2.9 Submerged aquatic vegetation

Seagrasses and macroalgae enhance biodiversity by providing habitat and shelter for many species.

Photo: NERI/Peter Bondo Christensen

Photo: Biofoto/S.E. Sørensen

Eelgrass (*Zostera marina*) is the most widely distributed marine angiosperm in shallow Danish coastal waters. On hard substrates the vegetation is dominated by macroalgae. Growth of short-lived nuisance species of macroalgae is a problem in inner parts of some estuaries.

Eelgrass in coastal waters

Eelgrass and seagrasses in general often represent a large standing biomass and a high primary production that influence the overall functioning of coastal ecosystems. Seagrasses enhance biodiversity by providing habitat and shelter for many species; they are nursery and foraging areas for commercially important species of fish; they improve water quality by reducing particle loads and absorbing dissolved nutrients; they stabilise sediments and influence global carbon and nutrient cycling (see Hemminga & Duarte 2000). The many ecosystem services provided by eelgrasses have created a concern for the response of this species to eutrophication. This concern has been intensified over the last few decades where large-scale reductions in seagrass meadows have been reported in response to eutrophication from many areas (see Short & Wyllie-Echeverria 1996).

The following paragraphs summarise the development of Danish eelgrass meadows during the last century. The presentation is based on early reports on eelgrass distribution covering the period 1900-1940, aerial photos from the archives of KMS (The National Survey and Cadastre Agency) covering the period 1945–1990s and data from the national Danish monitoring programme on marine vegetation covering the period 1989–2001.

Long-term changes in eelgrass distribution

In Denmark, records of eelgrass distribution date back to around 1900, and provide a unique opportunity to describe long-term changes. In 1900, eelgrass was widely distributed in Danish coastal waters, and covered approximately 6726 km^2 or 1/7 of all Danish marine waters (Petersen 1901, 1914) (Figure 2.32). The standing crop made up almost 1 kg dw m^{-2} in the densest stands and the total annual eelgrass production was estimated at 8 million tonnes dry weight (Petersen 1914). In the 1930s, the world wide wasting disease substantially reduced eelgrass populations, especially in north-west Denmark where salinity is highest (Blegvad 1935). In 1941, eelgrass covered only 7% of the formerly vegetated areas and occurred only in the southern, most brackish waters and in the low saline inner parts of Danish estuaries (Lund 1941, Rasmussen 1977)(Figure 2.32). No national monitoring existed between 1941 and 1989 but analyses of aerial photos from the period 1945–1990s, reveal an initial time lag of more than a decade before substantial recolonisation of the shallow eelgrass populations began. The photos also show that large populations had recovered in the 1960s (Frederiksen et al. subm.).

Today eelgrass again occurs along most Danish coasts (Figure 2.32) but has not reached the former area extension (Olesen 1993, Boström et al. in press). Comparisons of eelgrass area distribution in two large regions, Øresund and Limfjorden, in 1900 and in the 1990s, suggest that the present distribution area of eelgrass in Danish coastal waters constitutes approximately 20–25% of that in 1900. The area distribution of eelgrass in Limfjorden was estimated at 345 km^2 in 1900 (Ostenfeld 1908) and only at 84 km^2 in 1994 (based on aerial photography data from the Limfjord counties). In Øresund, eelgrass covered about 705 km^2 in 1900 (Ostenfeld 1908) and only about 146 km^2 in 1996–2000 (Krause-Jensen et al. 2001). Differences in methodology may, however, influence these comparisons.

The large reduction in area distribution of Danish eelgrass meadows is partly attributed to loss of deep populations. In 1900, colonisation depths averaged 5–6 m in estuaries, and 7–8 m in open waters while in the 1990s, colonisation depths were reduced about 50% to 2-3 m in estuaries and 4–5 m in open waters (Figure 2.33) (Boström et al. in press). The deep populations are most likely lost as a consequence of eutrophication. Increased nitrogen concentrations stimulate phytoplankton growth and thereby reduce the transparency of the water column and restrict the colonisation depth (Nielsen et al. 2002).

Recent inter-annual fluctuations in eelgrass distribution

Since 1989, the Danish Aquatic Monitoring and Assessment Programme has included annual surveys of colonisation depth and cover of eelgrass along depth gradients in a wide range of estuaries and coastal waters.

Photo: NERI/Peter Bondo Christensen

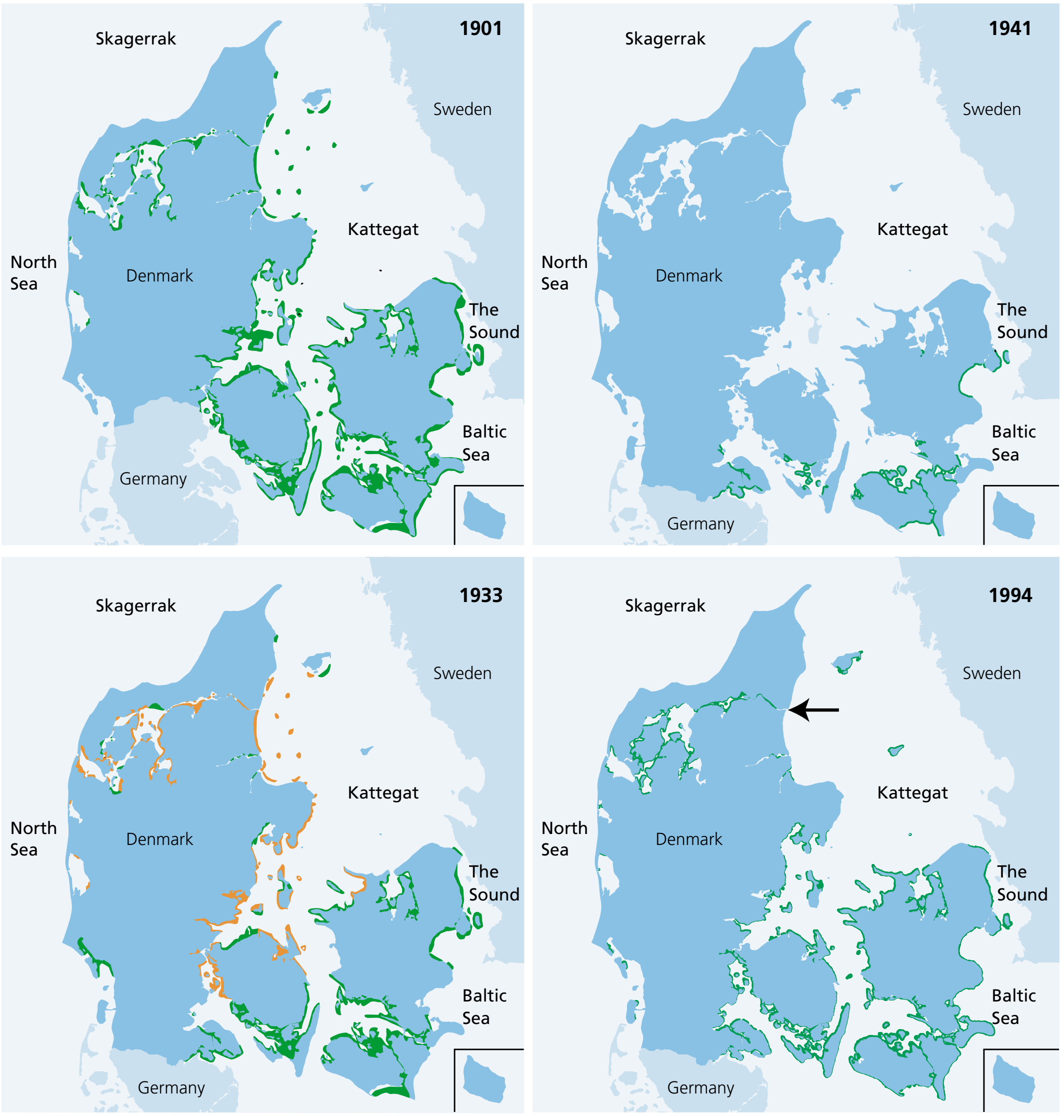


Figure 2.32

Map of eelgrass area distribution in Danish coastal waters in 1901 (redrawn after Petersen 1901), 1933 (redrawn after Blegvad 1935), 1941 (redrawn after Lund 1941) and 1994 (coarse map based on visual examination of aerial photos and data from the national Danish monitoring programme, produced by Jens Sund Laursen). Green areas indicate healthy eelgrass while orange areas (on the 1933 map) indicate where eelgrass was affected by the wasting disease but still present in 1933. The arrow shows the location of Limfjorden. (Boström et al. in press).

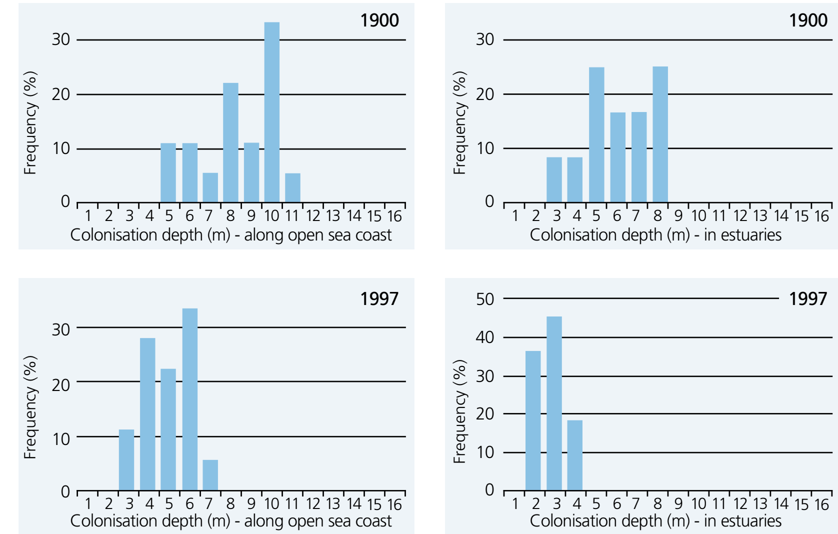


Figure 2.33

Maximum colonisation depth of Danish eelgrass patches along open coasts and in estuaries in 1900 and 1996/1997. Based on data from 18 sites along open coasts and 12 sites in estuaries investigated by Ostenfeld (1908) in 1900 and by the Danish Aquatic Monitoring and Assessment Programme in 1996/1997 (Boström et al. in press).

The colonisation depth of eelgrass reflects differences in water quality and physical setting along estuarine gradients. In the inner parts of estuaries, the annual mean colonisation depth ranged between 2.9 and 3.5 m, in outer parts between 3.3 and 4.3 m, and along open coasts between 4.7 and 5.8 m during the period 1989–2001 (Figure 2.34). Eelgrass colonisation depth shows no significant trend between 1989 and 2001 and does not reflect the sligh amelioration of water clarity observed through the same period (Ærtebjerg et al. 2002). Other regulating factors may blur the relationship between light and colonisation depth so that more marked changes in light climate are needed before colonisation depths increase. For example, eelgrass suddenly disappeared from several sites during the warm summers of 1992 and 1994 possibly due to combined exposure to anoxia, sulphide and extreme temperature (Goodman et al. 1995, Terrados et al. 1999).

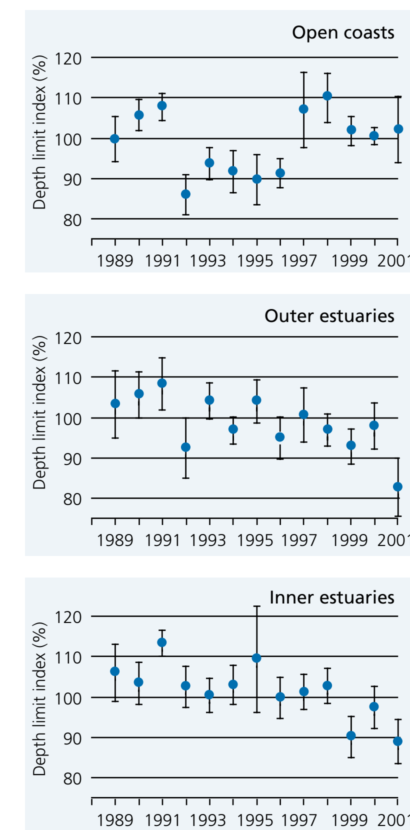
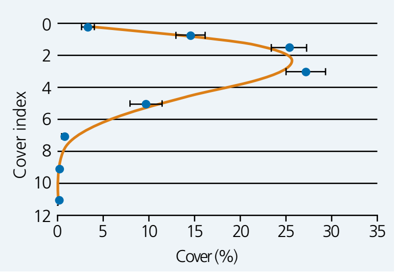


Figure 2.34

Index of maximal eelgrass colonisation depths (± s.e.) during the period 1989–2001 in inner estuaries (lower panel), outer estuaries (central panel) and along open coasts (upper panel). Index values represent colonisation depths of a given year relative to average colonisation depths of the period 1989–2001 (Ærtebjerg et al. 2002).

Figure 2.35
Eelgrass cover as a function of depth. Data represent average cover of 276 depth gradients monitored under the national Danish Aquatic Monitoring and Assessment Programme in 1994.



Eelgrass displays a bell-shaped distribution pattern along the depth gradient with maximum abundance at intermediate depth and lower abundance in shallow and deep water (Figure 2.35). Exposure, desiccation and ice-scour act to reduce eelgrass abundance in shallow water and render the populations extremely dynamic and unpredictable. In the deeper, more protected waters, reductions in eelgrass abundance towards the lower depth limit correlate with light attenuation (Sand-Jensen et al. 1997, Krause-Jensen et al. 2000) and are therefore more directly coupled to changes in eutrophication. The period 1989–2001 showed no significant trend in eelgrass cover at water depths above 2 m, but the cover of shallow populations was significantly reduced in inner estuaries and along open coasts (Henriksen et al. 2001). We have found no obvious explanation for this pattern.

In conclusion, eelgrass responds to several types of disturbances: changes in energy input (light), physical disturbances (e.g. wind, waves, extreme temperature, ice), chemical disturbances (e.g. anoxia, sulphide) and biological disturbances (e.g. the wasting disease). When eelgrass is used as a monitoring parameter to reflect changes in light climate due to eutrophication we should therefore be aware that other factors may affect the response. As the intensity of physical disturbances decline with depth, eelgrass colonisation depth and abundance from intermediate depths towards deeper waters are therefore likely to be better response parameters to eutrophication than the abundance of shallow populations.

Photo: NERI/Karsten Dahl

Short-lived nuisance macroalgae

Short-lived nuisance macroalgae are favoured by large supplies of nutrients (Pedersen 1995). Together with macroalgal species numbers and dominance patterns, the cover of nuisance species is therefore used as an indicator of the state of macroalgae in coastal areas in the Danish Aquatic Monitoring and Assessment Programme.

Information on these algae is relatively scarse and analyses of trends are therefore only possible for the inner part of estuaries at depth intervals of 0–1, 1-2 and 2–4 meter. During the monitoring period (1993–2001) average relative cover varied from 1–20% at 0–1 meter depth, 1–25 % at 1–2 meter depth and 0–20% at 2–4 meter depth. There were no significant changes during the period 1993 to 2001 (Kendalls-tau, p > 0,05) (see figure 2.36).

Macroalgae on reefs in open waters

As with seagrasses the macroalgae on the reefs in the open waters enhance biodiversity by providing habitat and shelter for many species. Furthermore they constitute a great part of the biodiversity in marine vegetation. The vegetation in the open waters consists of a multilayer of red and brown algal vegetation at water depths of 10–12 meters. At depths greater than 12–14 m total cover of upright algae decreases to a single layer with coverage less than 100% and cover decreases further with increased depth. In addition to upright forms, crust forming macroalgae cover stones and shells. The cover of crust forming algae is large, even at 24–25 meters depth.

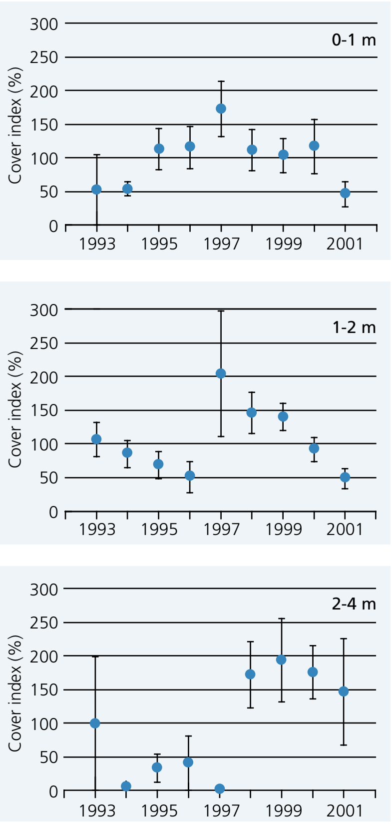


Figure 2.36
Cover index of loose lying nuisance macroalgal species in the inner parts of estuaries during the period 1993–2001. Data represent the depth intervals 0–1 m, 1–2 m and 2–4 m. The cover for each depth interval in a given year has been indexed relative to average cover of the depth interval during the period 1993–2001 (Ærtebjerg et al. 2002).

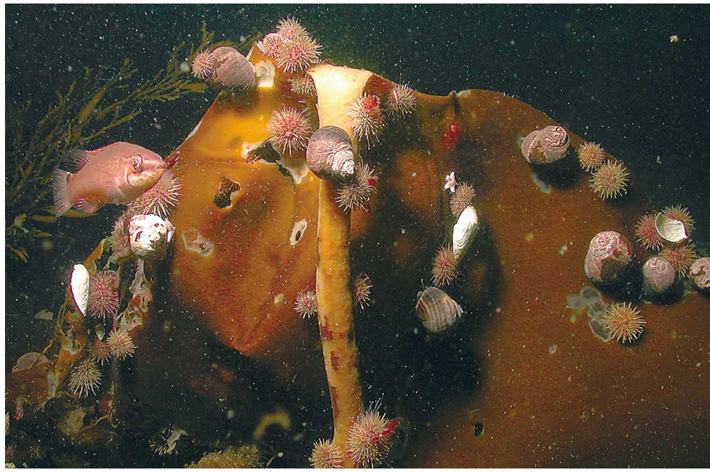


Photo: NERI/Karsten Dahl

Macroalgal vegetation is monitored at 9 stone-reefs and rocky bottom areas evenly distributed in the Kattegat, with one station located in the northern Belt Sea. The total relative cover of the upright forms is described along with specific cover of individual species including crust forming algae.

It has been shown that a good empirical correlation exists between the supply of the inorganic nutrients (nitrogen and phosphorus in freshwater inputs) and total cover of macroalgae at deep stations in the Kattegat during the period 1994 to 2000 (Henriksen et al. 2001). This means that high supplies of inorganic nutrients or freshwater leads to a reduced development of benthic vegetation. Exceptions are stations with an intense grazing pressure from the sea urchin *Strongylocentrotus droebachiensis*.

There is a corresponding significant correlation between the cover of benthic vegetation and the pelagic parameters, Secchi depth and chlorophyll concentration. This supports the hypothesis that the supply of nutrients influences light penetration to the bottom and thereby the environmental quality of the individual reef locations.

The total cover of the upright vegetation tended to increase in 2001 relative to the mean of the period 1994–2001 (Table 2.9). In general, algal coverage was low in years with relatively high runoff and high in years of low runoff. Accordingly there was no overall trend in the cover of macroalgae at the monitored reef stations (Figure 2.37).

Year	Month	Number of observations	Macroalgae coverage with reference to average coverage 1994-2001	P
1994	June	4	←→	–
	August	11	↓	–
1995	June	11	↑	–
	August	12	↓	–
1996	June	9	↓	–
	August	9	↑	*
1997	June	11	↑	***
	August	11	↑	***
1998	June	10	↓	*
	August	10	↓	**
1999	June	10	↓	*
	August	11	↓	***
2000	June	10	↓	–
	August	12	↓	–
2001	June	11	↑	–
	August	11	↑	–

Table 2.9

Macroalgal cover at a number of stonereefs in Kattegat relative to the average cover in the period 1994–2001.

↑ means that the majority of examined stations has a more developed vegetation cover.

↓ means that the majority of examined stations has a less developed vegetation cover.

←→ means that there is an equal number of stations with a less respectively more developed vegetation cover.

A signtest shows whether cover has increased or decreased relative to the mean for the period 1994-2001. * = P < 5%, ** = P < 1%, *** = P < 0,1%.

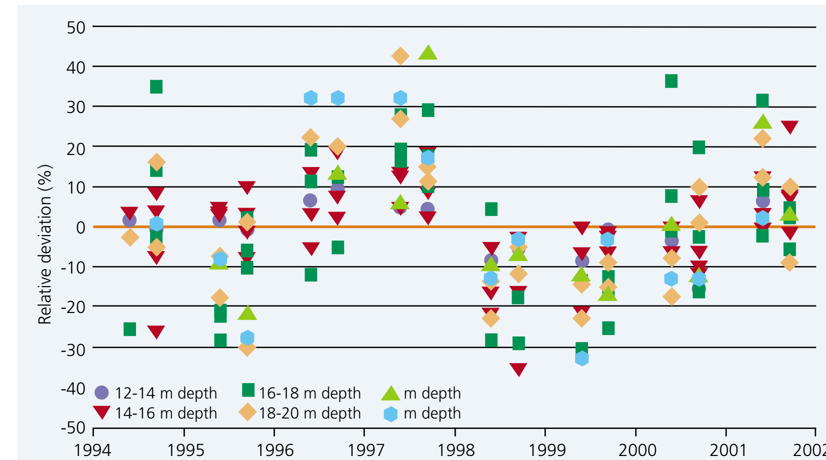


Figure 2.37
The relative deviation in macroalgal cover in relation to mean values for individual stations and time of investigation. Deviations are given in 2 meter depth intervals. Stations where algal abundance was limited by sea urchin grazing are left out.

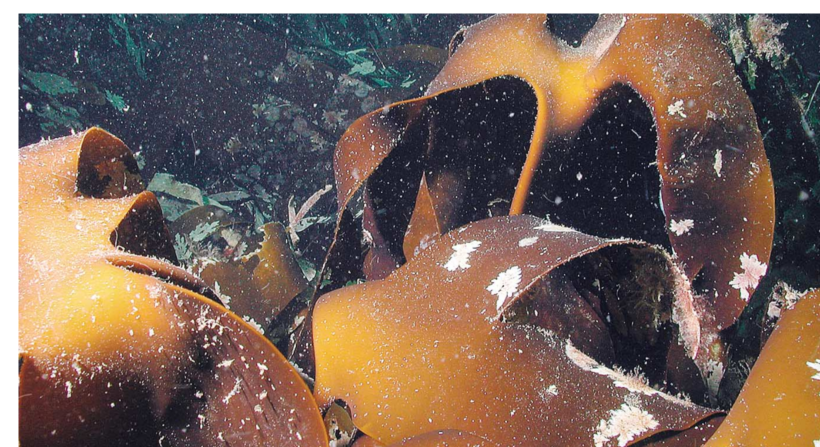


Photo: NERI/Karsten Dahl

2.10 Softbottom macrobenthos

Filter feeding bivalves are an important link between the pelagic and the benthic environment.

Photo: Biofoto/Michael Jensen

Benthic macrofauna in soft sediments plays an important role in the degradation of organic matter produced in the pelagic zone and several species serve as food for demersal fish. Since it is composed of many different species (in the Kattegat there are more than 500 species), benthic macrofauna furthermore constitute a great part of the biodiversity in the water surrounding Denmark. The macrofauna integrates environmental changes in the benthic environment and is a highly cost effective parameter to sample. Moreover, since this fauna often is limited by food, such as sedimenting phytoplankton biomass, macrozoobenthos is a useful parameter to describe changes in eutrophication.

Benthic macrofauna has been included in the Danish Monitoring Programme since 1979 and is measured with replicated quantitative sampling at fixed stations. Number of individuals and biomass (wet weight or dry weight) are measured for each species. Sampling occurs once a year typically in May/June. The number of fixed stations regularly visited at present include 24 stations in open sea areas mainly in the Kattegat and station grids in 22 estuaries and near coastal areas around Denmark.

Open sea areas

General trends in macrozoobenthos abundance in the Sound, the Kattegat and the Belt Sea follow a bimodal pattern over the last 20 years with peaks in the beginning of the 1980s and in the middle of the 1990s (Figure 2.38). While the first peak was dominated by both polychaetes and crustaceans, the peak in the 1990s was only dominated by polychaetes. Within the taxonomic groups different species dominated in different peaks and at different stations. Biomass did not show the same clear pattern as abundance. Biomass and total abundance showed the lowest values in the period 1998–2001 since the measurements started.

Over the two last decades changes in total abundance and biomass of macrobenthos are below the halocline in Danish open sea areas. Correlation analyses performed between the biological variables and two environmental variables related to climate, the North Atlantic Oscillation index (NAO-index) and runoff of freshwater from Denmark, showed significant positive correlation with a 1 or 2 years time lag, indicating influence of climate on benthic variations (Henriksen et al. 2001). In particular winter nutrient input, and the spring phytoplankton bloom, most likely influences the abundance of benthic macrofauna.

However, correcting for the linear effects of runoff, the general pattern in abundance is still there, suggesting the importance of other factors hitherto not identified. Some evidence indicates that reduced nitrogen nutrient concentrations and possibly reduced diatom abundance, when corrected for runoff, may have reinforced the decrease in benthos stocks in recent years.

Coastal areas

From 1998 the monitoring strategy for estuaries and coastal areas was changed from measurements on single stations to measurements on grids of stations in each area. Benthic fauna data sampled with consistent methods are now available from 25 different estuarine and bay areas from the period 1998 to 2001. Unlike the open Kattegat and the Belt Sea, where faunal changes in total abundance and biomass to a large degree are synchronised between stations, changes in coastal areas do not seem to be synchronised to the same degree. This is exemplified in Table 2.10 where total number of species in the 4 years are shown for each coastal area. The number of areas with increasing

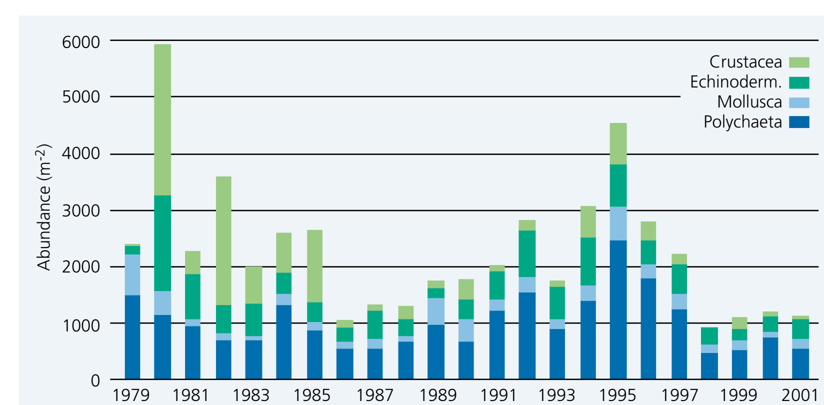


Figure 2.38
Temporal development of abundance at 3 stations of 17-55 m water depth, in the Kattegat-Sound-Belt Sea area partitioned on major taxonomic groups. Bristleworms (Polychaeta), Molluscs (Mollusca), Echinoderms (Echinodermata) and Crustaceans (Crustacea). Remaining taxa had a negligable part of the total abundance. Note the dramatic decrease in numbers of crustaceans over the 2 decades.

species numbers are about the same as the decreasing ones. Reasons for lack of synchrony between coastal areas is probably local factors like oxygen deficiency. For example, in Mariager Fjord fauna was more or less exterminated by oxygen deficiency in 1997 and thereafter a succession of species have taken place reflected by increasing number of species. In conclusion, there are no general synchronous changes in abundance, biomass or number of species in coastal areas over the last 4 years.

Kills of macrozoobenthos

The increase in oxygen consumption due to enhanced loads of organic matter to the bottom can lead to oxygen depletion and death of benthic fauna. Though these events are not part of the monitoring programme, many of the events are still recorded (see Table 2.11).

There are examples from Danish marine areas, like the southern Little Belt, where severe acute oxygen depletion and probably kills of macrobenthos have been an almost regular yearly event since the beginning of the last century. However, the area in the southern Little Belt where macrobenthos suffer from oxygen depletion is in recent years five times larger than 100 years ago (Marin ID 1988). Large scale oxygen depletion events with kills of macrobenthos belong to the present. A well known example is from 1986 when fishermen caught dead Norwegian lobsters (*Nephros norvegicus*) in the Kattegat. It is also likely that 2002 will be well-known for the extended oxygen depletion and resulting kills of macrobenthos.

Coastal area	m^2	1998	1999	2000	2001
Roskilde Bredning	0.51	29	29	23	30
Horsens Fjord	0.61	–	24	24	40
Vejle Fjord	10.07	–	52	36	55
Kolding Fjord	5.54	–	57	36	61
Ringkøbing Fjord	7.15	20	22	22	17
Nissum Fjord	6.44	30	29	33	28
Hevring Bugt	5.54	76	69	87	91
Øresund	6.44	68	51	54	52
Køge Bugt, central part	6.44	26	30	28	35
Odense Fjord	6.45	66	78	57	57
Ringgårdbassin	5.81	27	28	25	19
Roskilde Fjord, northern part	3.58	33	30	31	37
Isefjord	3.15	24	36	19	15
Kattegat	2.86	63	51	61	66
Lillebælt	5.54	35	50	51	27
Karrebæksminde Bugt	6.57	40	28	37	31
Skive Fjord	6.44	31	36	24	41
Nissum Bredning	6.44	33	31	49	34
Løgstør Bredning	6.44	42	34	32	35
Wadden Sea, northern part	26.66	43	43	41	47
Århus Bugt	6.03	62	46	54	57
Mariager Fjord	5.54	17	14	28	26
Flensborg Fjord	5.41	66	30	30	17
Wadden Sea, southern part	1.98	41	36	43	40
Nivå Bugt	3.58	–	62	65	57
Average species number		41.524	39.84	39.6	40.6

Table 2.10
Changes in macrofauna species numbers 1998-2001 in 25 coastal areas in Denmark. The first coloumn denotes total sampling area, red denotes a decrease compared to previous sampling, green denotes an increase and blue denotes no difference.

Year	Place	Cause
1981	North Sea, Hevring Bugt, Århus Bugt, Pæregård Strand, Dalby Bugt, Kalø Vig	Oxygen depletion
1983	Lillebælt, Kieler Bugt	Oxygen depletion
1985	Kattegat	Oxygen depletion
1986	Kattegat, Kieler Bugt	Oxygen depletion
1987	Sejrø Bugt, North coast of Sjælland, southern Lillebælt	Oxygen depletion
1988	Skagerrak, Kattegat	Harmful algal bloom
1988	Southern Kattegat, Limfjord, Århus Bugt,	Oxygen depletion
1989	Horsens Fjord	Oxygen depletion
1990	Århus Bugt, Northern Lillebælt	Oxygen depletion
1991	Haderslev Fjord	Oxygen depletion
1992	Haderslev Fjord	Oxygen depletion
1993	Vejle Fjord, Southern Lillebælt	Oxygen depletion
1994	Roskilde Fjord, Limfjord, Vejle Fjord, Nørrefjord, Flensborg Fjord, Sydfynske Øhav, Isefjord, Århus Bugt, Åbenrå Fjord, Genner Fjord, Flensborg Fjord, Lillebælt	Oxygen depletion
1995	Åbenrå Fjord, Roskilde Fjord	Oxygen depletion
1996	Limfjorden, Vejle Fjord, Lillebælt	Oxygen depletion
1997	Mariager Fjord, Limfjorden, Nakkebølle Fjord, South and east of Møn, Isefjord	Oxygen depletion
1998	Southern Lillebælt, Åbenrå Fjord Flensborg Fjord	Oxygen depletion
1999	Limfjord, Kalø Vig, Knebel Vig, Århus Bugt, Horsens Fjord, Vejle Fjord, Flensborg Fjord, sydlige Bælthav, Sydfynske Øhav	Oxygen depletion
2000	Langelandssund, Ebeltoft Vig, Århus Bugt, Odense Fjord, Limfjorden	Oxygen depletion
2001	Limfjorden, Århus Bugt	Oxygen depletion
2002	Limfjorden, Århus Bugt, Hevring Bugt, Mariager Fjord, Vejle Fjord, Ålborg Bugt, Kalø Vig, Åbenrå Fjord, Flensborg Fjord, coastal areas north and south of Fyn, Lillebælt	Oxygen depletion

Table 2.11
Recorded macrozoobenthos kills and their causes in Danish marine waters 1981–2002.

2.11 Fish kills in coastal waters

Fish kills are not the most common effect of eutrophication but illustrate severe effects of oxygen deficiency and harmful algal blooms.

Photo: Bent Lauge Madsen

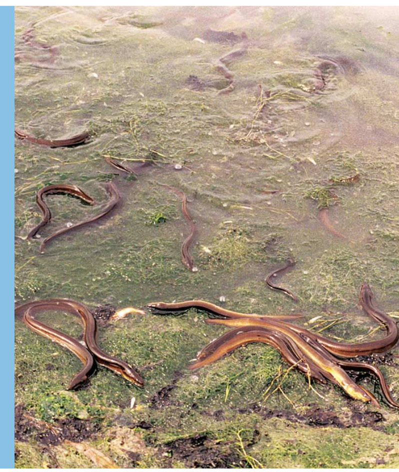


Photo: Århus County/Helene Munk Sørensen

Normally two types of events related to eutrophication can cause fish kills. It is either toxic algae or low oxygen concentrations. Low oxygen concentrations in the bottom waters result in the release of hydrogen sulphide (H_2S) from the sediments. The H_2S is lethal to most animals and the result of the sudden release is often extensive and leads to the immediate death of animals living at or near the sea floor as well as in the water column. This immediate effect, therefore, also includes fish. However, the most spectacular fish kills with dead or dying fish accumulating on the shores, as an example in 1981 and 2002, are due to upwelling of oxygen poor or H_2S containing bottom water to the surface along the coasts due to changing wind directions and forces, trapping the fish in these water masses with no chance to escape.

The Danish EPA published the first comprehensive national assessment of the effects of oxygen depletion and harmful algal blooms in 1984 (Danish EPA 1984) This report assessed fish kills in the Danish waters and concluded that fish kills were restricted, and were exceptional, but natural events, before the 1970s, and that the problem increased in the 1970- and early 1980s.

The four best known Danish examples of fish kills are from 1981, 1988, 1997 and 2002. In 1981 the kills were caused by severe oxygen depletion in most of the Danish waters (Danish EPA 1984). Fish kills in May and June 1988 were due to a large bloom of the flagellate *Chrysochromullina polylepis* (Lancelot et al. 1989). In 1997, almost all fish were killed in the inner part of Mariager Fjord due to oxygen depletion and release of H_2S in the entire water column. In October 2002 fish kills occurred several places along the Jutland east coast due to upwelling of oxygen poor and H_2S containing bottom water.

The ongoing national monitoring programme does not include fish monitoring. However, fish kills have been reported annually since 1989 as an integral part of the National Aquatic Monitoring and Assessment Program. Table 2.12 summarises the recorded

Year	Place	Cause
1981	Hevring Bugt, Århus Bugt, Pæregård Strand, Dalby Bugt, Båring Vig, Vejle Fjord, Limfjorden, including Thisted Bredning, Visby Bredning, Bjørnsholm Bugt, Risgårde Bredning, Skive Fjord and Lovns Bredning	Oxygen depletion
1981	Fjaltring Strand, Glyngøre Harbour	Harmful algal bloom
1982	North Sea, Køge Bugt, Holckenhavn, Kalø Vig, Isefjord	Oxygen depletion
1983	Lillebælt, Als Sund, Langelandssund	Harmful algal bloom
1987	Sejrø Bugt	Oxygen depletion
1988	North coast of Sjælland, Sejrø Bugt, Ringkøbing Fjord	Oxygen depletion
1988	North Sea, Skagerrak, Kattegat, and the northern part of Øresund	Harmful algal bloom
1989	Ringkøbing Fjord	Oxygen depletion
1990	Århus Bugt	Oxygen depletion
1990	Djurslands Østkyst	Oxygen depletion or harmful algal bloom
1991	Lillebælt	
1992	Lillebælt	Possible harmful algal bloom
1994	Årgab, Hovvig, Mariager Fjord, Vejle Fjord, Isefjord, Roskilde Fjord, Thisted Bredning, Skive Fjord, Lovns Bredning	Oxygen depletion
1995	Årgab, Hanstholm	Oxygen depletion
1997	Mariager Fjord, Vejle Fjord, Sydfynske Øhav, Korsør Nor, Isefjord	Oxygen depletion
1998	North Sea and Skagerrak	Harmful algal bloom
1999	Vejle Fjord	Oxygen depletion
2000	Kattegat, Smålandsfarvandet, Ebeltoft Vig, Sydfynske Øhav	Oxygen depletion
2001	Vejle Fjord	Oxygen depletion
2002	Ålborg Bugt, Hevring Bugt, Vejle Fjord, Kalø Vig, Øresund, Sydfynske Øhav	Oxygen depletion

Table 2.12 Recorded fish kills in Danish marine waters 1981–2002

Photo: NERI/Peter Bondo Christensen

fish kills in Danish marine waters. It might be argued that the situation has not improved since the mid 1980s, mainly because inputs of nutrients are still high, especially in wet years, and that the events of oxygen depletion that might trigger fish kills are tightly coupled to meteorological and hydrographic forcing.

Regional authorities have carried out studies on fish populations and fish kills in Limfjorden, Mariager Fjord, Ringkjøbing Fjord, Roskilde Fjord, the area around Rødsand, the Wadden Sea and Århus Bugt. Studies in Århus Bugt, which covers the period 1953–1998 and includes coherent data on fish, fisheries, zoobenthos and water quality, are summarised by Jensen (1999):

- The occurrence and composition of fish have changed significantly during the last 50 years, due to changes in fishing intensity and duration of oxygen depletion.
- Fishing intensity has had an impact on plaice (*Pleuronectes platessa*), which has decreased from the 1950s to the 1990s.
- Conversely, the dab (*Limanda limanda*) has become more frequent and is now the dominating flat fish in the bay.
- The mean length of dab has decreased since 1957 and is unambiguously linked to the duration of oxygen depletion. The growth is limited in years with longer periods of oxygen depletion compared to the situation in the 1950s. Less that 1% of dabs reach the minimum size limit of 25 cm.

Future possibilities to assess the health of coastal fish populations will improve from 2004, when monitoring of non-commercial fish in six representative coastal waters will be conducted on a regular basis within the monitoring and assessment programme.

Photo: NERI/Gunni Ærtebjerg