Limnol. Oceanogr., 51(1, part 2), 2006, 398–408

Coastal eutrophication and trend reversal: A Danish case study

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Abstract

In the past 2 decades significant measures have been taken to reduce nitrogen and phosphorus discharges from Denmark by 50% and 80%, respectively. These nutrient reduction targets now appear within reach after several consecutive reduction measures are fully implemented, particularly toward diffuse discharges, and reduced nutrient concentrations are beginning to be observed in estuaries and the Danish straits. Phosphorus concentrations have declined by 22% to 57% from the early 1990s, mainly owing to improved treatment of urban and industrial wastewater. Changes in nitrogen concentrations, following reduction measures toward diffuse sources, were more recent and partly masked by large interannual variations in freshwater discharge. The response in marine nitrogen concentrations was delayed relative to the decline in riverine concentrations, most likely owing to large internal loading from the sediments. Two consecutive dry years appeared to be the triggering mechanism for nitrogen concentrations to decline. In the last 5 yr, nitrogen levels in estuaries and coastal waters have decreased up to 44% when interannual variations in freshwater discharge were accounted for. These first signs of environmental recovery were most pronounced in estuaries and coastal waters but also were apparent in open waters of the Kattegat, the Sound, and the Belt Sea. This case study is the first to document significant decreases in nutrient concentrations on a large regional scale resulting from an active management strategy to reduce nutrients from both diffuse and point sources.

Eutrophication of coastal ecosystems from nutrient over-enrichment is widespread (Nixon 1995), with the effects manifested in a myriad of direct and indirect responses (Cloern 2001). Although the sources and pathways of nutrient inputs to aquatic ecosystems can be estimated with reasonable certainty, it has been difficult to achieve reductions in the different sources (Boesch 2002). However, some coastal ecosystems have experienced reductions in inputs of phosphorus and nitrogen primarily through improvement in treatment of wastewater and reductions in point sources (Butt and Brown 2000; Conley et al. 2000), although relatively little progress has been made in reducing diffuse sources of nutrients (Butt and Brown 2000; Boesch 2002). In eastern Europe a number of studies have shown decreasing nutrient concentrations in rivers and streams from reduced application of fertilizers after the economic collapse in eastern Europe and the Soviet Union in the early 1990s (for overview, see Stålnacke et al. 2003).

In 1987 a National Action Plan on the Aquatic Environment was enacted in Denmark to reduce nutrient inputs to the aquatic environment. This action plan was based on an agenda adopted in 1986 that aimed to reduce nitrogen and phosphorus discharges by 50% and 80%, respectively (Kronvang et al. 1993). The commitment to nutrient reductions was also made in multijurisdictional agreements with both Helsinki Commission for the Protection of the Marine Environment of the Baltic Sea Area (HELCOM) and Oslo-Paris Commission for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) (Conley et al. 2002*b*). This first action plan was most effective toward nutrient reductions from municipal wastewater, and it was soon recognized that new actions had to be taken toward the diffuse loading, in particular nitrogen. Another action plan for sustainable agricultural production followed in 1991 and a second National Action Plan on the Aquatic Environment in

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Acknowledgments
The present work is a contribution of the CHARM (EVK3-CT-2001-00065) and REBECCA (SSPI-CT-2003-502158) projects funded by the European Commission. We gratefully acknowledge the Danish counties responsible for data collection under the Danish Nationwide Aquatic Monitoring and Assessment Program and the Swedish Hydrological and Meteorological Institute for providing data from the Swedish monitoring programs. German loading data were provided by Heike Herata, Federal Environmental Agency in Berlin, and Thorkild Petenati from the federal state of Schleswig-Holstein. We thank Ole Hertel for providing atmospheric deposition data and Bo Riemann for comments on the manuscript. The manuscript was improved by valuable comments made by three anonymous reviewers.

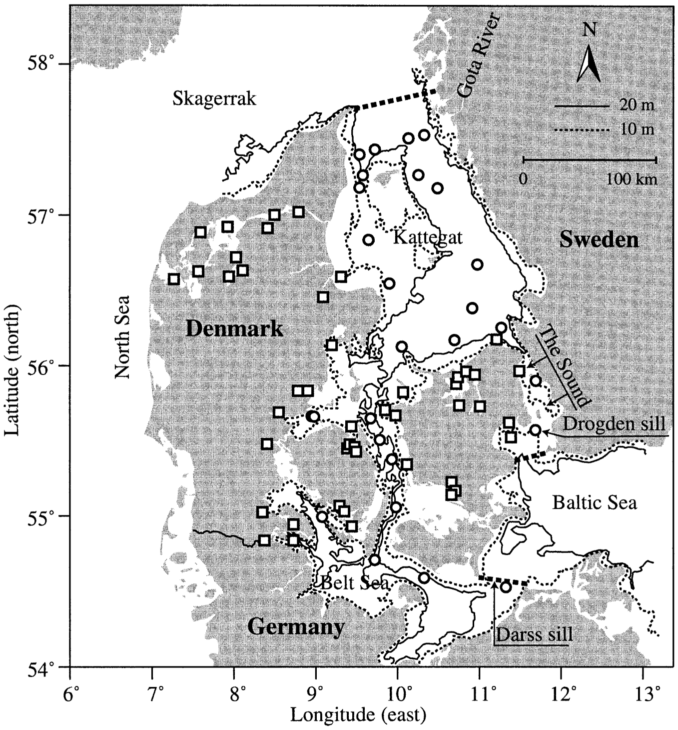

Fig. 1. Map of the Kattegat, the Sound, and the Belt Sea showing location of monitoring stations used in the study, partitioned into estuarine and coastal stations (squares) and open-water stations (circles). Boundaries of the study area are marked by dashed lines.

1998. The two latter action plans included a variety of strategies and measures to reduce diffuse nitrogen inputs, including fertilizer reductions, buffer strips, and restoration of wetlands (for further details, see Conley et al. 2002*b*).

The Danish National Aquatic Monitoring and Assessment Program (DNAMAP) was established in 1988 to monitor nutrient loading and ecological responses to the nutrient reduction targets. DNAMAP was organized with the aim of obtaining information on a wide range of eutrophication-related variables (e.g., nutrients, chlorophyll *a*, macrophytes, benthic macrofauna) covering many estuaries and coastal zones in Denmark. Monitoring in the open waters and selected coastal waters is a requirement of HELCOM and OSPAR, with DNAMAP and the Swedish national monitoring program operating in a coordinated joint effort.

The objective of this article is to demonstrate that the measures that have been taken in Denmark to reduce the load of nutrients from both point and nonpoint sources are successful. We have analyzed the trends in nutrient loading and concentrations after the first Action Plan on the Aquatic Environment and the establishment of DNAMAP in 1989 up to the most recent data from 2002, a period of 14 yr with large changes in nutrient loading that has allowed us to identify responses in the ecosystem to nutrient management. This analysis was possible owing to the extensive data set collected under DNAMAP, and our report is the first documentation of significant effects in the marine ecosystem that can be traced to nutrient reductions resulting from an active management strategy.

Study area—The Kattegat, the Sound, and the Belt Sea (the Danish straits) comprise a shallow transition area between the brackish Baltic Sea and the more saline Skagerrak/North Sea (Fig. 1). It is a coastal ecosystem with estuarine character dominated by advective transports and an almost permanent halocline located at ~15-m depth (Andersson and Rydberg 1993; Jakobsen and Trébuchet 2000). Transport in the surface layer is generally northward, whereas Skagerrak water penetrates southward into the Danish straits as a bottom current that gradually mixes with the surface layer. The residual flow from the Baltic Sea is ~500 $km^3\ yr^{-1}$ (Stigebrandt and Gustafsson 2003), and bottom water exchanges with the Baltic Sea occur over the sills at Drogden (8-m depth) and Darss (15-m depth). The area and volume of the Danish straits are 41,000 km^2 and 810 km^3, respectively (Gustafsson 2000).

A total land area of 64,135 km^2 discharges directly into the Danish straits with 47%, 37%, and 16% belonging to Denmark, Sweden, and Germany, respectively (Table 1). The Göta River (Fig. 1), which is the sixth largest river in the entire Baltic Sea area, is not included in the figures for the Kattegat, because discharges from this river mainly are carried northward out of the Kattegat into the Skagerrak. The Danish straits receive discharges from 70% of Denmark. Approximately 9 million people inhabit the catchment, the majority of these living in urban settlements. Land use is generally dominated by agriculture (Table 1), except for the Swedish catchment discharging into the Kattegat with 72% forest.

Local inputs of freshwater and nutrients are primarily discharged through productive estuaries and coastal regions (Carstensen et al. 2003). The Danish estuaries are for the

Table 1. Catchment area and land use for the Kattegat, the Sound, and the Belt Sea. Areas are from HELCOM (2002), and land uses were compiled from the GRID-Arendal database (Sweitzer et al. 1996).

	Denmark		Sweden		Germany	
	Catchment area (km^2)	Arable and pasture (%)	Catchment area (km^2)	Arable and pasture (%)	Catchment area (km^2)	Arable and pasture (%)
Kattegat	15,850	63	20,920*	18	—	—
The Sound	1,740	47	2,885	64	—	—
Belt Sea	12,340	68	—	—	10,400†	65
Total	29,930	64	23,805	23	10,400	65

* Excluding the catchment of the Göta River (50,233 km^2).
† The catchment area for the German federal state Schleswig-Holstein is 5,450 km^2.

most part shallow (<3 m deep) with relatively short residence times (Conley et al. 2000). The majority of estuaries have a well-mixed water column with intermittent periods of stratification during periods of calm winds or inflow of saline bottom water. Agricultural production in Denmark is very specialized and highly and consistently productive both per unit land and per unit resource (Porter and Petersen 1997). Over the past decade, there has been an increase in animal husbandry, which together with the measures in the action plans has precipitated a shift from chemical fertilizers to manure for crop production. Denmark is now the world's largest exporter of pork meat, with a standing stock of 13 million pigs in addition to 1.7 million cattle (2003 data from www.ddl.dk). For comparison, the human population of Denmark is 5.4 million.

Materials and methods

Detailed load compilations have been carried out in Denmark since 1989 as part of the first Action Plan for the Aquatic Environment. Data on the freshwater discharges from the Swedish catchment were obtained from the Swedish Meteorological and Hydrological Institute (SMHI), and nutrient loading figures were compiled from Stålnacke et al. (1999) and data from the Swedish University of Agricultural Sciences (www.slu.se). A long time series was available for nutrient and freshwater inputs from the German federal state of Schleswig-Holstein (1977–2002), whereas total inputs from Germany to the Belt Sea were available from 1994 onward (data source, Federal Environmental Agency, Berlin, Germany). We calculated the average ratio between total German input and that from Schleswig-Holstein for 1994–2002 and used this value for scaling up the inputs from Schleswig-Holstein during 1989–1993. Danish nutrient loading was partitioned into riverine and point source contributions, the latter combined of discharges directly to marine waters and discharges to freshwater carried with the riverine input. The diffuse nutrient loading was calculated as riverine input minus the point source input to freshwater. In this calculation we assumed that the retention of nutrients from point sources to freshwater was negligible, because freshwater point sources generally discharge in the downstream area. It should be recognized that certainty in the loading compilations is likely to have increased with time. The ratio between the diffuse nutrient loading and the freshwater discharge will hereafter be referred to as flow-weighted concentrations of TN and TP.

Atmospheric nitrogen deposition was calculated by means of a Lagrangian model with 96-h trajectories of air parcels to a net of receptor points having a resolution of 25 × 25 km (Hertel et al. 1995). The model was calibrated to deposition rates measured at two coastal gauges located in the northern and southern part of the study area. Atmospheric deposition of phosphorus has not been calculated on an annual basis, but Andersen et al. (2004) estimated it to be ~8 kg P km^{-2} for the study area and contended that temporal trends were unlikely. Based on these results a constant atmospheric phosphorus input of 328 × 10^3 kg yr^{-1} was assumed in this study.

Measurements of nutrient concentrations (NH_4^+, NO_2^-, NO_3^-, PO_4^{3-}, TN, and TP), collected within the framework of DNAMAP and the Swedish national monitoring program, were investigated in the present study. A total of 46 stations located in estuaries and the coastal region and 27 open-water stations (Fig. 1) were sampled with varying frequencies from one up to 103 times per year, however unevenly distributed both within and between years. Dissolved inorganic nitrogen (DIN) was calculated as the sum of ammonia, nitrite, and nitrate, whereas dissolved inorganic phosphorus (DIP) comprised phosphate only. For estuarine and coastal stations, we calculated the average concentration of all nutrient constituents over the entire water column, whereas average concentrations of samples ≤10 m and samples ≥20 m were used to characterize the surface and the bottom layer at open-water stations.

Yearly means of DIN, DIP, TN, and TP were computed through use of a general linear model that described variations between stations, years, and months after log-transformation of the variables. Thus, if Y_{ijkl} described the observations of any of the four considered variables, then

$$\begin{aligned} Y_{ijkl} &= station_i \times year_j \times month_k \times \varepsilon_{ijkl} \\ &\Updownarrow \\ \log(Y_{ijkl}) &= \log(station_i) + \log(year_j) + \log(month_k) \\ &\quad + e_{ijkl} \end{aligned} \tag{1}$$

where e_{ijkl} was a normal distributed random error with zero mean and variance σ^2, $station_i$ described the station-specific mean levels, $year_j$ described the year-specific mean level, and $month_k$ described the seasonal variation by monthly means. It was assumed that interannual and seasonal variations were multiplicative factors to each other and to the station-specific mean level, and that the error term on the original scale was lognormaly distributed.

The monitoring data were not balanced, i.e., uneven number of observations for different combinations of station, year, and month, and averaging all observations for a given year would result in values that depended on the differences in sampling frequencies. Comparable yearly means were calculated by computing the marginal distributions of $year_j$ as linear combinations of the parameter estimates in Eq. 1 (Searle et al. 1980) to account for differences in sampling frequencies. Yearly means obtained from Eq. 1 were back-transformed to the original scale by

$$E(year_j) = \exp\left\{E[\log(year_j)] + \frac{\sigma^2}{2}\right\}$$

In the following we shall refer to the back-transform of the marginal means computed from the model in Eq. 1 as the yearly nutrient means or levels.

Nitrogen and phosphorus concentrations were related to freshwater discharge, and point source nitrogen and phosphorus loading, respectively, by means of multiple linear regression models using yearly means from 1989–1997. The last 5 yr (1998–2002) were used to investigate deviations from those established relationships that could potentially accrue from changes in agricultural practices. Mean nutrient concentrations based on estuarine and coastal stations were

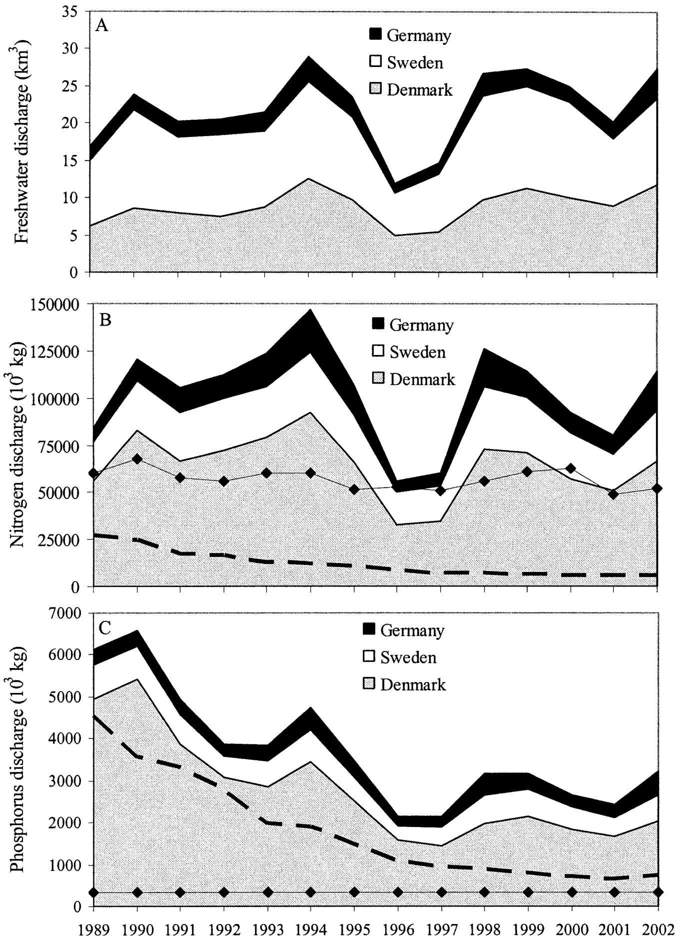

Fig. 2. Annual discharges of (A) freshwater, (B) total nitrogen, and (C) total phosphorus to the Danish straits (1989–2002). Atmospheric nitrogen and phosphorus depositions to the study area are shown by diamonds connected with a thin line. Danish nitrogen and phosphorus input has been partitioned into point sources (below dashed line) and diffuse sources (above dashed line). Inputs from the Göta River were not included in the Swedish figures.

related to input from Denmark only. In contrast, a combined freshwater discharge from Denmark, Sweden, and Germany was used to relate to concentrations found at open-water stations. We used point source loading from Denmark alone for the open-water stations, because annual figures of nutrient loading partitioned into point and diffuse sources were not available from Sweden and Germany.

Finally, the mean nutrient levels were adjusted to variations in freshwater discharge and in point source loading (denoted DIN_{ADJ}, TN_{ADJ}, DIP_{ADJ}, and TP_{ADJ}) by means of the multiple regressions models described above. Differences between the yearly means and predicted values from the multiple regressions were added to the predicted mean levels for DIN, TN, DIP, and TP corresponding to the average freshwater discharge over the entire period (1989–2002) and the average point source loading for the five most recent years (1998–2002). The adjusted nutrient levels would indicate changes in the diffuse inputs when random variations in freshwater discharge were taken into account. These adjusted means described the nutrient level in a given year if the loading from point sources had been low and if the freshwater discharge had an average level. Nutrient inputs, nutrient levels, and adjusted nutrient levels were analyzed for trends by means of linear regression.

Results

Nutrient loading—There were strong interannual variations in freshwater discharge as well as nutrient loading (Fig. 2). The total freshwater discharge varied from 12–29 km^3 yr^{-1}, and interannual variations in the Danish, Swedish, and German values were highly correlated ($r_{DK,SE} = 0.82$; $r_{DK,GE} = 0.86$; $r_{SE,GE} = 0.64$). Particularly, 1989, 1996, and 1997 were "dry" years, whereas the other years had freshwater discharges $>$20 km^3 yr^{-1}. During the entire period the freshwater discharge from Sweden was the largest (49%), followed by discharge from Denmark (40%) and Germany (11%). However, the loading from Denmark was clearly the largest for both total nitrogen (62%) and total phosphorus (74%). Interannual variation in the Danish, Swedish, and German contributions were highly correlated for nitrogen loading ($r_{DK,SE} = 0.87$; $r_{DK,GE} = 0.79$; $r_{SE,GE} = 0.86$) and less correlated for phosphorus loading ($r_{DK,SE} = 0.77$; $r_{DK,GE} = 0.22$; $r_{SE,GE} = 0.68$). All freshwater and nutrient discharges from the three countries were significantly correlated ($p < 0.05$), except $r_{DK,GE}$ for phosphorus loading ($p = 0.4552$).

Interannual variations in total nitrogen loading from land did not reflect any trends (Table 2) and were clearly linked to freshwater discharge (Fig. 2). Over the entire study period, the input from Danish point sources declined significantly, comprising ~50% in the dry year of 1989 to <10% of the total Danish contribution in the most recent years. This corresponded to a reduction of ~20,000 × 10^3 kg of nitrogen. The total phosphorus loading decreased significantly from

Table 2. Trend analysis of nutrient inputs to the Danish straits (1989–2002, $n = 14$) by linear regression ($F_{1,12}$). Significant trends (10^3 kg yr^{-1}) at the 5% significance level are highlighted by boldface type.

	Denmark		Sweden		Germany	
Source	Trend	*p*	Trend	*p*	Trend	*p*
TN diffuse sources	354	0.7558	—	—	—	—
TN point sources	**−1537**	**<0.0001**	—	—	—	—
TN total input	−1184	0.3088	−12	0.9718	193	0.5919
TP diffuse sources	27	0.3933	—	—	—	—
TP point sources	**−281**	**<0.0001**	—	—	—	—
TP total input	**−253**	**0.0001**	−16	0.0761	2	0.7048

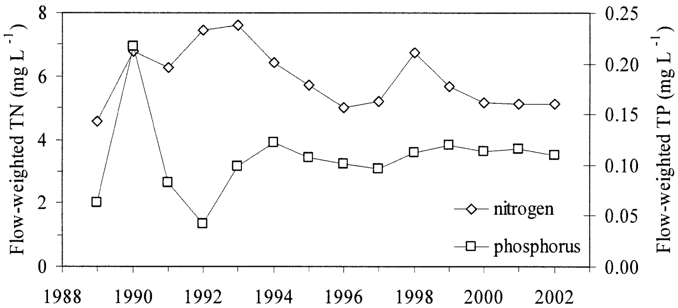

Fig. 3. The ratio between the diffuse nutrient loading and the freshwater discharge from Denmark, referred to as flow-weighted TN and TP concentrations in the text.

~6000 × 10^3 kg yr^{-1} in the beginning of the period to ~3000 × 10^3 kg yr^{-1} after 1994. This trend was attributed to reductions in Danish point sources from ~4500 × 10^3 kg yr^{-1} in 1989 to 600–800 × 10^3 kg yr^{-1} in recent years. Point source reductions have made the diffuse input of phosphorus the dominating source in Denmark from 41% in 1989 to ~80% in recent years. This change in dominating inputs from point to diffuse sources has also resulted in a gradual covariation of phosphorus loading with the freshwater discharge. There were no significant correlations between the freshwater discharges and point source loading data used in the multiple regression analysis below (*t*-test, all $p > 0.05$).

Atmospheric deposition of nitrogen was relatively stable (average of 55,000 × 10^3 kg yr^{-1}) over the entire study period, ranging from 47,000 × 10^3 kg yr^{-1} to 65,000 × 10^3 kg yr^{-1} with actually only 1 yr (1990) exceeding 60,000 × 10^3 kg yr^{-1} (Fig. 2). There was no trend in atmospheric deposition ($F_{1,12} = 3.66$, $p = 0.0799$), particularly if 1990 was excluded in the linear regression ($F_{1,11} = 1.55$, $p = 0.2394$). Consequently, atmospheric nitrogen deposition was considered constant over the study period and not included in the multiple regression. Atmospheric deposition of phosphorus was on average <10% of the land-based inputs and was not included in the multiple regression for the same reason.

The ratio between nitrogen and phosphorus diffuse input and freshwater discharge had the largest variations in the beginning of the period when loading compilations were considered more uncertain (Fig. 3). There were no significant trends in the flow-weighted concentrations for either TN ($F_{1,12} = 2.28$, $p = 0.1573$) or TP ($F_{1,12} = 0.06$, $p = 0.8136$) over the entire period, but TN decreased significantly ($F_{1,11} = 10.90$, $p = 0.0071$) if the first year with more uncertain loading figures was excluded. Flow-weighted TN concentrations from diffuse sources was about 7 mg L^{-1} in the beginning of the 1990s, decreasing to ~5.5 mg L^{-1} in recent years. Low levels were observed in the three dry years of 1989, 1996, and 1997. The flow-weighted TP concentration from diffuse sources was relatively constant, ~0.11 mg L^{-1} from 1993 and onward. There was no trend in TP levels during this period ($F_{1,8} = 0.95$, $p = 0.3594$).

Nutrient concentrations—The two wet years in 1994 and 1995 and the two dry years in 1996 and 1997 were clearly visible in the mean nitrogen levels in estuaries and coastal areas as well as for the open-water stations (Fig. 4A,C). DIN levels in surface waters decreased by ~30% from the two first years to the two last years, whereas there was no change in bottom water concentrations. TN levels decreased by 12–18% during the study period. However, nitrogen means during the last couple of years were almost at the same level as in 1996 and 1997 although the freshwater discharge was considerably higher. Trends were not significant for DIN, whereas TN levels decreased significantly by 8 μg L^{-1} yr^{-1} in estuaries and coastal waters and approximately at half the rate in open waters (Table 3).

The effect of dry and wet years on phosphorus levels was less pronounced; the phosphorus means decreased in the beginning of the study period and were more or less stationary in the last part of the investigated period (Fig. 4B,D). DIP levels decreased with significant changes in surface waters of 48–57% observed between 1989–1990 to 2001–2002 (Table 3), whereas the decline in the open-water bottom layer was more moderate (22%). Trends in TP levels were also significant with more similar changes using the same periods (30–39%) for the three considered water types.

The yearly means of DIN and TN were significantly related to freshwater discharge for estuaries and coastal stations as well as for the surface and bottom layer at open-water stations (Table 4). Nitrogen loading from point sources did not show a consistent pattern for explaining interannual variations in nitrogen concentrations, with significant relationships observed only for DIN levels in the surface layer of open-water stations and TN levels in the bottom layer of open-water stations. The yearly means of DIP and TP were clearly linked to point source loadings of phosphorus (Table 4), whereas interannual variations in freshwater discharges did not correlate significantly. The most significant relationships were obtained for estuarine and coastal stations, all having R^2-values >0.85.

In the last 5 yr of the study period, nitrogen did not exhibit the same behavior with respect to freshwater discharge as for the earlier period of 1989–1997 (Fig. 5). All the yearly means for both DIN and TN were below the regression lines except for the DIN level in the bottom layer at open-water stations that had 2 yr above the regression line (1999 and 2002). Although the phosphorus point source loading from Denmark was lower in 1998–2002 than in all the previous years, the phosphorus levels in these years were mostly above the extrapolation of the regression lines, particularly for DIP in estuaries and the coastal area as well as the open-water surface layer (Fig. 6).

Nitrogen levels adjusted for variations in freshwater discharge and nitrogen point source loading showed consistent decreasing trends (Fig. 7A,C), all significant but the adjusted DIN means at open-water stations (Table 3). Removing the interannual variation related to freshwater discharge improved the significance of the trends. The relative change in adjusted DIN levels from 1989–1990 to 2001–2002 varied from −3% in the open-water bottom layer to 14% in the open-water surface layer and 44% in estuaries and coastal waters. These trends were more similar for adjusted TN levels (15–18%). Adjusted phosphorus levels were generally more variable than were adjusted nitrogen levels (Fig. 7B,D), however, without any significant trend (Table 3). For both nitrogen and phosphorus, the highest rate of change and

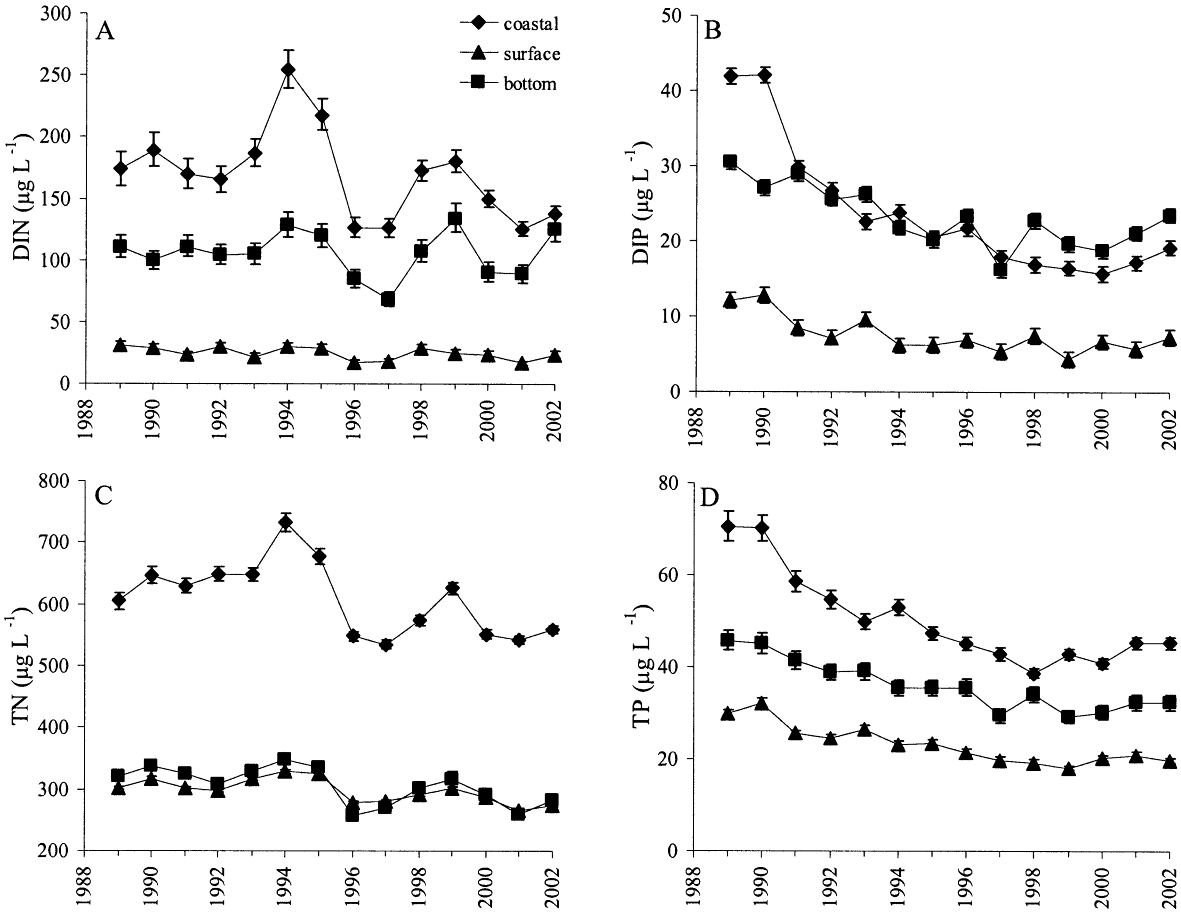

Fig. 4. Yearly means for estuarine and coastal stations, surface layer of open-water stations, and bottom layer of open-water stations. Error bars show the 95% confidence limits of the mean values.

the most significant trends were observed in estuaries and coastal regions.

Discussion

In this study we have identified strong relationships between land-based discharges and nutrient concentrations in the marine environment. This was possible owing to several reasons. First, yearly means of nutrient concentrations with a high precision were obtained by pooling data from a large number of stations, assuming that all stations had the estimated interannual variation in common. Second, the investigated period had large variations in both freshwater discharges and point source phosphorus loadings, yielding a high power for the multiple regression analysis. Third, interannual variations in freshwater discharge and point source loading were not correlated, and the estimates resulting from the multiple regression were consequently not biased. Finally, changes in nutrient concentrations attributed to management actions occurred at different times for nitrogen (end of

Table 3. Trend analysis of the nutrient means (1989–2002, $n = 14$) and means adjusted for variations in freshwater discharge and point source loading. Significant trends (μg L^{-1} yr^{-1}) at the 5% significance level are highlighted by boldface type.

	Estuarine and coastal		Open-water surface		Open-water bottom	
Variable	Trend	*p*	Trend	*p*	Trend	*p*
DIN	−3.93	0.1084	−0.60	0.0723	−0.22	0.8651
TN	**−8.17**	**0.0321**	**−2.94**	**0.0143**	**−4.55**	**0.0134**
DIP	**−1.78**	**<0.0001**	**−0.42**	**0.0040**	**−0.70**	**0.0032**
TP	**−2.05**	**<0.0001**	**−0.87**	**<0.0001**	**−1.15**	**<0.0001**
DIN_{ADJ}	**−6.36**	**0.0006**	−0.13	0.4899	−0.34	0.6851
TN_{ADJ}	**−11.02**	**0.0003**	**−3.70**	**0.0005**	**−4.04**	**0.0005**
DIP_{ADJ}	0.16	0.4023	0.11	0.2753	0.23	0.1435
TP_{ADJ}	0.22	0.2783	−0.17	0.1970	−0.08	0.5514

Table 4. Nutrient means (1989–1997, $n = 9$) related to freshwater discharge and nutrient loading from point sources by multiple regression. Freshwater discharge included only Danish data for estuarine and coastal stations, and the contribution from Denmark, Sweden, and Germany for open-water stations. Total nitrogen loading from Denmark was used for DIN and TN levels, and total phosphorus loading from Denmark was used for DIP and TP levels. Significant relations ($F_{1,6}$ at 5% significance level) are highlighted by boldface type.

			Freshwater discharge		Point source loading	
Variable	R^2	Intercept	Estimate (μg L^{-1} km^{-3})	p	Estimate (μg L^{-1} 10^{-6} kg^{-1})	p
DIN						
Estuarine and coastal stations	0.9535	33.86	**16.83**	**<0.0001**	0.805	0.1707
Open-water stations (surface)	0.7567	477	**0.604**	**0.0320**	**0.439**	**0.0355**
Open-water stations (bottom)	0.6610	42.74	**2.615**	**0.0211**	0.510	0.4491
TN						
Estuarine and coastal stations	0.9509	416.9	**25.556**	**<0.0001**	1.763	0.0750
Open-water stations (surface)	0.8784	237.5	**3.173**	**0.0007**	0.214	0.5900
Open-water stations (bottom)	0.9243	190.8	**5.039**	**0.0003**	**1.463**	**0.0289**
DIP						
Estuarine and coastal stations	0.8701	12.88	−0.2047	0.7297	**6.77**	**0.0007**
Open-water stations (surface)	0.6977	4.59	−0.0350	0.5795	**1.84**	**0.0100**
Open-water stations (bottom)	0.7676	18.13	−0.0735	0.5294	**3.25**	**0.0044**
TP						
Estuarine and coastal stations	0.9228	34.34	0.3019	0.5844	**8.32**	**0.0002**
Open-water stations (surface)	0.7637	16.40	0.1222	0.4497	**2.63**	**0.0061**
Open-water stations (bottom)	0.8650	28.62	0.0313	0.8424	**3.86**	**0.0009**

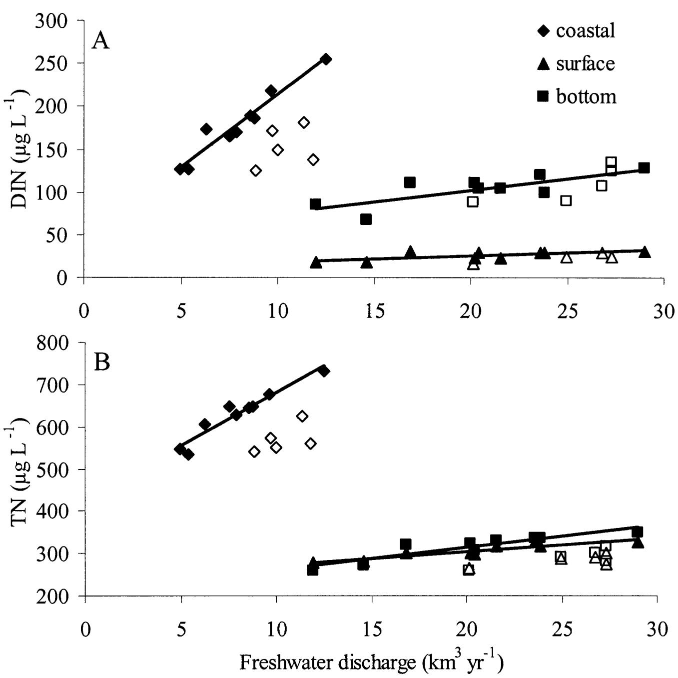

Fig. 5. Yearly (A) DIN and (B) TN levels for estuarine and coastal stations, surface layer of open-water stations, and bottom layer of open-water stations versus freshwater discharge. Freshwater discharges related to nitrogen levels for open-water stations included contributions from Denmark, Sweden, and Germany; coastal nitrogen levels were related to Danish discharges only. Open symbols show the recent levels (1998–2002) not included in the multiple regression.

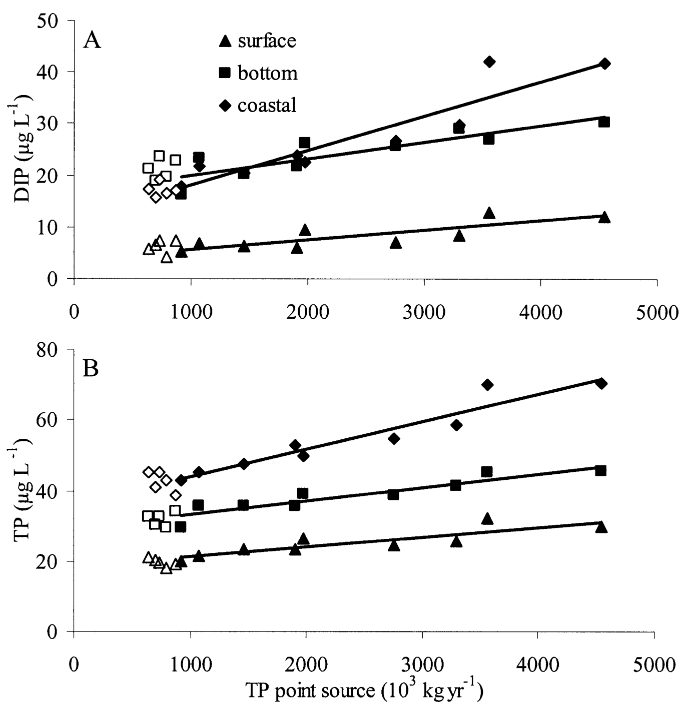

Fig. 6. Yearly (A) DIP and (B) TP levels for estuarine and coastal stations, surface layer of open-water stations, and bottom layer of open-water stations versus phosphorus point source loading from Denmark. Open symbols show the recent levels (1998–2002) not included in the multiple regression.

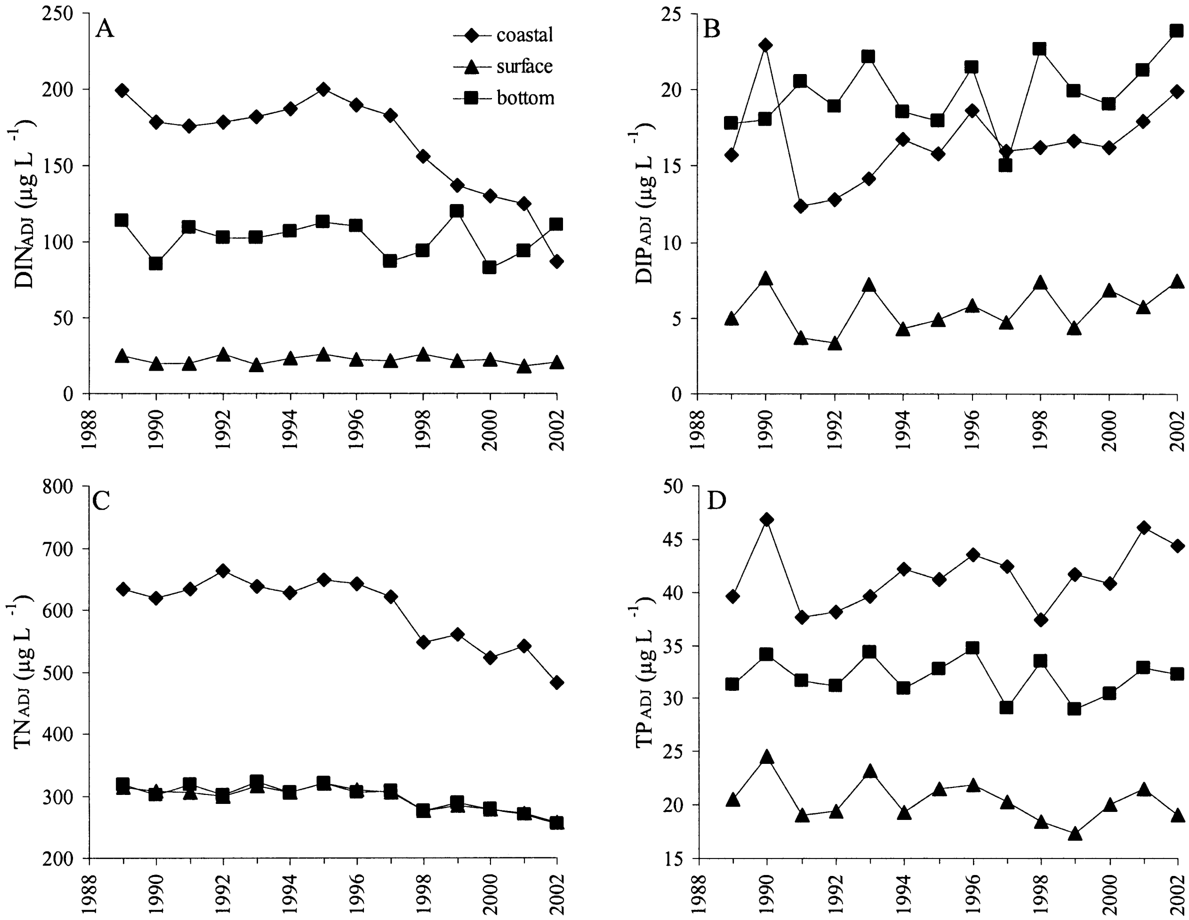

Fig. 7. Nutrient means adjusted for variations in both freshwater discharge and point source loading from the regression analysis.

the 1990s) and phosphorus (beginning of the 1990s), allowing us to separate out these different sources of variation (Conley et al. 2000)

Trends in nutrient concentrations—Nitrogen concentrations showed decreasing trends (Table 3) when variations in freshwater discharge were accounted for, and this decline can primarily be attributed to changes in diffuse loading. This is also shown by decreasing flow-weighted TN concentrations (Fig. 3), taking into account that nitrogen loading in the beginning of the period was more uncertain and that the high value in 1998 may be a consequence of the two dry years of 1996 and 1997 with nitrogen accumulating in the catchment.

Although flow-weighted concentrations of TN began to decline already in 1994 (Fig. 3), the response in nitrogen levels was not clearly identifiable until 1998 in the mean levels adjusted for variations in freshwater discharge (Fig. 7A,C). This delayed response may be partly owing to a substantial reduction in the internal recycling of nitrogen following the two dry years. In 1994 and 1995, nitrogen levels in the water column were, most likely, kept high by a large internal nitrogen release from the sediments. This may potentially also have been the case in 1996 and 1997, although these two dry years constitute one end of the scale in the relationship between nitrogen concentrations and freshwater discharge (Fig. 5). Coupling between nutrient loading, water-column production of organic matter, and recycling of nutrients from sediments occurs over time scales of several years or less (Boynton et al. 1995). Attempts to find significant correlations between nutrient load and system level responses in estuaries often succeeds only when annual nutrient loads are combined with some fraction of the previous year's nutrient load (Boynton et al. 1995; Conley et al. 2000), suggesting that an internal load is important.

Phosphorus concentrations declined substantially in estuaries and coastal areas as well as in the open-waters of the Danish straits during the beginning of the investigated period (Fig. 4B,D). This trend was clearly linked to reductions in point source loading, mainly through improved wastewater treatment. Changes in diffuse phosphorus loading should in principle, as shown for point source loading, show a similar variation in phosphorus concentrations in the water column, but these small-scale variations related to freshwater discharge are masked by larger fluxes such as phosphorus release during anoxic conditions from sediments. In the Baltic Sea, for example, annual variations in phosphorus release from sediments with variations in anoxia are over an order of magnitude larger than annual phosphorus loading (Conley et al. 2002*a*).

Flow-weighted TP concentrations in freshwater discharge corrected for point sources have remained almost constant,

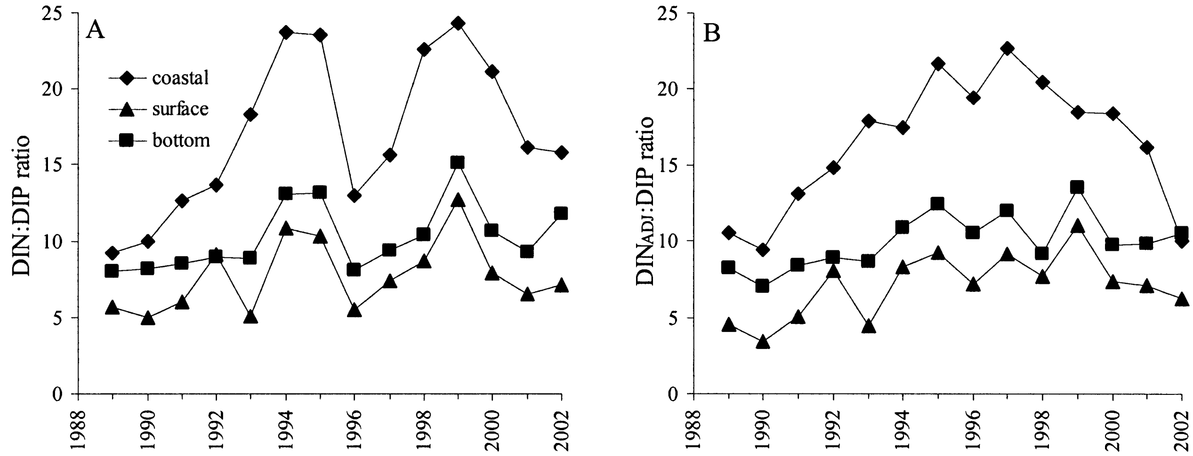

Fig. 8. The nitrogen/phosphorus molar ratio for the nutrient means (A) and nutrient means adjusted to variations in freshwater discharge (B).

whereas flow-weighted TN concentrations have declined. This has naturally altered the nitrogen/phosphorus ratio of diffuse loading and consequently that of the total loading. The increase in the DIN/DIP ratio in the beginning of the 1990s (Fig. 8) signals a change from potential nitrogen limitation in favor of phosphorus limitation. In the last 5 yr, the DIN/DIP ratio has declined in estuaries and coastal waters, which may potentially have led to more nitrogen limitation in favor of phosphorus limitation. Several Danish estuaries have shown spring phosphorus limitation switching to nitrogen limitation in early summer (Holmboe et al. 1999). There can still be large interannual variations in nitrogen and phosphorus limitation owing to changes in freshwater discharge, but nitrogen has become more important as the limiting nutrient over the last 5 yr (Fig. 8B). Although the decreasing trends in DIN and DIP may have resulted in changing patterns of nutrient limitation, it should be acknowledged that the combined effect has increased overall nutrient limitation (Ærtebjerg et al. 2003).

Bottom water concentrations—Advective transport dominates the open-water of the Danish straits, and the origin of the inflowing water from bottom waters in the Skagerrak may originate from different regions of the North Sea, with large variations in nutrient levels (Rydberg et al. 1996). The main source of inflowing bottom water is from the central North Sea having moderate nutrient levels and salinity of ~34, but occasionally nutrient-rich water with salinity of ~32 originating from the German Bight and carried with the Jutland Coastal Current spills into the Kattegat (Rydberg et al. 1996). This has happened to varying degrees in 1989, 1995, 1999, and 2002 (observed in winter–spring monitoring data of DIN vs. salinity) (data not shown), which could explain the relatively high values of adjusted DIN levels in these years (Fig. 7A). Another phenomenon that may influence bottom water concentrations in the Danish straits, particularly in the Kattegat, is the inflow of DIN-depleted surface water from the Skagerrak into the bottom layer (observed in 1990 and 1997) (data not shown).

The phosphorus pool in the bottom layer is considered to depend on local loading, water exchanges, and oxygen conditions (Rasmussen et al. 2003*b*). Oxygen depletion is a reoccurring phenomenon in the Danish straits (Andersson and Rydberg 1988, 1993; Babenerd 1991), and 2002 was the worst year ever recorded, with 21% of the area having oxygen concentrations <2 mg L^{-1} for extensive periods (HELCOM 2003). On the other hand, oxygen conditions were generally good in 1997 (Rasmussen et al. 2003*a*). These 2 yr, corresponding to the highest and lowest values for the adjusted DIP means, demonstrate that sediment phosphorus release during anoxic conditions increases the DIP levels in the bottom layer in open waters (Mortimer 1941; Conley et al. 2002*a*).

Nutrient levels in the open-water bottom layer were mainly determined by local inputs of both nitrogen and phosphorus, with relationships between concentrations and loads as strong as those in the open-water surface layer (Table 3). These interannual variations in nutrient levels cannot be explained by inflow from the central part of the North Sea, where nutrient levels are low (OSPAR Commission 2000) and presumably not directly influenced by land-based loading. This suggests a substantial vertical exchange of nutrients over the pycnocline through upwelling and entrainment (Gustafsson 2000) and remineralization of sedimenting particulate matter, particularly after the diatom spring bloom (Josefson and Hansen 2003). This supports the idea of the Danish straits being a marginal sea with estuarine character. Thus, interannual variations in bottom water nutrient levels were mainly determined by discharges from local sources, whereas changes in Skagerrak inflow and episodes of oxygen depletion only caused minor deviations from this pattern.

Nutrient management—Over the past 2 decades, coastal eutrophication of Danish marine waters has been a major concern, and substantial nutrient reductions have been achieved through national action plans, international marine conventions, and European Union legislation (Iversen et al. 1998; Conley et al. 2000; Ærtebjerg et al. 2003). The declining trends in nutrient concentrations documented here are to our knowledge the first successful large-scale effort to reduce inputs from both point and diffuse sources.

Table 5. Nitrogen and phosphorus inputs to the aquatic environment for the baseline (mid-1980s) with reduction targets partitioned into sectors. UWTPs, urban wastewater treatment plants; IDs, industrial discharges. Phosphorus discharges from the agricultural sector include losses from farmyards only. For details, please see Ærtebjerg et al. (2003) and Grant and Waagepetersen (2003).

	Nitrogen	Phosphorus
Sector	Baseline ÷ Reduction (%) = Target	Baseline ÷ Reduction (%) = Target
Agriculture	311,000 ÷ 152,400 (49) = 158,600	4,400 ÷ 4,000 (91) = 400
UWTPs	18,000 ÷ 11,400 (60) = 6,600	4,470 ÷ 3,250 (72) = 1,220
IDs	5,000 ÷ 3,000 (60) = 2,000	1,250 ÷ 1,050 (82) = 1,820
Total	334,000 ÷ 166,800 (50) = 167,200	10,120 ÷ 8,300 (80) = 3,440

The first national initiative was the 1985 NPo Action Plan with a suite of measures implemented in relation to the discharge of nitrogen (N), phosphorus (P), and organic matter (o) from agriculture and wastewater; however, specific reduction targets were not set. An event of widespread hypoxia in the Danish straits in 1986 led to the adoption of an agenda urging the government to reduce discharges and losses of nitrogen (by 50%) and phosphorus (by 80%) from agriculture, municipal wastewater treatment plants, and individual industrial discharges (Kronvang et al. 1993; Conley et al. 2002b). This strategic aim was formulated into sector-specific reduction objectives and targets for (1) discharges and losses from agriculture, (2) discharges from municipal wastewater treatment plants, and (3) direct discharges from industries (Table 5).

The reduction targets for both municipal wastewater treatment plants and industries were met in 1995, whereas the specific objectives and targets for the agricultural sector were difficult to meet within the original time frame. The Action Plan on Sustainable Agriculture adopted in 1991, focusing on reduction of losses from cultivated fields, was followed by the second Action Plan on the Aquatic Environment with additional measures in 1998 to fulfill the requirements of the European Union Nitrates Directive (Anonymous 1991). The decision for the second action plan was influenced by collapse of the Mariager Fjord estuary, which went completely anoxic in 1997 (Fallesen et al. 2000).

Total expected reductions in nitrogen root zone losses from agriculture in 2002 were estimated at $149{,}000 \times 10^3$ kg, corresponding to a reduction of 48% in the most recent assessment on the effectiveness of the measures (Grant and Waagepetersen 2003). However, the reduction estimates are associated with substantial uncertainty, and there are considerable time lags between changes in agriculture practice and water quality responses (Stålnacke et al. 2003; Tomer and Burkart 2003). These reductions would not have been achieved if the periodic assessment for reduction compliance had not been carried out and if the national monitoring program had not maintained focus on eutrophication. Reductions in nutrient overenrichment of coastal ecosystems will rely on implementation of an adaptive management framework (Boesch 2002). Further reductions in diffuse nitrogen loading may be needed, considering that Denmark has had one of the highest area-specific nitrogen loss rates (Conley et al. 2000) and that the Danish straits with an almost permanent stratification are prone to hypoxia.

Ecosystem perspectives—Decreasing nutrient concentrations is the first step toward reducing the adverse effects of eutrophication. Danish estuaries and coastal areas have had a long record of eutrophication symptoms from nutrient enrichment, including increased primary production (Richardson and Heilmann 1995), decreasing bottom oxygen concentrations (Andersson 1996), and loss of macrophytes (Borum and Sand-Jensen 1991). Interannual variations in nitrogen loading are reflected in summer chlorophyll *a* concentration and bloom frequency in the Kattegat (Carstensen et al. 2004) as well as primary production (Carstensen et al. 2003). Nielsen et al. (2002*b*) found that chlorophyll *a* and water transparency were significantly related to total nitrogen concentrations in Danish estuaries and coastal waters and, further, that the depth colonization of eelgrass and macroalgae were significantly related to water transparency (Nielsen et al. 2002*a*). These small components of the large complex ecosystem show that reduced nutrient concentrations are likely to improve the ecological status of estuaries and coastal and open waters in Denmark through direct or delayed responses or alternatively through threshold mechanisms (Scheffer et al. 2001). Some signs of ecosystem recovery have already been observed (Ærtebjerg et al. 2003).

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Received: 19 March 2004
Accepted: 29 April 2005
Amended: 1 May 2005