Baltic Sea Environment Proceedings No. 82B

Environment of the Baltic Sea area 1994-1998

Helsinki Commission
Baltic Marine Environment Protection Commission
2002


Kemijoki
Bothnian Bay
Bothnian Sea
NORWAY
SWEDEN
FINLAND
Oslo
Helsinki
Gulf of Finland
Neva
St. Petersburg
Tallinn
Stockholm
ESTONIA
Göta älv
Gulf of Riga
RUSSIA
Kattegat
Riga
DENMARK
LATVIA
Baltic Proper
Copenhagen
Sound
LITHUANIA
Daugava
Belt Sea
Nemunas
Kaliningrad
RUSSIA
Vilnius
Minsk
Vistula
Oder
Berlin
BELARUS
Warsaw
GERMANY
POLAND
Kiev
UKRAINE
Prague
CZECH REPUBLIC
0 100 200 km
SLOVAK REPUBLIC

Environment of
the Baltic Sea Area
1994–1998

Contents

Foreword

The Helsinki Commission has been assessing the effects of the various pollutants on the Baltic Sea's natural resources for the past twenty years. The resulting assessment reports are unique compilations of scientific facts born of outstanding co-operation among the scientific community in the Baltic region. The present Fourth Periodic Assessment "Environment of the Baltic Sea area – Background Document" (4PA) covers the years 1994–1998.

The main findings have already been published in March 2001 at the 22nd Meeting of the Helsinki Commission under the title "Environment of the Baltic Sea area 1994–1998", Baltic Sea Environment Proceedings No. 82A. This is a twenty-page, easy-to-read booklet aimed at providing the general public with a brief review of the current state and developmental trends in the Baltic marine environment. In contrast, the present Background Document provides the full details.

Work on the 4PA started in 1998, when the HELCOM Environment Committee established the following aims for the report:

- to describe the state of the marine environment
- to evaluate if there are any changes or trends
- to establish and/or describe cause-effect relationships
- to explain to what extent changes/trends are caused by human activities or natural variations
- to evaluate the effectiveness of various measures

As with the earlier periodic assessments, the 4PA had to pursue these aims when summarising and examining in depth a large number of aspects of present state of the Baltic Sea marine environment. It also had to review and present the findings in perspective, taking due account of the most recent monitoring data and literature.

In addition, though, it was agreed to widen the potential target group in an attempt to provide a direct source of information intended for the general public. It was therefore decided that the 4PA should be written in a consistent and easily readable form that would be attractive to the general public, administrators and scientists, among other means through applying the so-called DPSIR-format, i.e. Driving forces, Pressures, State, Impact and Response.

A set of guidelines was prepared to aid the authors, conveners, the Project Group and the Project Manager during the working process. The guidelines were developed based on discussions held on several occasions: The Ninth Meeting of the Environment Committee, 24 September–2 October 1998, on the Island of Vilm, Germany, the First Meeting of the Project Group for the 4th Periodic Assessment, 17–18 April 1999, in Gothenburg, Sweden and the Fourth Meeting of the EC Monitoring Group, 19–23 April 1999, in Gothenburg, Sweden.

More than 180 persons from the scientific community around the Baltic Marine Area have been involved in drafting the 4PA. During the drafting period, the Project Group met on three occasions: 17–19 April 1999 in Gothenburg, Sweden, 24–25 January 2000 on the Isle of Vilm Germany, and 8–11 November 2000 in Helsinki, Finland. In addition, a separate workshop was held on 13–15 June 2000 in Copenhagen, Denmark to prepare the initial draft of the conclusions.

In order to ensure high scientific quality and consistency, a separate review process was established whereby the conveners undertook cross checks of the different chapters. In addition, the 10 first chapters were reviewed by Professor Dietwart Nehring, formerly at the Baltic Sea Research Institute, Warnemünde, Germany.

The finalization of the background document proved to be a far more time-consuming process than originally anticipated. As the original production timetable could not be met, this gave rise to a series of unavoidable delays as several key-persons in the process were no longer available at the required times. The background document is nevertheless expected to be extremely useful, not least because it integrates the whole range of environmental problems facing the Baltic marine Area.

In addition to the scientists listed in Chapter 13 a number of other persons have been involved in preparation of the 4PA. At the HELCOM Secretariat, Professional Secretary Mr. Kjell Grip, Sweden, superseded in 2001 by Professional Secretary Mr. Juha-Markku Leppänen, Finland, assisted both the Project Group and the Project Manager, Mr. Tonny Niilonen, Denmark. Language revision of the 4PA was carried out by David I. Barry, On Line Activities, Denmark, and Mr. Lars Møller Nielsen, Studio-8, Denmark carried out the graphic layout.

Tonny Niilonen
Project Manager for the 4PA
Danish EPA
December 2002

Introduction

1

By: Tonny Niilonen

The Baltic Marine Area is one of the world's most extraordinary seas, not only because of its variability and beauty, but also because of the unusual natural history of the area and its associated catchment.

Over 100 million years ago, the present Baltic Marine Area was a warm saltwater sea characterized by vast coral reefs. Fossilized coral is thus found in many places in Denmark, Germany and Sweden, as evidenced at the spectacular white cliffs at Møn, Rügen and Gotland. Since the Ice Age, the Baltic Marine Area has undergone remarkable shifts between freshwater and saltwater periods (HELCOM, 1981). These periods often bear the names of mussels or snails characteristic of the period in question, e.g. the Baltic Ice Lake (freshwater), the Yoldia Sea (*Yoldia:* a saltwater mussel), the Ancylus Lake (*Ancylus:* a freshwater snail) and the Littorina Sea (*Littorina:* a saltwater snail). A few species still survive from the freshwater periods – so-called relicts that thrive in the parts of the Baltic Marine Area with the lowest salinity.

It was the last Ice Age approximately 10,000 years ago that formed the Baltic Marine Area as we know it (Jansson, 1980), i.e. embraced by land on all sides with only a narrow outlet to the North Sea-Skagerrak via the straits comprised of the Belt Sea and the Sound. The southern coasts are generally low and sandy, consisting of material from the last Ice Age. The remaining coasts are rocky, fringed with a band of rocky islands and skerries form-ing impressive archipelagos, especially in the Gulf of Bothnia and the Gulf of Finland. The ice sheet rounded the topography of the catchment creating numerous lakes and depositing glacial deposits, moraine, clay, sand and stones when it retreated (HELCOM, 1998).

The Baltic Marine Area has been used for marine transport and fishing since early times, and is now also used for many other purposes, e.g. tourism, leisure and recreation, and as a recipient for all kinds of wastewater.

Over the course of time the effects of pollution have become all too evident in the Baltic Marine Area, including eutrophication and the impact of oil spills and persistent hazardous substances on the fauna.

By way of introduction to the Baltic Marine Area and associated catchment, its physical geography and demography will be briefly described below.

1.1. Physical geography

The Baltic Marine Area encompasses 415,266 km^2, whilst the associated catchment is about four times as large. Baltic oceanographers traditionally divide the Baltic Marine Area into five main subareas (Table 1.1): The Baltic Proper, the Gulf of Bothnia, the Gulf of Finland, the Gulf of Riga and the Belt Sea-Kattegat. For some purposes the Baltic Marine Area can be divided into the following nine areas: the Bothnian Bay, the Bothnian Sea, the Archipelago Sea, the Gulf of Finland, the Gulf of Riga, the Baltic Proper, the Western Baltic, the Sound and the Kattegat.

From the straits comprised of the Belt Sea and the Sound, the Baltic Marine Area stretches to the east and north by way of a series of basins separated by submarine sills that restrict horizontal water exchange. The average depth is only a little over 50 m, with the maximum depth being 459 m in the Landsort Deep in the Baltic Proper (HELCOM, 1981b).

The vertical water exchange is hampered by permanent density stratification of the water column due to differences in salinity and temperature. At times, strong stratification and restricted horizontal water exchange result in areas with stagnant deep water in the Baltic Proper, which has the

Sub-area	Surface salinity range PSU	Sea area km^2	Sea volume km^3	Maximum depth m	Average depth m	Freshwater input km^3/yr
Baltic Proper		211,069	13,045	459	62.1	100
Gulf of Bothnia		115,516	6,389	230	60.2	193
Gulf of Finland		29,600	1,100	123	38.0	100-125
Gulf of Riga		16,330	424	>60	26.0	18-56
Belt Sea-Kattegat		42,408	802	109	18.9	37
Baltic Marine Area		415,266	21,721	459	52.3	

Table 1.1 Characteristic data on the Baltic Marine Area and its five main subareas (compiled from: HELCOM, 1996).

Sub-region/ Country	Bothnian Bay	Bothnian Sea	Archipelago Sea	Gulf of Finland	Gulf of Riga	Baltic Proper	Western Baltic	The Sound	Kattegat	Total
STATES THAT HAVE RATIFIED THE CONVENTION:										
Finland	146,000	39,300	9,000	107,000						301,300
Russia				276,100	23,700	15,000				314,800
Estonia				26,400	17,600	1,100				45,100
Latvia				3,400	50,100	11,100				64,600
Lithuania					11,140	54,160				65,300
Poland						311,900				311,900
Germany						18,200	10,400			28,600
Denmark						1,200	12,340	1,740	15,830	31,110
Sweden	113,620	176,610				83,225		2,885	63,700	440,040
Total	259,620	215,910	9,000	412,900	102,540	495,885	22,740	4,625	79,530	1,602,750
STATES THAT HAVE NOT RATIFIED THE CONVENTION:										
Belarus					25,800	58,050				83,850
Ukraine						11,170				11,170
Czech Rep.						7,190				7,190
Slovakia						1,950				1,950
Norway	1,055	4,855							7,450	13,360
TOTAL CATCHMENTS AREA										
Total	260,675	220,765	9,000	412,900	128,340	574,245	22,740	4,625	86,980	1,720,270

Table 1.2
Division of the Baltic Marine Area catchment (km^2) among Baltic Sea States for each of the nine subregions (km^2) (Source: HELCOM, 1998).

greatest volume of the five sub-areas (Table 1.1).

The catchment of the Baltic Marine Area covers 1,720,270 km^2. Of this, nearly 93% lies within the borders of the Baltic Sea States that have ratified the convention, while the remaining 7% lies within the borders of states that have not ratified it (HELCOM, 1999). Division of the catchment between the various Baltic Sea States is shown for each of the nine subregions of the Baltic Marine Area in Table 1.2.

The catchments of the Baltic Proper and the Gulf of Finland are the largest, covering approx. 580,000 km^2 and approx. 410,000 km^2, respectively, while those of the Archipelago Sea and the Sound are the smallest. The country with the largest part of the Baltic Marine Area catchment is Sweden (440,000 km^2), followed by Russia, Poland, and Finland (approx. 300,000 km^2), while Germany has the smallest part of the catchment (28,600 km^2). 117,520 km^2 of the total Baltic Marine Area catchment is situated outside the Baltic Sea States that have ratified the convention.

The total long-term mean run-off from all Baltic rivers is 15,190 m^3/s (479 km^3/yr), of which nearly 50% drains into the Baltic Marine Area via the seven largest rivers, namely the Neva, the Vistula, the Nemunas, the Daugava, the Oder, the Göta Älv and the Kemijoki. Their long-term mean flow and the division of the river catchments among the Baltic Sea States is summarized in Table 1.3.

Much of the pollution entering the Baltic Marine Area enters as riverine inputs, although atmospheric deposition is also important (see Chapter 4). Part of the pollutional load to the Baltic Marine Area originates from other countries, for example where part of the catchment area is situated upstream of the state, or where a river reach forms a border between two countries. The riverine pollutional load from the states that have not ratified the convention is relatively low except in the case of the river Nemunas, where 46% of the catchment is situated in Belarus.

Table 1.4
Population (millions) in the Baltic Marine Area catchment (Modified from: Partanen-Hertell et al., 1999).

Country	Population
Poland	38.1
Russian Fed.	10.2
Sweden	8.5
Finland	5.0
Denmark	4.5
Belarus	4.0
Lithuania	3.7
Germany	3.1
Latvia	2.7
Ukraine	1.8
Czech Rep.	1.6
Estonia	1.4
Slovak Rep.	0.2
Norway	0.0
Total	**85.0**

Table 1.3
Long-term mean flow and length of the seven largest rivers flowing into the Baltic Marine Area together with division of their catchments among the Baltic Sea States (Source: HELCOM, 1998).

River	Neva	Vistula	Nemunas	Daugava	Oder	Göta Älv	Kemijoki	Total
LONG-TERM MEAN FLOW (m^3/s)								
Period	*1859-1988*	*1951-1990*	*1811-1995*	*1895-1995*	*1951-1990*	*1961-1990*	*1961-1990*	
	2,488	1,081	664	633	574	572	553	6,565
LENGTH (km)								
	74(1)	1,047	937	1,020	854	90(2)	600	
STATES THAT HAVE RATIFIED THE CONVENTION – CATCHMENT AREA (km^2)								
Finland	56,200						49,470	105,670
Russia	215,600		3,170	27,000			1,660	247,430
Estonia				(4)				
Latvia			90	23,700				23,790
Lithuania			46,700	170				46,870
Poland		168,700	2,510		106060			277,270
Germany					5590			5,590
Denmark								
Sweden						42,780		42,780
STATES THAT HAVE NOT RATIFIED THE CONVENTION – CATCHMENT AREA (km^2)								
Belarus		12,600	45,450	3,330				91,350
Ukraine		11,170						11,170
Czech Rep.					7190			7,190
Slovakia		1,950						1,950
Norway						7,450		7,450
TOTAL CATCHMENT AREA (km^2)								
Total	271,800	194,420(3)	97,920	84,170	118,840	50,230	51,130	868,510

(1) Length of the Neva to Lake Ladoga
(2) Length of the Göta Älv to Lake Vänern
(3) Without delta
(4) Errata in BSEP 70 have been corrected

1.2. Demography

The population inhabiting the Baltic Marine Area catchment numbers nearly 85 million people (Table 1.4). Population density varies from over 500 inhabitants/km² in the urban areas of Poland, Germany and Denmark, to less than 10 inhabitants/km² in the northern areas of Finland, Sweden and Norway (Partanen-Hertell *et al.*, 1999).

Around 22 million people (26%) inhabit metropolitan areas, whilst 45% inhabit towns or small cities and 29% inhabit rural areas. Nearly 15 million people live within 10 km of the coast (Table 1.5 and Figure 1.1).

A remarkably large part (60–70%) of the catchment in Germany, Denmark and Poland consists of arable land (Figure 1.2). The percentage of arable land in Estonia, Latvia and Lithuania is 30–50%, while only about 10% of the catchment in Sweden, Finland and Russia is arable land (HELCOM, 1998). Forests, wetlands and lakes comprise 65–90% of the catchment in Finland, Russia, Sweden and Estonia, 30–50% in Poland, Lithuania and Latvia, and only 19–25% in Denmark and Germany.

Almost half of the catchment is forest (48%), while arable land accounts for 20%. The remainder is accounted for by non-productive open land (11%), pasture (6%), lakes (6%) and urban areas (1%) (see Figure 1.3).

The Gross Domestic Product (GDP) mainly derives from services (54%), industry (37%) and agriculture (9%) (Partanen-Hertell *et al.*, 1999).

Table 1.5
Population distribution in the Baltic Marine Area catchment (%).

Catchment:	Metro-politan	Small city /Town	Rural	Total
Bothnian Bay	0	1.25	0.4	1.65
Bothnian Sea	0	2.32	0.7	3.03
Gulf of Finland	7.09	5.48	2.35	14.92
Gulf of Riga	1.06	2.58	1.82	5.46
Baltic Proper	13.87	28.53	21.86	64.26
Kattegat	0.83	2.63	0.88	4.34
Belt Sea	3.12	2.03	1.18	6.34
Total	**25.98**	**44.83**	**29.19**	**100.00**

Metropolitan = Population >250,000
Small city/Town = Population 200–250,000
Rural = Population <200

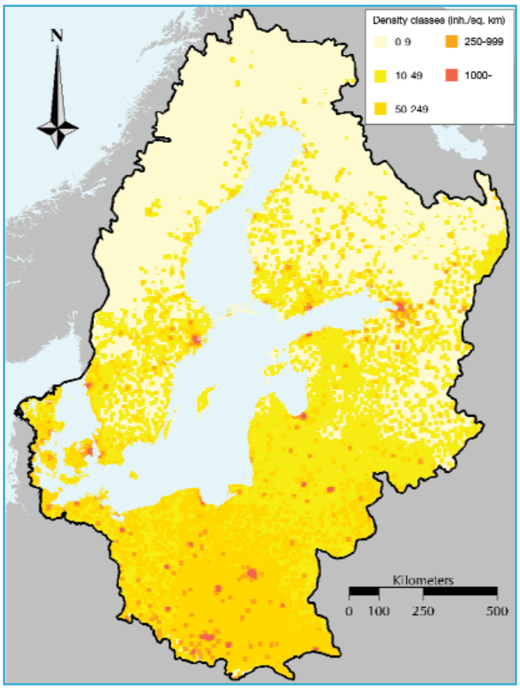


Figure 1.1
Population density classes in the catchment of the Baltic Marine Area.

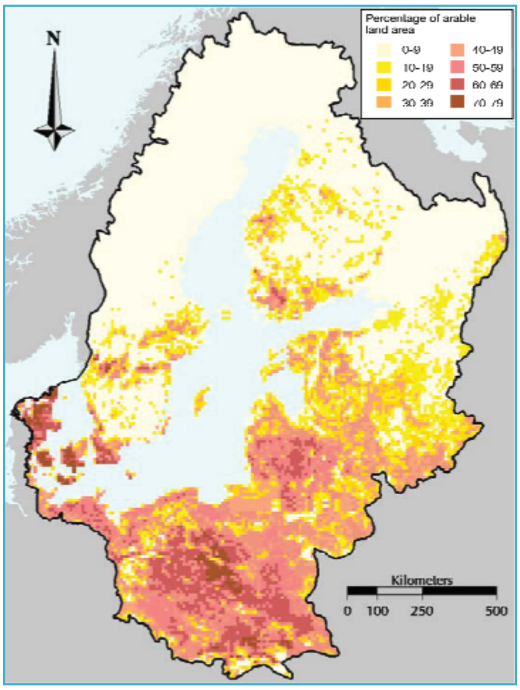


Figure 1.2
Percentage of arable land in the catchment of the Baltic Marine Area.

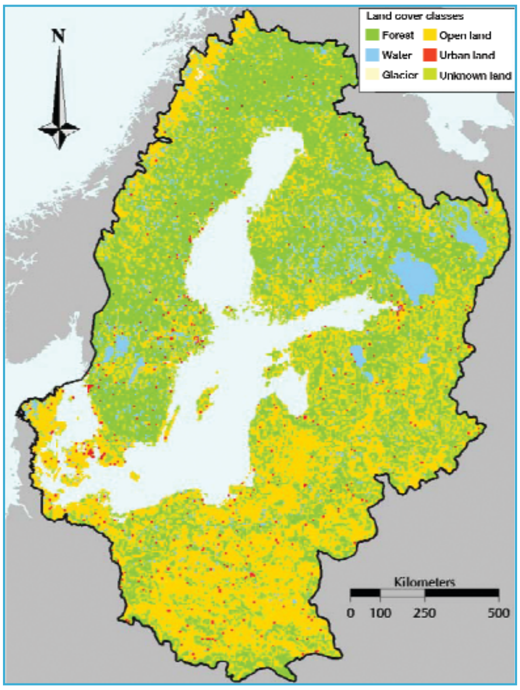


Figure 1.3
Land cover classes in the catchment of the Baltic Marine Area.

The main industries are metal working, machinery, vehicles, chemicals, textiles, food products and electronics. Agriculture accounts for a relatively large part of the economy in all the Baltic countries, while forestry is a major economic sector in Finland and Sweden.

Transport activity is currently increasing, and increased road traffic has caused environmental problems throughout the area. Marine transport has also expanded substantially during the 1990s (Magnusson, 1998).

Uses of the Baltic Marine Area

2

2.0. Scope and content

This chapter provides an overview of the human uses of the coastal and marine areas that it is necessary to consider when assessing the environmental quality of the Baltic Marine Area.

The Baltic Sea States exploit the coastal zone and the open sea for a wide range of uses, including industry, energy production, shipping, commercial fishing, extraction of oil, sand and gravel, dumping of dredged material, and the laying of submarine cables and pipelines. Moreover, human activities within the catchment itself, e.g. exploitation of raw materials, agriculture, transport and trade, have a major impact on the quality of the Baltic Marine environment. Man's modifications of the coastal areas are accompanied by alteration and loss of habitats and disturbed ecological balance. Pollution from former mining operations and marine dumping still exists.

For those human activities considered to have significant negative environmental impacts, HELCOM has taken a number of regulatory steps in the form of recommendations to be implemented by each Baltic Sea State via its national legislation. The intention of this chapter is not to scrutinize the implementation of these recommendations by the HELCOM countries, but to identify those activities that may affect the status of the Baltic Marine Area.

The territorial waters subject to coastal state jurisdiction generally extend 12 nautical miles from the baselines demarcating the inshore boundary of territorial waters. In accordance with the United Nations Convention on the Law of the Sea (UNCLOS), moreover, coastal states enjoy sovereign rights to the exploitation of natural resources in their exclusive economic zones (EEZ) extending beyond their territorial waters. Within its EEZ, each state is obliged to conduct fishery management and take precautionary measures to prevent, reduce and control pollution of the marine environment. In the enclosed Baltic Marine Area, the EEZs are delimited by intergovernmental agreements. Not all Baltic Sea States have yet settled their EEZs, however.

2.1. Tourism and recreation

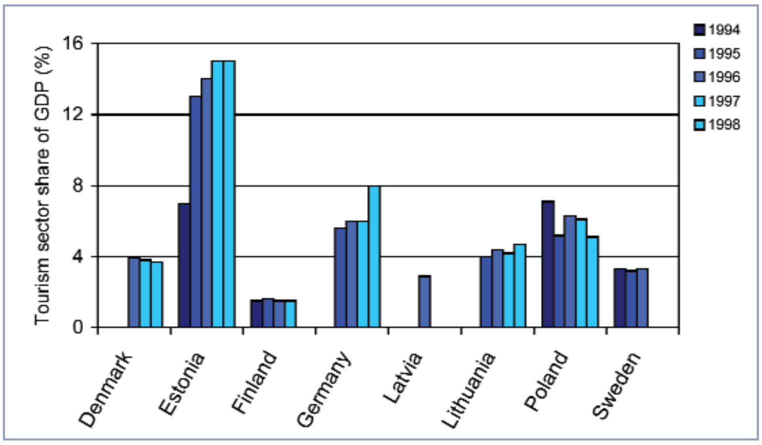
By: Gunnar Zettersten

By:
Gonvenor: Sverker Evans

Cooperation on tourism in the Baltic Marine Area started in the early 1980s and is one of the main sectors included in the Baltic 21 Programme (http:/www.ee/Baltic21). The tourism sector has the potential to contribute to sustainable development in the coastal zones by supporting the economy in regions where traditional activities are in decline, as well as to initiate good management practices in areas which could otherwise have a negative environmental impact. Important branches of the tourism sector are accommodation and catering, tour operators and travel agencies, national, regional and local tourism authorities, guides, educational tourism, nature tourism and cultural tourism. The percentage of GDP accounted for by the tourism sector in some Baltic Sea States during the assessment period is shown in Figure 2.1. World Tourism Organization forecasts up to the year 2020 indicate high growth of tourism in the Baltic Marine Area compared with other parts of Europe.

Tourism in the coastal areas peaks in the summer months. The establishment of bird and seal sanctuaries, EC Natura 2000 sites and Baltic Sea Protected Areas (BSPAs) contribute to the protection of wildlife during the sensitive breeding period. In addition, all countries have national legislation for the protection of a coastal strip outside urban areas. For further details see Chapter 10.

Figure 2.1
Percentage of GDP accounted for by the tourism sector in some Baltic Sea States.
Source: Baltic 21 and national data.
Note: The data for the different countries is not directly comparable as they define the tourism sector differently.



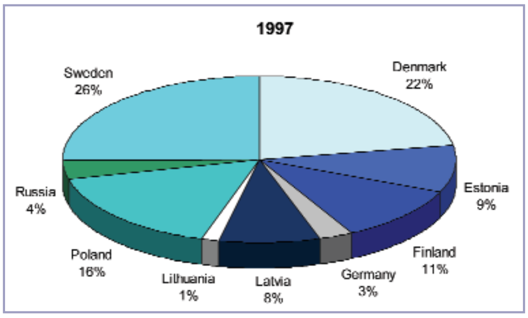


Figure 2.2
Nominal catch (tonnes) of herring, sprat, cod and flatfish in divisions III b, c, d and salmon in division III d in 1997 apportioned by Baltic Sea State. Source: ICES Cooperative Research Report No 236. Report to the ICES Advisory Committee on Fishery Management, 1999.

2.2. Fishing and mariculture

By: Sverker Evans

2.2.1. Fisheries

The Baltic Marine Area is an important fishing area and commercial fishery capacity has increased since the Second World War. In 1997 the total yield was 1,068,000 tonnes, corresponding to approx. 1% of the total world marine and freshwater catch (http://www.fao.org). The commercially most important fish species are herring, sprat, cod and salmon. Several other fish species, as well as Norway lobster (Nephrops norvegicus), blue mussel and oyster are important in the Kattegat and Belt Sea areas. Further details of fish stocks and landings are provided in Chapter 9.

The exploitation of fish resources in the Baltic Marine Area is regulated by the International Baltic Sea Fisheries Commission (IBSFC) within the Gdansk Convention in cooperation with the EU. Total Allowable Catch (TAC) and other management measures taken in various parts of the Baltic Marine Area are based on scientific advice provided by the International Council for the Exploration of the Sea (ICES). The TAC is divided among the countries as interchangeable national quota. Apportionment of the 1997 nominal catch of herring, sprat, cod and flatfish by country is shown in Figure 2.2. The fish catch statistics are relatively inaccurate, however, due to lack of data and incorrect reporting.

Herring and sprat landed in the countries on the eastern Baltic coasts are used mainly for human consumption while those landed in the countries on the western coasts are used mainly for the production of fishmeal. Denmark, Sweden and Finland conduct industrial fishery in the Convention area. In the period 1994–1998, 3.8 million tonnes of herring and sprat were landed by Baltic Sea States. Of this, 2.0 million tonnes (just over 50%) were used for industrial purposes (Figure 2.3).

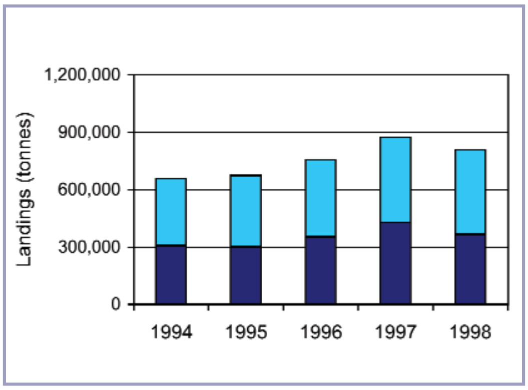


Figure 2.3
Total landings of Baltic herring and sprat in 1994–1998 apportioned by use (human consumption and industrial purposes). Source: ICES Cooperative Research Report No 236. Report to the ICES Advisory Committee on Fishery Management, 1999, and national information.

In spite of IBSFC and EU regulations aimed at controlling the fishery sector, commercial fishery has led to overfishing, depletion of resources, conflicts and ultimately to socioeconomic difficulties for fishing communities. Apart from effects on the target species, other aspects of the fishery have largely undocumented ecological effects on the marine ecosystem, e.g. selective fishing, bycatch, bottom trawling and the discarding of fish.

2.2.2. Mariculture

Among the Baltic Sea States, mariculture is undertaken in Denmark, Finland and Sweden. The main species farmed are salmon (Salmo salar) and rainbow trout (Onchorynchus mykiss). Most of the former salmon rivers in the Baltic Marine Area have been ruined by hydroelectric plants, habitat destruction and pollution. In order to compensate for these losses, smolt are released from hatcheries, mainly in the Gulf of Bothnia. A total of 6.4 million hatchery-reared smolt were released in rivers and at coastal release sites in 1998. In comparison, the estimated wild smolt production during the same year was about 0.5 million (cf. Chapter 10).

In 1998, mariculture production in the Baltic Sea States totalled approx. 22,400 tonnes (Table 2.1). The largest producer was Finland, where fish farming developed rapidly until the early 1990s. Thereafter the increase in production was curtailed by increased import of farmed salmon and rainbow trout, mainly from Norway. Blue mussel is cultured in Sweden, and blue mussel and oysters in Denmark. As mariculture still has growth potential, conflicts between the sector and other users of the coastal zone are likely to increase.

2.3. Shipping

By: Sverker Evans

Seaborne transport of goods is extensive in the Baltic Marine Area (Figure 2.4), with over 500 million tonnes of cargo being transported annually (Buch and Dahlin, 2000). In addition, there are approx. 50 ferry crossings on fixed routes between Baltic ports. For safety reasons, traffic separation schemes control the traffic in certain sea areas.

A high proportion of the larger ships consists of ferries and tankers. Bulk coal carriers of around

	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998
Denmark			5958	6798	7852	6793	7348	7802	5852	7089
Finland	13459	13181	15198	14673	13698	13319	13923	14757	13007	13269
Sweden	4336	4128	2978	2542	2269	2385	2941	2867	2166	2026
Total	17795	17309	24134	24013	23819	22497	24212	25426	21025	22384

Table 2.1
Mariculture production (tonnes) in the Baltic Marine Area in 1989–1998.
Source: National data.

150,000 dwt also operate in the area. In addition, hundreds of small cargo vessels operate throughout the year and there are large numbers of fishing vessels and pleasure craft near the ports and in the narrow passages during the summer months. Excluding small pleasure craft, some 2,000 vessels are at sea at any one time (based on port statistics, ferry information and Alexandersson, 1980). Moreover, 12 loaded chemical tankers with hazardous cargoes are estimated to be en route at any one time (HELCOM, 1990). Ships carrying packaged dangerous goods make about 4,000 voyages per month over short distances (HELCOM, 1993).

Shipping, and in particular the ballast water transported by ships, is widely recognized as a major vector for the unintentional spread of aquatic organisms and pathogens between water bodies in different parts of the world. Such introductions are a matter of serious concern as regards the nature protection and management in the Baltic Marine Area. Ballast water is carried by nearly all types of ship and a substantial pool of introduced organisms is already present in the area. Although harmful effects in most cases are non-existent or negligible, the further introduction and spread of these organisms should be minimised as a precautionary measure (cf. Chapter 10).

Oil spills at sea are experienced by the general public as one of the most conspicuous pollution problems. Approximately 13,000–14,000 harbour calls are made by oil tankers each year. Of these, about 65% are made by oil tankers with a cargo capacity of less than 10,000 m^3 (SSPA, 1998). One third of the oil transport is carried out by ships with a single hull, while 42% is carried out by ships with a double hull and 22% by ships with a double bottom (SSPA, 1996).

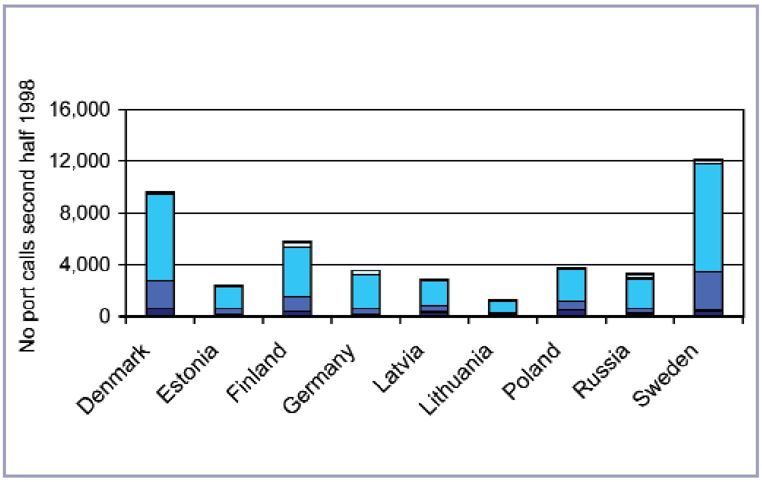
Some countries are planning to increase their oil transport at sea. The main route for oil transport is through the Kattegat and the Great Belt, via the southeastern part of the Baltic Marine Area and into the Gulf of Finland. In 1995, an estimated 77.4 million tonnes of oil was transported in the Baltic Marine Area, involving a total of more than 7,000 journeys (Baltic Marine Outlook 2000). Ventspils in Latvia is the largest oil harbour in the region. The volume of oil handled annually at the top eight Baltic oil ports in 1997–1998 is given in Figure 2.5.

Plans for new and enlarged oil terminals in Russia, Estonia, Latvia, Lithuania, Poland and Germany will increase oil handling capacity to about 137 million tonnes per year, as compared with 53 million tonnes per year at present (1995) (SSPA, 1996). If all the plans for new and enlarged oil terminals are fulfilled and the increased capacity is utilized, oil transport in the Baltic Marine Area will increase to about 177 million tonnes per year (SSPA, 1996).

Due to its status as an enclosed marginal sea, the Baltic Marine Area has been designated as a Special Area with regard to oil and waste. In Special Areas, stricter antipollution regulations apply than in other sea areas. In addition, the Baltic Sea States have agreed on a common strategy to prevent discharges of oil and other waste from shipping – the Baltic Strategy. This applies to fishing boats, working vessels and pleasure craft, as well as to various types of merchant vessel such as oil and chemicals tankers, dry-bulk ships and roll-on/roll-off freight and passenger ferries. By 1 July 2000, the Baltic Sea States have undertaken to ensure the provision of adequate reception facilities in ports and terminals. At present, however, some countries in economic transition do not yet have the technical or financial possibilities to fulfil the IMO and HELCOM requirements.

As financial incentives are considered to be the best way to promote environment-friendly development within the shipping sector, the Baltic Strategy calls for the harmonization of reception facility procedures and fees. The costs involved in the use of reception facilities should be included in the harbour dues, regardless of whether or not ships make use of the facilities. The “no special fee” system is now applied in Denmark, Estonia, Finland, Latvia, Russia and Sweden.

Figure 2.4
Number of port calls in Baltic Sea States in the second half of 1998 apportioned by vessel type.
Note: One port call = ship arrival and departure.
Source: Maritime Outlook 2000, Table 6, p 48.



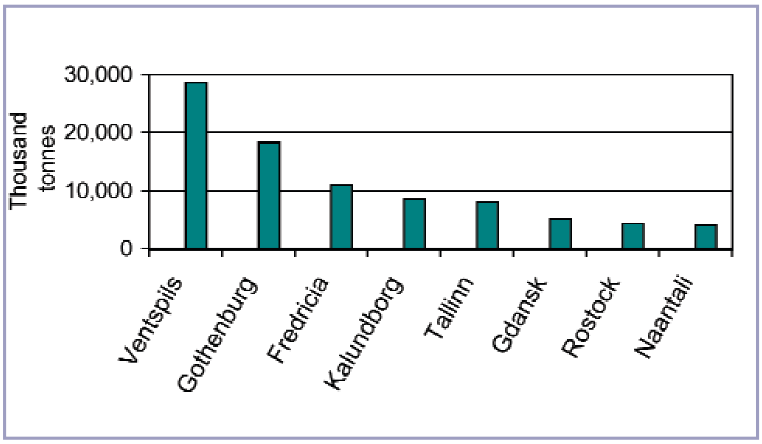


Figure 2.5
Volume of oil handled annually at the top eight Baltic oil ports in 1997–1998.
Source: Maritime Outlook 2000, Table 12, p. 40.

In order to stimulate the use of safer tankers to carry oil, Denmark, Finland, Germany and Sweden have introduced reduced navigational fees for double-hull construction tankers in their territorial waters. Despite the fact that the discharge of oil at sea is prohibited, violations frequently occur. The annual results of HELCOM´s Aerial Surveillance Programme, which is carried out by the national coast guards, do not reveal any significant decline in the number of observed oil slicks at sea (cf. Chapter 4).

2.4. Offshore activities and installations

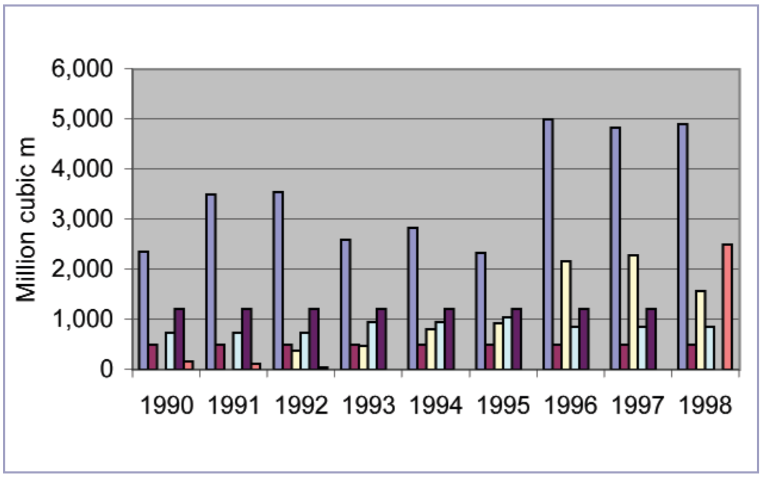
By: Eugeniusz Andrulewicz

2.4.1. Oil and gas, cables, pipelines

Oil and gas

Discharges from offshore oil production may include emissions of drilling mud and production water, dumping of drill cuttings, and incidental oil spills. Drilling muds consist of mixtures of water, clay and various chemicals and are used to lubricate the drill and transport the drilled rock fragments (cuttings) up to the surface. Production water consists of water from the reservoir and water injected to transport the crude oil to the surface.

Figure 2.6
Extraction of marine aggregates in the Baltic Marine Area in 1990-1998.
Source: J. Krause.



Commercial offshore oil exploitation in the Baltic Marine Area is taking place off the Polish coast. Since1994, two production platforms have been operating within the same crude oil field. In the period 1992–1998, approx. 700,000 tonnes of oil were extracted (Piliczweski, 1999). The estimated oil and gas reserves in the Polish sector total approx. 20 million tonnes and 10 billion cubic meters, respectively. Rock cuttings and drilling muds are collected and transported ashore, while clean coarse cuttings are disposed of at sea.

Additional oil reserves might possibly also exist in the southeastern region of the Baltic Proper (Poland, Russian Kaliningrad region, Lithuania and Latvia), in which case a considerable future increase in offshore oil and gas operations can be expected in those areas.

Cables

A number of telecommunications cables and high-voltage power cables are submerged in the Baltic Marine Area sediment. Direct current (DC) connections with sea cables for the transmission of electrical energy have been used in the area since the 1950s. The traditional technique for DC connections uses a cable for the power transmission, with the return current being carried through the seawater via electrodes placed in the sea. At present, there are seven DC high-voltage power cables in the Baltic Marine Area. The latest power links to become operational are the Baltic Cable between Sweden and Germany (250 km, 450 kV, 600 MW), which is the world's longest power transmission cable, and the new power link between Sweden and Poland, the SwePol Link, which entered into operation in 2000.

Pipelines

No offshore pipelines are currently in operation in the Baltic Marine Area. However, plans exist to lay a gas transmission pipeline from Russia through the Gulf of Finland and the Baltic Proper to Germany, as well as one between Germany, Denmark and Sweden.

2.4.2. Sand and gravel extraction

By: Jochen Chr. Krause

In the Baltic Marine Area, sand and gravel deposits are mainly of glacial origin or the results of slow postglacial erosion processes, and hence are considered as a finite resource. Marine aggregates are used for construction, infill and beach nourishment purposes (HELCOM, 1999). Since the 1990s, Denmark, Germany, Finland and the St Petersburg region of Russia have extracted increasing amounts of marine aggregates (Figure 2.6). Lithuania, Latvia, Estonia and the Kaliningrad region of Russia do not undertake any substantial extraction at present, but the situation could change once their economies start to grow. Sweden has not undertaken any excavation since 1992, with the exception of 1998, when sand was extracted in connection with the construction of the Øresund Fixed Link.

2.4.3. Offshore wind energy

By: Ole Norden Andersen

Only a few offshore wind farms have so far been established in the Baltic Marine Area. However, there are plans for additional expansion of offshore wind farms because the exploitation of wind power reduces consumption of fossil fuels and hence also emissions of CO_2, SO_2 and NO_X. A 2 MW turbine producing 5 GWh annually can substitute for 1,000 tonnes of coal and prevent the emission of up to 5,000 tonnes of CO_2, 6 tonnes of SO_2, 2.5 tonnes of NO_x, and the generation of several hundred tonnes of cinders, ashes and dust. However, offshore wind farms may conflict with a number of human interests such as the commercial fishery, shipping, aviation, military activities, extraction of minerals, oil and gas, archaeology, and recreation and nature conservation considerations. The visual impact of wind turbines on the landscape is obvious. Moreover, the turbine foundations occupy space and influence hydrographical conditions in the vicinity. Other disturbances are the generation of noise and vibrations above and below the water surface, as well as the formation of electromagnetic fields around the power cables.

Knowledge about the negative effects of wind farms on the marine and terrestrial biota is sparse. Such effects are presently being studied in Danish and Swedish waters.

2.5. Urbanization, including construction works

By: Sverker Evans

From a global point of view, the urban agglomerations in the Baltic catchment area are comparatively small. Apart from St Petersburg, with five million inhabitants, there are only four cities with more than one million inhabitants – Copenhagen, Minsk, Stockholm and Warsaw. Of the 85 million people who inhabit the catchment area, only a relatively small proportion (about 15 million) live close to the coast (cf. Chapter 1).

The coastal human populations nevertheless affect the water quality through the use of fresh water and subsequent discharge of wastewater into the sea. Direct municipal discharges of wastewater by the Baltic Sea States in 1995 are shown apportioned by source in Figure 2.7. The annual discharge from the 15 million inhabitants connected to a total of 430 municipalities discharging directly into the Baltic Marine Area totalled 3,490 million m^3. Approximately 13 million inhabitants were connected to 155 municipal wastewater treatment plants (MWWTPs) with a capacity exceeding 10,000 PE (Person Equivalents), producing nearly 2,950 million m^3 of treated wastewater annually. Nearly 500 million m^3 (14%) of this was discharged untreated, mainly into the Gulf of Finland from St. Petersburg and from the Leningrad region of Russia (430 million m^3/yr). The remaining 70 million m^3/yr derived from Poland, Latvia, Lithuania and Estonia and was discharged into the Baltic Proper, the Gulf of Riga and the Gulf of Finland. An additional 177 million m^3/yr of wastewater, of which 151 million m^3/yr were untreated, were discharged from the Kaliningrad region of Russia, but is not included here as the reliability of the information is questionable.

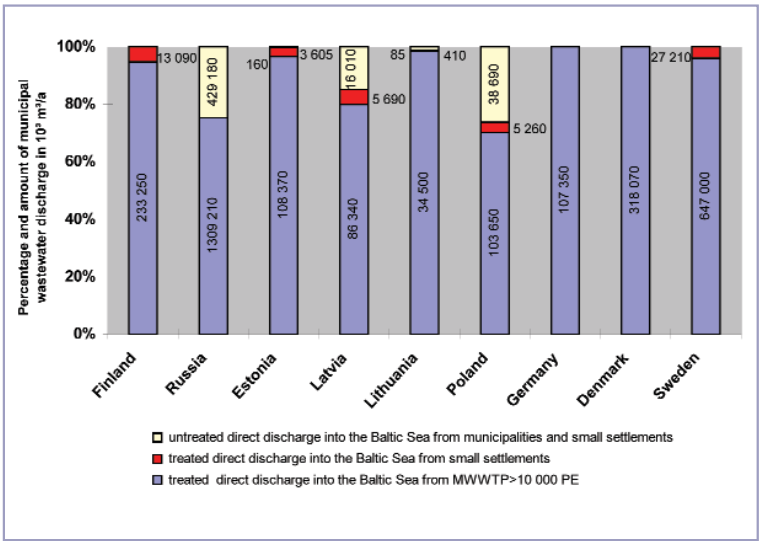


Figure 2.7
Direct municipal wastewater discharges into the Baltic Sea in 1995 apportioned by source.
Source: HELCOM 3rd Pollution Load Compilation, 1998.

A number of fixed transport links have been constructed in the 1990s. The Great Belt Fixed Link between the Danish islands Zealand and Funen opened in 1997–1998 and the Øresund Fixed Link between Denmark and Sweden opened in July 2000. Environmental studies were conducted in 1995–1999 as part of a feasibility study for a fixed link across Fehmer Belt between Denmark and Germany, but the decision to construct the link has not yet been made.

2.6. Disposal of dredged spoils

By: Katrin Grünewald

Dredging involves the removal of marine sediment in its natural condition, for example to keep waterways and harbours open for navigation (maintenance dredging) or in connection with the construction of new harbours or other coastal engineering projects (capital dredging). In addition,

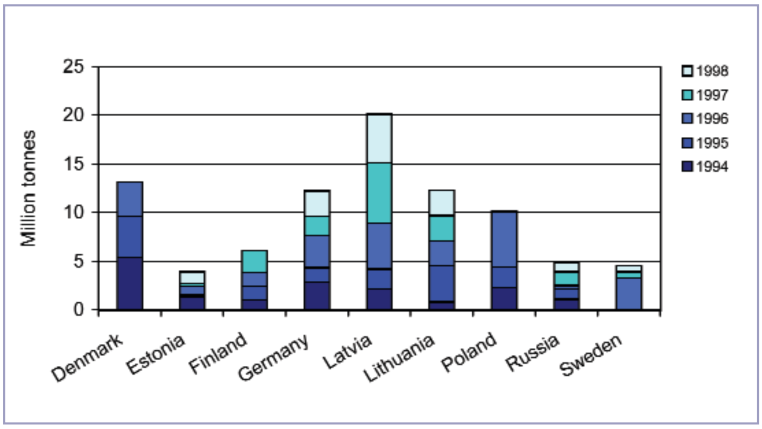


Figure 2.8
Dredged material (million tonnes) dumped in the Baltic Marine Area over the period 1994-1998 by the Baltic Sea States.
Source: K. Grunwald.

some dredged material can be used for beneficial purposes such as for beach nourishment. Due to land up-lift of approx. 9 mm/yr in the northern part of the Baltic Marine Area, dredging is frequently carried out along the coast of the Gulf of Bothnia.

During the period 1994–1998, 87.1 million tonnes of dredged material were dumped in the Baltic Marine Area. The majority of this derived from maintenance dredging (Figure 2.8). Comparison with the preceding period 1989–1993 is precluded due to a lack of information.

The quantities of heavy metals in the dredged material are considerable. Thus the material dumped contained 150 tonnes of copper, 350 tonnes of zinc and 500 tonnes of chromium. Part of the heavy metals content is natural in origin, however, and many operations simply relocate the material rather than constitute new input to the marine environment. The amounts of heavy metals in dumped materials dredged from harbours, estuaries and open sea areas vary considerably as a result of local geology, hydrography, sediment load and human influences.

As removal and disposal of dredged sediments may negatively affect the environment, the Baltic Sea States have been obliged since 1992 to exercise control over their dredging operations and in particular to minimize resuspension of contaminants and fine sediments.

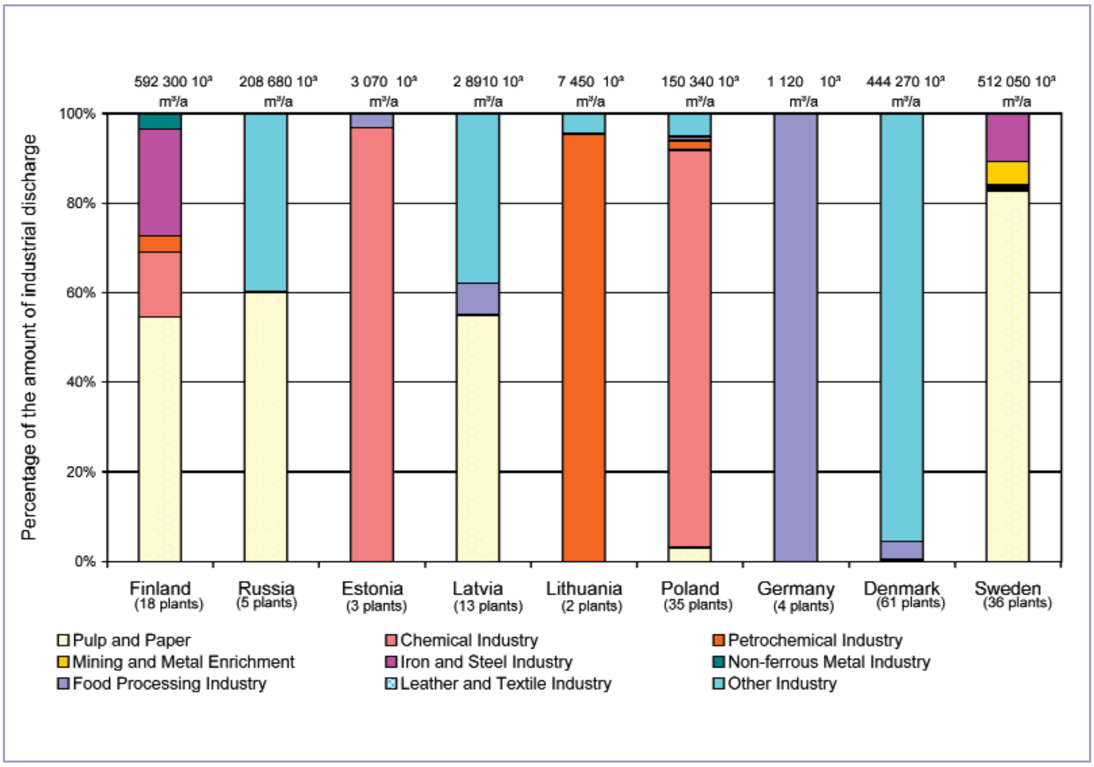
2.7. Coastal industry and industrial processes

By: Kaj Forsius

The Baltic Marine Area is influenced by industrial activities, both regionally and locally. Various types of industry are located along the Baltic coastline but these constitute only a very small part of all industries located in the Baltic catchment area. In 1995, direct wastewater discharges into the sea from 183 coastal industrial plants of various types totalled 1,950 million m^3 (HELCOM, 1998). Virtually all of this (99.9%) was treated. Direct discharges of treated wastewater to the Baltic Marine Area in 1995 apportioned by country are shown in Figure 2.9.

The largest share of the direct industrial wastewater discharge, 890 million m^3/yr, originated from 40 pulp and paper plants located in Finland, Russia, Latvia, Poland and Sweden, in particular from 24 Swedish plants and 9 Finnish plants. The second-largest share, 530 million m^3/yr, derived from various branches ("other industry" in Figure 2.9). Of this, 425 million m^3/yr derived from

Figure 2.9
Direct discharges of treated wastewater into the Baltic Marine Area by the Baltic Sea States in 1995 apportioned by industrial branch.
Source: HELCOM 3rd Pollution Load Compilation, 1998.



Oil refinery at Stigsnæs, Denmark. Photo: Biofoto/ Mr. Dieter Betz, 2000

25 Danish plants, and the remainder from plants in Latvia, Lithuania, Poland and Russia. Waste-water discharges from the chemicals industry amounted to 230 million m^3/yr, with Poland and Finland as the main sources. Wastewater from the iron and steel industry amounted to 195 million m^3/yr and was discharged into the Bothnian Bay and the Baltic Proper. One Finnish plant accounted for 140 million m^3/yr, while Sweden accounted for most of the remainder. The mining and metal extraction industry (one Swedish metal enrichment plant) discharged 27 million m^3/yr of waste-water into the Bothnian Bay while the nonferrous metals industry (two Finnish plants) discharged 21 million m^3/yr of waste-water into the Bothnian Bay. The petrochemical industry (eight plants) discharged 34 million m^3/yr, of which 21 million m^3/yr were discharged by two Finnish plants. All Baltic Sea States except Finland and Lithuania have food processing industries that discharge wastewater directly into the Baltic Marine Area. The 39 plants involved discharged 23 million m^3/yr of wastewater, of which 18 million m^3/yr was discharged from 27 Danish plants. The leather and textile industry discharged the least wastewater, 0.3 million m^3/yr; this derived from a single Russian plant discharging into the Gulf of Finland.

The Joint Comprehensive Environmental Action Programme (JCP) originally identified 132 so-called “Hot Spots” in the Baltic catchment area, of which 50 were industrial (http://www.helcom.fi). Of the latter, 37 were located in countries in economic transition. The highest priority industrial sectors identified were the pulp and paper industry, the chemicals industry and metals production and processing industry. These industries listed in the hot spots are mostly large enterprises in poor condition where privatization has been slow and competitiveness has remained low. Some of the industrial hot spots located in the countries in economic transition have closed down or have decreased production volume significantly. The reduction in discharges from the countries in economic transition is thus partly due to the decreased production volume and partly to industrial restructuring. All pulp and paper industry hot spots in Sweden and Finland have been deleted from the list as they have taken the recommended environmental measures and emissions have been reduced significantly.

2.8. Coastal defences

By: Ole Norden Andersen

Erosion occurs when more sediment is lost than is gained from a coastal area due to currents and waves. This can result in the loss of valuable recreational beach areas, and can constitute a serious threat to many infrastructure investments and interests located in or near coastal regions. Coastal erosion and subsequent flooding is of concern along several coasts in the Convention Area. Defence measures to prevent shoreline erosion or flooding events have been developed by those countries that suffer from coastal erosion.

In Latvia, regulations on the establishment of protected coastal belts pay special attention to coastal dynamics. Coastal protection deals primarily with restoration and strengthening of dunes by coating them with stones, clay or wickerwork or by planting willow (*Salix* sp.).

Table 2.2
Some results of beach cleaning operations carried out along the Polish coast in 1992-1999.
Source: Jóźwiak, T. 1994-1999. Coastwatch Observer, Polish report in Coastwatch Europé 1994-1999.

Year	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**
% of Polish coast surveyed	48	55	62	67	60	58	61	56
No. of surveyors	700	2000	3200	3800	4062	3181	3154	3579
NO. OF ITEMS FOUND PER 5 KM OF COAST								
Plastic bottles	86	120	174	230	213	320	170	180
Cans	62	59	78	72	90	115	100	137
Paper containers	56	56	72	61	95	95	60	60
Tires	3	4	3	5	4	6	6	2

About 22% of the Polish coast is presently protected by a number of defence structures such as storm flood embankments, sea walls, groynes and breakwaters, and by measures such as beach nourishment.

In Sweden, several areas along the southern, sandy coast of Sweden suffer from erosion. Over the past 150 years, 1–1.5 m of the land has disappeared each year in some affected areas. The most commonly used method of protection is to construct groynes close to settlements in order to stabilize the sandy sediment.

In Denmark, more than 33,000 ha or about 15% of the shallow marine areas with a water depth of less than 2 m have been dyked and drained over the past 200 years. These projects have resulted in the disappearance of 140 small islands and 14% of the original Danish coastline, which used to be about 8,000 km long. In some areas, the loss of shallow marine waters may have reduced total nitrogen retention capacity by up to 20%, thereby enabling a greater percentage of leached nitrogen to reach offshore areas. In Denmark, 38% of the coastline is protected by coastal defence measures. Since 1997, erosion protection is only permitted in urban areas in order to counteract immediate loss. Moreover, new coastal defence projects have to be aesthetically acceptable and more in compliance with nature. Renewals are kept at a minimum and old defences are demolished where possible.

2.9. Litter

Marine litter consists of solid waste from a large number of sources. Litter from ships, including fishing vessels, generally consists of normal household waste, waste from cargo holds, discarded fishing equipment, discarded medical and sanitary articles, etc. Plastic bags, bottles or cans are commonly left on beaches by tourists. Flooding of landfills located alongside rivers or on the coast may also contribute to waste contamination of the coastal environment. The presence of sanitary items is often attributable to the discharge of municipal sewage.

The use of plastics and other synthetic materials has increased markedly during the past 30 years, as is also reflected in the composition of the litter. Plastics are now the major component and may constitute more than 90% of the total waste volume.

Discharges of litter into the marine environment may cause appreciable environmental problems. Since 1992, yearly beach cleaning has been implemented along the Polish coast and about 50–100 m^3 of waste is removed each year (Jóźwiak, 1994–1999) (Table 2.2).

General meteorology, hydrology and hydrography

3

By:
Wolfgang Matthäus
Sten Bergström
Bengt Carlsson
L. Phil Graham
Lars Andersson

The environmental conditions of the Baltic Marine Area are strongly dependent on the meteorological forcing over the sea, the hydrological processes in the catchment and the hydrographic processes in the sea, as well as the interaction between them. These processes influence the water temperature and ice conditions, the regional inflow of fresh water from the rivers and the atmospheric deposition of pollutants on the sea surface. Moreover, they also govern water exchange with the North Sea and between the subbasins, as well as transport and mixing of water within the various subregions of the Baltic Marine Area.

The climatic, hydrological and hydrographic conditions of the Baltic Marine Area system have been subject to three previous assessments covering the periods 1980–85 (Launiainen *et al.*, 1987), 1984–88 (Astok *et al.*, 1990) and 1989–93 (Bergström and Matthäus, 1996). The conditions prevailing during the assessment period 1994–98 are summarized below.

3.1. Climate and database

The Baltic Marine Area is located within the westwind zone where cyclones coming from the west or southwest dominate the weather. Periodically, cyclones from a more southerly direction enter the region. The temperature climate of the region is largely coupled to the latitude of the main cyclonic tracks, although cloud cover also plays an important role, especially in winter.

In winter, most of the precipitation is frontal, especially inland. In summer, around half of the precipitation can be characterized as convective and is commonly greater inland than over the sea. 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, are almost exclusively connected to deep cyclones that form west of Scandinavia, and mainly occur from September to March.

The body of the Baltic Marine Area has a strong impact on the local climate within the region, in particular influencing the air temperature, precipitation, cloud cover, radiation and winds, and in coastal areas leading to pronounced gradients.

Most of the climate analysis used in the present assessment is based on observed data. Of particular importance is a gridded database for the entire catchment developed by the Swedish Meteorological and Hydrological Institute (SMHI). This database, which is based on synoptic observations, is the foundation for the analysis of air temperatures and precipitation. It has further been necessary to use a hydrological model to provide data

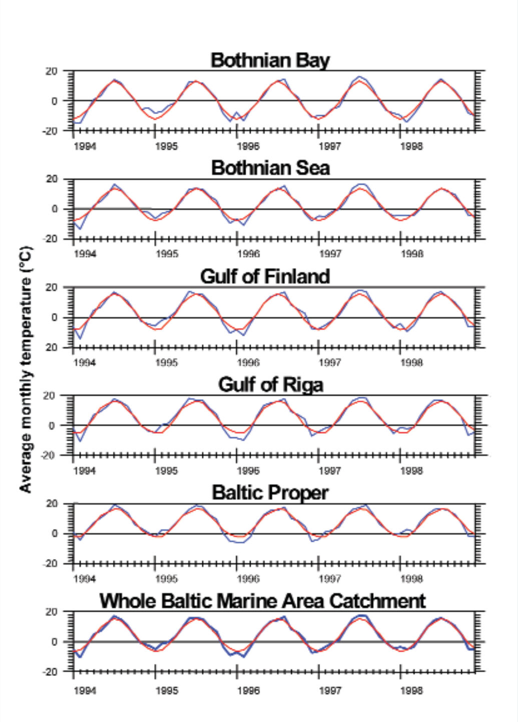


Figure 3.1
Mean monthly air temperatures in the Baltic Marine Area sub-catchments and whole catchment as compared with the long-term monthly mean for the reference period 1981–93.

	1981–93 (Mean)	1994	1995	1996	1997	1998	1994–98 (Mean)
Bothnian Bay	0.0	0.0	0.7	0.4	0.9	-0.4	0.3
Bothnian Sea	2.5	2.1	3.0	2.1	3.6	2.8	2.7
Gulf of Finland	3.4	3.1	4.7	3.4	3.9	3.4	3.7
Gulf of Riga	5.3	5.1	6.3	4.9	5.8	5.7	5.6
Baltic Proper	6.9	7.5	7.5	6.0	7.4	7.5	7.2
Belt Sea+Kattegat	7.1	7.6	7.3	6.0	7.6	7.2	7.1
Baltic Marine Area	4.3	4.4	5.0	3.9	4.9	4.5	4.6

Table 3.1
Areally averaged air temperature in the Baltic Marine Area sub-catchments and whole catchment (°C).

Figure 3.2
Monthly areally averaged precipitation in the Baltic Marine Area subcatchments and whole catchment as compared with the long-term monthly mean for the reference period 1981-93.

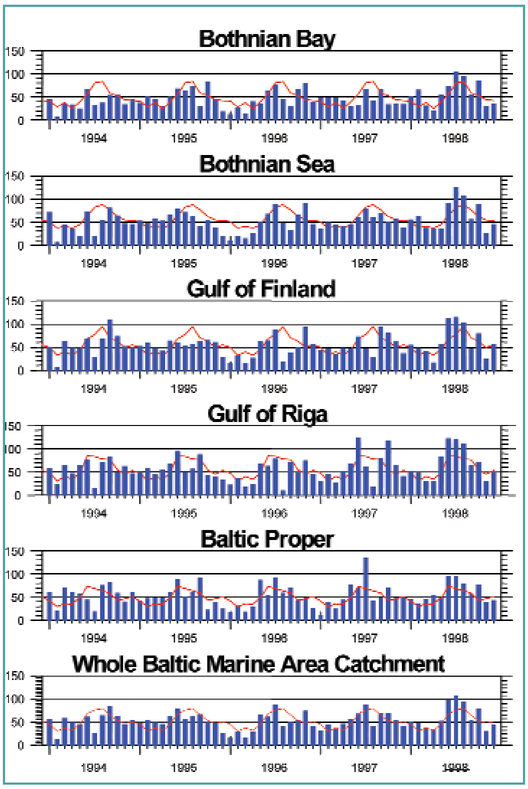
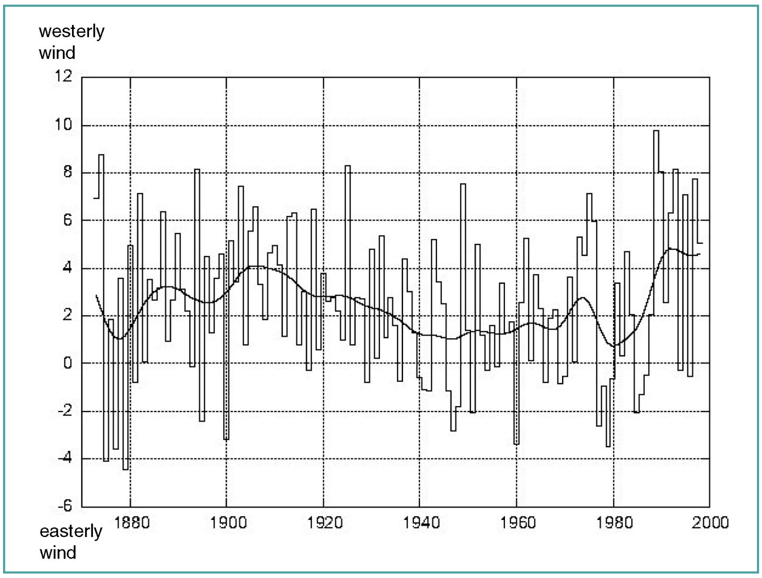


Figure 3.3
Average westerly geostrophic wind components (m/s) in winter (December-February) during the period 1871–98, based on triangulated observations of sea level pressure in Göteborg, Kalmar and Uppsala in Sweden. Prepared by H. Alexandersson, SMHI.



for parts of the analysis of riverine runoff. The availability of the hydrological HBV model for the entire Baltic Marine Area catchment (Graham, 1999) made it possible to fill in one missing year of runoff (1998) for the river Neva and to estimate snow storage for each of the subcatchments.

3.1.1. Air temperature

Average air temperature in the Baltic Marine Area catchment during the assessment period is summarized in Table 3.1, while the corresponding mean monthly air temperature is shown in Figure 3.1. Mean data for the reference period 1981–93 are shown for comparison. The relatively warm winter conditions during the reference period persisted during the assessment period, except for the relatively cold winter of 1995/96 (cf. Figure 3.5). Also worth noting is the warm summer of 1997 in most of the area, which coincided with catastrophic rainfalls in the south. In many parts of central and southern Sweden, 1997 turned out to be the warmest summer on record, at least since 1860.

3.1.2. Precipitation

Average precipitation conditions in the Baltic Marine Area catchment during the assessment period are summarized in Table 3.2 and monthly areally averaged precipitation is shown in Figure 3.2. Corresponding data for the reference period 1981–1993 are shown for comparison. If analysed catchment-wide, the wet summer of 1998 is the most remarkable feature, although the most dramatic event was the summer rains of 1997 in the catchment of the Baltic Proper, with disastrous flooding of the Oder and the Vistula. 1998 was the wettest year ever recorded in Sweden (since 1880). In contrast, summers during 1994–97 were drier than the reference mean for much of the catchment. Also worth noting is the low precipitation during the relatively cold winter of 1995/96, which negatively affected hydropower production in the north.

Averaged over the entire catchment, snow storage of the assessment period shows no remarkable deviations from the reference period of 1981–93. On a subcatchment basis, however, the winters in the south were less stable during the assessment period, with the exception of 1995/96.

3.1.3. Wind

Long-term trends in the wind climate can be analysed from geostrophic winds estimated by data on surface pressure. Such an analysis was carried

Table 3.2
Areally averaged precipitation in the Baltic Marine Area subcatchments and whole catchment (mm). Precipitation is uncorrected for catch losses.

	1981–93 (Mean)	1994	1995	1996	1997	1998	1994–98 (Mean)
Bothnian Bay	590	463	586	522	515	692	556
Bothnian Sea	690	548	634	547	616	755	620
Gulf of Finland	672	656	640	547	629	742	643
Gulf of Riga	680	654	676	553	720	804	681
Baltic Proper	595	651	627	574	658	714	644
Belt Sea+Kattegat	708	766	651	516	613	783	666
Baltic Marine Area	639	616	630	551	625	734	631

out for the zonal (westerly) and meridional (southerly) geostrophic wind components, based on triangulated observations between Göteborg, Kalmar and Uppsala in Sweden. The results are believed to be representative for at least the southern half of the Baltic Marine Area.

The most significant wind component in a climate analysis of the Baltic Marine Area is the zonal component for winter conditions, which is also the component exhibiting the most dramatic variability, as shown in Figure 3.3. The analysis shows that the present period is outstanding in the record and that the recent mild winters are strongly related to high zonal activity in the area.

An analysis of storm trends in northwestern Europe derived from a pressure data set by Alexandersson *et al*. (1998, 2000) revealed nothing remarkable during the assessment period apart from the weak storm year of 1996.

3.1.4. Cloudiness and radiation

The impact of the Baltic Marine Area on the cloud cover is strong, particularly in spring and summer when the cloud cover is much lower over the relatively cool water body than over land. An analysis of the total cloud cover for the summer months of the assessment period is shown in Figure 3.4 based on satellite information and a cloud classification algorithm (Karlsson, 1989). As is apparent, the summers of 1994 and 1997 were characterized by low cloudiness and high radiation, while the summer of 1998 was extremely cloudy.

3.2. Ice conditions

Ice conditions of the Baltic Marine Area are strongly related to the severity of the winters. The maximum extent of ice coverage of the Baltic Marine Area during the 20th Century is summarized in Figure 3.5 (Seinä and Palosuo, 1996). As can be seen, ice coverage remained very modest during the assessment period, the only exception being the relatively cold winter of 1995/96.

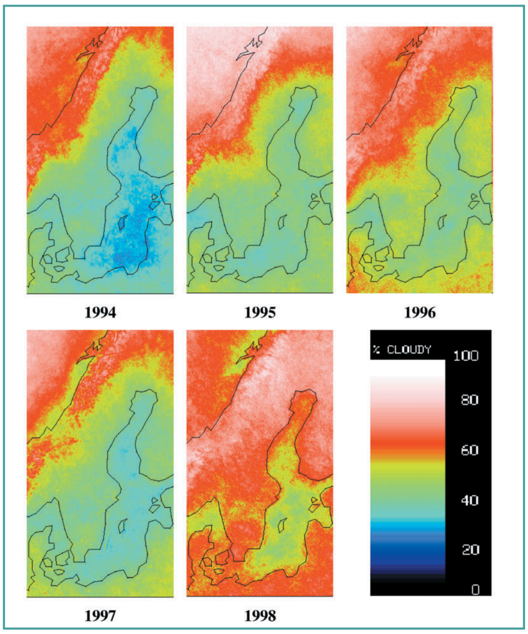


Figure 3.4
Seasonal mean cloudiness (%) for the months June, July and August during the period 1994–98 derived from NOAA AVHRR data. Note that the analysis for 1998 is based solely on data for July and August due to loss of the June data.
Prepared by K. G. Karlsson, SMHI.

3.3. Riverine runoff

Updating of riverine runoff to the Baltic Marine Area since the last assessment (HELCOM, 1996) has been made possible by support from the respective state hydrological agencies in the area. In a few cases, these data have been supplemented by modelled runoff from a hydrological model (Graham, 1999) or by estimates of specific runoff from nearby rivers. Figure 3.6 and Tables 3.3 and 3.4 summarize the period 1950–98 for the entire Baltic Marine Area catchment and its subcatchments.

Overall, the assessment period was wetter than normal with a mean riverine runoff of around 15,700 m^3/s as compared with 15,360 m^3/s for the period 1950–93. Worth noting is that 1998, with a mean of 18,720 m^3/s, stands out as the wettest year since 1950. Looking further back in the record (not shown here), 1998 is the second wettest year since 1920, only exceeded by 1924 with 19,500

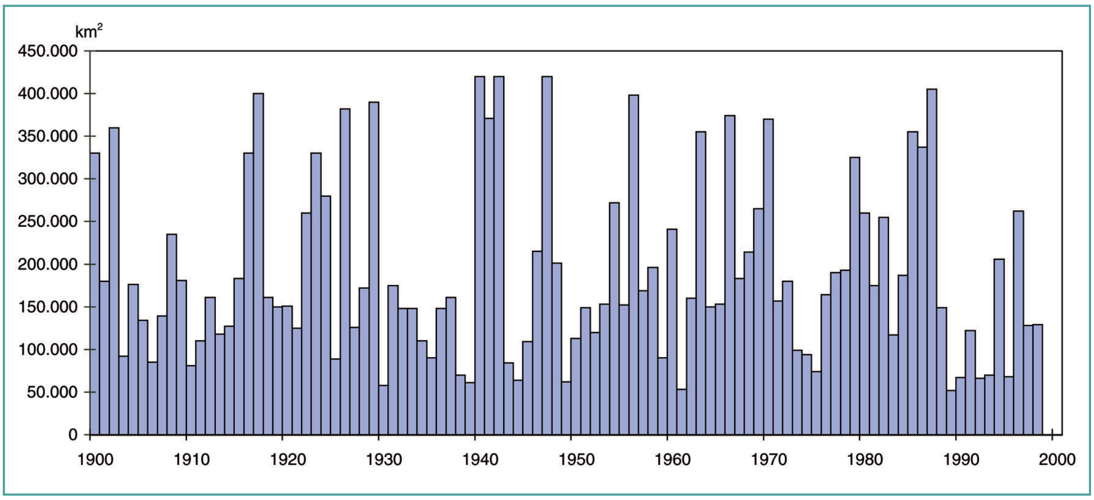


Figure 3.5
Maximum extent of ice coverage in the Baltic Marine Area during the 20th Century.
Source: Seinä and Palosuo,1996, supplemented by data from the archives of the Finnish Institute of Marine Research.

m³/s (Bergström and Carlsson, 1994).

The dramatic summer floods in Poland and Germany in 1997 are hardly noticeable in the annual means for that year. Nevertheless, the 1997 flood is one of the most dramatic hydrological events ever recorded in the Baltic area. Other extreme events during the assessment period were 100-year floods, which hit several parts of northern Sweden and parts of Norway in the spring of 1995, and intense summer floods in northern Sweden in 1998. The extreme conditions of 1995 and 1997 are reflected in the records of mean monthly flows from the river Vindelälven in northern Sweden and from the Oder in Poland and Germany, as shown in Figure 3.7.

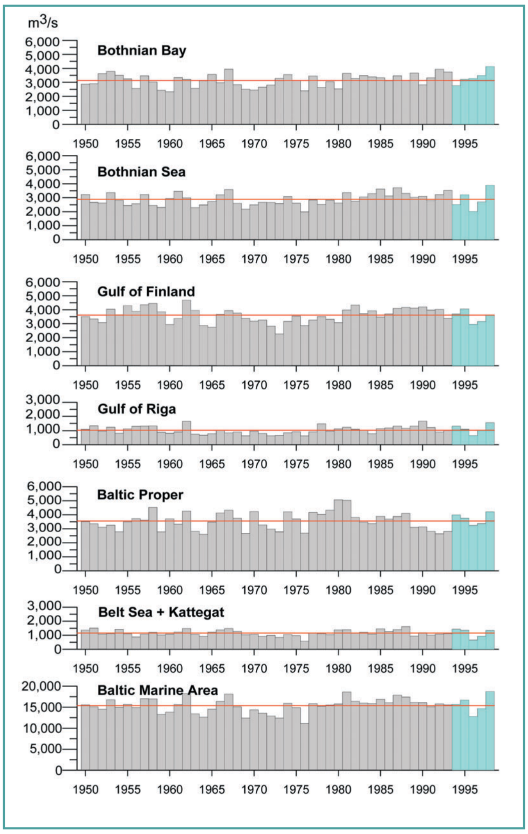


Figure 3.6
Riverine runoff to the Baltic Marine Area from the various subcatchments and whole catchment during the period 1950–98. The horizontal lines represent the means for the period 1950–93.

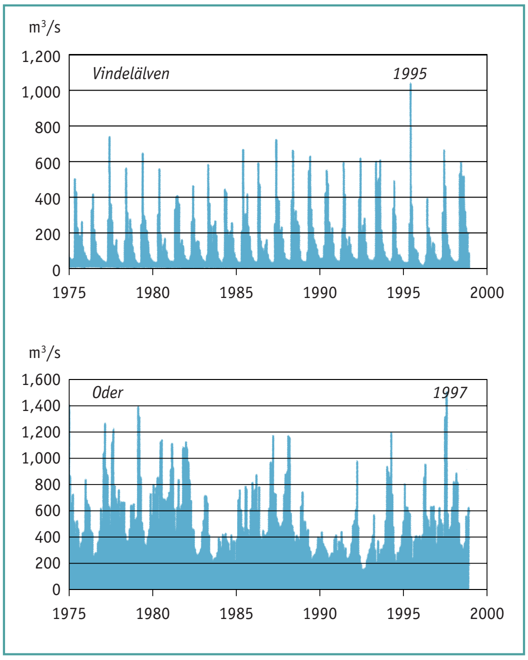


Figure 3.7
Mean monthly flow from the river Vindelälven and the river Oder during the period 1975–98.

3.4. Hydrography

3.4.1. Inflow activity

The weather situation and filling conditions in the Baltic Marine Area during the assessment period gave rise to several smaller inflows, but major inflow events did not occur. The decrease in frequency and intensity of extreme inflow events observed since the mid 1970s continued (Schinke and Matthäus, 1998) and was only interrupted by a very strong single event in January 1993 (Jakobsen, 1995; Matthäus and Lass, 1995; Fischer and Matthäus, 1996; cf. also Figure 3.8). Since the buffering capacity of the Bornholm Basin was exhausted by the very strong inflow in January 1993, smaller inflows crossing the sills into the Baltic in December 1993 and March 1994 could pass the Bornholm Basin quickly (Figure 3.9). Their water propagated relatively undisturbed and without significant losses into the central Baltic basins, thereby effectively renewing the deep water.

Similar effects were observed in the spring of 1996, 1997 and 1998 (Figure 3.9). Due to the regular filling of the Bornholm Basin with saline water >15 PSU to a depth of about 60 m during autumn of the previous years, smaller inflows of saline and oxygen-rich water crossing the sills in late autumn and early spring could pass the Bornholm Basin deep water at 50–60 m propagating relatively quickly into the eastern Gotland Basin.

Table 3.3
Average riverine runoff to the Baltic Marine Area apportioned by subregion (m³/s).

	1950–93 (Mean)	1994	1995	1996	1997	1998	1994–98 (Mean)
Bothnian Bay	3,135	2,760	3,218	3,272	3,486	4,128	3,373
Bothnian Sea	2,885	2,504	3,204	2,013	2,697	3,884	2,861
Gulf of Finland	3,611	3,700	4,048	2,951	3,149	3,627	3,495
Gulf of Riga	1,023	1,298	1,084	638	1,020	1,537	1,115
Baltic Proper	3,554	3,991	3,761	3,239	3,376	4,206	3,715
Belt Sea+Kattegat	1,152	1,428	1,345	656	917	1,337	1,137
Baltic Marine Area	15,362	15,681	16,660	12,770	14,645	18,719	15,695

Table 3.4 Seasonal riverine runoff to the Baltic Marine Area apportioned by sub-region (m^3/s).

JANUARY-MARCH

	1950–93 (Mean)	1994	1995	1996	1997	1998	1994–98 (Mean)
Bothnian Bay	1,655	1,918	1,837	2,192	2,527	1,645	2,024
Bothnian Sea	2,195	2,514	2,633	1,779	2,444	2,809	2,436
Gulf of Finland	2,635	2,290	3,502	2,071	2,942	2,555	2,672
Gulf of Riga	777	1,100	1,926	278	1,166	1,769	1,248
Baltic Proper	4,152	6,009	5,945	2,051	3,439	5,809	4,651
Belt Sea+Kattegat	1,552	2,202	2,354	604	1,417	1,588	1,633
Baltic Marine Area	12,966	16,033	18,197	8,975	13,935	16,175	14,664

APRIL-JUNE

	1950–93 (Mean)	1994	1995	1996	1997	1998	1994–98 (Mean)
Bothnian Bay	5,061	5,503	6,093	4,693	6,448	6,283	5,804
Bothnian Sea	4,067	3,903	5,911	2,219	3,801	4,046	3,976
Gulf of Finland	4,594	4,952	5,573	3,719	3,828	4,012	4,417
Gulf of Riga	1,911	2,996	1,664	1,341	1,468	1,537	1,801
Baltic Proper	4,431	5,481	4,640	4,962	3,436	4,227	4,549
Belt Sea+Kattegat	1,029	1,183	1,214	689	831	1,021	987
Baltic Marine Area	21,093	24,018	25,095	17,623	19,812	21,126	21,534

JULY-SEPTEMBER

	1950–93 (Mean)	1994	1995	1996	1997	1998	1994–98 (Mean)
Bothnian Bay	3,342	1,569	2,184	2,520	2,369	5,061	3,093
Bothnian Sea	2,655	1,304	1,796	1,386	1,842	5,197	2,580
Gulf of Finland	3,746	3,753	3,877	2,954	2,984	4,328	3,670
Gulf of Riga	540	317	243	212	280	1,456	576
Baltic Proper	2,397	1,484	1,851	2,674	3,195	2,724	2,585
Belt Sea+Kattegat	699	930	831	366	690	1,034	703
Baltic Marine Area	13,379	9,357	10,782	10,112	11,360	19,800	13,207

OCTOBER-DECEMBER

	1950–93 (Mean)	1994	1995	1996	1997	1998	1994–98 (Mean)
Bothnian Bay	2,483	1,715	2,462	2,858	2,179	3,640	2,545
Bothnian Sea	2,623	2,118	2,034	2,276	2,265	3,560	2,576
Gulf of Finland	3,470	3,683	3,046	2,833	2,749	3,645	3,046
Gulf of Riga	865	712	463	667	969	1,371	894
Baltic Proper	3,237	2,943	2,357	3,180	3,031	3,856	3,276
Belt Sea+Kattegat	1,329	1,566	950	949	806	1,847	1,348
Baltic Marine Area	14,007	12,737	11,312	12,763	11,999	17,919	13,685

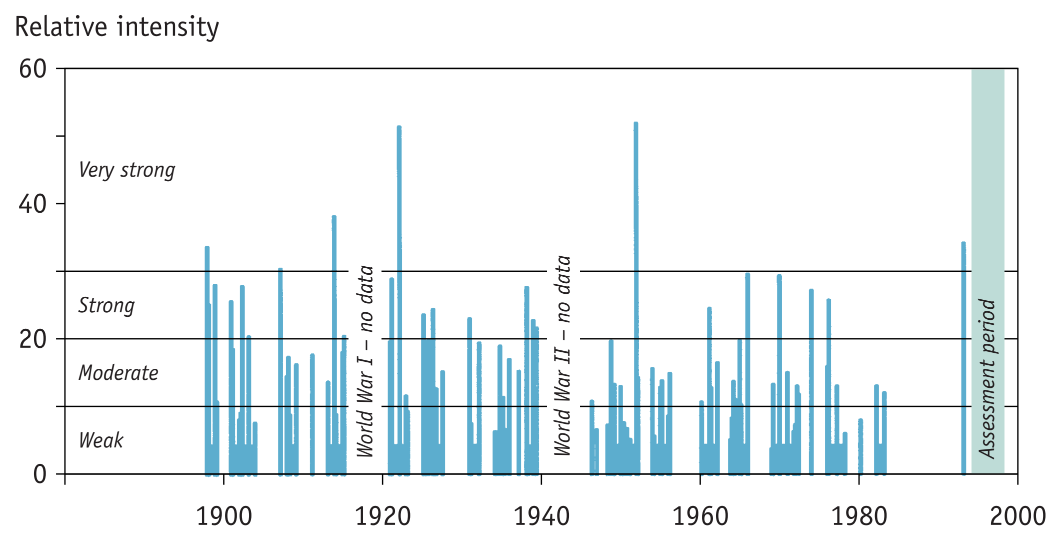


Figure 3.8 Major inflows of highly saline, oxygen-rich water into the Baltic Marine Area during the past century. The ordinate is a measure of the relative intensity of major events, the values used being calculated from the duration of the event and the salinity S of the water inflowing across the Darss and Drogden Sills when S≥17 PSU. Source: Matthäus and Franck, 1992; reassessed by Fischer and Matthäus, 1996.

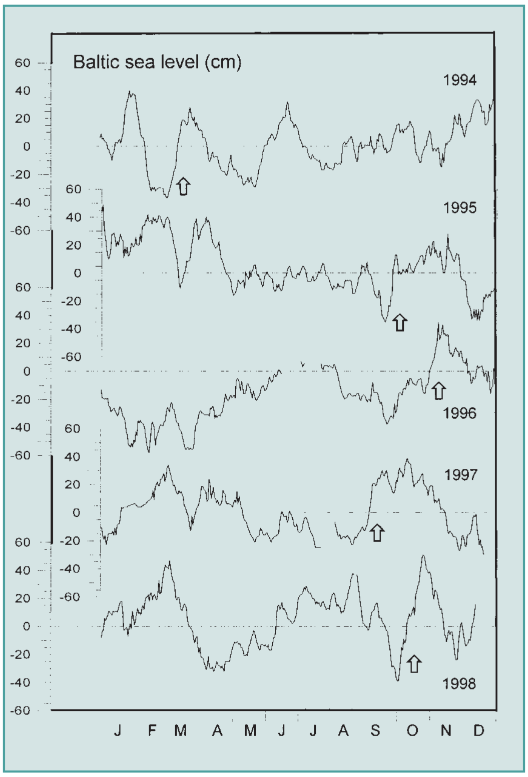


Figure 3.9
Variation in the filling stage of the Baltic Marine Area between 1994 and 1998, represented by the sea level at Landsort (data from SMHI).

The most intensive inflow of saline water to occur during the assessment period was in early November 1996, but the event just failed the criteria for a major Baltic inflow (Matthäus and Franck, 1992). At the end of October/beginning of November, the Baltic sea level started to rise very rapidly (Figure 3.9) due to prevailing westerly winds. Highly saline water (20–26 PSU) with a temperature of 8.5–10°C was transported through the Sound. The mean sea level in the Baltic Marine Area rose to 35 cm above the normal filling on 8 November. About 190 km^3 of water entered the Baltic Marine Area, 80 km^3 of it with a salinity ≥ 17 PSU. The effect of the inflow was restricted to the Bornholm Basin, however (Figure 3.10).

Between the end of May and the beginning of November 1997, persistent inflow was recorded in the near-bottom layers of Darss Sill. This culminated during two storms in September and at the beginning of October, transporting larger volumes of exceptionally warm and saline water across the sills into the Baltic Marine Area (Figure 3.10). The small inflows in the second half of 1998 only affected the Bornholm Basin deep water.

3.4.2. Variation above the halocline

Three of the four warmest summers of the present century occurred during the assessment period. However, they were characterized by very different temporal courses in air temperature and solar radiation and their impact on the development in sea surface temperature thus differed. The highest monthly means of up to >23°C were always found in August of the years 1997, 1994 and 1995. With respect to July, sea surface temperature in the central part of the Baltic Marine Area was highest in 1994 and 1997. In contrast to August 1995, July 1995 did not differ substantially from the other years.

During summer 1994, the highest surface temperatures were in the range 21–24.5°C in the western and central parts of the Baltic Marine Area. These temperatures correspond to positive anomalies of 5–6 K. The high temperatures were mainly restricted to the upper 10-metre water layer, which indicates that the water gained heat quickly and without disturbance in spring. Positive anomalies of 4–5 K were also found at a depth of 30 m in the Gotland, Fårö and Karlsö Deeps, however. High surface temperatures were not only measured in the Baltic Proper but also in the Gulfs of Finland and Bothnia.

In July 1997, the monthly mean sea surface temperature reached 17–18°C in the entire Baltic Marine Area and the maximum temperatures in the Gulf of Bothnia ranged between 18 and 22°C at the end of July, i.e. positive anomalies of 5–6 K. The temperatures in the western and central parts of the Baltic Marine Area increased rapidly at the end of July and during the whole of August. In August 1997, sea surface temperature exceeded 23°C in the open parts of the Baltic Proper and increased to 25°C in the western Baltic coastal areas except for the south and east coasts due to upwelling.

Anomalously high sea surface temperatures have previously been observed in 1938, 1959, 1975 and 1992. The surface temperatures in August 1997 still exceeded the values observed in the exceptionally warm summers of 1975, 1992 and 1994.

The exceptional warming in 1997 covered the whole layer above the summer thermocline of the western and central Baltic and resulted in temperatures >15°C to a depth of 20–25 m. Based on the long-term means, positive anomalies occurred of up to 6 K in the Arkona Basin, between 5 and 6 K in the Bornholm Basin and about 4 K in the Gotland Basin.

During the assessment period, relatively low salinities were observed in the central Baltic surface water due to the above-normal riverine runoff in winter and spring (Table 3.4). Surface salinity has decreased continuously in the eastern Gotland Basin since the late 1970s (Figure 3.11). Compared with the long-term mean (1961–90), the annual means in 1998 exhibited negative anomalies of about 0.5 PSU in the Bornholm Basin and 0.4 PSU in the eastern Gotland Basin (Table 3.5).

3.4.3. Changes in the deep water

In April 1993, a renewing process started in the deep water of the eastern Gotland Basin following major inflow in January 1993. This continued due to the effects of smaller inflows in December 1993 and March 1994. In May 1994, the deep waters of the Baltic Marine Area were free of hydrogen sulphide and oxygen concentrations of 3–3.8 ml/l were measured in the Gotland Deep at depths below 170 m, the highest recorded since the 1930s (Figure 3.10).

The absence of major inflow events thereafter has resulted in a new period of stagnation in the central Baltic deep waters. After the renewal, oxygen depletion started again in 1995 (Nehring *et al.*, 1995) and anoxic conditions developed (Figure 3.10). Hydrogen sulphide was detected in the Gdansk Deep in May 1995, in the Bornholm Basin in August 1995 and in the near-bottom layer of the Gotland Deep in February 1996 (Figure 3.12).

The current stagnation period in the eastern Gotland Basin deep water, however, shows several peculiarities compared with earlier periods of stagnation. During the years 1996, 1997 and 1998, the anoxic conditions were interrupted each winter and spring by the occurrence of low oxygen concentrations (Figure 3.10) reaching the Gotland Deep with small inflows of saline water.

In the deep water of the western Gotland Basin, oxygen depletion has continued since 1993 (Figure 3.10). The decrease in oxygen concentration, which is characteristic in this basin during the initial phase of stagnation periods (Matthäus, 1995; cf. also Table 3.6), led in 1998 to the lowest oxygen content since the mid 1980s.

The small inflow events in December 1993 and March 1994 transported cold, saline and oxygen-rich water into the central deep basins. The temperature at depths below 100 m in the Gotland Deep decreased from >5°C in the beginning of the year to 4–4.5°C in May 1994. Temperatures around 4°C are among the lowest values recorded in the deep water of the Gotland and Fårö Deeps during the present century (Figure 3.10).

The exceptionally high sea surface temperatures in summer 1997 led to an unusual increase in deep-water temperatures in late autumn due to a stronger inflow in September and early October (Figure 3.9). At the end of October 1997, positive temperature anomalies of about 2 K were measured in the Arkona Basin deep water and of 3–4 K in the Bornholm Basin deep water.

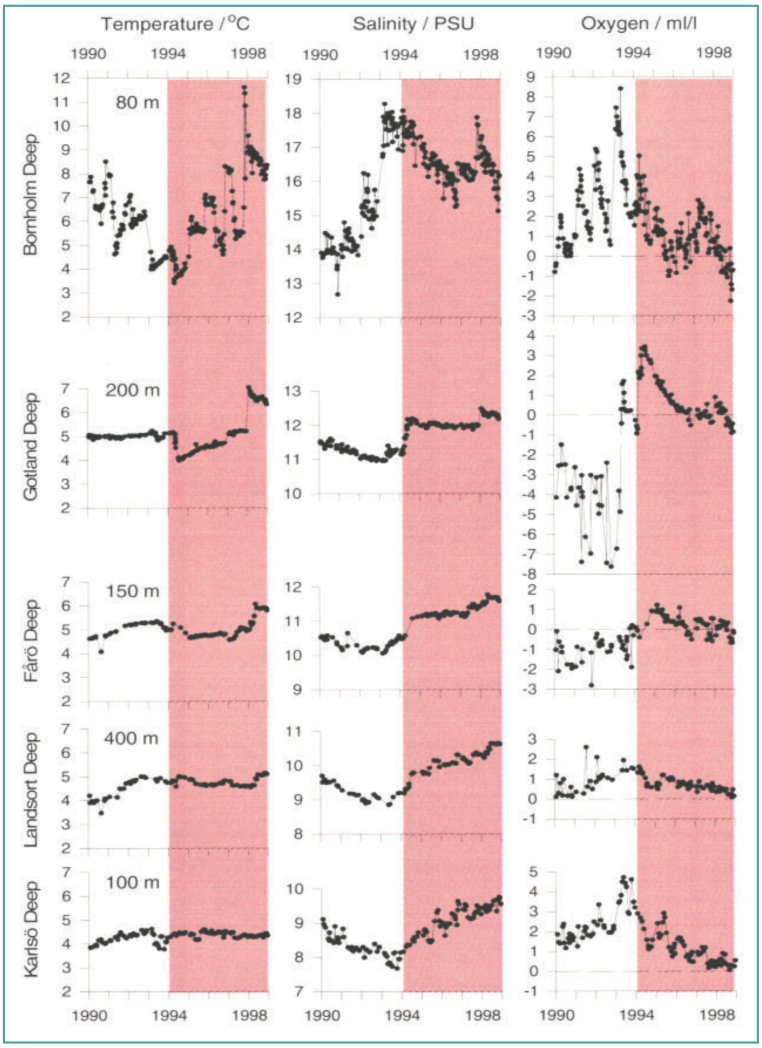


Figure 3.10
Variation in temperature, salinity, oxygen and hydrogen sulphide concentrations in the deep water of the Baltic Proper between 1990 and 1998. The hydrogen sulphide concentrations are expressed in terms of negative oxygen equivalents.

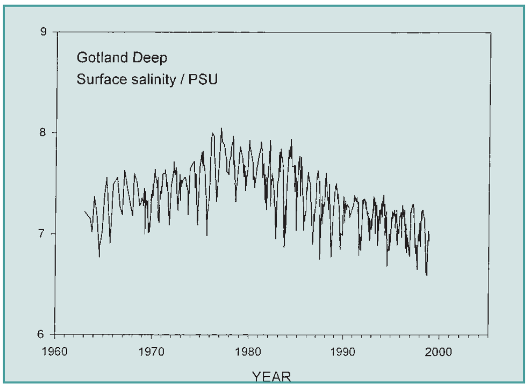
The inflow of warm (7–8°C), saline (14–15 PSU) water with an oxygen content of up to 2 ml/l observed in February 1998 at the southern slope of the eastern Gotland Basin (Matthäus *et al.*, 1999) reached the Gotland Deep in spring (Figure 3.10). As a result, temperature and salinity increased in the eastern Gotland Basin to >7°C and 12.7 PSU, respectively, in the near-bottom water of the Gotland Deep. The inflow process into the central Baltic ceased in May. The inflows across the sills into the Baltic Marine Area in 1998 did not significantly affect the central Baltic deep water, and temperature and salinity therefore decreased again (Figure 3.10). From July onwards, the Gotland Basin deep water became anoxic from a depth of 130 m to the bottom, and the areas of both oxygen deficiency and anoxic conditions were the largest observed since 1993 (Figure 3.12).

The inflow of warm, saline and oxygen-rich

Station	**1994**	**1995**	**1996**	**1997**	**1998**	**S_0** **(1961–1990)**
BMP K2 (Bornholm Deep)	7.35 ±0.13	7.50 ±0.21	7.28 ±0.21	7.45 ±0.09	*7.17 ±0.14*	7.72 ±0.25
BMP J1 (Gotland Deep)	7.09 ±0.21	7.14 ±0.09	7.08 ±0.15	7.05 ±0.20	*7.04 ±0.20*	7.41 ±0.46
BY 20 (Fårö Deep)	6.91 ±0.25	6.82 ±0.20	*6.74 ±0.24*	6.90 ±0.13	6.79 ±0.35	7.15 ±0.31
BMP H3 (Landsort Deep)		6.64 ±0.27	6.36 ±0.37	6.59 ±0.24	*6.30 ±0.41*	
BMP I1 (Karlsö Deep)		6.94 ±0.17	6.68 ±0.24	6.89 ±0.22	*6.60 ±0.15*	

Table 3.5
Annual and long-term (S_0) mean surface salinity (PSU) in the central part of the Baltic Marine Area. Standard deviation is also indicated (italic: minimum values).

Figure 3.11
Long-term variation in surface salinity in the eastern Gotland Basin (Gotland Deep).



water into the Baltic Marine Area in 1997 led to the highest annual mean temperature and salinity in the deep water of almost all regions of the central Baltic in 1998 compared with the annual means for the 1990s (Table 3.6). The annual mean oxygen concentration concomitantly reached the lowest value since 1994 in the deep water of the eastern Gotland Basin and since the mid 1980s in the western Gotland Basin deep water. For the first time during the 20th Century, annual mean oxygen content became negative at depths of 80 m in the Bornholm Deep in 1998 (Table 3.6).

3.5. Climate change

An attempt to analyse potential impacts of future climate change on the freshwater inflow to the Baltic Marine Area has been made by Graham *et al.* (2000). Based on regional down-scaling of Global Climate Models (GCMs) carried out within the Swedish regional climate modelling programme (SWECLIM), it appears that global warming may lead to a changed annual cycle and less pronounced spring floods as winters become less stable. According to these scenarios, freshwater inflow to the Baltic Marine Area will generally increase in regions fed by the northernmost parts of the catchment and decrease in the Baltic Proper.

The variation in water exchange between the North Sea and the Baltic Marine Area during the assessment period, which is reflected in the hydrographic conditions of the central Baltic deep water, is within the range of natural fluctuation, and can hardly be attributed to climate change.

Figure 3.12
Annual maximum extent of oxygen deficiency (≤2 ml/l) and hydrogen sulphide (dark blue areas) in the deep basins of the Baltic Proper between 1994 and 1998. Source: Matthäus et al., 2000, supplemented by data from the Finnish Institute of Marine Research.

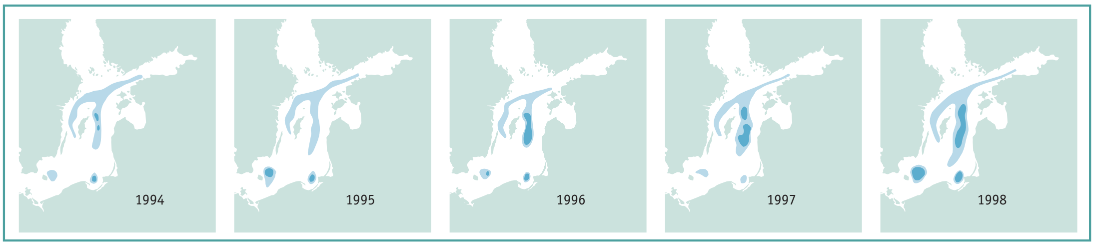


Table 3.6
Annual mean temperature, salinity and oxygen concentration in the deep waters of the central part of the Baltic Marine Area (1994–1998). Standard deviation is also indicated.

TEMPERATURE (°C; italic: maximum values)

Station	Depth/m	1994	1995	1996	1997	1998
BMP K2 (Bornholm Deep)	80	4.21 ±0.46	5.85 ±0.56	5.95 ±0.94	7.32 ±1.91	*8.53 ±0.45*
BMP J1 (Gotland Deep)	200	4.68 ±0.48	4.43 ±0.13	4.64 ±0.07	5.17 ±0.06	*6.62 ±0.15*
BY 20 (Fårö Deep)	150	5.03 ±0.13	4.72 ±0.04	4.79 ±0.03	4.84 ±0.20	*5.66 ±0.32*
BMP H3 (Landsort Deep)	400	4.86 ±0.13	4.74 ±0.11	4.70 ±0.06	4.66 ±0.08	*4.88 ±0.23*
BMP I1 (Karlsö Deep)	100	4.41 ±0.06	4.39 ±0.15	*4.45 ±0.07*	4.35 ±0.08	4.33 ±0.05

SALINITY (PSU; italic: maximum values)

Station	Depth/m	1994	1995	1996	1997	1998
BMP K2 (Bornholm Deep)	80	*17.43 ±0.34*	16.61 ±0.31	16.01 ±0.37	16.52 ±0.50	16.38 ±0.49
BMP J1 (Gotland Deep)	200	11.78 ±0.38	12.01 ±0.04	11.96 ±0.03	11.97 ±0.04	*12.32 ±0.07*
BY 20 (Fårö Deep)	150	10.75 ±0.29	11.17 ±0.03	11.23 ±0.04	11.32 ±0.13	*11.62 ±0.09*
BMP H3 (Landsort Deep)	400	9.49 ±0.23	9.94 ±0.16	10.09 ±0.11	10.22 ±0.11	*10.50 ±0.16*
BMP I1 (Karlsö Deep)	100	8.45 ±0.16	8.82 ±0.30	9.11 ±0.26	9.33 ±0.14	*9.51 ±0.13*

OXYGEN CONCENTRATION (ml/l; hydrogen sulphide converted into negative oxygen equivalents; italic: minimum values)

Station	Depth/m	1994	1995	1996	1997	1998
BMP K2 (Bornholm Deep)	80	2.43 ±1.07	0.76 ±0.91	0.54 ±0.59	1.34 ±0.72	*-0.20 ±0.81*
BMP J1 (Gotland Deep)	200	2.03 ±1.40	1.26 ±0.43	0.20 ±0.24	*-0.03 ±0.25*	0.02 ±0.48
BY 20 (Fårö Deep)	150	0.17 ±0.44	0.70 ±0.26	0.16 ±0.39	0.14 ±0.39	*-0.03 ±0.33*
BMP H3 (Landsort Deep)	400	1.12 ±0.36	0.92 ±0.22	0.69 ±0.11	0.60 ±0.14	*0.37 ±0.14*
BMP I1 (Karlsö Deep)	100	1.88 ±0.64	1.72 ±0.68	1.09 ±0.46	0.66 ±0.34	*0.43 ±0.22*

Inputs to the Baltic marine environment

4

By:
Heike Herata
Lars M. Svendsen
Tuija Ruoho-Airola
Anne Christine Brusendorff

The Baltic Marine Area receives pollutant inputs from both terrestrial sources including riverine runoff, direct discharges and atmospheric deposition, and from marine sources including ships and offshore installations.

These inputs are in principle controllable, and hence interesting from an administrative and political point of view – measures can be implemented to reduce them and hence improve environmental conditions where necessary.

It is not possible to directly judge the importance of these external inputs for the environmental conditions in the Baltic Marine Area, however, since a number of additional factors that we cannot control have to be taken into account before a complete picture can be ascertained. The magnitude of these inherent factors, e.g. biological fixation of nitrogen, denitrification, immobilization/mobilization of phosphorus in the sediments and long-range transport of nutrients by currents into or out of the Baltic Marine Area, is assessed in other chapters. In the case of nutrients, though, it is well known that the total inputs from land exceed internal loading from the above-mentioned processes.

This chapter mainly focuses on nutrients, organic matter and heavy metals, but also provides some information about oil. Hazardous substance loads cannot yet be compiled as they are not generally monitored on a routine basis.

The waterborne inputs are of two main types: *Direct loading* from sources discharging directly into the Baltic Marine Area such as coastal towns and industrial plants, and *riverine loading* comprising upstream discharges from the catchment area, including discharges from inland towns and industrial plants, diffuse loading from agriculture, forestry and sparsely built-up areas, and natural background loading. In HELCOM's Third Waterborne Pollution Load Compilation (HELCOM, 1998), total runoff was estimated to be 466,320 million m^3/yr, of which about 93% derives from monitored rivers, 6% from coastal areas and unmonitored rivers, and 1% from treated and untreated municipal and industrial discharges.

Air-borne inputs to the Baltic Marine Area are measured as atmospheric deposition. They derive both from sources within the Baltic Sea catchment area and from sources lying outside the catchment, from where they have been transported by atmospheric processes. In HELCOM's Airborne Pollution Load Compilation (HELCOM, 1997), these loads are

Spreading of animal manure on a cornfield. Photo: Biofoto/Mr. Lars Gejl.

Figure 4.1
Riverine organic matter (BOD_7) load to the Baltic Marine Area

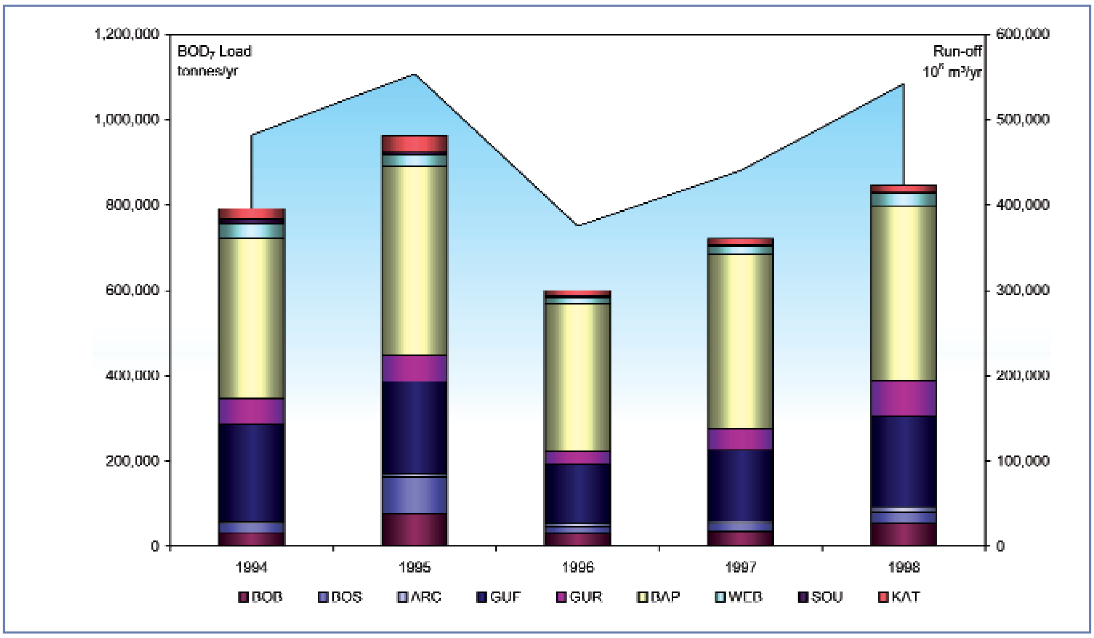


Figure 4.2
Riverine nitrogen (total-N) load to the Baltic Marine Area

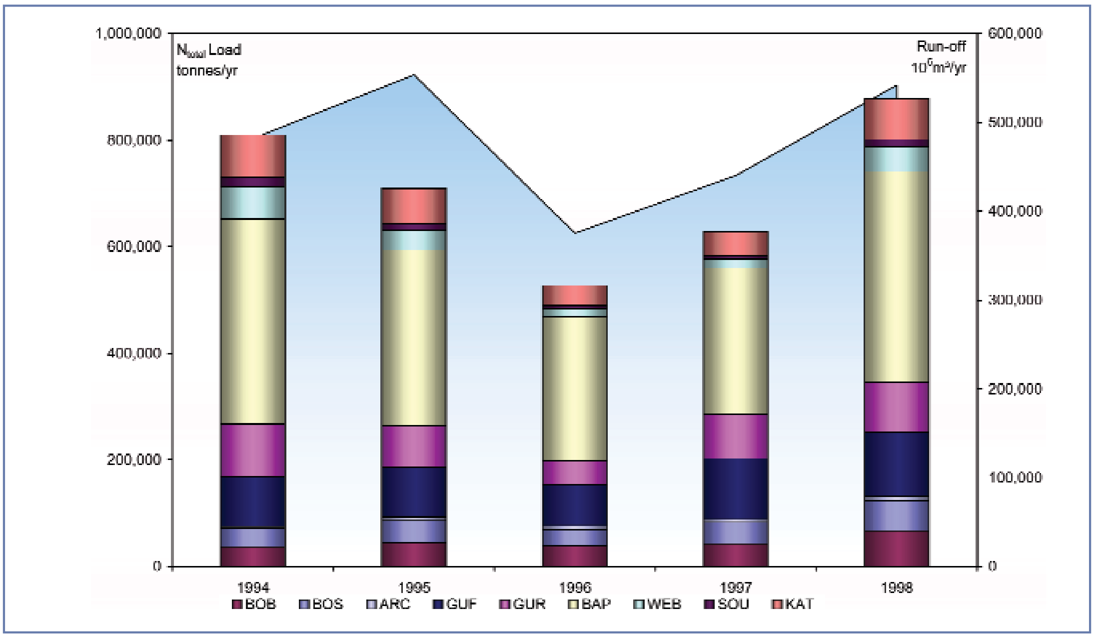
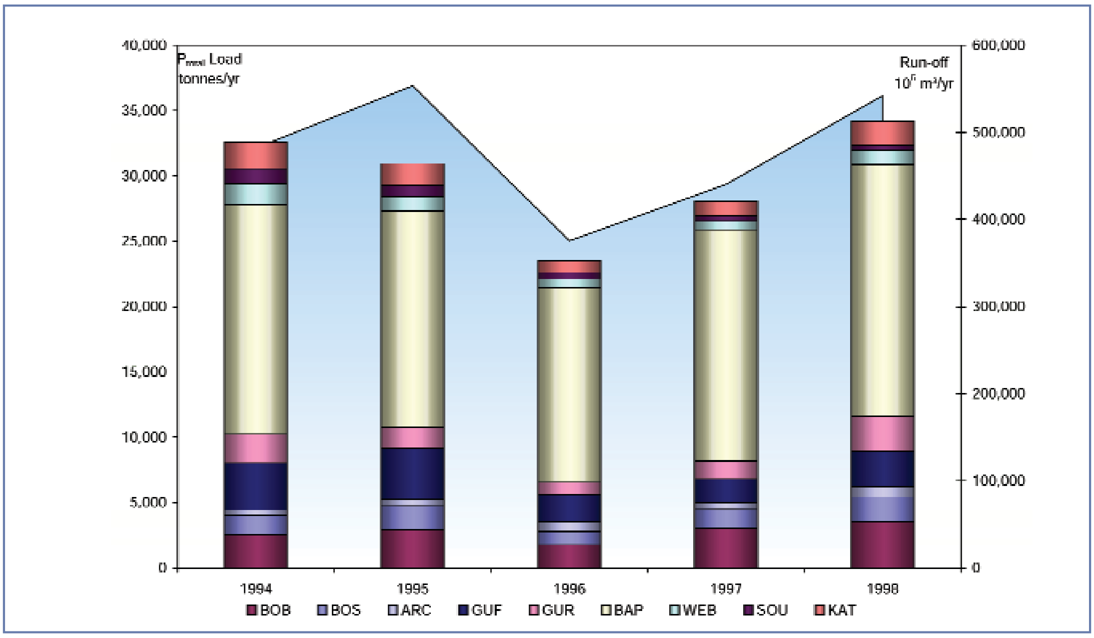


Figure 4.3
Riverine phosphorus (total-P) load to the Baltic Marine Area.



estimated on the basis of measurements and model calculations.

The inputs from ships and offshore installations consist of operational inputs from engine rooms and engine exhausts, illegal discharges and accidental discharges. Oil slick sightings made by surveillance aircraft provide an indication of these loads, but accurate estimation is very difficult.

4.1. Waterborne pollution load

The compilation encompasses nutrients, organic matter and heavy metal loads entering the Baltic marine environment via rivers, coastal areas and direct point sources such as municipal and industrial wastewater treatment plants. Only a small portion of the total point-source load within the Baltic Sea catchment area is considered separately, and an inventory of all point sources in the whole Baltic Sea catchment area is thus lacking.

4.1.1. Direct load from point sources

Direct load from municipal sources

In 1995, the discharge of wastewater from the 15 million inhabitants of the 430 municipalities discharging directly into the Baltic Marine Area totalled 3,490 million m^3/yr. Nearly 500 million m^3/yr (14%) of this was discharged untreated. The majority of this (430 million m^3/yr) originated from St Petersburg and from the Leningrad region of Russia with a population of 1.2 million and was mainly discharged into the Gulf of Finland. An additional 177 million m^3/yr of wastewater, of which 151 million m^3/yr were untreated, were discharged from the Kaliningrad region of Russia, which has 900,000 inhabitants.

The nutrient and organic matter loads from municipalities discharging directly into the Baltic Marine Area are shown in Table 4.1 apportioned by subregion into which they are discharged. The municipal organic matter (BOD_7) loads discharged directly into the Bothnian Bay, the Bothnian Sea, the Archipelago Sea, the western Baltic, the Sound and the Kattegat are low due to effective treatment of municipal wastewater in Finland, Sweden, Germany and Denmark, where the removal rate generally exceeds 90%. In addition to St Petersburg and the Leningrad region, municipal organic matter loading of the Baltic Marine Area is high from the large cities of countries in economic transition on the coast of the Baltic Proper, where the efficiency of wastewater treatment is suboptimal.

The untreated municipal nitrogen (total-N) and phosphorus (total-P) loads discharged directly into the Baltic Marine Area are quite small, accounting for only 3% and 7% of the total direct loads, respectively. In many subregions of the Baltic Marine Area the direct municipal nitrogen and phosphorus loads considerably exceed the corresponding industrial loads. The phosphorus load from municipal sources was greatest in the Gulf of Finland (2,610 tonnes total-P/yr), of which the St Petersburg region accounted for the majority (2,440 tonnes total-P/yr).

Subregion	BOD_7 tonnes/yr	Total-N tonnes/yr	Total-P tonnes/yr
BOB	2,615	1,907	32
BOS	1,603	1,941	40
ARC	867	1,157	26
GUF	47,481	25,389	2,612
GUR	5,224	1,249	326
BAP	40,617	15,552	1,259
WEB	4,965	5,177	215
SOU	3,292	4,733	578
KAT	3,660	3,728	204
Baltic Marine Area	110,324	60,833	5,292

Table 4.1
Organic matter (BOD_7), nitrogen (total-N) and phosphorus (total-P) loads (treated and untreated) from municipal sources discharging directly into the Baltic Marine Area in 1995 apportioned by subregion.

Mercury kg/yr	Cadmium kg/yr	Zinc kg/yr	Copper kg/yr	Lead kg/yr
1,140	6,590	360,660	75,880	32,940

Table 4.2
Heavy metal loads from municipal sources discharging directly into the Baltic Marine Area in 1995 (Data are incomplete. Heavy metals were not measured/estimated in Denmark in 1995. All Estonian figures are from 1994).

Heavy metals loading of the different subregions of the Baltic Marine Area varies depending on the population density, location of industries, exploitation of natural resources, efficiency and degree of wastewater treatment, etc. The anthropogenic load derives from industrial wastewater, release from products and pollution from various forms of land use, e.g. agriculture (cadmium content of fertilizers) and mining (mine waste deposits).

Due to the fact that the data are very incomplete, it is presently impossible to fully describe heavy metal loading from municipal sources discharging directly into the Baltic Marine Area. For technical reasons, comparable heavy metal load data are not available for all countries in 1995 (HELCOM, 1998). Cadmium loading from municipal sources in Russia discharging directly into the Gulf of Finland seems to be very high. Heavy metal loading of the Baltic Marine Area from direct municipal sources is summarized in Table 4.2, although the figures are very uncertain and undoubtedly underestimated.

Direct load from industrial sources

The 183 industrial plants along the Baltic coast discharged 1,950 million m^3/yr of wastewater directly into the Baltic Marine Area in 1995, of which more than 1,948 million m^3/yr was treated wastewater discharged by 177 of the industrial plants. The untreated wastewater discharged by the remaining 6 industrial plants amounted to 0.42

Table 4.3
Organic matter (BOD_7), nitrogen (total-N), phosphorus (total-P) loads from industrial sources discharging directly into the Baltic Marine Area in 1995 apportioned by subregion.

Subregion	BOD_7 tonnes/yr	Total-N tonnes/yr	Total-P tonnes/yr
BOB	14,606	1,508	125
BOS	43,202	2,261	271
ARC	150	763	91
GUF	3,996	4,919	822
GUR	173	155	26
BAP	20,397	2,384	434
WEB	6,844	803	75
SOU	2,900	954	73
KAT	9,492	829	93
Baltic Marine Area	101,760	14,576	2,010

Table 4.4
Heavy metals load from industrial sources discharging directly into the Baltic Marine Area in 1995 (Data are incomplete. No Danish measurements for 1995. All Estonian figures are from 1994).

Mercury kg/yr	Cadmium kg/yr	Zinc kg/yr	Copper kg/yr	Lead kg/yr
610	610	87,930	49,630	3,960

million m^3/yr, which is less than 0.1% of the total direct industrial load.

Organic matter, nitrogen and phosphorus loading from direct industrial sources in 1995 are summarized in Table 4.3. In the case of the Bothnian Sea, the industrial organic matter load is greater than the municipal organic matter load. The main industries discharging into this area are pulp and paper mills. The majority of the industrial organic matter load (BOD and COD) discharged into the Bothnian Bay, the Bothnian Sea and the Gulf of Finland also derived from the pulp and paper branch, mainly in Sweden and Finland. All the Finnish plants have biologically activated sludge removal treatment systems, but some Swedish plants still rely on mechanical wastewater treatment methods.

The untreated industrial nitrogen and phosphorus loads discharged directly into the Baltic Marine Area were relatively small, accounting for only 11% and 0.5%, respectively, of the total direct loads. The majority of the treated industrial nitrogen load is discharged into the Gulf of Finland (26%), the Baltic Proper (18%) and the Bothnian Sea (17%). Nearly all the untreated industrial nitrogen load, 1,570 tonnes N/yr, was discharged from Estonian industry directly into the Gulf of Finland. Most of the treated industrial phosphorus load was also discharged into the Gulf of Finland (41%), the Baltic Proper (21%) and the Bothnian Sea (13%), and derived from Russia (45%), Sweden (17%) and Finland (14%). Only Estonia, Latvia and Poland discharged untreated industrial phosphorus-containing wastewater directly into the Baltic Marine Area, but these inputs were negligible compared to the inputs from the other sources.

Due to the very incomplete data available, it has not been possible to determine the heavy metals load discharged directly into the Baltic Marine Area by industrial plants. The cadmium load data for Russian industrial plants discharging directly into the Gulf of Finland seems to be very high. Heavy metal loading of the Baltic Marine Area by direct industrial sources is summarized in Table 4.4, although the figures are very uncertain and undoubtedly underestimated.

4.1.2. Riverine load

More than 200 Baltic rivers are monitored or partially monitored. The total runoff from these rivers ranges from 400,000 to 600,000 million m^3/yr. The total catchment area is 1.5 million km^2, of which 1.2 million km^2 are located within the territory of Baltic Sea States. The discharge from coastal zones and unmonitored rivers amounts to approx. 6% of the total discharge. The corresponding catchment area is approx. 125,000 km^2.

The total riverine organic matter (BOD_7) load entering the Baltic Marine Area during the assessment period 1994–1998 ranged from 600,000 to 960,000 tonnes/yr. The majority of the organic matter load, 75%, entered the Baltic Marine Area via monitored rivers, while 15% entered via unmonitored rivers and coastal areas. The organic matter load from the rivers discharging into the northern part of the Baltic Marine Area (Bothnian Bay and Bothnian Sea) mainly derives from natural unmanaged areas with low anthropogenic activity and is attributable to the high content of humic matter (from forests, peat soils, etc.). In some rivers entering the Bothnian Bay the organic matter concentrations are close to or below the detection limit.

The total riverine nitrogen (total-N) load entering the Baltic Marine Area during the assessment period 1994–1998 was closely related to runoff. In years with high precipitation and hence high runoff, more nitrogen leached from cultivated areas than during dry years, thus resulting in a higher riverine nitrogen load. Approx. 50% of the nitrogen load entering the Baltic Marine Area was discharged from the catchment area of the Baltic Proper via the three main rivers: the Vistula, the Oder and the Nemunas.

The total riverine phosphorus (total-P) load entering the Baltic Marine Area during the assessment period 1994–1998 varied little from year to year with an overall tendency towards a slight increase. As with nitrogen, the majority of the riverine phosphorus load (more than 50%) entering the Baltic Marine Area was discharged from the catchment area of the Baltic Proper via the Vistula, the Oder and the Nemunas. The second main source of phosphorus input to the Baltic Marine Area is discharges from the catchment areas of the Gulf of Finland, the Bothnian Bay and the Gulf of Riga. While the phosphorus load from the catchment area of the Sound has decreased due to improved municipal wastewater treatment, that from the catchment area of the Archipelago Sea seems to be increasing.

Riverine runoff and nutrient/organic matter loading are highly correlated. The impact of differences in runoff during the years 1994 to 1998 can be partly eliminated by comparing flow-weighted concentrations calculated by dividing the transport with the corresponding flow for the various input sources (rivers and direct discharges). Since direct discharge data are only available for 1995, the flow-weighted concentrations shown in Table 4.5 are calculated on the basis of the monitored riverine inputs. No changes were seen in the flow-weighted organic matter, nitrogen and phosphorus concentrations during the assessment period 1994–1998.

This is attributable to the considerable reduction in the organic matter, nitrogen and phosphorus loads that took place between the end of the 1980s and 1995 in the majority of the subregions. Sweden, Finland, Germany and Denmark started to build effective treatment plants before 1995, and most of the reduction had already been achieved prior to 1995. The reduction in organic matter and phosphorus loads up to 1995 is attributable to the implementation of treatment measures at the major municipal wastewater treatment plants, industrial plants and fish farms within the catchment areas. Due to economic transition in the former socialist countries, many industrial plants – particularly large old-fashioned plants lacking in wastewater treatment plants – have been closed down. New industrial plants built since the beginning of the 1990s all now incorporate wastewater treatment. The amount of untreated municipal effluents from large cities and from small settlements discharged directly into rivers has also been reduced. Municipal wastewater treatment plants have been built incorporating chemical and biological treatment, including phosphorus removal, thus considerably reducing the organic matter and phosphorus loads discharged into the rivers. In most of the countries, moreover, there has been a switch to detergents with a low phosphate content.

In order to considerably reduce the nitrogen load, measures have been implemented in many of the Baltic Sea States to reduce nitrogen discharges from municipal wastewater treatment plants and agricultural sources.

Due to the scarcity of available data, the riverine heavy metals loads entering the Baltic Marine Area cannot be estimated. However, the few data available provide no indication of any decrease in the heavy metals load. The difficulties in obtaining comparable heavy metals load data are the same as explained above for point sources and apply to both countries in transition and the other Baltic Sea States. The marked differences in the size of the watercourses in the various countries also play a role. Thus while it is relatively easy to effectively monitor the few large rivers in southern Sweden, it is much more difficult to effectively monitor the hundreds of small streams and brooks in Denmark.

Table 4.5
Flow-weighted concentrations in the rivers of the Baltic Sea States.

	1994	1995	1996	1997	1998
BOD_7 (mg/l)	1.6	1.7	1.6	1.6	1.6
Total-N (mg/l)	1.7	1.3	1.4	1.4	1.6
Total-P (mg/l)	0.068	0.056	0.063	0.064	0.063

Table 4.6
Nitrogen emissions in 1997.

	Nitrogen oxides thousand tonnes	Ammonia thousand tonnes
Baltic Sea States	6,265	2,009
States emitting to the Baltic Marine Area (1996)	10,017	2,893

4.2. Airborne pollution load

Atmospheric deposition on the Baltic Marine Area is significant for nitrogen, certain heavy metals and certain persistent organic pollutants (POPs). The available data are best for nitrogen. The quality and amount of data on heavy metals and POP loads are increasing, and improvements are expected in measurements, emission inventories and model calculations. The results presented here are still preliminary in the case of many compounds other than nitrogen.


Waste water treatment plant at Rødbyhavn, Denmark.
Photo: Biofoto/Mr. Lars Havn Eriksen, 1994.

Figure 4.4 & 4.5 Calculated deposition of nitrogen oxide (NO_3-N) and ammonia (NH_4-N) on the Baltic Marine Area in 1998.

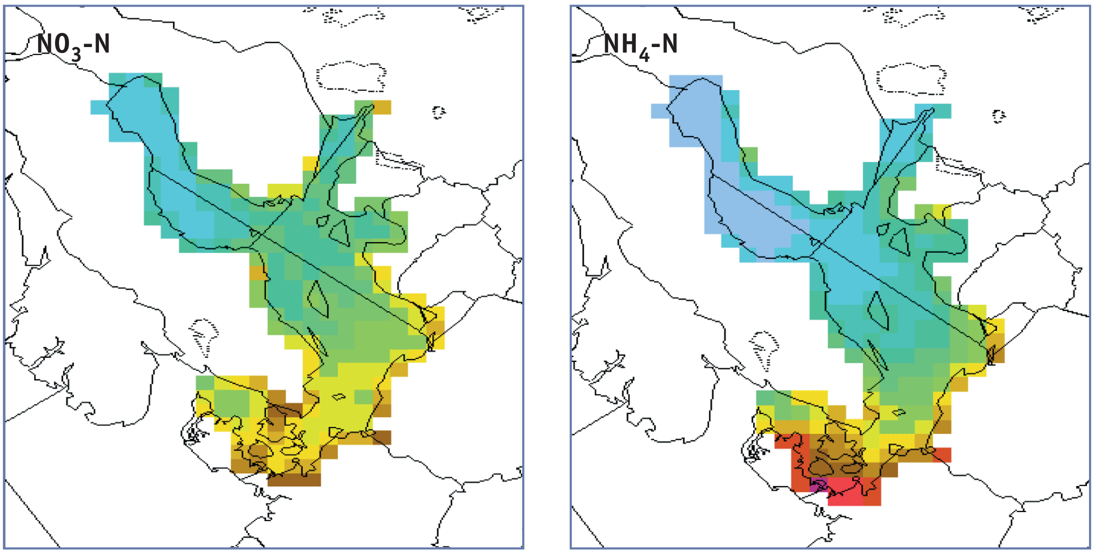
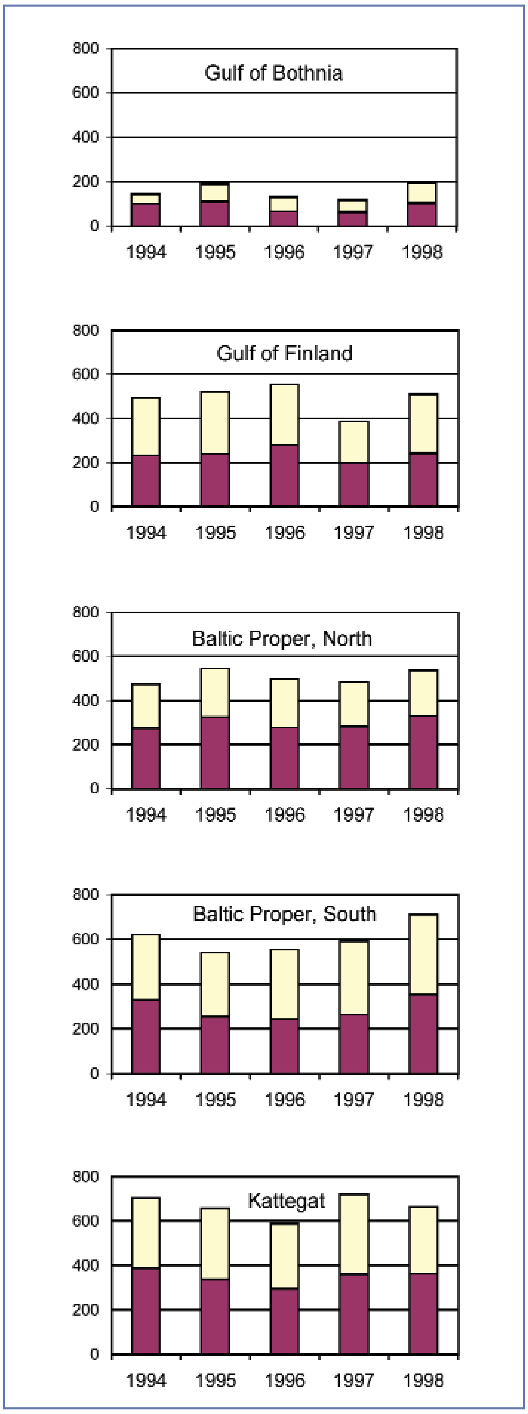


Figure 4.6 Wet deposition of nitrogen on subregions of the Baltic Marine Area in 1994-1998 in mg/m². (Examples from some coastal measuring stations).



4.2.1. Emissions to the air

Nitrogen emissions in 1997 are presented in Table 4.6 (Bartnicki *et al.*, 1998, Bartnicki *et al.*, 2000).

The designation "States emitting to the Baltic Marine Area" includes the Baltic Sea States as well as the United Kingdom, the Netherlands, France and the Czech Republic in the case of nitrogen oxides and Belarus and Ukraine in the case of ammonia (Bartnicki *et al.*, 1998). Nitrogen oxide and ammonia emissions in Europe have decreased slightly by 3–5% between 1994 and 1997 (Tarrason and Schaug, 1999).

The emissions situation as regards heavy metals is slowly improving, but is not yet as good as for nitrogen compounds. Best estimates are available for lead, cadmium and mercury. POP emission estimates for the European countries are mostly still preliminary and scarce (Ryaboshapko *et al.*, 1999, Tarrason *et al.*, 1997), and the modelled estimates of atmospheric deposition of heavy metals and POPs are thus uncertain.

4.2.2. Atmospheric deposition

The following estimates of atmospheric deposition are based on measurements made at coastal stations and on model calculations based on the emissions data officially reported by the EMEP countries combined with meteorological data.

Nitrogen

Nitrogen deposition on the Baltic Marine Area decreases from south to north. According to the model calculations, annual nitrogen deposition on the Belt Sea/Kattegat subregion amounts to about 1,100 mg N/m², while that on the Gulf of Bothnia is only about 200 mg N/m². In the case of ammonia, the gradient from south to north is even stronger. Calculated for the Baltic Marine Area as a whole, deposition of nitrogen oxide and ammonia is virtually equivalent (Bartnicki *et al.*, 2000). Detailed maps of the atmospheric deposition of nitrogen oxide and ammonia in 1998 are shown in Figures 4.4 and 4.5 (calculated by MSC-W; Tarrason and Schaug, 1999).

Calculated total deposition of nitrogen on the Baltic Marine Area has decreased slightly from 230,000 tonnes in 1994 to 210,000 tonnes in 1997 (Tarrason and Schaug, 1999). Deposition of nitrogen oxide is influenced by sources more remote than those influencing ammonia deposition. According to the model calculations, emissions from Germany and Poland were the main sources of nitrogen oxide deposition on the Baltic Marine Area in 1997, while Germany, Denmark and Sweden were the main sources of ammonia deposition (Bartnicki *et al.*, 2000). Other Baltic Sea States and non-Baltic Sea States were the main sources of atmospheric deposition on the various subregions of the Baltic Marine Area. Total atmospheric deposition of nitrogen on the whole Baltic Sea catchment area amounted to 1,766,000 tonnes in 1997 (Bartnicki *et al.*, 2000).

Wet deposition of nitrogen compounds on subregions of the Baltic Marine Area during the years 1994 to 1998 based on measurements at stations throughout the area are shown in Figure 4.6. No clear change can be identified during the assessment period. Compared to data from the previous five-year period, i.e. 1989–1993 (HELCOM, 1997), the total load has decreased by 20–30% in all subregions except for the southern Baltic Proper, where the level of deposition has remained unchanged.

Lead, cadmium, mercury and persistent organic pollutants

Annual lead deposition on the Baltic Marine Area in 1997 ranged from 0.1 mg/m² in the Bothnian Bay to 0.6 mg/m² in the Kattegat/Belt Sea, and totalled 150 tonnes. The main sources were emissions from Germany, Poland and United Kingdom. The model is able to reproduce the monthly variation in measurements, although these are 2- to 7-fold higher (Bartnicki *et al.*, 2000). Much higher deposition values have been reported for the years 1991–1996 based on different emission values (Bartnicki *et al.*, 1998).

Total atmospheric deposition of lead on the whole of the Baltic Sea catchment area was 1,750 tonnes in 1997 (Bartnicki *et al.*, 2000).

Wet deposition of lead on subregions of the Baltic Marine Area in 1994–1998 calculated on the basis of measurements made at stations around the Baltic Marine Area are presented in Figure 4.7. Deposition varied from year to year with no clear trend evident. According to a long time series at one station in the northern Baltic Proper, lead deposition has decreased about 20% from 1989–1993 to 1994–1998.

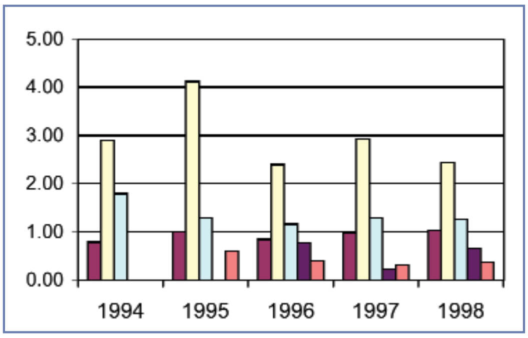


Figure 4.7
Wet deposition of lead on subregions of the Baltic Marine Area in mg/m² (Examples from coastal measuring stations).

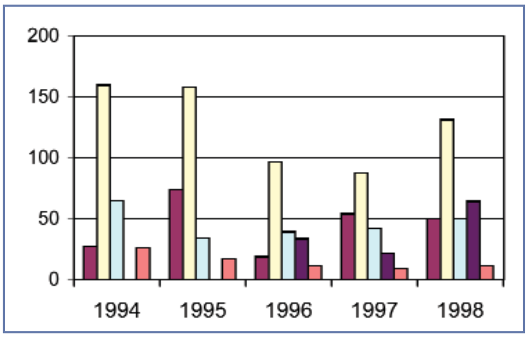


Figure 4.8
Wet deposition of cadmium on subregions of the Baltic Marine Area in µg/m² (Examples from coastal measuring stations).

Annual cadmium deposition within the Baltic Marine Area in 1997 ranged from 5 µg/m² in the Bothnian Bay to 25 µg/m² in the southern Baltic Proper, and totalled 5 tonnes. The main sources were emissions from Poland and Germany. As in the case of lead deposition calculations, there was a 2- to 9-fold discrepancy between measured and modelled cadmium values. In order to increase the accuracy of modelling, improved emission data are required. Previous calculations for the years 1991–1996 based on higher emissions yielded much higher deposition values (Bartnicki *et al.*, 1998).

Total atmospheric deposition of cadmium on the whole of the Baltic Sea catchment area amounted to 92 tonnes in 1997 (Bartnicki *et al.*, 2000).

Wet deposition of cadmium on subregions of the Baltic Marine Area in 1994–1998 calculated on the basis of measurements made at stations throughout the area is presented in Figure 4.8. Deposition varied from year to year with no clear trend evident. According to a long time series at one station in the northern Baltic Proper, cadmium deposition has decreased about 30% between 1989–1993 and 1994–1998.

Annual mercury deposition within the Baltic Marine Area ranged from 7 µg/m² in the Bothnian Bay to 25 µg/m² in the Kattegat/Belt Sea, and totalled about 5 tonnes. The main sources were emissions from Germany, Poland and Denmark. Natural emissions are also considerable. Observed and calculated mean annual mercury concentrations in precipitation only differed by a factor 2 or less (Bartnicki *et al.*, 2000).

Table 4.7
Total land-based inputs of nitrogen (total-N), phosphorus (total-P) and heavy metals to the Baltic Marine Area in 1995.

	Total-N tonnes/yr	Total-P tonnes/yr	Pb tonnes/yr	Cd tonnes/yr	Hg tonnes/yr
Monitored rivers	580,800	26,800	300	16.4	11.5
Riverine and direct load	760,740	37,650	337	23.6	13.3
Atmospheric deposition	230,000	n.i.	150*	5*	5*
Total input	990,000	40,000	490	30	20

n.i.: no information
* figures refer to 1997

Total atmospheric deposition of mercury on the whole of the Baltic Sea catchment area amounted to 53 tonnes in 1997 (Bartnicki *et al.*, 2000).

Total deposition of lindane on the Baltic Marine Area in 1997 was calculated to be approx. 3.4 tonnes. The main sources were emissions from France, United Kingdom and Belgium. Although only few data are available for lindane, they are in good agreement with the modelling results (Bartnicki *et al.*, 2000).

Recent MSC-E calculations provide the following estimates of POP deposition on the Baltic Marine Area in 1996: PCB 715 kg; B[a]P 7 tonnes; 2,3,4,7,8 PeCDF 65 g calculated as I-Teq. (Pekar *et al.*, 1999). Future improvement in emission inventories and measurements can be expected to provide more accurate figures than these preliminary estimates.

4.3. Total land-based pollution load

The majority of the anthropogenic load reaches the Baltic Marine Area via rivers and direct discharges from municipal and industrial sources, although atmospheric deposition on the Baltic waters is also an important source. The waterborne and airborne loads derive from discharges/emissions from municipalities and industrial plants as well as from the intensive agricultural production within the Baltic Sea catchment area. Total land-based inputs of nitrogen, phosphorus, lead, cadmium and mercury are summarized in Table 4.7. Atmospheric deposition accounts for about 25% of nitrogen inputs to the Baltic Marine Area, a few percent of phosphorus inputs, approx. 30% of lead inputs and approx. 20% of cadmium and mercury inputs. As regards heavy metals the riverine and direct loads are very uncertain. With all five substances, deposition rates are highest in the southwestern part of the Baltic Marine Area, decreasing towards the north-east.

Cleaning of beaches in Grønsund of oil washed ashore from the Baltic Carrier accident in 2001.

4.4. Atmospheric inputs from international ship traffic

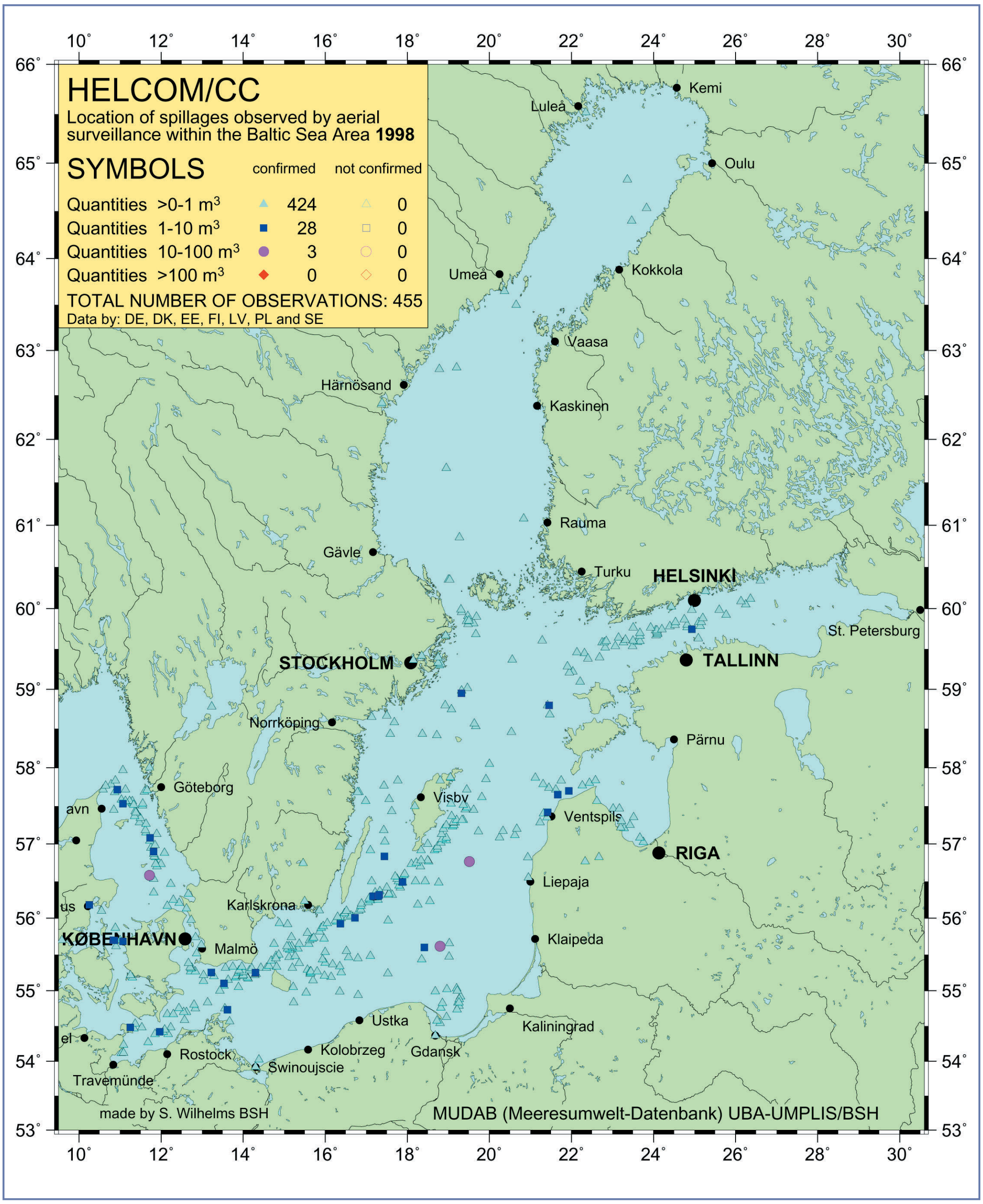
Emissions from international ship traffic in the Baltic Marine Area is a major source of nitrogen oxides. According to estimates made by Lloyd’s Register, emissions from international ship traffic amounted to 352,000 tonnes nitrogen oxides in 1990 – more than emissions from most of the Baltic Sea States, except from the Russian Federation, Poland and Germany (Bartnicki *et al.*, 2000).

Calculated total deposition of nitrogen on the Baltic Marine Area has decreased slightly during the 1990s, whereas the emissions from ship traffic have remained unchanged (Jonson *et al.*, 2000). According to model calculations, international ship traffic was the second largest source of nitrogen oxide deposition on the whole of the Baltic Marine Area in 1997 (Bartnicki *et al.*, 2000).

4.5. Illegal oil discharges

In spite of the designation of the Baltic Marine Area as a “special area” under MARPOL 73/78, in principle prohibiting the discharge of oil/oily mixtures from all ships, many illegal oil discharges are observed in the Baltic Marine Area. The locations, number and size of illegal oil discharges observed by national surveillance flights in 1998 are indicated in Figure 4.9. The highest incidence is seen in

Figure 4.9



the Sound, south and southeast of Sweden, and in the Gulf of Finland. The observations closely correlate with the major shipping lanes in the Baltic Marine Area and thus reflect the intensive ship traffic in these areas, which, however, are also the areas where regular aerial and remote sensing surveillance is carried out.

The Baltic Sea States carry out regular aerial surveillance activities to monitor illegal discharges at sea. An assessment of national oil spill reports over the last 12 years indicates an annual rate of approx. 500–700 illegal oil discharges.

The actual number of illegal discharges in some areas is undoubtedly higher than indicated on Figure 4.9, however, since a considerable number of observations of illegal oil discharges are usually detected during Coordinated Extended Pollution Control Flights (CEPCO Flights), where several Baltic Marine Area States carry out joint continuous surveillance of a busy shipping lane for 24 hours or more. It is not uncommon for up to ten such observations to be made during CEPCO flights.

4.6. Accidental oil spills

From 1969, about 40 major accidental oil spill accidents exceeding 100 tonnes have been registered in the Baltic Marine Area (HELCOM, 1996a). During the period 1993–1998, one accident occurred in the Baltic Marine Area involving the spillage of 180 tonnes oil (1995 Hual Trooper, the Sound, Sweden). Besides the acute effects of oil spills such as polluted beaches and seabird mortality, long-term effects of these spills have also been recorded, e.g. locally increased levels of PHC contamination in sediments. Following the sinking of the passenger ferry “Estonia” in 1994, the Finnish authorities carried out several underwater operations to remove bunker oil, etc. from the wreck.

From the statistical point of view, the main risks to ships are groundings, collisions and rammings (i.e. ship-object collisions) (HELCOM, 1984). Statistically, the number of oil-spill accidents in the Baltic Marine Area is estimated to be 2.9 per year. A risk assessment indicates that the statistical number of oil-spill accidents will rise to 3.2 if the present oil terminal capacities are fully utilized, and to 4.9 accidents per year if plans to construct new terminals and to enlarge existing terminals are implemented, and the terminals fully utilized. As a consequence, the statistically predicted amount of oil spilled annually will increase to 775 tonnes and 1,475 tonnes, respectively (HELCOM, 1996a, HELCOM, 1996b).

At the moment, only Poland is carrying out offshore activities in the Baltic Marine Area (two installations located in the Polish Exclusive Economic Zone). Germany has ceased its offshore activities and removed the offshore installations. Some other Baltic Sea States are also investigating their possibilities in this field, however.

Eutrophication and related effects 5

5.1. Baltic Proper

By: Convenor: G. Nausch. Co-convenor: W. Krzyminski

The location of the monitoring stations in the Baltic Proper is illustrated in Figure 5.1.

5.1.1. Meteorological, hydrological and hydrographic forcing

By: J. Elken, W. Matthäus, W. Krzyminski, J. Dubra

Surface layer

Weather conditions during the assessment period 1994–98 were generally warmer than the long-term average (cf. Chapter 3). The summers of 1994 and 1997 were unusually warm and calm (Table 5.1). Because of low wind-induced mixing and thermocline erosion, the upper mixed layer was only about 10–20 m thick in July–August in the open sea during these years. In the summer of 1997, however, moderate but persistent easterly winds caused upwelling of cold water along the eastern and southern coasts for several weeks. During the colder summer of 1998, stronger winds eroded the thermocline more effectively, and the thickness of the upper mixed layer exceeded 20 m in the Gotland Basin and 40 m in the Gdansk Basin. As a result, the maximum surface temperature in 1998 approached the long-term mean and was less than 16°C in the Gotland Deep.

The winters were generally mild (cf. Figure 3.5), and the minimum sea surface temperature did not fall below 2°C in 1993, 1995, 1997 and 1998. The cold winter of 1996 was severe enough to freeze some parts of the Northern Gotland Basin. As usual, the remains of winter water were found above the halocline during the subsequent summer. After the cold winters of 1994 and 1996, the summer minimum temperature of the intermediate layer in the Gotland Deep was less than 2.6°C as compared with more than 3.5°C after mild winters.

In summer 1997, the southern Baltic Proper received unusually high riverine runoff during short periods due to the exceptional flooding of the Oder and Vistula river basins. The Oder floodwaters (about 6.5 km^3 of additional water) covered the entire Pomeranian Bight up to Kolobrzeg and the Island of Rügen during the discharge of flood crests. Mainly westward winds guided the outflowing riverine runoff along the German coast into the Pomeranian Bight and the Arkona Sea. The maximum extension of the Vistula waters reached the geographical border of the Gdansk Basin. The overall impact of floods was moderate (Trzosinska and Andrulewicz, 1998) as compared with the annual discharge and had no long-term consequences for the Baltic ecosystem (Siegel *et al.*, 1998).

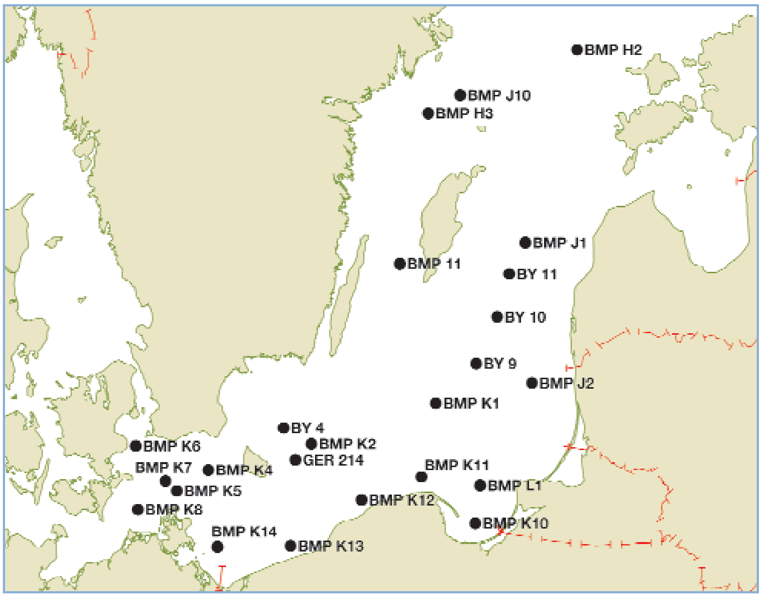


Figure 5.1
Map indicating the stations studied in the Baltic Proper. BMP: Stations under the Baltic Monitoring Programme. BY: Stations of the International Baltic Year.

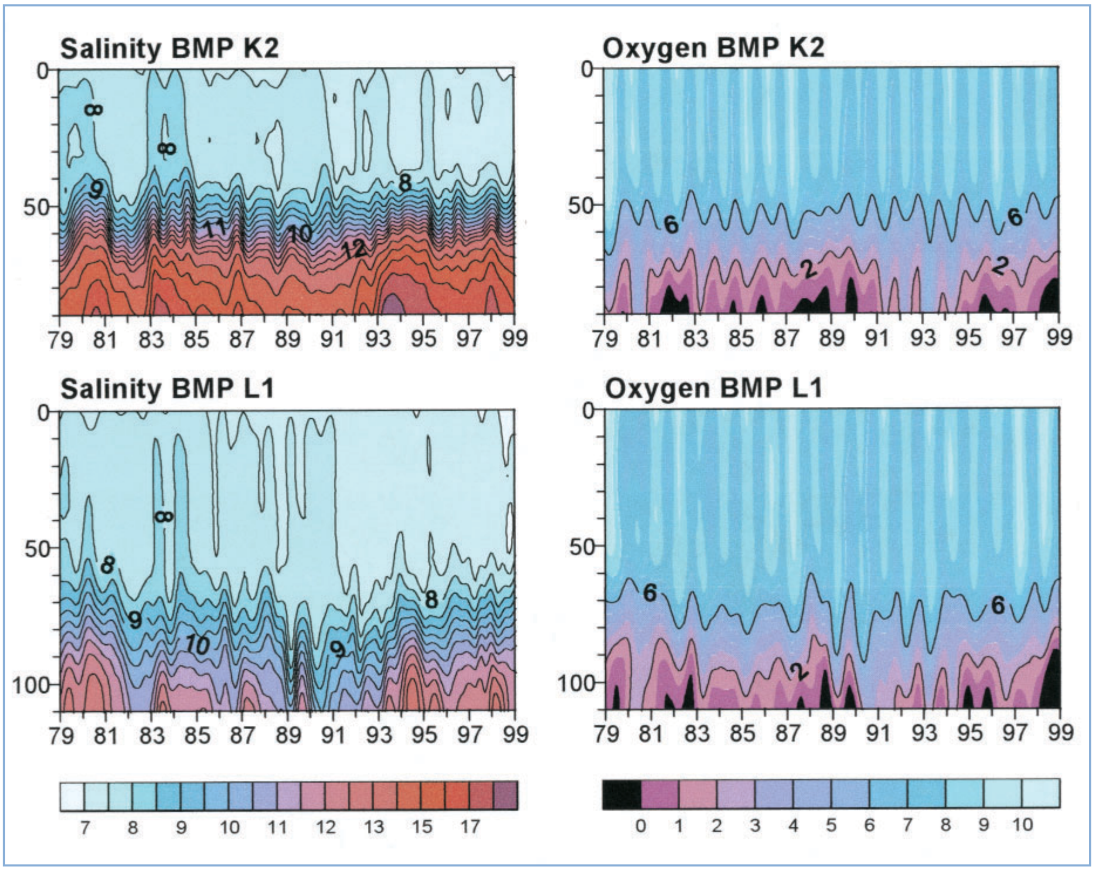
Halocline and deep layers

Permanent saline stratification of the Baltic Proper is formed by intermittent inflow of saline water from the Belt Sea and the Sound and by freshwater surplus from the adjacent basins (Gulfs of Bothnia,

Year	BMP K2		BMP L1		BMP J1		BMP H2	
	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer
1993	3.6	16.4	2.5	16.8	2.9	16.8	2.4	16.4
1994	1.8	21.3	1.7	21.9	1.6	24.0	0	19.8
1995	3.2	19.6	3.1	20.2	2.8	19.3	2.3	19.0
1996	0.1	17.8	0.8	18.5	0.6	19.5	0	17.0
1997	2.8	21.7	2.7	22.2	2.3	21.2	1.6	na
1998	3.5	15.4	2.8	16.4	2.9	15.7	3.0	15.1

Table 5.1
Annual minimum and maximum temperatures (°C) in the Bornholm Basin (BMP K2), Gdansk Basin (BMP L1), Eastern Gotland Basin (BMP J1) and Northern Gotland Basin (BMP H2) during the assessment period.

Figure 5.2
Salinity (left) and oxygen (right) as a function of time (1979–99) and depth (m) in the Bornholm Basin (BMP K2) and Gdansk Basin (BMP L1). Contours and colour scale are PSU for salinity and ml/l for oxygen. Black areas indicate anoxic layers.



Finland and Riga) and draining rivers (Vistula, Oder, Nemunas etc.). Inflowing saline water is spread and mixed in the sequence of deep sub-basins of the Baltic Proper. Starting from the Bornholm Basin, the basins work as buffers where incoming water may be trapped by the sill depth. Thereafter it is subject to several transformations before it can spread further to the next basin (Elken, 1996). Overflow from one sub-basin to another is dependent on the halocline height above the sill depth of the connecting channel and the density difference between the basins (Stigebrandt, 1983; Omstedt, 1990) as well as on the direction and speed of the dominant winds (Krauss and Brügge, 1992). Mixing across the halocline (Axell, 1998) is an essential component of the hydrographic and ecological cycle. Despite recent advances in numerical modelling (Lehmann and Hinrichsen, 2000), complex deepwater dynamics and related effects still have to be diagnosed on the basis of actual measurements.

The Bornholm Basin deep water exhibits a regular seasonal cycle in temperature, salinity and oxygen. During the assessment period, mean temperature increased from about 4°C to 8°C, while mean salinity decreased from about 17.5 PSU to 16.5 PSU, and oxygen content decreased from 2.5 ml/l to about zero (cf. Chapter 3.4) The general trend was interrupted by the inflow of saline, oxygen-rich and unusually warm water in autumn 1997, causing an increase in deep water temperature to 11.6°C (Matthäus *et al.*, 1999) and salinity to more than 17 PSU (Figure 5.2). Near-bottom oxygen concentrations decreased rather rapidly after the major inflow. By the end of 1995, the bottom layer containing hydrogen sulphide extended up to 75 m. Temporary improvement of oxygen conditions occurred following moderate inflows in autumn 1996 (Matthäus *et al.*, 1996) and 1997. A new anoxic period started in 1998.

In the Gdansk Deep, near-bottom salinity increased to 13.5 PSU in January 1994 as a result of inflow from the Bornholm Basin at the end of 1993 that also significantly increased the oxygen concentration (Figure 5.2). However, the enhanced halocline hindered cross-isopycnal mixing, thus leading to occasional oxygen deficiency as early as March of the same year. Parallel to the decrease in salinity, oxygen concentration decreased significantly resulting in oxygen depletion by the end of 1996. In 1997, strong mixing and inflow events improved oxygen conditions. In 1998, anoxic conditions recurred, and the hydrogen sulphide layer extended up to 90 m (IMGW, 1998), a process that started when the vertical gradient of salinity in the halocline increased as a result of deepening of the upper layer due to high Vistula river discharge in winter 1998.

In the Eastern Gotland Basin, a long period of stagnation since 1977 was terminated by the major North Sea water inflow in January 1993 (Matthäus and Lass, 1995) and smaller inflows in 1993/94. These completely renewed the deep water of the Gotland Deep by May 1994 and of the Northern

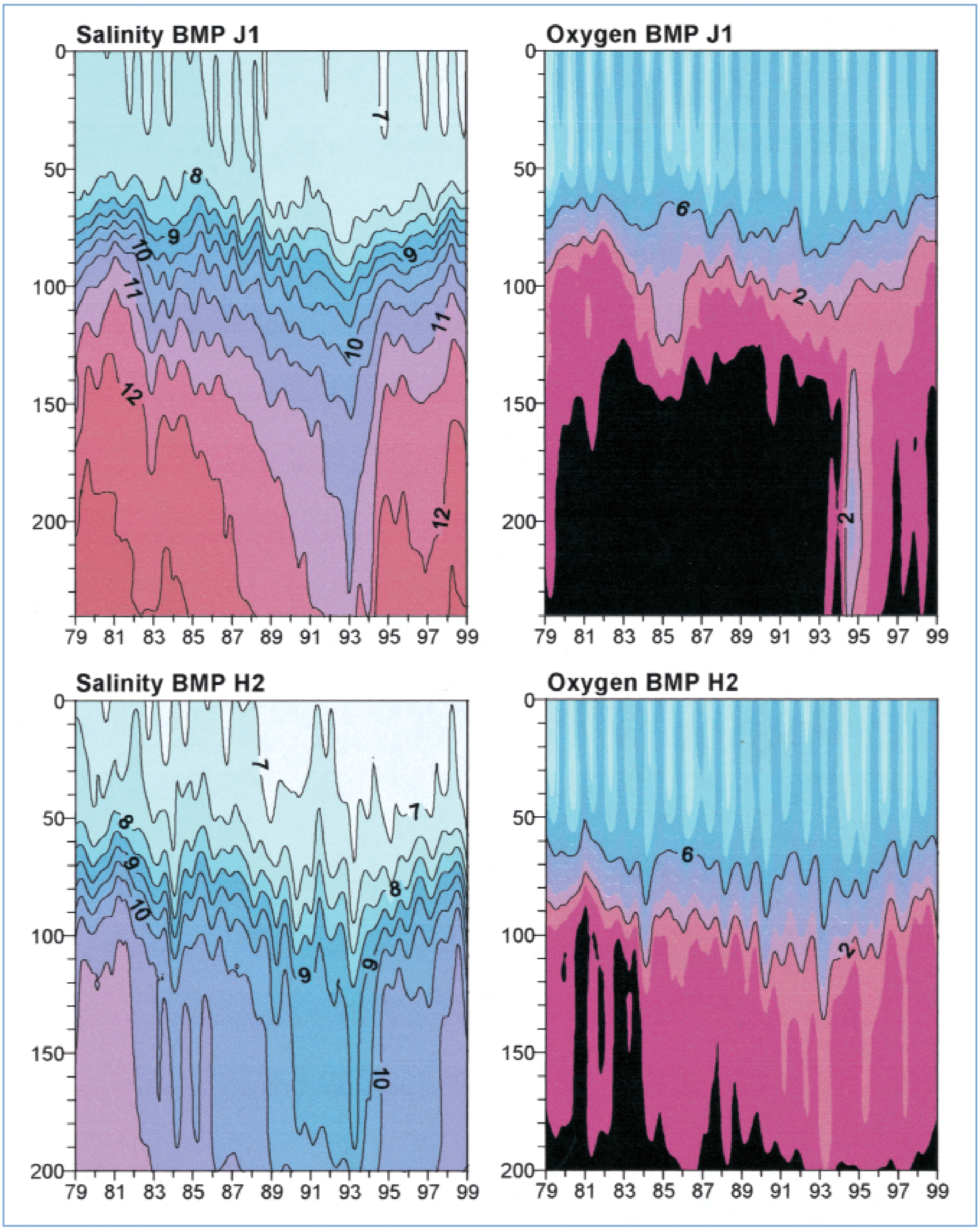


Figure 5.3
Salinity (left) and oxygen (right) as a function of time (1979–99) and depth (m) in the Eastern Gotland Basin (BMP J1) and Northern Gotland Basin (BMP H2). Contours and colour scale are as for Figure 5.2.

Gotland Basin by November 1994, and increased bottom salinity west of Gotland in October 1994. The amount of new water entering the Gotland Deep was so great that deep isohalines were raised by some 80–100 m, and the halocline was raised some 20–30 m as compared with winter 1993 (Figure 5.3). Isohaline movements of the same amplitude were also detected in the Northern Gotland Basin. The 150–240 m layer of the Gotland Deep, which had previously been anoxic, was filled by oxygen-rich water.

Following the major inflow in 1993, the Gotland Deep received smaller amounts of saline water each year that kept the bottom salinity above 12 PSU from May 1994 onwards. However, the amount of incoming water was small, and the water was oxygen-depleted, thus leading to reappearance of hydrogen sulphide in 1996. Anoxic and oxic conditions alternated at approximately six-month intervals between 1996 and 1998 due to the aeration effects of small inflows. This cycle was not changed by an inflow of unusually warm water into the Eastern Gotland Basin between January and April 1998. Besides noticeable salinity effects, the event increased the deep-water temperature from 4.3–5.3°C (a typical range for 1993–1997) to 7.4°C (Matthäus *et al.*, 1999).

In the Northern Gotland Basin, the volume of low-oxygen (<2 ml/l) water has increased since 1994 due to stronger density stratification preventing direct aeration by vertical mixing. At the same time, the halocline has moved upwards causing oxygen depletion west of Gotland due to the fact that these bottom layers receive waters from the Northern Gotland Basin at a depth of around 100 m. In 1994–98 there was a good negative correlation between salinity and oxygen in the Western Gotland Basin, i.e. the increase in bottom salinity caused by inflow from the Northern Gotland Basin was accompanied by a reduction in oxygen concentration.

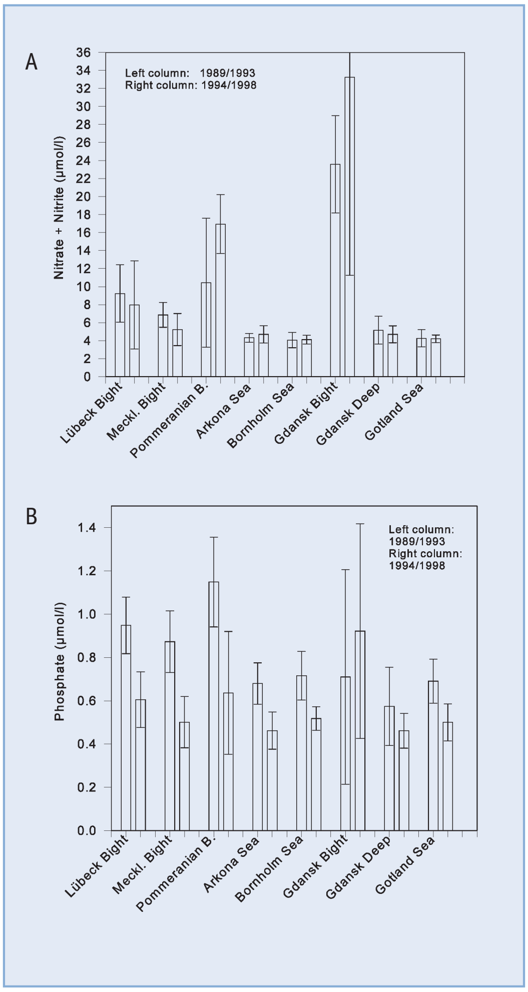


Figure 5.4
Mean nitrate + nitrite (A) and phosphate (B) concentrations in the mixed surface layer (0–10 m) in February in the western and central Baltic Marine Area during the periods 1989–93 and 1994–98.

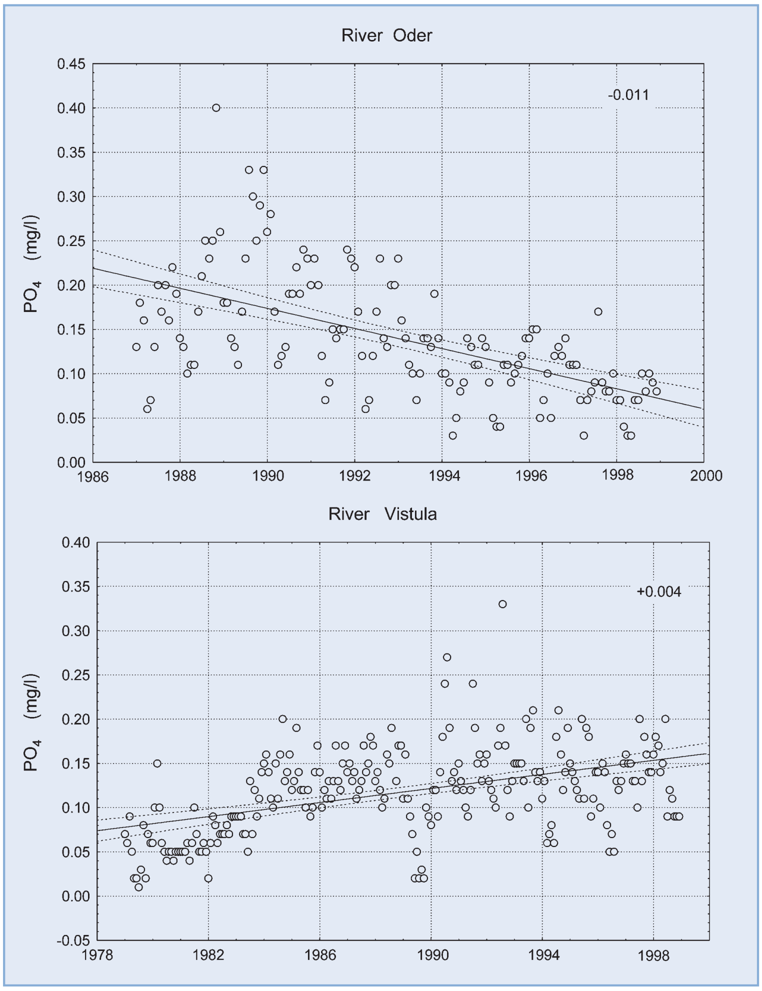


Figure 5.5
Phosphate outflow via the rivers Oder (1986–98) and Vistula (1979–98) (Niemirycz, 1999). Full line: Linear regression; broken lines: 0.95 confidence intervals. The statistical significance of the regression is indicated in the upper right-hand corner.

5.1.2. Hydrochemistry

By: G. Nausch, E. Lysiak-Pastuszak

Surface layer

Nutrient concentrations in the whole of the Baltic Proper are characterized by a pronounced seasonality with high concentrations in winter and a decrease to around the detection limit during the period of high biological productivity, beginning in spring and ending in late autumn (HELCOM, 1996). Nutrient trend studies can only be conducted in the surface layer in winter, when the biological activity is low, and nutrient concentrations are high (Nehring and Matthäus, 1991; HELCOM, 1996), the assumption being that a steady state has developed at this time between microbial mineralization, low biological productivity and high vertical exchange and mixing. This steady state is most discernible in the Eastern and Western Gotland Basins and lasts 3–4 months, where values characteristic of the winter situation are sometimes measured as late as in early April. In the other basins, nutrient concentrations start to diminish due to the onset of spring phytoplankton bloom already in March, whereas nutrient depletion is distinctly later in the Bornholm and Arkona Seas than in the Mecklenburg, Pomeranian and Gdansk Bights. The bloom starts when the upper mixed layer becomes shallower than the light penetration depth, which occurs earlier in shallower regions than in the deeper areas (Wasmund *et al.*, 1998).

In order to better differentiate the trophic status of all regions under investigation, only winter values from February were used. Phosphate and nitrate (+nitrite) are compared for the last two assessment periods in Figure 5.4. No clear trend is evident for the period 1994–98, and winter concentrations fluctuate around an average.

When comparing the two periods, however, especially in the coastal areas, but also in the central Baltic Proper, a significant decrease in phosphate concentration can be observed. Measures taken to reduce phosphate inputs from point sources in the catchment of the Baltic Marine Area seem to be effective in the coastal areas. In the open sea, however, these may also be influenced by changes in internal processes. A clear relationship can be identified to riverine input: The phosphate concentration in the Oder river has decreased between 1986 and 1999 (Figure 5.5). This general reduction in phosphate concentrations

	1979–83	1984–88	1989–93	1994–98
Arkona Sea	11.1 (13)	6.6 (29)	6.3 (60)	10.2 (124)
Bornholm Sea	7.4 (20)	6.3 (36)	5.7 (46)	7.9 (84)
Gdansk Deep	10.4 (15)	9.8 (25)	9.0 (16)	10.2 (24)
Eastern Gotland Sea	6.2 (15)	6.5 (33)	6.0 (79)	8.4 (121)
Landsort Deep	–	–	8.0 (14)	9.5 (35)

Table 5.2
Molar N:P ratios (NO_2 $+NO_3$ and PO_4) in the surface layer (0–10 m) during winter
Number of measurements in parentheses

is only not evident in the inner part of the Gulf of Gdansk, but this influences the whole bight, (Figure 5.4). It is attributed to increased riverine input of phosphate via the Vistula river (Figure 5.5) due to less effective reduction of phosphate discharge from point sources and inertia of the river catchment areas dominated by diffuse nutrient input resulting in delayed removal of phosphorus (Sapek, 1998; Tonderski, 1997). Similar to the period 1989–93 (Trzosinska and Lysiak-Pastuszak, 1996), seasonal variation and variation in winter concentrations were highest in the Pomeranian and Gdansk Bights, which are most exposed to discharges of phosphorus from land sources (IMGW, 1995–99). Finally, recent investigations by Matthäus *et al.*, (2000) indicate that a new equilibrium has established in the upper layer of the Baltic Proper, and that a further decrease in phosphate concentrations is unlikely.

Primary production in the Baltic Proper can be limited by nitrogen compounds, the most important of which is nitrate. In contrast to phosphate, in most of the investigated areas no significant differences between the two assessment periods can be seen (Figure 5.4). On the contrary, the winter nitrate concentrations are higher in the inner Gulf of Gdansk and in the Pomeranian Bight. Several reasons can be mentioned for the lack of reduction. Atmospheric input still plays an important role in the central Baltic Proper despite initial indications of a decrease in input from this source (HELCOM, 1997). Annual nitrogen input through nitrogen-fixing cyanobacteria in summer can not be overseen (Wasmund *et al.*, 2000). The most important reason, though, is that as nitrate mainly originates from diffuse sources in the catchment area (Figure 5.6), input to the coastal areas is closely related to freshwater runoff (Bachor, 1996; Nausch and Schlungbaum, 1995; Pastuszak *et al.*, 1996; Meyer and Lampe, 1999; Nausch *et al.*, 1999).

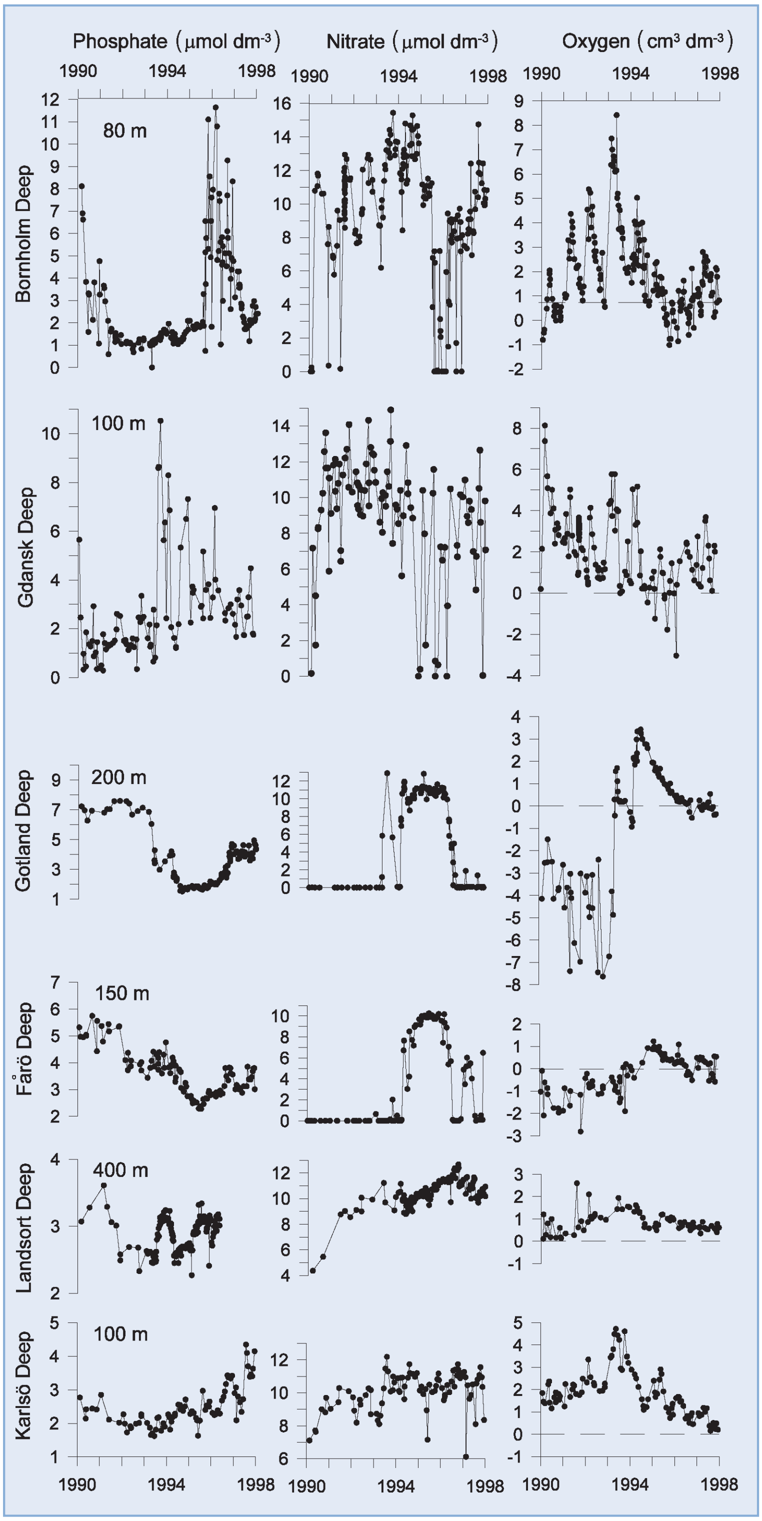
The molar N:P ratio is used as an indicator of the principal limiting factor. Phytoplankton incorporate nitrogen and phosphorus in a ratio of around 16:1 (Redfield *et al.*, 1963). This ratio is disturbed in the Baltic Proper due to denitrification processes in the deeper basins and at the sediment surface. The N:P ratios for all four assessment periods (1979–98) are summarized in Table 5.2. The above-mentioned reduction in phosphate concentration and the unchanged nitrate concentration led to higher N:P ratios in 1994–98, but still much below the Redfield value. During the spring bloom, the potential biomass production based on available phosphate is nearly twice the potential biomass production based on available inorganic nitrogen compounds assuming that these nutrients are incorporated by phytoplankton in the ratio 16:1 (Wasmund *et al.*, 1998). Phytoplankton development in the spring is thus nitrogen-limited. This is further evidenced by the nutrient situation observed in May, when the surface layer is already completely depleted of nitrate and phosphate is still present in a concentration of around 0.1 mmol/l (Matthäus *et al.*, 1999). Later in the year the nitrogen deficit is compensated for by cyanobacteria, which are able to assimilate atmospheric nitrogen, whereafter the primary production becomes phosphorus-limited.

A completely different picture exists in the

Figure 5.6
Nitrate outflow via the rivers Oder (1986–98) and Vistula (1979–98) (Niemirycz, 1999). Full line: Linear regression; broken lines: 0.95 confidence intervals. The statistical significance of the regression is indicated in the upper right-hand corner.

bights receiving major riverine inputs. Whereas the phosphate concentrations in the Pomeranian Bight and the inner Gulf of Gdansk are only slightly higher than in the open Baltic Sea, the nitrate concentrations are remarkably elevated (Figure 5.4). As a consequence the N:P ratios are much higher, reaching as much as 130:1 in the Pomeranian Bight and 73:1 in the Gulf of Gdansk in spring (Trzosinska and Lysiak-Pastuszak, 1996; IMGW, 1995–1999; Lysiak-Pastuszak and Drgas, 1999). In contrast to the open Baltic Sea, these bights are phosphorus-limited in spring. The nitrogen load can only be partly utilized for biomass production. After dilution by hydrographic processes, the nitrate contributes to the eutrophication of the Baltic.

Figure 5.7
Variation in phosphate, nitrate and oxygen/hydrogen sulphide (expressed as negative oxygen equivalents) in the deep waters of the Baltic Proper between 1990 and 1998.



Deep water

Nutrient conditions in the deep basins in the 1990s are mainly characterized by the major saltwater inflow in 1993, the inflow events in 1993 and 1994 and the subsequent stagnation period, which shows several peculiarities.

The Bornholm Basin, the westernmost of the deep basins, is subject to relatively frequent renewals due to vertical and horizontal exchange processes, resulting in oxygen input to the deep water from the Mecklenburg Bight and the Arkona Basin each year. Thus salt and oxygen-rich water was also able to penetrate into the Bornholm Basin prior to the major saltwater inflow in 1993. Anoxic conditions only prevailed in 1990. As a result of the inflow, however, the oxygen concentrations increased to 7 ml/l by the end of March 1993 (Nehring *et al.*, 1994). The deep water in the Bornholm Basin was thus oxic throughout almost the whole of the first half of the 1990s, as is also evidenced by the nutrient conditions.

In the presence of oxygen, phosphate is partly bound to sediment and suspended particles as an iron-III-hydroxophosphate complex, resulting in phosphate concentrations of only 1–2 µmol/l (Figure 5.7). If the redox state changes, the complex is reduced by hydrogen sulphide, and phosphate and iron-II-ions are liberated as reflected by an increase in their contents in the near-bottom water, especially in 1996, but also in 1998.

Areas such as the Bornholm Basin where the redox regime varies frequently, are characterized by intense phosphate accumulation in the presence of hydrogen sulphide, e.g. phosphate concentrations rise up to 12 µmol/l (Figure 5.7). The concentration of inorganic nitrogen compounds is also strongly influenced by the presence or absence of oxygen/hydrogen sulphide. Under oxic conditions, inorganic nitrogen compounds are present almost exclusively in the oxidized form as nitrate. Under anoxic conditions, the available nitrate is denitrified to dinitrogen gas. In contrast, ammonium, which is liberated by mineralization processes in both the water column and the sediments, cannot be oxidized and hence accumulates while nitrate vanishes completely.

The situation in the *Gdansk Deep* is very similar. Owing to inflow from the Bornholm Basin via the Stolpe Channel, water renewal processes are comparable to those in the central Baltic Proper. There is also evidence that vertical mixing can occasionally occur down to the bottom. Throughout the first half of the 1990s, conditions here were mainly oxic, with a low phosphate concentration of 1–2 µmol/l and a nitrate concentration of 10–14 µmol/l (Figure 5.7). Beginning from the end of 1994, anoxic situations appeared repeatedly interrupted by oxic phases, and nutrient concentrations fluctuated in accordance with the redox regime as described in the Bornholm Basin (Figure 5.7).

The water renewal in the first half of the 1990s and the subsequent development of a new stagnation period had the greatest impact on oxygen and nutrient conditions in the *Eastern Gotland Basin*. At the end of the 16-year stagnation period, which had started in 1977, the hydrogen sulphide concentration was extremely high (up to an oxygen equivalent of -8 ml/l) and the nutrient situation was characterized by the absence of nitrate and nitrite (Figure 5.7) and very high ammonium levels of up to 40 µmol/l (Nehring *et al.*, 1995). At the beginning of a stagnation period intensive phosphate accumulation normally occurs. The phosphate content of the water in the Gotland Deep thus increased from 2 to 6 µmol/l between 1977 and 1979. Thereafter, the concentration fluctuated around an average. It seems that the phosphate reserves in the surficial sediments are exhausted, and a further increase in hydrogen sulphide concentration does not result in further release of phosphate. A characteristic feature of the saltwater inflow in 1993 is its extremely fast penetration into the Eastern Gotland Basin. The nutrient distribution reacted immediately to the changes in the redox regime. The ammonium concentration decreased markedly and was negligible in June and August 1993, while the nitrate concentration increased and the phosphate concentration decreased (Nehring *et al.*, 1994). But already in November 1993 the layer below 200 m depth was again anoxic. Only the inflows at the end of 1993 and beginning of 1994 resulted in a longer improvement of the oxygen situation in the Gotland Deep. The current stagnation period exhibits several peculiarities compared to earlier stagnation periods, however. After the first appearance of hydrogen sulphide in March 1996 (Matthäus *et al.*, 1996) the thickness of the anoxic layer was 100 m during the second half of 1996, 1997 and 1998, extending from the bottom to a depth of about 150 m. The anoxic conditions were interrupted every winter and spring by the occurrence of low oxygen concentrations. The nutrients responded as described previously by Matthäus *et al.* (1999). From mid 1998 onwards, permanent anoxic conditions set in.

The development in the *Fårö Deep* can be compared to that in the Gotland Deep except that due to the presence of an additional sill, the processes are delayed and their intensity reduced. Although hydrogen sulphide concentrations were much lower in the Fåro Deep at the end of the previous stagnation period, the increase in oxygen after the water renewal was only moderate. The situation was most favourable here in 1994/1995, as with the whole of the Baltic Proper. The highest nitrate concentrations and lowest phosphate concentrations were measured at that time (Figure 5.7). Thereafter the stagnation period started to stabilize, as reflected by increasing phosphate and ammonium concentrations and nitrate depletion, initially interrupted by small oxygen pulses.

In the *Western Gotland Basin* (Landsort Deep and Karlsö Deep) the development was typical for the first part of stagnation periods in this basin (Matthäus, 1995). The oxygen concentration has decreased continuously since 1993 (Figure 5.7) leading to the lowest oxygen content since the mid 1980s. The nitrate concentration was stable over a longer period at around 10 µmol/l. At the end of 1998, however, oxygen conditions deteriorated so much that denitrification was initiated (Goering, 1968), resulting in a decrease in nitrate concentration as is clearly evident in 1999. The parallel increase in phosphate concentrations, especially in the Landsort Deep, was of lower intensity as compared to the Eastern Gotland Basin, probably due to the different structure of the sea floor.

5.1.3. Structure and function of the pelagic ecosystem

Phytoplankton

By: L. Edler, E. Sahlsten, N. Wasmund, B. Karlson

Seasonal variation of species composition. The spring bloom during the period 1994–98 generally started at the end of March in the southern Baltic Proper and some weeks later further north, and culminated in April or May. Diatoms together with dinoflagellates formed the bloom in the south, whereas dinoflagellates dominated in the northern parts. Large amounts of heterotrophic dinoflagellates succeeded the spring bloom in 1997. In 1998, the spring bloom in April began with a short peak of diatoms, followed by a dominance of the dinoflagellate *Peridiniella catenata*.

Cyanobacteria blooms. At the beginning of July 1994, cyanobacteria started to accumulate at the surface in the Gotland Sea and somewhat later in the southern part of the Baltic Proper. By the end of July 1994, the entire Baltic Proper (approx. 100,000 km^2) was covered by filamentous cyanobacteria, mainly *Nodularia spumigena*. Surface accumulations of cyanobacteria started to develop as early as the end of June 1995, and continued through August. In 1996, surface accumulations of filamentous cyanobacteria started to develop at the end of July and continued throughout September. Warm weather starting early in summer 1997 contributed to the considerable bloom of filamentous cyanobacteria, which covered large parts of the Baltic Proper. Toxic *N. spumigena* were very common. The summer of 1998 was characterized by blooms of *Aphanizomenon* sp. The cyanobacteria blooms occurring during the period were generally toxic.

Dinoflagellates. In early summer of 1994, the potentially toxic genera of *Chrysochromulina* and *Dinophysis* were present in the Baltic Proper. During May–June 1995, the genus *Chrysochromulina* was common along the German coasts. In 1995, large numbers of the dinoflagellates *D. norvegica, D. acuminata* and *Prorocentrum minimum* occurred concomitantly with the cyanobacteria blooms. In August 1995 and 1996, the dinoflagellate *Heterocapsa triquetra* coloured the surface water in many coastal areas of the Baltic Proper. September 1997 was characterized by the occurrence of the potentially toxic dinoflagellate *Alexandrium ostenfeldii*.

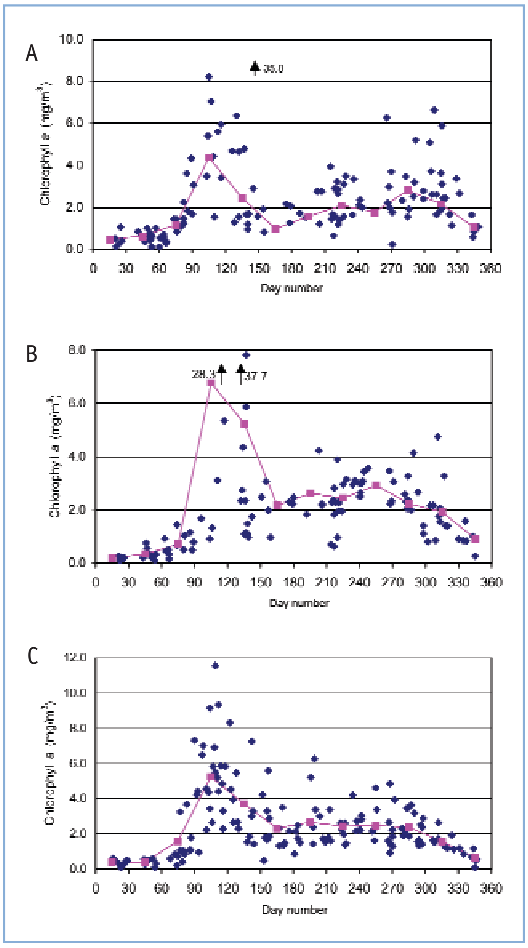


Figure 5.8
Mean seasonal variation of depth-integrated (0–10 m) chlorophyll a concentration in (A) the Bornholm Sea (BMP K2), (B) the Eastern Gotland Sea (BMP J1) and (C) the Northern Baltic Proper (BMP H3) during the period 1994–1998.

In 1998, *H. triquetra* formed a spectacular bloom in the northern part of the Baltic Proper. There are signs indicating that this species has become more common in the Baltic during recent years.

In 1994 and 1995, the autumn bloom of diatoms was dominated by *Coscinodiscus granii*. Cyanobacteria were common until November 1996, but no autumn diatom bloom developed that year. During autumn 1997, *P. minimum* formed considerable blooms in large parts of the Baltic Proper and was very common during early autumn 1998 in the southern part of the Baltic Proper. No diatom autumn bloom developed in 1998 either.

Chlorophyll

Seasonal variation. The seasonal development of phytoplankton biomass, reflected in the chlorophyll a concentration in the water, is characterized by three peaks (spring, summer and autumn blooms) in the Baltic Proper. These blooms occur every year but may be overlooked if the sampling network is insufficiently dense. Combining the five years of the assessment period into one annual time series provides a general picture of the seasonal variation (Figure 5.8). The peaks of the blooms occurred more or less at the same time as in previous assessment periods, i.e. spring bloom in March-April. In the Eastern Gotland Sea (BMP J1) there seems to be a delay until April/May, possibly due to the greater distance to the coast at this station (Wasmund *et al.*, 2000). Despite the spectacular surface accumulations of filamentous cyanobacteria in summer, only occasional chloro-

Table 5.3
Mean, standard deviation (SD) and number of samples (n) of all chlorophyll a data available from the Baltic Proper (0–10 m depth) for spring (March–May), summer (June–September), autumn (October–December) and winter (January–February) during the period 1979–98.

		1979–1983	1984–1988	1989–1993	1994–1998
Spring	Mean	1.98	2.79	2.63	3.28
	SD	1.50	2.96	2.98	4.35
	n	116	175	366	320
Summer	Mean	2.40	2.20	2.42	2.51
	SD	1.18	1.15	1.21	1.25
	n	182	184	295	342
Autumn	Mean	1.85	2.28	2.51	2.07
	SD	1.11	1.61	1.29	1.28
	n	103	134	173	231
Winter	Mean	0.93	0.46	0.86	0.53
	SD	0.49	0.31	0.42	0.47
	n	8	12	48	126

phyll peaks are observed in the southeastern part of the Baltic Proper. The autumn blooms are less pronounced than the spring blooms.

Mean chlorophyll *a* data from the Baltic Proper area are shown for the different assessment periods and the different seasons in Table 5.3. The seasons were defined according to the Third Periodic Assessment (HELCOM, 1996). Standard deviation is high, especially during the spring period, which reflects the wide range of data due to considerable dynamics with strong growth and decline of the spring bloom.

Long-term variation. The variation in chlorophyll a concentration in the total data set (1979–98) at selected stations is shown in Figure 5.9. It is obvious that any trend is masked by the high variation. A significant change (increase) over the whole period could only be detected in the Arkona Sea when pooling the three monitoring stations in the central Arkona Sea (Figure 5.9a). The chlorophyll a concentration also increased in the Bornholm Sea (Figure 5.9b) and the Eastern Gotland Sea (Figure 5.9c), but not significantly. These findings are supported by the comprehensive Alg@line Database (Figure 5.10). Comparison of the mean values for the different assessment periods reveals that the increase mainly occurs in the autumn (Table 5.3), as reported in the Third Periodic Assessment (HELCOM, 1996) for station BMP J1. In that assessment period, trend analysis performed using the Whirsch test yielded the same findings as a simple linear regression. The development in the Baltic Proper contrasts with that in the Kattegat/Belt Sea area, where chlorophyll a concentrations tend to decrease.

Spatial variation. The spatial variation in chlorophyll a concentrations is best recorded by the quasi-synoptic tracks of ships-of-opportunity (Leppänen *et al.*, 1995). These data are available from the homepage of the Alg@line monitoring programme coordinated by the Finnish Institute of Marine Research (http://meri.fimr.fi). By way of example, the spring data for each year of the assessment period are shown in Figure 5.11. The median chlorophyll a concentrations in the different regions of the Baltic Proper are similar but lower than in the western Gulf of Finland. The high sampling frequency attained using the ships-of-opportunity method also provides precise information on seasonal variations in the different regions.

In large river plumes, chlorophyll a concentrations are significantly higher than in the open Baltic Proper. A data compilation on coastal and open sea stations of the Baltic Proper is available in Wasmund *et al.* (2000).

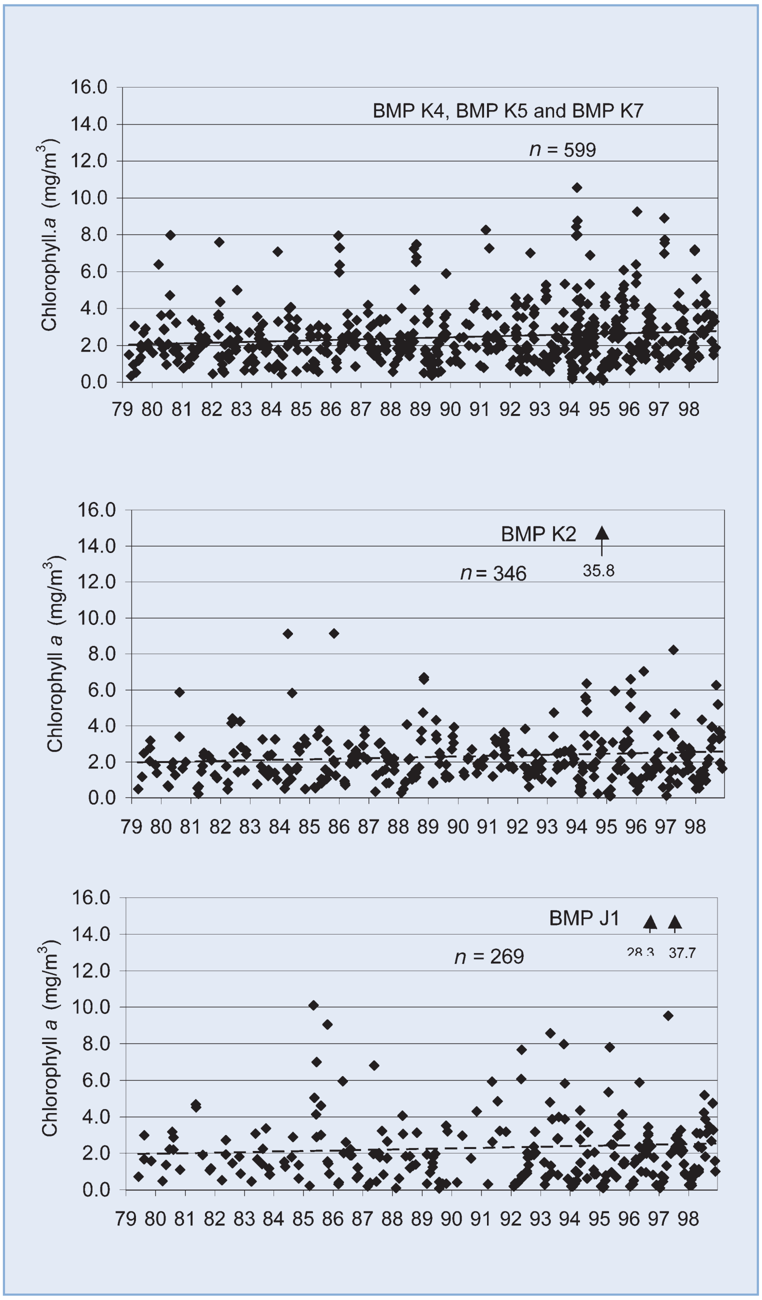


Figure 5.9
Long-term variation in depth-integrated (0–10 m) chlorophyll a concentration in (A) the Arkona Sea (BMP K4, K5 and K7), (B) the Bornholm Sea (BMP K2) and (C) the Eastern Gotland Sea (BMP J1) over the period 1979–1998 (all seasons).

In July 1997, heavy rains hit the southeastern part of central Europe. It is estimated that 17 km^3 of water had entered the Baltic Marine Area via the rivers Oder and Vistula during the period from 11 July to 31 August as compared to the normal runoff of about 6 km^3 (Brandt *et al.* ,1998). The anticipated increase in biological activity in the runoff water was seen both in the Pomeranian Bight and the Gulf of Gdansk. Elevated chlorophyll a concentrations in the range 20–30 mg/m^3 were measured, corresponding to a 10-fold increase. The fear that the nutrient-rich water might spread into large parts of the Baltic Proper and stimulate blooms of filamentous cyanobacteria proved unfounded.

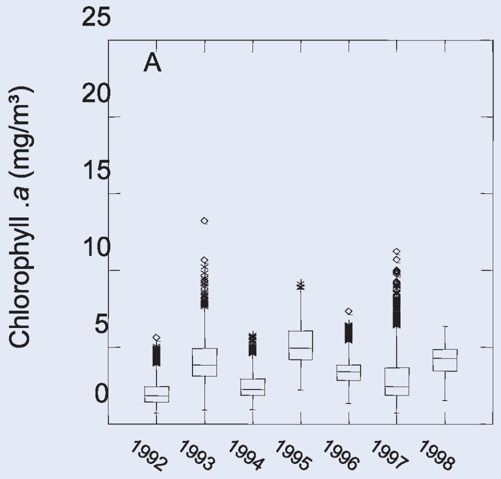


Figure 5.10 Annual variation (summer values July/August) in chlorophyll a concentration over the period 1992–99 in (A) the Western Gotland Sea and (B) the Northern Baltic Proper. The boxes indicate 50% of the observations with the median shown as a horizontal line. Data from the Alg@line Database, Finnish Institute of Marine Research.

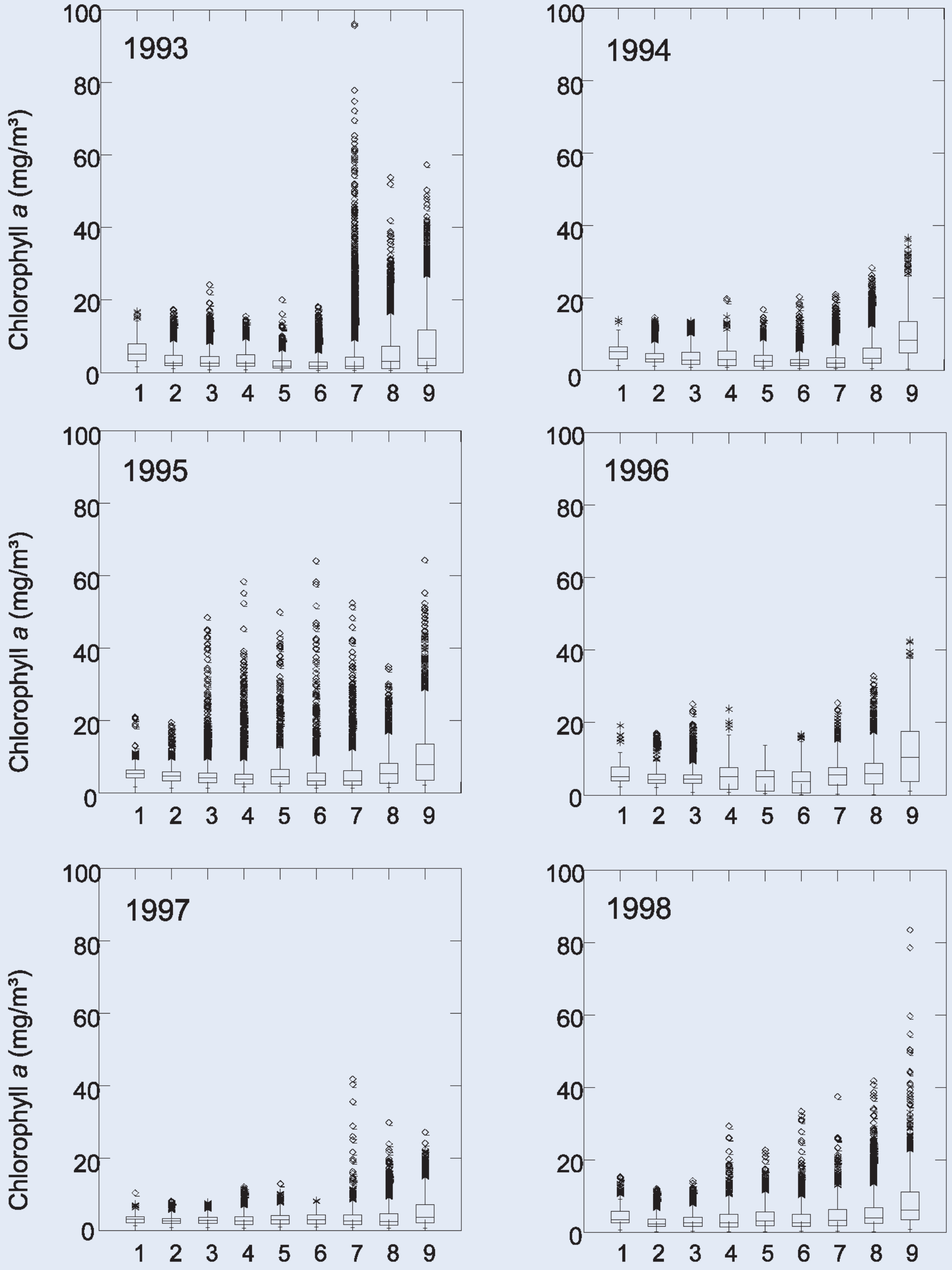


Figure 5.11 Distribution of chlorophyll a concentration along a transect from Travemünde to the Western Gulf of Finland in spring (March to May) in the years 1993–1998. The boxes indicate 50% of the observations with the median shown as a horizontal line.

The regions are

1: Travemünde (in Lübeck Bight),
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 and
9: Western Gulf of Finland.

Data from the Alg@line Database, Finnish Institute of Marine Research.

Mesozooplankton

By: A. Korshenko, K. Blachowiak-Samolyk, R. Opiola, T. Shchuka, N. Budaeva, A. Pöllumäe

Arkona Sea (BMP K3, K4, K5, K6, K7, K8). The composite structure of different water bodies in the Arkona Sea enabled co-existence of different zooplankton assemblages. The representatives of Baltic shallow water species were very abundant and formed a predominant complex. In summer time, 56–59% of the total number was accounted for by all age stages of *Acartia* species, mainly *A. bifilosa,* 17–21% by *Temora longicornis,* 3–8% by *Pseudocalanus minutus elongatus,* 4–6% by *Centropages hamatus* and 4–6% by cladoceran *Evadne nordmanii.* The effect of fresh and low-saline waters is expressed in the appearance of rotifers *Keratella quadrata*, calanoid *Limnocalanus macrurus* and a rather high abundance of *Eurytemora hirundoides* – up to 2% of the total. At the same time the southern Baltic Proper contained minor quantities of representatives of the North Sea group – crustaceans *Microsetella norvegica, Paracalanus parvus* and *Oncaea borealis.* The meroplanktonic larvae of benthic invertebrates such as Gastropoda, Pteropoda, Hydrozoa and Polychaeta played an important but temporary role in the community, indicating good conditions for the bottom fauna.

The previously reported long-term tendency towards a decrease in the abundance of copepods *P. minutus elongatus* in the waters (HELCOM, 1996) was not very apparent in the Arkona Sea during the present assessment period.

Bornholm Sea (BMP K2). Of the 22 or so species and forms comprising the mesozooplankton community during the assessment period, copepods have been the main component in terms of abundance and biomass. Including juvenile stages (nauplii and copepodites) they accounted for 50–90% of the total abundance. The most abundant were *Acartia* species (mainly *A. bifilosa* – up to 48–50%) and *T. longicornis* (23–33%). The less dominant forms were calanoids *Pseudocalanus minutus* and *C. hamatus* (about 5–9%), cyclopoids *Oithona similis* and cladocerans *E. nordmannii* (about 4%). Maximum annual abundance (about 60,000 ind./m^3) was observed in the subsurface layers in 1994 and 1995. In the three remaining years the figure was almost 50% less (Figure 5.12).

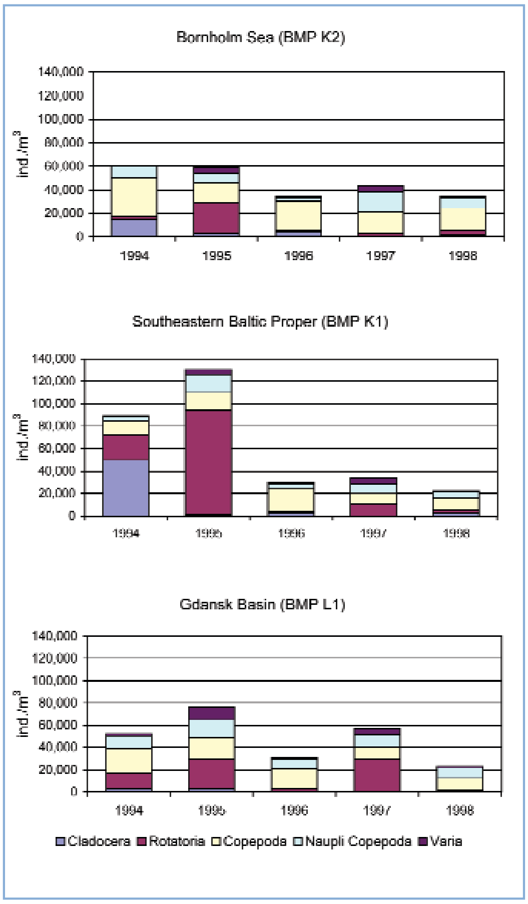


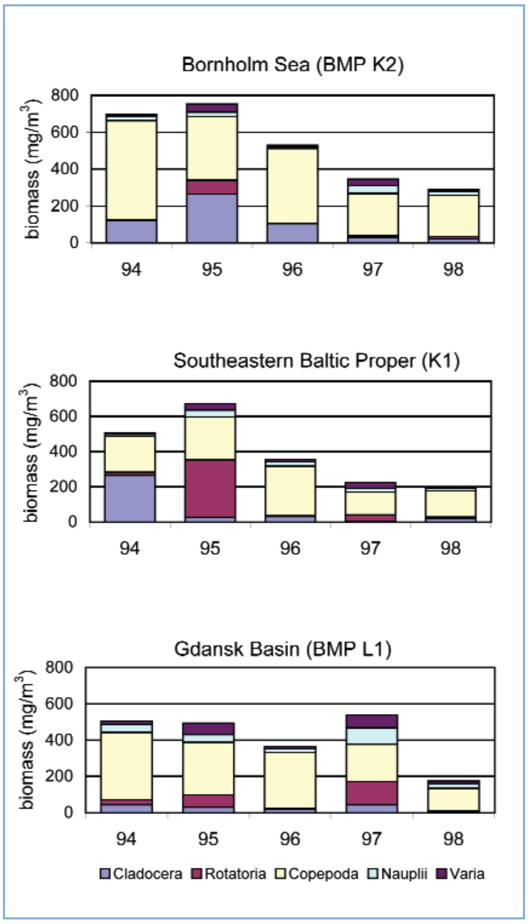
Figure 5.12
Mean abundance of mesozooplankton in the southern part of the Baltic Proper.

Gdansk Basin (BMP L1). The structure of the community was typical for offshore deep waters, with common calanoids dominating. *T. longicornis* accounted for about 38% of the total abundance, *Acartia longiremis f*or 24%, *P. minutus* for 20%, *C. hamatus* for 2% and rotatoria *Synchaeta spp*. for 14%. The average annual abundance was 80,000 ind./m^3 in 1995, about 50,000 ind./m^3 in 1994 and 1997, and about three times less in 1996 and 1998 (Figure 5.12). The North Sea species *M. norvegica* and *Calanus finmarchicus* were also detected. Compared with the shallow areas, the abundance of bivalve larvae in the basin was several times lower.

Southeastern Baltic Proper (BMP K1). In the southern part of the Baltic Proper, average annual abundance of mesozooplankton was 90,000 ind./m^3 in 1994, of which Cladocera constituted over 50%. In 1995, annual abundance was the highest recorded during the assessment period (130,000 ind./m^3) due to the mass occurrence of rotifers, which accounted for over 70% of the total zooplankton abundance (Figure 5.12). In 1996–98, total abundance was almost four-fold less than in 1995, the community usually being dominated by copepoda species during this period. In 1997, rotifers accounted for about 30% of the total abundance.

Eastern Gotland Sea (BMP J1). In summer time the number of species and groups in the community reached 22. The dominant complex consisted of copepoda species – *P. minutus elongatus, A. longiremis, T. longicornis* and rotifers *Synchaeta spp.* that are typical for the region. In accordance with its ecological requirements, *P. minutus* abundance increased with depth. Thus it accounted for 38% of the total abundance in the 0–50 m layer, 62% in the 50–100 m layer and 70% in the 100–200 m

Figure 5.13
Mean biomass of mesozooplankton in the southern part of the Baltic Proper.



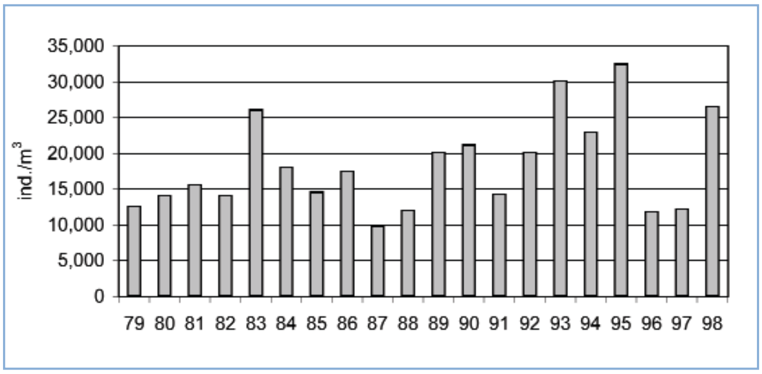


Figure 5.14
Long-term changes in annual mesozooplankton abundance in the southern part of the Baltic Proper.

layer. The opposite tendency is clearly visible for other calanoids and for less abundant cladocerans. The waters of the northern part of the Eastern Gotland Sea differ in origin and hence contain species indicative of various water masses. Besides common forms, representatives of the fresh-brackish-water assemblage such as *Limnocalanus macrurus* and *Keratella quadrata* and of the marine group such as *O. similis* and *Paracalanus parvus* are also present.

Interregional comparison. In the southern part of the Baltic Proper (Arkona, Bornholm, Gdansk and Gotland Seas) a clear trend towards decreasing annual mean abundance and biomass of mesozooplankton was seen during the assessment period (Figure 5.13, Figure 5.14). In 1998, mesozooplankton biomass was three times lower than in 1994.

In June 1998, mean mesozooplankton abundance and biomass varied from 3,948 ind./m^3 (111 mg/m^3) in the Bornholm Sea to 13,186 ind./m^3 (371 mg/m^3) in the Arkona Sea. Maximum total abundance in the water column ranged from 343 to 1,144 000 ind./m^3, productivity being highest in the Arkona Sea. Mesozooplankton was less abundant in the Gotland, Gdansk and Bornholm Seas, but the differences were not large – 3-fold at maximum.

Changes in the community. Zooplankton species composition in the southern part of the Baltic Proper has not changed significantly compared to the previous assessment periods (1979–93) (Wolska-Pys and Ciszewska, 1991; HELCOM, 1996; Ciszewska 1990).

However, there have been clear quantitative changes in the abundance of some copepod populations during the last two decades. Although *P. minutus elongatus* was markedly dominant from 1979 to 1988, it is decreasing in abundance in the southern parts of the sea. In the subsequent five-year period it was partly replaced by crustaceans with a higher tolerance to temperature and salinity changes, e.g. *T. longicornis* and *A. bifilosa* (Wolska-Pys, 1994). A small increase of Pseudocalanus population between 1995 and 1997 is probably attributable to improvement in hydrological conditions in the deep waters due to inflows from the North Sea in 1993 and 1994.

The increase in copepod biodiversity in the southern part of the Baltic Proper could be related to the occurrence of the new species *Acartia tonsa* from 1996 to 1998. The large calanoids *Limnocalanus grimaldii* were also noted in the waters in 1995 after a long-term absence (Wolska-Pys, 1996). The continuous increase of the population of halophilous *O. similis* in the deep waters of Gdansk Basin over the last five years is also important. The increase in Copepoda biodiversity was directly related to the reduction in the number of Cladocera species, which has decreased in the southern Baltic over the last five years from 4–5 (in 1994–96) to 1–3 (1997 and 1998) depending on the region. This decrease was especially notable in the Gotland Sea and in the Bornholm Sea.

In addition to the interregional difference in abundance and species composition, marked interannual variation in average abundance of mesozooplankton was also noted within regions (Figure 5.14).

5.1.4. Benthic conditions

Macrozoobenthos

By: H. Cederwall, V. Diziulis, A. Laine, A. Osowiecki, M. Zettler

Arkona Basin. Due to its proximity to the Danish

Straits the fauna of the Arkona Basin is very much influenced by minor inflows from the Kattegat, which often take place several times a year. Depending on the time of year and the conditions in the Kattegat, these inflows bring different kinds of pelagic larvae that settle in the Arkona Basin. The fauna in the deeper areas of the basin is therefore very variable, species turning up one year and then disappearing one or two years later to be replaced by other species.

At station BMP K4 (48 m) in the eastern part of the basin, the fauna increased in species richness and abundance as well as in biomass during the early 1990s. In the mid 1990s the fauna declined, but in 1997 increased strongly again following more than a year without any episodes of low oxygen (Figure 5.15), species number and abundance reaching the highest levels since 1982. Because of the low number of bivalves, the increase in biomass was less dramatic.

At this station the fauna has changed markedly since the early 1980s, at which time bivalves accounted for nearly 100% of the biomass (HELCOM, 1990). During the 1990s they often accounted for 50% or less of the biomass, and sometimes for only a few percent.

The change from a bivalve-dominated to a worm-dominated community is a response to deteriorating oxygen conditions in the deep area of the Arkona Basin.

The changes recorded at station BMP K4 during the 1990s are supported by results from national Swedish monitoring stations in the central part of the Arkona Basin (Cederwall and Sjöberg, 1995).

In the southern Arkona Basin (station BMP K3; 31 m) macrozoobenthos abundance, biomass and species richness increased until 1995. Whereas abundance only reached about 7,000 ind./m^2 in 1993, it increased to about 12,000 ind./m^2 in 1995. The dominant members of the community were the Polychaeta *Pygospio elegans*, the bivalve *Macoma balthica* and the crustacean *Diastylis rathkei*.

In 1997 the fauna declined, with abundance only reaching 5,000 ind./m^2 and only 11 species being present. In 1998, colonization occurred and species number increased again to 25.

Bornholm Basin. In the deepest part of the Bornholm Basin, new recruitment of fauna occurred as a consequence of water inflow from the Kattegat in 1993–94. This is exemplified by the findings at station BMP K2 (90 m, Figure 5.16). Species number, abundance and biomass increased, mainly due to settlement of polychaetes, of which several species had not been found there in over 12 years. A single specimen of *D. rathkei* and one small specimen of *Astarte sp.* were also found.

After 1994 the macrozoobenthos became successively impoverished, and by 1996 no animals were present. In 1997, recruitment of two polychaete species (*Harmothoe sarsi* and *Scoloplos armiger*) occurred. Two species were also detected in 1998 (surprisingly though the crustacean *D. rathkei* and the bivalve *Astarte borealis*), but in very low numbers.

In the shallower (70–80 m) northern part of the basin at Hanö Bay, polychaete recruitment occurred in 1994 followed by faunal decline in 1995 and an absence of fauna in 1996. In 1997, a few specimens of *H. sarsi* were found at a depth of 80 m, but no fauna was present at 70 m. In 1998, both depths were devoid of fauna. At a depth of 50–55 m the macrozoobenthos was normally rich during the whole assessment period.

Figure 5.15
Variation in total macrozoobenthos abundance and species number in the Arkona Basin (BMP K4).

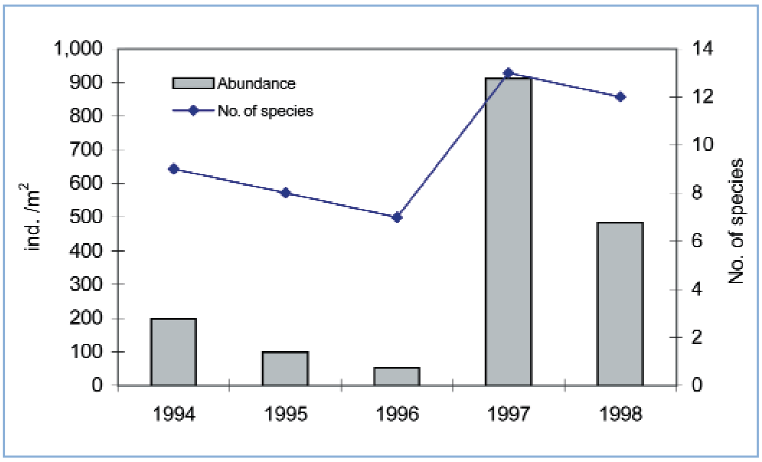
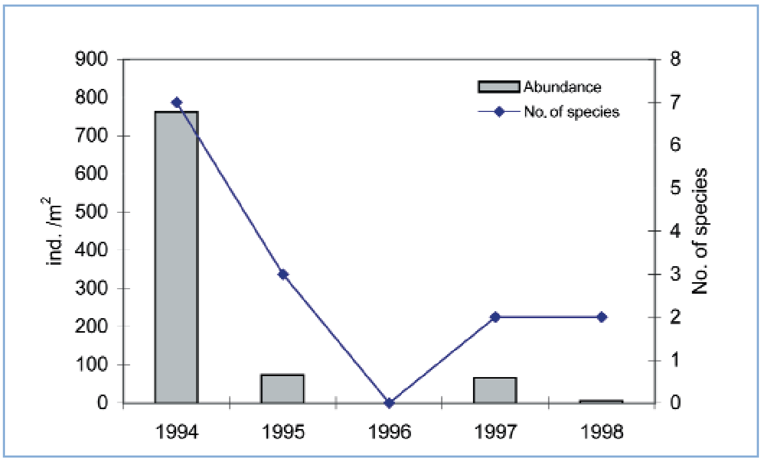


Figure 5.16
Variation in macrozoobenthos abundance and species number in the Bornholm Deep (BMP K2).



At three Polish coastal stations – BMP K12 (19 m), BMP K13 (34 m) and BMP K14 (13 m) – the number of taxa was lower during the period 1994–98 than during the period 1989–93. At all stations the biomass was dominated by molluscs. At station BMP K14 located in the Pomeranian Bight, total biomass exhibited a negative trend during the 1980s, while no trend was detected during the 1990s. At station BMP K13, total biomass tended to decrease during the 1990s. At station BMP K12, no biomass trend is detectable, although

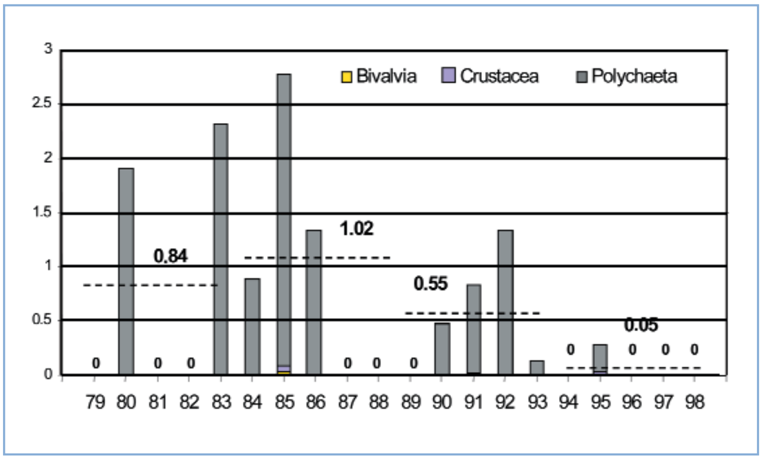


Figure 5.17
Variation in macrozoobenthos biomass and taxonomic composition in the Gdansk Deep (BMP L1). 0 indicates that no fauna was found. The figures above the horizontal bars are the mean values for the period in question.

total biomass was lower in the 1990s than in the 1980s.

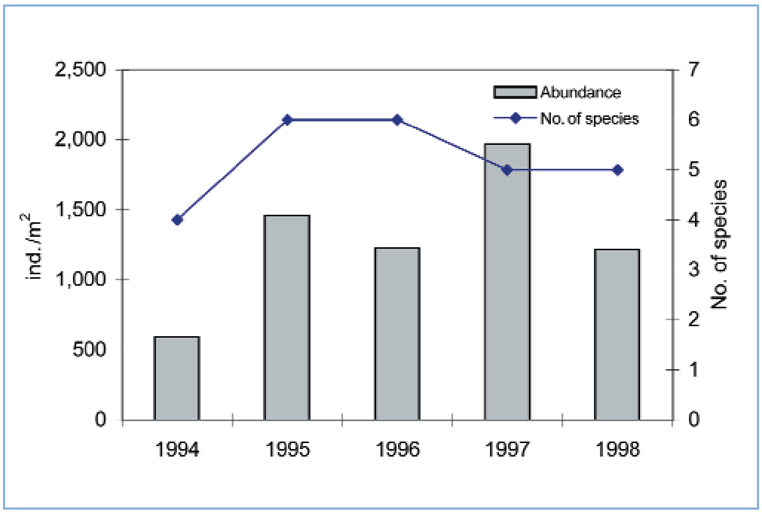
A new polychaete species of North American origin, *Marenzelleria viridis,* which appeared in Polish waters during the second half of the 1980s (Gruszka, 1991), was found at stations BMP K12 and BMP K14 for the first time in 1994–95. The biomass of the species increased between 1994 and 1998.

Gdansk Basin. On the bottom of the Gdansk Deep, periods lacking in benthic fauna alternate with periods with a species-poor fauna of low abundance. The macrofauna recolonizes irregularly due to inputs of denser, oxygen-rich water originating from the North Sea (HELCOM, 1996).

Macrozoobenthos investigations have been carried out in the Gdansk Deep (station BMP L1; 108 m) since 1979. During this time, macrozoobenthos was absent on ten sampling occasions (Figure 5.17). Over the period 1994–98, macrozoobenthos was only present in 1995 and consisted of small numbers of two species of polychaetes, *H. sarsi* and *S. armiger,* and one crustacean *Pontoporeia femorata*.

At station BMP K11 (19 m) in the coastal area near Zarnowiec, the total number of species present was higher in 1994–98 than in 1989–93. Molluscs dominated the total biomass. The species present in the greatest biomass were *Cardium glaucum* until 1991 and *M. balthica* from 1992 onwards. The shift in mollusc domination was very marked.

Figure 5.18
Variation in macrozoobenthos abundance and species number in the southern part of the Eastern Gotland Basin (BMP K1).



The biomass of all the macrozoobenthos phyla, with the exception of polychaetes, whose biomass oscillated irregularly, exhibited a significant decreasing trend.

The Gulf of Gdansk is far more eutrophic than other shallow areas in the open Polish coastal zone (Dubrawski *et al.*, 1998; Kruk-Dowgiallo and Dubrawski, 1998). At station BMP K10 (36 m) the number of taxa was approximately the same during 1994–98 as during 1982–93.

This station exhibited the greatest taxonomic diversity of the Polish coastal stations and the highest average biomass, which was strongly dominated by molluscs. Biomass increased at this station during the 1990s, with the highest values in 1995 and 1997. Between 1997 and 1998, however, the biomass declined considerably.

Eastern Gotland Basin. At station BMP K1 (90 m) in the southern part of the Eastern Gotland Basin, the effect of the 1993–94 inflow of water was only detected in 1995 when the marine species *D. rathkei* and *S. armiger* were found for the first time in many years, although only in low numbers. The species typical for the central and northern Baltic Proper (*P. femorata* and *M. balthica*) remained dominant, though. Abundance and biomass increased from 1994 onwards (Figure 5.18), peaking in 1997 at the highest values ever recorded.

At station BMP J2 (47 m) off the Lithuanian coast, biomass and abundance are mainly accounted for by *M. balthica*. Total biomass increased from 1981 until 1991, but has since decreased (Figure 5.19). After a period of increase during the period 1981–87, total abundance has been decreasing. Moreover, the number of species has been decreasing slightly since 1989.

Biomass and abundance of the crustacean *Monoporeia* affinis have fluctuated strongly: After an increase in the 1980s, the species has decreased drastically since 1989. *Halicryptus spinulosus* and *H. sarsi* have decreased since 1987 and *M. balthica* has decreased since 1988. All observed decreases occurred after a period of increase.

The abundance of *P. elegans* fell to a mean level after a three-year (1990–92) period of high abundance. No clear trend is evident in the abundance and biomass of oligochaetes and the crustacean *Saduria entomon*.

No occurrences of dead sea bottom were observed in the Lithuanian national monitoring region, but samples from station 5C (70 m) usually contained black mud with a strong smell of hydrogen sulphide. Some life was still present, however.

Within the Lithuanian national monitoring region, biomass and abundance have declined at depths below 40 m since 1990–91. At shallower nearshore stations, parallel fluctuations were observed but with less clear evidence of a decrease. At most stations, the number of species found has decreased since the mid 1990s.

At the Swedish coastal station BMP J10 (45 m), macrozoobenthos abundance is strongly dominated by *M. affinis,* and fluctuates markedly. During the period 1994–98, abundance was highest in 1995 (9,000 ind./m²) and lowest in 1997 (4,000 ind./m²). The number of species also varied. Thus eight species were found in 1994, but only four–five species during the following years. There is no clear long-term trend in abundance, but the values found during the period 1986–90 were higher than the ones found 1994–98.

In the late 1980s and early 1990s, gradual recovery of the communities was observed at bottoms deeper than 85 m in the Eastern Gotland Basin as a consequence of the positive effects of prolonged stagnation on the oxygen conditions in intermediate depths. The recovery of the benthic communities was most pronounced on the eastern slope of the basin extending down to about 155 m in the southern half of the basin (Laine *et al.*, 1997).

In 1994–98, the sampling sites at a depth of 90–100 m on the western slope were characterized by impoverished, *H. sarsi*-dominated communities. An exception was station HA1 (89 m) where *P. femorata* was observed in increasing numbers (abundance of up to 750 ind./m²).

On the eastern slope at 85–100 m (HA and HL transects), relatively abundant communities prevailed comparable to those at station BMP K1. These communities were dominated by *P. femorata*, *M. balthica* and *H. sarsi*, with a total abundance of 200 to more than 1100 ind./m².

D. rathkei occurred at one 101 m deep station (HL6) from 1994 to 1997. At a depth of 100–150 m only *H. sarsi* was occasionally found throughout the period. In 1998, the deeper stations (>100 m) were mostly devoid of macrofauna. An exception was station HL5 (135 m), where *S. armiger, H. sarsi*

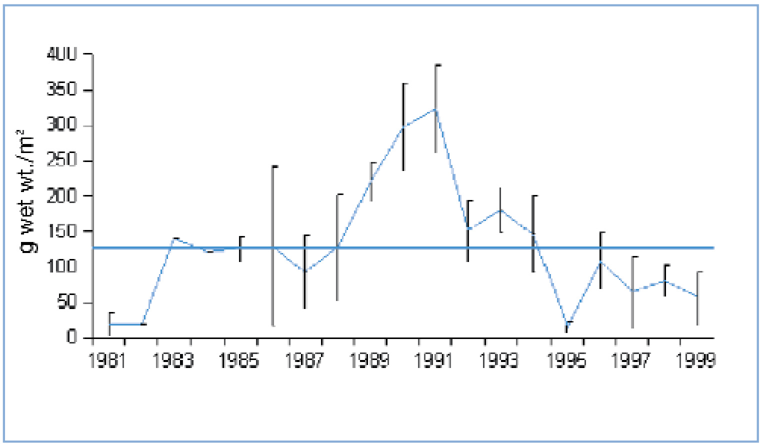
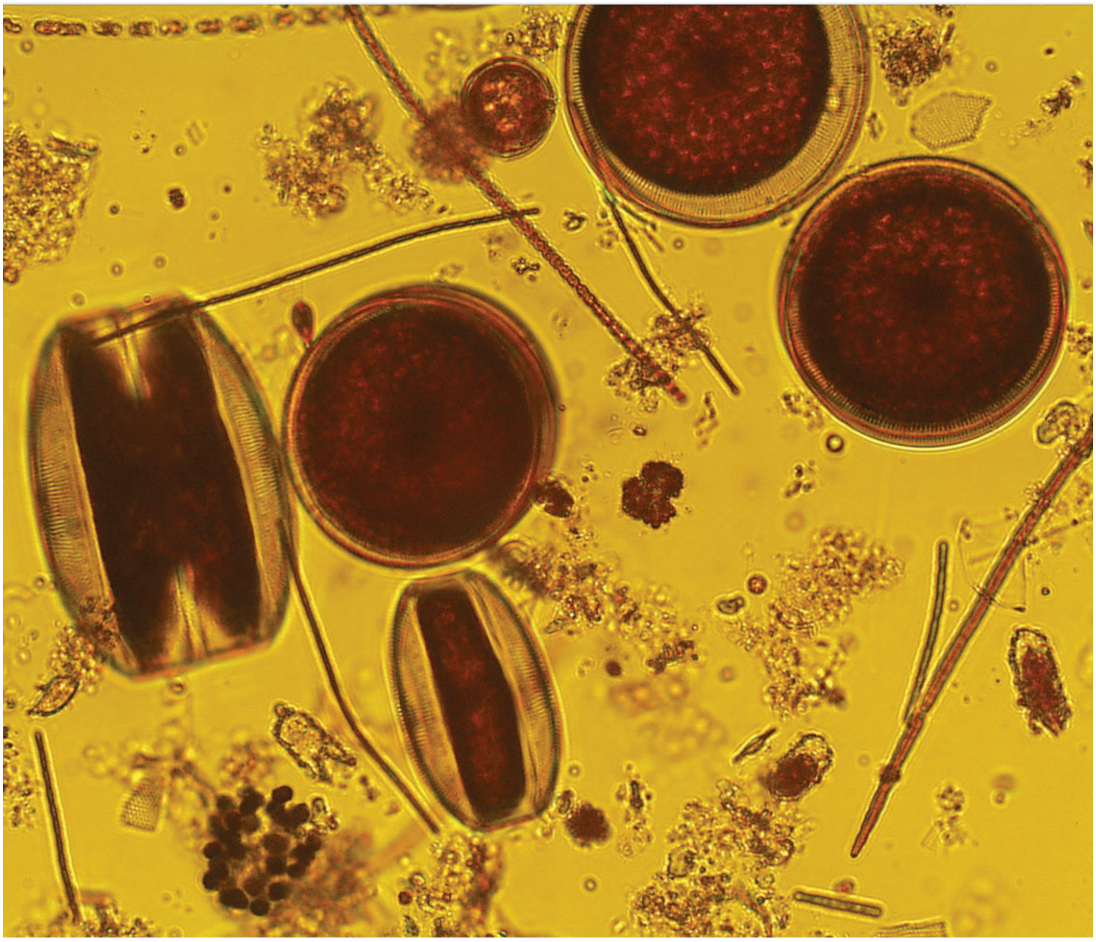


Figure 5.19
Variation in macrozoobenthos biomass off the Lithuanian coast (BMP J2).



Plankton sample from south western Baltic. Photo: Susanne Busch, Baltic Sea Research Institute, Rostock.

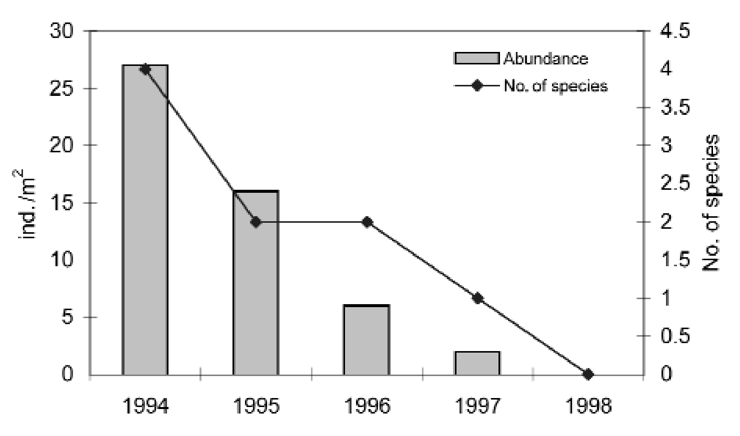


Figure 5.20
Variation in macrozoobenthos abundance and species number in the Western Gotland Basin (BMP I1).

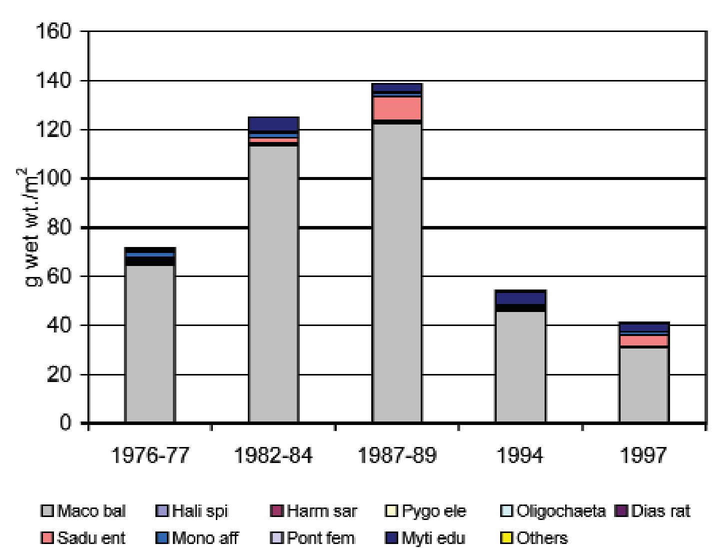
and *P. femorata* were recorded in low numbers.

Identification of the positive effects of the 1993–94 inflows on this depth zone is complicated by the improved oxygen conditions already present prior to the inflows, although they are clearly reflected in the distribution of marine macrozoobenthos species.

In the central part of the Eastern Gotland Basin, at station IVb 2 (112 m), no macrofauna was found in 1994. In 1995, two species were found in modest numbers (*H. sarsi* and *P. femorata*). In 1996, four species were found, but total abundance was lower. The macrofauna then gradually became impoverished, and in 1998 only one species (*Monoporeia affinis*) was found, in low numbers.

In the Gotland Deep (station BMP J1; 249 m), single specimens of *H. sarsi* were found in 1994 as a consequence of the inflow of new oxygenated water. *H. sarsi* (18 ind./m^2) was also recorded here in May 1995. The Gotland Deep was oxygenated in 1994–95, but hydrogen sulphide has been recorded annually in the deep layer since 1996.

Figure 5.21
Variation in macrozoobenthos taxonomic composition and mean biomass at six stations within 30–70 m depth in the Western Gotland Basin.



In the northern part of the Eastern Gotland basin, two intermediate depth stations – LF1 (67 m) and IBS5–10 (78 m) – exhibited species-rich but fluctuating communities with no clear trends in abundance. *P. femorata* and *M. balthica* dominated at these sites, with a lesser contribution by *M. affinis, H. spinulosus, H. sarsi* and *S. entomon.*

Somewhat deeper and more offshore, at stations LF2 (81 m), IBS5-9 (89 m) and LF3 (95 m), the increase, which had started earlier (Laine *et al.*, 1997), peaked in 1994–96 and was followed by a clear decline. The two first sites were strongly dominated by *P. femorata*, whereas *H. sarsi* was the most abundant species at the deeper station. Single specimens of *D. rathkei* were recorded at this site in 1996–97.

In the Fårö Deep area, single individuals of *H. sarsi* were occasionally present at 135 m (LF5). In 1995, the species was also recorded in the deep itself (F80; 193 m). Hydrogen sulphide has been recorded annually in the Fårö Deep at depths >150 m during 1994–98.

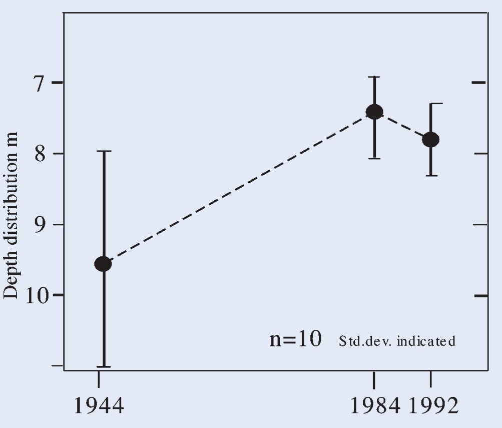
Northern Central Basin. In 1994–98, no hydrogen sulphide was recorded at station BMP H$_2$ (170 m). The depth where oxygen concentration fell below 1 ml/l varied annually from 90 m to 125 m. The polychaete *H. sarsi* has occasionally been detected at this site since the late 1970s (Laine *et al.*, 1997) and was observed in low numbers (<15 ind./m^2) in 1995–97. In addition, *P. femorata* was recorded at this site in 1996. At another 130 m deep open sea station (LL15) in the northern central basin *H. sarsi* was recorded only once, in 1996.

At 125 m, near the Landsort Deep, macrofauna were found in 1994–97 (mostly a few specimens of *H. sarsi*), but not in 1998. At 80–90 m in the Landsort area, several species were found in the mid 1990s in levels ranging from 20 to 150 ind./m^2. During the late 1990s the macrofauna disappeared from the 90 m level and was reduced at the 80 m level. In the shallower parts of the Askö-Landsort area biomass tended to increase during the 1990s, and there was a statistically significant increase in *M. balthica* in the area.

Western Gotland Basin. Some of the saline water that entered the Baltic Sea in 1993–94 flowed northwards from the Bornholm Basin and entered the Western Gotland Basin from the south (Cederwall and Sjöberg, 1995). This led to immigration of marine species to the deepest areas. Thus in 1994, the polychaete *S. armiger* was found for the first time in the Western Gotland Basin (BMP I1). After 1994 the fauna successively declined in the deepest part of the basin (BMP I1 and national Swedish stations) (Figure 5.20), and in 1998, macrofauna was absent at station BMP I1 for the first time since 1990.

In shallower (30–70 m) areas of the basin the faunal biomass decreased significantly in the beginning of the 1990s, following an increase during the 1970s and 1980s (Figure 5.21). In 1997, the situation was unchanged compared to 1994, with biomass being significantly lower than in the late 1980s.

Since the bivalve *M. balthica* accounts for over 80% of the biomass, the changes observed are mainly due to variations in this species. The individual mean weight of this species increased during the 1970s and 1980s, but then decreased strongly in the early 1990s. It seems that the biomass increase during the 1970s and 1980s was due to growth of a few successful age classes, which died off in the early 1990s, followed by recruitment of young small *M. balthica* in the early 1990s. The old population of *M. balthica* possibly hindered successful recruitment of young specimens since it is known that large *M. balthica* can have a negative effect on small, young *M. balthica* (Bonsdorff *et al.*, 1986).



T-test comparison probability (n=10):

	1944	1984	1992
1944	1		
1984	<0.001	1	
1992	0.013	0.013	1

Wilcoxon signed ranks test (two sided probability using normal approximation)

	1944	1984	1992
1944	1		
1984	0.007	1	
1992	0.012	0.022	1

Figure 5.22

Depth penetration of Fucus vesiculosus in the Gräsö archipelago, Åland Sea. Comparison of ten localities in 1944, 1984 and 1992. Results of a t test and Wilcoxon signed rank test of changes within each locality are shown (from Kautsky, 1995).

Phytobenthic plant and animal communities

By: H. Kautsky

The maximal depth distribution of plants is a measure of the ambient light conditions over time. The trends in the distribution of perennial plants in particular thus integrate environmental change over long periods. A strong correlation thus exists between the Secchi depth, reflecting the turbidity of the water, and maximal depth distribution of the phytobenthos. With increasing water quality (i.e. less turbidity) and the presence of suitable substrate, the attached plant species spread to greater depths.

In the Gräsö area of the Åland Sea, the depth distribution of the bladder wrack (*Fucus vesiculosus)* decreased by 3 m from 11.5 m in the 1940s to 8.5 m in the mid 1980s. During the same period the mean Secchi depth in the region also decreased by 3 m. Since the 1980s the depth distribution has increased again (Figure 5.22). The maximum depth distribution of the bladder wrack also seems to be increasing in the Askö area.

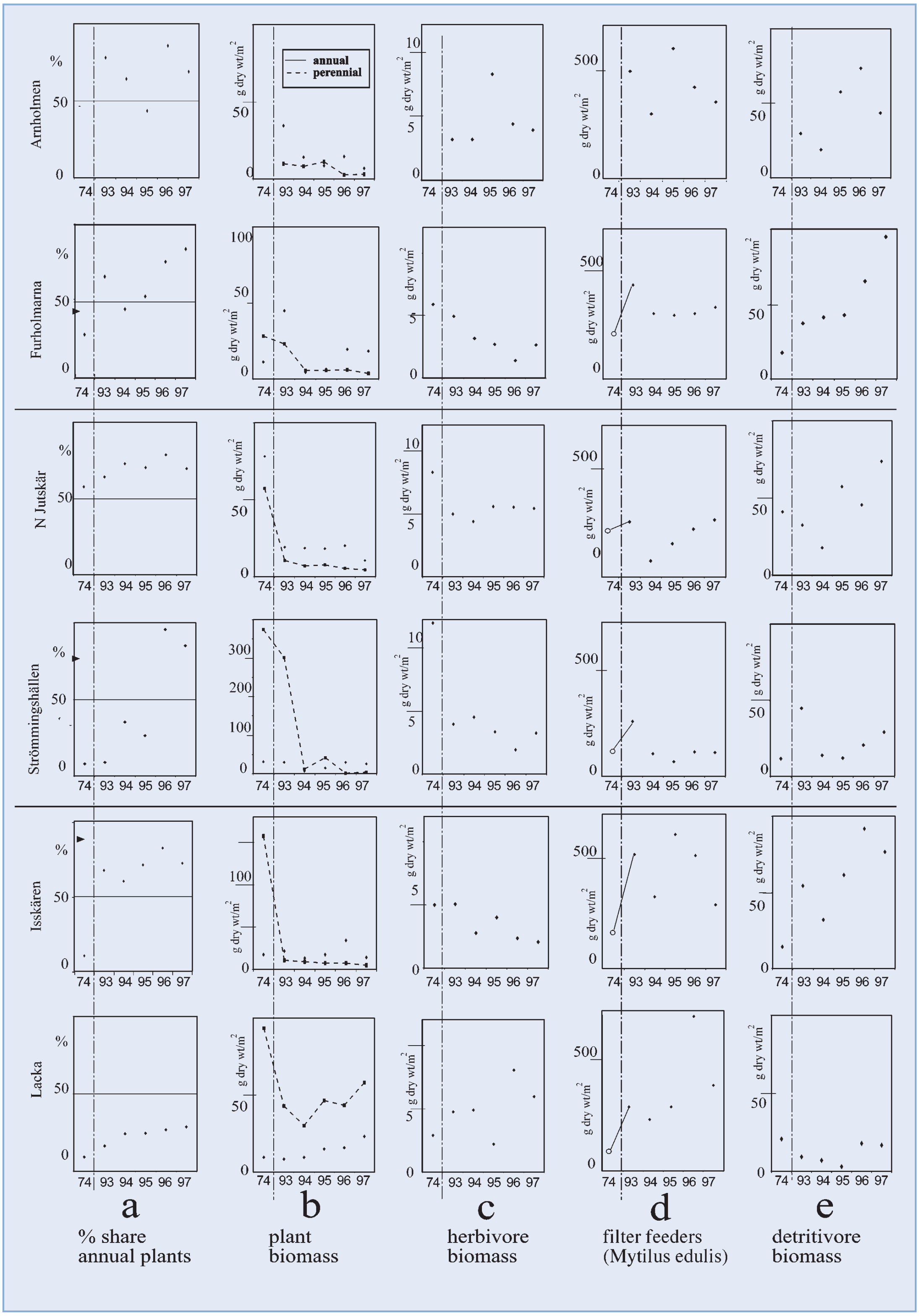
Eutrophication enhances the growth of opportunistic, filamentous, annual algae and the percentage of the total biomass accounted for by annual plants is expected to increase with increased eutrophication, as has been the case in the Askö area since the mid 1970s (Figure 5.23a). However, this is due not to an increase in annual plant biomass, but rather to a decrease in perennial plant biomass (Figure 5.23b).

The decrease of perennials could be attributable to several factors, e.g. shading and competition for nutrients from an increasing biomass of filamentous epiphytes could hamper the growth and distribution of the bladder wrack (*F. vesiculosus*). No increase of filamentous algae can be seen in the Askö area, though (Figure 5.23b). However, there are anecdotal indications reported from several places along the coasts of the Baltic Proper of an increased and seasonally prolonged occurrence of epiphytes on *Fucus*. An alternative explanation might be the life cycle of this perennial alga. *Fucus* has an intricate life cycle with only a few, short opportunities each year to reproduce – mainly during full moon and calm weather. The chance of reproductive failure is thus high, and young *Fucus* plants are only present in high abundance in occasional years. The populations might disappear from large areas when their life cycle is completed. The *Fucus* germlings seem to be poor competitors if no empty space is found. Recolonization may therefore take a long time due to competition for space from filamentous algae.

The disappearance of *F. vesiculosus* could be explicable by an increase in herbivore abundance since the herbivore population is thought to benefit from the eutrophication-induced increase in the biomass of annual plants, which are a preferred food item. It has been shown that the herbivores *Idothea spp*. might change their preference when in dense populations and instead consume *Fucus* fronds. However, there is no indication of any increase in herbivore biomass in the Askö area since the mid 1970s (Figure 5.23c).

Eutrophication enhances pelagic production, thereby increasing the organic matter content of the water column and hence sedimentation. It can thus be expected that the phytobenthic filter feed-

Figure 5.23
Quantitative changes within functional groups at six stations in the Askö area since the 1970s. The figure shows the percentage of annual plants (a) and the biomass in g dry weight/m^2 of annual and perennial plants (b), herbivores (c), filter feeders (mainly Mytilus edulis, including shells) (d) and detritivores (e). The lack of data between the years 1976 and 1992 is indicated by a broken vertical line. The stations Furholmen and Arnholmen are from the inner part of the archipelago, while N. Jutskär and Strömmingshällen are from the middle part and Isskären and Lacka are from the outer part.



ers and detritivores will increase due to the increased food availability. Only a weak and inconsistent trend can be seen in the various parts of the archipelago (Figures 5.23d and 5.23e), however. The inner parts are more dependent on the local riverine runoff while the outer parts more reflect the general status of the Baltic Proper.

At most stations in the Askö area, the biomass of the filter-feeding blue mussel (*Mytilus edulis*) almost doubled between the 1970s and 1993 (Figure 5.23d). This was followed by mass mortality in 1994 probably due to the unusually hot summer combined with a shortage of food (low pelagic production). Since then some but not all of the populations have increased again.

In conclusion, also comparing with the 1970s, some changes have taken place in the phytobenthic plant and animal communities since 1993 that are indicative of increased eutrophication, but the general trend seems to be stable conditions or trends indicating recovery.

5.2. Gulf of Bothnia

By: Convenor: H. Dahlin

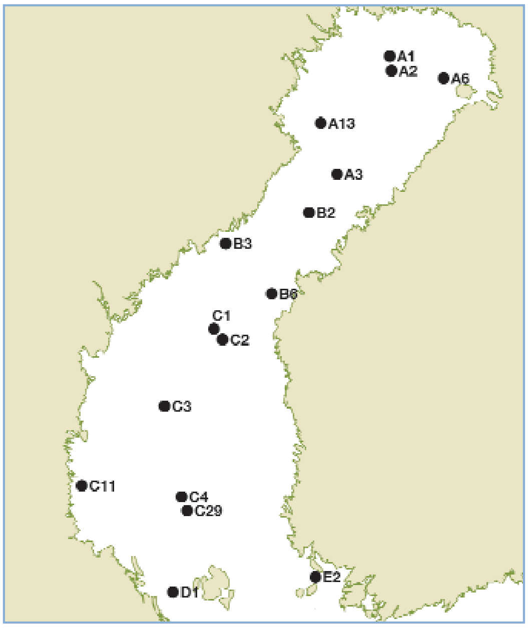


Figure 5.24
Map of the Gulf of Bothnia indicating the monitoring stations referred to in the text.

5.2.1. Meteorological, hydrological and hydrographical forcing

Weather and climate for the Gulf of Bothnia 1994–1998

By: H. Alexandersson

The period 1994–1998 has generally been mild and wet. The warmest year at Holmögadd (63°35’N, 20°45’E) in the middle of the Gulf was 1997 when the annual average temperature was 4.8°C, 1.4°C above the normal (the average value for the period 1961–90). In 1994 and 1996, the annual temperatures were very close to the normal. The coldest month was February 1994, with -11.2°C. This was also the month with the largest negative anomaly. The largest positive deviations occurred within the four-month period December 1994 to March 1995. The eight-month period August 1996 to April 1997 is also remarkable since none of the months had average temperatures below the normal (Figure 5.25).

For Sweden as a whole, there is a quite remarkable trend towards wetter conditions in the most recent 20 years although this is less clear in the easternmost areas. At Holmögadd there was a pronounced dry anomaly during the period July 1995 to April 1996. During all ten months, precipitation at Holmögadd was substantially less than normal. In 1998, in contrast, Holmögadd received 787 mm of precipitation, the largest annual amount measured since observations began in 1880 and almost 140% of the normal value. The wettest months in 1998 were June and August, with 106 mm in both months corresponding to about 300 and 160% of the normal value, respectively. Precipitation exhibited a much more complex spatial anomaly pattern than temperature, with 1998 being the wettest year since at least 1900 in substantial parts of northern and southwestern Sweden. The two wettest days at Holmögadd were 25 August 1995 and 4 July 1997, with 36.2 and 36.3 mm, respectively (Figure 5.26).

Global radiation measured at Umeå (63°50’N, 20°17’E) naturally deviates much more from the average in summer than in winter. Most remarkable are the strong negative deviations in summer 1998, which was cool and wet. In the warm summer of 1997, the positive deviations are somewhat smaller than could be expected. In several of the years the summers started cloudy in June and ended sunny in August (Figure 5.27).

During the years 1994–98, several hazardous weather events have occurred in the vicinity of the Gulf of Bothnia. Extraordinary precipitation and thunderstorms were recorded in the coastal area of Sweden in July and August 1997 and in December 1998.

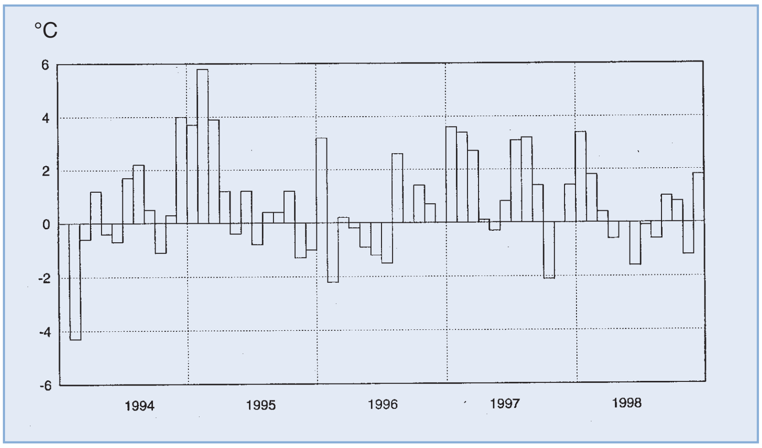


Figure 5.25
Monthly temperature anomalies in °C 1994–1998 at Holmögadd.

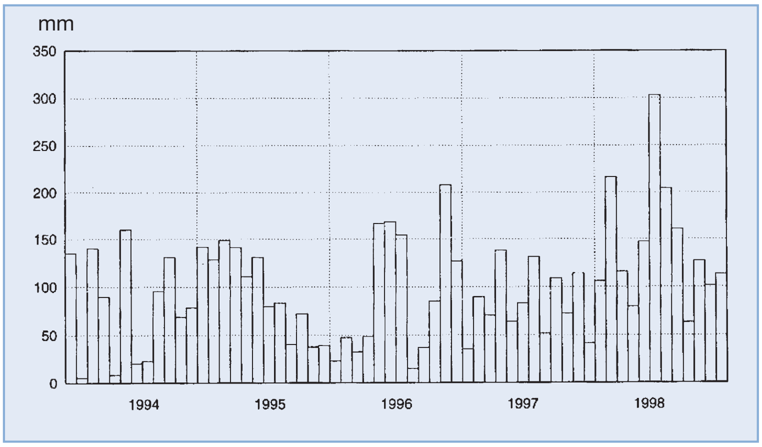


Figure 5.26
Monthly precipitation percentage anomalies 1994–1998 at Holmögadd.

Ice conditions

By: Hans Dahlin

Part of the Gulf of Bothnia is always covered by ice during winter, with high inter-annual variability.

The period 1994–98 begun with a “normal ice-winter”, the maximum ice extension in early March covering the whole of the Gulf of Bothnia. This was quite a change compared with the preceding five-year period with mild or extremely mild winters. In 1994–95, the winter was again extremely mild, but was followed by a normal winter in 1995–96. The remaining two winters of the assessment period were mild, with only parts of the Bothnian Sea being ice-covered in addition to the Bothnian Bay. The severity of the winters from 1981 to 1999 is illustrated in Figure 5.28 as an annual value calculated from observed air temperature. The figure also shows the large difference in the ice situation between the first two and the last two HELCOM assessment periods.

Figure 5.27
Monthly global radiation anomalies in kWh/m² 1994–1998 at Umeå.

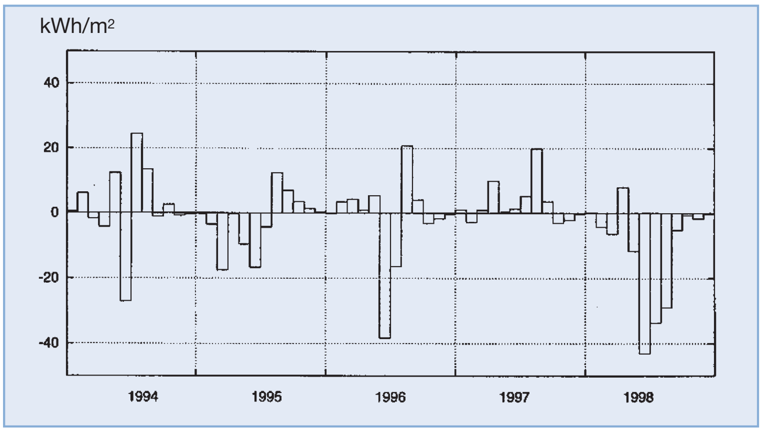
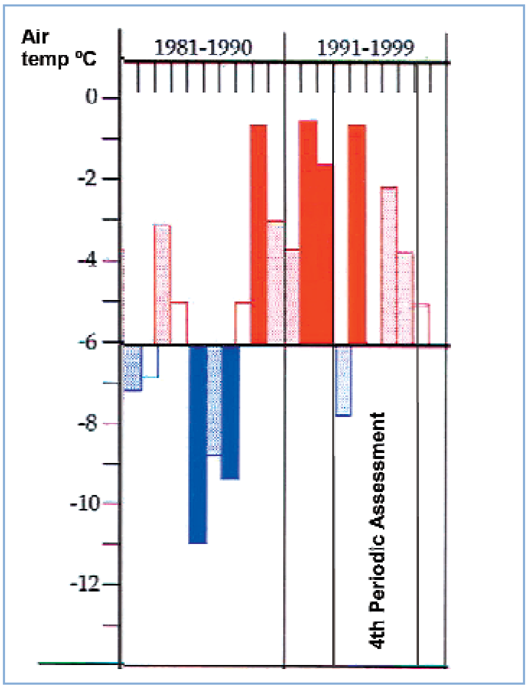


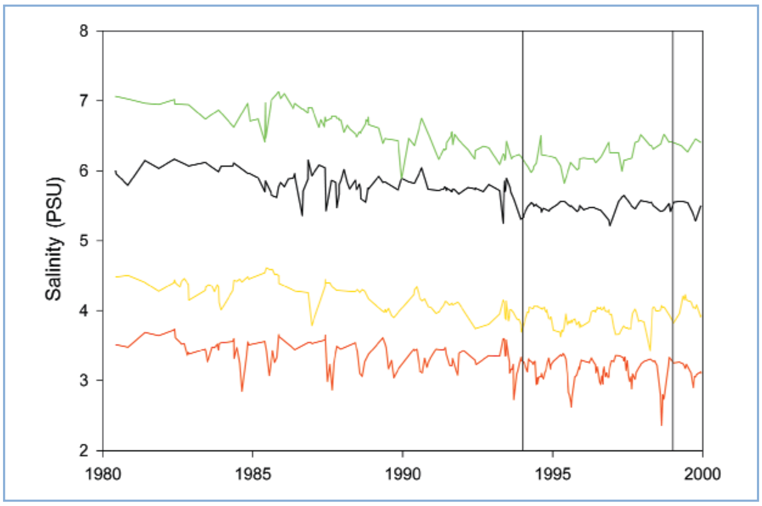
Figure 5.28
Severity of the winters in the Gulf of Bothnia from 1980 to 1998 as a function of air temperature.



🟥 Very mild winter
▨ Mild winter
▨ Strong winter
🟦 Very strong winter

Figure 5.29
Surface and deep-water salinity in the Bothnian Sea (BMP C4) and in the Bothnian Bay (BMP A13).

— C4 surface water
— A13 surface water
— C4 deep water
— A13 deep water



Hydrography

By: Pekka Alenius

The vertical salinity stratification is mostly a two-layer structure with a weak halocline at 50–60 m depth. Surface salinity expressed in practical salinity units (PSU) is typically between 5.5 and 6 PSU in the Bothnian Sea and slightly less than 3.5 PSU in the Bothnian Bay. The bottom salinity is generally one unit greater than the surface salinity, being around 6.5–7 PSU in the Bothnian Sea and about 4.3 in the Bothnian Bay. The deep waters of the Gulf originate from the more saline surface layer of the northern Baltic Proper that sinks when entering the Gulf through the Åland Sea. Thus the deep layers of the Gulf do not suffer from oxygen depletion.

The summer sea surface temperature near the coast of Finland was higher for the period 1994–97 in the Gulf of Bothnia, but around average in 1998. The year 1999 was warmer than normal in the whole of the Gulf of Bothnia. During the warm years, the duration of the summer conditions in the sea was some days longer than average.

Average sea surface salinity at the Finnish coast was 0.24 PSU less during the assessment period than normal, although the salinity varied considerably. The deep-water salinity was about 0.32 PSU lower than average. The deviations from normal in salinity were greatest at the beginning of the assessment period. Towards the end of the period, the salinity increased towards the long-term mean values.

In the open Bothnian Sea, the deep-water salinity decreased from 7 PSU in the 1980s to about 6 PSU in the 1990s, but is slowly rising again. The same tendency is also obvious in the surface layer, although the absolute reduction in salinity was smaller (from around 6 to around 5.5 PSU) than in the deep waters. Similar trends can be seen in the Bothnian Bay (Figure 5.29).

In the Gulf of Bothnia, where a majority of the biological species are stressed by extreme salinity (either too high or too low), a 10–15% change in salinity ought to have a significant impact on the ecosystem.

5.2.2. Hydrochemistry

By: B. Håkansson and N. Kajrup

The two sub-basins the Bothnian Bay and the Bothnian Sea are distinctively different in terms of hydrochemistry (Fonselius, 1978). In the northern basin the concentrations of nitrate+nitrite and silicate are rather high, whereas the phosphorus concentrations are low. In the northern basin the DSi:DIN:DIP (SiO_3-Si:NO_2+3-N:PO_4-P) ratios during winter far exceed the values for normal plankton uptake (Redfield *et al.*, 1963). In the Bothnian Sea, on the other hand, the Redfield ratios are normal for DIN:DIP but much higher for DSi:DIN, yielding Redfield ratios of approximately 67:16:1. The changes in concentrations and Redfield ratios between the preceding and the present assessment periods are small and within natural variation (Figure 5.30a and b). Seasonal variation is common for both sub-basins except for silicate, which is rather inert in the Bothnian Bay.

In the early autumn of 1995 and 1998, two very similar events occurred, resulting in larger than average anomalies. Low-salinity water reached the open sea of the Bothnian Bay, resulting in higher silicate and nitrate+nitrite concentrations than normal for the season. This demonstrates the importance of terrestrial inputs to the Bothnian Bay (Wulff *et al.*, 1996), although no corresponding change was noted for phosphate.

Long-term trends in the bottom waters

In the Bothnian Bay the deep water exhibited declining salinity over the period 1980–90, whereafter the salinity levelled off at about 3.8 PSU during the period 1991–1998. This can be compared with the typical value of 4.2 PSU, using the station F9 as an indicator. Hence, the present assessment period is characterized by lower than average, but stable deep-water salinity.

In the deep water a similar trend was seen for PO_4-P. In the 1980s, the PO_4-P concentration decreased towards 0.05 µmol/l and stabilized around this level from 1993 onwards. The nitrate+nitrite data from before 1990 are inadequate for assessment purposes although the concentrations seem to have decreased from 1993 onwards. No trend in silicate concentration is apparent since 1990, with the concentration having varied close to the average value of 29 µmol/l.

In the Bothnian Sea the deep water salinity has increased by approx. 0.5 PSU during the present assessment period, using the station SR5 as the indicator for this basin. The trend is similar for PO_4-P, which has increased by 0.2 µmol/l. In contrast, the nitrate+nitrite concentration tended to decrease during the assessment period. With silicate, no particular trend can be identified except that the concentrations are lower than the averages for the whole period 1980 to 1999. It is typical for the deep water that phosphorus is out of phase with nitrate+nitrite changes, a pattern seen for the past 20 years.

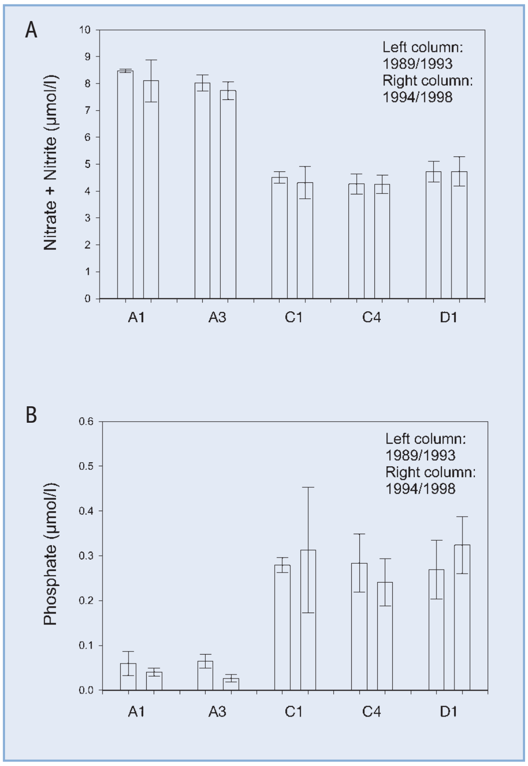


Figure 5.30
Mean (±SD) of
A) surface water nitrate + nitrite
B) phosphorus concentrations at selected stations in the Gulf of Bothnia during the last two assessment periods.

Limiting nutrients

The surface waters of the Bothnian Bay show no indications of eutrophication as determined from hydrochemistry data. Phosphorus levels are low, less than 0.1 µmol/l, whereas the nitrate+nitrite levels are rather high, slightly less than 8.5 µmol/l (Figure 5.30). The Redfield ratio is thus much greater than 16, demonstrating that this northernmost basin is phosphorus-limited (Figure 5.31a; Sandén *et al.*, 1991), these conditions remaining unchanged during the assessment period compared to the whole period 1980–1999. Silicate is rather independent of nitrate+nitrite changes, and the concentration is high, on average 29 µmol/l (Figure 3.31b). Compared to the past 20 years, the silicate concentration is unchanged.

The surface waters of the Bothnian Sea have freshened during the assessment period compared to the period 1980–1999. The salinity decreased 0.25 PSU at station BMP C4 (Figure 5.32). However, no corresponding change was detected for nitrate+nitrite and phosphorus, which exhibits the same variability during 1980–1999. Typical concentrations of nitrate+nitrite and phosphorus are presented in Figures 5.30a and b. The Redfield ratio for these two compounds is close to the expected value (Figure 5.31a), and nitrogen is only in deficit relative to phosphorus during the summer period. The Bothnian Sea is usually in hydrochemical balance and is only nitrogen-limited during the summer. In contrast, the silica concentration decreased relative to nitrate+nitrite during the assessment period (Figure 5.31b; Sandén *et al.*, 1991). The ratio nevertheless exceeds the Redfield

Figure 5.31a
Scatter plot of nitrate + nitrite versus phosphorus.

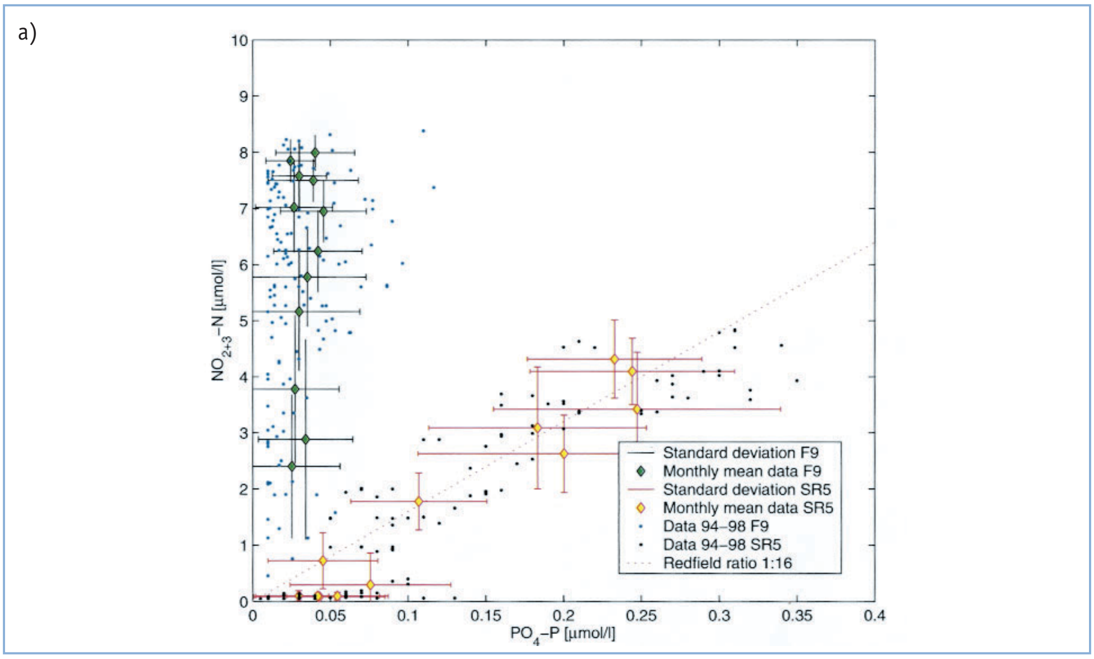


Figure 5.31b
Scatter plot of silicate versus nitrate + nitrite in surface water of the Bothnian Bay at station BMP A13 (=F9) and the Bothnian Sea, station, at station BMP C4 (=SR5).

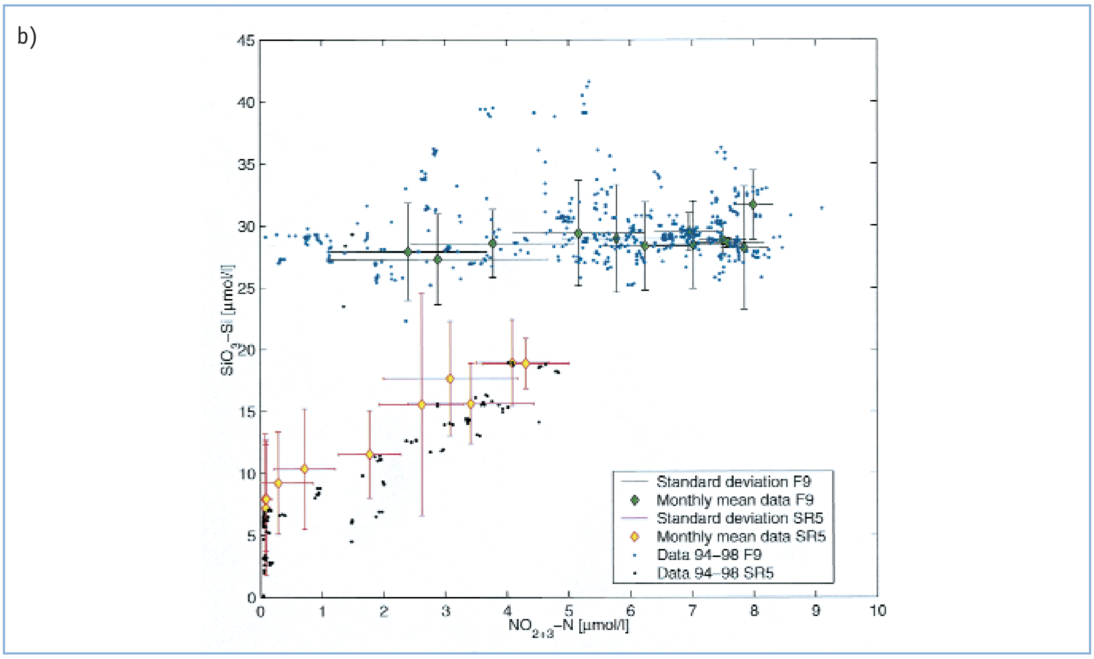


Figure 5.32
Scatter plot of nitrate+nitrite versus salinity in surface water of the Bothnian Bay BMP A13 (=F9) and Bothnian Sea BMP C4 (=SR5) during the present assessment period compared with monthly means during the period 1980 to 1999.

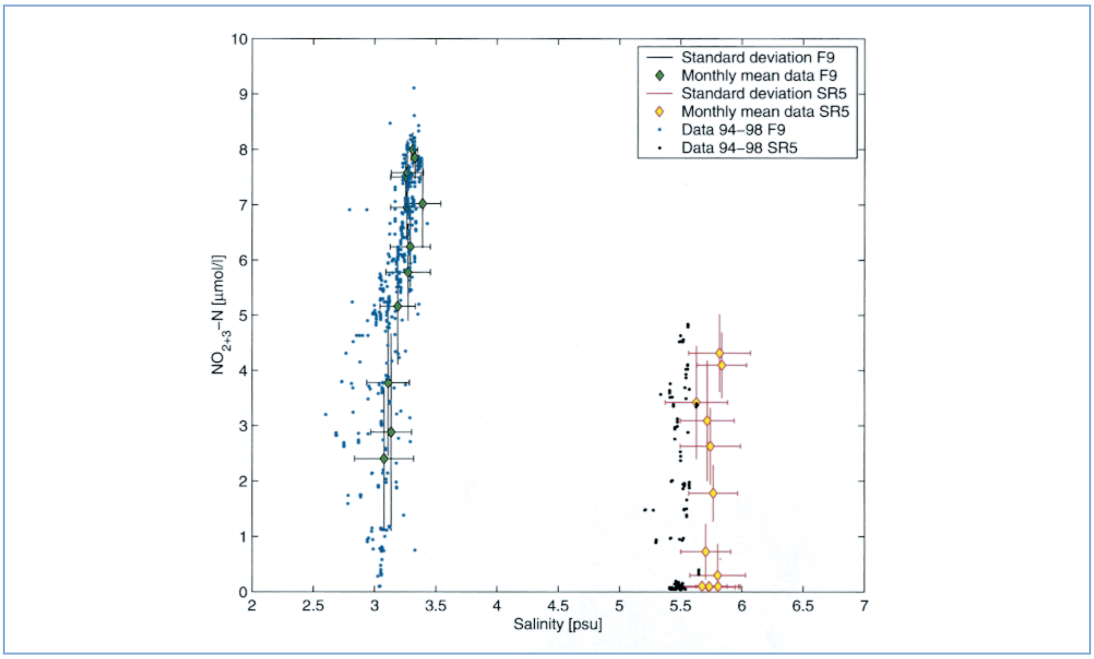


Table 5.4
The food web in the Bothnian Bay and the Bothnian Sea expressed as carbon biomass and flux. Pelagic values were calculated for the average depth of the water column in each basin.

CARBON BUDGET				
	Bothnian Bay		**Bothnian Sea**	
	Biomass	**Flux**	**Biomass**	**Flux**
Input of carbon (g C/m² • yr)				
River discharge and wastewater (1)	134	22	214	10
Pelagic primary production (1)	1.6	28	4.0	109
Benthic primary production (2)	-	3	-	3
Total	**136**	**52**	**218**	**122**
Secondary production				
Bacterioplankton (1)	0.68	20	0.98	41
Protozooplankton (1)	0.31	12	0.84	26
Micro- and mesozooplankton (1)	0.11	4.1	0.20	14
Fish (3)	0.30	0.26	0.70	0.96
Benthic bacteria (2) (3)	-	0.25	-	2.0
Benthic fauna (3)	0.28	1	4.2	8.0
Total pelagic	**1.4**	**36**	**2.72**	**81**
Total benthic	**0.28**	**1.3**	**4.2**	**10**

(1) Sandberg J, Andersson A, Johansson S, Wikner J. (2003). Pelagic food web structure and carbon budget in three brackishwater environments: Potential importance of terrigenous carbon. Submitted to Mar. Ecol. Prog. Ser.
(2) Wastenson L, et al., (1992) Hav och kust. 1992. Svensk National Atlas. Ed. Björn Sjöberg, Series Bra Böcker, Höganäs, Sweden
(3) Elmgren R, (1984). Trophic dynamics in the enclosed, brackish Baltic Sea. Rapp. P.-v. Réun. Cons. int. Explor. Mer. 183, 152-169. - Estimate lacking at present.

ratio (1:1), thus indicating that silica is not a limiting nutrient.

5.2.3. Structure and function of the pelagic ecosystem

The ecosystem of the Gulf of Bothnia differs mainly due to the high freshwater influence from the southern or western parts of the Baltic Marine Area. Primary production is here remarkably lower, and the abundance, size and diversity of plants and animals are generally less than in other parts of the Baltic Marine Area. Brackish water and freshwater species dominate the systems.

The main compartments of the ecosystem and their interrelationship expressed in terms of carbon biomass and carbon flux are illustrated for the Bothnian Bay and Bothnian Sea in Table 5.4 and Figures 5.33 and 5.34.

Bacterioplankton

By: J. Wikner

In terms of biomass, chemoorganotrophic bacterioplankton are the main component of the heterotrophic plankton community in the Gulf of Bothnia (Kuparinen *et al.*, 1996; Wikner and Hagström, 1999). This group of organisms also accounts for more than 50% of the planktonic secondary production. A major part of the carbon flow and oxygen consumption is consequently associated with bacterioplankton activity. The bacterioplankton community thus provides an important measure of the trophic state of aquatic environments (Cole *et al.*, 1988; Billen *et al.*, 1990).

Time series of bacterioplankton oxygen consumption and biomass in the Gulf of Bothnia reveal stable development during the assessment period (Figures 5.35, 5.37 and 5.39). No significant trends could be demonstrated, and there is nothing in the bacterial variables indicative of systematic changes in the trophic state of the Gulf of Bothnia.

The relationship between the bacterioplankton and phytoplankton communities does not indicate any systematic changes in ecosystem function (Figures 5.36, 5.38 and 5.40). The ratio between bacterial carbon production and phytoplankton CO_2 assimilation fluctuated between years, especially in the Öre estuary and less clearly at the offshore station in the Bothnian Sea. The difference in influence of terrigenous (i.e. originating from terrestrial sources) dissolved organic substrates, discharged by rivers, probably explains the different levels of the P_b:P_{ph} ratio in the basins. High values of the P_b:P_{ph} ratio compared to literature compilations (Cole *et al.*, 1988) may be interpreted as bacterioplankton growth based on other (e.g. allochthonous) sources than concomitant phytoplankton carbon production (i.e. resulting in net heterotrophy).

In 1999, bacterial oxygen consumption was markedly (2-fold) higher than average at the offshore station in the Bothnian Sea in line with an exceptional decrease in oxygen concentrations observed during summer (see the section above on hydrography). Elevated bacterial biomass was also observed at both the offshore and the coastal station in this basin, possibly due to the extreme precipitation and runoff during 1998. Although the immediate intra-annual effect 1998 appeared to be reduced productivity of both phytoplankton and bacterioplankton, the introduced dissolved terrigenous carbon may eventually have become more

Figure 5.33
Carbon flux and carbon in biomass in the Bothnian Bay (C g/m² • yr).

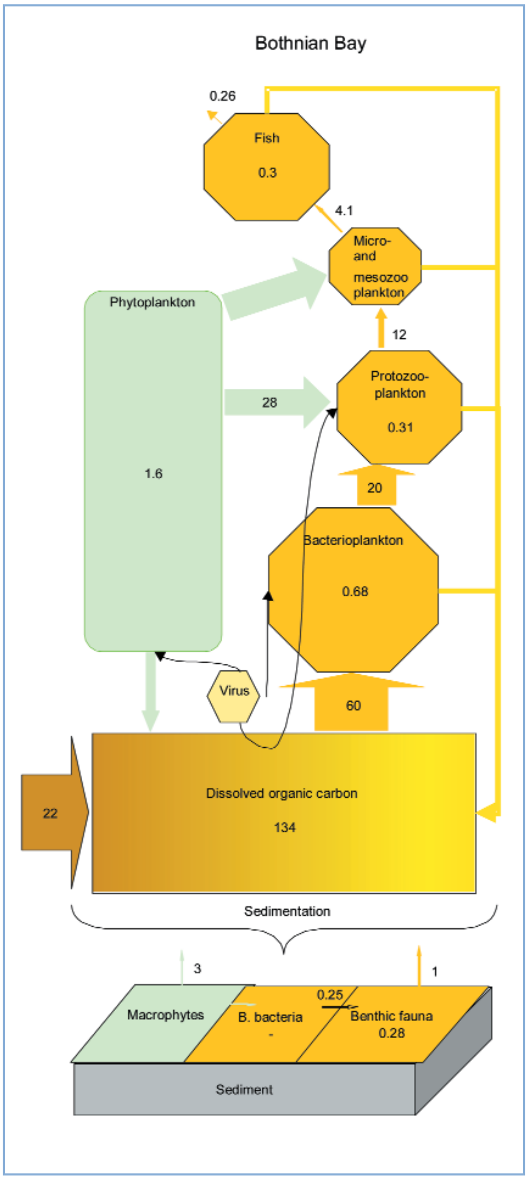
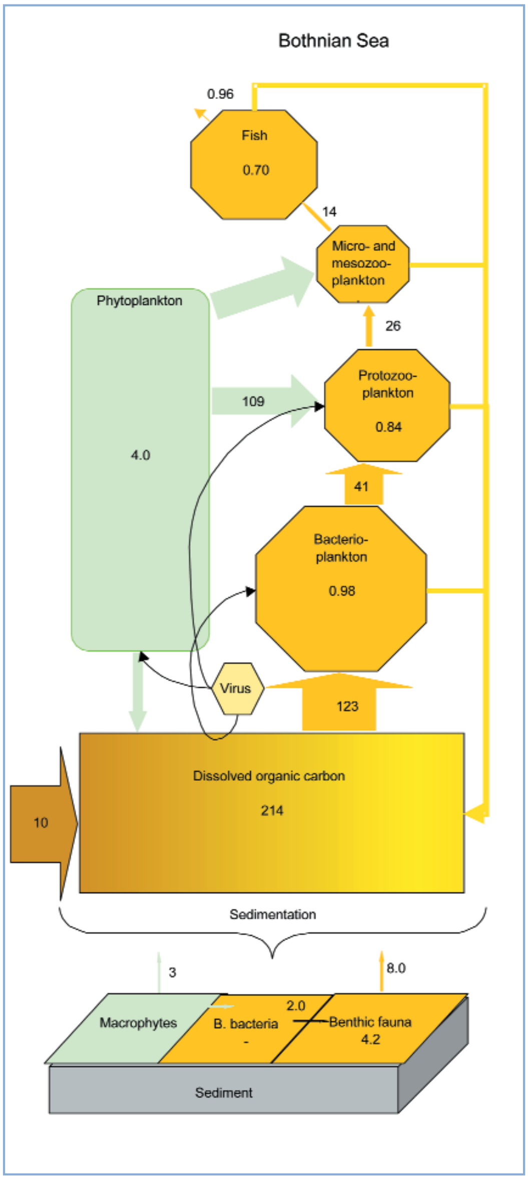


Figure 5.34
Carbon flux and carbon in biomass in the Bothnian Sea (C g/m² • yr).



bioavailable at the higher salinities and nutrient concentrations encountered in the Bothnian Sea (Wikner *et al.*, 1999). The more than 3-fold higher P_b:P_{ph} ratio in 1999 suggests that bacterioplankton productivity that year was supported by a source other than autochthonous phytoplankton carbon assimilation.

From volumetric values for both bacterial oxygen consumption and biomass it can be seen that carbon respiration was generally similar in both the Bothnian Bay and the Bothnian Sea (Table 5.5). The allochthonous supply of organic carbon thus seems to compensate for the lower phytoplankton carbon assimilation in the Bothnian Bay. When both autochthonous and allochthonous sources are included, both major basins appear to have a similar trophic state. As expected, the trophic state was highest in the coastal zone in terms of both bacterial oxygen consumption and biomass.

Differences in the quality of the carbon sources cause inter-basin differences in carbon flow, however. In the Bothnian Bay the allochthonous supply is mainly in the form of dissolved organic carbon, which promotes growth and remineralization primarily among bacterioplankton (cf. P_b:P_{ph} ratio

Table 5.5
Comparison of bacterioplankton variables in the basins of the Gulf of Bothnia. Bacterial oxygen consumption (BOx) and biomass, and their relation to phytoplankton carbon assimilation (Pb:Pph;) and chlorophyll a concentration are shown. Values are the average for the mean depth of each basin.

Basin (mean depth)	**BOx**(1) (mmol O2/m³ • yr)	**±CV BOx** (%)	**Bact. biomass** (mmol C/m³)	**±CV biomass** (%)	**P_b:P_{ph}** (-)	**±CV P_b:P_{ph}** (%)	**N_b:Chl**(2) ([µmolC/m³]/[µg/l])	**±CV N_b:Chl** (%)
Bothnian Bay (41 m)	64	43	1.5	6.6	0.8	47	68	15
Öre estuary (16 m)	153	23	2.1	13	0.5	48	27	35
Bothnian Sea (66 m)	67	43	1.6	17	0.6	78	82	16

(1) Bacterial oxygen consumption was calculated from carbon production using literature values for bacterial growth efficiency (30%, (Zweifel et al., 1993)) and respiration quotient (0.9, (Roy et al., 1999)). A thymidine conversion factor of 1.5 • 10^{18} cells/mol was used (Wikner and Hagström 1999).
(2) Chlorophyll a is often used as a proxy for phytoplankton biomass. However, the chlorophyll:biomass ratio is variable and the pigment content may also change due to light conditions. This variable should be interpreted with caution, especially when comparing different basins.

in Table 5.5). As a consequence the productivity of the larger zooplankton, and hence sedimentation, are low. In the Bothnian Sea, in contrast, a larger part of the carbon supply is in the form of large phytoplankton (Andersson *et al.*, 1996). The latter support zooplankton productivity and contribute to a higher sedimentation rate. Despite the similarity in volumetric bacterial carbon respiration in the basins, the source of bacterial substrates and carbon flow in the Bothnian Bay and Bothnian Sea food webs was thus clearly different.

Swedish coast.

By: Agneta Andersson

Stable environmental conditions in the Öre estuary in the Gulf of Bothnia. No significant trends were detected for either phytoplankton or physical-chemical parameters at the pelagic monitoring station BMP B3 in the Öre estuary in the Gulf of Bothnia from 1991–99, thus indicating that no major environmental changes have occurred in this marine area during the past ten years.

To assess the pelagic environmental condition in the Öre estuary, five commonly occurring phytoplankton species were selected for trend analysis (Figure 5.41). Two of these are spring bloom species (the diatom *Thalassiosira baltica* and the dinoflagellate *Peridiniella catenata*). One species peaks just after the spring bloom (the autotrophic ciliate *Mesodinium rubrum*), while the remaining two species are most abundant in late summer (blue-green algae of the genus *Synechococcus* and prasinophyceans of the genus *Pyramimonas*).

The spring bloom species show relatively large inter-annual variation (CV = 0.7; max:min = 10–40; Table 5.6). One important factor influencing the estimated average abundance is the sampling frequency. The spring bloom spans over a short period during which the phytoplankton increases exponentially with a generation time of about 2–3 days. It may be hard to find the absolute spring bloom maximum if sampling is performed once per 14 days. The duration of ice cover, which varied between 1.5 and 4 months per year during the period 1991–99, probably also influences the abundance of these algae. The summer phytoplankton exhibited lower inter-annual variation (CV = 0.3; max:min = 2.4; Table 5.6). The low abundance of *Synechococcus spp.* during 1998 may be due to the cold summer that year, these blue-green algae being known to be favoured by high water temperatures. None of the investigated phytoplankton exhibited significant trends during the 1990s.

The analysed physical-chemical parameters generally showed lower inter-annual variations than phytoplankton (CV = 0.03–0.1; max:min 1.1–1.7; Table 5.6, Figure 5.42). The yearly average temperature was approx. 4.4°C and the salinity 4.4 PSU

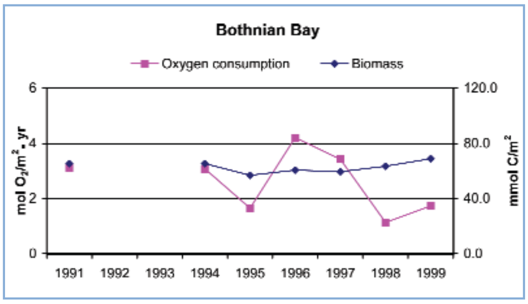


Figure 5.35
Annual bacterial community oxygen consumption and biomass integrated over the average depth of the Bothnian Bay (41m, stn. BMP A13).

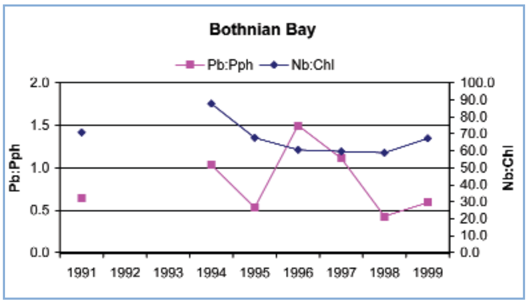


Figure 5.36
Bacterioplankton:phytoplankton ratio for production and biomass in the Bothnian Bay (station BMP A13). The bacterial carbon production:phytoplankton CO_2 assimilation ratio (Pb:Pph)) and bacterial biomass: chlorophyll a ratio (Nb:Chl) are shown. The ratios are primarily used as indicators of the balance between pelagic production based on allochthonous and autochthonous sources (c.f. footnote 2, Table 5.5). The ratios were calculated from estimates of annual values integrated over the average water column (41 m).

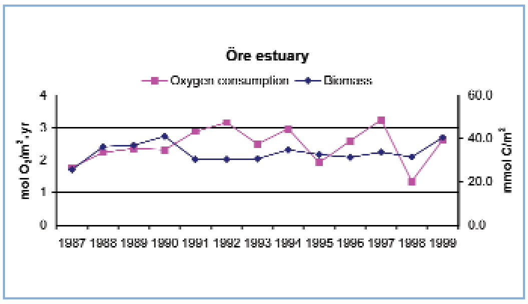


Figure 5.37
As for Figure 5.35, but for the Öre estuary (average depth 16 m, station BMP B3).

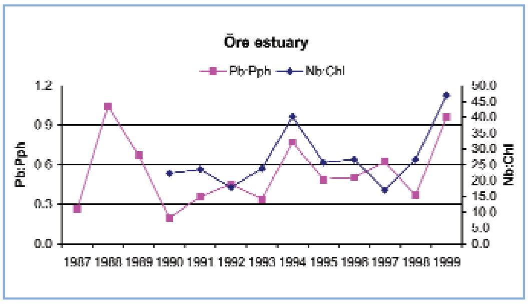


Figure 5.38
As for Figure 5.36, but for the Öre estuary (average depth 16 m, station BMP B3).

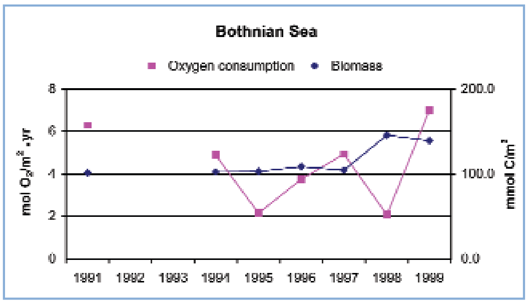


Figure 5.39
As for Figure 5.35, but for the Bothnian Sea (average depth 66 m, station BMP C1).

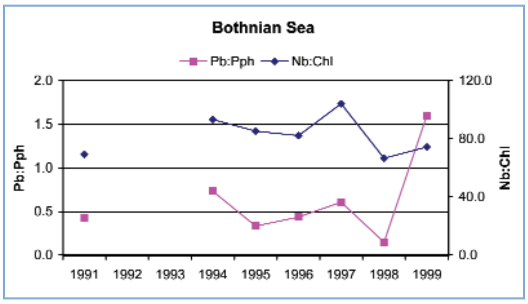


Figure 5.40
As for Figure 5.36, but for the Bothnian Sea (average depth 66 m, station BMP C1).

Figure 5.41 Phytoplankton abundance at station BMP B3 in the Öre estuary from 1991–1999. The figure shows annual mean abundance of some of the dominant phytoplankton expressed in cells per litre for all phytoplankton except for Synechococcus spp., which is expressed in cells per millilitre. Based on integrated seawater samples collected in the photic zone (0-20m) with a plastic hose on average 19 times per year.

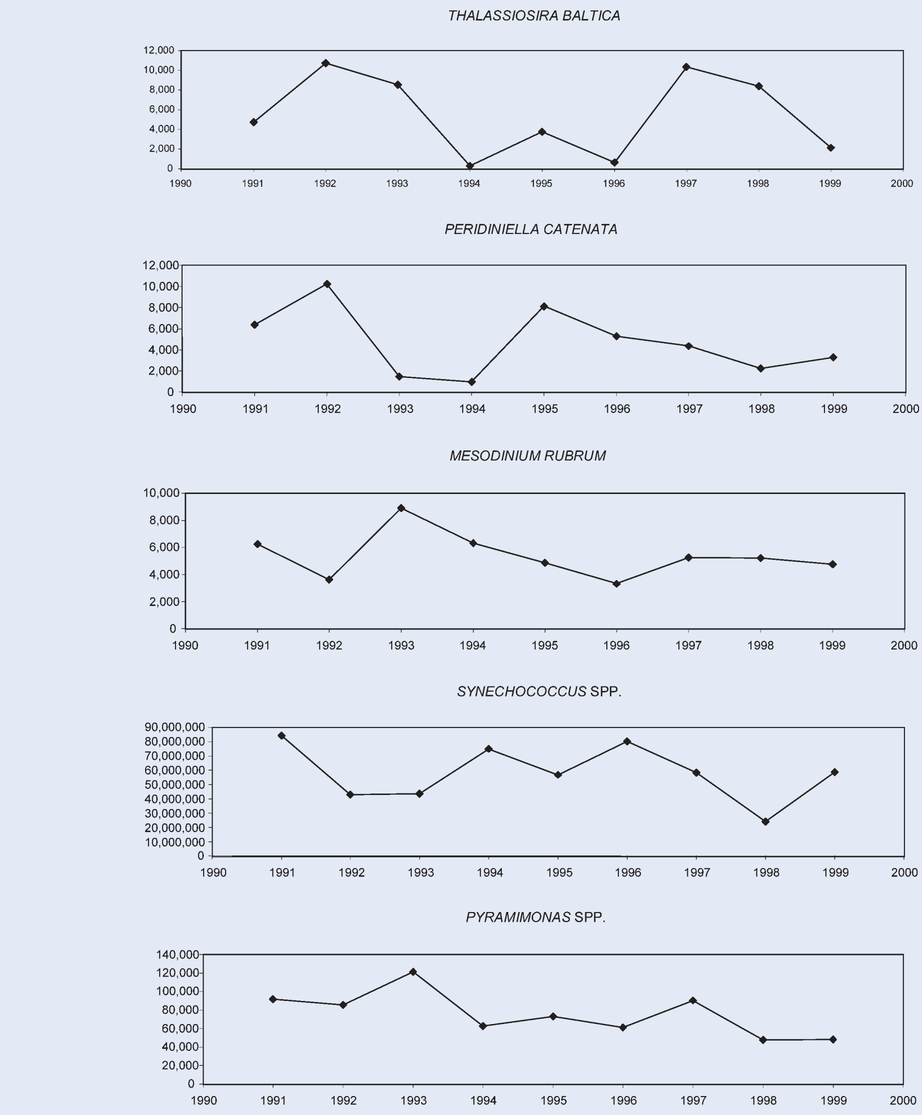


Table 5.6 The interannual variation is higher for phytoplankton than for abiotic factors at station BMP B3 in the Öre estuary. The table shows the coefficient of variation of annual mean values, the ratio of maximum:minium values and the detection limit for trend analysis. The assumption for the trend is that it will give 80% statistical power within a period of ten years. The samples were taken in the photic zone on average 19 times per year from 1991–99.

Phytoplankton	**CV**	**Max:Min**	**Detectable trend** (% change/year)
Thalassiosira baltica	0.7	36	21
Peridiniella catenata	0.7	11	21
Mesodinium rubrum	0.3	3	9
Synechococcus spp.	0.3	4	9
Pyramimonas spp.	0.3	2	9
Physical-chemical parameters			
Temperature	0.1	1.5	3
Salinity	0.04	1.1	1
Total P	0.1	1.3	3
Inorganic phosphorus	0.2	1.7	6
Total N	0.03	1.1	1
Inorganic nitrogen	0.1	1.4	3
Inorganic silica	0.1	1.2	3

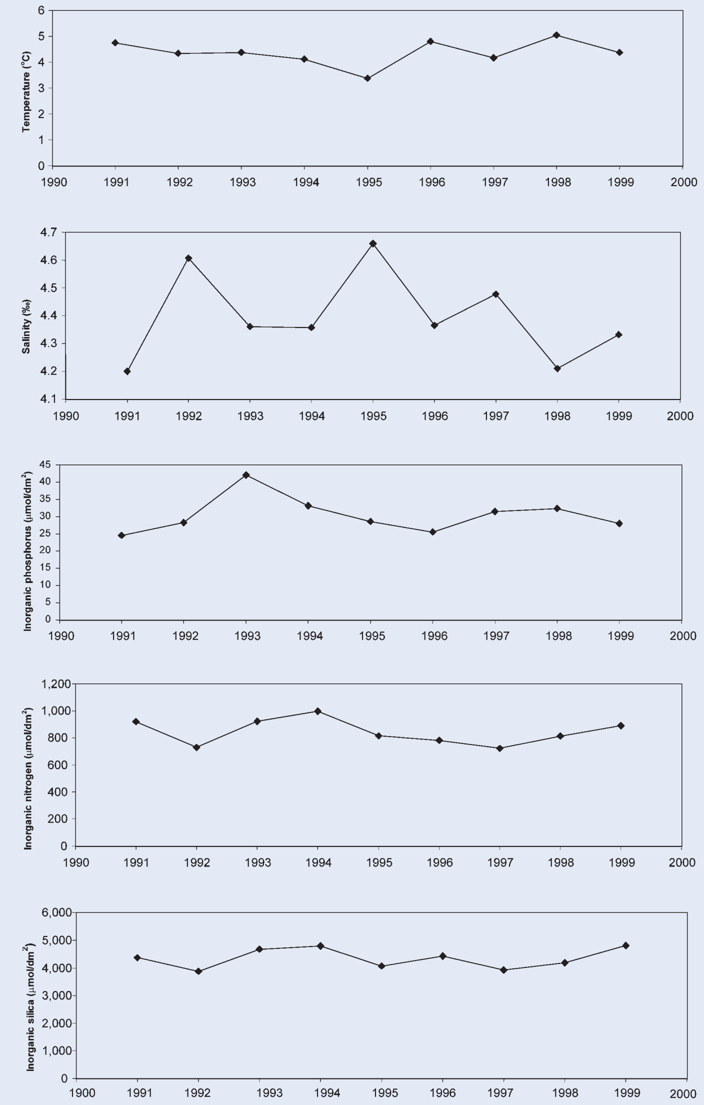


Figure 5.42
Physical-chemical parameters at station BMP B3 in the Öre estuary from 1991–1999. The figure shows annual mean values of temperature, salinity and inorganic phosphorus, nitrogen and silica based on integrated seawater samples collected in the photic zone (0-20 m) with a plastic hose on average 19 times per year.

at depths of 0–20 m. The average concentrations of inorganic nitrogen, phosphorus and silica were also relatively stable during the period studied. The molar ratio of inorganic nitrogen to phosphorus was about 30, indicating phosphorus limitation during the productive period of the year. None of the investigated physical-chemical parameters exhibited any significant trends during the 1990s.

Finnish coast.

By: H. Pitkänen, P. Kauppila, and P. Kangas

Areal distribution of nutrients in winter. In general, the late winter concentrations of both total and inorganic N and P are indicative of the trophic conditions and nutrient sources around the coast since the activity of primary producers is negligible. The winter level of inorganic nutrients also directly controls the level of the spring algal bloom. The phytoplankton spring bloom and summertime productivity decrease from south to north, being lowest in the phosphorus-limited Bothnian Bay.

The average DIN reserves in winter surface water in 1991–96 were below 7 µmol/l in the open Bothnian Sea and varied between 7 and 14 µmol/l in the open Bothnian Bay and in the whole of the Finnish coastal waters (Pitkänen and Kauppila, 2001). In the inner Archipelago Sea and innermost coastal areas receiving nutrient inputs from the land, the mean DIN concentratio-n ranged up to 35 µmol/l. Near the bottom the distribution was quite similar to that at the surface layer. However, the coastal areas receiving considerable amounts of river water rich in nitrogen exhibited lower concentrations in near-bottom waters than in the surface layer.

The highest winter phosphate reserves (0.6 –1.0 µmol P/l) were found in the surface layer off river mouths and towns in the Bothnian Bay and in the Archipelago Sea. In the whole of the Archipelago Sea the phosphate concentrations exceeded 0.3 µmol P/l, decreasing gradually to the north and being even less than 0.06 µmol P/l in the central open part of the Bothnian Bay. In spite of the major nutrient inputs from several large rivers to the Gulf of Bothnia, the concentration of bioavailable phosphorus in the Bothnian Bay is low due to effective chemical precipitation of phosphorus by iron compounds.

The winter distribution of the DIN:DIP ratio clearly indicates phosphorus-limited spring production in the Bothnian Bay due to the high N:P ratio of the river water (e.g. Alasaarela, 1980; Pitkänen, 1994) and effective phosphorus precipitation in the Bothnian Bay (Voipio, 1969). In the Bothnian Sea the ratio indicates co-limitation of nitrogen and phosphorus. In the outer Archipelago Sea, though, nitrogen seems to be the primary factor limiting the spring bloom. The role of N as the primary limiting factor has strengthened in the Archipelago Sea during the 1990s (Fig. 5.43, cf. Kirkkala *et al.*, 1998).

Nutrient trends in the Archipelago Sea. Eutrophication of the Archipelago Sea has proceeded during the 1990s (Bonsdorff *et al.*, 1997; Kirkkala *et al.*, 1998), although winter concentrations of phosphate and total phosphorus have increased slightly in the surface layers. DIN has clearly decreased (Fig. 5.43; Pitkänen and Kauppila, 2001). The concentration of both N and P increased during the 1970s and 1980s (Pitkänen *et al.*, 1987; Kirkkala *et al.*, 1998) due to intensification of agriculture and to fish farming. In contrast, the P concentration in waters near larger towns has decreased due to the introduction of phosphorus stripping at the municipal wastewater treatment plants.

The summer near-bottom concentrations of phosphate in the central Archipelago Sea increased markedly from 0.3 to 1.0 µmol P/l during the late 1980s and 1990s (Pitkänen and Kauppila, 2001). At the same time the deep-water oxygen concentrations continuously decreased, reaching 6–7 mg/l (approx. 50% saturation) in 1997. This indicates increased sedimentation of organic detritus caused by increased primary production and/or possible anoxia and internal loading at the sediment-water interface. At the same time, a slight increase in summer phosphorus concentration was also evident in the surface water.

The central Archipelago Sea is mainly limited by nitrogen during the mid and late summer months, although the situation can oscillate between nitrogen limitation and phosphorus limitation (Kivi *et al.* 1993; Kirkkala *et al.*, 1998). The generally increased DIP and de-creased DIN evidenced by the winter data indicate further strengthening of this trend towards the late 1990s (Pitkänen and Kauppila, 2001). The findings of Kirkkala *et al.* (1998) based on total nutrients from the middle and southern Archipelago Sea further support this conclusion.

A possible contributory factor is the increased flow of phosphorus from the Gulf of Finland, where

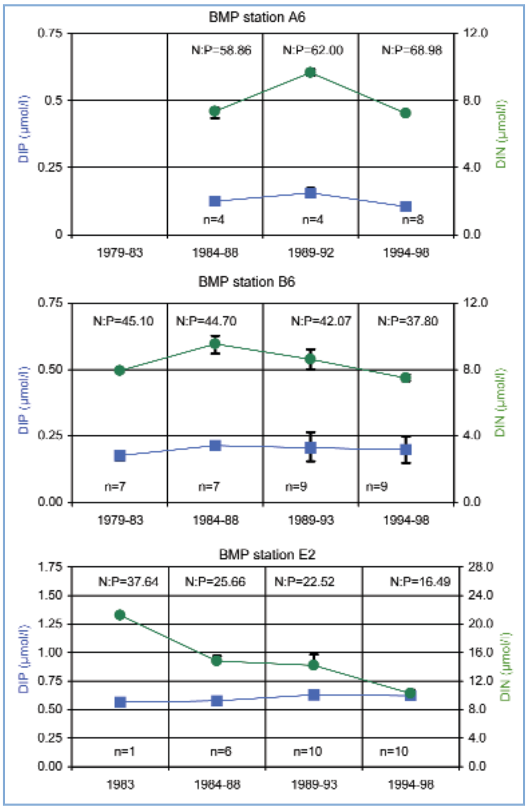


Figure 5.43
Areal distribution of chlorophyll a in the marine waters around Finland. Mean of records in July–September 1991–1996 (Kauppila and Lepistö, 2001).

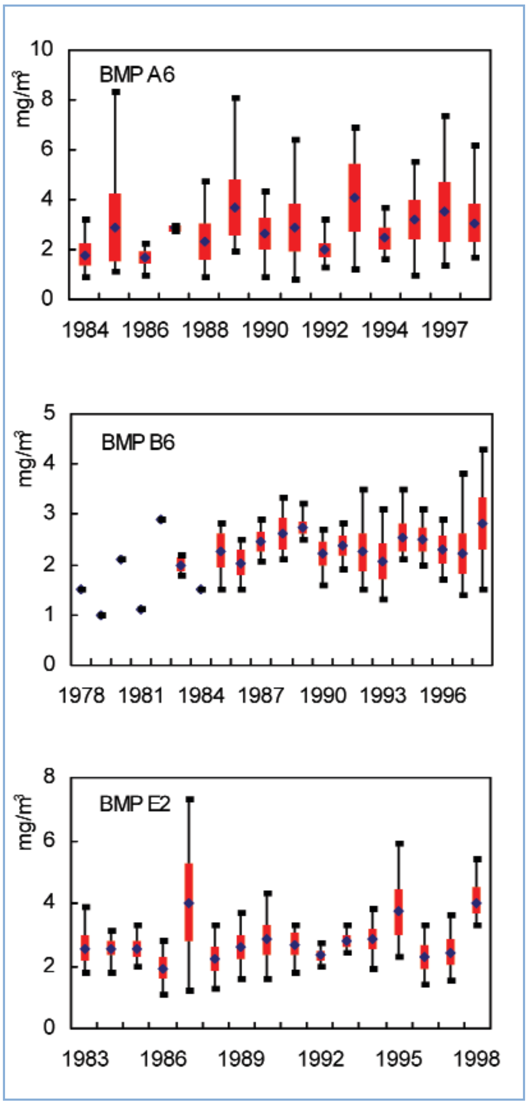


Figure 5.44
Chlorophyll a concentration at three stations of the Archipelago Sea (Seili), the Quark (Bergö) and the NE Bothnian Bay (Hailuoto) during the period 1979–1998 (Kauppila and Lepistö, 2001).

the phosphorus concentration increased markedly in the late 1990s. Another factor pulling in the same direction is the trend towards increasing near-bottom phosphate concentrations via vertical mixing in the autumn.

Nutrient trends in the Bothnian Sea and Bothnian Bay. In general, the changes seen in the Gulf of Bothnia were less clear than in the Archipelago Sea. Winter DIN levels tended to decrease in both the Finnish coastal areas of the Bothnian Sea and the Bothnian Bay during the 1990s (Fig. 5.43). The data from the open Gulf also suggest a similar conclusion when comparing the DIN concentrations in the 1990s to those in the early 1980s (Pitkänen *et al.*, 1987). The trend may be associated with the decrease in the riverine nitrogen load in the 1990s that is mainly attributable to decreased runoff. The balance between the limiting nutrients may also be changed due to the increased P concentrations in the open Bothnian Sea between the early 1980s and the 1990s (Pitkänen and Kauppila, 2001).

In the NE Bothnian Bay, the clear decreases in both nitrogen and phosphorus concentrations since the late 1980s were probably attributable to the improvement in sewage treatment and to smaller riverine runoff in the 1990s. The trend has been more pronounced for phosphorus than for nitrogen. Total phosphorus decreased by 0.13 to 0.16 µmol/l during the period, which is in accordance with the approx. 50% reduction in phosphorus load from local point sources (Pitkänen and Kauppila, 2001).

Phytoplankton

By: P. Kauppila and L. Lepistö

Chlorophyll a. In the central Archipelago Sea, the chlorophyll a concentration (below 3 mg/m^3) equalled that observed in the open sea. The average chlorophyll a concentration has slightly increased from 2.7 mg/m^3 in the early 1990s to 3.0 mg/m^3 in the present assessment period (Figure 5.44). The border of the slightly eutrophic area moved westwards during the 1980s and 1990s despite the reduction in anthropogenic nutrient loads. In the Quark, the average chlorophyll a concentration is low but has increased slightly from 2.2 mg/m^3 in the early 1990s to 2.5 mg/m^3 during the present assessment period. In the NE Bothnian Bay the chlorophyll a concentration is higher, and the mean has increased from 2.9 mg/m^3 to 3.1 mg/m^3 (Kauppila and Lepistö 2001).

Phytoplankton biomass and species composition. The mean biomass in the middle Archipelago Sea was 1.6 g/m^3. The spring mean biomass, dominated by diatoms, was moderately high, 2.8 g/m^3, but the summer biomass was only 0.4 g/m^3, with slight inter-annual variation. Dinoflagellates and cryptomonads dominated in summer. In 1997 and 1998, blue-green algae were abundant, probably due to increased N limitation. In the Quark, the diatom-dominated spring biomass was 0.8 g/m^3 as compared with only 0.3 g/m^3 in summer, with very low inter-annual variation. Like in oligotrophic lakes, different phytoplankton groups occurred in equal quantities in summer. In the NE Bothnian Bay, phytoplankton biomass was also dominated by diatoms and was 0.8 g/m^3 in both spring and summer (Kauppila and Lepistö 2001, Figure 5.45).

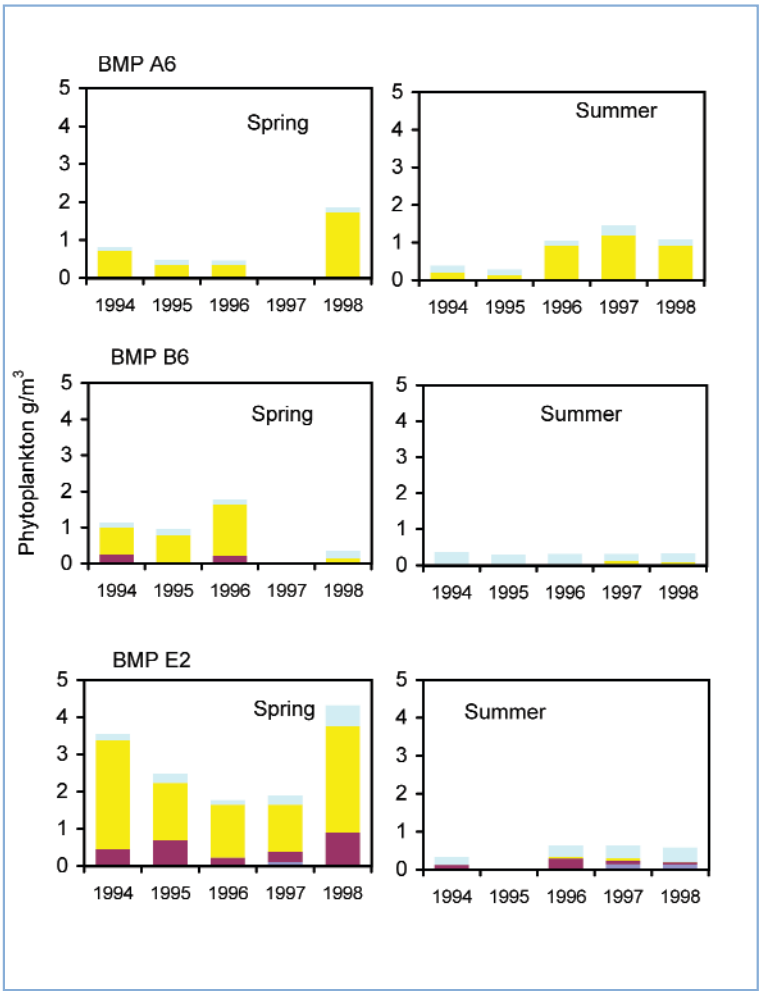


Figure 5.45
Phytoplankton abundance and composition at three sites in the Archipelago Sea (Seili), the Quark (Bergö) and the NE Bothnian Bay (Hailuoto) during the period 1994–1998 (Kauppila and Lepistö, 2001).

▨ Others
🟨 Diatomop.
🟥 Dinoph.
🟪 Cyanoph.

Achnanthes taeniata and *Chaetoceras wighamii* dominated the spring phytoplankton in the Archipelago Sea. Blue-green algae, especially the N$_2$-fixing *Aphanizomenon* sp., were occasionally observed already in the spring, but otherwise occurred in the summer in moderately high quantities at the same time as *Nodularia* sp. were relatively abundant. Further northwards in the Quark, however, blue-green algae were scarce in summer, when the community is dominated by cryptomonads such as *Plagioselmis prolonga*, chrysomonads such as *Uroglena* sp. and green algae such as *Hemiselmis* sp. and *Pyramimonas* sp. Due to the northerly location, the spring bloom in the NE Bothnian Bay develops in late May or in early June, and is dominated by *Diatoma tenuis* (Kauppila and Lepistö, 2001).

Exceptional phytoplankton blooms

By: L.Lepistö, P. Kauppila and J.-M. Leppänen

During the assessment period blue-green algal blooms were common in the Archipelago Sea, the Åland Sea and the Bothnian Sea, but were generally absent in the Bothnian Bay. The surface accumulations were less intense and extensive in the Bothnian Sea than in the Baltic Proper. In the Archipelago Sea, the blooms were local. In 1994, scattered cyanobacterial blooms covered most of the area of the Archipelago Sea. Intensive *Chlamydomonas* blooms in May and a diatom bloom in June-July were detected in coastal waters off Kokkola in the Bothnian Bay. Local *Chlamydomonas* blooms were observed in the Archipelago Sea at the end of April. In 1995, no exceptional blooms occurred. In 1996, the cyanobacterial blooms in the Bothnian Sea were more intensive than during previous years. In the Åland Sea, intensive local blooms formed by the dinoflagellate *Heterocapsa triquetra* coloured the water reddish-brown in August. In 1997, cyanobacterial surface accumulations were the most extensive and prolonged ever recorded in the area, as in other parts of the Baltic Marine Area. Large amounts of biomass drifted ashore, particularly in the Archipelago Sea. Cyanobacteria were also observed in large quantities in the open Bothnian Sea south of Kristiinankaupunki. The windy weather did not favour extensive cyanobacterial surface accumulations in 1998.

The cyanobacterial blooms are favoured by the lower inorganic N:P ratio. In recent years, when eutrophication has proceeded in the Archipelago Sea, production has shifted from one limited by both nutrients to one solely limited by nitrogen (Kirkkala *et al.*, 1998). During the fairly calm and rainy weather of the summer 1998 algal blooms were particularly abundant, including in the inner Archipelago Sea, where they are usually scarce. Mass occurrences of algae outside the normal season indicated continued eutrophication. In January 1998, mass occurrence of *Aphanizomenon* sp. was observed in the southeastern coast of the Archipelago Sea during long-lasting calm and mild weather (Kauppila and Lepistö, 2001).

5.2.4. Benthic conditions and macrofauna in the Gulf of Bothnia

Phytobenthos

By: S. Bäck, H. Kautsky, J. Mattila and A. Mäkinen.

As no long-term monitoring has been performed along the coasts of the Gulf of Bothnia, very little can be stated about the trends in the development of this somewhat impoverished ecosystem. Development in the area does not appear to be negative, however, as the depth distribution of submerged macrophytes is well in accordance and even somewhat deeper than found in comparatively clean areas of the Baltic Proper. On hard substrates several common algal species found in the Baltic Proper occur in high abundance. The bladder wrack (*Fucus vesiculosus*) is still belt forming up to its northern limit at the northern Quark, where it still covers 75% of the substrate.

Distribution and biomass of submerged macrophytes and drifting algae were studied in shallow, brackish (0–1 m), soft-bottom areas in the Åland archipelago during 1997–1999 by means of aerial photography and ground truth sampling. On average, 18 underwater macrophyte species were detected in the study areas. The most common species by weight and occurrence were *Chara aspera, Cladophora glomerata, Pilayella littoralis* and *Potamogeton pectinatus*. Mean vegetation coverage varied between 30 and 50%. No statistically significant differences were found between different grades of exposure, both sheltered and exposed bays being threatened by drifting algae.

Hard-bottom vegetation on Åland was previously studied in the 1970s. A follow-up study was carried out in 1999 by means of scuba diving. Six transects were revisited in the southeastern and southern parts – some of which are exposed to ferry traffic. Results from the current study show that the surge from the ferries keeps hard bottoms free of filamentous algae (Berglund and Roos, 2000). Compared with sites sheltered from the ferry traffic the number of species is lower but the biomass is similar. The latter can be seen as indicating eutrophication. Abundance of *Fucus vesiculosus* is clearly lower now than in the 1970s. Large drifting algal mats were also observed along the transects.

Mass occurrence of macroalgal mats. During the past ten years, the occurrence of drifting algal masses has become a problem in archipelago waters of SW Finland (Norkko and Bonsdorff, 1996a,b). The ecological impacts of these masses are drastic, severely affecting the perennial algal and animal communities (Norkko and Bonsdorff, 1996a,b). Even when only comprising a thin cover, drifting algal masses shadow and disturb the underlying vegetation.

The occurrences of drifting filamentous algal masses were mapped in the Archipelago Sea during the summers 1996–97 using an underwater video camera and Scuba diving (Vahteri *et al.*, 2000). The study covered depths from the shore down to 50 metres. The algal masses were described and classified according to the thickness of the mass and possible occurrence of anoxia (smell of hydrogen sulphide, black coloured sediment), i.e. indicators of their potential different ecological impacts. The succession of the masses was studied during one growing season. First, the algae aggregated on the sea floor as a thin cover. Later they started to drift downwards along the sloping bottom, developing into partly anaerobic mats and thereafter into totally anaerobic mats. The largest recorded drifting algal masses were approx. 0.3 km^2. Factors contributing to the growth of ephemeral algae, which result in drifting algal mats, include a high nutrient loading, good water transparency and appropriate bottom substrate.

Extensive mats of a single species of *Enteromorpha* spp. with high biomass have been reported to cover large sheltered areas of Uusikaupunki archipelago and Eurajoki estuary on the west coast of Finland (Lampolahti, 1997; Bäck *et al.*, 2000). *E. intestinalis* thalli had lost their tubular shape, spread, and formed unattached monostromatic sheets. *E. intestinalis* occurred in greatest abundance close to the shoreline, where the water was 1–2 m deep. It favoured the sheltered coastal areas and was not found in open shore locations. Black sulphurous sediments were occasionally found beneath the mats. Monostromatic *E. intestinalis* appeared to remain healthy even during severe winter conditions. In early spring, just after the ice had thawed, small amounts of *E. intestinalis* were found floating on the surface. Extensive floating mats were observed after periods of sunny, calm weather with low water level. On both occasions, *E. intestinalis* moved within the water column.

Monostromatic, sheet-like *E. intestinalis* occupies an unusual niche for macroalgae. It grows successfully unattached in shallow water on sediment bottoms, and can tolerate the harsh winter conditions of the Baltic Sea. On the basis of these findings it is believed that macroalgal mats may have the potential to cause serious changes to the Baltic shallow water ecosystem. The mats are extremely persistent and well able to tolerate a variety of environmental conditions and seasonal changes. If the occurrence of macroalgal mats is increasing in response to pollution as suggested by Fletcher (1996) and Bonsdorff *et al.* (1997), a significant threat to coastal ecosystems is indicated.

Macrozoobenthos

By: K. Leonardsson, A. Laine and A. Andersin

The present assessment period, 1994–1998. The benthic macroinvertebrate community in the open sea area of both basins of the Gulf of Bothnia is totally dominated by the amphipod *Monoporeia affinis* and the isopod *Saduria entomon*. The amphipod *Pontoporeia femorata* and the marine polychaete *Harmothoe sarsi* occur sparsely in the southern Bothnian Sea. According to the data available for this period the variations were small compared to the fluctuations observed earlier in the time series. The time series for macrofaunal total abundance and biomass at selected HELCOM stations in the open sea for which long time series exist are shown in Figure 5.46. Average abundance and biomass at these stations during the assessment period are given in Tables 5.7 and 5.8. *M. affinis* density has remained high at the two stations in the Bothnian Bay and was generally lower in the Bothnian Sea except at station BMP C2. Biomass of the species was nevertheless higher in the Bothnian Sea than in the Bothnian Bay due to the larger size of the individuals in the Bothnian Sea. *M. affinis* dominated the macrofauna numerically at all stations. At BMP D1, *P. femorata* also occurred at high densities. When considering the biomass, *S. entomon* dominated in the southern Bothnian Sea at stations BMP C4 and BMP D1. This

Table 5.7. Average density (individuals per square metre) during the period 1994-1998 of the most common macrofauna species at HELCOM stations in the Gulf of Bothnia.

Taxa	A3	A2	C3	C11	C4	D1
M. affinis	2012.1	1812.1	6262.9	717.0	1490.9	1275.6
P. femorata	0	0	0	0	6.0	691.7
S. entomon	3.1	1.2	6.4	5.1	22.7	14.5
H. sarsi	0	0	0	0	0.4	41.5
M. balthica	0	0	0	114.7	0	0
Others	0	0	0	1.0	0	0.4
Total	**2015.2**	**1813.3**	**6269.4**	**837.7**	**1519.9**	**2023.8**

Table 5.8. Average wet weight (grammes per square metre) during the period 1994-1998 of the most common macrofauna species in the Gulf of Bothnia.

Taxa	A3	A2	C3	C11	C4	D1
M. affinis	3.75	3.30	23.84	6.06	7.78	7.16
P. femorata	0.00	0.00	0.00	0.00	0.05	5.63
S. entomon	1.60	0.93	14.93	0.89	31.03	13.13
H. sarsi	0.00	0.00	0.00	0.00	0.01	0.09
M. balthica	0.00	0.00	0.00	12.92	0.00	0.00
Others	0.00	0.00	0.00	0.01	0.00	0.00
Total	**5.35**	**4.23**	**38.77**	**19.88**	**38.86**	**26.00**

Figure 5.46
Change in total macrofauna wet weight at selected open-sea HELCOM stations in the Gulf of Bothnia. Red lines denote total abundance, while blue lines denote total biomass. The average abundance and biomass for these stations during the present assessment period are given in Tables 5.7 and 5.8. The densities of M. affinis have remained high during the two periods. There are no significant differences between the current and the previous assessment periods as regards the overall medians for each basin (Table 5.11).

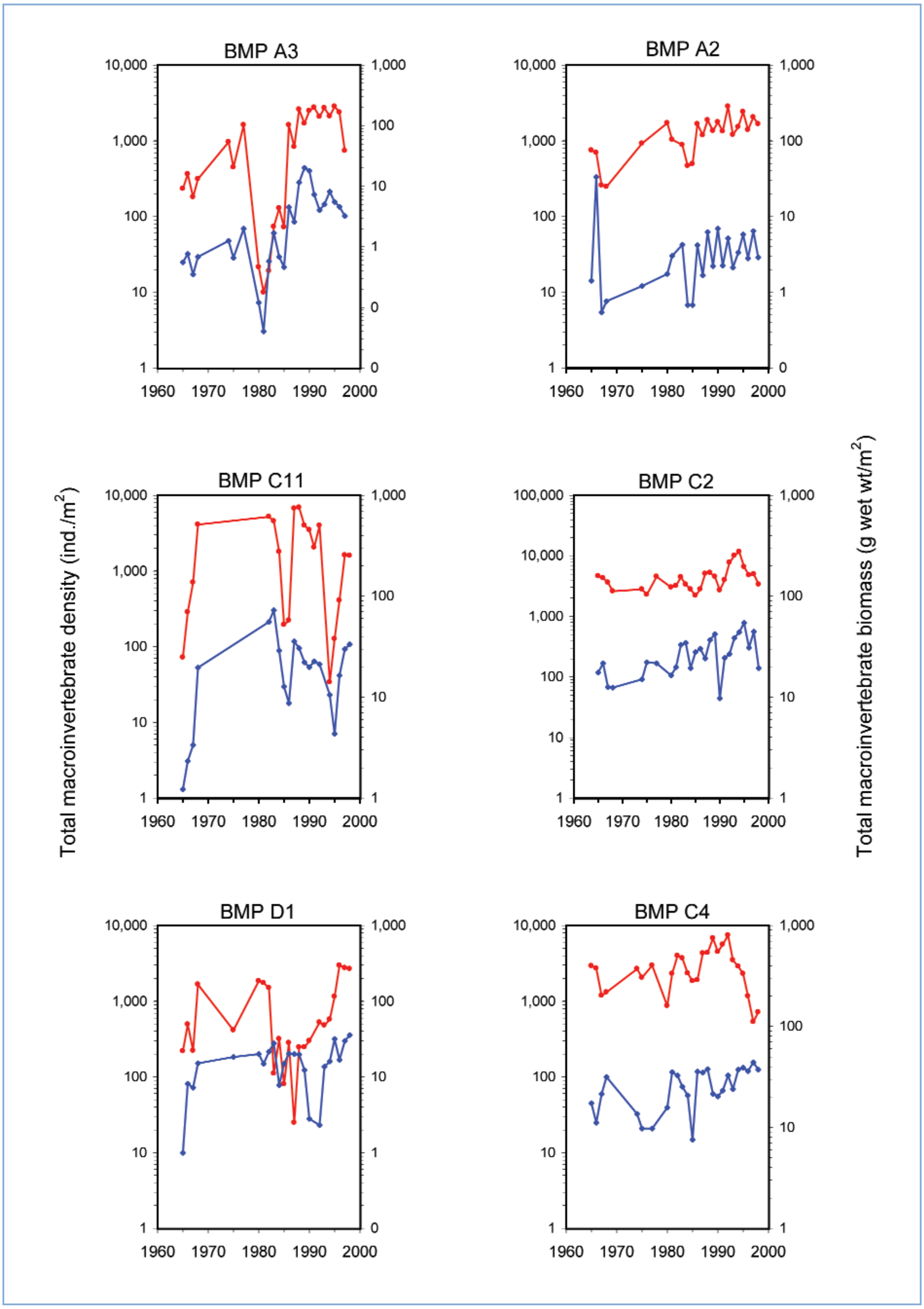


Table 5.9
Average abundance (ind./m²) during the period 1995–98 of the most common (>1% of total) macrofauna taxa along the Swedish coast. The different areas include 20 stations each with one replicate per station and year. The areas in the table are named from south to north.

	Bothnian Sea			Bothnian Bay	
Taxa	**Söderhamn**	**High Coast**	**Norrbyn**	**Piteå**	**Råneå**
Asellus aquaticus	0.8	0.0	0.0	15.7	0.0
Chironomidae	131.9	49.9	1.0	58.0	11.9
Gammarus sp.	0.3	0.4	4.7	25.3	0.0
Macoma balthica	521.2	121.9	240.4	0.0	0.0
Marenzelleria viridis	0.0	0.2	7.7	0.0	0.0
Monoporeia affinis	1,147.8	5,009.7	1,726.5	580.1	197.5
Oligochaeta	139.7	65.8	105.2	210.6	21.2
Saduria entomon	6.9	49.0	33.6	9.0	2.9
Others	20.2	4.1	4.4	7.4	1.0
Total	**1,968.8**	**5,300.9**	**2,123.5**	**906.2**	**234.6**

	Bothnian Sea			Bothnian Bay	
Taxa	**Söderhamn**	**High Coast**	**Norrbyn**	**Piteå**	**Råneå**
Chironomidae	2.33	1.12	0.00	0.09	0.03
Gammarus sp.	0.00	0.01	0.03	0.22	0.00
Macoma balthica	119.11	24.36	47.02	0.00	0.00
Marenzelleria viridis	0.00	0.00	0.54	0.00	0.00
Monoporeia affinis	10.25	22.03	9.09	2.31	0.60
Oligochaeta	0.33	0.11	0.16	0.72	0.07
Saduria entomon	1.39	7.57	3.55	1.78	0.77
Others	0.37	0.05	0.09	0.06	0.03
Total	**133.79**	**55.23**	**60.45**	**5.23**	**1.49**

Table 5.10
Average wet weight (g wet wt./m^2) during the period 1995–98 of the most common (>1% of total) macrofauna taxa along the Swedish coast. The different areas include 20 stations each with one replicate per station and year. The areas in the table are named from south to north.

species also occurred in high biomass at station US 6B in the northern Bothnian Sea. The bivalve *Macoma balthica* was only present at station BMP C11. *H. sarsi* occurred sparsely in the southern Bothnian Sea at BMP C4 and BMP D1 (Tables 5.7 and 5.8).

Information from four years of sampling beginning in 1995 provides a good description of the benthic community in the Swedish coastal region (Tables 5.9 and 5.10). Eight species each account for more than 1% of the total abundance or biomass in one or more of these coastal areas. In total, 49 species were recorded during the period, of which 26 were species of Chironomidae – midge larvae. *M. affinis* dominated numerically in all areas, while *M. balthica* was the numerically second most common species in the Bothnian Sea. In the Bothnian Bay, oligochaetes were second most common. In terms of biomass, *M. balthica* dominated in the Bothnian Sea, followed by *M. affinis*. In the Bothnian Bay, *S. entomon* and *M. affinis* were equally dominant. The total biomass was higher in the coastal zone in the Bothnian Sea than in the open sea, mainly due to the high abundance of *Macoma balthica*. In the Bothnian Bay, the total biomasses in the coastal zone were of comparable magnitude to those in the open sea. Species composition was influenced by fresh water, as reflected by the occurrence of the freshwater isopod *Asellus aquaticus* and many of the chironomid species.

During the assessment period the polychaete *Marenzelleria viridis* started colonizing the Swedish side of the Gulf of Bothnia (Figure 5.47). Thus the species was detected at three out of 155 stations in 1995, but at 17 out of 105 stations in 1998. Its main distribution is in the coastal region, although a few observations have also been made at deeper stations. *Marenzelleria* was first observed at the Finnish coast of the Bothnian Sea in 1992. The species seems to have dispersed northwards along the Finnish coast up to the northern Quark, and then west towards the Swedish coast in the northern Bothnian Sea. Its density is still relatively low, with the highest densities being found in the Norrbyn archipelago, where there was an average of ten individuals per square metre at the 20 coastal stations in 1998.

Basin	**Variable**	**U**	**P**
Bothnian Sea	Total abundance	5	0.117
	Total biomass	9	0.465
Bothnian Bay	Total abundance	21	0.076
	Total biomass	19	0.175

Table 5.11
Mann-Whitney U test comparison of the periods 1989–1993 and 1994–1998, (n=5 for each period). The test was performed on the index values.

Basin	***n***	**Z**	**P**	**Slope**
Bothnian Sea	19	-1.3	0.183	-0.026
Bothnian Bay	19	-1.5	0.134	-0.013
Total	**38**	**-2.0**	**0.043**	**-0.021**

Table 5.13
Results from the trend analyses based on the Hirsch and Slack non-parametric regression for average number of macrofauna taxa in the Gulf of Bothnia. The regression covers the time period 1980–98.

Comparison of assessment periods 1989–1993 and 1994–1998. There are no conspicuous differences between the two periods, neither in the Bothnian Sea, nor in the Bothnian Bay (Figures 5.46, 5.48 and 5.49), although there has been a tendency towards a decline in abundance and biomass during the past five years in the Bothnian Bay (Table 5.11).

Long-term trend, 1961–1998. The previously observed increasing long-term trends in total macrofauna abundance remain significant (Tables 5.12 and Table 5.13). However, the plateau seen since the late 1980s continued during the present assessment period although there is some indication of a decrease at the end of the period (Figures 5.48 and 5.49). The overall conclusion is that the

Basin	**Variables**	***n***	**Z**	**P**	**Slope**
Bothnian Sea	Total abundance	38	3.1	0.002	0.023
	Total biomass	38	4.1	<0.001	0.024
Bothnian Bay	Total abundance	38	4.6	<0.001	0.051
	Total biomass	19	1.8	0.069	0.094

Table 5.12
Trend analyses based on the Hirsch and Slack nonparametric regression for total abundance and biomass of the macrofauna in the Gulf of Bothnia. The regression covers the period 1961–98, except for total biomass in the Bothnian Bay, for which the period was 1980–98.

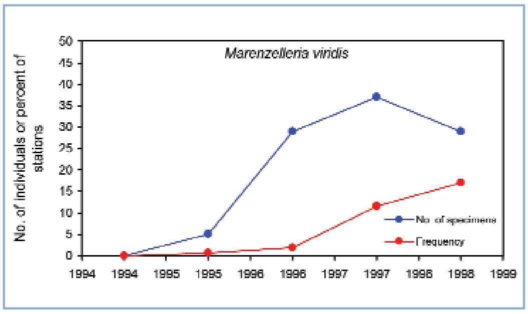


Figure 5.47
The occurrence of the invading polychaete Marenzelleria viridis in the Bothnian Sea and in the northern Quark. No observations have been made in the Gulf of Bothnia before 1994 in the BMP programme, or in the Swedish national or regional monitoring programmes.

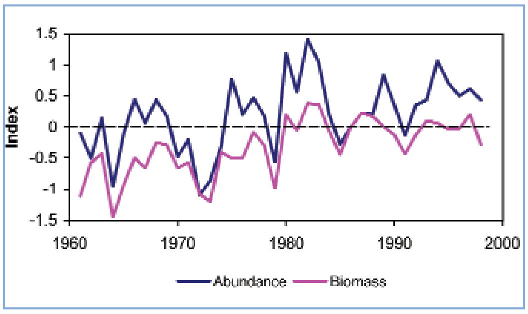


Figure 5.48
Total macrofauna index at stations in the Bothnian Sea over the past four decades. Base index year for abundance and biomass is 1986.

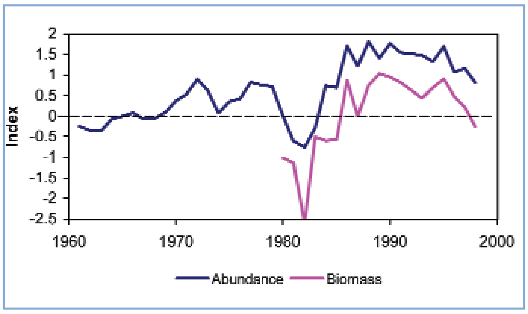


Figure 5.49
Total macrofauna index at stations in the Bothnica Bay over the past four decades. Base index year for abundance and biomass is 1986.

increase in the macrofauna community has ceased during the past ten years. As the earlier pronounced increase was mainly attributable to an indirect effect of increasing nutrient levels in the Gulf of Bothnia, the break in the trend could indicate that nutrient loading of the Gulf of Bothnia has stopped increasing during the last decade.

The average number of taxa per station seems to have declined weakly from 1980 to 1998 (Figure 5.50 and Table 5.13). Although the trends are not significant in the individual basins, the overall trend is significant. The species contributing to this pattern are those that occur at low densities, and their abundance therefore seems to have declined during the period, possibly due to the declining salinity in the region, especially in the Bothnian Sea. Species that suffer most from decreasing salinity are the marine *P. femorata* and *H. sarsi*. The more brackish water-tolerant species *M. balthica* has also declined during the period (1983–1998) at the stations where it is commonly found (Figure 5.51). The declining salinity may also account for the decline in *M. balthica*. At the station B6 in the northern Quark, massive recruitment of *M. balthica* took place at the beginning of 1980 from a previously low, but stable density. At this station the density now seems to be returning to the level observed during the 1970s.

Nutrient fluxes and sediment oxygen consumption in the Gulf of Bothnia. In the well-oxygenated Gulf of Bothnia, nitrogen fluxes were dominated by nitrate and ammonia efflux (Figure 5.52). Oxygen consumption correlated with the intensity of the dominant nitrogen flux in accumulation bottoms.

Phosphate fluxes across the sediment-water interface correlated with the phosphate concentration in the sediment pore water. Phosphate concentrations at a sediment depth of 2 cm and even deeper were influenced by the oxygen concentration of the near-bottom water. In the long term, oxygen consumption seems to be related to oxygen concentration in the near-bottom water, which may suggest that benthic communities are able to regulate their oxygen demand according to the environmental conditions (see Figure 5.57b in Chapter 5.3 on the Gulf of Finland).

High-resolution pore water nutrient profiles. Stations in the Gulf of Finland exhibited distinct seasonal patterns (see Gulf of Finland) whereas the nitrate peak at station BMP C4 in the Gulf of Bothnia was much smaller (5–10 µmol/l) (Figure 5.52), and no seasonal pattern was identified. At the deep, low-oxygen stations in the Baltic Proper, no nitrate peak was observed in the sediment.

Nitrogen losses through denitrification. At station BMP C4 in the Gulf of Bothnia, denitrification was between 250 and 300 µmol N/m^2 • d. These values are higher than in the low-oxygen deep stations in the northern Baltic Proper, but lower than in the central Gulf of Finland.

The bulk of the denitrification was based on NO_3- produced in the sediment by nitrification (Dn), i.e. coupled nitrification-denitrification. Compared to the Gulf of Finland and to the northern Baltic Proper, the Dn percentage was highest at station BMP C4 (97–100%).

Since the bulk of the denitrification was coupled to the NO_3- production by nitrification, factors regulating the rate of nitrification, e.g. supply of NH_4+ or O_2, are important for denitrification. Since nitrification is an oxic process while denitrification is mainly anoxic or suboxic, the most efficient coupling of nitrification and denitrification can be assumed to occur under moderate O_2 concentrations where the two processes can be located close to each other. In the Gulf of Bothnia, denitrification rates were low due to a low supply of organic matter to the benthos and resultant low NO_3 pools in the pore water (Figure 5.53).

Total denitrification was positively correlated with wet mass of benthic fauna and negatively correlated with depth. Since the abundance of benthic fauna fluctuates strongly in the Baltic Sea depend-

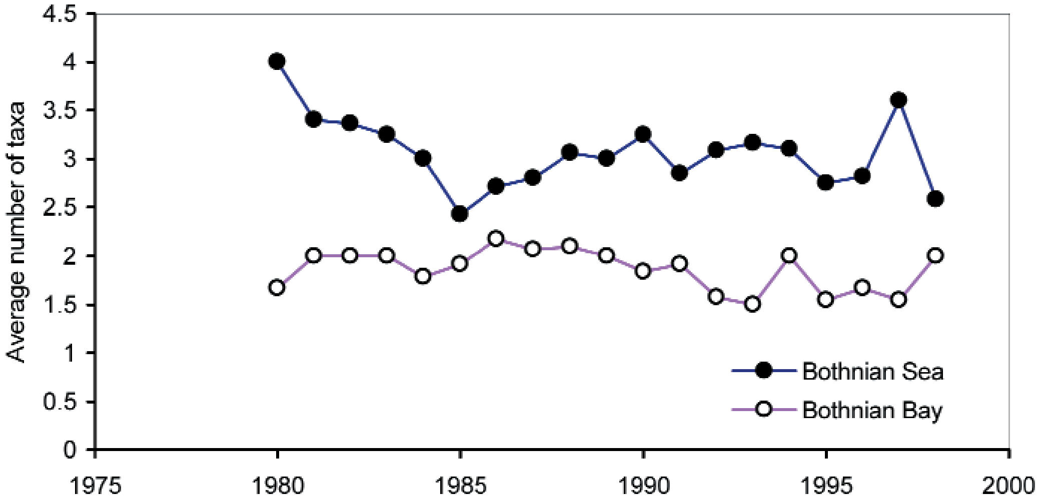


Figure 5.50
Average number of taxa in the Bothnian Bay and in the Bothnian Sea over the past two decades. Only data from open-sea stations are included. The information is based on 26 open-sea stations in the Bothnian Sea and 22 stations in the Bothnian Bay.

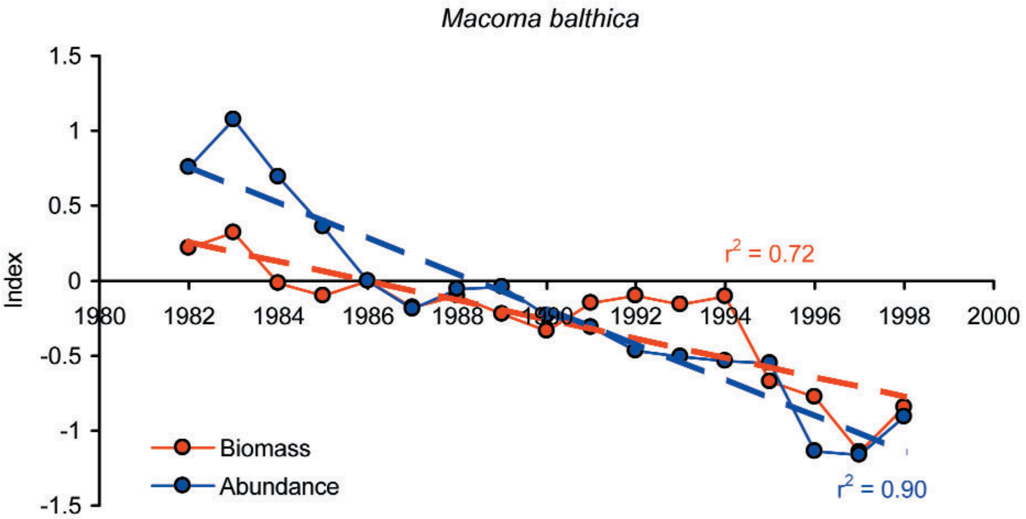


Figure 5.51
Abundance and biomass of Macoma balthica in the Bothnian Sea during the past 17 years. The base year is 1986. The data derive from the stations/areas in the Bothnian Sea at which M. balthica frequently occurs (BMP B2, BMP C11, Norrbyn archipelago and the open sea off Norrbyn). The broken lines denote the linear regression; r^2 is indicated.

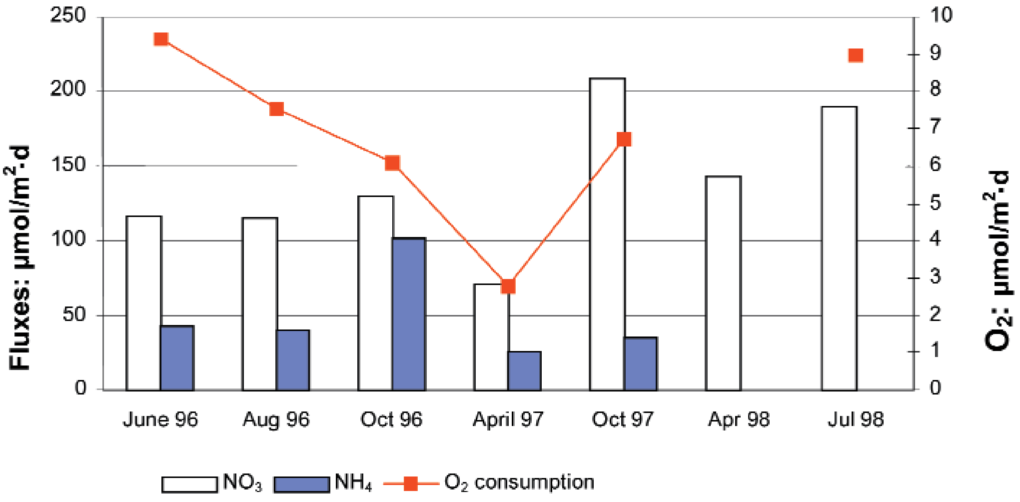
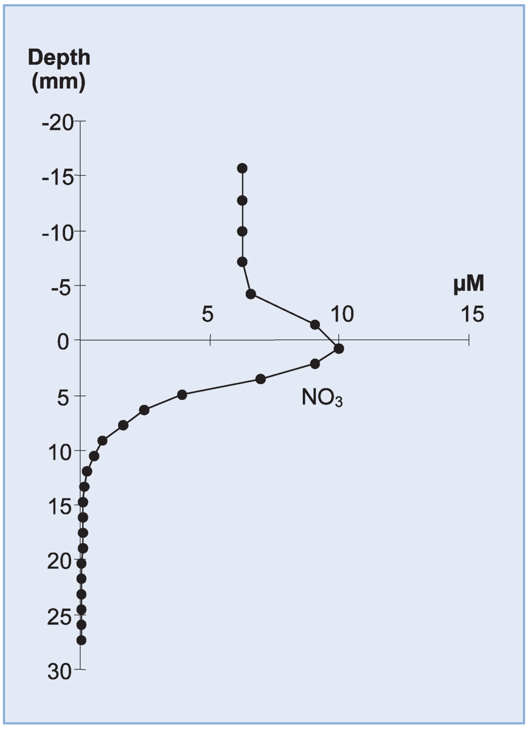


Figure 5.52
Nitrogen fluxes between sediment and water in the Gulf of Bothnia (station BMP C4).

Figure 5.53

Pore water nitrate concentration in the Bothnian Sea (station BMP C4) April 19, 1998.



ing on the hydrographic regime (Laine *et al.*, 1997), the rate of denitrification may also exhibit large temporal variation.

In summary, high organic loading and a high nitrate pool in the pore water coupled to active bioturbation favour denitrification. With increasing loading and decreasing oxygen supply, however, denitrification is depressed and the system loses its self-purification capacity (Tuominen *et al.*, 1998; Tuominen *et al.*, 1999). The Gulf of Bothnia is in a much healthier condition than the Gulf of Finland, where a small increase in settling organic matter may shift the sediment towards the loss of self-purification capacity.

5.3. Gulf of Finland

By: Convenor: U. Lips. Co-convenor: J.-M. Leppänen

The location of the monitoring stations in the Gulf of Finland is illustrated in Figure 5.54.

5.3.1. Meteorological, hydrological and hydrographical forcing

By: U. Lips, H. Pitkänen, P. Kauppila, P. Alenius and A. Nekrasov

Meteorology

The present assessment period was characterized by variable meteorological conditions. In general, mild winters (except for winter 1995/96) and warm summers (except for 1998) prevailed. The warmest summer during the assessment period was in 1997. In addition, northeasterly winds prevailed in the Gulf of Finland area in summer 1997 (in contrast to the common westerly-southwesterly winds). In general, the assessment period was characterized by drier than average weather conditions. The driest year of the assessment period was 1996, while the rainiest year was 1998.

Hydrology

Mean riverine runoff during the assessment period was close to the long-term average, but less than the mean of the two preceding periods. The highest riverine runoff into the Gulf of Finland was observed in 1995 and the lowest in 1996. The main freshwater source feeding the Gulf of Finland is the Neva river, which runs into the easternmost part of the Gulf. Total annual riverine runoff into the Gulf is 114 km^3 (ranging from 100 to 125 km^3), corresponding to about 25% of the total riverine runoff to the Baltic Marine Area.

Hydrography

The general circulation pattern in the Gulf of Finland can be described as follows: The saltier water from the northern Baltic Proper intrudes into the Gulf and flows eastwards along the Estonian coast, while the less saline (riverine) water flows westwards and out from the Gulf along the Finnish coast.

The series of winters with modest ice coverage that started in 1988 continued during the present assessment period. The exception was winter 1995/96, which was characterized by early formation of ice in November-December 1995 and the most extensive ice cover over the past ten years. The surface layer temperature maximum is usually observed in August. During the assessment period the surface layer (0–10 m) was warmest in 1994 (19.2°C) and in 1997 (18.4°C, but >20°C at 1 m depth), and lowest temperature maximum was observed in 1998 (16.5°C).

The increase in deep-layer salinity and associated strengthening of stratification in 1994–98 is related to the influxes of saline water from the deep layers of the Baltic Proper into the near-bottom layer of the Gulf of Finland. The long-term tendency towards a continuous decrease in deep-layer salinity reversed in 1993 (Figure 5.55). Stratification in the Gulf is usually strongest in July-early September. The total density difference between the near-bottom layer and the surface layer peaked in July 1997 at 4.67 kg/m³. Comparison of the mean stratification for the whole warm period in different years revealed that stratification was strongest in 1996 (estimated average density difference: 3.14 kg/m³). Among other things, the latter was related to incomplete vertical mixing of the water column in the Gulf of Finland in autumn-winter 1995/96.

Oxygen content

The mean oxygen concentration in the near-bottom layer of the central Gulf during the assessment period was less than the mean for the two preceding periods and close to that in 1979–83 (Figure 5.55). The deep-layer oxygen content depends on the near-bottom oxygen consumption and the stratification. The latter explains why the long-term trends in the deep-layer oxygen content oppose the deep-layer salinity trends.

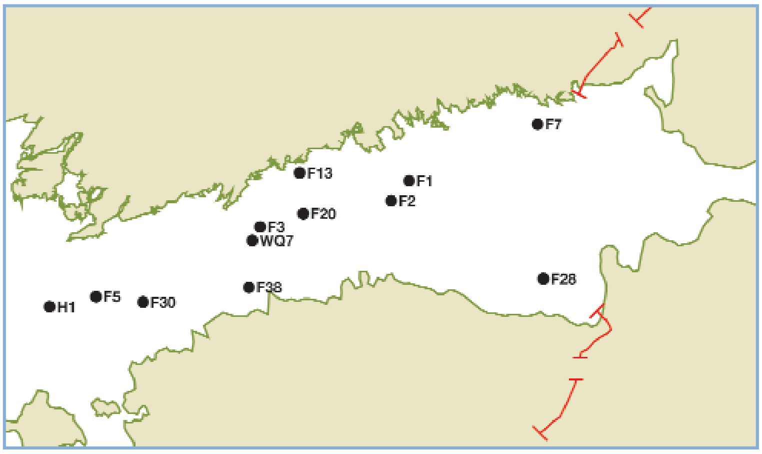


Figure 5.54
Map of the Gulf of Finland indicating the location of the monitoring stations referred to in the text.

In the semi-enclosed basins of the coastal areas, oxygen conditions during the assessment period were also at least as poor as in the early 1980s. In addition, extensive anoxia occurred at the sediment-water interface in the eastern Gulf of Finland in summer 1996 (Pitkänen and Välipakka, 1997). An ordinary seasonal minimum in oxygen concentration (observed in late summer due to the high sedimentation rates throughout the vegetation period and to restricted vertical exchange due to the developed seasonal thermocline) was strengthened in 1996 by an incomplete vertical mixing of the water column in the preceding autumn-winter.

Obvious reasons for the poor oxygen conditions

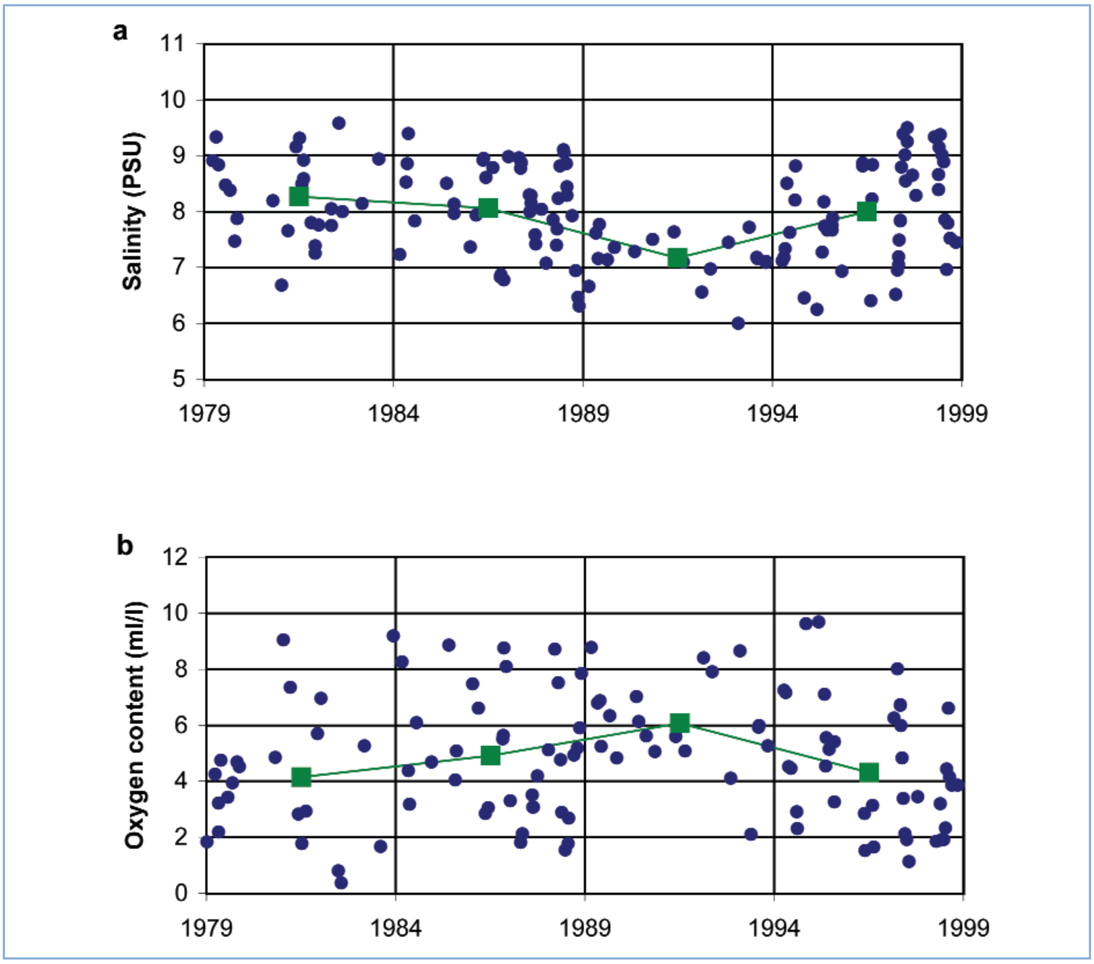


Figure 5.55
Long-term changes of salinity (a) and oxygen content (b) in the near-bottom layer at station F3 (LL7) in the central part of the Gulf of Finland.

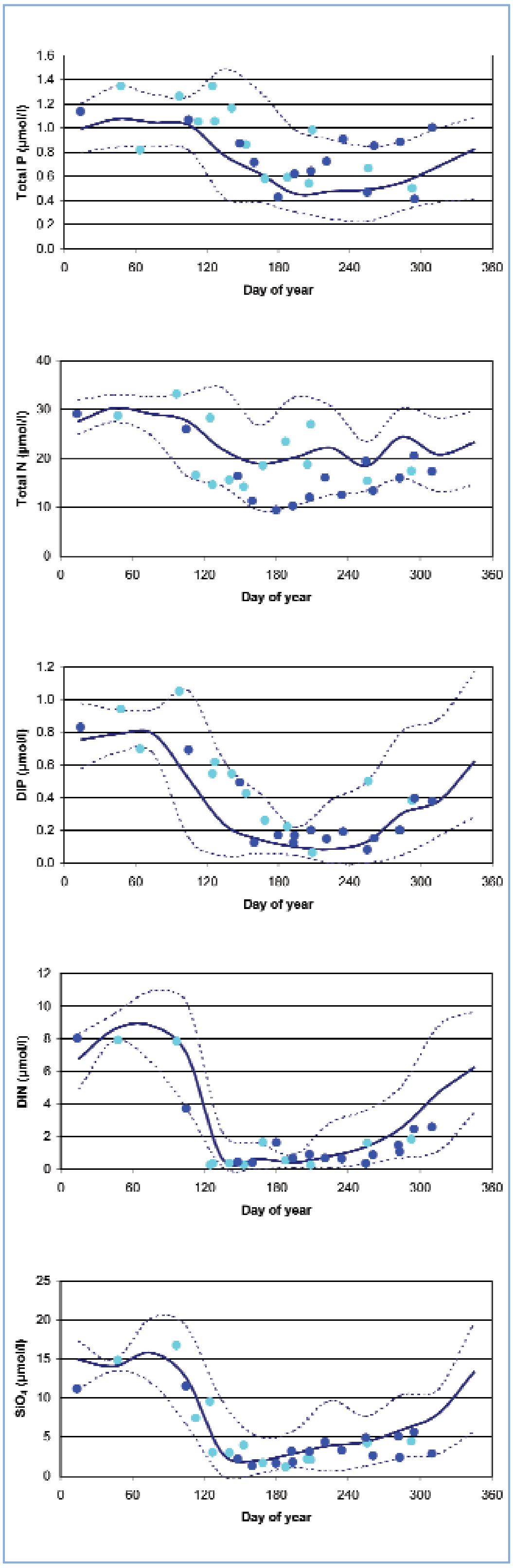


Figure 5.56
Seasonal variation in the concentration of total phosphorus (total P), total nitrogen (total N), dissolved inorganic phosphorus (DIP), dissolved inorganic nitrogen (DIN) and silicate silica (SiO4) in the surface layer (0–10 m) at station BMP F3 in the central part of the Gulf of Finland. Twenty-year monthly average (solid line), monthly minimum (lower broken line) and maximum (upper broken line) values are shown together with values for 1997 (green dots) and 1998 (blue dots).

are the high levels of accumulated organic matter in the sediments and the increased vertical stability due to the saltwater inflows. The fact that the deep-layer oxygen conditions in 1994–98 were at the level of the early 1980s, while the salinity stratification was still weaker than 15 years ago, indicates a general increase in eutrophication effects despite the reported decrease in nutrient loading of the Gulf (Pitkänen *et al.*,2001).

5.3.2. Hydrochemistry

By: U. Lips, H. Pitkänen, E.-L. Poutanen, P. Kauppila and S. Basova

Seasonal variation in the surface layer

All nutrient fractions exhibited pronounced seasonal variation. Concentrations are maximal in winter and minimal in spring-summer (Figure 5.56). Both DIN and silicates quickly reach yearly minimum concentrations (close to the detection limit for DIN) as early as May, while phosphates reach the minimum level in July–August. The latter indicates that the spring bloom is mainly nitrogen-limited in the open Gulf of Finland. The excess DIP remaining after the spring bloom is thus available for the development of the summer phytoplankton community.

Total P and DIP concentrations in 1997 and 1998 mainly exceeded the corresponding long-term monthly mean values, whereas total N and DIN concentrations were usually less than the corresponding mean values. In addition, several monthly maximum values of total P and DIP, and several monthly minimum values of total N and DIN for the past 20 years were recorded in 1997–98. The excess phosphate remaining after the spring bloom in 1997–98 exceeded the 20-year mean level.

Geographical variation

Due to the fact that the Neva river is the predominant source of nitrogen, the spread of river water from the easternmost end of the Gulf along the northern coast following the general estuarine circulation in the Gulf is clearly seen on the nitrate nitrogen distribution map (Figure 5.57). This pattern is less distinct on the phosphate phosphorus distribution map, although the prevailing west-east and south-north gradients are nevertheless observable. Areas of local maxima and minima in the northern Gulf sometimes correlated with local sources or the morphological peculiarities in the way the river water spreads (evidence of similar areas in the southern Gulf is unavailable due to the lack of data from coastal stations).

Long-term trends in the surface layer

The prevailing tendencies towards a decrease in DIP concentrations and an increase in DIN concentrations in the Gulf of Finland from the late 1970s to the early 1990s were supplanted by the opposite tendencies in the mid-late 1990s, i.e. an increase in DIP concentrations and a decrease of DIN concentrations (Figure 5.58). Although the number of winter measurements is usually quite low (less than 10 data points for every five-year period), and the variability within every period is higher than the estimated changes in mean values, the great similarity of the trends detected at all stations strongly supports the above conclusion.

While the long-term changes in the coastal areas of the northern Gulf were similar to those in the open Gulf, the concentrations in the southern Gulf seem to have remained more or less constant since 1993. The latter could be related to a combination of different trends observed in recent years in the Gulf of Finland (increasing P and decreasing N) and in the northern Baltic Proper (decreasing P and constant N). According to the information available (pers. com. by Svetlana Basova) from the easternmost part of the Gulf, the winter DIP concentrations in the Neva Bay were clearly higher in 1994–99 (1.38 µmol/l; 33 analyses) than in 1990–93 (0.99 µmol/l; 41 analyses).

The DIN:DIP ratio was below 16 in the open and southern Gulf during the whole monitoring period 1979–98. At Länsi-Tonttu the DIN:DIP ratio exceeded 16 only during the period 1989–93 and at Huovari during the period 1984-93. Based on the overall mean for all six stations analysed, the DIN:DIP ratio increased from 10.6 in 1979–83 to 12.8 in 1984–88 and to 14.8 in 1989–93. This trend was broken during the present assessment period, during which time the mean DIN:DIP ratio in the Gulf of Finland was only 10.4 (at least for the central part of the Gulf from the southern to the northern coast). This suggests that the vernal bloom in the Gulf is now nitrogen-limited, even in the investigated coastal areas.

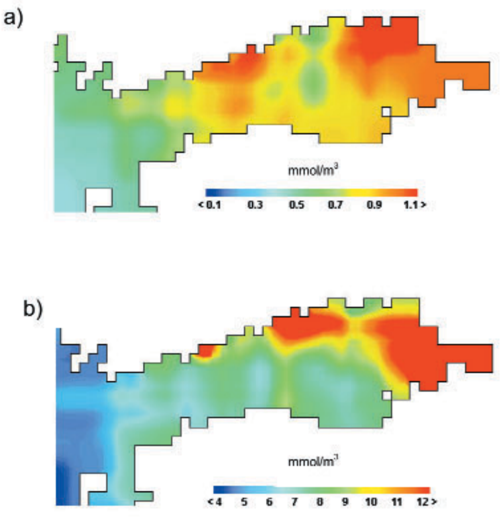


Figure 5.57
Horizontal distribution of phosphate phosphorus and nitrate nitrogen concentrations in the surface layer of the Gulf of Finland in winter 1998 (source: Finnish Institute of Marine Research, Uusimaa Regional Environment Centre, Southeast Finland Regional Environment Centre, City of Helsinki Environment Centre).

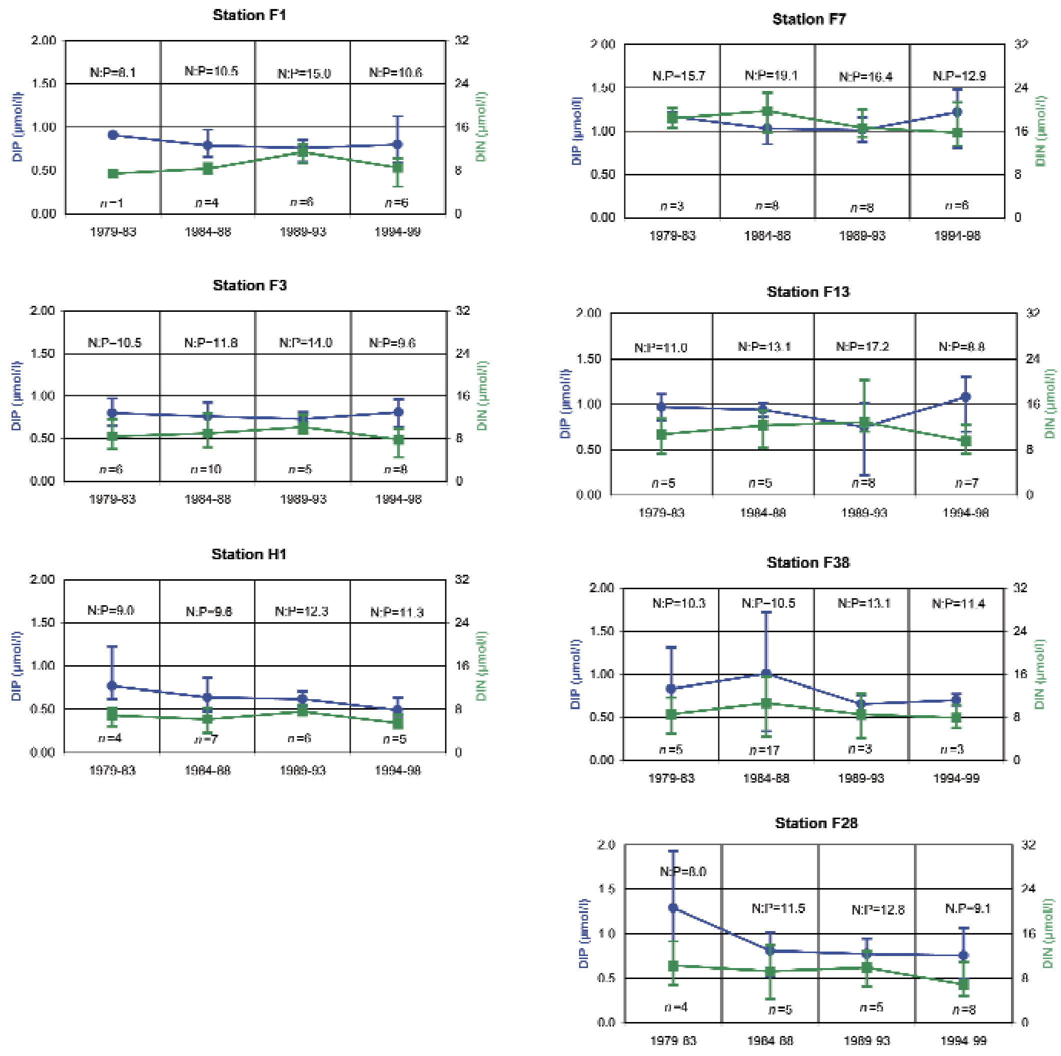
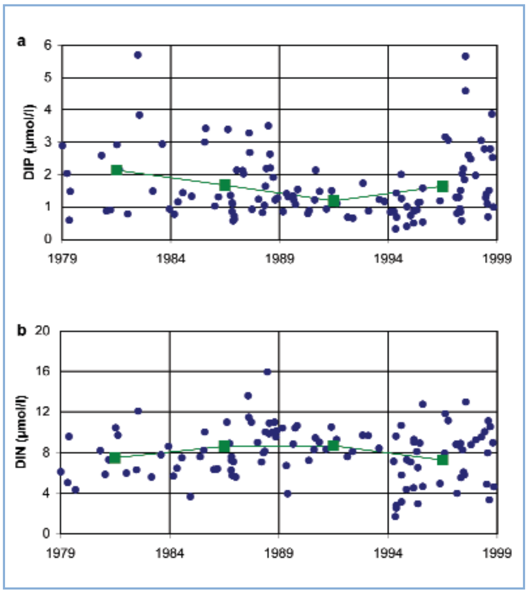


Figure 5.58
Long-term changes in winter concentration of DIP and DIN at stations F1, F3, H1, Huovari, Länsi-Tonttu, 57a and N12 in the Gulf of Finland. Five-year average concentrations are shown (DIP – blue; DIN – green). The maximum and minimum values are also indicated, as well as mean DIN:DIP ratios and number of data points (n).

Figure 5.59
Long-term changes in the DIP (a) and DIN (b) concentrations in the near-bottom layer at station BMP F3 in the central part of the Gulf of Finland.



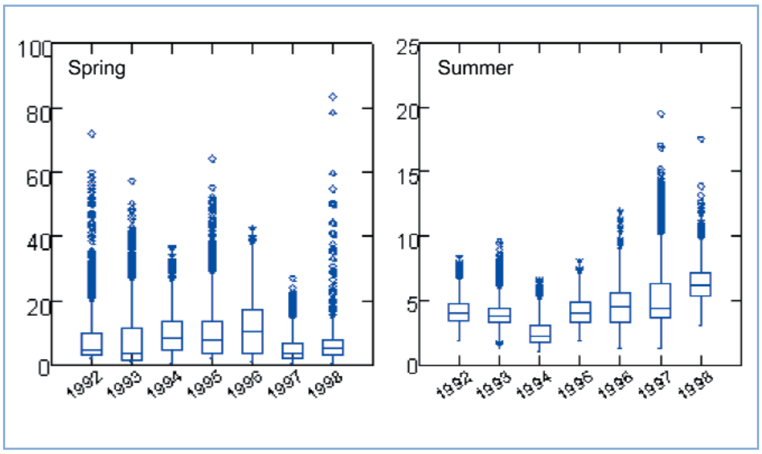


Figure 5.60
Interannual variation of chlorophyll a levels (in mg/m³) in the western Gulf of Finland in spring and summer on the basis of unattended measurements at ferries in 1992–98.

Long-term trends in the bottom layer

The DIN concentrations in the near-bottom layer of the Gulf increased during the 1980s while the DIP concentrations decreased. This development ceased during the 1990s (Figure 5.59). DIN concentrations started to slowly decrease, while DIP concentrations increased. In 1996–98, the summer phosphate phosphorus concentrations peaked at the same level as in the early 1980s.

The nutrient reserves in the near-bottom waters of the Gulf of Finland are much higher than in the other Baltic waters of similar depth. The obvious reason for the high deep-water accumulation of nutrients is intense production in the euphotic surface layer and consequent intense sedimentation of plankton detritus to the aphotic near-bottom layer (e.g. Heiskanen *et al.*, 2000). The DIP level probably increased in the Gulf of Finland in the late 1990s due to the benthic release of phosphorus caused by an extensive anoxia at the sediment-water interface in the eastern Gulf in summer 1996. Internal phosphorus loading is estimated to equal the annual external phosphorus loading to the Gulf (Pitkänen and Välipakka, 1997).

Nutrient inputs

The Gulf of Finland receives 7,600 tonnes of phosphorus and 139,000 tonnes of nitrogen annually according to estimates for the year 1995 (Pitkänen *et al.*, 1997), corresponding to a 25–30% decrease in nutrient load since the late 1980s/mid 1990s. However, due to the fact that the largest part of both phosphorus (46%) and nitrogen (53%) inputs originates from the mainly phosphorus-limited catchment area, the changes in the nutrient loads available for new production very much depend on the variation of riverine runoff. Taking into account that the riverine runoff was lower in the present assessment period than in 1981–92, the decrease of the DIN load into the Gulf should be more distinct than the decrease of the DIP load.

The opposing trend in inorganic phosphorus concentration seen in the Gulf of Finland in the 1990s (compared with the trend in loads) must thus be attributable to a marked increase in internal P loading in the mid 1990s or the export of nutrients via deep-water inflows from the Baltic Sea Proper. Due to the vertical mixing in autumn-winter, which reaches depths of 70 m and transports nutrients from the deeper layers to the upper layer, the phosphate concentration in the euphotic layer increased in the succeeding spring-summer. As a result, the nutrient limitation shifted towards N control in the 1990s, as clearly evidenced by the excess phosphate phosphorus in the surface layer of the open Gulf (Figure 5.56) in early summer 1997 and 1998. In the warm summer of 1997, this probably caused the strongest bloom of the toxic blue-green algae *Nodularia spumigena* seen in the whole of the central and western Gulf of Finland.

5.3.3. Structure and function of the pelagic ecosystem

Phytoplankton

By: A. Jaanus, J. Leppänen, S. Hällfors, L. Lepistö and P. Kauppila

The present assessment of the state of phytoplankton in the Gulf of Finland is based on data collected within the framework of both traditional and unattended monitoring by the Estonian Marine Institute and the Finnish Institute of Marine Research in the central and western parts of the Gulf, as well as on routine monitoring conducted by the Finnish Environment Institute along the Finnish coast. While automatically collected data have been available for the western Gulf since 1992, such measurements were not introduced until 1997 in the central Gulf of Finland between Tallinn and Helsinki. Some of the conclusions below regarding the easternmost part of the Gulf are based on literature data.

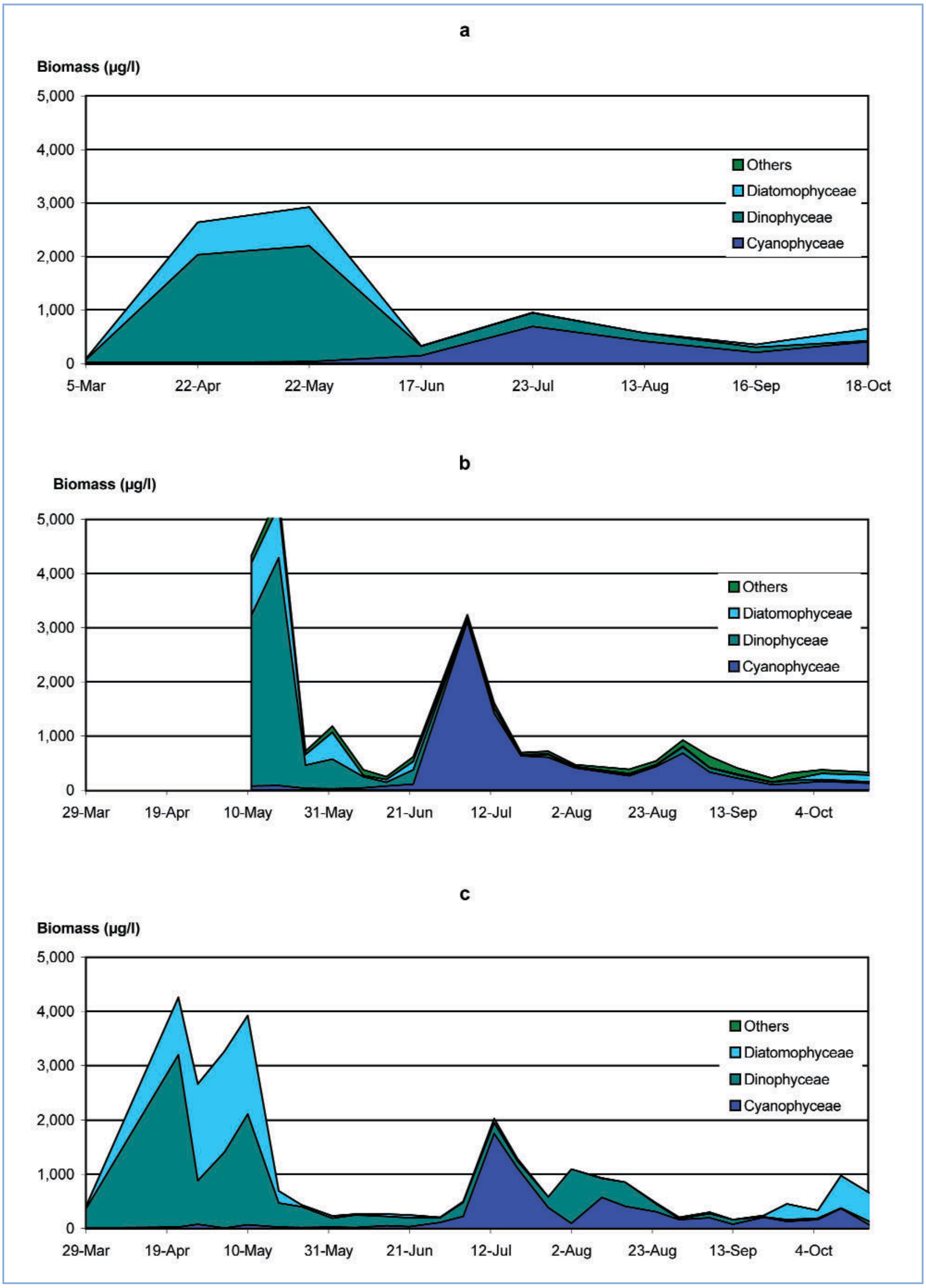


Figure 5.61 Phytoplankton dynamics in the central Gulf of Finland: (a) monthly mean biomass values (1994–98) at station BMP F3; measurements at automated sampling station WQ7 (situated close to station F3) in 1997 (b) and in 1998 (c).

Chlorophyll a. Compared with the other years, the seasonal pattern in 1997 differed for both the spring and summer periods according to the chlorophyll data collected using automated recordings on the ferries Finnjet and Finnpartner (Figure 5.60). Due probably to a cold and windy May, spring phytoplankton biomass remained low in most parts of the Gulf. The increase in cyanophytes observed in early July together with the high chlorophyll a levels indicate the development of an intensive bloom. The high summer chlorophyll a levels in 1996 and 1998 were probably caused by mass occurrence of *Heterocapsa triquetra*.

According to the coastal monitoring data, the eutrophic area (corresponds to 5–10 mg/m^3 chlorophyll a in the summer period) extended as far as 5 to 15 km from the coast. The border of the slightly eutrophic area moved westwards during the 1980s and 1990s, despite the reduction in anthropogenic nutrient loads. Recent data from the Neva Bay also revealed increasing eutrophication, as evidenced by a pronounced increase in primary production and the prevalence of the production process over the decomposition of organic matter (Telesh *et al.*, 1999).

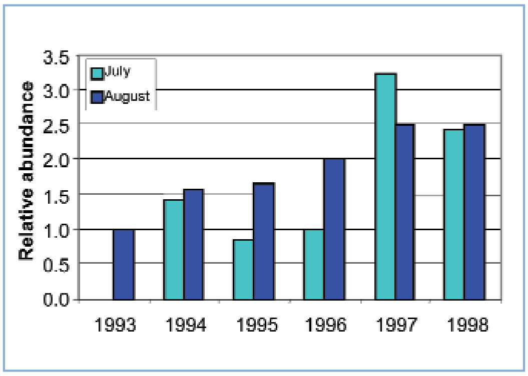


Figure 5.62 Interannual variation of relative abundance of cyanobacterium Nodularia spumigena in the western Gulf of Finland on the basis of ship-of-opportunity data collected in July and August 1993–98.

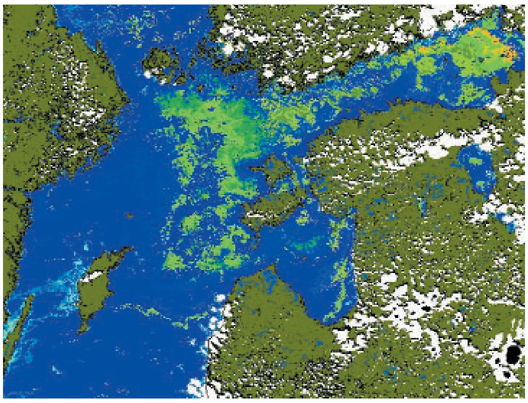


Figure 5.63 Cyanobacterial surface accumulations in the Gulf of Finland, the northern Baltic Proper and the Bothnian Sea in summer 1997. Original NOAA/AVHRR image processed by the Finnish Meteorological Institute, interpretation by the Finnish Institute of Marine Research.

Phytoplankton biomass and species composition. Mean monthly biomass of different algal groups at station BMP F3 is shown for the present assessment period in Figure 5.61a. The dynamics was rather low, with dinoflagellates being dominant in the spring and blue-green algae in the summer. In comparison, high-frequency automated sampling enables detection of almost all exceptional events in the marine environment, as well as identification of the phytoplankton bloom peaks. This is illustrated by the annual dynamics for 1997 and 1998 at a sampling station in the same area (Figures 5.61b and 5.61c).

As regards the dominant species, no significant changes have taken place compared to the preceding assessment periods. The spring bloom of the dinoflagellate *Scrippsiella hangoei* was intense, accounting for up to 95% of the total biomass during its peak in the central and western parts of the Gulf. Besides *S. hangoei*, the spring phytoplankton species occurring in the greatest biomass were *Achnanthes taeniata, Peridiniella catenata, Chaetoceros wighamii, Skeletonema costatum* and *Thalassiosira baltica*. In summer phytoplankton communities, *Aphanizomenon sp.* was the most stable species, dominating in 50–75% of analysed samples over the whole of the Gulf of Finland from June to September.

According to long-term Finnish coastal monitoring, phytoplankton biomasses have been increasing since the 1990s in both the eastern and western Gulf. In the eastern Gulf of Finland, cryptophytes and dinoflagellates (mainly *Dinophysis acuminata*) were succeeded by blue-green algae, *Aphanizomenon sp.* and *Planktothrix agardhii* in late summer (see also Telesh *et al.*, 1999). The dominance of *P. agardhii* coincided with the large outflow of nitrogen-rich and oligohaline water from the river Neva.

Nuisance algal blooms. Based on semi-quantitative analysis, the relative abundance of *Nodularia spumigena* has increased in the western Gulf of Finland during the 1990s (Figure 5.62). In summer 1997, the accumulations of blue-green algae in the whole of the Gulf were the most extensive and prolonged ever recorded (Figure 5.63), possibly due to the calm and warm weather conditions and the elevated DIP level (released from sediment during extensive oxygen deficit in summer 1996 and transported into the productive surface layer during the following autumn-winter). Very strong vertical stratification of the upper layer in summer 1997 further favoured the formation of surface accumulations.

Intensive blue-green algal blooms were also recorded in 1994 and 1998, although large surface accumulations did not form in 1998 due to wind-induced mixing. In addition to mass abundance of *Heterocapsa triquetra* in July–August 1996 and 1998, the first numerous appearance of the dinoflagellate *Prorocentrum minimum* in the western and central Gulf of Finland was recorded in September 1997.

Zooplankton

By: J. Flinkman, H. Kuosa, A. Põllumäe and T. Shchuka

Size structure of zooplankton communities in the Gulf of Finland has fluctuated during the assessment period. Most marked is the change in the proportion of larger zooplankton (over 20 µg wet weight), which includes adult copepods and larger cladocerans (Figure 5.64). When total zooplankton biomass is high, the proportion of large zooplankton is low. This coincides with periods of decreased salinity, the most important factor regulating zooplankton species composition and abundance (e.g. Segerstråle, 1969; Hernroth and Ackefors, 1979; Viitasalo *et al.*, 1995).

Larger, neritic copepods are the favoured prey of Baltic herring. Changes in zooplankton communities affect food availability for herring populations, and a decrease in the proportion of favoured prey may negatively affect their growth. This has been shown in the Gulf of Finland and northeastern Baltic Proper by Raid and Lankov (1995) and Flinkman *et al.* (1998).

According to the Estonian monitoring data, the abundance of marine calanoid species, especially *Pseudocalanus minutus*, was low in 1995 and 1996, but rose in 1997 and 1998. The glacial relict *Lim-*

nocalanus macrurus occurred in very low numbers in recent years and the abundance of *Bosmina coregoni*, a cladoceran of freshwater origin, had clearly decreased during the present assessment period. According to the data collected in June 1998, copepods dominate the zooplankton biomass in the eastern part of the Gulf of Finland, while rotifers constitute up to 90% of the abundance.

The cladoceran species *Cercopagis pengoi* was first encountered in the Gulf of Finland in 1992. The spatio-temporal dynamics of this large predatory species of ponto-caspian origin is strongly dependent upon climatic conditions. In cold summers, distribution area and population abundance of the cladoceran are remarkably smaller than in warm summers. According to the monitoring data, *C. pengoi* was found in several Estonian coastal and offshore stations in 1995, 1997 and 1998. The species was abundant in Estonian waters and in the whole of the open sea area of the Gulf of Finland in 1997. Reports have been received of fish net clogging in Finnish coastal waters, mainly in 1997.

Functioning of the pelagic food web

By: P. Kuuppo, A. Heiskanen, R. Lignell, T. Tamminen and P. Tuomi

The following section is based on experimental and monitoring studies performed on the southwest coast of Finland in spring (1991) and summer (July–August) 1994, 1996 and 1998 by the PELAG team (projects PELAG III, SILMU and COMWEB).

Nutrients imported to the pelagic system from external sources (i.e. new nutrients from the deep water, or allochthonous and atmospheric sources) support new production, which is either incorporated in the planktonic food web, or channelled into increased sedimentation (Figure 5.65). Major new production in the northern Baltic Marine Area takes place during the spring bloom in April–May and the cyanobacterial bloom in July–August. During the blooms, biomass is dominated by a few "key" species that govern the overall synthesis and channelling of organic material.

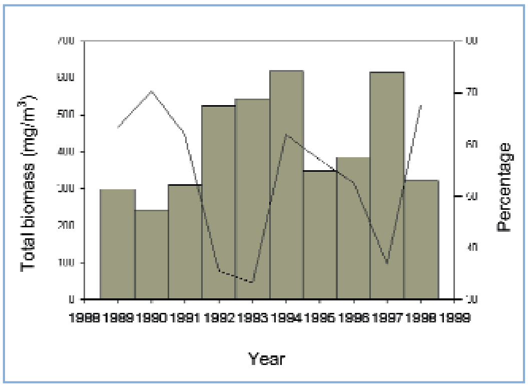


Figure 5.64
Long-term changes in total biomass of zooplankton and in the percentage of larger size zooplankton (over 20 µg wet weight, which includes adult copepods and larger cladocerans).

Spring. In April, before the onset of the spring bloom, phytoplankton primary production and bacterial production amounted to approx. 80 µg C/(m² • d).

During the spring bloom peak, dinoflagellates dominate in the water column, but the magnitude of spring bloom sedimentation is mainly governed by the diatoms, which are the major vehicle of the export flux from the pelagic system (Heiskanen, 1998; Tallberg and Heiskanen, 1998). The major part of the vernal dinoflagellate biomass produces slowly settling phytodetrital material after the bloom has terminated, and potentially retains more organic matter and nutrients in the planktonic food web than diatoms (Heiskanen, 1998). It is generally believed that due to the late development of mesozooplankton, a large part of the vernal bloom in the northern Baltic Marine Area is lost through sedimentation (Lignell *et al.*, 1993).

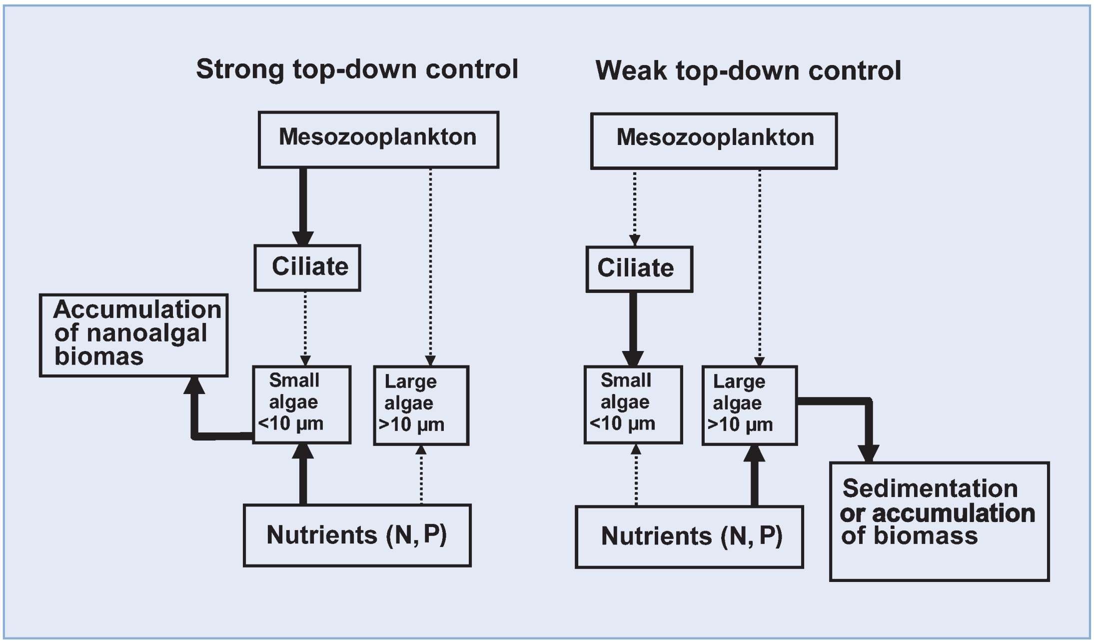


Figure 5.65
Potential impact of the top-down control by planktonic copepods (mesozooplankton) on the channelling of algal biomass in the pelagic system of the Gulf of Finland. Strong top-down control restrains the number of ciliates and leads to enhanced nutrient assimilation and biomass accumulation of small algae (<10 µm). Weak top-down control leads to increased ciliate numbers which suppress the biomass of small algae. Large phytoplankton (>10 µm) consequently attain a greater share of nutrients leading to increased sedimentation or accumulation depending on the species (Figure modified from Heiskanen, 1998).

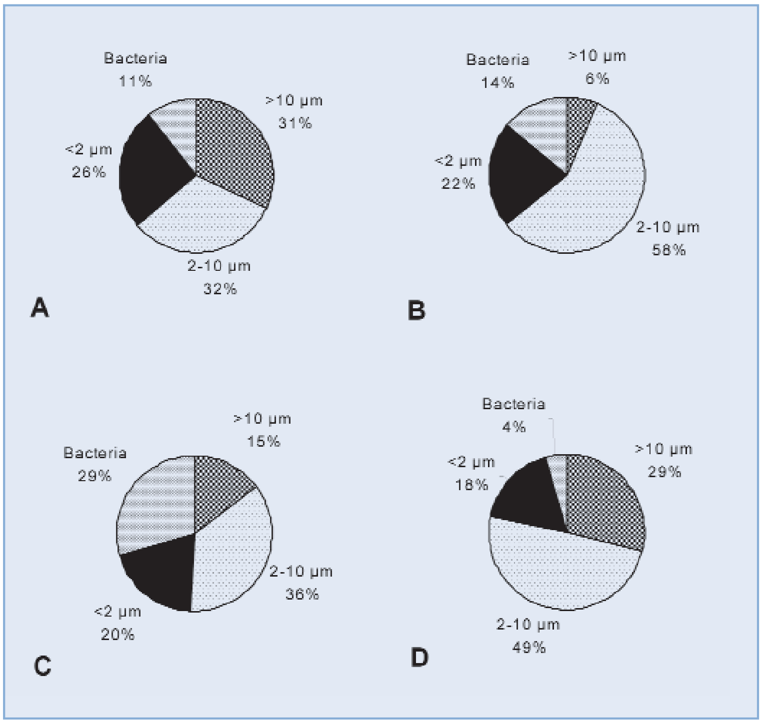


Figure 5.66
Fractionated primary productivity (^{14}C method) and bacterial secondary productivity (^{3}H-thymidine method) in early July 1994 (A), early August 1994 (B), and in corresponding periods in 1996 (C) and 1998 (D), off the SW coast of Finland expressed in percent of summed daily total primary productivity and bacterial productivity (PP+BP) in the mixed/euphotic upper 10-m water layer.

Summer. As the seasonal cycle proceeds from the vernal new production period to the regenerated production period in summer, most of the carbon produced by autotrophs and bacteria is channelled onwards in the microbial food web, and consequently respired by heterotrophs belonging to several trophic levels and size fractions (Kuosa, 1991; Kuuppo *et al.*, 1994).

In summer 1994 and 1996, the sum of phytoplankton and bacterial productivity (PP+BP) was typical for the season, ranging from 490 to 725 mg C/(m² • d). Bacterial productivity accounted for 11–29% of PP+BP, while picoalgal (<2 µm) primary productivity accounted for 18–26% (Figure 5.66). During most of the summer, the percentage of PP-BP accounted for by nanoalgae (2–20 µm), which dominated the phytoplankton biomass in the western Gulf of Finland during most of the summer, varied considerably, thus reflecting the different stages of summer phytoplankton succession.

Frequent upwellings in the coastal areas of the Gulf of Finland may cause drastic changes in local planktonic food web structure and function, both via flushing/advective transport of the local plankton community and input of nutrient-rich deep water. For example, the percentage of PP+BP accounted for by bacterial productivity, which was only 4% after an upwelling in 1998 (Figure 5.66), increased considerably during the subsequent three-week study period.

During the summer's regenerated production, nitrogen is the primary limiting nutrient for both total primary production and phytoplankton biomass (chlorophyll a) (Kivi *et al.*, 1993). Tight coupling exists between pico-size planktonic producers, algae and bacteria, and their nano-size predators, the heterotrophic nanoflagellates, which rapidly catch up with the increment in their prey (Kuuppo, 1994). The quantitative importance of viruses in the carbon cycle is as yet unclear.

Blooms of filamentous, diazotrophic cyanobacteria are recurrent phenomena during the late summer in the northern Baltic Marine Area (Kononen, 1992). The cyanobacterial biomass mostly disintegrates in the water column, fuelling mainly the microbial food web, and thus indirectly contributing to organic matter loading and subsequent oxygen depletion in the sediments (i.e. Heiskanen, 1998).

Top-down control and bottom-up limitation. Food web manipulation experiments conducted on the southwest coast of Finland during autumn have shown that crustacean zooplankton (mainly calanoid copepods and cladocerans) exert strong grazing/predation control on microprotozoa (Kivi *et al.*, 1996), which in turn has a cascading effect throughout the food web (Kivi *et al.*, 1996).

The effect of top-down control by mesozooplankton and the bottom-up limitation of nutrients on the development of the phytoplankton biomass fraction is illustrated schematically in Figure 5.65. The high number of crustacean zooplankton enhances retention of algal cells in the water column due to a shift towards dominance of a smaller algal size fraction, while large numbers of protozoa exerting higher grazing pressure on smaller phytoplankton would enhance potential sinking losses from a larger algal size fraction (Figure 5.66). The large size of phytoplankton may also promote accumulation of biomass, as during the cyanobacterial blooms in the Baltic Marine Area.

The mesocosm and monitoring studies carried out in the coastal areas of the Gulf of Finland during the 1990s have shown that the common properties of the new production systems – where nutrients are mostly supplied from external sources – seem to shift the planktonic system predominantly into nitrogen deficiency. Nitrogen limitation is maintained by concurrent sedimentation and effective sequestration of nitrogen into the body tissue of pelagic heterotrophs, while phosphorus is retained and recirculated in the pelagic system (Heiskanen *et al.*, 1996).

5.3.4. Benthic conditions

Sediments

By: J. Lehtoranta and K. Mäkelä

Sediments in the eastern Gulf of Finland are generally organic matter-rich. The mean total nitrogen concentration in the upper 2 cm of the sediment was 7.4 mg/g dry weight, similar to that obtained for the Baltic Proper by Koop *et al.* (1990). The mean surface total phosphorus concentration was 2.6 mg/g dry weight, almost twice that measured in the Baltic Proper (1.3 to 1.6 mg/g dry weight; Carman and Wulff, 1989). In the 10–15 cm layer, however, the mean total phosphorus concentration was similar to that in the Baltic Proper (1.0 to 1.2 mg/g dry weight).

Oxygen measurements with microelectrodes have revealed that the oxygen is depleted within the top few millimetres (2–4 mm) of the sediment-water interface in the Gulf of Finland (Conley *et al.*, 1996). Calculated sediment O_2 consumption varies from approx. 180 to 200 mg $O_2/(m^2 \bullet d)$. Chemical O_2 oxidation of upwards diffusing, reduced, dissolved Mn and Fe (and possibly nitrification) occurs in this thin surface layer.

Ammonium accumulates in high concentrations in the pore water, thus indicating that anaerobic mineralization of organic matter continues in the 10–15 cm layer of the sediment. High pore water ammonium and phosphate concentrations indicate high diffusive nutrient flux from the sediment towards the surface layer of the sediment in organic matter-rich areas.

There is evidently a large pool of mobile P (Fe and organic matter-bound P) in the 0–6 cm layer of the sediment in the Neva estuary and in the open Gulf of Finland. In the organic matter-rich coastal accumulation bottoms (the Finnish archipelago, the open area east of the archipelago and part of the Luga Bay), the pool of mobile P is concentrated in the surface layer (0–2 cm). The benthic fluxes of nutrients of the inner and outer parts of the Neva estuary are smaller than those in organic matter-rich accumulation bottoms.

Of the nutrients investigated, nitrate and nitrite exhibited characteristic distribution at the surface layer of the sediment. In 1996, when sampling was concentrated on stations GF2 and JML in the Gulf of Finland, a distinct seasonal pattern was observed (Figure 5.67). The estimate for the whole year will thus depend markedly on the extent to which the vernal or autumn bloom is encompassed by sampling.

Phosphate profiles exhibit distinct seasonal variation with higher concentrations in late summer-autumn. Silicate concentrations tended to increase towards autumn. In the central Gulf of Finland, silicate concentrations varied between 150 and 400 µmol/l. In the western part of the Gulf, the concentrations were 250–600 µmol/l.

Macrozoobenthos

By: J. Kotta, I. Kotta, P. Kangas, A. Laine and A. Andersin

The development of macrobenthic communities in deep, open sea areas. At the beginning of the 1990s, marked recolonization of previously desert-

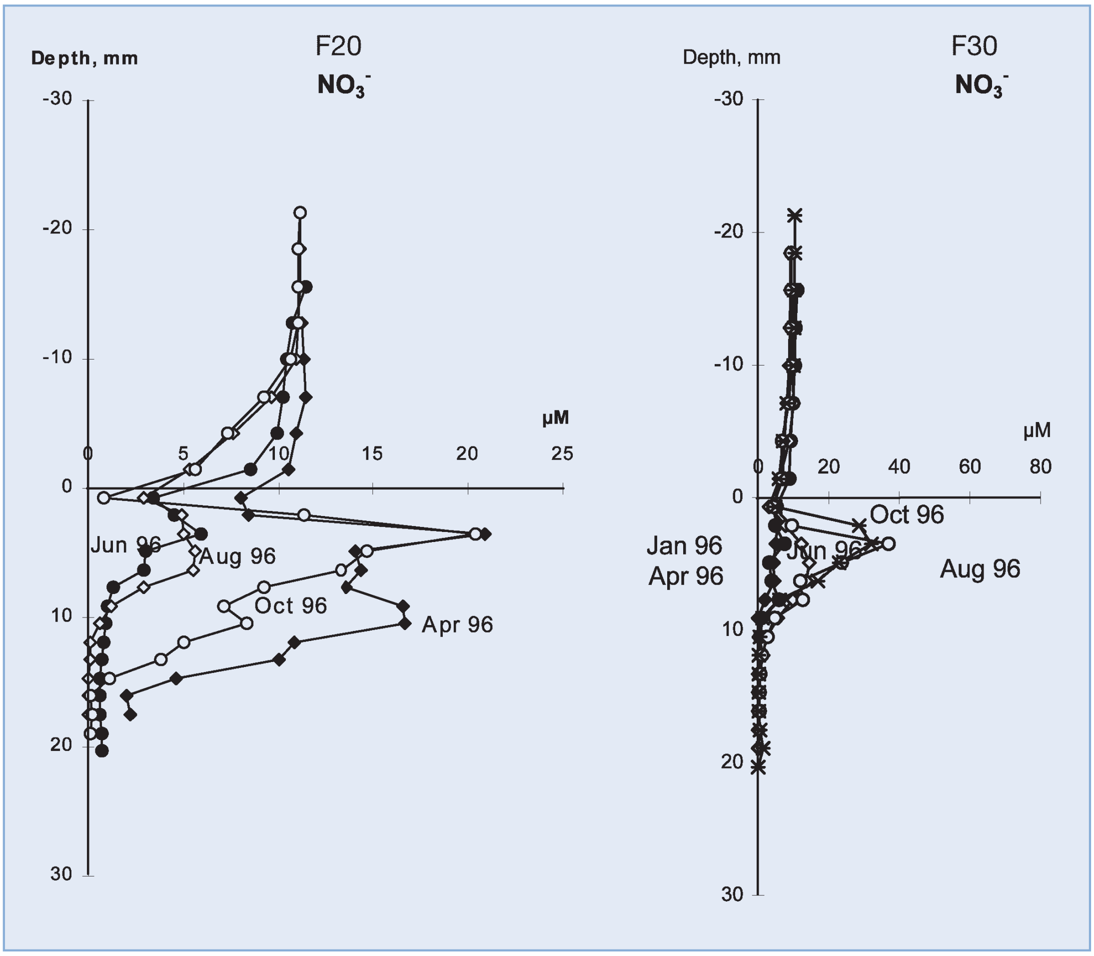


Figure 5.67
Seasonal variation in the pore water concentration of nitrate at stations F20 and F30 in the Gulf of Finland in 1996.

Figure 5.68
Changes in macrozoobenthos abundance (a) and biomass (b; wet weight) at station F5 in the western Gulf of Finland (depth 62 m) over the period 1966–98.

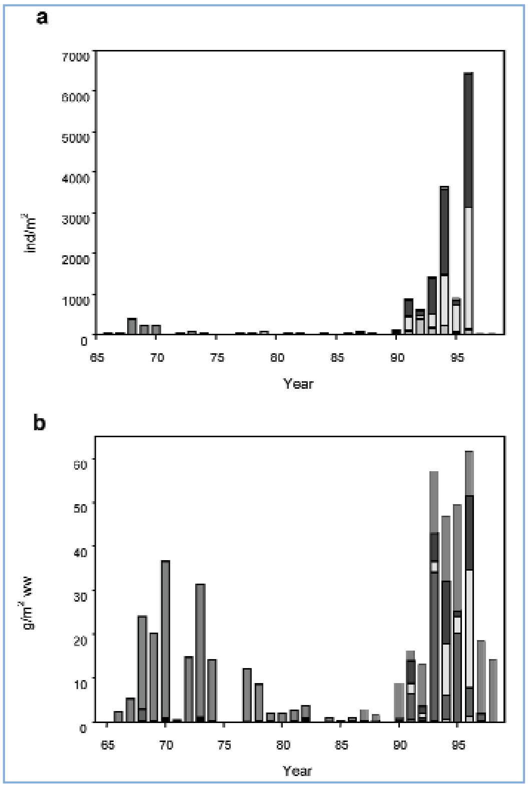
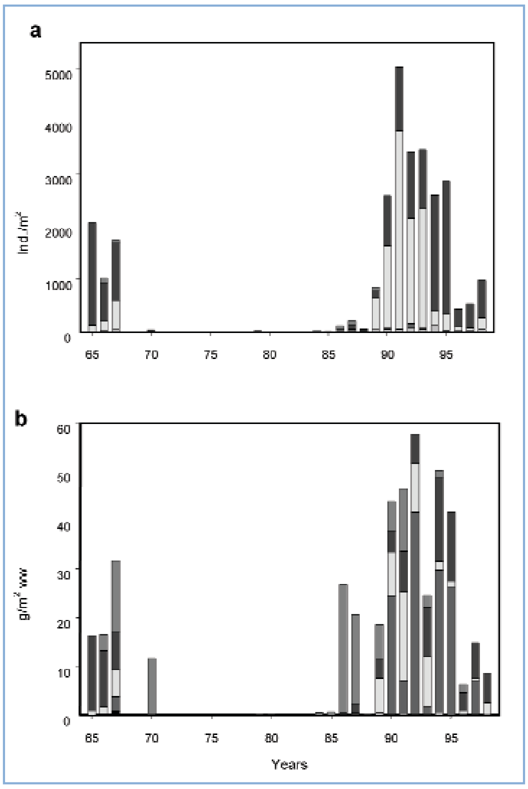


Figure 5.69
Changes in macrozoobenthos abundance (a) and biomass (b; wet weight) at station F2 in the central Gulf of Finland (depth 62 m) over the period 1965–98.



ed communities started in the Gulf of Finland. At the beginning of the present assessment period (1994–98), abundance and biomass were still high at many of the sampling sites. This was followed by a drastic collapse of the communities in 1996–97, however. By the end of the period communities were very sparse at most of the sites, and bottoms devoid of macrofauna were observed again in the deep western Gulf of Finland.

A common feature of the succession is the collapse of the previously abundant macrobenthic communities in the western and central parts of the open Gulf of Finland in 1996–97 (Figure 5.68 represents the western Gulf and Figure 5.69 the central part of the Gulf). Changes in community structure in the western part were characterized by the replacement of *Pontoporeia femorata* and *Monoporeia affinis* dominated communities by sparse, mainly *Macoma balthica* dominated assemblages. The abundance of the polychaete *Harmothoe sarsi* decreased, and the priapulid *Halicryptus spinulosus* has only occasionally been recorded in recent years. In terms of abundance, the communities in the central parts of the Gulf were typically dominated by *P. femorata* during the period 1994–98. As regards biomass, the patterns in biomass were more labile, *P. femorata* and *Saduria entomon* often being dominant. *M. affinis* and *M. balthica* accounted for only a minor part of the communities. Impoverished communities were predominantly observed in the deep parts (85–110 m) of the western half of the Gulf of Finland. Many of these sites were completely devoid of macrofauna in 1996–98.

In the eastern Gulf of Finland, no clear trend was detected. At the three sites sampled between Gogland and Neva Bay, both increases and decreases were recorded during the study period. In contrast to the western and central areas, the communities were dominated by the amphipod *M. affinis* (in terms of abundance) and the isopod *S. entomon* (in terms of biomass).

The recolonization of the bottoms since the beginning of the 1990s has been connected to changes in hydrography caused by the prolonged stagnation period (Laine *et al.*, 1997). Weak stratification and lowering of the halocline led to improved oxygen conditions in the deep Gulf of Finland, which allowed the development of abundant and species-rich communities. However, low oxygen concentrations (≤ 2 ml/l) have recurred in the central and western Gulf of Finland at 65–75 m in summer-autumn since 1996. This is evidently attributable to strengthening of the halocline caused by deep-water inflows from the open Baltic after major inflows of North Sea water in 1993–94. Hypoxia obviously caused the collapse of the macrozoobenthos communities in the open Gulf of Finland.

The development of the macrobenthic communities in coastal waters. Along the Estonian open coast, macrozoobenthos data were obtained from 75 stations at depths of 3–20 m depth. Most of the areas are not influenced by direct loading from land. In the easternmost Gulf of Finland, data from six stations in the shallow Neva Bay off St Petersburg were used. The Finnish data, except those from Tvärminne, derive from the pollution control studies (cf. Kangas *et al.*, 2000). The selected stations are the outermost ones of the control areas located at depths of 16–30 m, and hence less influenced by the direct wastewater loading.

Macrozoobenthos abundance and biomass have significantly decreased at central and easternmost localities in Finnish coastal waters since the 1980s (Figure 5.70), except off Kotka, where the sampling sites are located close to the pollution sources, from where loading has decreased markedly in the 1990s. Abundance has remained almost the same, and biomass has even increased in the westernmost part of the Gulf of Finland. The decline in macrozoobenthos abundance and biomass is probably related to oxygen deficiency at these bottoms in the late 1990s. On the other hand, the introduced polychaete *Marenzelleria viridis* has considerably expanded its area of distribution and is now found in most coastal areas.

Macrozoobenthos abundance and biomass have significantly increased in the central and western parts of the Estonian coastal waters during the assessment period, but have remained practically unchanged in the eastern part. In the 1980s, abundance and biomass reached minimum levels in the central part of the Gulf and peaked in the western and eastern parts. In the 1990s, abundance and biomass values gradually decreased from west to east. Owing to decreasing salinity, the number of macrozoobenthic taxa diminished towards the eastern side of the Gulf. The average number of macrozoobenthic taxa per station was 5.5 in the westernmost areas and 3.0 in the easternmost areas. No temporal changes in the benthic diversity was observed in the different sub-regions of the Gulf. However, the percentage of macrozoobenthos total abundance and biomass accounted for by *Mytilus edulis* has decreased, while that accounted for by *M. balthica* has increased.

Macrozoobenthos abundance has remained at a similar level, whereas biomass has decreased significantly in the Neva Bay, the easternmost part of the Gulf of Finland over the period 1980–98. These communities were characterized by relatively high abundance, very low biomass and low species diversity. The communities were dominated by *M. affinis, S. entomon,* oligohaetes and chironomids. As was the case in Finnish coastal waters, the percentage of abundance accounted for by *M. affinis* decreased considerably during the 1990s compared to the 1980s.

Phytobenthic communities in the littoral zone

By: S. Bäck, G. Martin, M. Orlova and L. Anokhina

Introduction. Coastal ecosystem diversity is richest in the littoral zone. The Finnish coastline is characterized by mosaics of islands and skerries, while the Estonian northern coastline is much more open and lacking in major archipelagos. The phytobenthic zone offers very different kinds of habitats supporting different types of vegetation and associated fauna. The main threats to phytobenthos are considered to be increasing eutrophication and physical destruction of habitats.

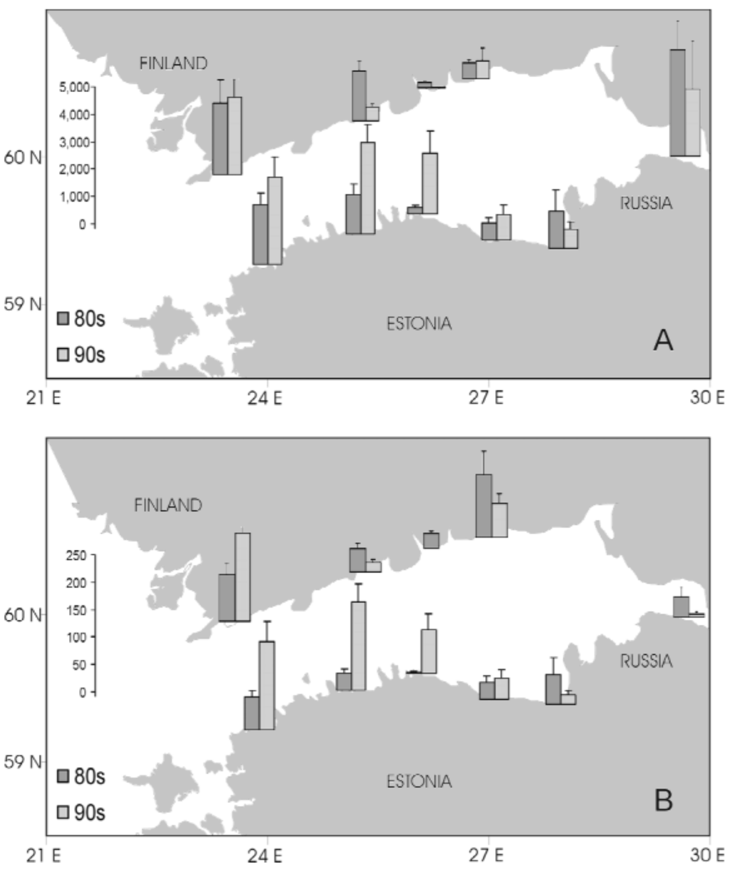


Figure 5.70
Changes in macrozoobenthos abundance (A) and biomass (B) in different areas of the Gulf of Finland from the mid 1980s to the mid 1990s.

Rocky bottoms are often dominated by macroalgae. A typical zonation of macroalgae is usually found on exposed rocky shores. Filamentous algae dominate close to the water level, brown algae *(Fucus vesiculosus)* in the shallow sublittoral zone, and red algae at the greatest depths. Sandy and muddy bottoms are characterized by phanerogams. The bottom vegetation in shallow bays may also be dominated by charophyte meadows.

Macroalgal species composition. In the outer archipelago of the eastern Gulf of Finland, the upper filamentous algal zone is dominated by *Cladophora glomerata,* the intermediate zone by *Fucus vesiculosus* (1–5 m depth) and the deep water filamentous algal zone by *Cladophora rupestris* (Bäck *et al.*, 1993; Valovirta and Porkka, 1996). Even at the semi-exposed sites, however, this filamentous algal zone became sparse or disappeared completely (Valovirta and Porkka, 1996). In semi-exposed localities the *Hildenbrandia* zone can start immediately below the *Fucus* zone at a depth of about 4 metres. A thick cover of the polyp

Cordylophora caspia is suggested to indicate eutrophication.

Along the southern coast of Finland, Kukk and Viitasalo (1997) have reported that the previously extensive *Enteromorpha spp*. belt in the inner archipelago off Helsinki receded and gave way to a *Cladophora glomerata* belt during the period 1978–1995. Phanerogams and filamentous brown algae appeared. The changes reflect the improvement in the treatment of wastewater from Helsinki. The recovery of the phytobenthos is also due to relocation of the outfalls from the adjacent urban areas away from the innermost bays to the open fringe of the archipelago waters.

General deterioration of the *F. vesiculosus* belt is of great concern because it is a key species in the Baltic Sea ecosystem. In the early 1990s (Bäck *et al.*, 1993), it had partly recolonized its former habitats on the Finnish coast, while the decline had continued in some parts of the archipelago. The recovery of *F. vesiculosus* has been restricted to the upper part of the littoral zone along the southern coast of Finland.

On the other side of the Gulf in the coastal waters off Tallinn, large-scale recovery of *F. vesiculosus* communities occurred in the mid 1990s. Large areas devoid of perennial vegetation were recolonized by *F. vesiculosus* communities in the Tallinn Bay and the Kopli Bay areas (Martin and Kukk, 1997).

Pronounced changes in the distribution and structure of the vegetation were found in Tvärminne with sparse occurrence of *Zostera marina*. The vegetation had increased significantly in terms of shoot density since the late 1960s, but not in terms of biomass (Boström *et al.*, 1997). In the Tallinn Bay area, *Z. marina* grows in dense stands quite near the Tallinn harbour area. This species was considered to have disappeared from the area during previous decades.

The Neva Bay littoral zone has been studied at one transect in the 1930s and at several transects in the early 1980s. As a result of heavy eutrophication, extensive macrophyte beds (mainly emergent phanerogams) have developed in the upper Neva Bay coastal zone in freshwater conditions, extending up to 500 m from the shoreline in some locations (Alimov, 1988; Alimov *et al.*, 1997; Golubkov *et al.*, 1987). The phytoperiphyton in those macrophyte beds (e.g. *Scirpus lacustris*) mainly consist of cyanophyta, bacillariophyta and chlorophyta. Green filamentous algae (*Cladophora glomerata, Spirogyra spp., st.* and *Oedogonium spp., st.*) dominate most phanerogam associations.

The investigations in 1998 were carried out in the resort district of St Petersburg, which covers around 30 km of shoreline along the northern coast including the lower portion of the Neva estuary. The study site includes sandy beaches, hard bottoms and mixed habitats. The area is characterized by heavy anthropogenic loading including a relatively high daily output of phosphorus (20–905 g P/day).

At the study site the aquatic living associations in coastal shallows were represented not only by benthic fauna and flora of the types common for sandy littorals, but also by well-developed fouling communities. At the zones of strong wave exposure and ice abrasion, a green filamentous alga, *Cladophora glomerata*, was the predominant fouling plant. In deeper portions, however, the algal community was gradually replaced by the sessile bivalve *Dreissena polymorpha* and associated species. Based on the seasonal dynamics of *Cladophora* biomass and data on vertical distribution and bottom coverage by hard substrata it was calculated that each linear meter of the shoreline could receive no less than 94–110 kg wet weight of filamentous algae after the storm event during summer 1998.

Mass occurrences of filamentous macroalgae. Small quantities of drifting macroalgae were encountered on the Finnish side of the Gulf of Finland in the 1990s. *Ectocarpus siliculosus* blooms on the SW coast of Finland have been suggested to be attributable to wind-induced upwelling and exploitation of brief nutrient pulses (Kiirikki and Blomster, 1996). Mass occurrences of filamentous macroalgae were recorded in an area from Helsinki to Virolahti in 1996. Altogether, 87 locations were visited and about 30 locations with *Cladophora*-dominated mats were found. The effects of these macroalgal mats have not been as marked as in the northern Gulf of Riga, however, where similar events resulted in marked decline in perennial species.

5.3.5. Possibilities of reducing harmful algal blooms in the Gulf of Finland

By: M. Kiirikki, L. Westerholm and J. Sarkkula

The effects of nutrient load reductions on the biomass of littoral filamentous algae, total phytoplankton and nitrogen-fixing cyanobacteria were evaluated for the Gulf of Finland as a whole. Two reduction scenarios were analysed: The Finnish national agenda “Water Protection Targets to 2005” (Anonymous, 1998) and improvement of phosphorus removal at the present wastewater treatment plants in St Petersburg. The Finnish national agenda aims to halve the anthropogenic N and P loads before the year 2005. The reference years for the load reduction are set to the beginning of the 1990s and the targets have thus already been part-

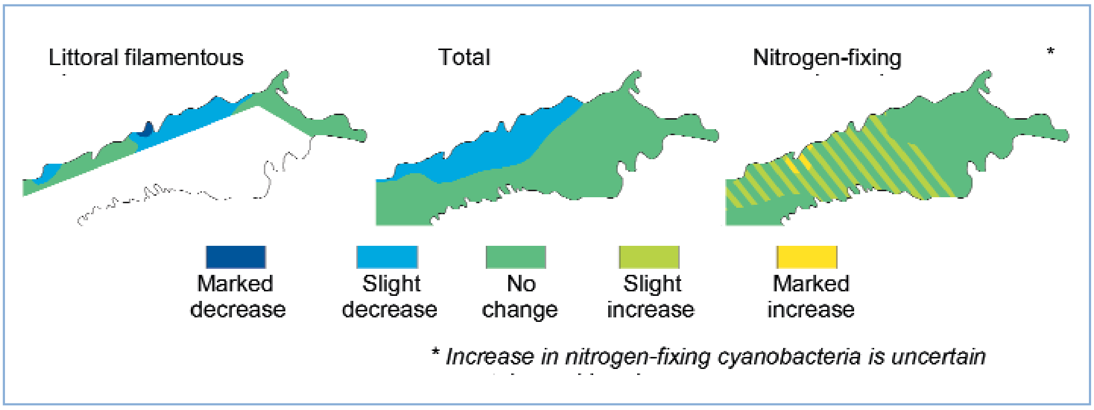


Figure 5.71
Ecological impact of the Finnish national agenda "Water Protection Targets to 2005" estimated using a 3D ecosystem model.

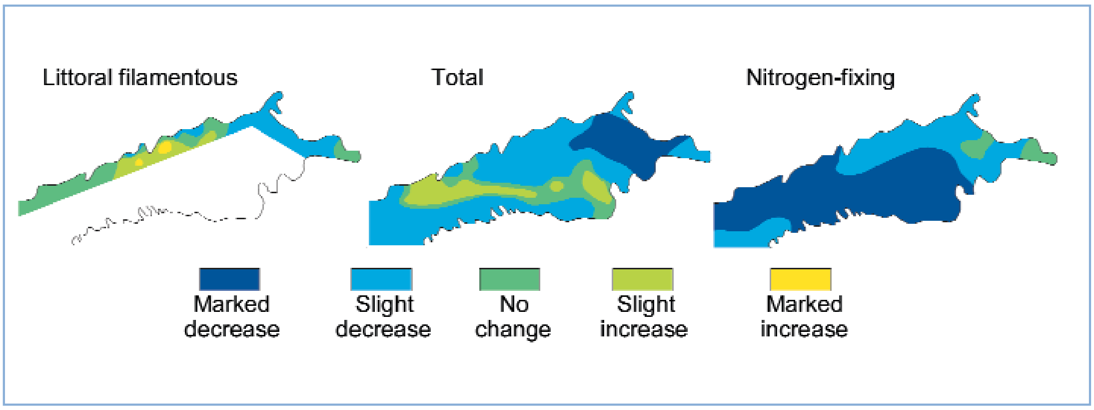


Figure 5.72
Ecological impact of improved phosphorus removal at the present wastewater treatment plants in St Petersburg estimated using a 3D ecosystem model.

ly attained. In St Petersburg, phosphorus removal at the present wastewater treatment plants could be improved by optimizing the biological process (Rantanen, 1994) or by introducing ferrosulphate precipitation. These procedures are relatively cheap and only require minor technical changes to the present treatment plants. The Finnish national agenda should cut the annual bioavailable nitrogen load to the Gulf of Finland by 2,100 tonnes and the phosphorus load by 62 tonnes. Improvements in St Petersburg should cut the bioavailable phosphorus load by 570 tonnes. The effects of load reduction scenarios were tested by using a 3D ecosystem model with a horizontal resolution of 5 km (Kuusisto *et al.*, 1998; Kiirikki *et al.*, 1998 and 2000). The six-year model runs started from the year 1999 thus describing the expected outcome in 2005. The calculated load reductions were subtracted from the load data for 1998.

According to the model calculations, the Finnish national agenda should reduce the biomass of total phytoplankton and littoral filamentous algae in the Finnish coastal waters (Figure 5.71). There is a possible risk of a slight increase in the nitrogen-fixing cyanobacteria, but this is considered to be temporary. The phosphorus reduction in St Petersburg will markedly reduce the nitrogen-fixing cyanobacteria in the central parts of the Gulf of Finland (Figure 5.72). It will also reduce the total phytoplankton biomass in the whole of the calculation area except for a narrow zone across the eastern or central parts of the Gulf of Finland, where a slight increase is anticipated. If both nutrient reduction scenarios are combined, their adverse effects seem to diminish. Thus the possible increase in the nitrogen-fixing cyanobacteria resulting from implementation of the Finnish national agenda is not detectable in the model simulation of the joint scenarios. Moreover, there will be less of an increase in the total phytoplankton biomass in the zone across the Gulf of Finland.

5.4. Gulf of Riga

By: Convenor: A. Yurkovskis. Co-convenor: G. Martin

The location of the monitoring stations in the Gulf of Riga is illustrated in Figure 5.73.

5.4.1. Meteorological, hydrological and hydrographical forcing

By: M. Treiliba, M. Mazmachs, I. Nikolushkina

The mean air temperature at the coast of the Gulf of Riga during the assessment period was 0.3°C above the normal while precipitation was 5% above the normal (Figure 5.74). The warmest years were 1995 (very warm winter and spring, and warm at the beginning of summer) and 1997 (very warm winter and hot, dry, sunny summer). The coldest year with the lowest amount of precipitation was 1996 (very cold, long winter with deep snow cover).

Offshore winds prevailed in the Gulf and the wind direction pattern was normal with a mean wind speed of about 0.4 m/s below the normal (Figure 5.74).

The total sunshine duration was 6% above the normal. Compared with the three previous assessment periods, the present assessment period was the sunniest and second warmest after the 1989–93 assessment period.

Riverine flow during the present assessment period was the second highest after the 1989–93 assessment period. The mean freshwater inflow to the Gulf of Riga was close to the mean for the period 1980–98, corresponding to 117% of the normal value for the period 1961–90. In 1998, riverine flow amounted to 161% of the normal due to a warm winter and a rainy summer, and was the second highest after 1990. In 1996, inflow was only 67% of the normal, the lowest value recorded during the period 1980–98.

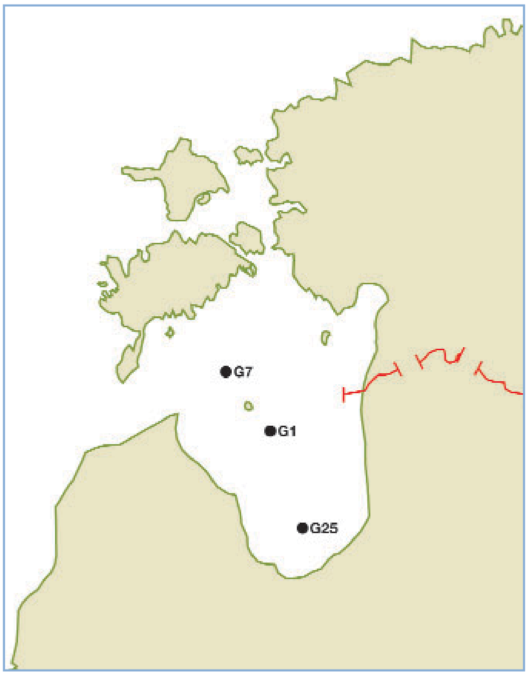


Figure 5.73
Map of the Gulf of Riga indicating the location of the monitoring stations referred to in the text.

More intensive saltwater inflows to the near-bottom layer in 1994 and 1996 (as evidenced by increased salinity values near the bottom) together with higher water temperatures in the upper layer during the last summers caused some increase in stratification during the assessment period. The basic stratification of the water column in the Gulf is determined by the thermocline, and hence is seasonal in character. A significant downtrend in the density gradient in the month of August was observed over the period 1980–93 ($p=0.02$; Figure 5.74). Spring flood of the large rivers that tends to increase the salinity gradient of the water column was elevated in 1994 and 1995, when fresh water reached the central part of the Gulf.

The mean oxygen concentrations below the thermocline tended to increase during the period 1980–95 ($p=0.02$; Figure 5.74). The decline in oxygen content over the period 1995–98 was not statistically significant. During the late summer of 1994 and 1996, oxygen concentrations near the bottom decreased to 1 ml/l.

5.4.2. Hydrochemistry

By: A. Yurkovskis

The nitrate level in winter decreased gradually from 1992 onwards (Figure 5.75). The slope of the reduction is 1.02 µmol/l•yr ($p=0.07$; $r^2=0.60$). The nitrate winter pool showed no clear tendency for the period 1981–91. Using a longer data set of 1974–91 revealed a significant increase in nitrate concentrations, however ($p=0.008$).

Winter phosphate increased linearly over the period 1981–98 (Figure 5.75), the average accumulation of phosphate amounting to 0.018 µmol/l•yr ($p=0.03$). For the whole monitoring period, i.e. 1974–98, the winter phosphate concentration fits a polynomial curve (Yurkovskis, submitted) with enhanced phosphate accumulation until 1996 and depletion of the pool thereafter. The latter finding is in concert with the pattern for nitrate except that there is a five-year delay for phosphate.

The silicate winter concentrations exhibited no tendency over the period 1988–98, although a sharp increase has been evident since 1995.

The summer nutrient pool below the thermocline has also been subjected to trend analysis (Yurkovskis, 1998). Phosphate was substituted by total phosphorus as this is independent of biological processes in the dark layer. Silicate decreased markedly over the period 1984–95 ($p=0.001$) con-

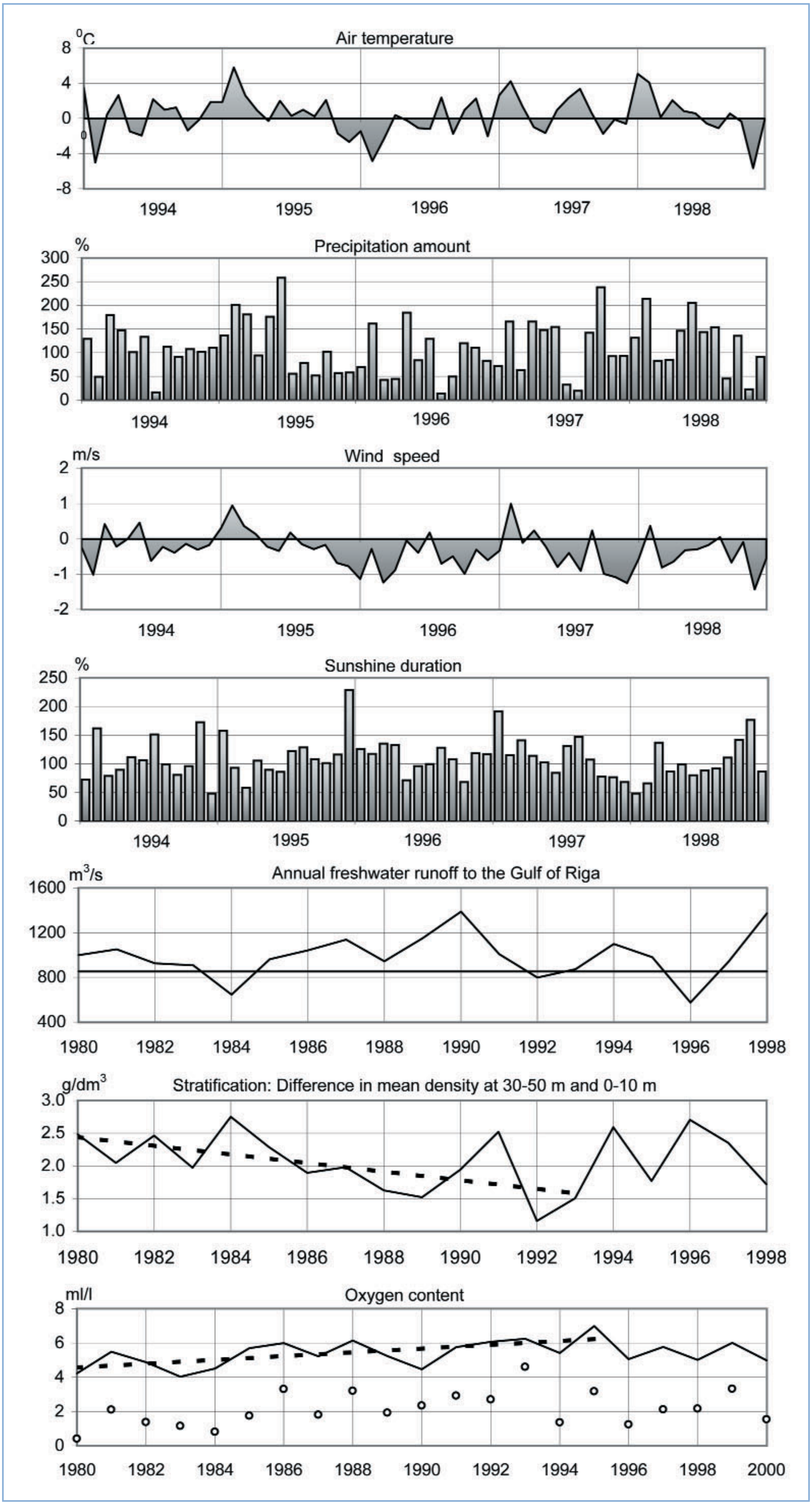


Figure 5.74
Differences in the meteorological characteristics relative to the mean for 1961–90; riverine runoff (horizontal line indicates the mean for 1961–90); stratification at station BMP G1 in August; oxygen content at stations of the central part (average for 30–50 m) and the annual oxygen minimum near the bottom (circles) in the Gulf of Riga (the broken lines indicate the trends).

Figure 5.75
Long-term changes in winter (February) phosphate and nitrate concentrations in the Gulf of Riga, 1981–98 (mean values of the homogenous 0–40 m layer for the offshore BMP stations, G25, G1 and G7; the broken lines indicate the trends).

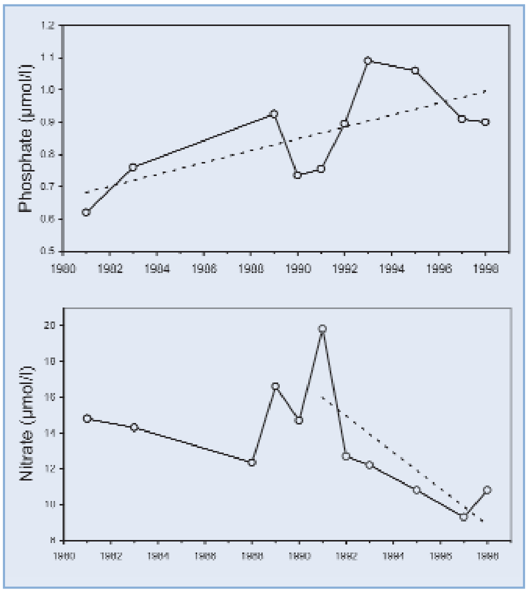
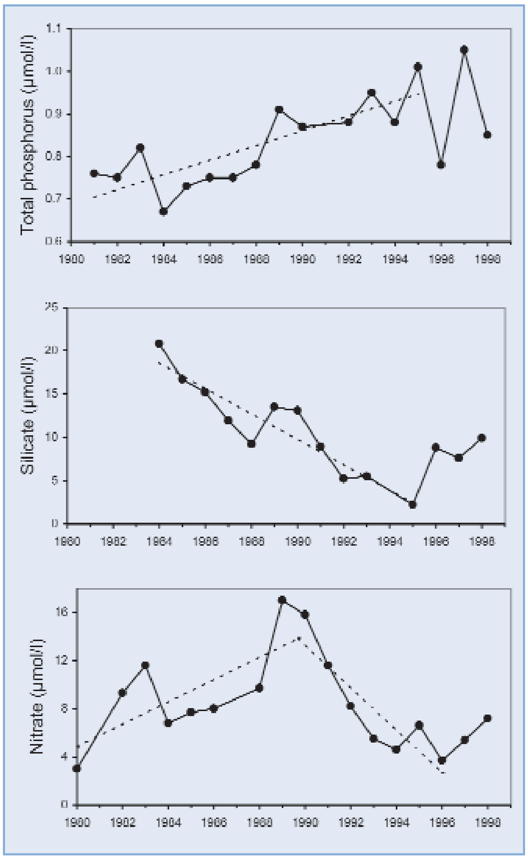


Figure 5.76
Long-term changes in summer (August) total phosphorus, nitrate and silicate concentrations in the Gulf of Riga, 1980–98 (mean values of the deep layer 20–40 m for nine offshore stations; the broken lines indicate the trends; for nitrate, a piecewise linear regression with break-point was used).



sistent with enrichment of the phosphorus pool thus proving enhanced burial of biogenic silicon in the bottom sediments as a result of eutrophication (Conley *et al.*, 1993). Phosphorus explains 83% of silicate variation over the period 1984–95. Further changes in total phosphorus, nitrate and silicate concentrations indicated a reversal in the long-term nutrient dynamics after 1995, when an increase of the silicate pool occurred in response to phosphorus depletion and was accompanied by enrichment of nitrate (Figure 5.76).

The periods of rising and falling nitrate concentrations coincide with corresponding changes in riverine runoff (Figure 5.74). The inflow from the river Daugava alone accounts for 54% of the variation in winter nitrate over the period 1981–98 ($p=0.01$). Together with the nutrient loading data (Laznik *et al.*, 1999), this shows that riverine input is a main factor driving the long-term dynamics of the nitrogen pool in the Gulf (Yurkovskis, 1998). In contrast, phosphorus enrichment in the Gulf lasted five years longer than the maximum of the freshwater input (Figures 5.74 and 5.76). The prolonged accumulation of phosphorus in the first half of the 1990s may be attributed to internal loading, by enhanced mo-bilization of phosphate from the bottom sediments controlled by physicochemical conditions. Analysis of the hydrochemical parameters indicates higher phosphorus release from the sediment in the 1990s than during the preceding period.

During the assessment period, complete inorganic nutrient depletion in spring was followed by a gradual increase of the silicate content from summer onwards and by renewal of the phosphate and nitrate pools during the autumn-winter period. Nutrient seasonal cycling in the surface layer was thus similar to that during the previous five-year period (Yurkovskis and Mazmachs, 1998).

The prespring N:Si:P ratios in the inorganic phase during the assessment period averaged 11:12:1 in the offshore areas, thus indicating nitrate and silicate deficiency relative to the Redfield value. In comparison the ratio was 22:14:1 during the period 1989–91, when nitrate was in excess. During the summer, when phosphate and nitrate are completely depleted in the surface layer, silicate increased steeply over the period 1996–98. In autumn, silicate was in excess relative to nitrate during the assessment period, which was opposite to the pattern in the 1980s.

Silicon was the first element to become limiting during phytoplankton spring blooms of the 1990s. In contrast, autumn growth of phytoplankton during the assessment period was limited by nitrate or phosphate. In the 1980s, in comparison, phosphorus and occasionally silicon tended to be limiting in spring and autumn.

Given that the Gulf of Riga is one large estuarine system, it is natural to find the highest nutrient levels in the southern part of the Gulf near the major river mouths and the lowest levels in its northwestern areas that are most affected by exchange of water with the Baltic Proper. During the assessment period the inorganic N:Si:P ratios varied around an annual mean value of 35:50:1, thus reflecting a surplus of N and Si in the vicinity of the Daugava river. The hydrochemical structure of the water masses in the northern part of the Gulf was determined by a complicated circulation pattern there. Water intrusions from the Moon-sund area and Pärnu Bay overlap water transfer by the main current system from the southern areas causing patchiness in nutrient distribution over the Northern Gulf. Overall, the marine systems of the areas neighbouring the Irbe Strait exhibited N limitation during the assessment period.

5.4.3. Structure and function of the pelagic ecosystem

Phytoplankton

By: B. Kalveka, A. Jaanus, I. Ledaine

In spring and autumn, no significant trends were observed for phytoplankton biomass and chlorophyll concentrations due to the low sampling frequency and intra-annual differences in the occurrence of the phytoplankton bloom. In summer, an increase in both chlorophyll a concentration and phytoplankton biomass was observed in the Gulf of Riga until the beginning of the 1990s. Thereafter the concentration started to decrease (Figure 5.76). The lower phytoplankton biomass seen during the present assessment period than during the assessment period 1989–93 applied to all seasons. The mean chlorophyll a concentration during the present assessment period (2.4 mg/m^3) was closer to that during the period 1984–88 (2.1 mg/m^3) than during the period 1989–93 (3.1 mg/m^3). A decrease in the spring phytoplankton biomass during the present assessment period coincided well with the decline in winter nitrate concentrations observed in the Gulf since the beginning of the 1990s (Figure 5.75).

In spring of 1994 and 1995, the phytoplankton biomass was lower than during the period 1988–91. In the cold spring of 1996, phytoplankton biomass was considerably lower than during analogous conditions in 1985–87. In 1997 and 1998, moreover, biomass was slightly lower than in 1981 and 1982, when winter conditions were similar. The bloom always started at the coastal areas, but peaked in the central part of the Gulf.

Although no changes occurred in total species composition, the relative abundance of the taxonomic groups fluctuated. Thus during the present assessment period the proportion of diatoms decreased compared to the previous periods, while the proportion of Dinophyceae (mostly *Peridiniella catenata*) increased in the phytoplankton biomass (Figure 5.77). In 1997 and 1998, *P. catenata* accounted for 96% of spring biomass due to silicate limitation of diatom growth as early as the end of April. The species composition of diatoms was the same as during the period 1980–93, although biomass was reduced. During spring of 1996, when diatom biomass during the assessment period reached its maximum, the biomass of *Achnanthes taeniata* had decreased more than 5-fold compared to the period 1985–87, while that of *Thalassiosira baltica* had decreased 2.5-fold and that of *Chaetoceros wighamii* had decreased 1.5-fold.

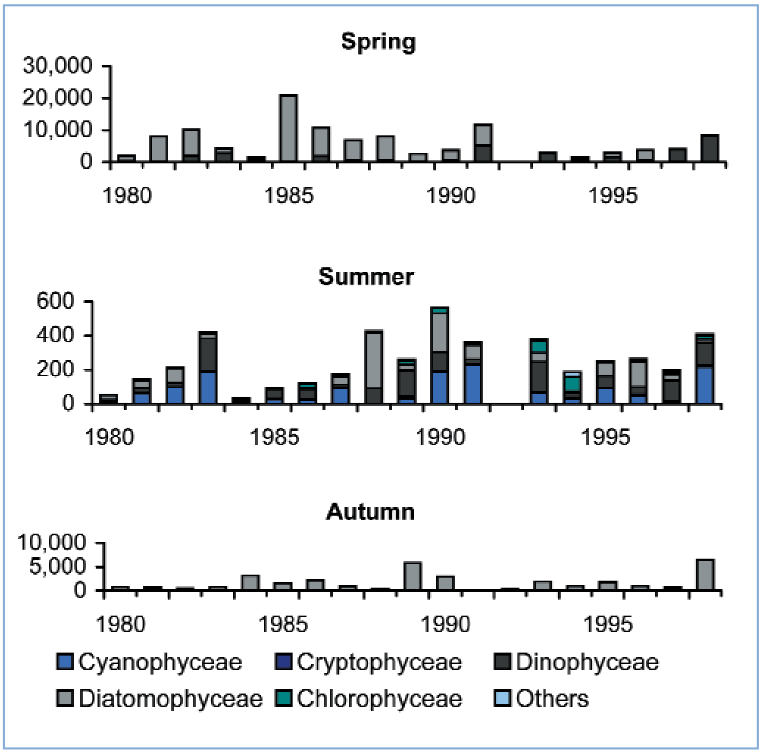


Figure 5.77 Distribution of the biomass of different algal classes in the central part of the Gulf of Riga, 1980–98 (station BMP G1).

In summer during the present assessment period the biomass of various summer algal groups – Cyanophyceae (*Aphanizomenon flos-aquae*), Dinophyceae (*Dinophysis acuminata*), Diatomophyceae (*Actinocyclus octonarius*) and Chlorophyceae (*Oocystis spp.*) – declined relative to the preceding assessment period 1989–93 (Figure 5.77). However, the biomass of Cryptophyceae remained unchanged while that of the group designated "other species" (*Pyramimonas sp.*) even increased. The chlorophyll a concentration and the number of species were higher in the coastal areas and declined towards the central part of the Gulf due to higher salinity and disappearance of the freshwater species.

Phytoplankton biomass in the autumn tended to decrease during the assessment period except in 1998 (Figure 5.77), when biomass was the third highest for the period 1980–98, thus demonstrating most clearly of all seasons the influence of increased freshwater runoff on the phytoplankton even in the central part of the Gulf. *Coscinodiscus granii* accounted for 96% of the total phytoplankton biomass, and its biomass was close to the maximal observed values. In previous years, the decline in biomass of *C. granii* occurred simultaneous-ly with the increase of *A. octonarius* and *Th. baltica*. Similar to spring, the autumn bloom also started earlier at the coastal areas, but peaked in the central part.

During the assessment period the following

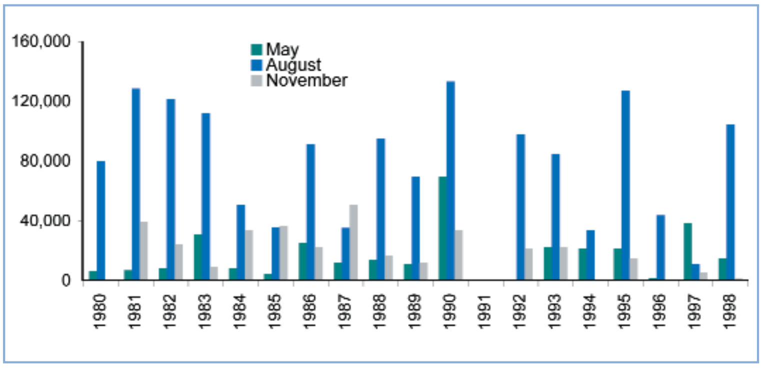


Figure 5.78 Total abundance of mesozooplankton in the open part of the Gulf of Riga (mean values of the 0–40 m layer for three offshore stations).

potentially toxic species were detected in the Gulf: *Aphanizomenon flos-aquae*, *Dinophysis acuminata* and *Chaetoceros danicus* as the dominant species, and occasionally also *Nodularia spumigena* and *Dinophysis acuta*. The biomass of the dinoflagellate *D. acuminata* only increased slightly in 1997 and 1998 compared to previous years and did not cause extreme situations in any part of the Gulf.

Mesozooplankton

By: A. Ikauniece, M. Simm

Mesozooplankton abundance during the assessment period was lower than during the 1989–93 assessment period for all seasons. The present description of the zooplankton community encompasses the coastal (Pärnu Bay) and offshore areas of the Gulf of Riga. The trends in zooplankton abundance differed for the two areas. A significant decreasing trend was evident in autumn over the period 1980–98 in the open part of the Gulf (Figure 5.78). Relative to the preceding assessment period (1989–93), mesozoobenthos abundance was lower during all seasons of the present assessment period. After declining in the early 1980s in the coastal area, zooplankton abundance increased and was higher during the present assessment period (except spring) than during the preceding assessment period.

No changes in zooplankton species composition were observed in spring and autumn during the present assessment period. Zooplankton abundance was dominated by copepods and rotifers, and mainly determined by temperature conditions. In 1996, after a severe winter, mesozooplankton abundance and biomass were the lowest for the whole period 1980–98 in the offshore part of the Gulf. In Pärnu Bay, spring mesozooplankton abundance and biomass were high during the 1990s, when warming of water took place earlier. In summer, copepods and cladocerans were the dominant groups and their abundance was controlled by water temperature and food availability. The relatively high copepod abundance in 1998 coincided with the increased phytoplankton abundance. Water temperature mainly affected cladocerans. The abundance of the introduced species *Cercopagis pengoi* has increased gradually since 1994, but only when average temperature has exceeded 15°C. During the warm summer of 1997, fishing net clogging by *C. pengoi* was reported in the Gulf for the first time.

Function of the pelagic food web

By: E. Kostrichkina, A. Andrushaitis

Based on the complex studies carried out during the present assessment period (Tamminen and Wassmann, 1999), the pelagic ecosystem of the Gulf of Riga is assessed as being highly effective in recycling heterotrophic organisms and having a high capacity to maintain nutrients and organic matter in the upper water layer.

Considerable seasonal changes in the structure of pelagic food webs are a characteristic feature of the Gulf. The main driving forces are the seasonal fluctuations in hydrographic conditions, the structure and size of nutrient pools for primary producers and the food availability for the consumers. In general, the seasonal dynamics of the plankton communities can be described as follows: increase in heterotrophy and in the complexity of the trophic structures from spring till summer and their simplification from summer to autumn due to changes in food availability. During the spring-summer period, the grazing type food web weakens, and the microbial loop gains in importance. The summer bloom of blue-green algae strengthens the microbial loop in mixed surface waters (Kononen, 1992; Sellner, 1997) and causes a decline in zooplankton biomass and the biomass of some phytoplankton fractions (Heerkloss *et al.*, 1984). The blue-green bloom does not affect the intensive development of *Bosmina longispina* and *Keratella spp.*, however, since these species exploit the blue-green algae (Horstman, 1975; Sellner, 1997).

Although inter-annual fluctuation was considerable, spring phytoplankton biomass tended to increase and exhibit diatom dominance during the period 1980–90 due to growth in the nutrient pools. In the 1990s, phytoplankton biomass tended to decrease and changes in a taxonomic composition were observed when the nutrient levels declined and silicon deficiency set in. Diatoms were replaced by dinoflagellates as the dominant species. The increasing significance of facultative heterotrophic species such as *Peridiniella catenata*, particularly since the mid 1990s, indicates that community trophic structure is becoming more complex. The increase in water temperature probably favoured the enhanced biomass of this algal species. The possible effect of these changes on zooplankton is difficult to assess because the amount of zooplankton in spring closely correlates with the water temperature (Kostrichkina *et al.*, 1990). Nevertheless, total zooplankton biomass was lower in the 1990s than in the 1980s.

During the 1990s the role of the microbial loop and nutrient recycling in the water layer became more significant. During the summers in the 1980s, when nutrient concentrations increased in the euphotic layer, total phytoplankton biomass tended to increase. Diatoms became more significant at the end of the 1980s when summer nutrient concentrations reached the maximum long-term values. A decline in the nitrogen and silicon pools in the 1990s and changes in the DIN:DIP ratio caused a decrease in total phytoplankton biomass and reshaped phytoplankton species composition. In response, zooplankton biomass and copepod biomass in particular decreased.

In general, the total plankton biomass was lower in the 1990s than in the 1980s. No crucial changes in autumn community structure were observed during the period 1980–98, although the proportion of some diatom species fluctuated within the phytoplankton community, and the proportion of the various copepod species fluctuated within the zooplankton community.

Nutrient uptake relative to existing pools and inputs. The present evaluation of plankton gross primary production (GPP) is based on data for the southern and central Gulf of Riga for the period 1991–97. Estimated average annual GPP was 255 g C/m^2 • yr (111 g C/m^2 in spring, 38 g C/m^2 during early summer phytoplankton decline, 83 g C/m^2 in summer and 23 g C/m^2 in autumn). This estimate corresponds to mean annual production of 2,015 kilotonnes C within the above-mentioned area of the Gulf of Riga (sea surface = 7,900 km^2, sea volume = 265.5 km^3). Based on a C:N:P ratio of 41:7.2:1 (by weight), 354 kilotonnes N/yr and 49 kilotonnes P/yr would be needed to support the estimated plankton production. Average nutrient pools in the respective sea area are 132 kilotonnes total N and 8.3 kilotonnes total P. The mean turnover time for the total N and total P pools in the whole water mass is thus 136 and 62 days, respectively. If nutrient pools locked within the upper mixed layer are considered separately, the turnover time of nutrients actually available for plankton autotrophs during productive period is as short as 38 days for total N and 14 days for total P.

Estimated nutrient input into the area examined (incl. rivers, coastal point sources, atmospheric deposition and N fixation) is 120 kilotonnes total N/yr and 2.6 kilotonnes total P/yr according to the data of Yurkovskis *et al.*, 1993, Andrushaitis *et al.*, 1995, Laznik *et al.*, 1999 and Granat, 2000. Input of new nutrients into the system may thus support production of 864 kilotonnes C/yr based on total N input or 108 kilotonnes C/yr based on total P input. At least 234 kilotonnes N and 46 kilotonnes P are recycled within the system each year.

Limitation. Experimental studies conducted in 1993 and 1994 (Seppälä *et al.*, 1999) revealed that phytoplankton in the Gulf of Riga is mainly N limited or N-P colimited in summer. P limitation has only been observed in the SE part of the Gulf, mainly influenced by river runoff. The depth of the mixed layer seems to influence the nutrient limitation pattern during the late summer/early autumn periods: A thin-moderate mixed layer favours N limitation, while deep mixing favours emergence of P limited phases (Tamminen and Seppälä, 1999). Evidence exists that termination of phytoplankton spring development during the 1980s was controlled by depletion of the Si pool.

Fate of the produced organic matter. Few estimates of the organic carbon budget exist for the Gulf of Riga system. Andrushaitis *et al.*(1992) estimated plankton GPP for the entire Gulf in 1989 to be 4.1 kilotonnes C, additional 0.9 kilotonnes C_{org} being loaded from external sources. During the same period of time 3.8 kilotonnes C_{org} were mineralized in the water column and 1.2 kilotonnes C settled on the bottom. Of this, 0.7 kilotonnes C_{org} were mineralized in sediments. Biogeochemical modelling of the same system for the period during 1993-95 yielded pretty similar estimates of pelagic production (4.1 kilotonnes C/yr), pelagic respiration (3.1 kilotonnes C/yr) and net sedimentation (1.2 kilotonnes C/yr) (O. Savchuk, personal communication). The ratio of annual export:primary production may thus be estimated at around 25%. Based on simultaneous plankton GPP and POC sedimentation measurements performed in different seasons of 1993–95, the export:plankton GPP ratio was 21% in spring and 9–12% in summer (Lundsgaard *et al.*, 1999; Olesen *et al.*, 1999).

5.4.4. Benthic conditions and macrofauna biology

Sediments

By: J. Aigars

In the soft sediments of the Gulf of Riga, lamination is absent due to bioturbation. Laminated sediments are usually located in the soft bottom areas undisturbed by biota or physical forcing.

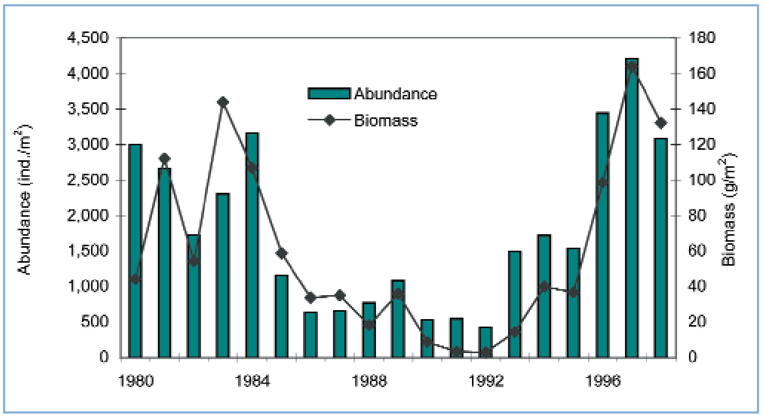
Redox measurements indicate that oxic conditions generally only prevail in the upper few millimetres to one centimetre of the sediment. Near-bottom waters are oxygenated throughout the year, however, and the sediment is subject to a certain degree of bioturbation. During summer, water column stratification and increased microbial activity caused by temperature increase and availability of fresh organic material reduce the oxygen concentration in near-bottom water, in turn causing a temporary decrease in the thickness of the oxic layer in the sediment. Reduced conditions have not hitherto been detected in surface sediments, however.

The organic matter content of the sediments, measured as loss on ignition (LOI) (Håkanson and Jansson, 1983), varies from 0.7% in erosion-transportation bottoms to 17.4% in accumulation bottoms (Carman *et al.*, 1996). LOI tends to decrease with increasing depth in sediments of the accumulation bottoms due to continuous decomposition of organic compounds. In the sediments of the erosion-transportation bottoms LOI tends to decrease down to 2–3 cm, whereafter it increases continuously. This is because sediments located at the transport-erosion bottoms are stiff glacial clay, rich with old organic material, overlain by more recently deposited coarser, organic matter-poor sediment material.

Inorganic carbon and nitrogen generally constitute less than 10% of the total carbon and nitrogen pools in the Gulf of Riga. In contrast, inorganic phosphorus, as in the other areas of the Baltic Sea, accounts for 45–100% of the total phosphorus pool. The concentration ranges for all types of sediment are 192–5,409 Mmol/g for total carbon, 20–520 Mmol/g for total nitrogen and 7–90 Mmol/g for total phosphorus (Carman *et al.* 1996). The average molar ratio of organic carbon and nitrogen in Gulf of Riga sediments is 10.2 and is almost independent of water depth, sediment characteristics and location. This ratio corresponds to that in similar sediment regions around the Baltic (Balzer, 1984; Koop *et al.*, 1990) and is slightly higher than that of marine particulate organic matter, indicating preferential N decomposition. The stability of this ratio suggests that fractionation mainly occurs prior to and immediately upon settling at the sediment surface, however.

Although limited, the data indicate slight seasonal variation in sediment surface concentrations of total carbon, total nitrogen and inorganic phosphorus. In spring and autumn, a small increase in carbon and nitrogen concentrations correlates with the sedimentation of phytoplankton blooms. In contrast, phosphorus accumulation has been observed during summer and seems to be microbially controlled. Since only one-year observations are available, it is not plausible to draw valid conclusions concerning seasonal dynamics of sediment nutrient concentrations.

Figure 5.79
Fluctuation in macrozoobenthos total abundance and biomass (formalin wet weight) in the southern part of the coastal zone of the Gulf of Riga.



Macrozoobenthos

By: M. Ceitlina, J. Kotta

In the southern coastal areas and western part of the Gulf of Riga, macrozoobenthos abundance increased during the assessment period and gradually returned to the level during the early l980s (Figure 5.79). Both quantitative distribution and species composition are heterogenous in the Gulf. The *Macoma balthica* community occupies the shallow coastal zone, whereas the *Monoporeia affinis* community inhabits the deep parts of the Gulf. During the period 1980–98, considerable changes were observed in the benthic community of the Gulf of Riga. Macrozoobenthos abundance and biomass fell in almost all regions of the Gulf during the period 1990–92. During the present assessment period, average macrozoobenthos abundance and biomass ranged from 60 to 2,500 ind./m^2 and from 0.5 to 180 g/m^2, respectively. Although no areas of the Gulf could be regarded as “dead bottoms” with a total absence of benthic animals due to the small depth of the Gulf, conditions in the central part were depressed during the assessment period. Moreover, the area with H_2S permanently present in the surface sediments has increased.

In the southern coastal part, polychaete biomass increased considerably and was 42-fold higher during the assessment period than during the period 1980–85. In contrast, amphipod biomass decreased and was 3-fold lower than in 1980–85. This part of the Gulf of Riga (observation depths 10–12 m) is regarded as the most eutrophic part; the bottom there is muddy sand with addition of detritus. Approximately 20 macrozoobenthos taxa occur here. Of these, 10 are constantly present while the rest are only found occasionally. During the assessment period, macrozoobenthos abundance and biomass were relatively high: 2,453 ind./m^2 and 94.3 g/m^2, respectively (Figure 5.79). *Macoma balthica* is the dominant member of the benthic community, accounting for 31% of total abundance and 87% of total biomass. The polychaetes *Manajunkia aestuarina, Marenzelleria viridis, Hediste diversicolor* and *Pygospio elegans* are the second most important species, together accounting for 65% of total abundance and 10% of total biomass. *M. aestuarina* and *M. viridis* dominate in terms of abundance, while *M. viridis* and

H.diversicolor dominate in terms of biomass. These species particularly increased in abundance during the assessment period. The introduced species *M. viridis,* which has occurred in the Gulf of Riga since 1989–90, increased markedly from 1993 onwards, while the abundance of the local species *M. aestuarina* and *H. diversicolor* has been increasing since 1994 and 1995, respectively. In contrast, the abundance of *P. elegans* declined, although the species remained a permanent component of the benthic community. The number of the amphipods – *Monoporeia affinis*, *Corophium volutator, Bathyporeia pilosa* has gradually decreased in the coastal zone, and they now account for only 5% of total abundance and 0.2% of total biomass. The dominant species in this group have changed. Thus while *B. pilosa* and *M. affinis* were dominant in the early 1980s, *C. volutator* and to a lesser extent *M. affinis* have dominated since the mid 1980s and during the present assessment period. The recovery in total abundance coincided with structural changes in the community (Figure 5.80).

Two tendencies had been observed in the northern coastal area, Pärnu Bay, namely a decline in macrozoobenthos biomass and invasion of the polychaete *M. viridis.* The bottom there is clayey sand with addition of mud, and the macrozoobenthos is influenced by freshwater inflow and effluents from Pärnu Town. *M. balthica* is dominant in terms of biomass, while *M. balthica* and *C. volutator* are dominant in terms of abundance.

In the deeper parts of the Gulf of Riga the pattern of long-term changes in species composition and quantitative parameters differs. In the central part (depth about 50 m), macrozoobenthos abundance and biomass decreased during the assessment period. The marked deterioration in benthic conditions started in 1991 and continued during the assessment period (Figure 5.81). In this muddy bottom area, the presence of H_2S has been observed since 1994. A total of 6 taxa have been found – *M. balthica, M. viridis, M. affinis, Pontoporeia femorata, Saduria entomon* and oligochaetes. Abundance and biomass were very low – 61 ind./m^2 and 10.5 g/m^2, respectively. *S. entomon* accounted for almost the whole biomass – without this species, average biomass did not exceed 0.5 g/m^2. Until the end of the 1980s the amphipods *M. affinis* and *P. femorata* were the most abundant – 1,500–8,000 ind./m^2. The amphipods present in the highest biomass were *M. balthica* and *S. entomon*.

In the western part (depth about 40 m) the macrozoobenthos declined in 1989–93, but recovered during the assessment period, returning to the level in the 1980s. In this part of the Gulf the type of bottom and the species composition do not differ from that in the central part. Quantitatively the situation is very different, however. Total average abundance and biomass during the assessment period were 2,138 ind./m^2 and 29.3 g/m^2, respectively. The oligosaprobic amphipod *P. femorata* was dominant, accounting for 62% of total abundance and 51% of total biomass. Near the Irbe Sound, *M. balthica* accounted for 60% of total biomass. The marine polychaete species *Harmothoe sarsi* was also found there in 1996 and 1997.

Figure 5.80
Variation in abundance of basic benthic groups in the southern part of the coastal zone of the Gulf of Riga.

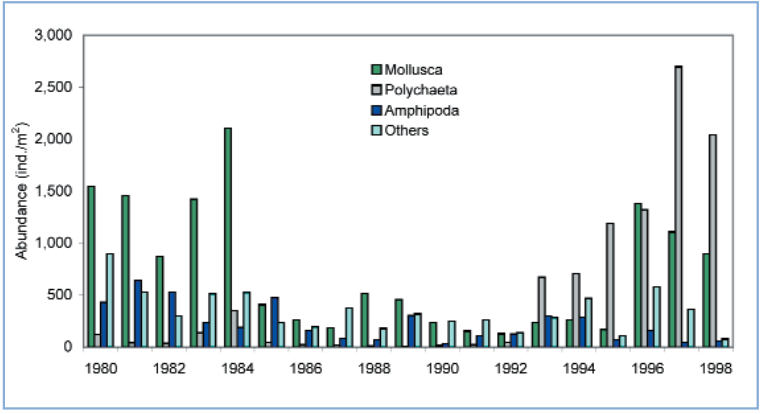
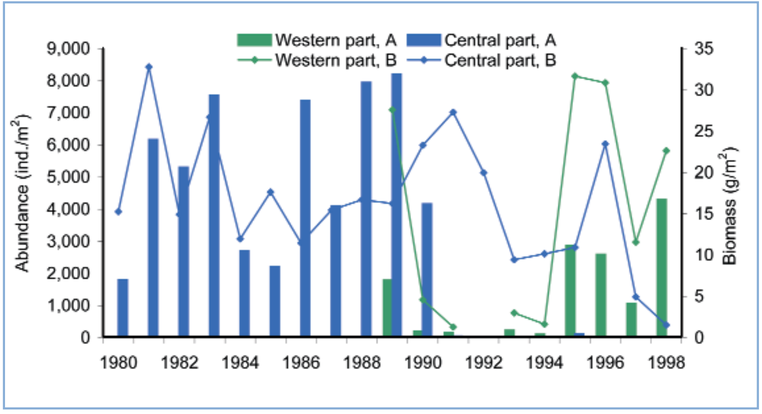


Figure 5.81
Fluctuation in macrozoobenthos total abundance (A) and biomass (B) (formalin wet weight) in the central and western parts of the deep area of the Gulf of Riga.



In the eastern part of the Gulf (depth about 40 m), macrozoobenthos abundance and biomass both increased gradually during the assessment period. The bottom type and the species structure there are the same as in the central and western parts of the Gulf. The average macrozoobenthos biomass during the assessment period varied from 0.7 to 23.8 g/m^2 depending on the presence of H_2S and oxygen conditions. Average abundance was 654 ind./m^2. The rise in abundance was mainly due to *M. affinis* and *P. femorata*, while the increase in biomass was mainly attributable to growth of *M. viridis.*

In the northern part (depth about 30 m) during 1994–98, the *M. balthica* population declined somewhat during the assessment period, while the *M. viridis* population increased. The bottom there is clayey with addition of sand, detritus and Fe-Mn concretions. The species composition is similar to the other parts of the Gulf except that the priapulid *Halicryptus spinulosus* is also present. Both abundance and biomass were high during the

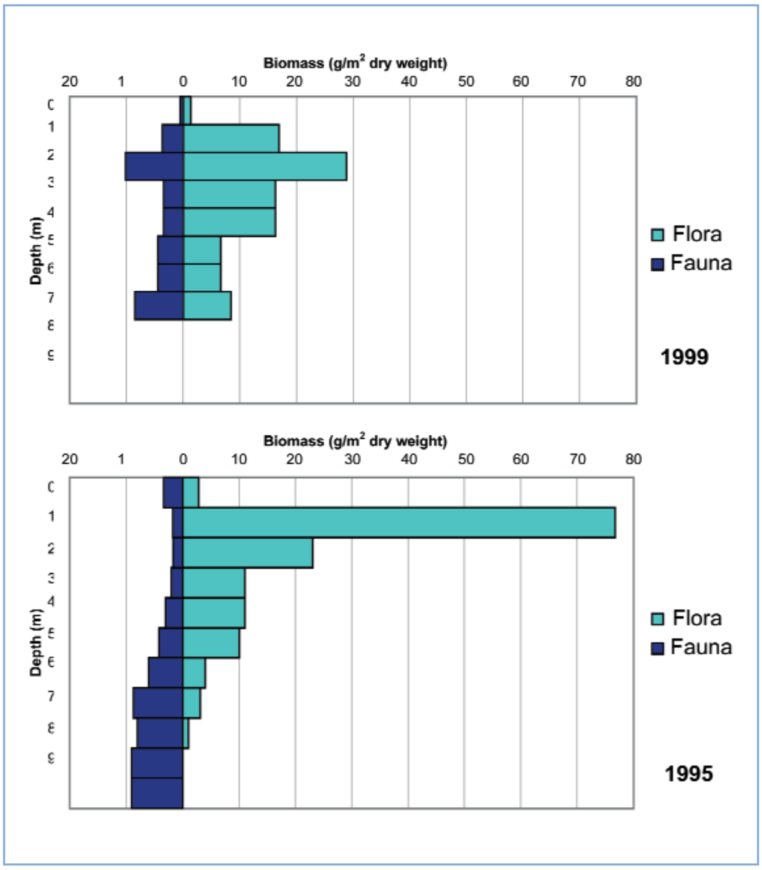


Figure 5.82
Depth distribution of phytobenthic plant and animal biomass in the Kõiguste area (southern coast of Saaremaa island).

assessment period – 1,828 ind./m^2 and 180.6 g/m^2, respectively. *M. balthica, M. affinis,* oligochaetes and *M. viridis* dominated in terms of abundance. *M. balthica* dominated in terms of biomass, accounting for 95–99% of the total.

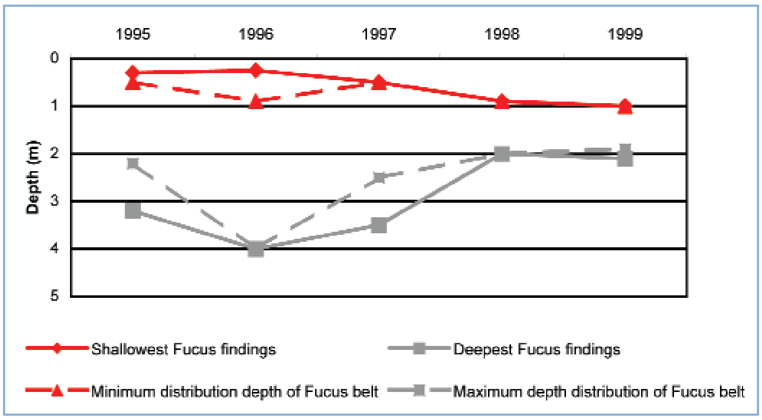
Benthic vegetation

By: G. Martin

The Gulf of Riga can be divided into two parts as regards conditions for the development of phytobenthic communities:

- The southern and eastern part of the Gulf, which includes the coastal areas south of Pärnu Bay up to the Kolka spit. Here the water transparency and trophic conditions are influenced by the riverine inflow, the shoreline is dominated by sandy beaches and stony bottoms, and the coastal slope is more steep.
- The northern part of the Gulf. This lacks any notable freshwater inflows and water quality is mainly influenced by the extensive water exchange processes taking place through the Irbe and Suur straits. The bottom composition is much more diverse ranging from muddy bottoms in the shallow closed or semi-enclosed bays to hard, rocky bottoms composed of limestone.

Figure 5.83
Decline of Fucus vesiculosus in the Kõiguste area (southern coast of Saaremaa island) illustrated by changes in depth distribution of Fucus vesiculosus and Fucus vesiculosus belt.



According to recent scientific publications, the phytobenthos of the Gulf of Riga includes 43 species of algae and flowering plants (Kautsky *et al.*, 1999; Martin, 1999). The changes in the phytobenthos species composition have been quite remarkable in the long term. Many species have disappeared from the area since the beginning of the century (Kukk, 1995).

In recent years, mass occurrence of filamentous macroalgae (mainly brown algae *Pilayella littoralis*) has been observed in the Gulf. This phenomenon have caused several drastic changes in the phytobenthic communities. The decline of *Fucus vesiculosus* in the northern part of the Gulf during the years 1997 and 1998 is most probably attributable to overgrowing by epiphytes. The decline of the commercially exploited loose *Furcellaria lumbricalis-Coccotylus truncatus* community in the inner sea of West Estonian Archipelago in 1997 is directly linked to the mass occurrence of epiphytic *Pilayella littoralis* in the area.

Long-term changes have occurred in the distribution of phytobenthic biomass in the area. These changes are more drastic in the northern part of the Gulf. Phytobenthic biomass peaked at a depth of 1–2 m in the range 5,000–6,000 g/m^2 wet weight during the 1960s and 70s. In the mid 1990s, the maximum biomass decreased to less than 4,000 g/m^2 wet weight in the same area. The depth distribution limit of phytobenthic communities in the Gulf has remained at 11 m.

During the assessment period changes in the structure and vertical distribution pattern of phytobenthic communities were recorded in both the southern and northern parts of the Gulf. In the Kõiguste area (southern coast of Saaremaa island), the high biomasses of phytobenthic communities observed here in 1995 at a depth of 2–3 m (up to 750 g/m^2 dry weight) have been replaced by communities of much lower biomass (max. 270 g/m^2 dry weight) (Figure 5.82). The change in biomass is attributable to a decline in *F. vesiculosus* in the area (Figure 5.83).

Changes in the southern part of the Gulf are less obvious than in the northern part. In Saulkrasti area the phytobenthic communities were extremely poor in the 1996. Only two species of green algae were observed in the area inhabiting the depth interval 1–2 m with quite low biomass (30 g/m^2 dry weight). In 1999, the phytobenthos of the same area was a bit more diverse (10 species). Biomass maximum was recorded at a depth of 2 m (73 g/m²). The phytobenthic biomass disappeared by a depth of 8 m. Marked changes have taken place in the quantitative parameters of the animal component of phytobenthic communities. The biomass of filter feeders currently exceeds 50 g/m² dry weight in the areas where suitable substrate was totally lacking in any sessile fauna in 1996 (Figure 5.84).

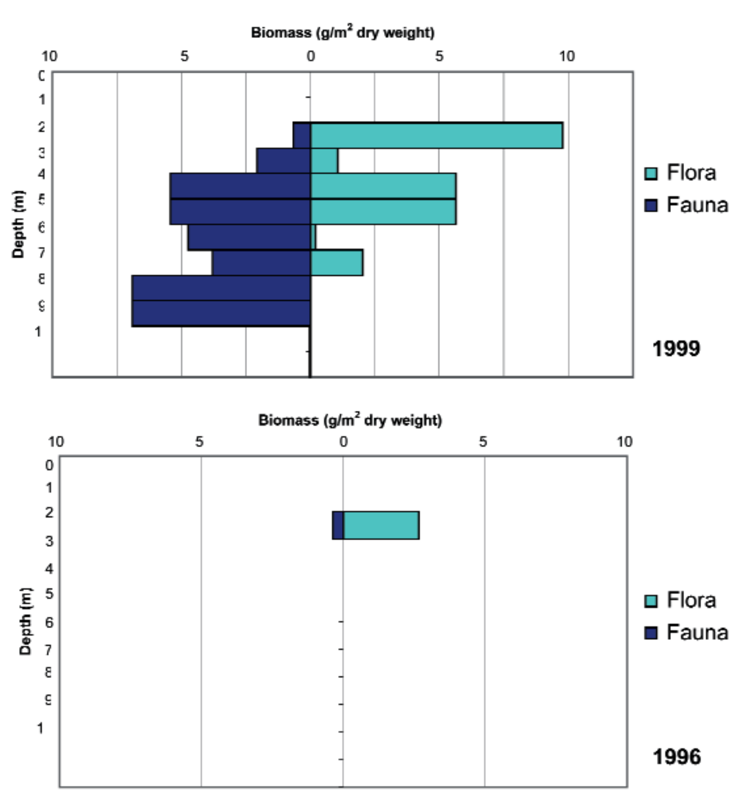


Figure 5.84
Depth distribution of phytobenthic plant and animal biomass in the Saulkrasti area (south-eastern coast of the Gulf of Riga).

Benthic-pelagic coupling

By: E. Kostrichkina, A. Yurkovskis

In the Gulf of Riga, the total flux of the deposited material during the productive seasons (spring-autumn) of the 1990s should have been lower than in the 1980s. As regards spring, the greatest changes in delivery of particulate matter to the sediment-water interface must have occurred in the second half of the 1990s as a result of the decrease in the amount of phytoplankton and shift from diatom dominance to dinoflagellate dominance. These changes were attributable to a decline in the nutrient pools (Figure 5.75), the occurrence of Si deficiency and perhaps also the increase in water temperature. The decrease in summer phytoplankton biomass after 1990 caused by a decline in the nutrient content in the photic layer led to reduced mesozooplankton biomass and consequently probably to decreased transfer of detritus to benthic macroinvertebrates. An analogous tendency might apply to autumn due to decreased phytoplankton biomass.

In the southern coastal zone of the Gulf of Riga, where freshwater inflow and consequently input of allochtonous detritus of low nutritional quality (Lehtonen, 1997) is considerable, the decrease in the amount of deposited material did not influence the dominant deposit feeders, namely the mollusc *Macoma baltica*, the polychaetes *Marencelleria viridis* and *Hediste diversicolor* and the omnivorous species *Saduria entomon*, which occasionally occurred in this area. Their biomass even increased in the 1990s relative to the second half of the 1980s, probably due to the increase in water temperature and the natural periodic fluctuations in species abundance. Despite the decline in transfer of material from pelagic to benthic communities, the total macrozoobenthos biomass (due to dominance of *M. balthica*) thus increased after 1993 relative to the second half of the 1980s. In contrast to the above-mentioned species, the biomass of seston- and deposit feeding amphipods (*Corophium volutator, Bathyporeia pilosa, Monoporeia affinis*) decreased in the 1990s.

In the deep zone of the Gulf, the decline of both sedimentation and the quality of deposited material in the 1990s together with the other factors (temperature, oxygen conditions) led to considerably diminished biomass of dominant species – amphipods seston- and detritus-feeding *M. affinis and Pontoporeia femorata*. According to Lehtonen (1997), the most important factor limiting these species in the Gulf is food availability. Moreover, amphipod abundance remained relatively high in the western part of the Gulf, where the oxygen conditions are better due to water intrusions from the Baltic Proper. The decrease in input of detrital material to the sediment did not influence the omnivorous species *S. entomon*, which has become the dominant species in the central part of the Gulf since the mid 1990s. The changes in zoobenthos species and trophic structures resulted in increased total biomass in the western part during the second half of the 1990s compared to the beginning of the decade. In the eastern part, the biomass remained at a low level.

The plankton and zoobenthos biomass is higher in the southern areas than in the central part of the Gulf due to land-based nutrient and organic matter loading, especially near the Daugava estuary. The studies of plankton and benthos temporal dynamics revealed an identical trend towards long-term changes in both the southern and central areas.

The reduction in species diversity in the areas with oxygen deficiency is attributable to the disappearance of oxygen-demanding species. At particularly low oxygen levels and in the presence of hydrogen sulphide, almost the only species present

(99–100% of the total biomass) is the highly mobile *S. entomon*, which is tolerant to oxygen deficiency. The long-term dynamics of zoobenthos biomass (1989–98) in the central part of the Gulf (Station G1; 56m depth) over the period 1989–98 thus did not correlate with the oxygen concentration near the bottom.

In the eutrophic Gulf of Riga, large inputs of phytodetrital material to the sediment cause oxygen depletion at the bottom followed by nitrogen losses due to denitrification. Low nitrate concentrations relative to oxygen consumption in the bottom-water fluxes indicated intensive dissimilatory nitrate reduction at the seabed over the past two decades. Comparison of normalized concentrations of both nitrate and ammonium in the fluxes confirmed that denitrification predominated in these reactions. Denitrification would thus seem to represent a relevant sink for nitrogen in the Gulf, probably removing a large part of deposited nitrogen from recycling.

Enhanced phytodetrital deposition caused a gradual long-term increase in deep-water oxygen consumption (Berzinsh, 1995). From the regression line, oxygen consumption was calculated to be 0.88 ml/l • month in May–August 1963 (20–50 m layer), and 1.26 ml/l • month in 1990. According to M. Mazmachs (personal communication), average oxygen consumption decreased by 10% over the assessment period (1.02 ml/l • month) relative to the period 1973-93. These eutrophication-related long-term changes in oxygen consumption were not crucial for oxygen dynamics in deep waters, however.

The alterations in oxygen conditions near the bottom forcing exchange processes in the sediment-water interface are principally dependent on the intrusion of more saline water from the Baltic Proper (Yurkovskis, 1998) and on summer stratification in the Gulf. The efficiency of exchange in the water is largely determined by the salinity maximum near the bottom. The vertical exchange processes thus tended to increase over the period 1980–93, thereafter to decrease again (Figure 5.74). The summer export flux of nutrients to the productive layer was consequently relatively weakened during the assessment period due to both higher density gradients and decreased nutrient content under the pycnocline (Figure 5.76).

Fish by the thousands washed ashore due to serious oxygen depletion at Øster Hurup in the western part of Kattegat, October 2002.
Photo: Christen Jensen, County of Nordjylland, Denmark

5.5. Kattegat and the Belt Sea

By: Convenor: G. Ærtebjerg. Co-convenor: G. Dave

The location of the monitoring stations in the Kattegat and Belt Sea area is illustrated in Figure 5.85.

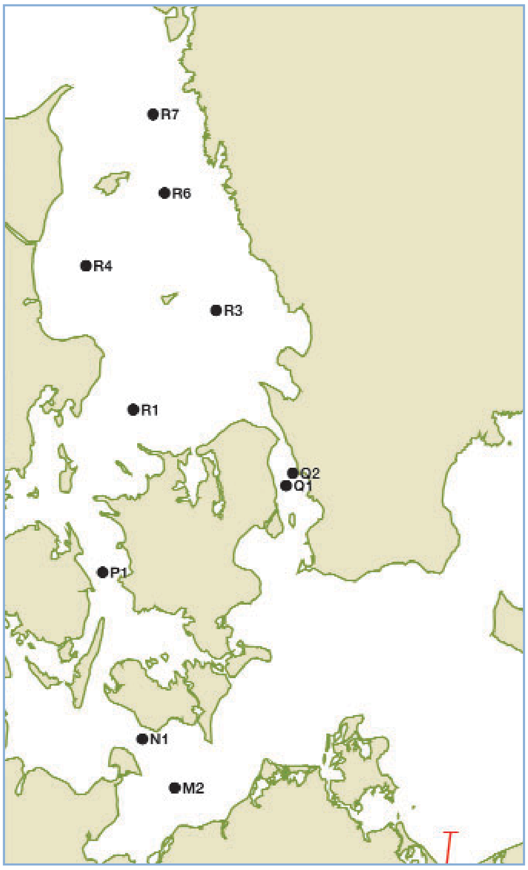


Figure 5.85
Map of the Kattegat and the Belt Sea indicating the location of the monitoring stations referred to in the text.

5.5.1. Meteorological, hydrological and hydrographical forcing

By: B. Sjöberg, H. U. Lass, H. Alexandersson, L. Andersson, B. Rasmussen

The hydrography of the Kattegat and the Belt Sea is characterized by pronounced salinity stratification, vertically as well as horizontally. In the Kattegat, a sharp halocline separates the surface layer (salinity 15–25 PSU) from the deep water (salinity 30–35 PSU). The average stratification reflects the net circulation in the Kattegat, which is essentially driven by wind-induced entrainment and horizontal advection of less saline Baltic water.

In out-flow situations three distinct water masses typically dominate the surface layer in the south, emanating from the Belt Seas, the Sound and the central Kattegat, with salinities of 14–20 PSU, 10 PSU and 18 PSU, respectively (Rasmussen, 1997; Pedersen, 1993). When the flow is reversed, mixing occurs, and only central surface Kattegat water persists. The Belt Sea is also usually strongly salinity-stratified.

Meteorology

The period 1994–98 is broadly characterized by wet and mild weather. While 1998 was extremely windy, wet and variable, the years 1995–97 had typically long periods of calm weather. The monthly mean temperature anomaly (represented by data from Varberg, 57°06'N, 12°16'E on the Swedish west coast) was maximum in early spring 1995, summer 1997 and winter-early spring 1998, and minimum in winter 1995/96. The mean annual temperature deviations in each year of the assessment period 1994–98 were 0.8, 0.5, -0.7, 0.6 and 0.3°C, respectively.

Precipitation was high in Denmark and western Sweden during 1994–95 and 1998 (see Table 3.2). The increase in precipitation was most marked in autumn and winter and is linked to a dominance of strong westerly winds. As regards the Kattegat and Belt Sea area, however, 1996 and 1997 were extremely dry years with exceptionally low runoff (see Tables 3.2 and 3.3).

Global radiation anomalies exhibit a clear positive correlation with temperature and a negative correlation with precipitation. In the years 1994–97 there was a clear tendency towards cloudy weather in May and June and more sunny weather in July and August. The pattern was reversed in 1998.

No severe storms hit the area during the assessment period 1994–98. The number of periods with windy weather (i.e. maximum ten-minute winds of at least 21 m/s) at Nidingen (57°18’N, 11°54’E) in the northern Kattegat was greatest in 1994 (12) and least 1996 (1). Not unnaturally, it is the wettest months that also are the windiest, e.g. September 1994 and October 1998.

Hydrography

The surface temperature is usually highest in August and lowest in February. Comparing the present assessment period with the past 20 years, monthly surface mean temperature has increased (from 17°C to 19°C) during July to September (Table 5.13). The increase is attributable to the warm and sunny summers 1994–97.

In the deep water, below the halocline, the temperature is lowest around April and highest in October. During the assessment period the temperature has increased slightly in summer and decreased from about 10°C to 9°C in September/October.

The average monthly surface salinity undergoes annual cycling with maximum values during autumn-winter and minimum during late spring and summer (Table 5.13). This is essentially an effect of the annual variation in runoff to the Baltic Marine Area causing volume flow maxima through the Belt Sea and Kattegat in April-May

	Area and depth \ Month	1	2	3	4	5	6	7	8	9	10	11	12
Temperature, °C	Fladen (1) – surface	1.9	1.7	2.7	5.6	9.8	13.5	18.4	19.6	15.3	11.2	8.0	4.3
	Anholt (2) – surface	1.5	1.1	2.6	5.5	10.1	14.6	18.8	19.7	15.1	11.4	7.9	4.1
	W Landsk (3) – surface	2.1	1.9	2.5	5.8	10.1	14.3	16.8	19.4	14.9	11.2	8.2	4.8
	Fladen – deep	6.4	5.6	4.9	5.1	5.8	6.6	7.6	8.5	9.0	9.4	10.2	8.6
	Anholt – deep	6.8	5.8	5.6	5.2	5.4	6.0	6.9	8.1	9.7	10.5	9.7	9.4
	W Landsk – deep	5.7	5.2	4.9				6.5		10.0			
Salinity, PSU	Fladen – surface	24.3	25.2	24.1	21.3	18.7	21.3	20.8	18.6	22.0	26.0	26.1	24.0
	Anholt – surface	22.1	21.3	21.7	19.2	16.2	17.0	18.0	17.8	20.3	21.7	22.7	21.2
	W Landsk – surface	14.2	11.0	15.3	10.0	9.3	9.4	8.5	11.8	11.9	15.6	12.1	12.8
	Fladen – deep	34.1	34.2	33.8	34.3	34.7	34.4	33.9	34.3	34.6	34.3	34.2	34.3
	Anholt – deep	33.2	33.4	33.4	33.4	34.0	33.9	33.7	33.6	33.5	33.7	33.1	33.2
	W Landsk – deep	30.7	30.7	31.7			33.1	33.3		32.6			31.9
O_2, ml/l	Fladen – deep	6.2	6.2	6.6	6.1	5.8	5.6	4.9	4.4	4.4	4.6	4.9	5.1
	Anholt – deep	5.9	6.0	5.9	5.5	5.5	5.3	4.4	3.6	3.3	3.9	4.3	4.8
	W Landsk – deep	5.7	6.2	5.8			4.9	3.6		2.0			3.8
O_2 saturation, %	Fladen – deep	89.5	88.3	92.1	85.9	83.3	82.3	72.5	67.5	68.1	70.6	76.9	80.0
	Anholt – deep	84.9	85.7	83.6	76.8	78.3	76.1	64.0	54.5	51.6	61.4	67.6	73.5
	W Landsk – deep	83.3	84.4	79.3				52.7		33.3			57.3

Significant increase (p<5%)

Significant decrease (p<5%)

(1) Fladen = BMP R6
(2) Anholt = BMP R3
(3) W Landsk = BMP Q2

Table 5.13
Monthly mean temperature, salinity, oxygen content and oxygen saturation together with trend test comparing the periods 1994–98 and 1979–93

(Rasmussen, 1995). The surface salinity varies between 20–25 PSU in the north, 15–20 PSU in the south and 10–15 PSU in the Belt Sea. The significant differences between the 1994–98 and 1979–93 assessment periods indicate a tendency towards greater dominance of Baltic outflow in the mild winters, warm summers and early autumns, and more mixing with North Sea water in the windy springs and autumns during the present assessment period.

Below the halocline, in the bottom water, the conditions are much more homogenous, and annual variation is much less pronounced. However, a bottom water salinity maximum is present during summer due to weakening of the horizontal density gradient in the calm summer months (Rasmussen, 1995). The salinity of the bottom water has increased significantly during the present assessment period, with most of the increase occurring in autumn.

Oxygen

The oxygen conditions vary during the course of the year with maximum values in February–March and minimum values in September–October. Conditions seem to have improved during the present assessment period although inter-annual variation is considerable. In 1994, severe oxygen deficiency occurred in estuaries and fjords, while the conditions in the open areas were relatively good due to an event with strong winds in late summer. In 1995, after a warm and calm summer, oxygen conditions were severe in open areas as well as in the estuaries and fjords. The two subsequent years, when runoff was low, oxygen conditions were quite good. Conditions deteriorated again in 1998, although they were not as bad as in 1995.

Time-series analyses (Agger and Ærtebjerg, 1996) revealed a general deterioration in oxygen conditions from the 1970s to the end of the 1980s. In the southern Kattegat and the Sound, conditions improved significantly during the period 1989–97. In the Belt Sea conditions have improved in spring, but deteriorated in July–October (Ærtebjerg *et al.*, 1998). The oxygen concentration is always the net result of several influences affecting both input and consumption, however. As the natural variability is therefore high, the future development is difficult to predict. The improved oxygen conditions seen during the present assessment period are probably related to the low runoff and nutrient load in the dry years 1996–97 (Rask *et al.*, 1999).

Light conditions, as measured by Secchi depth, have been improving (Markager *et al.*, 1999), and a statistically significant positive trend has been demonstrated for the periods 1977–98 and 1989–98 in the open areas and for the period 1977–89 in the coastal areas.

5.5.2. Hydrochemistry

By: D. Conley, L. Andersson, H. Gaul

During the 1994–98 assessment period, lower than average winter nutrient concentrations were observed during 1996 and 1997 in the Kattegat and the Belt Sea, especially for winter nitrate concentrations. The lower values could be attributable to reduced nutrient loading associated with the lower than average rainfall during those two years. In contrast, high winter nitrate concentrations were observed in 1994 due to high riverine flow. No consistent long-term trends in nitrogen concentrations were identified for the period 1980–98 (Figure 5.86).

Long-term trends in surface waters

A long-term decline in phosphate concentration is evident for the period 1980–98, especially during the present assessment period. The average Kattegat winter phosphate concentration is approximately 50% lower than the high phosphate concentration observed during the 1980s, and is now similar to that observed during the 1970s.

Denmark has significantly reduced its total phosphorus load, thereby significantly reducing the estuarine phosphate concentration. Whether or not the reduction in phosphorus load is the underlying cause of reductions in phosphate concentrations in the open Kattegat and Belt Sea is open to question, however. Significant declines in phosphate concentrations have also been reported in surface waters of the Arkona Basin, which is the source of much of the water passing through the Danish straits. The observed declines may therefore be partly attributable to large-scale biogeochemical changes in phosphate.

Changes in mean winter surface concentrations of DIN (DIN = nitrate + nitrite + ammonium) and DIP (DIP = phosphate) between the four 5-year assessment periods are shown in Figure 5.87. Averaging over years with very different load and hydrographic conditions masks the recent general development in DIP. However, a prevailing tendency towards decreasing DIP concentrations is seen at most stations, whereas no general tendency is apparent for DIN. The DIN:DIP ratio (mean 12.2; range 8.8–14.8) is somewhat below the Redfield ratio.

Long-term trends in bottom water nutrient concentrations have not been observed. Neither have long-term changes in the seasonal cycle of nutrients or in geographical variation relative to the previous assessment periods.

Limiting nutrients

A variety of indicators have been used to assess nutrient limitation, including determination of phytoplankton physiological state, inorganic nutrient stoichiometry and concentration, phytoplankton composition ratios, and measures of phytoplankton growth. A simple first order approach can be used to assess potential nutrient limitation by examining for time periods during which nutrient concentrations are below theoretical half-saturation constants (Ks) for uptake (2 µM for DIN, 0.2 µM for DIP and 2 mM for DSi (Fisher *et al.*, 1992), and comparing the stoichiometry to the expected Redfield ratio (16 for DIN:DIP). While crude, this has been found to be a robust approach for determining which nutrient is most limiting.

Both nutrient concentrations and nutrient ratios suggest that DIN continues to be the nutrient potentially most limiting to phytoplankton biomass in the Kattegat and the Belt Sea (Figure 5.88). An analysis of nutrient limitation revealed that phytoplankton was mainly potentially co-limited by DIN

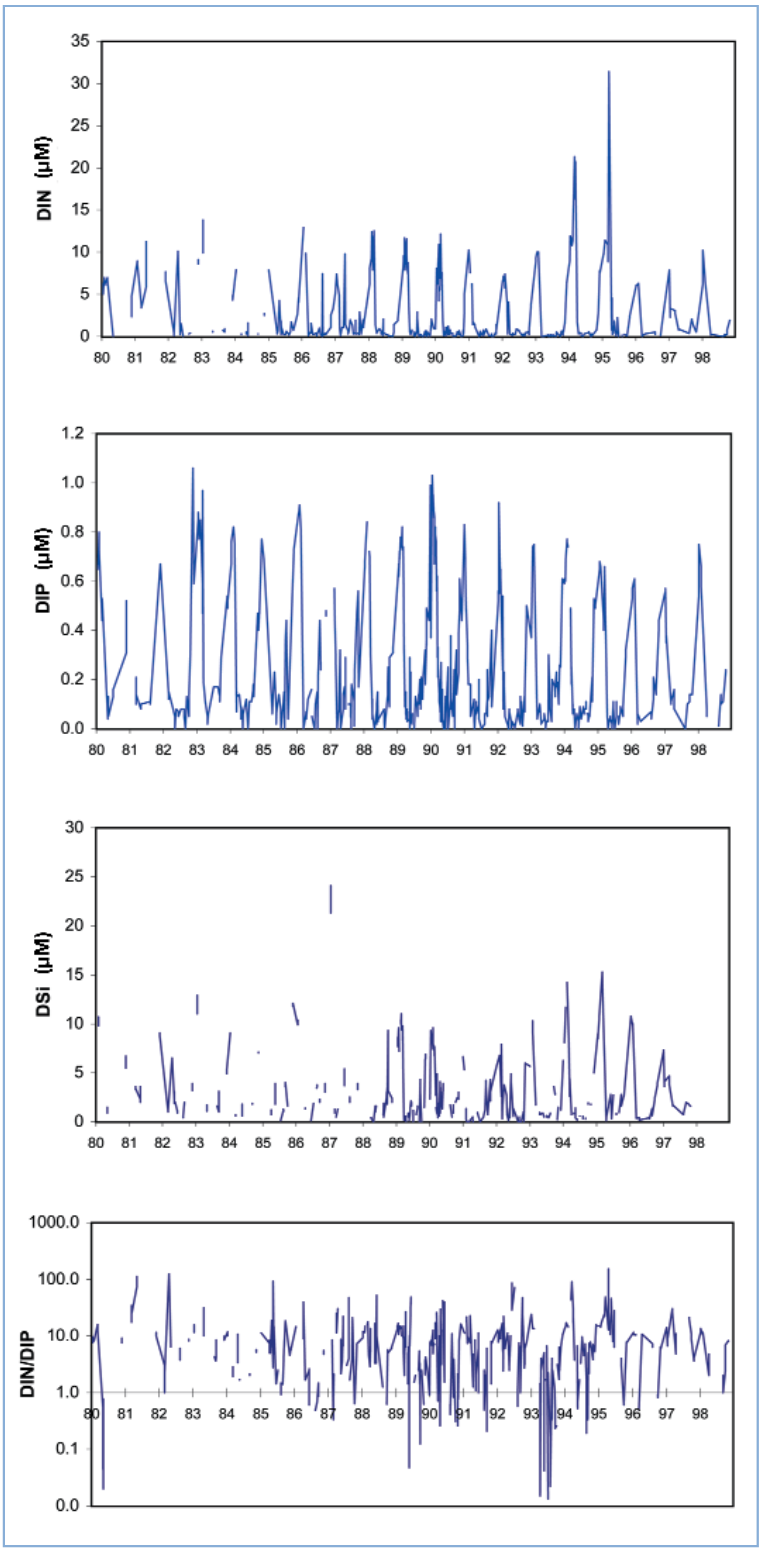
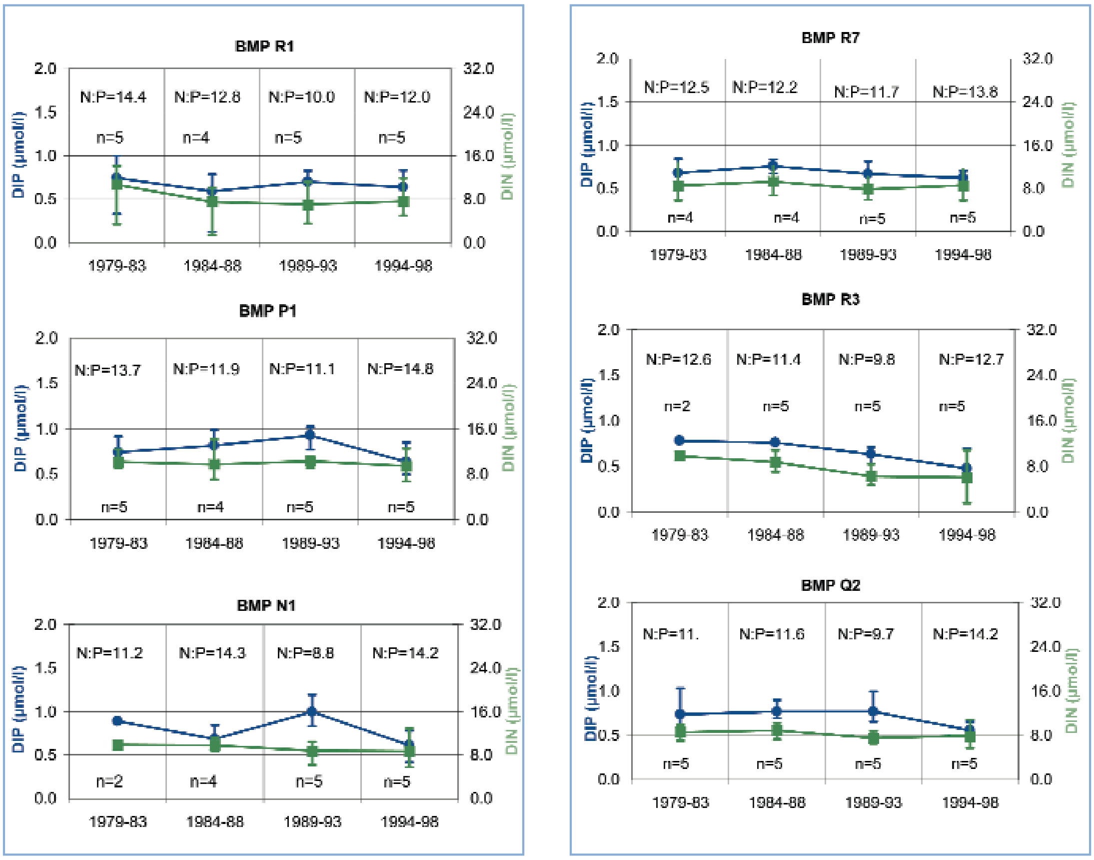


Figure 5.86
Dissolved inorganic nitrogen (DIN = nitrate + nitrite + ammonia), dissolved inorganic phosphate (DIP = ortho-phosphate) and dissolved silicate (DSi = silicate) in surface waters (0–5 m depth) at station BMP R7 (northeastern Kattegat) over the period 1980–98.

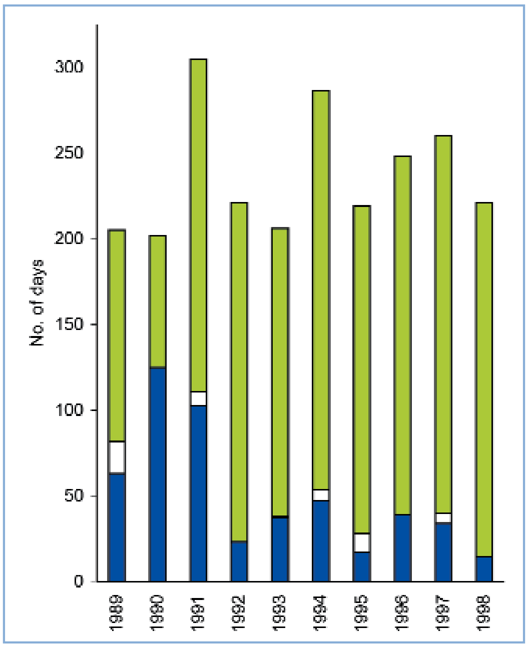
Figure 5.87
Long-term changes in winter concentrations of DIP and DIN at stations BMP N1, P1, Q2, R1, R3 and R7 in the Belt Sea and Kattegat showing five-year average concentrations, maximum and minimum values, mean DIN:DIP ratios and number of years covered by the period (n).



and phosphate concentrations or limited by DIN concentrations (Ærtebjerg *et al.*, 1998). The Redfield ratios suggest that when phytoplankton was co-limited by DIN and phosphate, DIN was often the main potentially limiting nutrient.

There were only a few days each year when phosphate concentrations were low enough and DIN concentrations high enough to render phosphate potentially limiting. The long-term decline in phosphate concentrations has also changed potential nutrient limitation in the Kattegat and Belt Sea areas. Although phosphate alone was only potentially limiting for limited periods each year, the number of days that the open sea areas are co-limited by low concentrations of DIN and phosphate has significantly increased over the years (Ærtebjerg *et al.*, 1998). The concentration of dissolved silicate is occasionally sufficiently low (<2 µM) to limit diatom populations.

Figure 5.88
Number of days with potential nutrient limitation in surface water at station BMP R1 in the southwestern Kattegat (from Markager et al., 1999). Potential nutrient limitation is defined as the amount of time a nutrient is below the theoretical half-saturation constant (K_S) for uptake. The K_S values used in the international literature (Fisher et al., 1992) are 2 µM for DIN (dissolved inorganic nitrogen = nitrate + nitrite + ammonium) and 0.2 µM for DIP phosphate).



5.5.3. Structure and function of the pelagic system

Chlorophyll a

By: N. Wasmund

Seasonal variation. Short but very intensive growth phases leading to high biomass peaks (blooms) of phytoplankton are the most conspicuous features of the annual cycle. As the timing of these blooms is rather constant, the five years of the assessment period were combined to a single annual time series for each station in Figures 5.89–5.91 with the chlorophyll a concentration serving as an approximation for total phytoplankton biomass. One representative station each was selected for the Kattegat (BMP R3), the Sound (BMP Q2) and Mecklenburg Bight (BMP M2).

The peak of the spring bloom normally occurred in mid March, which is much earlier than in the Baltic Proper. In the Kattegat, the bloom occurred even in January to February in 1997 and 1998, probably due to periods of calm and sunny weather. After the spring bloom, low chlorophyll a con-

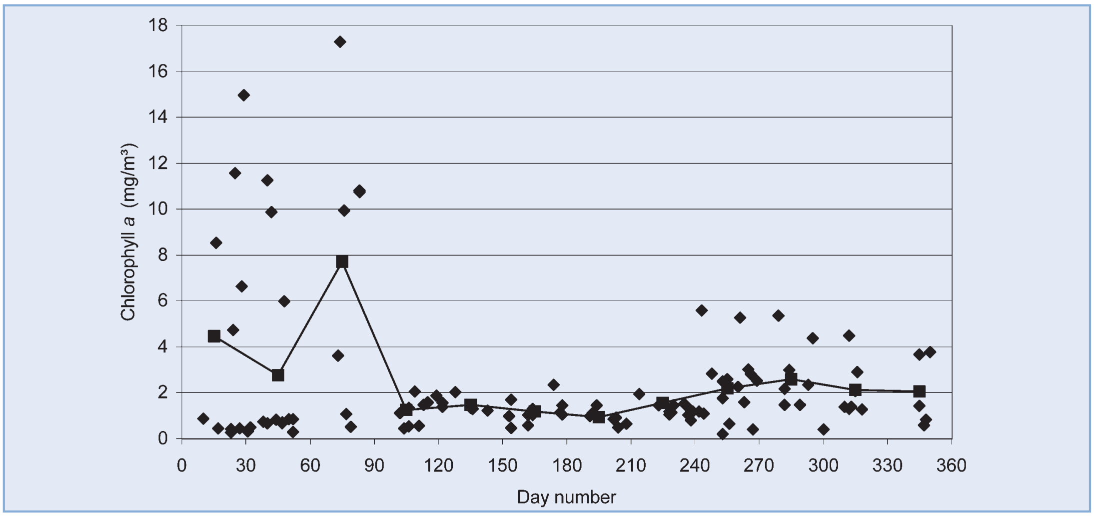


Figure 5.89 Measurements and monthly mean values for 0–10 m depth-integrated chlorophyll a concentrations in the Kattegat (BMP R3) from 1994–98.

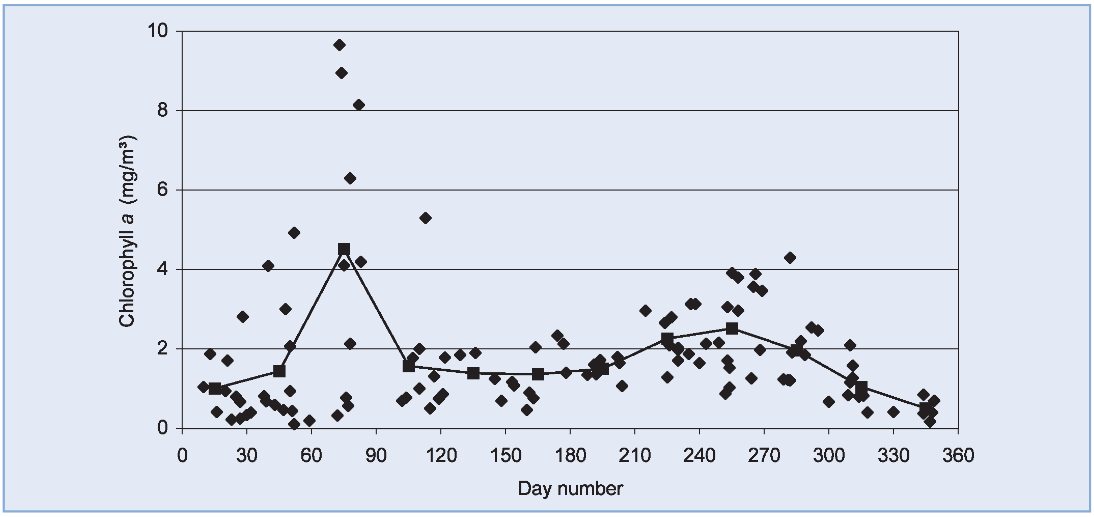


Figure 5.90 Measurements and monthly mean values for 0–10 m depth-integrated chlorophyll a concentrations in the Sound (BMP Q2) from 1994–98.

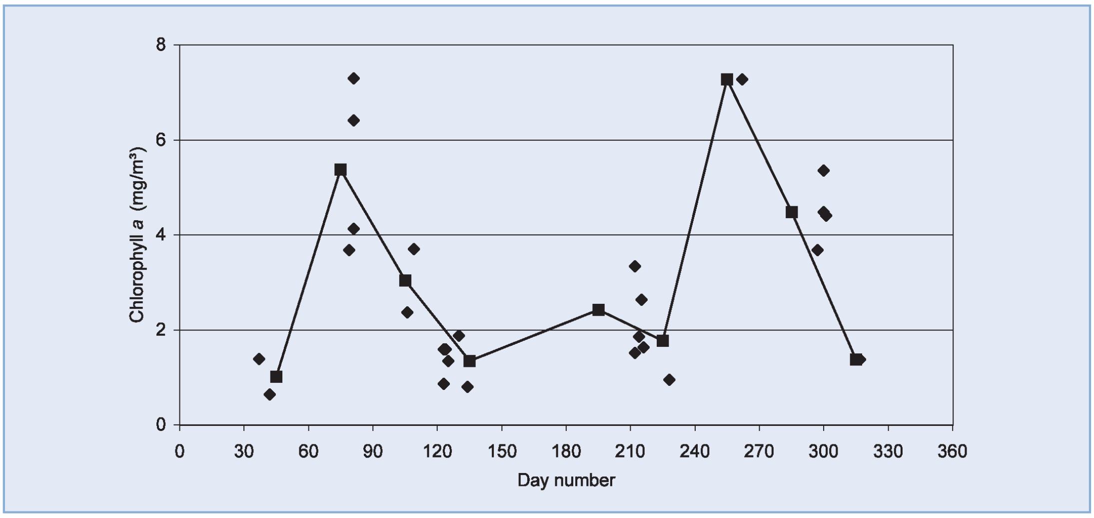


Figure 5.91 Measurements and monthly mean values for 0–10 m depth-integrated chlorophyll a concentrations in Mecklenburg Bight (BMP M2) from 1994–98.

centrations, rarely exceeding 2 mg/m³, predominated. They increased slowly during summer to form an autumn bloom in September–October. A summer bloom, as found in the Baltic Proper, obviously does not occur.

Long-term variation. The high intra-annual and inter-annual variability in chlorophyll *a* concentrations makes trend analysis less significant. Simple linear regression of all depth-integrated (0–10 m) chlorophyll *a* concentrations for the period 1980–98 from the Kattegat (BMP R1+R3+R4), the Sound (BMP Q2) and Mecklenburg Bight (BMP M2) reveals a non-significant tendency towards a decrease at all stations.

A comparison of Table 5.14 with the corresponding table in the Third Periodic Assessment (HELCOM, 1996) reveals that this decrease is mainly due to a decrease in the spring data. Due to a shift in spring bloom timing, however, the latest

Figure 5.92
Time series 1992–98 of chlorophyll a concentrations in Mecklenburg Bight.
A: Spring (March-May).
B: Summer (July-August).
The boxes represent 50% of the observations: The median is shown as a horizontal line. Data from the Finish Alg@line Project.

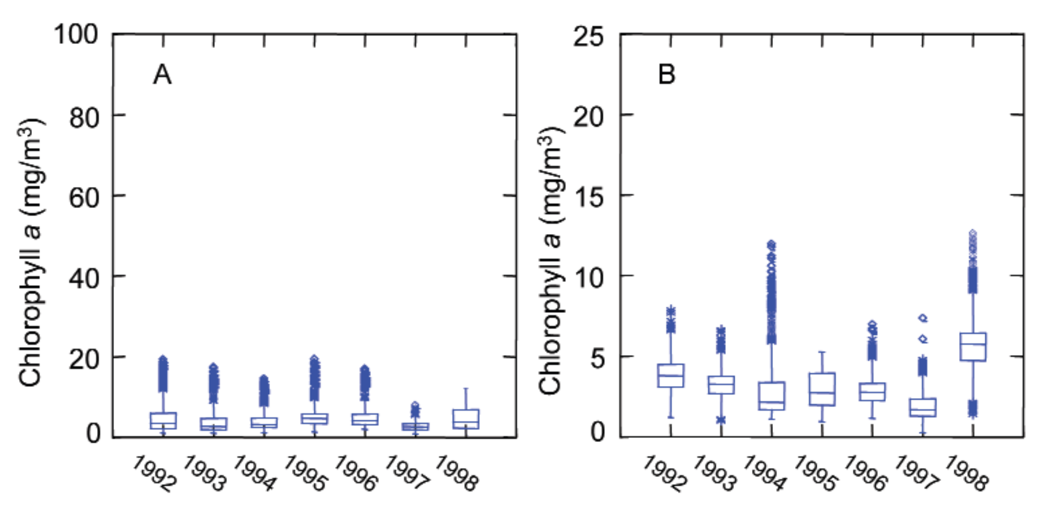


Table 5.14
Mean concentration, standard deviation and number of samples of all chlorophyll a data available from Kattegat/Belt Sea area (0–10 m depth) for spring (Feb–Apr), summer (May–Aug), autumn (Sept–Nov) and winter (Dec–Jan) from 1994 to 1998.

Chl *a*	Mean	SD	No. of samples
Spring	3.51	4.11	149
Summer	1.76	0.82	190
Autumn	2.61	1.93	178
Winter	2.59	3.53	59

spring blooms are included in the winter data in Table 5.14 rather than in the spring data. The tendency towards a decrease is particularly interesting in view of the tendency towards an increase in the Baltic Proper, thus confirming that the two marine waters differ significantly as regards eutrophication phenomena.

The spring (March–May) concentrations in the Lübeck and Mecklenburg Bights 1993–98 are shown in Figure 5.11 together with data from the Baltic Proper and western Gulf of Finland. Time series of chlorophyll a concentrations in spring (March–May) and late summer (July–August) in Mecklenburg Bight for the period 1992–98 reveal particularly low chlorophyll a concentrations in the dry year 1997 (Figure 5.92).

Primary production. This account is based on the conclusions of Ærtebjerg *et al.* (1998), who analyses pelagic variables collected from 1977 to 1997.

The trend analysis of the primary production versus time for the entire period revealed a reduction in spring and summer production in most of the areas under consideration from the northern Kattegat to the Arkona Basin (Ærtebjerg *et al.*, 1998). The reduction is attributable to the lower production capacity of the phytoplankton combined with decreasing concentrations of inorganic nitrogen nutrients and phosphate.

Phytoplankton species composition and biomass

By: L. Edler

The overall progression, as reflected by the development in chlorophyll *a* concentration during the course of the year, is also reflected in the species succession, which is characterized by the spring bloom of diatoms terminating in the development of dinoflagellates, many of which are heterotrophic. Late spring is dominated by small flagellates. Every now and then these flagellates burst into major blooms, e.g. in 1988, when *Chrysochromulina polylepis* reached cell densities up to 100 million cells/l, and in 1998, when the *Chattonella sp*. bloom in the Skagerrak also penetrated into the Kattegat. The summer is a period of low phytoplankton biomass, interrupted by sudden and short peaks of different species, mainly diatoms. During this period autotrophic dinoflagellates start to develop in the deep water below the halocline. The dinoflagellates peak at the beginning of autumn, thereafter to be followed by an autumn diatom bloom.

The long-term (1982–98) development in phytoplankton biomass in spring (February–April) exhibits a significant negative trend in the Kattegat (Figure 5.93). Most obvious is the decline in dinoflagellate and cryptophycean biomass, which fell more than 90% from the peak in 1982–84 to the present assessment period. Diatoms have also declined – by about 80% from the peak in 1984–88. Despite the decline, diatoms still constituted the dominant algal group in spring, accounting for 64% of the biomass during the present assessment period.

The summer period from May to August shows much the same trend as the spring. Thus all groups have declined 70–90% in the present assessment period relative to earlier peak levels. The decline is also seen during the autumn period, but is not as great as during spring and summer. Autumn 1993 is an exception, phytoplankton biomass being considerably greater than in both the preceding and subsequent years, probably due to an extensive bloom of the large diatom *Guinardia flaccida* covering the entire Kattegat.

Comparison of the dominating species in each season and assessment period reveals a relatively stable pattern. The spring periods are dominated by typical cold-water diatoms such as *Detonula confervacea, Porosira glacialis* and *Thalassiosira spp*. A shift towards spring dominance by the warm-water species *G. flaccida* and *Proboscia alata* was observed during the present assessment period, however. The summer periods have been more stable, characterized by small flagellates and other nanoplankton together with the warm-water diatoms *Dactyliosolen fragilissimus, G. flaccida, Leptocylindrus danicus* and *Proboscia alata*. At certain times dinoflagellates, such as *Ceratium spp.* and *Noctiluca scintillans,* have been abundant. The autumn periods have also been relatively stable over the four assessment periods, with typical autumn dinoflagellates (*Ceratium spp.* and *Gymnodinium mikimotoi*) and diatoms (*Cerataulina pelagica, G. flaccida* and *Coscinodiscus spp.*). The last two assessment periods have also seen occasional

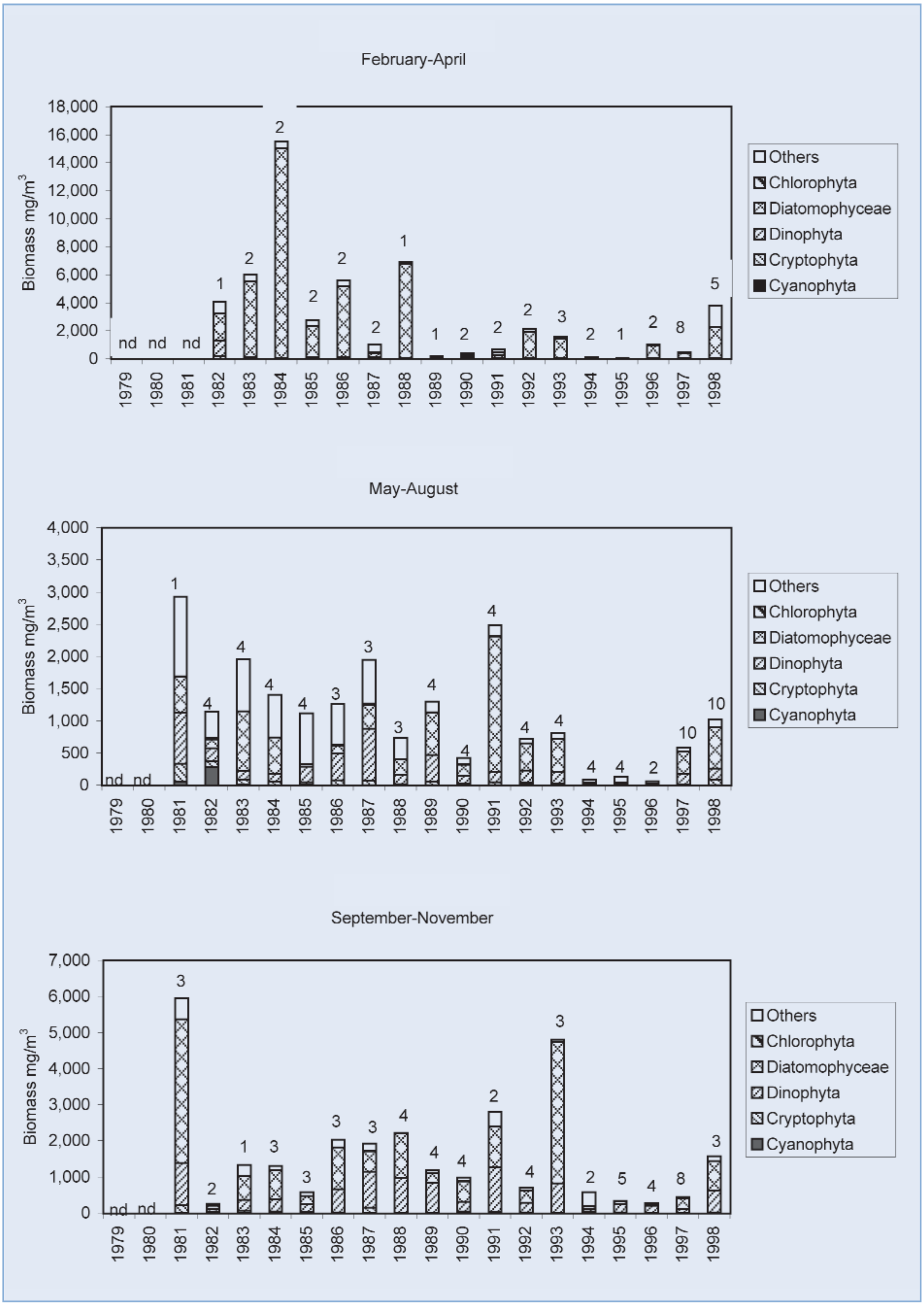


Figure 5.93
Long-term development in seasonal means of phytoplankton biomass (wet weight, mg/m^3) at station BMP R3 in the Kattegat divided into 6 groups of species.

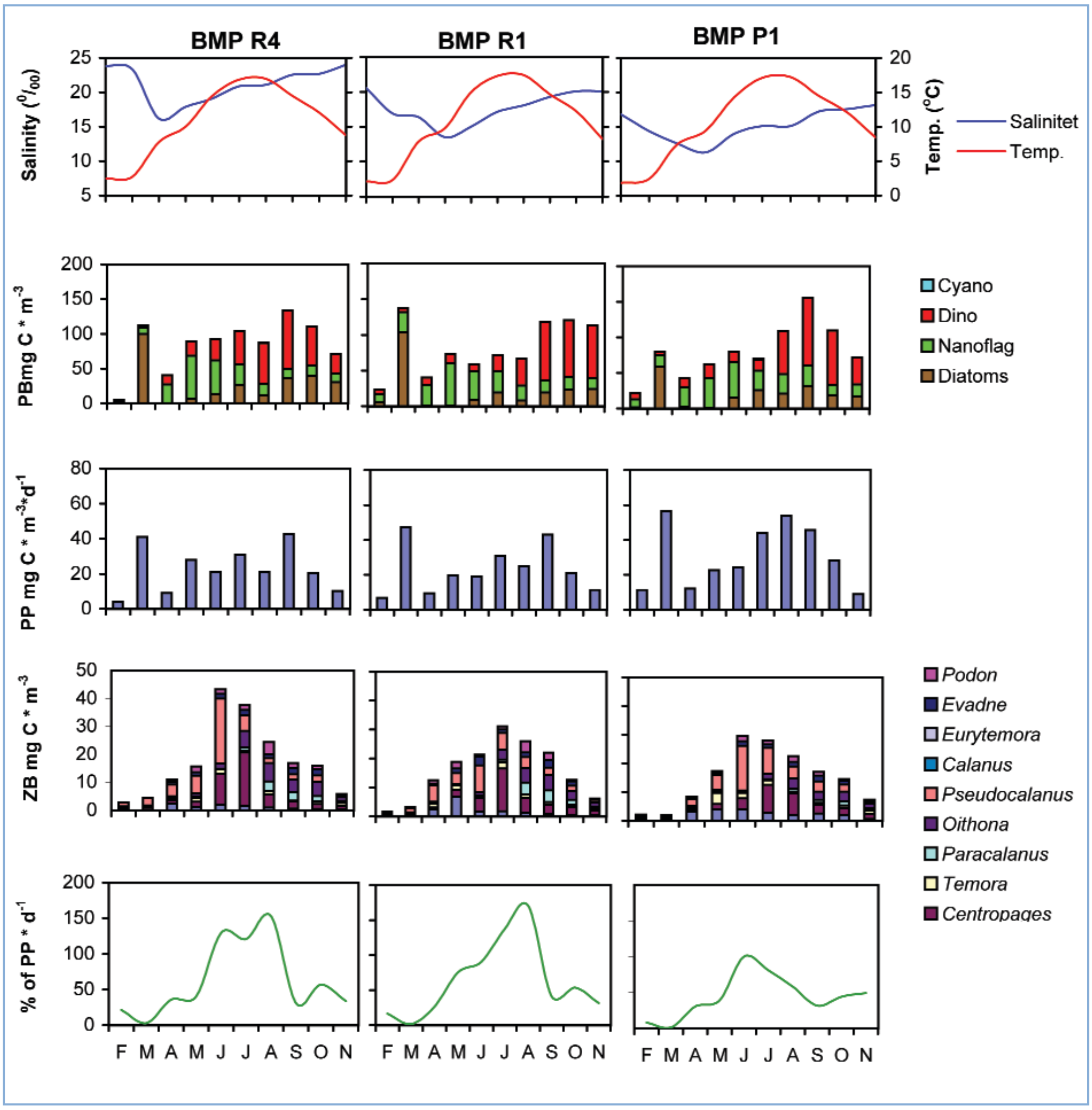


Figure 5.94 Seasonal variation in the average (1983–96) temperature and salinity in the surface layer (0–10 m), phytoplankton biomass (PB), primary production (PP), mesozooplankton biomass (ZB) and grazing potential of mesozooplankton (% of PP) at six stations in the Kattegat-Belt Sea area (from Ærtebjerg et al., 1998).

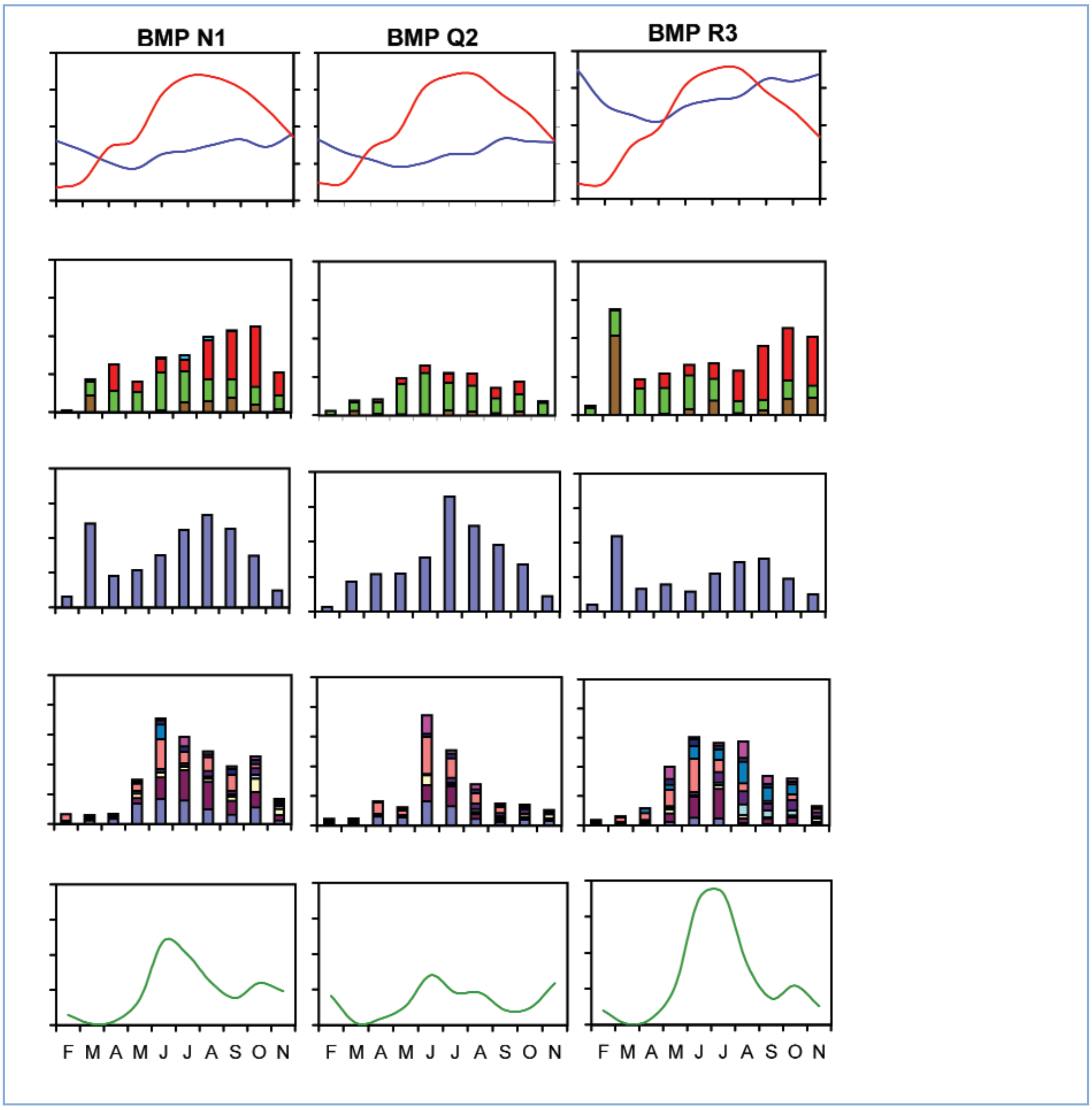
blooms of *Polykrikos schwarzii* and *N. scintillans*.

The most obvious changes in the phytoplankton during the assessment periods are structural in character rather than changes in species abundance. The occurrence of spectacular blooms in late spring is one example. The *Chrysochromulina* bloom in April–June 1988, which resulted in fish kill, has been repeated in some years. In 1992, fish kill associated with a *Chrysochromulina* bloom occurred in the southern part of the Kattegat. In May 1998, two small flagellates bloomed in the Kattegat-Skagerrak area that have not previously been reported there. These two Raphiodiophyceans – *Chattonella sp.* and the unidentified *Flagellate B* – caused fish kill among various wild species of fish. Another structural change that has taken place during the present assessment period is the occurrence of winter blooms. In January 1997, an unusually early diatom bloom developed in the Kattegat. The spring bloom normally develops at the beginning of March, but this year the bloom occurred as early as late January. A winter bloom also occurred in January 1998, but in this case it was mainly autumn species that formed the peak, and the typical spring bloom species did not develop until mid February. The reason for the changed pattern of blooms remains to be elucidated.

Zooplankton

By: T. G. Nielsen

The only source of zooplankton data available for this assessment is the Danish monitoring programme. Only the mesozooplankton biomass has been estimated for the open waters. The biomass samples originate from vertical net hauls made using a 100 µm net hauled from a depth of 25 m to the surface. The figures thus represent the average biomass of the water column. The average annual distribution of mesozooplankton biomass constructed from all available data for the period 1983–96 are summarized in Figure 5.94. The same unimodal population development of the mesozooplankton biomass was observed at all stations. In general, the winter biomass was low (2–5 mg C/m^3) increasing to a seasonal peak of about 30–40 mg C/m^3 in June. Inter-regional differences were minor. Two areas can be identified on the basis of the summer biomass measurements, however: 1) The eastern part of the Kattegat and the Sound, and 2) The western part of the Kattegat



and the Great Belt. Summer biomass in the two areas was 30 C/m³ and 40 mg C/m³, respectively (Ærtebjerg *et al.*, 1998).

The mesozooplankton community in the Kattegat-Great Belt area is dominated by small neritic copepods, which is typical for shallow coastal ecosystems (Kiørboe and Nielsen, 1994; Ærtebjerg *et al.*, 1998). Benthic larvae (Meroplankton) also occur periodically, and the zooplankton community in the Kattegat-Belt Sea area is episodically influenced by the import of brackish water species from the Baltic Sea (e.g. *Eurytemora affinis* and *Bosmina spp.*) or of Atlantic species in the bottom water from the North Sea (e.g. *Calanus finmarchicus* and *Metridia spp.*).

With respect to species composition, no significant differences were observed between the six stations considered. The mesozooplankton community was dominated by *Acartia spp.*, *Centrophages spp.*, *Temora longiremis, Oithona spp.*, *Pseudocalanus spp.*, *Paracalanus spp.* and *Eurytemora affinis* as well as by the cladocerans *Podon* and *Evadne*. The timing of their respective annual peaks varies. The *Acartia* and *Temora* peak in early summer (May–June) followed by *Centrophages* and *Pseudocalanus* in June–July and *Oithona* and *Paracalanus* in August–September. In contrast to the copepods, the cladocerans have a bimodal seasonal distribution with peaks in both spring and autumn.

During the period 1978–97, primary production in the Kattegat-Belt Sea area decreased significantly in the spring and summer (Ærtebjerg *et al.*, 1998). Investigations in the Kattegat have shown that the egg production rate of the copepod community is food-limited during the summer period (Kiørboe and Nielsen, 1994). The copepod grazing potential is very high during the summer months (Figure 5.94). Considering that the copepods primarily exploit cells larger than 11 µm, a reduction in the primary production will influence the mesozooplankton biomass. This assumption is supported by an analysis of copepod biomass data (Ærtebjerg *et al.*, 1998), which revealed a significant reduction in mesozooplankton biomass at two out of three Kattegat stations.

Functioning of the pelagic ecosystem

By: T. G. Nielsen

In general, the diversity of the plankton community decreases with salinity from the North Sea to

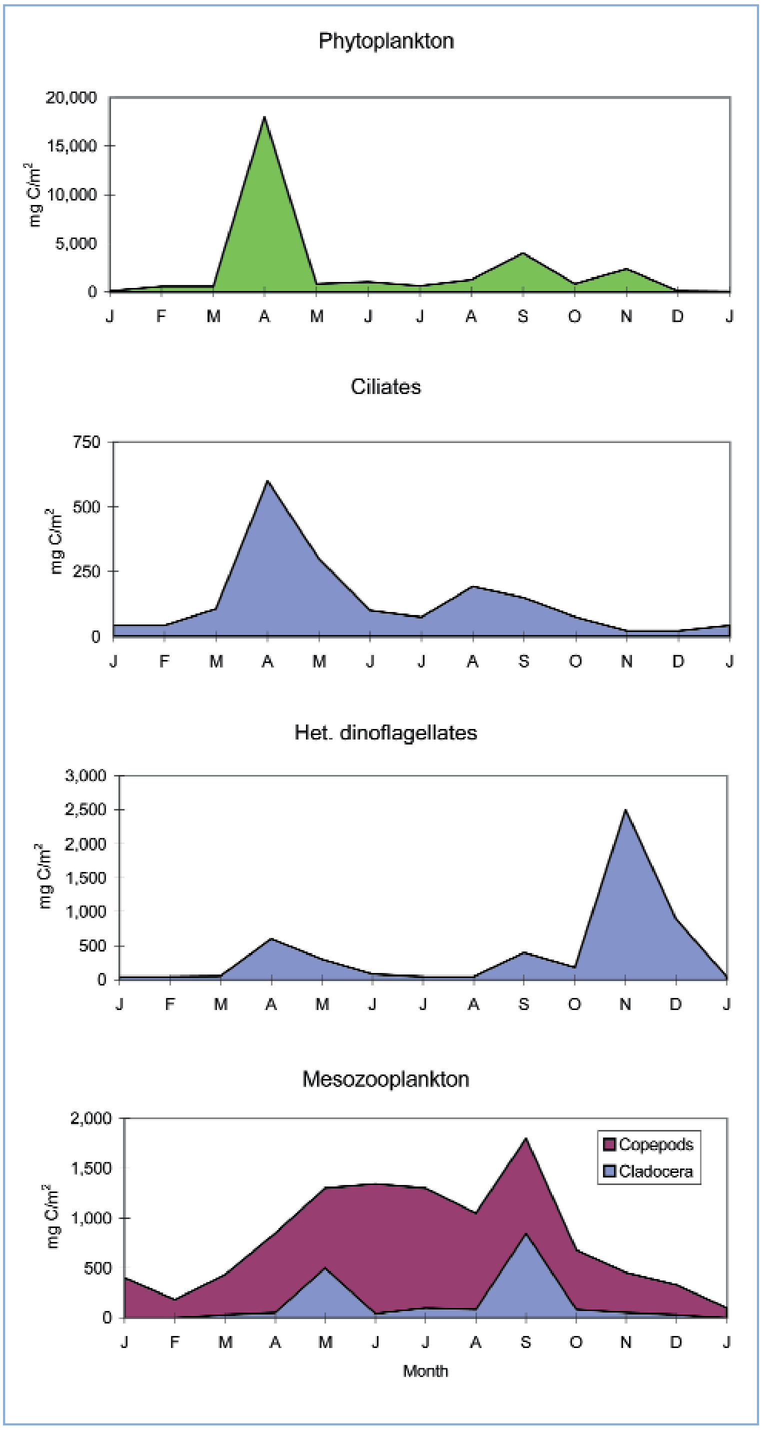


Figure 5.95 Seasonal variation in biomass (mg C/m^2) of phytoplankton, ciliates, heterotrophic dinoflagellates and mesozooplankton in the southern part of Kattegat (modified from Nielsen and Hansen, 1999).

the Baltic Sea. However, diversity is high in the Kattegat and the Belt Sea due to the influence of both brackish and Atlantic species as a result of the inflow of water from the Baltic Proper and the Skagerrak-North Sea, respectively.

During winter, primary production is light-limited, and the biomass of phytoplankton is low (Richardson and Christoffersen, 1991). The increasing light during early spring triggers the development of a diatom bloom, the exact timing of which is very dependent on the light and wind regime during the year in question. In the more shallow areas, where light penetrates to the bottom or where the water column is stratified throughout the year, the spring bloom starts earlier than in deeper areas where a thermocline first has to be established (e.g. in the North Sea and the Baltic Proper). When the spring bloom has become established, the diatoms quickly deplete the surface layers of nitrate and phosphate, thereafter sedimenting out within a few weeks (Olesen and Lundsgaard, 1995).

In the Kattegat and the Great Belt, the spring blooms can appear from January to April, but normally develop in March (Ærtebjerg *et al.*, 1998). After the spring bloom, the phytoplankton biomass in the nutrient-depleted surface water is low.

In the summer, the plankton is normally dominated by nanoplankton (i.e. 2–20 µm). During this period, phytoplankton biomass and production are usually highest in connection with the pycnocline (Richardson and Christoffersen, 1991; Nielsen *et al.*, 1990, 1994). Here, the phytoplankton community is supplied with light from above the pycnocline and with nutrients from below.

In the autumn, irradiance decreases and wind stress increases causing erosion of the pycnocline and introducing nutrients into the surface layer. This nutrient input stimulates a bloom of dinoflagellates and/or diatoms during the period from August to October (Thomsen, 1992; Nielsen, 1991). Thereafter primary production gradually decreases to the winter levels as a result of declining solar irradiance.

In the Kattegat-Belt Sea area, the microzooplankton (0.02–0.2 mm) is dominated by ciliates, heterotrophic dinoflagellates, rotifers and the smallest developmental stages of copepods. The most important groups in the open water are the ciliates and the heterotrophic dinoflagellates, which contribute equally to the standing stock of protozooplankton (Figure 5.95). From the size point of view, both groups belong to microplankton. Functionally, though, their biology is very different. Ciliates prey upon nanoplankton while the dinoflagellates prefer prey similar to their own size (Hansen *et al.*, 1994). The importance of the dinoflagellates is often ignored because they are treated as phytoplankton despite being heterotrophic.

In the open waters the occurrence of protozooplankton follows the variation in phytoplankton, with a peak in biomass associated with the spring and autumn blooms (Nielsen and Kiørboe, 1994; Hansen, 1991). During the blooms the feeding of the low copepod community is saturated, and the protozooplankton can build up due to low grazing pressure from the copepods.

Throughout the summer, protozooplankton biomass is low. After the spring bloom, phytoplankton biomass is dominated by cells too small for the copepods to exploit, and the copepods become strongly food-limited. During the summer the ciliates and heterotrophic dinoflagellates are thus an important link between the small primary producers and the larger zooplankton. In the late autumn, protozooplankton biomass often peaks after the copepod population has decreased (Nielsen and Kiørboe, 1994; Hansen, 1991).

The winter population of copepods in the Kattegat and Great Belt area is very low, and most of the population is present as resting eggs. In association with the spring bloom, the resting eggs hatch, and the few over-wintering copepods start to produce eggs. The combination of low temperature during the spring bloom and the many stages in the life cycle of the copepods creates a miss-match between the spring bloom and the annual peak of the copepods. In contrast, the cladocerans have the ability to respond quickly to the spring bloom through parthenogenetic reproduction (Figure 5.95). After a peak in the summer the number of copepods gradually decreases to the winter levels.

Grazing impact of zooplankton

In the spring, when both the temperature and copepod population are low, it is primarily ciliates and heterotrophic dinoflagellates that are responsible for pelagic grazing. In contrast to the mesozooplankton (copepods), which typically have generation times of weeks, the protozooplankton have population growth rates comparable to those of the phytoplankton. This can be observed in the spring when there is a tight coupling between the phytoplankton and the protozooplankton (Figure 5.95). After the spring bloom, when the population of mesozooplankton has been established, the copepods take over the role as the most important pelagic grazer for the rest of the stratified season. During summer, the dominant phytoplankton species are too small to be exploited directly by the copepods, so a significant fraction of the primary production is channelled trough the protozoa before it is available to the copepods. In the autumn, when the copepod population decreases, the relative importance of the microzooplankton increases once again.

On an annual scale the grazing impact of protozooplankton on primary production is as important as the grazing pressure performed by the mesozooplankton (Hansen, 1991; Kiørboe and Nielsen, 1994; Nielsen and Kiørboe, 1994). Thus, if we want to understand nutrient and organic matter cycling in our marine environment, both groups have to be considered.

5.5.4. Benthic conditions and macrofaunal biology

Surface sediments

By: I. Cato, B. Larsen

The Kattegat is characterized by a number of shallow banks generally consisting of uncovered old, folded prequaternary sediments. Many small banks are also almost uncovered, as is the substratum of glacial tills (moraine and late glacial clays) in the southern part of the Belt Sea.

The remainder of the seafloor can be divided into three main areas of bottom sediments attributable to differences in bottom dynamics. Sand and coarse silt, with an organic carbon content of less than 1%, dominate the sediment surface of the shallow (<20–25 m) areas influenced by wave action (Floderus, 1988). These sediments are found within the main part of the western Kattegat and along the Swedish coast from the Sound in the south to about 30 km south of Gothenburg. These sand sediments are often less than one metre thick and form a mobile cover above older sediments.

The Deep Trench (50–150 m deep) that intersects the eastern Kattegat is dominated by recent postglacial clayey sediments with an organic carbon content of about 2–3% dry matter (Cato, 1997; Kuijpers *et al.*, 1992). The trench forms the main area of accumulation of fine-grained sediments dominated by suspension deposition (Bengtsson and Stevens, 1996).

On both sides of the trench, at depths between 25 and 50 m, the bottom sediment consists of postglacial silty clays, the uppermost part of

Table 5.15
Range, mean and median concentrations of total organic carbon, total nitrogen, total phosphorus and total iron in the surface (0–2 cm) fine-grained sediments of the Swedish sector of the Kattegat.

Element	Years of sampling	Unit	No. of samples	Range	Mean ±1 SD	Median	References
Organic C	1970	%	66	0.4–2.5	1.1		Olausson, 1975
	1986–90	%	108	0.28–7.5	2.1 ± 1.2	2.2	Cato, unpubl.
	1987–91	%	32	0.37–3.7	1.8		Kuijpers et al., 1992
Total N	1970	%	16	0.08–0.37	0.23		Olausson, 1975
	1986–90	%	28	0.032–0.32	0.22 ± 0.087	0.24	Cato, unpubl.
	1987–91	%	24	0.15–0.33	0.23		Kuijpers et al., 1992
Total P	1970	g/kg	39	0.37–1.19	0.79		Olausson, 1975
	1986–90	g/kg	143	0.45–3.0	0.92 ± 0.31	0.89	Cato, 1997
Fe	1986–90	g/kg	143	11–56	33 ± 9.2	34	Cato, 1997

Figure 5.96
Long-term trends (1979–98) in total macrozoobenthos abundance at stations in the Kattegat and Belt Sea area. Left: All stations. Right: HELCOM stations only.

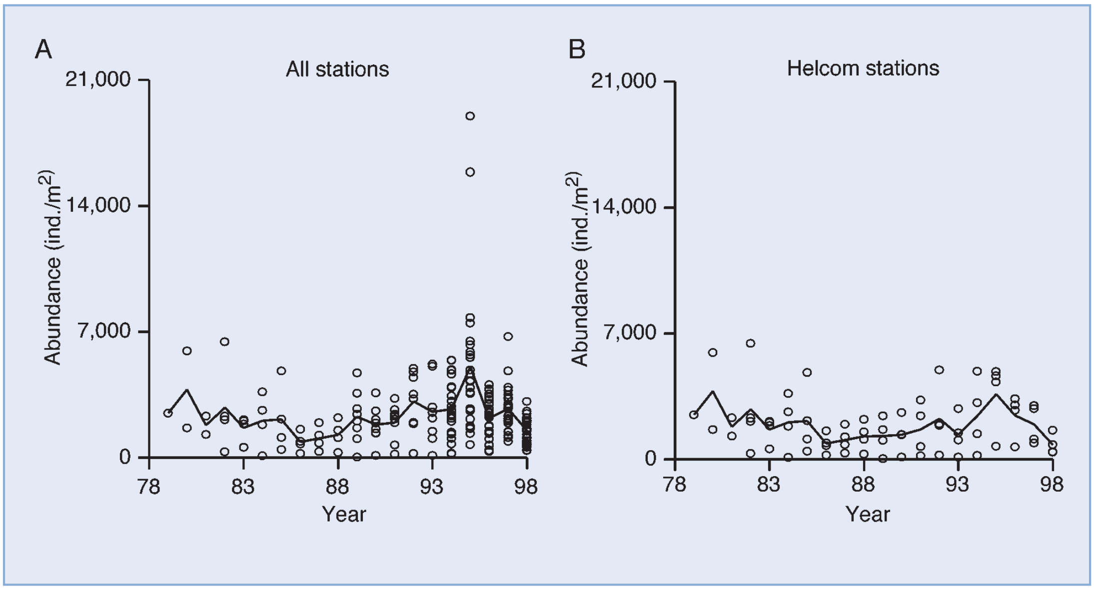
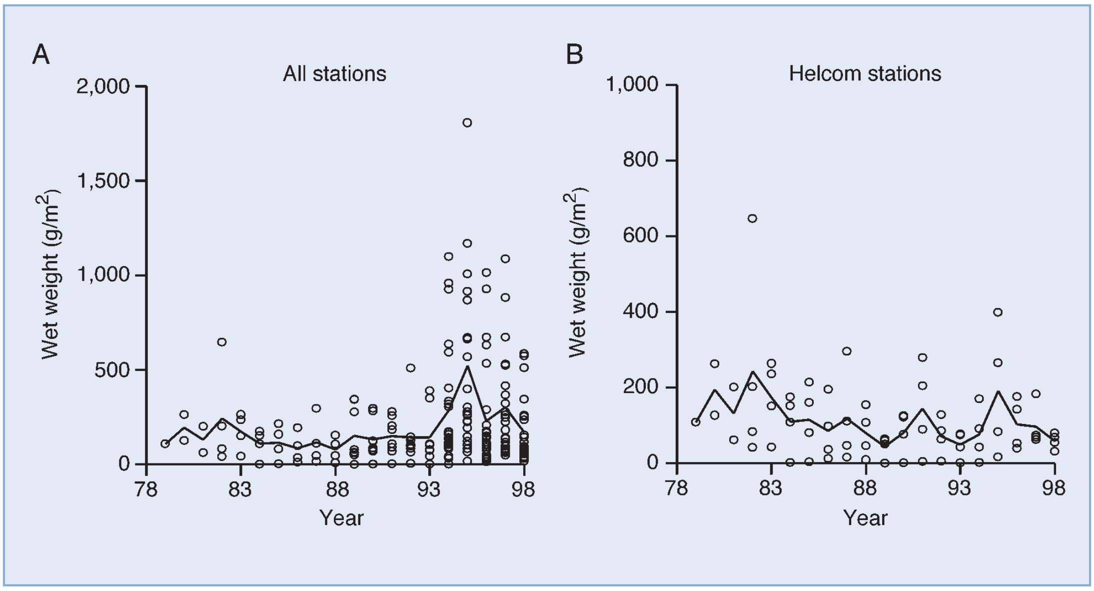


Figure 5.97
Long-term trends (1979–98) in macrozoobenthos biomass at stations in the Kattegat and Belt Sea area. Left: All stations. Right: HELCOM stations only.



which, depending on the depth, is sometimes resuspended and oxygenated. The sediments in these areas thus do not represent time series of continuous sedimentation.

Other important clay and silt sedimentation areas are found along the rugged Swedish coastlines of the northeastern Kattegat in certain sheltered bottom areas (Cato, 1997). In the northern Sound, the Great Belt and especially the Little Belt many small mud deposition basins are also present.

Recent laminated sediments are rare in the Kattegat and have only been observed in the deeper sheltered areas in its northeastern part and occasionally in the Deep Trench. The range, mean and median concentrations of organic carbon, total nitrogen, total phosphorus and total iron in fine-grained surface sediments of the Swedish sector of the Kattegat during different periods are summarized in Table 5.15.

Macrozoobenthos

By: T. Forbes, R. Rosenberg, S. Phil-Baden

The only quantitative macrozoobenthos data available for trend analysis for the assessment period (1994–98) derive from the Danish monitoring programme which encompasses 24 permanent stations, including HELCOM stations BMP P1, BMP Q1 and BMP R3. Long-term time series of macrozoobenthos data (1979–98) are also available for the three HELCOM stations. Important changes in the sampling strategy are described in Ærtebjerg *et al.* (1998) and Markager *et al.* (1999).

Long-term trends in total macrozoobenthos abundance are shown for all stations (HELCOM plus Danish national monitoring stations) in Figure 5.96a and for the three HELCOM stations alone in Figure 5.96b. The trend lines are locally-weighted, robust regression lines (Cleveland, 1979). The overall pattern suggests a distinct peak beginning in the early to mid 1990s. This is seen at the HELCOM stations as well as in the complete Danish data set. Macrozoobenthos abundance is greatest in the early to mid 1990s. From 1994 to 1998 there is a consistent trend towards an overall decrease in abundance throughout the Kattegat and Belt Sea regions. Long-term trends in macrozoobenthos biomass indicate a peak in the mid 1990s in the complete data set, but not at the three HELCOM stations (Figure 5.97a and b), although the variation in biomass between stations masks any clear temporal trends.

There have also been notable changes in macrozoobenthos community structure with regard to some of the major taxonomic groups over the past 20 years (1979–98) as summarized in Figure 5.98. The most noticeable shift in species composition involved a decrease in crustacean abundance and a concomitant rise in mollusc and echinoderm abundance. This structural change occurred in the late 1980s and early 1990s. Crustacean and echinoderm abundance were lowest in 1986 when serious hypoxia affected many areas of the Kattegat. Both echinoderms and especially crustaceans are sensitive to serious hypoxia. The echinoderms recovered in 1987, a year with only mild hypoxia. Crustacean abundance has remained relatively low since 1986, but echinoderms and molluscs have increased steadily, peaking in 1992 and 1995, respectively.

Spatially detailed, gridded sampling in the northern and southern Kattegat reveals a clear distinction in community structure between the two geographical groups. At the southern station, inter-region community composition was relatively homogenous, whereas multivariate statistical analyses of the northern station revealed that the community structure changed along a depth gradient (Markager *et al.*, 1999).

Benthic-pelagic coupling

By: R. Rosenberg

Information on the significance of benthic-pelagic coupling in the Kattegat-Belt Sea area as a whole is not available, although several studies have been made at different localities over the past 20 years. As this is the first time benthic-pelagic coupling is included in a periodic assessment, the results of a number of these studies are referred to below irrespective of whether they derive from before the assessment period. The findings indicate that benthic-pelagic coupling is important, especially in the relatively shallow waters of the Kattegat-Belt Sea area and associated estuaries.

Fronts between water masses of different salinity and temperature are often characterized by enhanced production of plankton. In the fronts of the northern Kattegat, primary production can be up to 25-fold greater than in adjacent waters (Richardson, 1985). A survey of the benthos there in the early 1990s indicated strong benthic-pelagic coupling with a significant correlation between polychaete/echinoderm biomass and chlorophyll a/phaeopigment concentrations (Josefson and Conley, 1997). Similar indications of strong benthic-pelagic coupling were also found on the western slopes of the Deep Trench. The very high benthos abundance and biomass in this area were suggested to be attributable to high food availability due to advection into the area in the near-bottom water (Rosenberg, 1995).

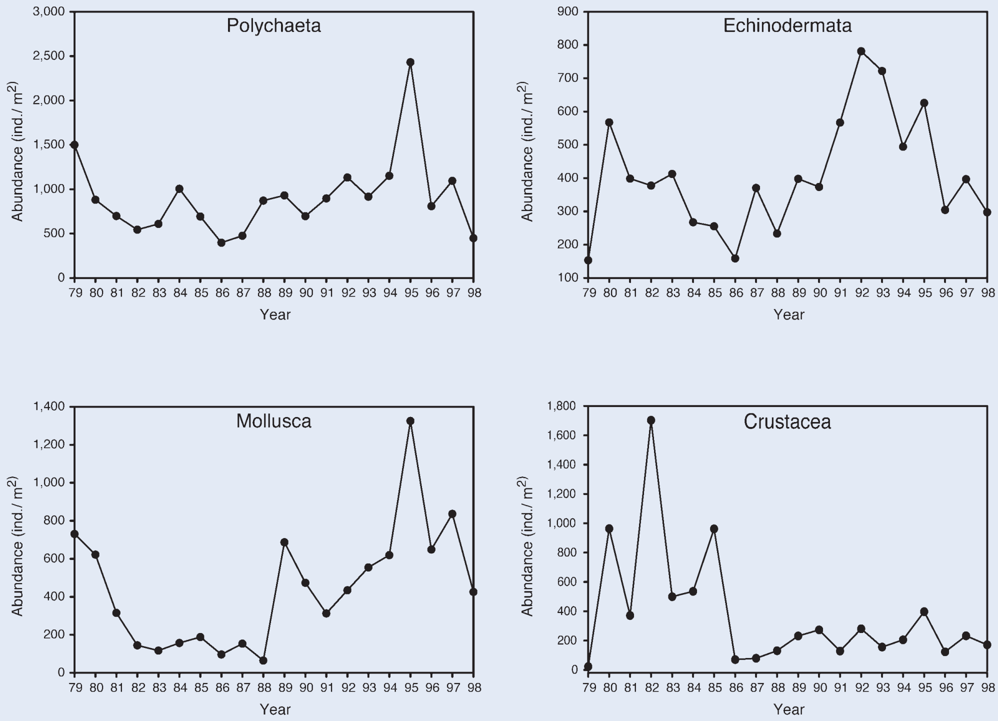


Figure 5.98
Long-term temporal changes (1979–98) in abundance of four key macrozoobenthos taxonomic groups in the Kattegat and Belt Sea area.

Epifauna and flora at the reef Schulz's Grund in the transition area between Great Belt and Kattegat. Photo: Karsten Dahl, National Environmental Research Institute, Denmark

The results from these offshore studies, where high inputs of organic matter support very high benthic faunal abundance and biomass, strongly indicate that benthic sublittoral communities are generally food-limited.

A detailed one-year study in the Sound at a depth of 30 m in 1983/84 showed that phytoplankton sedimenting out after the spring bloom provided sufficient energy to support about 2/3 of the annual benthic faunal growth increment (Christiansen and Kanneworff, 1985). In a recent study in the straits between Malmö and Copenhagen, a 20 km long bed of filter-feeding blue mussels (*Mytilus edulis*) was found to reduce the phytoplankton biomass passing through this bed by 74%. Beyond the mussel bed the phytoplankton biomass increased again, but was shifted towards smaller cell sizes (Norén *et al.*, 1999).

The Laholm Bay in the southeastern Kattegat has been affected by eutrophication over the past 25 years. The water at depths between 0 and 10 m has been estimated to be filtered twice weekly by the dominant filter-feeding bivalves *Mya arenaria* and *Cerastoderma edule* (Loo and Rosenberg, 1989). In that study, however, these bivalves filtered only about half of their capacity, and the majority of the phytoplankton was exported from the bay. The bivalves produced about 235 metric tonnes of nitrogen as biodeposits annually, much of which was probably exported to offshore accumulation bottoms.

Pearson and Rosenberg (1992) developed an energy flow model of the southeastern Kattegat ecosystem based on investigations over the years 1985–89. The model clearly demonstrates that most of the energy is channelled through benthic suspension feeders. The area is particularly susceptible to seasonal hypoxia in the bottom water. The authors state that prolonged severe hypoxia in that area will reduce the energy flow through the benthic compartment and result in accumulation of sedimentary carbon prior to the initiation of anaerobic pathways, mainly through prokaryotic components.

Loo and Rosenberg (1996) found that annual production of the bivalve *Mytilus edulis* cultured on vertical lines was 6-fold greater than primary production while that of the bivalves *Mya arenaria* and *Cerastoderma edulis* was equal to primary production. With the deeper-living brittle star *Amphiura filiformis,* the secondary production was only 0.003 times the primary production. This study emphasized the ecological importance of horizontal advective processes for energy transfer from the pelagic to the benthic system.

In Kerteminde Fjord, a shallow, enclosed fjord on the Danish island of Funen, Petersen and Riisgård (1992) found that the ascidian *Ciona intestinalis* had the potential to filter 10–100% of the water volume daily. With high abundance of *C. intestinalis* in late summer-early autumn, this species has an important impact on the phytoplankton biomass. The polychaete *Nereis diversicolor* was also found to filter-feed in Kerteminde Fjord (Riisgård *et al.*, 1996), and it was suggested that the species might play a significant role in reducing phytoplankton in the near-bottom water.

Several of the studies mentioned above clearly demonstrate the ecological importance of benthic-pelagic coupling. The sedimentation and advective transport of phytoplankton are important food sources for benthic animals. Moreover, benthic suspension feeders can efficiently reduce phytoplankton and other seston in the water column. In shallow waters this will increase the transparency of the water and counteract eutrophication. Cultured bivalves are a special case, with an extraordinary potential to clear phytoplankton from the water.

Benthic vegetation

By: J. Karlsson, K. Dahl

Data for the assessment period is available from the Danish national monitoring programme, which includes benthic vegetation and covers both coastal and open sea areas. The Swedish national monitoring programme does not include phytobenthos in the Kattegat-Belt Sea area, although regional monitoring programmes including benthic vegetation have been run by local authorities.

Macroalgae – Species composition. Species number decreases from about 325 in the northern Kattegat to approximately 150 at the entrance of the Baltic Proper (Nielsen *et al.*, 1995), partially as a result of decreasing salinity and light penetration. The major part of the reduction in species number takes place over a relative short geographic distance in the Sound and Belt Sea area (Nielsen *et al.*, 1995; Dahl *et al.*, 2001).

No significant changes in species number have been found during the assessment period in the Kattegat and Belt Sea area. The local and temporal variation is considerable, however. Thus, a decrease in the number of annual species was reported in 1994 from Ærø, the Great Belt and adjacent waters. In the Little Belt, the number of both annual and perennial species decreased during the same year, and did not recover during the remainder of the assessment period (Ærtebjerg *et al.*, 1998).

No significant temporal variation was found for species richness at the offshore localities monitored along the northern Kattegat-western Baltic Proper gradient (Dahl *et al.*, 2001).

Macroalgae – Depth distribution and vegetation cover. The depth distribution of macroalgae normally increases with increasing distance to the coast because of decreasing water turbidity. At most coastal and open water localities investigated in the Danish monitoring programme, the lack of hard-bottom substrate *per se* limited the maximum depth distribution of benthic algal communities.

The maximum depth hitherto recorded for erect macroalgae (*Coccotylus truncata, Delesseria sanguinea, Bonnemaisonia hamifera* (tetrasporophyte)) at localities in the offshore central part of the Kattegat was 29 m at Lille Middelgrund (Karlsson, 1998), which is comparable to conditions in the open Skagerrak (Karlsson, 1995). Somewhat shallower levels, partially explained by differences in the hard-bottom depth distribution, have been reported from Kim's Top and Store Middelgrund (Nielsen and Dahl, 1992; K. Dahl, personal observation), and from the Swedish Kattegat coast, where erect macroalgae have been recorded down to 24–25 m depth (Gustafsson, 1999; Karlsson *et al.*, 2000). In the open parts of the Belt Sea, attached erect algae have been found down to a depth of 21 m (Ærtebjerg *et al.*, 1998).

In offshore parts of the Kattegat and Belt Sea, the upright vegetation covers about 95–100% of the hard-bottom substrate in a complex multilayered pattern down to depths of approximately 15 m (Ærtebjerg *et al.*, 1998). Below 15 m the erect vegetation is generally single-layered, and the cover gradually becomes thinner, ending up solely as red and brown crusts.

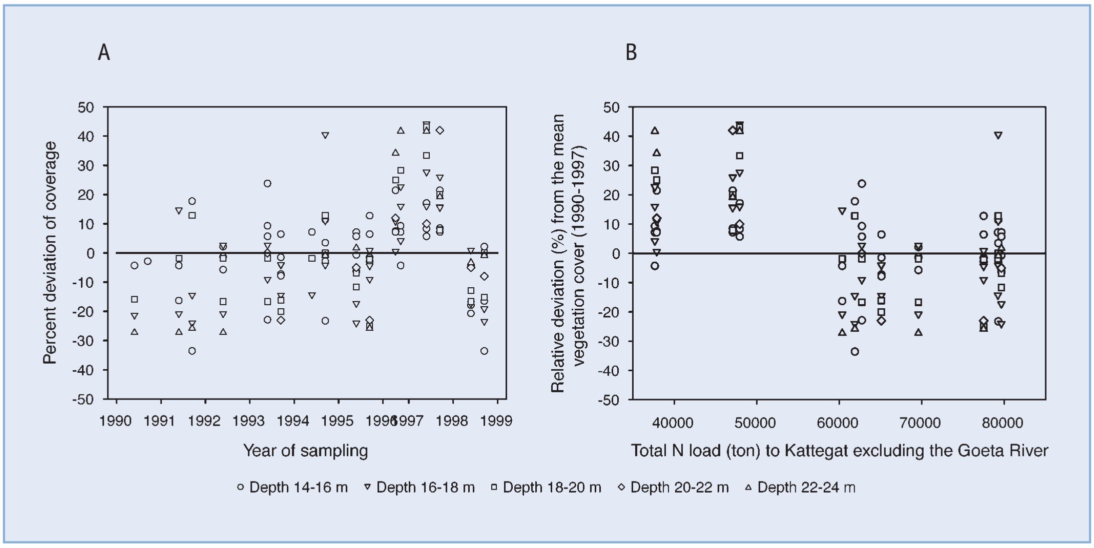
An overall significant increase in vegetation cover was observed at localities in the open northern and central parts of the Kattegat in 1996–97 relative to the average for the period 1990–98. The increase in total area covered was not correlated with an increase in species number (Ærtebjerg *et al.*, 1998). In June 1998, on the other hand, vegetation cover decreased significantly (Markager *et al.*, 1999).

Vegetation cover did not correlate with inter-annual variation in irradiation measured as hours of sunshine over land. The observed increase in vegetation cover (Figure 5.99) may be attributable to the low nutrient loading of the Kattegat in the two dry years 1996 and 1997 as this may have reduced pelagic plankton biomass and hence increased light penetration to the bottom (Markager *et al.*, 1999).

In the open southern Kattegat and in the Belt Sea, no inter-annual changes were observed, partly because of the intense grazing by sea urchins (*Strongylocentrotus droebachiensis*) (Markager *et al.*, 1999).

In 1994, the areal distribution of perennial red and brown algae decreased in many Danish fjords

Figure 5.99
Percentage deviation in vegetation coverage compared to the average coverage at each sampling station and sampling month. The deviation is given for five depth intervals. A) Data for the period 1990–1998 and B) Data for the period 1991–1997 shown plotted against total nitrogen load to the Kattegat in the previous 12 months, exclusive input from Goeta river. Swedish data from 1997 are extrapolated based on the relationship between the nitrogen load from Denmark to the Kattegat, including the fjords, and the Swedish load to the Kattegat, excluding via the Goeta river, for the period 1991–1996 (Relationship S/DK = 1/1.61. SD = 0.39).



and coastal waters, particularly in the island sea south of Funen and in the Little Belt (Dahl *et al.*, 1995; Kaas *et al.*, 1996). Like in the open Kattegat, an increase in the depth distribution of benthic macroalgae was observed during 1996–97 in some coastal regions in the Danish Belt Sea (Ærtebjerg *et al.*, 1998). The temporal variations seen were small, though, compared with the spatial variation between localities (Ærtebjerg *et al.*, 1998).

The common eelgrass (Zostera marina). The general distribution of the common eelgrass has decreased significantly since 1900 (Rasmussen, 1977). The main reduction took place in the 1930s, when epidemic eelgrass disease harried Europe. Since the 1950s, eutrophication has further reduced the distribution of eelgrass in Danish waters. In addition, the mean maximum depth distribution of eelgrass has decreased by about 2.5–3 m from 1908 to 1991–93 (Petersen *et al.*, 1999).

Since 1989 neither the maximum depth distribution nor the lower limit of continuos eelgrass stands have changed significantly (Figure 5.100) (Ærtebjerg *et al.*, 1998; Markager *et al.*, 1999). Temporal variation during the period 1989–98 includes a significant increase in mean maximum depth of 0.5 m in 1996 and 1 m in 1997 relative to a mean maximum depth of 3.5 m in 1994. This was most pronounced at the mouths of the estuaries and in the open coastal areas, corresponding to the positive development observed in the macroalgae during the same period (Ærtebjerg *et al.*, 1998; Markager *et al.*, 1999). In 1998 this weak trend was succeeded by a decrease in maximum depth to the 1995 level (Markager *et al.*, 1999).

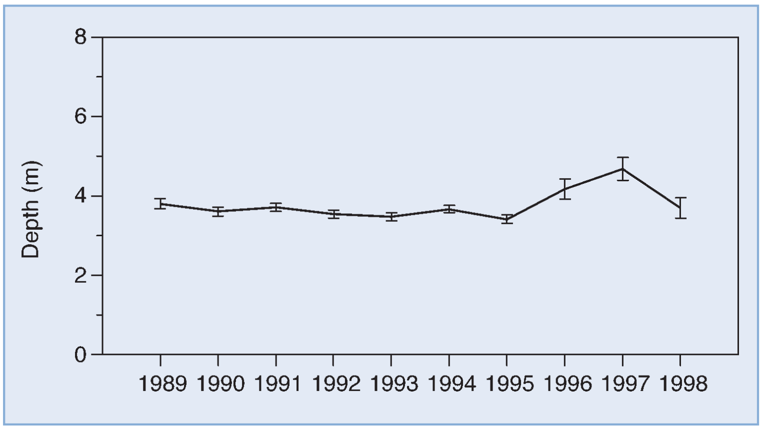
In the period 1989–98, the eelgrass vegetation underwent thinning in the Kattegat-Belt Sea area as a whole. Significantly thinner vegetation cover was observed at depth intervals between 0–1 m and 4–6 m, i.e. at the upper and lower ends of the *Zostera* depth distribution range (Markager *et al.*, 1999).

Macrophytobenthos and eutrophication. Eutrophication-related macroalgae vary both temporally and spatially in the Kattegat-Belt Sea area, but may include *Erythrotrichia* sp., *Ectocarpus* sp., *Pilayella littoralis, Chaetomorpha* sp., *Cladophora sp., Enteromorpha* sp., *Percursaria percursa, Rhizoclonium* sp., *Ulva* sp. and *Ulvaria* sp.

During the assessment period no significant reduction was detected in the area covered by eutrophilic macroalgae in Danish waters, the development being divergent at different depth intervals (Markager *et al.*, 1999). During the period 1990–98 there was a significant reduction at depths between 1 and 4 m (Markager *et al.*, 1999). However, considerable temporal, regional and local variation has been reported (Kaas *et al.*, 1996; Ærtebjerg *et al.*, 1998; Markager *et al.*, 1999). Ærtebjerg *et al.* (1998) also reported a decrease in the number of eutrophilic species in the Great Belt and adjacent waters during 1997.

During the period 1994–96, floating mats of filamentous algae occurred during summer in shallow bays (<1 m) of the northeastern part of the Kattegat, covering between 25–50% of the area avail-

Figure 5.100
Mean maximum depth distribution of the common eelgrass (Zostera marina) in Danish waters over the period 1989–98. N=1,743, n varies between years. Bar= SE. From Markager et al., (1999).



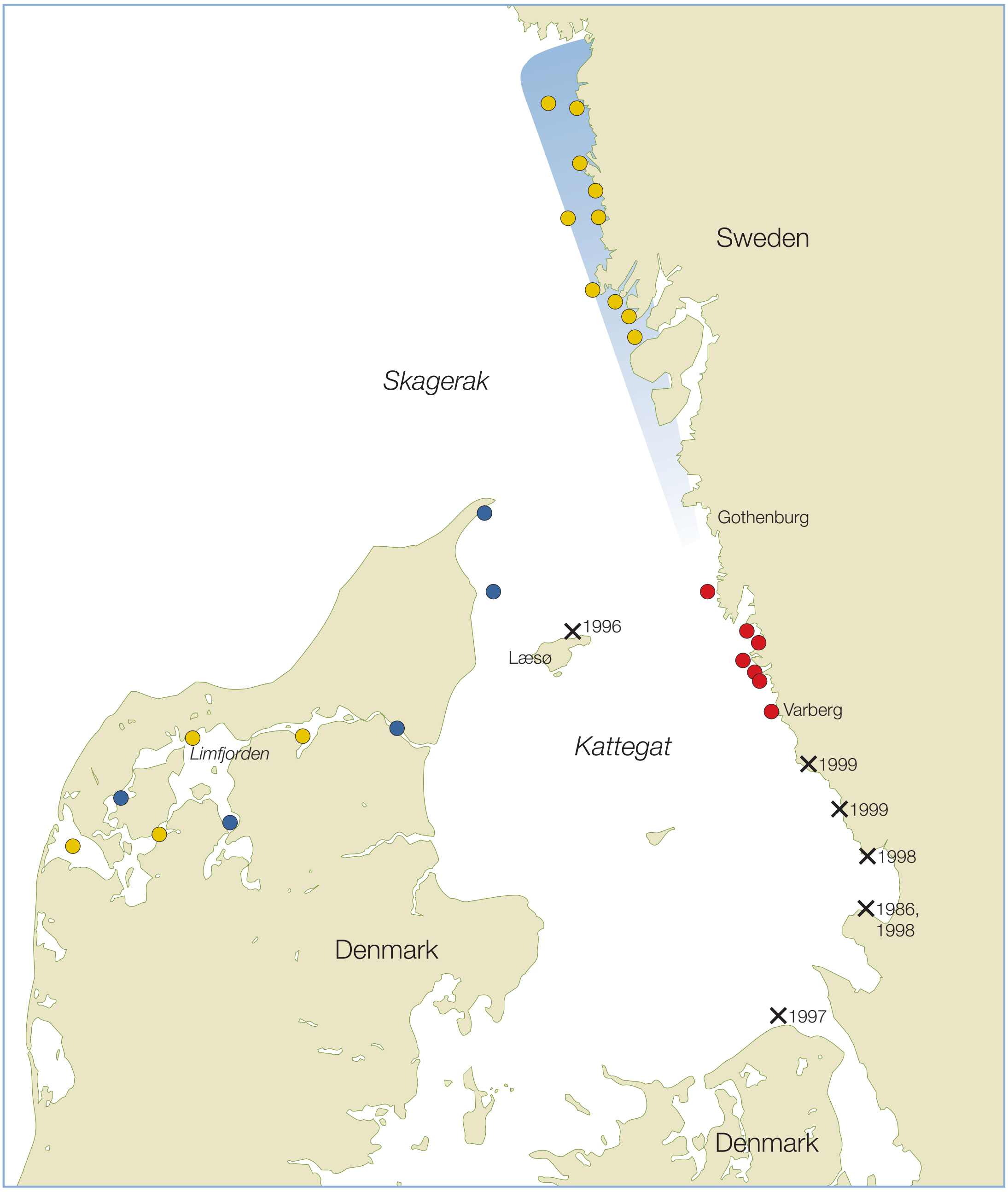


Figure 5.101 Colonization progress of the brown alga Sargassum muticum. Yellow dot = localities pre-1990, Blue dot/field = progress between 1990–94, Red dot = localities after 1995, X = records of drifting plants. Compiled from Anon. (1992), Christensen (1984), Karlsson et al. (1992), Karlsson and Loo (1999), K. Dahl (personal observation) and I. Wallentinus (personal communication).

able but with no significant inter-annual variation (Pihl *et al.*, 1997). In 1998, however, the area covered decreased dramatically and biomass increased – contradictory findings that probably reflect methodological constraints in periods with strong winds (Pihl and Svensson, 1998). The bulk of the floating mats consisted of *Cladophora sp.* and *Enteromorpha sp.* (Pihl *et al.*, 1997; Pihl and Svensson, 1998). During the 1994–98 period, filamentous brown algae, mainly *Ectocarpus* sp., have increased during August–September in sheltered to moderately exposed coastal areas at depths in the range 1–6 m (Karlsson *et al.*, 2000; J. Karlsson, personal observation).

Anoxia and the occurrence of the sulphur bacteria *Beggiatoa sp.* were widespread in shallow parts along the northern part of the Swedish Kattegat coast in the summers of 1994–97 (Karlsson, 1995; Karlsson *et al.*, 2000; Loo *et al.*, 1996).

***Exotic species and/or new additions to the flora*.** The brown alga *Sargassum muticum*, first recorded as attached in Denmark in 1984 and in Sweden in 1987, has extended its distribution to both sides of the Kattegat and in particular along the Swedish coast (Figure 5.101) (Karlsson and Loo, 1999). Considerable inter-annual variation has been recorded in population and plant size (Dahl *et al.*, 1995; Karlsson *et al.*, 1995), with taller populations apparently being favoured by relative high water temperatures during the winter, which leads to earlier onset of annual growth (J. Karlsson, personal observation). In Sweden, *S. muticum* is very common along the Skagerrak coast and in the northern part of the Kattegat, whereas its distribution is rather disjunct and patchy further south in the Kattegat, and its southern distribution limit was close to Varberg in 1998 (Figure 5.101) (Karlsson and Loo, 1999).

During the 1994–98 assessment period, some new species have been added to the flora (*Cryptopleura ramosa, Schmitzia hiscockiana*) (Karlsson, 1998; Karlsson *et al.*, 2000), and a number of species previously considered as rare (e.g. *Halarachnion ligulatum, Lomentaria orcadensis, Phymatolithon calcareum, Porphyra cf. miniata, Tsengia bairdii, Sphacelaria plumula*) have been confirmed as still being present, sometimes occurring in high abundance (Karlsson, 1998; Karlsson *et al.*, 2000; K. Dahl, personal observation; R. Nielsen, *personal communication*).

Contaminants

6

6.0. Scope and contents

Contaminants are substances that either do not occur naturally in the environment or occur at concentrations exceeding natural levels. Those that are harmful to the ecosystem are referred to as pollutants or hazardous substances. Contaminants have previously been shown to have had a number of detrimental effects on Baltic biota. The decreasing populations of Baltic fish predators were attracting attention as early as the 1950s and 1960s, observations that were soon correlated with the high concentrations of contaminants in the Baltic marine environment.

This chapter assesses the state of the Baltic Marine Area with regard to hazardous substances in seawater (Chapter 6.1), sediment (Chapter 6.2) and biota (Chapter 6.3), as well as with regard to the effects of contaminants on "top-predators" (Chapter 6.4). In accordance with the recommendation of the Helsinki Commission, the contaminants encompassed by the Fourth Periodic Assessment are as follows:

Heavy metals
Arsenic (As), Cadmium (Cd), Copper (Cu), Lead (Pb), Mercury (Hg), Zinc (Zn), Chromium (Cr) and Nickel (Ni).

Organic contaminants
DDT (dichlorodiphenyltrichloroethane and its metabolites and derivatives). PCBs (polychlorinated biphenyls), HCHs (hexachlorocyclohexanes), HCB (hexachlorobenzene), PCDDs (polychlorinated dibenzo-p-dioxins) and PCDFs (polychlorinated dibenzofurans), flame retardants (polybrominated compounds), organotins (tributyl tin (TBT), dibutyl tin (DBT) and monobutyl tin (MBT)), petroleum hydrocarbons and PAHs (polycyclic aromatic hydrocarbons).

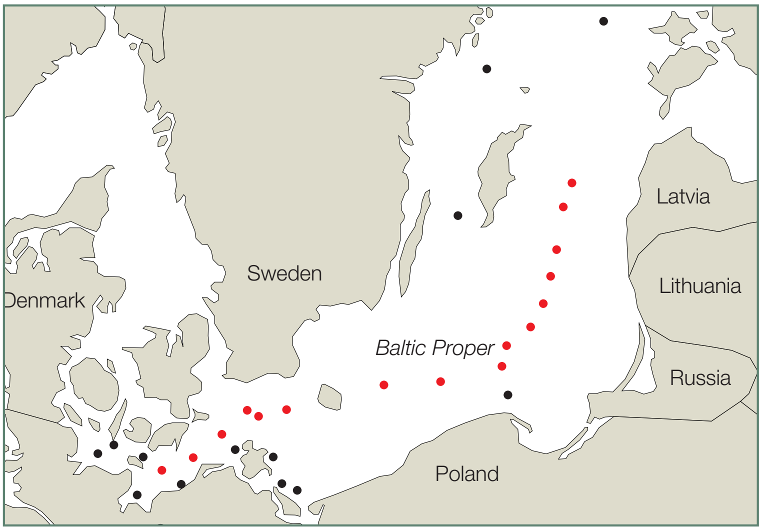


Figure 6.1
Location of the monitoring stations for the measurement of heavy metals in seawater (Red: monitoring stations; Black: additional stations in 1998).

6.1. Surface seawater

By: Bernd Schneider, C. Pohl, Norbert Theobald

6.1.1. Heavy metals in surface seawater

Monitoring data
Surface seawater concentrations of dissolved and particulate Cd, Cu and Zn in Baltic Marine Area surface waters were reported in the Third Periodic Assessment. A trend towards decreasing concentrations over the period 1980–93 was identified for Cd and Cu. In the case of Cd, this has since been confirmed by Kremling and Streu (2000). There was no evidence of any long-term change in the Zn concentration but the findings were possibly biased by methodological artefacts.

In the present assessment, we present data for the period 1994–98. For the first time, moreover, the assessment also encompasses Pb and Hg. The findings are discussed in the context of the previous findings and conclusions. Despite the fact that contaminant concentrations in seawater have been accorded the status of a "main parameter" in the revised HELCOM COMBINE monitoring programme, data are only available for a single Baltic State. Moreover, the data are restricted to the area between the Mecklenburg Bight and the Gotland Sea.

Because many heavy metals (e.g. Cd) exhibit nutrient-like biogeochemistry, their concentrations follow a seasonal cycle with maximum concentrations during the winter (Schneider and Pohl, 1996). Since 1995, therefore, the annual sampling

Figure 6.2
Surface seawater Cd concentrations and trend for the period 1980–1993 (black) and the data for the period 1994–98 (red). The broken red line indicates the mean concentration for 1994–98. Open symbols refer to the Mecklenburg Bight/Arkona Sea, closed symbols to the Bornholm Sea/Gotland Sea.

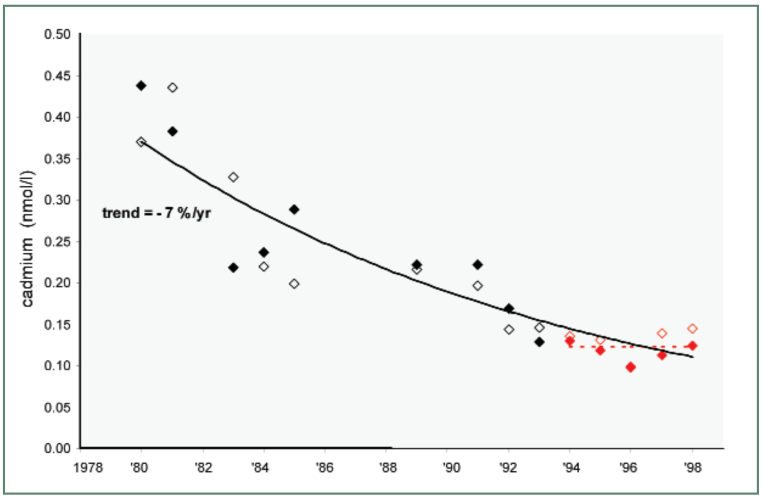
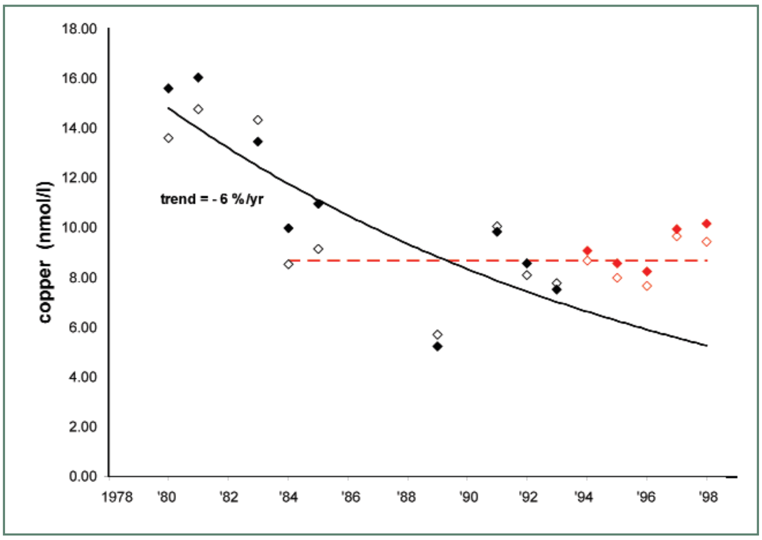


Figure 6.3
Surface seawater Cu concentrations and trend for the period 1980–1993 (black) and the data for 1994–98 (red). The broken red line indicates the mean concentration for 1984–98. Open symbols refer to the Mecklenburg Bight/Arkona Sea, closed symbols to the Bornholm Sea/Gotland Sea.



has been performed in February. The monitoring station net consisted of 16 stations in the Mecklenburg Bight and the Gotland Sea. In some years, they were supplemented by additional stations (Figure 6.1). Depending on the depth of the mixed layer, either one or two water samples were collected at each station. Except for Hg, the concentrations in the dissolved and particulate phases were determined separately.

Data and trends

Cadmium and copper

The trend towards decreasing Cd and Cu concentrations identified for the period 1980–93 seems to have ceased and the concentrations seem to be stabilizing at a low level.

Since no significant regional differences could be detected, the concentration trend curves for the period 1980–1993 are based on data from all the stations in the area between the Mecklenburg Bight and the Gotland Sea (Figure 6.2 and Figure 6.3). The concentrations include the particulate fraction, which averaged about 5%. The data for the period 1994–98 were added to the graphs to allow comparison. Taking into account the interannual variability, the Cd concentration during the period 1994–98 does not differ significantly from the projected 7%/yr decrease identified for the period 1980–93. Taking the period 1994–98 alone, however, the Cd concentrations seem to have stabilized at a level of approx. 0.12 nmol/l (Figure 6.2).

In the case of Cu, the data for the period 1980–93 indicate a 5%/yr decrease in the surface seawater concentration. However, including the data for the period 1994–98 and only analysing for the period 1984–98 reveals that the Cu concentration has remained almost constant at about 9 nmol/l since 1984 (Figure 6.3).

To relate long-term concentration trends to input changes is extremely difficult. The riverine input data are still highly uncertain because the data record is incomplete and no data are available prior to 1990. Atmospheric deposition estimates based on model calculations are susceptible to uncertainties associated with the emission inventories, while measurement-based estimates of deposition are impaired by the low number of stations with long time series of deposition measurements and difficulties in extrapolating coastal data to the entire Baltic Marine Area. Nevertheless, the available Cd deposition estimates for the period 1986–89 and 1997 of 77 tonnes/yr (Schneider, 1993) and 33 tonnes/yr (Schneider, 2000), respectively, are consistent with the observed decrease in the surface seawater Cd concentration in the Baltic Marine Area.

Zinc, lead and mercury

This is the first assessment for which reliable concentration data are available for Zn, Pb and Hg. The time series are as yet too short to enable trend analysis, however.

The mean total Zn concentrations, which include a 5–10% particle-bound Zn fraction, are presented in Figure 6.4. The mean concentration for the period 1994–98 was slightly higher in the Mecklenburg Bight/Arkona Sea (15.5 nmol/l) than in the Bornholm Sea/Gotland Sea (12.0 nmol/l).

In the case of Pb, the dissolved and the particulate fraction contribute equally (30–70 %) to the total concentrations. The mean total concentrations varied considerably from year to year (Figure 6.5), but the mean for the period 1994–98 was consistently higher in the Mecklenburg Bight/Arkona Sea (0.42 nmol/l) than in the Bornholm Sea/Gotland Sea (0.19 nmol/l). This may be attributable to a southwest/northeast gradient in atmospheric deposition combined with the short residence time of Pb in the Baltic Marine Area (Schneider *et al.*, 2000). Although the 5-year record is too short to permit trend analysis, Kremling and Streu (2000) have reported a significant decrease of dissolved Pb in the surface waters of the Baltic Marine Area during the period 1982–1993/95.

In the case of Hg, the dissolved and particulate phases were not analysed separately. The mean total concentrations in the Mecklenburg Bight/Arkona Sea and the Bornholm Sea/Gotland Sea did not differ significantly and were around 0.008 nmol/l and 0.012 nmol/l in 1997 and 1998, respectively.

Sinks and sources

Atmospheric deposition and riverine runoff are equally important as sources of the heavy metals inputs to the Baltic Marine Area. Schneider *et al.* (2000) estimated that atmospheric deposition accounts for about 60% of the total input of Cd and Pb. As exchange with the North Sea is negligible (Schneider *et al.*, 2000), sedimentation is thus the final sink for Cd and Pb in the Baltic Marine Area (see also Chapter 6.2).

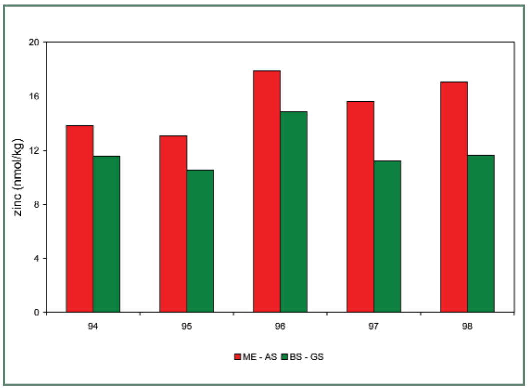


Figure 6.4
Surface seawater Zn concentrations during the period 1994–98. Red bars refer to the Mecklenburg Bight/ Arkona Sea, green bars to the Bornholm Sea/ Gotland Sea.

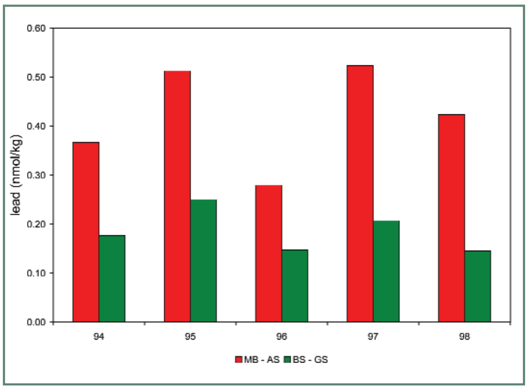


Figure 6.5
Surface seawater Pb concentrations during the period 1994–98. Red bars refer to the Mecklenburg Bight/ Arkona Sea, green bars to the Bornholm Sea/ Gotland Sea.

The situation with Hg is complicated by the fact that oxidized Hg introduced into the Baltic Marine Area via riverine runoff and atmospheric deposition may subsequently undergo reduction and be released to the atmosphere as gaseous elemental Hg (Wängberg *et al.*, 1999). At present, the data record is too uncertain to enable development of a comprehensive balance of the Hg fluxes into and out of the Baltic Marine Area.

The distribution of heavy metals in sediment cores may be used to distinguish between the natural and the anthropogenic fractions. Neumann *et al.* (1996) found that the concentrations of heavy metals were uniform in sediment layers dating back prior to 1900. Taking these concentrations as representing the natural background level, the difference up to the present concentrations can thus be assumed to represent anthropogenic perturbation of the heavy metals budget for the Baltic Marine Area. Given that the sediment contains an inert lithogenic heavy metals fraction that does not contribute to biogeochemical cycling of heavy metals in seawater, anthropogenic inputs can be estimated to have enhanced the concentrations of Cd, Pb, Zn, and Cu by a factor of roughly 2–4 (see Chapter 6.2).

6.1.2. Organic contaminants in surface seawater

Surface seawater concentrations of halogenated hydrocarbons (PCBs, DDT, HCHs, HCB) and petroleum and other hydrocarbons in Baltic Marine Area water have previously been reported in the Third Periodic Assessment. A trend towards a concentration decrease during the period 1975–93 was identified for HCH isomers. Despite the fact that surface seawater contaminant concentrations have

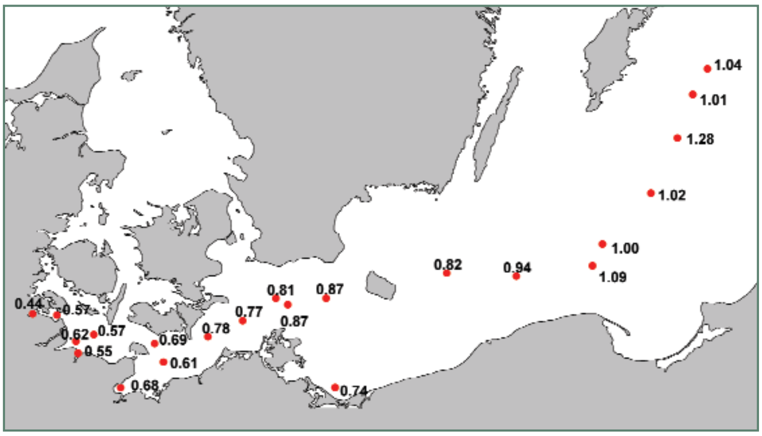


Figure 6.6
Surface (3–5 m) seawater concentrations of α-HCH (ng/l) in the Baltic Marine Area. The values are means for 1997–98.

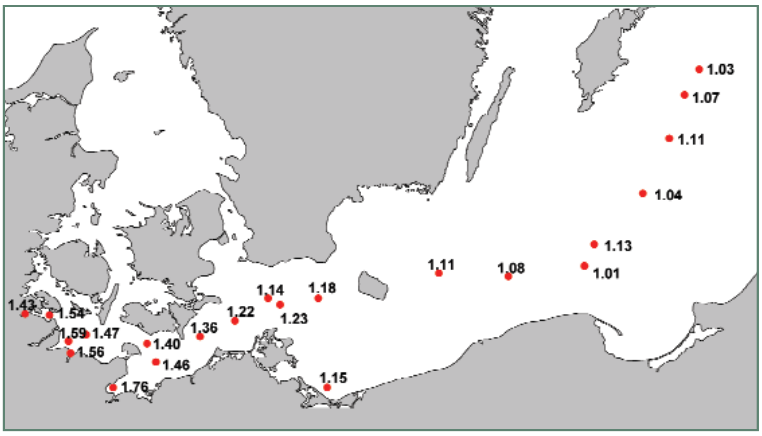


Figure 6.7
Surface (3–5 m) seawater concentrations of γ-HCH (ng/l) in the Baltic Marine Area. The values are means for 1997–98.

been accorded the status of a "main parameter" in the revised HELCOM COMBINE monitoring programme, data are only available from a few Baltic States. Concentrations of halogenated hydrocarbons and PAHs have been reported by one country only and are largely restricted to the area between the western part of the Baltic Marine Area and the Gotland Sea (Anonymous, 2000; Anonymous, in press). Total petroleum hydrocarbon concentrations have been reported for nearly the whole of the Baltic Marine Area.

PCBs

Surface seawater PCB concentrations were rather low. Thus the concentration of PCB 153 – one of the main congeners – ranged from 10 to 24 pg/l (median values for the period 1994–98). Due to the low concentrations and the fact that many samples were below the analysis detection limit, the data set is limited and variability is high (min <10 pg/l, max 34 pg/l). Thus it is not possible to identify a temporal trend for the period 1994–98. Neither was there any evidence of geographical variation, except for a general increase in concentration towards the coasts. Due to the high lipophilicity of the PCBs they are enriched in suspended matter and sediments.

DDTs

Surface seawater DDT concentrations ranged from 2 to 77 pg/l. The highest concentrations were observed in the Pomeranian Bight near the mouth of the river Oder, where the values for DDD and DDE ranged from 30 to 77 pg/l. In the rest of the southern and western parts of the Baltic Marine Area, the concentration range was 2–30 pg/l.

Due to the low concentrations, the data set is rather limited and variability is high. Thus there has been no observable change in concentrations since the preceding assessment periods.

Hexachlorobenzene (HCB)

Surface seawater HCB concentrations ranged from <5 to 10 pg/l. Due to the low concentrations, no changes have been observed relative to the preceding assessment periods. Neither was there any evidence of any geographical variation within the Baltic Marine Area.

Hexachlorocyclohexane (HCH isomers)

Surface seawater concentrations of the HCH isomers exhibited distinct geographical variation within the Baltic Marine Area.

In 1997 and 1998, the concentration of α-HCH ranged from 0.43 ng/l in the Bights of Kiel and Flensburg to 1.1 ng/l in the Baltic Proper. A clear concentration gradient was observed from east to west (Figure 6.6). Such a gradient has previously been detected and is attributable to the mixing of the less contaminated North Sea water with the

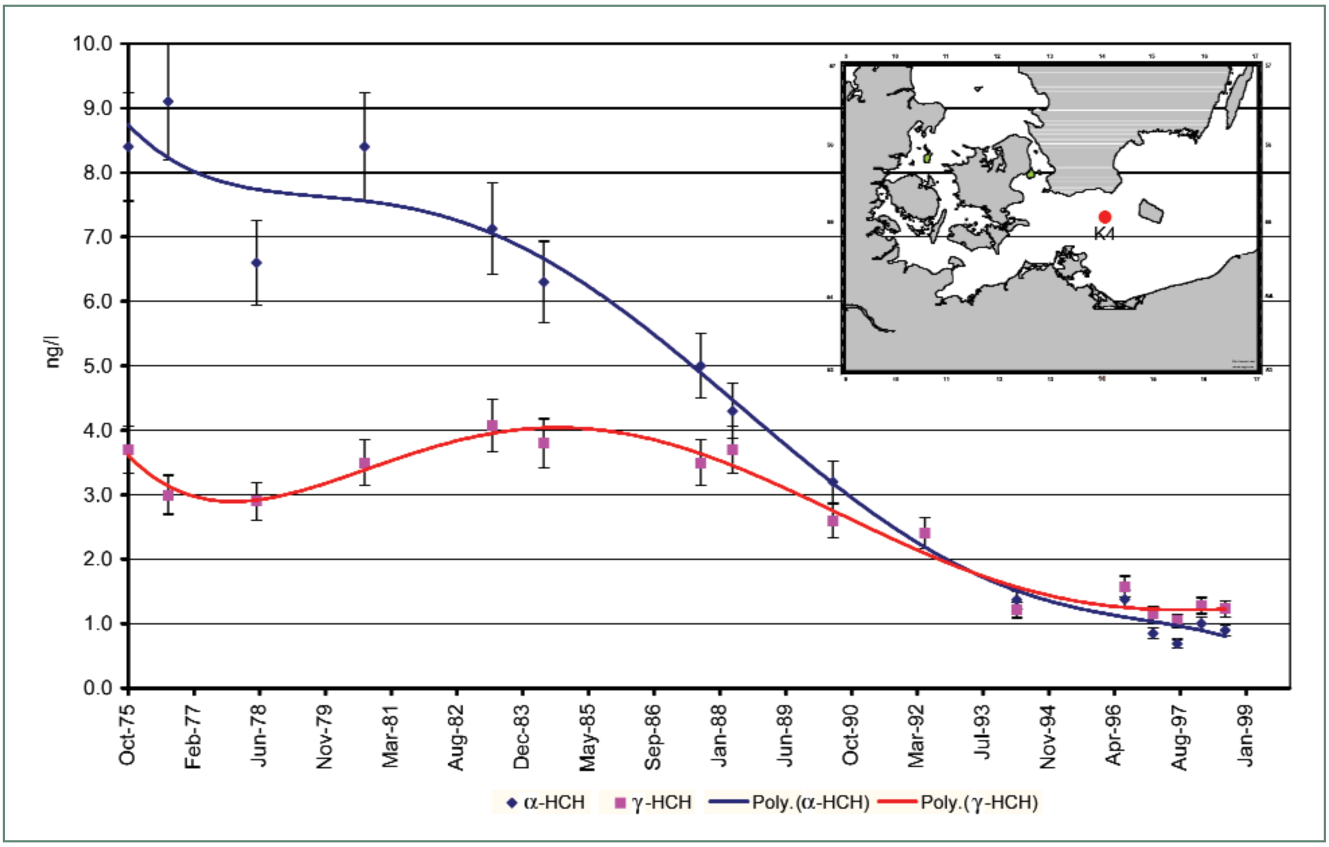


Figure 6.8
Temporal trend in HCH isomer concentrations in the surface water (3–5 m) of the Arkona Basin (station BMP K4). The curves indicate the polynomial equation representing the data.

more contaminated water from the Baltic Marine Area, which is still affected by past HCH loading. The concentration difference in the two water bodies is also evident in the water layers of the Kattegat: thus while the surface seawater concentration (outflow from the Baltic Marine Area) ranged from 0.54 to 0.75 ng/l, that in the deep water (inflow from the North Sea) was only 0.25–0.31 ng/l.

The surface seawater concentrations of Lindane (γ-HCH) in the Baltic Marine Area ranged from 0.9 to 2.6 ng/l (median of 1997 and 1998). The highest concentrations (2.4–2.6 ng/l) were observed in early summer at the coast of Schleswig-Holstein (western Baltic). Lindane exhibited seasonal variation, with elevated concentrations (about 30% higher) being observed in the early summer months. As shown in Figure 6.7, there is an increasing lindane concentration gradient from the central to the western part of the Baltic Marine Area. In the central part the concentrations are relatively uniform with mean values around 1 ng/l (range 0.88–1.3 ng/l). In the western part the concentrations are higher, ranging from 1.4–2.1 ng/l.

With both HCH isomers, distinct changes were observed in the concentrations relative to the preceding assessment periods. Long-term temporal trends can be observed for the HCH isomers as reliable monitoring data is available back to 1975. As shown for the station BMP K4 (Arkona Basin) in Figure 6.8, the α-HCH concentration started to decline rapidly in 1981, falling from 8 ng/l that year to 1.3 ng/l in 1994. Thereafter, the decline slowed considerably, with the concentration falling to 0.9 ng/l by 1998. This is still significantly higher than the concentrations in the North Sea (0.1–0.2 ng/l) and the Atlantic Ocean (0.05–0.1 ng/l). With γ-HCH the trend was less distinct, the concentration first starting to fall around 1988, declining from 3.7 ng/l that year to 1.3 ng/l in 1994. Thereafter the decline slowed, with the concentration reached in 1998 being 1.2 ng/l, similar to that in the southern North Sea.

Petroleum and other hydrocarbons

Total hydrocarbon concentrations (THC: relative concentrations calibrated to topped Ekofisk crude oil) are relatively uniform in the western and central parts of the Baltic Marine Area, ranging from 0.5 to 1.6 µg/l in the summer months of 1997 and 1998. Higher values were observed in coastal regions such as the inner bights or the Oder mouth. In winter time, the concentrations were significantly higher, ranging from 1.1 to 3 µg/l. The concentrations in the Gulf of Bothnia and the Gulf of Finland were similar, the yearly average ranging from 0.2 to 2.1 µg/l (E.-L. Poutanen, personal communication). The concentrations in the Gulf of Finland were slightly higher than those in the adjacent waters. Due to the large variation, no long-term temporal trend could be detected in any region of the Baltic.

The concentrations of single n-alkanes ranged from 1 to 4 ng/l, with some maximal values reaching 10 ng/l. In the summer months, higher concentrations (up to 178 ng/l) were observed for n-C_{17} and n-C_{15}, both of which originate from algae. In addition, the prevalence of odd-numbered n-alkanes above C_{21} indicates that a great amount of these compounds are of biogenic origin (metabolites of terrestrial plants). Thus, – except for acute oil spills – the majority of the alkanes are biogenic in origin.

Polycyclic aromatic hydrocarbons (PAHs)

In the western and central parts of the Baltic Marine Area, the surface seawater concentrations of single PAHs ranged from 4.5 to <2 pg/l. The median concentration of the 2- to 4-ring aromatics (naphthalene to chrysene) in the open sea ranged from 0.02 to 2.1 ng/l. The mean concentrations of the more lipophilic 5- to 6-ring PAHs (benzofluoranthene to benzo[ghi]perylene) were only <0.005–0.15 ng/l, though. Variation in the concentrations is high, especially for the more lipophilic compounds, which are strongly adsorbed on surfaces (suspended matter, biota, sediments). Thus even the median concentrations can differ two-fold from year to year. The variation is smaller with the more water-soluble aromatics like naphthalene and fluorene. Significantly higher concentrations are observed in winter, this being attributable to higher inputs from combustion sources, slower degradation and – in shallow waters – to a higher suspended matter content. Because of the high variability, no changes have been detected relative to the preceding assessment periods.

6.2. Contaminants in sediments

Convenor: Eeva-Liisa Poutanen

Co-convenor: Eugeniusz Andrulewicz

Sediments from the Baltic Marine Area are not included in the HELCOM joint monitoring programmes. Although much data is available in the literature, not all of this is suitable for the purposes of the present assessment due to lack of comparability. The information on geographical variation and contaminant concentrations in surface sediments is therefore based almost entirely on the results of the 1993 Baltic Sea Sediment Baseline Study (ICES, 2000). Additional data from other publications have been included if sample pre-treatment, analytical methods, etc. are similar.

The sea floor in the Baltic Marine Area is extremely variable as regards both topography and quality. The sampling sites are usually located in large uniform basins in order to maximize sample representativeness. The effects of bioturbation and wave- and current-induced resuspension of the sediment surface may severely distort the assumed continuous deposition in such areas.

Contaminant concentrations in sediment are often controlled by factors other than the level of direct inputs. Thus sediment is chemically and sometimes also biologically very active and the hydrochemistry of the near-bottom water layer affects contaminant chemistry on the sediment surface.

Sediments may act as either sinks or sources of material depending on the hydrochemical factors in the water and the conditions pertaining within the sediments. The signal from changes in inputs may therefore be completely obscured by other factors.

6.2.1. Trace metals

By: Eeva-Liisa Poutanen, Mirja Leivuori and Eugeniusz Andrulewicz

Surface distribution of trace metals

Concentrations in surface sediments vary markedly depending on the local conditions, notably the oxic-anoxic variation. Under anoxic conditions, concentrations of trace metals such as Cu, Hg and Cd and to a lesser degree Zn and Pb decrease in water due to the formation of rather insoluble sulphides. In contrast, metals such as Fe, Cr, As, Mn and Co are released from the sediment.

The distance that contaminants are transported from the source depends on the local conditions as well as on the organic matter content and the affinity of the contaminants for particles (quick sedimentation). Substances that remain in solution can be transported over long distances (e.g. Cd). Depending on the temperature, some of the

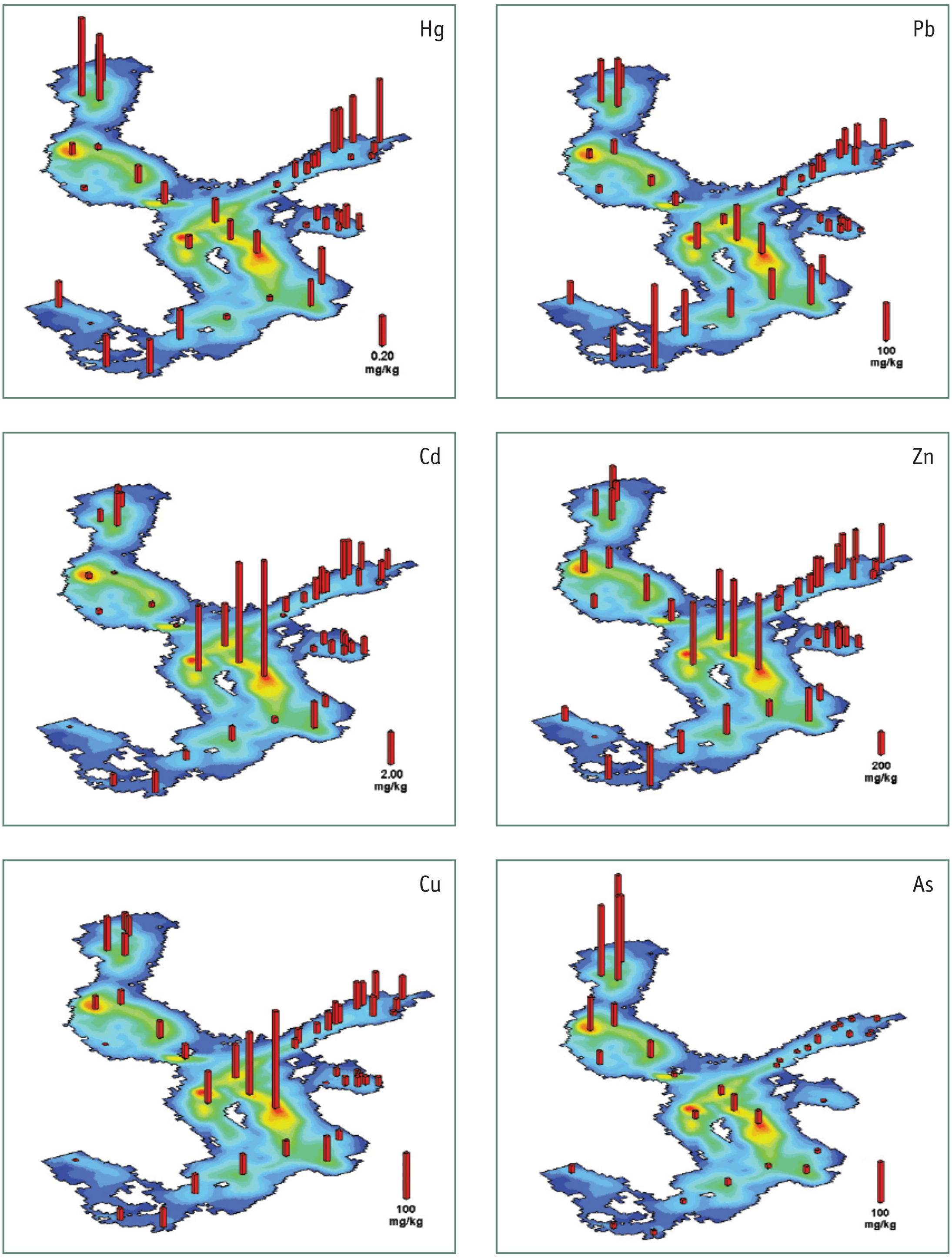


Figure 6.9
Hg, Pb, Cd, Zn, Cu and As levels in surface sediment in the Baltic Marine Area (mg//kg dw; salt-corrected values). Data from ICES (2000), extended by data from Leivuori (1998) and Leivuori et al. (2000). The yellow and orange areas indicate the deep basins.

molecules will even favour the gas phase (e.g. Hg and some organic pollutants).

Mercury

The highest mercury levels (0.42 mg/kg dw) were found in the Bothnian Bay and in the easternmost part of the Gulf of Finland (0.35 mg/kg dw) (Figure 6.9). The overall pattern indicates that mercury is not transported very far away from the original recipient waters. The main sources of the mercury contamination are industrial, for example the pulp and paper industry (for bleaching) and the chlor-alkali industry. In the Bothnian Bay, however, the high mercury levels might be partly due to geochemical release of mercury in connection with the construction of large artificial inland reservoirs for hydroelectric power in Finland in the 1970s and/or due to drainage of large areas of swamp.

Lead

As lead contamination of marine sediments mainly derives from airborne pollution, elevated levels are found in several locations (eastern Gotland Basin, Bothnian Bay, Arkona Basin, Gulf of Gdansk and the eastern end of the Gulf of Finland). The pattern of atmospheric deposition of lead in the subregions of the Baltic Marine Area (i.e. high deposition in the south, decreasing towards north) does not correspond to the lead distribution pattern in sediment, however (cf. Chapter 4). The elevated sediment lead content in the northern subregions must therefore be due to local sources. The highest level (198 mg/kg dw) was recorded in Lübeck Bay (Figure 6.9), possibly due to its coastal proximity and land-based pollution from local sources. The lowest levels were measured in the Bothnian Sea. Taking into account the high total deposition of lead on the Baltic Proper (about 87.3 tonnes/yr in 1997; Bartnicki *et al.*, 2000) and the prevalence of anoxic conditions in the central basins of the Baltic Proper, one would have expected even higher lead levels in the eastern Gotland Basin. The observed moderate concentrations in the eastern Gotland Basin may be due to unsuitability of the sediment core for contaminant measurements as stated by Jensen on the basis of dating (ICES, 2000).

Cadmium

As is apparent from Figure 6.9, the distribution of cadmium is very uneven, ranging from very low levels (0.22 mg/kg dw) in the Bothnian Sea to high levels in the Gotland Basin (7.16 mg/kg dw), the Fårö Deep (6.20 mg/kg dw) and the western Gotland Deep (4.12 mg/kg dw). Slightly elevated levels were observed in the northern central basin (2.70 mg/kg dw), the eastern Gulf of Finland (2.24 mg/kg dw) and the southern Bothnian Bay (2.15 mg/kg dw). The data indicate that cadmium is effectively transported to and trapped in areas where the bottom waters are anoxic.

Zinc and copper

Both zinc and copper appear to be effectively transported towards the central basins of the Baltic Proper (Figure 6.9). The highest levels were found in the Gotland Deep (Zn: 518 mg/kg dw; Cu: 176 mg/kg dw), the Fårö Deep (Zn: 525 mg/kg dw; Cu: 119 mg/kg dw), the western Gotland Basin (Zn: 440 mg/kg dw; Cu: 71 mg/kg dw) and the northern central basin (Zn: 482 mg/kg dw; Cu: 73 mg/kg dw). Moderate levels were also observed at almost all near-coastal stations such as in Lübeck Bay, the Gulf of Gdansk, the Gulf of Riga, the Gulf of Finland and in the Bothnian Bay.

Arsenic

Regional variation was greatest with arsenic (Figure 6.9). Very high levels were found in the Bothnian Bay (222 mg/kg dw), with a noticeable decline further southwards. These high arsenic levels are clearly attributable to former major discharges from the metal industry (Rönnskärsverket) in Skellefteå, Sweden. Although the discharges have been substantially reduced, the arsenic is still present in the sediment.

Additional information

Further information on the recent distribution of trace metals in Baltic Marine Area sediments is available in the literature, e.g. the Gulf of Bothnia and the Gulf of Finland (Leivuori and Niemistö, 1995; Leivuori, 1998; Vallius and Leivuori, 1999; Vallius, 1999), the Gulf of Gdansk (Szczepanska and Uscinowicz, 1994; Uscinowicz *et al.*, 1998), the Arkona Basin and Lübeck Bay (Leipe *et al.*, 1998; Neumann *et al.*, 1996) and the Gulf of Riga (Seisuma *et al.*, 1998; Leivuori *et al.*, 2000).

Historical distribution of trace metals

Downcore sediment profiles from undisturbed sediments may be used to describe the history of the pollution (e.g. Niemistö and Voipio, 1981; Gingele and Leipe, 1998; Emeis *et al.*, 1998). Interpretation of the vertical profiles is hindered by variability of the redox potential, mixing due to bioturbation, hydrodynamic effects, diagenetic processes, dredging etc., and thus has to be undertaken with great care. Several locations in the Baltic Marine Area have been identified as areas useful for monitoring changes in pollution inputs with a five-year or even more locations if a ten-year sampling frequency is applied (ICES, 2000). The undisturbed laminated sediments found in some areas could even be used to study inter-annual variations (e.g. Borg and Jonsson, 1996; Jonsson *et al.*, 2000; Jonsson, 2000).

The vertical distribution of Pb, Hg, Cd and As in some surface sediment cores from different subregions of the Baltic Marine Area are shown in Figure 6.10. In most cores, a clear distinction can be made between natural geochemical background levels and contaminants from anthropogenic sources. In the Gulf of Gdansk, the sedimentation rate is rather high and the core analysed was not long enough to reach down to sediment layers containing background levels. In the case of arsenic, the vertical distribution is only shown for three subregions since the sediment profiles from all regions except the Bothnian Bay and to a lesser degree the Bothnian Sea were very similar.

The lead profile in the Lübeck Bay sediments shows a strong increase in the lead concentration after the beginning of the 1900s. The concentration peaked around the middle of the 1950s and has remained more or less stable. Similar results have been reported by others (e.g. Leipe *et al.*, 1998).

In several of the sediment cores the trace metal concentrations decrease in the upper few centimetres (2 to 5 or 10 cm), thus indicating a general decrease in concentration levels since the early 1980s. This is especially noticeable for arsenic in the Bothnian Bay (A1), for mercury in the Gulf of Finland (F20) and in the Gulf of Bothnia (A1, EB1)

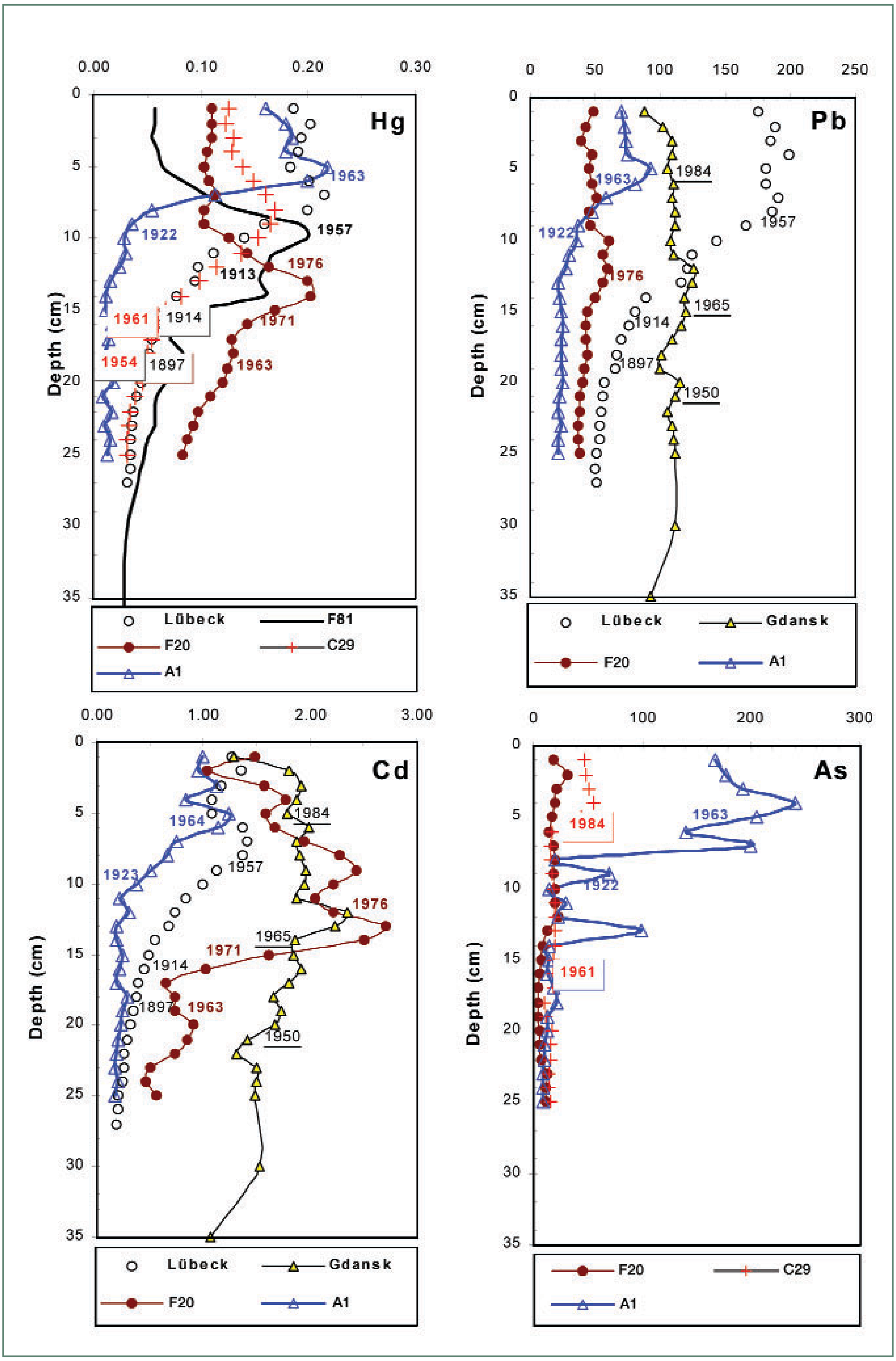


Figure 6.10
Vertical distribution in 1993 of Hg, Pb, Cd and As (dry-weight basis) in sediment from
- *Lübeck Bay,*
- *Gdansk Bay,*
- *Gotland Deep (F-81),*
- *Gulf of Finland (F20),*
- *Bothnian Bay (A1),*
- *Bothnian Sea (C29)*

The age of the sediment is indicated (based on Jensen, 1997).
Position of BMP-stations are shown in figures 5.24 and 5.54.

and for cadmium in the Gulf of Finland. Several recent publications confirm these findings (Uscinowicz *et al.*, 1998; Leivuori, 1998; Vallius and Leivuori, 1999; Gingele and Leipe, 1998; Neumann *et al.*, 1998).

In addition to contamination history, the trace metal profiles reflect the result of postdepositional diagenetic redistribution. New findings from the Bornholm Basin (Leipe *et al.*, 1995) in which sampling stations were reinvestigated after 20 years indicate that such postdepositional changes probably have negligible impact on the distribution of trace metals in several areas of the Baltic Marine Area and that trace metal concentrations change little during the course of time. This is demonstrated, for example, by the fact that the curves for Zn and Pb are virtually identical. Similar behaviour of Cu, Pb, Zn and Cd has been demonstrated in sediment samples from the Bothnian Bay collected 10 years apart (Leivuori and Niemistö, 1995). Because the reactions and mobilization of trace metals take place in the uppermost zone of the sediment, trace metals are subject to variable conditions until they become “buried” deeper in the sediment, where environmental conditions are more stable and the metal concentrations no longer change.

6.2.2. Organic contaminants

By: Eugeniusz Andrulewicz and Eeva-Liisa Poutanen

Organic contaminants may reach the sea via riverine runoff, atmospheric deposition and direct discharge of effluents. They are usually adsorbed onto fine-grained organic particles in the water mass, where sedimentation processes carry them to the bottom, trapping them in the sediment. The concentrations of organic contaminants in sediment accumulation areas are generally several orders of magnitude higher than those in the overlying water mass.

The concentrations of organic contaminants (e.g. PCBs, DDTs and PAHs) may vary widely both between and within subregions of the Baltic Marine Area since they are mainly associated with organic matter in sediment.

Figure 6.11
PCB levels (dry-weight basis) in surface sediment in the Baltic Marine Area. Data from ICES (2000).

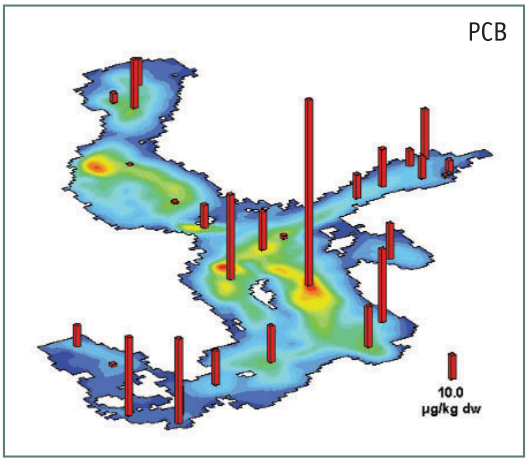
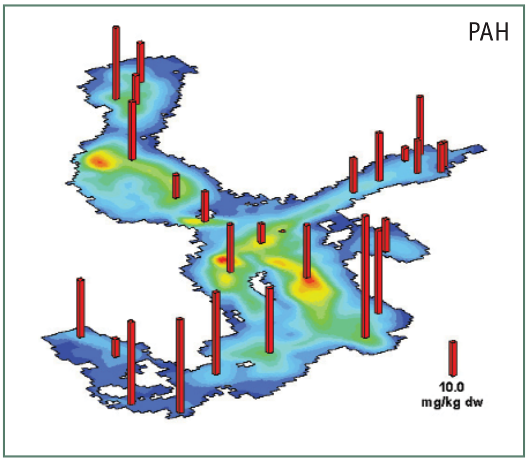


Figure 6.12
PAH levels (dry-weight basis) in surface sediment in the Baltic Marine Area. Data from ICES (2000).



Surface distribution of organic contaminants

Only limited conclusions can be drawn from the available data regarding the geographical distribution of organic contaminants. Moreover, comparison between different studies and study areas is hampered by the varying number of individual substances (congeners) examined. Interpretation of the findings is further hampered by differences in geochemical properties between and possibly also within the basins.

PCBs

In general, the concentrations of PCBs were found to increase from north to south (Figure 6.11). The highest level of PCB contamination was observed in the eastern Gotland Basin, the western Gotland Basin and in Lübeck Bay. Recent investigations of PCBs in sediments along the Swedish coast of the Baltic Proper reveal concentrations ranging from about 2 to 33 µg/kg dw. Elevated concentrations (up to 100 µg/kg dw) were found near Stockholm, however (Meili *et al.*, 2000). Concentrations in surface sediment from the Landsort Deep were low (3-10 µg/kg dw) (de Wit *et al.*, 1990). Low concentrations of 6 CB congeners (excluding 105) were found by Kowalewska and Konat (1999) in sediment from the Gulf of Gdansk: 27 µg/kg dw in the Deep of Gdansk, 14 µg/kg dw near Gdynia harbour and 7 µg/kg dw near the mouth of the Vistula River. Rather low concentrations ranging from about 2 to 14 µg/kg dw have also been reported for the Gulf of Bothnia (ICES, 2000; Bavel *et al.*, 1995).

DDTs

Recent information on the spatial distribution of DDTs in surface sediments is even more limited than for PCBs. The sediment concentrations of p,p′-DDE in the Gulf of Gdansk is reported to be 1.55 µg/kg dw, and the sum of DDT to be 2 to 5 µg/kg dw (Sapota, 2000). A decade earlier in 1986, the total DDT concentration in the surface sediment in the Gulf of Finland was reported to be 10–40 µg/kg dw (Perttilä and Haahti, 1986).

PAHs

The highest PAH concentrations (up to 35.2 mg/kg dw) have been found in the southern-most part of the Baltic Marine Area (Gulf of Gdansk, Lübeck Bay, Mecklenburg Bay, Arkona Basin; Figure 6.12). Somewhat elevated concentrations were recorded in the eastern Gulf of Finland as well as in the northern Gulf of Bothnia (up to 17.0 mg/kg dw and 20.9 mg/kg dw, respectively). As PAHs reach the marine environment via effluent discharges, urban run-off, atmospheric deposition and spillage or disposal of oil and petroleum products, these elevated concentrations are probably attributable to coastal proximity. Slightly lower levels (sum of 15 individual compounds) ranging from 0.01-7.0 mg/kg dw (average 1.83 mg/kg dw) have been recorded in the Gulf of Gdansk (Kowalewska and Konat, 1997). Sediment PAH levels are reported to range from 0.8 mg/kg dw to 1.9 mg/kg dw in the Belt Sea and Arkona Sea (Witt, 1995; Witt and Trost, 1999) and from 2.18 mg/kg to 0.845 mg/kg dw in three Bodden areas, (Witt, 1995; Witt and Trost, 1999).

While low-molecular weight PAHs can be acutely toxic, the major concern is for some of the higher-molecular weight compounds as some of their metabolites are active carcinogens in marine animals. At least four of the PAHs found in high concentrations in the southern Baltic Marine Area sediments (benz[*a*]anthracene, benzo[*k*]fluoranthene, chrysene, and benzo[*a*]pyrene) are known to be carcinogenic in mammals, and are almost certainly so in fish (Woodhead *et al.*, 1999).

Organotins

Tributyltin (TBT) and to some extent also other organotins are still used as antifoulants on large sea-going vessels. Release from ship hulls is the main source of marine TBT contamination. Organotin compounds occur in sediments as tri- (TBT), di- (DBT) and monobutyltin (MBT), of which DBT and MBT are degradation products of TBT. TBT has oestrogenic effects and can cause imposex or intersex in many snail species.

The highest concentration of organotin is typically found in sediments from harbours, port channels, near shipyards and along shipping lanes (Table 6.1). In several locations the majority of butyltin compounds were present as TBT, thus suggesting recent exposure of the local aquatic environment to TBT.

Historical distribution of organic contaminants

The general concentration trend in the Baltic Proper suggests an increase of PCB concentrations from the early 1970s onwards (ICES, 2000). This contrasts with the decreasing concentration trends for PCBs in biota from the Baltic Proper (HELCOM, 1996). In the Gulf of Finland, PCB concentrations were found to be slightly lower in the early 1990s than in the 1970s. In some areas (e.g. Gotland Basin, Gulf of Gdansk, Bornholm Basin) PCB concentrations remain more or less constant or even increase after the 1970s (ICES, 2000; Nylund *et al.*, 1992). A decrease in PCB concentrations in the early 1990s has been reported in the Gulf of Gdansk (Sapota, 2000; Andrulewicz *et al.*, 2000).

Earlier studies of anoxic sediments from the Baltic have often shown no evidence of decreasing concentrations in recent years. However, several investigations indicate a tendency towards decreasing concentrations in the most recent lamina of sediment from the northwestern part of the Baltic Proper (for a review see Olsson *et al.*, 2000). In the offshore area of the northwestern Baltic Proper, PCB increased during the 1940-60s, with maximum concentrations of around 60 µg/kg dw being recorded in the early 1970s (Jonsson *et al.*, 2000). PCB levels also increased in the inner Stockholm archipelago area from the late 1960s, peaking in 1975. PCB concentrations were remarkably high in lamina from the 1940s and 1950s, and the decrease since the early 1970s is less than that recorded in biota, possibly due to diffusion processes. Recent studies of between-year variation in PCB deposition rates clearly indicate the importance of sediment resuspension during windy periods (Eckhéll *et al.*, 2000). This may explain why the lamina of anoxic sediment cores are poor matrices to study temporal trends of both nonpolar substances and trace metals.

A recent study shows that DDT degrades in sediment and DDT is probably transported through the sediment core by diffusion mechanisms (Olsson *et al.*, 2000). The trends thus differ in sediment cores and biota, possibly explaining why considerable amounts of DDT and PCB were found in sediment lamina representing the 1940s, long before the two substances were in widespread use.

The downcore PAH concentrations generally exhibit peak concentrations in the 1970s and early 1980s, thereafter decreasing towards the sediment surface. In the sediment cores from the Mecklenburg Bight, PAH levels were enhanced at a depth

Location	Description	mg/kg dw			Reference
		TBT	DBT	MBT	
Gulf of Gdansk	Shipyards	2.6-40	2.0-42	1.2-46	Szpunar *et al.*, 1997; Steinthikumar *et al.*, 1999
Aarhus Bay	Harbour/marina	0.3-6.3	0.1-0.9	0.01-0.3	Aarhus County, 2000
Aarhus Bay	Outside harbour	0,01-0.19			Aarhus County, 2000
Aarhus Bay	Reference site	0.01	<0.001	<0.001	Aarhus County, 2000
Haderslev Fjord	Reference site	0.28-1.17			South Jutland County, 1998
Augustenborg Fjord	Reference site	0.004-0.007			South Jutland County, 1998
Hjelsminde Fjord	Reference site	0.001			South Jutland County, 1998
Flensborg Fjord	Reference site	0.001-0.04			South Jutland County, 1998
Odense Fjord	Reference site	0.12-0.56			South Jutland County, 1998
Odense Fjord	Harbour	2.6-5.0			South Jutland County, 1998

Table 6.1. Organotin compounds in marine sediments (mg/kg dw).

of 6 and 10 cm (1965 and 1945), reflecting the high anthropogenic loading during that period (Witt and Trost, 1999).

6.2.3. Other contaminants

For a number of years chlorine was widely used as a bleaching agent in the pulp and paper industry. During bleaching, different chlorinated compounds are created and finally discharged as effluents. To assess areas contaminated by the pulp and paper industry, extractable organic chlorine (EOCl), extractable organic halogen (EOX), total organic halogen (TOCl) are usually determined. Detailed information about these compounds is available from the literature (Håkanson *et al.*, 1988; Södergren *et al.*, 1993; Jonsson *et al.*, 1993; Kankaanpää, 1996 and 1997; Kankaanpää *et al.*, 1997; Palm and Lammi, 1995).

Information about sediment concentrations of polybrominated diphenylethers (PBDE), dioxins and furans in the Baltic Marine Area sediments is very limited. From Swedish studies it appears that dioxins and furans are present in Baltic sediments at concentrations of a few ng/kg (Kjeller and Rappe, 1995; de Wit *et al.*, 1990).

6.3. Contaminant concentrations in biota

Convenor: Mats Olsson

Co-convenor: Anders Bignert

This chapter presents the data on spatial distribution and temporal variation in contaminant concentrations in Baltic biota. The locations and species investigated during the assessment period are indicated in Figure 6.13. Spatial variation is described using mean concentrations for the last 5 years or, when unavailable, shorter time periods during the assessment period. In cases where long-term monitoring reveals a statistically significant change in concentration over time, an estimated value for the last year of observation has been calculated on the basis of a log-linear regression analysis. For reference purposes, information from the Baltic Marine Area is sometimes compared with findings made at Väderöarna, a small group of islands located in the Skagerrak. When studying temporal trends, shorter periods than 10 years have not normally been used since they do not provide an adequately solid basis for trend analysis of environmental data (Bignert *et al.* 1993, Bignert *et al.*, 1994).

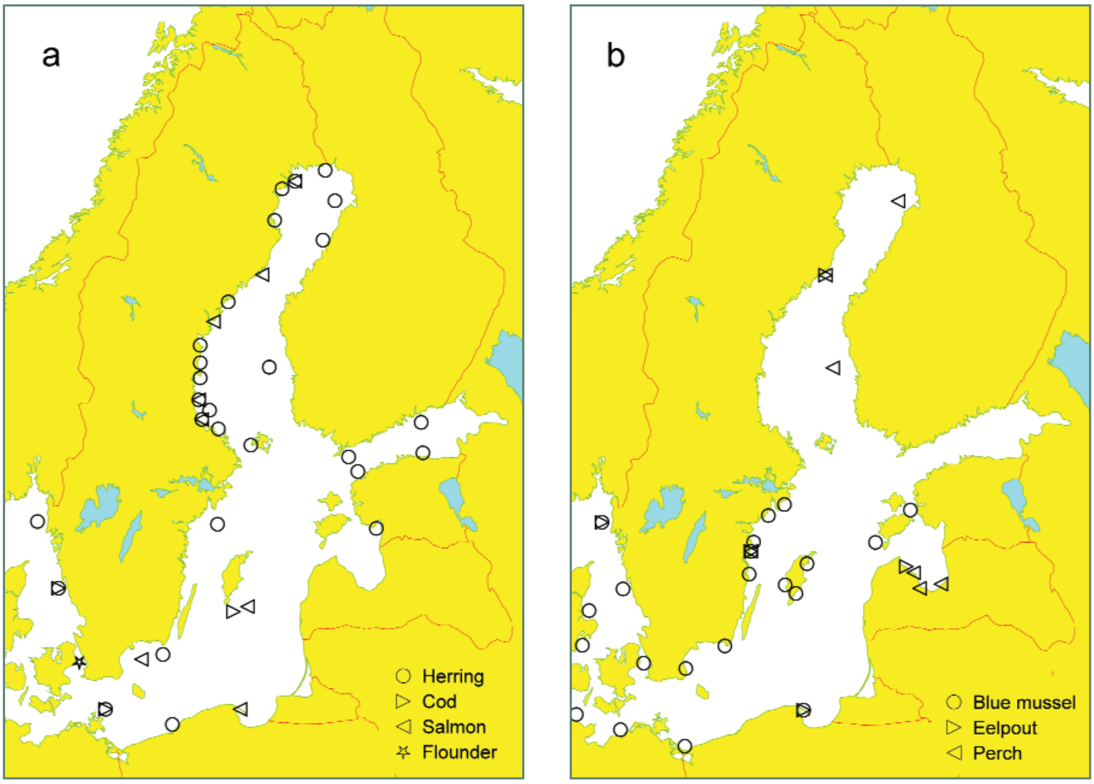


Figure 6.13
Map indicating sampling locations for species belonging to a) the open sea and b) coastal waters that have been analysed for concentrations of heavy metals and organic compounds during the assessment period.

6.3.1. Heavy metals

By: Mats Olsson, Anders Bignert, Mintauts Jansons, Markku Korhonen, Mirja Leivuori, Britta Pedersen, Eeva-Liisa Poutanen and Mart Simm

Heavy metal concentrations in fish are determined in liver except in the case of mercury, which is always determined in muscle. With invertebrates, the concentrations are determined in homogenized soft tissue.

The most comprehensive material is available for herring, both with respect to spatial distribution and temporal variation. It is the only organism for which data are available for the entire Baltic Marine Area from the Kattegat to the Bothnian Bay. Other species are used to support the discussions.

Background concentrations of heavy metals in pristine areas of the Baltic Marine Area have been determined using blue mussel (ICES, 1997). In the following, the background concentration range is referred to as the "OSPAR range".

Cadmium

From the spatial distribution of cadmium (Cd) in the Baltic Marine Area (Figure 6.14) it can be seen that the concentration is generally lower in biota from the Kattegat than from the Baltic Proper. The concentration in herring (*Clupea harengus*) liver varies from 0.16 to 0.91 µg/g ww, the lowest concentration being found in the Kattegat and the highest in the central part of the Bothnian Sea. The Cd concentration in blue mussel (*Mytilus edulis*) is also lower in the Kattegat than in the Baltic Proper, where the concentration range is about 0.25–0.8 µg/g ww (Figure 6.15), hence exceeding the "OSPAR range" (0.070–0.11 µg/g ww). No obvious pattern of spatial differences was apparent in the Baltic Proper (Figure 6.14). Although it has been suggested that decreasing salinity increases the bioavailability of Cd ions (Bengtsson *et al.*, 1975), there is no evidence from the available data to indicate a concentration increase from the southern to the northern parts of the Baltic Marine Area in line with the decreasing salinity.

The temporal trend over the past 18 years is a statistically significant 4–6% annual increase in liver Cd levels in pelagic herring (Figure 6.16). This is apparent in Swedish waters of the Baltic Proper and in the southern part of the Bothnian Sea, but not in the Bothnian Bay or in the Kattegat. Recent data on herring liver are available from both Finnish and German waters. The concentrations measured in German waters are almost identical to those found in the Swedish samples from corresponding areas after several years of increase (0.60 µg/g ww), while those measured in Finnish waters are slightly lower (0.40 µg/g ww). A Danish time series (1969–99) based on flounder (*Pleuronectes flesus*) from the Sound also indicates a significant increase of about 5% annually (Anonymous, 1999) (Figure 6.17). Two shorter time series (5 and 4 years) based on coastal perch (*Perca fluviatilis*) and blue mussel from Swedish waters of the Baltic Proper both show a significant increase exceeding 10% annually. The findings thus indicate that the biota Cd levels are increasing in the Baltic Proper, the only exception being the liver of demersal Baltic cod (*Gadus morhua*), where the concentration has decreased by 6% over the corresponding time period.

Liver Cd concentrations were fairly similar in herring and eelpout (*Zoarces viviparus*), the means ranging from 0.13 to 0.91 µg/g ww, but lower in perch liver. The Cd concentration was lowest (0.021–0.043 µg/g ww) in lipid-rich cod liver from both German and Swedish waters. The wet weight concentration in blue mussel is of the same magnitude as the corresponding concentration in herring liver.

Whereas the Cd concentration in most biota has increased over the past 20 years, the seawater Cd level in the Baltic Proper is decreasing in the water above the halocline and remaining constant in deep waters (Schneider, 1996). This remarkable dif-

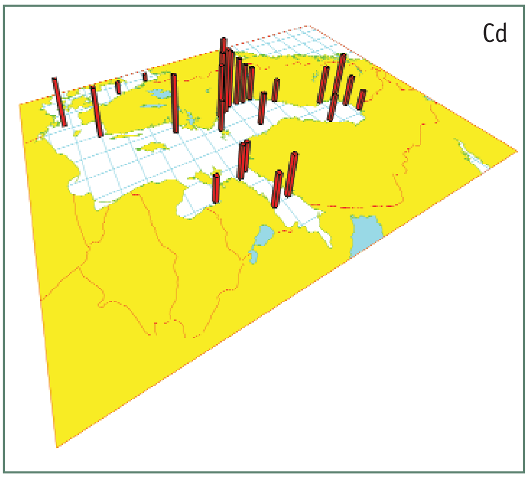


Figure 6.14
Spatial distribution in the concentration of Cd in herring liver. The concentration was highest (0.91 µg/g ww) along the western coast of the Bothnian Sea and lowest (0.16 µg/g ww) in the Kattegat. In the adjacent waters of the Skagerrak, the concentration was even lower (0.10 µg/g ww).

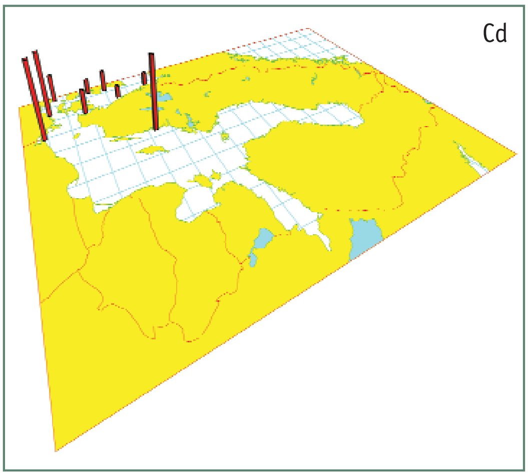


Figure 6.15
Spatial distribution in the concentration of Cd in blue mussel (dw). The concentration was highest (5.3 µg/g dw) outside the Pomerian coast and lowest (0.83 µg/g dw) in the Kattegat.

Figure 6.16
Cd concentration in herring liver (µg/kg ww) collected along the Swedish coast during the 1980s and 1990s:
a) the Bothnian Bay,
b) the southern Bothnian Sea,
c) the northern Baltic Proper,
d) the southern Baltic Proper and
e) the Kattegat.
The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is shown.

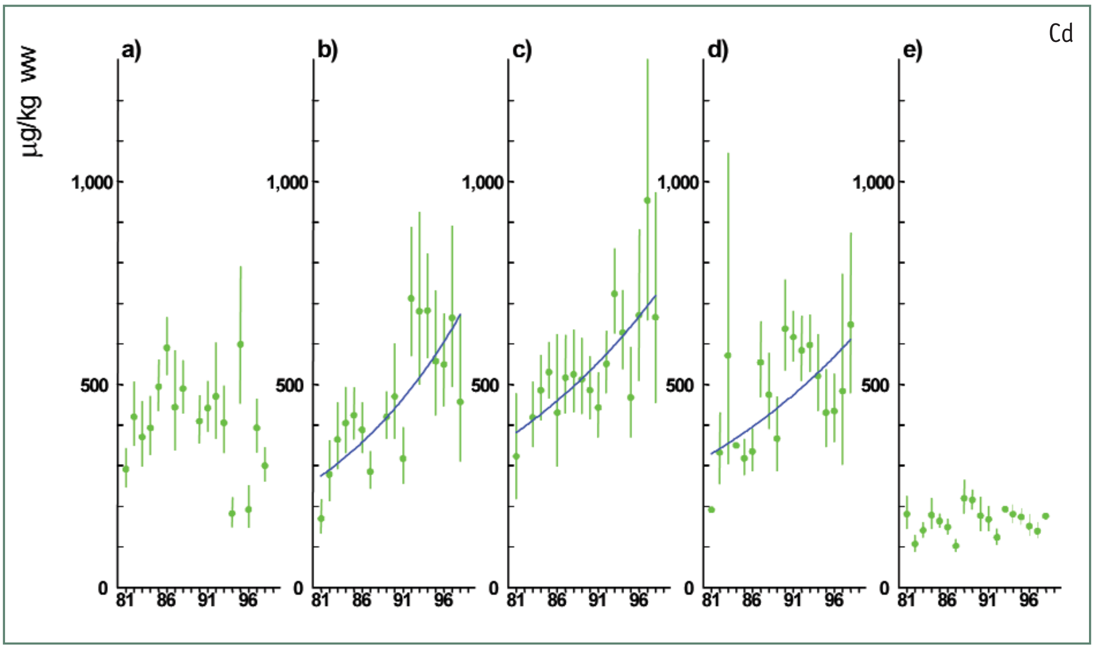
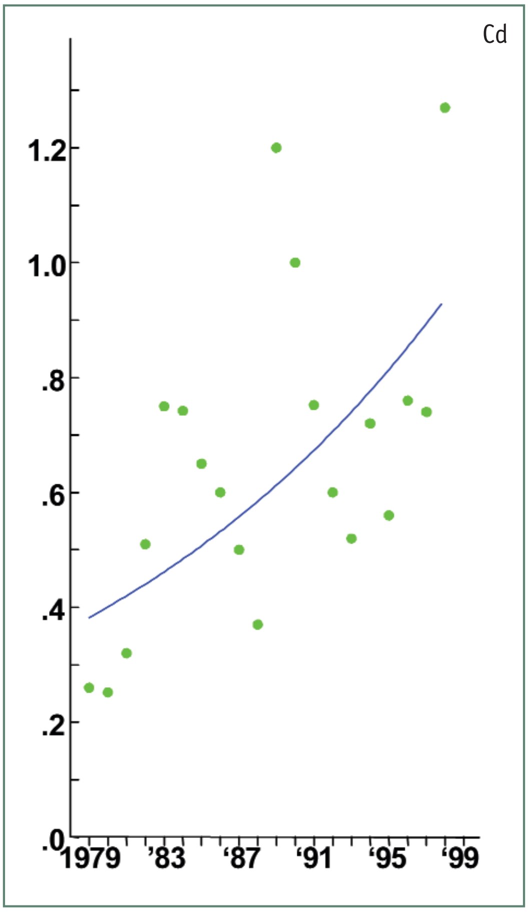


Figure 6.17
Cd concentration (µg/g dw) in flounder liver from the Danish part of the Sound over the period 1979–98 (Data from Markager et al., 1999).



ference has not yet been explained. Reanalysis of old herring samples (saved in a specimen bank) using the same analytical method as for the recent material has confirmed the increase.

Although the reason for the increasing biota Cd concentration in the Baltic Marine Area has not been clarified, a number of tentative explanations have been proposed. Thus, Harms (1995) suggests that the increase in herring liver Cd concentration is a result of recent decreasing salinity in the Baltic reported by Bergström and Mattäus (1996). No simple correlation between Cd concentration and the south–north axis of decreasing salinity has been demonstrated, though. In the northern part of the Baltic Marine Area in particular, bioavailability might be influenced by complexing agents such as humic acids that hamper Cd uptake in herring in the northernmost areas. Borg *et al.* (1988) have suggested increased mobility of Cd ions due to acidification of poorly buffered fresh waters in Baltic precipitation areas, but no time series of Cd concentrations in freshwater fish are available to confirm or reject this hypothesis. Finally, since Cd is bound to the protein metallothionin (MT), increased production of metallothionin by the fish could lead to increased Cd levels in their tissues (da Silva and Williams, 1994). Changes in Cd concentrations in fish tissues could be the result of both inhibition and induction of MT production caused by physiological changes rather than an increase in environmental Cd levels.

Copper

There is no apparent pattern to the spatial distribution of biota copper (Cu) concentrations with respect to either region or salinity. In herring liver the Cu concentration ranges from 2.1 to 4.8 µg/g ww, with the highest concentrations (4.7 and 4.8 µg/g ww) being found in the central Bothnian Sea in the vicinity of fairly industrialized areas of the Swedish coast. All other concentrations ranged from 2.1 to 3.9 µg/g ww. The concentration found in blue mussel from Denmark and Sweden (5.4–10 µg/g dw) is lower than at a reference site at Kobbefjord, Greenland (Riget *et al.*, 1993) and lower than the concentration levels regarded as “high background levels at diffuse loading”, i.e. <10 µg/g dw (Knudsen and Skie, 1992).

The time series (>15 years) indicate that the Cu concentration has remained stable during the assessment period in both fish species studied, i.e. herring and cod.

Liver Cu concentration is remarkably consistent (2–4 µg/g ww) in several fish species (herring, perch, eelpout), an exception being lipid-rich cod liver, where the concentration is 3–4 times greater (7–32 µg/g ww).

As for Cd, the seawater Cu concentration decreased in the Baltic Marine Area over the period 1980–93 (Schneider, 1996), a trend that is not reflected in the biota Cu concentrations.

Lead

There is no systematic pattern to the spatial distribution of biota lead (Pb) concentrations with respect to either region or salinity. The liver Pb concentration in herring ranges from 0.020 to 0.064 µg/g ww. With both cod and blue mussel, the Pb concentration is similar in both the saline waters of the Kattegat and the more brackish waters of the Baltic Proper. The Pb concentrations in blue mussel from the Kattegat and the Baltic Proper are similar to those measured at a reference site at Kobbefjord, Greenland (Riget *et al.*, 1993), and lower than the concentration levels regarded as “high background levels at diffuse loading”, i.e. <5 µg/g dw (Knudsen and Skie, 1992).

The long-term temporal trend for liver Pb concentration in pelagic herring (from about 1980 onwards) is a statistically significant 3–6% annual decrease in the Gulf of Bothnia and in the northern Baltic Proper along the Swedish coast (Figure 6.18). A significant 4% annual decrease over the past 10 years has also been demonstrated in herring from the southern Baltic Proper. In the Kattegat, a decrease is also apparent, but is not signifi-

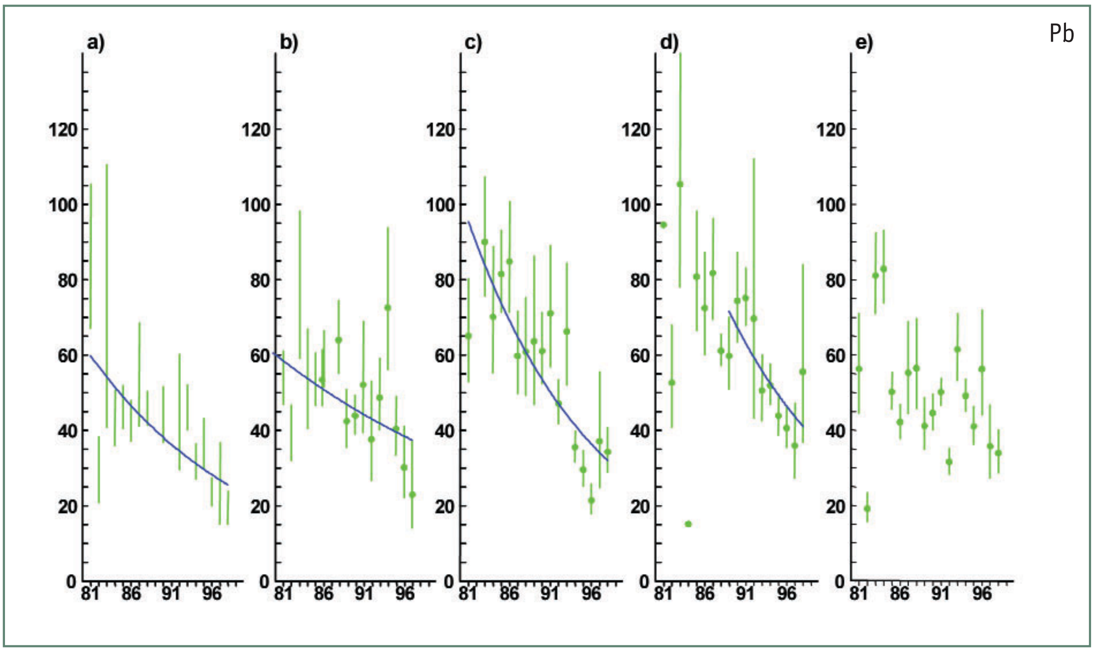


Figure 6.18
Pb concentration in herring liver collected along the Swedish coast during the 1980s and 1990s:
a) the Bothnian Bay,
b) the southern Bothnian Sea,
c) the northern Baltic Proper,
d) the southern Baltic Proper and
e) the Kattegat.
The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated (blue for the entire period, red for the last 10 years).

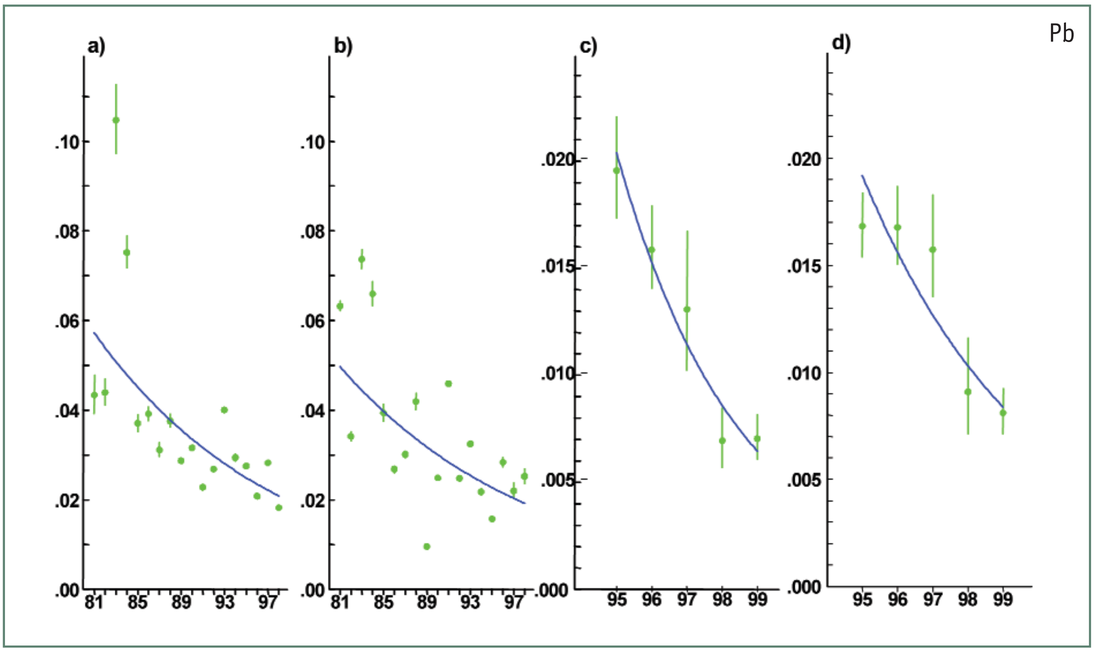


Figure 6.19
Pb concentration (µg/g ww) in fish liver during the 1980s and 1990s:
a) cod, the Baltic Proper,
b) cod, the Kattegat,
c) perch, the Gulf of Bothnia, and
d) perch, the Baltic Proper.
The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated.

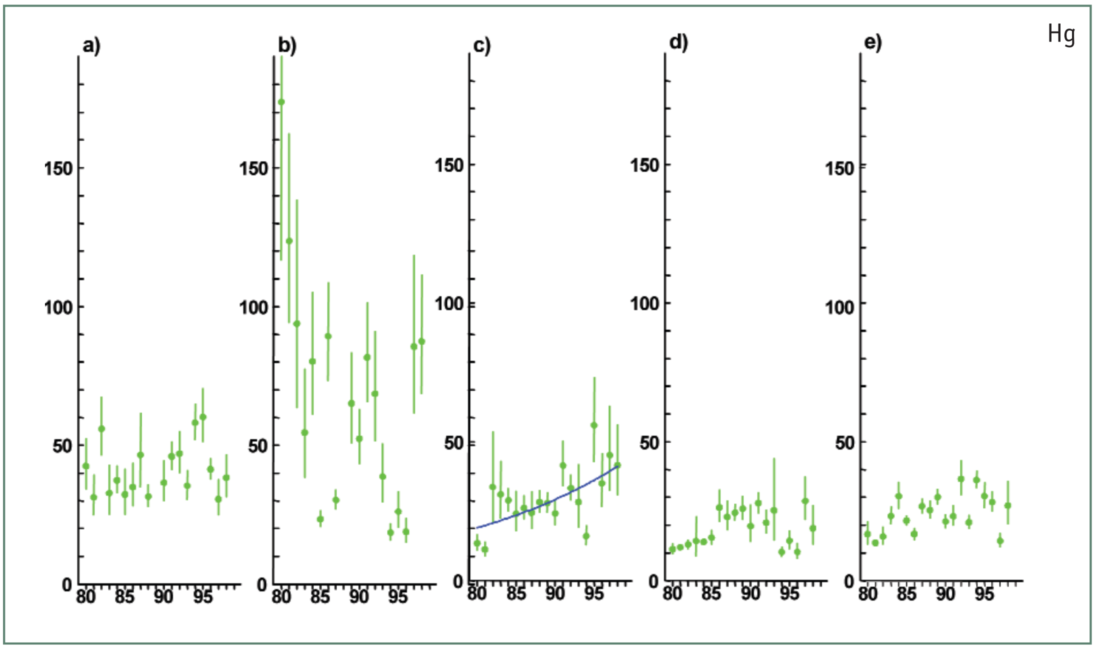


Figure 6.20
Hg concentration in herring muscle collected along the Swedish coast during the 1980s and 1990s:
a) the Bothnian Bay,
b) the southern Bothnian Sea,
c) the northern Baltic Proper,
d) the southern Baltic Proper and
e) the Kattegat.
The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated.

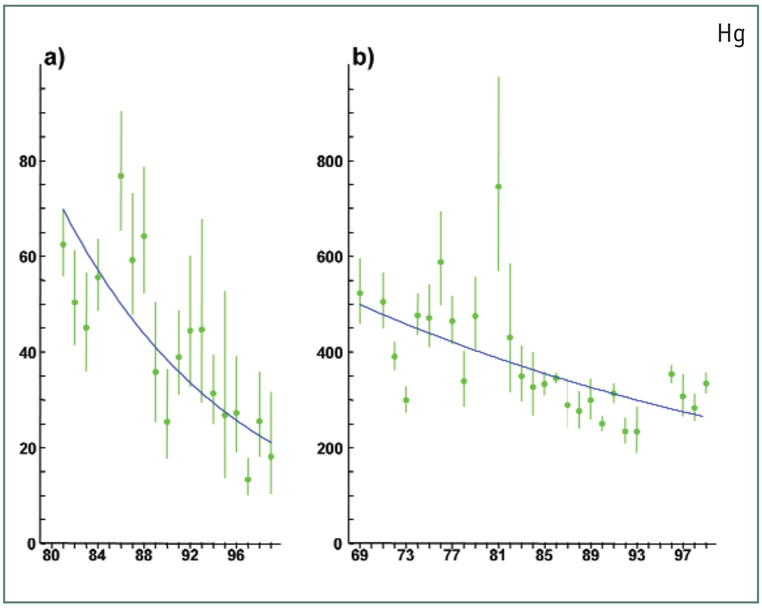


Figure 6.21
Hg concentration (µg/g dw) in a) perch muscle from the Baltic Proper over the period 1980–98 and b) guillemot egg from the central Baltic Proper over the period 1969–98. The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated.

cant. Liver Pb concentrations are also decreasing by about 6% annually in desmeral cod in both the Kattegat and the Baltic Proper. (Figure 6.19). Shorter time series (only five years) for Pb concentration in perch liver indicate a significant annual decrease of more than 20% in both the Gulf of Bothnia and the Baltic Proper (Figure 6.19).

Studies of Pb concentrations in liver of different fish species indicate fairly consistent concentrations irrespective of the species investigated (0.011–0.064 µg/g ww). However, the concentrations in the soft tissue of blue mussel are up to one order of magnitude higher than in fish liver (0.160–0.250 µg/g ww).

Mercury

Neither is there any systematic pattern to the spatial distribution of biota mercury (Hg) concentrations in the Baltic Marine Area. Muscle Hg concentration in herring ranges from 0.016 to 0.083 µg/g ww. The lowest concentration is found at one sampling site in the central part of the Gulf of Bothnia, while the highest is found at a site in the northernmost part of the same area. Comparing the Kattegat and the Baltic Proper, the Hg concentration in blue mussel and cod liver is similar in the two areas. The Hg concentration in blue mussel from the Baltic Proper and Kattegat slightly exceeds the upper limit of the “OSPAR range” (0.005–0.010 µg/g ww; ICES, 1997). The estimated mean concentrations for 1998 in herring and cod muscle are both below the range proposed as the “present background concentrations in pristine areas within the Baltic Marine Area” (0.010–0.050 µg/g ww) in round fish (ICES, 1997).

The long-term temporal trend for Hg concentration differs in the various subregions and hence is confusing and difficult to explain. In the northern Baltic Proper along the Swedish coast the Hg concentration in herring muscle is increasing significantly by about 4% annually (Figure 6.20). In the southwestern part of the Gulf of Bothnia, the concentration was fairly high in the beginning of the 1980s but subsequently declined until 1996. During the last two years, however, the concentration has returned to the level in the early 1980s indicating that attention should be paid to this area. In contrast, the Hg concentration is significantly decreasing by 6.3% annually in perch muscle in the central Baltic Proper, and by 2.3% annually in guillemot eggs from the same area (Figure 6.21).

No obvious difference in fish muscle Hg concentrations are apparent between the species investigated, with all values lying in the range 0.016–0.091 µg/g ww. One would expect higher concentrations in the predatory species cod and perch, but the available data do not reveal any clear difference. This could in part be due to local variation in the Hg level to which the individual fish have been exposed. The Hg concentrations in blue mussel (0.011 µg/g ww) and in perch (0.023 µg/g ww) from the same locality in the central Baltic Proper are remarkably similar. The concentration in guillemot egg is much higher (0.260 µg/g ww), however, reflecting biomagnification in the terrestrial part of the food web.

Zinc

The concentration of zinc (Zn) in fish liver is fairly constant irrespective of sampling area and species, ranging from 19 to 42 µg/g ww. Similar although slightly lower concentrations (17–36 µg/g ww) are also detected in blue mussel. The latter concentrations are below the proposed background level for the North Sea (ICES, 1997).

Other heavy metals

Some recent data are available concerning the concentration of chromiun (Cr) and nickel (Ni) in Baltic biota over the period 1995–98. These can be used for a preliminary evaluation of spatial and species differences. The mean Ni concentrations during the four-year period are as follows: herring: about 65 mg/kg ww, cod: about 80 mg/kg ww, eelpout: about 60 mg/kg ww and perch: about 30 mg/kg ww. The corresponding Cr concentrations are: herring, eelpout and cod: about 60–80mg/kg ww; and perch: about 40 mg/kg ww. The concentrations in blue mussel were considerably higher: Ni: about 300 µg/kg ww and Cr: about 400mg/kg ww.

6.3.2. Organic contaminants

By: Mats Olsson, Anders Bignert, Marie Aune, Michael Haarich, Uwe Harms, Markku Korhonen, Eeva-Liisa Poutanen, Ott Roots and Grazyna Sapota

The data for the halogenated organic compounds are expressed on a lipid weight basis (lw) because they are all highly lipophilic and hence accumulate in body lipids. The concentrations of organic contaminants in fish are determined in muscle except in the case of cod, where liver is always analysed instead. With invertebrates, the concentrations are determined in the soft tissue.

The environmental transport and fate of organochlorines have been thoroughly discussed over the last 10 years. As a result of their volatility they are transported from warmer to colder areas of the globe, in part explaining why the Baltic has been seriously polluted by these compounds in the past.

Less volatile compounds will move more slowly than more volatile compounds and concentrations will thus decrease more rapidly in warmer areas of the world than in colder. The model explaining the expected global transport of persistent chemicals is referred to as Global Chromatography (Wania and Mackay, 1995). The long-term monitoring of DDT, PCB, HCB and HCH in biota samples carried out in the Baltic since the 1970s has been used to verify the model. Interestingly, though, these unique temporal trends do not support the model since the concentrations decrease at a similar rate in the warmer south as in the colder north (Bignert *et al.*, 1998).

DDT

The concentration of ΣDDT[1] in herring muscle varies between 0.06 and 1.2 µg/g lw. It is highest in the southern Baltic Proper and decreases to the north and west, with similar levels being found in the Kattegat and the Bothnian Bay (Figure 6.22). This spatial distribution pattern also applies to blue mussel (Figure 6.23), cod and eelpout, species that cover different parts of the assessment area. The ΣDDT concentration in cod liver was 1.0 µg/g lw in the Arkona Sea and 0.22 µg/g lw in the Kattegat. The highest concentration in perch was found in the inner part of the Gulf of Finland (1.0 µg/g lw) while the lowest was found on the Swedish coast of the Baltic Proper (0.071 µg/g lw).

The temporal trend for ΣDDT in herring is a decrease over the entire period. The longest time series going back to the late 1960s/early 1970s reveal an annual decrease of about 7% in herring from the southern Bothnian Sea and of 11–12% in both herring and guillemot egg from the Baltic Proper (Figure 6.24). Time series starting in the early 1980s or later all show a similar decrease in the range 9–14% in the Kattegat and the Gulf of Bothnia and from 5–11% in the Baltic Proper, irrespective of the species studied, i.e. herring, cod, perch and blue mussel (Figure 6.25). The decrease in concentration is statistically significant during the last 10 years for most studied biota irrespective of whether they are coastal like perch, pelagic like herring and guillemot or demersal like cod. The ΣDDT concentration also decreases at a similar rate in long-term monitoring studies going back to the late 1960s in Swedish fresh waters from the south to the northernmost Subarctic regions of Sweden. This indicates that decreases in atmospheric deposition probably also play an important role in explaining the processes in the Baltic (Olsson and Reutergårdh, 1986; Bignert *et al.*, 1998).

On a lipid weight basis the concentrations found are remarkably similar regardless of species

---

[1] The amount of the pesticide DDT in the environment is often given as the sum of the technical product DDT and its persistent bioaccumulative degradation products DDE and DDD. The sum of these compounds is here denoted ΣDDT.

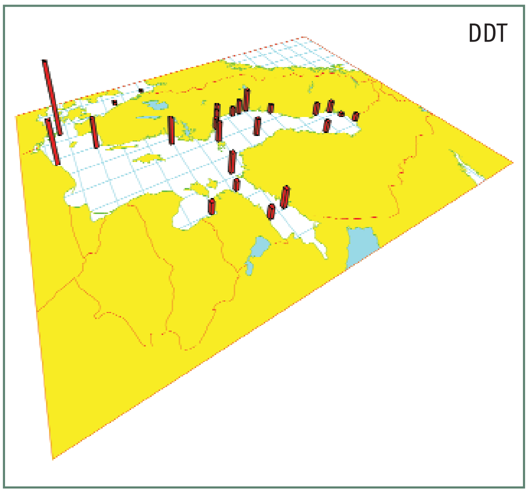


Figure 6.22
Spatial distribution in the concentration of ΣDDT in herring muscle (μg/g lw). The concentration is lowest in the northern part of the Bothnian Bay (0.061 μg/g lw) and highest outside the German coast (1.2 μg/g lw).

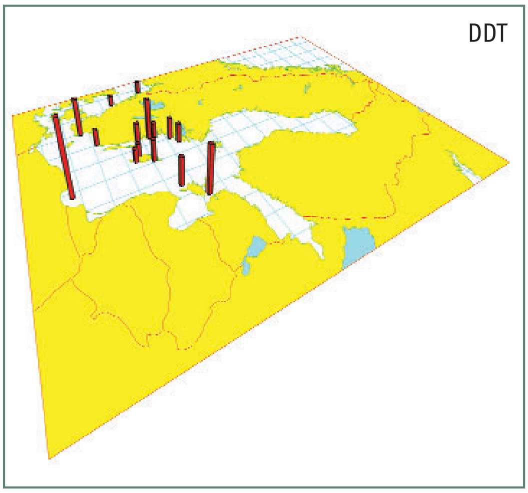


Figure 6.23
Spatial distribution in the concentration of ΣDDT in blue mussel. The concentration is lowest in the Kattegat (0.028 μg/g lw) and highest in the Gulf of Gdansk (0.20 μg/g lw).

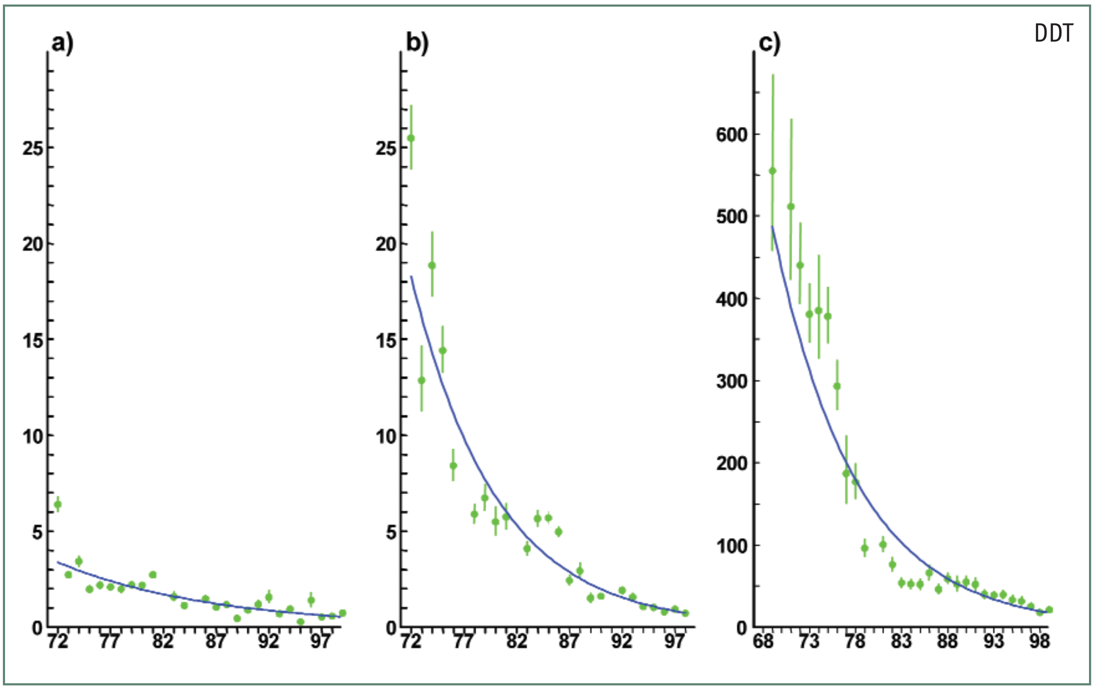


Figure 6.24
ΣDDT concentration (μg/g lw) in herring muscle over the period 1972–98 in
a) the southern Bothnian Sea and
b) the southern Baltic Proper, as well as in
c) guillemot egg from the central Baltic Proper over the period 1968–98.
The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated.

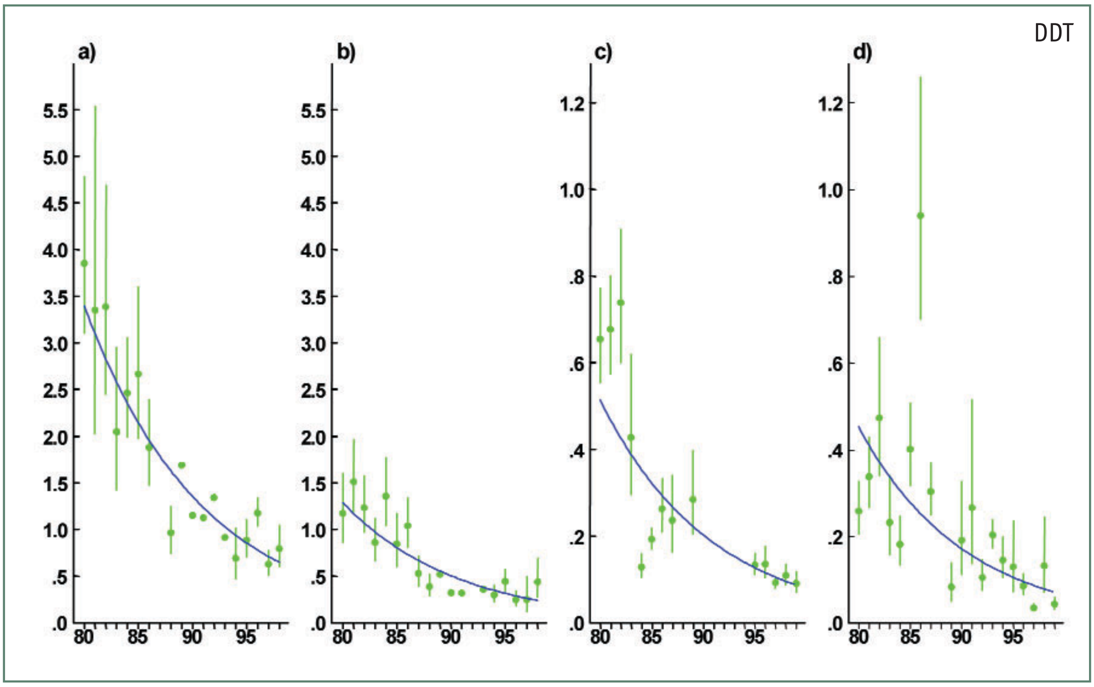


Figure 6.25
ΣDDT concentration (μg/g lw) in fish over the period 1980-98:
a) cod liver from the Baltic Proper,
b) cod liver from the Kattegat,
c) perch muscle from the Gulf of Bothnia and
d) perch muscle from the Baltic Proper.
The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated.

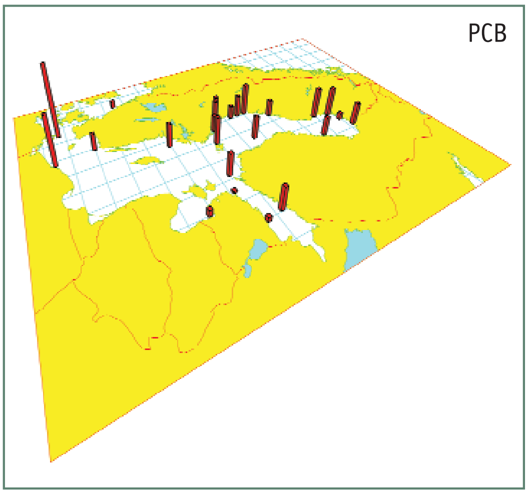


Figure 6.26
Spatial distribution in the concentration of PCB in herring muscle. The concentration is lowest outside the Gulf of Finland (0.12 µg/g lw) and highest outside the German coast (4.0 µg/g lw).

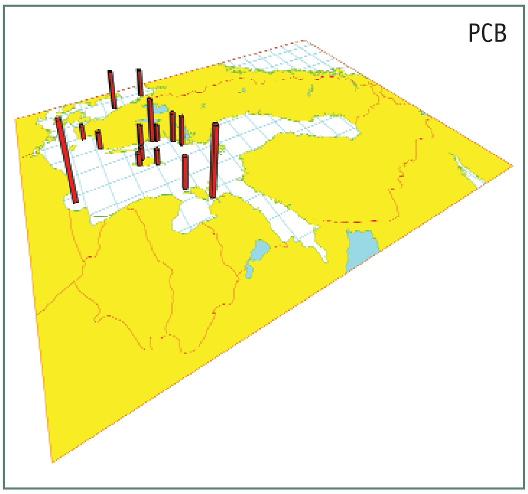


Figure 6.27
Spatial distribution in the concentration of PCB in blue mussel. The concentration is lowest off Gotland (0.16 µg/g lw) and highest in the Gulf of Gdansk (0.93 µg/g lw).

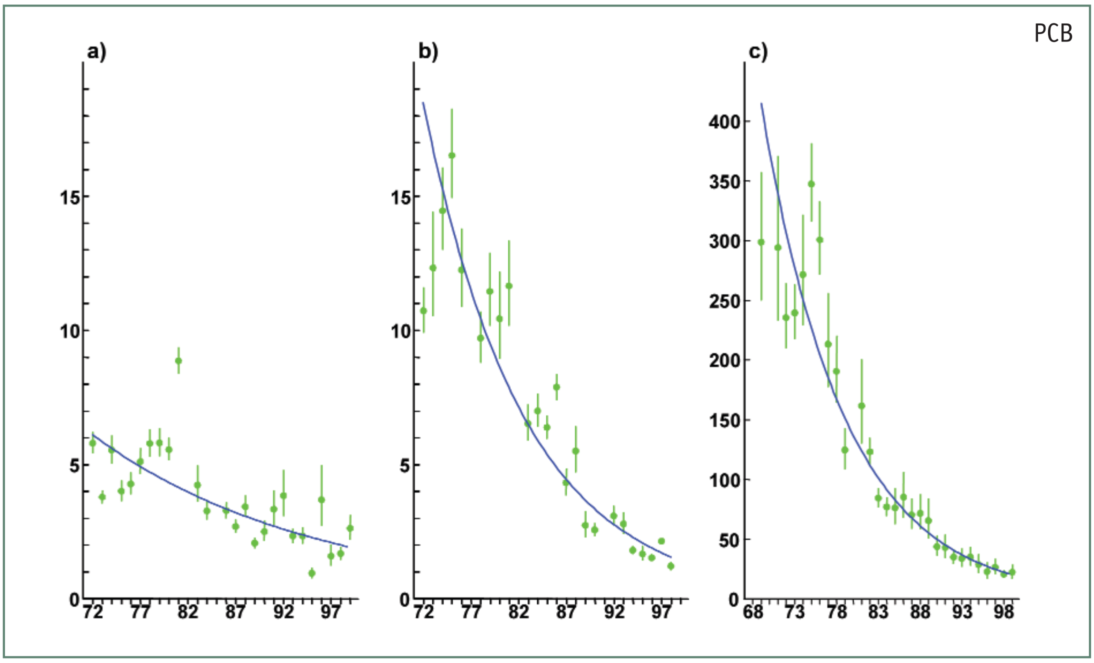


Figure 6.28
PCB concentration (µg/g lw) in herring muscle over the period 1972–98 in
a) the southern Bothnian Sea and
b) the southern Baltic Proper, as well as in
c) guillemot egg from the central Baltic Proper over the period 1968–98.
The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated.

and position in the food web. Thus the concentrations found in the planktivorous herring (0.06–1.2 µg/g lw) are quite similar to those found in the piscivorous perch (0.071–1.0 µg/g lw). Moreover, the concentrations are also often quite similar to those in blue mussel, *Macoma sp.* and *Saduria sp.* (0.036–0.39 µg/g lw). Slightly higher concentrations are found in salmon (1.3–2.1 µg/g lw) (Aune *et al.*, 2000).

That concentrations are often similar irrespective of position in the food web except for the lipid-rich salmon contradicts the models suggesting biomagnification of lipophilic compounds in the food web (Gobas *et al.*, 1993; Rolff *et al.*, 1993). The results indicate that the partition equilibrium process is more important than biomagnification via the food web in explaining bioconcentration in aquatic biota, a hypothesis further supported by a recent field study in fish (Olsson *et al.*, 2000). In species lacking gills and hence not exposed to partition equilibrium processes, e.g. guillemot and seals, consumption of aquatic organisms from the Baltic is accompanied by significant biomagnification. Thus the concentration of xDDT in guillemot egg lipids is about 20 µg/g lw as compared with less than 1 µg/g lw in the guillemot's main food, the herring.

PCB

The concentration of PCB[2] outside the German coast herring muscle ranges from 0.12 to 4.0 µg/g lw. The lowest concentrations are found in the northern Bothnian Bay, along the Estonian coast and in the Kattegat and the highest outside the German coast. The spatial distribution does not reflect any polarization along the north-south axis,

---

[2] Polychlorinated biphenyl (PCB) is a mixture of different chlorinated biphenyl congeners (CBs) that have been used in various industrial products. The total amount or the sum of these congeners is normally used to report PCB concentrations. Sometimes, however, specific congeners are reported instead. To facilitate the comparison of concentrations, the concentration of the persistent congener CB-153 has been divided by a constant (0.11) to provide an approximate value of the total amount of PCB (Atuma *et al.*, 1996).

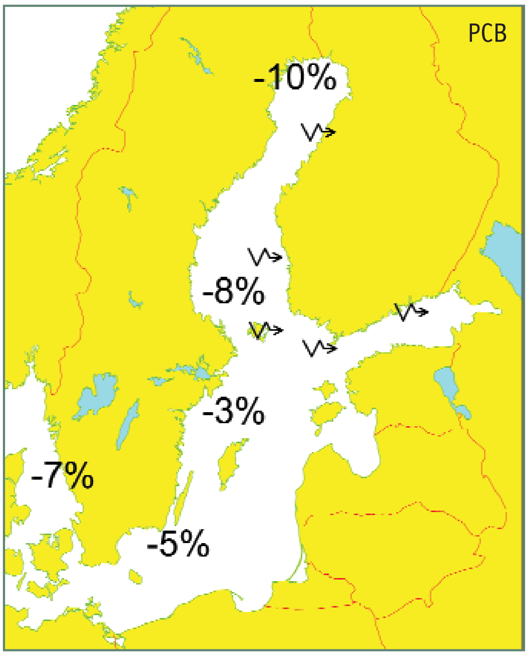


Figure 6.29
Annual percentage decrease in PCB concentration in herring muscle in different parts of the Baltic Marine Area determined from long-term monitoring data (>15 years). The arrow indicates the lack of any significant change.

but merely the influence of local sources (Figure 6.26). The PCB concentration is even higher in demersal cod from the Kattegat (3.8 µg/g lw) than from the Baltic Proper (Arkona 2.0 µg/g lw and SE Gotland 1.8 µg/g lw). The importance of local sources is further supported by studies of the coastal species blue mussel, perch and eelpout. Biota PCB concentrations in the Kattegat are often similar to those in the Baltic Proper, e.g. in blue mussel (Figure 6.27), where the highest concentration (0.93 µg/g lw) is found along the Polish coast and the lowest in the central Baltic Proper (0.16 µg/g lw).

Biota PCB concentrations have been decreasing for some time. The longest time series going back to the late 1960s/early 1970s thus reveal an annual decrease of about 4% in herring from the southern Bothnian Sea and about 9% in herring and guillemot egg from the Baltic Proper (Figure 6.28). Shorter time series for the pelagic herring starting in the early 1980s or later also reveal an annual decrease ranging from about 10% in the Bothnian Bay to only about 3% in the Baltic Proper and about 7% in the Kattegat (Figure 6.29). The concentration of PCB in the desmeral cod liver is decreasing by about 9% annually in the Kattegat and the Baltic Proper, while that in the coastal perch is decreasing about 7% annually in the Bothnian Bay and 11% annually in the Baltic Proper.

The decrease in concentration over the last 10 years is statistically significant in herring from the Gulf of Bothnia and the Kattegat. In the Baltic Proper, however, the decrease has ceased. Moreover, no statistically significant decrease can be detected in cod liver over the last 10 years. The relative amount of less chlorinated and less persistent PCB congeners increased in Baltic herring samples during the 1990s, thus indicating recent discharge of less persistent and more volatile congeners to the Baltic environment (Bignert *et al.*, 1999) (Figure 6.30).

No species differences in the long-term decrease (>15 years) are apparent and PCB is decreasing at the same rate in the Baltic as indicated by long-term monitoring studies in Swedish fresh waters going back to the late 1960s (Olsson and Reutergårdh, 1986; Bignert *et al.*, 1998) indicating a general decrease in atmospheric deposition of PCB.

On a lipid weight basis the PCB concentration is remarkably uniform regardless of species except for the lipid-rich salmon, where the concentration is often higher. The concentration found in the planktivorous herring (0.12–2.8 µg/g lw) is quite similar to that in the piscivorous perch (0.65–1.7 µg/g lw) or the eelpout (1.0–2.5 µg/g lw). The

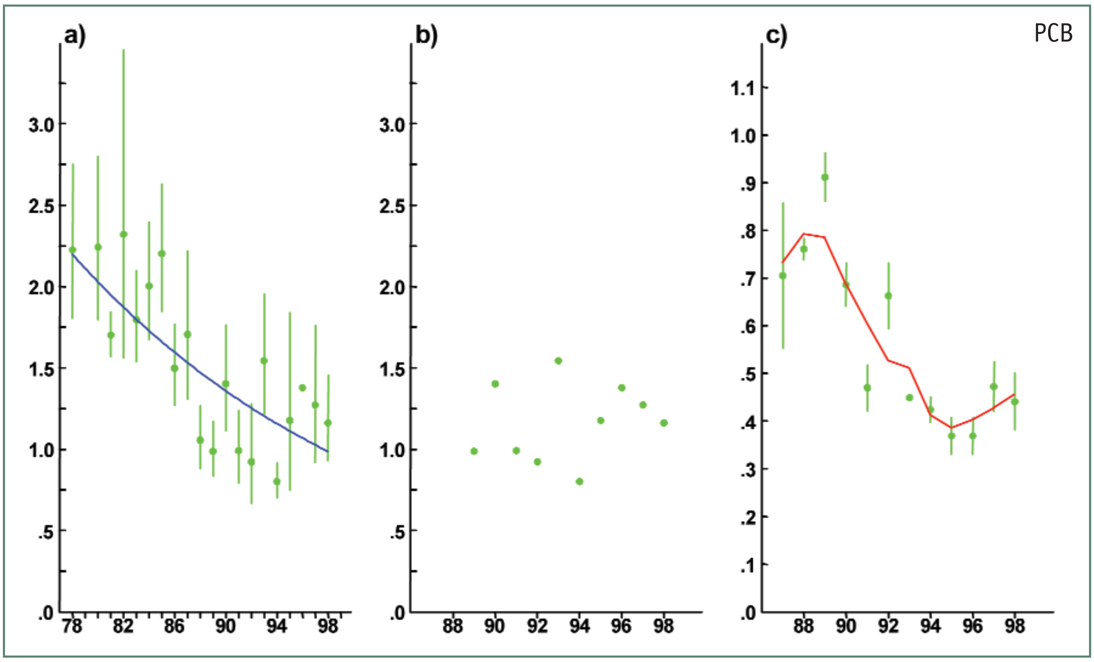


Figure 6.30
PCB concentration in herring muscle in the northern Baltic Proper over
a) the entire period 1978–98 and
b) the period 1989–98.
Taking the last 10 years alone it seems that the trend towards decreasing PCB concentrations seen earlier has now ceased. The concentration ratio for the less persistent congener CB-118 and the more persistent congener CB-153 is shown for the period 1987–98 in c). Statistical analysis using the smoother technique (red line) reveals a significant relative increase in the less persistent congener in the last few years indicating recent discharge of PCB. The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated.

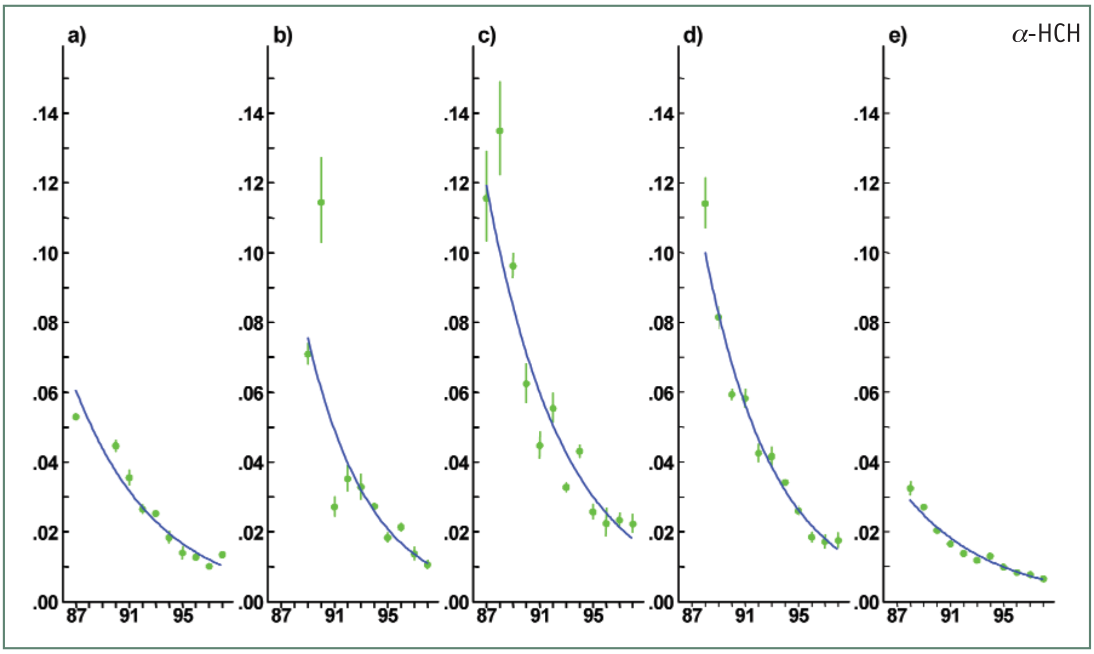


Figure 6.31
α-HCH concentration in herring muscle (µg/kg lw) collected along the Swedish coast during the 1980s and 1990s:
a) the Bothnian Bay,
b) the southern Bothnian Sea,
c) the northern Baltic Proper,
d) the southern Baltic Proper and
e) the Kattegat.
The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated.

concentration measured in cod is higher, ranging from 1.8 to 3.8 µg/g lw. The concentration in blue mussel, *Macoma sp.* and *Saduria sp.* (0.16–1.2 µg/g lw) is similar to that in the herring. The PCB concentration in the lipid-rich predator the salmon from 8 different locations (Atuma *et al.*, 1998) ranged from 1.3 to 4.0 µg/g lw. These observations partly contradict the suggested biomagnification of lipophilic compounds in the aquatic food web (see Section 6.3.2.1). PCB is biomagnified in the terrestrial part of the Baltic food web, however. Thus the concentration of PCB in guillemot egg lipids is about 30 µg/g lw as compared with about 1 µg/g lw in the guillemot's main food, the herring.

HCH[3]

The concentration of α-HCH and lindane (γ-HCH) in herring muscle range from 0.006 to 0.030 µg/g lw and 0.008 to 0.034 µg/g lw, respectively. The concentration of α-HCH appears to be higher in the southern Baltic Proper and lower in the Kattegat, whereas the concentration of lindane seems to be the same in the Kattegat and the central parts of the Baltic Proper (0.017 µg/g lw). With cod, somewhat higher concentrations of lindane are found in the Arkona Sea (0.029 µg/g lw) than in the central Baltic Proper (0.017 µg/g lw) and the Kattegat (0.018 µg/g lw).

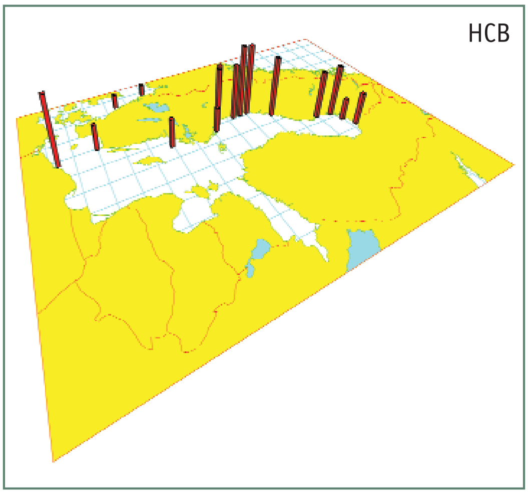
The annual decrease in α-HCH concentration ranges from 12 to 22% in herring, cod, perch and blue mussel, with no species-specific or area-specific variation being evident (Figure 6.31). With lindane the annual decrease is more varied, ranging from 6 to 8% for blue mussel, herring and cod from the Kattegat and 8 to 24% for herring, cod, blue mussel, perch and guillemot egg in the Baltic Proper and the Gulf of Bothnia. That the concentration of lindane is decreasing more slowly in the western part of the Baltic Marine Area seems to be attributable to extensive use of lindane in south-western Europe in recent times.

There are no obvious species difference in HCH concentration expressed on a lipid-weight basis. Thus lindane concentration in the predator species cod (0.018–0.029 µg/g lw), perch (0.005–0.018 µg/g lw), eelpout (0.006–0.016 µg/g lw) and salmon (0.010–0.027 µg/g lw) is roughly the same as in herring (0.008-0.019 µg/g lw) and blue mussel (0.003–0.027 µg/g lw). With HCH too, biomagnification seems to be negligible in the aquatic part of the Baltic food web. In the terrestrial part of the food web the concentration of both lindane and α-HCH is slightly lower in guillemot egg lipids than in the guillemot's main food, the herring. This is opposite to the case with DDT and PCB, where the concentration is much higher in the predator. β-HCH biomagnifies, however, the concentration in guillemot egg lipids being about 15–20 times higher (0.30 mg/kg lw) than in herring (<0.020 mg/kg lw).

---

[3] Lindane is the commercial name of the γ-isomer of HCH (hexachlorocyclohexane) that is still used in some European countries as a pesticide. Production of this pesticide generates extensive amounts of by-products. More than 80% of the industrial product HCH consists of the α- and β-isomers. If not removed during the production process, these persistent and bioaccumulative isomers are also spread into the environment together with the active pesticide. Since the 1970s, most European countries have used the pure γ-isomer lindane. Since the beginning of the 1990s, this is also true for a number of countries in the developing part of the world.

Figure 6.32
Spatial distribution in the concentration of HCB in herring muscle. The concentration is lowest in the Kattegat and Skagerrak (0.010 µg/g lw) and highest in the Bothnian Sea (0.056 µg/g lw).



HCB[4]

The HCB concentration in herring muscle ranges from 0.011 to 0.056 µg/g lw. The concentration is slightly higher along the Swedish coast of the Gulf of Bothnia than indicated by Finnish data from the same area, as well as along the Swedish coast of the Baltic Proper and in the Kattegat. Enhanced concentrations are also seen along the Polish coast (Figure 6.32). The higher concentrations along the Swedish coast of the Gulf of Bothnia may be attributable to fairly high industrial activity in this part of Sweden.

The annual rates at which the concentration of HCB is decreasing ranges from 6 to 18% (Figure 6.33). The decrease is generally lower in the very north and in the Kattegat than in the Baltic Proper. However, where the annual decrease is low, the concentration is also usually low.

There are very few clear indications of biomagnification of HCB in the Baltic aquatic food web when measured on a lipid weight basis. The concentrations found in salmon (0.044–0.11 µg/g lw) are slightly higher than in the planktivorous herring (0.011–0.056 µg/g lw). In cod (0.010–0.038 µg/g lw), perch (0.005–0.067 µg/g lw) and eelpout (0.009–0.031 µg/g lw) the concentration is similar to that in herring. In blue mussel and *Macoma sp.*, slightly lower concentrations (0.002–0.007 µg/g lw) are found. In the terrestrial part of the food web the concentration of HCB is higher in guillemot egg lipids (0.85 µg/g lw) than in herring.

Dioxins[5]

The dioxin concentration (expressed as the TEQ value[6]) in 7 pools of herring muscle along the Finnish coast ranged from 165 to 329 pg/g lw (Vartiainen *et al.*, 1997). The concentration was highest in the inner part of the Gulf of Finland. On a wet weight basis this corresponds to 2.9–24 pg/g. The highest wet weight concentrations were found in fat specimens of old age from the Bothnian Sea and the Gulf of Finland. The samples studied consisted of pools of herring of age classes 2–18. The TEQ values for coplanar PCBs were of the

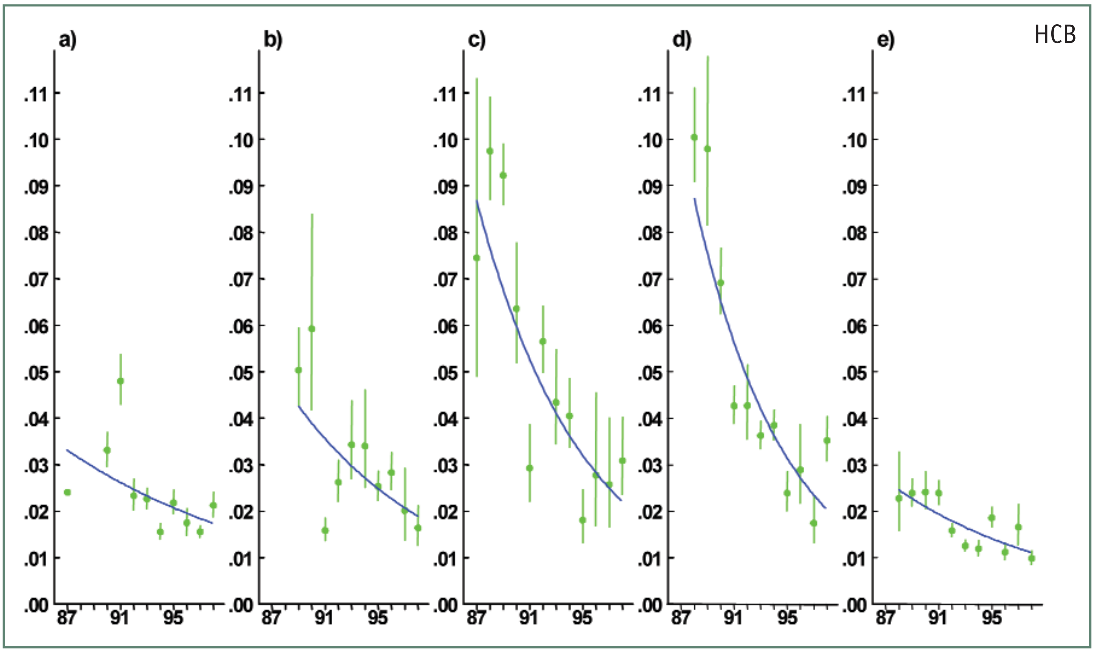
---

[4] HCB was originally used as a fungicide. Since the 1970s, though, most of the HCB introduced into the environment derives from a by-product of the chlorine industry and from inappropriate combustion of certain chlorine-containing wastes.

[5] Dioxins are contaminants produced during industrial processes as well as at combustion of wastes. The term dioxins is used here as a common name for a number of congeners of polychlorinated dibenzo-*p*-dioxins (PCDD) and polychlorinated dibenzofurans (PCDF).

[6] Dioxin quantities are expressed here as the TEQ value (Toxic Equivalent). The coplanar molecules of dioxins block a specific receptor-protein (Ah-receptor), thereby causing a series of various toxic effects on the organism. The different congeners have different capability to block the Ah-receptor. The most toxic congener is the 2,3,7,8-tetrachloro dibenzo-*p*-dioxin (TCDD). The toxicity of other congeners is related to the toxicity of TCDD. A measure of the total toxic capability of a dioxin congener in a tissue is the TEQ value, which is the congener concentration multiplied by its specific Toxic Equivalent Factor (TEF). The addition of all TEQ values from the different congeners indicates the total toxicity of dioxins in the tissue. Also coplanar PCB congeners block the Ah-receptor: Their toxicity can be estimated in a similar way as for the dioxins.

Figure 6.33
HCB concentration in herring muscle (µg/kg lw) collected along the Swedish coast during the 1980s and 1990s:
a) the Bothnian Bay,
b) the southern Bothnian Sea,
c) the northern Baltic Proper,
d) the southern Baltic Proper and
e) the Kattegat.
The values are annual geometric means with the 95% confidence interval shown. If statistically significant, the log-linear regression is indicated.



same magnitude as those for the dioxins. As for dioxins, moreover, the concentration (ww) was highest in samples from the Bothnian Sea and the Gulf of Finland. Swedish data on dioxin concentrations (TEQ) in individual young herring from the Bothnian Bay range from 8.5 to 57 pg/g lw, with the corresponding values for coplanar PCB being 6.8 and 35 pg/g lw (mean 20 pg/g lw). In the southern Baltic Proper the corresponding values are 1.9–56 pg/g lw (mean 22 pg/g lw) and 14–105 pg/g lw (mean 43 pg/g lw), respectively. In the Kattegat, the corresponding values are 4–16 pg/g lw (mean 8 pg/g lw) and 8.5–40 pg/g lw (mean 18 pg/g lw), respectively (de Wit *et al.*, 2000). The higher concentrations found in the Finnish material are probably mainly attributable to the higher age classes analysed.

The temporal trend is slower for dioxins than for PCB and, as for PCB, the dioxin concentration has levelled off during the last 10 years (Figure 6.34). In guillemot eggs from Stora Karlsö covering the period 1969–98 the TEQ values for dioxins have decreased from about 3.5 mg/kg lw in the late 1960s to about 1 mg/kg lw at the end of the 1980s. Since then, no further decrease has occurred. Swedish annual data on herring collected in the southern part of the Bothnian Sea, the southern part of the Baltic Proper and in the Kattegat since 1990 fail to show any evidence of a decrease in dioxin concentrations (Anonymous, 2000).

The available data do not permit a thorough study on the rate at which dioxins might bioaccumulate in the aquatic food web. However, TEQ values are about 50-fold higher in guillemot egg lipids than in herring. Although the Baltic grey seal also feeds on herring, studies from the early 1990s indicate that biomagnification of dioxins does not occur in this species (Bergek *et al.*, 1992).

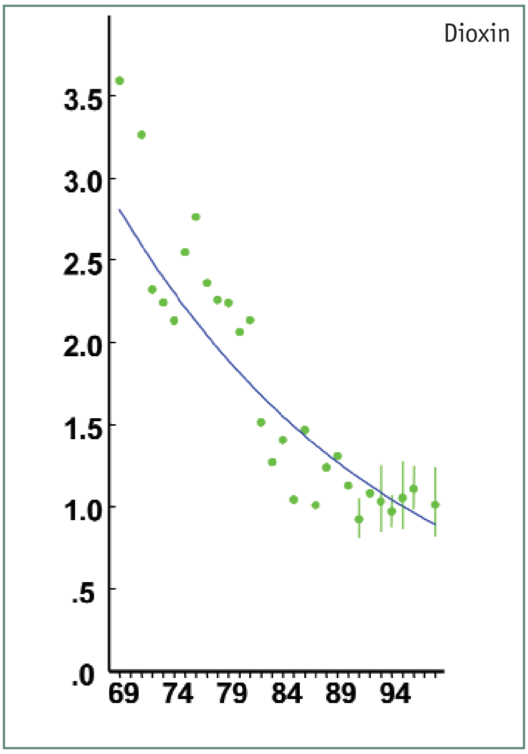


Figure 6.34
Dioxin concentration (TEQs, µg/kg lw) in guillemot egg from Stora Karlsö, west Gotland in the central Baltic Proper, over the period 1969–98. In years when individual eggs (normally 10 a year) have been analysed, the 95% confidence interval is given for the annual geometric mean. Other values are annual pooled values for 10 eggs. If statistically significant, the log-linear regression is indicated.

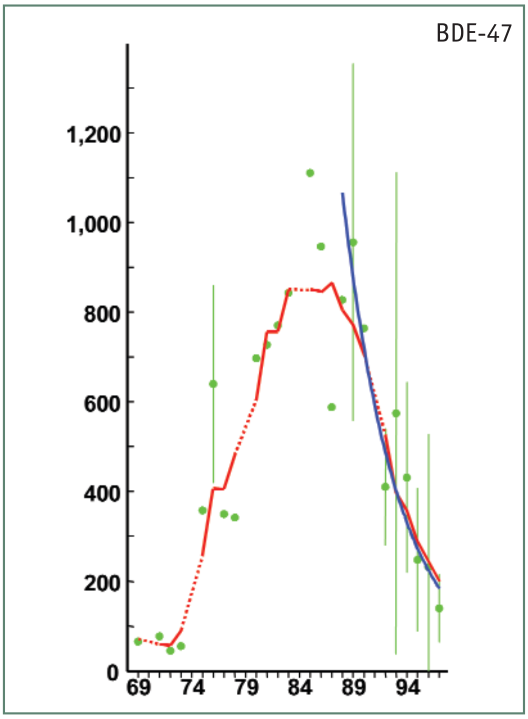


Figure 6.35
Concentration of the polybrominated compound BDE-47 (µg/kg lw) in guillemot egg from Stora Karlsö, west Gotland in the central Baltic Proper, over the period 1969–98. In years when individual eggs (normally 10 a year) have been analysed, the 95% confidence interval is given for the annual geometric mean. Other values are annual pooled values for 10 eggs. If statistically significant, the log-linear regression is indicated. The red line indicates the trend determined using the smoother technique while the blue line indicates the log-linear regression over the last 10 years (Data from Sellström et al., 1999).

Polybrominated compounds[7]

As yet, very little information is available concerning spatial or species differences as regards polybrominated compounds in the aquatic food web in the Baltic Marine Area.

Concentrations of BDE47 (2,2´,4,4´-tetrabromodiphenyl ether) and BDE99 (2,2´,4,4´,5-pentabromodiphenyl ether) increased in guillemot eggs from the Baltic Proper in the early 1970s, peaked in the mid 1980s, and decreased in the 1990s (Sellström *et al.*, 1999). The temporal trend for BDE47 is presented in Figure 6.35. Data on human breast milk for the corresponding period indicate a continuous increase since the early 1970s (Meironyté *et al.*, 1999), while data from Swedish freshwater fish indicate that the concentration of the PBDE has stabilized since the end of the 1980s (Kirkegaard *et al.*, 1999). The concentrations of hexabromocyclododecane (HBCD), another polybrominated flame retardant, has increased over the entire period according to a recent report (Kirkegaard *et al.*, 1999), including in guillemot

---

[7] Various polybrominated compounds have been used as flame retardants in a number of products including high-tech products such as computers and other electrical equipment. The international use of the polybrominated flame retardants has increased since the beginning of the 1970s and world-wide use was estimated to be 600,000 tonnes in 1992 (OECD 1994). During the 1990s, increasing concern has been expressed for this group of contaminants and awareness of possible negative effects on wildlife and humans has developed. One important group of compounds among the polybrominated flame retardants are the polybrominated diphenyl ethers (PBDEs), which are lipophilic.

eggs. The temporal trend for this group of compounds is fairly uncertain.

Organotin compounds[8]

Recent data on Baltic marine birds and mammals clearly demonstrate the presence of tributyl tin (TBT) in the Baltic Marine Area. Thus, Kannan and Falandysz (1997) reported liver concentrations of 35–4,600 mg/kg ww in Baltic sea birds and 18–27 mg/kg ww in neonatal harbour porpoises. In Danish marine waters, the tin (Sn) concentration in blue mussel ranged from 2 to 94 mg/kg ww (Markager *et al.*, 1999). The concentrations correlate to the activity of ship traffic, with the highest concentrations being found in inner coastal waters. The concentration is generally highest for tributyl tin, lower for dibutyl tin and lowest for monobutyl tin.

6.4. Effects of contaminants on Baltic Biota

Convenor: Mats Olsson

Co-convenor: Anders Bignert

By: Mats Olsson, Anders Bignert, Anders Bergman, Björn Helander and Tero Härkönen

As early as in the 1950s and 1960s it became apparent that the populations of a number of fish predators inhabiting the Baltic Marine Area were decreasing – including the ringed seal (*Pusa hispida*), the grey seal (*Halichoerus gryphus*), the harbour seal (*Phoca vitulina*), the otter (*Lutra lutra*) and the white-tailed sea eagle (*Haliaeetus albicilla*). In the late 1960s it further became apparent that the Baltic Marine Area was polluted (Jensen *et al.*, 1969), and the earlier observations of decreasing predator populations were soon correlated to this pollution.

As contaminants bioaccumulate in fish inhabiting polluted waters, they pose a certain risk to organisms exploiting the latter as a food resource. In the Baltic Marine Area a greater risk is generally posed by pelagic fatty species than by the leaner coastal species, irrespective of whether consumed by seals, predatory birds or man, and irrespective of whether the contaminant is DDT, PCB or dioxins. These contaminants detrimentally affect biota due to their toxicity. Being lipophilic they accumulate in body lipids. Since the Baltic pelagic food web is rich in lipids whereas the coastal zone fauna often is leaner, concentrations of lipophilic compounds on a fresh weight basis are often higher in pelagic fish species than in coastal species. Thus while the muscle lipid concentration is 3–10% in herring (*Clupea harengus*) and 10–20% in salmon (*Salmo salar*), it is often less than 1% in coastal species such as perch and pike.

6.4.1. Detrimental effects on Baltic fauna and consumer organisms

A common effect of contaminants on the Baltic fauna is reproductive disturbance in higher vertebrates (Helle *et al.*, 1976; Olsson *et al.*, 1975; Helander *et al.*, 1982; Helander and Bignert, 1992). With some species such as the grey seal and the ringed seal, a disease complex has been identified that is probably related to adrenal lesions (Bergman and Olsson, 1986), although lesions are also found in the skin, claws, bone, kidneys, intestines, blood vessels and uterus.

In the 1970s, the impact of contaminants together with former heavy seal hunting had reduced the ringed seal population to about 2,000 individuals (Helle, 1986), the grey seal population to about 2,000 individuals (Almkvist, 1986) and the Baltic Proper harbour seal population to slightly more than 100 individuals (Helander and Bignert, 1992). These factors made the seal, the otter and the white-tailed sea eagle rare species throughout the Baltic Marine Area.

Contaminants have also been suspected of harming the populations of a number of other species in the past, including fish and invertebrates. This was often attributed to local discharges of contaminants, however, and apart from the Baltic salmon the effects seen in the Baltic Marine Area have mainly concerned top predators.

A disease complex in Baltic salmon entailing a high prevalence of fry mortality has been described and named M 74 (For a review see Bengtsson *et al.*, 1999). While it has been suggested that contaminants could have rendered the salmon susceptible to the disease, this has not yet been confirmed.

Significantly enhanced levels of the detoxification enzyme EROD have been detected in Baltic fish over the period 1988–98. At the end of that period, EROD activity was 2–3 times higher than at the beginning (Åjders *et al.*, 1999), thus indicating that the fish have been exposed to something that has induced production of this detoxification enzyme. Since the concentrations of contaminants known to induce EROD production have concomitantly decreased, some unknown contaminants would seem to be responsible.

In Danish waters, imposex has been detected in red whelk (*Neptunea antiqua*) and common whelk (*Buccinum undatum*) (Markager *et al.*, 1999). Imposex is attributable to chemicals such as trib-

---

[8] TBT (tributyl tin) has been used as an antifouling agent in hull and fishing gear paints. It has also been used in plastics as a stabilizer in PVC, and as an industrial catalyst in the production of polyurethane and silicones.

utyl tin that cause disruption of the endocrine system (endocrine disruptors). The Danish findings are taken as evidence of an effect of TBT on coastal biota.

Infants born to Swedish fishing families with a diet high in fish from the Baltic Marine Area appear to have a slightly lower birth weight than infants born to other Swedish families (Rylander *et al.*, 1995). Due to the high concentrations of contaminants in fish from Baltic waters, the Swedish authorities have introduced restrictions on their consumption. The concentrations of dioxins and compounds having a similar toxic effect, namely the coplanar polychlorinated biphenyls (some of the congeners found in commercial PCB), are still too high in herring and salmon, and the Swedish Food Administration has thus recommended women of reproductive age not to consume large amounts of these species and pregnant women not to consume them at all (Anonymous, 1996).

6.4.2. Development of fish predator populations

The census work on seal and white-tailed eagle started during the 1970s. The census was originally based on single-year observations, but has been carried out annually since the mid 1980s. Census work on the Baltic guillemot started even earlier in the 1960s. The size of the otter population has only been determined occasionally. See also Chapter 10.

Grey seal

The grey seal inhabits the entire Baltic Marine Area but is distributed mainly in the central and northern parts. The population in the Kattegat only numbers about ten individuals. The population in the northern parts (north of latitude 59°) has increased more than that in the southern part (south of latitude 59°) (Figure 6.36). Over the period 1982–97, moreover, the population increased by about 12% annually in the northern part but only by about 5% in the southern part (Helander, 1998). There are reasons to believe that these population increases are overestimates attributable to demographic factors or behavioural alterations rendering the seals gradually less shy and easier to count during the course of time since they became protected. Over the last ten years, the annual increase is only slightly less than 8% in the northern parts of the Baltic Marine Area and slightly less than 5% in the south (Helander and Härkönen, 1999). The reason for the lower increase in the south is unknown, but could be attributable to residual effects of contaminants on reproduction (note that the concentrations of PCB and dioxins in biota are no longer decreasing in the Baltic Proper) or to a higher relative incidence of fishery mortality among young grey seals that become entangled in fishing gear.

In 1998, about 3,200 seals were counted along the Swedish coast and about 5,000 in Finnish and Swedish waters (Helander and Härkönen, 1999). Surveys in 1999 including Estonian and Russian waters indicated a total population of 7,600 seals, although this is probably an underestimate. Estimates made using three different approaches to counting the seals each indicate a total population in the order of 10,000 in the entire Baltic Marine Area.

In contrast, a total population of at least 100,000 grey seals is estimated to have existed at

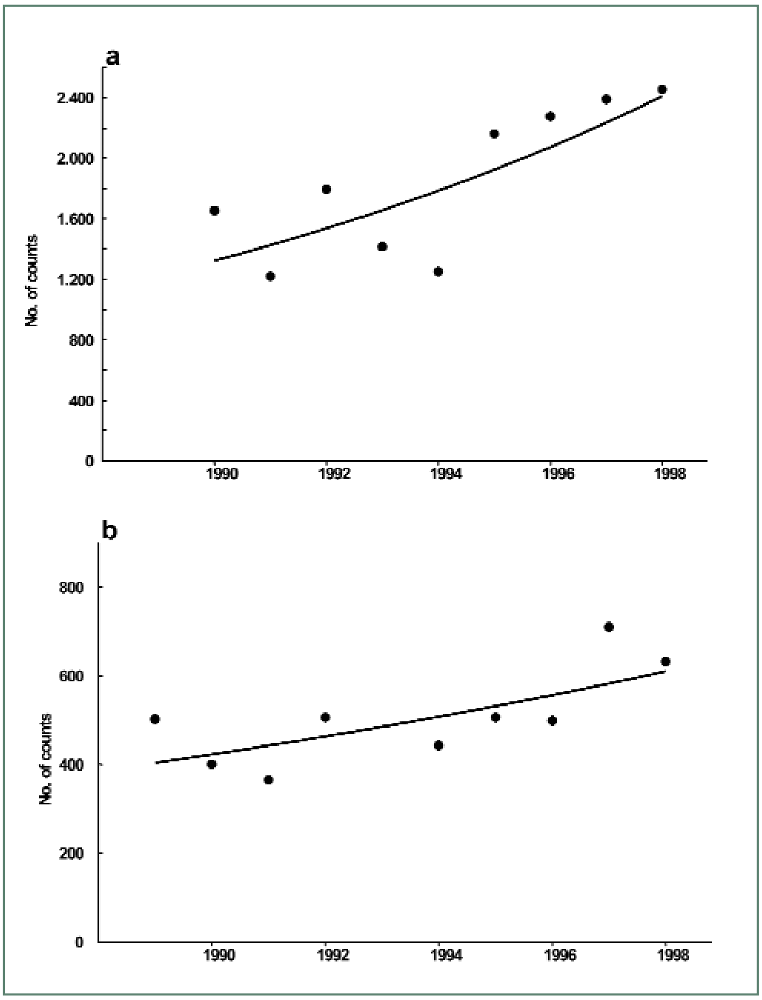


Figure 6.36
Number of grey seal observed during the annual census along the Swedish Baltic coast
a) in the waters north of 59°N and
b) south of 59°N.

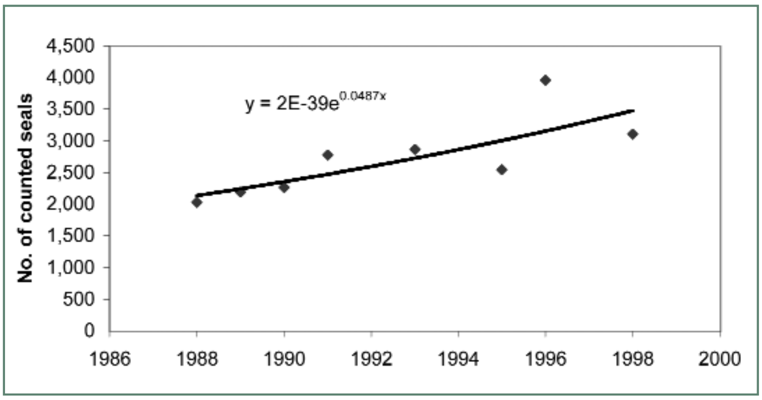
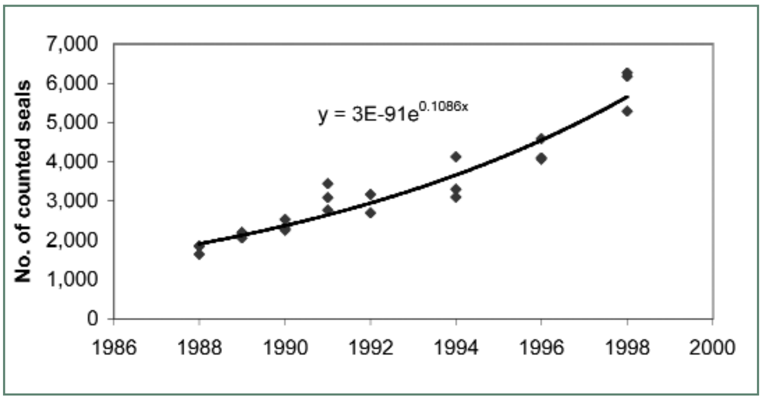


Figure 6.37
Number of ringed seal observed during the annual census in the Gulf of Bothnia.



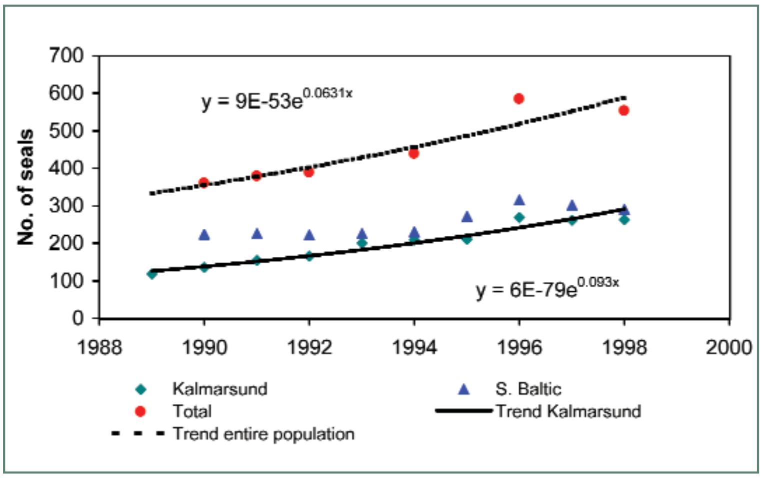


Figure 6.38
Number of harbour seal observed during the annual census in the Kattegat

Figure 6.39
Number of harbour seal observed during the annual census in the Baltic Proper.
l Entire Baltic Proper,
m Eastern subpopulation in Kalmarsund.

the beginning of the 20th Century based on hunting statistics and a model for recalculating the earlier population size (Hårding and Härkönen, 1999). Hunting was the main cause of the population decrease up to the 1940s while reproductive failure and disease has been responsible for the decrease since the 1960s.

Ringed seal

The ringed seal is found in the northern parts of the Gulf of Bothnia and in the eastern parts of the Baltic Proper, the Gulf of Finland and the Gulf of Riga, and there is a small population in the Archipelago Sea. Overall, the ringed seal population is increasing by about 5% annually (Figure 6.37). As with the grey seal, the first reports on the size of the ringed seal population were based on single-year counts (Helle, 1986). Since the late 1980s, a census has been carried out on a regular annual basis in the Gulf of Bothnia (Härkönen and Lunneryd, 1992). The lower increase of the ringed seal population relative to that of the grey seal population is attributable to reduced fecundity caused by continued uterine occlusion in old seals and new development of uterine occlusion in young seals (Helle and Nyman, 2000).

Recent census figures indicate that there are about 4,000 ringed seals in the Gulf of Bothnia, 200-300 in the Gulf of Finland and about 1,400 in the Gulf of Riga (Härkönen *et al.*, 1998).

Based on recent modelling of hunting statistics in Finland, Sweden, Estonia and Russia it is estimated that there were just over 200,000 ringed seals in the Baltic Marine Area at the beginning of the 20th Century (Hårding and Härkönen, 1999).

Harbour seal

The harbour seal is found in the Kattegat and in the southern parts of the Baltic Proper, mainly along the Danish and Swedish coasts. The population in the Baltic Proper numbers about 600 (Helander and Härkönen, 1999), while that in the Kattegat numbers about 4,000.

The rate of population increase varies depending on the time period investigated because an outbreak of phocine distemper in 1998 caused gross mortality, killing about 60% of the harbour seals along the European coasts (Heide-Jørgensen *et al.*, 1992 a and b). The disease was first observed in the Kattegat area, and thereafter spread to the west and north as well as to the south into the Baltic Proper, where it was affecting the colonies in the southwestern parts of the area. The colonies located in the inner part of the Baltic Proper were not affected – or at least the virus did not cause gross mortality.

Since the virus outbreak the populations have recovered (Härkönen *et al.*, 2000). In the Kattegat area, for example, the harbour seal population has increased by about 11% annually (Figure 6.38), a rate elevated as a result of demographic changes following the 1988 epizootic. It is nevertheless lower than the 16% annual increase recorded in another population hit by the epizootic – the Skagerrak population. In the Baltic Proper the population at Kalmarsund has increased slightly less, by about 7% annually (Figure 6.39).

Based on modelling of hunting statistics it is estimated that the harbour seal population in the Baltic Proper numbered 5,000 at the beginning of the 20th Century (Hårding and Härkönen, 1999).

The harbour seals in the inner part of the Baltic Proper, Kalmarsund, are genetically distinct from all other populations, while those inhabiting Måkläppen and the Danish locations seem to be closely related to the Kattegat population (Härkönen and Hårding, 2000). The harbour seals in the Baltic Proper should thus be regarded as a population of special concern.

Health of the Baltic seals

Recent studies indicate that health conditions among Baltic grey seals are improving, although as discussed below, some problems still remain and some are even worsening (Bergman, 1999). The disease complex suffered by ringed seal and grey seal reduces fecundity and causes metabolic, hormonal and immunological disorders (Bergman and

Olsson, 1986). In a recent study, Bergman (1999) found that the prevalence of some of these disorders has decreased while that of others has increased. In particular, improvement has been seen as regards female reproductive health, e.g. occlusions of the uterine horn and pregnancy rate. The prevalence of the benign uterine tumours (leiomyomas) is still high but improvement has been seen with respect to cortical hyperplasia and adrenal adenoma. These disorders are nevertheless still found in animals born after the dramatic decrease in DDT, PCB and dioxin concentrations in biota in the 1970s. The prevalence of skull bone lesions, which were frequent in seals examined during the 1960s to 1980s (Bergman *et al.*, 1992), seem to have decreased in animals born after the 1970s, however (Olsson and Bergman, 2000).

Of serious concern is the finding that the prevalence of intestinal ulcers has increased among grey seals over the last 15 years – from about 10% among young grey seals (<3 years of age) examined in 1977–86 to more than 50% in 1987–96 (Figure 6.40). As much as 7% of the autopsied animals had died due to colonic ulcers. Although associated with hookworm (*Corynosoma sp.*) infections, the ulcers are believed to result from immunosuppression. Studies on harbour seal in the Netherlands support this interpretation and indicate the presence of immunosuppressive agents in fish from the Baltic Marine Area compared to fish from the North Atlantic (De Swart *et al.*, 1994 and 1995; Ross *et al.*, 1995 and 1996).

Finnish studies of ringed seals show that uterine horn occlusions are still prevalent, also among fairly young individuals (Helle and Nyman, 2000), thereby reducing their fecundity.

Otter

Levels of contaminants such as PCB and dioxins still seem to be too high in fish from the Baltic Marine Area to permit recovery of the otter (*Lutra lutra*) population in the coastal zone. Although several reports indicate thriving and improving otter populations in fresh waters in several Baltic States, there is no evidence of recovery of coastal populations (Stjernberg and Hagener-Wahlsten, 1994; Sjöåsen *et al.*, 1997; Roos *et al.*, 2000).

The musteline otter was a common species in the archipelagos of the Baltic Marine Area in the past, inhabiting the shallow parts of the coastline. Past excessive hunting and in more recent times the effects of contaminants, mainly PCB (Roos *et al.*, 2000), made the species rare and extinct in the coastal parts of the Baltic Marine Area.

The American mink (*Mustela vison*) has been used as a model for piscivorous mammals in experimental studies. The range of PCB body tissue concentrations common in wild otters is the same as that known to detrimentally affect reproduction in mink fed on PCB-treated food (Aulerich *et al.*, 1977; Jensen *et al.*, 1977; Kihlström *et al.*, 1992). Field studies on otter (Mason and MacDonald, 1986; Olsson and Sandegren, 1983; Leonards *et al.*, 1996; Smith *et al.*, 1996; Roos *et al.*, 2000) long-term experimental low-dose exposure studies on mink (Lund *et al.*, 1999) and *in vitro* studies on otter liver (Brunström *et al.*, 1998) all clearly indicate that reproduction is impaired in mustelines exposed to fairly low concentrations of PCB. The *in vitro* studies on otter liver (Brunström *et al.*, 1998) revealed EROD-inducing potencies as high or higher than those related to strong reproductive impairments in the mink (Lund *et al.*, 1999).

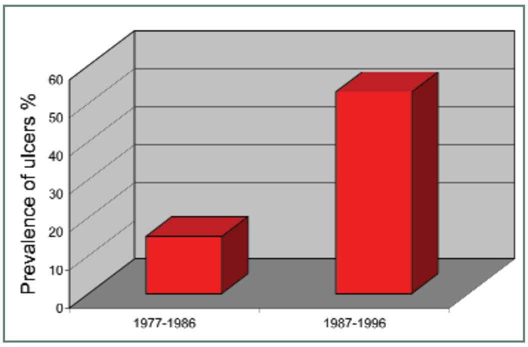


Figure 6.40
Prevalence of moderate to severe intestinal ulcers in young grey seal (<3 years of age) in the Baltic Marine Area.

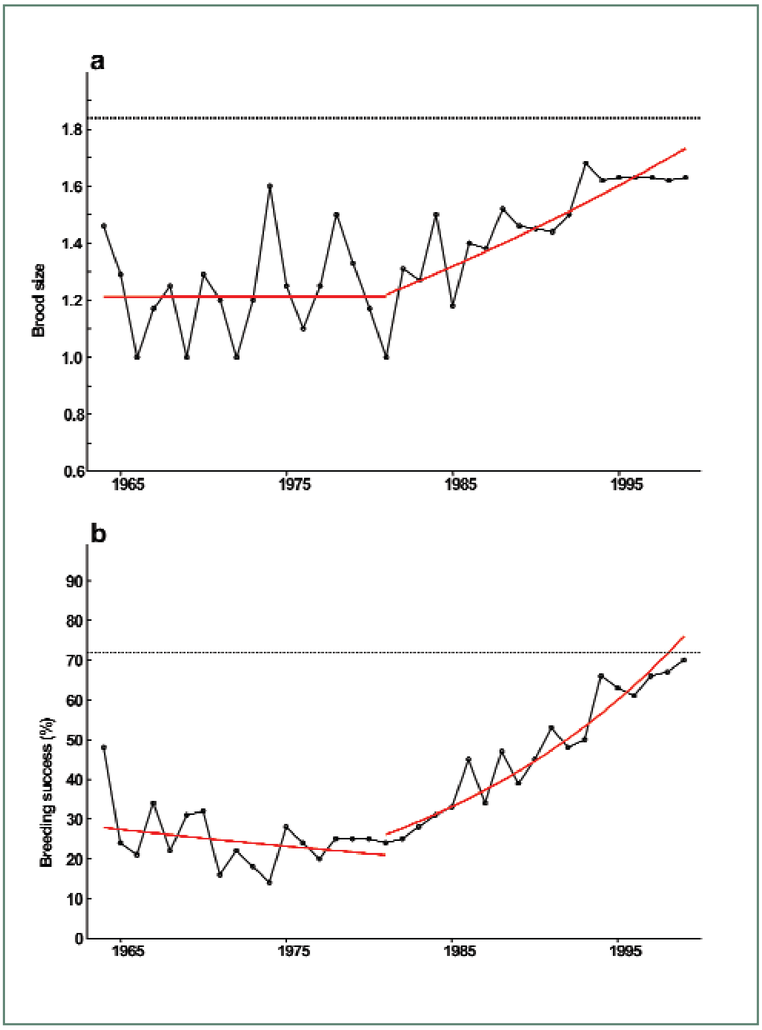
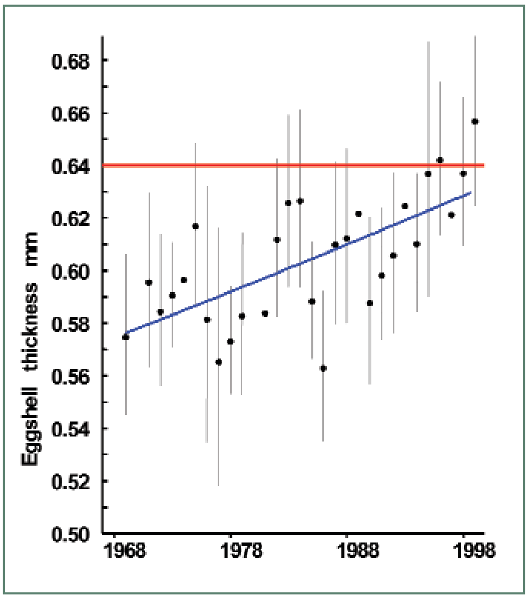


Figure 6.41
Breeding statistics for white-tailed sea eagle along the Swedish Baltic coast.
a) Mean brood size.
b) Proportion of successfully reproducing pairs.
Conditions prior to the 1950s are indicated by the solid black line. Note that brood size seems to have stabilized at a level lower than that prior to the 1950s.

Figure 6.42

Temporal trend in the thickness of guillemot eggshells collected in Stora Karlsö in the central Baltic Proper. The solid red line indicates the thickness prior to 1940.



White-tailed sea eagle

The fecundity of the white-tailed sea eagle population has increased since the beginning of the 1980s (Helander, 1998), the proportion of successful reproducing pairs having increased to a level close to that prior to 1950 (Figure 6.41). The number of known territories along the Swedish Baltic coast has increased three-fold to 150 in 1998. The mean brood size has also increased since the beginning of the 1980s (Figure 6.41). DDT and PCB threatened the species as early as in the 1950s and both the proportion of reproducing pairs and brood size decreased (Helander, 1985). Since measures have been taken to reduce pollution of the Baltic Marine Area the concentrations of DDT and PCB have decreased (see Chapter 6.3) and the population has started to recover. However, brood size seems to have stabilized over the past six years at a level significantly lower than that prior to 1950, probably because contamination still affects embryo survival. Interestingly, brood size levelled off at the same time as PCB and dioxin levelled off following a period of decreasing concentrations in Baltic biota (see Chapter 6.3).

Old females remain reproductively impaired even though the concentrations of both PCB and DDT have decreased in their eggs (Helander *et al.*, 1999). A problem that still persists is desiccation of eggs due to changes in eggshell quality such that the present productivity of old females better correlates to desiccation of their eggs than to the concentration of DDT and PCB in the eggs. Earlier studies correlated the reproductive impairment mainly to the presence of DDE in the eggs whereas recent findings indicate that it is attributable to the presence of both DDT and PCB.

Guillemot

The shell thickness of guillemot eggs from Stora Karlsö in the central Baltic Proper has increased since the 1970s (Bignert *et al.*, 1995), and has now returned to the level prior to 1940 (Figure 6.42). That the eggshells were much thinner in the 1960s is attributable to severe DDT pollution of the Baltic Marine Area.

Radionuclides in the Baltic Marine Area

7

By:
Convenor: Sven P. Nielsen

This chapter is based on information collected by the participants of the HELCOM project on Monitoring of Radioactive Substances in the Baltic Sea (MORS-PRO).

The Baltic States have been monitoring radionuclides in the Baltic Marine Area since 1984. The data encompass radioactivity in different compartments of the Baltic marine environment as well as data on discharges from nuclear installations (nuclear power plants and nuclear research facilities) in the catchment of the Baltic Marine Area. In addition, the following important anthropogenic sources of radionuclide inputs to the Baltic Marine Area have been considered: Atmospheric fallout from nuclear weapons testing and the Chernobyl accident, and discharges to sea from the European reprocessing facilities at Sellafield in the UK and La Hague in France.

Due to the relatively small exchange of water between the Baltic Marine Area and the North Sea, the residence time of contaminants including man-made radionuclides in the Baltic Marine Area is rather long. The levels of ^{90}Sr and ^{137}Cs are therefore still high in the Baltic Marine Area compared with other water bodies in the world. The ^{90}Sr derives from atmospheric nuclear weapons testing, which peaked in the 1960s and resulted in direct input to the Baltic Marine Area from atmospheric fallout and to delayed input via riverine runoff from the Baltic Marine Area catchment. The ^{137}Cs also derives from atmospheric nuclear weapons testing, but this input is small compared to the direct input from the Chernobyl accident in 1986. The delayed input of ^{137}Cs to the Baltic Marine Area from rivers is smaller than the direct atmospheric fallout of ^{137}Cs and relatively much smaller than that of ^{90}Sr. This is attributable to the fact that for chemical reasons, caesium is less mobile in the environment than strontium.

The man-made radionuclides present in the Baltic Marine Area derive from several sources and modes of input. Direct atmospheric fallout has accounted for the main input, partly from the atmospheric nuclear weapons testing and partly from the Chernobyl accident. Riverine runoff has contributed significantly in the case of ^{90}Sr and to a lesser extent with ^{137}Cs. Hydro-dynamic transport has played an important role as regards inputs of radionuclides from the European reprocessing facilities, which are located beyond the North Sea far from the Baltic Marine Area. Although only a few percent of the total marine discharges of radioactivity from these facilities are estimated to reach the Baltic Marine Area, the total input to the area is far from negligible on a relative basis. Direct discharges to the Baltic Marine Area also occur in coastal waters from routine operation of the nuclear facilities on the Baltic coast. These discharges are very low and authorized by national regulatory authorities, however. Total inputs of ^{137}Cs to the Baltic Marine Area during the period 1950–96 are summarized in Table 7.1. Other man-made radionuclides are also present in the Baltic Marine Area, but ^{137}Cs dominates with respect to the radiation dose to man.

Table 7.1
Input of ^{137}Cs (TBq) to the Baltic Marine Area during the period 1950–96

Source	Input mode	^{137}Cs input (TBq)
Weapons fallout	Atmospheric deposition	1,800
Weapons fallout	Riverine runoff	100
Chernobyl fallout	Atmospheric deposition	4,400
Chernobyl fallout	Riverine runoff	300
European reprocessing	Hydrodynamic transport	400
Nuclear facilities	Coastal discharge	2

Since the Chernobyl accident in 1986, the levels of man-made radionuclides in the Baltic Marine Area have generally been declining, mainly due to

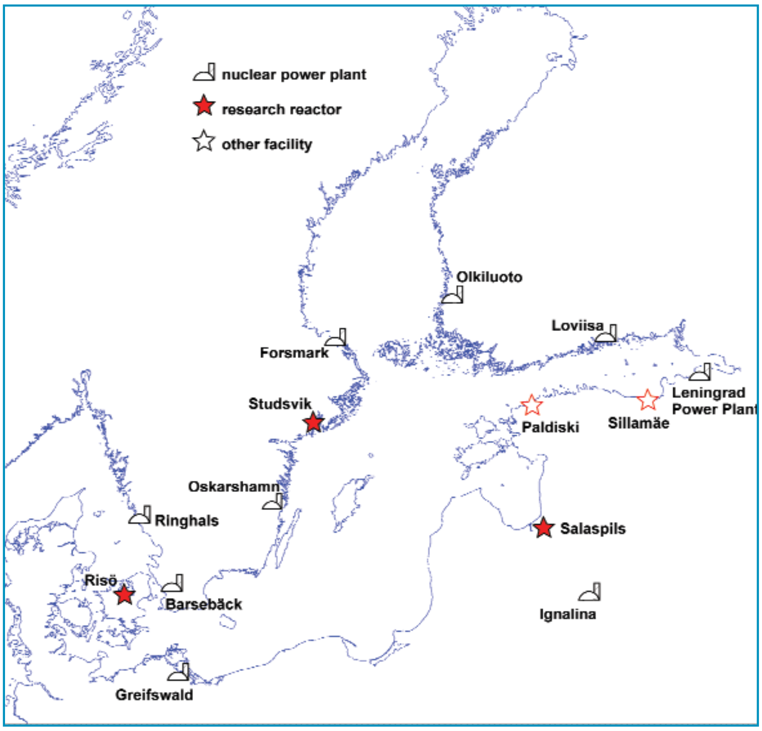


Figure 7.1
Nuclear power plants, nuclear research reactors and other nuclear facilities in the Baltic Marine Area.

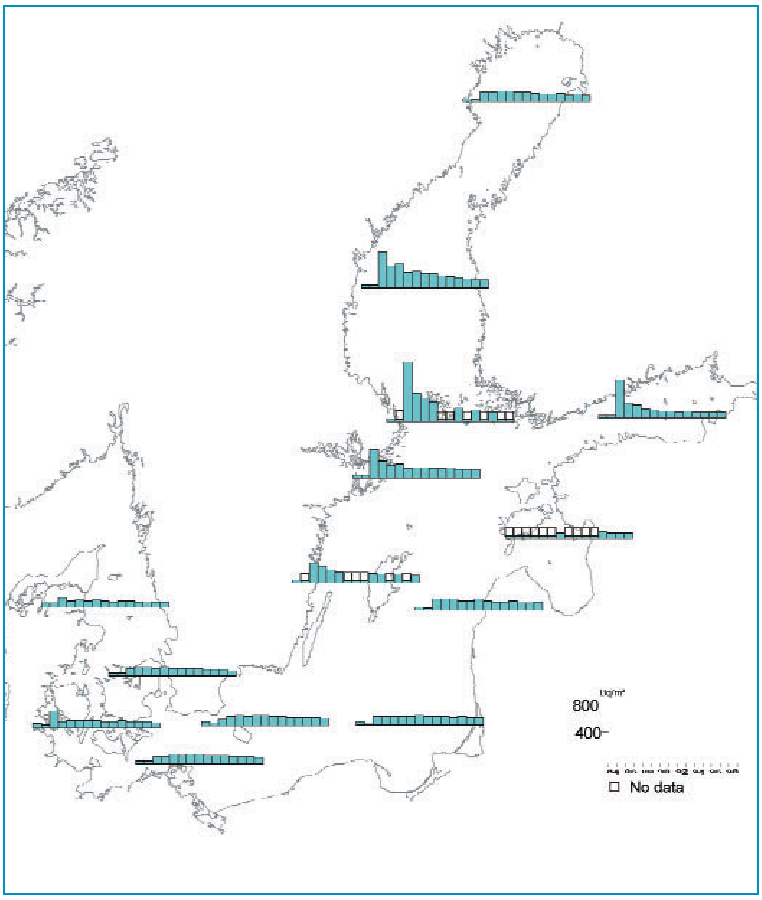


Figure 7.2
Temporal evolution in seawater ^{137}Cs concentrations in the Baltic Marine Area (Bq/m^3).

radioactive decay and outflow of water via the Belt Sea and Kattegat. However, the increase in marine discharges of radionuclides from Sellafield since 1994 affects European coastal waters, including the Baltic Marine Area. Thus small but measurable amounts of ^{99}Tc are transferred to the Baltic Marine Area from Sellafield via inflow from the North Sea.

7.1. Biota

Levels of radionuclides in marine biota are linked to the corresponding levels in seawater and sediment via accumulation through food chains. The complexity of food chains increases with the trophic level of the species considered. Fish, the biota type in the Baltic Marine Area of greatest importance as regards human consumption mainly accumulate radionuclides via their food rather than from the seawater.

Much of the radionuclide content of the marine biota in the Baltic Marine Area derives from the Chernobyl accident in 1986, and predominantly consists of ^{137}Cs and ^{134}Cs. The ^{134}Cs:^{137}Cs ratio in biota fits very well with that of the fallout from Chernobyl. Time trends for ^{137}Cs levels in fish do not always closely follow the corresponding trends in seawater. Thus species high up the food chain, e.g. predators such as cod and pike, exhibited the highest ^{137}Cs levels, although there was some delay before this peaked relative to the seawater peak in 1986. In the long term, though, the time trend for ^{137}Cs in biota will follow that in seawater.

The levels of ^{137}Cs in fish samples (mainly herring and pike) from the northern regions of the Baltic Marine Area, where the initial concentrations in seawater following the Chernobyl accident were highest, have decreased since the end of the 1980s. In the second half of the 1990s, the levels in herring and pike from the Bothnian Bay were around 20 and 40 Bq/kg, respectively, with slightly lower levels being recorded in the Gulf of Finland.

In the Baltic Proper, the subregion of the Baltic Marine Area with the highest production of fish for human consumption, ^{137}Cs levels in fish increased until the beginning of the 1990s and then decreased, although rather slowly. In the more southern part of the Baltic Marine Area where the density of sampling stations is highest, ^{137}Cs levels in herring and cod varied between 10 and 20 Bq/kg in the second half of the 1990s. The corresponding levels in benthic flat fish and small sprat were generally lower – around 10 Bq/kg.

Radionuclide levels in fish from the Belt Sea were even lower. Thus average ^{137}Cs levels in herring have been less than 3 Bq/kg since 1990, a level previously observed during the period 1965–1974. The pattern was similar with plaice. ^{137}Cs levels in cod have remained around 10 Bq/kg since 1987, nearly twice as high as during the period 1965–74.

The lowest ^{137}Cs levels in fish are seen in the Kattegat area, where the level in herring and cod has been below 4 Bq/kg since about 1990, without showing any obvious impact of the Chernobyl accident.

The marine alga (*Fucus vesiculosus*), which is used as bio-indicator of radionuclide levels at nuclear power plant (NPP) monitoring stations in the Baltic Marine Area, reveals that the level of radionuclides representative of NPP discharges, e.g. ^{60}Co, ^{58}Co, ^{54}Mn, ^{65}Zn and ^{110}mAg, are low in Baltic waters. Some of these radionuclides are also found in samples of benthic invertebrates (e.g. *Mytilus edulis*, *Macoma baltica*, *Saduria entomon*). The ^{137}Cs level in *Fucus vesiculosus* from various parts of the Baltic Marine Area closely follows the ^{137}Cs time trend for surface seawater.

7.2. Sediments

Bottom sediments play an important role in the marine environment because they act as a sink for certain radionuclides in the sea. Large amounts of radioactive substances entering the sea are adsorbed on suspended particulate matter in the course of time and deposited in bottom sediments.

The most important event affecting the sediment content of man-made radionuclides in the Baltic Marine Area is the Chernobyl accident in 1986. Since then the sediment content of ^{137}Cs and ^{134}Cs has increased considerably, especially in the seabed of the Bothnian Sea and the eastern Gulf of Finland.

During the 1990s the deposition of long-lived fallout nuclides from the water phase into the sediments has continued, but at a decreasing rate. At the same time, the ^{137}Cs deposits derived from the Chernobyl accident have become more deeply buried in the sediment and can be used as a marker to date sediment and determine sedimentation rates. As the most contaminated water was eventually transported southwards by sea currents from the Gulf of Bothnia and the Gulf of Finland, the deposition of fallout nuclides into sediments also increased in the Southern parts of the Baltic Marine Area.

The highest recorded sediment ^{137}Cs level in the Baltic Marine Area was 125 kBq/m^2 in the northern Bothnian Sea. The total inventory of ^{137}Cs in the seabed of the Baltic Marine Area was estimated to be 1,940–2,210 TBq in 1998 (depending on the calculation method used). Considerable differences were still found between the various subregions

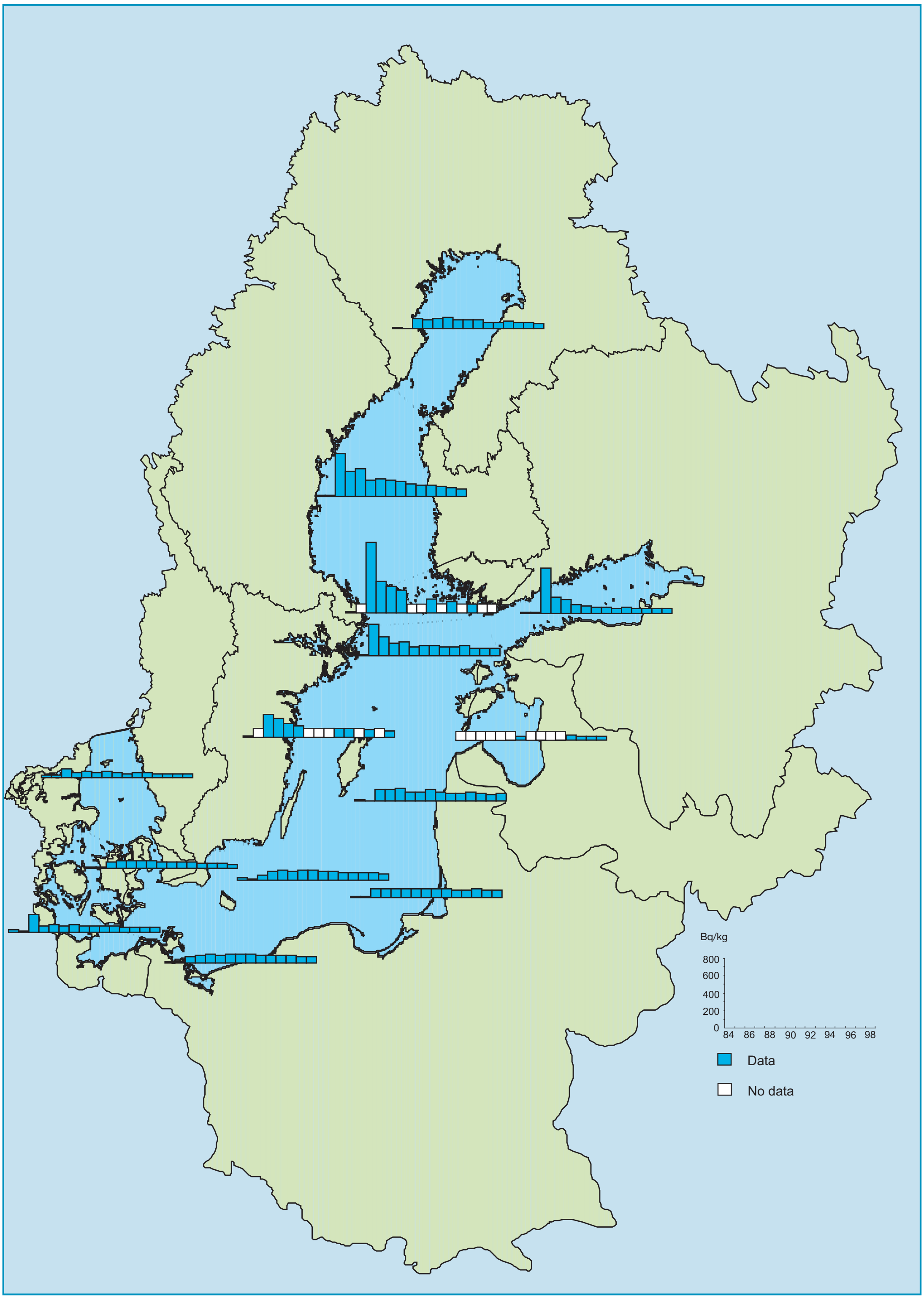


Figure. 7.3
Sediment ^{137}Cs levels at different sampling stations of the Baltic Marine Area in 1998.

Table 7.2
Sediment ^{137}Cs inventories in the subregions of the Baltic Marine Area in 1998

Sub-region	137Cs Inventory (TBq)	Sampling stations
Bothnian Bay	140–165	8
Bothnian Sea	1,370–1,560	13
Gulf of Finland	230–255	78
Gulf of Riga	23	1
Baltic Proper	175–205	35

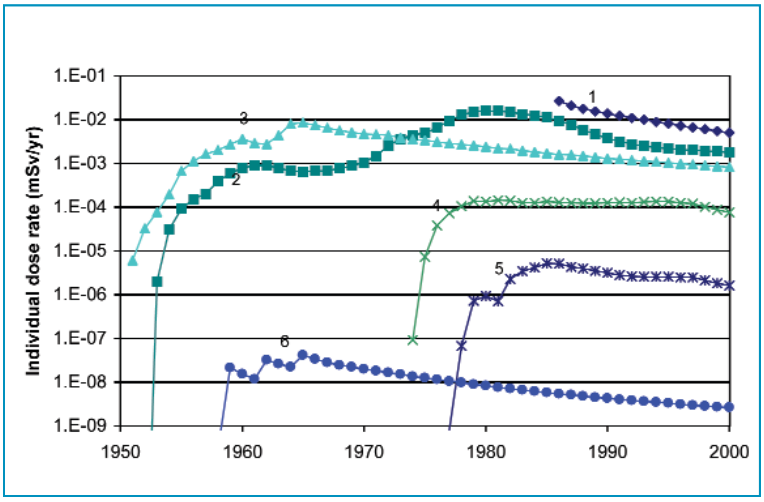


Fig. 7.4
Annual doses from marine pathways (mSv/yr) to individuals of critical groups in the Kattegat area shown apportioned by source (Chernobyl fallout, Sellafield and La Hague reprocessing plants, atmospheric nuclear weapons tests, nuclear power plants, nuclear research reactors, dumping of radioactive waste).

and even between sampling stations situated very close to each other. The sediment ^{137}Cs levels at individual sampling stations in the Baltic Marine Area are shown in Figure 7.3, while the inventory in each subregion is listed in Table 7.2.

7.3. Radiation doses to man

The radiological consequences of radioactivity in the Baltic Marine Area have been assessed based on information on inputs and observed levels of radioactivity in the Baltic Marine Area during the period 1950–96 (EC, 2000). Doses to man were calculated using a model covering the North Atlantic waters including the Baltic Marine Area. The quality of the model predictions was investigated by comparing predicted levels of ^{137}Cs and ^{90}Sr in water and fish with observed levels. The predicted concentrations of both radionuclides were generally in good agreement with the observations. The reliability of the model predictions must be characterized as satisfactory for the purposes of the present radiological assessment.

The doses to which members of critical groups are exposed as a result of the ingestion of radionuclides contained in seafood produced in the Baltic Marine Area and from exposure to radioactivity in coastal areas were calculated for the time period 1950–2000. This period includes source contributions from nuclear weapons testing, the Chernobyl accident and the two European reprocessing plants Sellafield and La Hague, and nuclear installations bordering the Baltic Marine Area. The individual dose rates from marine pathways for persons in the Kattegat region are shown in Figure 7.4.

These dose rates were calculated based on rates of annual intake of marine produce and beach occupancy time. The dose rates to which individuals inhabiting the Bothnian Sea and Gulf of Finland catchments are exposed are predicted to be higher than in other subregions of the Baltic Marine Area due to the pattern of fallout from the Chernobyl accident. The dose rates are predicted to have peaked in 1986 at around 0.2 mSv/yr.

A radiological assessment of the dumping of low-level radioactive waste in the Baltic Marine Area in the 1960s by Sweden and the Soviet Union has revealed that the human dose is negligible.

Doses from naturally occurring radionuclides in seafood (^{210}Po) have been calculated on a similar basis and compared with the doses from man-made radionuclides accumulated via marine pathways. This comparison shows that dose rates and doses from natural radionuclides exceed those from man-made radionuclides except for the year 1986, when the individual dose rates from Chernobyl fallout approached that from natural radionuclides in some regions of the Baltic Marine Area.

The maximum annual dose to individuals from any critical group in the Baltic Marine Area during the period 1950–2000 is estimated to be 0.2 mSv/yr, which is below the dose limit of 1 mSv/yr for exposure of the general public set out in the Basic Safety Standards (EC, 1996). Despite the uncertainties involved in the assessment, it is unlikely that marine pathways have exposed any individual to doses above this limit. Doses to man due to discharges from nuclear power plants in the Baltic Marine Area are estimated to be at or below the levels stipulated in the Basic Safety Standards as being of no regulatory concern (individual dose rate of 10 µSv/yr and collective dose rate of 1 man-Sv/yr). It should be noted that as the assumptions made throughout this assessment are deliberately realistic rather than conservative, the estimated radiation doses to man are also likely to be realistic.

8

Dumping of chemical munitions

By:
Kjeld F. Jørgensen

After the Second World War, the Allied forces dumped chemical munitions found in Germany in the Baltic Marine Area and in the Skagerrak. Information on the chemical munitions dumped in the Baltic Marine Area was last reviewed by a special working group on dumped chemical munitions (HELCOM CHEMU) on the basis of national reports submitted to the Helsinki Commission at the end of 1993 (HELCOM, 1994). Unless otherwise stated, the information in this chapter derives from that review. Another recent review has been made by Karlsson *et al.*, 1998.

Around 34,000 tonnes of chemical munitions containing about 12,000 tonnes of chemical warfare agents were dumped east of Bornholm and near Gotland in 1947 and 1948 on the order of the Soviet Military Administration in Germany. In 1960, the tabun shells dumped at the southern entrance to Little Belt were retrieved, leaving about 5,000 tonnes of chemical munitions (phosgene and nerve gas) on the seabed. The dumping sites in the Baltic Marine Area are shown in Figure 8.1.

In addition, 26 known and 6-8 unknown vessels were sunk in the Skagerrak by American and British forces together with an estimated 130,000 tonnes of chemical munitions and conventional ammunition at a position 25 nautical miles southeast of Arendal in the Norwegian Channel.

There are indications that some munitions were dumped overboard while the ships were en route to the dumping areas east of Bornholm and southeast of Gotland. Moreover, some of the munitions were dumped in wooden crates and could possibly have drifted outside the dumping areas. Information on other dumping areas in the Baltic Marine Area has never been verified.

8.1. Properties and fate of chemical warfare agents

Due to the many factors involved, theoretical considerations and calculations cannot be used to predict the condition of the munitions in a particular dumping area. The investigations hitherto undertaken have revealed both intact munitions and completely corroded casings that have lost their warfare agents. It can therefore be assumed that some of the chemical warfare agents are still intact within their containers, while others have been dissolved, transported and diluted.

Relocation of munitions due to hydrographic factors is unlikely and there is therefore little threat of residues of warfare agents or chemical munitions being washed ashore on the coasts of

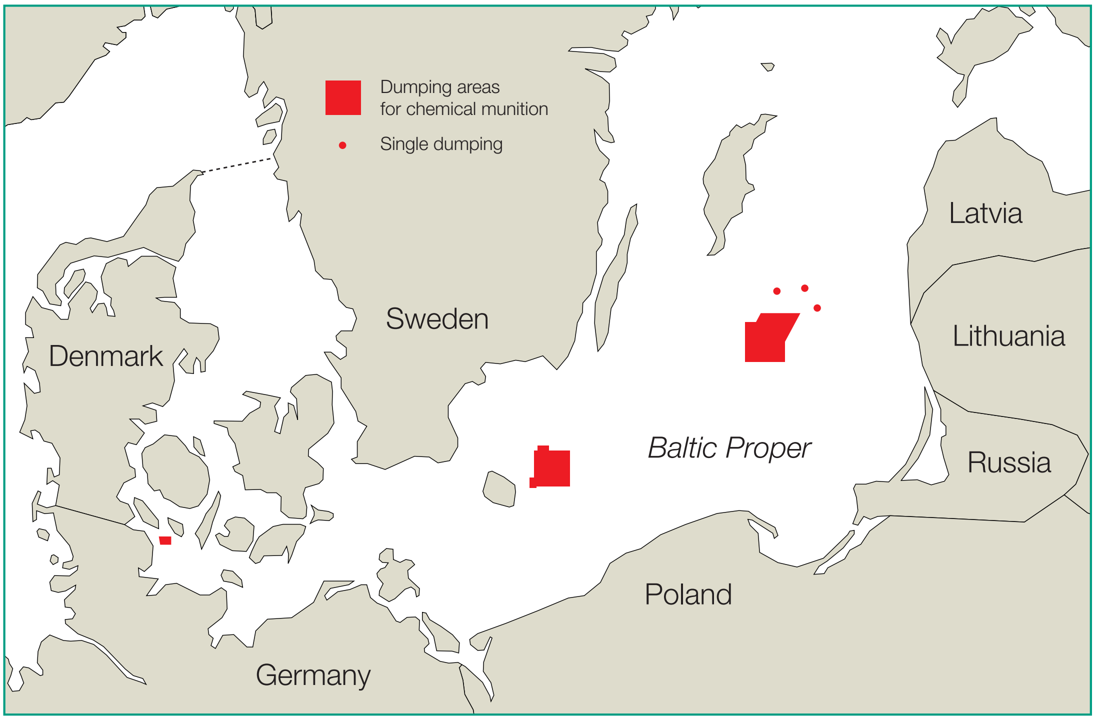


Figure 8.1
Chart indicating munitions dumping positions in the Baltic Marine Area

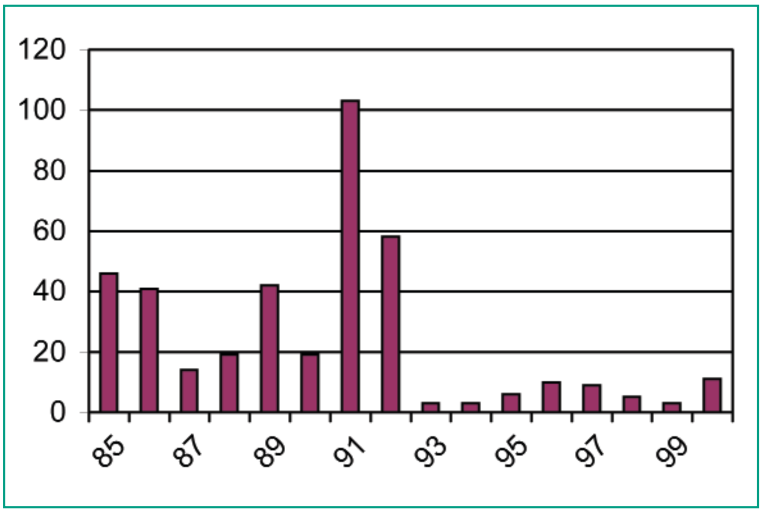


Figure 8.2
Number of chemical munitions caught by Danish fishermen, 1985–2000

the Baltic Marine Area. Fishermen fishing in the Baltic Marine Area occasionally catch dumped munitions. Compared to a peak in 1991, the number of such catches has been low since 1992 (Figure 8.2).

Most warfare agents have only limited water solubility and generally degrade in seawater. Thus they cannot exist in high concentrations for very long. Those of the agents that degrade slowly (Clark I and II, Adamsite and viscous mustard gas) have a very low water solubility and the concentrations reached are thus very low and according to present knowledge have no ecological effects. A wide-scale threat to the marine environment from dissolved chemical warfare agents can thus be ruled out, although elevated levels of sparingly soluble Clark, Adamsite or viscous mustard gas might occur in the sediment in the immediate vicinity of dumped munitions.

Little information is available concerning the arsenic-containing warfare agents and chlorinated additives. These compounds are often lipid soluble, though, and hence might bioaccumulate.

Investigations of the behaviour of warfare agents under the conditions pertaining in the Baltic Marine Area have only been conducted with a few of the agents. Thus their behaviour can usually only be described qualitatively, information on the rates at which the processes occur usually being lacking. Almost all warfare agents are broken down at varying rates into less toxic, water-soluble substances.

8.2. Effects

8.2.1. Mustard gas

Knowledge of the ecological effects and chemical behaviour of dumped chemical warfare agents is limited. The effects of mustard gas on marine organisms at various trophic levels have been investigated under conditions simulating the physical/chemical conditions in the Baltic Marine Area. Most investigations have focused on mustard gas and nitrogen mustard exposure directly in the water column or as lumps of viscous gas in the bottom of the test aquarium. As expected, different organisms react with varying degrees of sensitivity.

Mustard gas present in the water column was found to be toxic to planktonic algae such as *Phaeodactylum tricornutum* and *Rhodomonas balthica* at concentrations of approx. 1 mg/l (1 ppm). At this concentration photosynthesis was inhibited by 10–20%. At a concentration of about 10 ppm, photosynthesis was inhibited by 80–90%. Mustard gas also inhibited egg hatching in the crustacean *Artemia salina* at concentrations ranging from 10 to 100 ppm, with almost total inhibition at 100 ppm. With juvenile plaice exposed for 96 hours, the LC_{50} was approx. 3 ppm. Studies undertaken with flounder indicate that mustard gas does not bioaccumulate in fish (Danish EPA, 1986).

In a test of percentage mortality within one day in freshwater organisms, mustard gas had no effect at a concentration of 1.8 mg/l in fish (*Poecelia reticulata*) and gastropod molluscs (*Lymnaea stagnalis*). With the zooplanktonic crustacean (*Daphnia sp.*), 33.3% mortality within 3 days occurred at a concentration of 0.033 mg/l (HELCOM, 1993).

8.2.2. Warfare agents containing arsenic

In view of their physical and chemical properties, it cannot be ruled out that Clark and Adamsite might accumulate in biota. However, it should be borne in mind, that even following complete degradation of arsenic-containing warfare agents, the arsenic will persist in inorganic form, albeit that such inorganic arsenic compounds are less acutely toxic than the parent warfare agent. Taking into account the mass reduction in the warfare agents during degradation (200 g of Clark contains 75 g of arsenic) and the lower acute toxicity of inorganic arsenic compounds, degradation can be expected to be accompanied by partial detoxification. Inorganic arsenic compounds undergo further reaction in algae and fish to form non-toxic organic arsenic compounds.

In a series of investigations with freshwater organisms parallel to those with mustard gas, no toxic effects were found either with Adamsite or chloroacetophenone, probably due to their low water solubility (HELCOM, 1993).

Clark I and Clark II have recently been shown to have a 48-hour EC50 of around 0.025 mg/l in *Daphnia magna* (Muribi, 1997).

The warfare agent additive monochlorobenzene is potentially environmentally hazardous due to its stability and ecotoxicity. Mustard gas can contain

20% monochlorobenzene and Tabun may contain up to 50% monochlorobenzene.

8.3. Field investigations

Since the previous assessment (HELCOM, 1994), no field investigations of the chemical warfare dumping areas have been reported to the HELCOM lead country (Denmark) except for German investigations of the transport routes to the dumping areas of Bornholm and Gotland aimed at detecting possible munitions on those routes.

In connection with a video survey of the seabed in the dumping area east of Bornholm in November 1992, two sediment samples were collected from nearby locations in the middle of the dumping field. The Danish Civil Defence Analytical-Chemical Laboratory found mustard gas in one of the samples and the more stable by-product of mustard gas production, 1,4-dithiane, in both samples. The Danish National Environmental Research Institute analysed the samples for arsenic and found higher levels (185 and 210 mg As/kg dry weight) than in samples taken from other parts of the Baltic Marine Area. No other traces of chemical warfare agents or chemical compounds related to such agents were found in the sediment samples.

In 1992, the Norddeutscher Rundfunk (North German Radio Station) had 18 sediment samples analysed. These had been collected at 6 different positions, 5 of them in the Bornholm dumping area. One of the samples contained Clark I in a concentration of 10 mg per kg of sediment (10 ppm) while nothing was found in the other samples from the same area. No other warfare agents were found in any of the samples. The arsenic concentrations – even in the sample containing Clark – did not exceed the values usually observed in the Baltic Marine Area (up to 100 mg per kg of sediment).

Investigations by the German Hydrographic Institute in 1987 showed that the arsenic content of seawater from the Baltic Marine Area, including near-bottom water, does not exceed 1 µg/l (0.001 ppm). Concentrations in the dumping areas were not higher than those measured elsewhere.

8.4. Conventional munitions

Besides chemical warfare agents, explosives have also been dumped, either separately or together with chemical warfare agents. The explosives were not dealt with in the HELCOM CHEMU report (HELCOM, 1994) and the amounts dumped and their fate has never been reviewed.

Recent studies on the dumping of munitions that only contain explosives (mainly TNT) have not progressed sufficiently to enable hazard assessments to be made. Toxicity tests indicate that TNT is acutely toxic at concentrations of a few mg/l, and that its environmental fate is rather complex (Dave *et al.*, 2000).

8.5. Concluding remarks

No new information on the chemical munitions dumped in the Baltic Marine Area has been pub-

Cleansing of fishing gear from a fishing vessel at Bornhom having had mustard gas in its trawl.
Photo: Nordfoto/Mr. Poul Erik Rath-Holm, 1991.

lished since the last review in 1993 (HELCOM, 1996). Neither has such information been reported to the HELCOM lead country – Denmark. Information to HELCOM shows that Danish fishermen have occasionally caught mustard gas munitions since 1992.

There is still a general lack of ecotoxicological data for most chemical warfare agents. However, it can be concluded that nothing impinges on the recommendation of the Helsinki Commission that attempts should not be made to recover the dumped chemical munitions.

Marine, migratory and freshwater fish in The Baltic Marine Area

9

By:
Gonvenor: J. Pawlak
Co-convenor:
M. Sparholt

9.1. Fish stocks and fisheries in the Baltic Marine Area

Due to the brackish environment, the ichthyofauna of the Baltic Marine Area is characterized by low species diversity with the dominant species being cod (*Gadus morhua*), herring (*Clupea harengus*) and sprat (*Sprattus sprattus*). Sprat and herring are the dominant zooplanktivorous fish in the ecosystem while cod is the most abundant piscivorous fish. Sprat and herring are the most important prey both for adult cod and salmon (*Salmo salar*).

There is substantial commercial fishery directed at cod, herring and sprat in the Baltic Marine Area. The main fisheries for cod are those using demersal trawls, high-opening trawls (operating both pelagically and demersally) and gillnets. Gillnet fishery increased in the 1990s and the share of the total cod catch taken by gillnets has been about 50% in recent years. Baltic herring is mainly exploited using pelagic trawls, demersal trawls and, during the spawning season, coastal trap nets and pound nets. The main part of the sprat catch is taken in mixed fisheries by pelagic trawls. More than half of the herring and sprat catches are presently used for fishmeal and fish oil. Baltic salmon is exploited offshore using drift nets and longlines and coastal gillnets and traps during the spawning run.

Coastal fishery has been carried out at a fairly constant rate in the Baltic Marine Area for many years. Herring, eel (*Anguilla anguilla*), salmon, trout (*Salmo trutta*), flounder (*Platichthys flesus*), pike (*Esox lucius*), perch (*Perca fluviatilis*), pike-perch (*Stizostedion lucioperca*), smelt (*Osmerus eperlanus*), blue mussels (*Mytilus edulis*), whitefish (*Coregonus lavaretus*) and shrimp (*Crangon crangon*) are the most important species exploited. Since the mid 1980s, fishery has diversified and effort has increased in several coastal regions.

Officially reported catches in the Baltic Marine Area over the period 1973–98 are shown in Figure 9.1. The total catch has remained rather stable at around 0.9–1.0 million tonnes per year with the exception of the period 1986–95, when the total catch was somewhat lower. The individual species catches have varied considerably more, especially the cod and sprat catches. ICES subdivides the Baltic Marine Area into fishing areas as shown in Figure 9.2.

9.1.1. Sprat (*Sprattus sprattus*)

Sprat is distributed throughout most of the Baltic Marine Area and is regarded as one stock unit without clear subunits. It is a batch spawner, spawning pelagic eggs from March to August. It reproduces in the Baltic Proper, the Gulf of Riga and the Gulf of Finland, whereas no offspring are produced in the Gulf of Bothnia due to the low salinity. It is fished both for human consumption and for reduction to fishmeal and oil. Annual landings decreased from high values in the 1970s to <50,000 tonnes in the early 1980s (Figure 9.1). Sprat catches have increased since the early 1990s, however, peaking at 530,000 tonnes in 1997. A much decreased mortality from predation by cod together with a period of favourable environmental

Figure 9.1
Nominal fish catches in the Baltic Marine Area from 1973–98, anadromous species are not included.

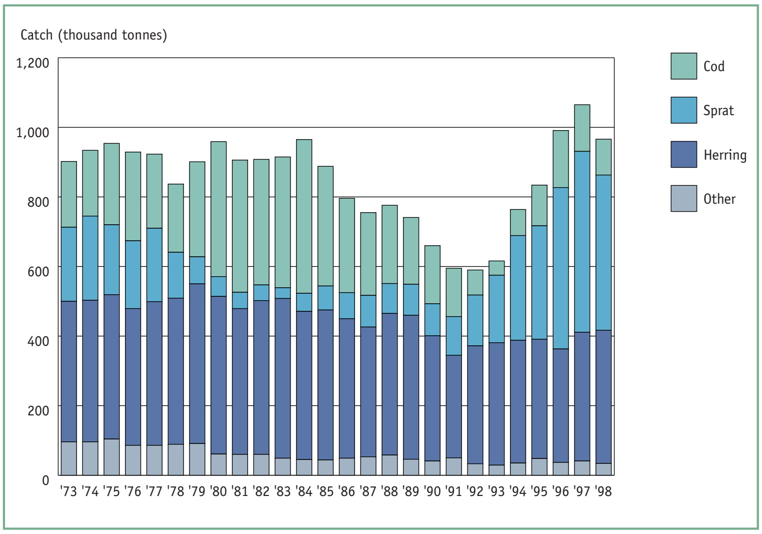
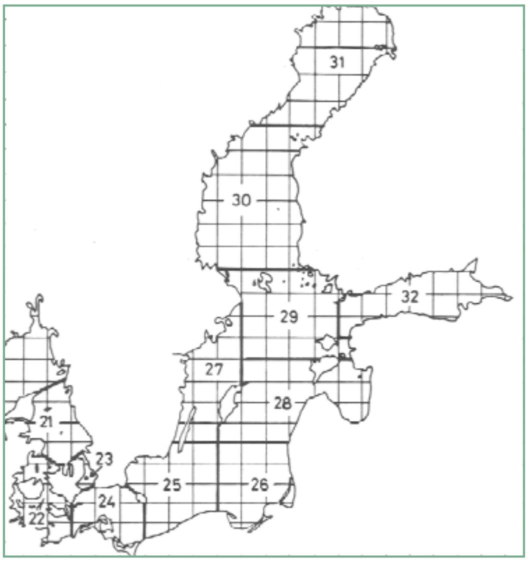


Figure 9.2
ICES subdivisions of the Baltic Marine Area used in fishery assessment.



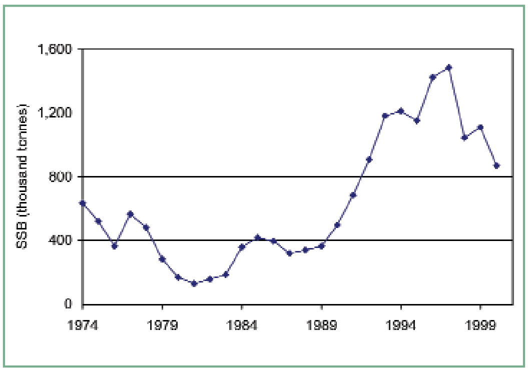


Figure 9.3
Sprat spawning stock biomass (SSB) in the Baltic Marine Area, 1974–2000

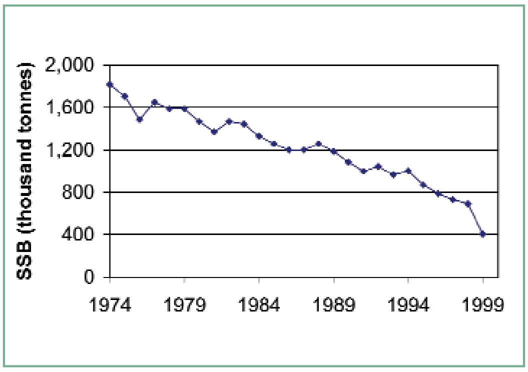


Figure 9.4
Herring spawning stock biomass (SSB) in ICES subdivisions 25–29+32 of the Baltic Marine Area.

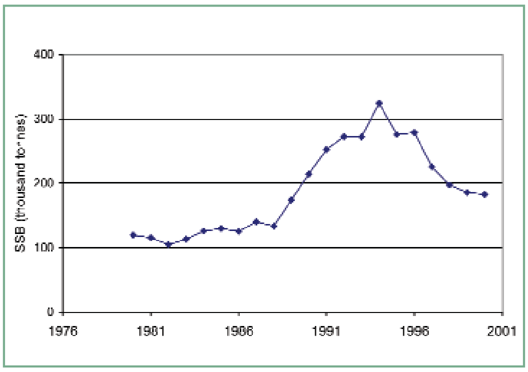


Figure 9.5
Herring spawning stock biomass (SSB) in ICES subdivision 30 of the Baltic Marine Area.

conditions for hatching of abundant year classes in the 1990s contributed to the increase in the sprat stock during the past ten years (Figure 9.3). The stock is considered to be within safe biological limits, although the fishing mortality exceeds the recommended level.

9.1.2. Herring (*Clupea harengus*)

Herring, overwhelmingly spring-spawning herring, is distributed throughout the Baltic Marine Area, forming several local stocks. Autumn-spawning herring was abundant during the first half of the 20th Century, but has been scarce since the 1960s.

Spawning of the spring-spawning herring generally starts around Rügen Island in March/April and ends in the northern areas in July. Spawning of autumn-spawning herring starts in the northern and central areas of the Baltic Marine Area in August–September and ends in the southwestern part in November. The Rügen herring migrate out of the Baltic Marine Area into the Skagerrak and the North Sea after spawning in the spring. There they feed on the abundant plankton until the autumn, whereafter they migrate to the Sound area and Arkona Basin to overwinter. These herring are fast-growing. The growth rate of other stocks gradually decreases towards the northern parts of the Baltic Marine Area, probably due to deterioration of feeding conditions (Sparholt *et al.*, 1994). The growth and condition of herring in the Baltic Proper, the Gulf of Finland and the Gulf of Riga (more recently also of sprat) have deteriorated since the early 1980s (Raid and Lankov, 1995), the changes correlating with decreases in the abundance of large zooplankton species caused by changes in hydrography and possibly also by increased predation on zooplankton (Flinkman *et al.*,1998; Ojaveer *et al.*, 1998).

Under ICES, routine assessment of herring is based on the following stock assessment units: Herring in the Kattegat and Skagerrak (Division IIIa) and the Western Baltic (subdivisions 22–24); Herring in the Baltic Proper (subdivisions 25–29) and the Gulf of Finland (subdivision 32); Herring in the Bothnian Sea (subdivision 30); and Herring in the Bothnian Bay (subdivision 31).

A greater number of populations has been differentiated on the basis of biological and ecological criteria, however (Ojaveer *et al.*, 1981; Ojaveer and Elken, 1997; ICES, 2000a).

The size of the herring stocks has generally decreased over the past 5–10 years (Figure 9.4), the decrease having been most severe in the central Baltic (in the assessment unit comprised of subdivisions 25–29+32). Annual landings in this unit fluctuated around 300,000 tonnes in the 1970s and the early 1980s, but have since decreased to around 200,000 tonnes. Fishing mortality has tended to increase since 1994. The herring stock in this unit is considered to be outside safe biological limits, meaning that this stock suffers increased risk of low recruitment. Other stocks are in better condition, however, and some of them have increased in recent years (ICES, 2000a).

9.1.3. Cod (*Gadus morhua*)

Cod is distributed over the entire Baltic Marine Area except the Bothnian Bay. In the Bothnian Sea and the Gulf of Finland, the amount and distribution are rather limited, except in periods when the cod is generally abundant. Cod reproduction is dependent on the hydrographic conditions, cod needing a certain level of salinity (>11 PSU) and oxygen concentrations (>2 ml/l) for successful fertilization and survival of eggs. The main spawning areas are thus situated in the western and southern areas of the Baltic Marine Area. The spawning area only extends into the Gotland Deep when oxygen and salinity conditions are good (after strong influxes of North Sea water).

Cod spawning takes place over several months. The spawning intensity varies from year to year, depending to some extent on the stock structure. Peak spawning takes place in the spring if most of the spawners are four years of age or older. In recent years, though, spawning has shifted to the late summer as the spawners are now mainly younger than four years old.

Discards of undersized cod are large in Baltic cod fishery, especially in the western areas. This significantly reduces the potential stock production and influences commercial fishery in the long term. An increase in the minimum permitted mesh size would improve the situation. Cod in the Baltic Marine Area consist of two separate stocks, the western stock (ICES subdivisions 22–24) and the eastern stock (ICES subdivisions 25–32).

The western stock (subdivisions 22–24) is rebuilding from an historically low spawning stock biomass (SSB) in 1992 and by 1998, SSB had exceeded the long-term average level of 37,000 tonnes (Figure 9.6). The stock is probably within safe biological limits, but fishing mortality has been too high. In the past decade the fishery has been very much dependent on recruitment, and the fishing pressure on the young ages has increased. The stock appears to have been able to withstand an unusually high fishing mortality compared to other cod stocks around the world. However, the fishing mortality of this stock may be overestimated due to its mixing with neighbouring cod stocks.

The eastern stock (subdivisions 25–32) declined from the historically highest SSB during 1980–82 and reached the lowest recorded in 1992 (98,000 tonnes) (Figure 9.7). This decrease was a combined result of increasing traditional fishing effort and the effect of stagnation processes causing poor reproduction conditions due to a lack of saltwater inflow from the North Sea and no major water exchange in the eastern Baltic Marine Area since the mid 1970s. The stock is outside safe biological limits and high priority needs to be accorded to reducing the fishing mortality and rebuilding the stock.

9.1.4. Flatfish

The commercially most important flatfish in the Baltic Marine Area are plaice (*Pleuronectes platessa*), flounder, turbot (*Psetta maxima*) and dab (*Limanda limanda*). The total landings of plaice fluctuate, probably due to the immigration of plaice from the Kattegat. The western Baltic is the main fishing area for plaice. From the landings it can be inferred that flounder is moderately exploited and that the stock is more or less stable. The total landings of turbot increased from around 100 tonnes in the 1960s to around 1,200 tonnes in the mid 1990s. After 1996, landings decreased continuously to a level of about 600 tonnes in 1999. In the late 1990s, dab landings decreased compared with the period 1994–96.

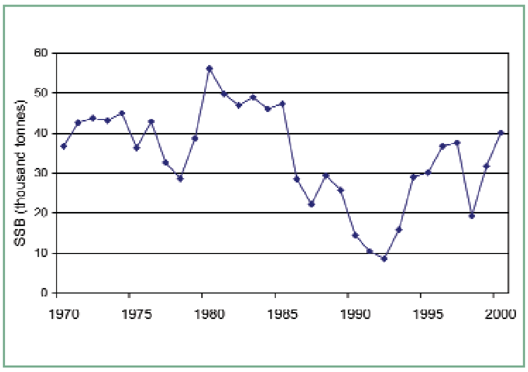


Figure 9.6
Cod spawning stock biomass (SSB) in ICES subdivisions 22–24 of the Baltic Marine Area.

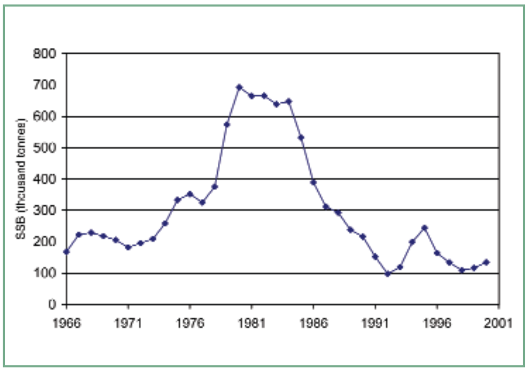


Figure 9.7
Cod spawning stock biomass (SSB) in subdivisions 25–32 of the Baltic Marine Area.

9.1.5. Salmon (*Salmo salar*)

Human impact (intensive exploitation, pollution, damming of rivers, etc.) has substantially decreased the population size of salmonids and deteriorated their reproduction conditions. To compensate, hatcheries have been built on these rivers and stocking initiated. In the past, discrimination between wild salmon rivers with and without reared releases of salmon has been inadequate. If the releases in a river are of a large magnitude compared to the wild production, they will inevitably influence the status of the population, thereby possibly biasing the assessment of whether the population is or could be self-sustaining.

In 1998, as part of the “Salmon Action Plan 1997–2010”, the International Baltic Sea Fishery Commission (IBSFC) decided on a list of rivers with naturally reproducing salmon populations where enhancement should preferably not occur after 2005.

Baltic salmon and sea trout rivers (Figure 9.8) have been divided into three categories (ICES, 2000b): 1) Rivers with naturally reproducing salmon or sea trout populations and without releases of reared fish (wild salmon river or wild sea trout river); 2) Rivers without substantial naturally reproducing salmon or sea trout populations

River names with a slash (/) show main river/tributary. River names with hyphen (-) show names in different countries.

Figure 9.8
Baltic salmon rivers divided into three categories. Only lower parts of rivers with current salmon production or potential for production of wild salmon are shown. The presence of dams preventing upstream migration is indicated by lines across rivers. Rivers with names in bold support wild smolt production. Rivers with names underlined are deemed to have a potential for the re-establishment of wild salmon. The remaining rivers listed receive releases of reared salmon smolt, with no natural production (ICES, 2000b).

and with long-term releases of reared fish (reared salmon river or reared sea trout river); 3) Rivers with potential for the establishment of naturally reproducing salmon populations (potential salmon river).

Total catches of salmon in the Baltic Marine Area have declined from 5,636 tonnes in 1990 to 2,148 tonnes in 1999. At the same time, there has been a shift from offshore fishery towards coastal and river fishery. During the same period, offshore catches in the Baltic Proper and the Gulf of Bothnia decreased from 3,647 tonnes to 1,348 tonnes (Figure 9.9).

In the rivers flowing into the Gulf of Bothnia that have wild smolt production, parr (freshwater phase of salmon) production in the hatching years 1992–96 was low, although the spawning run was fairly good. In those years, M74 syndrome caused high mortality that decreased parr production considerably. In the hatching years 1997–99, parr densities increased to a very high level, about five to ten times higher than in earlier years and the highest levels ever recorded in some rivers. These high parr year classes were caused by large spawning runs in 1996–97. Nonetheless, parr densities remain very low in some small rivers in the southern part of the Bothnian Sea. In these rivers, only a few spawners ascend the river and it will probably take a long time before reproduction increases to target levels. The target in the "Salmon Action Plan 1997–2010" is to rebuild the stocks to at least 50% of their potential smolt production. This seems to be possible for the large rivers in the area, but not for the small ones.

The salmon populations in Swedish rivers flowing into the Baltic Proper are in a better state than those in the Gulf of Bothnia. In the southeastern Baltic, most salmon rivers are small. There is no evidence that M74 syndrome occurs in salmon from these rivers. Salmon production is low in many of the southeastern salmon rivers, but data on potential salmon production are lacking, as are detailed data on the actual state of the population. It seems that the target in the "Salmon Action Plan 1997–2010" is attainable for the rivers in this area.

In 1998, salmon landings from the Gulf of Finland totalled 191 tonnes, less than half that of the previous two years. This decrease appeared to be due partly to seal damage to caught salmon in gear and partly to a very low initial survival of smolt over the past three years. It does not seem that the target in the "Salmon Action Plan 1997–2010" is attainable for the rivers in this area.

9.1.6. Sea trout (*Salmo trutta*)

In the Gulf of Bothnia, sea trout populations previously existed in numerous small rivers and brooks and in most salmon rivers. At present, wild sea trout populations have been verified in more than fifty rivers or brooks. Some of these populations are supported by releases. The disappearance of sea trout populations is partly attributable to human activities such as damming, dredging, pollution, and silting. Most of the populations in the Bothnian Bay are so small that only a few spawners enter these rivers annually.

The situation for the sea trout populations in the Gulf of Finland is similar to that in the Gulf of Bothnia. Although 53 rivers discharging into the Gulf of Finland previously supported sea trout populations, the number has now decreased to about 44. The wild sea trout populations in the northern Gulf of Bothnia and in the Finnish part of the Gulf of Finland are in very poor condition. The main threat is a high exploitation rate in gillnet fishery, including by-catches in whitefish fishery, especially in the Gulf of Bothnia.

It is estimated that at least 160 rivers discharging into the Baltic Proper still support sea trout populations. Many of the Baltic countries have enhancement programmes for sea trout, with many of the releases being carried out directly off the coast independently of rivers.

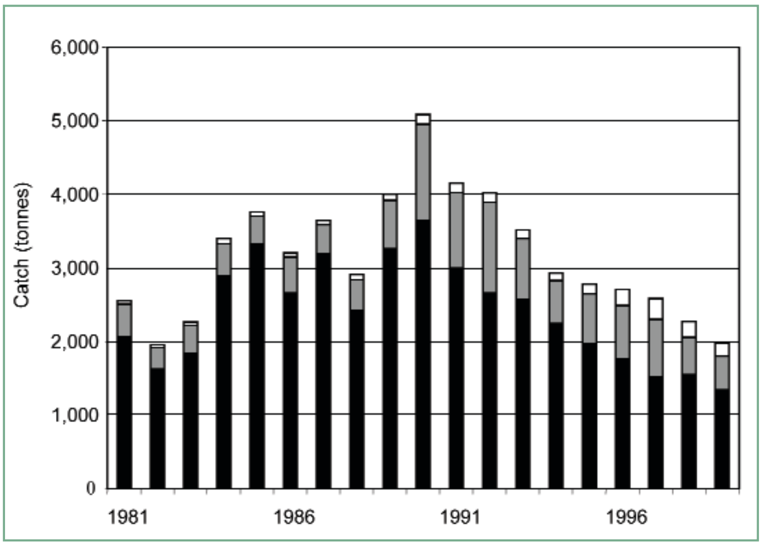


Figure 9.9
Nominal catches of Baltic salmon from the sea, coasts and rivers of the Baltic Proper and the Gulf of Bothnia, 1981–99.

9.1.7. Eel (*Anguilla anguilla*)

The commercial eel catch has decreased since 1955, the first year for which reliable data are available (Figure 9.10). There is no estimate of stock size and some landings are not reported. Indications from catch per unit effort suggest that the stock has decreased in accordance with the catch. In general, the eel stock has decreased over its entire distribution area, which is western Europe. There are indications that the eel stock is depleted and recruitment impaired. ICES has recommended that the fishery for eel be as low as possible in all areas and that a recovery plan should be implemented.

9.2. Diseases and parasites affecting Baltic fish

This section provides information for the period 1994–98 on new trends in the occurrence of the most studied fish diseases and parasites affecting wild fish species in the Baltic Marine Area. Other diseases/parasites that have received attention due to their possible link to marine pollution or their suspected impact on fish stocks are also highlighted.

9.2.1. Impact of anthropogenic factors on fish diseases and parasites

Despite improvements over the past years with regard to anthropogenic impact on the environmental quality of the Baltic Marine Area, a number of disease problems very likely attributable to human activities still affect Baltic fish species, as

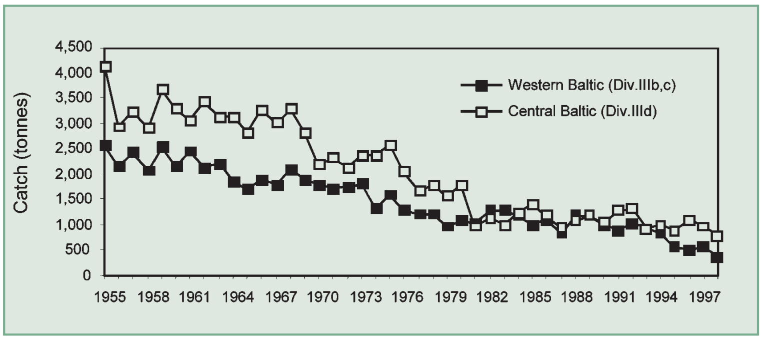


Figure 9.10
Commercial catches of eel in the western and central parts of the Baltic Marine Area, 1955–99

outlined below.

The M74 syndrome continues to occur in Baltic salmon and trout and threatens the survival of the few still naturally reproducing salmon stocks in the Baltic Marine Area.

Swedish studies indicate that the prevalence of parasitic nematodes in perch (*Perca fluviatilis*) and eelpout (*Zoarces viviparus*) might be increased in polluted areas (ICES, 2000c).

Local differences in the occurrence of skin parasites (ciliates) on cod and flounder attributable to eutrophication were recorded in the Kiel Bight (Palm and Dobberstein, 1999).

The parasitic nematode *Anguillicola crassus*, introduced to Europe from Asia, still persists in eel (*Anguilla anguilla*) from the Baltic Marine Area.

A new herring parasite (digenean trematode *Pseudobacciger harengulae*) has been observed on the Swedish west coast, possibly introduced via ballast water discharges (Rahimian, 1998).

Reproductive disorders indicating reduced reproductive capacity have been observed in female perch, roach (*Rutilus rutilus*) and pike (*Esox lucius*) from coastal Swedish and Lithuanian brackish areas affected by thermal effluents (Luksiene *et al.*, 2000).

An elevated prevalence of liver lesions has been found in perch along a PAH gradient in Swedish coastal waters (Ericson *et al.*, 1998).

An increased prevalence of liver neoplasms in flounder from areas in the Gulf of Finland (Lang *et al.*, 1999; Bogovski *et al.*, 1999; Bylund, personal communication) has been attributed to the presence of contaminants.

In the case of certain other contaminant-associated diseases described in the HELCOM Third Periodic Assessment (HELCOM, 1996), the evidence indicates that the reduction in industrial discharges has led to a decrease in disease prevalence. Examples are skeletal deformities in perch and pike and fin erosion in perch and ruffe (*Gymnocephalus cernuus*) in Swedish coastal areas affected by pulp mill effluents (Lindesjöö and Thulin, 1990, 1992; Lindesjöö *et al.*, 1994; Lindesjöö, personal communication).

9.2.2. Impact of fish diseases/parasites on fish stocks in the Baltic Marine Area

There is little conclusive information regarding the impact of fish diseases or parasites on fish stocks in the Baltic Marine Area. In only a few cases has mortality attributable to specific diseases or environmental factors been so obvious that it seems justified to assume that these conditions might have significant effects on stock size of the fish species in question (HELCOM, 1996). The *Ichthyophonus hoferi* epizootic in the beginning of the 1990s was an example of such a case. Since it ceased shortly thereafter, though, it is not currently regarded as a problem for the herring stocks in the Baltic Marine Area. Nonetheless, the possibility cannot be excluded that future epizootics might affect fish stocks in the area.

The M74 syndrome in Baltic salmon remains a significant problem. Although thiamine treatment of early life stages of salmon and more careful selection of adult females for breeding purposes have improved the situation in hatcheries, the syndrome undoubtedly continues to threaten the survival of the wild stocks and, unfortunately, there is as yet no clear indication of a significant and longer-lasting decrease in the occurrence of the syndrome in wild salmon.

The high prevalence of acute skin ulceration in Baltic cod observed during the past few years is causing concern that this disease might affect the stocks by significantly increasing the mortality and possibly also by reducing the fitness and reproductive potential of affected but surviving cod. Mellergaard and Bagge (1998) suggest that the ulcers, which they considered to be caused by mechanical damage due to contact with fishing gear, might have been lethal for the fish due to the large proportion of the body area affected. A high prevalence of skin ulcers not only potentially affects the stock due to increased mortality, but also the fishery since affected fish in most cases are not marketable due to the occurrence of conspicuous disease signs on the body surface.

A new finding has been the occurrence of viral haemorrhagic septicaemia (VHS)-like rhabdovirus and infectious pancreatic necrosis (IPN)-like birnavirus in wild Baltic fish species, pathogens originally thought to be restricted to farmed fish.

Some diseases, e.g. those considered to be associated with effluents from pulp mill industries, have decreased in prevalence, reflecting improvements in the state of the marine environment of the Baltic Marine Area with regard to industrial discharges and their biological effects. With other diseases and parasites, however, human activities probably continue to affect their prevalence and distribution, and studies on fish diseases/parasites thus need to be continued as part of national and international monitoring and assessment programmes in the Baltic Marine Area.

9.3. Effects of mariculture

9.3.1. Use of chemicals and medicines

The use of medicines in mariculture is not considered to pose a significant environmental problem in the Baltic Marine Area. In contrast to countries with salmonid production in oceanic salinities, the low salinity in the Baltic Marine Area means that there are no parasite problems requiring the use of antiparasitic agents. The use of antimicrobial agents in Sweden and Finland (but not in Denmark) has declined in recent years due to increased utilization of vaccines against the most common bacterial diseases, as is also the case in many other countries including Norway, Ireland, and Scotland (ICES, 2000c).

9.3.2. Impact of mariculture on eutrophication in the Baltic Marine Area

Most fish farming in the Baltic region is carried out in fresh water, with mariculture accounting for only a very small part of the total nutrient input to the Baltic Marine Area. The nitrogen load from marine fish farms comprises less than half of one percent of the nitrogen input to the Baltic Marine Area, while the phosphorus load comprises less than two percent of the total phosphorus input (ICES, 2000c).

Mariculture activities are very unevenly distributed in the Baltic Marine Area, while highly localized eutrophication could potentially occur in the immediate vicinity of a concentration of marine fish farms, the overall impact of mariculture on the Baltic Marine Area can be considered to be negligible (ICES, 2000c).

9.3.3. Genetic issues with regard to Baltic salmon

Sweden and Finland have conducted extensive restocking programmes for Baltic salmon for a number of years, particularly in relation to rivers with hydroelectric power plants. As a consequence, only about 10% of the salmon living in the Baltic Marine Area are wild. This gives rise to questions concerning the effect of restocking on the genetic integrity of wild Baltic salmon through interbreeding between wild and reared salmon.

On the basis of estimated production in 1999, the two largest stocks – the river Torne stock and the river Kalix stock – together produce 28% of the total wild production, with rivers in the Bothnian Bay area accounting for about 70% of all wild production. Two phylogeographically different lineages of Atlantic salmon occur within the Baltic Marine Area, the older one originating from eastern glacial lake populations, the Ice Lake lineage, and the younger one from Atlantic populations, the Atlantic lineage (Koljonen *et al.*, 1999). Current production levels of wild smolt and potential reproduction habits suggest that the Ice Lake lineage is in greater danger of becoming extinct than the Atlantic lineage. In genetic terms, however, each salmon river contains (at least) one unique stock, which should be protected.

Some interbreeding between wild and cultured

Mariculture showing a row of netcages for trout.
Photo: Biofoto/Mr. Peter Bang.

salmon takes place, especially when wild stocks are supported by enhancement releases. In these cases, however, releases are based on river-specific broodstocks to ensure that no non-native genetic material is imported into the wild stocks. In some cases, though, the hatchery production is based on an effective population size that is smaller than that of the wild stock to be supported. The releases as such may then reduce the total genetic diversity of the stocks (Ryman and Laikre, 1991).

In general, very little genetic information is available on the wild stocks, although it is evident that some, but not substantial, changes have occurred in both the diversity levels of the marker genes and the inherited life-history traits (ICES, 2000c).

9.4. Ecosystem effects of fishing

The environmental effects of fishery in the Baltic Marine Area are not known with certainty, but are considered to be substantial, particularly at the top of the food web. Cod, which is the dominant piscivorous fish, has been reduced to low biomass by the fishery and weak recruitment. Changes in the cod stock have undoubtedly influenced stocks of its primary fish prey, namely sprat and herring. These species are themselves the target of directed fishery, however, and it is difficult to differentiate the direct effects of fishery on their population dynamics from responses to reduced abundance of cod. Sprat and herring are the dominant zooplankton predators in the ecosystem, and zooplankton populations are expected to be affected by the changes in abundance of their predators. Again, however, it is not easy to differentiate the effects of changes in zooplankton predator abundance from the effects on zooplankton of changing nutrient availability and hydrographic conditions.

The effects of fishing on invertebrates, marine mammals and birds are poorly known. Multispecies assessment models have been used to evaluate predator-prey interactions among cod, herring and sprat, and how these interactions may be affected by fishery. However, these models do not include lower trophic levels and hence cannot provide an insight into certain types of possible effects of fishing on the entire food web.

Harbour porpoise (*Phocoena phocoena*) ***caught in gill-net. Photo: Biofoto/Mr. Svend Tougaard.***

9.4.1. Effects on target species and the food web

The main ecological effect of fishery is the removal of large quantities of fish, in particular target species. Three potential consequences of this are reductions in offspring production, changes in the area occupied, and changes in food web dynamics. In the Baltic Marine Area there are strong indications that fishery has caused all three types of change (Hansson, in press). When hydrographic conditions adversely affect the reproductive success of cod, which is the dominant piscivorous fish in the Baltic Marine Area (Sparholt, 1996; Jarre-Teichman *et al.*, 1997), fishery exasperated the situation and depressed the eastern cod stock to a biomass outside safe biological limits (ICES, 1999a, 1999b). As the cod stock decreased, the predation pressure on herring and sprat was reduced and sprat in particular increased substantially (ICES, 1999b). It has been proposed that the Baltic offshore fish community has gone through a regime shift from one controlled by cod predation to a clupeid-controlled system attributable to herring and sprat predation on cod eggs and larvae (Köster and Möllmann, 1997). The growth and condition of herring deteriorated concomitantly with the increase in sprat stock (Raid and Lankov, 1995). These changes in herring correlate with decreases in the abundance of large zooplankton species, but it is unclear whether this decrease is attributable to increased zooplankton predation or to changes in hydrography (Flinkman *et al.*, 1998).

With regard to effects on distribution, a consequence of the decreased cod stock is that the species has virtually disappeared from some of its normal distribution areas, i.e. the Archipelago Sea, the Bothnian Sea and coastal areas of the Baltic Proper (O. Sandström, personal communication).

With regard to effects on progeny production, ICES has determined spawning stock limit values for western Baltic cod and sprat stocks. These stipulate the spawning stock biomasses (SSB) below which there is evidence of impaired recruitment (ICES, 1999a, 1999b). The biomass of the eastern Baltic cod stock is presently below this limit value, as it was for a period during the early 1990s. Excessive exploitation contributed greatly to the depressed condition of the stock in both cases. As noted above, however, environmental conditions were also unfavourable for cod reproduction, and the effect of fishing on reduced offspring production has not been clearly distinguished from environmental influences on recruitment. Sprat SSB was below the current limit value in the early 1980s, when excessive fishing combined with poor recruitment and high predation by cod moved the stock to a condition where production of recruits was impaired.

Coastal fishery by anglers and commercial fishermen has probably also influenced ecosystem structures (Hansson *et al.*, 1997). The impact is generally more local than that of the offshore fishery, however, since most of the coastal fish species are relatively sedentary.

Studies in other intensively fished areas encompassing a variety of species have shown that fishery can induce changes in fish growth rates and fish population sex ratios, maturity ogives, and genetic composition (Jørgensen, 1990; ICES, 1996; Rice and Gislason, 1996). Given the exploitation intensity in the Baltic Marine Area, such changes are likely to have occurred.

9.4.2. Effects of by-catch and discard

The total by-catch of fish among the Baltic Marine Area fisheries is unknown. Earlier reports generally contained few data on this topic and hence quantitative estimates could not be made. The EU has supported several very recent studies of by-catch, however, the results of which have been compiled by ICES (2000c). These studies primarily concern the major fisheries for cod, herring and sprat. Some smaller and coastal fisheries also have a very high proportion of by-catch, though. In vendace *(Coregonus albula)* roe fishery in the Bothnian Bay, for example, the by-catch is 92% by weight (O. Sandström, personal communication).

In addition to the discarding of non-target species, another type of discarding activity is the return of fish offal to the sea. Degradation of this organic matter consumes oxygen and it has been speculated that this might contribute to oxygen depletion in bottom waters. A recent analysis indicates that the problem is negligible, however (ICES, 2000c).

9.4.3. Impact of fishing gear on benthic organisms

All towed demersal fishing gears disturb the seabed and thus may detrimentally affect benthic structures and processes. ICES has recently conducted a comprehensive review and provided advice on these effects in the North Sea (ICES, 2000c). In the Baltic Marine Area the effects of bottom gears are expected to be smaller than in the North Sea because beam trawling is not carried out and the benthos is generally dominated by smaller organisms than in the North Sea.

Fishing vessel at Alter Strom in the old habour of Warnemünde. Photo: Susanne Feistel, Baltic Sea Research Institute, Rostock.

Towed gears that penetrate the sediment considerably affect both the benthic infauna and the benthic epifauna, whereas other types of gear (e.g., gillnets, trap nets, Danish seines) may affect the epifauna in some circumstances. Local studies are often needed to quantify the impacts of specific fisheries.

Some specific case studies are available concerning the impact of trawl gears on the benthos and benthic habitats in the Kiel Bight. Evidence was found for sediment disturbance, including remobilization of nutrients and increased release of nutrients and organic material followed by an increase in oxygen consumption (Krost, 1990). Biological responses of the invertebrate community to trawling activity in the Kiel Bight were also documented. There were obvious biological impacts on thin-shelled bivalves, and starfish suffered heavy damage, whereas little or no damage occurred to the solid-shelled bivalves. Trawling reduced the mean population size of the solid-shelled *Arctica islandica*, however. For several species of mussels the proportion of damage increased with increasing body size. Many epibenthic organisms suffered sublethal damage. An increase in predatory and scavenging species feeding on dying and dislocated fauna was also documented (Rumohr and Krost, 1991). The applicability of these findings to the larger Baltic Marine Area requires further study, however.

9.4.4. Effects on seabirds

Fishing nets, in particular set nets, cause considerable seabird mortality in the Baltic Marine Area. Oldén *et al.* (1988) estimated that between 1982 and 1988, about 25,000 seabirds, chiefly common guillemots *(Uria aalge)*, died in set net fishery for cod in the Kattegat. Similarly, Stempniewicz (1994) calculated that about 16,000 long-tailed ducks *(Clangula hyemalis)* and velvet scoters *(Melanitta fusca)* are killed annually in the set net fishery for flatfish and cod in the Gulf of Gdansk, representing 10–20% of their local wintering populations. In the Kiel Bight, up to 17% of the maximum winter population of eiders (*Somateria mollissima*) and black scoters (*Melanitta nigra*) were estimated to drown in the same type of fishery (Kirchhoff, 1982). There are also reports of guillemot and razorbill *(Alca torda)* mortality in driftnet fishery for salmon (ICES, 1995).

Fishing activities will also affect the seabird community in other ways, for example through the discarding of unwanted catch and offal, which become important sources of food. Studies indicate, for example, that over 50% of the offal discarded in the Baltic Marine Area will be consumed by seabirds (ICES, 2000c). This food source is believed to be responsible for increases in seabird populations over time in the North Sea (ICES, 1996).

9.4.5. Effects on marine mammals

Harbour porpoises

During the period 1987–96, an average of five harbour porpoises per year were by-caught in fisheries in Polish waters, mostly by salmon fisheries. 3–5 porpoises are also by-caught each year in bottom-set gillnets and driftnets along the Swedish Baltic coast. Moreover, an estimated 111 porpoises were by-caught in German fisheries in ICES Division IIIc (the Kiel and Mecklenburg Bays and inner Danish waters) during the period 1987–95 (Kock and Benke, 1996), while six were by-caught in 1996. These reports suggest that fisheries by-catches amount to 0.5–0.8% of the porpoise population in the southwestern part of the Baltic Marine Area each year, as well as 1.2% of the porpoise population in the Kiel and Mecklenburg Bays and inner Danish waters. Estimates of the harbour porpoise population are uncertain, however, and the number of porpoises by-caught in fisheries is probably underestimated. The loss of porpoises to fishery in the Baltic Marine Area may be too high to sustain the population (ICES, 1997).

Seals

About 200 seals per year are estimated to have been by-caught in Estonian waters, predominantly in fyke nets. Grey seals appear to be more vulnerable to entanglement than ringed seals and constitute more than 80% of the seals by-caught in fisheries in Estonian waters. In Poland, a total of nine seals, of which a majority were grey seals, were by-caught in 1996, predominantly in salmon nets. In German parts of the area, one harbour seal was by-caught by trawl and one in a fyke net in 1995; no cases were reported in 1996.

Salmon driftnets are also responsible for seal mortalities in Swedish waters. Thus Swedish data for the period 1974–1990 show that of the 216 grey, ringed and harbour seals by-caught in fishery, salmon driftnets were responsible for 29 (14%) (ICES, 1995). A survey along the Swedish coast north of Åland showed that a minimum of 250 grey seals were by-caught in fishery in 1996. A realistic approximation of the number of seal mortalities in Swedish coastal fishery is 400 animals (O. Sandström, personal communication). A further 338 seals were by-caught in Finnish fishery during the period 1986–95, including 113 (33%) in driftnet fishery in the open sea. Most of the latter were grey seals.

Most of the seals by-caught in fisheries in the Bothnian Bay were entangled in traps set for salmon and whitefish, while those by-caught in the Bothnian Sea were entangled in nets set for salmon and whitefish. In the Baltic Proper, most of the seals were by-caught in eel traps and turbot nets; of these, half were pups of the year.

Although recorded data almost certainly underestimate the total number of seals by-caught in fishery, this added mortality does not appear to threaten the seal populations since their numbers are increasing (Helander and Härkönen, 1997).

Seals are also known to take salmon and other fish from gillnets and traps (ICES 1999b,c).

Biodiversity and nature conservation 10

By:
Convenor:
Dieter Boedeker
Co-convenor:
Ole Norden Andersen

Biodiversity is low in the Baltic Marine Area due to its character as a geologically very young brackish sea with a prehistory as a freshwater lake. Many species are precluded due to the low oxygen levels and to fluctuating and progressively lower salinities as one moves from the outer to the innermost parts.

Nature conservation and sustainable use of natural resources were incorporated into the Helsinki Convention with a new Article 15 in 1992, this being the first time that the parties to a regional marine convention had jointly agreed on programmes and measures aimed at conserving marine and coastal species and their biotopes, protecting biodiversity and promoting ecological processes.

10.1. Biotopes

By: Ole Norden Andersen and Dieter Boedeker

A "biotope" is an area or habitat of a particular type that can be defined by the organisms (plants, animals, microorganisms) that typically inhabit it, for example "white dunes". A "biotope complex" is a typical combination of specific biotopes occurring in a characteristic spatial distribution, for example "sandy coasts" consisting of sandy beaches and different types of dune, etc. In order to facilitate nature protection in the Baltic marine and coastal areas, an all-encompassing biotope classification was performed, segregating 133 biotopes (66 marine and 67 coastal) and 13 biotope complexes (HELCOM, 1998).

A HELCOM Red List based on this classification provides the first state-by-state and Baltic-wide assessment of the degree of threat posed to biotopes and biotope complexes by loss of area or by change, i.e. threats to biodiversity, species interactions, ecological processes and environmental conditions. The Baltic-wide assessment revealed that 88% of the biotopes are either "Heavily endangered", "Endangered" or "Potentially endangered" (Figure 10.1), but that none of the biotopes are "Immediately threatened".

In a country:biotope matrix, however, where 52% of the biotopes are rated as "Endangered" or "Potentially endangered" by loss of area and 74% are correspondingly threatened by change, one marine biotope occurring in 2 Baltic Sea States and 12 coastal biotopes occurring in either a single Baltic Sea State (9 out of 12) or in 2 Baltic Sea States (3 out of 12) are classified as "Immediately threatened" by change. In the individual Baltic Sea States the corresponding percentages range from 13 to 71% and from 54 to 96%, respectively.

An indication of which human activities are most harmful to biotopes is provided by the number of adverse impact entries for each activity in relation to specific biotopes on the Red List (HELCOM, 1998). This is shown in Figure 10.2 for five main biotope groups (2 marine and 3 coastal).

The most common threat factors are pollution, eutrophication, construction (mainly local) and fishing which affect 75%, 70%, 57% and 47%, respectively, of the 133 biotopes. Marine biotopes are the most extensively threatened. All marine biotopes are threatened by change due to eutrophication or pollution in at least one Baltic Sea State. Each of these two threat factors account for 24% of the scores in a marine biotope:threat matrix. Fishing and construction threaten 74% and 70% of the marine biotopes, respectively. Of the coastal biotopes, 61% are threatened by recreational activities and 51% by pollution.

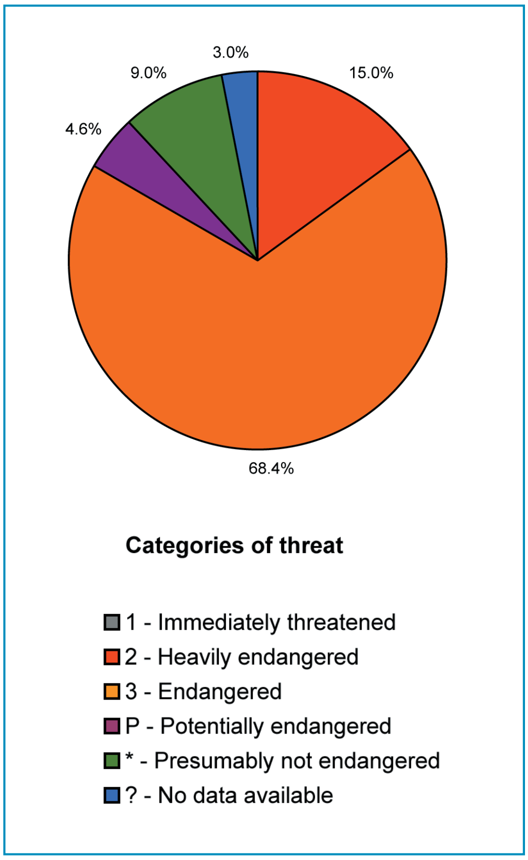


Figure 10.1
Threat assessment for all 133 types of coastal and marine biotopes found in the Baltic Marine Area. (Compiled by: D. Boedeker, after HELCOM, 1998).

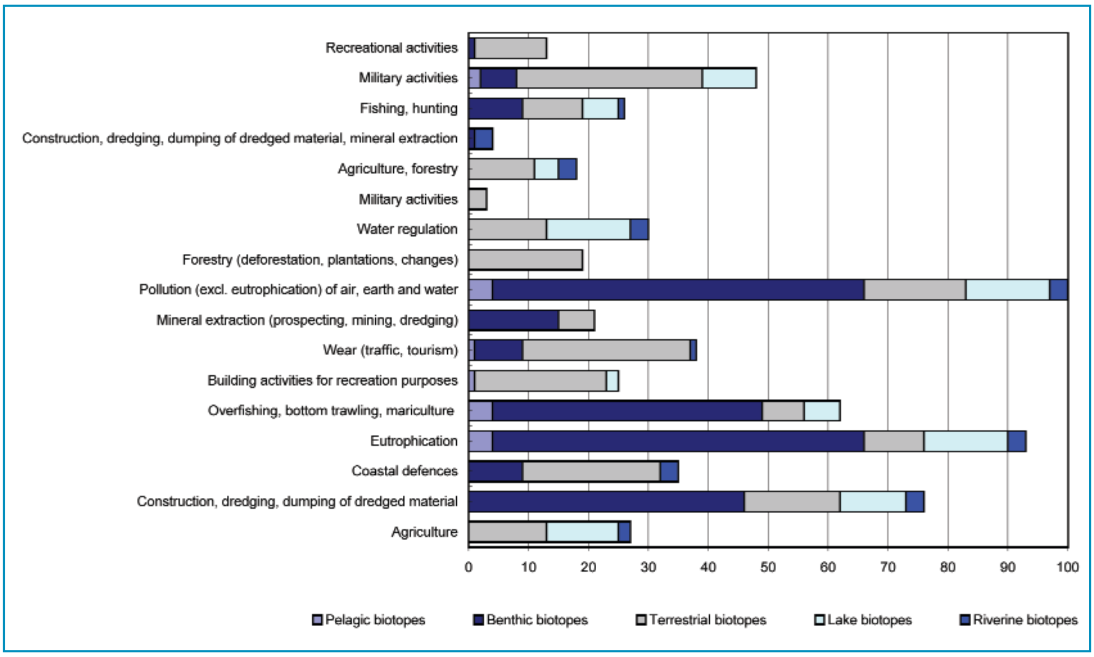


Figure 10.2
Number of adverse impact entries for various human activities in the HELCOM Red List of Marine and Coastal Biotopes
(Compiled by: D. Boedeker, after HELCOM, 1998).

10.2. Species

10.2.1. Plankton

By: Irena Olenina and Ole Norden Andersen

Phytoplankton monitoring under the HELCOM Baltic Monitoring Programme (BMP), which started in 1979, has recorded the presence of 670 species, i.e. one third of the approx. 2,000 species known to inhabit the Baltic Marine Area. The number of species recorded annually merely reflects the sampling effort. Moreover, the constant addition of new species and gradual accumulation of far more species than are found annually reflects an inadequate annual sampling frequency and/or a highly inter-annually fluctuating species composition and perhaps inconsistent taxonomic expertise (Figure 10.3).

Differences in seasonal diversity patterns within various subregions of the Baltic Marine Area and shifts in dominant species assemblages can be ascertained, however (Figure 10.4). These reflect environmental differences or changes, e.g. in nutrient levels. Compared to the previous assessment period, the dinoflagellate *Gymnodinium lohmanii (= Gyrodinium spirale)* emerged as a new dominant in the spring and summer phytoplankton communities in the Kattegat, the Belt Sea, the Arkona Basin and the Bornholm Basin. In the Gotland Basin, the cyanobacteria *Noduliaria spumigena* and *Aphanizomenon spp*. created summer blooms as usual, but with the latter species the bloom was prolonged into the autumn. The potentially toxic dinoflagellate *Prorocentrum minimum* also emerged as a new dominant in the autumn phytoplankton. In the northern Baltic Proper and in the Gulf of Finland the dinoflagellate species *Scrippsiella hangoii* and *Heterocapsa triquetra* dominated in spring and summer, respectively.

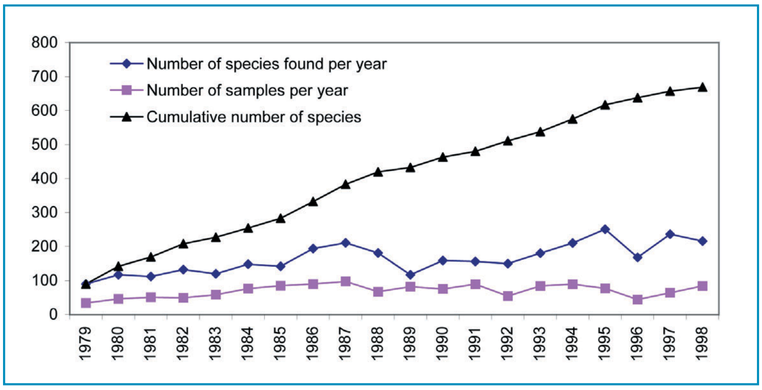


Figure 10.3
Number of phytoplankton species recorded at the HELCOM BMP stations over the period 1979–98.
(I. Olenina).

Eutrophication has also led to a general increase in zooplankton abundance. The increase is attributable not only to the massive diatom blooms, but also to the greater abundance of smaller phytoplankton channelling energy via microzooplankton to the larger mesozooplankton by way of the "microbial loop". These relationships, including effects on higher organisms and fish in particular, should be investigated further.

10.2.2. Phytobenthos

By: Saara Bäck, Hans Kautsky, Lidia Kruk-Dowgiallo and Daiva Jurgilaite

Species composition

The number of macroscopic plant species, primarily algae, declines rapidly with decreasing salinity from the Kattegat to the innermost parts of the Baltic Marine Area. In the Baltic Proper, where the salinity in the photic zone is fairly stable, the inner distribution limits of the various species coincide more or less with their freshwater tolerance limits, and they remain relatively constant. An increase in species number due to the occurrence of freshwater elements only occurs in the innermost parts of bays and in estuaries. The number of macroscopic marine algae in the Baltic Marine Area decreases from more than 356 species in the Kattegat (salinity 23‰) to less than 100 species in the low-salinity (5-6‰) waters of the Stockholm archipelago (83 species), around the Åland Islands (97 species) and off southern Finland (97 species). Species numbers were systematically higher along the rocky coasts of Sweden than along the predominantly moraine or sandy coasts elsewhere (Nielsen *et al.*, 1995; Middelboe *et al.*, 1997). Of the 32 species recorded in the northernmost part of the Bothnian Bay (Råneå archipelago), all but one are freshwater species.

Vegetation zonation

As salinity drops towards the east in the Gulf of Finland, the red algae belt common in the southwestern part of the Gulf is replaced by a green algae belt dominated by *Cladophora*. In the vicinity of the Vyborg Bay in Russia, the deep-water vegetation is dominated by the encrusting red alga *Hildenbrandia prototypus*. The *Cladophora rupestris* zone is lacking east of the Fiskar Islands, and the few *C. rupestris* individuals found are often brown in colour due to a dense cover of epiphytic diatoms (Lehvo and Bäck, 2000).

Endangered biotopes

The most endangered phytobenthic biotopes in the Baltic Marine Area are found on level sandy bottoms (HELCOM, 1998), where eelgrass *Zostera marina* is common. The small, but compact meadows of *Potamogeton perfoliatus* found in the inner parts of Puck Bay are considered to be unique (Kruk-Dowgiallo, 1994). In many areas of southern Danish and of Swedish waters, charophyte meadows also suffer from eutrophication effects (Blindow, 2000).

Fucus vesiculosus

Baltic *Fucus vesiculosus* occurs in the upper sublittoral zone (Kautsky *et al.*, 1992). Between the 1940s and the 1980s the depth limit decreased from 11 m to 7–8 m along the Swedish coast (Waern, 1952; Kautsky *et al.*, 1986). Since then, it has increased again by 1 m (Kautsky, 1995). Along many parts of the Finnish and Estonian coasts *F. vesiculosus* disappeared at the end of the 1970s (Kangas *et al.*, 1982; Rönnberg *et al.*, 1985; Kukk, 1985), and the depth distribution decreased along most open shores. Partial recovery has taken place (Kangas and Niemi, 1985), but *F. vesiculosus* does not reach the same depths as it did previously. In the Tvärminne archipelago, for example, individual *F. vesiculosus* were recorded down to depths of 8–10 metres in the 1930s (Purasjoki, 1936) and the 1970s (Strandström, 1980), but only down to 5 metres in 1994 (Ruuskanen and Bäck, 1999).

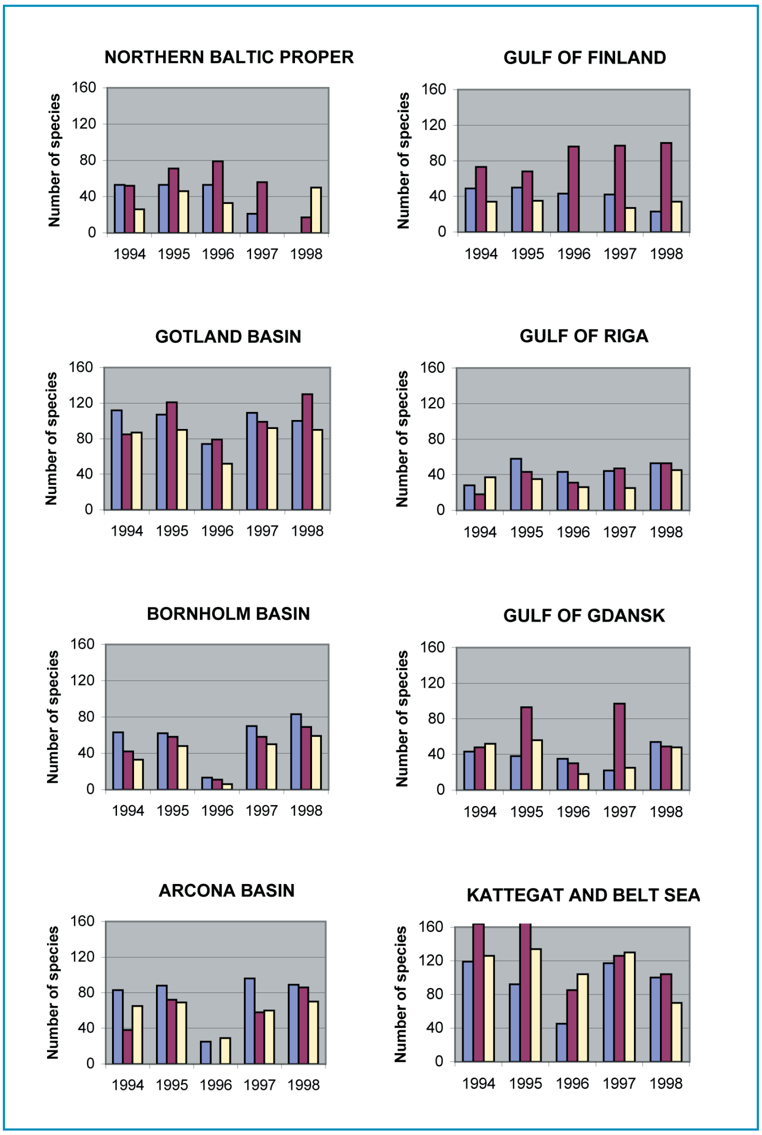


Figure 10.4
Phytoplankton species diversity in spring, summer and autumn in the various subregions of the Baltic Marine Area over the period 1994–98.
(I. Olenina).

Threats to biodiversity

The main threats to the phytobenthos are considered to be eutrophication and, locally, habitat destruction. These threats have caused the number of taxa occurring in many areas to decrease. In Klif Redlowski and Inner Puck Bay in Poland, for example, the decrease has been about 50% since the beginning of the 20th Century. The disappearance of many species is related to mass occurrences of filamentous brown algae of the genera *Pilayella* and *Ectocarpus*, which, for example, began in the mid 1970s along the Polish coast and continued until the end of the 1980s (Ciszewski *et al.*,1992; Kruk-Dowgiallo,1996). In the Kattegat coastal areas, however, the biomass of eutrophication-dependent filamentous algae has generally decreased slightly during the 1990s.

10.2.3. Zoobenthos

By: Thomas Forbes and Ole Norden Andersen

The marine zoobenthos complies with a characteristic biogeographical feature of the Baltic Marine

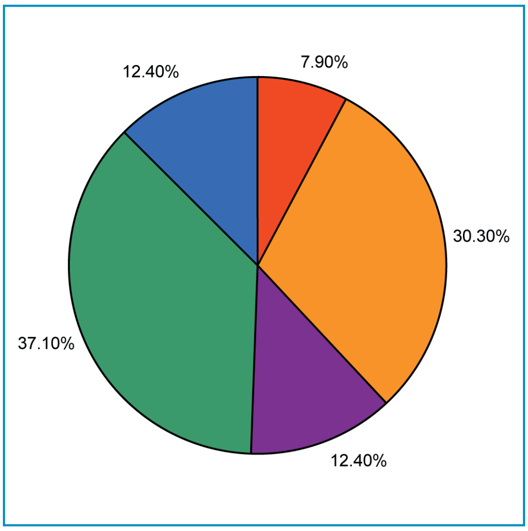


Figure 10.5
Assessment of the threat by loss of area posed to 432 benthic Baltic marine biotopes, as analysed in a 62 biotope:10 state matrix. The percentage of the 432 biotopes falling into each category of endangerment is shown.
(O. Norden Andersen, after HELCOM, 1998).

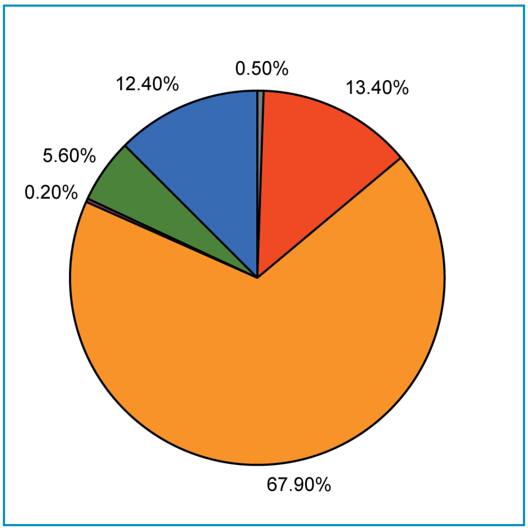


Figure 10.6
Assessment of the threat by change posed to 432 benthic Baltic marine biotopes, as analysed in a 62 biotope:10 state matrix. The percentage of the 432 biotopes falling into each threat category of endangerment is shown.
(O. Norden Andersen, after HELCOM, 1998).

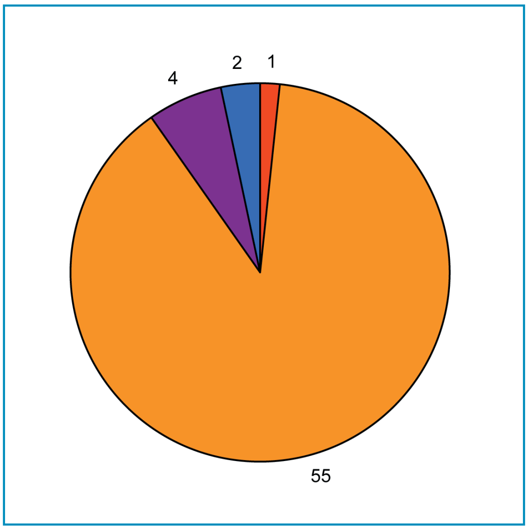


Figure 10.7
Baltic-wide assessment of the threat posed to the 62 benthic marine biotope types in the Baltic Marine Area. The number of biotopes falling into each category of endangerment is shown.
(HELCOM, 1998).

Categories of threat

- ⬛ 1 - Immediately threatened
- 🟥 2 - Heavily endangered
- 🟧 3 - Endangered
- 🟪 P - Potentially endangered
- 🟩 * - Presumably not endangered
- 🟦 ? - No data available

Area in that it is relatively species-rich in the western regions such as the Kattegat, becoming progressively poorer in (marine) species as one moves towards the east. This is attributable to falling and fluctuating salinities and suggests geographical differences in community susceptibility to eutrophication, chemical contamination and invasion by alien species. Other species-poor regions include hypoxic/anoxic basins, most notably in the deep central Baltic Sea Proper. During the prolonged stagnation period in the early 1990s, the near-bottom oxygen conditions in the central basins improved at intermediate depths as a consequence of the descending halocline. Thus a recolonization of macrofauna was observed in the 70–100 m depth zone, an area which corresponds to approximately 19% of the total (Laine *et al.*, 1997). This succession continued until the stratification became stronger again, and shallower, due to major inflows re-occurring from 1993–1994 onwards, which periodically improved oxygen conditions in the deepest layers. The long-term decrease of salinity before 1993, however, caused a decline in the abundance of marine species (especially polychaetes), which was most notable in the southern Baltic Marine Area (Osowiecki and Warzocha 1996; Persson *et al.*, 1996; Laine *et al.*, 1997).

Hypoxia in shallow coastal waters also seriously affects biodiversity, however, and seems to be an increasing problem – especially in the archipelagos of the northern Baltic Sea. These irregular events are caused by local topography, hydrography and drifting algal mats.

Of the benthic biotopes included in the HELCOM Red List of Biotopes (HELCOM, 1998), all are considered to be threatened by change due to eutrophication or pollution in at least one Baltic Sea State. Each of these two factors is responsible for 25% of the scores in a benthic biotope:threat factor matrix. The proportion falling into each of the various categories of endangerment mentioned in Section 10.1 is shown in Figures 10.5 and 10.6. Only one biotope, “Sublittoral level sandy bottoms dominated by macrophyte vegetation”, is classified as “Immediately threatened”, however, and only in the relatively limited coastal waters of Poland and Lithuania. This accounts for less than 0.5% of the 432 biotopes scored in the matrix.

Majority or equal scoring among the Baltic Sea States in which each biotope occurs reveals that the biotopes fall into only three threat categories (Heavily endangered, Endangered and Potentially endangered) (Figure 10.7). Thus more than 90% of the benthic biotopes (56 out of 62) are “Endangered” or “Heavily endangered” in a majority of the Baltic Sea States for which information on each biotope is available. The threats are primarily related to change (81%), with only 50% being threatened by loss of area. Moreover, none of the biotopes are in danger of disappearing entirely from the Baltic Marine Area (Figures 10.5 and 10.6).

The cockle *Cerastobyssum (Cardium) hauniense* once found throughout the Danish straits and estuarine fjords where salinities ranged from 8–16‰, now only occurs in the southwesternmost Baltic Marine Area, perhaps due to increased fluctuation in salinity and incidents of anoxia.

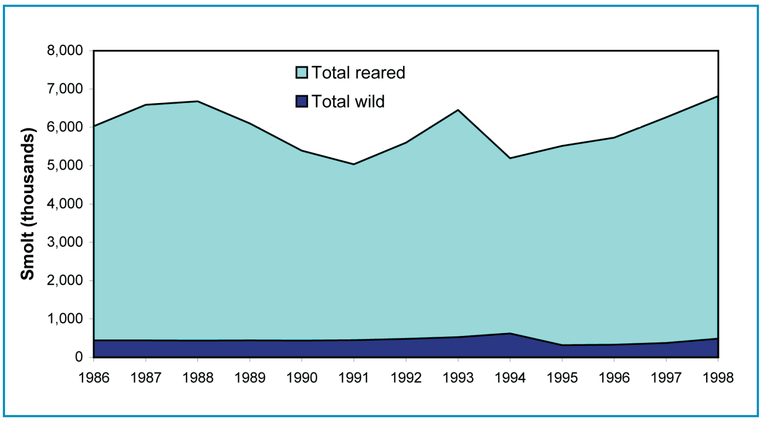


Figure 10.8
Annual production of reared and wild salmon smolt in the Baltic Marine Area over the period 1986–98. (Source: National Board of Fisheries, Sweden).

10.2.4. Fish

By: Jörn Gessner and Dieter Boedeker

The distribution of the roughly 100 fish species inhabiting the Baltic Marine Area is largely governed by salinity. Marine species (some 70 species) dominate in the Kattegat and in the Baltic Proper, while freshwater species (some 30-40 species) occur in coastal areas and in the innermost parts (Nellen and Thiel, 1996). The shallow coastal areas also serves as a nursery ground for many marine fish species, although little is known about species composition, habitats, genetic diversity, ecology and endangerment, particularly in the case of non-exploited species. Moreover, little is known about the potential threats from acoustic disturbances, electric and magnetic fields around marine cables, and future construction and operation of offshore windmill farms, etc. There is, nevertheless, cause for concern because of eutrophication, pollution and fishing pressure, as illustrated by the current situation for three of the fish species inhabiting the Baltic Marine Area.

Cod (*Gadus morhua*)

Cod is perhaps the most important commercially exploited fish species in the Baltic Marine Area. Cod fishery at its present level is not sustainable (see Chapter 9.1.2.1), and the eastern Baltic cod stock is in immediate danger of collapse because landings are often above recommended biologically safe levels and contain too many immature cod. Furthermore, some catch declarations are rather inaccurate (see Chapter 2.2; ENVIRO, 2000).

Salmon (*Salmo salar*)

Anadromous fish such as the salmon spend most of their lives in the sea but migrate to fresh water to spawn. In the Baltic Marine Area, anthropogenic disturbances and pressures such as obstacles to free passage in rivers, commercial and recreational fishing and the M74 syndrome (see Chapter 9) have brought the Baltic wild salmon close to extinction. Hatchery rearing and release of fry have been used to compensate for the loss of natural spawning areas and prevent total extinction. The consequence, though, is that less than 8% of salmon smolt produced in the Baltic Marine Area in 1998 derived from the natural, wild population. This is considerably less than the corresponding figure for 1994 of 13.5% (Figure 10.8).

In 1997, HELCOM and IBSFC launched the “Salmon Action Plan 1997–2010“ the Baltic salmon and prevent it from becoming extinct (see chapter 9).

Common sturgeon (*Acipenser sturio*)

An HELCOM project to protect and enhance the common (Baltic) sturgeon population in the Baltic Marine Area was established in 1997. Since the common sturgeon is presumed to be extinct in the Baltic Marine Area, the aim is to pave the way for its re-establishment.

The diadromous sturgeon, which migrates between saltwater and freshwater, was once abundant throughout the southern parts of the Baltic Marine Area, where it spawned in the tributaries of the main rivers (Figure 10.9). Only occasional catches have been reported since 1915. The drastic decline in the population is attributable to dams etc. hindering migration to spawning grounds, destruction or damage to spawning and nursery grounds in connection with river maintenance, the impact of pollution on spawning success, and the

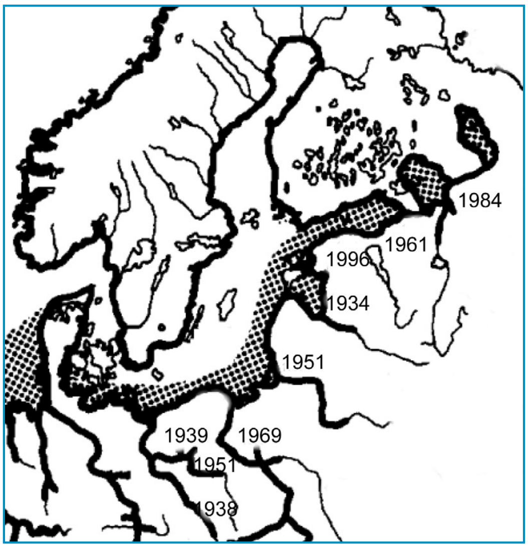


Figure 10.9
Historic distribution of the common sturgeon Acipenser sturio in the Baltic Marine Area. The main area of distribution is shaded. The spawning rivers are indicated by solid lines extending to the uppermost point of migration. The years indicate the most recently recorded observation. (Source: Holcik et al., 1989).

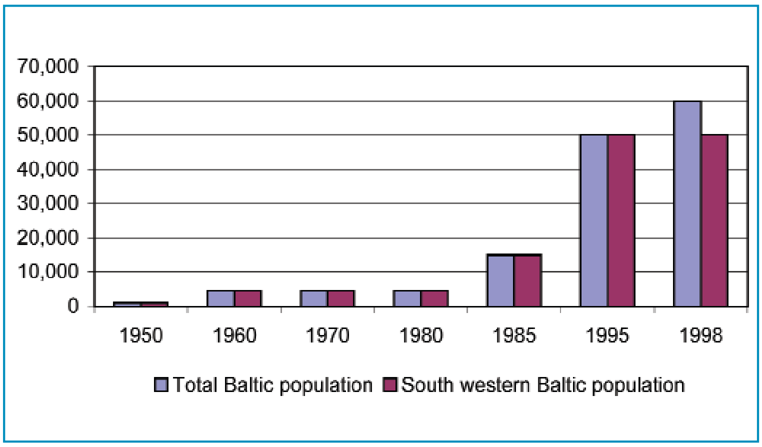


Figure 10.10 Development of the cormorant breeding population in the Baltic Marine Area. The trend shows that the southwestern population levelled off at approximately 50,000 breeding pairs, while the species expanded its distribution range to the northern Baltic Marine Area. (Compiled by: D. Boedeker, from several sources).

increasing pressure of fishery for both meat and caviar.

The only known remaining population inhabits the Gironde (France). It is widely accepted that only ex situ measures such as the rearing of breeding stock in captivity to preserve genetic heterogeneity and long-term reproduction plans will provide some chance of success.

A comprehensive fish monitoring programme should be drawn up, and a “Red List of Fish Species” for the Baltic Marine Area prepared, in order to enable adequate HELCOM assessments of fish biodiversity.

10.2.5. Birds

By: Christof Hermann, Dieter Boedeker and Henrik Skov

The Baltic Marine Area encompasses many highly important staging areas for sea birds, and more than 30 species breed along the shores. The population trends for characteristic species first presented in the preceding assessment (HELCOM, 1996b) have been updated and widened for the current assessment period.

Cormorant (*Phalacrocorax carbo sinensis*)

After having been hunted near to extinction during the 19th and early 20th Century, the cormorant returned to the southwestern Baltic Marine Area in the 1930s to 1950s. In the beginning of the 1980s the cormorant population increased again at an exponential rate (Figure 10.10). The southwestern population levelled off at approximately 50,000 breeding pairs during the 1990s, and at the same time the cormorant expanded its distribution range to the eastern part of the Baltic Marine Area. The first breeding in Finland was reported in 1996, since when the eastern cormorant population has developed rapidly to 8,600–11,000 pairs.

Table 10.1 Number of white-tailed eagle breeding pairs in the Baltic Sea States 1991 and 1998. Compiled by D. Boedeker (unless otherwise stated, the population refers to the whole country or region).

Country	Breeding pairs 1991	Breeding pairs 1998	Current population trend
Denmark	0	5	↗
Estonia	40	60	↗
Finland	78	151	↗
Germany (Baltic Sea population)	35 (1994)	57	↗
Latvia	5–8	11	↗
Lithuania	7	25–30	↗
Poland	245	Ca. 500	↗
Russia, Kaliningrad region	1–4	5–6	↗
Russia, St Petersburg region	25	32	↗
Sweden (Baltic Sea population)	73	150	↗

This extraordinary expansion is in part attributable to reduced human persecution and improvement in food resources due to eutrophication-enhanced growth of specific fish populations, e.g. cyprinids (Knief, 1998). The exponential increase coincides with the recovery of other species in the Baltic Marine Area previously affected by contaminants (Olsson, M., 2000. Personal communication). Given the present state of management and food availability, most experts expect the southwestern population to stabilize at about the current level, whereas that in the eastern parts of the Baltic Marine Area is expected to develop like the southwestern population with a delay of about a decade.

As cormorants feed exclusively on fish, fishermen view it as a competitor as well as a pest and have for some time been calling for a drastic reduction in the population. Although the cormorants in the Baltic Marine Area feed mainly on fish of sizes and species that are of minor economic value (Knief, 1998), their ravaging of pound nets and fish farms has necessitated the implementation of specific cormorant management measures. In Denmark, for example, cormorants can be shot all year round near fishing nets, fish farms and "put and take" facilities. Moreover, new colonies may be destroyed. An international "Action plan for the management of the great cormorant in the African-Eurasian region" presented to the Bonn Convention in 1998 advocates for appropriate site-specific fisheries management as a first priority and thereafter local management, including controlled hunting and measures such as those mentioned above.

White-tailed eagle (*Haliaeetus albicilla)*

The white-tailed eagle is representative of predators whose populations have been detrimentally affected by DDT and other pollutants, but which are now improving following bans on the use of these substances (see also Chapter 6.4). Severe persecution strongly reduced the population density and distribution range of this species up to the first decades of the 20th Century. In some parts of the Baltic Marine Area such as Denmark, Russia (the Kaliningrad region) and Lithuania, it did not occur for several decades.

Protection measures implemented since then first permitted population growth, but the increase was reversed from the mid 1950s to the early 1980s by the harmful effects of chemical pollutants (DDT and PCBs) on reproduction success. The ban on DDT and other pesticides in the early 1970s led to population recovery from the beginning of the 1980s. During the 1990s the growth rates were remarkably high, and most populations doubled between 1991 and 1998 (Table 10.1, Figure 10.11). The increase in the number of breeding pairs was also related to expansion of the distribution range, with the species returning to breed in areas where it had not bred for decades (e.g. Lithuania in 1987 and Denmark in 1996). The white-tailed eagle still requires attention, like other predators being vulnerable to certain pollution hazards. A breakdown in reproduction around the year 1985 in Germany, Finland and Sweden (Figure 10.11) was perhaps due to illegal use of DDT in GDR forestry (Bignert *et al.*, 1989) (see also Chapter 6.4).

Dunlin (*Calidris alpina schinzii*)

The dunlin is representative of waders that are declining in many regions of the Baltic Marine Area as a consequence of habitat loss, alterations in area management and increased predation.

At the beginning of the century, the dunlin was relatively common along the coasts. Since then the population has declined dramatically in all Baltic Sea States and is still declining, except perhaps in Denmark. Only a few pairs remain in Germany, Poland, Lithuania, Latvia and Russia and the species could well disappear entirely from these regions in the very near future (Table 10.2).

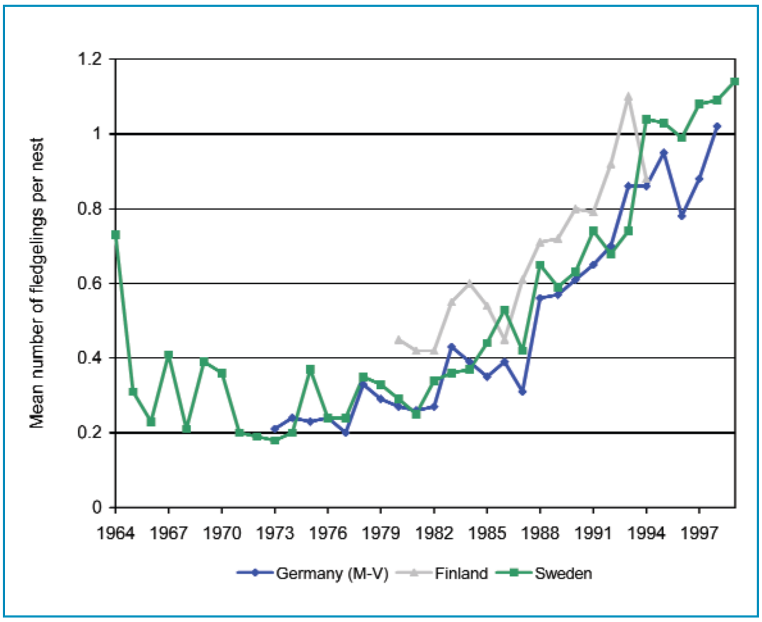


Figure 10.11
Fecundity (hatched young per occupied nest) of the white-tailed eagle in Sweden, Finland and the Federal German State Mecklenburg-Vorpommern (M-V) over the past few decades.
(Compiled by C. Herrmann, after Hauff, 1998; Helander, 2000; Stjernberg and Koivussari, 1995).

Barnacle goose (*Branta leucopsis*)

The barnacle goose breeds in coastal plains and valleys of East Greenland, the Svalbard archipelago and arctic Russia. Colonies were established in the Baltic Marine Area as recently as 1975 and have since grown rapidly to approx. 2,000 breeding pairs in 1993. Breeding sites are located in Sweden, Finland, Estonia and Denmark (Hagemeijer and Blair, 1997).

The expansion of the breeding area to include the Baltic Sea States can be attributed to a considerable population increase, among other things, due to cessation of hunting in Denmark in 1954 and in all other EC countries in 1977 in connection

Eider ducks flying in formation.
Photo: Johnny Kahlert, National Environmental Research Institute, Kalø, Denmark.

Table 10.2
Dunlin current breeding population, population trend and threat status in the Baltic Marine Area.
Compiled by D. Boedeker from several sources.

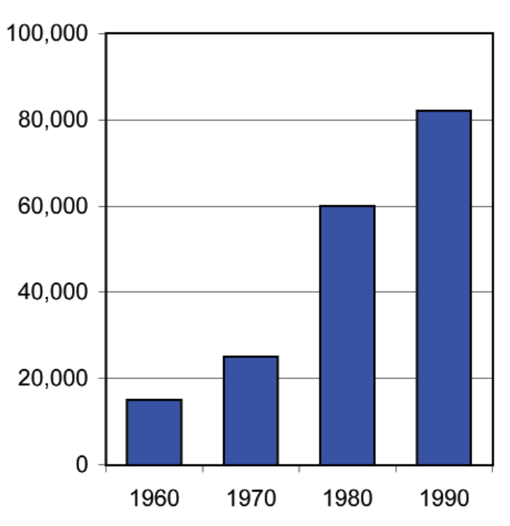
Country	**Current breeding population**	**Current population trend**	**Remarks**	**Red Data Book of the Baltic Region (Ingelög *et al.*, 1993)**
Denmark	800–1200	Constant or slow decline	50,000-75,000 breeding pairs estimated for the beginning of the century	4 Care demanding
Estonia	400–500	Constant decline estimated		Not represented
Finland	ca. 100	Constant increase	First breeding in the early 1980s	1 Endangered
Germany	34–35	Rapid decline	Fringe population in S-H (4 breeding pairs)	1 Endangered
Latvia	2–5	Rapid decline	In the 1970s: about 20 breeding pairs	1 Endangered
Lithuania	5–50	Current trend unknown		1 Endangered
Poland	30–40	Rapid decline	In the 1970s: 80–100 breeding pairs	1 Endangered
Russia, Kaliningrad region	5–8	Decline	Only one of former five regular breeding sites left	1 Endangered
• Russia, St Peters-burg and Pskov oblast • Novgorod oblast	20–30 few	Current trend unknown	Data from beginning of the 1990s	1 Endangered
Sweden	300–320	Decline	Data from 1994	2 Vulnerable
Baltic Sea States	**1,800–2,500**			

with the adoption of the EC Wild Birds Directive and presumably to the fact that it is far less energy-consuming to stop in the Baltic area to breed than to fly all the way to the shores of the Barents Sea (Ganter *et al.*, 1999). In Estonia, through which almost the entire Barents Sea population (approx. 250,000 individuals) passes, the numbers of staging geese during spring migration has increased steadily (Figure 10.12).

Eider (*Somateria mollissima*)

The eider has its main breeding sites at the coast. The population in the Baltic Marine Area increased ten-fold between 1949 and 1985. This increase is thought primarily to be attributable to improved feeding conditions resulting from eutrophication, to the ban on egg collecting in all Baltic Sea States, to the designation of waterfowl reserves and to an increase in hunting restrictions (Hagemeijer and Blair, 1997).

Figure 10.12
Estimated development in the number of staging barnacle geese during spring migration in Estonia.
(Data collected by: C. Herrmann).



The current trends in the various parts of the distribution area differ: In the main breeding area, the Archipelago Sea (about 150,000 breeding pairs), the trend is unclear. In the Gulf of Bothnia (8,400 breeding pairs), population growth continues. In the Gulf of Finland (20,000 breeding pairs), an annual decline of 6–10% has followed the 10% annual population increase of the 1970s and 1980s. In Estonia, the current population numbers 15,000–20,000 breeding pairs. In the St Petersburg region of Russia and in Latvia, Lithuania, Poland and Germany, the species is rare. Regular breeding in low numbers has recently been recorded in some of these areas, however. In Denmark, the steady population increase during much of the 20th Century to 20–25,000 breeding pairs in 1988–90 has been curbed by the loss of 4,000 out of 5,000 pairs in the southwestern Kattegat due to a *Pasteurella* epidemic in 1996, as well as by a subsequent reduction in brood size (Christensen *et al.*, 1997).

In 1996, mass mortality of nestlings occurred throughout the breeding range in the northern Baltic Area. Intestinal helminths, primarily Acanthocephalan worms, may cause large-scale eider mortality, but there is little evidence that these parasites act as a population-regulating factor. Viral pathogeneses are currently being studied intensively. Lead and other trace elements seem also to be a serious mortality factor locally.

Important bird areas (IBAs)

While most of the more important IBAs in lagoons

Table 10.3
General conservation issues for marine mammals in the Baltic Marine Area
Compiled by D. Boedeker based on Chapter 6.4; Haddow, P. 2000. Personal communication; ICES, 1997; Uhd Jepsen, 2000.

	Population beginning of the 20th Century	Estimated population/trend	International protection	Conflict with coastal fishery	Major threats
Harbour porpoise	?	• Baltic Proper: ?/probably slightly increasing • Belt Sea and Kattegat: ca. 40,000/increasing	ASCOBANS, Habitats Directive	Very little	• Contaminants/diseases • Human disturbances • Entanglement in fishing gear
Harbour seal	5,000 (Baltic Proper)	• Baltic Proper: 550/increasing (100 in the 1970s) • Kattegat: 5,700/stable (2,200 in 1976)	Bern/Bonn Conventions, Habitats Directive	Kattegat, Belt Sea and Western Baltic Proper	• Contaminants/diseases • Entanglement in fishing gear • Human disturbances
Grey seal	100,000	• North of latitude 59º: 7,000/increasing (2,000 in the 1970s) • South of latitude 59º: 640/slightly increasing	Bern Convention, Habitats Directive	Severe	• Entanglement in fishing gear • Contaminants/diseases • Human disturbances
Ringed seal	200,000	• Gulf of Bothnia: 3,900/slightly increasing • Bay of Riga: 1,400 • Gulf of Finland: 300	Bern Convention	Minor	• Contaminants/diseases

and other coastal habitats are already protected to some extent, the most important areas for sea ducks located offshore are almost entirely unprotected, in part due to legal constraints. The NATURA 2000 network for the current and upcoming EU Member States in the Baltic Marine Area provides new prospects for protecting marine sites. Very few of the areas designated so far address the conservation priorities of offshore waters, however.

As a first step to obtaining a comprehensive overview of the conservation status of birds in the Baltic Marine Area, the existing national monitoring programmes for marine birds in the Baltic Sea states need to be harmonized and coordinated.

10.2.6. Marine mammals

By: Dieter Boedeker, Harald Benke, Ole Norden Andersen and Rüdiger Strempel

Four species of marine mammals breed in the Baltic Area – three species of seal (harbour (common), grey and ringed) and one species of whale (harbour porpoise). The general conservation issues in the Baltic Marine Area pertaining to these mammals are summarized in Table 10.3.

Harbour porpoise (Phocoena phocoena)

Two distinct populations of harbour porpoises that differ significantly from those in the North Sea are believed to inhabit the Baltic Marine Area, one in the Baltic Sea transition area encompassing the Skagerrak, the Kattegat, the Belt Sea, the Sound and the westernmost Baltic Proper, and one in the central Baltic Proper (Huggenberger *et al.*, 1999).

Comprehensive surveys of population sizes in the North Sea and the Baltic Sea transition area were conducted as an EU project (SCANS) in 1994 and 1995. The results establish the Belt Sea and the Kattegat as an area of European interest and importance for the harbour porpoise (Hammond *et al.*, 1995; Sonntag *et al.*, 1995). Harbour porpoise numbers in the Sound, the Kattegat north of Zealand and the eastern Skagerrak were estimated at about 36,000, while those in the Belt Sea and southern-western Kattegat were estimated at 5,000 (Table 10.3).

The harbour porpoise population in the central Baltic Proper is uncertain, but historical data indicate that it is vulnerable, and possibly facing extinction. One major threat is fishery mortality as indicated by the following figures, albeit that these are uncertain and possibly underestimated (ICES, 1997):

- In Polish waters an average of five harbour porpoises were taken annually in fisheries during the period 1987–1996, mostly in salmon fisheries.
- A further 3–5 porpoises are taken annually in bottom-set gillnets and drift nets along the Swedish Baltic coast.
- An estimated 111 porpoises were taken in German fisheries operating in ICES division IIIc during the period 1987–1995 (Kock and Benke, 1996), and six were taken in 1996.

Other threats in the Baltic Marine Area include pollution, acoustic disturbances, shipping and electric and magnetic fields created by marine cables.

The harbour porpoise population in the Baltic Marine Area has declined markedly over the past 100 years (Siebert *et al.*, 1996). The Baltic Sea States have, therefore, agreed on HELCOM Recommendation 17/2 aimed at protecting the harbour porpoise in the Baltic Marine Area. Highest priority is accorded to reducing mortality in fisheries, especially in salmon drift nets and bottom-set gillnets, and to establishing protected areas for harbour porpoises if and when scientific information indicating efficacy of such areas becomes available.

Seals

The seal species that live and breed in the Baltic Marine Area are the harbour or common seal (*Phoca vitulina*), the grey seal (*Halichoerus grypus*) and the Baltic ringed seal (*Phoca hispida botnica*). Detailed information on distribution, population developments, health status and threats to the three Baltic seal species is provided in Chapter 6.4 and summarized in Tables 10.3 and 10.4.

Until recently, seals were subjected to intensive persecution which led to a drastic decline in populations. In Denmark, hunting of grey seals brought the population near to extinction, and hunting in the Baltic Proper as well as in German coastal waters led to the extinction of the grey seal population in the southernmost Baltic Sea. Moreover, pollution (DDT and PCBs) and diseases have affected the health and reproduction of seals in the entire Baltic Marine Area (Uhd Jepsen, 2000; Chapter 6.4).

The populations of all three seal species in the Baltic Marine Area began to recover in the early 1980s, although the grey seals did not recover as much as previously estimated (Helander and Härkönen, 1999). Population sizes are still far from those estimated at the beginning of the 20th Century (Chapter 6.4; Table 10.3). In 1988 the Baltic Sea States therefore agreed to ban all seal hunting in the Baltic Marine Area until natural health conditions and normal reproduction rates can be demonstrated scientifically (HELCOM Recommendation 9/1).

Although reproduction has improved and the prevalence of most pollution-related disorders seems to be declining, sterility is still found in young ringed seals, and the prevalence of some pollution-related disorders is still high in grey seals, and some are even on the increase (Chapter 6.4). In most Baltic Sea States seals are, therefore, still on their national Red Lists, where the classifications range from “Extinct” to “Vulnerable” (Table 10.4). On the IUCN World Red List of subspecies and populations (1996) the Baltic ringed seal is

Table 10.4
National issues pertaining to seal protection

	Species	Red List	Protection	Reserves	Estimated fisheries mortality	Conflict with coastal fishery
Denmark	Harbour seal	4, Care demanding	General (grants for exemptions)	All major haul-outs	?	Yes, increasing
Sweden	Grey seal	2, Vulnerable	General	21	420 (1996)	Yes,increasing
	Harbour seal	4, Care demanding		15	?	Only Kattegat increasing
	Ringed seal	-		-	ca. 30/yr	No proven evidence
Finland	Grey seal	2, Vulnerable	General (grants for exemptions)	?	338 (1986–1995)	Yes, increasing
	Ringed seal	1, Endangered		?		Minor
Russia	Ringed seal	-	General	2	?	Minor
	Grey seal	1, Endangered				
Estonia	Grey seal	2, Vulnerable	General	2	200–250 (1995)	Yes, increasing
	Ringed seal	-		3	? (few)	None
Latvia	Grey seal Occasional visitor	2, Vulnerable	General (grants for exemptions)	-	Several tens	Minor
	Ringed seal Occasional visitor	-			Probably some	
Lithuania	Grey seal Occasional visitor	1, Endangered	In protected areas	No specific	?	?
Poland	Grey seal Occasional visitor	0, Extinct	General	2 for re-colonization	? (1998: 1)	None
Germany	Grey seal Occasional visitor	0, 4	General	-	(0–2)	None
	Harbour seal Occasional visitor	0, 2	No hunting		(0–1)	

Compiled by D. Boedeker based on Chapter 6.4; Haddow, P. 2000. Personal communication; ICES, 1997; Ingelög *et al.*, 1993; Uhd Jepsen, 2000

Table 10.5
Invasive alien species known to have been introduced into the Baltic Marine Area by 1998 apportioned by introduction vector and region of origin.

Vector	SE Asia & Indo-Pacific	North America	Pacific Ocean	Ponto-Caspian	Other regions	Unknown	Total
Aquaculture (target and non-target species)	4	2	3	-	1	1	11
Intentional (stocking, ornamental, etc.)	1	13	7	10	6	-	37
Shipping	6	12	3	10	8	4	43
Unknown	2	1	-	1	-	-	4
Total	13	28	13	21	15	5	95

Source: Baltic Marine Biologists WG NEMO, 2000. Continuously up-dated list is available on : http://www.ku.lt/nemo classified as “Vulnerable”, while the grey seal population is listed as “Endangered” in the Baltic Sea States. Based on the assumption that the grey seals disappeared from the southernmost Baltic Sea, mainly due to persecution, scientific reintroduction projects were launched in the mid 1990s along the German coast and are presently being considered for the Polish coast. A similar project has been running in Sweden since the early 1980s. The results of these projects will perhaps settle the uncertainties about the viability of the grey seal in the southern Baltic.

Only the population size and the natural health condition of the harbour seal population in the outer Baltic Sea transition area can be considered to have been normal or close to normal during the assessment period (Chapter 6.4; Table 10.4). Priority should be given to further reducing pollution and to elaborating guidelines for a synoptic seal monitoring within the HELCOM Combine Programme in order to increase seal counts.

Seals raid and damage fishing gear and fish-rearing facilities, in some areas rendering fishing with gillnets virtually impossible. So although most people welcome the presence of seals in the Baltic Marine Area, even if they rarely see one, the need to address the conflicts between seals and fisheries is recognized. In an effort to address management and conservation issues pertaining to the Baltic seal populations, HELCOM established a seal project in 1998, among others with the aim of drawing up an action plan.

10.2.7. Recently introduced non-indigenous species

By: Sergej Olenin, Erkki Leppäkoski, Stephan Gollasch, Piotr Gruszka, Krzysztof Skóra, Kai Hoppe, Henn Ojaveer and Marina Orlova

Introduced non-indigenous (non-native, alien, exotic) species are species that have been transferred into areas outside their native range by man. If they are able to establish themselves in and adjacent to the location of introduction, they are regarded as invasive (Olenin and Leppäkoski, 1999). By the year 1998, 95 species of animals and plants that are known or believed to be non-indigenous had been recorded in the Baltic Marine Area and adjacent coastal lagoons and lakes (Figure 10.13). These species include zoobenthos (42), fish (23), phytoplankton (9), phytobenthos (9), nektobenthos (4), parasitic invertebrate (3), zooplankton (3), mammal (2) and bird (1) species. Approximately 66 of these are invasive, that is have established themselves.

Most of the alien marine species that have established themselves in the Baltic Marine Area derive from North American and Ponto-Caspian regions, where the climate and often also salinities are very similar (Table 10.5).

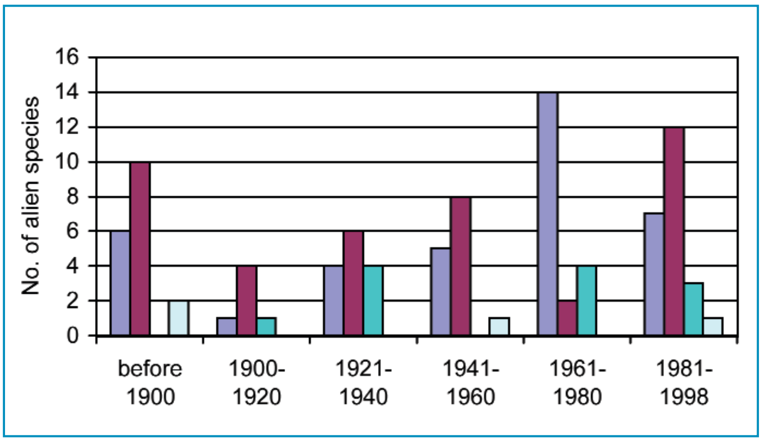


Figure 10.13 Numbers of alien species introduced into the Baltic Sea by various vectors in the 20th Century. (Compiled by S. Olenin).

The dinoflagellate *Prorocentrum minimum* has successfully established itself in much of the Baltic Marine Area over the past two decades (Hajdu *et al.*, 2000). During the previous assessment period it was recorded as a dominant in summer phytoplankton in the Kattegat and in the Arkona and Bornholm Basins. During the present assessment period it bloomed several times in the Gulf of Gdansk (Hajdu *et al.*, 2000).

The rate of invasions has increased during the past two decades, primarily due to increasing ship traffic and intentional introduction for stocking purposes (Figure 10.13). Inland waterways connecting the Baltic Marine Area with the Black and Caspian Seas are of great importance for the dispersal of freshwater and brackish water species colonizing the Baltic estuaries (Gruszka, 1999; Panov *et al.*, 1999). The rate of secondary spread of non-indigenous species within the Baltic Marine Area has been estimated at 30–480 km/yr (Leppäkoski and Olenin, 2000).

The environmental and economic impacts of these introductions are largely unknown. Recent examples of “negative” impacts are outlined below.

The zebra mussel *Dreissena polymorpha* competes with native species for food and space, and it changes benthic habitats. It creates problems by clogging water intake pipes and by fouling ships’ hulls and hydrotechnical constructions in fresh and oligohaline water bodies (Karataev *et al.*, 1997; Olenin *et al.*, 1999). The zebra mussels invaded freshwater parts of the southeastern Baltic coastal lagoons nearly two centuries ago, and in the 1990s spread further into the oligohaline waters of the Gulf of Finland and the Gulf of Riga (Orlova *et al.*, 1998). In the Neva estuary it smothers the shells of the native unionids *Colletopterum piscinale*, *Unio conus* and *Tumidiana tumidus*. In the Gulf of Riga, it co-exists with the native bivalve *Mytilus trossulus*.

Another bivalve, the ship worm *Teredo navalis*, which is not demonstrably native or introduced

Figure 10.14
The HELCOM system of marine and coastal Baltic Sea Protected Areas indicating protection and implementation status.
(D. Boedeker, R. Grunewald, H. von Nordheim).

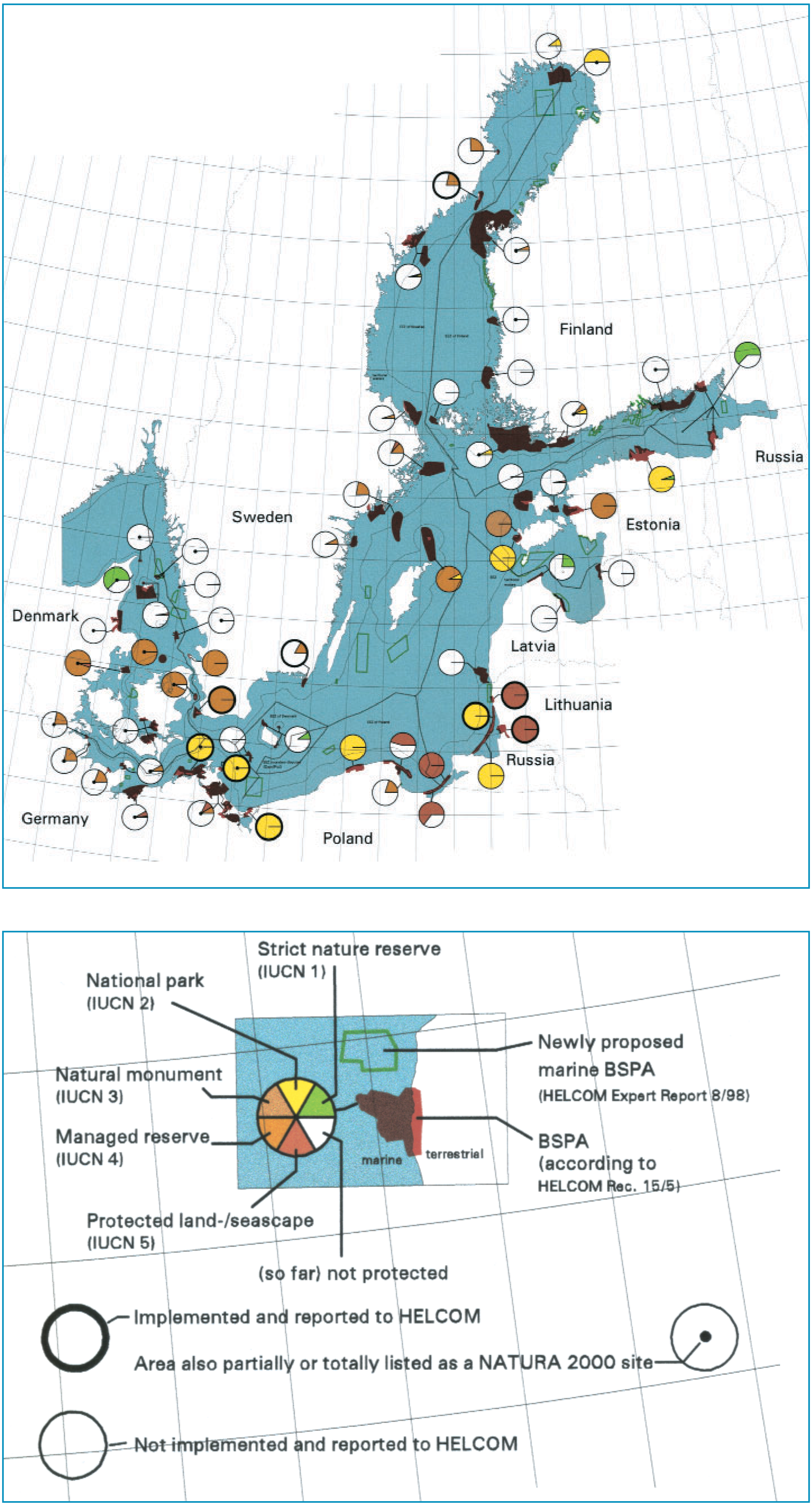


Table 10.6
Baltic Sea Protected Areas: Implementation status of HELCOM Recommendation 15/5 in the various Baltic Sea States.

	Denmark	Estonia	Finland	Germany (M-V/S-H)		Latvia	Lithuania	Poland	Russia	Sweden
Definite maps (1)				2 areas			3 areas	1 area		3 areas
Status	GR NR SAC SPA	NP NR	NP NR (parts)	NP		SNPA	NP RP	NP NR LR		NP NR (parts)
Management plans	Some	1	5 (some partly)	0	0	1	NP	NP	-	3
Establishment	Some	3	None	2	4	3	1	Some		3
Monitoring	Extensive	Extensive	No (but some for birds)	Some	Some	Irregular	Some	Some	-	3
Legislation	No	No	Yes	Yes	No	Yes	No	No	-	No
New BSPA	No	No	15 (SCI and SPAs)	No	No	1	No	1	No	No

Source: Swedish Environmental Protection Agency, 2000

Abbreviations: NP= National Park, NR= Nature Reserve, SAC= Special Area of Conservation (EC-Habitats Directive Area), SPA= Special Protection Area (EC Bird Directive Area), SNPA= Special Nature Protected Area, LR= Landscape Reserve, GR= Game Reserve (1)= All Baltic Sea States have sent general maps to EC4.

(cryptogenic, sensu Carlton, 1996), has been known earlier from the westernmost parts of the Baltic Marine Area. During the present assessment period, it has however expanded its range eastward to the mesohaline parts of the Baltic Marine Area, it has established self-sustaining populations from Kiel Bight to the island of Hiddensee (Gollasch and Rosenthal, 2000). During this period, it has caused damage to vessels and coastal wooden installations in Germany estimated at approx. 50 million ECU.

The North American spionid polychaete *Marenzelleria viridis* spread further into the inner parts of the Gulfs of Riga, Finland and Bothnia in the period 1994–1998 (Kotta and Kotta, 1997; Stigzelius *et al.*, 1997). In the Vistula Lagoon, it has come to constitute up to 95% of benthic macrofaunal biomass (Zmudzinski, 1996). It competes for food and space with the native amphipod *Corophium volutator* and feeds on larvae of native species (Daunys *et al.*, 1999).

Bio-fouling (clogging) of fishing gear by a Ponto-Caspian water flea *Cercopagis pengoi* has become a serious problem in the eastern Gulf of Finland and in Lithuania (GAAS, 2000). Bio-fouling and a drastic decline in fish catches due to the mass occurrence of this species has cost a single Russian fishery company operating off the Neva estuary an average of more than 50,000 ECU annually in 1996–1998 (GAAS, 2000). This predacious cladoceran, which was first observed in 1992 in the Gulfs of Riga and Finland (Ojaveer and Lumberg, 1995), had invaded the Baltic Proper near the Stockholm archipelago and Gotland by 1998 (Gorokhova *et al.*, 2000). Recent reports describe it as a welcome new source of food for pelagic fish, however.

The round goby *Neogobius melanostomus* is still increasing in number in the Puck Bay in the Gulf of Gdansk (Skóra and Stolarski 1993). By 1999, it had spread to the mouth of the river Vistula, adjacent canals (Skóra, 1999), the Vistula Lagoon (Borowski, 2000) and the Rügen area. In the Gulf of Gdansk, this species competes with native fish for space and food and reduces stocks of native molluscs and crustaceans.

New species found for the first time in the Baltic Marine Area during the period 1994–1998 are the oligochaetes *Paranais frici*, *Potamothrix heukeri* and *P. vejdovskyi* (Panov *et al.*, 1999), the crustacean *Gmelinoides fasciatus* occurring in the eastern part of the Gulf of Finland, and the fish *Mugil labrosus* occurring in Puck Bay in the Gulf of Gdansk (Skóra, 1999). The introduction vector for the oligochaetes and fish is unclear, while the crustacean *G. fasciatus* was introduced intentionally into adjacent water bodies (Panov, 1996).

10.3. Baltic Sea Protected Areas and protection of a coastal strip

10.3.1. Baltic Sea Protected Areas

By: Dieter Boedeker and Anna Helena Lindahl

As a first step towards the establishment of a system of marine and coastal so-called Baltic Sea Protected Areas (BSPAs) the Baltic Sea States provisionally notified 62 such coastal and marine areas through HELCOM Recommendation 15/5 62 (HELCOM, 1996a). Consideration is being given to extending the BSPAs to include a number of offshore areas (Hägerhäll and Skov, 1998).

The current situation as regards the establishment of BSPAs is illustrated in Figure 10.14 and Table 10.6. As yet the Baltic Sea States have finally notified nine BSPAs to HELCOM. Only Lithuania made a final notification of all provisionally notified BSPAs (3 areas). From Figure 10.14 it can be seen that several of the BSPAs that have not yet been finally notified already hold national protection status. Denmark has finalized the process so far by notifying 14 of 19 provisionally notified areas (some modified) as EU Nature 2000 areas.

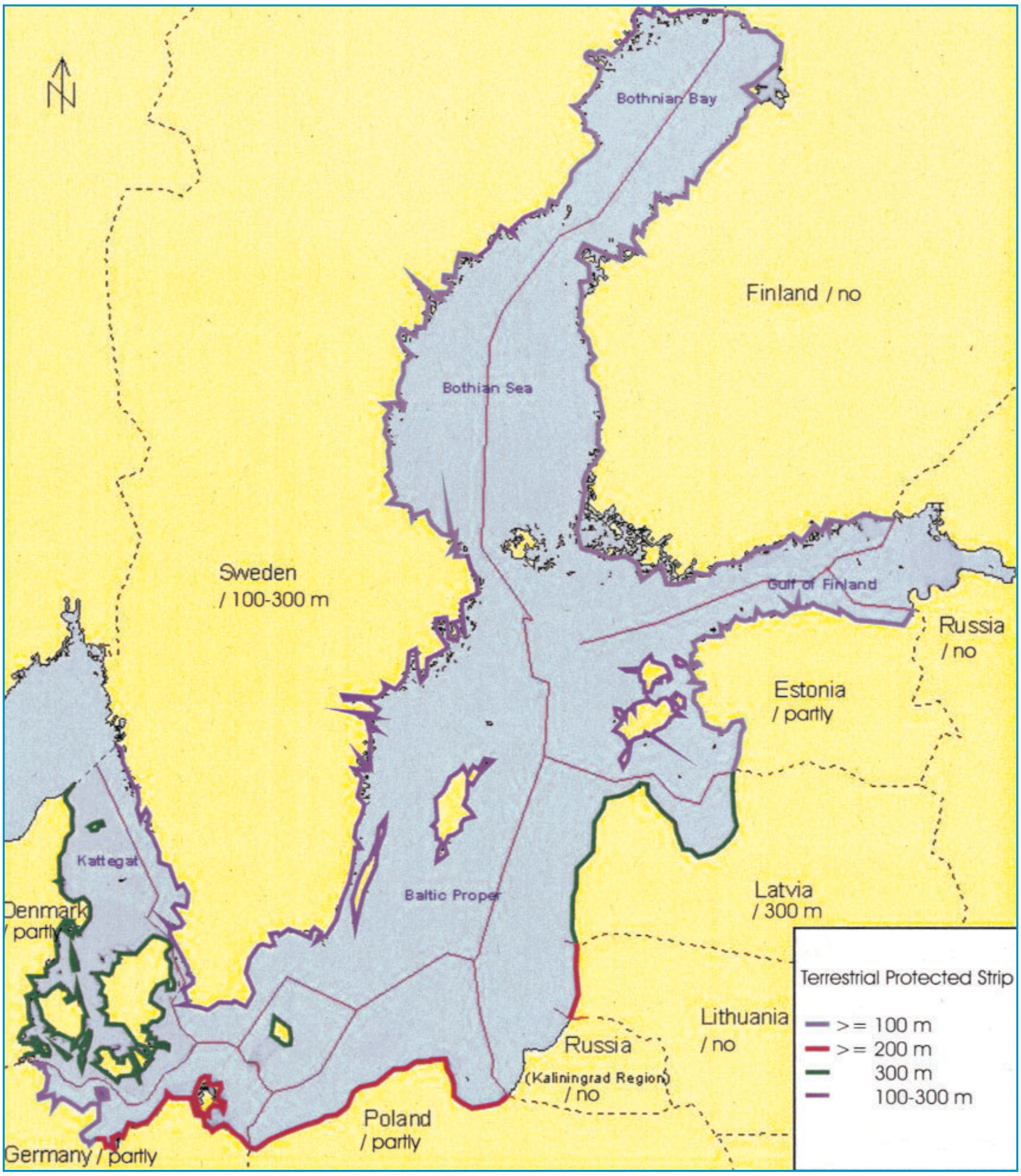


Figure 10.15 Protection of the coastal strip in Baltic States indicating the actual protection status of the seaward coastal strip. (D. Boedeker, based on Nordberg, 1994, and map in Seifert and Kaiser, 1995).

10.3.2. General protection of a coastal strip

By: Lauri Nordberg

In order to stop further degradation of natural coasts, the Baltic Sea States have agreed to protect a coastal strip outside urban areas and existing settlements (HELCOM Recommendation 15/1, 1994). The strip shall extend at least 100 to 300 meters from the mean water line, both landwards and seawards. Within the protected strip, activities that would permanently change the nature and landscape should only exceptionally be permitted, and intensive forestry and intensive farming are to be restricted. A zone of at least 3 kilometres from the mean water line shall be established as a coastal planning zone where major construction projects must be preceded by a land use plan, including an environmental impact assessment.

The recommendation has been largely implemented as far as the terrestrial part is concerned. The situation is less satisfactory as regards the marine part (Figure 10.15). Most countries have established a coastal planning zone or regard the whole country as a planning zone. Most Baltic Sea States have also implemented restrictions on intensive forestry in a coastal strip. Fewer restrictions have been imposed on intensive farming, incentives often being instead used to achieve the same goal.

Overall assessment and conclusions

11

11.1. Conclusions

11.1.1. Natural conditions

The natural forces in the Baltic Marine Area varied considerably during the assessment period, which was characterized by several extreme periods and events, e.g.:

- Unusually warm summers in 1994 and 1997.
- Precipitation in 1998 was the highest ever recorded, at least in Sweden, and basin-wide riverine runoff was the greatest since 1924.
- A long period of stagnation was interrupted by strong inflows of oxygen-rich saltwater to the Baltic Marine Area in 1993 and 1994, followed by renewed stagnation in 1995.
- Inflow of warm saline water resulted in extraordinarily high temperatures in the deep waters of the central Baltic.
- There was extreme river flooding in Sweden in 1995 and in Poland and Germany in 1997.

This large inter-annual variation in natural conditions influenced the assessment work, making description of general changes during the assessment period difficult.

The conditions in the deep waters of the central Baltic during the assessment period were very similar to those following high levels of saltwater inflow in the mid 1970s. Thus after the renewal process and resultant high oxygen concentrations in the central Baltic deep waters in 1994, oxygen depletion recommenced and large areas were affected by oxygen deficiency in 1998.

11.1.2. Pollutant inputs

The environmental quality of the Baltic Marine Area is largely determined by waterborne and airborne transport of substances from the adjoining land areas:

- Total direct and riverine inputs of phosphorus and organic matter to the Baltic Marine Area have decreased considerably since the end of the 1980s because of measures taken to reduce pollutant inputs from towns and industry.
- Total riverine inputs of nitrogen to the Baltic Marine Area have not decreased correspondingly. Diffuse sources thus constitute the main sources of nitrogen input, in contrast to the situation with phosphorus.
- Annual atmospheric nitrogen deposition on the Baltic Marine Area has decreased by nearly 40% in the period 1985–1997.

In order to reduce the nitrogen load to the Baltic Marine Area, additional measures need to be implemented in the next few years. In particular, regulatory measures are needed to reduce anthropogenic loads within as well as outside the Convention Area with regard to:

- Inputs from municipal wastewater treatment plants.
- Diffuse loading from agriculture and transport.
- Inputs from shipping, which comprise one of the largest sources accounting for 12–20% of the total nitrogen deposition.
- Illegal discharges of oil by shipping, which must be terminated.

11.1.3. Eutrophication

Nutrient levels are still generally high compared to the background levels during the 1950s. During the assessment period, the eutrophication situation has developed differently in the various basins of the Baltic Marine Area, e.g.:

- Phosphate levels have continued to decrease in the Kattegat/Belt Sea and Baltic Proper. In the Gulf of Finland, changing stratification and oxygen depletion have caused phosphate release from the sediment of the magnitude of the annual land-based loads.
- Nitrogen levels show no indication of any general change and remain high. The annual variation in the nitrogen load is closely related to the variation in riverine runoff and mirrors the fluctuations in water discharge.
- The extreme riverine flooding events in 1997 in the Oder and Vistula had only a local and temporary influence on the conditions in the Pomeranian Bight and the Gulf of Gdansk.
- Undesirable, toxic and potentially toxic phytoplankton blooms occurred in all parts of the Baltic Marine Area. The surface accumulations of blue-green algae that occurred in summer 1997

were the most extensive ever recorded.
- The inflows of oxygenated saltwater during 1993 and 1994 considerably affected the benthic communities in the Baltic Proper and western part of the Gulf of Finland, as reflected by short-term increases in biomass and abundance.
- The saltwater inflow in the Gulf of Finland also restored the halocline, leading to low oxygen levels in the bottom water and major collapse of the benthic communities during the period 1996–1997.

11.1.4. Hazardous substances

Concentrations of most of the hazardous substances monitored have decreased over the past 30 years in the whole of the Baltic Marine Area, although many of these still cause concern:

- Cadmium concentrations in organisms in the Baltic Proper and in the southern Bothnian Sea are increasing despite the reduction measures hitherto implemented. No simple explanation for this increase has yet been found. Cadmium levels in biota are also generally higher in the Baltic Proper and in the Gulf of Bothnia than in the Kattegat.
- The spatial distribution and concentration of heavy metals in organisms and sediments are generally quite similar throughout the Baltic Marine Area, with the few remaining problems being local and related to past and present-day sources.
- Concentrations of dioxins and PCBs in biota have not decreased during the 1990s in the Baltic Proper, which indicates continuous input and/or resuspension.
- Decreasing concentrations of a number of organochlorine compounds have resulted in concomitant general improvement in the health of birds of prey and mammals. However, ongoing studies show that health and reproduction are impaired, possibly indicating that the present levels of organochlorine compounds such as PCBs and dioxins are still too high.
- Particular consideration should be given to the impacts of organochlorine compounds such as dioxins in Baltic Sea fish, also with regard to the food chain.
- Organotin compounds used as antifouling agents on ships have been detected in sediment and organisms in the Kattegat and Belt Sea area, where damage has been detected in the reproductive organs of certain species (imposex). Present knowledge of organotin distribution in the Baltic Marine Area is very scanty.
- The levels of non-natural radioactivity in the Baltic Marine Area are still high compared to other marine areas due to radioactive fallout from the Chernobyl accident in 1986. However, the concentrations are decreasing and are not considered to comprise a health concern as the associated human radiation doses are around the limits considered to be of no regulatory concern in the EU Basic Safety Standards.
- Dumped chemical warfare agents are not considered a widespread risk to the marine environment. However, a group at risk could be the crews of fishing vessels. Although the dumping areas are clearly marked on nautical charts, chemical munitions are occasionally caught in bottom trawls and hauled on board.

11.1.5. Fishery

Although the International Baltic Sea Fishery Commission regulates the offshore fishing efforts within the Convention Area, over-exploitation is common practice, even resulting in exploitation of stocks beyond safe biological limits. Man-made obstacles and loss of spawning grounds in rivers also constitute threats to migratory fish species such as wild salmon and the near-extinct sturgeon. Other major findings of the assessment include:

- Fishery of the main target fish species such as cod, herring, salmon and eel is presently unsustainable due to over-exploitation and impairment of conditions for reproduction. ICES advice shall focus on this issue.
- Bycatches of marine mammals, sea birds (top predators) and non-target fish species are too high, thus threatening the ecological function and biodiversity of the Baltic Marine Area.
- More accurate catch statistics for commercial species is a prerequisite for any assessment of the fish stocks and of the impact of fishery.
- Permanent and seasonal closed areas and restrictions on certain fishing practices in specific areas should be more widely applied.

11.1.6. Nature conservation and biodiversity

The biodiversity of the Baltic Marine Area is under pressure, not only due to natural variations in hydrographical conditions, but increasingly due to the impact of human activities. Approximately 90% of the marine and coastal biotopes in the Baltic Marine Area are to some degree threatened, either by loss of area or reduction in quality. Specifically, it may be concluded that:

- The most threatened biotopes and species need to be legally protected.
- The Baltic Marine Area Protected Areas need to

be implemented without further delay. These should even be further developed to include the most important bird areas at sea.
- Baltic top predators such as marine mammals and several bird species suffer from pollution, bycatches in fishery, and destruction of their habitats. The use of salmon drift nets and bottom-set gill nets should therefore be restricted in the Baltic Marine Area.
- Invasion of the Baltic Marine Area by non-indigenous species has increased markedly over the past few decades. In this context HELCOM should encourage and support the work of the Baltic Marine Biologists, and the Contracting Parties should work closely together within IMO/MEPC on the elaboration of an International Convention for the Control and Management of Ships' Ballast Water and Sediments.
- Survival of the Baltic wild salmon, *Salmo salar*, is affected by threats of its genetic resources.
- Protection, enhancement or reintroduction of the sturgeon, *Acipenser sturio*, should be actively pursued in order to save it from extinction in the Baltic Marine Area.

11.2. Overall Assessment

11.2.1. Hydrography and climate change

The environmental conditions of the Baltic Marine Area are strongly dependent on the meteorological forcing in the area, the hydrological processes in the drainage basin and the hydrographic processes in the Baltic Marine Area and their interaction. These processes influence the water temperature and ice conditions, the regional inflow of fresh water from the rivers and the atmospheric deposition of pollutants on the sea surface. The exchange of water with the North Sea and between the subbasins together with the transport and mixing of water within the water bodies of the Baltic Marine Area are governed by these processes.

The recent record of mild winters and high zonal activity in the atmosphere prevailed during the present assessment period. Ice coverage remained very modest with the exception of the relatively cold winter in 1995/96. The Gulfs of Bothnia, Finland and Riga were completely ice-covered in winter 1993/94 and 1995/96, however, while only the inner and coastal parts of the gulfs were ice-covered during the remaining winters. The summers of 1994, 1995 and 1997 were warmer than average. Due to low cloudiness and high radiation the water temperature in the upper 15 m layer of the Baltic Marine Area was high in summer 1994 and 1997. The surface temperature sometimes exceeded 23°C in the open parts of the Baltic Proper, corresponding to positive anomalies of up to 5°C.

The summer of 1997 was one of the most dramatic in the area in recent times, with extreme heat and disastrous flooding in the Oder and Vistula drainage basins. 1998 was one of the wettest years on record, with the largest basin-wide river runoff since 1924. In contrast, a wet year in 1994 and dry years in 1996 and 1997 were observed in the Kattegat and Belt Sea area, resulting in large interannual differences in discharge to this marine area during the assessment period. Since the late 1970s, surface salinity has decreased due to the high level of riverine runoff.

The general tendency towards a decreasing frequency and intensity of extreme saltwater inflows observed since the mid 1970s continued during the assessment period. A major inflow in January 1993 in combination with smaller events in 1993–94 and 1997–98 markedly affected conditions in the Baltic deep basins. The entire deep water was renewed in 1993–94 by saline, oxygen-rich but relatively cold water with a temperature of 4–4.5°C. Temperatures of 4°C are among the lowest observed in the eastern Gotland Basin deep water during the 20th century. In May 1994, the whole area was free of hydrogen sulphide, and the oxygen concentrations in the range 3–3.8 ml/l measured in the Gotland Deep at depths below 170 m were the highest recorded since the 1930s. The inflow of very warm, saline and oxygen-rich water in autumn 1997 resulted in temperatures 1–1.5°C above the long-term mean, e.g. above 7°C in the near-bottom layer of the Gotland Deep.

Due to the low inflow activity, a new period of stagnation started in 1995. Oxygen deficiency and anoxic conditions affected large areas of deep water in 1998. In the western Gotland Basin, oxygen depletion continued in the deep water during the assessment period leading in 1998 to the lowest oxygen concentrations recorded since the mid 1980s. The inflows in 1993–94 mainly affected the Baltic Proper, but also strengthened the stratification in the deep layers of the Gulf of Finland, in part accounting for the low oxygen concentrations in the near-bottom layer in summers 1996–98.

The variations in water exchange between the North Sea and the Baltic Marine Area during the assessment period are within the range of natural variation and can hardly be attributed to the climate change. Modelling studies show that global warming may lead to an increase in freshwater inflow to the Baltic Marine Area from the northern parts of the drainage basin and a decrease in the Baltic Proper.

11.2.2. Development in atmospheric and waterborne inputs

Riverine transport of discharges from point and diffuse sources within the river basins accounts for much of the anthropogenic load to the Baltic Marine Area. Direct loading from point sources is also important. A further significant source of pollutant input is deposition on the marine waters and the catchment area.

As the atmospheric phosphorus load is negligible compared to the riverine and direct point-source loads, only the atmospheric nitrogen load will be considered here.

At present it is not possible to provide any reliable information concerning changes in the waterborne pollution load to the Baltic Marine Area for heavy metals and persistent organic pollutants as comparable data are lacking for most of the Baltic States. This is also the case concerning changes in deposition of persistent organic pollutants since monitoring has only been conducted for a few years, and the models for calculating deposition are still incomplete and under development.

Atmospheric inputs

According to model calculations, the annual atmospheric nitrogen load to the Baltic Marine Area has decreased from approx. 330 ktonnes to approx. 200 ktonnes over the period 1985–97. A downward trend has been detected at many stations. Apart from the Baltic States, other European countries and shipping are also important sources of nitrogen loading of the Baltic Marine Area.

Based on model calculations for the Baltic Marine Area for 1997, the estimated total lead load is approx. 150 tonnes/yr, while the estimated cadmium load is approx. 5 tonnes. The emission inventories needed for the load calculations are improving but have not yet been finalized. The lead and cadmium concentrations measured in precipitation at some monitoring stations around the Baltic Marine Area have decreased during the 1990s.

With each of the substances mentioned the atmospheric input is greatest in the southern parts of the Baltic Marine Area and decreases towards the north.

Waterborne Inputs

Comparison of riverine load data for the years 1994–98 indicates that there have not been any significant changes in BOD_7, phosphorus (P_{total}) and nitrogen (N_{total}) loads during the assessment period. The waterborne BOD_7 and P_{total} loads (riverine and point-source) to the Baltic Marine Area have decreased considerably since the end of the 1980s, however, mainly due to improved wastewater treatment and resultant considerable reduction in discharges of untreated wastewater, especially into the major rivers. The improvement is attributable to closure of old industrial plants and construction of new wastewater treatment plants and, in the case of phosphorus, to the installation of phosphorus removal units at most old major municipal wastewater treatment plants and at new plants. The use of detergents with a low phosphate content has also markedly reduced the phosphorus load since 1990.

In contrast, the waterborne nitrogen load to the Baltic Marine Area has not decreased since the end of the 1980s. The nitrogen load is highly dependent on freshwater runoff, to which it is highly positively correlated. In years with high precipitation and hence high runoff, more nitrogen is leached from cultivated areas than in dry years, and the riverine Ntotal load is thus higher. Moreover, in contrast to phosphorus, where wastewater has hitherto been the major source, the majority of the nitrogen load derives from diffuse sources. In most of the Baltic States, measures aimed at reducing N loading from diffuse sources have only just been initiated, whereas measures aimed at reducing wastewater loading of rivers and the sea have been in operation for some years in many of the Baltic States. If the nitrogen load is to be reduced markedly, additional measures will have to be taken in the coming years directed at both municipal wastewater treatment plants and in particular at diffuse sources.

11.2.3. Eutrophication and related effects

Nutrients in the surface layer

As nutrient concentrations in the surface layer are highest and rather stable during winter until the onset of the spring bloom, the winter surface concentrations are used for trend analyses. The concentrations depend on nutrient inputs via rivers, the atmosphere and direct sources, as well as via internal loading from sediments and subsequent mixing into the surface layer. The magnitude of the internal load is determined by the oxygen conditions at the sediment-water interface.

The phosphate concentration has generally decreased significantly in the Kattegat-Belt Sea area and in the Baltic Proper. In the estuaries and bights this is probably attributable to improved wastewater treatment, but in the open marine waters it is probably mainly due to internal processes. In the Gulf of Riga and the Gulf of Bothnia, no clear change in phosphate levels was observed during the assessment period. The winter phosphate pool fluctuated at a high level, probably due a high level of input from diffuse sources and internal loading. In the Gulf of Finland, the phosphate concentration increased markedly after 1996 due to phosphate release from the sediment during the assessment period. In the Gulf of Bothnia, no changes in phosphate concentration were observed.

The nitrogen load closely correlates to runoff, and variation in nitrogen concentration in the coastal areas of the Baltic Marine Area during the assessment period mainly mirrors the variation in runoff, with no general trend being detectable. The nitrogen level is still high due to atmospheric nitrogen deposit-ion and nitrogen fixation by bacteria, especially in the open marine waters. In the Gulf of Riga and in the Gulf of Finland, however, the nitrate concentrations during the present assessment period were the lowest for 20 years.

Extreme flooding of the Oder and Vistula rivers occurred in Summer 1997. Within six weeks, the water transported large amounts of nutrients and contaminants into the Baltic Marine Area. The effects were restricted to the Pomeranian Bight and the Gulf of Gdansk, however, and the situation normalized after about two to three months.

Potential nutrient limitation

Low N:P ratios (8–10:1) are observed in winter in the open Baltic Proper and open Gulf of Finland. This indicates that spring development of the phytoplankton is nitrogen-limited. In summer, nitrogen fixation by cyanobacteria compensates for the nitrogen deficit, and the primary production becomes potentially colimited by phosphorus. Although the N:P ratio has increased in the Kattegat-Belt Sea area and in the Baltic Proper due to a reduction in the phosphorus concentration, potential nitrogen limitation is still dominant in the open sea.

In the bights and estuaries receiving nitrogen-rich fresh water, the nitrate concentrations are high resulting in potential phosphorus limitation in the spring. In the Bay of Bothnia, phytoplankton production is generally phosphorus-limited throughout the productive season. In the Gulf of Riga, nitrogen or both nitrogen and phosphorus potentially limit primary production in the summer. In autumn, nitrogen and phosphorus are potentially colimiting.

A long-term decrease in silicate concentrations is apparent in most parts of the Baltic Marine Area, and silicate now limits diatom growth in the spring in the Gulf of Riga, even though silicate has increased somewhat during the assessment period. Silicate limitation changes the structure of the phytoplankton community rather than limiting the total production.

Near-bottom water

The nutrient situation in the deep basins of the Baltic Proper is mainly affected by the saltwater inflows in January and December 1993 and early 1994 and by the subsequent stagnation period. The Bornholm and Gdansk basins are subject to relatively frequent renewals, however, due to vertical and horizontal exchange processes causing more frequently alternating oxic and anoxic conditions. Under anoxic conditions, nitrate is removed by denitrification and phosphate is released from the sediment and accumulates in the water, as does remineralized ammonium. Under oxic conditions, phosphate is again bound in the sediment and the concentration in the bottom water is low, while ammonium is nitrified and accumulates as nitrate.

The response to the water renewal and subsequent new period of stagnation was clearest in the eastern part of the Gotland Basin. The nutrient distribution in the eastern Gotland Basin changed immediately following the inflow of oxygen-rich water during the assessment period. Ammonium decreased markedly, nitrate increased and the amount of phosphate decreased. The years 1996 to 1998 were characterized by repeated changes in the redox regime, which is a peculiarity of the recent stagnation period, causing alternating distribution of nutrients as described above.

In the western Gotland Basin, oxygen concentrations have decreased continuously since 1993 due to increased stratification of the water column leading to the lowest oxygen content since the mid 1980s. At the end of 1998, oxygen conditions had deteriorated so much that denitrification was initiated, and the phosphate concentrations started to increase.

In the Gulf of Finland, enhanced stratification during the assessment period caused a rapid decline in deeper-layer oxygen conditions. The mean oxygen concentration in the near-bottom layer during the assessment period was less than the mean for the two previous assessment periods and close to that in the 1979–83 assessment period. Extensive anoxia occurred at the sediment-water interface in the eastern Gulf of Finland in summer 1996 resulting in phosphate release from the sediment in amounts close to the annual riverine load. This additional nutrient supply was then available to the phytoplankton growth cycle.

In the Kattegat, Belt Sea and Gulf of Bothnia, the nutrient concentrations in the near-bottom water remained unchanged during the assessment period.

Pelagic system

The onset of the growth season varies from area to area. Normally, an intense spring bloom starts in March in the Kattegat and Belt Sea, in April in the southern Baltic Proper, in April-May in the central and northern Baltic Proper, Gulf of Riga and Gulf of Finland and in May-June in the Gulf of Bothnia. In the southern and western parts the spring bloom is dominated by diatoms, while in the central and northern Baltic Proper, Gulf of Riga, Gulf of Finland

and Gulf of Bothnia it is mostly dominated by dinoflagellates.

The long-term (1979–98) development of the phytoplankton biomass in spring exhibits a significant positive trend in the central and western Baltic Proper, mainly due to the increase in the dinoflagellate biomass. In the Gulf of Finland, no change was observed in the spring bloom. In the northern part of the Baltic Proper the tendency is the opposite: Total mean biomass has decreased by 50%, mainly due to the major reduction in the diatom biomass. In the Gulf of Riga the share of diatoms has decreased in comparison with previous periods, while that of dinoflagellates has increased due to silicate deficiency in spring. No change in the spring bloom has been observed in the Gulf of Finland. In the Kattegat and Belt Sea, the biomass of most phytoplankton groups has decreased in all productive seasons compared to the 1980s.

A summer bloom of nitrogen-fixing cyanobacteria is normal in the Baltic Proper, Bothnian Sea, Gulf of Finland and Gulf of Riga. In warm and calm summers in particular, this bloom is often dominated by potentially toxic *Nodularia spp*. In the warm summer of 1997, blooming cyanobacteria were transported into the Belt Sea, and large numbers were even observed in the southern and central Kattegat.

During the assessment period several large phytoplankton blooms occurred in the Baltic Proper, the adjacent gulfs and the Kattegat and Belt Sea. These included not only cyanobacterial blooms, but also blooms of dinoflagellates such as *Scrippsiella hangoei, Heterocapsa triquetra, Prorocentrum minimum* and *Gymnodinium mikimotoi,* which caused reddish discoloration of the water, and of G. mikimotoi, which caused benthic faunal mortality and oxygen deficit. These blooms were usually relatively short in duration and occurred in all parts of the Baltic Marine Area in summer and early autumn. It seems as if blooms of this kind became more common during the present assessment period than in the previous periods.

An event that has not previously been observed is the “winter blooms” in the Kattegat, Belt Sea and southern Baltic Proper. During the winters 1996/97 and 1997/98 the spring bloom developed in January-February, thereby depleting the nutrient pool and hence preventing development of the “normal” spring bloom in March.

Chlorophyll *a* and primary production exhibited large seasonal and interannual variability, and no changes were observed in most parts of the Baltic Marine Area. In the Kattegat-Belt Sea area both parameters tended to decline over the period 1989–98, mainly due to low biological activity in the extreme dry years 1996 and 1997. In the Gulf of Riga, mean phytoplankton biomass and chlorophyll *a* were lower in all seasons of the present assessment period than in the previous assessment period. The decrease in spring biomass coincided well with the decline in winter nitrogen concentrations observed since the early 1990s.

Mesozooplankton abundance and biomass tended to decrease during the assessment period relative to earlier periods in the Kattegat-Belt Sea, the southern Baltic Proper and the Gulf of Riga, whereas no changes were observed in the Gulf of Finland. No information is available as to whether the increase in biomass observed in the Gulf of Bothnia during the previous assessment period has changed during the present assessment period. No significant changes in species composition have been observed, although the dominant species varied from year to year, mostly in accordance with the variation in temperature and salinity. The Ponto-Caspian cladoceran species *Cercopagis pengoi* was abundant in the warm summer 1997, causing fish net clogging in the Gulfs of Finland and Riga.

Benthic conditions

The inflows of oxygen-rich salt water during 1993 and 1994 considerably affected the benthic communities throughout the open sea areas of the Baltic Proper and the western Gulf of Finland as reflected by short-term increases in biomass and abundance. The subsequent stagnation resulted in considerable decimation of the macrozoobenthos, in some cases even causing total extinction. None of the changes in the open sea benthic conditions could be linked to changes in eutrophication level.

In coastal areas, benthic species diversity and biomass decreased in the Baltic Proper but increased significantly in the Gulf of Riga and the southern Gulf of Finland. The colonization of coastal areas by a new bristleworm for the area, *Marenzelleria viridis*, is noteworthy. Perennial benthic vegetation declined in the northern part of the Gulf of Riga, as well as in many Danish fjords. At the same time, recovery of the bladder wrack *Fucus vesiculosus* population was observed in the Gulf of Finland and the southern Gulf of Riga.

11.2.4. Contaminants, including radioactive substances and dumped chemical munitions

Spatial variation

The concentration of metals and organic pollutants has been investigated in sediment and biota samples throughout the Baltic Marine Area. Of the metals studied in biota (Cd, Cu, Pb, As, Hg and Zn), only Cd exhibited systematic spatial variation, with the highest concentrations being found in the southern part of the Gulf of Bothnia and in the Baltic Proper. With the other metals, local variation is observed that is probably related to urban activities, but this is generally less than one order of magnitude. Sediment concentrations of Hg were highest in the Bay of Bothnia and the eastern Gulf of Finland, while the concentration of Cd, Zn and Cu was highest in the central basin of the Baltic Marine Area. High sediment As concentrations were only recorded in the Bay of Bothnia. Pb seems to be evenly distributed.

DDT compounds exhibited systematic spatial variation. Concentrations were 5–10 fold higher in the central part of the Baltic Marine Area than in the Bay of Bothnia and the Kattegat. The PCB concentration in herring was lower in the Bay of Bothnia and the Kattegat than in the central part of the Baltic Marine Area. With coastal species there were no significant interregional differences in PCB concentration. Local sources obviously play an important role in explaining the spatial variation at a more local level. Data from both water and biota indicate that HCH concentrations are generally higher in the southern parts of the Baltic Marine Area. Moreover, comparison of the distribution of the various HCHs provides evidence for the recent use of lindane in some West European countries. HCB concentrations vary along the Baltic coast and indicate ongoing industrial contamination of the Baltic Marine Area. Sediment data indicate that PCBs are rather evenly distributed, although with the highest concentration in the Eastern Gotland Deep. High concentrations of PAHs were observed in southern basins such as the Gulf of Gdansk, the Lübeck Bight, the Mecklenburg Bight and the Arkona Basin. As regards other important contaminants such as DDT, HCB, organotins and dioxins, insufficient comparable data are available to provide an overall picture of their spatial distribution in sediment.

Organotins have been detected in Baltic abiotic and biotic matrices including marine birds and mammals, mainly in the southernmost parts. Present knowledge is too limited to allow a comprehensive evaluation, however.

The most significant source of artificial radioactivity (^{137}Cs and ^{134}Cs) in the Baltic Marine Area is the fallout from the 1986 Chernobyl accident. Other important sources are global fallout from atmospheric nuclear weapons tests performed during the late 1950s and early 1960s and discharges from nuclear reprocessing plants in Western Europe (Sellafield and La Hague). Presently, the reprocessing plants are of minor importance as sources of radioactive contamination of the Baltic Marine Area.

Temporal trends

The concentration of lead is decreasing in almost all the biological matrices investigated, which is an encouraging confirmation of the positive impact of international measures to protect the environment. Copper and zinc concentrations in biological material seem to be stable over time. The copper concentration in water has also been stable for the past 10 years. The trend analyses for mercury concentrations are contradictory, both statistically significant increases and decreases having been found.

The cadmium concentration seems to be increasing in many biological matrices from the central Baltic Marine Area, but not in water samples from the southern Baltic Proper, where a decrease was observed during the 1980s. The increase is an unexpected trend since many data indicate that discharges have decreased during the past 20 years. At present there is no simple explanation for the increasing concentrations observed.

The sediment concentrations of the heavy metals investigated (Hg, Pb, Cd, As, Cu and Zn) generally tend to increase towards the sediment surface, although a decrease is sometimes observed within the upper few centimetres. The latter could be partly attributable to lower input (discharges) of contaminants, but may also be partly due to diagenetic processes within the highly dynamic superficial sediment. As regards sediment it can be concluded that although the laboratories are able to provide reliable data from an analytical point of view, interpretation of these data (such as correlation to the water phase or discharge data or bioavailability and fate of deposited contaminants) is presently only at the discussion and development stage. To improve the situation there is a need for basic research on relevant processes.

With all the chlorinated organic compounds studied (DDT, PCB, HCH, HCB, and dioxins) the concentrations have decreased over the past 30 years irrespective of the biological matrix and region of the Baltic Marine Area in question. As regards water samples, reliable time series are only available for HCHs, but these confirm the decrease. In biota samples, concentrations have continued to decrease over the past 10 years except for PCB and dioxins in the Baltic Proper, where the concentrations have remained more or less constant, indicating continuous release of the compounds into the environment.

The polybrominated compounds used as flame retardants have generally decreased in concentration in guillemot eggs over the past 10 years (PBDE), although the concentration of some of them has increased (HBCD).

Since the Chernobyl accident the levels of manmade radionuclides in the Baltic Marine Area have generally been declining, mainly due to radioactive decay and outflow of water through the Belt Sea and the Kattegat. In contrast, discharges of radionuclides from Sellafield have increased since 1994, affecting European coastal waters including the Baltic Marine Area, to which small but measurable amounts of ^{99}technetium and ^{129}iodine are transferred via inflow from the North Sea.

The situation concerning the dumping of munitions has not changed appreciably since the last assessment. Small amounts of mustard gas are still inadvertently caught each year by Danish fishermen in the major dumping areas near Bornholm. New information on the toxicity of the chemical warfare agents to aquatic organisms has only been obtained for Clark I and II, and no new chemical measurements of chemical munitions have been reported. Consequently there is no new information available that changes the estimated risk to man and the environment since the previous assessment (HELCOM, 1996).

Effects

Widespread effects of pollutants encompassing the entire Baltic Marine Area are mainly seen in marine top predators feeding on Baltic fish. Following the decrease in environmental concentrations of organochlorines in particular, health improvements have been recorded. The size of the Baltic seal population has thus increased, although problems remain. The low rate of recovery of the ringed seal is probably attributable to the prevalence of uterine occlusions in female seals, including young individuals, possibly due to excessive dietary concentrations of PCB and dioxins. The rate of increase of the grey seal and harbour seal populations in the Baltic Proper is somewhat lower than in the Gulf of Bothnia and in the Kattegat, respectively, possibly due to contaminants, habitat destruction or fishery bycatch.

The prevalence of the lesions that comprised the disease complex found in Baltic seals generally seems to be declining and their health seems to be improving. This is especially true for the reproductive organ lesions. The prevalence of moderate to severe chronic intestinal ulcers is rapidly increasing among young grey seals, however, probably due to immunosuppression caused by environmental pollutants, experimental studies having demonstrated immunosuppression in harbour seals fed on Baltic fish. The concentration of the detoxification enzyme EROD in perch liver from the Baltic Proper has increased since the end of the 1980s, although the reason for this is unclear.

The white-tailed sea eagle population has increased and its reproductive capacity is now close to that in the pre 1950s. Moreover, eggshell thickness has returned to the pre 1950 level in the fish-feeding Baltic guillemot. The mean brood size of the Baltic white-tailed sea eagles has levelled off at a lower number than prior to 1950, however. The latter coincides with cessation of the decrease in environmental concentrations of dioxins and PCB.

At present there are no data indicative of a significant ongoing fall in dioxin and PCB concentrations in the Baltic Proper. Because of the present concentrations of these compounds in Baltic fish, the Swedish Food Administration recommends women of reproductive age to cut back on consumption of Baltic herring and salmon. Present conditions thus hamper utilization of fatty Baltic fish in Sweden.

Comparison of the radiation doses in seafood arising from naturally occurring radioactivity and man-made radioactivity reveals that the dose rates and doses are highest from natural radioactivity except for the year 1986, when the dose rate from Chernobyl fallout approached that from natural radioactivity in some regions of the Baltic Marine Area. The maximum individual annual dose to which critical groups in the Baltic Marine Area were exposed during the period 1950–98 is estimated to be 0.2 mSv/yr, which is less than the limit of 1 mSv/yr set for the general public in the Basic Safety Standards (EC, 1996). The radiation dose to man resulting from liquid discharges from nuclear power plants in the Baltic Marine Area are estimated to be equivalent to or less than the levels considered to be of no regulatory concern mentioned in the Basic Safety Standards.

11.2.5. Fishing, mariculture and fish diseases

Impact on stocks

Fishing mortality has increased as regards Baltic cod (in ICES fishing areas 25–32) and herring (in ICES fishing areas 25–29, 32), and fishery of the two species is now unsustainable. The sprat stock has increased considerably over the past 10 years due to reduced predation from cod and favourable environmental conditions for hatching. As regards salmonids, human impact (intensive exploitation, pollution, damming of rivers, etc.) has substantially reduced the wild population size and deteriorated the reproduction conditions. Despite some indications of a reduction in prevalence of the M74 syndrome in 1998–99, this disease continued to pose a major threat to the survival of the few remaining naturally reproducing salmon stocks in the Baltic Marine Area. Moreover, the eel stock is depleted and recruitment is impaired over its entire area of distribution.

Fishery affects the species composition and the size distribution of both the main target species and the non-commercial fish stocks. Discard levels are high and bycatches of marine mammals, sea birds and non-target fish are of great concern for the ecosystem and among other things counteract the intentions of the HELCOM recommendations concerning the protection of seals, harbour porpoise and wild Baltic salmon.

Mariculture may lead to nutrient enrichment problems locally. Moreover, the escape of mariculture fish and the introduction of non-native species may have ecosystem consequences in terms of genetic change and distorted competitive relationships. The mass rearing and release of salmon has reduced the genetic variability within hatchery stocks and might influence the genetic variability of the few remaining wild populations.

Effectiveness of measures

IBSFC regulates the offshore fishing efforts within the Convention Area according to the "FAO code of conduct of responsible fisheries". For socioeconomic reasons the TACs set by IBSFC have often exceeded the scientifically recommended limits. In view of the uncertainties in total stock assessments, such high TACs have frequently contributed to the exploitation of stocks beyond safe biological limits. Despite these regulations, over-exploitation is common practice among fishing fleets in the Baltic Marine Area.

Knowledge limitations

More accurate fish catch statistics, including data on discards and bycatches of harbour porpoise, seals and sea birds are needed. In particular the development of reliable fish catch statistics for the coastal areas, which are presently almost non-existent, is a prerequisite for any assessment of coastal fish stocks.

Further studies on the ecosystem effects of commercial fishery are urgently needed in order to assess both the effects of removal of target species on food web structure and the physical impact caused by fishing gears. As an example, the M74 syndrome may be linked to changes in the food web as a result of the high exploitation of certain fish stocks.

The impact of fishery practices on the marine ecosystem is a matter of considerable concern. A proposed action plan has thus been drawn up for the eastern cod stock. An action plan also exists for sal-mon that indicates an urgent need for regulatory action. In addition, recent concern for the Baltic eel population has resulted in calls for a recovery plan.

Selective fishing gears need to be developed to avoid bycatches of marine mammals and birds.

There is a need to consider the introduction of geographically and seasonally closed areas for fishing as an important conservation measure to ensure sustainable management of the living resources.

11.2.6. Biodiversity and nature conservation

Biodiversity is low in the Baltic Marine Area due to the low and variable salinity as well as to its bathymetry and prehistory as a freshwater lake.

As a major activity, HELCOM developed a red list in 1998 specifying all 133 Baltic marine and coastal biotopes and 13 selected biotope complexes. The overall threat assessment for the entire Baltic Marine Area gives some cause for concern because most (88%) of the biotopes and all biotope complexes are considered to be threatened to some degree.

A corresponding red list for Baltic Marine Area coastal and marine species is under preparation. Several activities and pressures considered harmful to biotopes according to the red biotopes list are also considered to threaten species or groups of species.

Eutrophication has caused changes in plankton species composition, at times involving toxic phytoplankton species, and shifts in the occurrence of blooms, all of which can affect other biota. Phytoplankton also incorporates hazardous substances into the pelagic food chains, ultimately negatively effecting higher organisms such as mammals and birds.

Phytobenthos species diversity and depth distribution has declined markedly over large areas during the past few decades due to pollution. Although the situation for the bladderwrack, *Fucus vesiculosus,* has improved, it is still not found at the depths where it was previously recorded. Eelgrass, *Zostera marina*, which is found on level sandy bottoms, is an example of one of the most endangered biotopes in the Baltic Marine Area. While on one hand the zoobenthos has benefited from greater phytoplankton production, it has also suffered from eutrophication-enhanced oxygen depletion and human activities at sea.

Invasions of non-indigenous species have had profound negative as well as positive effects on native biota and human interests, and more invasions can be expected in the future, e.g. via ballast water. The environmental and economic impacts of these invasions are largely unknown, although extensive damage has been recorded e.g. in the Gulf of Finland.

As with other taxa, the number of marine fish species declines the further the distance from the North Sea. Among the widely distributed anadromic species, the wild salmon is highly endangered. An inventory of salmon rivers has been made. A reintroduction project for the Atlantic Sturgeon – possibly no longer present in the Baltic Marine Area – is now running under HELCOM.

The Baltic Marine Area encompasses many highly important staging areas for sea birds, and more than 30 species breed along the shores. Proper conservation of these birds may call for coordination of national monitoring programmes and for the establishment of protected areas.

The harbour porpoise, the only cetacean occurring regularly and breeding in the Baltic Marine Area, is numerous in the outer parts of the area, but very scarce in the Baltic Proper, where bycatch may pose a major problem.

Baltic seal populations – harbour seals in the outer parts and grey seals in the inner and central parts, and the ringed seals in the innermost parts – are generally increasing. In the Gulf of Finland, however, development of the ringed seal population is alarming. Its former population size and distribution have not yet been regained, and its health condition and fecundity have not yet returned to normal.

Although many are already protected nationally, only nine of 62 marine and coastal Baltic Sea Protected Areas (BSPAs) notified according to HELCOM Recommendation 15/5 have been nationally implemented and reported to HELCOM. Some 20 additional, mainly offshore marine areas were identified for possible inclusion in the BSPA system in 1998. Management plans for five major coastal wetlands in the southeastern Baltic Marine Area proposed by HELCOM also have nature conservation implications.

11.2.7. Environmental effects of measures implemented

The following section present examples of long-term changes in the Baltic marine environment resulting from measures implemented to protect it. The focus is on three of the major Convention issues: Eutrophication, contamination and biodiversity, including the impact of fishery and shipping, the aim being to illustrate the need for joint international action to combat transboundary pollution and verify the effectiveness of the measures implemented. The following are only examples, however, and do not represent a complete and comprehensive summary of available information from the monitoring activities.

The examples are presented below grouped according to the outcome of the measures implemented.

Examples where the measures implemented have led to environmental improvements

Eutrophication. The reduction in point-source phosphorus loading from the catchment area has resulted in reduced phosphorus concentrations in estuaries and coastal areas. The decrease in winter phosphate concentrations in seawater was particularly pronounced in the second half of the 1990s. The reduction in point-source nitrogen loading has only had local effects since most nitrogen comes from diffuse sources.

Contaminants. As a result of the joint international measures taken, the formerly so serious DDT pollution has decreased substantially in the Baltic Marine Area, and concentrations of the substances have decreased more than 90% in all investigated matrices.

Biodiversity. The recolonization of coastal areas of Tallinn Bay by *Fucus vesiculosus* shortly after the reconstruction of a wastewater treatment plant is a good example of a measure having a beneficial effect on the biological community.

Another example is recovery of certain fish-eating bird species that previously suffered from thin egg shells and reduced fecundity.

This improvement also implies restoration of the top of the food web, i.e. the very part of the food web that used to be extinct in most places in the Baltic Marine Area.

Examples where the measures implemented had an initial effect that has now slowed down

Contaminants. After joint measures were implemented to reduce the environmental concentrations of dioxins and PCB, concentrations of the two substances decreased rapidly in the Baltic Marine Area. During the past 10 years, however, the environmental concentrations of PCB and dioxins have levelled off, suggesting that some input of these compounds continues.

Biodiversity. As a consequence of the measures to reduce environmental concentrations of dioxin and PCB, some of the most threatened species, namely the Baltic grey seal, the ringed seal and the white-tailed sea eagle, have recovered to some extent. In the case of the ringed seal and white-tailed sea eagle, however, the improvement ceased when the dioxin and PCB concentrations in Baltic fish levelled off.

In the southwestern Baltic Proper the grey seal and harbour seal populations are improving at a slower rate than in the rest of the Baltic Marine Area. Habitat destruction and disturbance combined with fishery bycatch probably account for the delay in restoration of the ecosystem of the Baltic Marine Area.

Examples where the measures implemented have had no discernible effect.

Eutrophication. Despite the decisions taken to reduce diffuse nitrogen loading, yet no significant changes have hitherto been observed in the pelagic ecosystem of the Baltic Marine Area.

Contaminants. Despite the measures implemented to reduce Cd discharge to the marine environment, the concentration of this metal is tending to increase in many fish species. This development is unexpected since the concentrations in the seawater are decreasing, and a number of studies indicate reduced discharge of Cd to the environment in recent years. At present there is no simple explanation for the increase in biota Cd concentrations, and further investigation is needed.

Under MARPOL 73/78 the Baltic Marine Area has been designated as a Special Area with regard to oil and waste from ships. In addition, Baltic Sea States have agreed on a common strategy to prevent discharges of oil and other waste from shipping. From the year 2000, adequate reception facilities should be available in ports and terminals.

Despite existing restrictions to prevent discharges of oil into the sea, violations are still frequent, however, and the number of observed oil slicks has not decreased significantly over the years. This calls for attention as future substantial increases in the total amount of oil transported at sea are anticipated.

Biodiversity. Despite decreasing concentrations of hazardous substances in the Baltic Marine Area, the sea-going otter is still absent or locally very rare. In the past, the otter was a regular inhabitant of the archipelagos and the shore line areas of the Baltic Marine Area. The concentration of PCB in particular is still so high in coastal fish that they do not comprise a healthy food resource for the otter, and the food web of the archipelagos therefore lacks a mammalian species.

Despite regulatory efforts to reduce the impact of fishery upon the commercial fish stocks, evidence is still lacking of improvement in the Baltic cod population, the herring population and the wild salmon population. The spawning stocks of these species are beyond safe biological limits.

Examples where measures have been implemented but where the environmental response has not been followed due to lack of appropriate methods or resources

Eutrophication. More reliable forecasts of the effects of induced changes in nutrient loading require a more comprehensive knowledge of the mechanisms of the pelagic system and interaction with the sediment than is available today. Moreover, many of the components of the system are not included in the monitoring programme or are only included at few stations, e.g. nitrogen fixation, bacterial biomass and production, microzooplankton biomass, sedimentation of organic matter, oxygen consumption and others.

Contaminants. A number of measures have been implemented to reduce oil pollution of the Baltic Marine Area, but at present there is no appropriate monitoring of the environmental responses.

Biodiversity. The state of biodiversity in a marine area reflects the eutrophication level, the degree of contamination, the climatic conditions and management of the area's natural resources. Efficient methods and the financial resources to follow biodiversity have hitherto been lacking and as a consequence, suitable time series for determining past temporal changes in biodiversity are also lacking.

Thus even though appropriate measures to improve biodiversity may have been implemented in the past, the Baltic Sea States lack the appropriate data to demonstrate the effects of these measures.

11.3. Perspectives and areas where monitoring needs to be improved

HELCOM presently has two separate monitoring programmes – one for inputs and one for their effects in the sea. The present monitoring and assessment activities have hitherto resulted in four comprehensive high-quality assessment reports on the state of the environment in the Baltic Marine Area, as well as three Pollution Load Compilations. The timing of the scientific assessment reports and Pollution Load Compilations has been out of step, thus making analysis of causal relationships difficult. Moreover, the assessment procedures have been rather cumbersome, although they have steadily improved. The concept of five-year assessments will inevitably fail to provide timely reports, however.

The COMBINE Programme is based on scientific traditions of the 1960s and 1970s, predominantly encompassing traditional sampling approaches and techniques. It has resulted in a valuable data set, including high-quality, long-term time series of several variables representing different ecosystem parameters.

11.3.1. Improvement of the monitoring and assessment work

During the present assessment procedure, several shortcomings were identified that should be improved or supplemented:

- The present programme is not guaranteed but is susceptible to the varying scientific priorities of the responsible institutes and changing financial priorities of the Baltic Sea States.
- Pressures and effects can seldom be linked due to the different monitoring strategies employed in the two components of the monitoring programmes.
- Regional coverage of the data is uneven due to the varying capacity and interest of the Baltic Sea States in carrying out their commitments.
- All the essential parameters and compartments are not necessarily included in the programmes due to the lack of financial resources or capacity. Methods for monitoring the biological impact of hazardous substances are particularly lacking.
- Sampling frequency is often too low to allow proper statistical trend analysis, especially in the case of the pelagic variables.
- Quality assurance (QA) procedures are not satisfactory to guarantee the comparability of all data.
- A fully functioning joint database has not been realized, and the data flow from sampling and analysis to the database is sporadic.
- Access to all the available data is usually difficult.
- All Baltic Sea States should carry out the same basic programme, and few attempts have been made to share the sampling and analysis work according to the capacity of the Baltic Sea States.
- The present programme does not sufficiently encompass biodiversity issues, particularly as regards fish, birds and marine mammals.

11.3.2. Proposals for action and further development

To improve the present situation, HELCOM's Monitoring and Assessment Group (MONAS) should consider the following actions:

- The present monitoring programme and time series should be guaranteed in the future; despite the varying scientific priorities of the responsible institutes, Baltic Sea States should honour their commitments to support the joint programme.
- Monitoring and assessment should be a regular and integrated process including regular data flow from measurements and QA procedures to assessments.
- The workload should be divided in a more effective manner.
- Numeric models, automatic sampling and recording systems including satellite imagery should be developed and implemented more effectively.
- A fully functioning database as well as effective and continuous data exchange are prerequisites for effective assessment work.
- Socioeconomic driving forces should be included.

The assessment procedures could be developed more effectively and timely as follows:

- Agreed assessment criteria should be clear and include reference values and environmental quality criteria.
- More frequent thematic assessments and indicator reports with clear targets and schedules should be produced whereafter the frequency of background assessments could be considered.
- Thematic assessments and indicator reports should be made available as printed publications supplemented by information on the Internet.

A data exchange and information system could be developed as follows:

- Interactive databases with possibilities to produce maps (incl. GIS) and graphs (with statistical analysis) and to extract necessary information, etc.
- Direct Internet connection is essential.
- Databases should provide possibilities to submit all data.
- Document databases should be established.

References

12

Agger CT and Ærtebjerg G (1996). Long-term development of autumn oxygen concentrations in the Kattegat and the Belt Sea areas. Proceedings of the 13th Symposium of the Baltic Marine Biologists: 29–34.

Alexandersson G (1980). Shipping in the Baltic. AMBIO 9(3): 158–162.

Alexandersson H, Schmith T, Iden K and Tuomenvirta H (1998). Long-term variations of the storm climate over NW Europe. The Global Atm. and Ocean System, 6: 97–120.

Alexandersson H, Tuomenvirta H, Schmith T and Iden K (2000). Trends of storms in NW Europe derived from an updated pressure data set. Clim. Res. 14: 71–73.

Alimov AF (Ed.) (1988). Communities of freshwater invertebrates in the macrophyte beds. Proceedings of the Zoological Institute, Leningrad, 186 pp. (in Russian).

Alimov AF, Telesh IV, Välipakka P and Viitasalo I (1997). Biological state of the coastal waters of the Gulf of Finland. Report of the Working Group "Biological State and Changes". In: Proceedings of the Final Seminar of the Gulf of Finland Year 1996, March 17–18, 1997 Helsinki. pp. 161–166.

Almkvist L (1986). Survey of Baltic grey seal along the Swedish coast. Proceedings from the Symposium on the Seals in the Baltic and Eurasian Lakes. Savonlinna, Finland, 5–8 June, 1984. Finnish Game Research 44: 19–27.

Andersin A-B (1986). The question of eutrophication in the Baltic Sea – results from a long-term study of the macrozoobenthos in the Gulf of Bothnia. Publ. Water. Res. Inst. Finland 68: 102–106.

Andersson A, Hajdu S, et al. (1996). Succession and growth limitation of phytoplankton in the Gulf of Bothnia (Baltic Sea). Marine Biology 126: 791–801.

Andrushaitis A, Seisuma Z, Legzdina M and Lenshs E (1995). River load of eutrophying substances and heavy metals into the Gulf of Riga. In: Ojaveer E (Ed.) Ecosystem of the Gulf of Riga between 1920 and 1990. Estonian Academy of Sciences, Tallinn, Estonia, pp. 32–40.

Andrushaitis G, Andrushaitis A, Bitenieks J et al. (1992). Balance of organic carbon of the Gulf of Riga. In: Proceedings of the 17th Conference of Baltic oceanographers, Norrköping, 1990, SMHI, 12 pp.

Anonymous (1992). Japansk tang i Kattegat – Havbiologerne forundrer sig, Vendsyssel Tidende, 7. November 1992 (In Danish).

Anonymous (2000). BLMP Meeresumwelt 1994–96, BSH, Hamburg and Rostock, pp.120–124.

Anonymous (2002). BLMP Meeresumwelt 1997–98, BSH, Hamburg and Rostock

Anonymous (1998). Water Protection Targets to 2005 (in Finnish with English summary). Finnish Environment Institute, Suomen ympäristö 226: 1–82.

Anonymous (1996). Mat för två. Information to the public. Swedish Food Administration.

Anonymous (2000). Data from the Swedish Contaminant Monitoring Programme.

Astok V, Matthäus W, Berzins V, Carlberg S, Cyberska B, Elken J, Lange W, Launiainen J, Nomm A, Suursaar Y, Tamsalu R and Vihma T (1990). Hydrography. In: Second periodic assessment of the state of the marine environment of the Baltic Sea, 1984–88; Background Document. Baltic Sea Environmental Proceedings, 35B, pp. 21–68.

Atuma SS, Linder C-E, Andersson Ö, Bergh A, Hansson L and Wicklund-Glynn A (1996). CB153 as indicator for congener specific determination of PCBs in diverese fish species from Swedish waters. Chemosphere. Vol 33: 1459–1464.

Aulerich RJ and Ringer RK (1977). Current status of PCB toxicity to mink, and effect on their reproduction. Arch. Env. Contam. Toxicol. 6: 279–292.

Aune M et al. (2000). Personal communication. Data on DDT concentrations in Baltic salmons during late 1990s from the Swedish Food Administration.

Aune M, Atuma S, Darnerud PO, Wicklund-Glynn A and Cnattingius S (1999). Analysis of Organochlorine Compounds in Human Milk. In: Organohalogen Compounds 41: 87–90.

Axell LB (1998). On the variability of Baltic Sea deepwater mixing. J. Geophys. Res. 103: 21667–21682.

Bachor A (1996). Nährstoffeinträge aus Mecklenburg-Vorpommern in die Ostsee 1990–95. Wasser und Boden 48 (8): 33–36.

Baltic Maritime Outlook 2000. Swedish Maritime Administration (1999). ISSN 91–86502–15–8.

Balzer W (1984). Organic matter degradation and biogenic element cycling in a near shore sediment (Kiel Bight). Limnol. Oceanogr. 29: 1231–1246.

Bartnicki J, Barrett K, Tsyro S, Erdman L, Gusev A, Dutchak S, Pekar M, Lukewille A and Krognes T (1998). Atmospheric supply of nitrogen, lead, cadmium, mercury and lindane to the Baltic Sea. EMEP Centres Joint Report for HELCOM EMEP/MSC-W Note 3/98.

Bartnicki J, Gusev A, Pavlova N, Ilyin I, Lükewille A and Barrett K (2000). Atmospheric supply of nitrogen, lead, cadmium, mercury and lindane to the Baltic Sea in 1997. EMEP Centres Joint Report for HELCOM, EMEP/MSC-W Note 6/99, Norwegian Meteorological Institute Research Report No. 97.

Bavel van B, Näf C, Bergqvist P-A, Broman D, Lundgren K, Papakosta O, Rolff C, Strandberg B, Zebûhr Y, Zook D and Rappe C (1995). Levels of PCBs in the aquatic environment of the Gulf of Bothnia: Benthic species and sediments. Marine Pollution Bulletin 32 (2): 210–218.

Bengtsson B-E (1975). Accumulation of cadmium in some aquatic animals from the Baltic Sea. In: 3rd Soviet-Swedish Symp. on the Baltic Sea pollution, Stockholm. NBL-report.

Bengtsson B-E, Hill C, Bergman Å, Brandt I, Johansson N, Magnhagen C, Södergren A and Thulin J (1999). Reproductive Disturbances in Baltic Fish: A Synopsis of the FiRe Project. Ambio 28: 2–8.

Bengtsson H and Stevens R (1996). Heavy-mineral provinces in southern Skagerrak and northern Kattegat. Norges Geologiska Undersökelse Bulletin 430: 47–55.

Bergek S, Bergqvist P-A, Hjelt M, Olsson M, Rappe Ch., Roos A and Zook D (1992). Concentrations of PCDDs and PCDFs in seals from Swedish waters. Ambio 21(8): 553–556.

Berglund J and Roos C (2000). The effects of ferry traffic and long-term changes on algal vegetation in the Åland archipelago. Forskningsrapporter från Husö biologiska station 101, 33 pp.

Berglund J, Mattila J, Rönnberg O and Bonsdorff E (2000). Seasonal and inter-annual variation in macrophyte and drift algal occurrence and biomass in shallow bays in the Åland archipelago, N Baltic Sea (manuscript)

Bergman A (1999). Health condition of the Baltic grey seal (Halichoerus grypus) during two decades. Gynaecological health improvement but increased prevalence of colonic ulcers. APMIS 107: 270–282.

Bergman A and Olsson M (1986). Pathology of Baltic Grey Seal and Ringed Seal females with special reference to adrenocortical hyperplasia: Is environmental pollution the cause of a widely distributed disease syndrome? Proceedings from the Symposium on the Seals in the Baltic and Eurasian Lakes. Savonlinna, Finland. 5–8 June, 1984. Finnish Game Research 44: 47–62.

Bergman A, Olsson M and Reiland S (1992). Skull-bone lesions in the Baltic grey seal (Halichoerus grypus). Ambio 21: 517–519.

Bergström S and Matthäus W (1996). Meteorological and hydrographical conditions. In: Third periodic assessment of the state of the marine environment of the Baltic Sea, 1989–93: Background document. Baltic Sea Environmental Proceedings No. 64B, HELCOM 1996.

Bergström S and Carlsson B (1994). River runoff to the Baltic Sea: 1950–1990. Ambio 23: 280–287.

Bergström S and Matthäus W (1996). Meteorology, hydrology and hydrography. In: Third periodic assessment of the state of the marine environment of the Baltic Sea, 1989–93: Background document. Baltic Sea Environmental Proceedings No. 64B, HELCOM 1996, pp. 9–18.

Berzinsh V (1995). Dynamics of hydrological parameters of the Gulf of Riga. In: Ojaveer E (Ed.) Ecosystem of the Gulf of Riga between 1920 and 1990. Estonian Academy of Sciences, Tallinn, Estonia, pp. 32–40.

Bignert A, Cleeman M, Dannenberger D, Baul H and Roots O (1997). Halogenated hydrocarbons. In: Third periodic assessment of the state of the marine environment of the Baltic Sea, 1989–93: Background Document. Baltic Sea Environmental Proceedings No. 64B, HELCOM 1996, pp.130–138.

Bignert A, Göthberg A, Jensen S, Litzén K, Odsjö T, Olsson M and Reutergårdh L (1993). The need for adequate biological sampling in ecotoxicological investigations: a retrospective study of twenty years pollution monitoring. Sci. Tot. Env. 128: 121–139.

Bignert A, Litzén K, Odsjö T, Olsson M, Persson W and Reutergårdh L (1995). Time-related factors influence the concentrations of DDTs, PCBs and shell parameters in eggs of Baltic Guillemot (Uria aalge), 1861–1989. Environmental Pollution 89: 27–36.

Bignert A, Olsson M, Asplund L, Eriksson U and Häggberg L (1999). Altered CB congener composition in Baltic herring indicates ongoing pollution. In: Organohalogen Compounds 41: 383–386.

Bignert A, Olsson M, Persson W, Jensen S, Zakrisson S, Litzén K, Eriksson U, HäggBerg L and Alsberg T (1998). Temporal trends of organochlorines in Northern Europe, 1967–95. Relation to global fractionation, leakage from sediments and international measures. Environ. Pollut. 99: 177–199.

Billen G, Servais P et al. (1990). Dynamics of bacterioplankton in oligotrophic and eutrophic aquatic environments: bottom-up or top-down control? Hydrobiologia 207: 37–42.

Bogovski S, Lang T and Mellergaard S (1999). Histopathological examinations of liver nodules in flounder (Platichthys flesus L) from the Baltic Sea. ICES Journal of Marine Science 56: 152–156.

Bonsdorff E, Blomqvist EM, Mattila J and Norkko A (1997). Coastal Eutrophication: Cause, consequences and perspectives in the archipelago areas of the northern Baltic Sea. Estuarine, Coastal and Shelf Science 44: 63–72.

Bonsdorff E, Blomqvist EM, Mattila J and Norkko A (1997a). Long-term changes and coastal eutrophication. Examples from the Åland Islands and the Archipelago Sea, northern Baltic Sea. Oceanologica Acta 20(1): 319–329.

Bonsdorff E, Mattila J, Rönn C and Österman C-S (1986). Multidimensional interactions in shallow soft-bottom ecosystems; testing the competitive exclusion principle. Ophelia Suppl. 4: 37–44.

Borg H, Andersson P and Johansson K (1988). Influence of acidification on metal fluxes in Swedish forest lakes. The Science of the Total Environment, 87/88 (1989): 241–253.

Borg H and Jonsson B (1996). Large-scale metal distribution in Baltic Sea sediments. Mar. Poll. Bull. 32: 8–21.

Borowski W (2000). Babka bycza w Zalewie Wislanym (in Polish) [Round goby in the Vistula Lagoon]. Mag. Przem. Ryb. 4(12): 39.

Boström C, Bonsdorff E, Kangas P and Norkko A (1997). Long-term changes in an eelgrass (Zostera marina L) community in the northern Baltic Sea. Abstracts of the BMB 15 and ECSA 27 Symposium, Mariehamn, Finland, 9–13 June 1997, p.35.

Brandt M, Edler L and Andersson L (1998). Översvämningar längs Oder och Wisla sommaren 1997, samt effekterna i Östersjön. SMHI Oceanografi nr. 68, 33 pp.

Brunström B, Olsson M and Roos A (1998). 2,3,7,8–TCDD equivalent concentrations in livers from Swedish otters determined with a bioassay. Proceedings from the 18th Symposium on Halogenated Environmental Organic Pollutants, Stockholm, Sweden, 17–21 August, 1998. In: DIOXIN-98. Environmental Levels III. Johansson N, Bergman Å, Broman D, Håkansson H, Jansson B, Klasson Wehler E, Poellinger L and Wahlström B (Eds.) Organohalogen Compounds 39: 149–151.

Buch E and Dahlin H (Eds.) (2000). The BOOS PLAN: Baltic Operational Oceanographic System 1999–2003. EUROGOOS Publication No. 14, Southampton Oceanographic Centre, Southampton. ISBN 0-904175–41–3.

Bäck S, Lehvo A and Blomster J (2000). Mass occurrence of unattached Enteromorpha intestinalis on the Finnish Baltic coast. Ann. Bot. Fennici 37: 155–161.

Bäck S, Mäkinen A, Rissanen J and Rönnberg O (1993). Phytobenthos monitoring. Two case studies from the Finnish Baltic coast in summer 1993. Ministry of the Environment. Finland. Memorandum 1993. 1: 1–24.

Carlton J T (1996). Biological invasions and cryptogenic species. Ecology 77: 1653–1655.

Carman R and Wulff F (1989). Adsorption capacity of phosphorus in Baltic Sea sediments. Estuar. Coast. and Shelf Sci. 29: 447–456.

Carman R, Aigars J and Larsen B (1996). Carbon and nutrient geochemistry of the surface sediments of the Gulf of Riga, Baltic Sea. Marine Geology 134: 57–76.

Cato I (1997). Contaminants in the Skagerrak and Kattegat sediments. In: Cato I and Klingenberg F (Eds.) Proceedings of the Fourth Marine Geological Conference: The Baltic. 24–27 October 1995, Uppsala, Sweden. Sveriges Geologiska Undersökning Ser. Ca 86: 21–35.

Cederwall H and Sjöberg B (1995). Colonization of marine benthic fauna – an effect of saltwater inflows to the Baltic Sea in 1993–94. Östersjö ´94, 4–10, Marine Research Centre, Stockholm University, Stockholm (In Swedish).

Christensen T (1984). Sargassotang, en ny algeslægt i Danmark (in Danish). Urt 1984(4): 99–104.

Christiansen H and Kanneworff E (1985). Sedimenting phytoplankton as major food source for suspension and deposit feeders in the Øresund. Ophelia 24: 223–244.

Ciszewska I (1990). Quantitative changes in the Baltic mesozooplankton on the basis of monitoring within the 1979–83 period. Oceanologia 29: 77–90.

Ciszewski P, Ciszewska I, Kruk-Dowgiallo L, Osowiecki A, Rybicka D, Wiktor J, Wolska-Pys M, Mudzinski L and Trokowicz D (1992). Trends of long-term alterations of Puck Bay ecosystem. SiMO No. 60/8 KBM, PAN, Gdansk, pp. 199–33.

Cleveland WS (1979). Robust locally-weighted regression and smoothing scatterplots. J. Am. Stat. Assoc. 74: 829–836.

Cole JJ, Findlay S, et al. (1988). Bacterial production in fresh and saltwater ecosystems: a cross systems overview. Mar. Ecol. Prog. Ser. 43: 1–10.

Conley DJ, Schelske CL and Stoermer EF (1993). Modification of the biogeochemical cycle of silica with eutrophication. Mar. Ecol. Prog. Ser. 101: 179–192.

Conley J, Stockenberg A, Carman R, Johnstone RW, Rahm L and Wulff F (1996). Sediment fluxes along an eutrophication gradient in the Gulf of Finland, Baltic Sea. Estuar. Coast. Shelf. Sci. 45: 591–598.

da Silva F and Williams RJP (1994). The biological chemistry of the elements. The Inorganic Chemistry of Life. Clarendon Press. Oxford.

Dahl K, Hansen J, Helmig SA, Nielsen R and Larsen HS (2001). Naturkvalitet på stenrev. Hvilke indikatorer kan vi bruge? (in Danish). National Environmental Research Institute, Technical Report No. 352.

Dahl K, Ærtebjerg G, Jensen JN, Nielsen TG, Lisberg D, Krause-Jensen D and Christensen PB (1995). Marine områder – Fjorde, kyster og åbent hav. Vandmiljøplanens Overvågningsprogram 1994 (in Danish). National Environmental Research Institute, Technical Report No. 142, 123 pp.

Daunys D, Zettler M and Gollasch S (1999). Marenzelleria viridis (Verrill, 1873). In: Gollasch, S, Minchin, D, Rosenthal, H and Voigt, M (Eds.) Case histories on introduced species: their general biology, distribution, range expansion and impact. Berlin. Logos-Verlag, pp. 61–67.

Dave G, Nilsson E and Wernersson A-S (2000). Sediment and water phase toxicity and UV-activation of six chemicals used in military explosives. Aquatic Ecosystem Health and Management 33 (3): 291–300.

De Swart RL, Ross PS, Vedder LJ, Timmerman HH, Heisterkamp SH, van Loveren H, Vos JG, Reijnders PJH and Osterhaus ADME (1994). Impairment of immune function in harbour seals (Phoca vitulina) feeding on fish from polluted waters. Ambio 23: 155–159.

de Wit C (2000). Data from the Swedish dioxin survey. Personal communication.

de Wit C, Jansson B, Strandell M, Jonsson P, Bergqvist P-A, Bergek S, Kjeller L-O, Rappe C, Olsson M and Slorach S (1990). Results from the first year of the Swedish dioxin survey. Chemosphere 20 (10–12): 1473–1480.

Dubrawski R, Witkowska K and Kula A, (1998). Properties of surface of bottom sediments in the coastal zone of the Gulf of Gdansk (1994–95). Oceanological Studies 4: 23–47.

EC (1996). European Council Directive 96/29/Euratom. Basic safety standards for the protection of the health of workers and the general public against the dangers arising from Ionising radiation.

EC (2000). European Commission. Radiation Protection No. 110. The radiological exposure of the population of the European Community to radioactivity in the Baltic Sea – Marina-Balt Project. Proceedings of a seminar held at Hasseludden Conference Centre, Stockholm, 9–11 June 1998. EUR Report 19200, Luxembourg.

Echéll J, Jonsson P, Meili M and Carman R (2000). Storm influence on the accumulation and lamination of sediments in deep areas of the northwestern Baltic Proper. Ambio 29: 238–245.

Elken J (Eed) (1996). Deep-water overflow, circulation and vertical exchange in the Baltic Proper. Report Series, Estonian Marine Institute, Tallinn, 6, 91 pp.

Elmgren R (1984). Trophic dynamics in the enclosed, brackish Baltic Sea. Rapp. P-v. Réun. Cons. Int. Explor. Mer. 183: 152–169.

Emeis K-C, Neumann T, Endler R, Struck U, Kunzendorf H and Christiansen C (1998). Geochemical records of sediments in the eastern Gotland Basin – products of sediment dynamics in a not-so-stagnant anoxic basin? Applied Geochemistry 13: 349–358.

ENVIRO (2000). Baltic Cod – A natural resource at high risk. Report No. 3, April 2000. Swedish Ministry of the Environment, p. 6.

Ericson G, Lindesjöö E and Balk L (1998). DNA adducts and histopathological lesions in perch (Perca fluviatilis) and northern pike (Esox lucius) along a polycyclic aromatic hydrocarbon gradient on the Swedish coastline of the Baltic Sea. Can. J. Fish. Aquat. Sci. 55: 815–824.

Fischer H and Matthäus W (1996). The importance of the Drogden Sill in the Sound for major Baltic inflows. J. Mar. Syst. 9: 137–157.

Fisher TR, Peele ER, Ammerman JW and Harding LW (1992). Nutrient limitation of phytoplankton in Chesapeake Bay. Mar. Ecol. Prog. Ser. 82: 51–63.

Fletcher RL (1996). The occurrence of “Green tides”. A review. In: Schramm W and Nienhuis PH (Eds.) Ecological studies, Vol. 123. Marine Benthic Vegetation. Springer-Verlag, Berlin Heidelberg. pp. 7–43.

Flinkman J, Aro E, Vuorinen I and Viitasalo M (1998). Changes in northern Baltic zooplankton and herring nutrition from 1980s to 1990s: top-down and bottom-up processes at work. Marine Ecology Progress Series 165: 127–136.

Floderus S (1988). On the spatial distribution of wave impact at Kattegat seabed. Geografiska Annaler 70 A: 269–272.

Fonselius SH (1978). On nutrients and their role as production limiting factors in the Baltic. Acta Hydrochim. Hydrobiol. 6: 329–339.

Gingele FX and Leipe T (1998). Distribution and enrichment of redox-sensitive metals in Baltic Sea sediments. Baltica 11: 5–16.

Gobas FAPC, Zhang X and Wells R (1993). Gastrointestinal magnification: The mechanism of biomagnification and food chain accumulation of organic chemicals. Environ. Sci. Technol. 27: 2855–2863.

Goering JJ (1968). Denitrification in the oxygen minimum zone of the eastern tropical North Pacific. Deep Sea Res. 15: 157–164.

Gollasch S and Rosenthal H (2000). National report for Germany. ICES Working Group on Introductions and Transfers of Marine Organisms. Pärnu, Estonia, 27–30 March, 2000, p. 3.

Golubkov SM, Umnova LP, Anokhina LE, Panov VE (1987). The processes of production and decomposition in the littoral macrophytes. In: Winberg GG and Gutelmaher BL (Eds.) The Neva Bay: Hydrobiological investigations. Leningrad. pp. 70–75 (in Russian).

Gorokhova E, Aladin N and Dumont H (2000). Further expansion of the genus Cercopagis (Crustacea, Branchiopoda, Onychopoda) in the Baltic Sea, with notes on the taxa present and their ecology. Hydrobiologia 429: 207–218.

Graham L P (1999). Modelling runoff to the Baltic Sea. Ambio 28: 328–334.

Graham L P, Rummukainen M, Gardelin M and Bergström S (2000). Modelling climate change impacts on water resources in the Swedish regional climate modelling programme. Contr. to: Detection and Modelling of Recent Climate Change and its Effects on a Regional Scale, Tarragona, Spain, 29–31 May 2000.

Granat L (2000). Deposition of nitrate and ammonium from the atmosphere to the Baltic. In: Wulff F, Rahm L and Larsson P (Eds.) A systems analysis of the changing Baltic Sea. Springer-Verlag (accepted).

Gruszka P (1999). The river Odra estuary as a gateway for alien species immigration to the Baltic Sea Basin. Acta hydrochim. hidrobiol. 27(5): 374–382.

Gruszka P (1991). Marenzelleria viridis (Verill, 1873) (Polychaeta, Spionidae) – a new component of shallow water benthic community in the Southern Baltic, Acta Ichthyol. Pisc. 21 (Suppl.) 57–65.

Gustafsson B (1999). Dominerande makroalgers djuputbredning i områdena Varberg och sydvästra Onsala/Nidingen, i norra Halland. Examensarbete i marin botanik. Botaniska institutionen, marin botanik, Göteborgs universitet. (in Swedish)

GAAS (2000). Cercopagis pengoi (Ostroumov, 1891). Group on Aquatic Alien Species. Zool. Inst., St. Petersburg, Russia.
http://www.zin.ru/projects/invasions/cerpen.htm
_(2000-05–16, downloaded)

Haddow P. Seal Conservation Society
http://www.greenchannel.com/tec/sealbalt.htm

Hagemeijer WJM and Blair MJ (Eds.) (1997). The EBCC Atlas of European breeding birds: Their distribution and abundance. T & A D Poyser, London, 903 pp.

Hajdu S, Edler L, Olenina I and Witek B (2000). Spreading and establishment of the potentially toxic dinoflagellate Prorocentrum minimum in the Baltic Sea. Int. Rev. Hydrobiol 85: 207–218.

Hammond PS, Benke H, Berggren P, Borchers DL, Buckland ST, Collet A, Heide-Jørgensen MP, Heimlich-Boran S, Hiby AR, Leopold MF and Oeien N (1995). Distribution and abundance of the harbour porpoise and other small cetaceans in the North Sea and adjacent waters. Final Report, LIFE 92–2/UK/027, Cambridge, 240 pp.

Hansen B, Hansen PJ and Bjørnsen PK (1994). The size ratio between planktonic predators and their prey. Limnol. Oceanogr. 39: 395–403.

Hansen PJ (1991). Quantitative importance and trophic role of heterotrophic dinoflagellates in a coastal pelagic food web. Mar. Ecol. Prog. Ser. 73: 253–261.

Hansson S (1999). Human effects on the Baltic Sea ecosystem-fishing and eutrophication. In: Ecosystem considerations in fisheries management. Proceedings of the International Symposium on Ecosystem Considerations in Fisheries Management. Alaska Sea Grant Program Report, pp. 405–407, University of Alaska Fairbanks, Fairbanks, Alaska.

Hansson S, Arrhenius F and Nellbring S (1997). Food web interactions in a Baltic Sea coastal area. In: Forage fish in marine ecosystems. Proceedings of the International Symposium on the Role of Forage Fish in Marine Ecosystems. Alaska Sea Grant Program Report 97–01, pp. 281–291. University of Alaska Fairbanks, Fairbanks, Alaska.

Harms U (1997). Biota. In: Third periodic assessment of the state of the marine environment of the Baltic Sea, 1989–93: Background Document. Baltic Sea Environmental Proceedings No. 64B, HELCOM 1996, pp.149–153.

Hauff P (1998). Bestandsentwicklung des Seeadlers Haliaeetus albicilla in Deutschland seit 1980 mit Rückblick auf die vergangenen 100 Jahre. Vogelwelt 119. 47–63.

Hav och kust. (1992). Svensk National Atlas. Sjöberg B (Ed.), Series Eds. Wastenson L, et al. Bra Böcker, Höganäs, Sweden.

Heerkloss R, Börner R, Krummhaar B, Arndt H, and Schnese, W (1984). Growth inhibition in estuarine zooplankters due to blue-green algae. Ophelia Suppl. 3: 45-49.

Heide-Jørgensen MP, Härkönen T and Åberg P (1992b). Long-term Effects of Epizootic in Harbour Seals in the Kattegat-Skagerrak and Adjacent Areas. Ambio 21: 511–516.

Heide-Jørgensen MP, Härkönen T, Dietz R and Thompson P (1992a). Retrospective of the 1988 European seal epizootic. Diseases of Aquatic Organisms 13: 37–62.

Heiskanen A-S (1998). Factors governing sedimentation and pelagic nutrient cycles in the northern Baltic Sea. Monogr. Bor. Env. Res. 8: 1–80.

Heiskanen A-S, Gran V, Lehtoranta J and Pitkänen H (2000). Fate of nutrients (N, P) along the estuarine gradient of the River Neva in the eastern Gulf of Finland, Baltic Sea. Short communication. ICES Journal of Marine Science 56 (Supplement): 161–164.

Heiskanen A-S, Tamminen T, Gundersen K (1996). The impact of planktonic food web structure on nutrient retention and loss from a late summer pelagic system in the coastal northern Baltic Sea. Mar. Ecol. Prog. Ser. 145:195–208.

Helander B (1985). Reproduction of white-tailed sea eagle in Sweden. Holarctic Ecology 8: 211–227.

Helander B (1998). Toppkonsumenter. Östersjö '97, pp 17–21. Stockholms Marina Forskningscentrum (SMF).

Helander B (2000). Havsörn och gråsäl – fortsatt ökande bestånd. pp. 37–40. In: Tidlund A and Brunner A (Eds.) Östersjö '99 (with English summary). Stockholms Marina Forskningscentrum (SMF).

Helander B and Bignert A (1992). Harbour seal (Phoca vitulina) on the Swedish Baltic Coast: Population Trends and Reproduction. Ambio 21: 504–510.

Helander, B och Härkönen, T (1999). Säl och havsörn. Östersjö ´98, pp 47–52. Stockholms Marina Forskningscentrum (SMF).

Helander B and Härkönen T (1997). Marina toppkonsumenter. Östersjö ‘96: 45–48.

Helander B, Olsson A, Bignert A, Asplund L and Litzén K (1999). White-tailed sea eagle Haliaeetus albicilla in Sweden: Reproduction in relation to DDE, PCB, coplanar PCB and eggshell parameters. In: Olsson A (1999). Applications of various analytical chemical methods for exposure studies of halogenated environmental contaminants in the Baltic environment (Paper V). Doctoral Dissertation. Environmental Chemistry, Stockholm University.

Helander B, Olsson M and Reutergårdh L (1982). Residue levels of organochlorine and mercury compounds in unhatched eggs and the relationships to breeding success in white-tailed sea eagles Haliaeetus albicilla in Sweden. Holarctic Ecology 5: 349–366.

HELCOM (1981). Geological History of the Baltic Sea. In: Assessment of the effects of pollution on the natural resources of the Baltic Sea, 1980. Baltic Sea Environmental Proceedings No. 5B: 80–82.

HELCOM (1981b). Major characteristics of the Baltic Sea. In: Assessment of the effects of pollution on the natural resources of the Baltic Sea, 1980. Baltic Sea Environmental Proceedings No. 5B: 82–88

HELCOM (1984). Studies on ship casualties in the Baltic Sea. Baltic Sea Environmental Proceedings No. 11.

HELCOM (1990). Second periodic assessment of the state of the marine environment of the Baltic Sea, 1984–88; Background Document. Baltic Sea Environmental Proceedings No. 35B.

HELCOM (1990). Study of the risk for accidents and related environmental hazards from the transportation of chemicals by tankers in the Baltic Sea Area. Baltic Sea Environmental Proceedings No. 34.

HELCOM (1993). Transportation of packaged dangerous goods by sea in the Baltic Sea Area. Baltic Sea Environmental Proceedings No. 51.

HELCOM (1993). Complex analysis of the hazard related to the captured German chemical weapons dumped in the Baltic Sea. CHEMU 2/2/1Rev.1 (27 September 1993) as amended 11 October 1993 – Document submitted by Russia.

HELCOM (1994). Report on chemical munitions dumped in the Baltic Sea. HELCOM 15/5/1.

HELCOM (1996). Third periodic assessment of the state of the marine environment of the Baltic Sea, 1989–93: Background document. Baltic Sea Environmental Proceedings No. 64B, HELCOM 1996, 252 pp.

HELCOM (1996a). Coastal and marine protected areas in the Baltic Sea Region. Baltic Sea Environmental Proceedings No. 63. 104 pp + tables and maps.

HELCOM (1996b). Transportation of oils in the Baltic Sea Area 1993, SSPA, 1996.

HELCOM (1997). Airborne pollution load to the Baltic Sea 1991–95. Baltic Sea Environmental Proceedings No. 69, pp. 1– 55.

HELCOM (1998). The third Baltic Sea pollution load compilation (PLC-3). Baltic Sea Environmental Proceedings No. 70: 1–133.

HELCOM (1998). Red List of marine and coastal biotopes and biotope complexes of the Baltic Sea, Belt Sea and Kattegat – Including a comprehensive description and classification system for all Baltic marine and coastal biotopes. Baltic Sea Environmental Proceedings No. 75, 115 pp.

HELCOM (1999). Marine sediment extraction in the Baltic Sea – Status report. Baltic Sea Environmental Proceedings No. 76.

Helle E (1986). The decrease in the ringed seal population of the Gulf of Bothnia in 1975–84. Proceedings from the Symposium on the Seals in the Baltic and Eurasian Lakes. Savonlinna, Finland, 5–8 June, 1984. Finnish Game Research 44: 28–32.

Helle E and Nyman M (2000). Unpublished Data. Game and Fisheries Research Institute, Helsinki, Finland.

Helle E, Olsson M and Jensen S (1976). DDT and PCB levels and reproduction in ringed seal from the Bothnian Bay. Ambio 5: 188–189.

Hernroth L and Ackefors H (1979). The zooplankton of the Baltic proper. A long-term investigation of the fauna, its biology and ecology. Rep. Fish. Swedish Inst. Mar. Res. 2:1–60.

Hirsch R M and Slack J R (1984). A nonparametric trend test for seasonal data with serial dependence. Water Resources Research 20: 727–732.

Horstmann U, (1975). Eutrophication and mass production of blue-green algae in the Baltic. Merentutkimuslaitoksen Julk. Havsforskningsinst. Skr. 239: 83-99.

http://www.ee/Baltic21

http://www.fao.org

http://www.helcom.fi

Huggenberger S, Benke H and Kinze CC (1999). Geographical variations of the harbour porpoise (Phocoena phocoena, L) populations in the North and Baltic Seas using morphometric comparisons. European Research on Cetaceans 13: 362–366.

Hägerhäll B and Skov H (1998). Proposal for offshore Baltic Sea Protected Areas (BSPAs). Unpublished expert report to HELCOM EC-NATURE, 103 pp.

Hällfors G, Viitasalo I and Niemi A (1987). Macrophyte vegetation and trophic status of the Gulf of Finland. A review of Finnish investigations. Meri 13: 111–158.

Härkönen T and Hårding KC (2000). (Personal communication).

Härkönen T and Lunneryd SG (1992). Estimating abundance of ringed seals in the Bothnian Bay. Ambio 21: 497–503.

Håkanson L and Jansson M (1983). Principles of lake sedimentology. Springer-Verlag, Berlin, p. 316.

Håkanson L, Jonsson P, Jonsson B and Martinsen K (1988). Distribution of chlorinated organic substances from pulp mills. Wat. Sci. Tech. 20(1): 25–36.

Hårding K and Härkönen T (1999). Developments of the Baltic grey seal (Halichoerus grypus) and ringed seal (Phoca hispida) populations during the 20th Century. Ambio 28: 619–627.

ICES (1995). Report of the Baltic Salmon and Ttrout Aassessment Working Group. ICES CM 1995/Assess:16.

ICES (1996). Report of the Working Group on Ecosystem Effects of Fishing Activities. ICES CM 1996/Assess/Env:1.

ICES (1997). Report of the ICES Advisory Committee on the Marine Environment, 1997. ICES Cooperative Research Report, 222: 78–80 and 93.

ICES (1997). Report of the OSPAR/ICES Workshop on the overall evaluation and update of background/reference concentrations for nutrients and for contaminants in sea water, biota and sediment. SIME 97/7/2–E Ostend 3–7 February 1997.

ICES (1999a). Report of the Baltic Fisheries Assessment Working Group. ICES CM 1999/ACFM:15.

ICES (1999b). Report of the ICES Advisory Committee on Fishery Management, 1998. ICES Cooperative Research Report, 229.

ICES (1999c). Report of the Baltic Salmon and Trout Assessment Working Group. ICES CM 1999/ACFM:16.

ICES (2000). Contaminants in the Baltic Sea sediments. Results of the 1993 ICES/HELCOM Sediment Baseline Study. In prep.

ICES (2000a). Report of the ICES Advisory Committee on Fishery Management 1999. ICES Cooperative Research Report, 236, Part 2. 405 pp.

ICES (2000b). Report of the Baltic Salmon and Trout Assessment Working Group. ICES CM 2000/ACFM:12. 123 pp.

ICES (2000c). Report of the ICES Advisory Committee on the Marine Environment, 2000. ICES Cooperative Research Report, 241: 9–26, 91–95, 111–117, 118–120, 201–213.

IMGW 1995–99. Environmental conditions in the Polish zone of the southern Baltic Sea in 1994, 1995, 1996, 1997, 1998. Cyberska B, Lauer Z, Trzosinska A (Eds.) Materialy Oddzialu Morskiego, Gdynia 1995, 1996, 1997, 1998, 1999 (in Polish).

Ingelög T, Anderssen R and Tjernberg M (Eds.) (1993). Red Data Book of the Baltic Region. Part 1: List of threatened vascular plants and vertebrates. Swedish Threatened Species Unit, Uppsala in cooperation with the Institute of Biology, Riga. 95 pp.

IUCN (1994). IUCN Red List of Threatened Animals. Compiled by the World Conservation Monitoring Centre. 285 pp.

IUCN (1996). IUCN Red List of Threatened Animals Database. Cambridge. http://www.wcmc.org.UK/cgi-bin/arl_output.p

Jakobsen F (1995). The major inflow to the Baltic Sea during January 1993. J. Mar. Syst. 6: 227–240.

Jansson Bengt-Owe (1980). Natural Systems of the Baltic Sea. Ambio 9(3–4):128–136.

Jarre-Teichman A, Wieland K, MacKenzie RR, Aro E and Plikshs M (1997). Stock-recruitment relationships for cod (Gadus morhua) in the central Baltic Sea incorporating environmental variability. Paper presented at the ICES Symposium on the Recruitment Dynamics of Exploited Marine Populations: Physical-Biological Interactions.

Jensen A (1997). Pb-210 dating of sediment cores from the Baltic Sediment Baseline Study. VKI Report No. 904740.

Jensen S, Johnels AG, Olsson M and Otterlind G (1969). DDT and PCB in marine animals from Swedish waters. Nature 224: 247–250.

Jensen S, Kihlström JE, Olsson M, Lundberg C and Örberg J (1977). Effects of PCB and DDT on mink (Mustela vison) during the reproductive season. Ambio 6: 239.

Jonson JE, Tarrason L and Bartnicki B, (2000). Effects of international shipping on European pollution levels. EMEP EMEP/MSC-W Note 5/2000. 22p.

Jonsson P (2000). Sediment burial of PCBs in the offshore Baltic Sea. Ambio 29: 260–267.

Jonsson P, Eckhéll J and Larsson P (2000). PCB and DDT in laminated sediments from offshore and archipelago areas of the NW Baltic Sea. Ambio 29: 268–276.

Jonsson P, Rappe C, Kjeller L-O, Kierkegaard A, Håkanson L and Jonsson B (1993). Pulp mill-related polychlorinated organic compounds in Baltic Sea sediments. Ambio 22: 37–43.

Josefson AB and Conley DJ (1997). Benthic response to a pelagic front. Mar. Ecol. Prog. Ser. 147: 49–62.

Jóźwiak, T 1995–99. Coastwatch Europe, Polish report ocean. Wyd. Sopockiego Instytutu Ekologicznego, Sopot 1995, 1996, 1997,1998, 1999.

Jørgensen T (1990). Long-term changes in age at sexual maturity of northeast Arctic cod (Gadus morhua L). ICES Journal of Marine Science 46: 235–248.

Kangas P and Niemi Å (1985). Observations of recolonization by the bladderwrack, Fucus vesiculosus, on the southern coast of Finland. Aqua Fennica 15: 133–141.

Kangas P, Autio H, Hällfors G, Luther H, Niemi Å and Salemaa H (1982). A general model of the decline of Fucus vesiculosus at Tvärminne, south coast of Finland in 1977–81. Acta Bot. Fennica 118: 1–27.

Kangas P, Byholm L and Stigzelius J (2001). Changes in zoobenthic communities. In: State of the Finnish coastal waters in the 1990s. Kauppila P and Bäck S (Eds.). The Finnish Environment 472: 79–89.

Kankaanpää HT (1996). Analysis of the ICES-HELCOM sediment baseline study samples for EOX gives more evidence of natural organohalogen production in the Baltic Sea. ICES CM 1996/R:1. ICES Annual Science Conference, 84th statutory meeting 27 September to 4 October, 1996, Reykjavik, Iceland.

Kankaanpää HT (1997). Sedimentation, distribution, sources and properties of organic halogen material in the Gulf of Finland. Monographs of the Boreal Environment Research, No. 6.

Kankaanpää HT, Laurén MA, Saares RJ, Heitto LV and Suursaar ÛK (1997). Distribution of halogenated organic material in sediments from anthropogenic and natural sources in the Gulf of Finland catchment area. – Environmental Science and Technology 31(1): 96–104.

Kannan K and Falandysz J (1997). Butyl tin residues in sediment, fish, fish-eating birds, harbour porpoise and human tissues from the Polish coast of the Baltic. Mar.Poll. Bull. 34(3): 203–207.

Karataev AY, Burlakova LE and Padilla DK (1997). The effects of Dreissena polymopha (Pallas) invasion on aquatic communities in Eastern Europe. J. Shellfish Research, 16(1): 187–203.

Karlsson J, Kuylenstierna M and Åberg P (1992). Contribution to the seaweed flora of Sweden: New or otherwise interesting records from the west coast. Acta Phytogeographica Suecia 78: 49–63.

Karlsson J (1995). Inventering av marina makroalger i Østfold 1994: Området Heia-Torbjørnskjær (in Swedish). Rapport till Østfold fylke, Norge. Tjärnö Marinbiologiska Laboratorium, 21 pp.

Karlsson J (1998). Inventering av marina makroalger i Halland 1997 (in Swedish): Lilla Middelgrund. Länsstyrelsen i Hallands län. 14 pp. excl. appendices.

Karlsson J and Loo L-O (1999). On the distribution and the continuous expansion of the Japanese seaweed – Sargassum muticum – in Sweden. Bot. Mar. 42(3): 285–294.

Karlsson J et al. (1998). Miljöeffekter av kemiske vapen, FAO-R –98-00839-170 – SE, August 1998 (in Swedish).

Karlsson J, Loo L-O and Loo-Lutterwall P-L (2000). Inventering av marin fauna och flora i Halland 1997: Nidingen-Hållsundsudde-Fjärehals (In Swedish). Länsstyrelsen i Hallands län, Livsmiljöenheten. Meddelande 2000.1. 47 pp. excl. appendices.

Karlsson J, Valentinsson D and Loo L-O (1995). Sargassosnärja – Sargassum muticum – vid svenska västkusten (On the distribution, growth and associated biota of the Japweed – Sargassum muticum – in Sweden) (In Swedish with English summary). Rapport till Statens Naturvårdsverk, 16 pp.

Karlsson K-G (1989). Development of an operational cloud classification model. Int. J. Remote Sensing 10: 687–693.

Kauppila P and Bäck S (2001). The state of the Finnish coastal waters in the 1990s. The Finnish Environment 472.

Kauppila P and Lepistö L (2000). Changes in phytoplankton. In: Kauppila, P and Bäck, S (eds). State of the Finnish coastal waters in the 1990s. The Finnish Environment 472: 61–70.

Kautsky H, (1992). The impact of pulp mill effluents on phytobenthic communities of the Baltic Sea. Ambio 21 (4): 308–313.

Kautsky H (1995). Quantitative distribution of sublittoral plant and animal communities in the Baltic Sea gradient. In: Eleftheriou, A, Ansell, A and Smith, CJ (Eds.) Biology and Ecology of Shallow Coastal Waters. 28th EMBS, Crete 23–28 Sept 1993. Olsen and Olsen. pp. 23–31.

Kautsky N, Kautsky H, Kautsky U and Waern M (1986). Decreased depth penetration of Fucus vesiculosus (L) since the 1940's indicates eutrophication of the Baltic Sea. Mar. Ecol. Prog. Ser. 28: 1–8.

Kautsky H, Kautsky L, Kautsky N, Kautsky U, and Lindblad C (1992). Studies on the Fucus vesiculosus community in the Baltic Sea. Acta phytogeogr. Suec. 78: 33–48.

Kautsky H, Martin G, Mäkinen A, Borgiel M, Vahteri P and Rissanen J (1999). Structure of phytobenthic and associated animal communities in the Gulf of Riga. Hydrobiologia 393: 191–200.

Kihlström JE, Olsson M, Jensen S, Johansson Å, Ahlbom J and Bergman Å (1992). Effects of PCB and different fractions of PCB on the reproduction of the mink (Mustela vison). Ambio 21: 563–569.

Kiirikki M and Blomster J (1996). Wind-induced upwelling as a possible explanation for mass occurrences of epiphytic Ectocarpus siliculosus (Phaeophyta) in the northern Baltic Proper. Marine Biology 127: 353–358.

Kiirikki M, Haapamäki J, Koponen J, Ruuskanen A and Sarkkula J (1998). Linking the growth of filamentous algae to the 3D-Ecohydrodynamic model of the Gulf of Finland. Environmental Modelling and Software 13: 503–509.

Kiirikki M, Westerholm L and Sarkkula J (2000). Possibilities of reducing harmful algal blooms in the Gulf of Finland. Suomen ympäristö 416: 1–36. Finnish Environment Institute (in Finnish with English summary).

Kirchhoff K (1982). Wasservogelverluste durch die Fischerei an der Schleswig-Holsteinichen Ostseeküste. Die Vogelwelt 103: 81–89.

Kirekegaard A, Sellström U, Bignert A, Olsson M, Asplund L, Jansson B and de Wit C (1999). Temporal trends of a polybrominated diphenyl ether (PBDE), a methoxylated PBDE, and hexabromocyclododecane (HBCD) in Swedish Biota. Organohalogen compounds 40: 367–370.

Kirkkala T, Helminen H and Erkkilä A (1998). Variability of nutrient limitation in the Archipelago Sea, SW Finland. Hydrobiologia 363: 117–126.

Kivi K, Kaitala S, Kuosa H, Kuparinen J, Leskinen E, Lignell R, Marcussen B and Tamminen T (1993). Nutrient limitation and grazing control of Baltic plankton community during annual succession. Limnol. Oceanogr. 38: 893–905.

Kivi K, Kuosa H and Tanskanen S (1996). An experimental study on the role of crustacean and microprotozoan grazers in the planktonic food web. Mar. Ecol. Prog. Ser. 136: 59–68.

Kiørboe T and Nielsen TG (1994). Regulation of zooplankton biomass and production in a temperate, coastal ecosystem. I. Copepods. Limnol. Oceanogr. 39: 493–507.

Kjeller L-O and Rappe C, (1995). Time trends in levels, patterns and profiles for polychlorinated dibenzopdioxins, dibenzofurans, and biphenyls in a sediment core from the Baltic proper. Environmental Science and Technology 29: 346–355.

Knief W (1998). Zur Situation des Kormorans (Phalacrocorax carbo sinensis) in Deutschland. Ber. Z Vogelschutz 35: 91–105.

Knutzen J and Skei J (1992). Preliminary proposals for classification of marine environmental quality respecting micropollutants in water, sediments and selected organisms. Norwegian Institute for Water Research, Report O-862602/O-89266, 22 pp.

Kock K-H and Benke H (1996). On the by-catch of harbour porpoise (Phocoena phocoena) in German fisheries in the Baltic and the North Sea. Arch. Fish. Mar. Res. 44: 95–114.

Koljonen M-L, Jansson H, Paaver T, Vasin O and Koskiniemi J (1999). Phylogeographic lineages and differentiation pattern of Atlantic salmon in the Baltic Sea with management implications. Can. J. Fish. Aquat. Sci. 56: 1766–1780.

Kononen K (1992). Dynamics of the toxic cyanobacterial blooms in the Baltic Sea. Finnish Mar. Res. 261: 3–36.

Koop K, Boynton WR, Wulff F and Carman R (1990). Sediment-water oxygen and nutrient exchange along a depth gradient in the Baltic Sea. Mar. Ecol. Prog. Ser. 63: 65–77.

Kostrichkina E M, Yurkovskis A K, Line R J, and Berzinsh V, (1990). Long-term changes of zooplankton in relation to eutrophication of the Gulf of Riga (in Russian). Gidrobiol. Zhurn., No. 26: 10-16.

Kotta I. and Kotta J (1997). Changes in zoobenthic communities in Estonian waters between the 1970s and 1990s. An example from the southern coast of Saaremaa and Muuga Bay. In: Ojaveer, E (Ed.) Proceedings of the 14th Baltic Marine Biologists Symposium. Estonian Academy Publishers, Pärnu, Estonia, pp. 70–79.

Kowalewska G and Konat J (1999). Determination of polychlorinated biphenyls in the marine sediments using gas chromatography. Chem. Anal. 44: 223.

Kowalewska G, Konat J (1997). Distribution of polynuclear aromatic hydrocarbons (PAHs) in sediments of the southern Baltic Sea. Oceanologia 39 (1): 83–104.

Krauss W and Brügge B (1992). Wind-produced water exchange between the deep basins of the Baltic Sea. J. Phys. Oceanogr. 21: 373–384.

Kremling K and Streu P (2000). Further evidence for a drastic decline of potentially hazardous trace metals in Baltic Sea surface waters. Mar. Poll. Bull. 40: 674–679.

Krost P (1990). Der Einfluss der Grundschlepp-netzfischerei auf Nahrsalz-Freisetzung aus dem Sediment und Makrofauna der Kieler Bucht (Westl. Ostsee). Berichte aus dem Institut für Meereskunde an der Christian-Albrecht Universität Kiel, Nr. 200.

Kruk-Dowgiallo L and Dubrawski R (1998). The state of the environment of the Gulf of Gdansk coastal zone in autumn 1994 and summer 1995. Oceanol. Stud. 27 (4): 137–159.

Kruk-Dowgiallo L (1994). Rozmieszczenie i biomasa fitobentosu wewn trznej Zatoki Puckiej, lato 1987. [Distribution and biomass of phytobenthos in inner part of Puck Bay in summer 1987]. In: Aging: Zatoka Pucka. Mo liwo ci rewaloryzacji. [Puck Bay – possibilties of recultivation]. IO Warszawa. 109–122.

Kruk-Dowgiallo L (1996). The role of filamentous brown algae in the degradation of the underwater meadows of the Gulf of Gdansk. Oceanol. Stud. 25 (1–2): 125–137.

Kuijpers A, Dennegård B, Albinsson Y and Jensen A (1992). Sediment transport pathways in the Skagerrak and Kattegat as indicated by sediment Chernobyl radioactivity and heavy metals. In: Dennegård B and Kuijpers A (Eds.) Marine geological environmental investigations in the Skagerrak and northern Kattegat. Univ. of Göteborg, Inst. of Marine Geology Report 7, 27 pp.

Kukk H (1985). The influence of anthropogenic factors on the composition and distribution of bottom vegetation in the Gulf of Finland. In: Problems concerning bioindication of the ecological condition of the Gulf of Finland. pp. 123–126. Valgus, Tallinn.

Kukk H (1995). Phytobenthos. In: Ecosystem of the Gulf of Riga between 1920 and 1990. Estonian Academy Publishers, Tallinn, pp. 131–138.

Kukk H and Viitasalo I (1997). The convalescence of Seurasaari Bay: recovery of macrophyte vegetation took fifteen years after closing the wastewater sewers. Abstracts of the BMB 15 and ECSA 27 Symposium, Mariehamn, Finland, 9–13 June 1997. p.13.

Kuosa H (1991). Picoplanktonic algae in the northern Baltic Sea: seasonal dynamics and flagellate grazing. Mar. Ecol. Prog. Ser. 73: 269–276.

Kuparinen J, Leonardsson K, Mattila J and Wikner J (1994). Bottniska vikens ekologi: Näringsvävsstruktur, materialflöden och trender (with English summary). Vatten 50: 201–219.

Kuparinen J, Leonardsson L, et al. (1996). Food web structure and function in the Gulf of Bothnia, the Baltic Sea. Ambio Special Report 8: 12–20.

Kuuppo P (1994). Heterotrophic nanoflagellates in the microbial food web of the SW coast of Finland, the Baltic Sea. Grazing on planktonic bacteria. Ph.D thesis, University of Helsinki, Walter and Andrée de Nottbeck Foundation Scientific Reports 10, pp. 20.

Kuuppo-Leinikki P, Autio R, Hällfors S, Kuosa H, Kuparinen J and Pajuniemi R (1994). Trophic interactions and carbon flow between picoplankton and protozoa in pelagic enclosures manipulated by nutrients and a top predator. Mar. Ecol. Prog. Ser. 107: 89–102.

Kuusisto M, Koponen J and Sarkkula J (1998): Modelled phytoplankton dynamics in the Gulf of Finland. Environmental Modelling and Software 13: 461–470.

Köster FW and Möllmann C (1997). Predation by sprat and herring on early life stages of cod and sprat in the central Baltic. In: Forage fish in marine ecosystems. Proceedings of the International Symposium on the Role of Forage Fish in Marine Ecosystems. Alaska Sea Grant Program Report: 97–01, pp. 41–69. University of Alaska Fairbanks, Fairbanks, Alaska.

Kaas H, Møhlenberg F, Josefson A, Rasmussen B, Krause-Jensen D, Jensen HS, Svendsen LM, Windolf J, Middelboe AL, Sand-Jensen K and Foldager Pedersen M (1996). Marine områder. Danske fjorde – status over miljøtilstand, årsagssammenhænge og udvikling (in Danish). National Environmental Research Institute, Technical Report No. 179, 205 pp.

Laine AO, Sandler H, Andersin A-B and Stigzelius J (1997). Long-term changes of the macrozoobenthos of the Eastern Gotland Basin and the Gulf of Finland (Baltic Sea) in relation to the hydrographical regime. J. Sea Res. 38: 135–159.

Lampolahti J (1997): Uudenkaupungin merialueen kasvillisuuden kehitys 1990–96. Tutkimusraportti Kemira Agro Oy:n Uudenkaupungin tehtaille. 30 p. + 36 figures.

Lang T, Mellergaard S, Wosniok W, Kadakas V and Neumann K (1999). Spatial distribution of grossly visible diseases and parasites in flounder (Platichthys flesus) from the Baltic Sea: a synoptic study. ICES Journal of Marine Science 56: 138–147.

Launiainen J, Matthäus W, Fonselius S and Francke E (1987). Hydrography. In: First periodic assessment of the state of the marine environment of the Baltic Sea, 1980–85; Background Document. Baltic Sea Environmental Proceedings, 17B, pp. 7–34.

Laznik M, Stålnacke P, Grimvall A and Wittgren HB (1999). Riverine input of nutrients to the Gulf of Riga – temporal and spatial variations. J. Marine Systems 23(1–3): 11–25.

Lehmann A and Hinrichsen HH (2000). On the wind-driven thermohaline circulation of the Baltic Sea. – Phys. Chem. Earth (B), 5 (2): 183–189.

Lehtonen, KK (1997). Ecophysiology of two benthic amphipod species from Northern Baltic Sea. Helsinki, Monograph of the Boreal Environment Research 7: 9–33.

Lehvo A and Bäck S (2000). Notes on sublittoral vegetation in the vicinity of the Vyborg Bay, Russia. Memoranda Soc. Fauna Flora Fennica 76: 7–13.

Leipe T, Neumann T and Emeis K-C (1995). Schwermetallverteilung in holozänen Ostseesedimenten. Untersuchungen im einflußbereich der Oder (in German). Geowissenschaften 13: 470–478.

Leipe T, Tauber F, Brügmann L, Irion G and Hennings U (1998). Schwermetallverteilung in Oberflächensedimenten der westlichen Ostsee (Arkonabecken, Mecklenburger/Lübecker Bucht und Kieler Bucht). Meyniana 50: 137–154.

Leivuori M (1998). Heavy metal contamination in surface sediments in the Gulf of Finland and comparison with the Gulf of Bothia. Chemosphere 36(1): 43–59.

Leivuori M and Niemistö L (1995). Sedimentation of trace metals in the Gulf of Bothia. Chemosphere 31(8): 3839–3856.

Leivuori M, Jok_as K, Seisuma Z, Petersell V, Larsen B, Pedersen B and Floderus S (2000). Distribution of heavy metals in sediments of the Gulf of Riga, the Baltic Sea. Boreal Environment Research 5: 165–185.

Leonards PEG, Elmeros M, Cofino WP, Conroy JWH, Gutleb AC, Mason CF, Murk AJ, Olsson M, Smit MD, van Hattum BGM and Madsen AB (1996). Toxic PCBs in European otter populations in relation to biological factors and health status. In: Smit MD, Leonards PEG, Murk AJ, de Jongh AWJJ and van Hattum B (Eds.) Development of otter-based quality objectives for PCBs. Inst. for Env. Studies, Vrije Universiteit, Amsterdam, pp. 47–62.

Leonardsson K (1999). Mjukbottenfauna. Bottniska viken 1998 – årsrapport från den marina miljööver-vakningen. Wiklund K (Ed.) (In Swedish). Ljungbergs. pp. 12–13.

Leppäkoski E and Olenin S (2000). Non-native species and rates of spread: lessons from the brackish Baltic Sea. Biological Invasions 2(2): 151–163.

Leppänen J-M, Rantajärvi E, Hällfors S, Kruskopf M and Laine V (1995). Unattended monitoring of potentially toxic phytoplankton species in the Baltic Sea in 1993. J. Plankton Res. 17: 891–902.

Lignell R, Heiskanen A-S, Kuosa H, Gundersen K, Kuuppo-Leinikki P, Pajuniemi R and Uitto A (1993). Fate of phytoplankton spring bloom: Sedimentation and carbon flow in the planktonic food web in the northern Baltic. Mar. Ecol. Prog. Ser. 94: 239–252.

Lindesjöö E and Thulin J (1990). Fin erosion of perch (Perca fluviatilis) and ruffe (Gymnocephalus cernua) in a pulp mill effluent area. Dis. aquat. Org. 8: 119–126.

Lindesjöö E and Thulin J (1992). A skeletal deformity of northern pike (Esox lucius L) related to pulp mill effluents. Can. J. Fish. Aquat. Sci. 49: 166–172.

Lindesjöö E, Thulin J, Bengtsson B-E and Tjärnlund U (1994). Abnormalities of a gill cover bone, the operculum, in perch (Perca fluviatilis) from a pulp mill effluent area. Aquat. Toxicol. 28: 189–207.

Loo L-O and Rosenberg R (1989). Bivalve suspension-feeding dynamics and benthic-pelagic coupling in an eutrophicated marine bay. J. Exp. Mar. Biol. Ecol. 130: 253–277.

Loo L-O and Rosenberg R (1996). Production and energy budget in marine suspension feeding populations: Mytilus edulis, Cerastoderma edule, Mya arenaria and Amphiura filiformis. J. Sea Res. 35: 199–207.

Loo L-O, Isasson I, Loo-Lutterwall P-L and Hansson L-J (1996). Inventering av grunda havsområden vid Hallands Väderö 1994 (In Swedish). Länsstyrelsen i Kristianstads län, 36 pp.

Luksiene D, Sandström O, Lounasheimo L and Andersson J (2000). The effects of thermal effluent exposure on the gametogenesis of female fish. J. Fish Biol. 56: 37–50.

Lund B-O, Örberg J, Bergman Å, Larsson C, Bergman A, Bäcklin B-M, Håkansson H, Madej A, Brouwer A and Brunström B (1999). Chronic and reproductive toxicity of a mixture of 15 methylsulfonyl-polychlorinated biphenyls and 3-methylsulfonyl-2,2-bis-(4-chlorophenyl)-1,1-dichloroethene in mink (Mustela vison). Environmental Toxicology and Chemistry 18: 292–298.

Lysiak-Pastuszak E and Drgas N (1999). Nutrients. In: Assessment of the state of the Polish sector of the Baltic Sea between 1994–98. Ministry of Environment, Institute of Meteorology and Water Mangement, Gdynia 1999, 171–188 (in Polish).

Magnusson K (1998). Oil handling in the Baltic Sea Area, 1996–2001. 24 pages. SSPA Maritime Consulting AB report 7935–2 (http://www.helcom.fi/SSPA-reports/REPORT-red.htm).

Markager S, Nielsen TG, Carstensen J, Conley D, Dahl K, Hansen J, Henriksen P, Josefson A, Larsen MM, Pedersen B, Rasmussen B, Strand J, Ærtebjerg G, Fossing H, Lauersen JS, Hertel O, Skov H, Svendsen LM, Cleemann M and Pritzl G (1999). Marine områder. Status over miljøtilstanden i 1998. NOVA 2003 (in Danish with an English summary). National Environmental Research Institute, Technical Report No. 290, 164 pp.

Martin G (1999). Distribution of phytobenthos biomass in the Gulf of Riga (1984–91). Hydrobiologia, 393, pp. 181–190.

Martin G and Kukk H (1997). Benthic littoral vegetation as indicator of the eutrophication situation in the coastal urban area of Tallinn Bay. In: Ecological investigations in Tallinn Bay area in 1993–97. Esonian Marine Institute Report Series 8: 1–10.

Mason CF and MacDonald S (1986). Otters: Ecology and Conservation. Cambridge University Press, Cambridge.

Matthäus W (1995). Natural variability and human impacts reflected in long-term changes in the Baltic deep water conditions – a brief review. Dt. Hydrogr. Z. 47: 47–65.

Matthäus W and Franck H (1992). Characteristics of major Baltic inflows – a statistical analysis. Cont. Shelf Res. 12: 1375–1400.

Matthäus W and Lass H-U (1995). The recent salt inflow into the Baltic Sea. J. Phys. Oceanogr. 25: 280–286.

Matthäus W, Nausch G, Lass HU, Nagel K and Siegel H, (1996). The Baltic Sea in 1996 – Continuation of the stagnation and decreasing phosphate concentrations. Dt. Hydrogr. Z. 48: 161–174.

Matthäus W, Nausch G, Lass HU, Nagel K and Siegel H (1999). The Baltic Sea in 1998 – Characteristic features of the current stagnation period, nutrient conditions in the surface layer and exceptionally high deep water temperatures. Dt. Hydrogr. Z. 51: 67–84.

Matthäus W, Nausch G, Lass HU, Nagel K and Siegel H (2000). Hydrographisch-chemische Zustandseinschätzung der Ostsee 1999. Mar. Sci. Rep./Meereswiss. Ber., Warnemünde, 39: 1–73.

Matthäus W, and Schinke H, (1999). The influence of river runoff on deep water conditions of the Baltic Sea. Hydrobiologia 393: 1-10.

Meili M, Jonsson P and Carman R (2000). PCB levels in laminated coastal sediments of the Baltic Sea along gradients of eutrophication revealed by stable isotopes (* 15N, (*13C). Ambio 29: 282–287.

Meironyté D, Norén K and Bergman Å (1999). Analysis of polybrominated diphenyl ethers in Swedish human milk. A time- related trend study, 1972–97. J. Toxicol. Environ. Health, Part A 58: 101–113.

Mellergaard S and Bagge O (1998). Fish gear-induced skin ulcerations in Baltic cod, Gadus morhua L .J. Fish Disease 21: 205–213.

Meyer H and Lampe R (1999). The restricted buffer capacity of a south Baltic estuary – the Oder estuary. Limnologica 29: 242–248.

Miljøstyrelsens Havforureningslaboratrium; Levnedsmiddelstyrelsen; Civilforsvarets Analystisk-Kemiske Laboratorium, 1986: “Report on accumulation tests of war gas agents in fish.” January 1986 (In Danish).

Middelboe AL, Sand-Jensen K and Brodersen K (1997). Patterns of macroalgal distribution in the Kattegat-Baltic region. Phycologia 36: 208–219.

Murubi M (1997). Toxicity of mustard gas and two arsenic-based chemical warfare agents on Daphnia magna. Projektrapport från utbildningen i Ekotoxikologi, Ekotoxikologiska avdelningen, Nr. 53. Uppsala universitet. 30 pp.

Nausch G, Nehring D and Ærtebjerg G (1999). Anthropogenic nutrient load of the Baltic Sea. Limnologica 29: 233–241.

Nausch G and Schlungbaum G (1995). Nährstoffdynamik in einem flachen Brackwasser-system (Darß-Zingster Boddengewässer) unter dem Einfluß variierender meteorologischer und hydrographischer Bedingungen. Bodden 2: 153–164.

Nehring D and Matthäus W (1991). Current trends in hydrographic and chemical parameters and eutrophication in the Baltic Sea. Int. Revues ges. Hydrobiol. 76: 297–316.

Nehring D, Matthäus W, Lass HU, Nausch G (1994). Hydrographisch-chemische Zustandseinschätzung der Ostsee 1993. Meereswiss. Ber., Warnemünde 7: 1–68.

Nehring D, Matthäus W, Lass H-U, Nausch G and Nagel K (1995). The Baltic Sea in 1995 – beginning of a new stagnation period in its central deep waters and decreasing nutrient load in its surface layer. Dt. Hydrogr. Z. 47: 319–327.

Nellen W and Thiel R (1996). Fische. In: Rheinheimer G (Ed.) Meereskunde der Ostsee. Berlin, Heidelberg, New York (Springer), pp. 190–196.

Neumann T, Leipe T and Shimmield G (1998). Heavy-metal enrichment in surficial sediments in the Oder River discharge area: source or sink for heavy metals? Applied Geochemistry 13: 329–337.

Neumann T, Leipe T, Brand T and Shimmield G (1996). Accumulation of heavy metals in the Oder estuary and its off-shore basins. Chem. Erde 56: 207–222.

Nielsen R and Dahl K (1992). Macroalgae at Briseis Flak, Schultzs Grund and Store Middelgrund, stone reefs in southern and eastern part of Kattegat, Denmark. In: Proceedings of the 12th Baltic Marine Biologists Symposium. Bjørnestad E, Hagerman L and Jensen K (Eds.), Olsen and Olsen. pp. 109–118.

Nielsen R, Kristiansen A, Mathiesen l and Mathiesen H (1995). Distributional index of the benthic macroalgae of the Baltic Sea area. Acta Bot. Fennica 155: 1–51.

Nielsen TG and Kiørboe T (1994). Regulation of zooplankton biomass and production in a temperate, coastal ecosystem. II Ciliates. Limnol. Oceanogr. 39: 508–519.

Nielsen TG (1991). Contribution of zooplankton grazing to the decline of a Ceratium bloom. Limnol. Oceanogr. 36(6): 1191–1206.

Nielsen TG and Hansen PJ (1999). Dyreplankton i danske farvande (in Danish). National Environmental Research Institute, TEMA-rapport No. 28, 62 pp.

Nielsen TG, Bjørnsen PK, Kaas H, Rasmussen B and Cohen A (1994). Plankton-dynamik omkring springlaget i Kattegat (in Danish). Danish EPA, Havforskning fra Miljøstyrelsen 33, 166 pp.

Nielsen TG, Kiørboe T and Bjørnsen PK (1990). Effects of a Chrysochromulina polylepis subsurface bloom on the plankton community. Mar. Ecol. Prog. Ser. 62: 21–35.

Niemistö L and Voipio A (1981). Notes on the sediment studies in the Finnish Pollution Research in the Baltic Sea. Rapp. P-V Rzun. Cons. int. Explor. Mer. 181: 87–92.

Nordberg L (1994). Baltic Coastal Conservation. General legal protection of the coastal strip and coastal habitats in the Baltic Sea States and Norway. Memorandum 1. 12 pp.

Norén F, Haamer J and Lindahl O (1999). Changes in the plankton community passing a Mytilus edulis mussel bed. Mar. Ecol. Prog. Ser. 191: 187–194.

Norkko A and Bonsdorff E (1996a). Rapid zoobenthos community responses to accumulations of drifting algae. Mar. Ecol. Prog. Ser. 131: 143–157.

Norkko A and Bonsdorff E (1996b). Population responses of coastal zoobenthos to stress induced by drifting algal mats. Mar. Ecol. Prog. Ser. 140: 141–151.

Nylund K, Asplund L, Jansson B, Litizen K and Sellström U (1992). Analyses of some polyhalogenated organic pollutants in sediments and sewage sludge. Chemosphere 24: 1721–1730.

Ojaveer E and Elken J (1997). On regional subunits in the Baltic Sea. In: Proceedings of the 14th Baltic Marine Biologists Symposium, Parnu, Estonia, 5–8 August 1995, pp. 156–169. Estonian Academy Publishers, Tallinn. 302 pp.

Ojaveer E, Lindroth A, Bagge O, Lehtonen H and Toivonen J (1981). Fish fauna of the Baltic Sea. In: Voipio A (Ed.) The Baltic Sea, pp. 275–292. Elsevier Oceanography Series. 418 pp.

Ojaveer E, Lumberg A and Ojaveer H (1998). Highlights of zooplankton dynamics in Estonian waters (Baltic Sea). ICES Journal of Marine Science 55: 748–755.

Ojaveer H and Lumberg A (1995). On the role of Cercopagis pengoi (Ostroumov) in Pärnu Bay and the NE part of the Gulf of Riga ecosystem. Proc. Estonian Acad. Sci. Ecology 5: 20–25.

Olausson E (1975). Man-made effect on the sediments from Kattegat and Skagerrak. GFF 97: 3–12.

Oldén B, Peterz M and Kollberg M (1988). Sjöfågeldöd i fisknät i nordvästra Skåne. Naturvårdsverket Rapport 3414. 51 pp. Solna, Sweden.

Olenin S, (1997). Benthic zonation of the Eastern Gotland Basin, Baltic Sea. Neth.J.Aquat.Ecol. 30: 265-282.

Olenin S and Leppäkoski E (1999). Non-native animals in the Baltic Sea: alteration of benthic habitats in coastal inlets and lagoons. Hydrobiologia 393: 233–243.

Olenin S, Orlova M and Minchin D (1999). Dreissena polymorpha (Pallas, 1771). In: Gollasch S, Minchin D, Rosenthal H and Voigt M (Eds.) Case histories on introduced species: their general biology, distribution, range expansion and impact. Berlin. Logos-Verlag. pp. 37–42.

Olesen M and Lundsgaard C (1995). Seasonal sedimentation of autochtonous material from the euphotic zone of a coastal system. Estuar. Coast. Shelf Sci. 41: 474–490.

Olesen M, Lundsgaard C and Andrushaitis A (1999). Influence of nutrients and mixing on the primary production and community respiration in the Gulf of Riga. J. Marine Systems, 23 (1–3): 127–143.

Olsson M, Bignert A, Eckhéll J and Jonsson P (2000). Comparison of temporal trends (1940s-1990s) of DDT and PCB in Baltic sediment and biota in relation to eutrophication. Ambio 29: 195–201.

Olsson A and Bergman A (2000). Data on prevalence of skull bone leisons. Personal communication.

Olsson A, Valters K and Burreau S (2000). Concentration dynamics of organochlorine substances in relation to fish size and trophic position – A study on perch (Perca fluviatilis). Environ. Sci. Technol. 34: 4878–4886.

Olsson M and Reutergårdh L (1986). DDT and PCB pollution trends in the Swedish environment. Ambio 15(2): 103–109.

Olsson M and Sandegren F (1983). Is PCB partly responsible for the decline of the otter in Europe? Proceedings from the 3rd International Otter Symposium, Strasbourg, 24–27 November, 1983.

Olsson M, Johnels AG and Vaz R (1975). DDT and PCB levels in seals from Swedish waters. The occurrence of aborted seal pups. Proceedings from the Symposium on the Seal in the Baltic. 4–6 June, 1974, Lidingö, Sweden. SNV PM, No. 591, pp. 43–65.

Omstedt A, (1990). Modelling the Baltic Sea as thirteen sub-basins with vertical resolution. Tellus, 42A, pp. 286–301.

Orlova MI, Khlebovich VV and Komendantov AYu (1998). Potential euryhalinity of Dreissena polymorpha (Pallas) and Dreissena bugensis (Andr.) Russ. J. Aq. Ecol. 7: 17–28.

Osowiecki A, and Warzocha J, (1996). Macrozoobenthos of the Gdansk, Gotland and Bornholm basins in 1978-1993. Oceanlogical Studies 1-2: 137-149.

Palm H and Lammi R (1995). Fate of pulp mill organochlorines in the Gulf of Bothnia sediments. Environ. Sci. Technol. 29(7): 1722–1727.

Palm HW and Dobberstein RC (1999). Occurrence of trichodinid ciliates (Peritricha: Ureolariidae) in the Kiel Fjord, Baltic Sea, and its possible use as a biological indicator. Parsitol. Res. 85: 726–732.

Panov V, Krylov P and Telesh I (1999). The St Petersburg harbour profile. In: Gollasch S and Leppäkoski E (Eds.) Initial risk assessment of alien species in Nordic coastal waters. Nordic Council of Ministers. Nord 8: 225–244.

Panov VE (1996). Establishment of the Baikalian endemic amphipod Gmelinoides fasciatus Stebb. in Lake Ladoga. Hydrobiologia (Berlin) 322: 187–192.

Partanen-Hertell M et al. (1999). Raising environmental awareness in the Baltic Sea Area. Finnish Environment 327: 1–128.

Pastuszak M, Nagel K, Nausch G (1996). Variability in nutrient distribution in the Pomeranian Bight in September 1993. Oceanologia 38: 195–225.

Pearson T and Rosenberg R (1992). Energy flow through the SE Kattegat: a comparative examination of the eutrophication of a coastal marine ecosystem. Neth. J. Sea Res. 28: 317–334.

Pedersen FB (1993). Fronts in the Kattegat: The hydrodynamic regulating factor for biology. Estuaries 16: 104–113.

Pekar M, Pavlova N, Gusev A, Sha-talov V, Vulikh N, Ioannisian D, Dutchak S, Berg T and Hjellbrekke A-G (1999). Long-range transport of selected persistent organic pollutants. Joint report of EMEP Centres: SC-E and CCC EMEP/MSC-E 4/99.

Persson L-E, Osowiecki A, Warzocha J, Olenin S, Solovjova S and Andersin A-B, (1996). Benthic biology. In: HELCOM. Third preiodic assessment of the state of the marine environment of the Baltic Sea, 1989-93; Background document. Balt.Sea Env.Proc. 64B: 100-102.

Perttilä M and Haahti H (1986). Chlorinated hydrocarbons in the water and sediments of the seas around Finland. Publications of the Water Research Institute, National Board of Waters, Finland, No. 68: 197–200.

Petersen JK and Riisgård HU (1992). Filtration capacity of the ascidian Ciona intestinalis and its grazing impact in a shallow fjord. Mar. Ecol. Prog. Ser. 88: 9–17.

Petersen MF, Borum J and Brøgger L (1999). Etablering af ålegræs og samspil mellem plante og miljø. In: Lomstein B (Ed.) Havmiljøet ved årtusindskiftet (In Danish), Olsen and Olsen, pp. 1–176.

Pihl L and Svensson A (1998). Förekomst, utbredning och biomassa av fintrådiga grönalger i grunda mjukbottenområden i Bohuslän under 1998 (In Swedish). Report to the Water Quality Association of the Bohus Coast, Uddevalla, Sweden. 8 pp.

Pihl L, Svensson A, Moksnes P-O and Wennhage H (1997). Utredning av fintrådiga grönalger i grunda mjukbottensområden i Göteborgs och Bohus län under 1994–96 (In Swedish). Länsstyrelsen i Göteborgs och Bohus län, Miljöavdelningen 1997 22: 18 pp.

Piliczewski B (1999). Dzialalnosz ludzka i wykorzystanie Baltyckiego. In: Stan Srodowiska polskich obszarów Morza Baltyckiego, IMGW, Gdynia. (Human Activity and Uses of the Baltic Sea. In: The environmental state of the Polish areas of the Baltic Sea, IMGE, Gdynia.

Pitkänen H, Lehtoranta J and Räike A (2001). Internal: nutrient fluxes counteract decreases in external load: The case of the estuarial Gulf of Finland. Ambio 30: 195-201.

Pitkänen H and Kauppila P (2000a). Hydrography and oxygen conditions. In: Kauppila P and Bäck S (Eds.) State of the Finnish coastal waters in the 1990s. The Finnish Environment 472: 30–36.

Pitkänen H and Kauppila P (2000b). Nutrients. In: Kauppila P and Bäck S (Eds.) State of the Finnish coastal waters in the 1990s. The Finnish Environment 472: 37–60.

Pitkänen H and Välipakka P (1997). Extensive deep water oxygen deficit and benthic phosphorus release in the Eastern Gulf of Finland in late summer 1996. In: Sarkkula J (Ed.) Proceedings of the Final Seminar of the Gulf of Finland, Year 1996, March 17–18, 1997 Helsinki. Suomen Ympäristökeskuksen moniste 105, 378 pp.

Pitkänen H, Kangas P, Miettinen V and Ekholm P (1987). The state of the Finnish coastal waters in 1979–83. Vesi- ja ympäristöhallinnon julkaisuja No. 8, 167 pp.

Pitkänen H, Kondratyev S, Lääne A, Gran V, Kauppila P Loigu E, Markovets I, Pachel K and Rumyantsev V (1997). Pollution load on the Gulf of Finland from Estonia, Finland and Russia in 1985–95. In: Sarkkula J (Ed.) Proceedings of the Final Seminar of the Gulf of Finland Year 1996, March 17–18, 1997 Helsinki. Suomen Ympäristökeskuksen moniste 105. 378 pp.

Purasjoki K (1936). Merilevien vertikaalisesta esiintymisest. (Vertical distribution of marine algae in Tvärminne region). Tvärminnessä. MSc thesis, Univ. of Helsinki. 126 pp. (In Finnish).

Rahimian H (1998). Pseudobacciger harengulae (Yamaguti, 1938) (Digenea: Fellodistomidae) from the Atlantic herring Clupea harengus L: A new host and locality record. In: On parasites of herring, Clupea harengus L, from the west coast of Sweden. Doctoral dissertation from Göteborg University, Faculty of Natural Science, Sweden.

Raid T and Lankov A (1995). Recent changes in the growth and feeding of the Baltic herring and sprat in the northeastern Baltic Sea. Proc. Est. Acad. Sci. Ecol., 5(1/2): 38–55.

Rantanen P (1994). Biological phosphorus removal study at the Suomenoja research station. Vatten 50: 321–328.

Rask N, Pedersen SE and Jensen MH (1999). Response to lowered nutrient discharges in the coastal waters around the island of Funen, Denmark. Hydrobiologia 393: 69–81.

Rasmussen B (1995). Stratification in Kattegat. PhD Thesis. National Environmental Research Institute, Denmark, 145 pp.

Rasmussen B (1997). The near-surface horizontal buoyancy flux in a highly stratified region, Kattegat. Estuar. Coast. Shelf Sci. 45: 405–414.

Rasmussen E (1977). The wasting disease of eelgrass (Zostera marina) and its effects on environmental factors and fauna. In: McRoy CP and Helffrich C (Eds.) Seagrass ecosystems, a scientific perspective. Marcel Dekker, New York. pp. 1–51.

Redfield AC, Ketchum BH and Richards FA (1963). The influence of organisms on the composition of sea water. In: Hill MN (Ed.) The Sea Vol. II, New York, Wiley, pp. 26–77.

Rice J and Gislason H (1996). Patterns of change in the size spectra of numbers and diversity of the North Sea fish assemblage, as reflected in surveys and models. ICES Journal of Marine Science 53: 1214-1225.

Richardson K (1985). Plankton distribution and activity in the North Sea/Skagerrak-Kattegat frontal area in April 1984. Mar. Ecol. Prog. Ser. 26(3): 233–244.

Richardson K and Christoffersen A (1991). Seasonal distribution and production of phytoplankton in the southern Kattegat. Mar. Ecol. Prog. Ser. 78: 219–227.

Riget F, Johansen P and Asmund G (1993). Naturlig variation af kobber, cadmium, bly og zink i blæretang og blåmusling ved Nuuk. Teknisk rapport. Grønlands Miljøundersøgelser.

Riisgård HU, Poulsen L and Larsen PS (1996). Phytoplankton reduction in near-bottom water caused by filter-feeding Nereis diversicolor – implications for worm growth and population grazing impact. Mar. Ecol. Prog. Ser. 141: 47–54.

Rolff C, Broman D, Näf C and Zebuhr Y (1993). Chemosphere 27: 461–468.

Roos A, Greyerz E, Olsson M and Sandegren F (2000). The otter (Lutra lutra) in Sweden – population trends in relation to SDDT and total PCB concentrations during 1968–98. Environmental Pollution 111: 457–469.

Rosenberg R (1995). Benthic marine fauna structured by hydrodynamic processes and food availability. Neth. J. Sea Res. 34: 303–317.

Ross PS, De Swart RL, Reijnders PJH, van Loveren H, Vos GJ and Osterhaus ADME (1995). Contaminant-related suppression of delayed-type hypersensitivity and antibody responses in harbour seals fed herring from the Baltic Sea. Environmental Health Perspect. 103: 162–167.

Ross PS, De Swart RL, Timmerman HH, Reijnders PJH, Vos JG, van Loveren H and Osterhaus ADME (1996). Suppression of natural killer cell activity in harbour seals (Phoca vitulina) fed Baltic Sea herring. Aquatic Toxicology 34: 71–84.

Roy SO, Packard TT, et al. (1999). Impact of acetat, pyruvate, and physiological state on respiration and respiratory quotients in Pseudomonas nautica. Aquatic Microb. Ecol. 17: 105–110.

Rumohr H, Bonsdorff E, and Pearson T, (1996). Zoobenthic succession in Baltic sedimentary habitats. Arch.Fish.Mar.Res. 44: 19-214.

Rumohr H and Krost P (1991). Experimental evidence of damage to benthos by bottom trawling with special reference to Arctica islandica. Meeresforschungen 33: 340–345.

Ruuskanen A and Bäck S (1999). Morphological variation of northern Baltic Fucus vesiculosus. Ophelia 50: 43–59.

Ryaboshapko A, Ilyin I, Gusev A, Afinogenova O, Berg T and Hjellbrekke A-G (1999). Monitoring and modelling of lead, cadmium and mercury transboundary trans-port in the atmosphere of Europe. Joint report of EMEP Centres MSC-E and CCC EMEP/MSC-E 3/99.

Rylander L, Strömberg U and Hagmar L (1995). Decreased birth weight among infants born to women with a high dietary intake of fish contaminated with persistent organochlorine compounds. Scandinavian Journal of Work, Environment and Health 21: 368–375.

Ryman N and Laikre L (1991). Effect of supporting breeding on the genetically effective population size. Conservation Biology 3: 325–329.

Rönnberg O, Lehto J and Haahtela I (1985). Changes in the occurrence of Fucus vesiculosus L. in the Archipelago Sea SW Finland. Annales Bot Fennici 22: 231–244.

Sandén P, Rahm L and Wulff F (1991). Non-parametric trend test of the Baltic Sea data. Environmetrics 2(3): 263–278.

Sapek A (1998). Phosphorus cycle in Polish agriculture. In: Sapek A (Ed.) Phosphorus in agriculture and water quality protection. Institute of Land Reclamation and Pasture (IMUZ), Falenty 1998: 8–18.

Sapota G (2000). Bioaccumulation of chlorinated hydrocarbons in the food chain of the Gulf of Gdansk environment. Institute of Meteorology and Water Management – Maritime Branch Gdynia. Ph.D. dissertation thesis.

Schinke H and Matthäus W (1998). On the causes of major Baltic inflows – an analysis of long time series. Cont. Shelf Res.18: 67–97.

Schneider B (1993). Untersuchung und Bewertung des Schadstof-feintrags über die Atmosphäre im Rahmen von PARCOM (Nordsee) und HELCOM (Ostsee). GKSS 93/E/53, Geesthacht.

Schneider B (1996). Heavy Metals. In: Third periodic assessment of the state of the marine environment of the Baltic Sea, 1989–93: Background Document. Baltic Sea Environmental Proceedings No. 64B, HELCOM 1996, pp. 144–153.

Schneider B and Pohl C (1996). Time series for dissolved cadmium at a coastal station in the western Baltic Sea. J. Mar. Syst. 9: 159–170.

Schneider B, Ceburnis D, Marks R, Munthe J, Petersen G and Sofiev M (2000). Atmospheric Pb and Cd input into the Baltic Sea: A new estimate based on measurements. Mar. Chem. 71: 297–307.

Segerstråle SG (1969). Biological fluctuations in the Baltic Sea. Prog. Oceanogr. 5: 169–184.

Seinä A and Palosuo E (1996). The classification of the maximum annual extent of ice cover in the Baltic Sea 1720–1995. Based on the material collected by Risto Jurva (winters 1720–1940) and the material of the Ice Service of the Finnish Institute of Marine Research (winters 1941–95). MERI – Report series of the Finnish Institute of Marine Research No. 27: 79– 91.

Seisuma Z, Pedersen B and Larsen B (1998). Vertical distribution of heavy metals and total C and N in sediments of the Gulf of Riga. Proc. Latvian Acad. Sci. Section B, Vol 52.

Sellner K G, (1997). Physiology, ecology and toxic properties of marine cyanobacteria blooms. Limnol. Oceanogr., Vol. 42: 1089-1104.

Sellström U (1999). Determination of some polybrominated flame retardants in biota, sediment and sewage sludge. Doctoral Dissertation, Environmental Chemistry, Stockholm University, Sweden.

Seppälä J, Tamminen T and Kaitala S (1999). Experimental evaluation of nutrient limitation of phytoplankton communities in the Gulf of Riga. J. Marine Systems 23(1–3): 107–126.

Siebert U, Benke H, Schulze G and Sonntag RP (1996). Über den Zustand der Kleinwale. In: Lozan J-L, Lampe R, Matthäus W, Rachor E, Rumohr H and von Westernhagen H (Eds.) Warnsignale aus der Ostsee. Berlin. pp. 242–248.

Siegel H, Matthäus W, Bruhn R, Gerth M, Nausch G, Neumann T and Pohl C (1998). The exceptional Oder flood in summer 1997 – Distribution patterns of the Oder discharge in the Pomeranian Bight. Dt. Hydrogr. Z. 50: 145–167.

Sjöåsen T, Ozolins J, Greyerz E and Olsson M (1997). The Otter (Lutra lutra) Situation in Latvia and Sweden related to PCB and DDT levels. Ambio 26: 196–201.

Skóra KE (1999). Nowa ryba w Pucyfiku (in Polish) ([New fish in the Puck Bay). Mag. Prz. Ryb. Gdynia. 4(8): 7.

Skóra KE and Stolarski J (1993). New fish species in the Gulf of Gdansk. Neogobius sp. (cf. Neogobius melanostomus Pallas 1811), Bulletin of the Sea Fisheries Institute, Gdynia, 1(128): 83.

Smit MD, Leonards PEG, Madsen A, Hatti BGM, Murk AJ, Jongh AWJJ (1996). Bioaccumulation of PCBs in Danish otter habitats. In: Smit MD, Leonards PEG, Murk AJ, de Jongh AWJJ and van Hattum B (Eds.) Development of otter-based quality objectives for PCBs. Inst.for Env. Studies, Vrije Universiteit, Amsterdam. pp. 7–31.

Sonntag R, Benke H and Coljin F (1995). EG-weite und gemeinsame Sichtung von Kleinwalen in der Nordsee per Schiff und Flugzeug zur Bestandserfassung und Verteilung von Kleinwalen. Forschungsvorhaben Nr. 808 05 63. Endbericht, FTZ Büsum.

South Jutland County (1998). Heavy metals and anthropogenic substances in ports and fjords. South Jutland County (Danish report), pp. 38.

Sparholt H (1996). Causal correlation between recruitment and spawning stock size of central Baltic cod? ICES Journal of Marine Science 53: 771–779.

Sparholt H, Aps R, Aro E, Fetter M, Flinkman J, Kornilovs G, Raid T, Shvetsov F, Sjöstrand B and Uzars D (1994). Growth changes of herring in the Baltic. TemaNord 1994: 532.

SSPA (1996). Transportation of oils in the Baltic Sea Area 1995. Inventory of transportation patterns and risk estimation of oils carried in the Baltic Sea Area. SSPA Maritime Consulting AB, Report 7596–1.

SSPA (1998). Oil handling in the Baltic Sea Area, 1996–2001. SSPA Maritime Consulting AB, Report 7935–2.

Stempniewicz L (1994). Marine birds drowning in fishing nets in the Gulf of Gdansk (southern Baltic): numbers, species composition, age and sex structure. Ornis Svecica, 4: 123–132.

Stenthilkumar K, Duda CA, Villeneuve DL, Kannan K, Falandysz J and Giesy JP. (1999). Butyl tin Compounds in Sediment and Fish from the Polish Coast of the Baltic Sea. ESPR Environ. Sci. and Pollut. Res. 6(4): 200–206.

Stigebrandt A (1983). A model for the exchange of water and salt between the Baltic and the Skagerrak. J. Phys. Oceanogr. 13: 411–427.

Stigzelius J, Laine A, Rissanen J, Andersin AB and Ilus E (1997). The introduction of Marenzelleria viridis (Polychaeta, Spionidae) into the Gulf of Finland and the Gulf of Bothnia (Northern Baltic Sea). Ann. Zool. Fennici 34: 205–212.

Stjernberg T and Koivussari J (1995). The recovery of the white-tailed sea eagle population in Finland. Linnut 3: 5–14.

Stjernberg T and Hagner-Wahlsten N (1994). Saukon levinneisyys Suomessa vuosina 1975 ja 1985 Abstract in English. Suomen Riista 40: 42–49.

Strandström T (1980). Studier över blåstången (Fucus vesiculosus L) I Tvärminneområdet. MSc Thesis, Univ. of Helsinki. 95 pp.

Szczepanska T and Uscinowicz S (1994). Geochemical Atlas of the Southern Baltic (1: 500 000). Panstwowy Institytut Geologiczny, Warszawa. pp. 23–34.

Szpunar J, Falandysz J, Schmitt VO and Oberska E. (1997). Butyl tins in marine and freshwater sediments of Poland. Bull. Environ. Contam. Toxicol. 58: 859–864.

Södergren A, Adolfsson-Erici M, Bengtsson B-E, Jonsson P, Lagergren S, Rahm L and Wulff F (1993). Bleached Pulp Mill Effluents. Fate and Effects in the Baltic Sea. Swedish Environmental Protection.

Tallberg P and Heiskanen A-S (1998). Species-specific phytoplankton sedimentation in relation to primary production along an in-shore-offshore gradient. J. Plankton Res. 20 (11): 2053–2070.

Tamminen T and Kivi K (1996). Typpikuormitus, ravinnekierrot ja rannikkovesien rehevöityminen. PELAG III loppuraportti. 76 pp.

Tamminen T and Seppälä J (1999). Nutrient pools, transformations, ratios, and limitation in the Gulf of Riga, Baltic Sea, during four successional stages. J. Marine Systems, 23(1–3): 83–106.

Tarrason L and Schaug J (Eds.) (1999). Transboundary Acid Deposition in Europe. EMEP/MSC-W Report 1/99.

Tarrason L and Schaug J (Eds.) (2000). Transboundary acidification and eutrophication in Europe. EMEP/MSC-W Report 1/00.

Tarrason L, Barrett K, Erdman L and Gusev A (1997). Atmospheric supply of nitrogen, lead and cadmium to the Baltic Sea. EMEP Summary Modelling Report for 1991–95. EMEP/MSC-W, Norwegian Meteorological Institute Research Report, No. 58.

Telesh IV, Alimov AF, Golubkov SM, Nikulina VN and Panov VE (1999). Response of aquatic communities to anthropogenic stress: a comparative study of Neva Bay and the eastern Gulf of Finland. Hydrobiologia. 393: 95–105.

Thomsen HA (1992). Plankton i de indre danske farvande (in Danish). Danish EPA, Havforskning fra Miljøstyrelsen 11. 336 pp.

Tonderski A (1997). Control of nutrient fluxes in large river basins. Univ. Linköping, Linköping Studies in Arts and Science. PhD Thesis. No. 157.

Trzosinska A and Andrulewicz E (Eds.) (1998). Short-term effects of 1997 flood on the marine environment of the Gulf of Gdansk and Pomeranian Bay. Wydawnictwo MIR, Gdynia, (in Polish).

Trzosinska A, Lysiak-Pastuszak E (1996). Oxygen and nutrients. Oceanological Studies 1–2: 41–76.

Tuominen L, Heinänen A, Kuparinen J and Nielsen LP (1998). Spatial and temporal variability of denitrification in the sediments of the northern Baltic Proper. Mar. Ecol. Prog. Ser. 172: 13–24.

Tuominen L, Mäkelä K, Lehtonen KK, Haahti H, Hietanen S and Kuparinen J (1999). Nutrient fluxes, pore water profiles and denitrification in sediment influenced by algal sedimentation and bioturbation by Monoporeia affinis. Estuar. Coastal Shelf Sci: 49: 83–97.

Uhd Jepsen P (2000). Draft action plan for the implementation of the HELCOM project on seals (unpublished). National Forest and Nature Agency, Denmark.

Uscinowicz S, Ebbing J, Laban C and Zachowicz J (1998). Recent muds of the Gulf of Gdansk. Baltica 11: 25–32.

Vahteri P, Mäkinen A, Salovius S and Vuorinen I (2000). Are drifting algal mats conquering the bottom of the Archipelago Sea, SW Finland? AMBIO 29: 338–343.

Vallius H (1999). Heavy metals deposition and variation in sedimentation rate within a sedimentary basin in the Central Gulf of Finland. Chemosphere 38(9): 1959–72.

Vallius H and Leivuori M (1999). The distribution of heavy metals and arsenic in recent sediments in the Gulf of Finland. Boreal Env. Res. 4: 19–29.

Valovirta I and Porkka M (1996). The distribution and abundance of Dreissena polymorpha (Pallas) in the eastern Gulf of Finland. Memoranda Soc. Fauna Flora Fennica 72: 63–78.

Vartiainen T, Parmanne R and Hallikainen A (1997). Ympäristömyrkkyjen kertyminen silakkaan. YMPÄRISTÖ JA TERVEYS-LEHTI. 7–8/97.

Viitasalo M, Vuorinen I and Saesmaa S (1995). Mesozooplankton dynamics in the northern Baltic Sea: implications of variations in hydrography and climate. J. Plankt. Res. 17: 1857–1878.

Voipio A (1969). On the cycle and balance of phosphorus in the Baltic Sea. Suomen Kemistilehti 42: 48–54.

Waern M (1952). Rocky shore algae in the Öregrund archipelago. Acta phytogeogr. Suec. 30: 1–298.

Wania F and Mackay D (1993). Global fractionation and cold condensation of low volatility organochlorine compounds in Polar regions. Ambio, Vol 22, No 1.

Wasmund N, Nausch G, Matthäus W (1998). Phytoplankton spring blooms in the southern Baltic Sea – spatio-temporal development and long-term trends. J. Plankton Res. 20: 1099–1117.

Wasmund N, Nausch G, Postel L, Witek Z, Zalewski M, Gromisz S, Lysiak-Pastuszak E, Olenina I, Kavolyte R, Jasinskaite A, Müller-Karulis B, Ikauniece A, Andrushaitis A, Ojaveer H, Kallaste K, Jaanus A (2000). Trophic status of coastal and open areas of the southeastern Baltic Sea based on nutrient and phytoplankton data from 1993–1997. Meerswiss. Ber., Warnemünde, 38, 83 pp.

Wasmund N, Voss M and Lochte K, (2000). Annual nitrogen fixation patterns in the Baltic Proper. Mar. Ecol. Progr. Ser. (in press).

Wikner J and Hagström Å (1999). Bacterioplankton intra-annual variability at various allochthonous loading: Importance of hydrography and competition. Aquat. Microb. Ecol. 20: 245–260.

Wikner J, Cuadros R, et al. (1999). Differences in consumption of allochtonous DOC at limnic and estuarine conditions in a watershed. Aquatic. Microb. Ecol. 17: 289–299.

Witt G and Trost E (1999). Polycyclic aromatic hydrocarbons (PAHs) in sediments of the Baltic Sea and of the German coastal waters. Chemosphere 38(7) 1603–1614.

Witt G (1995). Polycyclic aromatic hydrocarbons in water and sediment of the Baltic Sea. Marine Pollution Bulletin 31(4–12): 237–248.

Wolska-Pys M (1994). Zmiennosc mezozooplanktonu wod otwartych poludniowego Baltyku jako wynik zmian rodowiska w: Problemy ochrony srodowiska przyrodniczego Morza Baltyckiego i strefy nadmorskiej (Mesozooplankton variations in the open waters of the southern Baltic Sea as the result of environmental changes. In: Problems of protection of the natural environment of the Baltic Sea and coastal area), Wyd. IOS, Warszawa, pp. 82–90.

Wolska-Pys M (1996). Mezozooplankton (w:) Warunki srodowiskowe polskiej strefy poludniowego Baltyku w 1995 roku, (Mesozooplankton. In: Environmental conditions of the Polish zone of the southern Baltic in 1995), IMGW Gdynia, pp. 163–168.

Wolska-Pys M, Ciszewska I. (1991). Changes in mesozooplankton of the southern Baltic in 1979–88, Act. Ichth. et Pisc. 21 (Suppl., Szczecin): 249–257.

Woodhead RJ, Law RJ and Matthiessen P (1999). Polycyclic aromatic hydrocarbons in surface sediments around England and Wales, and their possible biological significance. Marine Pollution Bulletin 38: 773–790.

Wulff F, Perttilä M and Rahm L (1996). Monitoring, mass balance calculation of nutrients and the future of the Gulf of Bothnia. AMBIO Special Report No. 6: 28–35.

Wängberg I, Schager P, Munthe J, Iverfeldt Å, Schmolke SR, Kock HH, Otten S, and Ebinghaus R, (1998). Estimates of the Air-Sea Exchange of Mercury in the Baltic Sea Derived from Summer and Winter Measurements. EU Baltic Sea Study ”BASYS” Subproject ”Atmospheric Loading” Proceedings of the Second BASYS Annual Scientific Conference 23-25 September 1998 Stockholm.

Yurkovskis A (1998). Course and environmental consequences of eutrophication in the Gulf of Riga. Proc. Latvian Acad. Sci., Section B, 52 (Suppl.): 56–61.

Yurkovskis A and Mazmachs M (1996). Hydrochemistry Inc. Third periodic assessment of the state of the marine environment of the Baltic Sea, 1989–93: Background document. Baltic Sea Environmental Proceedings No 64B, HELCOM 1996, pp. 64–66.

Yurkovskis A, Wulff F, Rahm L, Andrushaitis A and Rodriguez-Medina M (1993). A nutrient budget of the Gulf of Riga; Baltic Sea. Estuarine, Coastal and Shelf Sciences, 37: 113–127.

Zettler ML (1997). The newcomer Marencelleria viridis (Verrill, 1873), its development and influence on the indigenous macrozoobenthos in a coastal water of the Southern Baltic. Proceedings of the 14th Baltic Marine Biol. Sympos. Tallinn 5–8 August 1995, pp. 280–296.

Zmudzinski L (1996). The effect of the introduction of the American species Marenzelleria viridis (Polychaeta: Spionidae) on the benthic ecosystem of Vistula Lagoon. Mar. Ecol. 17: 221–226.

Zorita E and Laine AO (2000). Dependence of salinity and oxygen concentration in the Baltic Sea on the large-scale atmospheric circulation. Clim. Res. 14: 25–41.

Zweifel UL, Norrman B, et al. (1993). Consumption of dissolved organic carbon by marine bacteria and demand for inorganic nutrients. Mar. Ecol. Prog. Ser. 101: 23–32.

Ærtebjerg G, Carstensen J, Conley D, Dahl K, Hansen J, Josefson A, Kaas H, Markager S, Nielsen TG, Rasmussen B, Krause-Jensen D, Hertel O, Skov H and Svendsen, LM (1998). Marine områder. Åbne farvande – status over miljøtilstand, årsagssammenhænge og udvikling. Vandmiljøplanens Overvågningsprogram 1997 (in Danish with an English summary). National Environmental Research Institute, Technical Report No. 254, 248 pp.

Ådjers K, Bignert A and Förlin L (1999). Fisk och fiske. Östersjö´98, pp 42–46. Stockholms Marina Forskningscentrum.

Aarhus County (2000). Tributyl tin (TBT) in the marine environment and malformation of marine snails in Aarhus County 1998–99. Aarhus County (Danish report), 54 pp.

13

Institutions and persons involved in the preparation of the Fourth Periodic Assessment

Country	Institution	Contributing persons
DENMARK	Ministry of the Environment Danish Environmental Protection Agency Strandgade 29 DK-1401 Copenhagen K	Mr. Kjeld F. Jørgensen Mr. Tonny Niilonen
	Ministry of the Environment Geological Survey of Denmark and Greenland Dept. Environmental History and Climate Thoravej 8 DK-2400 Copenhagen NV	Mr. Birger Larsen
	Ministry of the Environment National Environmental Research Institute Dep. of Marine Ecology Frederiksborgvej 399 P.O. Box 358 DK-4000 Roskilde	Mr. Alf Josefson Mr. Daniel Conley Mr. Karsten Dahl Mr. Thomas Forbes Mr. Stiig Markager Mr. Torkel Gissel Nielsen Ms. Britta Pedersen Mr. Bjarke Rasmussen Mr. Gunni Ærtebjerg Nielsen
	Ministry of the Environment National Environmental Research Institute Vejlsøvej 25 P.O. Box 314 DK-8600 Silkeborg	Mr. Lars M. Svendsen
	Ministry of the Environment National Forest and Nature Agency Haraldsgade 53 DK-2100 Copenhagen Ø	Mr. Ole Norden Andersen Ms. Anne Grethe Ragborg
	Ministry of the Environment National Forest and Nature Agency Nature and Wildlife Section Ålholtvej 1 DK-6840 Oksbøl	Mr. Palle Uhd Jepsen
	Risø National Laboratory Nuclear Safety Research Dept. NUK-114 P.O. Box 49 DK-4000 Roskilde	Mr. Sven P. Nielsen
ESTONIA	Environmental Information Center Mustamäe St. 33 EE-10616 Tallinn	Mr. Ott Roots
	Estonian Marine Institute Deptartment of Fisheries Research Viljandi Rd 18 B EE-11216 Tallinn	Mr. Ahto Järvik Mr. Tiit Raid
	Estonian Marine Institute Department of Marine Biology Marja St. 4D EE-10617 Tallinn	Mr. Andres Jaanus Mr. Ilmar Kotta Mr. Jonne Kotta Mr. Henn Kukk Mr. Georg Martin Mr. Henn Ojaveer Mr. Arno Pollumäe Mr. Mart Simm
	Estonian Marine Institute Department of Marine Physics Paldiski St. 1 EE-10137 Tallinn	Mr. Jüri Elken Mr. Laur Mäki Mr. Urmas Lips

Country	Institution	Contributing persons
FINLAND	Department of Environmental and Marine Biology Åbo Akademi University FIN-20014 Turku	Mr. Erkki Leppäkoski Ms. Johanna Mattila
	Finnish Environment Institute P.O. Box 140 FIN-00251 Helsinki	Ms. Saara Bäck Ms. Anna-Stinna Heiskanen Mr. Pentti Kangas Ms. Pirkko Kauppila Mr. Mikko Kiirikki Mr. Markku Korhonen Ms. Pirjo Kuuppo Mr. Jouni Lehtoranta Ms. Liisa Lepistö Mr. Heikki Pitkänen Mr. Juha Sarkkula
	Finnish Institute of Marine Research P.O. Box 33 FIN-00931 Helsinki	Mr. Pekka Alenius Ms. Ann-Britt Andersin Mr. Juha Flinkman Ms. Seija Hällfors Mr. Harri Kankaanpää Mr. Harri Kuosa Mr. Jorma Kuparinen Mr. Ari Laine Mr. Kari Lehtonen Ms. Mirja Leivuori Mr. Risto Lignell Mr. Kalervo Mäkelä Ms. Anita Mäkinen Ms. Eija Rantajärvi Mr. Ari Seinä Ms. Pirjo Tuomi
	Finnish Meteorological Institute Air Quality Department P.O. Box 503 FIN-00101 Helsinki	Ms. Tuija Ruoho-Airola
	Ministry of the Environment P.O. Box 380 FIN-00131 Helsinki	Mr. Lauri Nordberg Ms. Eeva-Liisa Poutanen
	STUK - Radiation and Nuclear Safety Authority P.O. Box 14 FIN-00881 Helsinki	Mr. Erkki Ilus
GERMANY	Agency for Environment, Nature Conservation and Geology Mecklenburg-Vorpommern Wampenerstrasse D-17498 Neuenkirchen	Mr. Rainer von Goerne
	Baltic Sea Research Institute Seestrasse 15 D-18119 Warnemünde	Mr. Jürgen Alheit Mr. Hans Ulrich Lass Mr. Thomas Leipe Mr. Wolfgang Matthäus Mr. Günther Nausch Mr. Lutz Postel Mr. Bernd Schneider Mr. Norbert Wasmund
	Bundesanstalt für Gewässerkunde Schnellerstr. 140 D-12493 Berlin	Ms. Katrin Grünewald
	Department of Marine Biology University of Rostock Freiligrathstrasse 7-8 D-18055 Rostock	Mr. Jochen C. Krause
	Federal Agency for Nature Conservation International Academy for Nature Conservation Isle of Vilm D-18581 Lauterbach/Rügen	Mr. Dieter Boedeker Mr. Henning von Nordheim
	Federal Environmental Agency Bismarckplatz 1 P.O.Box 330022 D-14191 Berlin	Ms. Heike Herata
	Federal Maritime and Hydrographic Agency Bernhard-Nocht-Str. 78 P.O. Box 301220 D-20305 Hamburg	Mr. H. Albrecht Mr. Horst Gaul Mr. Norbert Theobald
	Federal Research Centre for Fisheries Institute for Fishery Ecology Markmannstrasse 129b D-20539 Hamburg	Mr. Michael Haarich Mr. Uwe Harms
	Society to Save the Sturgeon Institute of Freshwater Ecology and Inland Fisheries / IGB-Berlin Mueggelseedamm 310 / PF 19 D-12587 Berlin / D-12561 Berlin	Mr. Joern Gessner

Country	Institution	Contributing persons
LATVIA	Centre for Systems Analysis University of Klaipeda H. Manto gat. 84 LT-5808 Klaipeda	Mr. Sergej Olenin
	Department of Marine Ecology Institute of Aquatic Ecology University of Latvia 3 Miera St. LV-2169 Salaspils	Mr. Andris Andrushaitis
	Marine Monitoring Centre Institute of Aquatic Ecology University of Latvia 8 Daugavgrivas str. LV-1007 Riga	Mr. Juris Aigars Ms. Maija Ceitlina Ms. Anda Ikauniece Mr. Mintauts Jansons B. Kalveka Ms. Jelena Kostrichkina Ms. Guna Luksha Mr. Mikelis Mazmachs Ms. I. Nikolushkina Mr. Aivars Yurkovskis
	Latvian Hydrometeorological Agency Maskavas Str. 165 LV-1019 Riga	Ms. Marta Treiliba
	Spatial Development Planning Centre Elizabetes Str. 2A-423 LV-1340 Riga	Mr. Aigars Kuskis
LITHUANIA	Environmental Protection Ministry Center of Marine Research Taikos St. 26 LT-5802 Klaipeda	Ms Natalija Budajeva Mr. Viktoras Didziulis Mr. Juozas Dubra Ms. Lina Gecaite Ms. Aldona Jasinskaite Ms. Rasa Jokubauskaite Ms. Daiva Jurgilaite Ms. Irina Olenina Ms. Regina Ragauskaite Mr. Algirdas Stankevicius Ms. Zoja Stukova Mr. Ovidijus Stulpinas Mr. Ignas Vysniauskas Ms. Renata Zajacauskaite
	Institute of Ecology Lab. Marine Ecology Akademijos 2 LT-2600 Vilnius	Mr. Rimantas Repecka
POLAND	Hel Marine Station University of Gdansk P.O. Box 37 PL-84 150 Hel	Mr. Krzysztof Skora
	Institute of Meteorology and Water Management Maritime Branch Ul. Waszyngtona 42 PL-81 342 Gdynia	Ms. Barbara Cyberska Ms. Natalia Drgas Mr. Wlodzimierz Krzyminski Ms. Elzbieta Lysiak-Pastuszak Mr. Miroslaw Mietus Ms. Grazyna Sapota Ms. Anna Trzosinska
	Institute of Meteorology and Water Management ul. Jordana 10/11 PL-40 056 Katowice	Mr. Waldemar Jarosinski
	Institute of Oceanography University of Gdansk ul. Pilsudzkiego 46 PL-81 378 Gdynia	Mr. Maciej Wolowicz
	Polish Academy of Sciences Marine Biology Center in Gdynia Ul. Swietego Wojciecha 5 PL-81 347 Gdynia	Ms. Lidia Kruk-Dowgiallo Mr. Zbigniew Mudryk Ms. Elzbieta Niemkiewicz Mr. Radoslaw Opiola Mr. Andrzej Osowiecki S. Uschinowicz
	Polish Academy of Sciences Ornithological Station Nadwislanska 108 PL-80 680 Gdansk	Mr. Maciej Gromadzki
	Sea Fisheries Institute Department of Oceanography Kollataja 1 PL-81 332 Gdynia	Mr. Eugeniusz Andrulewicz Mr. Jan Warzocha

Country	Institution	Contributing persons
RUSSIA	Institute of Global Climate and Ecology Glebovskaya 20 B RU-107 258 Moscow	Mr. Sergei Mosharov Mr. G. Panov Ms. Elena Serova T. Shchuka Ms. Irina Umbrumyants
	Regional Centre of Monitoring of Arctic Bering str. 18 RU-199 226 St. Petersburg	Mr. Sergei Melnikov
	State Oceanographic Institute (SOI) The Russian Federation Kropotkinski per. 6 RU-119 838 Moscow	Mr. Serguei Kirianov Mr. Alexandr Korshenko Ms. Irina Matveeva Mr. Semen Y. Oradovski Ms. Irina Telesh
	Zoological Institute St. Petersburg	Ms. Marina Orlova
SWEDEN	Geological Survey of Sweden Box 670 S-751 28 Uppsala	Mr. Ingemar Cato
	Institute for Applied Environment Research University of Gothenburg Box 464 S-405 30 Gothenburg	Mr. Göran Dave
	Institute for System Ecology University of Stockholm S-106 91 Stockholm	Mr. Hans Cederwall Mr. Hans Kautsky
	Kristineberg Marine Research Station S-450 34 Fiskebäckskil	Ms. Susanne Pihl-Baden Mr. Rutger Rosenberg
	National Board of Fisheries Institute of Coastal Research Gamla Slipvägen 19 S-740 71 Öregrund	Mr. Erik Neuman Mr. Olof Sandström
	Swedish Environmental Protection Agency Blekholmsterrassen 36 S-106 48 Stockholm	Mr. Carl-Elis Boström Mr. Sverker Evans Mr. Per Jonsson Ms. Anna-Helena Lindahl Mr. Lars Thorell Mr. Anders Widell Mr. Gunnar Zettersten
	Swedish Meteorological and Hydrological Institute (SMHI) Doktorsgatan 9 D S-262 52 Ängelholm	Mr. Lars Edler
	Swedish Meteorological and Hydrological Institute (SMHI) S-601 76 Norrköping	Mr. Hans Alexandersson Mr. Sten Bergström Mr. Bengt Carlsson Mr. Hans Dahlin Mr. Phil Graham
	Swedish Meteorological and Hydrological Institute (SMHI) Oceanographic Laboratory Building 31 Nya Varvet S-426 71 Västra Frölunda	Mr. Lars Andersson Mr. Nils Kajrup Ms. Elisabeth Sahlsten Mr. Björn Sjöberg
	Swedish Museum of Natural History Contaminant Research Group Box 50007 S-104 05 Stockholm	Mr. Anders Bergman Mr. Anders Bignert Mr. Björn Helander Mr. Tero Härkönen Mr. Mats Olsson
	Swedish Radiation Protection Institute S-171 16 Stockholm	Ms. Maria Lüning
	Swedish Threatened Species Unit Swedish University of Agricultural Sciences P.O. Box 7007 S-750 07 Uppsala	Mr. Roger Andersson
	Tjärnö Marine Biological Laboratory S-452 96 Strömstad	Mr. Jan Karlsson
	Umeå Marine Science Center Umeå University Norrbyn S-910 20 Hörnefors	Ms. Agneta Andersson-Nordström Mr. Kjell Leonardsson Mr. Johan Wikner
BIRDLIFE	BirdLife, Ornis Consult 140 Vesterbrogade DK-1620 Copenhagen V	Mr. Henrik Skov
HELCOM	HELCOM Secretariat Katajanokanlaituri 6 B FIN-00160 Helsinki	Ms. Anne Christine Brusendorff Mr. Kaj Forsius Mr. Juha-Markku Leppänen
ICES	International Council for the Exploration of the Sea (ICES) Palaegade 2-4 DK-1261 Copenhagen K	Ms. Janet Pawlak Mr. Henrik Sparholt

Colophon

Published by:
Helsinki Commission,
Katajanokanlaituri 6 B
FIN-00160 Helsinki, Finland
E-mail: Helcom@helcom.fi
Internet: http://www.helcom.fi

Editor:
Tonny Niilonen, Danish Environmental Protection Agency, Denmark.

Scientific editors:
Ole Norden Andersen, National Forest and Nature Agency, Denmark; Eugeniusz Andrulewiscz, Sea Fisheries Institute, Poland; Sten Bergström, Swedish Meteorological and Hydrological Institute, Sweden; Anders Bignert, Swedish Museum of Natural History, Sweden; Dieter Boedeker, Federal Agency for Nature Conservation, Germany; Hans Dahlin, Swedish Meteorological and Hydrological Institute, Sweden; Göran Dave, University of Gothenburg, Sweden; Sverker Evans, Swedish Environmental Protection Agency, Sweden; Heike Herata, Federal Environmental Agency, Germany; Kjeld Frank Jørgensen, Danish Environmental Protection Agency, Denmark; Wlodzimierz Krzyminski, Institute of Meteorology and Water Management, Poland; Juha-Markku Leppänen, Finnish Institute of Marine Research, Finland; Urmas Lips, Estonian Marine Institute, Estonia; Georg Martin, Estonian Marine Institute, Estonia; Wolfgang Matthäus, Baltic Sea Research Institute, Germany; Günther Nausch, Baltic Sea Research Institute, Germany; Sven Nielsen, Risø National Laboratory, Denmark; Mats Olsson, Swedish Museum of Natural History, Sweden; Janet Pawlak, International Council for the Exploration of the Sea; Eeva-Liisa Poutanen, Ministry of the Environment, Finland; Tuija Ruoho-Airola, Finnish Meteorological Institute, Finland; Bernd Schneider, Baltic Sea Research Institute, Germany; Henrik Sparholt, International Council for the Exploration of the Sea; Gunni Ærtebjerg, National Environmental Research Institute, Denmark; Aivars Yurkovskis, Marine Monitoring Centre, Latvia

Language revision: David I. Barry, On Line Activities, Raklev Høje 40, DK-4400 Kalundborg, Denmark

Cover photo: Brännskar, Tvärminne Archiepelago, south-western coast of Finland
(photo: J. Ekebom)

Back cover photo: Björkviksflada, Ekenäs Archiepelago, southern coast of Finland
(photo: J. Ekebom)

Design and layout: Lars Møller Nielsen, Studio-8, St. Kongensgade 110 C, DK-1264 Copenhagen K. Denmark

Number of pages: 216

Printing: Schultz Grafisk A/S

Number printed: 1000

Source reference: For bibliographic purposes this document should be cited as *HELCOM, 2002. Environment of the Baltic Sea Area 1994-1998. Balt. Sea Environ. Proc. No. 82B.*

ISNN: 0357-2994

NORDIC ENVIRONMENTAL LABEL
PRINTED MATTER
541 176

Helsinki Commission,
Katajanokanlaituri 6 B
FIN-00160 Helsinki, Finland
Telephone: +358-(0)9-6220 220
Telefax: +358-(0)9-6220 2239
E-mail: Helcom@helcom.fi
Internet: http://www.helcom.fi