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Nitrogen in the Environment: Sources, Problems, and Management
J.L. Hatfield and R.F. Follett (Eds)

Chapter 9. Nitrogen Transport and Fate in European Streams, Rivers, Lakes, and Wetlands

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This chapter provides an overview of the present state of nitrogen (N) pollution in European streams, rivers, and lakes. The main focus of this chapter is on diffuse sources of N. Diffuse sources are today the main concern in many European catchments, and measures need to be developed to protect drinking-water supplies and maintain the environmental quality of rivers, lakes, and coastal waters. In 1991, the European Union (EU) adopted the Nitrate Directive aimed to reduce or prevent nitrate pollution of water due to application and storage of inorganic fertilizers and manure on farmland. The EU countries have identified zones vulnerable to nitrate pollution, and most countries have adopted action plans to reduce N pollution. The newly adopted EU Water Framework Directive (WFD) aims at protecting different surface water bodies to prevent further deterioration and to protect and enhance the status of aquatic ecosystems (European Parliament, 2000). The implementation of the WFD involves different steps where River Basin Authorities shall: (i) perform an analysis of pressures and impacts and develop monitoring programs (before 2007) and (ii) implement mitigation strategies in the form of River Basin Management Plans (before 2009). Thus, throughout Europe, catchment managers are combating N pollution from both point and diffuse sources, a task requiring thorough knowledge of the mechanisms governing N loss arising from different uses of land, as well as of the fate of N in groundwater and surface waters. Based on examples from different European catchments and comprehensive datasets gathered from ongoing nutrient monitoring programs in Denmark and the Netherlands, this chapter illustrates the most important aspects to be considered.

This chapter is dedicated to our great friend and collaborator Jens Peder Jensen, who died in March 2006.

1. INTRODUCTION

Elevated nitrogen (N) concentrations in European surface waters have mostly been related to modern agricultural practices, in particular, the use of N fertilizers (Neill, 1989; Edwards et al., 1990; Wright et al., 1991; Stibe and Fleischer, 1991; Kronvang et al., 1995). However, in some countries and catchments, N discharge from point sources such as sewage treatment plants and industry still contributes significantly to riverine N loading (Kristensen and Hansen, 1994; Iversen et al., 1997; Bøgestrand et al., 2005).

The elevated riverine N loading has been associated with increased primary production and nuisance algal growth in coastal zones and semi-enclosed and enclosed areas of European seas (Mee, 1992; OSPARCOM, 1992; Kronvang et al., 1993; Conley et al., 2002). Examples of the consequences of eutrophication are increased frequency of algal blooms (sometimes toxic), increased water turbidity, oxygen depletion in deeper waters, and mass kills of fish and benthic fauna (Kronvang et al., 1993; European Environment Agency, 1995).

The estimated European riverine gross flux of N amounts to 2.5–6.5 million metric tons per year (European Environment Agency, 1995). Part of the riverine N loading is, however, removed during its passage from source to the open sea (Kronvang et al., 1999; Kronvang et al., 2005). Knowledge of the fate of the riverine N transport (mostly as nitrate) is important for allowing accurate estimations of N emissions and the resulting net escape to the open sea. Hence, quantification of the transformation of nitrate under anoxic conditions into N_2O and N_2 gases in rivers, lakes, wetlands, and estuaries is an important issue.

2. PROBLEM IDENTIFICATION

2.1. Nitrogen Sources in European Catchments

Around 2000, the input from agriculture to the total emission of N to the aquatic environment in nine larger river basins in Europe ranged between 44% and 64%, the highest values being recorded in the drainage basin to the North Sea and the lowest in the Daugava river basin. The remaining N sources are point source emissions and N emissions from undisturbed land (forest, mountains, and tundra). The relative importance of the different N emissions in different European river basins is illustrated in Table 1. In densely populated regions, point source discharge of N contributes significantly to the riverine N loading (e.g., River Odra, Poland/Germany and River Po, Italy). In intensively agri-cultivated regions such as in the North Sea Basin, diffuse N loss from arable land is the dominant N source, whereas diffuse N loss from undisturbed land dominates in sparsely populated and cultivated regions such as the Daugava river draining parts of Belarus, Lithuania, and Latvia.

2.2. Nitrogen Consumption in European Agriculture

Today, more than 40% of the European land area is used for agricultural production (Table 2). The land use varies, however, much from country to country, and may

Table 1.
Sources of total nitrogen for the nitrogen loading of major European water bodies (Bøgestrand et al., 2005).

Region	Period	Catchment area (km²)	Annual loading (kg N/ha)	Point sources (%)	Anthropogenic diffuse losses (%)	Background (%)
Baltic Sea	2000	1,600,000	5.4	14	56	30
North Sea	2000	530,000	14.4	36	64	10
Danube	1999	817,000	8.6	31	45	23
Vistula	1999	194,000	9.6	22	59	19
Rhine	1999	185,000	28.7	31	54	14
Elbe	1999	148,000	15.5	34	55	12
Odra	1999	119,000	10.5	43	49	9
Daugava	1999	88,000	6.8	13	44	43
Po	1999	69,000	35.6	36	54	10

Table 2.
Distribution of land use in Europe including the European part of the Russian Federation (Veldkamp et al., 1995).

Type of land use	Area (km²)	Percentage
Arable land	3,343,838	24.3
Grassland	1,513,557	15.7
Permanent crops	144,508	1.5
Coniferous + mixed forest	2,488,327	25.8
Deciduous forest	664,587	6.9
Urban areas	117,887	1.2
Inland waters	203,196	2.1
Other (mountain, tundra, etc.)	2,175,516	22.5

even vary from region to region within each individual country. For example, arable land constitutes about 59% of the total land area in Denmark, 40% in Spain and Italy, 34% in Germany, 24% in The Netherlands, 6% in Sweden, and 3% in Norway. Similarly, farming intensity and crop types vary from region to region. The amount of arable land per inhabitant in Europe (0.38 ha) is only half that found in the United States of America (0.76 ha) and only slightly higher than in Africa (0.27 ha).

The N surplus per hectare agricultural land increased in Europe with 75% between 1970 and 1989, followed by a decrease during the 1990s in most European countries (Figure 1). The socioeconomic and political reforms in the Eastern

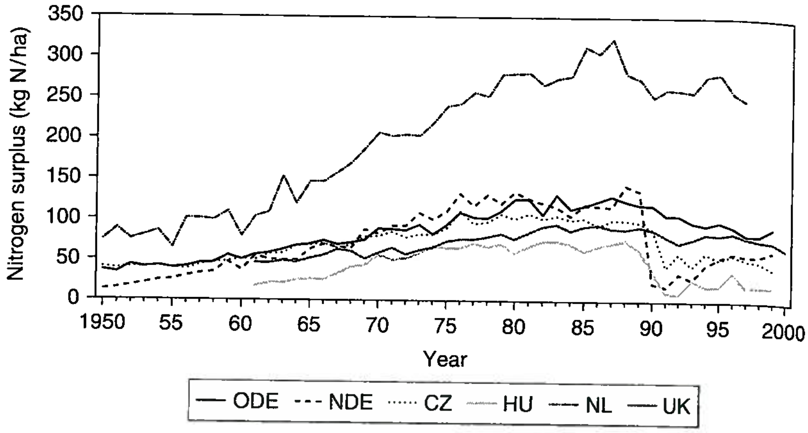


Figure 1. Changes of the nitrogen surplus in agriculture of different European countries from 1950 to 1999 (from Behrendt, 2004).

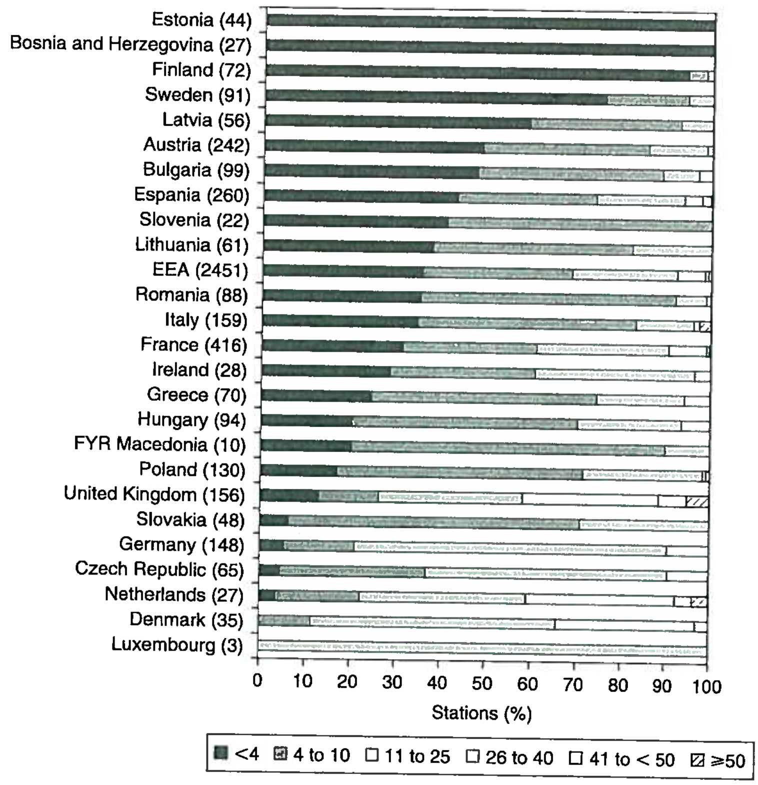


Figure 2. Concentrations of nitrate (mg nitrate per liter) at monitoring stations in European rivers shown in concentration classes. The number of monitoring stations within each country is given in the bracket following the country name. (Data from European Environment Agency, 2005b)

European countries in the late 1980s resulted in a strong decrease in the N surplus which is now steadily rising as it is the case for the former eastern part of Germany (Figure 2). However, large variations in the N surplus are found, the highest values being recorded in the north-western part of Europe and the lowest in the southern and eastern part of Europe (Table 3). The differences in N surplus varies widely between European countries, the highest values occurring in Belgium and The Netherlands (>200 kg N/ha) and the lowest – and less than 40 kg N/ha – in northern or southern European countries (Table 3).

Table 3.
Average nitrogen surplus in different European countries in 1990 and 2000 calculated as balances of inputs (mineral fertilizers, manure, biological fixation, and atmospheric deposition) and outputs (harvested crops).

	1990	2000
	(Kg N/ha)	
Belgium	264	224
Netherlands	263	226
Luxembourg	167	117
Germany	147	105
Greece	83	69
Denmark	80	77
Finland	74	51
United Kingdom	63	45
France (EEA)	55	39
Sweden	52	38
Austria	48	43
Italy	44	37
Portugal	43	42
France	36	25
Ireland	36	44
Spain	27	39
EU-15	65	55

Source: European Environment Agency, 2005a.

2.3. Nitrogen Concentration and Trends in European Rivers

Descriptive statistics for the concentration of total-N, nitrate, and ammonium in European rivers is shown in Table 4. Dissolved inorganic N (nitrate and ammonium) constituted 88% of the total-N concentration in river water. The median total-N concentration in 43 near pristine European rivers was much lower (0.33 mg N/L) than the

Table 4.
Descriptive statistics of the annual mean nitrogen concentration in European rivers.

	Number of river stations	Percentage of river stations with concentrations not exceeding (mg N/L)					
		Mean	10%	25%	50%	75%	90%
All rivers							
Total-N	329	3.07	0.30	0.80	2.12	4.50	7.07
Nitrate-N	654	2.63	0.25	0.70	1.80	3.90	5.72
Ammonium-N	580	0.67	0.03	0.07	0.18	0.45	1.42
Near pristine rivers							
Total-N	43	0.40	–	0.19	0.33	0.39	–
Nitrate-N	39	0.30	–	0.05	0.10	0.22	–

Redrawn from Kristensen and Hansen, 1994.

median total-N concentration reported for all 329 European rivers (2.12 mg N/L), indicating an anthropogenic influence from, for instance, agriculture and sewage effluents. The concentration of nitrate in rivers in different European countries is shown in Figure 2 for the year 2002. The number of stations with high nitrate concentrations (>25 mg nitrate per liter) is generally highest in the western European countries (The Netherlands, Denmark, United Kingdom), whereas countries with a dominance of monitoring stations with low nitrate concentrations (<10 mg nitrate per liter) are generally lying in northern and eastern Europe (Figure 2).

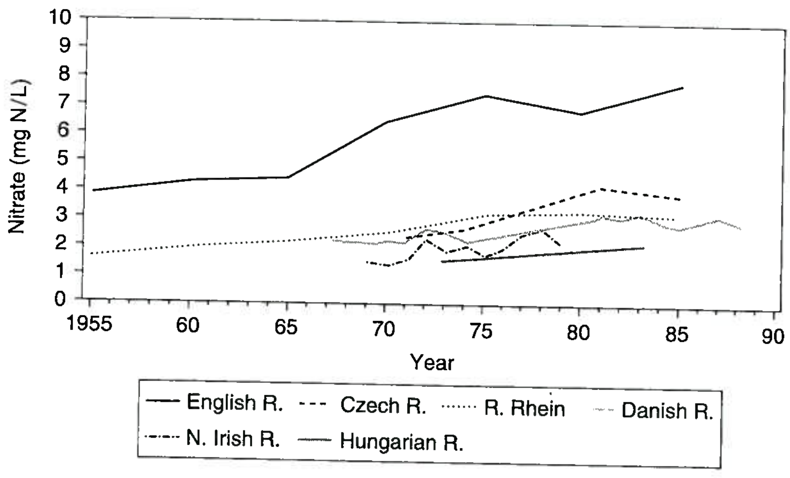
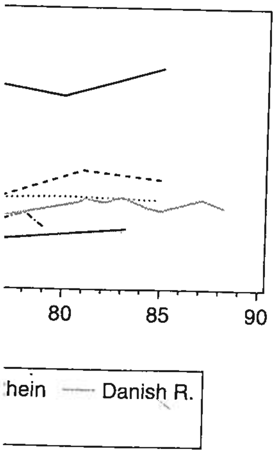


Figure 3. Trends in the nitrate concentration of selected European rivers. (Redrawn from Kristensen and Hansen, 1994).

oncentration in European rivers.

f river stations with ns not exceeding (mg N/L)			
25%	50%	75%	90%
0.80	2.12	4.50	7.07
0.70	1.80	3.90	5.72
0.07	0.18	0.45	1.42
0.19	0.33	0.39	–
0.05	0.10	0.22	–

European rivers (2.12 mg N/L), nstance, agriculture and sew- in different European countries er of stations with high nitrate ighest in the western European om), whereas countries with a centrations (<10 mg nitrate per e (Figure 2).



ed European rivers. (Redrawn

The concentration of nitrate-N has increased in many European rivers, especially during the period 1950–1990 (Figure 3). The increase in riverine nitrate-N concentrations is mainly attributable to a corresponding increase in the N surplus in most European countries (Figure 1). Also, during the same period, enhanced emission of N from the burning of fossil fuels and agricultural activity has, however, increased the atmospheric dry and wet deposition of N. This has mainly resulted in an increase in riverine N concentrations in remote and sparsely populated areas of Europe.

On average, 70% of European rivers experienced an increase in nitrate concentrations between 1978–1988 and 1988–1990 (Kristensen and Hansen, 1994). The increase was most pronounced in eastern and southern Europe, possibly because the use of N fertilizers peaked later here than in the north-western European countries (European Environment Agency, 1995). Since the early 1990s, a general downward trend in riverine nitrate-N concentrations has been recorded in many European countries, being most widespread for monitoring stations in Denmark, Czech Republic, and Germany (Figure 4). The reason for this pattern is because Denmark

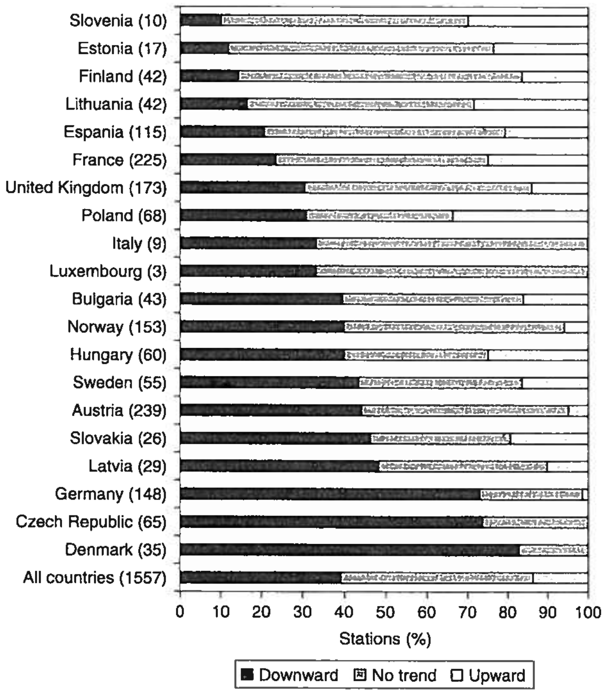


Figure 4. Recent trends (1992–2001) in nitrate concentrations (mg nitrate per liter) in rivers in different European countries. The number of monitoring stations within each country is given in the bracket following the country name. (Data from European Environment Agency, 2005b).

has many regulations on the use of N in chemical fertilizer and manure adopted by the Danish Government (Andersen et al., 1999). Since 1990, the economic change in many eastern European countries has also led to a strong decline in the use of N fertilizers. A concomitant decrease in riverine nitrate concentrations can also be seen in some of these countries (e.g., Latvia), whereas the nitrate concentration is only decreasing in a few rivers in Lithuania.

3. SOURCES AND RETENTION OF NITROGEN IN A LARGE RIVER – THE RHINE

The catchment area (186,000 km^2) of the Rhine (length 1,220 km) covers parts of nine countries with a total of 50 million inhabitants. The average discharge at the Dutch–German border is 2,300 m^3/s (Figure 5). Table 5 summarizes the land use structure of the catchment.

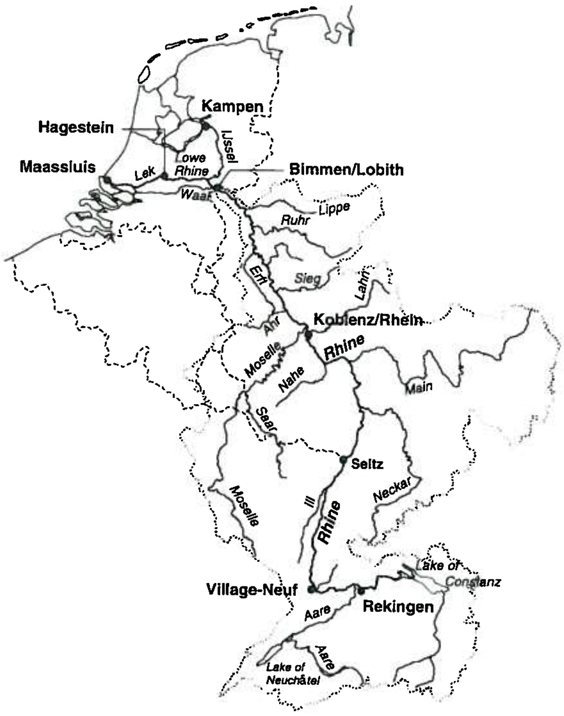


Figure 5. Catchment area of the Rhine showing tributaries and sampling stations.

Table 5.
Land use in the Rhine catchment.

						Agricultural area			
	Total area[1]	Urban area		Forest area		Arable land		Grass land	
Country	×1000 km^2	×1000 km^2	%	×1000 km^2	%	×1000 km^2	%	×1000 km^2	%
CH	34	3	9	11	33	5	15	14	41
D	102	22	22	40	39	27	26	13	13
L	2.6	0.3	12	0.9	35	0.5	21	0.7	27
F	23	1.3	6	8.8	40	5.9	27	5.0	23
NL	24	7	31	2	9	4	16	9	36
TOTAL	188	34	18	63	34	42	23	42	22

CH: Switzerland; D: Germany; L: Luxembourg; F: France: NL: The Netherlands.
[1]Note that total area does not equal the sum of urban, forest, and agricultural areas. This is due to both uncertainties in area estimates and the fact that other, less-important forms of land use is not included in the table.

Concentrations of N-compounds are measured at several stations along the Rhine (Figure 5). Especially for Lobith, long time series are available. The average concentrations vary along the length of the river. In 1993, the measured concentrations ranged from 1.7–5.1 mg total-N per liter. The highest concentrations were measured downstream in the Netherlands.

The concentrations also vary with time and are partly correlated with discharge (Van Dijk et al., 1996). Annual average concentrations are calculated as the discharge-weighted means. Total-N concentrations measured at Lobith increased to about 7.5 mg N/L (1970–1975) followed by a decrease to 3.4 mg N/L (2001–2005) (Figure 6). The similar trend is also observed in the case of the concentration of nitrate-N in the river Rhine (Figure 6).

The elevated N concentrations in the Rhine originate from emission. Estimates of the most important sources of N for the year 2004 are summarized in Table 6. Agriculture and domestic wastewater are the most important sources of N. The total emission in 1992 was about 470 ktons N per year; 427 ktons was emitted upstream of Lobith. The same year, the riverine loading at Lobith amounted to 303 kton N. The most important reasons for the difference are probably retention in the catchment and errors in emission estimations.

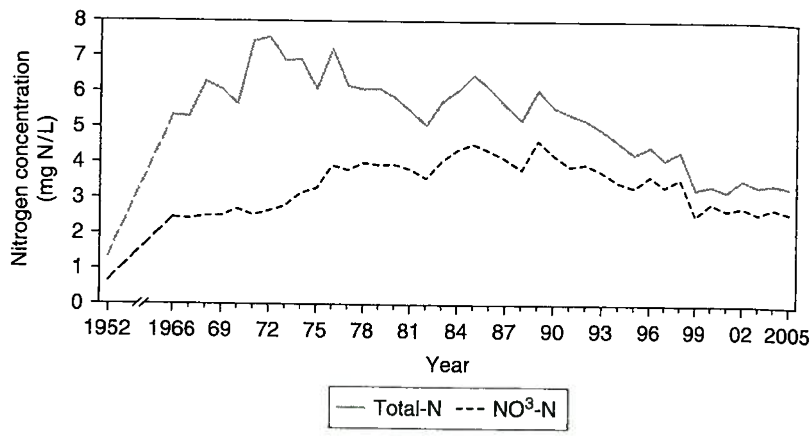


Figure 6. Annual average concentrations of total-N and nitrate-N in the Rhine at Lobith from 1960 to 2005.

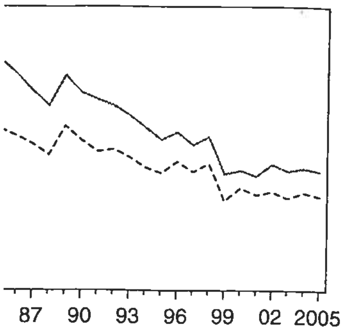
Table 6.
Estimates of diffuse and point sources of nitrogen for the year 2004 in the river Rhine.

Nitrogen emissions (kton N/year)					
WWTPs	Industry	Agriculture	Other diffuse sources	Atmospheric deposition	Total
27	4	68	2	7	108

4. NITROGEN CYCLING IN SMALL CATCHMENTS

The N cycling in 21 small and predominantly arable catchments was investigated during the period 1989–1996 (Figure 7). The monitored catchments are representative of the typical Danish farming systems ranging from plant production on the predominantly loamy soils on the islands to animal production on the sandy soils in Jutland. The catchments also represent the typical soil types of Denmark and the gradients in climate and hydrology (Table 7). Agricultural practices at field level were investigated by a questionnaire survey conducted in 1993/1994. The data collected were used for calculating N leaching from the root zone applying an empirical leaching model (Andersen et al., 1999).

Nitrogen leaching from the root zone on agricultural areas was calculated for all 21 catchments both for 1993/1994 and the 7-year period, 1989–1990 to 1995–1996. The latter was calculated by assuming same agricultural practice as in 1993–1994 and letting climate vary from year to year in each of the catchments. Measurements of total-N concentrations and discharge in the stream draining each catchment were conducted



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for the year 2004 in the river

diffuse :s	Atmospheric deposition	Total
	7	108

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tly arable catchments was investi- [T]he monitored catchments are rep- ms ranging from plant production to animal production on the sandy the typical soil types of Denmark e 7). Agricultural practices at field /ey conducted in 1993/1994. The ng from the root zone applying an

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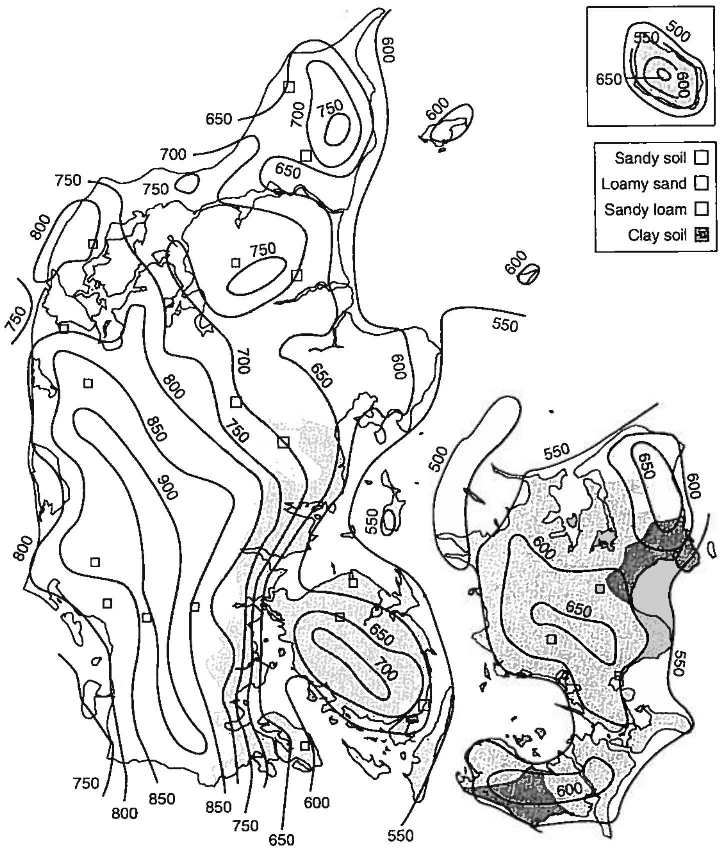

Figure 7. The 21 small catchments incorporated as parts of the Danish Aquatic Monitoring Programme are situated in different parts of Denmark and cover the dominant soil types and gradients in precipitation.

during the whole study period, enabling us to calculate the N loss from the catchment. Moreover, the annual N loss from agricultural areas within the catchments was calculated by applying measured annual N losses from nonagricultural areas within the catchments (Kronvang et al., 1995, 1996). Five of the 21 catchments have been monitored up to recent years (1989–2005) and two of these catchments are representing sandy regions and three for the catchments loamy regions of Denmark.

4.1. Nitrogen Budgets

The monitoring data from five small catchments enable us to establish an overall N budget for the catchments situated in the sandy western region and the loamy eastern region for the period 1998/1999 to 2002/2003 (Figure 8). The N balance reveals that the application and surplus of N is highest on agricultural areas within the sandy

Table 7.
Description of catchment area, soil type, runoff, and hydrological regime of the two classes of catchments situated in the sandy western and loamy eastern regions of Denmark.

	Sandy western region of Denmark	Loamy eastern region of Denmark
Number of catchments	10	11
Average catchment area	12.6 km^2	13.6 km^2
Average proportion of sandy soils	86%	25%
Average proportion of organic soils	7.6%	1.1%
Average annual runoff	289 mm	242 mm
Baseflow index	0.74	0.55
Average annual interflow[1]	30%	55%
Average annual groundwater flow[1]	70%	45%

[1]Calculated by setting up a precipitation-runoff model for each catchment.

western catchments, mainly due to higher production and use of N in animal manure (Figure 8). The higher N surplus for the sandy catchments than the loamy catchments is also found in the modeled average annual N leaching from the root zone being higher for the sandy (85 kg N/ha) than for the loamy catchments (53 kg N/ha). The N retention in the root zone seems to be of greater significance in the loamy catchments than in the sandy catchments possibly because of high soil water content in the latter.

The average N export from the agricultural areas within the catchments measured at stream monitoring stations was considerably lower for the sandy catchments (12 kg N/ha) than from the loamy catchments (23 kg N/ha). For both the sandy and loamy catchments, average N export from the agricultural areas within the catchments was lower than the modeled average N leaching (Figure 8). The difference obtained can be ascribed to both subsurface N-removal processes (denitrification) or, during this short-term period, may be also to hydrological inertia (time lag from N in groundwater) within the catchments. The latter presumably being most pronounced in sandy catchments where the proportion of stream water derived from groundwater is highest (Table 7).

4.2. Subsurface Nitrogen Removal

The year of 1994 was a relatively wet year (mean annual precipitation 880 mm) compared to the average (694 mm) for the 7-year period, 1989–1996. Therefore, both the modeled N leaching from the root zone on agricultural land and the measured N

f, and hydrological regime of the
' western and loamy eastern regions

region of	Loamy eastern region of Denmark
	11
	13.6 km^2
	25%
	1.1%
	242 mm
	0.55
	55%
	45%

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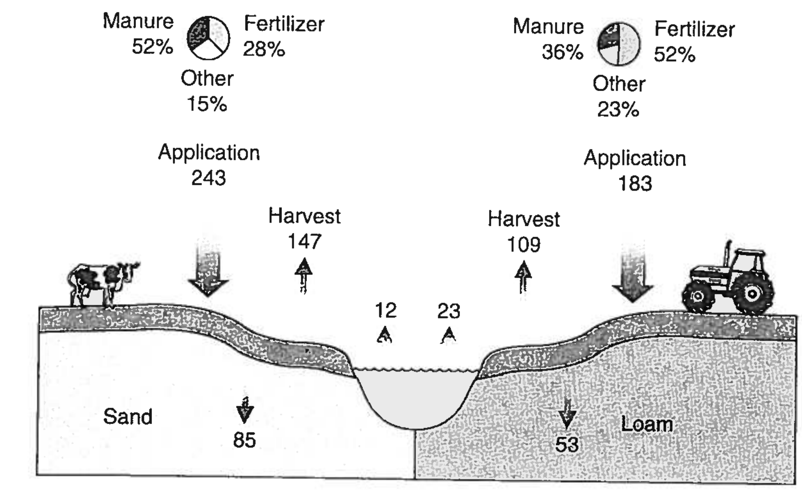

Figure 8. The average nitrogen input balance and nitrogen flows for 2 sandy catchments and 3 loamy catchments in Denmark from 1998/1999 to 2002/2003.

export from the agricultural areas within the catchments were considerably higher than the average for the period 1989–1996. To obtain a better description of the missing link between N leaching and N export, we investigated the differences for the period 1989–1996. Average subsurface N retention (measured as the difference between N leaching and N export divided by the former) was significantly ($P < 0.001$) correlated with the proportion of sandy soils (S) and average runoff in the 21 catchments (Figure 9).

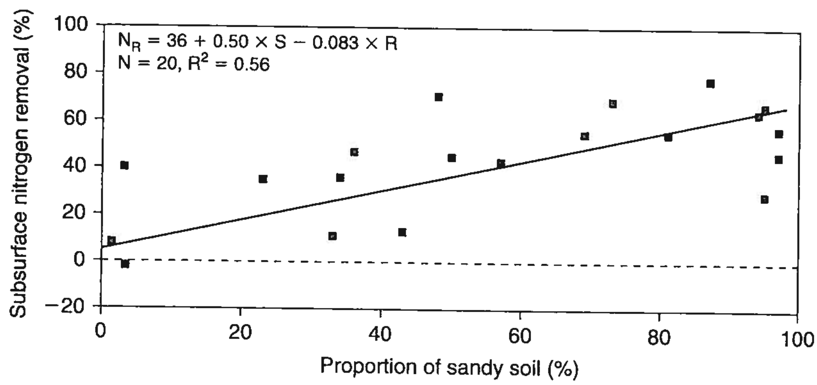

Figure 9. Relationship between average annual subsurface nitrogen removal and the proportion of sandy soils within 21 small Danish headwater catchments.

We believe that the N retention in subsurface soils is due to subsurface removal of nitrate-N in groundwater rather than a hydrological time lag (groundwater residence time) in the small headwater catchments investigated. However, in larger catchments groundwater residence time may be of great significance in the comparison between N leaching and N export. Subsurface N retention is in these cases of vital importance for the linkage between changes in agricultural practices and trends in riverine N concentrations or loading.

5. IMPORTANCE OF AGRICULTURAL LAND FOR NITROGEN EXPORT

The average annual export of total-N, nitrate-N and ammonium-N from 1989–1998 in relation to the proportion of agricultural land in 70 Danish catchments is shown in Table 8. The N export from the monitored catchments reveals an increase concurrently with an increase in the proportion of agricultural land (Table 8). Thus, the average annual export of total-N, nitrate-N and ammonium-N increases by a

Table 8.
Average annual export of total-N, nitrate-N, and ammonium-N from catchments with different proportions of agricultural land and where nitrogen emission from point source is less than 0.5 kg N/ha during the 10-year period, 1989–1998.

Proportion of agricultural land (%)	Number of catchments	Catchment area (km^2)	Runoff (mm)	Total-N export (kg N/ha)	Nitrate-N export (kg N/ha)	Ammonium-N export (kg N/ha)
<20	5	7.4	183	2.4 ± 1.9	1.5 ± 1.6	0.09 ± 0.06
20–40	5	3.3	185	6.4 ± 3.4	4.7 ± 2.6	0.26 ± 0.47
40–60	6	8.7	284	14.5 ± 6.2	12.2 ± 5.2	0.22 ± 0.08
60–70	8	37.6	200	14.9 ± 6.0	14.2 ± 5.1	0.25 ± 0.17
70–80	20	30.2	286	21.9 ± 6.7	20.5 ± 9.8	0.41 ± 0.28
>80	26	12.5	234	21.1 ± 10.8	19.0 ± 9.2	0.41 ± 0.28

Also shown are number of catchments, mean catchment area, and mean annual runoff from each of the land use classes.

factor 9, 14, and 5, respectively, when the proportion of agricultural land increases from 0–20% to >70%. Although the average catchment area and average runoff also experience an increase, there is no doubt that a substantial proportion of the agricultural N surplus is lost to surface waters.

If we correct the measured N export for differences in average annual runoff (discharge-weighted concentration) we still see a significant increase in the N input to surface waters with increasing proportions of agricultural land (Table 9). The observed increase in average annual discharge-weighted concentration of N is again most pronounced for nitrate-N (factor 11), followed by total-N (factor 7), and ammonium-N (factor 3.5) (Table 9).

Table 9.
Average annual discharge-weighted concentration of total-N, nitrate-N, and ammonium-N from catchments with different proportions of agricultural land and where the point source nitrogen emission is less than 0.5 kg N/ha during the 10-year period, 1989–1998.

Proportion of agricultural land (%)	Number of catchments	Total-N (mg N/L^{-1})	Nitrate-N (mg N/L^{-1})	Ammonium-N (mg N/L^{-1})
<20	5	1.4 ± 0.8	0.8 ± 0.7	0.05 ± 0.04
20–40	5	3.5 ± 1.7	2.7 ± 1.5	0.11 ± 0.18
40–60	6	5.4 ± 1.8	4.4 ± 1.5	0.09 ± 0.04
60–70	8	7.8 ± 2.2	6.9 ± 2.2	0.12 ± 0.05
70–80	20	8.4 ± 2.8	7.5 ± 2.9	0.15 ± 0.10
>80	26	9.8 ± 3.9	8.6 ± 3.6	0.18 ± 0.10

The loss of N from agricultural land within 17 European catchments is shown in Table 10. The N loss from agricultural land to surface waters varies greatly from catchment to catchment (Table 10). However, the N loss is generally highest in the catchments lying in the north-western part of Europe and lowest in the catchment in the southern and eastern part of Europe (Table 10). The lack of subsurface N-removal in catchments such as Vansjø-Hobøl in Norway, Rönne Å in Sweden, and Eurajoki in Finland may be the cause to the very high N losses from agricultural land found in these regions.

6. NITROGEN REMOVAL IN LAKES

Lakes play an important role as sinks in the transport of land-based N to downstream coastal and marine areas. The reported rates of N-removal have, however, varied significantly (8–81%, Seitzinger, 1988), although quite constant and high loss rates have been found in 69 Danish shallow eutrophic lakes (Jensen et al., 1990). The present section aims to bring an overview of the magnitude of N-removal in shallow lakes and the important controlling factors. Quantification of the N-removal is based on mass balances from 22 Danish lakes with 10 years data and four Dutch lakes with 11–13 years data. For more details on methods and calculations, we refer to Jensen et al. (1990, 1992) and Jeppesen et al. (1998).

Most of the N-removal in lakes is due to denitrification, and only a minor part to permanent burial in the sediments (Dudel and Kohl, 1992; Van Luijn et al., 1996). In Danish Lake Søbygård, Jensen et al. (1992) estimated that around 90% of the N-removal could be attributed to denitrification. In the remaining part of this section, the term "N-removal" will be used for the sum of denitrification and burial of N in lakes.

Table 10.
Average annual loss of total nitrogen from agricultural land to surface waters within 17 European catchments as estimated applying source apportionment.

	Catchment area (km^2)	Model estimated loss of total nitrogen from agricultural land to surface waters (kg N/ha)
Vansjø-Hobøl, Norway	690	68.4
Yorkshire-Ouse, England	3315	36.8
Enza, Italy	901	12.2
Eurajoki, Finland	1336	50.6
Rönne Å, Sweden	1900	64.5
River Odense, Denmark	486	39.7
Vechte, Germany/The Netherlands	2400	25.8
Uecker, Germany	2430	22.0
Susve, Lithuania	1165	12.6
Lough Derg and Ree, Ireland	10600	27.3
Attert, Luxembourg	254	42.8
Gurk, Austria	2574	31.6
Zelivka, Czech Republic	1189	27.1
Kapos, Hungary	3170	13.1
Vilaine, France	10134	41.0
Guadiamar, Spain	1356	4.7
Pinios, Greece	2797	19.8

lthough quite constant and high
w eutrophic lakes (Jensen et al.,
verview of the magnitude of N-
lling factors. Quantification of the
nish lakes with 10 years data and
details on methods and calcula-
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trification, and only a minor part
ohl, 1992; Van Luijn et al., 1996).
stimated that around 90% of the
n the remaining part of this sec-
m of denitrification and burial of

ural land to surface waters
ing source apportionment.

m^2)	Model estimated loss of total nitrogen from agricultural land to surface waters (kg N/ha)
	68.4
	36.8
	12.2
	50.6
	64.5
	39.7
	25.8
	22.0
	12.6
	27.3
	42.8
	31.6
	27.1
	13.1
	41.0
	4.7
	19.8

Average N-removal for the 26 lakes is estimated to 92.3 mg N/m^2/day (Table 11). A substantial amount of the N entering the lakes is removed, the relative N-removal being as high as 38.8%. This value corresponds well with the value reported for 69 shallow Danish lakes (mean: 43%) (Jensen et al., 1990).

Table 11.
Average annual nitrogen loading and retention in 23 Danish shallow lakes and 4 lakes in the Netherlands.

Name of lake	Mean depth (m)	Lake area (km^2)	Retention time (years)	Nitrogen loading (mg N/m/day)	Nitrogen retention (mg N/m/day)	Nitrogen retention (% of loading)
Denmark						
Lake Arreskov	1.9	3.2	1.46	39.4	23.2	58.3
Lake Arresø	3.1	39.9	4.14	32.1	17.9	58.6
Lake Borup	1.1	0.1	0.07	409.1	48.1	15.5
Lake Bryrup Langsø	4.6	0.4	0.23	523.2	241.6	48.1
Lake Dons Nørresø	1.0	0.4	0.05	450.8	113.8	27.6
Lake Engelsholm	2.6	0.4	0.23	195.2	125.9	64.9
Lake Fuglesø	2.0	0.1	0.17	513.8	226.2	49.4
Lake Fårup	5.6	1.0	0.46	123.1	71.7	57.7
Lake Gundsømagle	1.2	0.3	0.09	552.6	151.6	28.7
Lake Hejrede	0.9	0.5	0.15	246.0	53.4	23.7
Lake Hinge	1.2	0.9	0.05	400.2	55.7	14.6
Lake Jels Oversø	1.2	0.1	0.02	1592.6	147.2	9.0
Lake Kilen	2.9	3.3	0.77	98.0	69.8	71.4
Lake Langesø	3.1	0.2	0.58	229.5	108.1	49.7
Lake Lemvig	2.0	0.2	0.09	590.1	172.9	30.5
Lake Ravn	15.0	1.8	2.33	190.8	100.4	60.1
Lake St. Søgård	2.7	0.6	0.23	445.7	109.7	25.2
Lake Søgård	1.6	0.3	0.07	1005.4	226.1	20.5
Lake Søholm	6.5	0.3	1.78	90.9	50.2	57.7
Lake Tissø	8.2	12.3	1.06	224.6	145.5	68.1
Lake Tystrup	9.9	6.6	0.55	650.3	243.1	40.5
Lake Vesterborg	1.4	0.2	0.08	743.6	121.2	18.5
Lake Ørn	4.0	0.4	0.05	357.6	37.1	10.3
Mean	3.6	3.2	0.64	421.9	115.7	39.5

(*Continued*)

Table 11. (Continued)

Name of lake	Mean depth (m)	Lake area (km^2)	Retention time (years)	Nitrogen loading (mg N/m/day)	Nitrogen retention (mg N/m/day)	Nitrogen retention (% of loading)
The Netherlands						
Lake Veluwemeer	1.3	32.4	0.13	126.7	65.5	51.4
Lake Woldewijd	1.5	18.0	0.32	49.5	18.1	37.1
Lake Nuldernauw	1.5	9.5	0.12	163.5	69.5	42.6
Lake Drontermeer	1.1	5.4	0.02	611.8	122.0	21.0
Mean	1.4	16.3	0.15	237.9	68.8	38.0
Grand mean	2.5	9.8	0.4	329.9	92.3	38.8

Included is also a description of lake morphology and average hydraulic retention time.

The N-removal rate ranges from 17.9 to 243.1 mg N/m^2/day as the N loading ranges from 32.1 to 1,592.6 mg N/m^2/day, while the relative N-removal ranges from 9.0 to 71.4% removal of the N loading. No significant differences are found between the Dutch and the Danish lakes in either absolute or relative N-removal rates. The intra-lake variation seems to be as high as the inter-lake variation in the two countries (Table 11).

6.1. Abiotic Factors Controlling Nitrogen Removal

Nitrogen removal depends on the N loading to the lake, and the rate of removal is generally higher with higher loading ($P < 0.0001$, Figure 10). The removal rate

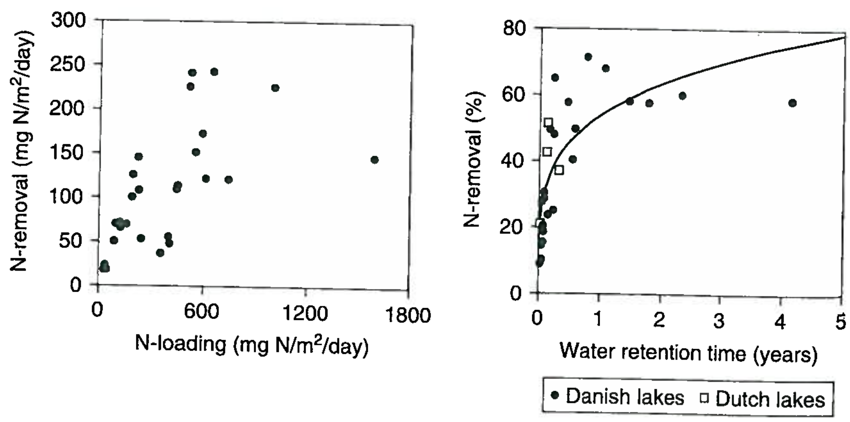

Figure 10. Nitrogen removal versus nitrogen loading in Danish and Dutch shallow lakes and the relative nitrogen removal compared to hydraulic retention time.

Nitrogen loading (mg N/m/day)	Nitrogen retention (mg N/m/day)	Nitrogen retention (% of loading)
126.7	65.5	51.4
49.5	18.1	37.1
163.5	69.5	42.6
611.8	122.0	21.0
237.9	68.8	38.0
329.9	92.3	38.8

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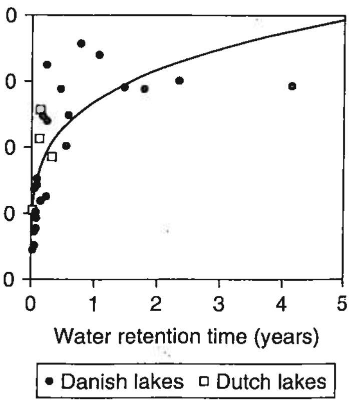

ling in Danish and Dutch shallow
:o hydraulic retention time.

can, however, vary markedly, even at similar N levels of loading. The main reason for this variation is chiefly explained by differences in hydraulic retention time, a factor upon which relative N-removal is highly dependent (Figure 10).

A simple empirically derived relationship, depending solely on the hydraulic retention time in the lake (τ_w), may account for 60% of the observed variation in the relative N-removal, $N_{ret}(\%)$:

$$N_{ret}\,(\%) = 53.9 \times q_w^{0.235},\ r^2 = 0.60,\ P < 0.0001$$

This relationship is comparable with the model proposed by Jensen et al. (1990) and Windolf et al. (1996).

6.2. Biotic Factors Controlling Nitrogen Removal

Besides the abiotic factors controlling N-removal, the biological structure of a lake may also markedly influence N-removal. In two of the Danish lakes included, the biological structure changed dramatically during the investigation period (Jeppesen et al., 1998). In Lake Arreskov, fish kill in winter 1991–1992 caused a shift from a turbid plankton-dominated stage to a clear water and hence macrophyte-dominated stage. The relative N-removal increased from 26–38% before to 48–62% afterward the fish kill. Similarly in Lake Engelsholm, N-removal increased from 49–53% to 59–66% following a partial removal of the planktivorous fish stock in 1992–1994.

Various factors resulted in the increased N-removal in the two lakes (Jeppesen et al., 1998): (i) a decrease in organic N in the lakes and outlets due to the decrease in the N incorporated in the phytoplankton; (ii) reduced resuspension due to a decrease in the number of fish foraging in the sediment and an increase in benthic algal growth; and (iii) higher denitrification in the sediment, reflecting less competition between denitrifiers and phytoplankton for nitrate, enhanced N retention by phyto- and zoobenthos and enhanced sediment nitrification due to higher oxygen concentrations. These two cases clearly demonstrate the very complicated interactions between N-removal and lake biological structure. Differences in biological structure may thus be part of the explanation of differences in the reported N-removal rates, especially in shallow lakes.

7. NITROGEN REMOVAL IN FRESHWATER WETLANDS

In Denmark, studies of N-removal in freshwater wetlands have been undertaken since the mid-1980s. Both natural fens and meadows with different hydrological regimes as well as restored and constructed wetlands have been investigated.

7.1. Natural Wetlands

In natural freshwater wetlands receiving groundwater recharge (i.e., minerotrophic wetlands), N-removal varies from 57 kg N to >2,100 kg N/ha/year. The relative removal varies only from 56% to 97%, without any clear correlation between loading and efficiency (Tables 12 and 13). Thus, there is a huge capacity of Danish

Table 12.
Nitrate removal in different riparian wetlands with groundwater recharge (flow through) (Hoffmann, 1998b).

Locality	Nitrate removal rates (kg NO_3^-N/ha/year)	Reduction (%)[1]
River Stevns, meadow	57	95
Rabis brook, meadow	398	56
River Gjern		
A, meadow	140	67
B, fen (1993)	2100	97
B, fen (5 years)	1079	97
C, meadow (5 years)	541	96
D, meadow (5 years)	398	97

[1]Percentage of incoming nitrate loading removed.

Table 13.
Examples of nitrogen removal rates measured in two Danish river valleys with groundwater recharge following re-meandering of the river channel (Hoffmann et al. 1998, 2000a).

Locality	Kg nitrate-N/ha/year	Percentage
River Brede, large-scale, meadow (63 ha)	92	71
Headwaters of River Gudenaa, large-scale, meadow (57 ha)	8.4	57

Both studies were conducted in the first year following the river restoration (1995/1996).

freshwater wetland soils for N-removal through denitrification. The N loading to the wetlands reflects upland characteristics such as land use, precipitation surplus, drainage conditions, soil type, the groundwater flow pattern, and so on. Some of the overall factors characterizing wetlands recharged by groundwater and examples on their ability to remove N will be given in this section.

At the River Stevns, the concentration of nitrate in recharging groundwater to the meadow varied from 15 to 30 mg nitrate-N per liter. Nearly 74% of all upland groundwater was discharged directly into River Stevns through drainage pipes. Only 57 kg N was retained in the meadow, of which 31 kg N/ha/year was denitrified, while on average 89 kg N was removed from the meadow through haymaking.

with groundwater recharge (flow

noval rates N/ha/year)	Reduction (%)[1]
	95
	56
	67
	97
	97
	96
	97

ed.

n two Danish river valleys with
of the river channel (Hoffmann

rate-N/ha/year	Percentage
	71
	57

lowing the river restoration

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itrate in recharging groundwater to per liter. Nearly 74% of all upland er Stevns through drainage pipes. which 31 kg N/ha/year was denitri- n the meadow through haymaking.

Thus, N-removal from the meadow was higher than the input the meadow (total input – 60 kg N; total output – 120 kg N).

The wet meadow at the Rabis brook is recharged by nitrate-rich groundwater, which at the valley slope breaks through to the soil surface and irrigates the meadow naturally (Table 12). This is possibly the reason why the relative N-removal is so low (56%), since the nitrate has to move by advective flow or by diffusion to the active denitrification sites close to the soil surface.

In River Gjern catchment area, studies have been made for several years of wetlands with different hydraulic regimes (Table 11). Especially in a 73-m wide water-covered fen (area B, Table 12) N turnover has been studied intensively, and particularly high rates of denitrification have been found in the area around the river valley slope. Over a distance of only 13–17 m, the nitrate concentration falls from approximately 25 to 0.01 mg nitrate-N per liter, corresponding to a denitrification rate of 1–5 g N/m^2/day, depending on where in the zone of enhanced denitrification sampling is undertaken (Blicher-Mathiesen, 1998; Hoffmann, 1998a; Hoffmann et al., 2000b). Only a few other studies have hitherto reported denitrification rates of this magnitude, for example, Cooper (1990) (8.1 g N/m^2/day), Haycock and Burt (1993) (0.74 g N/m^2/day), Haycock and Pinay (1993) (up to 10 g N/m^2/day), and Jørgensen et al. (1988) (2.1 g N/m^2/day).

7.2. Rehabilitation of Fens, Wetlands, and Wet Meadows in Floodplains

In most European countries, watercourses have been modified by man to improve certain features, for example, flood control, drainage of surrounding land, navigation, and so on. In countries such as Denmark with an intensive agricultural production, more than 90% of the total river network has been regulated to some extent (Iversen et al., 1993). Straightening and channelization of watercourses was conducted to ensure sufficient drainage of the floodplains.

Today we are restoring many of our rivers by reinstating their former meandering course (Kronvang et al., 1998). Hence, former fens, wetlands, and wet meadows are reinstated in our river valleys by re-meandering the river channel, elevating the river bed and disconnecting drains and ditches which also lead to increased N-removal (Table 13). The N-removal rates obtained at two river restoration sites in Denmark are shown in Table 13. Considerable variation in the groundwater flow patterns both along the river and from riverside to riverside was found, implying that N transport and removal vary significantly (Hoffmann et al., 1998; Hoffmann et al., 2000a).

7.3. Irrigation of Meadows with Drainage or River Water

Experiments involving the irrigation of meadows and reed forests with drainage water or river water have also yielded promising results with respect to nitrate reduction. The extent of nitrate removal primarily depends on the amount of water infiltrating the soil. Therefore, the size of the area to be irrigated needs to be adjusted to the amount of water it is expected to receive. In an irrigation experiment alongside the River Stevns, where all the water input to the area infiltrated the soil, 99% of the nitrate was denitrified in the uppermost 2 cm of the soil profile (Table 14) (Hoffmann et al., 1993). In the same study, nitrate reduction was also measured on a plot of the meadow that had been fed with tile drainage water for approximately 100 years via a drainage conduit terminating on the river valley slope. Over a distance of 45 m, the nitrate concentration fell from 11.3 mg nitrate-N per liter at the conduit outflow to 0.1 mg nitrate-N per liter midway into the meadow, that is, a reduction of 99% (Hoffmann et al., 1993).

Table 14.
Nitrate removal by irrigation with time drainage water or stream water.

Locality	Kg NO_3-N/ha/year	%
Glumsø, reedswamp[1]	520	65
Glumsø, reedswamp[1]	975	62
Glumsø, reedswamp[1]	2725	54
Glumsø, large-scale, reedswamp[1]	569	94
River Stevns, meadow[2]	350	99
River Stevns, meadow with drainage pipes[3]	(conc.) 11.3	99
Syv brook, meadow	300	72
River Storå, restored meadow	530	48
River Gjern, meadow1,2 (min)	34	88
River Gjern, meadow1,2 (max)	200	98

The removal rates obtained during irrigation with tile drainage water takes into account the periodicity of tile drainage runoff. In Denmark, highest runoff in winter and spring: October to May.

[1]Different hydraulic loading and different nitrate loading.

[2]Short-term experiment.

[3]Concentration given in mg nitrate-N per liter.

When part of the irrigation water is discharged as surface runoff, the relative nitrate removal decreases to between 48% and 72% according to the studies hitherto undertaken at Syv Bæk brook (Hoffmann, 1991), Lake Glumsø (Hoffmann, 1986; Jørgensen et al., 1988), and the River Storå (Fuglsang, 1993, 1994) (Table 14), this being attributable to the fact that denitrification does not occur in the surface water due to the prevailing oxic conditions. Any N-removal occurring in the surface water must therefore take place through algal uptake, uptake into the microbial pool or sedimentation of particulate N.

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uppermost 2 cm of the soil profile
me study, nitrate reduction was also
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luit terminating on the river valley
centration fell from 11.3 mg nitrate-
nitrate-N per liter midway into the
et al., 1993).

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Kg NO_3-N/ha/year	%
520	65
975	62
2725	54
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300	72
530	48
34	88
200	98

tile drainage water takes into
Denmark, highest runoff in winter

loading.

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% according to the studies hitherto
Lake Glumsø (Hoffmann, 1986;
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Irrigation of riparian areas with tile drainage water can only be undertaken in periods during which water is flowing in the drains. In consequence, the operational period highly depends on local conditions such as the amount of precipitation, soil type, etc. The annual rates given for the meadows at the River Stevns and Syv brook shown in Table 14 are thus calculated for the period during which they were actually irrigated by drainage water, that is, 120 and 200 days, respectively.

A short-term irrigation and flooding study at the lower reaches of the River Gjern shows that nitrate reduction occurs in the uppermost part (0–2 cm) of the soil in areas subjected to regular flooding or irrigation events (Hoffmann, 1996, 1998b). Although the infiltration capacity is low at the Gjern study site, the results show reduced nitrate values even during short-term flooding/irrigation events. It means that apart from being important for sedimentation, naturally meandering rivers and their riparian areas serve as a functional unit and a stabilizing ecological factor for the aquatic environment.

8. MEASURES TO REDUCE EMISSIONS OF NITROGEN FROM POINT AND DIFFUSE SOURCES IN EUROPE

Measures to reduce point source emissions of N focus on urban wastewater treatment (UWWT) plants as well as various key industries. The Urban Wastewater Treatment Directive (91/271/EEC) adopted by the European Commission in 1991 is a key directive for water management in the EU. The Directive sets minimum standards for the collection, treatment, and disposal of wastewater dependent on the size of discharge, and the type and sensitivity of receiving waters. In the case of N, a maximum annual average threshold of 15 mg N/L is set for smaller UWWT plants (10,000–100,000 PE), whereas the threshold is lower (10 mg N/L) for larger UWWT plants (>100,000 PE).

Measures to reduce N emissions from agriculture are more difficult to implement, both in a political and practical perspective. A variety of policies has been applied in the EU and the different countries: (i) legislation; (ii) economic instruments; and (iii) information. In 1991, the European Commission adopted the Nitrate Directive aimed to reduce or prevent the nitrate pollution of water caused by the application and storage of inorganic fertilizers and manure on farmland. The Nitrate Directive requires that member states: (i) identify vulnerable areas to nitrate pollution; (ii) establish Action Plans governing the time and rate of fertilizer and manure application, and conditions of manure storage in vulnerable zones; (iii) implement monitoring programs to assess the effectiveness of action programs; and (iv) establish Codes of Good Agricultural Practice to be implemented by farmers on a voluntary basis in other areas.

The impact of the Nitrate Directive will of course depend on the interpretation of requirements by the EU Member States, especially that of the area extent of vulnerable zones. As an example, Austria, Denmark, Germany, and The Netherlands have designated the whole territory, France 46% vulnerable zones of the whole territory, while Portugal has designated five, and the United Kingdom 69 vulnerable zones.

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, 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 programs (before 2007); and (iii) implement mitigation strategies in the form of River Basin Management Plans (before 2009). A good ecological quality should be reached in surface water bodies before 2016 and this will in many river basins demand further reductions in N loadings from especially diffuse sources.

The measures applied in selected European countries for combating N pollution from point and diffuse sources are shown in Table 15. Most countries have initiated tertiary wastewater treatment and imposed regulations on manure restrictions and storage.

In Denmark, the government has adopted five major Action Plans since the early 1980s to reduce by 50% N pollution of the aquatic environment (Table 16). The Action Plan on the Aquatic Environment I from 1987 demanded N-removal at all major sewage treatment plants (>15,000 PE). The regulations imposed to ensure a reduction in N pollution from agriculture have been reiterated in all subsequent Action Plans (Table 15). Besides the multiple measures implemented, Denmark has also adopted three other plans: (i) a Plan for Afforestation (1987) aimed to double the forest area during the next 60–80 years (5,000 hectares per year); (ii) the Strategy on Marginal Land (1987) aimed to restore 20,000 ha of former wetlands over a 10 to 20 year period; and (iii) EU Set-Aside schemes. These plans will of course also contribute to the reduction of the N pollution from farmland.

9. CONCLUSIONS AND PERSPECTIVES

Concern about elevated N concentrations and loading to groundwater and surface water in Europe has prompted the introduction of many reduction strategies at international, national, and local levels. Measures to reduce point source discharges of N focus on tertiary treatment of urban wastewater as well as various key industries have been implemented in most countries within the EU (see Tables 15 and 16).

Today, a high proportion of the total anthropogenic N loading to the aquatic environment consists of N loading from agriculture (cf. Table 1). However, reducing the N input from agriculture is difficult for both technical and political reasons. Measures may be introduced/implemented at the source to reduce the large N surplus (as seen in many European countries, see Table 3). Another approach may be to introduce transport measures to reduce the level of N before it reaches surface and thus increase the N-removal potential along the routes followed by N from soil to water.

The latter may be allowed by reinstating formerly drained lakes and wetlands in which the potential for nitrate removal through the denitrification process normally is high (see Tables 11–14). So far results obtained in Denmark and The Netherlands

Table 15.
Measures applied across Europe to combat nitrogen pollution.

Countries approach	A	B	D	DK	E	FIN	F	GR	IRL	I	NL	N	P	PL	S	UK
Point sources																
UWWT tertiary treatment		×	×	×	×	×	×		×		×	×		×	×	×
Industry – BAT	×		×	×												×
Agriculture																
Manure restrictions	×	×	×	×	×		×	×	×	×	×	×			×	×
Manure storage	×	×	×	×			×	×	×		×	×			×	×
Silage storage			×	×				×	×		×	×			×	×
Fertilizer restrictions	×	×	×	×			×	×	×		×	×			×	×
Atmospheric input																
Atmospheric sources	×	×	×	×	×						×	×	×	×	×	×
Industry – BAT	×		×	×										×	×	×

A: Austria; B: Belgium; D: Germany; DK: Denmark; E: Spain; FIN: Finland; F: France; GR: Greece; IRL: Ireland; I: Italy; NL: The Netherlands; N: Norway; P: Portugal; PL: Poland; S: Sweden; UK: United Kingdom.

Table 16.
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 nitrogen pollution
NPO Action Plan	1985	Elimination of direct discharges from farms Livestock harmony on the farm level
Action Plan on the Aquatic Environment I	1987	9-month storage facility for slurry 65% winter green fields Crop and fertilizer plans
Plan for sustainable agricultural development	1991	Standard nitrogen fertilization values for crops Standard values for nitrogen in animal manure Required utilization of nitrogen in animal manure (30–45%) Fertilizer 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 ha farmland to wetlands Reiterated demands for utilization of nitrogen in animal manure (40–55%)
Action Plan on the Aquatic Environment III	2003	Halving of the Danish P surplus before 2015 Reiterated demands for catch crops and utilization of N in animal manure Establishment of 50,000 ha buffer zones to reduce P loss from agricultural fields Demands for transforming 4,000 ha of agricultural land to wetlands

have shown that the potential for N-removal in lakes and freshwater wetlands is high. In Denmark, the Second and Third Action Plans on the Aquatic Environment (see Table 16) included the following initiative: to remove a total of 7,000 tons N/year by rehabilitating 20,000 hectares of drained and otherwise reclaimed wetlands within 5 years.

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