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Nutrient pressures and ecological responses to nutrient loading reductions in Danish streams, lakes and coastal waters

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Abstract

The Danish National Aquatic Monitoring and Assessment Programme (NOVA) was launched in 1988 following the adoption of the first Danish Action Plan on the Aquatic Environment in 1987 with the aim to reduce by 50% the nitrogen (N) loading and by 80% the phosphorus (P) loading to the aquatic environment. The 14 years of experience gathered from NOVA have shown that discharges of total N (TN) and P (TP) from point sources to the Danish Aquatic Environment have been reduced by 69% (N) and 82% (P) during the period 1989–2002. Consequently, the P concentration has decreased markedly in most Danish lakes and estuaries. Considerable changes in agricultural practice have resulted in a reduction of the net N-surplus from 136 to 88 kg N ha^{-1} yr^{-1} (41%) and the net P-surplus from 19 to 11 kg P ha^{-1} yr^{-1} (42%) during the period 1985–2002. Despite these efforts Danish agriculture is today the major source of both N (>80%) and P (>50%) in Danish streams, lakes and coastal waters. A non-parametric statistical trend analysis of TN concentrations in streams draining dominantly agricultural catchments has shown a significant ($p<0.05$) downward trend in 48 streams with the downward trend being stronger in loamy compared to sandy catchments, and more pronounced with increasing dominance of agricultural exploitation in the catchments. In contrast, a statistical trend analysis of TP concentrations in streams draining agricultural catchments did not reveal any significant trends. The large reduction in nutrient loading from point and non-point sources has in general improved the ecological conditions of Danish lakes in the form of increased summer Secchi depth, decreased chlorophyll *a* and reduced phytoplankton biomass. Major changes have also occurred in the fish communities in lakes, with positive cascading effects on water quality. In Danish estuaries and coastal waters only a few significant improvements in the ecological quality have been observed, although it is expected that the observed reduced nutrient concentrations are likely to improve the ecological quality of estuaries and coastal waters in Denmark in the long term.

Keywords: Nutrients; Pressures; State; Trends; Ecological impacts; Freshwater; Estuaries

1. Introduction

Excess nitrogen (N) and phosphorus (P) loading from point and non-point sources is considered one of

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the main factors damaging the ecological quality of streams, lakes and estuaries and the deteriorating quality of ground water (Meybeck, 1982; Iserman, 1990; Sabater et al., 1990; European Environment Agency, 1995, 1999; Jordan et al., 1997). The measures introduced in various countries to reduce nutrient pollution and hence improve water quality have had varying success. In many larger rivers the ecological quality has improved due to a reduction in point source discharges (European Environment Agency, 1999), whereas the ecological quality of smaller streams, being ecologically important for the aquatic biota, has seldom been improved (Pieterse et al., 2003). Thus, the efforts to reduce point source nutrient inputs to rivers, lakes, estuaries and coastal waters has been successful in many countries world wide, but improvements in ecological quality were in many cases dampened by nutrient losses from non-point sources (e.g. Thornton et al., 1999).

The newly adopted EU Water Framework Directive (WFD) aims at protecting different water bodies to prevent further deterioration and to protect and enhance the status of aquatic ecosystems (European Parliament and of the Council 2000/60/EC, 2000). The implementation of the WFD involves different steps where River Basin Authorities shall (i) perform an analysis of pressures and impacts (before 2005); (ii) develop monitoring programmes (before 2007); and (iii) implement mitigation strategies in the form of River Basin Management Plans (before 2009). An important part of the WFD is that reference conditions of different water body types should be detailed and applied in the target setting of ecological quality criteria in water bodies for judgement of the fulfilment of quality objectives (guideline).

It is essential to document the chemical and ecological responses to previous reductions of nutrient loading to the aquatic environment in order to improve our knowledge of important issues such as: (i) time lag and inertia in nutrient turnover from soil to surface water (e.g. Stålnacke et al., 2003); (ii) quantitative responses to different management measures against non-point pollution (Kronvang et al., 1999); and (iii) ecosystem responses to reduced nutrient pollution, including system resilience (Jeppesen et al., 1999).

Many countries have developed monitoring programmes and protocols that enable a reliable quantification of nitrogen (N) and phosphorus (P) loadings and concentrations in the aquatic environment (e.g. Kronvang et al., 1993; Kronvang et al., 1995). Data from such monitoring programmes can be of great help in understanding the various hydrological and biogeochemical processes governing N and P cycling in terrestrial, freshwater and marine environments and their ecological impacts (Kronvang et al., 1993). Together with existing models the experience gathered can assist catchment managers in making predictions of nutrient reductions and ecological effects in the aquatic environment (e.g. Heathwaite et al., 2000; Pieterse et al., 2003).

In Denmark, the first River Basin Management Plans for reduction of N, P and organic matter pollution of surface waters were adopted in the early 1970s (Andersen, 1994). The Danish Parliament adopted the first National Action Plan in 1987 with the aim to reduce by 50% the N-loading and by 80% the P-loading of surface waters, and at the same time the Danish National Aquatic Monitoring and Assessment Programme (NOVA) was launched (Kronvang et al., 1993). The 14 years of experience from the NOVA programme serve as a multiple catchment scale experiment for documenting nutrient responses to changes in point sources discharges and agricultural practices and in land–water interactions. As shown by Stålnacke et al. (2003) in an analysis of relationships between intensity of agricultural production and resulting N loads in Latvian rivers, there seems to be inertia between soil-surface water N interactions, such that nutrient loads carried by rivers were not reduced despite the large decrease in fertilizer input to agricultural land. Moreover, resilience in lakes, estuaries and coastal marine ecosystems can greatly influence ecological improvements being either accelerated or dampened following nutrient reductions, depending on the biological structure and sediment–water interactions (Jeppesen et al., 1999). Finally, changes in nutrient emissions will influence N/P-ratios in streams, rivers, lakes and estuaries in such a manner that a new nutrient limitation situation may occur in the water bodies Conley, 1999). Knowledge of such ecosystem responses is vital to catchment managers in Europe who are challenged with the task of implementing the WFD.

This paper describes and documents the effects of four Danish Action Plans adopted since the 1980s on N and P pollution of streams, rivers, lakes and estuaries. The paper includes quantitative information on the impact of the Action Plans on driving forces causing nutrient pollution, changes in different nutrient pollution pressures, trends in the nutrient state and changes in the ecological quality of surface waters.

2. Materials and methods

2.1. River network

A network of 180 Danish river monitoring stations were established in Denmark in 1988 covering the existing gradients in climate, soil types, geology, land use and agricultural practices (Kronvang et al., 1993). The sampling stations were selected to obtain reliable results on the state and trends in nutrient loadings to water bodies, on changes in nutrient sources and trends in nutrient concentrations and composition of surface waters. The river network is designed to obtain information on nutrient pollution at three levels: (i) a nation-wide monitoring of nutrient sources and nutrient loading to freshwater and marine water at a total of 130 sampling stations; (ii) monitoring of nutrient sources, concentrations and export from catchments where, respectively, point sources (86 catchments), agricultural non-point sources (48 catchments) and nutrient losses from natural areas as a reference (7 catchments) are the only nutrient source; and (iii) monitoring of nutrient cycling in 6 smaller agro-ecosystems. The river stations are instrumented with equipment for the continuous recording of water stage, and discharge is measured fortnightly or monthly to enable calculation of daily discharge. Standardised protocols have been developed for water sampling, laboratory analysis and load estimation to obtain reliable and comparable monitoring results. Concentrations of total and inorganic N and P fractions are measured at weekly, fortnightly or monthly intervals, the interval between sampling dates being longest in baseflow dominated rivers (Kronvang and Bruhn, 1996). Annual data on nutrient discharges from different point sources (WWTP's, industrial plants, urban runoff, freshwater fish farms and scattered dwellings) are available on catchment level based on standardised sampling and load estimation protocols. A detailed description of the river monitoring programme and methods can be found in Kronvang et al. (1993).

2.2. Lake network

Twenty-seven Danish lakes have been monitored intensively since 1989. A detailed description of the lakes and methods is found in Jeppesen et al. (2002, 2004a,b)), and only a brief overview will therefore be given here. Following the standard protocol prescribed by the Danish Nation-Wide Monitoring Programme on the Aquatic Environment (Kronvang et al., 1993), biweekly samples are taken during summer (1 May–1 October) and monthly samples during the rest of the year for analyses of TP, phytoplankton and zooplankton communities. TP, TN and chlorophyll *a* are analysed on a depth-integrated sample from the photic zone sampled at a mid-lake station and analysed according to Søndergaard et al. (1992) and Jespersen and Christoffersen (1987), respectively. Mass balances are established based on 18–26 annual measurements in main inlets and outlets of the lakes (Søndergaard et al., in press).

Zooplankton densities are determined using depth-integrated water samples taken with a Patalas sampler and pooled from 1 to 3 stations, and phytoplankton is counted on Lugol-fixed sedimented water samples (pooled sample from the photic zone at a mid-lake station) using an inverted microscope. Biomass is estimated from length–weight relationships (zooplankton) and geometric forms (phytoplankton) according to standard methods (Jeppesen et al., 2003).

The composition and relative abundance of the pelagic fish stock in the lakes are determined by standardised test fishing (Mortensen et al., 1991) with multi-mesh sized gill nets (6.25, 8, 19, 12.5, 16.5, 22, 25, 30, 33, 38, 43, 50, 60, 75 mm) conducted in each lake between 15 August and 15 September. The nets are set in late afternoon and retrieved the following morning. Catch per unit effort (CPUE) of planktivorous fish is calculated as mean catch night^{-1} net^{-1}. Test fishing is only conducted every five years, sampling years differing among the lakes.

2.3. Estuarine and coastal network

A total of 40 estuaries and coastal regions are monitored as part of NOVA at various frequencies for the different stations and parameters considered. Physical, chemical and biological variables in the water column are measured monthly in some areas up to weekly in specific designated estuaries with intensive sampling, whereas the benthic parameters are monitored annually or bi-annually. Water samples are analysed for ammonia, nitrite, nitrate, total nitrogen, phosphate and total phosphorus as well as a number of other chemical constituents. A detailed description of the monitoring activities can be found in Kronvang et al. (1993) and Conley et al. (2002). Yearly means of nutrient concentrations were computed by means of a general linear model that described variations between stations, years and months after log-transformation of the variables (Rasmussen et al., 2003). Because of an uneven number of observations, comparable yearly means were calculated by computing the marginal distribution of the mean values, where differences in sampling frequency was taken into account.

2.4. Statistical methods applied

Annual nutrient transport is estimated by means of an interpolation method (e.g. Kronvang and Bruhn, 1996) given that nutrient concentrations have been measured at various times over the year and that mean daily runoff values exist for the monitoring station. Concentrations were interpolated linearly in time for days without nutrient measurements.

If nutrients were measured at Julian days denoted t_i, $i=1,2,\ldots,n$ having concentrations c_i, $i=1,2,\ldots,n$, and let t_0 and t_{n+1} denote the first and last day of the year with nutrient concentrations equal to the first and last measurement of the year, $c_o=c_1$ and $c_{n+1}=c_n$.

Then the annual transport was estimated by

$$\hat{L}=\sum_{i=0}^{n-1}\sum_{t_i<t\le t_{i+1}} q_t \frac{c_i(t_{i+1}-t)+c_{i+1}(t-t_i)}{t_{i+1}-t_i} \quad (1)$$

The source apportionment method was applied to calculate the importance of point and non-point sources for the N and P export at a given river monitoring station. The method assumes that the TN or TP transport at a selected river measurement site (L_{river}) represents the sum of the components of the nutrient discharges from point sources (D_{P}), the nutrient losses from non-point sources (LO_{D}) and the natural background losses of nutrients (LO_{B}). Furthermore, it is necessary to take into account the retention of nutrients in the catchment after their emission to surface water (R). This may be expressed as follows:

$$L_{\text{river}}=D_{\text{P}}+\text{LO}_{\text{D}}+\text{LO}_{\text{B}}-R \quad (2)$$

The aim of the source apportionment is to evaluate the contributions of point and non-point sources of nutrients to the total riverine nutrient load, i.e. to quantify the nutrient losses from non-point sources (LO_{D}) as follows:

$$\text{LO}_{\text{D}}=L_{\text{river}}-D_{\text{P}}-\text{LO}_{\text{B}}+R \quad (3)$$

Trend analysis of time series of TN and TP concentrations was undertaken using the Mann–Kendall seasonal test with correction for serial correlation (Hirsch and Slack, 1984). This is a robust non-parametric site-specific statistical test for monotone trends. The number of seasons per year was set at 12, one for each month of the year. A test statistic was calculated for each season, the seasonal statistics being combined to one overall test statistic, thereby eliminating seasonal effects. The test statistic identifies whether the trend is upward (positive) or downward (negative). Normally, both N and P concentrations depend heavily on the discharge at the time of measurement. To detect trends attributable mainly to anthropogenic interactions, it is necessary to compute discharge-adjusted concentrations, and discharge adjustment was hence performed by applying the robust curve fitting procedure LOWESS (Locally Weighted Scatterplot Smoothing; Cleveland, 1979). The magnitude of the trend was estimated by the robust and non-parametric Sen's slope estimator (Hirsch et al., 1982).

3. Results and discussion

3.1. Measures implemented to reduce nutrient pollution in Denmark

During the last two decades several Action Plans have been implemented in Denmark to reduce nutrient loading to the aquatic environment (Table 1). Action Plan I included measures against both point source and non-point source nutrient pollution and had the overall

Table 1
Action plans and their major elements adopted in Denmark to combat nitrogen pollution from agriculture

Action plans	Year of adoption	Major implemented measures to combat diffuse nutrient pollution
NPO action plan	1985	Elimination of direct discharges from farms
		Livestock harmony at the farm level
Action plan on the aquatic environment I	1987	9-Month storage facility for slurry
		65% Winter green fields
		Crop and fertiliser plans
Plan for sustainable agricultural development	1991	Standard nitrogen fertilisation values for crops
		Standard values for nitrogen in animal manure
		Required utilisation of nitrogen in animal manure (30–45%)
		Fertiliser accounts at farm level
Action plan on the aquatic environment II	1998	Demands for an overall 10% reduction of nitrogen application to crops
		Demands for catch crops
		Demands for transforming 16,000 hectares farmland into wetlands
		Reiterated demands for utilisation of nitrogen in animal manure (40–55%)

aim of reducing N-loading and P-loading to the aquatic environment by 50 and 80%, respectively, within a five year period. Action Plan I was reinforced by the Plan for Sustainable Agricultural Production in 1993 and Action Plan II in 1998, both being directed against nutrient losses from agricultural non-point pollution. Action Plan II included, for the first time, mitigation measures directed against sources of non-point nutrient losses (change in agricultural practices) as well as transport pathway measures by restoring formerly drained wetlands in order to increase the self purification potential through increased denitrification of nitrate (Table 1). The demand for better utilisation of nitrogen in animal manure has had the highest effect on nitrogen losses to surface waters. Consequently, the consumption of N and P in chemical fertilizer decreased dramatically from 392 to 206 mill. kg N and from 48,000 to 15,000 kg P during the period 1985–2002 in Danish agriculture owing to the restrictions on farmers in the four Action Plans. The Action Plans have reduced the annual net N-surplus (total nutrient input to soil minus nutrients removed with harvested crops) on Danish agricultural land from 136 to 88 kg N ha^{-1} (41%) and the net P-surplus from 19 to 11 kg P ha^{-1} (42%) (Grant et al., 2003).

3.2. Background nutrient concentrations and losses

Since 1989, values of background nutrient concentrations and losses to surface water have been established from the monitoring of 7 small streams not directly impacted by agriculture and covering gradients in climate and geology in Denmark (Table 2). As requested for reference conditions in the WFD, the anthropogenic pressures on the catchments are low; although they receive nutrient inputs from atmospheric deposition (long-range and regional). The rates of dry deposition of ammonia nitrogen resulting from

Table 2
Average annual runoff, export coefficients and flow-weighted concentrations of nitrogen and phosphorus fractions measured in 7 small streams draining mostly forested catchments with low anthropogenic pressures during the period 1989–2002

	Mean (± standard error)	Range
Runoff (mm)	184±11	112–250
Export coefficients		
Total N (kg N ha^{-1})	2.02 (0.15)	1.15–3.10
Nitrate N (kg N ha^{-1})	1.25 (0.08)	0.82–1.77
Ammonium N (kg N ha^{-1})	0.070 (0.006)	0.032–0.111
Total P (kg P ha^{-1})	0.090 (0.007)	0.040–0.136
Dissolved reactive P (kg P ha^{-1})	0.035 (0.003)	0.018–0.056
Flow-weighted concentration		
Total N (mg N l^{-1})	1.25 (0.03)	1.13–1.45
Nitrate N (mg N l^{-1})	0.79 (0.03)	0.67–1.00
Ammonium N (mg N l^{-1})	0.041 (0.002)	0.028–0.058
Total P (mg P l^{-1})	0.053 (0.001)	0.042–0.059
Dissolved reactive P (mg P l^{-1})	0.021 (0.001)	0.014–0.026

The standard error of the mean is shown in parenthesis.

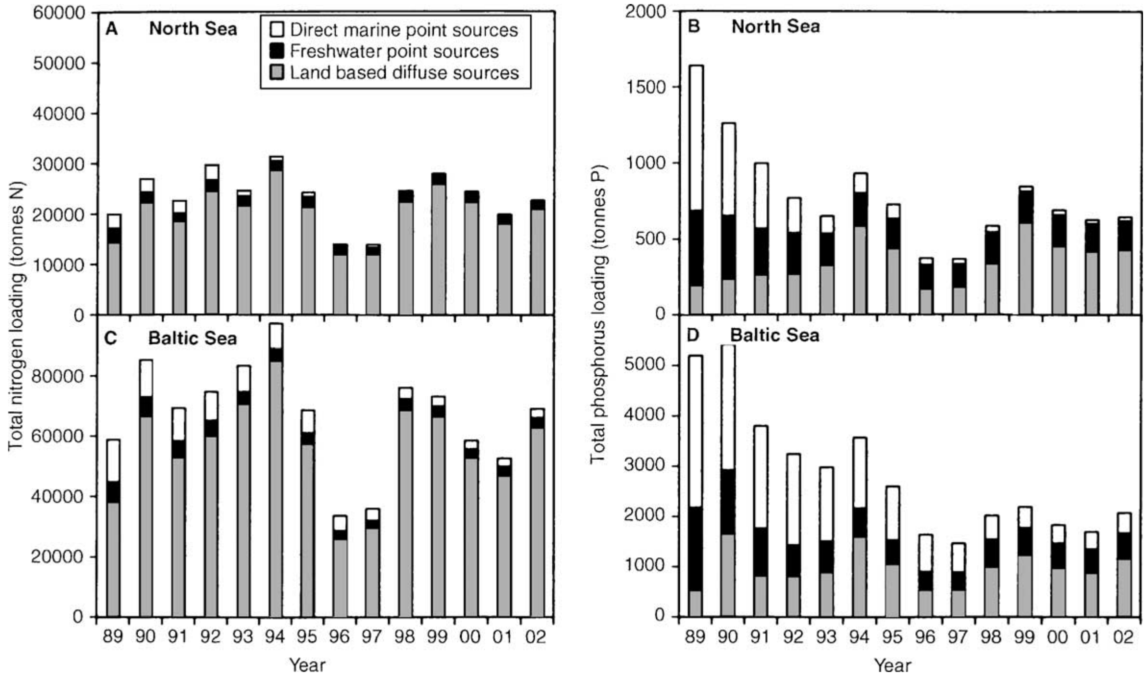

Fig. 1. Annual total nitrogen (A,C) and total phosphorus (B,D) loading of freshwater and coastal waters in the North Sea and Baltic Sea regions of Denmark during the period 1989–2002, as derived from 3 main sources: point sources discharging to freshwater, point sources discharging directly to coastal marine waters and non-point sources.

agriculture is quite high in Denmark due to animal husbandry (Hertel et al., 1995), such that agriculture still has a substantial indirect impact even in ‘undisturbed’ catchments. To a lesser degree catchments are also disturbed from other types of land use than agriculture ($<10\%$ agricultural land). The background losses of P in Danish streams are in the same range as the 0.117 kg P ha^{-1} reported by Dillon and Kirchner (1975) for forested sedimentary areas. However, the background export of TN from Danish catchments is higher than the 0.087–0.678 kg N ha^{-1} reported by Dillon et al. (1991) for forested stream catchments in Central Ontario and the 1 kg N ha^{-1} reported by Mulder et al. (1997) for a Norwegian forest ecosystem.

3.3. Trends in nutrient loading to freshwater and coastal waters

The loading of nutrients to freshwater and coastal waters to the Danish North Sea and Baltic Sea regions has undergone substantial inter-annual variations during the period 1989–2002 (Fig. 1). During the period, discharge of nutrients from point sources to freshwater and coastal waters has decreased due to improved treatment at sewerage treatment plants and, to a minor degree, reductions in nutrient discharges from industrial plants, fish farms and scattered dwellings (Fig. 1). Consequently, the majority of the land-based nutrient loading of Danish freshwater and coastal waters is today delivered from non-point sources (Fig. 1 and Table 3). A relationship between annual runoff and annual non-point nutrient loading of Danish coastal waters shows that meteorological fluctuations is one of the main driving factors for non-point nutrient losses (Fig. 2).

The annual emission of nutrients to Danish streams, rivers and lakes from agricultural areas has been calculated by applying the source apportionment method, which considers both background losses and nutrient retention in surface waters (Table 4). Retention of N and P in surface water is clearly of great importance for N and of minor importance for P when calculating the contribution from agriculture

Table 3
Changes in the importance of non-point nitrogen and phosphorus losses for the total nitrogen and total phosphorus loading of Danish freshwater and coastal waters in the North Sea and Baltic Sea regions during the period 1989–1990 to 2000–2001

	Total nitrogen (%)		Total phosphorus (%)	
	1989–1990	2000–2001	1989–1990	2000–2001
North Sea				
Freshwater	87	92	32	69
Coastal waters	77	91	15	66
Baltic Sea				
Freshwater	88	94	40	68
Coastal waters	71	89	20	54

(Table 4). Despite a decrease in N and P losses from point sources, the total average annual riverine nutrient loading to Danish freshwaters does not markedly change during three periods of calculation (Table 4). This is mainly attributable to an increase in the average annual TN and TP loss from agriculture to freshwater during the three periods (Table 4). The documented increase in TN and TP losses from agriculture can, however, be explained by an increase in the average annual runoff during the three periods, amounting to 401 mm (1989–93), 406 mm (1994–98) and 510 mm (1999–02) in the North Sea region, and 260, 280 and 347 mm in the Baltic Sea region of Denmark.

3.4. Nutrient state in streams

The flow-weighted N and P concentrations generally increases with the proportion of agricultural land in Danish catchments without major point sources (>30 person equivalents) (Fig. 3). The dominant N-fraction in the streams draining the different catchment types is dissolved inorganic N (DIN) which increases with enhanced proportions of agricultural land (Fig. 3). The flow-weighted concentration of dissolved reactive P (DRP) generally constitutes less than 50% of the flow-weighted TP concentration in all stream and catchment types (Fig. 3B).

A seasonal Mann–Kendall trend analysis of TN and TP concentrations in streams with flow-adjustment reveals a general downward trend in TN concentrations during the period 1989–2002 (Table 5). The downward trend is in general stronger for loamy than for sandy catchments and increases with an increasing proportion of agricultural land in the catchments (Table 5). Thus, nitrogen concentrations in streams draining catchments without larger point source discharges (<0.5 kg N ha^{-1}) respond very strongly to the mitigation measures implemented to reduce N-leaching from agricultural production. Of the 70 streams tested 48 showed a statistically significant downward trend ($p<0.05$). This finding can be explained by a shorter residence time for water

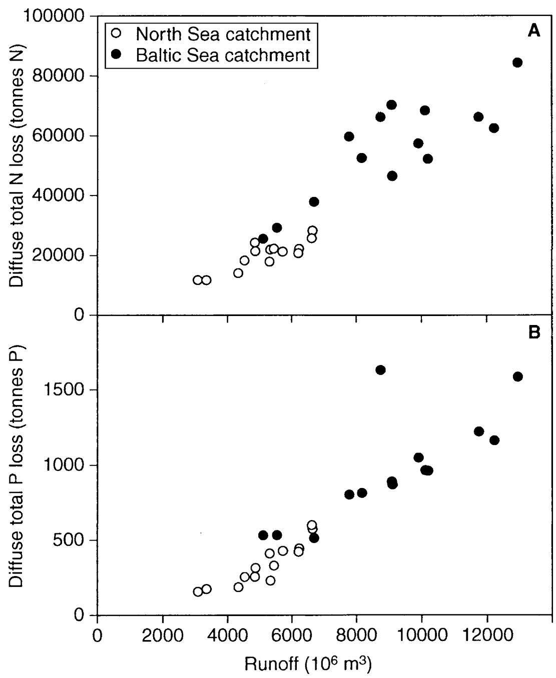

Fig. 2. Relationship between annual runoff and, respectively, annual non-point total nitrogen (A) and total phosphorus (B) losses during the period 1989–2002.

Table 4
Average annual losses of total nitrogen and total phosphorus from agriculture to freshwater in the North Sea and Baltic Sea regions of Denmark during the three periods: 1989–1993, 1994–1998 and 1999–2002, calculated using the source apportionment method

	North Sea Region			Baltic Sea Region		
	1989–93	1994–98	1999–02	1989–93	1994–98	1999–02
Total nitrogen (Tonnes N)						
Riverine transport	22,320	20,920	23,430	63,150	56,600	60,480
Point sources	2260	1790	1720	5700	3510	3520
Retention in surface water	6970	6450	6150	18,730	16,050	14,660
Background	2240	2400	2640	5840	6250	6870
Agriculture	24,790	23,180	25,220	70,340	62,890	64,750
Total phosphorus (Tonnes P)						
Riverine transport	595	527	674	1957	1404	1565
Point sources	343	191	202	1024	468	508
Retention in surface water	16	14	16	81	39	29
Background	102	99	127	265	257	332
Agriculture	166	252	361	748	717	754

and hence N in the soil due to a larger proportion of the net-precipitation going into rapid subsurface drainage (tile drainage) in loamy catchments as compared to sandy catchments (Kronvang et al., 1996).

The statistical trend analysis of TP concentrations in streams with low point source discharges from the catchments (<0.025 kg P ha^{-1}) shows a general downward trend in streams draining loamy catchments; in streams draining sandy catchments the trend points both ways (Table 5). Only 7 of the 45 streams tested experienced a statistical significant downward trend ($p<0.05$). The trend results show no obvious relationship with the proportion of agricultural land in the catchments (Table 5). The reason for the general downward trend in streams draining loamy catchments may possibly be attributed to a decline in P discharged from scattered dwellings via tile drains.

The significant downward trend observed for N in Danish streams draining agricultural catchments in response to changes in agricultural practices and fertilisation is interesting. Changes in agricultural management do not automatically lead to changes in nutrient loading and it may take decades in some watersheds to record reductions in nutrient loading because of high groundwater nitrate concentrations from previous heavy use of fertilisers (Tomer and Burkhart, 2003). Stålnacke et al. (2004) examined changes in nitrogen and phosphorus loading by rivers in response to the significant decreases in agricultural intensity and fertiliser use in Eastern Europe following the collapse of the Soviet Union. Downward trends were found in only 2 of the 4 rivers examined suggesting that factors other than reduced fertilizer use influenced the inertia of the water quality response. The relatively rapid response observed in

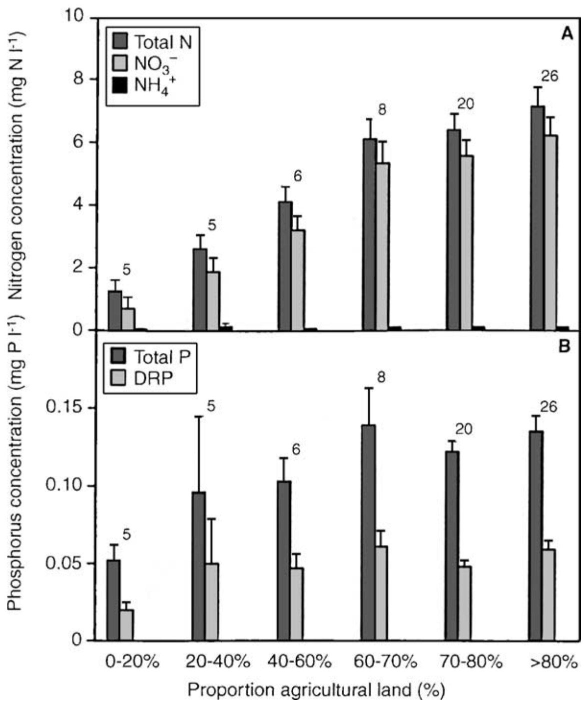

Fig. 3. Average annual flow-weighted concentrations of total N, nitrate and ammonium (A) and total phosphorus and dissolved reactive phosphorus (B) in streams draining catchments with an increasing proportion of agricultural land.

Table 5
Mean trends in total nitrogen concentrations and total phosphorus concentrations in streams draining sandy and loamy catchments with almost absence of point sources (N: <0.5 kg N ha^{-1}; P: <0.025 kg P ha^{-1}) during the period 1989–2002

	Proportion of agricultural land in the catchment			
	0–40%	40–70%	70–80%	80–100%
Total nitrogen				
Sandy catchments	−17.4% (8.1%) ($n=8$)	−23.0% (3.8%) ($n=8$)	−28.1% (4.6%) ($n=12$)	−33.0% (5.0%) ($n=13$)
Loamy catchments	−23.4% (17.0%) ($n=2$)	−34.8% (6.4%) ($n=6$)	−35.3% (4.1%) ($n=8$)	−32.6% (3.8%) ($n=13$)
Total phosphorus				
Sandy catchments	4.0% (9.2%) ($n=8$)	5.1% (9.1%) ($n=5$)	−21.7% (7.3%) ($n=5$)	0.4% (7.5%) ($n=9$)
Loamy catchments	−9.3% (17.9%) ($n=2$)	−23.9% (11.0%) ($n=3$)	−12.5% (7.6%) ($n=4$)	−12.9% (5.2%) ($n=9$)

The non-parametric Seasonal Mann–Kendall test was applied with flow-adjustment of the observed nutrient concentrations (LOWESS). The trend results cover 4 classes of catchments depending on their proportion of agricultural land and two dominant soil types. The standard error of the mean is given in parenthesis and the number of streams in each class is shown beneath the trend result.

Danish catchments may be due to the combination of great reductions in fertilizer use and improved use of animal manure, and intensive crop production with high yields.

The N/P ratio for both TN/TP and DIN/DRP has decreased in streams draining undisturbed and agricultural catchments during the period 1989–2002, whereas no change was detected in larger streams receiving nutrients from point sources (Table 6). The changes observed in N/P-ratios in undisturbed catchments may be caused by a variety of factors including a decrease in atmospheric deposition of nitrogen and/or increased runoff between the three periods compared, as this would increase erosional P-losses more than N-leaching. The observed decrease in the N/P-ratio in streams draining agricultural catchments must be attributed to the downward trend in N-losses from agricultural land. Thus, measures taken to combat nutrient pollution will alter the nutrient stoichiometry with potential seasonal changes in nutrient limitation in aquatic environments (e.g. Downing, 1997; Conley, 1999).

3.5. Nutrient impacts in lakes

During the period 1989–2002, the monitored lakes responded to the nutrient loading reduction by significant reductions in median and mean TP, TN and phytoplankton biomass (expressed as chlorophyll *a*) and increased water transparency (Fig. 4). The lower phytoplankton biomass can be attributed to lower nutrient input and reduced internal loading (Fig. 5; Søndergaard et al., 2002) and thus to enhanced resource control. The reduction in TP and chlorophyll *a* was first observed in spring and autumn, and later in summer as well, for most of the lakes included in the data set (shallow lakes) (Jeppesen et al., in press a; Søndergaard et al., in press). This response indicates that internal loading primarily declines during cold periods rather than in late summer in the early recovery phase. This is partly due to a gradually reduced exchangeable P pool in the sediment and improved redox conditions occurring concurrently with a reduction in algal sedimentation (Søndergaard et al., in press). The duration of the period with excess

Table 6
Changes in average annual TN/TP-ratio and DIN/DRP-ratio (standard error of the mean) in streams draining three different catchment types in Denmark calculated for three periods: 1989–1993, 1994–1998 and 1999–2002

Period	1989–1993 ($n=7$)		1994–1998 ($n=48$)		1999–2002 ($n=86$)	
Catchment types	TN/TP	DIN/DRP	TN/TP	DIN/DRP	TN/TP	DIN/DRP
Streams draining undisturbed catchments	56 (4)	174 (25)	54 (2)	130 (11)	42 (2)	109 (3)
Streams draining agricultural catchments without major point sources	81 (8)	177 (23)	73 (3)	159 (9)	59 (1)	136 (9)
Streams draining catchments with point sources	38 (5)	81 (11)	41 (2)	86 (6)	39 (1)	89 (2)

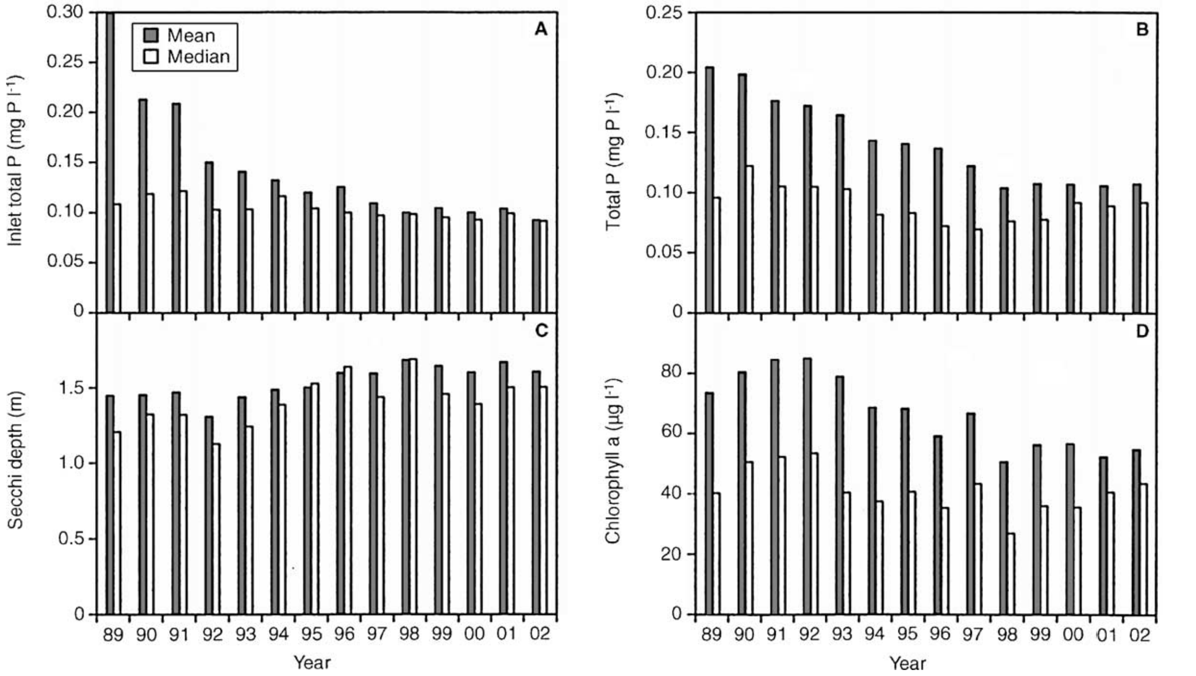

Fig. 4. Changes in average annual inlet total P concentration (A), in-lake total phosphorus concentration (B), Secchi depth (C) and chlorophyll *a* (D) in 27 Danish lakes during the period 1989–2002.

internal loading after P loading reduction varies among lakes depending on the extent and duration of the period with high loading, and on retention time and depth (Jeppesen et al., 1991; Søndergaard et al., 2001). A recent meta-analysis of 35 European and North American case studies has shown that a new equilibrium adapted to the lower loading generally occurs after 10–15 years (Jeppesen et al., in press b); however, longer response times have been recorded in several cases.

While the decline in phytoplankton biomass and the increase in transparency are clearly associated with the decline in nutrients, cascading effects from the top of the food web have most likely been a contributory factor. Supporting this, test-fishing with multiple mesh-sized gill nets have revealed a shift from dominance by cyprinids (especially bream, *Abramis brama*, and roach, *Rutilus rutilus*) to larger quantitative importance of particularly perch (*Perca fluviatilis*). Also, the CPUE of more littoral species, such as tench (*Tinca tinca*), rudd (*Scardinius erythrophthalmus*) and pike (*Esox lucius*), rose importantly during the study

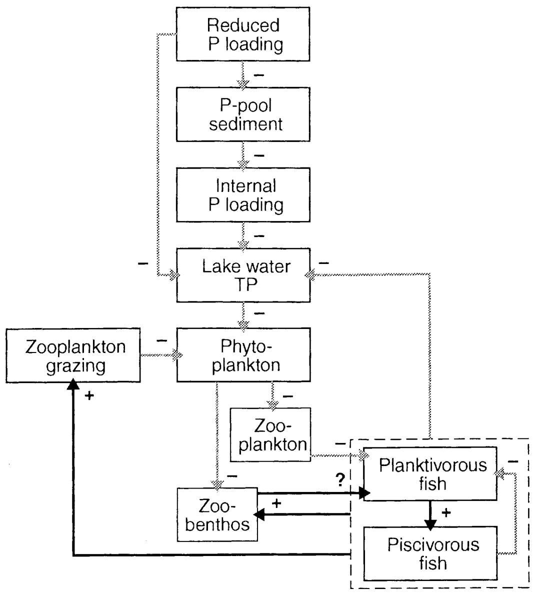

Fig. 5. Conceptual diagram illustrating the influence and cascading effects of decreased nutrient loading on chemical and biological variables in lakes.

period. The contribution of potential piscivores (pike, perch and pikeperch, *Stizostedion lucioperca*) increased in lakes with major alterations in summer mean TP, thereby enhancing top-down control on benthi-planktivorous fish (Jeppesen et al., in press a). The changes occurred over a 5- to 10-year period, which indicates that fish may respond rapidly to lower nutrient levels and that these changes largely follow the trajectory known from lakes undergoing eutrophication. With reduced abundance of planktivorous fish, predation on zooplankton decreased, as expected, and, accordingly, the zooplankton:phytoplankton ratio increased as did the grazing on phytoplankton. Thus, it can be concluded that changes at the top of the food web (fish) also affect phytoplankton biomass in the recovery phase (Fig. 5). The meta-analysis (Jeppesen et al., in press b) also showed pronounced changes in fish biomass expressed as a reduction in benthi-planktivorous fish biomass and an increase in the percentage of piscivores, and often in the zooplankton:phytoplankton ratio as well, thus confirming the findings from Danish lakes.

3.6. Nutrient impacts in estuaries and coastal waters

Long-term changes have occurred in nutrient loading and nutrient concentrations in Danish estuaries and coastal waters, rapid increases occurring due to the intensification of agriculture after World War II (Richardson, 1996). Reconstruction of nutrient concentrations using diatom-based transfer functions has shown that TN concentrations in Roskilde Fjord ranged from 701–813 μg N l^{-1} during the time period 1850–1950, with rapidly increasing TN concentrations; today, the average level is ca. 1260 μg N l^{-1} in the mid-1990s (Clarke et al., 2003). Concerted efforts have been made to reduce N loading to the marine environment (Kronvang et al., 1993; Conley et al., 2002), and significant declines in dissolved inorganic nitrogen concentrations (DIN = nitrate + nitrite + ammonium) and TN concentrations have recently been observed in the marine environment (Rasmussen et al., 2003), when interannual variations in the runoff were taken into account.

Significant reductions in dissolved reactive phosphorus (DRP) and TP concentrations have been reported in estuaries and coastal areas throughout Denmark following P loading reductions. Roskilde Fjord, in particular, had some of the highest P concentrations in Denmark (Conley et al., 2000), but after the construction of a combined advanced sewage treatment plant significant reductions in nutrient concentrations have been observed (Fig. 6). Prior to P reductions, summer DRP concentrations in surface waters ranged between ca. 620–930 μg P l^{-1} due to both excessive nutrient loading and high rates of internal loading from the sediment during the warm summer months. However, following P removal between the years 1991–1995, summer DRP values greatly decreased by nearly a factor 2–3 and are ca. 310 μg P l^{-1} today.

Nutrient concentrations in Danish estuaries generally follow the annual cycle of phytoplankton production, with the lowest DIP concentrations usually being observed during the spring bloom and the lowest DIN concentrations occurring during the

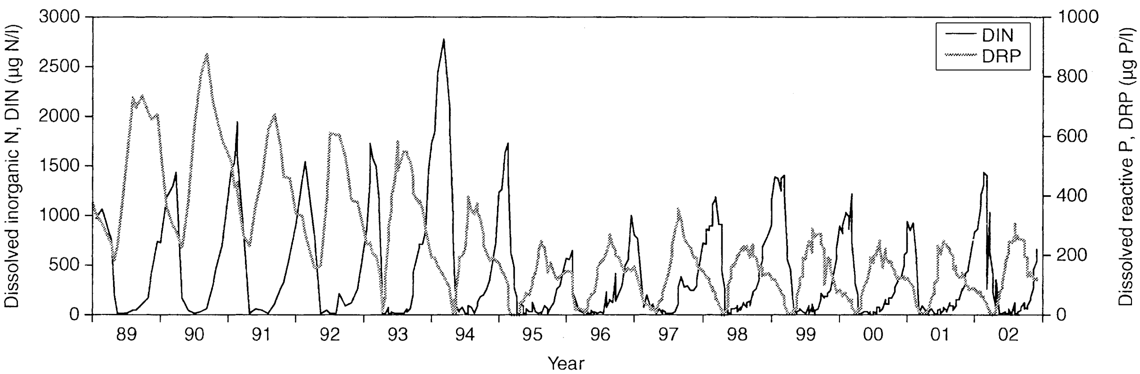

Fig. 6. Dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP) concentrations in surface waters at station 60 in Roskilde Fjord, Denmark, from 1989 through 2002.

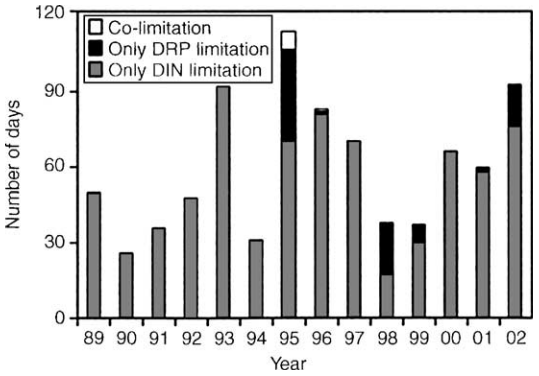

Fig. 7. Potential nutrient limitation of DIN, DRP and co-limitation by both DIN and DRP during the productive period from March-September in surface waters at station 60 in Roskilde Fjord, Denmark, from 1989 through 2002.

warmer summer months (Conley et al., 2000). DRP concentrations less than 6.2 μg P l^{-1} and DIN concentrations less than 28 μg N l^{-1} can be used as indicator concentrations that may potentially limit nutrient concentrations for phytoplankton growth (Fisher et al., 1992). In Danish estuaries, the number of days with potential nutrient limitation has increased following nutrient reductions (the development in Roskilde Fjord is depicted in Fig. 7). In addition, the number of days of potential limitation by DRP has increased tremendously both in Danish estuaries in general (Conley et al., 2002) and in Roskilde Fjord in particular (Fig. 7). DRP limitation occurs primarily in the spring when concentrations are low, switching to DIN limitation during the summer. This pattern of switching nutrient limitation between DRP in spring and DIN in summer is common to estuaries in general (Conley, 1999). Due to the large reductions in TP loading with the building of advanced sewage treatment plants, DIP limitation is common in Danish estuaries (Rasmussen et al., 2003).

On a nationwide basis sharp declines in yearly mean total P and DRP concentrations were first observed in 1991 in estuarine and coastal marine areas (Fig. 8). Low DIN and total N concentrations were observed during the low freshwater flow years of 1996 and 1997 and lower N concentrations have been observed thereafter. These two dry years appeared to be the triggering mechanism for N concentrations to decline (Rasmussen et al., 2003). However, declines in N concentrations are partly masked by large interannual variations in freshwater discharge. When interannual variations in freshwater discharge are accounted for (Rasmussen et al., 2003), nitrogen concentrations in estuaries and coastal waters have decreased by up to 44% in the last 5 years (1998–2002), although no reductions have been observed in the open waters around Denmark.

Despite the fact that P loading has been reduced by nearly 90% with subsequent decreases in DRP levels in estuaries and coastal waters of Denmark and the recently observed reductions in N concentrations, there have been few significant effects on the ecosystem. In Roskilde Fjord, no long-term trends of water quality variables (oxygen depletion,

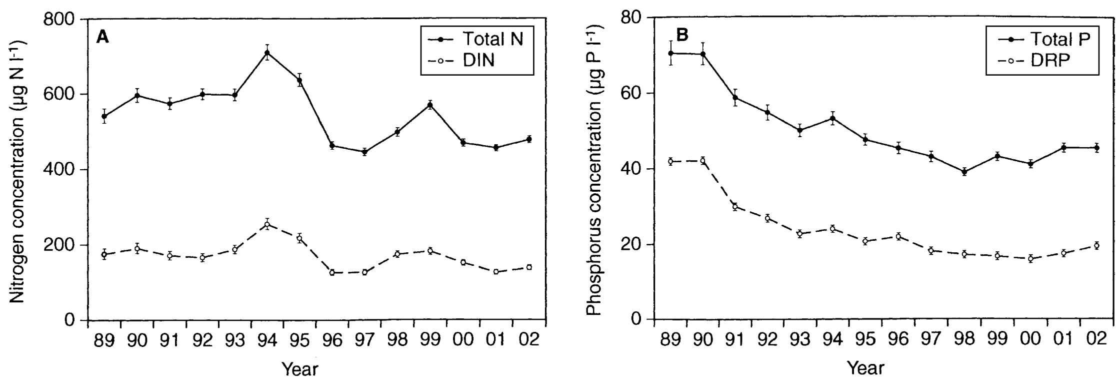

Fig. 8. Yearly mean nutrient concentrations from 1989–2002 for estuarine and coastal stations. A. Dissolved inorganic nitrogen concentrations (DIN) and total nitrogen (TN) concentrations. B. Dissolved reactive phosphorus (DRP) and total phosphorus (TP) concentrations. Error bars show the 95% confidence limits of the mean values.

chlorophyll, primary production, phytoplankton biomass, macrophytes and benthic fauna) have occurred, despite the tremendous reduction in TP loading and DRP concentrations, although there has been some reductions in macrophyte abundance especially that of *Ulva* (Conley, 1999).

Although significant relationships between nutrient loading and/or nutrient concentrations with biological indicators are difficult to discern in individual Danish estuaries, significant relationships have been observed on a national basis. For example, significant linear relationships were found for Danish estuaries between Secchi depth (a parameter for water clarity) and TP loading ($r^2=0.59$) and for TN loading ($r^2=0.44$) (Conley et al., 2002). For the open waters around Denmark a significant relationship was found only between Secchi depth and TN loading ($r^2=0.71$), but not TP loading (Conley et al., 2002). Given that more variance was explained by TP loading than by TN loading in estuaries, these data suggest that TP loading was more important in Danish estuaries and that TN loading is more important for light penetration in the open waters around Denmark.

Finally, given that the rate of nutrient loading to Danish estuaries and coastal areas ranks among the highest in the world, it is surprising that average chlorophyll *a* in Danish estuaries is only $8\ \mu g\ l^{-1}$ during summer (Conley et al., 2000). The low mean chlorophyll levels are due to grazing by the blue mussel *Mytilus edulis* and the sea squirt *Ciona intestinalis* that are known for their immense filtration capacities. Potential filtration rates calculated for Danish estuaries during summer are commonly 2–3 times that of the estuary volume, thus explaining the low chlorophyll levels observed in Danish estuaries (Rasmussen et al., 2003).

4. Conclusions

A thorough assessment of a 14-year series of monitoring data from streams, rivers, lakes and estuaries has shown that major changes have occurred following the adoption of several Action Plans to combat nutrient pollution of the Danish aquatic environment. A major reduction of point source discharges to Danish freshwater, estuaries and coastal marine waters has been achieved, amounting to 69% for TN and 82% for TP during the period 1989–2002. During the same period, the Action Plans have resulted in major reductions in the annual net input of N to agricultural land decreasing by 41%, from 148 to $88\ kg\ N\ ha^{-1}$, during 1985–2002. A statistical non-parametric trend analysis shows a significant downward trend in 48 streams draining agricultural catchments without major point sources during 1989–2002. The downward trend became more evident with increasing proportions of agricultural land in the catchment and was more pronounced for loamy than for sandy catchments. In contrast, no significant trends could be detected for TP concentrations in streams draining agricultural catchments.

Danish lakes have responded to a major nutrient (particularly P) loading reduction. This has led to an increase in water transparency, lower algal biomass, lower biomass of plankti-benthivorous fish, and an increased percentage of piscivores fish. The improved clarity can be attributed to both enhanced resource control (less input of nutrients, reduced internal loading) and enhanced top-down control by zooplankton mediated by changes in the fish community.

In contrast, only few significant ecological effects on Danish estauries and coastal marine ecosystems have been observed with the reductions in point source nutrient loadings. The main reason that total riverine loading of nutrients has not decreased significantly during the same period is probably due to higher runoff and increased non-point loss of both N and P from agricultural areas. Short-term and long-term changes in the climatic conditions may delay and potentially counteract the measures adopted to decrease non-point nutrient pollution of aquatic ecosystems. Other factors like inertia in catchment responses to management measures and resilience in lakes and estuaries due to sediment P-release may also delay the full effects of Action Plans. Catchment managers should be aware of the impacts of climate change on nutrient losses, inertia in catchments and resilience in water bodies for nutrient management measures when implementing River Basin Management Plans under the European Water Framework Directive to achieve a good ecological quality in water bodies in 2015.

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