- other parasites [654],
- the infestation of Baltic eel (*Anguilla anguilla*) with the swimbladder nematode (*Anguillicola crassus*) [199,254,563],
- a jaw erosion syndrome affecting smelt (*Osmerus eperlanus*) mainly in polluted areas off the Finnish west coast [704],
- fin erosion of perch (*Perca fluviatilis*) and ruffe (*Gymnocephalus cernua*) [417], and skeletal deformities in northern pike (*Esox lucius*) [418] and perch [416] in Swedish coastal waters, due to effluents from pulp mill industries,
- the eyefluke *Diplostomum* spp. in perch (*Perca fluviatilis*) and roach (*Rutilus rutilus*) from warm-water effluents along the Swedish coast [255], and
- reproductive disorders in roach (*Rutilus rutilus*) in brackish waters on the Finnish coast [702], probably due to a myxosporidian infection (*Pleistophora mirandellae*).

6.3.3 Impact of anthropogenic activities on the prevalence and geographical distribution of fish diseases and parasites

From a number of studies, carried out in the Baltic Sea, there is strong indication of a link between anthropogenic environmental changes, e.g., industrial and urban pollution, eutrophication and thermal discharges, and the occurrence of elevated prevalences of certain fish diseases and parasites in certain areas. Some of these studies dealt with

- increased prevalence of the ulcus-syndrome in cod from polluted areas in the Danish Belt Sea [288,289],
- increased skeletal deformities in perch and pike in Swedish coastal areas affected by pulp-mill effluents [416,418],
- increased prevalence of skeletal deformities in four-horned sculpin in polluted areas of the Gulf of Bothnia [61],
- the occurrence of fin erosion in perch and ruffe in Swedish coastal areas affected by pulp mill effluents [417],
- increase in the prevalence of externally visible diseases of dab due to oxygen deficiency in the Kattegat [457],
- the occurrence of an elevated prevalence of a jaw-erosion syndrome in smelt on the Finnish west coast [704],
- the occurrence of liver neoplasia in flounder from polluted areas in the Gulf of Finland [94,114],
- changes in the parasite fauna of flounder in the Gdansk Bight due to effects of pollution on the abundance of intermediate and final parasite hosts [568], and
- increased prevalences of the eel parasite *Anguillicola crassus* in waters affected by thermal discharges [254].

The results of these studies support the assumption, that changes in the prevalence or intensity of diseases and parasites in wild fish can be used as an indicator of biological effects of contaminants and other anthropogenic activities, and that studies of fish diseases/parasites in wild fish should, therefore, be included in Baltic Sea (and other) monitoring and assessment programmes on changes of the quality of the marine environment.

6.3.4 Impact of fish diseases and parasites on fish stocks

There is little conclusive information regarding the impact of fish diseases or parasites on mortalities in Baltic Sea fish stocks. In only a few cases have mortalities, induced by specific diseases or environmental factors, been so obvious that the assumption seems justified that these conditions might have had a significant effect on stock size in the affected fish species. Examples are mortalities in garfish (*Belone belone*) at the south coast of Sweden due to gas super-saturation associated with thermal discharge effluents from a nuclear power plant, mortalities in perch (*Perca fluviatilis*) attributed to effects of pulp-mill effluents, and mortalities in western spring-spawning herring due to the *Ichthyophonus* sp. epidemic. The most recent case is the heavy mortalities in yolk-sac fry of the Baltic stock of Atlantic salmon (*Salmo salar*) due to the *M-74* syndrome.

The *Ichthyophonus* sp. epidemic in herring has so far been the only disease, affecting parts of the Baltic Sea, in which attempts have been made to estimate mortality due to the infestation and the resultant impact on the stock. However, attempts to quantify disease-induced mortalities in general suffer from a number of confounding factors, such as

- large fluctuations in fish stock sizes due to natural and anthropogenic causes other than diseases may mask and/or exceed disease-induced changes,
- methods employed in fish-stock assessments do not permit a distinction to be made between mortality due to specific diseases, natural factors, or fishing effort,
- the prevalence of a disease, on which mortality estimates are based, only gives information at the time of sampling, and does not provide important information from the past (e.g., number of fish that have already died) or for the future (number of fish that will become infected),
- the prevalence estimated from a sample may not reflect the true prevalence in a fish population or stock, since diseased fish may be over- or under-represented due to differences in the spatial distribution of infected and uninfected fish, and/or because the gear used selects for infected or uninfected fish, and
- the prevalence may not be an appropriate measure of mortality, at least as long as there is no clear evidence of how lethal a disease is under natural conditions and the length of the survival time of infected fish.

Because of these variables, it has so far not been possible to provide conclusive evidence for an *Ichthyophonus*-induced herring-stock decline nor to develop realistic models for mortality rates due to the infestation. From this experience with a disease condition, which has been very prevalent in the affected herring stocks, and which is considered to be highly lethal for infected individuals, it seems questionable whether disease-induced mortalities and associated effects on the population can be estimated by applying current methods of epidemiology and stock assessment.

NATURE CONSERVATION AND BIODIVERSITY

7

S. ANDRULEWICZ, 1993

The vascular plant Water Milfoil (Myriophyllum sp.) on the sandy bottom of the Puck Lagoon.

HELCOM

7.1 GENERAL CONSERVATION ASPECTS

7.1.1 History of nature conservation in the Baltic Sea Area

Activities of nature conservation in marine areas are of much more recent origin than in terrestrial and limnic environments. Even though environmental problems have been observed and discussed, it was not until recent decades that scientists and the public became concerned about marine environmental problems and aware of the fact that the marine environment is not indestructible. Bodies within the United Nations, e.g., UNEP and UNESCO, as well as the International Union for the Conservation of Nature (IUCN), recognised the need for coordinating and systematising work to combat the growing threats to marine environments. For example, a resolution from the 17th General Assembly of IUCN in Costa Rica 1988 states that the primary goal of marine conservation is *"...to provide for the protection, restoration, wise use, understanding and enjoyment of the marine heritage of the world in perpetuity through the creation of a global, representative system of marine protected areas and through the management in accordance with the principles of the World Conservation Strategy of human activities that use or affect the marine environment..."*.

The main goal is hence to prevent further deterioration of the marine environment, and especially of particularly valuable marine areas, even when used by man. Areas that are already deteriorated should, if possible, be restored. In all cases, wise management, including monitoring to follow trends, and using special protective measures when necessary, is essential.

Until the late 1980s, international work concentrated mainly on environmental issues. Nature conservation issues, *sencu strictu*, were considered as national concerns and were generally not coordinated between countries. However, interest in nature conservation issues within the Baltic Sea Area became gradually more pronounced. Bodies and organisations like the "Nordic Council", "Coalition Clean Baltic" (CCB), "Baltic Marine Biologists" (BMB) and "World Wide Fund for Nature" (WWF) became increasingly involved in nature conservation issues related to Baltic Sea ecosystem and species.

This chapter has been prepared by *Dieter Boedecker*[1], Germany, and *Ola Jennersten*, WWF, and has been reviewed by the Working Group on Nature Conservation and Biodiversity (EC NATURE).

7.1.2 HELCOM, EC NATURE and Article 15

Since the signing of the Helsinki Convention in 1974, HELCOM has become the main forum for handling environmental issues in the Baltic Sea Area. During the first 15 years, work was focused on environmental problems. The environmental efforts, however, could not stop the ongoing loss of habitats and biotopes, and hence the permanent threat to the biodiversity in many parts of the Baltic Sea Area. However, new thinking strongly suggested that other measures were necessary in order to protect biodiversity and ecologically important areas and processes, and thus a revision of the Convention was called for.

As a result of recommendations from the Ministerial Meeting at Ronneby, Sweden, in September 1990, and from the First International Seminar on Nature Conservation and Biodiversity in the Baltic Region at Runö, Sweden, in May 1991, a new article was drafted for the revised Helsinki Convention of April 1992. Article 15 of this new Convention states: *"The Contracting Parties shall individually and jointly take all appropriate measures with respect to the Baltic Sea Area and its coastal ecosystems influenced by the Baltic Sea to conserve natural habitats and biological diversity and to protect ecological processes. Such measures shall also be taken in order to ensure the sustainable use of natural resources within the Baltic Sea Area. To this end the Contracting Parties shall aim at adopting subsequent instruments containing appropriate guidelines and criteria"*.

Hence, not only the Baltic Sea itself is included within the frame of HELCOM, but also terrestrial biotopes of the Baltic Sea coasts, as far as nature conservation aspects are concerned. The protection of habitats, species and ecological processes, both in marine and coastal areas, are important in the improvement of the environmental situation of the Baltic Sea Area in future. With "Article 15 - Nature Conservation and Biodiversity", environmental work in the Baltic Sea was elevated to a different level and has since included nature conservation as an important tool.

7.2 NATURE CONSERVATION STATUS OF TAXONOMIC GROUPS

7.2.1 Introduction

This sub-chapter briefly summarises the current nature conservation status for a number of important species in the marine and brackish areas, including present distribution and abundance and the most important factors leading to a deterioration of the conditions for the species and the threats to their future survival. A series of important historical, physical and chemical factors determine the composition of species in the Baltic Sea. Because the Baltic Sea is geologically very young, a very limited brackish-water flora and fauna has developed. Many of its inhabitants entered after the last glaciation, either through the very narrow and shallow connection to the North Sea or out of rivers and periglacial water systems. Due to the salinity gradient from the Skagerrak to the innermost parts of the Baltic Sea, salt-water animals and plants occurring in the Kattegat have been gradually replaced by brackish water and even fresh-water species as one moves inward along the gradient. The number of species of marine animals >1mm also changes from about 1,500 in the Skagerrak, e.g., *Saduria entomon* and *Cardium hauniense*, to 800 in the Kattegat, 150 in the Southern Baltic Proper, 80 in the central parts of the Baltic Proper, 50 in the Bothnian Sea, and to still fewer numbers in the northernmost parts of the Bothnian Bay. The biodiversity of the Baltic Sea is characterised by few species, but many individuals of each species. Where the species live at the limit of what they can tolerate with respect to salinity, they may be extra sensitive to other stress factors and disturbances.

An interesting result of the salinity gradient is the phenotypic differences found among some species which live both in the Kattegat/Skagerrak and in the Baltic Sea. For example, the shell size of adult common mussels, *Mytilus edulis*, decreases inward in the Baltic Sea, and certain species of algae, occurring in the Baltic Proper as well as in the northern Kattegat, cannot be interchanged. An important part of the Baltic marine communities are the glacial relicts, e.g., the four-horned sculpin *Myoxocephalus quadricornis*, the amphipod *Monoporeia affinis*, the isopod *Saduria entomon*, the mysid *Mysis relicta* and the ringed seal *Phoca hispida botnica*. One can even talk about a relict food web including all these species. A recent drastic decline in the abundance of several relict species in the Gulfs of Riga and Finland needs investigation.

7.2.2 Description of the present situation including threats

7.2.2.1 Benthic vegetation

As already mentioned, in the brackish water of the Baltic Sea, fresh- and salt-water plants exist side by side, e.g., *Phragmitis* and *Fucus*. Being a salt-water algae, bladder wrack *Fucus vesiculosus* is affected by decreasing salinity, which results in the Baltic Proper being the northern border for this species. It is also affected by competition with other species of algae.

This competitive situation increases with higher nutrient levels, causing an increase in phytoplankton, limiting light penetration, and the growth of epiphytes on, e.g., *Fucus* and *Zostera*. Therefore, *Fucus* and *Zostera* can only grow in shallower water. Green algae like *Cladophora* and *Enteromorpha* are also positively affected by eutrophication, which results in intense competition with *Fucus* for space in shallow water. Very likely for these reasons, the distribution of *Fucus* has decreased. Except for the Polish coastal zone, at some locations, however, during recent years a decrease in nutrient input has caused *Fucus* and *Zostera* to return to former habitats in deep regions from where they disappeared 10 to 15 years ago.

7.2.2.2. Invertebrates

Among the Baltic Sea invertebrate fauna three groups dominate, i.e., molluscs, polychaetes and crustaceans. In the Baltic Proper, four species are particularly common and often constitute almost 100 % of the benthic-fauna biomass, the bivalves *Mytilus edulis* and *Macoma baltica*, the amphipod *Pontoporeia affinis* and the isopod *Saduria (Mesidotea) entomon*. *Macoma* constitutes the main biomass in the Bothnian Sea, while *Pontoporeia* is most common in the Bothnian Bay, even though its total biomass is extremely low. The majority of species are found in shallow areas. Deeper areas are indeed very poor in species, i.e., less than 10 species in the Baltic Proper and not more than 1-2 species in the Bothnian Bay.

Oxygen deficiency is caused by the decomposition of sedimenting matter originating from pulp and paper factories, other sewage discharges and algal deposition. This is a main threat to the fauna of deep bottoms below the halocline. However, increased primary production in shallow water results in an increase of the production of the benthic fauna.

7.2.2.3 Fish

Along the coasts and within the archipelagos of the Baltic Proper, the fish fauna are dominated by fresh-water species like pike, *Esox lucius*, perch, *Perca fluviatilis*, burbot, *Lota lota*, and roach, *Rutilus rutilus*. Several salt-water species can also be found along the coasts, e.g., stickleback, *Pungitius pungitius*, gobies, *Pomatoschistus microps* and *P. minutus*, and pipe fish, *Nerophis ophidion*. Like other taxa, the number of fish species decreases northwards. Herring, *Clupea harengus*, is the most common pelagic fish species in the Baltic Proper, constituting about 40 % of the fish biomass. Cod, *Gadus morhua*, and sprat, *Sprattus sprattus*, are also common species in offshore waters, and contribute together about the same percentage to the total fish biomass as herring does. These three species are the most important commercial fish species in the Baltic Sea. Salmon, *Salmo salar*, sea trout, *Salmo trutta*, eel, *Anguilla anguilla*, pike and bleak, *Coregonus albula*, are also commercially important. Examples of glacial relict fish species are the four-horned sculpin, *Myoxocephalus quadricornis*, the sea scorpion, *Myoxocephalus scorpius*, and the sea snail, *Liparis liparis*.

Salmon, Salmo salar - The salmon reproduces high up in rivers, and young fishes spend 1-3 years in fresh water before migrating to the sea. After living at sea for 2-4 years, maturing adults return to their home stream. This behaviour results in several local stocks of salmon, each being adaptable to conditions at the hatching site.

Numerous salmon stocks in the countries bordering the Baltic Sea have become extinct during recent decades. Impassable hydro-power dams, pollution, poaching and over fishing of the Baltic Wild Salmon at sea, or combinations of these factors, are the most important causes

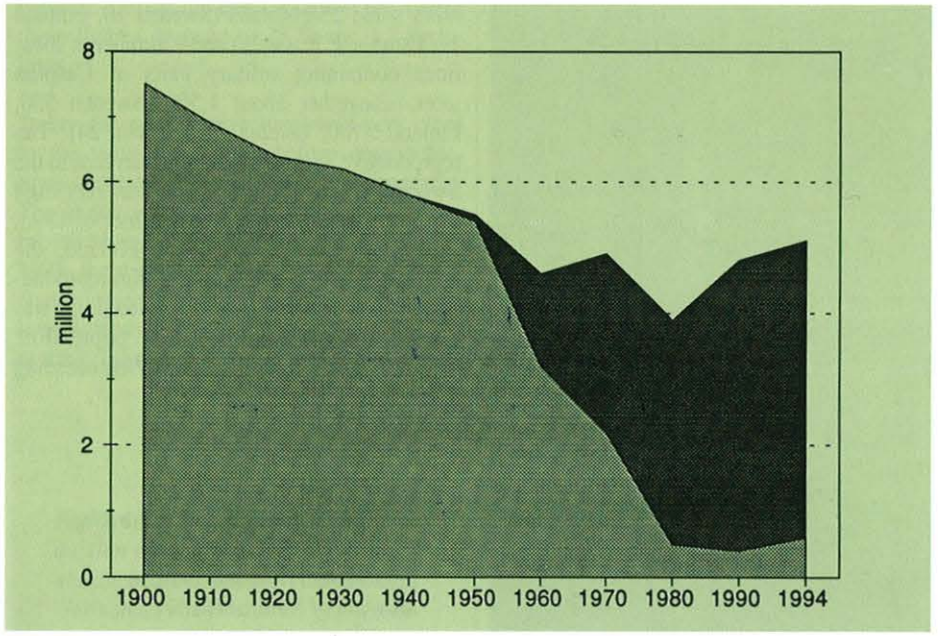


Fig. 7.1 Estimated production of salmon fry in the Baltic Sea, 1900-94 (light area - wild salmon, dark area - artificially reared salmon; data from the Salmon Research Institute, Älvkarleby, Sweden)

[1] see ANNEX 11.3 for addresses of authors

and permanent threats. Rearing and release of fry has been the method used to compensate the loss of natural spawning areas. Today, most of the rivers, previously used as spawning grounds by salmon and sea trout, are regulated and no longer passable. As a result, a dramatic decrease in Baltic Wild Salmon is observed, and less than 10 % of Baltic salmon is a natural, wild population (Fig. 7.1). A serious reproductive disorder causing high mortality among reared fry in hatcheries was described in 1974 as *M-74* syndrome. High number of sprat, high winter temperatures and high incidence of *M-74* are correlated. During warm winters, salmons eat more (higher metabolic rates), and especially during years of high sprat production, sprat forms major part of the salmon's diet. Sprat is a fatty fish with a high concentration of certain hazardous substances. One plausible hypothesis explaining the disease is related to food intake. From 1992, the number of one-year-old salmon has decreased drastically in rivers with natural reproduction. This implies, that the same disorder is now also present among wild salmon. Computer simulations show, that the combination of today's high catches and present mortality level will lead to extinction of wild salmon in the near future, and that to prevent this, a moratorium on commercial catches is needed. Such a ban on the catch must also include the sea trout, because salmon and sea trout are often caught together and can normally only be distinguished by experts.

Cod, Gadus morhua - Cod probably entered the Baltic Sea some 7,500 years ago. The Baltic cod has over these years adapted to the low salinity, resulting in, for example, a much better survival of sperms and eggs in waters of low salinity compared with the Atlantic cod, without becoming a new species. Salinity and oxygen content are factors determining the hatching success of cod. A sufficient level of salinity, i.e., >10 psu, keeps the eggs suspended and may keep them above depths with oxygen deficiency. Cod-reproduction peaks cooccur with inflow of water of high oxygen and salinity levels. There are two more or less distinct populations of cod in the Baltic Sea. The western stock reproduces south-west of Bornholm and the eastern in deep areas of the south-eastern parts of the Baltic Proper.

Cod fishing is economically important in the Baltic Sea. However, cod is very susceptible to over-fishing, because of limited reproduction areas and sensitivity to low salinity and low oxygen levels. From 1965 to 1985, catches of cod increased from 150,000 t to 450,000 t, but have then decreased to less than 50,000 t yr^{-1}. In 1994, surveys showed a good supply of cod eggs as a result of the 1993 inflow of oxygen-rich water. The number of cod larvae produced was, however, low, indicating low hatching success. During the 1990s, cod, like salmon, have experienced substantial disorders in embryonic development and high egg mortality has been recorded. Also other fish species in the Baltic Sea have suffered from different reproduction and recruitment disorders, e.g., perch, sea trout and burbot. As a conclusion, fishing practices should follow the principles of sustainable use. Better knowledge about reproductive disorders of fish is needed, and further investigations are necessary on both issues.

7.2.2.4 Sea birds

More than 30 species of water birds breed along the coasts of the Baltic Sea. Common eider, *Somateria mollissima*, tufted duck, *Aythya fuligula*, red-breasted merganser, *Mergus serrator*, redshank, *Tringa totanus*, sandpiper, *Actitis hypoleucos*, as well as herring gull, *Larus argentatus*, arctic tern, *Sterna paradisaea*, and common tern, *Sterna hirundo*, are all characteristic species along the Baltic Sea coasts. Razorbill guillemot, *Alca torda*, black guillemot, *Cepphus grylle*, and guillemot, *Uria aalge*, are truly marine bird species nesting on small islands or steep rocks. During recent years, the levels of PCBs and DDTs have decreased, resulting in higher reproductive success for these birds and birds of prey. Eutrophication, causing a higher production of algae and of some fish, has probably been positive for some species, e.g., cormorants, *Phalacrocorax carbo sinensis*. Barnacle goose, *Branta leucopsis*, has become a new breeding species with about 2,000 breeding pairs which, during migration, nest on islands off Gotland, Hiiumaa and Saaremaa.

Surveys during winter have found impressive amounts of sea birds wintering in Baltic Sea waters, making these areas of utmost importance and thus at risk from oil spills and other discharges of pollutants. Large amounts of diving sea birds are incidentally caught in fishing gear every year. For example, gill-net fishing for cod can result in large by-catches of birds. During 1982-88, about 25,000 sea birds were caught in gill nets in the south-eastern Kattegat. Guillemots are probably very vulnerable to gill-net fishing according to reports from the Karlsö Islands outside Gotland. In Schleswig-Holstein, Germany, the annual water-fowl losses, caused by set-net fishery at the end of the 1970s, were calculated to be at least 15,000 diving ducks.

Caspian tern, Sterna caspia - The Baltic Sea is one of two breeding areas for the Caspian tern, one of the largest of all terns, in Europe. The terns breed on small, isolated sandy islands or cliffs in the outer part of the Swedish, Finnish or Estonian archipelagos, and in the Ladoga area, Russia. The Baltic Sea hosts some 25 colonies (Sweden 10, Finland 12, Estonia 2, Russia 1) and a number of locations containing solitary pairs of Caspian terns, altogether about 1,500 (Sweden 550, Finland >700, Estonia 270, Russia 24). The reproductive success of the Caspian tern in the Swedish part of the Baltic Sea was very high in 1992, but low during the summer of 1993, and even lower in 1994. In Finland, the Caspian terns had a more successful reproduction in 1994, and the number of fledglings was double that of the Swedish population. Caspian terns face threats both at the breeding sites in Europe and at the wintering sites in tropical West Africa. The presence of the American mink and occasional parasite outbreaks cause problems at the breeding sites, while hunting and droughts take their toll in West Africa.

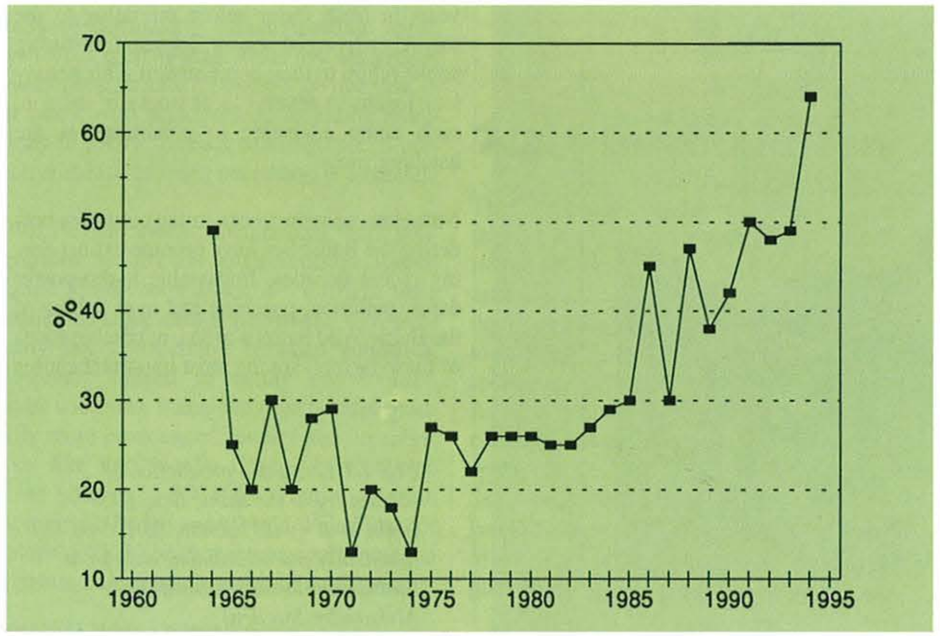


Fig. 7.2 Nesting success of the white-tailed eagle in the Baltic Sea Region(data from the Swedish Museum of Natural History)

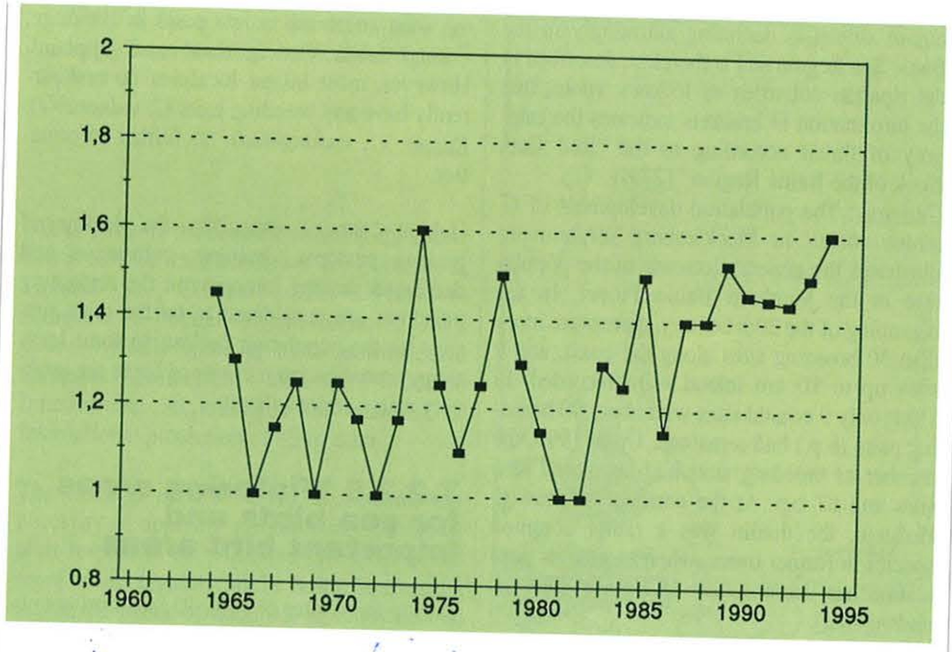


Fig. 7.3 Number of fledglings per nest of the white-tailed eagle in the Baltic Sea Region (data from the Swedish Museum of Natural History)

White-tailed eagle, Haliaeetus albicilla - The white-tailed eagle was the first species, associated with the aquatic environment, showing negative reproductive success due to pollutants. During the mid 1950s, the reproductive success drastically decreased in the whole Baltic Sea Area. The main problems were lowered hatching rates of eggs caused by high contents of organochlorines, especially DDTs and PCBs, but also disturbances by man including illegal hunting. Following the ban on DDTs and PCBs, the reproductive success of these eagles in the Baltic Sea has increased. Before the 1950s, the nesting success was about 72 %, with an average of 1.8 fledglings per nest. During the late 1960s and the 1970s, these figures dropped to 25 % and 1.2 fledglings.

Today, the reproductive success is very close to the values prevailing before organochlorines occurred in the Baltic Sea environment. The Baltic white-tailed eagle population has increased and regained old territories. During 1994, these eagles showed the highest reproduction rate of the last 30 years, i.e., 68 % of investigated eagle pairs succeeded with their nesting (Figs. 7.2, 7.3).

Since the early 1970s, biologists and environmentalists in Sweden, Finland, Estonia and Germany have worked hard to save the white-tailed eagle. Annual surveys of nesting pairs and their reproductive success, winter feeding with non-toxic meat for the wintering birds and nest-site protection have been important parts of this successful work. Figure 7.4 illustrates this positive trend in Mecklenburg-Vorpommern, Germany. In the other parts of Germany and in Denmark, the white-tailed eagle is still regaining territories that had been abandoned at the turn of the century. The breeding population in Germany has doubled between 1980 and 1994/95. The current reproduction rate is about 50-60 %, and in areas with a lower population of eagles, for instance Schleswig-Holstein, the rate is even higher. One can conclude that it has taken 15 years following the ban of DDTs to abolish the negative effects of DDTs, and a further 10 years for the situation for the white-tailed eagle to return to 'normal'.

Cormorant, Phalacrocorax carbo sinensis - The cormorant has two sub-species in the Baltic Sea. The nominate race, *Phalacrocorax carbo carbo*, which breeds or nests in the Atlantic Ocean, is often found in the Baltic Sea during winter. The other race, *Phalacrocorax carbo sinensis*, breeds in the Baltic Sea Area and is probably the most successful bird in the area in terms of recovery of numbers (Fig. 7.5). After being hunted down to near extinction in the Baltic Sea during the 19th century, the cormorant returned in the 1940s and nested for the first time after its disappearance. During the last decade, the increase in individual numbers has been impressing. For example, the number of breeding birds increased in Denmark in the 1980s by 25 % per year. The data from the 1990s shows, however, that this population growth is levelling off. As a result of the increase, cormorants have become a pest for fishermen because of its feeding on fish. Illegal destruction of colonies has occurred in some countries during recent years.

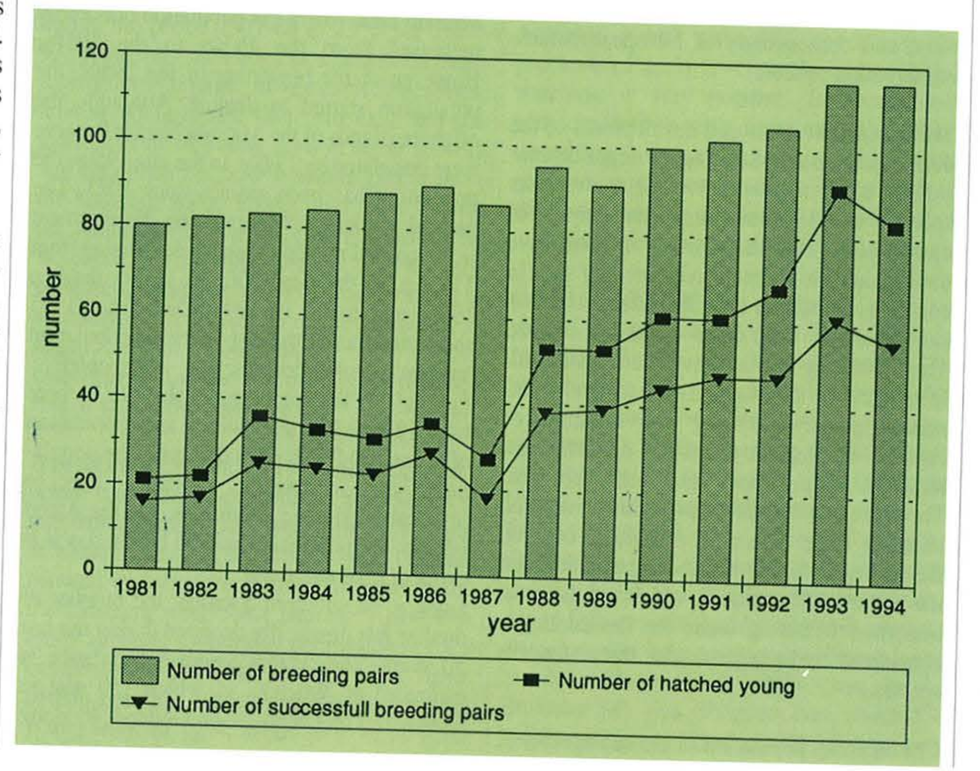


Fig. 7.4 Population size and reproduction of the white-tailed eagle population in Mecklenburg-Vorpommern, Germany, 1981-94

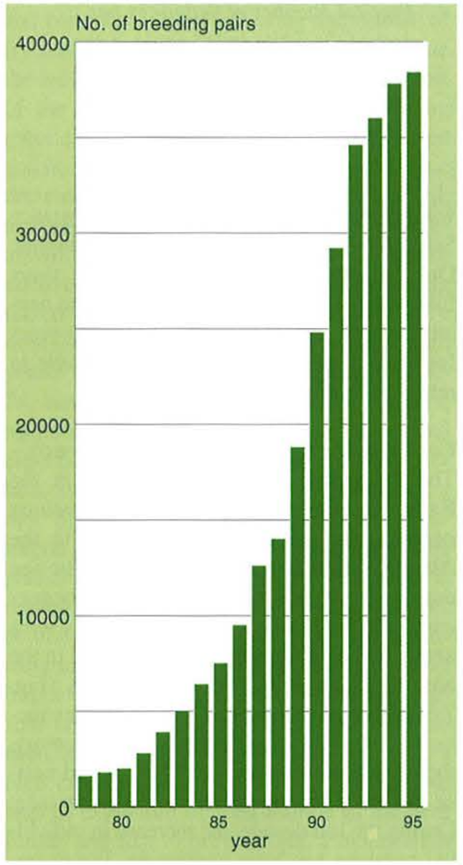


Fig. 7.5 Development of the cormorant population in Denmark

Dunlin, Calidris alpina - The dunlin is a common bird with a circumpolar distribution and a stable breeding population in northern temperate and Arctic latitudes. The European breeding sites are known to be the most southern ones, and consequently of European nature-conservation concern.

The population status and development of the dunlin in the Baltic Sea Region might be considered to be representative for most wader species, a typical and important group of coastal birds. Nowadays, many of them have their main or even exclusive breeding sites in coastal habitats like beaches, islets, marshes, salt meadows and other coastal wetlands. These breeding habitats have been dramatically reduced or deteriorated due to land-reclamation projects, dyking and drainage of coastal wetlands, touristic disturbances, increase of predators and other reasons. Therefore, most waders have to be considered as being under threat on a regional or even Baltic Sea-wide basis. Land reclamation and the pollution of estuaries, which are the most important wintering areas for the dunlin, is considered to be a particular threat for this species.

The breeding population of the sub-species *C. alpina shinzii* is declining alarmingly in the Baltic Sea Region and is therefore described in the riparian countries as follows: (note, that the information in brackets indicates the category of threat according to the "Red Data Book of the Baltic Region" [*279*])

Germany: The population development of *C. alpina shinzii* in Mecklenburg-Vorpommern illustrates the general decrease of the population in the Southern Baltic Proper. In the beginning of the 20th century, there were more than 30 breeding sites along the coast, and 9 sites up to 80 km inland were recorded. In 1981, only 9 coastal sites with about 90 breeding pairs (b.p.) had remained. Up to 1995, the number of breeding sites had decreased to 5 sites and 67 b.p. At the coast of Schleswig-Holstein, the dunlin was a rather common species in former times, whereas now its population has decreased to 12 b.p. in 1984 (*1, endangered*).

Denmark: About 600 b.p. in 1971, declining to 490 in the mid 1980s (*2, vulnerable*)

Poland: At the mid 1980s, the Polish population amounted to 80-100 b.p. (*1, endangered*)

Lithuania: A drastic population decrease was observed during the last decades, and only a few breeding pairs are now present (*1, endangered*)

Latvia: About 10-15 b.p. until the mid 1980s; no proven breeding has been recorded since 1991. In the last few years, several pairs of dunlin, exhibiting territorial behaviour, were recorded in the eastern part of Teici Bog, situated about 170 km away from the coast; but there is no evidence of breeding at this time (*1, endangered*),

Estonia: Estonia is the only Baltic country, where the dunlin is still rather common, with an estimated total population of 500-700 b.p. One of the most important breeding sites is the Matsalu area, where the population noticeably increased from the 1930s to the 1950s. However, at the beginning of the 1960s, the population started to decline. Although, the offshore islands of the Matsalu Nature Reserve were populated by 13 b.p. in the late 1950s, the population has been extinct since 1981. The general tendency of population development in the coastal meadows is also decreasing from 315 b.p. in the late 1950s to 155 in the late 1960s and to 85 in 1993. Data from 1995 show, that during the last years the population has considerably decreased (*2, vulnerable*).

Finland: At the beginning of the 1960s, there were 13 breeding sites with a total population of about 150-200 b.p. At that time, the population was increasing, probably due to favourable climatic conditions. On the Åland Islands, the population was 30 b.p., today it is extinct (*1, endangered*).

Sweden: In southern Sweden, the number of dunlins has drastically declined during the last 50 years. The current Swedish population is estimated at 390 b.p., of which 150 nest in Skåne, 150 on Öland, 40 on Gotland, 40 along the west coast and a few pairs in Blekinge, Östergötland, Västergötland and Uppland. However, most inland localities do not currently have any breeding pairs (*2, vulnerable*).

Russia: (*1, endangered*), no further information.

Habitat changes, especially the decline of grazing pastures, draining, cultivation and decreased grazing intensity on the remaining meadows are considered to be the main reasons for the population decline. In some locations, predation from crows or foxes are probably also a serious threat.

7.2.2.5 Wintering areas for sea birds and important bird areas

During the winter of 1992-93, sea-bird counts were conducted from ships and aircrafts in the Baltic Sea. The survey showed, that nine million sea birds of some 30 species use the Baltic Sea as a wintering area. The long-tailed duck, *Clangula hyemalis*, common eider, *Somateria mollissima*, velvet scoter, *Melanitta fusca*, and black scoter, *Melanitta nigra*, dominate and constitute some 70 % of all birds. The birds were not evenly distributed over the area. Thirty nine locations were identified as important wintering areas, and ten of these harboured 70 % of all birds (Fig. 7.6).

Four areas are of outstanding importance for wintering sea birds:

• Shallow lagoons and estuaries along the German-Polish coast, i.e., the Vorpommern- and Odra lagoons (*area 1* in Fig. 7.6), contain almost half a million birds. The area is particularly important for smew, *Mergus albellus*, greater scaup, *Aythya marila*, goosander and tufted duck.

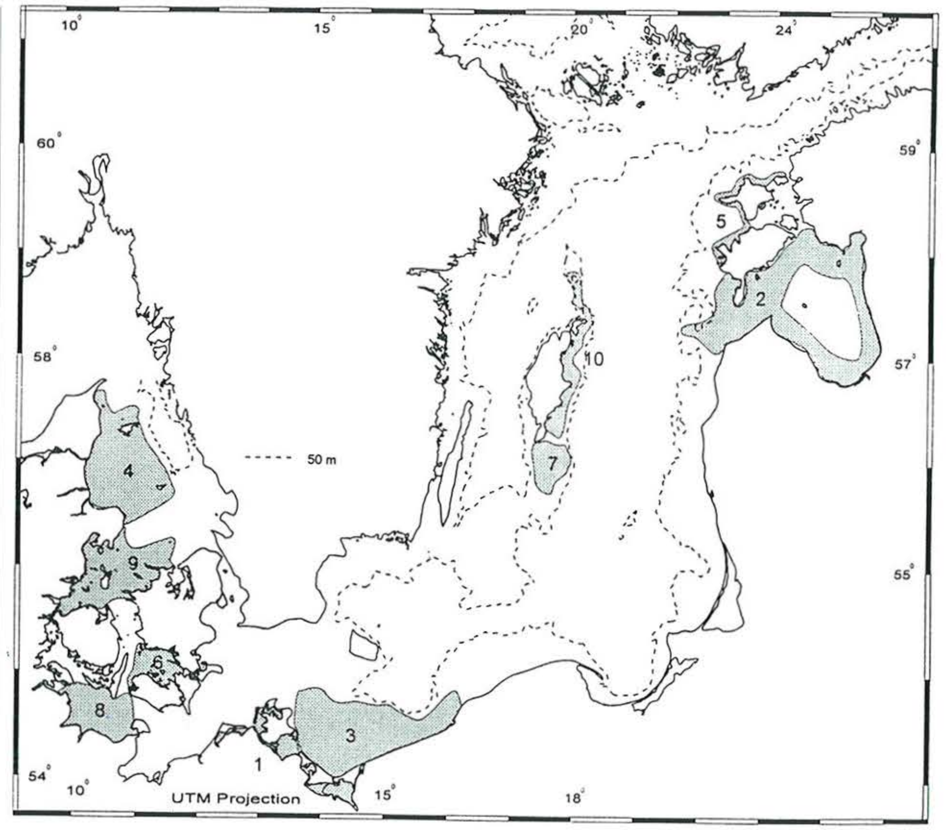
• The shallow and sandy bottom waters of the Pomeranian Bight between Denmark, Germany and Poland (*area 3* in Fig. 7.6) are wintering sites for 1.3 10^6 sea birds. The Pomeranian Bight is particularly important for velvet scoter, Slavonian grebe, *Podiceps auritus*, long-tailed duck and black guillemot.

• Sand and gravel banks along the Estonian and Latvian coasts in the Gulf of Riga (*area 2* in Fig. 7.6) support some 1.5 10^6 sea birds. This area is of global significance for divers (*Gaviidae*) with for example 24 % of the north-west European population of red-throated diver, *Gavia stellata*. It is also a very important wintering area for long-tailed duck (>1 10^6), velvet scoter and black guillemot.

• The north-western Kattegat (*area 4* in Fig. 7.6), especially the shallow north-western part, harbours >1.2 10^6 birds during the winter. The area is of great importance for common eider, razorbill, red-necked grebe and black scoter.

The distribution of Important Bird Areas (IBAs[2]) among the countries sharing the Baltic Sea is as follows: Denmark - 42, Estonia - 5, Finland - 22, Germany - 23, Latvia - 6, Lithuania - 2, Poland - 15 and Russia - 2 [*187*]. Currently, only some of these areas receive adequate protection, although BirdLife International recommends the national or international protection of all of them.

Fig. 7.6 The ten most important wintering areas for birds in the Baltic Sea Area



The findings of this survey accentuates the necessity of applying precautionary measures with respect to oil pollution. Oil spills occur more or less regularly in the Baltic Sea, resulting in the death of hundreds to thousands of sea birds each year. During the present period, when several new oil terminals are under construction or planned, especially in the Gulf of Riga area, risks of large-scale oil spills are indeed high. The foreseeable consequences of a massive oil spill within any of the most important areas are frightening, since one single spill may eliminate large portions of the entire north-west European sea-bird populations. Control of and legal action against deliberate oil spils are also necessary. Strict regulations for all oil transports, especially in protected areas, and safe transport corridors are of immense importance. Since many of these important wintering areas for birds are situated offshore, e.g., Slupsk Bank, Midsjö Bank, Öland South Bank or Hoburgen Bank, the need for legal means to protect offshore areas is also of highest priority.

7.2.2.6 Marine mammals

Three species of seals, harbour (common), grey and ringed seal, and one whale species, harbour porpoise, constitute the species of marine mammals breeding in the Baltic Sea Area. A recent publication [*730*] dealing with the state of the Baltic Sea states *"...There seems to be reason for some cautious optimism regarding the possible long-term recovery of the grey seal and ringed seal in the Baltic Sea..."*. But the situation for the ringed seal in the Gulfs of Finland and Riga is still particularly vulnerable, and *"...For the very small population of common seal, the situation is alarming..."* [*730*]. It should also be kept in mind, that in the southern part of the Baltic Sea the grey seal and the harbour seal had nearly disappeared. With the *"Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas* (ASCOBANS)", the Contracting Parties (Denmark, Germany, Poland and Sweden as HELCOM Contracting Parties) are *"...aware that the population of harbour porpoise in the Baltic Sea has drastically decreased."* They also stated that further investigations for management and protection measures are necessary.

Following a history of decreasing populations and severe problems with organochlorines (DDTs, PCBs), resulting in skeleton and uterine deformations leading to sterility, the situation for the Baltic seals is now slowly improving. The proportion of seals with uterine deformaty has decreased from an average of 36 % during the period 1977-86 to 25 % for 1987-93. However, pathological changes in non-reproductive organs, for example, intestines, arteries, kidneys and skin, still exist and the mortality rate of young seals in some areas is still very high, i.e., >50 % among grey seals in the Southern Baltic Proper, and 95 % among the southernmost Swedish common seals. The incidence of intestinal lesions has actually increased during the last 5-10 year period. It is alarming, when previously unknown persistent organochlorine compounds like *Tris,4-chlorophenyl-methan* are found in the tissue of ringed seal in the Baltic Sea.

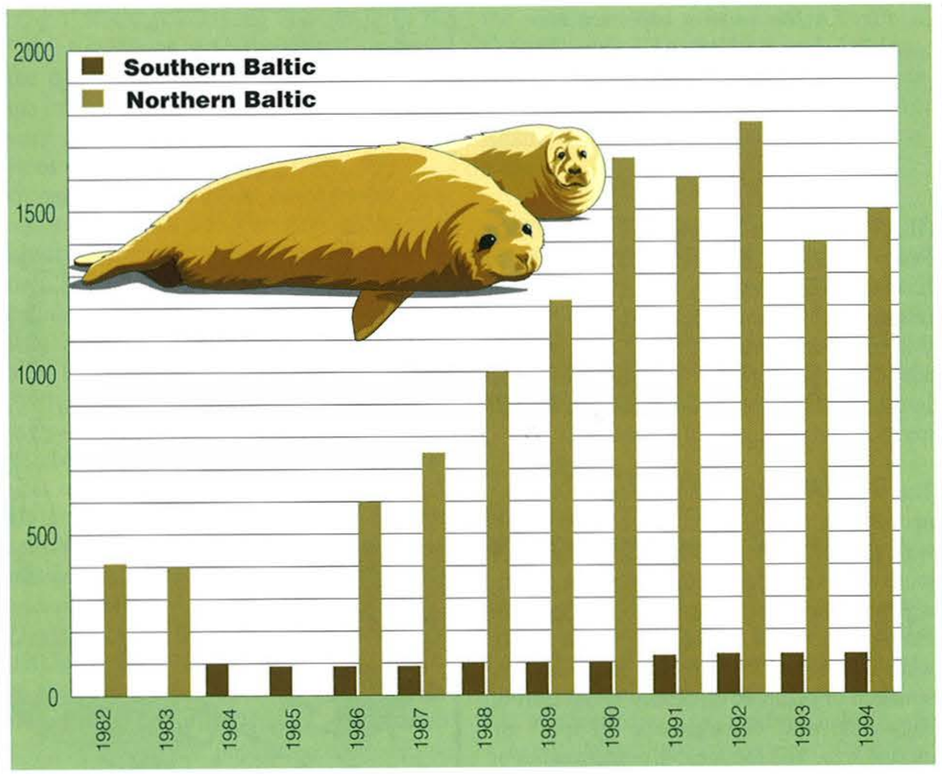
A peaceful coexistence between seals and fishermen seems to be very difficult at present. On one hand, seals eat fish from fishing nets and as a result, nets are often destroyed. This has resulted in fishermen in the Gulf of Bothnia, the Estonian west coast and the Åland Sea either applying for hunting permission or for a compensation payment. On the other hand, fishing gear also causes high mortality among young seals. In 1994, about 80 % of all juvenile grey-seal mortality was caused by drowning in fishing gear. Conflicts between seals and fishermen are increasing with the recovery of the seal stocks and changes in seal behaviour, although the number of seals in the Baltic Sea has so far not reached the population of the early 20th century. Hunting was a major reason for the drastic decline in the number of seals until the 1970s. Within the HELCOM framework, the Contracting Parties have agreed on a ban of hunting seals.

Although public awareness concerning the problems of seals is relatively high and seal watching is very popular, the recreational activities disturb seals more and more. Such disturbances are particularly important when seals are resting on shore or giving birth to their pups, and this can lead to a withdrawal of the animals from such areas.

Grey seal, Halichoerus grypus - The major part of the grey seal population is found in Estonian, Finnish and Swedish waters. The total number of grey seals counted was about 5,300. Of these, less than 200 were found south of 58° N. The majority of the animals were found along the western coast of Estonia (1,200-1,500), the south-western coast of Finland (mainly Åland, 1,000), and along the Swedish coast of the Northern Baltic Proper and the Åland Sea (1,700 specimens). In the Åland Sea, some of the animals haul out of the water at the border between Finland and Sweden. Along the Swedish coast to the Bothnian Sea and Bothnian Bay, about 550 specimens have been counted. Fewer grey

[2] The IBA Programme is a conservation initiative of BirdLife International without legal consequences

Fig. 7.7 Number of grey seals in the more southern (left columns) and northern parts (right columns) of the Baltic Sea Area



seals were found in the Gulf of Finland (50-100) and along the Finnish coast of the Bothnian Sea and Bothnian Bay (200 specimens). Very few grey seals occurred in Danish waters, and none have been observed breeding in these waters.

However, the population trend is promising. The Swedish population north of 59° N has increased annually by 12 % during 1982-94 (Fig. 7.7). A similar increase of almost 14 % yr^{-1} during 1985-94 has been found in south-western Finland. However, south of 59° N, only a minor and statistically insignificant increase of 3 % yr^{-1} was found by Swedish investigators for 1984-94 (Fig. 7.7). Since data from the Estonian coast are lacking, no estimate of earlier trends can be made for this part of the Baltic Sea. However, a slight increase in the populations can be observed from the data collected since 1990.

The rapid increase in the number of grey seals in the northern parts of the Baltic Sea Region is obviously a result of improvements in reproductive ability. However, part of the increase might be explained by better censuses in the recent period and by changes in the grey seal behaviour, i.e., they become less shy due to the absence of hunting. Although the population estimates were based on counts during the moulting period, when the seals are believed to be much more stationary than at other times, the possibility of migrations and thus of double counts of animals cannot be excluded. Satellite tracking of grey seals in the Baltic Sea after the moulting period has shown that individuals can move quickly between haul-out sites which are far apart.

Unlike the northern population, the southernmost population has not increased, and this finding cannot be explained at present. One reason could be that more young grey seals are being drowned in fishing gears in this area.

Baltic ringed seal, Phoca hispida botnica - Ringed seals are mainly found in Arctic areas where it is the most common seal species. The ringed seal population is distributed in the Bothnian Bay, Eastern Gulf of Finland and Gulf of Riga. There are also observations of individual ringed seals in the archipelago between Turku and Åland where even pups have been reported during the last years. Annual surveys in the Bothnian Bay have been carried out since 1988 (Fig. 7.8). In 1994 and 1995, the populations in the Gulfs of Finland and Riga were studied in a joint effort by Estonian, Finnish, Russian and Swedish scientists. However, the data from the last two areas are not in all aspects comparable to the data from the Bothnian Bay. Surveys in the inner part of the Gulf of Finland were carried out by helicopter, whereas in other areas aircrafts were used.

In the Gulf of Bothnia, the number of animals seems to have slightly increased, by about 5 % yr^{-1} during 1988-95, from 2,300 to 3.000 specimens (Fig. 7.8). Studies in the Gulf of Finland in 1994 gave a much lower number of ringed seals than that reported earlier by the Soviet Union. The total number counted in 1994 was 170-200, and in 1995 about 150 specimens. Comparing this figure with the mass mortality of ringed seals in the winter 1991/92, when 100-150 specimens were found dead, this suggests an alarming situation for the population in the Eastern Gulf of Finland. So far, no cause of death has been identified. In a study in the Gulf of Riga in 1994, 500-800 seals were found to be present. In 1995, there was no ice on the Gulf, so it was not possible to conduct a census that year.

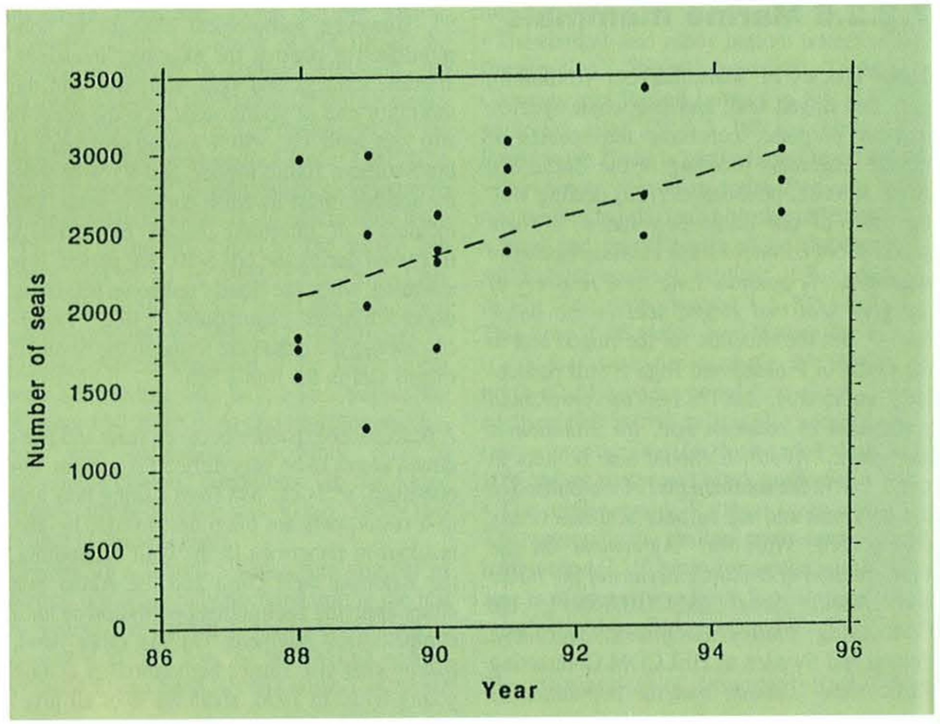


Fig. 7.8 Number of counted ringed seals hauled out during the moulting period on ice in the Bothnian Bay

Following glaciation, several populations were land-locked and exist today as geographically isolated relicts in the Baltic Proper and in the lakes Saima and Ladoga. Since the Baltic ringed seal is a distinct and endemic subspecies, it should be added to the Bern Convention and the EU Habitat Directive. The ringed seal is the least known of the Baltic Sea seal species. An international project, with input from Sweden (Swedish Environmental Protection Agency), Finland (Wildlife Research Institute), Estonia (Estonian Marine Institute), Latvia (Latvian Academy of Sciences) and Russia (University of St. Petersburg), has recently started surveys including radio-tracking (satellite telemetry) of seals.

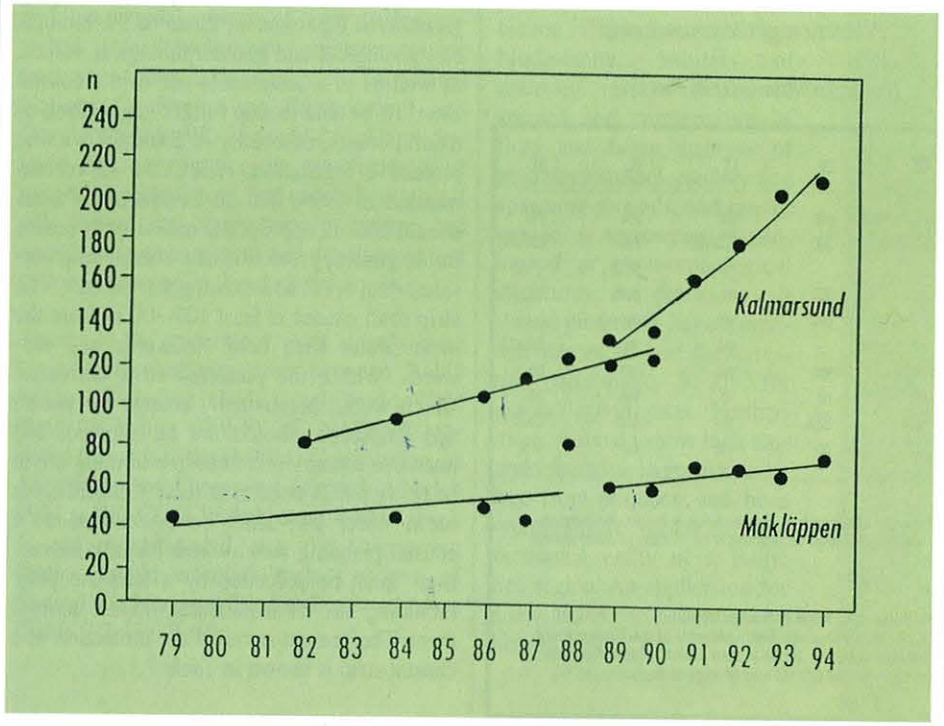
Common seal, Phoca vitulina - The majority of Baltic common seal are found in the Kattegat, and a smaller number in Swedish waters, where they are located in two areas, in south-western Sweden (Måkläppen) and in the Kalmar Sound. The population at Måkläppen was hit by the *phocine distemper virus* in 1988. In 1994, a total of 75 animals were counted in the Måkläppen area. An increase in population corresponds to about 4 % yr^{-1}. In the Kalmar Sound, 209 animals were counted in 1994. In this area, the census changed somewhat in 1989 to a more detailed one. The annual increase of the population during 1982-90 was 5.8 %, but 11 % for 1989-94 (Fig. 7.9). The lower growth rate of the population at Måkläppen is partly due to predation by foxes on new-born pups. Surveys of the pupping rate in the two areas revealed 17 pups at Måkläppen in 1994 and 54 pups in the Kalmar Sound area. These figures correspond to a pupping rate of 23 % and 26 %, respectively. The annual observed pup mortality at Måkläppen was 5 pups or 29 %, which is substantially lower than in previous years. In the Kalmar Sound, the corresponding figures are 4 pups or 7 %. Obviously, the reproductive rate and pup survival in the Kalmar Sound are good.

In 1994, in the Danish part of the Kattegat, the number of common seals had surpassed 2,000, while the number counted in the Limfjord was 600. Within and south of the Danish Straits, the seal numbers have been consistently low since 1990 suggesting that the species is still vulnerable in this area.

Harbour-seal stocks in the Kattegat and Skagerrak have now recovered from the *phocine distemper virus* disease of 1988, and the size of these stocks equals that of the pre-disease times, i.e., about 6,500 animals. A mitochondrial DNA analysis among North European harbour seals reveals, that different sub-populations (Kattegat/Skagerrak, Wadden Sea, English west coast, Scottish east coast, southern Swedish and Kalmar straits) are genetically distinct from each other. The Baltic Sea sub-population represents a specific local race of high conservation value.

Harbour porpoise, Phocoena phocoena - The state of the harbour porpoise in the Baltic Sea is unclear. An international survey of harbour porpoises, SCANS (Survey of Cetaceans of the Atlantic North Sea), has included the Baltic Sea during 1995. First results are available from observations in 1994. In the waters of the Great Belt and north of the Danish island Fünen, a total of 8,100 or 0.99 individuals per km^2 were observed. Such a high population indicates an area of European interest and importance, but it does not mean that the species is not endangered in this area. Experts believe that only some thousand individuals may occur in the whole Baltic Sea (588 in Kiel Bight according to SCANS), and suspect that they are genetically specific and reproduce exclusively within the Baltic Sea. In view of this, some scientists have concluded that there is a possibility of total extinction of harbour porpoises in the Baltic Proper. During 1989-91, 175 porpoises were caught in fishing gear in the Kattegat, and 13 in the Baltic Sea. Most of these were caught in gill nets, 72 % by cod gill-nets in the Kattegat and 54 % by salmon drift-nets in the Baltic Sea. Thus, gill nets comprise a serious threat for harbour porpoises.

Summary - Protection of fishing gear, development of new 'proof' gear, and research on the behaviour of seals and harbour porpoises around fishing gear is needed, to ensure the future of Baltic marine mammals. Seal sanctuaries, such as those in Danish waters, seem to be successful in assisting the recovery of seal populations. Such reserves should include strictly protected potential resting places, like gently sloping sandy beaches, boulder beaches, bars, banks or spits. It is essential to give the new southern Baltic Sea population of harbor seals a chance to survive. One step forward would be the full implementation of the HELCOM Recommendation on the coastal strip (15/1). According to HELCOM Recommendation 9/1, the general ban on hunting shall be retained. Research on porpoise in the Baltic Sea (by-catches, observer programme, stock identity) is urgently needed, as stated in ASCOBANS. The ringed seal should be added to the Bern Convention.



7.2.2.7 Introduction of new species

During the last century, at least 50 new species have been introduced into the Baltic Sea because of human activities. Most of these species have entered with ballast water or attached to the hulls of ships, but species have also been deliberately introduced. The species-poor communities of the Baltic Sea are probably more sensitive to introductions of alien species than more species-rich marine areas. It is to be expected that more species will be

Fig. 7.9 Number of harbour seals counted during the moulting period in the Kalmar Sound and at Måkläppen introduced in the future due to increased shipping.

A few alien species have become common enough to affect the original natural communities. The polychaete *Marenzelleria viridis* was first found along the German coast and has since spread to both Sweden and Finland. It inhabits muddy sediments located in water depths of up to 60 m, and may constitute up to almost 100 % of the macrofauna biomass. Its appearance might affect native species like polychaetes, e.g., *Nereis diversicolor* and amphipods. Other successful invading species are the jellyfish *Cordylophora caspia*, the barnacle *Balanus improvius* and the bivalves *Dreissena polymorpha* and *Mya arenaria*.

Among fishes, the round goby, *Neogobius melanostomus*, is a successful invader. It has probably been introduced with ballast water from the Black Sea and Caspian Sea. Occupying similar niches as, for example, flounder, *Platichthus flesus*, black goby, *Gobus niger*, and eelpout, *Zoarces viviparus*, the round goby is expected to compete with these species. The barnacle goose is a successful self-introduced newcomer to the Baltic Sea fauna of breeding birds. Among mammals, the American mink, *Mustela vison*, is a good example of a deliberate introduction which severely affects natural communities. Several islands, serving as nesting sites for sea birds in the Stockholm Archipelago, have recently experienced drastic reductions in the number of breeding birds, probably due to predation by mink.

7.3 NATURE CONSERVATION AND SUSTAINABLE USE

7.3.1 Current activities in nature conservation

Although the 1992 Helsinki Convention is still not ratified by all Contracting Parties, all Baltic Sea States have agreed to act as if it was already in force. Therefore, HELCOM 14 decided to establish a working group on Nature Conservation and Biodiversity, EC NATURE, under the Environment Committee, to implement the earlier mentioned new Article 15.

During the present assessment period, EC NATURE has concentrated its work; among others, mainly on the following subjects:
• guidelines for the identification of coastal ecosystems, influenced by the Baltic Sea and their inland limits, as an implementation of Article 15, 1992 Helsinki Convention (As a result of this, Article 15 covers the whole Baltic Sea, its coastal and inner waters and its coastal zone, influenced by the Baltic Sea),
• a general protection of the coastal strip (HELCOM Recommendation 15/1),
• the establishment of a 'System of coastal and marine Baltic Sea Protected Areas (BSPA)' (HELCOM Recommendation 15/5),
• guidelines for the establishment of new BSPAs,
• guidelines for the management of BSPAs,
• the preservation of natural coastal dynamics (HELCOM Recommendation 16/3), and
• a project, to compile a Red Data Book of Biotopes in the Baltic Sea Area, started in 1994.

In relation to species protection, the following items are currently included on the agenda:
* seals,
* Baltic Wild Salmon,
* common sturgeon,
* cormorants, and
* other tasks.

Some of the above mentioned topics are explained in more detail in the following sections.

7.3.2 Protection of the coastal strip

The increasing pressure on the coastal zone through anthropogenic activities, such as building of new roads, harbours, marinas, hotels, private summer houses and camping grounds, is a permanent threat to the biological, geological and geomorphological values, as well as to a sustainable use of the coastal zone. To be able to stop further degradation of natural coasts, especially in countries without protective legislation, HELCOM 15 recommended in 1994, that all Contracting Parties should take all appropriate measures to ensure future general protection of a coastal strip outside urban areas and existing settlements. The strip shall extend at least 100-300 m from the mean water line, both landwards and seawards. Within the protected strip, activities, which would permanently change the nature and landscape, should not be allowed, and intensive forestry and intensive farming are to be restricted. A zone of at least 3 km from the mean water line shall be established as a coastal planning zone, where major constructions must be preceded by a land-use plan, including an environmental impact assessment. The present status of the protection of a coastal strip is shown in Table 7.1.

Table 7.1 Implementation of HELCOM Recommendation 15/1 "Protection of the coastal strip", 1995 (No information from Russia)

	DK	EE	FI	DE	LV	LT	PL	SE
Legislation	yes	yes	yes	yes	yes	yes	yes	yes
Land-strip* (m)	300 (100)	100 (200)	no	M-V:200; S-H:100	300	200-700	200 (100)	200-300 (100)
Sea strip (m)	partly	partly	no	no	300	no	partly	100-300
Admin. prot. meas.	yes		yes		yes	yes	yes	
Other prot. meas.	m1, m2		m2, m3	m1, m2, m4		m4	m4	m1, m2
Restrictions on forestry	partly	yes	no	yes	yes	yes	yes	no
Restrictions on farming	no	yes	no	no**	no	no	yes	no
Exceptions	CC	Min	REC	SEA	SEA	yes	Mar	CAM
Coast. plan. zone (km)	3	no	no	no***	no	yes	no	yes

* In case of different widths, the less dominant is shown in brackets (In Sweden, the distance is measured from the mean water mark, in Denmark the distance is measured from where terrestrial vegetation is unbroken; in other states from the high water line expressed in different ways)
** Incentives: yes
*** Whole State is planning zone
m1 - biotope protection, m2 - land-use plans, m3 - municipal by-laws, m4 - large protected areas
M-V - Mecklenburg-Vorpommern, S-H - Schleswig-Holstein, CC - County Council, Min - Ministry of the Environment, REC - Regional Environmental Centre, SEA - State Environmental Authority (in M-V also District Authority), Mar - Maritime Office with Regional Authority, CAM - County Administration Board (70 %) and Municipal Authority (30 %)

DENMARK
• Bornholm
- Davids Banke/Hameren to the north
- Ertholmene to the north-east
- Dueodde/Salthammer Rev to the south
• Adler Grund
• Smålandsfarvandet
- North-western Smålandsfarvandet
- North-eastern Smålandsfarvandet
- Southern Smålandsfarvandet
• Waters around Saltholm
• Stavns Fjord and adjacent waters
• Waters around Hesselø
• Store Middelgrund
• Randers Fjord and neighbouring waters
• Waters around Læsø
• Waters around Hirsholmene
• Læsø Trindel/Tønneberg Bank/Kummel Banke
• Hertha's Flak

ESTONIA
• Lahemaa National Park
• Matsalu Nature Reserve
• Kopu Peninsula on Hiiumaa
• Vilsandi Nature Reserve
• Hiiumaa Islets
• Harilaid

FINLAND
• Bothnian Bay National Park
• Outer Bothnian Threshold Archipelago (The Quark)
• Oura Archipelago
• Uusikaupunki Archipelago
• Southern Archipelago Sea
• Tammisaari Archipelago-Hankoniemi-Pojo Bight
• Eastern Gulf of Finland National Park
• Åland Area: Signilskär/Märket

GERMANY
• Strelasund Sound/Greifswald Bodden/Island Greifwald Oie/Odra Mouth Area
• Jasmund National Park
• Vorpommern Lagoon Area/Waters around Westrügen
• Wismar Bight/Salzhaff Area
• Graswarder-West coast of Fehmarn incl. Flügger Sand
• Part of Hohwacht Bight with Lagoons
• Oehe-Schleimünde with Shallow Waters
• Geltinger Birk and Noor incl. Kalkgrund

LATVIA
• Northern Vidzeme Region Nature Protection Complex: Coastal Section Dzeni-Ainazi
• Kaltene/Engure Area
• Lierlirbe/Kolka Area
• Papa/Perkone Area

LITHUANIA
• Kursiu Nerija (Curonian Spit) National Park
• Pajuris Regional Park
• Nemunas Delta Regional Park

POLAND
• Vistula Spit Landscape Park (Park Krajobrazowy Mierzeja Wislana) incl. surrounding water areas
• Redlowo Reserve (Kepa Redlowska) incl. surrounding water areas
• Nadmorski Landscape Park (Nadmorski Park Krajobrazowy) incl. surrounding water areas
• Slowinski National Park (Slowinski Park Narodowy) incl. surrounding water areas
• Wolinski National Park (Wolinski Park Narodowy) incl. surrounding water areas

RUSSIA
• Gulf of Finland National Park (Russian part)
• Curonian Spit State National Park
• Vistula Spit Landscape Park

SWEDEN
• Haparanda Archipelago
• Bjuröklubb Area
• Holmö Islands
• Trysunda/Ullånger/Ulvöarna/Ulvö Depth
• Gräsö/Singö-Archipelago
• Storö/Bockö/St. Nassa/Sv. Högarna/Sv. Björn
• Landsort/Hartsö/Askö/Landsort Deep
• St. Anna - Missjö Archipelago
• Kopparstenarna/Gotska Sandön/Salvorev Area
• Torhamns Archipelago
• Falsterbo Peninsula with Måkläppen
• Kullaberg
• Nidingen/Sönnerbergen/Mönster

Table 7.2 Areas proposed to be included in a system of Baltic Sea Protected Areas (BSPAs) as adopted by HELCOM 15

7.3.3 Baltic Sea Protected Areas (BSPAs)

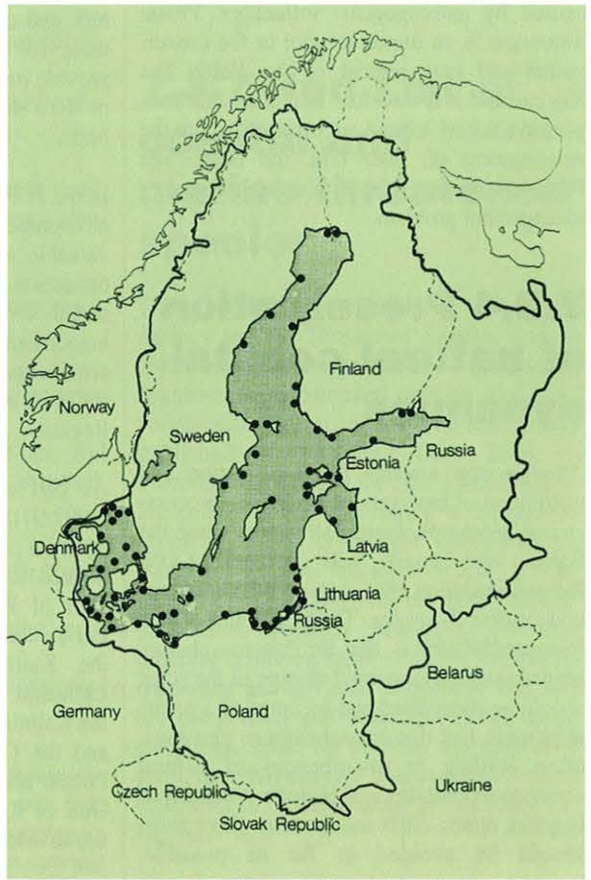
During the 1990s, several international bodies have urged governments to establish systems of protected coastal and offshore marine areas. The WWF/IUCN/UNEP document *"Caring for the Earth - A Strategy for Sustainable Living"* notes, that the establishment of such areas lags behind its terrestrial counterparts, and suggests, that a global system of protected marine areas should be established no later than 2010. The Caracas Declaration and Action Plan from the 1992 World Parks Conference developed these ideas by suggesting, that this system of areas should include major biogeographic types and ecosystems, and that national systems of large marine protected areas, including complete marine ecosystems, should be established. The United Nations Conference on Environment and Development in Rio (*Agenda 21*) also underlined the necessity for the establishment of marine protected areas, and pointed out, that it will require new approaches to marine and coastal area management and development, at national, sub-regional and global levels.

Thanks to the preparatory work already done by a joint working group between Baltic Marine Biologists (BMB) and World Wide Fund for Nature (WWF), EC NATURE proposed 62 areas to the Helsinki Commission, as a starting point. These were adopted in March 1994 at HELCOM 15, held at ministerial level. It was recommended, that the Contracting Parties should establish such a system of Coastal and Marine Baltic Sea Protected Areas (Table 7.2, Fig. 7.10).

7.3.3.1 New BSPAs

According to Recommendation 15/5, the 62 BSPAs should be the first step in establishing such a system which could then be gradually extended to include new coastal and marine areas. All Contracting Parties should individually and jointly take all appropriate steps to fulfil this Recommendation. Guidelines for designating new BSPAs were elaborated by EC NATURE and adopted by HELCOM 16 in March 1995. Following this guideline, under BSPA, particular protection shall be given to the species and natural habitats and nature types of the marine and coastal ecosystems of the Baltic Sea Area, to conserve biological and genetic diversity and to protect ecological processes. The areas for protection include those with high biodiversity, habitats of endemic, rare or threatened species and communities of flora and fauna, habitats of migratory species, nursery and spawning grounds, and rare or unique or representative geological or geomorphological structures or processes. It should preferably have a minimal size of 10 km² for terrestrial parts and/or 30 km² for marine/lagoon parts. Furthermore, it should show high naturalness, to a large extent be free from pollution and be a representative ecological and functional entity of a Baltic Sea region. An application for a new BSPA can be sent at any time to HELCOM.

Fig. 7.10 System of Baltic Sea Protected Areas (BSPAs)



7.3.3.2 Management plans for BSPAs

According to HELCOM Recommendation 15/5, management plans should be established for each BSPA, "*...to ensure nature protection and sustainable use of natural resources*". In 1995, such guidelines were adopted by the Environment Committee. As stated above, the aim for management in BSPAs is to ensure the conservation and/or restoration of biotopes and habitats, in order to preserve or to restore biodiversity and to ensure a sustainable use of natural resources. All imaginable conflicts of interest, especially between the aims of nature conservation and anthropogenic demands of land- and sea-use, e.g., extraction of sand, gravel and stone, oil and gas exploration and exploitation, dumping of waste, constructions, agriculture and forestry, tourism, transport, wind-mill parks, submarine cables, and military activities, may occur in BSPAs. Hence, such activities should be regulated where necessary, whereas some anthropogenic activities, on the other hand, should be maintained and developed in order to achieve certain conservation goals. A good example is the maintenance of traditional agricultural landscapes, with several species adaptable to disturbances, but demanding environmentally friendly farming practices.

A management plan must include monitoring of important physical, chemical and biological factors, to register changes or improvements as a result of planned actions, or deterioration, caused by anthropogenic influences. Public awareness is an important tool in the conservation and management of the Baltic Sea Region, and it is therefore important that people have access to, and are informed about the development of, *inter alia*, the Baltic Sea Protected Areas. This is to a high degree a management problem.

7.3.4 Preservation of natural coastal dynamics

Another step towards the preservation and restoration of biodiversity, ecological processes and geomorphological structures along the Baltic Sea coast was the HELCOM Recommendation 16/3 on preservation of natural coastal dynamics. The major aim of this Recommendation is, that the dynamic character and continuous natural change of the coast should be recognised and accepted as a natural process, and that coastal defence measures, often leading to disturbances of natural ecosystem processes and biotope structures of beaches, dunes, cliffs and the near- shore zone, should be avoided as far as possible. Consequently, HELCOM further recommended

• that new coastal defence measures outside settlements should not normally be executed, except when integrated coastal-zone-management plans provide otherwise,

• that active cliffs as sediment supplier, and natural coastal flood areas as potential nutrient traps, should not be subject to any new coastal defence measures, except when integrated coastal-zone-management plans provide otherwise, and

• that coastal areas outside settlements, that have been subject to episodic flooding before they were dyked for land-use purposes, should be restored as coastal wetlands through removal or relocation of dykes further inland, wherever possible.

7.3.5 Management of coastal lagoons and wetlands

Coastal lagoons, estuaries and other natural wetlands comprise very important key biotopes with high productivity. The crucial role of coastal lagoons and wetlands as 'multi-purpose ecosystems' is clearly recognised in the HELCOM Baltic Sea Joint Comprehensive Environmental Action Programme (HELCOM JCP), adopted at the HELCOM Ministerial Meeting in April 1992. These habitats can serve as important buffers for pollution to the Baltic Sea, by acting as natural traps and filters, and can provide variable levels of treatment for bio-degradable wastes. They also provide important habitats for a rich diversity of flora and fauna, including many migratory birds.

In the JCP, area-focused activities, to manage environmentally sensitive and economically valuable areas, were proposed. Accordingly, the action plan is designed to support the conservation of critical habitats, especially coastal lagoons and wetlands, and the development of criteria and guidelines for the identification of the most important wetlands in the Baltic Sea Region.

As part of the implementation of Phase I (1993-97) of the JCP, it is proposed, that management plans should be developed and implemented for the Matsalu Bight, Estonia, the Gulf of Riga, shared between Estonia and Latvia, the Curonian Lagoon, shared between the Kaliningrad Region of Russia and Lithuania, the Vistula Lagoon, shared between the Kaliningrad Region of Russia and Poland, and the Odra Mouth Area, shared between Poland and Germany. It was agreed, that the Gulf of Riga, because of its size, should be sub-divided into areas suitable for the development of integrated management plans. Consequently, two sub-projects, for the Käina Bight in Estonia and the Engure-Kemeri area in Latvia, have been established. Furthermore, this action plan shall support the development of a series of demonstration activities concerning the use of natural and constructed wetlands for waste-water treatment, storm-water retention, and as traps for nutrients and other contaminants.

Finally, the action plan is supposed to support the identification and evaluation of selected wetland areas in a variety of locations within the Baltic Sea drainage area. Particular attention should be given to the identification of priority sites for wetland conservation in the upper, middle and lower portions of the major drainage basins, as well as in coastal areas. Under the HELCOM Programme Implementation Task Force (HELCOM PITF), the Working Group on Management of Coastal Lagoons and Wetlands (PITF-MLW) was established in accordance with a decision of HELCOM 14 in February 1993. The World Wide Fund for Nature (WWF) provides secretariat facilities to this Working Group.

In addition to other duties, the PITF-MLW should initiate, facilitate and co-ordinate the development of 'Integrated Coastal Zone Management Plans' for the priority areas identified in the JCP. In accordance with its Terms of Reference, PITF-MLW is obliged to work in close collaboration with EC NATURE. PITF-MLW has decided, that the development of management plans for the six designated areas should be done through a decentralised process, by setting up 'Area Task Teams (ATTs)'. An ATT is a national (Matsalu Bight and sub-areas in the Gulf of Riga) or bilateral group of experts (scientists, technical experts, etc.) and representatives from national, regional and local authorities from the participating countries.

The development of integrated management plans should be an open and transparent process. Thus, the ATTs will encourage the active involvement of local communities, groups, interests and individuals in the development of the plans. In June 1995, the ATT secretariat was established and became operational in all six Task Areas. Integrated management plans were expected to be ready in early 1996. One important element in the management plans will be the development of concrete proposals on the conservation and sustainable use of the rich biodiversity of the areas.

I Landscape types and large biotope complexes	II Biotope types	
• archipelagos	A. Coastal Areas	B. Marine and brackish areas
• coastal dune complexes	• islets, skerries	• offshore stony and rocky bottoms
• large spits of sand separating a lagoon from the Baltic Sea	• boulder beaches	• offshore bottoms of shell gravel, gravel or coarse sands
• upheaval areas	• rocky shores	• offshore muddy or sandy bottoms with comparatively high biodiversity and/or bioproductivity
• coastal lakes	• coastal caves	• inshore stony and rocky bottoms
• riverine areas (under back-water influence by sea)	• active cliffs	• inshore bottoms of shell gravel, gravel or coarse sands
• sills	• inactive cliffs	• inshore muddy or sandy bottoms with comparatively high biodiversity and/or bioproductivity
• shallow inlets, bights, lagoons, "boddens" (hard, sandy and soft bottoms)	• sand banks, bars and small spits	• offshore banks and stony or rocky reefs
• fjords / fjärds	• beaches of sand, pebbles and shingles	• inshore banks and stony or rocky reefs
• r iver mouth areas	• beach ridges	• gas vent communities
	• reed stands	• mussel beds
	• pools, rock pools, fladas and glo-lakes	• seagrass meadows
	• coastal wetlands (bogs, swamps and fens)	• large *Fucus* stands
	• coastal marshes and meadows	• other large stands of macrophytes
	• coastal dry grasslands, heath and shrubberies	• mud flats, sand flats, rock flats (so called "hydrolittoral")
	• natural wet forests under marine influence	
	• virgin and natural coastal forests (e.g., beech, pine, alder, ash) up to 1 km from the mean-water line	
	• primary forests on upheaval areas	

Table 7.3 'First Aid List' of landscape types, large biotope complexes and biotope types of high nature value in the Baltic Sea Area

7.3.6 Red Data Book

HELCOM 15 requested EC NATURE to elaborate a 'Red Data Book' on biotopes in the Baltic Sea Area. The work started with two workshops on the island Vilm in Germany, one in December 1994 and one in October 1995. It was agreed to compile two types of lists: A 'Complete List of Coastal and Marine Biotopes' will serve as basis for a coming Red Data Book. A list of coastal and marine landscape types, large biotope complexes and biotope types of high value in the Baltic Sea Area will function as a 'First Aid List' (Table 7.3). It is up to each country, to use the latter immediately as a nature-conservation instrument, e.g., for the general protection of certain biotopes. This list is not considered to be a complete list of all possible biotopes and their sub-structures in the Baltic Sea Area. Among the listed biotopes are those which are of particular ecological value and which show natural or next to natural conditions, i.e., little or no direct modification or direct disturbances by human activities.

Nevertheless, the elaboration of the Red Data Book, based on a complete list of Baltic biotope types, remains the major task for the experts. Meanwhile, the project group finalized the work on

• a "Complete List of Biotopes" and explanations of all Baltic biotope types,
• a criteria system for the threat evaluation to be used for the work on the Red Data Book, and on
• the following list of common threat factors to be used in the Red List:
- agriculture (intensive, changing, land reclamation, stop of traditional farming),
- construction, dredging, dumping,
- climate change (largely irreversible to man),
- eutrophication (fertilisation, sewage, combustion),
- fishing, aquaculture, hunting,
- building activities for recreation purposes, e. g., summer houses, marinas,
- wear (traffic, tourism),
- mineral extraction (prospecting, mining, dredging),
- pollution (non-eutrophication) of air, soil and water (pesticides, waste disposal, sewage, combustion, oil),
- forestry (deforestation, changes),
- water regulation (drainage, rerouting, extraction/desiccation, land reclamation),
- military activities (shooting, bombing, etc.), and
- coastal defence, e.g., dyking, stabilisation of sand.

One future task is to establish national working groups to compile the national data. Due to the lack of detailed biotope mapping in most countries, the experts in each national working group have to evaluate threats according to their personal scientific knowledge. The finalization of the Red Data Book is expected in 1998.

7.3.7 Red List of coastal and marine threatened species

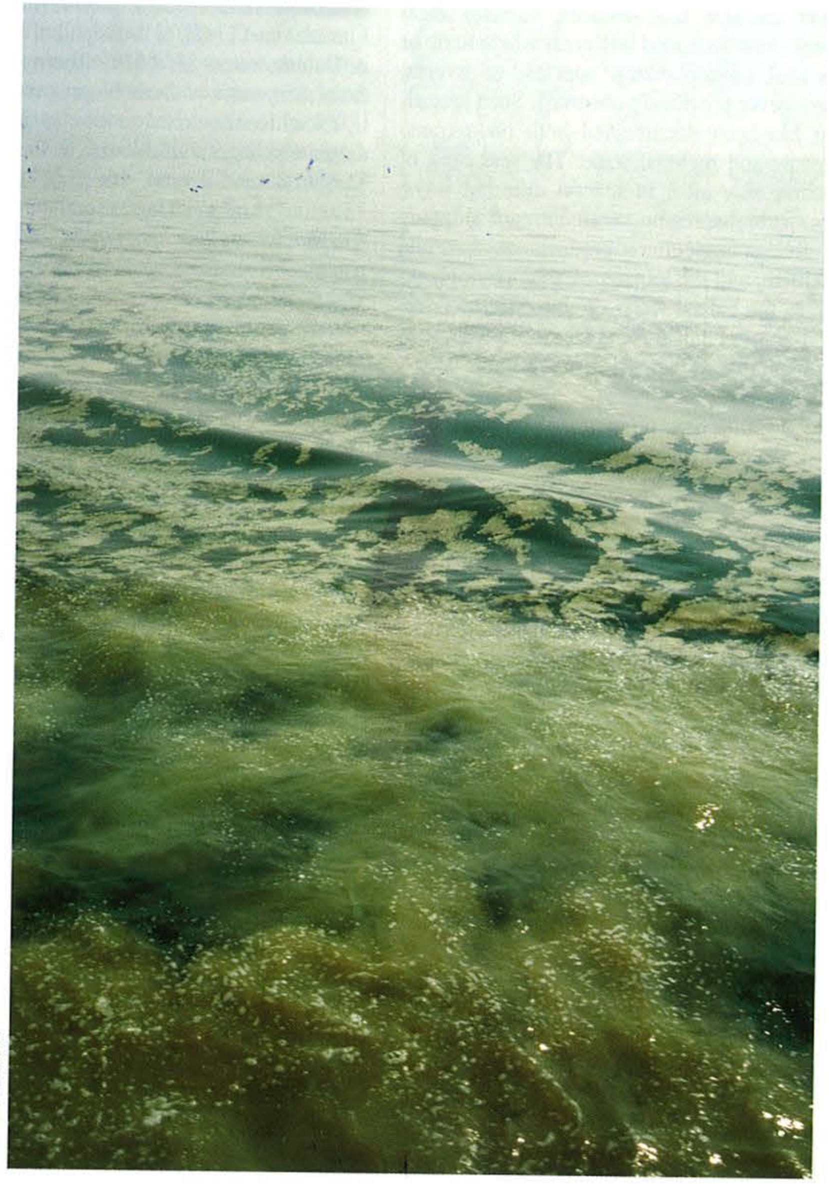
The Swedish Environmental Protection Agency and the Threatened Species Unit, in collaboration with specialists of most Baltic countries, have compiled data on threatened vertebrate and plant species. The results were published in the 'Red Data Book on Threatened Species in the Baltic Sea Region'. Presently, a survey has started of the state of invertebrate and cryptogam species in the Baltic Sea Region. The work includes a compilation of existing national red lists, establishment of co-operation between experts from different countries, agreements on criteria for red-listing and red data categories. The taxa are categorised according to IUCN's threat categories. Table 7.4 presents the number of plant, mammal and bird taxa of different threat categories in different areas of the Baltic Sea Region.

	Åland Sea	Finland	St. Petersburg Region	Estonia	Latvia	Lithuania	Kaliningrad Region	Poland	Mecklenburg-Vorpommern	Schleswig-Holstein	Denmark	Sweden
Extinct												
P	27	23	20	16	14	12	15	40	107	101	30	34
M	3	2	0	0	1	1	1	1	6	6	0	2
B	4	1	2	2	4	5	15	5	15	18	15	8
Endangered												
P	42	43	59	47	90	57	66	54	227	194	38	79
M	1	5	2	4	5	1	2	7	9	7	3	3
B	5	8	22	9	17	12	11	12	21	26	12	5
Vulnerable												
P	39	57	85	37	77	57	37	142	199	150	77	86
M	2	2	2	2	1	0	0	2	5	5	9	9
B	13	8	33	8	12	15	20	19	19	15	7	14
Rare												
P	34	69	138	44	69	62	9	146	37	45	123	142
M	2	4	6	9	9	3	17	23	8	0	4	4
B	16	4	23	16	38	20	28	37	12	22	36	13
Care demand.												
P	32	41	75	5	25	-	-	-	132	162	-	79
M	1	3	5	-	-	-	0	-	10	12	3	5
B	13	13	28	2	-	-	2	-	35	31	19	51
Indetermin.												
P	-	6	1	6	-	7	-	36	27	-	-	-
M	0	-	0	3	8	12	2	3	-	-	-	-
B	1	-	2	4	7	16	3	7	-	-	-	-
Total												
P	174	239	378	155	275	195	127	418	729	652	268	420
M	9	16	15	18	24	17	22	36	38	30	19	23
B	52	34	110	41	78	68	79	80	102	112	89	91

Table 7.4 *Number of red-listed vascular plants (P), mammals (M), and birds (B) in different areas arranged according to threat category [279]*

SPECIAL PROBLEMS

8



RIITTA OLSONEN

HELCOM

8.1 HARMFUL ALGAE

L. Edler[1] (Convener), *K. Kononen*, *H. Kuosa*

8.1.1 General overview

Among the approximately 5,000 species of marine phytoplankton, 75 to 150 may be harmful. They may be harmful by contaminating marine organisms used for human consumption, resulting in human poisoning. They may kill marine organisms, such as wild and cultured fish, as well as birds and land-based wild and domestic animals. They develop such a high biomass that the recreational value of coastlines is jeopardized, and are as aerosols along the coast which may threaten human health. They may also be harmful without producing toxic substances, by the clogging of fish gills, and develop into such quantities that oxygen depletion, resulting in bottom animal and fish mortality, is caused during their decomposition.

The threats from harmful algae can thus be summarized as
* human health problems,
* poisoning of land-based wild and domestic birds and animals,
* poisoning and mortality of marine organisms,
* ecosystem damage,
* threats to the biodiversity of the sea, and
* economic losses.

Blooms of phytoplankton are common in the sea on a world-wide basis. In recent years, harmful algal blooms or events have become an increasing problem in coastal waters around the world. It is generally agreed that the increased frequency of blooms and harmful events, as well as increased spatial distribution of phytoplankton blooms and indigenous species all over the world is real [*19,626,627*]. At many places, this increase has been linked to the increase of the anthropogenic impact on coastal waters.

The Intergovernmental Oceanographic Commission is presently attempting to evaluate the economic impact of the harmful algal events on a world-wide basis. This, however, is extremely difficult, as many intoxications are never reported. Moreover, much of the impact of harmful algae is difficult to estimate in economic terms. For instance, human intoxication due to toxic algae is considerable. It is roughly estimated that there are 50,000-150,000 cases of human poisoning every year.

Over the last few decades, harmful algal events have extended into areas where toxic or harmful phytoplankton species, or events, were never previously observed. Such spreading has been documented both on a trans-oceanic and regional scale. The spreading of species may arise in several different ways, e.g., natural spreading with currents, shipping of living mariculture products as fish and shellfish, and the transfer of alien phytoplankton in the ballast water of ships. Independent of the mechanism of species transfer, this problem is also increasing in the Baltic Sea.

In the Baltic Sea, problems of harmful algae also occur. In fact, considerable phytoplankton blooms during summer are a characteristic feature of the Baltic Sea. In the Baltic Sea, there are about 30 different species of phytoplankton which have proved to be harmful in the Baltic Sea and elsewhere. Among these, the blue-green algae are most common, forming spectacular blooms nearly every summer. These blooms may result in mortality of domestic animals, mainly cattle, dogs and ducks, through the drinking of water. Very large blue-green algal blooms have been reported from the Baltic Sea since the middle of the last century. Satellite imagery shows that areas up to 60,000 km^2 of the Baltic Sea may be covered by blue-green algal blooms during the summer [*325*].

Toxic events and poisoning caused by blue-green algae in the Baltic Sea have been reported since the beginning of the 1960s (Table 8.1.1, [*152,313,337*]). In addition to ducks, cattle and dogs dying from intoxication, there are also reports that people swimming in the sea during blue-green blooms have suffered from stomach complaints, headaches, eczema and inflammation of the eyes. In some years, there has also been prohibition of swimming along certain shores where blue-green algae have accumulated [*420*].

Small phytoplanktonic flagellates, e.g., *Prymnesium parvum*, have caused fish kills in coastal areas of the Baltic Sea, such as the Stockholm Archipelago [*30,711*], along the Finnish coast [*342*], at the island of Rügen and in Danish waters [*314,526*]. There are indications, that some of these blooms were caused by exceptional nutrient conditions. In 1988, an extremely large algal bloom hit the Kattegat, Skagerrak and Sound, covering an area of 75,000 km^2. The small flagellate *Chrysochromulina polylepis*, which was responsible for the bloom produced a toxin which killed or affected all sorts of marine organisms, ranging from other phytoplankton and zooplankton to seaweeds, mussels and wild and cultured fish. There was also some indication that the *Chrysochromulina* bloom had a direct or indirect negative effect on the survival of eider ducklings. The economic impact due to the bloom was considerable, especially for the fish-farming industry.

There are indications that a mass mortality of sea birds in the Gulf of Finland in 1992 may have been caused by algal toxins, although so far no definite link has been established. A special group of phytoplankton, the dinoflagellates, contains many toxic species. Several of them occur regularly in the Baltic Sea, especially in the Kattegat. Some of the toxins produced by these dinoflagellates have been found in the Baltic Proper, but toxic effects have not been reported. In the Kattegat and Skagerrak, however, mussels accumulate the toxins almost every year, causing economic loss following the harvest ban. Other dinoflagellates in the Kattegat occasionally form blooms with such a high biomass, that oxygen deficiency develops, when they sink to the deep water and consume oxygen during their decomposition. Since the beginning of the 1980s, oxygen deficiency in the Kattegat during the early autumn is a recurrent phenomenon which is spreading over larger areas.

Year	Location	Species	Affected animals	Reference
1900-33	Waterneversdorf (SBP)	*Prymnesium parvum*	fish	[*250*]
1963	Rügen (SBP)	*Nodularia spumigena*	~400 ducks	[*313*]
1970	Darß (SBP)	*Prymnesium parvum*	fish	[*314*]
1975	Fehrmarn (Belt Sea)	*Prymnesium parvum*	fish	[*250*]
1975	Danish coast	*Nodularia spumigena*	30 dogs sick, 20 died	[*420*]
1982	Swedish coast	*Nodularia spumigena*	9 dogs	[*152,414,430*]
1983	German coast	*Nodularia spumigena*	16 young cattle	[*338*]
1984	Finnish coast	*Nodularia spumigena*	1 dog, 3 puppies	[*538*]
1988	Skagerrak Kattegat, Sound	*Chrysochromulina polylepis*	large-scale effects	[*338*]
1990	small coastal inlet, Finland	*Prymnesium parvum*	fish	[*419*]
1990	Rügen (SBP)	*Prymnesium saltans*	fish	[*330*]
1991	shallow coastal lake	*Prymnesium parvum*	sea birds	[*256*]
1992	Eastern Gulf of Finland	?		[*326*]

***Table 8.1.1** Toxic plankton events reported from the Baltic Sea Area [338] (SBP - Southern Baltic Proper)*

[1] see ANNEX 11.3 for addresses of authors

8.1.2 Species distribution

Species distribution in 1993 - In the unattended monitoring system [*409*], 22 potentially toxic phytoplankton species were found. The material, covering 1993, shows the spatial and temporal distribution of the species. In 1993, *Aphanizomenon flos-aquae* was present in all sea areas during the growth season. Other blue-green algal species were more concentrated in the late summer and autumn. Many potentially toxic dinoflagellates were encountered in most areas of the Baltic Sea, and *Chrysochromulina* species were almost constantly present in all areas. The material clearly indicated differences in the distribution patterns.

The analysis of the distribution of harmful species material was based on information in the HELCOM database and on data gathered from national sources. The results are presented as species-distribution maps (Fig. 8.1.1).

Diatoms and other chrysophytes - A number of potentially harmful diatoms were found in the Kattegat, Belt Sea and Arkona Basin (Fig. 8.1.1). Only *Chaetoceros danicus* has been reported from the whole Baltic Sea. For some reason, *Chaetoceros borealis* was mainly reported from samples taken between 1979 and 1985. Since then it has been identified as *C. danicus* and thus the earlier reports may be false. For the Gulf of Finland, the misidentification of *C. danicus* as *C. decipiens* is also possible as the distribution pattern of *C. decipiens* appears to be rather unique, i.e., with plenty of reports just from the Kattegat and the Gulf of Finland, and no reports from other areas of the Baltic Sea. *Pseudonitzschia-pungens* has been a permanent part of the phytoplankton community in the Kattegat, Belt Sea and Arkona Basin. Possibly because its identification started only in 1990, *P. pseudodelicatissima* was reported only from 1986 to 1993.

Two potentially toxic small flagellated chrysophytes have been reported, *Chrysochromulina polylepis* and *Prymnesium parvum*. *C. polylepis* had a widely distributed bloom in 1988. However, it is not likely that the present material can give a full picture of the distribution of these two species, as their positive identification requires electron microscopy. A toxic strain of the silico-flagellate *Dictyocha speculum* caused fish kills in Danish waters in the 1980s. This species has been present in the area during the last ten years.

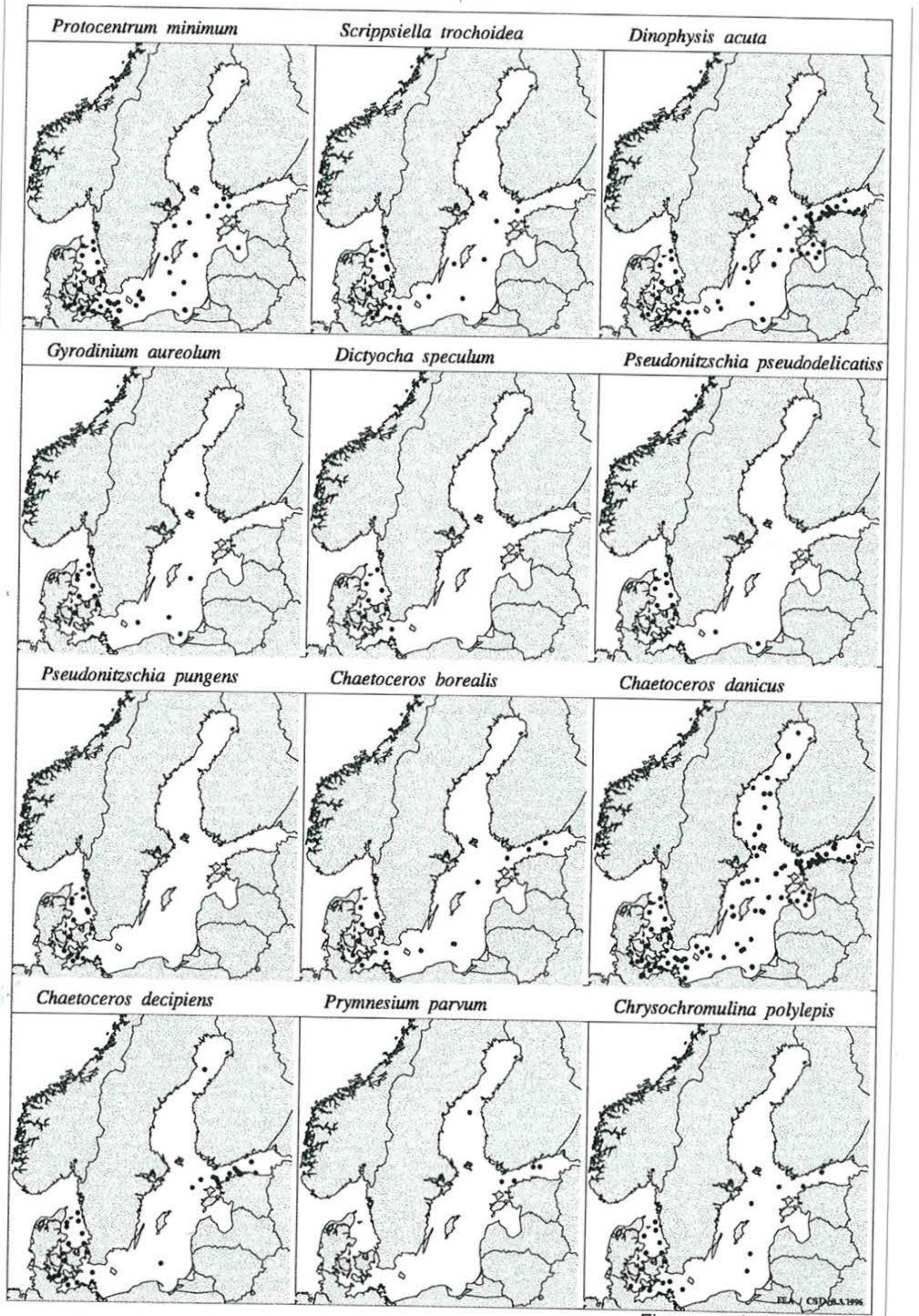
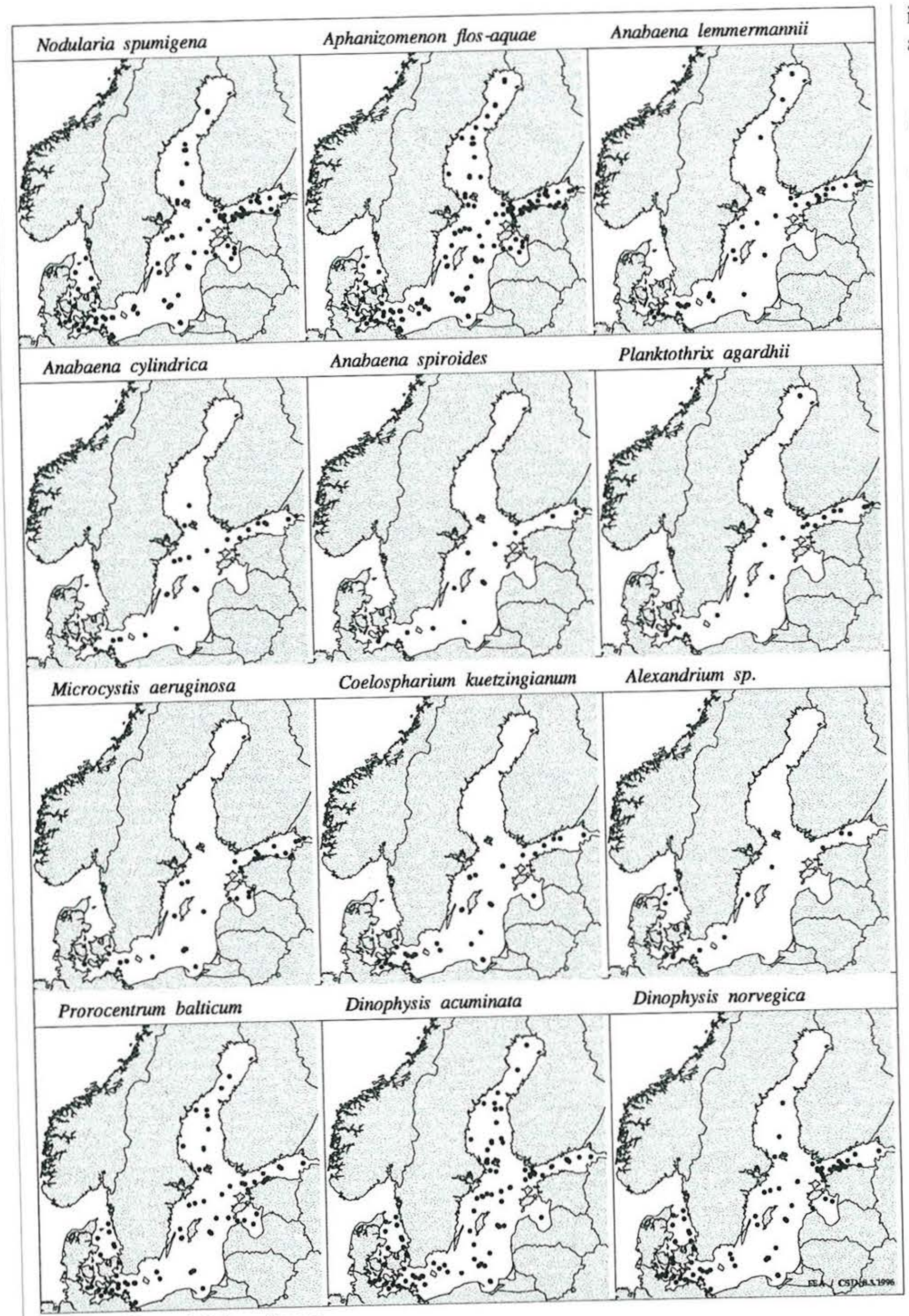


Figure continues

***Fig. 8.1.1** Species-related occurrence of harmful algal blooms in the Baltic Sea*

Dinoflagellates - Dinoflagellates are a group of phytoplankton with potential harmful effects on Baltic Sea ecosystems. Several genera of dinoflagellates consist of toxic species. Two widely distributed genera can be found over most of the Baltic Sea. These are *Dinophysis* and *Prorocentrum*. From the three *Dinophysis* species, *D. acuminata* may be found over the whole Baltic Sea and *D. norvegica* from all other areas except the Bothnian Bay. *D. acuta* is lacking from the Gulf of Bothnia. The distribution patterns of *D. acuminata* and *D. norvegica* showed no changes, but *D. acuta* has changed its distribution by expanding into the Baltic Proper and the Gulf of Finland during 1986-93. However, this should be subjected to a more detailed analysis as such a drastic change could also be due to problems in the taxonomy of the genus.



→ *Figure 8.1.1 continues*

Prorocentrum balticum has been a common species throughout the monitoring period, but it seems that *P. minimum* was more common in the Central Baltic Proper in 1986-93.

Another group of potentially harmful dinoflagellates includes marine species found more frequently from the Kattegat during 1986-93. These species include *Alexandrium* spp., *Gyrodinium aureolum* and *Scrippsiella trochoidea*.

Blue-green algae - Blue-green algal blooms are a common part of the Baltic Sea ecosystem. Although the heaviest blooms are not common in the Bothnian Bay, bloom-forming species can be found throughout the Baltic Sea. The species *Anabaena lemmermannii*, *Aphanizomenon flos-aquae* and *Nodularia spumigena* have a wide distribution (Fig. 8.1.1). Anabaena lemmermannii seems to occur over most of the Baltic Sea. However, no clear changes in the distribution of blue-green algae can be observed. Another component of the Baltic Sea blue-green algal community are species favouring fresh- or diluted brackish-water environments. These species include Coelosphaerium kuetzingianum, Microcystis aeruginosa and Planktothrix agardhii. In the assessed material, these three species showed the same distribution pattern. They were reported from the whole Baltic Sea except the Gulf of Bothnia. Two other species, Anabaena cylindrica and Anabaena spiroides, were more frequently encountered during the period 1986-93. However, the material lacks information on coastal waters, in which they are commonly found.

8.1.3 Summary

The assessment has shown that potentially harmful algae are present in all parts of the Baltic Sea. Thus concern on their distribution and the monitoring of their abundance is highly justified. Most of the potentially harmful species appear to be found mainly in the Kattegat, Belt Sea and Arkona Basin, but clear cases of distribution northwards have been documented.

The nature of the material restricts detailed analysis of certain species. However, some recommendations are possible:

- The identification of some species requires special techniques. If the distribution of small flagellates, e.g., *Chrysochromulina polylepis* and *Prymnesium parvum*, is to be monitored, their identification must be based on electron microscopic analysis. Also the species analysis of many dinoflagellates requires special techniques. A quality-controlled system should therefore be established for the analysis of taxa requiring special techniques.
- Some species have been reported to the database only for a certain period, e.g., from 1979 to 1985, and not later. This may be due to changes in the identification and/or taxonomy. However, when these changes occur, the existing database should be critically evaluated and corrected. A critical evaluation should also be made if a taxon is divided into two separate taxa, or if two or more are joined to one taxon.
- All records of potentially harmful species should be reported annually to an expert group which should confirm the records.

8.2 SANITARY CONDITIONS IN COASTAL WATERS

The hygienic quality of the beaches is of great importance for general recreational purposes, tourism and other uses. There are national as well as international standards and recommendations for control of the bacteriological quality of water. Among international guidelines can be mentioned those of the European Community (EC). One of the first directives within the EC in the environmental field was adopted in December 1975 (76/160/EEC).

The systems used for the determination of the sanitary conditions, e.g., for bathing and swimming, are generally based on examinations of bacteria, such as coliforms, streptococci, salmonella and enteroviruses in the waters. This control is often combined with analyses of physical and chemical parameters, in addition to visual inspections. The faecal coliforms, often being the key parameter in the assessments, have a limited survival in marine waters. This directs the problems to near-shore areas and especially to the mixing zone between fresh water and sea water.

All countries reporting on sanitary conditions have pointed out that the conditions at some beaches are particularly dependent on meteorological conditions, mainly the directions of wind or, e.g., a temporary cleaning of the beach during a storm. In these cases, the period for prohibition of a beach is dependent on the frequency of regular controls or initiatives taken by local authorities. Within a wide archipelago, where the isolation and temporary stagnation of water exchange is frequent, the beaches are especially sensitive.

The various control routines, the application of standards and the categorisation of conditions differ between the countries. It is therefore necessary, to present and comment data on an individual national basis. The text presented here is compiled by *Eeva-Liisa Poutanen*, Environment Secretary of HELCOM, on the basis of information provided by the countries.

Denmark

The classification of water quality for bathing mainly follows the EC directive, with the exception of the number of faecal coliforms, 1,000 per 100 cm^3, which is half the value accepted by EC. This limit must not be exceeded for more than 5 % of the bathing season.

For 1995, the data were presented together with a detailed bathing-water map. The situation in Denmark as a whole, including the western and inland waters, showed improved conditions compared to 1994. In 1995, there were 1,301 (1994: 1,288) monitoring stations, of which 1,225 (1994: 1,232) met the water quality requirements. Based on the results obtained at 22 (1994: 23) monitoring stations, the number of banned sites was 20 (1994: 21), but there were 52 (1994: 33) sites with doubtful bathing water quality.

A calculation of figures relating to the Convention Area for 1995 gives a coastline of approximately 6,000 km, of which about 4,000 km are suitable for bathing. Taken together, the sites, where bathing is prohibited, cover a coastline of 10 km in length.

The data presented in Table 8.2.1 show that 22 monitoring stations have prohibited bathing, which corresponds to 20 banned sites, while the number of sites with doubtful bathing water quality is 52, including a total of 54 monitoring stations with doubtful quality.

Table 8.2.1 *National reports on total number of beaches controlled mainly during 1995, and decisions taken for bans on bathing*

Country	Total number controlled	Doubtful quality	Bathing prohobited
Denmark *	990	33	19
Estonia	33	2	4
Finland	100	1	0
Germany	306	1**	0
Latvia	36	9	0
Lithuania	20	1	4
Poland	82	15	4
Sweden	500	30	1

* - monitoring stations (Denmark without the western and inland waters)
** - Schleswig/Holstein

Estonia

Health protection authorities control the sanitary conditions at 100 bathing places, including 24 larger beaches. Although, bathing conditions have been improved during the last years, improvement has been faster at the beaches of lakes and rivers than at the marine beaches. The *Coli-index* has been in a range of 10,000-70,000, with a pathogenic microflora being absent in most of the cases. Conditions at the former most polluted area, the Pärnu Bight, have been improved.

In July 1995, *Chloreae inaba* and *Chloreae ogava* were found on the northern coast at the Aa beach. After identification of the non-pathogenic character of the vibrions, the beach was re-opened. The *Heiberg vibrio* pathogens were only detected in occasional cases at the Aa beach and the Pärnu beach, but rather frequently at the beaches of the western coast of the Estonian mainland.

The Paralepa beach in Haapsalu Bight and the Stroomi beach (not listed for bathing) were closed during the summer of 1995. All the other beaches were opened for bathing.

Finland

The national classification of the Finnish beaches is based on the number of thermotolerant coliform bacteria and faecal streptococci per 100 cm^3. Values below 100 indicate good conditions, 100-1,000 moderate, while those exceeding 1,000 indicate poor quality.

The first national review concerning the hygienic quality of bathing waters in 1995 was recently published. 73 % of all beaches were in excellent condition according to the Finnish standards and only 1 % were poor. One hundred individual coastal beaches are included in the EU Bathing Water Directive reporting system. The quality criteria were met by 80 of them, while only one did not fulfil the obligatory requirements (although the poor quality of this beach was just temporary).

A short compilation of the hygienic status of 50 beaches of the five largest coastal cities was done in 1993. In general, the hygienic quality of water was good on most study occasions, and occasionally poor quality occurred in inlets with poor water exchange. During the last 4-5 years (from 1992), bans have been imposed on only two beaches in these towns (once because of the breakdown of a pumping station). The water protection measures conducted are reflected in the hygienic status of bathing water. For example, a third of beaches of Helsinki were repeatedly classified as poor in the 1960s. Since then, in the 1970s and 1980s, only a few similar occasions have been reported, and during recent years, more than half of the beaches are classified as good in Helsinki.

Germany

Almost 200 bathing sites along the 340 km of coastline of Mecklenburg-Vorpommern and 21 sites on the banks of Bodden areas have been investigated by government laboratories in Greifswald and Rostock, once a fortnight during the bathing season (15 May to 10 September) in 1990-95. Since 1990, investigations and assessments have been based on the EC guideline from 8 December 1975 for the quality of bathing water. The guideline states, how bathing waters are to be judged in accordance with hygienic criteria. Total coliform bacteria and faecal coliform bacteria, facultatively *Streptococcus faecalis* and, where justified, salmonella and intestinal viruses, are regularly investigated (cf. Table 8.2.2).

From a sanitary standpoint, chemical investigations are of secondary importance. They are carried out specifically only when a 'tendency towards eutrophication of bathing waters' exists.

The site of each sample was visited and an appraisal made of its standard hygiene with the following result:

a) Bathing sites found to be very suitable for bathing, i.e., no contamination was found: 1993 - 113, 1995 - 133;
b) Bathing sites found to be suitable for bathing, i.e., slight contamination was occasionally found: 1993 - 20, 1995 - 3;
c) Bathing sites found still suitable for bathing,

Table 8.2.2 Results of studies on the sanitary conditions at about 200 beaches at the shores of Mecklenburg-Vorpommern (Germany), 1993 and 1995

Parameter	Number of studies		Samples <Guide value		Samples >Guide value & <Imperative value		Samples >Imperative value	
	1993	1995	1993	1995	1993	1995	1993	1995
Total coliform bacteria	1,625	1,604	1,457 (89.7 %)	1,458 (90.0 %)	152 (9.3 %)	126 (7.8 %)	16 (1 %)	20 (1.2 %)
Faecal coliform bacteria	1,625	1,604	1,363 (83.9 %)	1,456 (90.8 %)	238 (14.6 %)	148 (9.2 %)	24 (1.5 %)	16 (1 %)
Faecal streptococci	23	150	13	148	10	2		

Guide values: total coliform bacteria per 100 cm^3 <500; faecal coliform bacteria per 100 cm^3 <100; *Streptococcus faecalis* per 100 cm^3 <100;

Imperative values: total coliform bacteria per 100 cm^3 <10,000; faecal coliform bacteria per 100 cm^3 <2,000

i.e., contamination was found:
1993 - 11, 1995 - 3.
Storms and strong accumulations of algae deposited on the beach impeded the collection of samples and occasionally influence the results in a negative way. No bathing sites were closed.

The tendency for the microbial contamination to decrease could possibly be explained by the installation of, or improvements in waste-water treatment plants (Greifswald, Stralsund, Bergen, Kühlungsborn, Wismar, island Poel).

Twenty one bathing sites on the banks of the 'boddens' (Darß Bodden Chain, Achterwasser /Usedom) have permanently a transparency <1 m, caused by eutrophication.

Latvia

The water quality at beaches is monitored from May to September. Bathing is not prohibited if the lactose-positive coliform bacteria (LPCB) content does not exceed 5,000 cells per liter, *E. coli* <1,000, staphylococci <100, coliphages <100, and no pathogenic microflora are found.

Beaches of good quality exist in the north-western part of the Gulf of Riga. Most polluted beaches were from the mouth of the river Daugava to the Estonian border, and bathing in this territory is prohibited. The water quality of Liepaja City beaches occasionally does not meet the requirements, and bathing was prohibited for several weeks.

A tendency for microbial contamination to decrease is also observed at Jūrmala beaches. This could be explained by the recent operation of the Riga water treatment installations, as well as by a decrease in the number of visitors to the area. Bathing in Jūrmala was prohibited only in some cases in year 1993.

Lithuania

According to the 1988 requirement, the *Coli-index* was 500 per 100 cm^3. After 1992, this standard was changed to 100-500. The hygienic service permits beach activities (e.g., bathing) when the *Coli-index* does not exceed 2,500/100 cm^3 and there is an absence of pathogens.

During 1992-93, beaches in the vicinity of Klaipeda were periodically closed. In 1994, this happened only on two occasions, but in 1995, there were no closures. The sanitary conditions are affected by discharges from the town and by the direction of the wind. Maximum mean values for the season, 10,000/100 cm^3, were reported from beaches at Melnrage during 1991. Via the Klaipeda Strait, the mixture of river water and waste products closely follows the coastline, moving mainly northwards. The best water quality for bathing is reported from the Neringa beaches which are located along the coast south of Klaipeda, facing the Baltic Proper.

Poland

The faecal coliform index is used as the key parameter of bacterial pollution level. Waters are classified into four quality groups, according to the value of this index, with the number of faecal coliform bacteria per 100 cm^3 increasing in a logarithmic scale, i.e., water class
- *I* - ≤100,
- *II* - 100-1,000,
- *III* - 1,000-10,000, and
- *IV* - >10,000.

In compliance with legal regulations, only the water quality groups *I* and *II* are allowed to be used for bathing and water sports.

The frequency of collecting water samples, the length of the research period and the method of determining bathing usability are regulated by the Ministry of Health and Social Care. According to them, the examination of beaches needs to be continued for a period of 9 months (March-November), and only at those sites, where at least 70 % of the results comply with the criteria for the water quality groups *I* and *II*, are allowed to be used for bathing.

On the basis of the examination results and the above mentioned method for estimating sanitary conditions of sea water, all controlled bathing beaches are classified as either
a) very clean (if all results comply with the criteria for groups *I* and *II*),
b) clean (if at least 70 % of the results comply with the criteria for groups *I* and *II*),
c) uncertain (in case of minimal excess of the item b) requirements),
d) polluted (if more than 10 % of the results comply with the citeria for group *IV*, but if there have been also results complying with the quality groups *I* and *II*), or
e) seriously polluted (if more than 10 % comply with the citeria for group *IV*, and there have not been results complying with the criteria for groups *I* and *II*).

Irrespective of the level of faecal coliform bacteria pollution, the finding of the *Salmonella* sp. classifies the bathing beach as polluted or seriously polluted.

At the majority of stations, the highest level of pollution was observed in 1992, and the lowest one in 1995. Increasing numbers of clean bathing beaches were noted in the Slupsk, Gdansk and Elblag voyvodships. It is worth stressing, that the increase at Elblag voyvodship was particularly high, i.e., from 35 % to 71 %. Positive changes noted since 1995 are the results of efforts made by local authorities to decrease the pollution load discharged into the sea. The scale of the improvement recorded in 1995 is compared with the results of the four-year research cycle (1992-95) and presented in Figure 8.2.1.

Generally, the sanitary state of the water at the Polish coastal zone in 1992-95 may be described as 'satisfactory' based on the fact that 72 % of all bathing beaches were 'very clean' and 'clean', 16 % 'uncertain', 9 % 'polluted', and only about 3 % 'seriously polluted'. The sanitary state of the bathing beaches situated near the open sea is considerably better than those situated in Gdansk Bight. This is shown below in percentage of the total number of bathing beaches for each category.

	very clean and clean	uncertain	polluted	seriously polluted
open sea	84	14	2	0
Gdansk Bight	42	21	27	10

A permanent improvement of the sanitary state of the Polish bathing beaches has been recorded for the last three years.

Sweden

A summary of the bacteriological quality of the Swedish bathing areas has been done for 1993-95, and bathing maps for each county were produced. During that period, the EC directive was not followed and no chemical analyses were done. The frequency of sampling varies considerably, from once a week to once a year. The recommendation is, that the samples should be taken twice a month during the bathing season.

The limits for classification of Swedish beaches are given in numbers per 100 cm^3:

	Suitable	Suitable with reservations	Unsuitable
Presumptive *E. coli*	<100	100-1,000	>1,000
Presumptive faecal streptococci	<30	30-300	>300
Coliform bacteria 35 °C		>1,000	

Around 500 beaches are controlled along the Swedish coast. Generally, the water quality is very good, but there are a few places with higher amount of faecal bacteria, especially along the most southern part of Sweden. This happens occasionally, and often in connection with heavy rain. During the period 1993-95, 30 beaches have had doubtful quality during one or more years. Only one beach is classified as not suitable for bathing.

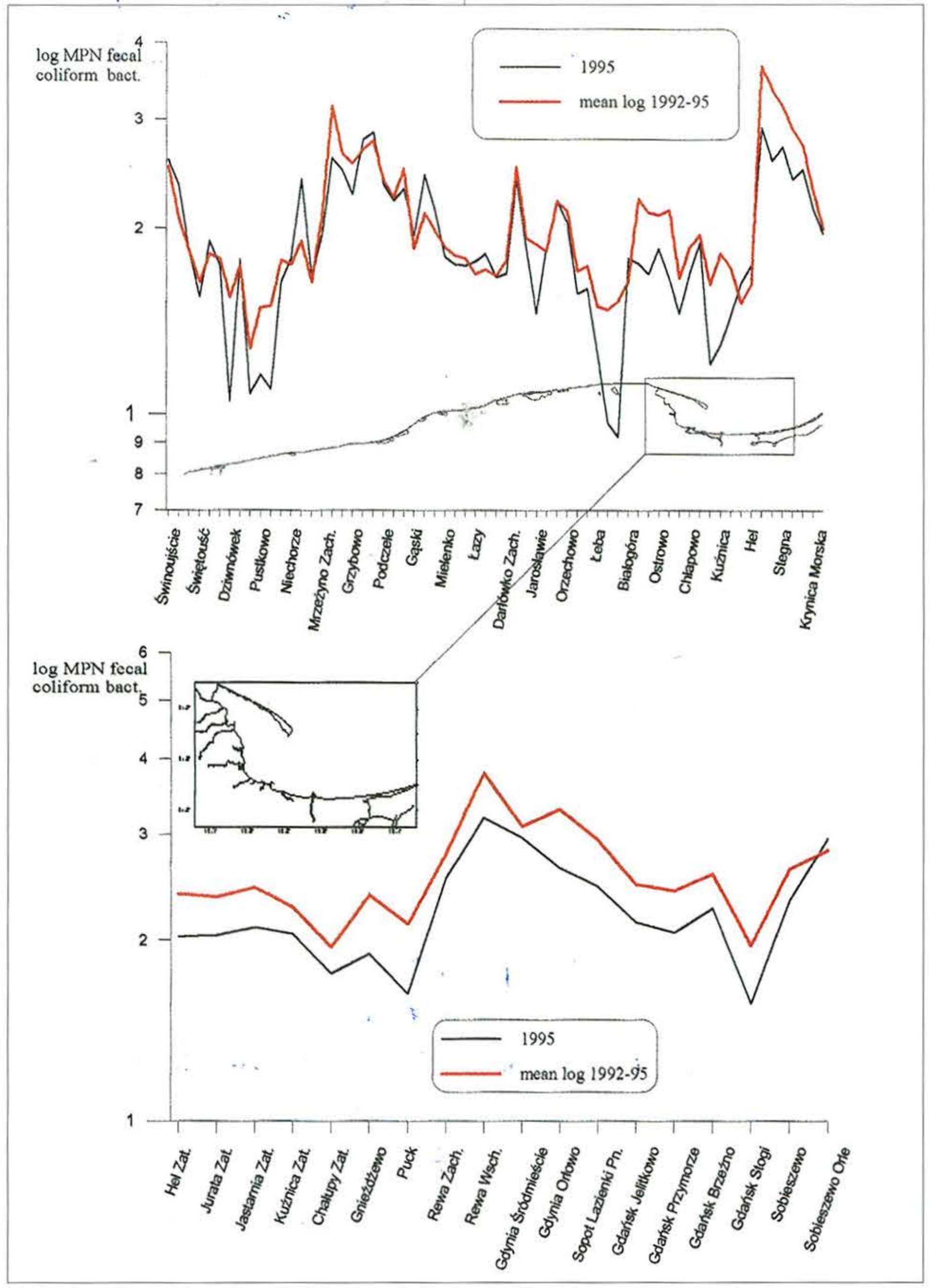


Fig. 8.2.1 Sanitary conditions of coastal waters along the Polish shore of the Baltic Sea, and of the Gdansk Bight in more detail, 1995, compared to the mean value during the period 1992-95

8.3 DUMPING OF CHEMICAL MUNITION

8.3.1 Introduction

The various problems to the environment, arising from dumping of chemical munitions in the Baltic Sea Area many years before the Helsinki Convention was signed, have been dealt with at several meetings in the framework of the Convention. In 1992, the Helsinki Commission decided to convene a special working group to deal with problems related to dumped chemical munitions, taking into account that, in the beginning of the 1990s, intensive rumors circulated in the press and elsewhere about new dump areas for chemical munitions in the Helsinki Convention Area, and that >300,000 t of chemical munitions had been dumped in the Convention Area.

Information on the chemical munitions dumped in the Baltic Sea was compiled by the special working group (*ad hoc* Working Group on Dumped Chemical Munition, HELCOM CHEMU), under the leadership of Denmark, on the basis of national reports provided to the Helsinki Commission by all the Contracting Parties and observers, including from United Kingdom, United States of America and Norway, as by end of 1993. The 'Report on Chemical Munitions Dumped in the Baltic Sea' was submitted for approval by the Commission in 1994 [*239*].The report does not contain information on munitions dumped after World War I, neither does it contain information on dumping of conventional ammunition. Information on dumpings up to 1947 is included in the report (except for the dumping of 200 t by the former GDR in the 1950s).

The mandate of the working group was prolonged by the Commission in 1994, with a request, to follow and implement the substantial recommendations provided in the report. All the Baltic Sea States were requested to provide the Commission by 1995 with information and official documentation concerning chemical munitions dumped after 1947.

Following the decision by the Commission, Denmark, Estonia, Finland, Germany, Latvia, Lithuania, Poland and Sweden subsequently reported that there were no dumping activities of chemical munitions by their respective countries after 1947. Information on the dumping activities by the former GDR after 1947 was already included in the report to the Commission [*239*]. Russia confirmed that the

Russian national report submitted to HELCOM contained all available information on dumping activities by the former USSR [230].

The 16th meeting of the Commission endorsed the final report of the working group [242] and agreed on several proposals for further actions, which mainly dealt with studies and investigations of the chemical processes of warfare agents and ecological effects of such processes. Furthermore, the Commission confirmed that, according to existing knowledge, dissolved warfare agents are not a wide-spread risk to the marine environment.

8.3.2 Types, quantities and properties of dumped munition

Chemical warfare agents are chemical compounds which through chemical or biochemical reactions interfere with the physiological functions of the human organism in such a way, that the combat capability of soldiers is impaired or that death is caused. Chemical warfare agents are gaseous, liquid or solid substances for anti-personnel use, they are mostly contained in shells and bombs. They are released in the air or sprayed.

Chemical weapons were used in World War I and caused the deaths of around 100,000 men and disabled about 1.2 10^6. Although large amounts of chemical warfare agents were produced and developed during World War II (around 65,000 t of warfare agents in Germany), they were never used in Europe. Mustard gas was the most widely produced, accounting for around 39 % of total production. In Table 8.3.1, the quantities of chemical warfare agents produced in Germany are shown.

Based on their effects, the chemical warfare agents can be classified into
- tear gases (lachrymators): chloroacetophenone (CAP),
- nose and throat irritants: Clark I, Clark II, Adamsite,
- lung irritants: phosgene, diphosgene,
- blister gases (vesicants): sulphur mustard, nitrogen mustard, Lewisite,
- nerve gases: tabun, and
- additives, such as monochlorobenzene, are made to the warfare agents in order to change their physico-chemical properties.

Furthermore, the dumped chemical munitions might also contain certain amounts of explosives. Leaching of persistent and bioaccumulable substances (N-compounds) from this material might occur. It is worth noting, that most specified quantities of the dumped chemical warfare materials referred to in this chapter are gross weights based on the weight specifications for the material when it was dumped.

Around 34,000 t of chemical munitions, containing about 12,000 t of chemical warfare agents, were dumped east of Bornholm and near Gotland in 1947 and 1948, on orders of the Soviet Military Administration in Germany (SMAD) [230]. In Tables 8.3.2 to 8.3.4 are given the types of chemical munitions and the amounts of chemical warfare agents that were dumped.

Earlier, it was estimated that between 36,000 t and 50,000 t of munitions have been dumped east of Bornholm and south-east of Gotland (south-west of Liepaja). These munitions contained chemical warfare agents of the types blister-, vomiting-, tear agents and phosgene. Based on the information currently available, the estimate of the quantity of chemical munitions dumped east of Bornholm and south-east of Gotland can be reduced to around 34,000 t. However, due to the new information [230] and the high proportion of aircraft bombs, the average chemical agent content was higher than the 15 % hitherto assumed. In earlier estimates, the quantities of chemical agents have been calculated to be about 6,000 t based on a 15 % level of chemicals in the munitions. Table 8.3.5 gives an overview of quantities of chemical munitions and warfare agents dumped in the Helsinki Convention Area.

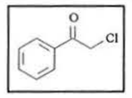
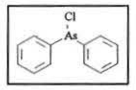
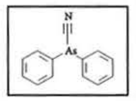
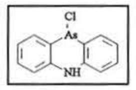
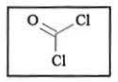
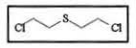
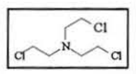
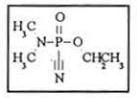
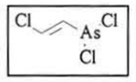
Warfare agent	Quantity [t]	Structure	Melting point [°C]	Boiling point [°C]	Vapour pressure [mmHg] 20 °C	Density [g cm⁻³]	Aqueous solubility [g dm⁻³]
Chloroaceto-phenone (2-Chloro-1-phenyl-ethanone)	7,100		54-56	244	$13x10^{-3}$	1.32	1
Clark I (Diphenyl arsine chloride)	1,500		38-44	307-333	$16x10^{-4}$	1.422	2
Clark II (Diphenyl arsine cyanide)	100		30-35	290-346	$47x10^{-6}$	1.45	2
Adamsite (10-Chloro-5-hydrophen-arsazine(10))	3,900		195	410	$2x10^{-13}$	1.65	$2x10^{-3}$
Arsinic oil*	7,500						
Phosgene (carbon dichloride oxide)	5,900		-128	7.6	1,178	3.4	9
Mustard gas (2,2'-Dichloro-diethyl-sulfide)	25,000		14	228	0.72	1.27	0.8
Nitrogen mustard (2,2',2''-Trichloro triethylamine)	2,000		-4	235	$11x10^{-3}$	1.24	0.16
Tabun (P-Cyano-N,N-dimethyl phosphonamid acid ethyl ester)	12,000		-50	246	0.07	1.07	120
Lewisite (dichlor-(2-chlorvinyl)-arsane)	Production small, but unknown		-18	190	0.35	1.89	0.5

Table 8.3.1 Important chemical warfare agents produced in Germany between 1935 and 1945 [229]

*Mixture of arsenic-containing compounds with the main ingredients being Pfiffikus (phenyldichloroarsine), Clark I, arsenic trichloride and triphenyl-arsine

About 200-300 t of chemical munitions residues, discovered after 1952 in the former GDR, were dumped east of Bornholm [229]. In addition, it should be mentioned that witnesses reported that four ships containing around 15,000 t of chemical munitions were dumped south-west of Rønne (Bornholm) in 1946. Witnesses also reported that, in 1956, four decommissioned East German coastal patrol vessels were loaded with chemical munitions (around 50 t) and were sunk south-west of Rønne, and unconfirmed reports claim that about 8,000 t of chemical munitions were dumped east of Bornholm in addition to those mentioned in Table 8.3.4 [229]. None of these dumpings have been confirmed by other sources.

In 1960, the tabun shells, which were sunk at the southern entrance to the Little Belt, were raised, leaving a residue of about 5,000 t of chemical munitions (phosgene and nerve gas). However, to date, no catch or findings of chemical munitions or parts thereof have been registered in the specific area. Only phosgene and tabun munitions were dumped, and these substances are rapidly degraded in sea water. In addition, mostly thin-walled bombs, whose casings rust fairly quickly, were dumped at this location [229].

Types of chemical munitions and quantities of warfare agents dumped in the Helsinki Convention Area under control of the former Soviet Union -in t- [230]

	Mustard gas	As-cont.	Adamsite	CAP	Others	Total
Aircraft bombs	6,432	984	642	520	-	8,578
Artillery shells	729	-	66	39	-	834
High-explosive bombs	341	-	-	-	-	341
Mines	46	-	-	-	-	46
Encasements	87	221	753	-	80	1,141
Smoke grenades	-	-	71	-	-	71
Containers	-	1,004	-	-	-	1,004
Drums	-	-	20	-	-	20
Total	7,635	2,209	1,552	559	80	12,035

Table 8.3.2 Total amounts

	Mustard gas	As-cont.	Adamsite	CAP	Others	Total
Aircraft bombs	512	78	51	41	-	682
Artillery shells	58	-	5	3	-	66
High-explosive bombs	27	-	-	-	-	27
Mines	4	-	-	-	-	4
Encasements	7	18	60	-	6	91
Smoke grenades	-	-	6	-	-	6
Containers	-	80	-	-	-	80
Drums	-	-	2	-	-	2
Total	608	176	124	44	6	958

Table 8.3.3 Southeast of Gotland (southwest of Liepaja)

	Mustardgas	As-cont.	Adamsite	CAP	Others	Total
Aircraft bombs	5,920	906	591	479	-	7,896
Artillery shells	671	-	61	36	-	768
High-explosive bombs	314	-	-	-	-	314
Mines	42	-	-	-	-	42
Encasements	80	203	693	-	74	1,050
Smoke grenades	-	-	65	-	-	65
Containers	-	924	-	-	-	924
Drums	-	-	18	-	-	18
Total	7,027	2,033	1,428	515	74	11,077

Table 8.3.4 East of Bornholm

	CMs (t)	WAs (t)	WA type
Bornholm Basin (East of Bornholm)	~32,000	~11,000	mustard gas, viscous mustard gas, Clark I, Clark II, Adamsite, chloroacetophenone; (less certain: phosgene, nitrogen mustard, tabun)
East of Bornholm	8,000 (n.v.)	-	no information
Area SW of Bornholm	~15,000 (n.v.)	-	no information
Gotland Basin (SW of Liepaja)	~2,000	~1,000	mustard gas, Adamsite, chloroacetophenone
Little Belt	~5,000	750*	tabun, phosgene
Måseskär, W of Sweden (outside the Helsinki Convention Area)	~20,000 (q.n.v.)	-	mustard gas (other types n.v.)

* (estimated at 15%)
n.v. - not verified
q.n.v. - quantity not verified

Table 8.3.5 Quantities of chemical munitions (CMs) and types and quantities of chemical warfare agents (WAs) dumped in the Helsinki Convention Area [228-230], and at Måseskär (west of Sweden) in the southern part of Skagerrak [231]

Dumping outside the Helsinki Convention Area - In 1964, 462 tabun shells were recovered from Wolgast harbour (former GDR), set in concrete blocks and dumped in the Norwegian Sea [229]. From 1945 to 1948, on the orders of the British and American occupation forces, confiscated German merchant ships were loaded with large quantities of chemical munitions and these were subsequently sunk. Records show, that one ship was sunk in the Norwegian Sea and 26 named and 6-8 unnamed vessels were sunk in the Skagerrak, together with an estimated 130,000 t of chemical munitions and conventional ammunition, at a position 25 nautical miles south-east of Arendal in the Norwegian Trench [228,229,232,241].

In addition, the wrecks of eight rather small naval vessels and a medium-large cargo vessel were sunk at a depth of 200 m at a position west of Måseskär lighthouse in the Skagerrak, i.e., just outside the Convention Area. The quantity here has been estimated by Swedish authorities to be approximately 20,000 t of chemical munitions containing mustard gas. However, the presence of other types of chemical warfare agents, including nerve gas, cannot be ruled out [*231*]. It has been confirmed by United Kingdom [*232,241*], that ships with chemical munitions were sunk in this area. However, no information about quantities and types of chemical munitions has been obtained.

Furthermore, another two vessels with chemical munitions were sunk at a position close to the parallel of the Skew in the Skagerrak, close to the border of the Helsinki Convention Area. These are probably the ships, which were sunk on the orders of the French occupation authorities, and 1,500 t of chemical munitions were dumped in connection with this operation [*229*].

In summary, it can be stated that with relative certainty around 40,000 t of chemical munitions have been dumped in the Helsinki Convention Area. It is estimated, that the chemical munitions contained no more than 13,000 t of chemical warfare agents. This figure does not take into account the dilution and degradation which have subsequently taken place. No information on types of hitherto unknown chemical munitions or warfare agents has been revealed.

8.3.3 Dumping areas

Dumping areas in the Helsinki Convention Area were identified at locations south-east of Gotland (south-west of Liepaja), east of Bornholm and south of the Little Belt (Fig. 8.3.1). In the Little Belt, the water depth is around 30 m, and the sea bed is covered with mud up to 8 m thick. The rates of sedimentation here are 1-2 mm yr^{-1}. Accordingly, a thickness of up to 10 cm may be reached after 50 years. Therefore, the munitions can be expected to have sunk into the soft and muddy sediments. The surface current mainly flows north-west and south-east at a speed of 0.3-0.5 m s^{-1}.

Dumping east of Bornholm in the Bornholm Basin was primarily inside a circular area with a radius of 3 nautical miles. However, it must be assumed that the chemical munitions were spread over a considerably larger area during dumping. Several factors indicate this, e.g., the positions where fishermen have caught munitions in their nets and the circumstances of the dumpings [*228-230*].

The dumping operations south-east of Gotland took place in the Gotland Basin within several positions as specified in [*239*]. The water depth in the dumping area is between 70 m and 120 m. In general, the hydrographic conditions are similar to those in the Bornholm Basin, with very stable stratification of the water masses and only a slight bottom current.

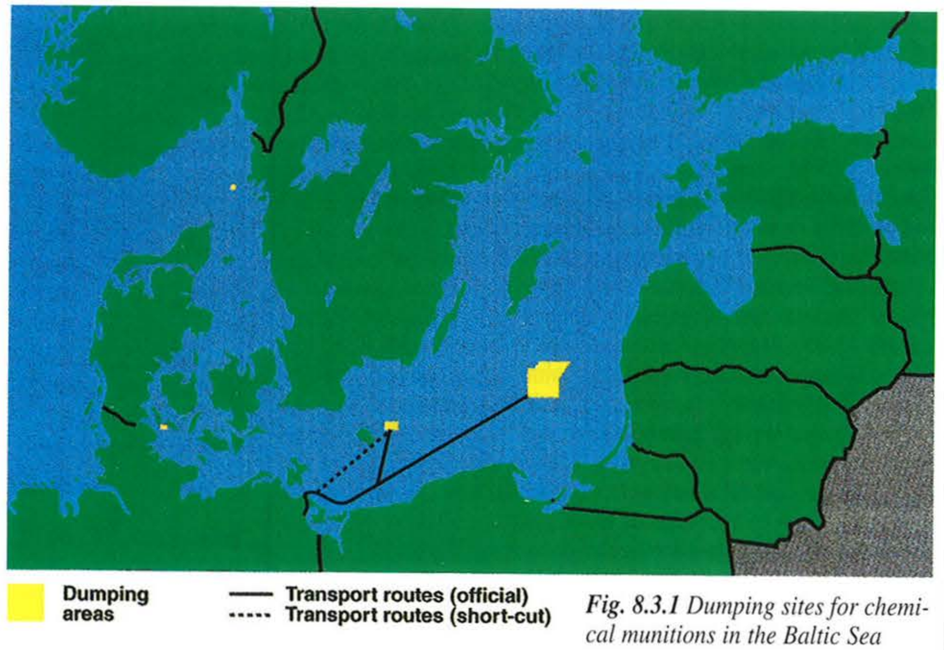


Fig. 8.3.1 *Dumping sites for chemical munitions in the Baltic Sea*

8.3.4 Dumping methods

During transport to the dumping area east of Bornholm, munitions were sometimes thrown overboard while the ships were en route. Therefore, warfare agents are assumed to be spread over a considerable area along the transport routes. Furthermore, the actual dumping of munitions may have taken place while the vessels were either drifting or underway. The first dumping operations took place while the munitions were still packed in wooden boxes, which sometimes were observed to drift around before sinking to the bottom of the sea. It is stated, that in some cases the boxes were washed ashore on Bornholm and on the Swedish coast.

Buoys marking the dumping positions were laid out relatively late, i.e., after the dumping had been completed. At that time the dumping vessels were only equipped with strictly necessary navigation equipment, therefore in many cases the exact dumping positions are uncertain.

The chemical warfare agents in the eastern part of the Helsinki Convention Area were mainly dumped in the form of munitions or contained in containers. This is in contrast to the method used in the Skagerrak and southern Little Belt, where complete ships were sunk. The very nature of this latter dumping operation has apparently prevented munitions being dumped outside the area where the ships were sunk. The fact that the warfare agents were inside a ship hull has also prevented further spreading.

8.3.5 Present conditions of the dumped chemical munition

The chemical warfare agents in the Helsinki Convention Area were mainly dumped in munitions, mostly in bombs and shells. In addition, warfare agents that had not been loaded into bombs were dumped in containers.

The munitions pose a threat only when the warfare agent inside is released. This can occur suddenly in an explosion, e.g., caused by mechanical stress during a recovery operation, or slowly as the walls of the shells corrode. In the case of the handling of warfare munitions, that had been dumped in the Helsinki Convention Area, such an explosion has never occurred as far as is known.

The condition of the munitions varies since it depends on a number of factors. These factors include the original wall thickness, the material of which the body of the munitions and the igniter is made, i.e., iron or aluminium alloys, and the nature of the dumping area such as solid ground, where munitions lie exposed to the water or mud, where munitions lying buried in the sediment are cut off from an oxygen supply.

In autumn 1971 and spring 1972, the West German Army raised 28 bombs and 15 shells, which contained phosgene and tabun, from the southern Little Belt. The recovered munitions had sunk about 50 cm into the mud. An examination revealed, that most had been corroded and no longer contained warfare agents. No traces of warfare agents were found in the sediment and water samples taken in the immediate vicinity [*229*].

In 1989, the research institute of the Norwegian Ministry of Defence (Forsvarets Forskningsinstitutt) undertook an extensive investigation of the ships loaded with munitions that had been sunk in the Skagerrak. Most of the bombs found in the wrecks or nearby had not yet completely rusted through, but some were found in this condition [*167*]. Off Bornholm, according to the Danish authorities, greatly corroded munitions and lumps of viscous mustard gas were mainly found.

In conclusion, due to the large number of parameters, theoretical considerations or calculations cannot be used to comment on the condition of the munitions in a particular dumping area. Investigations so far have found either intact munitions and completely corroded casings, which do not contain warfare agents.

8.3.6 Behaviour in the environment

An overview of the most important physico-chemical properties, influencing the behaviour of warfare agents in the environment, is given in [*239*]. The behaviour of chemical substances in the marine environment depends both on the chemical and physico-chemical properties of the substances and on environmental factors, such as temperature, salinity and the pH value of the water. As the pH value of sea water is rather constant around 8, salinity and temperature are the main environmental parameters that influence chemical reactions here. The solubility of the compounds and the speed of chemical reactions both increase with a rise in temperature. With an increase in temperature of 10 °C, the speed of reactions generally doubles. Water temperatures in the Baltic Sea vary between 0 and 20 °C, i.e., reactions occur 4 times faster at 20 °C than at 0 °C. However, in the water above the seabed in the Baltic Sea, the temperature variation is much less, typically between 2 and 12 °C.

Dissolution of the chemical warfare agents into the sea is considered as the crucial first step in the degradation of the compounds. Besides a rise in temperature, the process of dissolution is increased by current movement. The solubility of the various chemical warfare agents varies from good (tabun) to very poor (Adamsite, viscous mustard gas). However, it should be noted, that poor solubility retards the process of degradation.

The behaviour of warfare agents in the marine environment is additionally influenced by the physical properties of the agents. For instance, a warfare agent in viscous or highly viscous form, or in lump form, can be caught in nets. This cannot happen to substances in liquid or powder form. This is one reason why most accidents with warfare agents so far have involved viscous mustard gas. Because of the admixture of thickeners, viscous mustard gas is the only warfare agent occuring in large lumps that are mechanically relatively stable. Other warfare agents are also resistant to sea water, e.g., Clark and Adamsite.

All warfare agents react with sea water, but reaction rates can vary enormously depending on the chemical structure of the different agents. Through hydrolysis, new compounds are formed which have different properties from those of the original warfare agents. Such reaction products are usually no longer toxic or are less toxic and generally dissolve better in water. Investigations on the behaviour of warfare agents under Baltic Sea conditions exist only for a few substances. For this reason, their behaviour can often only be described qualitatively, as details of the rates at which the processes occur are not available.

Almost all warfare agents are broken down at varying rates into less toxic, water-soluble substances. Some compounds, however, show an extremely low solubility and slow degradability (viscous mustard gas, Clark I and II, and Adamsite). These, however, cannot occur in higher concentrations in the water, therefore a wide-scale threat to the marine environment from dissolved chemical warfare agents can be ruled out. However, elevated levels of sparingly soluble Clark, Adamsite or mustard gas in viscous form might occur in the sediment in the immediate vicinity of dumped munitions.

Relocation by currents and threat to the coast - Two ways of relocation of dumped chemical munitions have been considered by the working group [*239*], i.e., relocation by hydrographic conditions and relocation by fishing activities. The possibility, that chemical munitions or lumps of viscous mustard gas can be washed ashore, is extremely unlikely. Almost all of the dumped chemical warfare agents have a density >1. The only exception is tabun with a density close to 1. Near-bottom currents in the dumping areas are too weak to move the heavy munitions, which are mostly covered by mud, or to force them into upper layers of water. Likewise, lumps of viscous mustard gas, which have a density of about 1.3-1.5 g cm^{-3}, will not be shifted far by the currents. Except for the few cases referred to in [*239*], there have not been any confirmed reports of bombs or bomb remains being washed ashore on Danish, Swedish, Polish or German territories, since the dumped warfare objects were settled on the seabed. Again, except in a few cases, rumours about mustard gas finds on beaches did not stand up to later investigation.

The conclusion, that warfare agent residues from the dumping areas in the central part of the Baltic Sea cannot be washed ashore by currents, is supported by the fact that the seabed currents in the area are rather weak and mainly easterly. Material released from the seabed will thus move into the Baltic Sea. In addition, the dumped material needs to be moved upwards from a depth of up to 100 m in order to be washed ashore.

A relocation by hydrographic conditions is unlikely. Therefore, a threat to coastal areas of the Helsinki Convention Area from residues of warfare agents or chemical munitions washed ashore is unlikely.

8.3.7 Threats

Based on present knowledge, a widespread risk to the marine environment from dissolved warfare agents can be ruled out. Elevated levels of sparingly soluble Clark, Adamsite or viscous mustard gas may, however, occur in the sediment in the immediate vicinity of dumped munitions. Because of the very limited extent of the agents, however, no threat is posed to marine flora and fauna according to current information. No detrimental effects on the marine environment due to warfare agents have so far been observed.

Insufficient ecotoxicological data is available for most of the chemical warfare agents. Further investigations should be carried out with a special emphasis on mustard gas, chlorinated additives and arsenic compounds.

Discoveries of warfare agents during fishery outside the dumping areas happen from time to time. The problem is recognised especially in the area east of Bornholm. Here, fishermen operating repeatedly find bombs, shells and fragments thereof, and lumps of mustard gas in their bottom-trawl nets. There are several explanations for the spread of munitions. During transport to the dumping area east of Bornholm, munitions have been thrown overboard while the ships were en route. Warfare agents are assumed to be spread over a considerable area along the transport routes. Furthermore, the munitions have sometimes been thrown overboard from drifting or sailing vessels. The first dumping operations took place while the munitions were still packed in wooden boxes, which sometimes were observed to drift around before sinking to the bottom of the sea. However, spreading of the chemical munitions are also done unintentionally by fishing vessels when trawling. In this way, chemical munitions can be dragged about in the trawl over the sea bed without being caught. Furthermore, on some occasions munitions recovered by fishermen have probably been thrown back into the sea, possibly a long way from the position where they were initially dumped.

The dumping areas are pronounced as foul with an "anchoring and fishing not recommended" on nautical charts. However, since fishing in these areas is not prohibited, commercial fishing can occur. In order to avoid problems, considerably larger areas encircling the dumping grounds have been designated as risk areas on the nautical charts. In Denmark and Sweden, rules have been laid down for fishing in these areas through various legislations. For example, it is obligatory for the vessel to carry protective clothing and first aid for chemical warfare agents.

The detailed regulations on the handling of caught chemical munitions include the designation of appropriate authorities to, among other things, assess the chemical munitions caught and to advise fishermen whether to re-dump or bring the munitions ashore. If the chemical munitions pose a risk of explosion, they will be re-dumped after consultation with the appropriate authorities. This is done in accordance with the decision at the 9th Meeting of the Helsinki Commission in 1988 (HELCOM 9/16, Paragraph 8.12). On the other hand, if, after close examination of the material caught, experts do not find that it poses any risk of explosion, it will be brought ashore and deposited in special depots, until it can be transported to a destruction facility. Afterwards, the vessel and its gear will be decontaminated following specialized procedures, and the appropriate authority has to approve the vessel and gear before fishing is re-commenced.

As mentioned above, spreading of dumped chemical warfare material is to some degree caused by fishermen re-dumping chemical warfare equipment which had been caught in fishing nets, possibly a long way from the position where it was dumped originally. Fishermen from Denmark, Greenland and the Faroe Islands have access to *ex gratia* compensation from the Danish State, but only when chemical warfare agents are caught outside the areas pronounced "anchoring and fishing not recommended", and on condition that the catch of fish, contaminated by chemical warfare agents, is destroyed. It should be added, that of 103 cases of chemical warfare material catches registered by Denmark in 1991, only 5 cases involved "foreign" fishermen.

It is generally accepted that since Denmark compensates its fishermen if they destroy contaminated catches, fairly reliable Danish statistics exist about reported finds of warfare agents. Fishermen from other nations bordering the Helsinki Convention Area are not obliged to notify the authorities of such findings. Accordingly, only incomplete figures exist on warfare agent finds by fishermen from other countries. Information has been provided by Germany, Latvia, Lithuania, Poland and Sweden.

Table 8.3.6 gives the figures for total amount of chemical munitions caught and registered by Denmark east of Bornholm during the period 1985-92. The reasons for the increase in 1991 are still unknown. This is probably due to a combination of different factors, like the spreading of dumped munitions, and increased fishing as a result of decreased cod stocks in the Baltic Sea. It should be mentioned that the number of catches of munitions registered by Denmark in 1993 is less than ten.

Table 8.3.6 *Total numbers and weight of chemical munitions (CMs) caught east of Bornholm and registered by Denmark in the period 1985-92 [228]*

	Number of "catches"	CM mass (kg)	Landed CMs (kg)
1985	46	2,695	585
1986	41	1,830	370
1987	14	582	175
1988	19	1,044	115
1989	42	1,966	120
1990	19	979	182
1991	103	5,378	269
1992	58	2,597	100
total	342	17,072	1,917

Germany has reported 13 cases. Only the incidents in which crews were injured are known, so far with no major fatalities. All 13 incidents occurred east of Bornholm in the area pronounced "Foul chemical munitions" and "Anchoring and Fishing Dangerous", or in the immediate vicinity [229]. Sweden has reported 4 incidents with mustard gas from this area since 1980, one involving a fishing vessel from Estonia [233]. Due to the fact that in the Gotland Basin the composition of the munitions is similar to that in the Bornholm Basin, a similar assessment of the risks to fisheries applies, but on a smaller scale.

Latvia has reported fishermen's contact with chemical munitions. The contacts took place from the 1950s to the 1970s, and in some cases later. The locations of these catches were within the dumping area south-east of Gotland. Most findings were in the 1950s, and in some instances the contacts have led to fishermen being injured [*234*].

Sweden has reported 4 fishing vessel incidents involving dumped chemical warfare agents south-east of Gotland since 1980. Two incidents involved mustard gas and the others Clark I and CAP [*231*]. Likewise, Lithuanian fishermen occasionally have had contact with chemical weapons in the area. One episode from 1986 is reported (56°20' N / 19°48' E), where fishermen after contact with a mustard gas bomb were hospitalized [*235*].

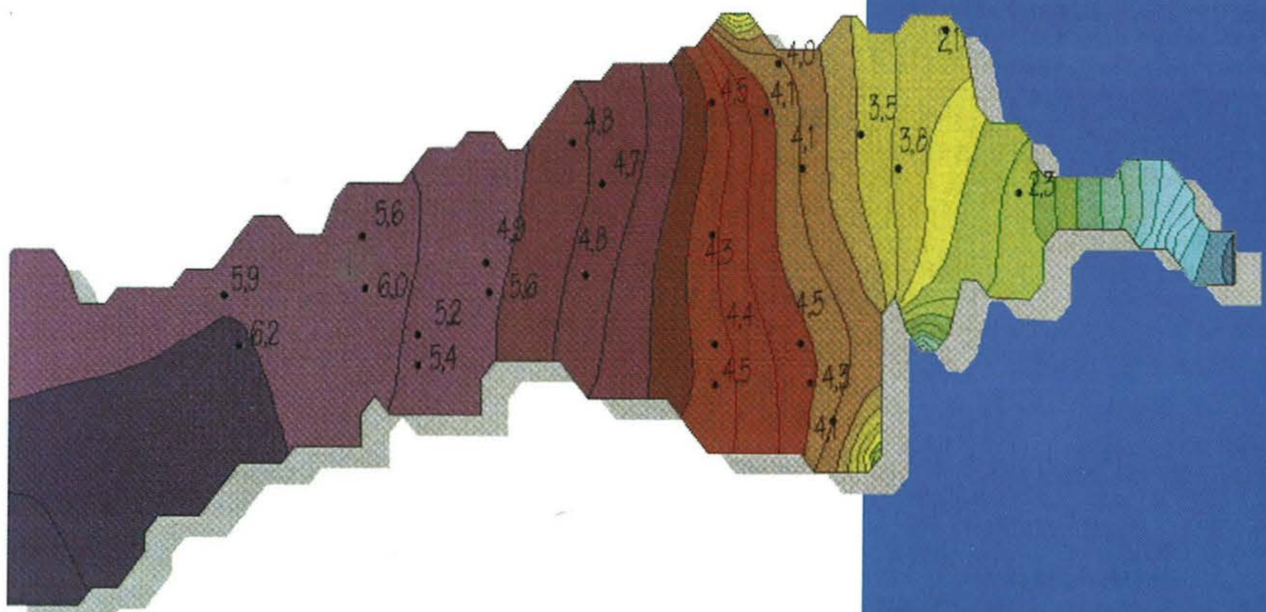
In the Polish exclusive economic zone, there have been 16 identified findings of outdated ammunition and weapons. Chemical munitions have occurred in one of those areas. Judged from the coordinates given (54°37' N / 15°39' E), this area is on the route which the ships used to the dumping area south-east of Gotland [*236*].

The very nature of the dumping operations in the area south of the Little Belt has apparently prevented accidental or deliberate dumping of munitions outside the area where the ships were sunk. The fact that the chemical munitions were inside a ship hull, has also prevented further spreading. In this area, only easily degradable warfare agents (tabun and phosgene) were dumped.

Potential risks to consumers - Contracting Parties have control procedures for fish and other types of seafood, before they reach consumers. According to the existing knowledge, no content of mustard gas or other chemical warfare agents have been found in edible fish or other types of seafood. Given the present knowledge, the chemical warfare agents do not constitute a problem in terms of food toxicology.

NEW MONITORING ASPECTS

9



Surface layer salinity in Gulf of Finland as estimated with a two-layer hydrodynamic model (Kai Myberg, Finnish Institute of Marine Research)

9.1 STUDIES ON PELAGIC BIOLOGY FROM FERRIES

E. Rantajärvi[1], S. Hällfors, J.-M. Leppänen

9.1.1 Introduction

The aim of this paper is to demonstrate the importance of appropriate sampling frequency in evaluating regional differences and long-term trends in phytoplankton. Furthermore, it gives proposals to improve the current BMP phytoplankton monitoring programme in order to reliably measure the changes in phytoplankton.

The Second Periodic Assessment of the State of the Marine Environment of the Baltic Sea [*220*] revealed the difficulty of using the phytoplankton data (chlorophyll, primary production, biomass), collected by the Contracting Parties according to BMP Guidelines [*216*], in long-term trend analysis. With that data set it was even difficult to show regional differences in the trophic levels. The main reason for this was the very limited spatial and temporal sampling frequency in the data collection as well as difficulties in comparing the results from different laboratories (cf. [*223*]). The increase in the sampling frequency was a major suggestion in the Second Periodic Assessment [*220*] in order to improve the possibility of using the BMP data to evaluate the state of the Baltic Sea.

The small-scale variability in phytoplankton in the Baltic Sea was demonstrated by many studies (cf., e.g., [*306,308,309,311,341*]). A method to detect the lateral distribution in plankton, was described [*310*]. It was based on flow-through fluorometry (cf. [*424*]). In the Baltic Sea, the method was first used on research vessels [*308,407*]. For economic and practical reasons, it is not possible to increase the temporal sampling frequency by using only the research vessels for data collection. An unattended sampling device and data collection procedure was tested [*408*], and further developed [*409,560*]. The unattended sampling device has been in operational use since 1992, and is the basic data source for a comprehensive early-warning network on plankton blooms in the Baltic Sea.

9.1.2 Material and methods

In this chapter, data from 1992 and 1993 are used to make the assessment. The data were collected using automated equipment housed on the passenger ferry *Finnjet*. The *Finnjet* crosses the whole Baltic Proper and the western Gulf of Finland from Helsinki to Travemünde twice a week from September till May, and three times a week from June till August (Fig. 9.1.1).

A sampling device, consisting of a flow-through fluorometer, a thermo-salinograph, a GPS navigator, a desktop PC and a refrigerated water sampler, was installed on board the ship. Water was pumped constantly through the sensors from a fixed depth (about 5 m) while the ship was moving. The spatial resolution of the frequently recorded parameters (*in vivo* fluorescence, temperature, salinity) was 100-200 m, while the temporal frequency was about 1-3 days depending on the schedule of the ferry. Every week, 24 water samples were collected during one voyage of the ship.

Nutrient concentrations in water samples were analyzed at the laboratory of the Finnish Institute of Marine Research after the ferry arrived in Helsinki harbour. For details on the sampling device and methods cf. [*406,560*].

For practical reasons, the recorded data was averaged over one nautical mile, giving in 1992 and 1993 a total of 68,183 and 75,958 records, respectively. Although this treatment decreased the original data by almost tenfold, this did not negatively influence the spatial distribution pattern [*560*].

The data compiled are divided according to growth seasons and sub-regions. The growth seasons are spring (March-May), summer (June-August) and autumn (September-October). The division of the sub-regional areas is presented in Figure 9.1.1, and the regional and seasonal distribution of the number of observations (1 nautical mile average values) in Figure 9.1.2.

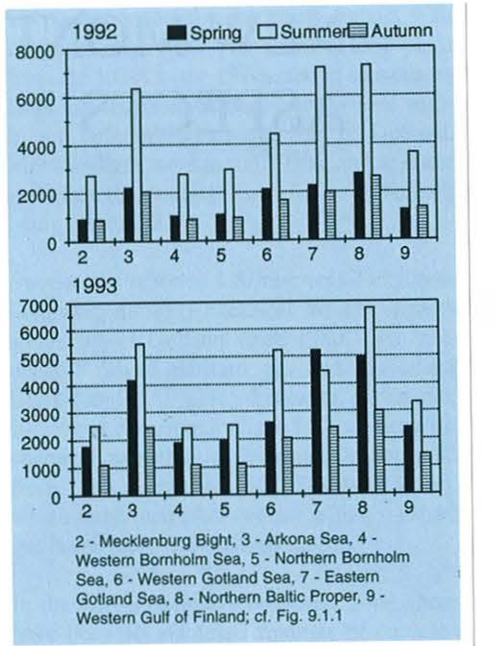


Fig. 9.1.2 Number of chlorophyll observations (1 n.m. average values) in different regions of the Baltic Sea during the growth periods

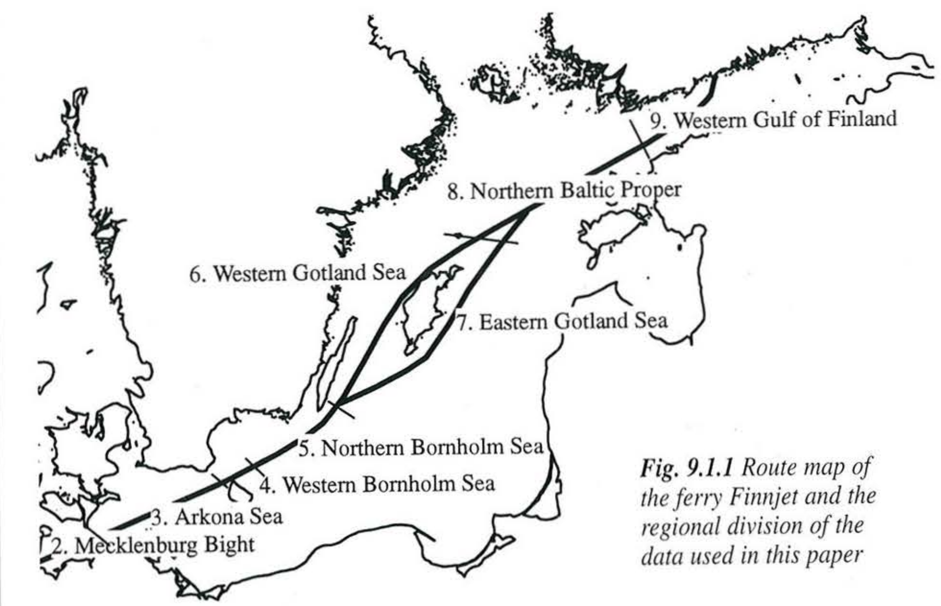


Fig. 9.1.1 Route map of the ferry Finnjet and the regional division of the data used in this paper

[1] see ANNEX 11.3 for addresses of authors

9.1.3 Results

9.1.3.1 Variation in chlorophyll *a* concentrations

The frequency distribution of the chlorophyll *a* concentrations in spring (Fig. 9.1.3) is very skewed and close to the log-normal distribution, i.e., low concentrations predominate while high concentrations are infrequent, even during the peak phase of the bloom. In summer and autumn, the concentration distribution was closer to the normal one (Fig. 9.1.3).

The regional coefficient of variation ranged in spring between about 40 % and 170 %. In summer and autumn, the coefficient of variation were about 40 % and 10-40 %, respectively. The regional differences in total variation were pronounced (Fig. 9.1.4).

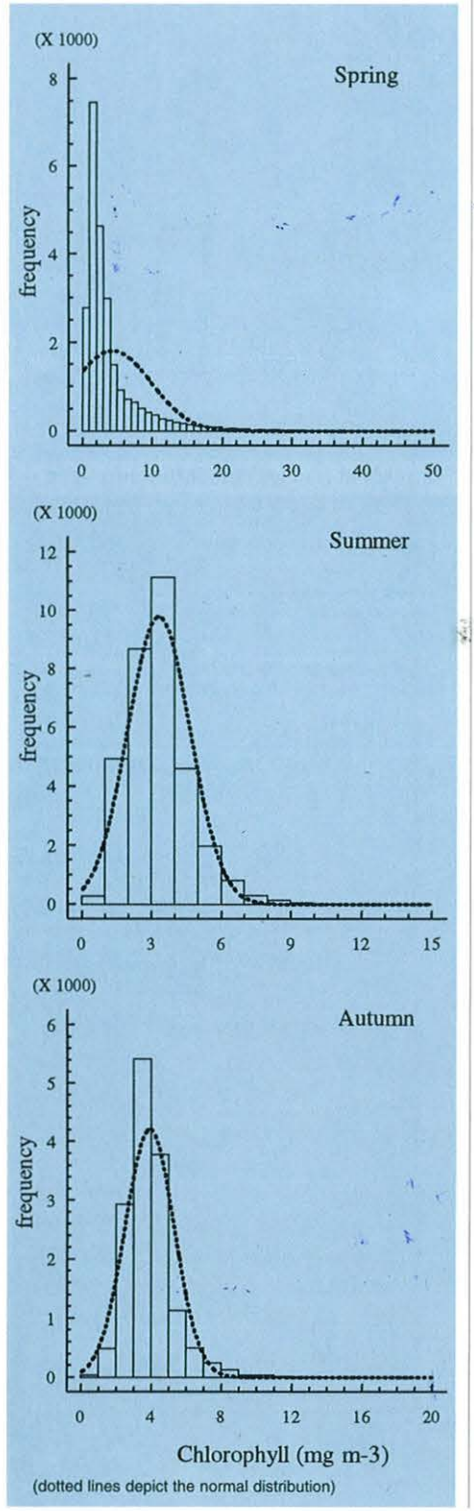
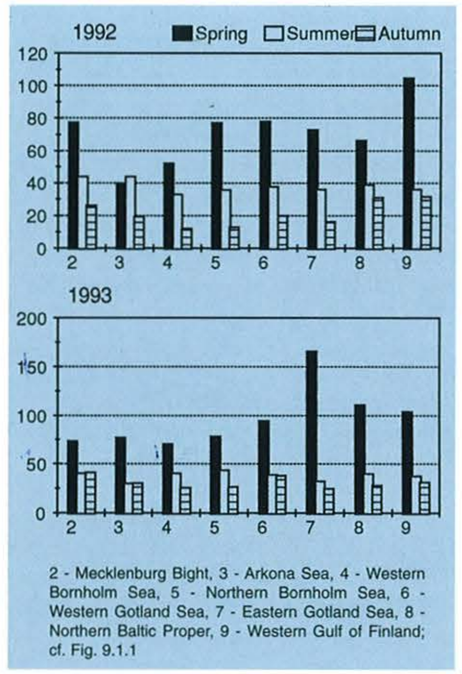


Fig. 9.1.3 Frequency distribution of chlorophyll concentrations (mg m^{-3}, 1 n.m. averages) during different growth periods in 1993, as measured with the unattended high-frequency recordings

Fig. 9.1.4 Coefficient of variation (%) of chlorophyll concentrations in different regions during the growth periods in 1992 and 1993, respectively.



9.1.3.2 Regional distribution of chlorophyll *a* and nutrient concentrations

The differences in the regional distribution pattern of chlorophyll concentrations were more distinct in spring than in summer and autumn (Fig. 9.1.5). In addition, the regional distribution pattern varied between subsequent years for spring much more than it did for the other growth periods.

In 1993, the highest nitrate nitrogen concentrations before the onset of the vernal bloom were measured in the Western Gulf of Finland. In summer and autumn, nitrate-nitrogen concentrations close to the detection limit were recorded in all areas (Fig. 9.1.6). Highest initial phosphate-phosphorus concentrations were recorded in the Western Gotland Sea (Fig. 9.1.6), and phosphate was almost depleted only in the southernmost and northernmost regions.

9.1.3.3 Regional representativeness of a fixed station

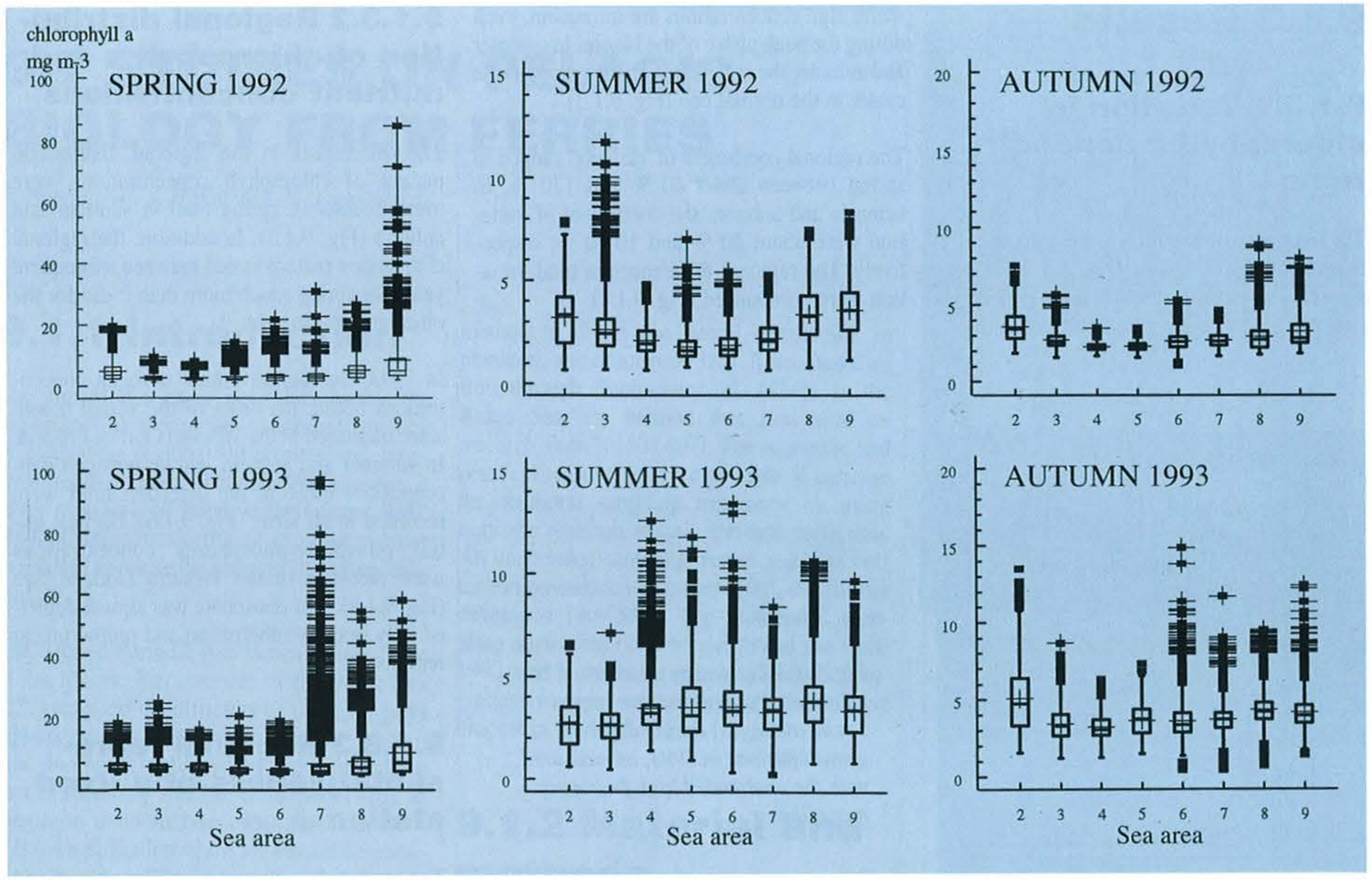
The regional representativeness of a fixed sampling station was studied using the high-frequency data collected in 1993 in the Arkona Sea and the Northern Baltic Proper. Three different data sets were used, i.e.,

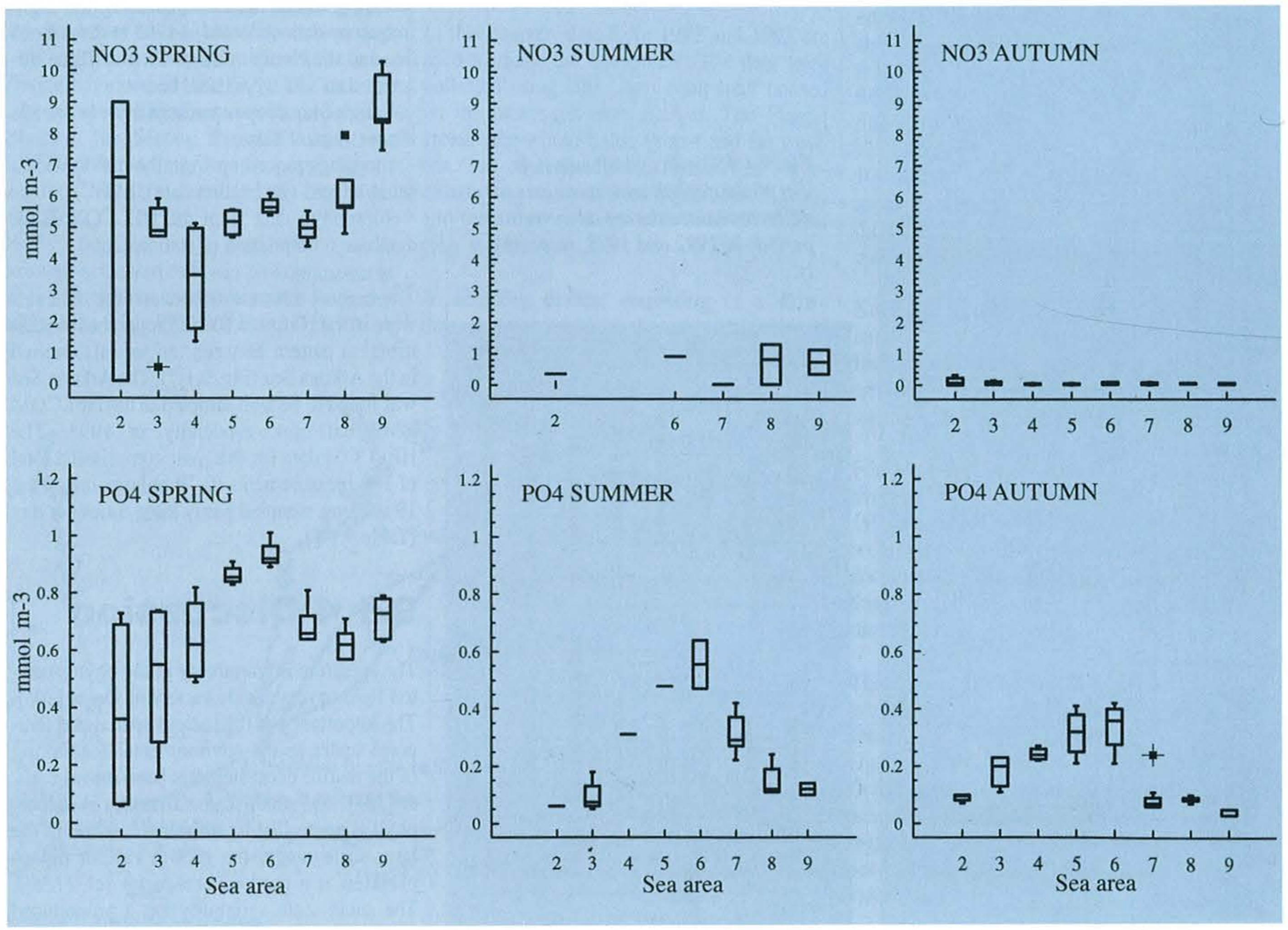
- all high-frequency recordings done in one sea basin (transect data),
- recordings picked up from the previous data set at a fixed site (station data), and
- chlorophyll data from the HELCOM BMP database (compilation of stations data).

The highest differences between the data sets were in the Northern Baltic Proper, but the distribution pattern between the sets also varied in the Arkona Sea (Fig. 9.1.7). The Arkona Sea was found to be well sampled in the HELCOM BMP data set, especially in 1993. The HELCOM data for that year comprised a total of 138 measurements (0-10 m layer means) at 19 stations, sampled partly three times per day (Table 9.1.1).

9.1.4 Discussion

The spectrum of variability in the phytoplankton field covers a wide variety of scales [*208*]. The importance of the various spatial and temporal scales to the environmental monitoring of the marine ecosystem has been shown earlier [*337*]. The small-scale variability in chlorophyll is controlled by turbulence, while in the large-scale range, the growth rate of phytoplankton is a predominant factor (cf. [*138*]). The small-scale variability has a pronounced





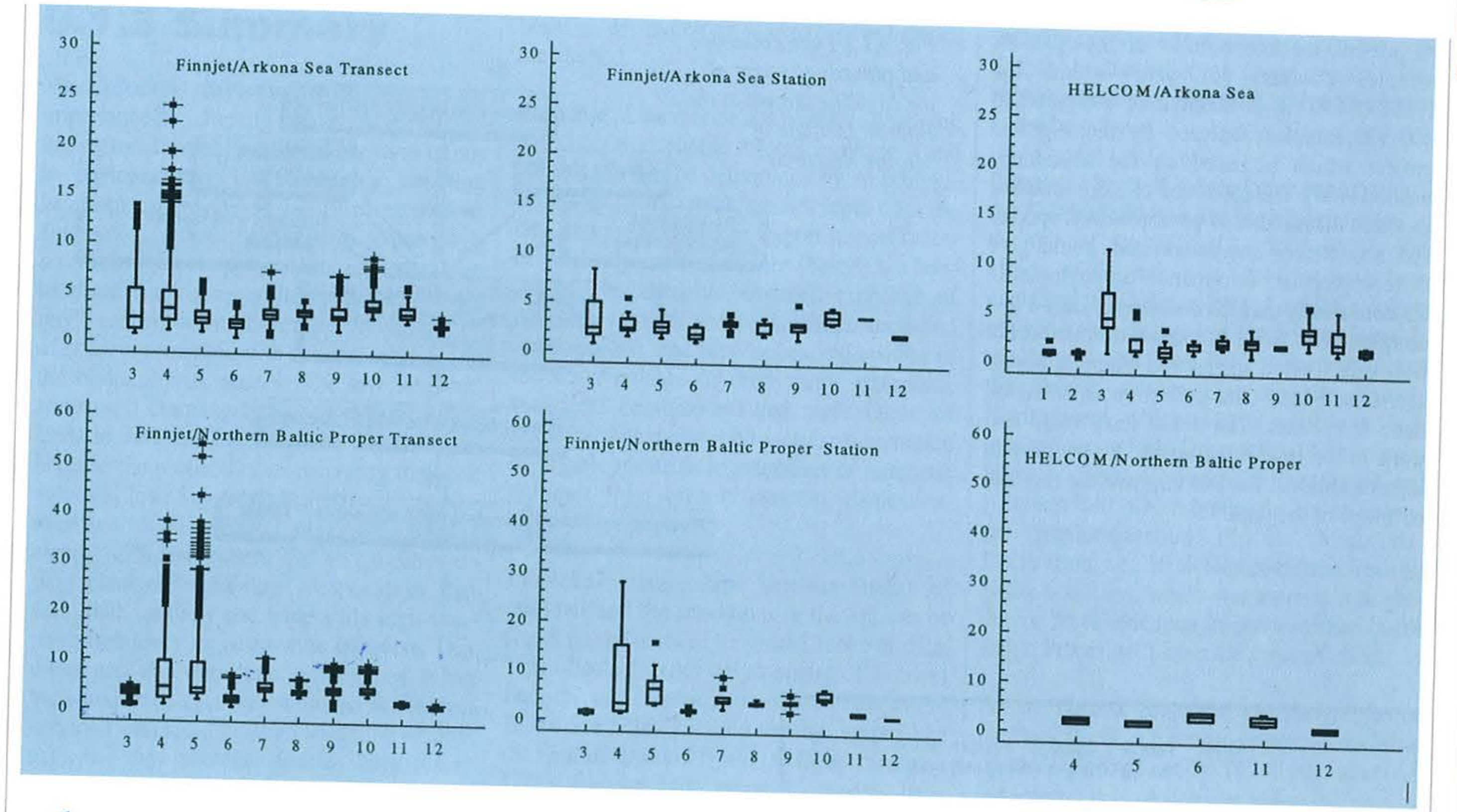


Fig. 9.1.5 Box-and-whisker plots on the seasonal variability in the chlorophyll concentrations (mg m^{-3}) in the different regions in 1992 and 1993, respectively

The plotting procedure divides the data into four parts of equal frequency. The box encloses the middle 50 %. The horizontal line inside the box represents the median. The vertical lines extend from the first and third quartile to the smallest and largest data point within 1.5 interquartile ranges, respectively. 'Outliers' are indicated with dots and + signs. 2 - Mecklenburg Bight, 3 - Arkona Sea, 4 - Western Bornholm Sea, 5 - Northern Bornholm Sea, 6 - Western Gotland Sea, 7 - Eastern Gotland Sea, 8 - Northern Baltic Proper, 9 - Western Gulf of Finland; cf. Fig. 9.1.1

Fig. 9.1.6 Variability in nitrate-nitrogen (NO_3) and phosphate-phosphorus (PO_4) concentrations (mmol m^{-3}) in different areas of the Baltic Sea, 1993

2 - Mecklenburg Bight, 3 - Arkona Sea, 4 - Western Bornholm Sea, 5 - Northern Bornholm Sea, 6 - Western Gotland Sea, 7 - Eastern Gotland Sea, 8 - Northern Baltic Proper, 9 - Western Gulf of Finland; cf. Fig. 9.1.1

effect on the chlorophyll concentrations measured on each sampling occasion. Therefore, the sampling has to be designed according to the scales relevant to phytoplankton, in order to get appropriate data for the evaluation of basin-wide and long-term variability.

Eutrophication increases the occurrence, intensity and duration of the blooms [*189,190*], rather than leading to an even increase in the chlorophyll concentrations. The present chapter reveals, that the frequency distribution of chlorophyll concentrations shows the predominance of small concentrations in the sea, even during the bloom events. This decreases the probability of recording the peak values using low-frequency sampling, and produces very low average values. The present data confirm also, that sampling at fixed stations, even with high temporal frequency, gives clearly different patterns in the chlorophyll variability compared to those obtained for both spatially and temporally high sampling frequency on basin-wide transects. This phenomenon is even more pronounced when these data are compared to data obtained with low-frequency sampling according to the present BMP of HELCOM.

Furthermore, it is unlikely that sparse sampling frequency could detect exceptional blooms. The present reporting system of HELCOM is not helpful for rapid information exchange on any exceptional event, and cannot act as an early-warning system for, e.g., harmful algal blooms. The *Algaline* Project, carried out by the Finnish Institute of Marine Research, has developed an extensive and fast data acquisition and reporting system, which is based on the unattended recordings and samplings on ferries (Fig. 9.1.8). The high-frequency data used in the preparation of this chapter have been collected under project.

Information on the phytoplankton species composition is a prerequisite in order to understand changes in the plankton community. In an unattended measuring system, the fluorescence recordings enable the preselection of the water samples for time-consuming phytoplankton species determination. Thus, although the number of samples analyzed can be reduced, the necessary information on the bloom-forming species is still obtained. In addition, systematic monitoring of phytoplankton species composition yields comprehensive data for the analysis of phytoplankton community changes and represents a reliable method of mapping harmful species [*409*].

Fig. 9.1.7 Monthly variability in the chlorophyll a concentrations (mg m^{-3}, 1 n.m. averages) according to the spatial and temporal sampling frequency in the Arkona Sea and in the Northern Baltic Proper, respectively, in 1993

The uppermost figures represent data sets with all high-frequency recordings done in the sea basins (transect data). The middle figures represent data sets created by picking up records from the transect data at fixed single sites (station data). The lowest figures represent HELCOM BMP data (compilation of stations data).

At present, the total number of phytoplakton samples counted per year within the HELCOM BMP is about 200, compared to 400-500 samples analyzed by the *Algaline* project which is based on the abundance method [*409*]. The collection of more samples, in which the number of phytoplankton species and abundances are determined, could give more appropriate information on phytoplankton community than the counting of only a few samples, with exact biomass information but with very limited spatial and temporal coverage. Unattended data collection is based on three ferry lines. The dense ferry route network in the Baltic Sea makes it very easy to select additional lines to improve the regional coverage of the data.

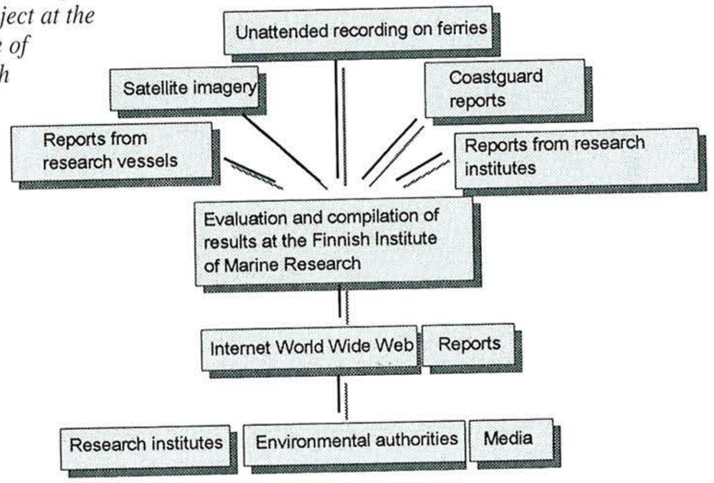


Fig. 9.1.8 *Data collection and reporting system of the Algaline project at the Finnish Institute of Marine Research*

	Month	Sample number	Average (mg m-3)	Minimum (mg m-3)	Maximum (mg m-3)	C V (± %)
Arkona Sea a) *Finnjet* Total transects	3	1,415	**3.6**	0.7	13.8	81
	4	1,135	**4.5**	1.2	23.9	76
	5	1,628	**2.6**	1.1	6.2	33
	6	1,892	**1.9**	0.5	4.0	30
	7	2,033	**2.9**	0.7	7.2	25
	8	1,569	**3.1**	1.7	4.3	15
	9	1,512	**3.1**	1.2	6.8	33
	10	936	**3.9**	2.6	8.7	21
b) *Finnjet* Fixed station	3	9	**3.3**	0.8	8.3	73
	4	9	**2.9**	1.6	5.4	43
	5	12	**2.6**	1.5	4.3	35
	6	11	**2.0**	1.1	2.8	31
	7	13	**2.9**	1.9	3.9	19
	8	12	**2.7**	1.8	3.5	21
	9	10	**2.8**	1.4	5.0	31
	10	6	**3.8**	3.0	4.7	18
c) HELCOM stations	1	5	**1.2**	0.7	2.1	49
	2	6	**1.0**	0.5	1.4	30
	3	22	**4.9**	0.9	11.5	56
	4	10	**2.1**	0.7	5.2	75
	5	20	**1.3**	0.3	3.4	61
	6	10	**1.7**	0.9	2.4	28
	7	10	**2.1**	1.4	2.9	25
	8	19	**2.1**	0.6	3.5	36
	9	2	**1.8**	1.7	2.0	10
	10	18	**3.3**	1.5	5.8	38
	11	12	**2.5**	1.0	5.3	52
	12	4	**1.4**	1.0	1.8	26
Northern Baltic Proper a) *Finnjet* Total transect	3	1,768	**1.2**	0.8	4.9	47
	4	1,174	**7.2**	1.2	38.3	86
	5	2,018	**7.4**	2.2	54.0	81
	6	2,197	**2.5**	0.8	7.21	39
	7	2,578	**4.5**	2.2	10.7	30
	8	1,971	**3.9**	2.8	6.6	15
	9	1,852	**4.3**	0.7	9.5	31
	10	1,159	**4.5**	3.0	9.4	22
b) *Finnjet* Fixed station	3	6	**1.4**	0.9	2.1	32
	4	6	**8.9**	1.4	28.0	120
	5	9	**7.0**	3.1	15.8	60
	6	8	**2.0**	1.2	3.0	25
	7	10	**4.7**	2.4	9.0	40
	8	7	**3.8**	3.4	4.5	10
	9	8	**3.9**	12.0	5.6	25
	10	7	**5.2**	4.0	6.7	18
c) HELCOM stations	4	2	**2.1**	1.5	2.8	41
	5	3	**1.8**	1.2	2.2	29
	6	3	**3.2**	2.3	3.9	25
	11	5	**2.6**	1.5	3.8	38
	12	3	**0.8**	0.4	1.2	47

Table 9.1.1 *Basic statistics of chlorophyll data*

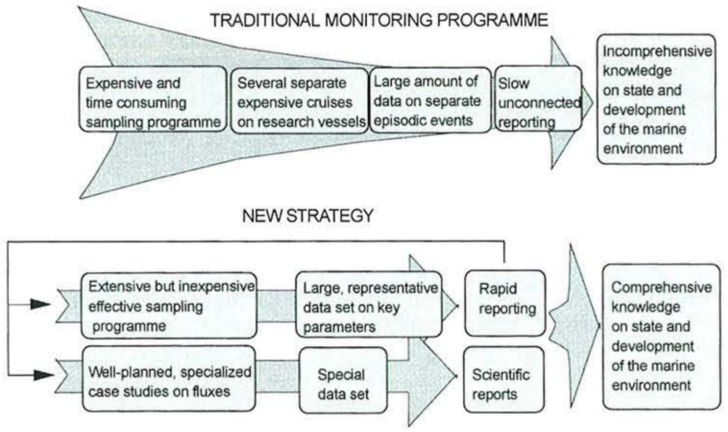


Fig. 9.1.9 *Achievements of the new monitoring strategy compared to the present one*

If systematic monitoring of some phytoplankton parameters, agreed upon by the HELCOM Contracting Parties, is concentrated at research institutes with appropriate QA procedures, the data compatibility would improve pronouncedly by comparison with the present situation. The data acquisition on ferries cannot totally replace the sampling made on research vessels. However, it can provide essential and supplementary information for the traditional programmes. It could even be an alternative to the traditional phytoplankton monitoring programme of HELCOM (Fig. 9.1.9).

9.1.5 Summary

The aim of this chapter was to demonstrate the importance of sampling frequency in evaluating regional differences and long-term trends in phytoplankton. High-frequency sampling has been carried out on ferries using various automated analyzers and sensors. Chlorophyll *a* is used here as an estimate for phytoplankton biomass. The frequency distribution of chlorophyll concentration shows the predominance of small concentrations in the sea, even during the biomass peak phases. The low temporal and spatial sampling frequency most probably leads to underestimation of the actual levels, because the probability of recording the peak values is low. Sampling at the fixed stations, even with high temporal frequency, gives a clearly different pattern for the basin-wide phytoplankton variability compared to that from both spatially and temporally high-sampling frequency on basin-wide transects. This difference is even more pronounced when high-frequency data are compared to the data obtained with low-frequency sampling according to the present Baltic Monitoring Programme of HELCOM.

The data acquisition on ferries can give essential supplementary information to the traditional programmes and it could even be an alternative to the phytoplankton monitoring programme of the HELCOM.

9.2 ECOLOGICAL MODELS IN EUTROPHICATION STUDIES

F. Wulff[1], *J. Elken*

Anthropogenic impact on the Baltic Sea ecosystem is influenced by a complex system of biogeochemical processes [*283*]. The response is modulated by significant natural changes linked to variable meteorological and hydrological forcing (cf. Chapter 2), appearing also as stagnation periods following the major salt water inflows. The regional sub-systems of the Baltic Sea, from the Kattegat to the Bothnian Bay, are intimately connected and present a gradient from a nitrogen limited eutrophic system to a phosphorus limited oligotrophic system. From a management perspective, efforts to reduce loads of eutrophic substances are likely to have highly different cost-benefits, depending on where these reductions are applied.

Response of the marine ecosystem to changing inputs of the eutrophic substances nitrogen and phosphorus can be determined by an ecological model [*549*], which handles input from the land and the atmosphere, export-import fluxes between adjacent sea regions (based on a budget or hydrodynamic module), exchange of nutrients with the seabed (sedimentation and resuspension), and biogeochemical cycling in the water column (hydrodynamic transports, biological consumption and regeneration of nutrients within the food web, sedimentation of organic material, regeneration of inorganic nutrients from organic material, denitrification).

Empirical relationships between inputs of nutrients and the conditions in the sea can be found on the basis of long-time series of data. This allows for the development of nutrient budgets and empirical models for the entire Baltic Sea [*724,725*], and for sub-regions like the Gulf of Riga [*731*] and the Gulf of Bothnia [*728*]. Results from empirical models have been used to develop management strategies for the Baltic Sea [*723*]. While such models have a good analytical capability, their predictive power is rather limited.

Predictive process-oriented 'mechanistic' models depend critically on the way in which the exchange fluxes and biogeochemical cycling processes are reproduced by the model state variables. The first realistic simulation of long-term nutrient changes in the Baltic Proper was made using a coupled physical-biogeochemical 'filling box' 1D model for nitrogen and oxygen dynamics [*641*]. There is also a family of ecological models applied to the entire Baltic Sea or its sub-regions that contain different ecological approaches and mathematical formulations [*144,145,165,596,650*]. The long residence time of water in the entire Baltic Sea system of several tens of years results in rather long response times for the load changes. This requires high-quality standards for the models. A common agreement between scientists about the consequences of the nutrient-loading scenarios for the whole Baltic Sea system, however, is still missing. Integrated modelling approaches, combining different models, are under development within the *EU-MAST3* project *BASYS* [*726*].

Local sea areas allow for easier implementation of ecological models as management tools. Due to shorter residence times for water and nutrients, the models may be calibrated on a seasonal cycle to simulate the variations throughout a typical year. The models may be run with the nutrient boundary conditions obtained from observations, assuming that variable nutrient export will not alter the conditions in the adjacent larger region. This requires measurements with a fine resolution at the boundaries.

Potential effects of reduced nitrogen loading to the Kattegat and Belt Sea have been studied by the use of the *MIKE12* model [*606*]. The effects of the reduction scenarios have been evaluated using the criteria of annual minimal oxygen concentrations in the bottom layer. In the Kattegat and Great Belt, implementation of the Danish Action Plan on the Aquatic Environment, which aims to achieve a 50 % load reduction from Denmark, will have a significant effect. However, in the Sound and Fehmarn Belt, which border the Baltic Proper, the implementation of the Ministerial Declaration, i.e., 50 % load reduction from all Baltic countries, which was estimated to produce a 20 % reduction in concentration in the Baltic Proper, will have the greatest effect.

For the Gdansk Bight and an external part of the Puck Bight, it has been shown on the basis of the *DELWAQ* model [*674*] that after the implementation of nutrient reduction scenarios, the conditions in the bights will not be significantly better than they were in 1991. However, if no measures are taken, the situation will significantly get worse.

A box model, comprising basic physical fluxes and biogeochemical processes, has been applied to evaluate the effects of nutrient-reduction scenarios in the Gulf of Riga [*128*]. In the areas influenced by large rivers, high N/P ratios in the river run-off have made the local primary production limited by phosphorus. The model simulations indicate that a strategy, solely restricted to the reduction of phosphorus load to the Gulf of Riga, will lead to an increase of the net export of nitrogen to the Baltic Proper. Thus, while improving the situation in the Gulf of Riga, such a measure could lead to further undesired eutrophication of the Baltic Proper.

In the Tallinn and Muuga Bights, the effect of reduction of nitrogen and phosphorus loading in wastewater has been evaluated [*146*] on the basis of the model *SYSTEM3* [*145*]. An extensive monthly field measurement programme for basic water quality parameters was run in 1994 (Fig. 9.2.1a). Load data were compiled in detail throughout the year. A special field programme, consisting of hydrographic and current measurements, was carried out to establish the water exchange boundary conditions. Measurements of nutrient enrichment revealed the dominance of nitrogen limitation [*433*] in the area. The ecological part of the model was calibrated to follow the monitored seasonal cycle of nutrients and biological parameters. For the primary production, calibration results are given in Figure 9.2.1b. Annual primary

[1] see ANNEX 11.3 for addresses of authors

production and chlorophyll *a* concentrations were used as criteria for the reduction effects.

Results from the three load-reduction scenarios, set up by the Estonian environmental authorities, are given in Figure 9.2.2 in terms of primary production response in the Tallinn Bight. The conditions observed during 1994 were defined as a reference state. Load reduction from diffuse outlets through the rainwater system will decrease the eutrophication level in the coastal areas of the inner bights (Fig. 9.2.2a). The introduction of biological treatment in the Tallinn WWTP in 1993 produced a load reduction of 85 % for P and 75 % for N. Further treatment of the wastewater has minor effects both on the local conditions (Fig. 9.2.2b) and on the nutrient export to the Gulf of Finland. A relocation of the outfall, which is presently near station *65*, to the shallow coastal areas will lead to a deterioration of the quality of these waters. At present, there is a major nutrient load from the Pirita River which enters east of station *57*. Reduction of this load would have the largest positive effect on the Tallinn Bight (Fig. 9.2.2c).

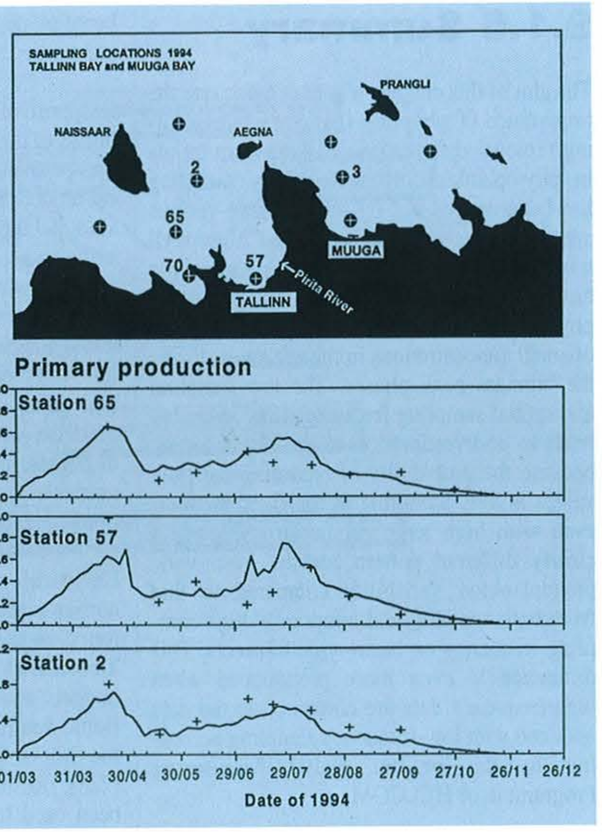


Fig. 9.2.1 Water-quality sampling stations in Tallinn and Muuga Bights in 1994 (a), and calibration results of the eutrophication model shown for the primary production -in g C m^{-2} day^{-1}- as a function of time (b) [146]

solid lines - modelled results
markers - observed data

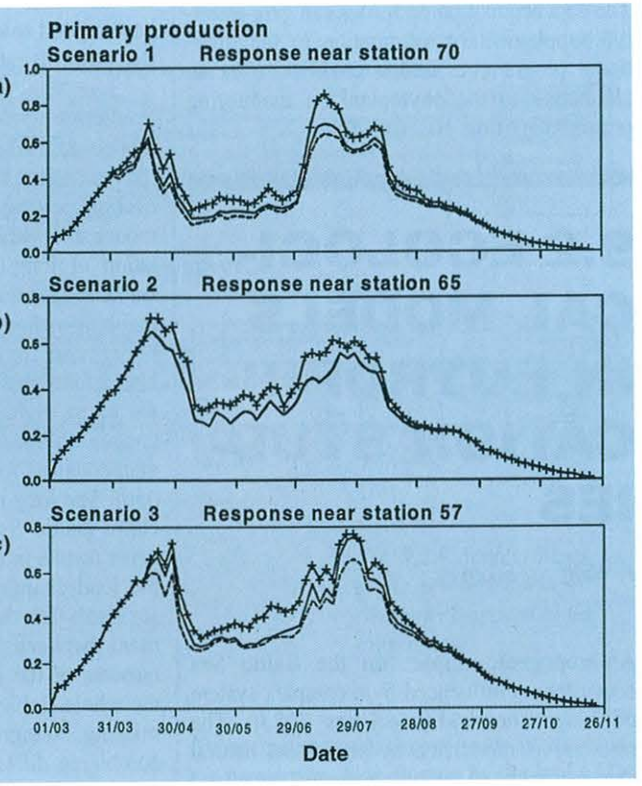


Fig. 9.2.2 Modelling results on the seasonal response of the primary production –in g C m^{-2} day^{-1}– at three coastal water stations off Tallinn, Estonia [146]

(cf. Fig. 9.2.1 for positions of stations)
(—) reference situation,
(+++) before implementation of biological waste-water treatment, and
(——) further waste-water management scenarios, i.e.
Scenario 1: 50 % load reduction from diffuse outlets (near station *70*),
Scenario 2: further load reduction from the waste-water treatment plant (WWTP, near station *65*), and
Scenario 3: 50 % load reduction from the Pirita River (near station *57*)

OVERALL ASSESSMENT

10

HELCOM

10.1 HYDROCHEMISTRY

H.-P. Hansen[1]
with additional contributions by *W. Matthäus*, *G. Nausch* and *D. Nehring*

10.1.1 Nutrients

Unlike the variety of harmful substances like organic compounds or heavy metals, no direct hazardous effects on individuals are caused by nutrients in the Baltic Sea environment. On the contrary, their absence would stop growth and production. Excessive or unbalanced nutrient concentrations, often in combination with unfavourable hydrographic conditions, may exaggerate individual processes and alter or break down the natural ecosystem balance. Excessive production, and finally oxygen deficiency and formation of hydrogen sulphide during decomposition and mineralization, are the main concerns in many areas of the Baltic Sea. It should be remembered that the nutrient levels *per se* do not determine the eutrophication level, but it is the ratio between the nutrient loads and the natural budget of the respective ecosystem which is the key factor. The pronounced regional differences with respect to hydrography, and nutrient loads and budgets, complicate a general assessment of the eutrophication state of the Baltic Sea as a whole. One of the main features involved in budget calculations for example, the residence time, may vary from weeks (estuaries) to a few months (e.g., Kattegat and Belt Sea), to years and even decades (basins of the Baltic Proper). Volumes and residence times of saline water in various deep basins have been listed in Table 4.4.1.

A generalized description of the nutrients in the Baltic Sea can explain gradients and variabilities by a number of systematic features in loads, sinks, distribution and modification processes. This includes spatial, i.e., horizontal and vertical, nutrient variabilities, and temporal changes which are dominated by seasonal biogeochemical cycles. Long-term changes in loads as well as meteorological and hydrographic processes cause long-term changes in nutrients including trends. Some of these processes, in particular the dynamics of the water masses, simultaneously affect the main nutrients (inorganic nitrogen compounds, phosphate, silicate), while several biological and chemical processes, e.g., remineralization and redox processes, lead to asynchronous concentration changes.

The majority of the Baltic Sea areas is characterised by a surplus of phosphate relative to the *Redfield* N/P ratio of 16, thus the phytoplankton production is mostly nitrogen-limited, with cyanobacteria as the exception, since they are able to fix also molecular nitrogen. Phosphorus limitation plays an important, if not dominating, role in the Bothnian Bay, in the western coastal areas of the Bothnian Sea, in some local areas, and is still active in the Gulf of Riga where it was strongly pronounced during the 1980s. Taking into account the additional loads of nitrogen from the atmosphere and from nitrogen fixation, this nutrient is the variable of highest concern with respect to considerations on eutrophication in the Baltic Sea.

Horizontal distribution - As the dominating sources of nitrogen and phosphorus are land-based, a generally decreasing tendency of nutrient concentrations is found with increasing distance from the coast, and from the inner parts of gulfs and bights towards the open sea. The atmospheric deposition, i.e., mainly wet deposition of NO_x and NH_4, represents a more 'global' load, decreasing along a line in north-easterly direction from central Europe towards central Sweden [*613*], and decreasing as well from the coast towards the open sea.

In the Kattegat and Belt Sea area, an additional nutrient source is water inflow from the Skagerrak with about 33.5 psu and 10-11 µmol dm^{-3} nitrate. A correlation between winter nitrate concentrations and salinity is observed. However, the major part of the nitrogen load can be related to local sources. Thus, positive deviations from the linear relationship can be detected. The nitrate concentrations (winter surface values) decrease from high levels in the Kattegat/Belt Sea area (6-9 µmol dm^{-3}) along a transect through the Bornholm and Gotland Seas (around 4 µmol dm^{-3}), while phosphate concentrations are rather uniform (0.6-0.9 µmol dm^{-3}) along the transect. Areas influenced by coastal sources exhibit elevated concentrations, e.g., the Mecklenburg Bight, Gdansk Bight, Gulf of Riga, Bothnian Bay and Eastern Gulf of Finland.

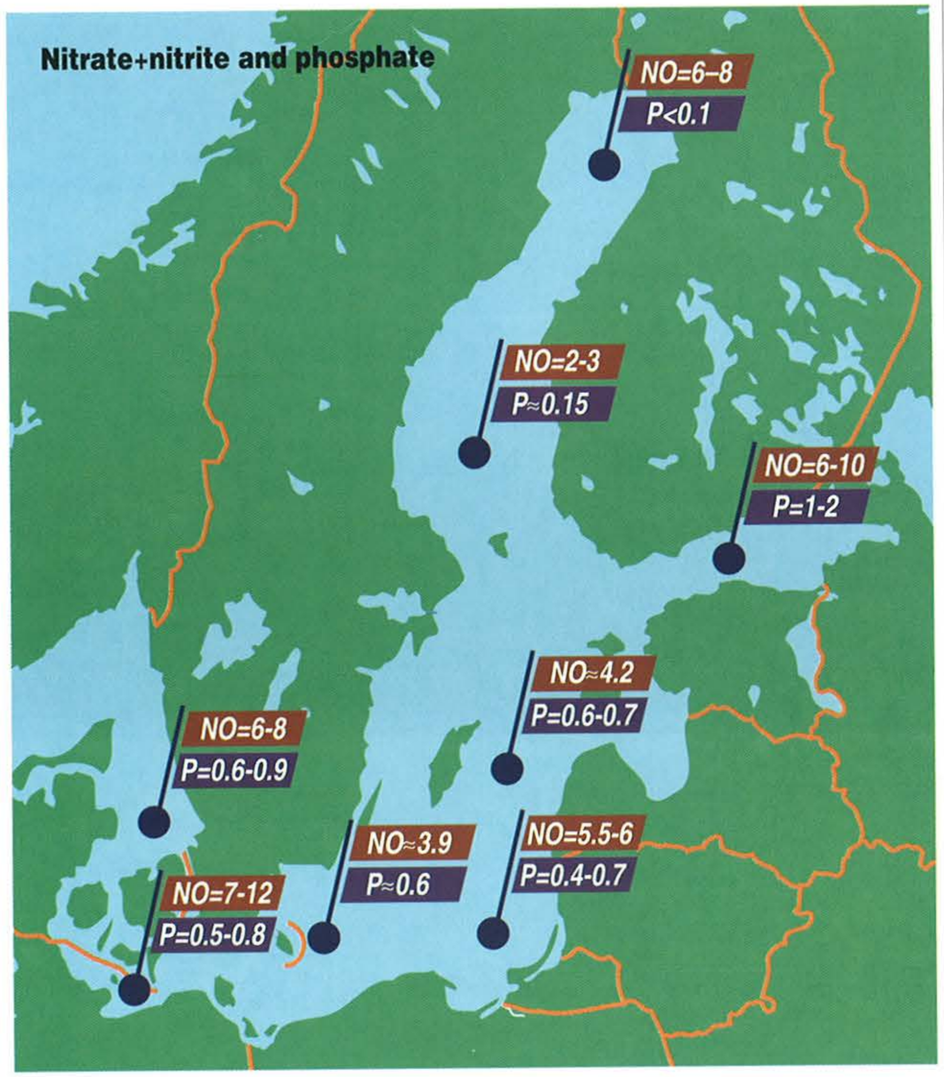


Fig. 10.1.1 Nitrate+nitrite (NO) and phosphate (P) winter surface concentrations in the Baltic Sea, 1989-93 (values in µmol dm^{-3})

[1] see ANNEX 11.3 for addresses of authors

In addition, nitrate (but not phosphate) concentrations in the surface layer are influenced by fresh water run-off. The run-off is high in winter and low in summer, i.e., the nitrogen load is high during the non-productive season when nutrients are accumulating in the surface water. For the period 1975-94, a significant correlation has been shown between winter run-off from Denmark and winter concentrations of nitrate+nitrite in the surface water of the transition area, excluding the central Sound, where most of the nutrient load originates from sewage (Fig. 4.5.5). Similar behaviour has been described for the Gulfs of Bothnia, Finland and Riga. Correlations indicating the significance of nitrogen inputs from non-point (diffuse) sources were not found in the Central Baltic Proper in the absence of direct coastal influence. Figure 10.1.1 displays winter surface concentrations of nitrate and phosphate in the Baltic Sea.

Vertical distribution - Vertical nutrient distributions depend strongly on the regional hydrography. The type of haline stratification, i.e., missing, temporary or permanent haloclines, determines the flux and vertical distribution of nutrients. In general, haloclines are also 'nutriclines', separating surface water with lower nutrient concentrations from bottom water with enriched concentrations. As the mineralization takes place mainly in the deeper layers and at the bottom, the deep layers represent nutrient reservoirs. In the range of the redoxcline, nitrate is rapidly removed at low oxygen concentrations by denitrification, while phosphate is released from the sediments in the presence of hydrogen sulphide and becomes enriched. It has been shown [*479,482*], that the phosphate release from the sediment during stagnation periods and anoxia in the Gotland Basin decreases with progressing duration of the anoxia. Bottom phosphate concentrations level out after an initial increase.

Seasonalities - The annual production of organic material in the euphotic layer is determined by the amount of nutrients available at the start of the production (spring-diatom bloom), and by the fluxes into the euphotic layer from deeper layers, inputs from the land and from the atmosphere, as well as from internal nutrient cycles (turnover) during the productive season. Except for some local processes, the annual development of the phytoplankton production is rather similar throughout the Baltic Sea, showing a bimodular curve, with a short peak-spring-bloom and a broader maximum at the late summer to autumn bloom. The spring bloom is dominated by new production, while the proportion of regenerated production is increasing during summer and autumn.

The nutrient concentrations observed in the productive layer are controlled throughout the course of the production cycle. There is a rapid decrease to low or zero levels during the spring bloom. This low level is maintained until autumn when the fluxes of regenerated nutrients into the photic layer exceed the productive consumption (Fig. 4.4.7). Intermediate increases can be related to short-term events as, e.g., heavy rainfall and run-off, upwelling and advective transport.

Winter concentrations of phosphate and nitrate develop distinct maxima in the surface layer of the Kattegat/Belt Sea area and the Arkona and Bornholm Basins, and are characterised by plateaus of high concentrations in the Eastern, Northern and Western Gotland Basins. Trend studies of these nutrients in the surface layer are therefore restricted to this widely accepted 'winter period', i.e., before the start of the more pronounced annual biological activity, the spring bloom of the phytoplankton. This happens about mid of February in the Kattegat and Belt Sea, and as late as the end of April/beginning of May in the Northern Baltic Proper. Therefore, the winter period is restricted to 1-2 months only in the Arkona and Bornholm Basins, but to 2-3 months in the Eastern and Western Gotland Basins (Fig. 4.4.7). In coastal areas and in the western parts of the Baltic Sea, 'winter concentrations' are more difficult to establish. Before constant concentrations are established, the increase of the nutrient concentrations in the photic zone, which is mainly due to fluxes of regenerated nutrients from beneath, may be stopped by the onset of the spring bloom. However, in general, the development of the nutrient concentrations in the surface layer can be described as a steep increasing slope from November to January, followed by a 'winter plateau'. In the northern parts of the Baltic Sea, the ice coverage may last until the onset of the spring bloom, thus complicating proper sampling.

Long-term variations including trends - As already mentioned, the analysis of long-term trends for nutrients in surface waters should be based on measurements during the less productive season, and must cover periods of more than 10 years to take into account intra- and interannual variabilities. The winter concentrations of phosphate and nitrate showed positive overall trends in the surface layer of all sub-regions of the Baltic Proper for the period 1958-93 and 1969-93, respectively. These trends mainly result from the considerable increase between 1969 and 1978. In contrast to phosphate, the increase of nitrate continued until 1983. Thereafter, the concentrations of both nutrients fluctuate strongly at a high level without significant trends.

In the Landsort Deep area, the increase of phosphate and nitrate winter concentrations continued in the recent assessment period. Also in the Gulf of Bothnia, both total and inorganic nitrogen concentrations continued to increase. Total phosphorus also showed a long-term increase in all areas except the Bothnian Bay. The reason for this increase was primarily a consequence of high riverine discharges during the 1980s.

In the Gulf of Riga, the situation is different. The nitrate values dropped sharply in the 1990s due to a lower land-based impact. As a result, the highest phosphorus values appeared during 1989-93, caused by both the increased release from sediments and the reduced uptake in the water column by the plankton. For some areas, the seasonal variability of nutrients in the surface layer has been considered too large for trend analyses, e.g., in the Gulf of Finland [*305*] and in the Kiel Bight [*202,203*].

In the intermediate water layers of the Bornholm Basin and the Eastern and Western Gotland Basins, the nitrate concentrations increased significantly during the previous three assessment periods. Probably as a consequence of eutrophication, the silicate concentrations decreased significantly during recent decades in all depths below the halocline in the Baltic Proper. Decreasing silicate concentrations could also be detected in the Kattegat, Kiel Bight, Gulf of Riga and Gulf of Bothnia.

Fertilisers used in the drainage area are the most important source of eutrophication in shelf seas. The phosphorus and nitrogen fertilisers annually used in the drainage area of the Baltic Sea are compared with the averaged phosphate and nitrate winter concentrations in the surface layer of the Bornholm Basin (Fig. 10.1.2). The strong increase in the use of fertilisers, which began in the early 1960s, is followed, with a delay of 5-10 years, by increasing phosphate and nitrate winter concentrations. Taking this delay into account, the correlations are obvious. The drastic reduction in the use of fertilisers, mainly caused by the great economical changes in the countries in transition, began in the late 1980s, and is thus not yet significantly reflected by decreasing nutrient concentrations. But first hints are appearing with respect to the averaged phosphate concentrations. They now exhibit a decreasing tendency in surface waters of the Central Baltic Proper. This behaviour is even more pronounced in the Arkona and Bornholm Basins, with stations located nearer to the coast than in the Eastern Gotland Basin. The situation is not so clear for nitrate.

Decreasing trends of phosphorus have also been observed in the Danish fjords, but not yet in the Kattegat and the open Belt Sea. The reduction in the use of fertilisers is the main reason for the decrease of the nitrate concentrations in the Gulf of Riga.

10.1.2 Oxygen and hydrogen sulphide

The concentrations of oxygen in water are controlled by its fluxes between sea and atmosphere, by its assimilative production, and by respiration. Because of the rapid changes in temperature and salinity, which produce variations in oxygen solubility, the degree of oxygen saturation provides a more informative figure for surface waters than that provided using data on concentrations. On the other hand, deep waters below the pycnoclines cannot equilibrate with the atmosphere and are generally under-saturated. They are therefore best characterised by their absolute oxygen amounts, i.e., expressed in concentrations, whereby hydrogen sulphide is very often given as 'negative oxygen'.

The oxygen exchange between surface waters and the atmosphere depends on the mixing of the surface water layer. The compensation of imbalances can take hours to days and weeks. Therefore, the saturation degree provides a measure of the direction of the oxygen flux, rather than of the local oxygen production or consumption. Trends for oxygen saturation in surface waters have only been reported for the 0-2 m layer of the Vistula Estuary, with +1.45 % yr^{-1} from a mean of 101.7 %. This may be interpreted as a result of increasing phytoplankton production, but could also be due to meteorological or hydrographic changes.

Oxygen and hydrogen-sulphide concentrations in the deep waters of the Baltic Proper depend on the duration of stagnation periods and on advective water exchanges. They also reflect the microbial oxygen consumption which is increasing as a consequence of eutrophication. Therefore, the stagnation period since 1977 has led to extremely low oxygen and high hydrogen-sulphide concentrations, respectively, in the deep waters of the Eastern Gotland Basin (c.f. Table 2.2 and Fig. 2.18). While the Gotland Deep exhibited hydrogen-sulphide concentrations of more than 150 µmol dm^{-3}, the Bornholm Deep suffered from increasingly frequent events of anoxia, causing layers of *Beggiatoa* to cover the bottom. The series of salt-water inflows, which started in the spring of 1993, has changed the situation considerably. The Gotland Deep temporarily displayed oxygen concentrations of up to 3.8 cm^3 dm^{-3}, and hydrogen sulphide disappeared completely. Investigations from 1994-96 show, however, that both basins are returning to anoxic conditions.

The lack of salt water inflows into the deep waters since 1976 has reduced the bottom salinity and thus the vertical stability. Positive bottom-oxygen trends of about 0.5 % yr^{-1} in the Bothnian Bay and Bothnian Sea have been explained by improved mixing due to reduced vertical stability.

During the stagnation period, water masses from the minor inflow events, i.e., those, not reaching the intensity of major inflows, pass through the Central Baltic Proper along an intermediate layer below the halocline. Consequently, oxygen concentrations have increased around the 100 m depth in the Eastern Gotland Basin [479,482]. The density of these water masses is high enough to displace the bottom water of the Western Gotland Basin and to supply it with oxygen (cf. Table 2.2, Fig. 2.18). The deep waters of the Gdansk Basin show a general trend of increasing oxygen cencentrations. The maximum increase was observed between 75 and 90 m depth.

Oxygen/nutrient relationship - Oxygen deficiencies in bottom waters are common phenomena in many coastal areas with seasonal or permanent haline stratification. The oxygen concentrations display a seasonal variability, with minima between July and October. Generally, these minima reflect the biochemical oxygen demand of organic matter resulting from preceding phytoplankton production, though direct links, i.e., quantitative correlations, are difficult to establish. In the Belt Sea, high autumn/winter fresh water run-off, which is a good measure of the riverine nutrient load, seems to induce oxygen deficiency in the bottom water during the following autumn.

Before 1988, several authors had reported significantly decreasing oxygen concentrations in the bottom water during autumn for the Kattegat and Belt Sea area [5,50].

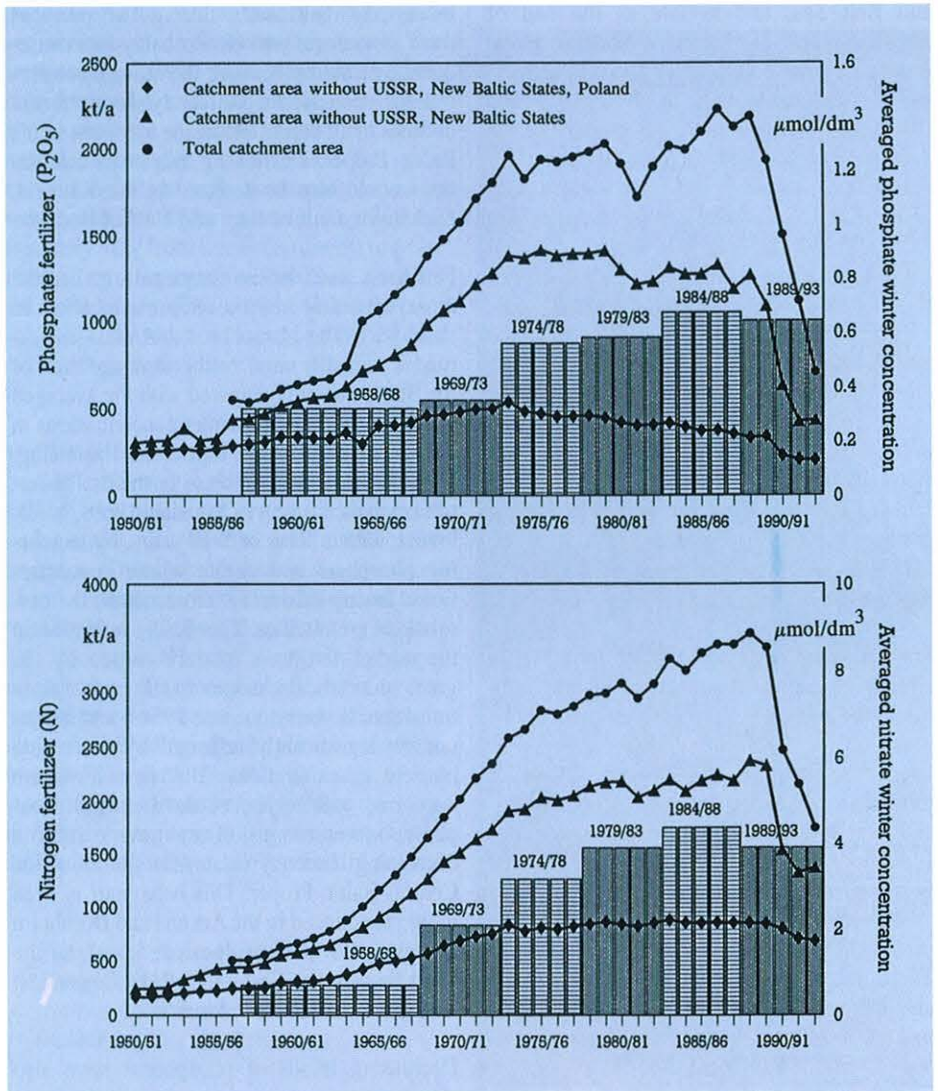


Fig. 10.1.2 Use of synthetic phosphorus and nitrogen fertilisers in the drainage area of the Baltic Sea (lines, pro-rated by drainage area), and phosphate and nitrate winter concentrations in the 0-10 m layer of the Bornholm Basin, averaged for 5 (or 11) years (cf. Table 4.4.4, [164])

10.2 PELAGIC BIOLOGY

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J. Kuparinen (bacterioplankton)
L. Postel, G. Behrends, R. Olsonen (zooplankton)

10.2.1 Introduction

The present BMP data on phyto- and zooplankton provide a globally unique baseline information on the regional distribution of plankton species in the Baltic Sea. This information is of special value, because the occurrence of potentially toxic species and the introduction of alien species have increased the threats to human health and the marine environment. These items are closely related to the biodiversity and the carrying capacity of the sea. The species information is of importance in evaluating possible reasons for new, unexpected environmental problems, such as the *M-74* syndrome.

The statistical *power* of the plankton data in defining trends is poor due to the combination of natural high spatial and temporal variability in plankton and the low sampling frequency within the monitoring programme. For phytoplankton, the taxonomic difficulties, especially within the group of flagellates, and the changes in nomenclature have added additional complications to the comparability of data from different sources.

Bacterioplankton data with sufficient temporal sampling frequency have only been generated within national programmes for the Southern Baltic Proper, the Gulf of Riga and the Bothnian Sea. The statistical *power* of the bacterioplankton-biomass variable was estimated in the Bothnian Sea to detect a 5 % annual change with 80 % probability (n=21). For the bacterial production, it was estimated to be 12 % yr^{-1} (p=0.80, n=10). No significant trend could be demonstrated for either variable in the Bothnian Sea. Similar results were obtained for the Kiel Bight and Fehmarn Belt, except for a negative trend in bacterial production at *Boknis Eck* (Kiel Bight). In the Gulf of Riga, an increase of the bacterial biomass was reported only for May. Further details on bacterioplankton are found in Chapter 10.3.

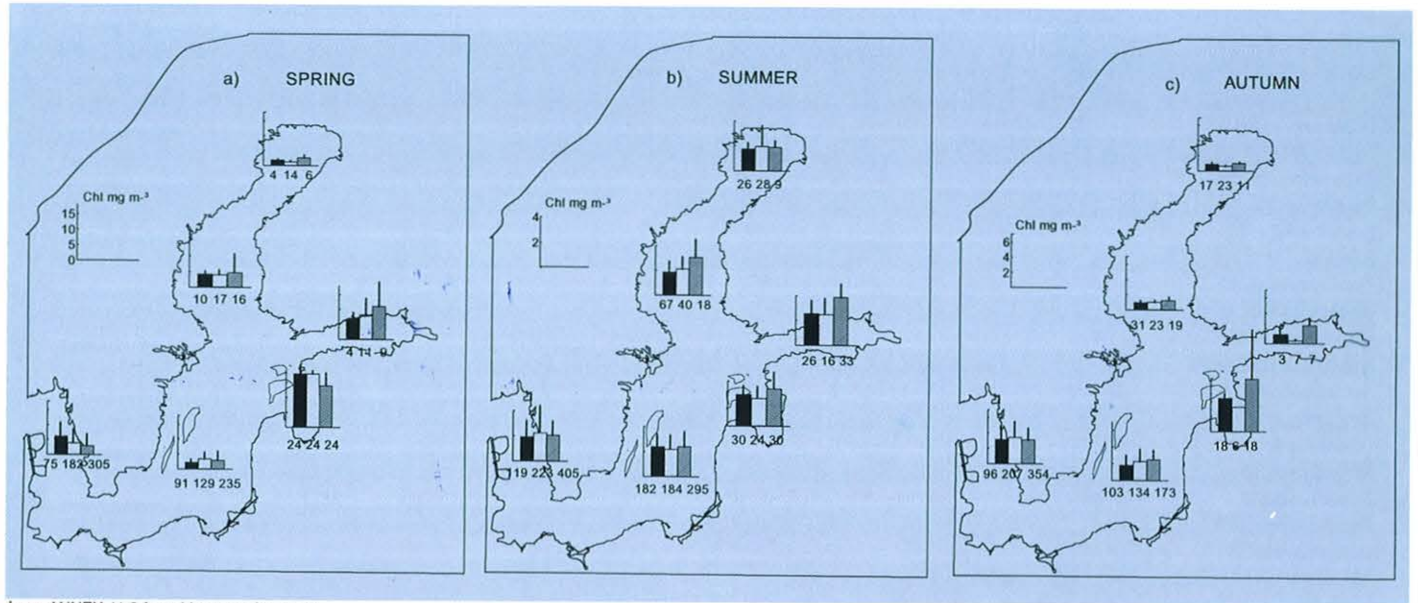
10.2.2 Regional differences

Phytoplankton and bacterioplankton - In the combined 15 years' chlorophyll *a* data (Fig. 10.2.1), the Gulfs of Finland and Riga exhibited a higher spring and summer chlorophyll *a* biomass than that in the Baltic Proper, reflecting the higher nutrient loading in relation to the size of the area.

The salinity gradients, and possibly also the duration of the growth period, were reflected in the species which dominated the composition of the different sub-areas. Only few species appeared as dominants throughout the Baltic Sea. In spring, such ubiquitous species were *Achnanthes taeniata, Peridiniella catenata* and *Chaetoceros wighamii*, in summer different cryptomonads, and in autumn *Thalassiosira baltica* and *Rhodomonas minuta* (Table 10.2.1). In the southernmost parts of the Baltic Sea, the 'oceanic impact' on the species composition was clearly seen in the dominance of several *Ceratium* and diatom species.

Zooplankton - The salinity gradient in the Baltic Sea leads to naturally induced differences between the sea areas. The Darß Sill is known as a strong faunistic boundary. West of this sill, the marine species, such as the copepods *Oithona similis* and *Paracalanus parvus*, dominate the mesozooplankton community. In the shallow waters, bivalves and other meroplanktonic larvae also play an important role. East of the Darß Sill, a progression from brackish water species to fresh water species can be found. The succession of dominant forms follows the order *Pseudocalanus minutus elongatus, Acartia* spp., *Eurytemora affinis* and *Limnocalanus macrurus*. The latter species is a cold-stenotherm glacial relict and dominates the plankton in the Bothnian Bay during spring. In summer, the cladoceran species *Bosmina coregoni maritima* generally occupies the first rank in the surface layer of the entire Baltic Proper. *Bosmina coregoni maritima* is a typical brackish water species with a temperature optimum of >15 °C. In

Fig. 10.2.1 Mean chlorophyll a concentrations in the surface water layer of the Baltic Sea, averaged by areas, seasons and monitoring periods (black column - 1979-83, white column - 1984-88, grey column - 1989-93; number of observations is indicated below those columns)



[1] see ANNEX 11.3 for addresses of authors

***Table 10.2.1** Dominating phytoplankton species in different parts of the Baltic Sea during three seasons.*

The species list is based on the BMP data base, averaged over 1979-83, 1984-88, and 1989-93, respectively. The scores 1, 2 and 3 indicate the number of assessment periods in which the species was dominant.
BB - Bothnian Bay, BS - Bothnian Sea, NBP - Northern Baltic Proper, GoF - Gulf of Finland, WGB - Western Gotland Basin, EGB - Eastern Gotland Basin, GoR - Gulf of Riga, SBP - Southern Baltic Proper, S - Sound, BBS - Belt Sea, K - Kattegat. CYA - Cyanophyceae, CRY - Cryptophyceae, DIN - Dinophyceae, DIA - Diatomophyceae, CHR - Chrysophyceae, PRY - Prymnesiophyceae, EUG - Euglenophyceae, PRA - Prasinophyceae, CHL - Chlorophyceae, ZOO - endosymbiotic microzooplankton, MIX - mixture of species belonging to several taxonomic groups

SPRING Species	BB	BS	NBP	GoF	WGB	EGB	GoR	SBP	S	BBS	K
Thalassiosira baltica DIA	3	3	1	3			3			1	
Achnanthes taeniata DIA	3	2	2	3		2	3		2		
Gymnodinium sp. DIN	2	1			1						
Peridiniella catena DIN	3	3	3	3	3	2	3	2			
Diatoma elongatum DIA	1										
Nanoplankton MIX	2	2						2	2		
Mesodinium rubrum ZOO	1	1	1		1	1		1			
Chaetoceros wighamii DIA		2	1		1	1	3	1	1		1
Pyramimonas sp. PRA	2										
Protoperidinium sp. DIN			1			1					
Thalassiosira levanderi DIA			1						2	1	
Glenodinium sp. DIN			2								
Dinophysis acuminata DIN			1		3	2					
Skeletonema costatum DIA			2	3	3	2		3	2	3	3
Dinobryon balticum CHR					1						
Eutreptiella sp. EUG					1				1		
Dinophysis norvegica DIN					1						
Gymnodinium vestificii DI						1					
Chaetoceros subtilis DIA						1					
Gymnodinium cf. lohmannii DIN						1					
Dinophysis baltica DIN						1					
Melosira nummuloides DIA							3				
Chaetoceros holsaticus DIA								1	1	1	
Chaetoceros ceratosporus DIA								1			
Detonula confervacea DIA								1	2	2	3
Chaetoceros sp. DIA								1		1	
Chaetoceros debilis DIA								1	1		
Nitzschia longissima DIA								1			
Peridinium sp. DIN				1							
Cryptomonadales spp.				2							
Thalassiosira decipiens DIA										2	
Melosira arctica DIA										1	
Porosira glacialis DIA										1	2
Chaetoceros teres DIA										1	
Flagellates MIX										1	
Thalassiosira sp. DIA									1		2
Thalassiosira gravida (?) DIA											1
Thalassiosira polychorda DIA											1
Coscinodiscus concinnus DIA											1
Rhizosolenia setigera DIA											1
SUMMER											
Diatoma elongatum DIA	3										
Flagellates MIX	1	1	1	2	1	2			1	1	1
Cryptomonadales spp. CRY	2	2	2	2	1	1	3		2		
Microcystis sp. CYA	1	1				1					
Pyramimonas spp. PRA	3				1		3				
Monads MIX	1										
Chrysochromulina spp. PRY	1	2			1						
Teleaulax amphioxeia CRY		1									
Chaetoceros wighamii DIA		1									
Aphanizomenon flos-aquae CYA		3	3	2	3	3	3	2			
Nodularia spumigena CYA		2	2	1		3		3			
Proteceratium reticulatum DIN		1									
Chrysidiastrum sp.		1									
Gymnodinium spp. DIN		1			1						
Snowella lacustris CYA				2							
Dinophysis acuminata DIN			2	2		1	3				
Coscinodiscus granii DIA				1							
Achnanthes taeniata DIA				1							

Species	BB	BS	NBP	GoF	WGB	EGB	GoR	SBP	S	BBS	K
Dinophysis norvegica DIN			2	1				1			
Gomphosphaeria sp. CYA						2	3	2	1		
Aphanothece sp. CYA								2	1		
Microcystis spp. CYA						1		1			
Nanoplankton MIX			1		1	1		2	2	1	2
Prorocentrum minimum DIA								1	1	2	
Glenodinium spp. DIN					1						
Eutreptiella sp. EUG					1						
Mesodinium rubrum ZOO			1		2						
Thalassiosira baltica DIA					1						
Chaetoceros danicus DIA					1						
Peridiniella catena DIN			1								
Actinocyclus octonarius DIA							3				
Oocystis spp. CHL							3				
Distephanus speculum CHR											1
Rhizosolenia alata DIA											2
Rhizosolenia fragilissima DIA									3	3	3
Ceratium tripos DIN									2	1	3
Peridinium sp. DIN											1
Chrysochromulina polylepis PRY											1
Guinardia flaccida DIA									1	1	1
Ceratium furca DIN									1		
Rhodomonas minuta CRY										1	
Gymnodinium simplex DIN										1	
Ceratium fusus DIN										1	
Skeletonema costatum DIA										1	
AUTUMN											
Cryptomonas baltica CRY	1	1	1								
Cryptomonadales spp. CRY	3			1				1	3		1
Thalassiosira baltica DIA	1	2	1	1	1	1		1			
Rhodomonas minuta CRY	2	2	1	1	1	2		2			
Monoraphidium contortum CHL	2										
Nanoplankton MIX	1	1			1	2		3	1	1	2
Monads MIX	2	1									
Thalassiosira pseudonana DIA	1										
Aphanizomenon flos-aquae CYA		2	2	1	2		3				
Gomphosphaeria pusilla CYA		1						1			
Rhodomonas lens CRY		1									
Coscinodiscus granii DIA		1	3	3	1	3	3	3			
Actinocyclus octonarius DIA		1	3	1	2	3	3	1			
Mesodinium rubrum ZOO		1			2	1					
Dinophysis norvegica DIN			2	1	1						
Dinophysis baltica DIN			1								
Eutreptiella sp. EUG					1						
Nanonplankton MIX					1	2		3	2	1	2
Flagellates MIX				1	1	2		1	1		
Protoperidinium spp. DIN						1					
Gomphosphaeria spp. CYA							3	1			
Chaetoceros danicus							3				
Peridinium spp. DIN				1							
Melosira varians DIA				1							
Dinophysis acuminata DIN				2							
Snowella lacustris CYA				1							
Cerataulina pelagica DIA									3	2	3
Ceratium tripos DIN										3	3
Ceratium furca DIN										2	
Prorocentrum micans DIN										2	
Ceratium lineatum DIN										1	2
Guinardia flaccida DIA									1	2	2
Ditylum brightwellii DIA										1	
Ceratium fusus DIN										1	
Gyrodinium aureolum DIN									1		1
Coscinodiscus sp. DIN									1		
Porosiga glacialis DIA									1		
Skeletonema costatum DIA									2		
Thalassiosira eccentrica DIA											1

January - March	BB I	BB II	BB III	BS I	BS II	BS III	AAS I	AAS II	AAS III	NBP I	NBP II	NBP III	GoF I	GoF II	GoF III	WGB I	WGB II	WGB III	EGB I	EGB II	EGB III	GoR I	GoR II	GoR III	SBP I	SBP II	SBP III	BoG I	BoG II	BoG III	S I	S II	S III	BBS I	BBS II	BBS III	K I	K II	K III
No.of taxon. groups				9	6	5	6	8	9	15	14	16	10	14	14				7	13	11			6	22	18	22				20	13		26	24	26	28	22	17
Acartia spp.				97	87	37	72	75	20	53	53	12	72	57	47				23	25	39			70	20	22	24				27	39					15		25
Eurytemora affinis						43			50						18									19															
Pseudocalanus m.e.						17			26	30	25	20							62	50	29				62	25								29	14	17		12	29
Synchaeta spp.							21						20																										
Limnocalanus m.								13						11																									
Fritillaria borealis										14	15	39							24	17	19						18										11		
Temora longicornis												14																											
Copepod Nauplii														22												31	38					27						27	
Polychaeta Larvae																															43			20	24	14	19		
Oithona similis																																		19	26	38		11	22
Tintinnids																																		12					
Paracalanus parvus																																				15			
Bivalvia Larvae																																					20	22	

April - June	BB I	BB II	BB III	BS I	BS II	BS III	AAS I	AAS II	AAS III	NBP I	NBP II	NBP III	GoF I	GoF II	GoF III	WGB I	WGB II	WGB III	EGB I	EGB II	EGB III	GoR I	GoR II	GoR III	SBP I	SBP II	SBP III	BoG I	BoG II	BoG III	S I	S II	S III	BBS I	BBS II	BBS III	K I	K II	K III
No.of taxon. groups	10	7	10	13	10	11	17	15	10	20	17	23	19	17	20	13	16	20	12	13	18			9	22	21	26				20	10		28	23	24	27	25	22
Limnocalanus m.	54	58	16																																				
Synchaeta spp.	29		73	42	45	61	26	25	61		28	75	38	51	37			45	12		83			33	15	23	60							50	43	30			20
Eurytemora affinis	14	39	10	23	12	19	17	25	14																														
Acartia spp.				33	43	15	21	46			23		32	13	11		19	13		14				21	14	27	13				47	72			12	10	22	11	13
Bivalvia Larvae							27		20						13		14																		10	16		24	16
Pseudocalanus m.e.										52	22		20			59	38		57	49					40	14								20			10		
Fritillaria borealis										13	11								11						10														
Copepod Nauplii														31																		10						22	
Keratella spp.															19									23															
Temora longicornis																11	11			15						11													
Evadne spec.																	14	17													18						21		
Oithona similis																																			11	24	13	18	25

July - September	BB I	BB II	BB III	BS I	BS II	BS III	AAS I	AAS II	AAS III	NBP I	NBP II	NBP III	GoF I	GoF II	GoF III	WGB I	WGB II	WGB III	EGB I	EGB II	EGB III	GoR I	GoR II	GoR III	SBP I	SBP II	SBP III	BoG I	BoG II	BoG III	S I	S II	S III	BBS I	BBS II	BBS III	K I	K II	K III
No.of taxon. groups	19	13	14	27	13	16	24	11	13	27	22	28	27	19	34	17	16	15	20	16	17			8	25	23	24		13	18	29	19		32	28	26	32	26	
Eurytemora affinis	31	38	27	24	20	18	11	38	19			11	19	16	17									20															
Keratella spp.	40	23	32		18	47	11		20	17	11	24	15	15	29									48															
Bosmina c.m.	15	27			11				21	31	49	27	43	49	24	70	86	61	67	60	79			22	51	43	37		11	32									
Synchaeta spp.			17						13																														
Acartia spp.				50	35	22	48	40	23	21	25	13				10		10							16	24	22		60	12	45	10		18	14				
Centropages hamatus							17																											12	14	12			
Pseudocalanus m.e.										11								14														16		18			13	15	
Bivalvia Larvae																											12							17	19	20	16		
Oithona similis																															14	40		19	21	29	23	41	
Copepod Nauplii																													17	24		17						14	
Temora lonigicornis																													11	10									
Oikopleura dioica																																					11		

October - December	BB I	BB II	BB III	BS I	BS II	BS III	AAS I	AAS II	AAS III	NBP I	NBP II	NBP III	GoF I	GoF II	GoF III	WGB I	WGB II	WGB III	EGB I	EGB II	EGB III	GoR I	GoR II	GoR III	SBP I	SBP II	SBP III	BoG I	BoG II	BoG III	S I	S II	S III	BBS I	BBS II	BBS III	K I	K II	K III
No.of taxon. groups	9	6	9	9	11	10	8	11	6	15	14	22	14	14	22	13	15	10	14	15	14			10	27	23	25		15	18	29	20		29	28	26	28	29	24
Eurytemora affinis	47	45	79	44	42	57	32	44	67					11	24									35															
Limnocalanus m.	37	53																																					
Cyclopoida			12																																				
Acartia spp.				46	56	40	57	53	27	46	40	32	55	36	15	35	34		30	26	18			48	34	36	17		17		12	34		10					
Temora longicornis										27	23	23	14	11		20	26	40	34	25	26				19	21	26			10									
Synchaeta spp.											20	14	18	15	47	15		18	13	29	12				22	16	18		19	23									
Fritillaria borealis												14						24			17									11									
Copepod Nauplii														24												11	13		45	37		29						12	
Pseudocalanus m.e.																20	18																	31	13	12	10	11	
Centropages hamatus																					11													15					
Oithona similis																															31	17		26	30	35	19	33	42
Bivalvia Larvae																															20					12	27	15	15
Oikopleura dioica																																					13		
Paracalanus parvus																																							10

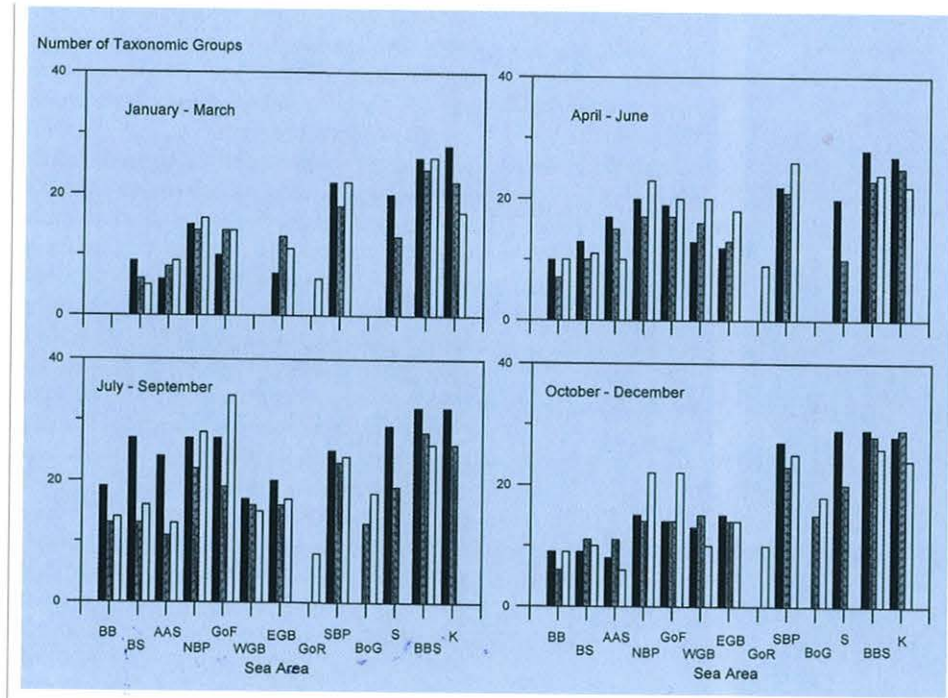


Fig. 10.2.2 Mean number of taxonomic groups of mesozooplankton in the surface water layer, averaged by areas, seasons and monitoring periods
(black column - 1979-83, white column - 1984-88, grey column - 1989-93; for acronyms of areas see Table 10.2.1)

summer, the copepod *Eurytemora affinis* and rotifers of the genus *Keratella* dominate the waters north of the Åland Islands. The Gulf of Finland is affected by inflow of the river Neva, the largest single fresh-water source of the Baltic Sea. As a result of this, higher numbers of fresh-water species occur here, particularly cladocerans (*Daphnia* spp., *Bythotrepes longimanus*, etc.) and rotifers (*Keratella* spp., *Polyarthra* spp., etc.).

Species numbers in the surface layer of the different sea areas follow the *brackish-water rule* [*565*], with a species minimum in the salinity regime of 5-7 psu, and increasing numbers towards the marine and fresh water regions. Table 10.2.2 and Figure 10.2.2 include counts of taxonomic groups reported to HELCOM. The highest numbers, 28-32 taxonomic groups, were found in the south-western parts of the Baltic Proper, which are most influenced by the North Sea. The lowest numbers, 13-20 taxonomic groups, occurred in the

***Table 10.2.2** Compilation of mesozooplankton species contributing >10 % to the total area-averaged mean abundance in the surface water layer of the Baltic Sea*

BB - Bothnian Bay, BS - Bothnian Sea, AAS - Åland and Archipelago Seas, NBP - Northern Baltic Proper, GoF - Gulf of Finland, WGB - Western Gotland Basin, EGB - Eastern Gotland Basin, GoR - Gulf of Riga, SBP - Southern Baltic Proper, BoG - Gdansk Bight, S - Sound, BBS - Belt Sea, K - Kattegat

Monitoring periods: I - 1979-83, II - 1984-88, III - 1989-93
Dominant species are pronounced by a frame, those contributing with >50 % are additionally in bold.

region of the Central Baltic Proper with 5-7 psu. In the Northern Baltic Proper and Gulf of Finland, due to the above mentioned input of fresh water species, higher numbers, 16-34 taxonomic groups, were reported. Exceptions to this rule are the Bothnian Bay and Bothnian Sea, where low species numbers (9-27) occurred. This is probably caused by the shorter production period.

Summer is the best season for regional comparisons of total zooplankton abundances. In summer, the average abundances in the Bothnian Sea, Åland and Archipelago Seas, Gulf of Finland and Northern Baltic Proper were higher than those in the other parts of the Baltic Proper. The highest abundance values were recorded, however, in the Gulf of Riga (Fig. 10.2.3c).

The mean abundance during winter and autumn is relatively low, and sometimes hard to separate from the noise in the surface layer data (Figs. 10.2.3a,d). The only exception with higher values was the relatively enclosed area of the Gulf of Riga. In the period from April to June, a latitudinal shift of the seasonal signal superimposes the regional differences (Fig. 10.2.3b). Again, the highest abundance values were recorded for the Gulf of Riga.

10.2.3. Long-term variations

Phytoplankton - As in the long-term data series for phytoplankton [*336,339,712*], the phytoplankton species composition showed pronounced variations between different sampling occasions and consecutive years. However, when considering the means over 5 year periods (Table 10.2.1), there were no remarkable differences in the species dominance. For all periodic assessment periods and in all sea areas but the Kattegat, the bulk of the algal biomass was formed by the same three to five dominating species. The Kattegat showed a pronounced variability in the averaged dominating species composition. It is suspected that this is due to the highly variable hydrography, as well as to the heavy anthropogenic nutrient loading to that area.

A decrease in the proportion of diatoms in the spring bloom, which benefited flagellates, was observed in the southern Gotland and Bornholm Basins since 1989/90, possibly due to mild winters or the exhaustion of silicates during the latter parts of the bloom.

With the exception of the Gulf of Riga, coherent patterns of eutrophication could not be demonstrated in the plankton community. The only significant trend observed was an increase in chlorophyll *a* during summer in the Bothnian Sea. However, this could not be confirmed by data on the phytoplankton biomass or carbon fixation from that area. This was in accordance with the lack of negative trends of oxygen concentrations in the deep layers of the Bothnian Sea. The low sampling frequency and the high variance of pelagic variables resulted, however, in a low statistical *power* hampering the detection of trends. In the Western Gulf of Finland, an increase in the peak values during the vernal blooms were observed for a time series covering the period 1968-88 [*189,190*].

In some areas, changes in nutrient loading, caused mainly by the removal of phosphorus from sewage water, or by an increasing riverine load, were reflected in the nutrient concentrations and their ratios. As diazotrophic, nitrogen fixing, cyanobacteria are mainly controlled by inorganic phosphorus [*340*], it could be expected that the major changes in the DIN/DIP ratio would be reflected in the intensity of the cyanobacterial blooms. In the Gdansk Bight, the DIN/DIP ratio in the bottom water increased due to river impact and favourable oxygen conditions. This is probably the reason why intense blooms of dia-

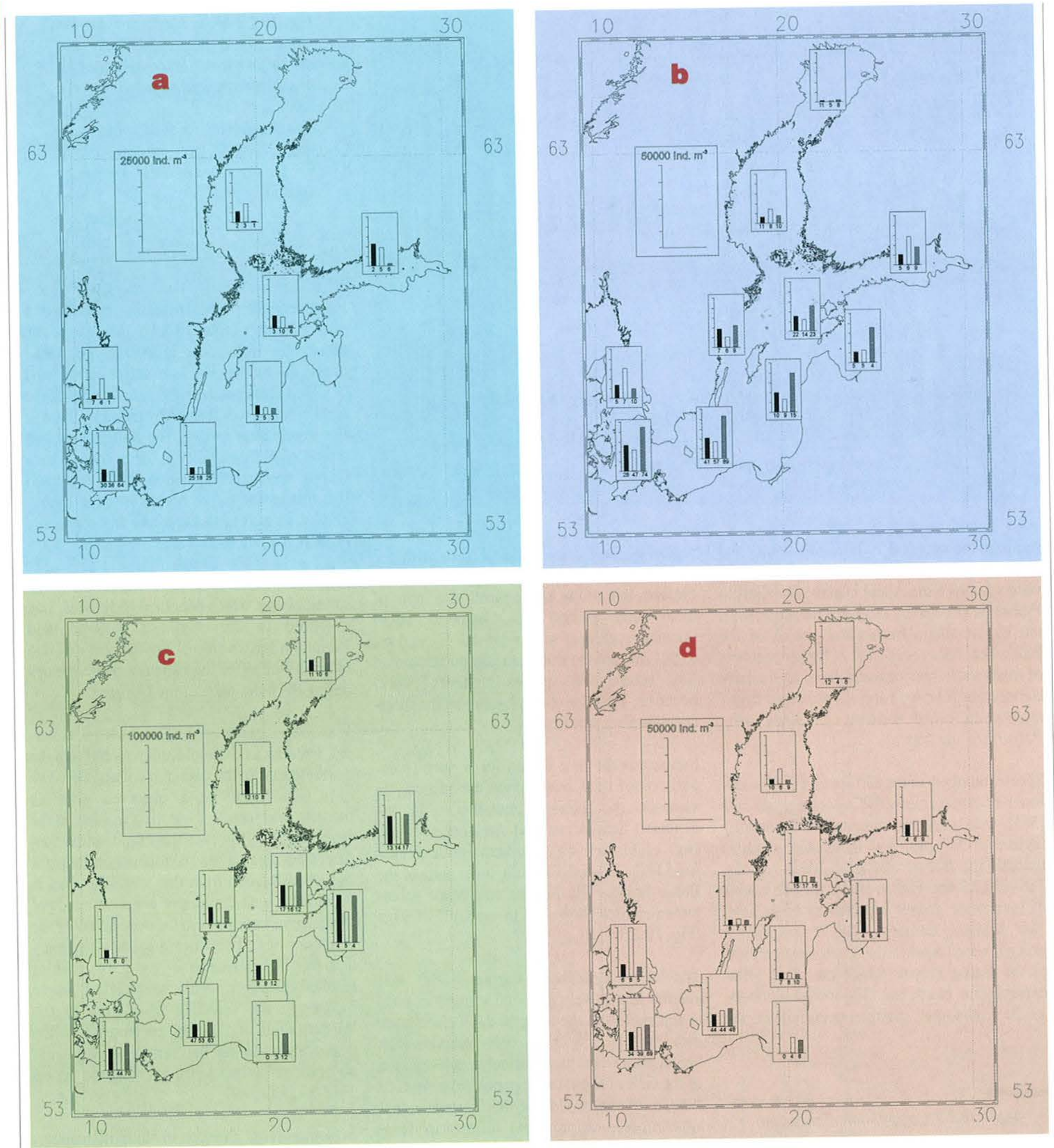


Fig. 10.2.3 Mean total mesozooplankton abundance in the surface water layer of the Baltic Sea, averaged by areas, seasons and monitoring periods (black column - 1979-83, white column - 1984-88, grey column - 1989-93)
a) winter (January-March)
b) spring (April-June)
c) summer (July-September)
d) autumn (October-December)

zotrophic cyanobacteria have not been reported for this area during the recent years. Opposite developments were observed in the Gulf of Riga, where a decrease of the nitrogen pool, due to a lower fresh water input and reductions in the use of fertilisers, was observed. This was probably reflected by distinct cyanobacteria blooms which appeared in the 1990s. Due to the completely different cycling mechanisms of nitrogen and phosphorus compounds, it has to be emphasised, however, that these observations do not support any 'cause-effect' speculations regarding the role of the removal of these nutrients from sewage water. The biogeochemical cycling of phosphorus is much more simple and contains only one major, oxygen controlled sink in the sediments. In contrast, nitrogen cycling includes sinks due to denitrification, which at present are poorly quantified, as well as a large, mainly unknown pool of organic nitrogen compounds.

Mesozooplankton - During the first half of this century, the mesozooplankton was most numerous in the southwestern parts of the Baltic Sea, decreasing successively to the north [*174*]. The eastern and northern regions were classified as oligotrophic. During that time, the open ocean was considered to be the main nutrient source needed for a successful plankton development. The closer the area to the North Sea, the higher was the amount of plankton. These conditions have changed significantly [*553*]. Contrary to the earlier description [*174*], the previous assessments showed that the summer abundance of zooplankton in the surface water layer of the Gulf of Finland and Northern Baltic Proper were higher than the Baltic Sea average (Fig. 10.2.3c). Additionally, during the 1980s, the Åland and Archipelago Seas showed increased zooplankton concentrations. For 1989-93, there was now a coherent area with abundances above the average, including the Northern Baltic Poper, Gulf of Finland, Åland and Archipelago Seas and Bothnian Sea. Here, the average concentrations exceeded those of the other regions by one third (Fig. 10.2.3c).

Eutrophication in the Gulf of Finland and in the Åland and Archipelago Seas was shown to exist at the beginning of the BMP. This fact is underlined by measurements on the abundance of the eutrophication indicating diatom *Thalassiosira hyperborea* var. *pelagica*, on the cladoceran *Bosmina coregoni* and on the carbon content in dated sediments from the Southern Bothnian Sea [*586*]. According to these results, the process of eutrophication started in the 1950s. In the following period, this process seems to have continued because both the average amount of zooplankton and the affected areas have increased (Fig. 10.2.3c). The highest zooplankton abundances were measured during summer and autumn in the Gulf of Riga (Fig. 10.2.3c,d).

Temporal changes of the mesozooplankton composition have been explained by fluctuations in salinity and temperature rather than by eutrophication [*678*]. Though not always statistically significant, those changes were distinct. The Darß Sill is not only a faunistic boundary. The dynamics of salt water inflows also provide temporal changes in environmental conditions west and east of this topographic border. East of it, the previous long stagnation period led to a decreasing salinity, while west of it, several weak salt water inflows have occurred. The change in salinity was mirrored by the zooplankton community. West of the sill, the marine cyclopoid *Oithona similis* is dominant, which does not occur in noteworthy numbers in the other parts of the Baltic Sea. Due to a number of inflow events, *Oithona similis* became even more important during 1989-93.

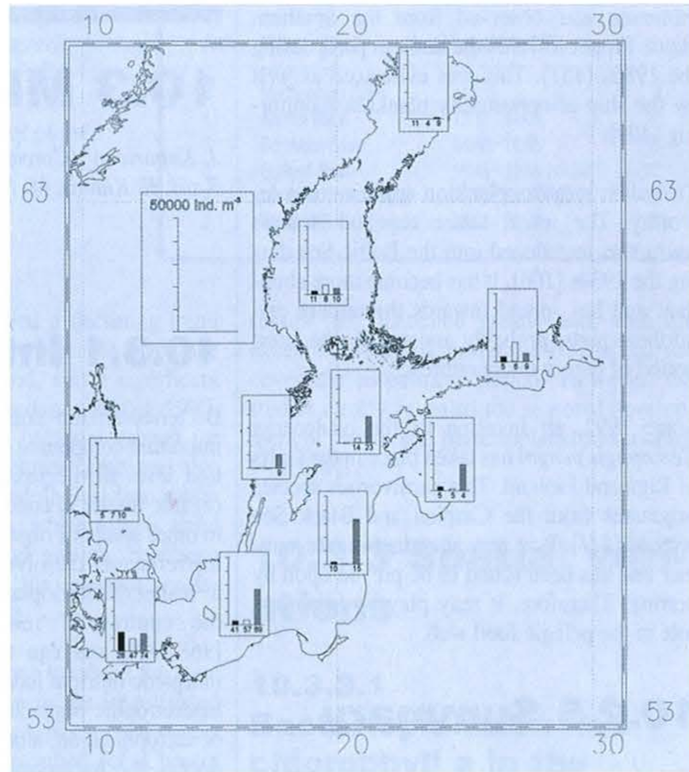


Fig. 10.2.4 Mean abundance of Synchaeta spp. in spring, averaged by areas and periods (black column - 1979-83, white column - 1984-88, grey column - 1989-93)

East of the Darß Sill, from 1984-88 to 1989-93, the stagnation period led to a shift in both the species numbers and in dominant species. The species number in the Northern Baltic Proper and Gulf of Finland increased (Fig. 10.2.2). This was partly due to the introduction of euryhaline fresh water species, (e.g., 8 species of the rotifer genus *Keratella* were observed in 1989-93, compared to only 3 in 1984-88), or even of true fresh water species as the cladoceran species *Bythotrepes longimanus* or the rotifers of the genus *Polyarthra* and *Kellicottia longispina*. The dominant plankton also changed. In the Central Baltic Proper, the dominant halophilic cold water species *Pseudocalanus minutus elongatus* decreased, and *Acartia* spp. took its place. In the northern parts of the Baltic Proper, where *Acartia* spp. were dominant during 1984-88, they have been replaced by *Eurytemora affinis*, a more brackish water species. These successive substitutions are clearly induced by the salinity decline. In summer, however, they are masked in the Baltic Proper by the dominance of the cladoceran *Bosmina coregoni maritima*, and in spring during 1989-93 by rotifers of the genus *Synchaeta*.

The salinity-induced changes were accompanied by temperature effects. The disappearance of the large cold-stenotherm copepod *Limnocalanus macrurus* in the Bothnian Bay can be explained by increasing temperatures which led to the decline of the total mesozooplankton biomass in the northernmost parts of the Baltic Sea. This is also confirmed by the decreasing stocks of this species in the Gulf of Riga since the beginning of the 1980s. Other species were favoured by the series of very mild winters during 1989-93, particularly during spring (Fig. 10.2.3b), and in the southwestern parts during winter (Fig. 10.2.3a). An example for an earlier seasonal scheduling is the appendicularian *Fritillaria borealis*. Regularly, this species was found to be abundant in the Northern Baltic Proper from April to June. During the assessment period, it also became dominant during winter (Table 10.2.2). An extreme example are the rotifers *Synchaeta* spp., which in 1989-93 increased in abundance by up to tenfold and dominated all over the Central Baltic Proper (Fig. 10.2.4). The lowered salinity, together with the mild winters, are possibly the main reasons for such visible changes in the population of *Synchaeta* spp.

On the other hand, the development of the rotifers may provide hints on eutrophication as well because with their rapid reproductive cycle, they can respond very quickly to increasing phytoplankton concentrations. Therefore, they have been considered as indicators for changes of the trophic state [*586*].

10.2.4 Alien species

The phytoplankton species data do not allow a reliable evaluation of the possible introduction of new species into the Baltic Sea. A gradual intrusion of the potentially toxic *Prorocentrum* *minimum* was observed from the Southern Baltic Proper towards the nothern parts during the 1980s [*151*]. This was evidenced as well by the ship-of-opportunity plankton monitoring [*409*].

Two alien mesozooplankton species are noteworthy. The warm water copepod *Acartia tonsa* was introduced into the Baltic Sea during the 1930s [*106*]. It has become more abundant and has spread towards the eastern and northern parts, probably assisted by the latest period of higher temperatures.

Since 1992, an invasion of the cladoceran *Cercopagis pengoi* has taken place in the Gulfs of Riga and Finland. This carnivorous species originates from the Caspian and Black Sea regions [*511*]. It is very abundant in late summer and has been found to be preyed upon by herring. Therefore, it may play an important role in the pelagic food web.

10.2.5 Summary

The phytopankton data do not reveal any drastic changes in the open Baltic Sea ecosystem. Pronounced effects of eutrophication are most evident in areas near to the major local nutrient sources. In some areas, the changes of the nutrient ratios, which are due to changes in waste-water technologies, in the use of fertilisers, or in the riverine loads, and the inter-annual differences of the winter temperatures were reflected by the phytoplankton as shifts of dominance between functional groups, such as diazotrophic cyanobacteria, diatoms and flagellates.

During the assessment period, the changes detected in the total abundance as well as the species composition of the mesozooplankton community, are probably explained mainly by changes in salinity and temperature rather than by eutrophication. The increase in the number of the rotifers species and the *Synchaeta* spp. abundance can be interpreted as sign of a changing trophic state. On the other hand, these increases were also favoured by mild winters and decreasing salinity.

In the first half of this century , the Northern Baltic Proper, Gulf of Finland, Åland and Archipelago Seas and Bothnian Sea were described as poor in zooplankton. Today, in these areas higher abundance values are found than those exhibited in the remaining Baltic Sea.

10.3 MICROBIAL FOOD WEB

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10.3.1 Introduction

Bacterioplankton communities contribute an important component of the secondary production level in the seas, by utilising dissolved organic matter in concentrations, not available to other levels of organization, and by degrading refractory dissolved and particulate organic matter. Bacterioplankton also participate in the control of inorganic nutrient pools [*368,739*], and can thus channel effects of inorganic nutrient loading from autotrophic to heterotrophic organisms. Thus, in the process of eutrophication, along with the primary producers, the bacterioplankton community contributes to the response at the base of the food chain. In a general sense, bacterioplankton activity (production and biomass increase) is an integrated part of the dynamics of organic and inorganic matter pools, and well reflects the state of the aquatic environment. In offshore areas, coupled phyto- and bacterioplankton responses are expected from the input of inorganic nutrients. In coastal areas, the responses may be uncoupled, as land-based particulate and dissolved products can give rise solely to bacterial production and biomass.

Despite the non-obligatory status of bacterioplankton measurements in the BMP, reliable data sets are available from the Baltic Sea due to local research (cf. Chapter 4.1.4.2) and monitoring activities [*177*]. The main features of the seasonal development of the bacterioplankton can be described for the main basins of the Baltic Sea. This chapter presents some of the available data and tries to draw some conclusions about trends, although the number and quality of data are only sufficient for trend analysis at specific sites, such as in the Gulf of Bothnia (cf. Chapter 4.1.4.2) and in the Kiel and Mecklenburg Bights [*177*].

10.3.2 Temporal variations

10.3.2.1 Gulf of Bothnia

Seasonal bacterioplankton data are available for the period 1984-94 from the Öre Estuary, Bothnian Sea (cf. Chapter 4.1.4.2). An increasing trend of the annual bacterioplankton growth was recorded during 1987-92, after which the values started to decrease. This trend was correlated with the inter-annual variation of the phytoplankton productivity. The subsequent decrease during 1992-94 was similar to the trend of chlorophyll *a* for that period. Within each year, the bacterioplankton show two intensive growth periods, in spring and summer. The peak values during both seasons appear in a narrow temporal scale, and no clear trend is evident for which of the two growth periods contribute more to the yearly production estimate. The pooled seasonal chlorophyll *a* data show higher spring than summer averages at all measured localities. However, this trend was not seen clearly in the bacterioplankton spring data.

In an overall assessment, no pronounced increase was observed in organic production in the Bothnian Bay or Bothnian Sea until 1993. This was in accordance with the trends seen for bacterioplankton measures. The strongest predictor of the bacterial specific growth rate was the phytoplankton productivity (cf. Chapter 4.1.4.2), which could be expected as bacteria obtain carbon and energy sources from phytoplankton processes. However, as pointed out by system-ecological studies in the Gulf of Bothnia [*373*], heterotrophic processes dominate the plankton community, especially in the Bothnian Bay, and bacteria obtain much of their carbon and energy sources from allochthonous matter, for which tight coupling of autotrophic and heterotrophic processes is not expected in the Gulf of Bothnia.

The pelagic measures showed no trends of the organic production during 1984-94, and thus no signs of eutrophication during that period. The measurements showed, however, that intensive autotrophic and heterotrophic growth occurs on a relatively small temporal scale, and that annual estimates are sensitive to both the overall sampling frequency, but particularly the frequency around the most intensive growth periods. In the pelagic system, eutrophication is indicated by an increased intensity and frequency of growth occasions, rather than by a linear trend in the integrated annual values [*189*]. The data thus suggests that in future monitoring activity more emphasis should be placed on recording short-term, intensive growth periods. This, on the other hand, calls for a more interactive monitoring programme, and the application of research strategies to trace short-term processes.

10.3.2.2 Kiel and Mecklenburg Bights

Clear seasonal trends on bacterioplankton variables are also available from measurements over a relatively long period, 1988 to 1995, in the southwestern parts of the Baltic Sea, i.e., at *Boknis Eck*, Kiel Bight, Fehmarn Belt and Mecklenburg Bight, localities which represent a gradient from the inner fjords to the open sea [*176,177*]. A high seasonal variability in bacterioplankton production, with maximum values during summer and/or autumn, was recorded at all localities [*177*]. Regression analysis between the rate variables of phyto- and bacterioplankton showed a tendency to an increasing correlation along a gradient from coastal to offshore stations, with a best fit at the open sea station Mecklenburg Bight [*177*]. The decoupling between phyto- and bacterioplankton at the coastal stations, and their coupling at the offshore stations, is conceptually self-evident, but this had not been previously demonstrated on a larger scale by field data. Inter-annual decoupling of phyto- and bacterioplankton at the coastal stations of the Gulf of Bothnia (cf. Chapter 4.1.4.2) also suggests that bacterioplankton measures are not fully phytoplankton-dependent, and that bacterioplankton should be used in assessing the organic production of the Baltic Sea.

In the open sea areas, the bacterioplankton is more dependent on the phytoplankton-carbon production, and thus may provide for monitoring purposes a methodologically independent estimate of the organic production in the trophic layer. However, due to the dynamic nature of the open sea plankton processes, bacterioplankton-rate measurements, together with chlorophyll *a* measurements, also give early signals on the development of organic production (eutrophication) in the open sea areas. Furthermore, bacterioplankton growth may provide a rough estimate of the respiration in the aphotic zone.

Table 10.3.1 Sea areas used for the analysis of bacterioplankton and chlorophyll a data
n - average number of observations for bacteria

Area	Name	Longitude (°E)	n
0	Mecklenburg Bight	- 12.15	3
1	Arkona Basin	12.15 - 14.40	17
2	Bornholm Basin	14.40 - 16.45	13
3	Gotland Basin	16.45 - 19.45 (19.30)	27
4	Northern Baltic Proper	19.45 (19.30) - 23.25	32
5	Western Gulf of Finland	23.25 - 25.00	10

The trend analysis showed a declining trend for the bacterioplankton biomass over the whole investigation period, and a significant increase in bacterial abundance during 1990-95. A declining trend in bacterial growth for *Boknis Eck*, but not for Fehmarn Belt, was also demonstrated. The methodological inaccuracy in cell-volume estimates, which affects trend analyses on biomass and growth, has been thoroughly discussed in the regional assessment (cf. Chapter 4.5.4.3).

Despite the lack of obvious trends in the state of the organic production in the investigated area, the changes in the environmental conditions have been well described (cf. Chapter 4.5.4.3). The observations of the water column temperature, resulting in prolonged stagnation conditions during major growth seasons, and observations of the precipitation conditions, resulting in changes in bacterioplankton-substrate supply, clearly explain the observed trends during the 1990s and stress the need for a system-analytical approach in the assessment of changes in the environment. The decoupling of phytoplankton and bacterioplankton processes in enclosed bights and at coastal sites, observed even at those sites which are eutrophic, support the conclusions obtained from the Gulf of Bothnia studies [*708*] that bacterioplankton-carbon production should be used in assessing organic production of the Baltic Sea.

10.3.2.3 Other areas

Several independent studies in the Southern and Northern Baltic Proper and in the Western Gulf of Finland were carried out during the assessment period (e.g., [*412,435,649*]). These studies are not suitable for trend analyses, since the sampling frequency over the growth season has fluctuated greatly and, with the exception of a few years, the studies did not cover the assessment period. However, the studies clearly revealed the seasonal development of phyto- and bacterioplankton variables [*435,649*].

10.3.3 Spatial variations

10.3.3.1 Bacterioplankton and chlorophyll a in the Baltic Proper

The spatial variability of bacterioplankton and chlorophyll *a* was studied during 1993 with the automated measuring system installed on the passenger ferry *Finnjet*, supplemented with on-board laboratory measurements of bacterioplankton, i.e., including incubations for leucine incorporation and preservation of the samples for microscopy. The *Finnjet* route is given in the Figure 9.1.1.

Bacterioplankton measurements, performed on subsamples in triplicate or duplicate, were run every 15 minutes, giving a spatial resolution of 1.3 to 2 km depending on the ships speed. Triplicate incubations were performed every full hour to check the inter-sample variability. On other sampling occasions, measurements were based on duplicate incubations. Chlorophyll *a*, salinity and temperature measurements were taken with a spatial resolution of 100-200 m. For correlation analysis between chlorophyll and bacterioplankton, chlorophyll *a* values were averaged over one mile on each automatic sampling occasion for

Table 10.3.2 Summary statistics of measurements on 5 Finnjet cruises, 1995
(average values underlined, n - sample number)

	Spring	Summer	Autumn	All samples
Chlorophyll *a* ($\mu g\ dm^{-3}$)	6.19 ± 7.55 (n=*223*)	3.66 ± 1.43 (n=*151*)	4.98 ± 0.80 (n=*92*)	5.13 ± 5.44 (n=*466*)
Leucine incorp. ($pmol\ dm^{-3}\ h^{-1}$)	34.1 ± 24.6 (n=*219*)	286.5 ± 157.6 (n=*145*)	159.8 ± 59.1 (n=*86*)	139.5 ± 146.5 (n=*450*)
Bacteria (10^6 cells cm^{-3})	1.06 ± 0.52 (n=*60*)	3.74 ± 1.12 (n=*38*)	3.01 ± 0.63 (n=*23*)	2.27 ± 1.45 (n=*121*)

[1] see ANNEX 11.3 for addresses of authors

bacterioplankton, whereby sea water flows into the bottle during one minute. Chlorophyll *a* data from the *Finnjet* from 1992 and 1993 are presented in this assessment elsewhere (cf. Chapter 9.1). The division of sea areas used here is similar to that in Chapter 9.1, however, the divisions to Western and Northern Bornholm Sea (nos. 4 and 5, Fig. 9.1.5) were combined and the Mecklenburg Bight data (no. 2, Fig. 9.1.5) were omitted (Table 10.3.1). During summer, the ship sailed along the eastern side of Gotland, and during other seasons along the western and more sheltered side of the island.

Five cruises were completed during 1993 to cover the major seasons (Fig. 10.3.1). Cruise 1 was completed in 35 hours and the other cruises in 20-24 hours, which provided a synoptic overview of the central basins of the Baltic Sea. Diel variability is included in the data, but its magnitude cannot be extracted from the present data. However, independent studies of diel variability suggest significant diel signals only on occasions when bacterioplankton is tightly coupled to phytoplankton and limited by phytoplankton-carbon supply. Since during the course of one transect, the ship crossed several sea areas and several plankton patches, which were in different successional stages, the diel signal is masked by the spatial variability of the data. During the spring period (March-April), high values of chlorophyll *a* and low values of bacterioplankton-leucine incorporation were recorded (Fig. 10.3.1). During the summer period (June-July), low chlorophyll *a* and high leucine-incorporation values were recorded throughout the central basins (Fig. 10.3.1). During autumn (September-October), chlorophyll *a* and leucine-incorporation rates were relatively on a 'similar level' as expressed in the units of Figure 10.3.1. Summary statistics of the results of this study is given in Table 10.3.2.

The high spring chlorophyll *a* values, i.e., the peak values, show a transfer from south to north according to the spring bloom development in the different basins. This movement of the peak values is described in more detail in Chapter 9.1.

The spring values clearly point out the spatial and temporal decoupling of phyto- and bacterioplankton (R=0.058, df=217, p=0.396) as expressed with a state variable (phytoplankton chlorophyll *a*) and a rate variable (bacterioplankton growth). During the spring period, patches of phytoplankton develop rapidly, supplying food and energy for the bacterioplankton growth, which follows the phytoplankton development with a time lag of a few days [*367,370,388,709*] depending on the sea area. When the two state variables, bacterioplankton-cell counts and chlorophyll *a*, are compared with each other, the correlation improves considerably (R=0.679, df=59, p=0.000). The decoupling of rate and state variables is also shown in Figure 10.3.2, which describes bacterioplankton-cell numbers and leucine incorporation during cruises.

During summer, when the inorganic nutrients limit the growth of phytoplankton (e.g., [*333*]) and bacterioplankton [*368,740*] in the open sea, and the bacteria are primarily dependent on the phytoplankton-carbon supply [*368*], the growth rate of bacteria and chlorophyll *a* are significantly correlated (R=0.762, df=144, p=0.000) (Fig. 10.3.3). Also the state variables, cell number of bacteria and chlorophyll *a*, correlate significantly (R=0.597, df=37, p=0.000). The coupling of phyto- and bacterioplankton is shown in Figure 10.3.1 for the transition zone from the Arkona Basin to the Bornholm Basin (longitude 14.25°E) during cruise 4, and by the peak values for the Eastern Gotland Sea (Fig. 10.3.1). During both summer cruises, a patch of high bacterioplankton production was recorded, which covered several kilometers (Fig. 10.3.1).

The extent of maximum values over the seasonal average and the frequency of high values in a specific sea area suggest that temporarily and spatially restricted phyto-and bacterioplankton patches have a pronounced effect on the open sea carbon production in summer. The spatial scales of peak values also imply that contemporary monitoring strategies, with a low number of fixed offshore stations and a low sampling frequency, produce data for the organic production, which cannot be extrapolated over a basin-wide scale.

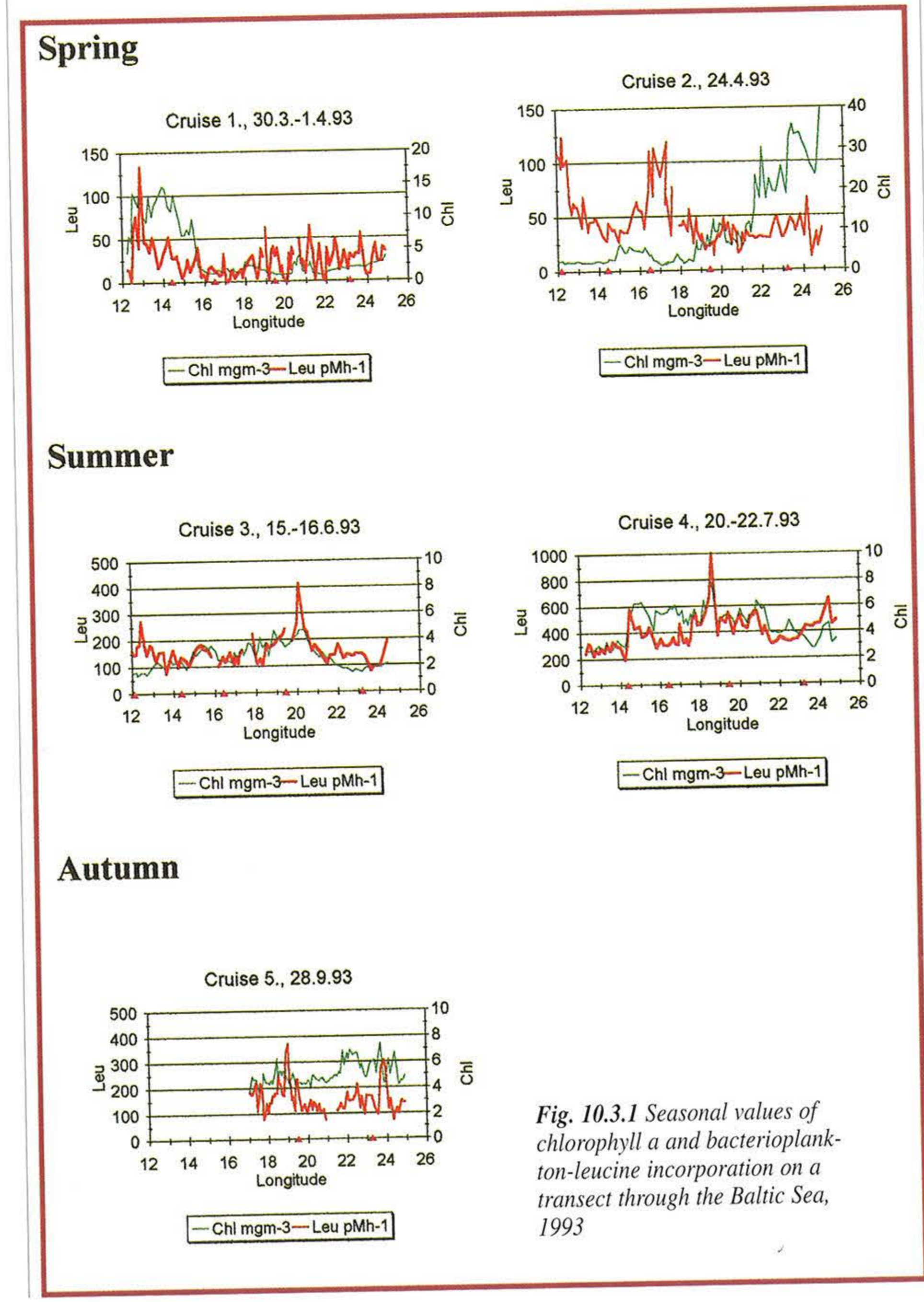


Fig. 10.3.1 Seasonal values of chlorophyll a and bacterioplankton-leucine incorporation on a transect through the Baltic Sea, 1993

10.3.3.2 Microbial food web during late summer

The joint multinational cruises into the open Baltic Sea were introduced by HELCOM in 1987 due to the fact, that studies of any single component of the ecosystem alone with periodic sampling had been proved to be of little value in the analysis of the development of the state of the Baltic Sea, as it did not allow an analysis of the state of the planktonic system based on the dynamical transitions, e.g., from both bottom-up to top-down controls, within the food web. The working hypothesis for these joint summer cruises was, that during mid-summer, i.e., during a period of regenerated production, the oligotrophic systems regenerate nutrients within the community, which is dominated by small organisms, and leave very little excess matter for sedimentation. In eutrophic systems, however, basic elements are stored and accumulated into biomass of the community, which is dominated by large organisms, accumulating organic matter for sedimentation. In each sea area, nutrient-enrichment experiments were run at the sampling sites to evaluate the state of the community. This extensive task could not be carried out by any single Baltic Sea institute, and therefore, the efforts and expertise were gathered for joint multinational cruises. The initial aim was to organize joint cruises on an annual basis. However, this soon became a biannual attempt, and finally, only Germany (1988 and 1994) and Finland (1990) have provided ship time for so far three studies.

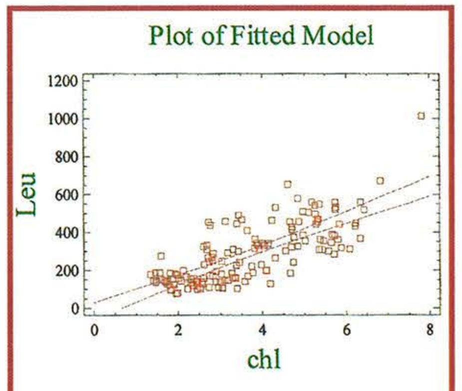


Fig. 10.3.3 Correlation between the growth rate of bacteria (in pmol leucine inc. dm^{-3} h^{-1}) and chlorophyll a (in mg m^{-3}) during summer

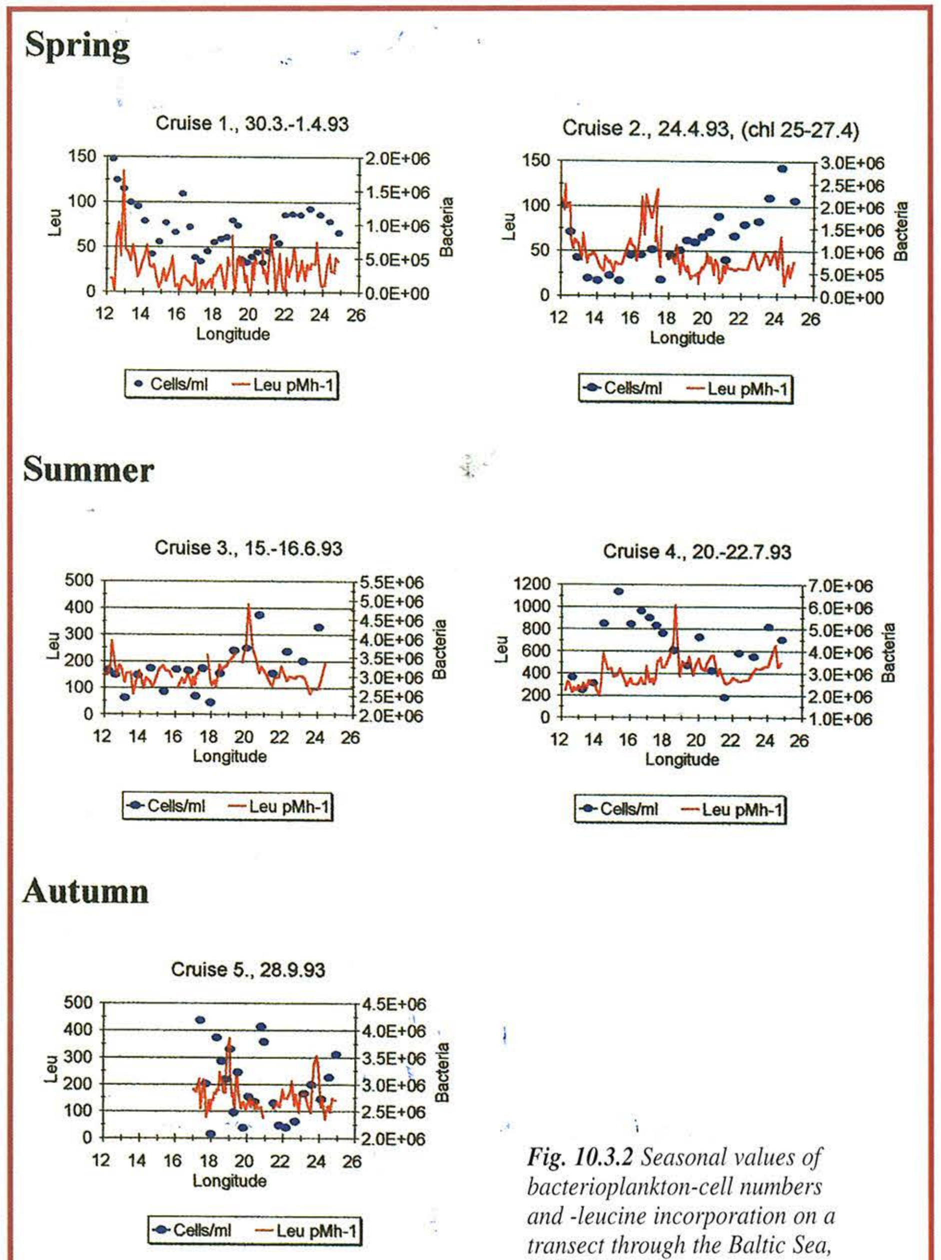


Fig. 10.3.2 Seasonal values of bacterioplankton-cell numbers and -leucine incorporation on a transect through the Baltic Sea, 1993

During the cruise in 1990, five stations were visited in different basins of the Baltic Sea, i.e., in the Kattegat (*R3*), Arkona Basin (*K4*), Bornholm Basin (*K2*), Gotland Basin (*J1*) and the Gulf of Finland (*LL11*). In 1994, cruises were made also to the Southern Gotland Basin (*K1*) and the Northern Gotland Basin (*H2*). During the 1990 cruise, low levels of NO_3-N and PO_4-P concentrations were recorded in all basins in the upper mixed layer (Fig. 10.3.4), and this represented the regenerating production period for the planktonic community, evidence for which was provided by the slightly elevated NH_4-N concentrations (Fig. 10.3.4). During the 1994 cruise, the nutrient concentrations were slightly higher, especially the nitrate concentrations in the Bornholm Basin and Southern Gotland Basin (Fig. 10.3.5). Warm and calm days prevailed during the week of the 1990 cruise, creating a secondary thermocline below 5 m depth at all stations but *LL11*, which experienced wind mixing prior to the sampling date (Fig. 10.3.4).

During the 1990 cruise, the phytoplankton-chlorophyll *a* maximum was recorded at the pycnocline (22 m) in the Kattegat, and above the thermocline (15 m) at all other stations. The vertical distribution of the planktonic variables followed the distribution of the primary production with maximum values at 2-5 m. Heterotrophic nanoflagellates dominated over autotrophic nanoflagellates in numbers at all stations except *LL11*. During the 1994 cruise, the pycnocline was at a much higher level (8 m) in the Kattegat, whereas in the Baltic Proper, the water-column stratification and the chlorophyll *a* distribution were similar to those observed during the 1990 cruise.

In 1990, the southernmost basin, the Arkona Basin, was most productive with a decreasing trend to the north (Fig. 10.3.6). More than 80 % of the phytoplankton-carbon production at all sites was due to organisms <10 µm.

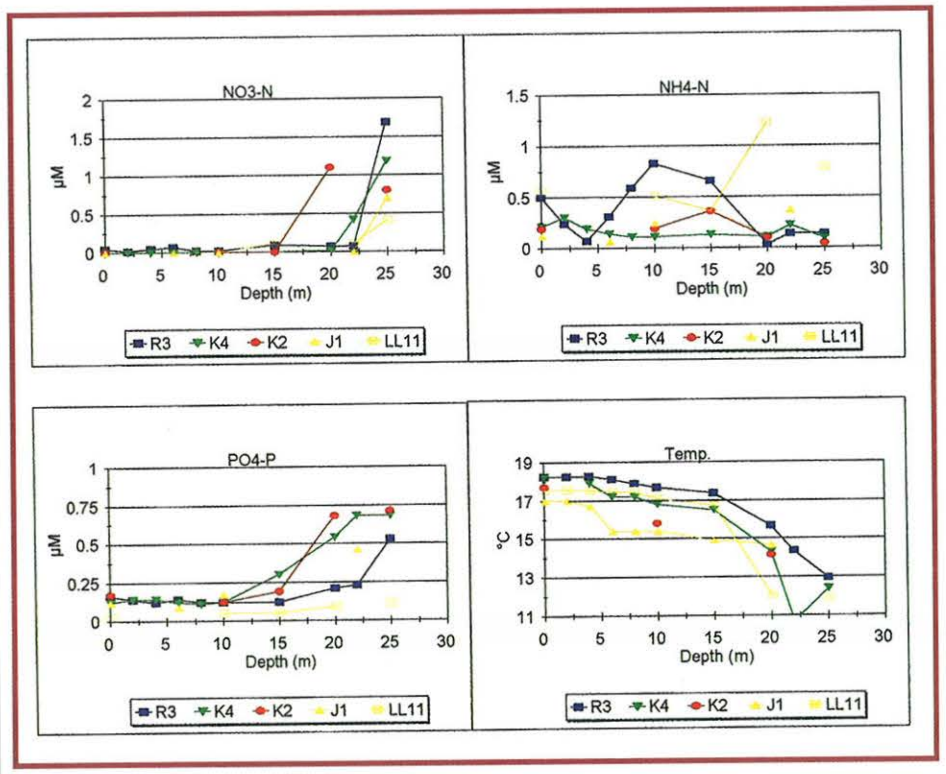


Fig. 10.3.4 *Temperature and nutrient concentration profiles at five stations, on a transect between the Kattegat and the Gulf of Finland, during the multinational cruise Joint-90, summer 1990*

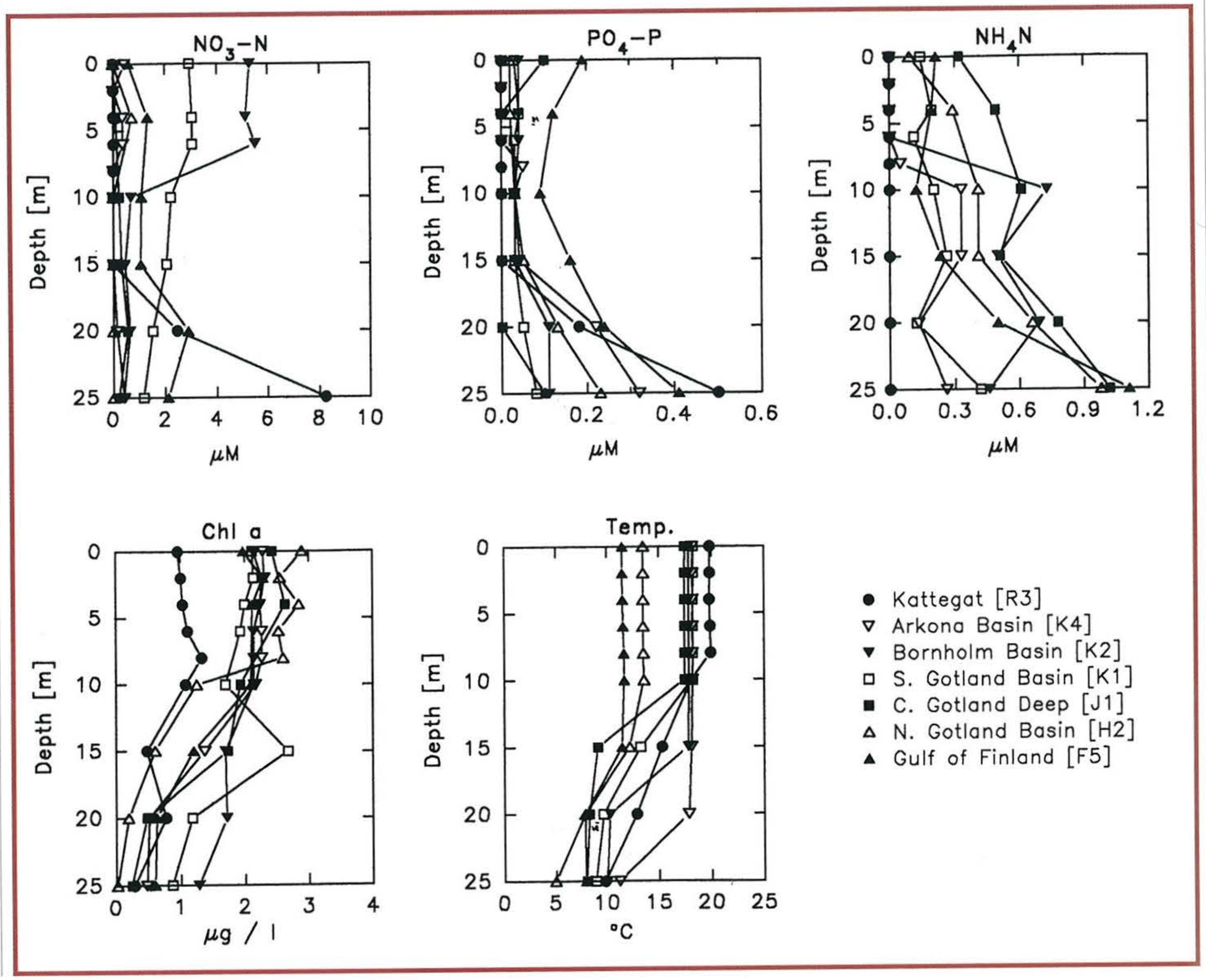


Fig. 10.3.5 *Temperature, chlorophyll a and nutrient concentration profiles at five stations, on a transect between the Kattegat and the Gulf of Finland, during the multinational cruise Joint-94, summer 1994*

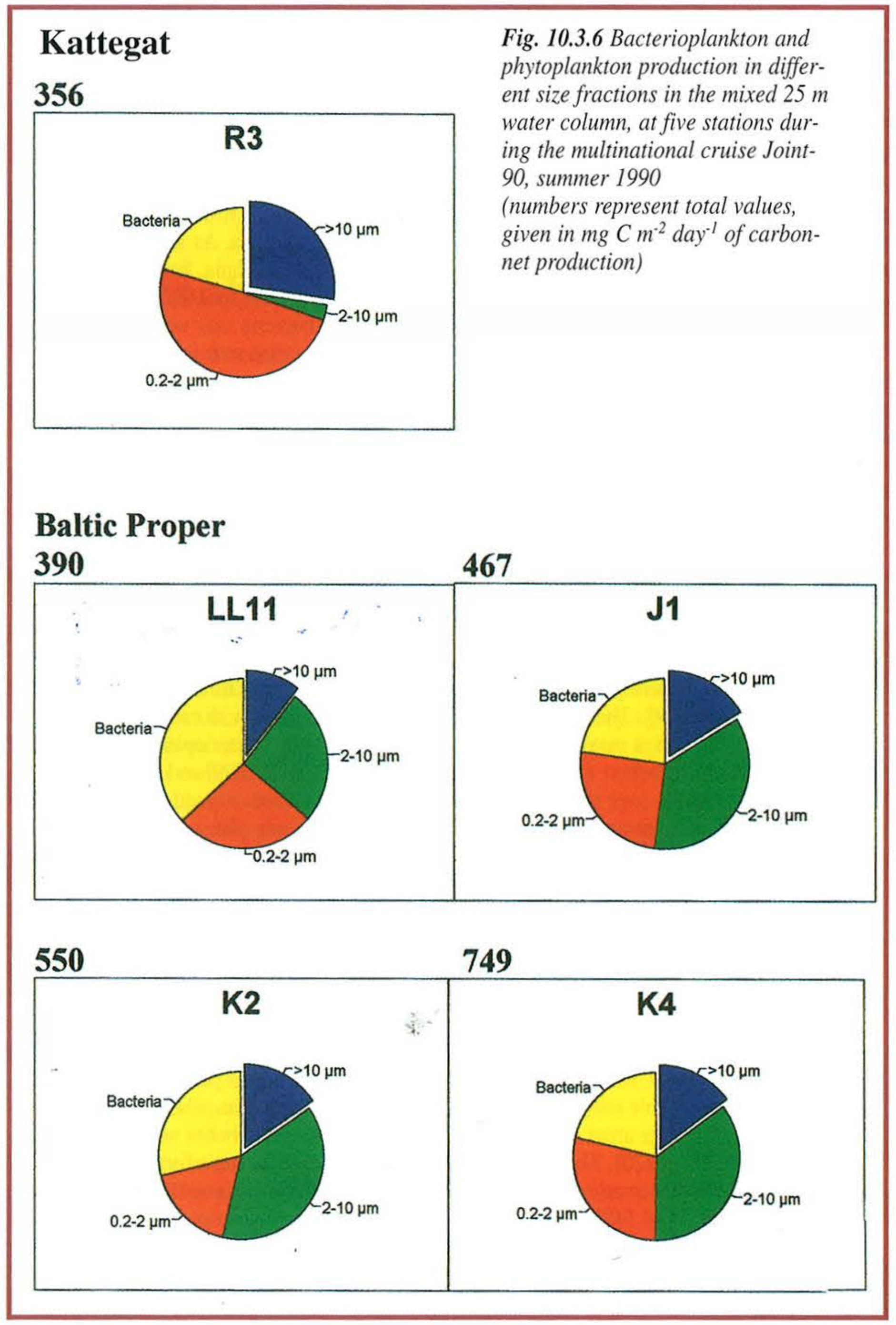
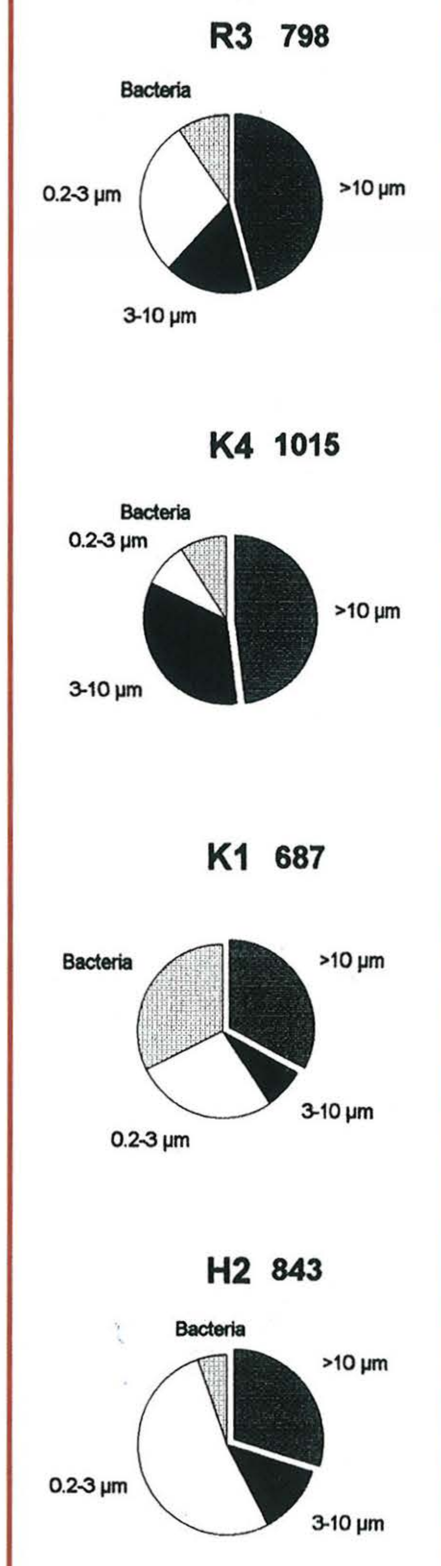


Fig. 10.3.6 *Bacterioplankton and phytoplankton production in different size fractions in the mixed 25 m water column, at five stations during the multinational cruise Joint-90, summer 1990 (numbers represent total values, given in mg C m^{-2} day^{-1} of carbonnet production)*

Fig. 10.3.7 *Bacterioplankton and phytoplankton production in different size fractions in the mixed 25 m water column, at five stations during the multinational cruise Joint-94, summer 1994 (numbers represent total values are given in mg C m^{-2} day^{-1} of carbonnet production)*



Basin	Phytoplankton production g C m^{-2} yr^{-1}	References	Bacterioplankton production g C m^{-2} yr^{-1}	Reference	Bacterio/Phytoplankton production ratio
Bothnian Bay, coast	16-29	[390,709]	-	-	-
Bothnian Bay, open sea	31	[705]	13	[373,708]	0.42
Bothnian Sea, coast	28-90	[390,709,740]	17-21	[707,740]	0.23 - 0.61
Bothnian Sea, open sea	92	[740]	22	[740]	0.24
Gulf of Finland, coast	65-126	[189,367,369,374,411,649]	9-19	[367,369,374,411,649]	0.14 - 0.15
N. Baltic Proper, coast	93-160	[387,388]	29	[387]	0.18 - 0.31
Belt Sea	150-270	[180]	22-43	[176,177,180]	0.14- 0.2

Table 10.3.3 *Annual primary and bacterioplankton production in different basins of the Baltic Sea*

Organisms in the size fraction 2-10 µm dominated the production at all other stations, but were practically absent in the Kattegat. When bacterioplankton production was included, 50 % or more of the carbon production took place in the picoplankton (<2 µm) size fraction (Fig. 10.3.6). The near absence of the component 2-10 µm size fraction in the Kattegat ecosystem is an indication of an imbalanced system, which is subject to rapid changes, e.g., blooms, as the large (>10 µm) and the small (<2 µm) organisms are at different levels in resource competition (mineral nutrients) and under the control of predators. The other localities showed a more balanced system structure and functioning and thus, despite the higher level of primary productivity could remain in the recycling status leaving very little excess matter for sedimentation.

In 1994, all stations were at a higher level of total carbon production, with a low contribution of bacteria, except in the Southern Gotland Basin. Due to differences in the pore size of the fractionation filters used during the two cruises, 2 µm in 1990 and 3 µm in 1994, direct comparison was possible only for certain size categories, i.e., for bacteria, for the 0.2-10 µm fraction and for the >10 µm fraction. At all stations, the larger plankton had dramatically increased and bacteria had decreased, except at station *K1*, where bacteria was present at a similar magnitude to that of the larger phytoplankton (Fig. 10.3.7). The dominance of larger phytoplankton (>10 µm) over the pico- and lower nanoplankton (<10 µm) implies a higher nutrient status in 1994 than in 1990. The difference between the two years may be explained by several factors, such as water-column stability, frequency of upwelling and atmospheric load, and may not be related to changes in the anthropogenic impact.

10.3.4 Discussion

Due to the short generation time of microbes, e.g., for bacterioplankton 1-2 days, even at temperatures so low as 1 °C, and the bi-directional control of bacteria (bottom-up and top-down) in the food web, strict constraints are given to the acquirement of information, particularly to the spatial and temporal coverage of sampling. Based on low-frequency sampling, it was stated [*210*] that bacterioplankton variations were connected more to the seasonal characteristics of the community development than to the spatial scale of sampling, and that variations over the Central Baltic Proper were greater and more random in spring than in summer, suggesting that phytoplankton and bacterioplankton variation was closely linked in the open areas of the Baltic Sea [*210*]. The *Finnjet* study confirms this basic concept of close coupling of phyto- and bacterioplankton in the open Baltic Sea throughout summer, and the apparent decoupling during the intensive growth period in spring (Fig. 10.3.1).

The basins of the Baltic Sea seem to have different levels in annual phytoplankton and bacterioplankton productivity (Table 10.3.3), with increasing values towards south. The few available data cannot, however, reveal the coupling or decoupling of the two processes in an annual scale, nor clearly point out the differences between coastal and offshore areas.

The coupling of the phyto- and bacterioplankton production in the open sea areas is well illustrated in Figure 10.3.1 and in the correlation coefficient from the summer cruises (Fig. 10.3.3). The significance of this correlation coefficient is, however, relatively low. These relationships may suggest that bacterioplankton is bottom-up regulated during the summer period, and that pools of organic carbon from phytoplankton regulate the bacterioplankton development. However, inorganic nutrients may primarily limit bacterioplankton growth during summer [*368,739*]. The high overall DOC level in the Baltic Sea may support part of the bacterioplankton-carbon requirements, making them competitive over phytoplankton in oligotrophic, low nutrient regimes as exhibited in the Gulf of Bothnia (cf. Chapter 4.1.4.2). Thus, during the regenerated summer period in the open parts of the Baltic Sea, bacteria are the first to react to low-level nutrient spiking, e.g., from upwelling, in the open pelagic systems.

The second periodic assessment stated that, on the basis of the regional surveys, bacterioplankton showed remarkable uniformity in the mixed surface layer in large areas of the Baltic Sea during late summer [*220*]. This is also in accordance with the observations from the *Finnjet* study in 1993 and [*48*], which point out that because of the bi-directional control of the bacterioplankton community, i.e., by grazing control from the top and by nutrient/carbon limitation from the bottom, cell counts may exhibit unexpected uniformity throughout the open-sea pelagic systems.

The variability analysis for the *Finnjet* study resulted in the same main conclusions as those presented in [*210*]. Phytoplankton (chlorophyll *a*) and bacterioplankton production, measured as leucine incorporation, had a total variance of 106 % and 107 %, respectively, with the main contribution from the spring variability. Bacterioplankton-cell number variability was only 64 % for all cruises, and much lower than the productivity for spring (49 % *vs.* 72 %) and summer (30 % *vs.* 55 %). The low annual and seasonal variability of the bacterioplankton-cell numbers imply an efficient and stable top-down control by the predators and a dependency on bacterioplankton-carbon sources in the food web, as suggested in the regional assessment for the Gulf of Bothnia (cf. Chapter 4.1.4.2).

The coupling of phyto- and bacterioplankton processes is disrupted in the coastal and gulf areas of the Baltic Sea, as pointed out by the correlation analysis from the southwestern parts of the Baltic Sea. As shown by the data from the Gulf of Bothnia, bacteria respond to organic loading from land, and secondary production from bacteria may well exceed that of zooplankton in areas rich in organic matter. A new concept, presented recently [*12*], suggests that high-molecular weight dissolved organic matter (DOM) can be more bioreactive than the low-molecular weight DOM, which previously has been considered as the main carbon source for bacterial growth. Pronounced nitrogen release from humic substances has recently been recorded [*112*], which supports that concept [*12*], for which further support has been obtained by studies from brown-water lakes. The views given for the Gulf of Bothnia that a high riverine DOC contribution, fuelling the marine food web with carbon, may support exceptional high bacterioplankton-secondary production [*373,708,709*] and thus, secondary-production measures should be examined in assessing the eutrophication status of the engulfed and coastal sea areas, rather than strictly focusing on phytoplankton as the key measure for eutrophication.

The studies throughout the Baltic Sea point out, that the lower part of the microbial food web, the pico- and nanoplankton, dominate the carbon and energy flow during the late summer season in the open parts of the Baltic Sea. The results suggest that, where eutrophication during summer periods has to be investigated in offshore areas, the main focus should be put on studies of the carbon flows from pico-, nano- and microplankton, since only in a developed eutrophic system, larger organisms will dominate to produce excess organic material for sedimentation. From this point of view, the basic concept given by the *ad hoc* Working Group on Microbiology in 1980 is still valid, i.e., that the size-fractionated production and the chlorophyll *a* measurements reflect the status of the open-sea pelagic system, and that the functional analysis of the system can be used to indicate early stages of eutrophication.

10.4 BENTHIC BIOLOGY

A.B. Josefson[1]

10.4.1 Macrozoobenthos

This chapter is an extraction of general features from data presented in the sub-regional assessments.

Spatial patterns - The distribution of mean total abundance in the assessment period shows similar values in the northern and eastern gulfs and in the western area, Kattegat and the Belt Sea (Fig. 10.4.1). Abundances in the Baltic Proper are generally much lower than in these gulf areas, except in the Eastern Gotland Basin, where values are highly variable. Benthic biomass, on the other hand, shows a totally different picture, in that values in the northern and eastern gulfs are considerably lower than in the western area (Fig. 10.4.2). This difference is likely to be even greater than shown in this figure, since the large-sized sea-urchins and Ocean Quahog, which do not occur in areas east and north of Arkona, are excluded from the biomass values. Since benthic biomass may be a good predictor of benthic secondary production [*666*], it is likely, that benthic production is higher in the Kattegat area. The difference may be attributed to differential effects of eutrophication at present and in the past, and/or different degree of coupling between pelagic and benthic production. The low values of both abundance and biomass in the basins of the Baltic Proper certainly result from adverse oxygen conditions.

Temporal changes - In general, there are few or no changes in the macrozoobenthos communities during the last assessment period, that can be related to deteriorating oxygen conditions in the bottom water. There are two examples of the opposite situation, in the Gulf of Finland and on the slope of the Eastern Gotland Basin, where macrofauna has colonised previously "dead" bottoms, apparently due to improved oxygen conditions. In other previously defaunated areas in the Baltic Proper, the situation is unchanged, with no or impoverished fauna present during the assessment period.

In previously (before 1989) faunated areas, where permanent hypoxic conditions do not prevail, the total macrofaunal densities and biomasses do not show a general change compared to the previous assessment. These areas include the Kattegat, the Bothnian Sea, the Bothnian Bay and the Gulf of Riga. However, considering a longer time period, i.e., since the 1970s, the trend of increasing biomass has levelled off. In the Kattegat, animal groups dominated by short-lived species, crustaceans and polychaetes, showed decreases in abundance.

A conspicuous feature during the last 15 years, which occurred in the Kattegat, the Bothnian Sea, the Bothnian Bay and the Gulf of Riga, is a bimodal pattern of abundance, and to some extent biomass, in macrozoobenthos, in particular for crustaceans. A strong peak occurred in the beginning of the 1980s, which continued through to the middle of this decade. Another peak occurred in the late 1980s/early 1990s. The large scale of this pattern, together with the fact that the major actors (species) in this pattern are different in different areas, suggests the importance of external factors acting on a large scale, such as climate and eutrophication. This pattern resembles the pattern for land run-

Fig. 10.4.1
Distribution of total macrofaunal abundance among different sub-areas in the Baltic Sea

(height of bars denote maximum value, and filled area the range in the period 1989-94;

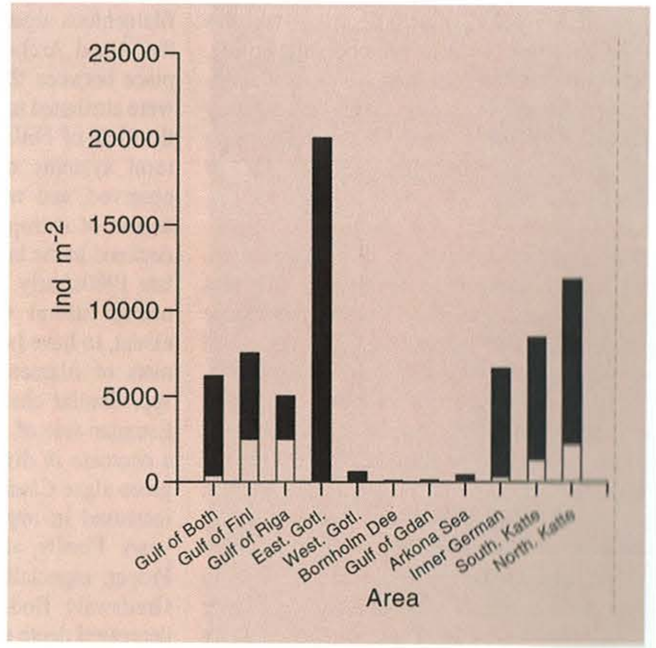
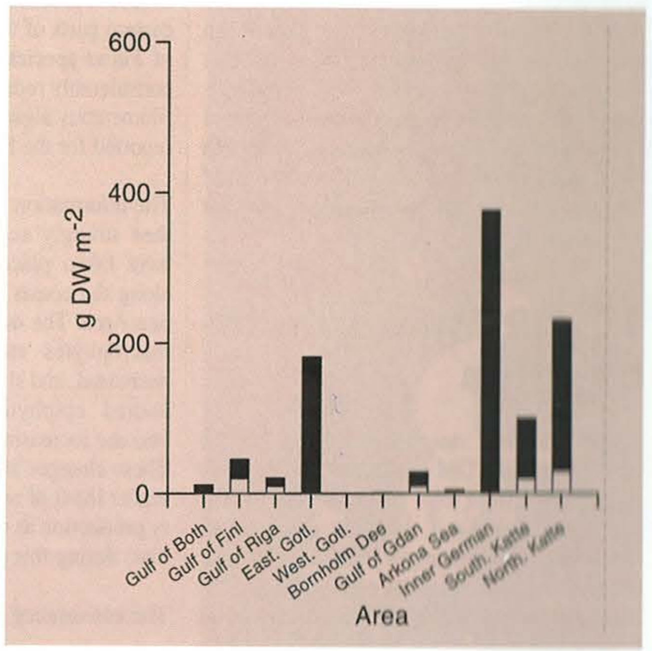


Fig. 10.4.2
Distribution of macrofauna biomass (dry-weight), excluding the large-sized sea-urchins (Echinoida) and Ocean Quahog (Arctica islandica), among sub-areas of the Baltic Sea, 1989-94 (cf. Fig. 10.4.1)



[1] see ANNEX 11.3 for addresses of authors

off during the same period, which may suggest a causal link for the sequence nutrient input - primary production - sedimentation. In the Kattegat, where the second peak for crustaceans was much weaker than the first one, this pattern correlates positively with the diatom weight in the 0-10 m layer for the previous year at most stations. Diatoms are considered to be a very important link between pelagic and benthic production, mainly because they show high sinking rates and have high nutritional value. Since it is likely that the macrofaunal standing stock is often food limited, one cannot exclude the possibility that the decreased abundance of crustaceans, and to some extent also polychaetes, in the 1980s and early 1990s, as well as the levelling off of biomass, may be related to a decreased sedimentation of phytoplankton to the bottom during that period. In support of this explanation is also the fact, that short-lived taxa, such as many crustaceans (having life spans ranging from 1 to 3 years) show a decrease, whereas long-lived taxa, such as echinoderms, do not. When food is less abundant, which may inhibit recruitment, the latter forms simply stay longer on the bottom, because they have a long life span, e.g., for *Amphiura filiformis* >6-7 or even 10-25 years.

Decreased sedimentation of phytoplankton, particularly diatoms, would be predicted from both decreased run-off, which may influence nutrient input to the sea, and also decreased concentrations of dissolved silica [*121,558*]. As in the central parts of the Baltic Sea [*558*], decreasing trends of dissolved silica also seem to be apparent in the Kattegat area during the same period. In the Kattegat, the diatom weights showed a dramatic decrease in spring during the 1980s, in particular in the end of the 1980s, and similar changes have occurred in the Gotland and Bornholm Basins. In other areas, however, such changes have not yet been detected. It is at present premature to attribute faunal changes on the bottom to either of the two factors, decreased nutrient load or silica depletion. This should be a relevant topic for future work. Apparently, several of the benthic changes are in accordance with the hypothesis of decreased sedimentation of organic matter to the bottom during the last assessment period.

10.4.2 Macrophytobenthos

Measurements of macrophytobenthos are not part of the HELCOM monitoring programme. Information from the previous assessment period is scattered and, therefore, precludes an assessment of changes since 1984-88. However, several data sets on benthic vegetation exist, enabling comparisons to be made over longer time periods.

In the Kattegat and the Belt Sea, two major changes were observed in this century, (*a*) a decreasing depth distribution of eelgrass (*Zostera marina*) and other perennial macroalgae, and the disappearance of one *Fucus* species in some parts of the Belt Sea and the southern Kattegat, and (*b*) an increased coverage by quick-growing epiphytic or drifting algae which are favoured by nutrient enrichment. Both types of changes have been attributed to increasing nutrient loads, and decreased light penetration as a consequence of increased primary productivity in case (*a*), and increased nutrient concentrations in case (*b*).

In the Gulf of Bothnia, the major feature was a decreased coverage and depth distribution of the perennial bladder wrack (*Fucus vesiculosus*) and increased coverage of fast growing filamentous algae in the Archipelago Sea and the Åland Archipelago. These changes took place between the 1960s and the 1980s, and were attributed to increasing eutrophication. In the Gulf of Finland, great changes in the littoral systems on the Finnish coast were observed, and were attributed to a changing degree of eutrophication. The bladder wrack declined in the late 1970s and recovered in the late 1980s/early 1990s. The previously dominating littoral vegetation seems, to some extent, to have been replaced by drifting algal mats of filamentous algae like *Cladophora* spp. Similar changes were reported from the Estonian side of the Gulf. In the Gulf of Riga, a decrease in diversity was reported with the green algae *Cladophora glomerata*, which has increased in importance over the last 30-70 years. Finally, along the coasts of the Baltic Proper, especially in the Gdansk Bight and Greifswald Bodden, a general feature is a decreased depth distribution of the phytal zone in recent decades. In some of the southern and eastern parts of the Baltic Proper, the biomass of *Fucus* species and of *Zostera marina* was considerably reduced. Increased dominance of filamentous algae (*Ectocarpus* spp.) has been reported for the Polish coast.

The information available on macrophytobenthos strongly suggests, that general changes have taken place during the recent decades along the coasts of virtually the whole Baltic Sea Area. The depth distribution of perennial macrophytes attached to the bottom has decreased, and short-lived filamentous or thin-bodied epiphytic or drifting algae have become increasingly important in recent time. These changes are most easily explained by a higher input of nutrients with increased primary production as one consequence (eutrophication) during this period.

The consistency of the reports and the apparent significance of the observed changes call for a serious consideration, as to whether or not macrophytes should be part of a future HELCOM monitoring programme. However, since most of the information so far is 'soft', in some cases even anecdotal, further work is needed to standardise methods, to make the observations more objective than they are today.

11 ANNEXES

HELCOM

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11.2 ACRONYMS

AAS	Atomic absorption spectroscopy
ACFM	(ICES) Advisoy Committee on Fishery Management
ACME	(ICES) Advisory Committee on the Marine Environment
ADCP	Acoustic Doppler Current Profiler
ASCOBANS	Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas
AVHRR	Advanced Very High Resolution Radiometer
BBM	Bacterial biomass
BCF	Biological concentration factor
BETA	Steering Group for the Coordination of the Third Periodic Assessment
BMB	Baltic Marine Biologists
BMP	Baltic Monitoring Programme
Bq	Becquerel (1 disintegration s^{-1})
BSPA	Baltic Sea Protected Areas
BY	(International) Baltic Year (1969/70)
CB	Chlorobiphenyl
CCB	Coalition Clean Baltic
CFB	Colony-forming bacteria
COMBINE	Cooperative Monitoring in the Baltic Marine Environment
CTD	Conductivity-Temperature-Depth (Density) record
DDD	Dichloro-diphenyl-dichloro-ethane
DDE	Dichloro-diphenyl-dichloro-ethylene
DDT	Dichloro-diphenyl-trichloro-ethane
DNA	Deoxyribonucleic acid
DOC	Dissolved organic carbon
EGAP	Group of Experts on Airborne Pollution of the Baltic Sea Area
EMEP	European Monitoring and Evaluation Programme
GC	Gas chromatography
GC/MS	Gas chromatography/Mass spectrometry
HPLC	High-performance (-pressure) liquid chromatography
IAEA-MEL	International Atomic Energy Agency - Marine Environmental Laboratory
IBSFC	International Baltic Sea Fishery Commission
ICES	International Council for the Exploration of the Sea
IOW	Baltic Sea Research Institute
IUCN	International Union for the Conservation of Nature
IMO	International Maritime Organization
JCP	Baltic Sea Joint Comprehensive Environmental Action Programme
kT	Kilo-tons (10^3 t)
MORS	Group of Experts on Monitoring of Radioactive Substances in the Baltic Sea
NOAA	(US) National Oceanic and Atmospheric Administration
NOx	Nitrogen oxides
OSPARCOM	Oslo and Paris Commissions
PAHs	Polycyclic aromatic hydrocarbons
PBq	Peta-Becquerel (10^{15} Bq)
PCBs	Polychlorinated biphenyls
PE	Population Equivalent
PITF	Programme Implementation Task Force
PLC	Pollution Load Compilation
POPs	Persistent organic pollutants
psu	practical salinity unit
RSD	Relative standard deviation
RTG	Radionuclide thermoelectric generator
S	Salinity
SCANS	Survey of Cetaceans of the Atlantic North Sea
SD	Standard deviation
STUK	Finnish Centre for Radiation and Nuclear Safety
TBN	Total bacterial number
TBq	Tera-Becquerel (10^{12} Bq)
TCDD	Tetra-chloro-dibenzo-dioxine
TeBDE	Tetra-bromo-diphenyl-ether
TOC	Total organic carbon
UNEP	United Nations Environmental Programme
UVF	Ultraviolet fluorescence
VPA	Virtual population analysis
WWF	World Wide Fund for Nature

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