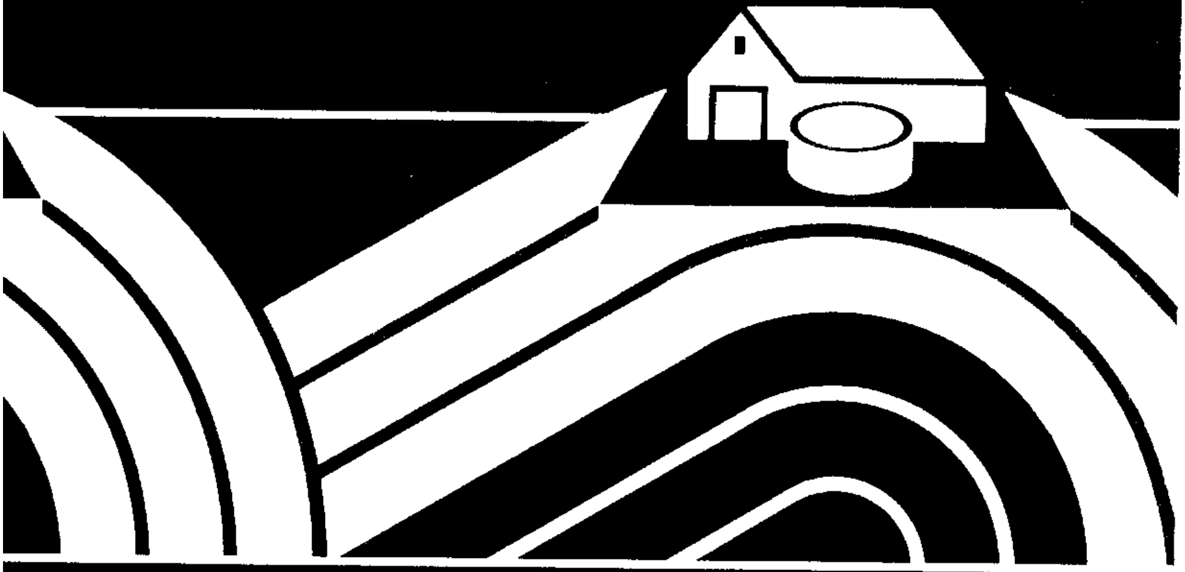
NPo-forskning
fra Miljøstyrelsen

C - abstracts

Nitrogen and Phosphorus in Fresh and Marine Waters



Miljøministeriet Miljøstyrelsen

Danish Research Programme on Nitrogen, Phosphorus and Organic Matter (NPO)

The aim of the NPO Research Programme is to gather knowledge on the decomposition of Nitrogen (N), Phosporus (P) and organic matter (O) in the soil, and on their impact on lakes, watercourses, inlets, groundwater and the sea.

This report is one of a total of about 50 reports to be issued in connection with the implementation of the NPO Research Programme. The National Agency of Environmental Protection (NAEP) is responsible for the programme, under which about 70 NPO projects have been launched, carried out at 25-30 institutions.

In the 1970's and the beginning of the 1980's there was a growing awareness of the threats to life in watercourses etc. presented by discharges of nutrients – and of the risk of nitrate contamination of groundwater. In 1984 a report was prepared, synthesising existing knowledge in this field. The report, known by the name of NPO Report, was published by the NAEP.

To follow up this report the Dansih Parliament took the first steps in 1985 to reduce pollution with nutrients – laying down requirements for storage and application of farm yard manure in the agricultural sector.

For the purpose of improving our knowledge on the impact of nutrients in nature, the Danish Parliament also reserved 50 million DKK for the research programme, running from 1985 to the end of 1990.

The significance of the NPO Research Programme was further underlined with the Danish Parliament's adoption of the Action Plan on the Aquatic Environment in 1987. The results of the NPO Research Programme will play a vital role in the evaluation of the effects of the Action Plan.

To safeguard the technical and economic interests relating to the research activities a steering group was set up, having the overall responsibility for the implementation of the NPO Research Programme. Furthermore, three coordination groups were formed, each of them responsible for one of the three fields: soil and air, groundwater, and surface water.

The reports are published in the series »NPo-forskning fra Miljøstyrelsen« (NPO Research in the NAEP), divided into three sections:

A: reports on soil and air
B: reports on groundwater
C: reports on watercourses, lakes and marine waters.

The NAEP has been secretariat for the research programme. The reports published in this series are edited by the Agency with the assistance of the coordination groups.

NPo-forskning fra Miljøstyrelsen
C - abstracts

Nitrogen and Phosphorus in Fresh and Marine Waters

Project abstracts
of the Danish NPo Research Programme

Miljøministeriet
Miljøstyrelsen

List of contents

INTRODUCTION

Emissions of the nutrients nitrogen and phosphorus from cities, industries and agriculture are associated with environmental impacts in atmosphere, soils, groundwater and aquatic ecosystems.

In Denmark, the dominating source of nitrogen is related to agricultural production. In 1985, the Danish Parliament established the NPo-action plan with its primary objective to reduce nitrogen losses from agriculture, in particular in connection with application of manure and slurry. At the same time it was realized that the scientific basis was insufficient to provide a reliable evaluation of the effects of the proposed measures.

Accordingly, the NPo research program was initiated to provide a multidisciplinary and comprehensive study of the entire transport- and transformation cycle of nitrogen (N) and phosphorus (P), as schematically shown in figure 1.1.

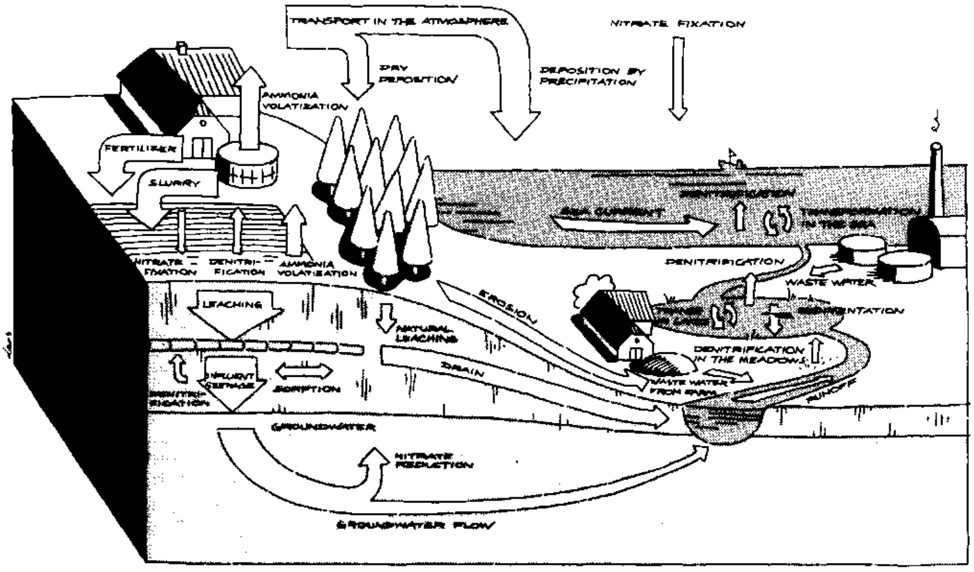


Figure 1.1 Transport and transformation of nitrogen and phosphorus in the environment

The Parliament allocated 50 million Danish kroner for the program and put the National Agency of Environmental Protection in charge of the execution. After a planning phase in the winter of 1986 the program was initiated by spring 1986 with about 50 interrelated projects and participation of about 25 Danish research institutes.

The projects were divided in three groups:

- Group A: Soil and atmosphere:
  - * Farming practices in application of mineral fertilizers and manure
  - * Ammonia volatilization
  - * Nitrogen transport and transformation in the atmosphere
  - * Nitrogen transformation in soil
  - * Nitrogen leaching from the rootzone

- Group B: Groundwater:
  - * Surveys of extent of nitrate contamination
  - * Nitrate transport and transformation in unconfined alluvial aquifers
  - * Nitrate transport and transformation in confined limestone aquifers
  - * Nitrate reduction processes in groundwater
  - * Modelling of nitrate transport and transformation in regional aquifers.

- Group C: Meadows, streams, lakes and marine waters:
  - * Nutrient transport and accumulation in sediments
  - * Internal phosphorous load from sediments in lakes
  - * Denitrification in sediments
  - * Effects of nutrient load on biological structures.

The results have been published as a summary report (Dyhr-Nielsen et al. 1991: Nitrogen and Phosphorus in Soil and Water, NPo-forskning fra Miljøstyrelsen, National Agency of Environmental protection, Denmark), as three abstract volumes in English and as 56 project reports, mainly in Danish.

The project reports may be grouped as presented in table 1.1-1.3.

Table 1.1 Reports in Group A: Soil and atmosphere

	Measurements and interview	Modelling
Farming practices in application of mineral fertilizers and manure	A2, A21	A1
Ammonia volatilization	A7, A11, A12, A15	
Nitrogen transport and transformation in the atmosphere	A4, A5, A6	A18, A22
Nitrogen transformation in soil	A3, A9, A13, A16, A19	A10, A20
Nitrogen leaching from the rootzone	A8, A9, A13, A17	A10, A14, A20

Table 1.2 Reports in Group B: Groundwater

	Unconfined aquifers	Confined aquifers
Surveys of nitrate contamination	B4, B16, B18	B4, B16, B18
Climate and discharge data	B9, B17	B9, B17
Nitrate reduction processes	B3	B1, B2, B6
Nitrate transport and transformation	B5, B7, B8, B10, B11, B12,B17	B7, B17, B19
Modelstudies	B8, B13, B15	B14, B15

Table 1.3 Reports in Group C: Meadows, streams, lakes and marine water

	Meadows	Streams	Lakes	Marine Waters
Erosion and leaching	C12,C13 C14	C12		
Transport and accumulation	C10, C11		C11	C6
Internal load			C4, C5	C8
Denitrification	C13,C14 C15	C2	C9	C1
Effects on biological structure		C7, C10	C9	C3, C5, C8.

In the planning of the research programme, a close integration of field- and laboratory studies with mathematical modelstudies has been emphasized. Accordingly, the program was established around three main activities:

- Detailed investigations of particular subsystems, performed as field and laboratory studies,
- Integrated studies of total systems in selected catchments,
- Synthesis of field- and laboratory findings in mathematical models

The field studies of particular subsystems have been located in representative regions all over Denmark, as shown in figure 1.2.

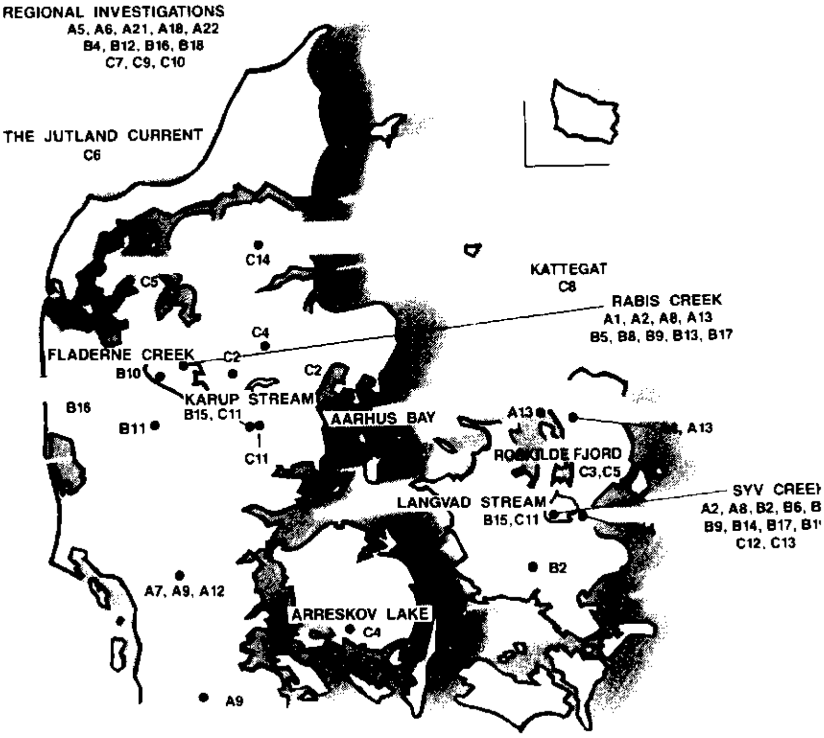


Figure 1.2 Location of the NPo-projects.

The integrated studies of transport and transformation processes have been situated in two small catchments, Syv creek and Rabis creek.

Rabis creek is located in western Jutland, as shown in figure 1.3. The geology and agricultural practices are characteristic for large regions in western and central Jutland. Ground water aquifers are found in unconfined aquifers in glacial outwash plains, highly vulnerable to nitrate leaching. The runoff is dominated by ground water baseflow. Manure and slurry are important nitrogen sources, as some of the highest concentrations of animal production are found in these regions.

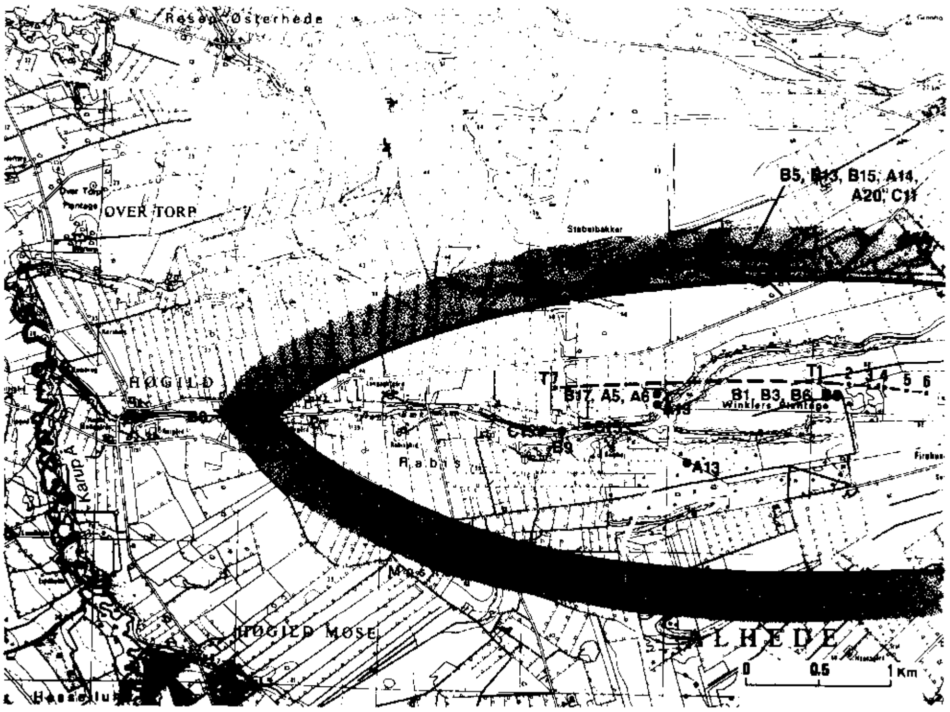


Figure 1.3 The study basin Rabis creek in Jutland.

Syv creek is located in Sealand, as shown in figure 1.4. The geology and the agricultural practices are characteristic for the eastern islands Sealand and Funen. Ground water aquifers are confined and protected by thick layers of moraine till. Runoff is dominated by contributions from tile drains of the loamy soils. Mineral fertilizer is the dominating nitrogen source, as the region has a relatively small animal production.

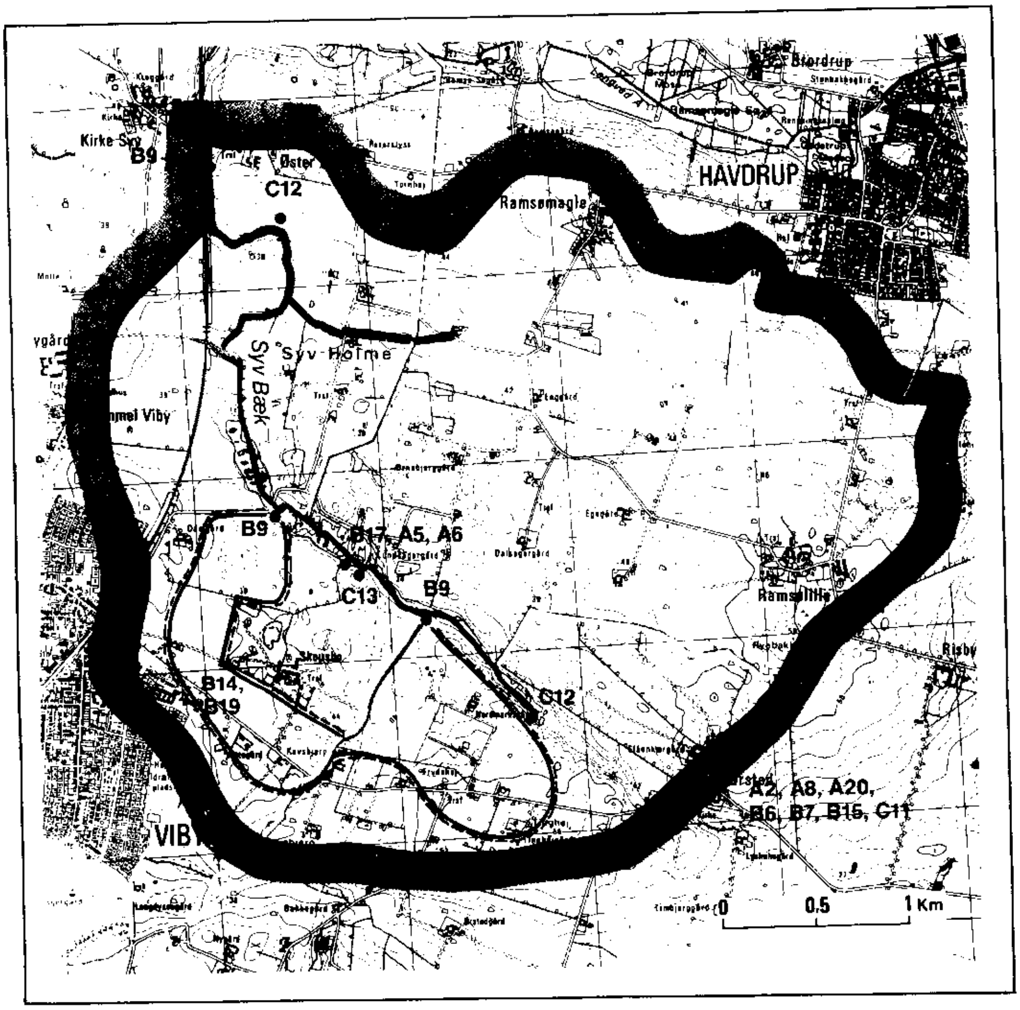


Figure 1.4 The study basin Syv creek on Sealand.

The present volume is one of three abstract volumes containing abstracts from Group A on Soil and Atmosphere.

Ministry of the Environment
National Agency of Environmental Protection
Denmark

The N, P and Organic Matter Research Program 1985-1990

Number C1

Denitrification and Nitrogen Mineralization in Coastal Marine Sediments

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SUMMARY

In the Aarhus Bight, Denmark, high productivity of micro-algae (phytoplankton) in early spring (March-April) results in a large, momentaneous input of organic matter to the sediment. Mineralization begins immediately and NH_4^+ release to the bottom water increases dramatically in a short period following the sedimentation. Denitrification increases during this period, but a high rate is also dependent on a high NO_3^- concentration in the bottom water. When the latter pool is depleted in early summer, denitrification is dependent on NO_3^- production by nitrification in the sediment and the rate is relatively low. From mid-summer to the next sedimentation event in early spring, both NH_4^+ and NO_3^- are released at similar and constant rates. The sampling station in Aarhus Bight released 3.4 g N m^{-2} $year^{-1}$ as NH_4^+ and NO_3^- from the sediment for recirculation by new production in the water column; denitrification representing a loss of N from the ecosystem was only 25 % of this value, but is expected to vary significantly between years.

1. INTRODUCTION

In the coastal waters of Kattegat, Denmark, a large fraction of the spring phytoplankton bloom (February-March) is expected to reach the seabottom by sedimentation. The latter takes place in March-April and initiates a number of degradative processes by bacteria, which eventually produce and release the deposited N to the overlying water as NH_4^+ (after mineralization), NO_3^- (after NH_4^+ conversion to NO_3^- by nitrification), or N_2 (after NO_3^- conversion to N_2 by denitrification). As both NH_4^+ and NO_3^- are important nutrients for further production in the water column, while N_2 represent a loss of N from the ecosystem, it is important to know the seasonal cycles and quantitative importance of the bacterial processes in the sediment.

2. MATERIALS AND METHODS

Three localities in the Aarhus Bight were selected for the study; all had water depths of about 15 m (Andersen et al., 1984). Sampling was done at regular intervals of 2-4 weeks during 1985-86 (denitrification) and 1987-88 (NH_4^+ and NO_3^- flux at sediment-water interface, etc.). In the following we present results from Station 16, which was typical for a large area of the Aarhus Bight and selected for intensive studies during 1987-88.

NH_4^+ and NO_3^- concentrations in the water column (surface and bottom) and sediment were followed; a fine-resolution cutting technique was developed for analysis of the pools in 2-mm fractions of the uppermost sediment (Lomstein et al., 1990).

Flux rates of NH_4^+ and NO_3^- (transport rates between sediment and water phase) were performed in the laboratory using intact cores and 0.5-4 h incubations at near-in situ conditions of temperature, darkness and stirring of the water phase (Jensen et al., 1990).

Denitrification rates were measured in the laboratory using intact cores and 0.5-1.5 h incubations with C_2H_2 (acetylene blockage technique) to inhibit the N_2O reduction by denitrifying bacteria; near-in situ conditions of temperature, darkness and stirring were used (Jensen et al., 1990).

3. RESULTS

Nitrate is the primary N source for the phytoplankton production in early spring and the depletion of NO_3^- in surface water is therefore a good indicator of bloom termination. In Figs 1 and 2 described below, the hatched area indicates the period of rapid NO_3^- consumption in the surface water and subsequent sedimentation of the micro-algae.

3.1. Hydrography

In Kattegat, the water column often stratifies due to differences in salinity between north-going (low-salinity) surface water and south-going (high-salinity) bottom water; the halicline is typically at 15 m (Andresson and Rydberg, 1988). At the 15-m station in Aarhus Bight, the stratification in summer (April-May to July-August) is interrupted by periods of mixing.

Most important for the N cycling in the sediment are the small yearly variation in timing of the bloom of productivity, sedimentation and mixing/stratification. In a period after sedimentation, the maintenance of high NO_3^- concentrations in bottom water due to early stratification (April-May) has a major impact on denitrification and direction of the N fluxes at the sediment-water interface.

3.2. NH_4^+ and NO_3^- concentrations in water and sediment

Winter concentrations of NH_4^+ and NO_3^- in the mixed water column are typically about 3 µM and 10-15 µM, respectively. As mentioned earlier, both pools are very low (less than 0.2 µM) in the surface water after the algal bloom and its sedimentation in March-April; in contrast, both pools remain high in the bottom water until July-August (Jensen et al., 1990). From the fall (October) both NH_4^+ and NO_3^- concentrations increase again in the water column, and eventually reach the winter concentrations.

In the sediment, the input of organic material by sedimentation resulted in rapid mineralization and immediate increase of NH_4^+ concentrations from 10-20 µM to about 40 µM in the surface sediment (upper 2 mm). The sedimentation also increased NO_3^- consumption, however, and the NO_3^- concentrations decreased from about 25 µM to 5-10 µM in the same surface layer (upper 2 mm).

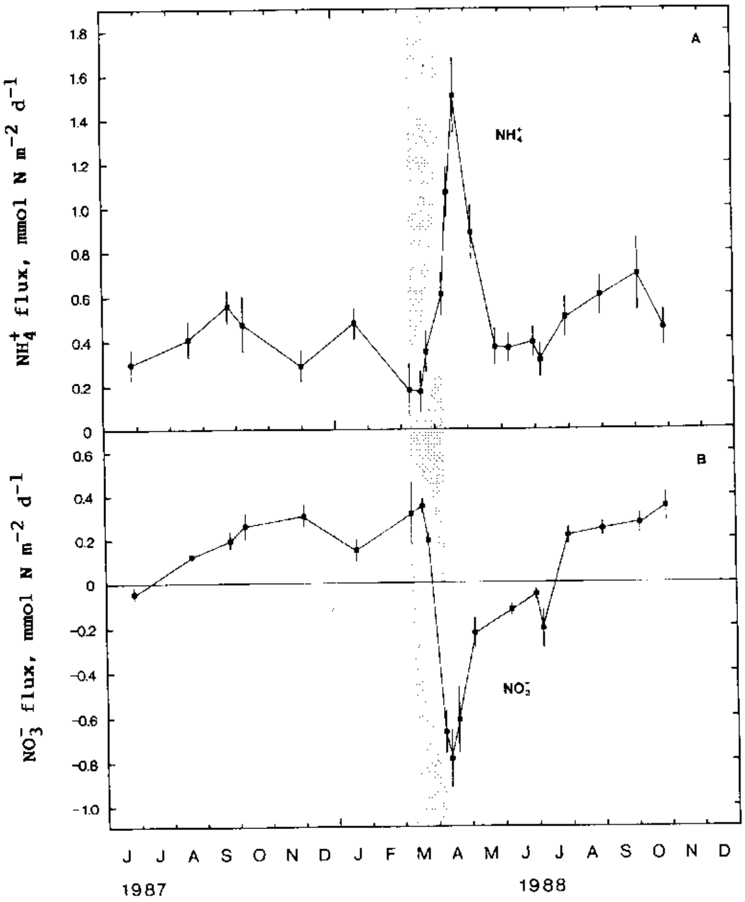

Figure 1. Seasonal variation of NH_4^+ (A) and NO_3^- (B) flux rates (transport across sediment-water interface) at Station 16 in Aarhus Bight. Negative flux indicates uptake by the sediment. Hatched area indicates sedimentation period after phytoplankton bloom in spring.

From the concentrations measured in bottom water and surface sediment it was possible to predict the directions of NH_4^+ and NO_3^- fluxes at the sediment-water interface. A year-round release of NH_4^+ from the sediment could be predicted, while an uptake of NO_3^- apparently took place in a period over several months after the sedimentation event, when the NO_3^- concentrations were still relatively high in the bottom water.

3.3. NH_4^+ and NO_3^- flux rates

Figure 1 shows the seasonal variation of measured flux rates, which were in accordance with the directions of NH_4^+ and NO_3^- transport as predicted above. Average NH_4^+ release rate for a large part of the year was about 0.4 mmol N m^{-2} day^{-1}, but during a 1-2 month period after bloom sedimentation the release rate was as high as 1.5 mmol N m^{-2} day^{-1}. By comparison, the NO_3^- release rate was about 0.3 mmol N m^{-2} day^{-1}, except for a period of 3-4 months after sedimentation when a NO_3^- uptake by the sediment was observed; maximum NO_3^- uptake immediately after sedimentation (March-April) was about 0.8 mmol N m^{-2} day^{-1}.

3.4. Denitrification rates

Figure 2 shows results from the investigation in 1985-86 on seasonal patterns of denitrification rates in the Aarhus Bight sediments. It was observed that denitrification at Station 16 was highest (about 0.7 mmol N m^{-2} day^{-1}) during or immediately after the sedimentation event in March-April. After this, the rate declined for 3-4 months to reach a minimum value (about 0.1 mmol N m^{-2} day^{-1}) in the fall.

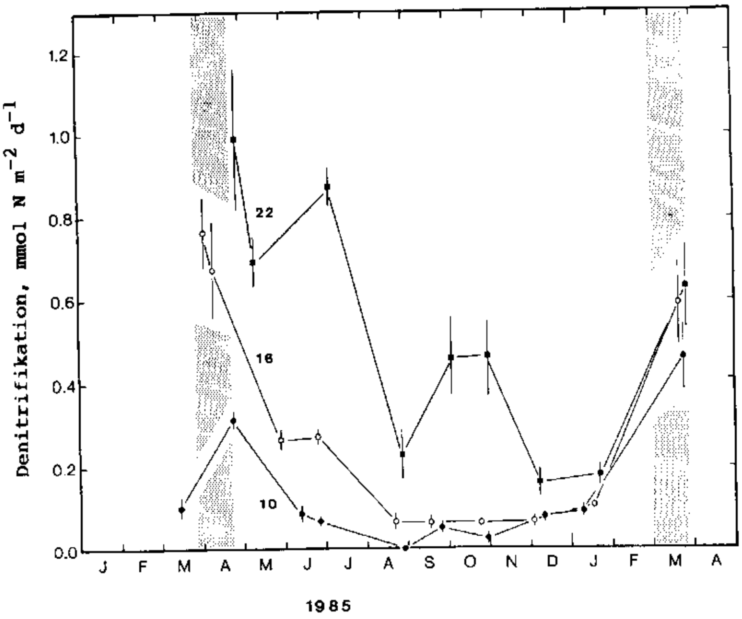

Figure 2. Seasonal variation of denitrification rate at Stations 10, 16 and 22 in Aarhus Bight. Hatched area indicates sedimentation period after phytoplankton bloom in spring.

4. DISCUSSION AND CONCLUSION

4.1. Sedimentation of phytoplankton bloom in spring

A large part of the annual primary production in the Aarhus Bight (100-200 g C m^{-2} $year^{-1}$) is due to the phytoplankton bloom in February-March; productivity rates at this time may be >1 g C m^{-2} day^{-1}. In the absence of efficient grazing by zooplankton at this time most of the bloom sinks out of the water column and reaches the sediment as easy-degradable phyto-detritus.

The sudden input of organic matter to the sediment surface initiates high microbial activity, even if the temperature is still relatively low at this time (<5 °C) (Graf, 1987; Jensen et al., 1990).
Mineralization processes at the sediment surface results in release of NH_4^+ from degradation of organic N pools in the micro-algae.
This study clearly demonstrated NH_4^+ accumulation in the upper few mm of the sediment and subsequent NH_4^+ release of NH_4^+ to the bottom water immediately after sedimentation of the phytoplankton bloom (Jensen et al., 1990).

Among the degradative processes in the surface sediment, bacterial denitrification is localized immediately below the oxic zone, which is only a few mm thick, at a maximum. The denitrification zone is never deeper than 1 cm (Jensen et al., 1990) and the overall rate of the process is therefore dependent, not only on the input of supply of organic matter for degradation, but also on NO_3^- supplied by diffusion from the oxic sediment surface or from the bottom water. Thus, in Aarhus Bight, the bottom water remained relatively rich in NO_3^- until mid-summer (July-August), which was important to support the elevated denitrification rates in a period after the sedimentation event in March-April.

4.2. Comparison of nutrient (NH_4^+ and NO_3^-) flux and denitrification rates

The annual net release of NH_4^+ and NO_3^- from the sediment to the water column was about

210 mmol N m^{-2} $year^{-1}$ or 3.0 g C m^{-2} $year^{-1}$. Release of NH_4^+ comprised more than 80 % of the total.
Denitrification was about 70 mmol N m^{-2} $year^{-1}$ or 1.0 g N m^{-2} $year^{-1}$ and was thus 25 % of the total release of inorganic N (NH_4^+, NO_3^- and N_2), which comprised about 280 mmol N m^{-2} $year^{-1}$ or 3.9 g N m^{-2} $year^{-1}$. Blackburn and Henriksen (1983) similarly estimated that denitrification comprised 22 % of total release of inorganic N in the western part of the Kattegat area. In the present study we were able to demonstrate that denitrification in a period immediately after sedimentation in spring was up to 55 % of the total release, compared to only 15 % during the rest of the year (Jensen et al., 1990). Furthermore, it was demonstrated that NO_3^--rich bottom water supplied 85 % of the NO_3^- required for denitrification in the period after sedimentation (April-July), while the importance of the supply from bottom water was less during the rest of the year. On a yearly basis the bottom water supplied more than 50 % of the NO_3^- for denitrification in the sediment.

In conclusion, the study has demonstrated dramatic changes in the N cycling at the sea bottom after the sedimentation of a phytoplankton bloom in early spring. Nutrient fluxes increase (and change direction in the case of NO_3^-) and denitrification is strongly stimulated by both the input of organic matter and by the maintenance of high NO_3^- concentrations in the bottom water during this period. The pool of NO_3^-, which accumulates in the water column during fall and winter has fundamental importance for the N cycling: 1. The bloom of algal productivity in spring is largely dependent on the NO_3^- resource in the surface water. 2. The sedimentation of the bloom in turn stimulates O_2-demanding degradative processes and may cause shortage of O_2 both in the bottom waters and the sediment. 3. The denitrification, which is the single natural sink of N in the sea, is largely dependent on the NO_3^- resource in the bottom water.

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Ministry of the Environment
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The N, P and Organic Matter Research Program 1985-1990

Number C2

Denitrification and Oxygen Metabolism in Stream Sediments

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SUMMARY

Denitrification and oxygen metabolism were studied intensively during a seasonal cycle in Gelbæk, a lowland stream in eastern Jutland, Denmark. A microsensor technique, using small electrodes to measure nitrous oxide (an intermediate of bacterial denitrification) and oxygen, was developed to study the processes (specific activity, zonation, regulation, etc.) in the surface sediment. Degradative processes by bacteria, notably the respirations with oxygen and with nitrate (denitrification), were shown to increase soon after a spring bloom of organic matter production by benthic micro-algae. Denitrification rates remained high throughout the summer provided that the NO_3^- concentrations in the water column and thus the potential diffusion to the sediment were also high. Actual denitrification rates were also dependent on the depth of oxygen penetration, which determines the upper boundary of the oxygen-free denitrification zone. A model, which predicted denitrification rates from oxygen uptake rates in the sediment (a reflection of oxygen penetration) and concentrations of oxygen and NO_3^- in the water column was developed.

1. INTRODUCTION

Danish streams are often rich in NO_3^- and there is much interest in studying the capacity of NO_3^- removal by denitrification in the stream sediments. A community of microalgae typically developes where the water depth is low enough to allow light penetration to the bottom. Such a bloom of benthic primary production in spring could thus have significant effects on the microbial metabolism in the streambed; increased availability of organic substrate should stimulate denitrification but NO_3^- concentrations may at the same time depend on e.g. NO_3^- diffusion from the water column or NO_3^- production by a nitrification process within the surface sediment.
A major purpose of this study was to measure a natural, seasonal cycle of denitrification rates in a NO_3^--rich stream and to describe the primary control factors of the process.

This rapport summarizes the most important results from the study and more details must be sought in the publications which have been published:

Christensen, P.B., L.P. Nielsen, N.P. Revsbech og J. Sørensen (1989) Microzonation of denitrification activity in stream sediments as studied with a combined oxygen and nitrous oxide microsensor. Appl. Environ. Microbiol. 55, 1234-1241.

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Nielsen, L.P., P.B. Christensen, N.P. Revsbech og J. Sørensen (1990) Denitrification and oxygen respiration in biofilms studied with a microsensor for nitrous oxide and oxygen. Microbial Ecol. 19, 63-72.

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2. MATERIALS AND METHODS

Sampling of intact cores from the stream sediment in Gelbæk was done at regular intervals of 2-4 weeks during 1987.

For detailed studies of the control of O_2 metabolism (respiration and photosynthesis) and denitrification a new microsensor (micro-electrode) was developed to measure O_2 and N_2O concentration profiles in the sediment at very high resolution (Revsbech et al., 1988).

In cores, which were incubated with the inhibitor C_2H_2 to block reduction of N_2O to N_2O by the denitrifying bacteria (acetylene blockage technique), the N_2O produced by denitrification activity must distribute in the surface sediment according to simple diffusion-reaction kinetics. Using the microsensor during the incubation with C2H2 both the zone and the rate of N_2O production (denitrification) could be determined with great precision as indicated by a curvature of N_2O profile; by comparison, a linear concentration gradient (diffusion only) is observed in the oxic layer where denitrification is absent (Christensen et al., 1989).

Seasonal variation of flux rates of O_2 and NO_3^- (transport rates between sediment and water phase) were determined in cores incubated at near-in situ conditions of temperature, light (200 µEinstein m^{-2} s^{-1}) or darkness, and stirring of the water phase (Christensen et al., 1990). Oxygen uptake in dark was assumed to represent the O_2 respiration rate, while the sum of O_2 uptake in dark and O_2 release in light was assumed to represent the photosynthesis rate.

In the seasonal study, denitrification rates were determined in cores incubated with C_2H_2 to inhibit the N_2O reduction by denitrifying bacteria; near-in situ conditions were used as described above. The determination of N_2O accumulation (denitrification) by this whole-core technique (Sørensen, 1978) agreed well with that obtained by the micro-sensor technique described above.

3. RESULTS

3.1. Microzonation of O_2 respiration and denitrification

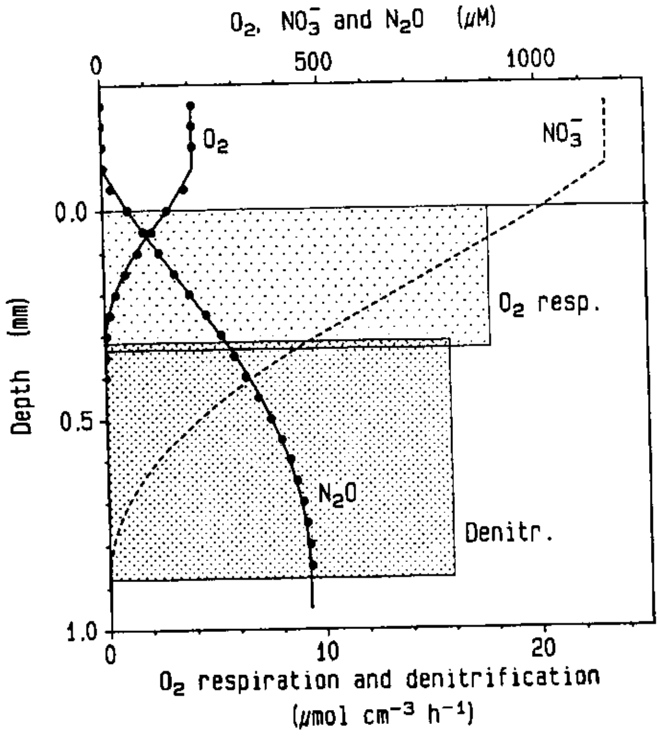

Figure 1. Steady state profiles of O_2 and N_2O in a biofilm incubated with C2H2. (See text for determination of profiles of NO_3^- concentration, O_2 respiration and denitrification).

Figure 1 shows steady state profiles of O_2 and N_2O in C2H2-treated biofilm (dense community of micro-algae, bacteria, protozoa, etc.) from a shallow stream. The biofilm was incubated in the dark and the O_2 profile (curvature) indicates consumption (O_2 respiration) in the upper 0-0.3 mm. Nitrous oxide accumulated with depth and reached a constant maximum concentration at about 0.9 mm depth; the profile was linear in the upper 0-0.3 mm, however, indicating absence of denitrification in this oxic surface layer. Activities of O_2 respiration (0-0.3 mm depth) and denitrification (0.3-0.6 mm depth) could be calculated from the curvatures of O_2 and N_2O profiles, respectively; both were constant with depth within their respective activity zones (Fig. 1).

By assuming that NO_3^- was absent below the denitrification zone and that denitrification was the only NO_3^--consuming process in the dark-incubated biofilm, the NO_3^- concentration profile could be calculated from that of N_2O. As shown in Fig. 1 the curvature of the NO_3^- profile indicates that consumption takes place only in the denitrification zone; NO_3^- diffuses into the biofilm from the water column and the oxic surface layer comprises a diffusion barrier for NO_3. The linear NO_3^- gradient in the oxic layer therefore also suggests that NO_3^- production (nitrification) is of minor importance compared to the NO_3^- supply by diffusion from the water phase.
Figure 2 shows a similar incubation, including a parallel incubation in the light (200 µEinstein m^{-2} s^{-1}), of sediment from Gelbæk.

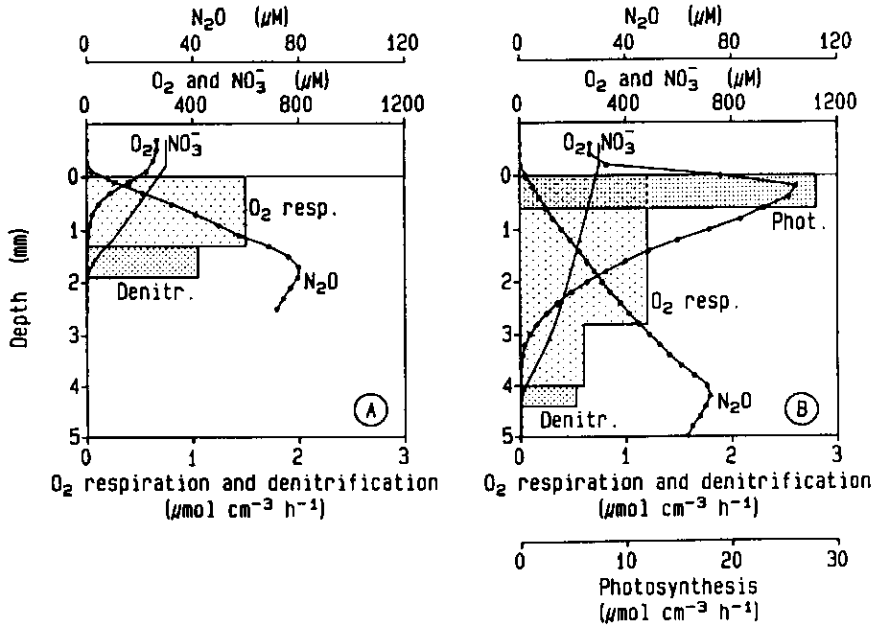

Figure 2. Effect of photosynthetic O_2 production on denitrification in Gelbæk sediment. A: Dark. B: Light (200 µEinstein $m^{-2}s^{1}$).

The light-incubated sediment clearly demonstrates that photosynthesis by the microalgae results in a marked accumulation and deeper penetration of O_2 (Fig. 2 B). The extended O_2 profile forced the denitrification zone into deeper layers, in this case from about 1.2-1.8 mm depth to about 4.0-4.4 mm depth. It was easy to see that the deeper oxic zone (longer distance of diffusion) resulted in a lower NO_3^- concentration gradient (and thus a lower supply rate) at the upper edge of the denitrification zone. This explains many earlier observations of lower denitrification rates in sediments in the light than in the dark (Andersen et al., 1984; Christensen and Sørensen, 1986; Jørgensen and Sørensen, 1988; Sørensen et al., 1988).

3.2. Seasonal variation of O_2 respiration and denitrification

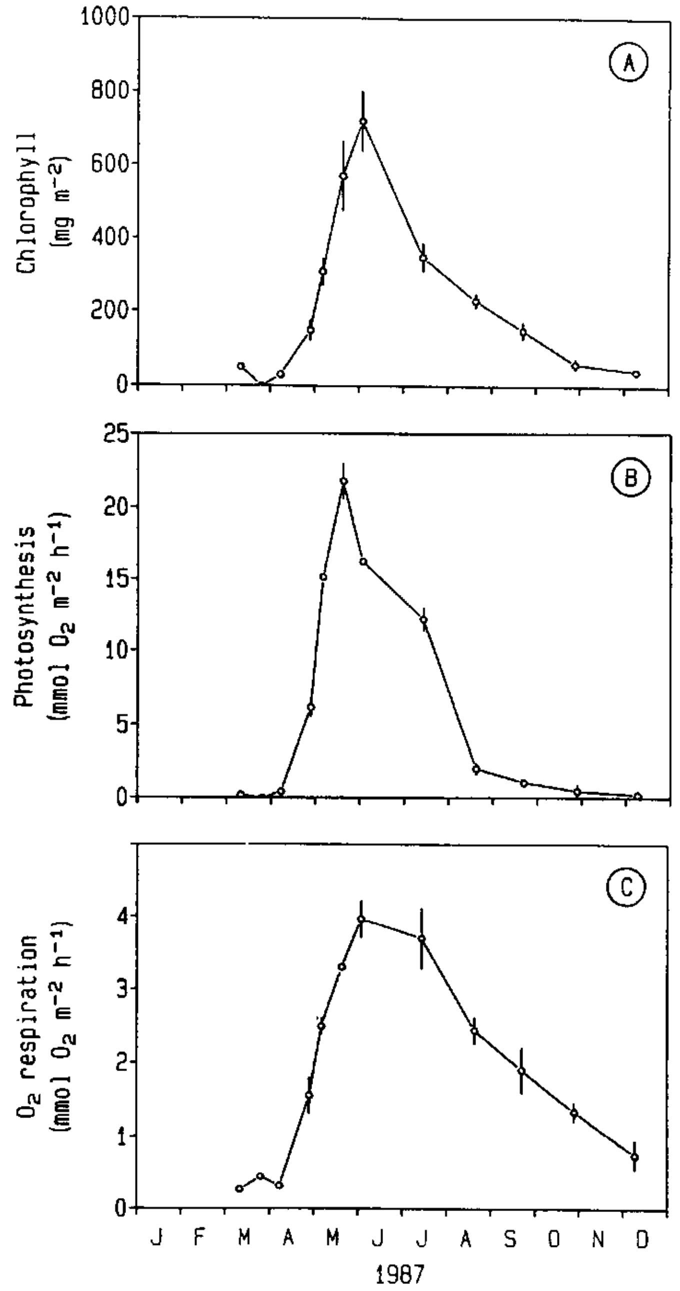

Figure 3. Seasonal variation of chlorophyll content (upper 5 mm) (A) and rates of photosynthesis in light (200 µEinstein m^{-2} s^{-1}) (B) and O_2 respiration in dark (C) in Gelbæk sediment.

The study of seasonal patterns of primary production, O_2 respiration and denitrification in the streambed is illustrated in Fig. 3 and described in the following. A rapid development of the benthic micro-algal community in spring (April-May) can be observed from the chlorophyll content in the surface sediment (upper 5 mm); maximum development was observed ultimo May, after which the content decreased throughout the summer and fall (Fig. 3 A).
Photosynthesis rates showed a similar pattern: rapid development to a peak value in May, followed by a decrease throughout the summer (Fig. 3 B).

As expected the development of the algal community stimulated the degradative processes, e.g. O_2 respiration and denitrification, in the sediment. Oxygen respiration rates (dark incubation) increased during the spring period (April-May) and decreased throughout the rest of the summer and fall (Fig. 3 C). It may be noted that O_2 respiration rates were maintained at a relatively high level in the fall compared to photosynthesis rates; this is clearly a reflection of a shift to predominantly heterotrophic community structure at this time of the year.

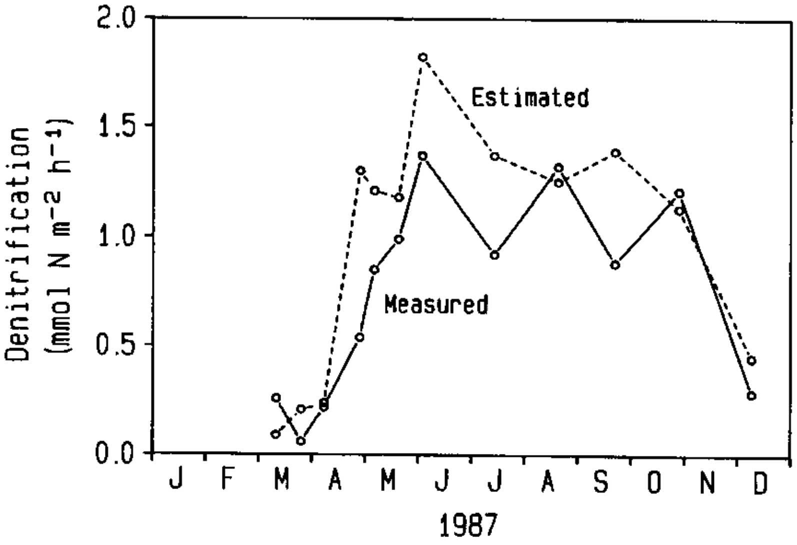

Figure 4. Seasonal variation of measured and calculated rates of denitrification in dark in Gelbæk sediment. (see text on calculation of denitrification rates from O_2 uptake rates and concentrations of O_2 and NO_3^- in water phase).

As shown in Fig. 4 measured denitrification rates (dark incubation) also increased in the spring (April-May) and reached a peak value ultimo May. (The difference between measured and estimated denitrification rates indicated in Fig. 4 is explained below). As for O_2 respiration, relatively high rates were maintained throughout the summer and fall; the increasing NO_3^- concentration in the water column in the fall may explain the surprisingly high denitrification rates at this time of the year (Nielsen et al., 1990).

Reduced denitrification rates were observed in the cores incubated in the light (200 μEinstein m^{-2} s^{-1}); most pronounced was the effect during the period of high primary production in spring (April-May) when the rates in the light were only about 20 % of those in the dark; later in the summer and fall the degree of inhibition gradually became smaller (Nielsen et al., 1990).

4. DISCUSSION AND CONCLUSION

4.1. Importance of O_2 respiration and denitrification

On an annual scale denitrification was of little importance in terms of the removal of N from the Gelbæk stream. In 1987, the total denitrification was estimated to be 600 kg N ha^{-1} of stream $year^{-1}$, which corresponds to less than 1 % of the amount being transported to the stream from its surrounding 1000 ha of agricultural land (Kronvang and Thyssen, 1987). In spring, however, the removal may transiently be as high as 47 %; the value was 6 % at a maximum when calculated for the summer period (May-September). These results are in agreement with our previous study in 1985, which gave values of less than 1 % on an annual scale and about 10 % calculated for the summer period (Christensen et al., 1986).

4.2. Regulation of O_2 respiration and denitrification

Both the detailed studies with microsensor technique and the seasonal records using a conventional whole-core technique demonstrated that the annual pattern of O_2 respiration and denitrification were primarily controlled by the availability of new organic substrate from the primary production in spring. On the daily scale, however, the rates were also controlled by distribution of O_2 and NO_3^- in the water column and the surface sediment.

We developed a simple model for prediction of denitrification rates in the dark based on rates of O_2 respiration (O_2 uptake in the dark) and concentrations of O_2 and NO_3^- in the water phase. The model, which is presented in detail elsewhere (Christensen et al., 1990), gave estimates of denitrification rates which were very close to the measured ones (Fig. 4) and may thus be used to predict denitrification from relatively simple assays. Most important in this model is the observation that NO_3^- production (nitrification) is negligable and denitrification is dependent on NO_3^- supply by diffusion from the water phase. As noted by the microsensor technique the denitrification activity is therefore dependent on both the NO_3^- concentration in the water column and the thickness of the O_2-containing surface layer (in turn controlled by O_2 respiration rate), which comprises a diffusion barrier for the NO_3^-.

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Number C3

Eutrophication: Effects in Marine Plant Communities

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Resume.

The objective of this project was to examine the effect of nutrient enrichment on composition, depth distribution, biomass and production of plant communities and the subsequent effects on oxygen conditions in a shallow estuary. This was accomplished by comparing two large broads with different nutrient availabilities in the outer and inner parts of Roskilde Fjord, Denmark. By tradition, nutrient loading is expected to increase total plant biomass and primary production. However, even nutrient-poor coastal areas may be highly productive due to dense stands of macrophytes. We therefore hypothesized that increasing phytoplankton production at high nutrient loadings would be more or less balanced by reduced macrophyte production. This hypothesis was supported by our findings. Nutrient enrichment favours phytoplankton at the expense of benthic microalgae and perennial macrophytes. Total plant biomass was reduced with increasing nutrient richness but total production was unchanged. The oxygen regime was highly variable at different depths. Oxygen depletion occurred both in shallow and deeper regions and tended to be more severe in the nutrient-rich region due to qualitative and not quantitative changes in total primary production. Finally, we established empirical relationships between nutrient richness and macrophyte depth penetration based on data from a variety of coastal Danish areas. Increasing nitrogen richness stimulated phytoplankton biomass and reduced the depth penetration of eelgrass and macroalgae while no relationships were found with phosphorus richness.

1. Introduction.

The photoautotrophic communities in coastal marine areas are formed by phytoplankton, periphytic microalgae, fast-growing annual macroalgae (ephemeral algae), slow-growing perennial macroalgae and rooted macrophytes (mostly seagrasses). Under natural, nutrient-poor conditions phytoplankton biomass is low, but the slow-growing macrophytes form dense and very productive communities in shallow waters down to 10-20 meters. The primary production of such communities may exceed 1000 gC m^{-2} y^{-1} (Mann, 1972; McRoy and McMillan, 1977). At greater depths, the benthic macroflora is replaced by benthic microalgae extending further down.

Nutrient enrichment of coastal areas stimulates phytoplankton biomass and production and accumulation of ephemeral algae in shallow waters, while depth penetration of perennial macroalgae and rooted macrophytes is reduced (Sand-Jensen and Borum, 1983). Accordingly, the integrated effect of eutrophication on community composition and total primary production of shallow, coastal areas is complex.

In this paper we present some of the general findings from a comparative study of composition, depth distribution, biomass and production of photoautotrophic communities and the subsequent effects on oxygen conditions in two basins of different nutrient status in a Danish estuary, Roskilde Fjord (Borum, 1985).

2. Localities and methods.

Roskilde Fjord is connected to the open waters of Kattegat and consists of several basins of similar morphometry but different nutrient loading and richness. The mean depth of the two selected basins was 3.2 and 3.4 m. Both regions were dominated by dense eelgrass (*Zostera marina* L.) meadows before eutrophication accellerated in the 1950's (Nielsen, 1951). Today, the estuary is more nutrient-rich and exhibits a marked nutrient gradient (Borum, 1985). The differences in nutrient richness between the two study areas represent a doubling of the annual mean concentration of total nitrogen (0.75 and 1.61 mgN l^{-1}) and a 5-fold increase of total phosphorus (0.11 and 0.57 mgP l^{-1}). Though both areas are eutrophied we refer to the basins as "nutrient-poor" and "nutrient-rich", respectively. Nitrogen is the most important limiting nutrient for the phytoplankton biomass and growth in both areas.

Data on composition, depth distribution, biomass and production of plant communities were sampled along transects covering the depth interval from 0 to 5 meters in each basin. Sampling was performed by SCUBA diving in spring (April), summer (June-July) and autumn (September). Diel changes in oxygen concentrations were measured along the same transects and during the same periods.
More general relationships between nutrient richness, phytoplankton development and depth penetration of benthic macrophytes were established with data from a variety of coastal, Danish areas kindly provided by regional environmental authorities.

3. Results.

3.1. Composition and depth distribution of plant communities.

Phytoplankton biomass was low in the nutrient-poor basin 1 and the Secchi-depth ranged between 4 and 5 m throughout the year. In the more nutrient-rich basin 2, Secchi-depth varied between 2 m in spring and 3.5 m during summer and autumn. Eelgrass (*Zostera marina*) was an important component in both areas but occurred at higher densities and extended further down (5 versus 3 m) in basin 1. Ephemeral macroalgae were found down to 5 m depth in both areas, and genera like *Cladophora* and *Chaetomorpha* formed floating mats in shallow water of basin 1 during summer. Mass accumulation was not observed in basin 2, but ephemeral macroalgae like *Ulva* and various red algae were as abundant as perrenial macrophytes.

Data from a large number of coastal Danish areas showed that depth penetration of macrophytes can be predicted from nutrient richness (Figure 1). No significant relationships were found with phosphorus, but with increasing nitrogen richness, phytoplankton biomass increased, the Secchi-depth decreased, and the lower depth limit of eelgrass, perennial brown algae and other macroalgae decreased.

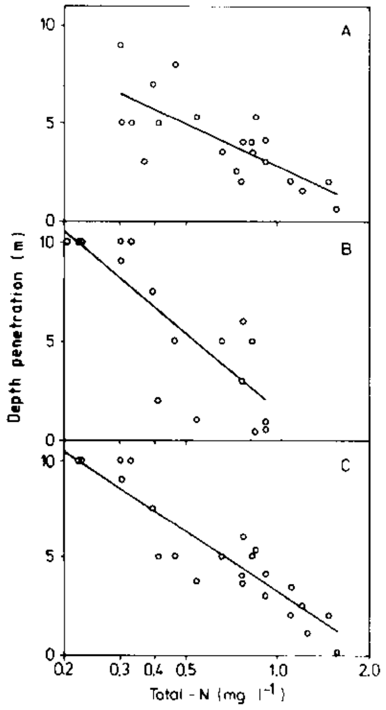

Figure 1. Depth penetration of (A) eelgrass, (B) perennial brown algae, and (C) other macroalgae versus total-N (log-scale). Mean values (March 1 - October 31) from different Danish coastal areas. After Nielsen _et_ _al_. (1989).

3.2. Plant biomass.

Total plant biomass varied substantially with season and with water depth. In both basins, maximum biomass was found in autumn and at depths of 2-3 m. Mean biomass (time- and depth-weighted annual mean) of the entire photoautotrophic community was higher in the nutrient-poor basin than in the nutrient-rich (28.5 versus 17.6 gC m^{-2}; Table 1). Perennial macrophytes contributed with more than 65% of total biomass in basin 1 but only with 47% in basin 2. Although the biomass of ephemeral macroalgae was approximately the same in the

Table 1. Mean biomass of different groups of plants in the nutrient-poor basin 1 and the nutrient-rich basin 2 in Roskilde Fjord.

	Plant biomass in gC m^{-2}	
	basin 1	basin 2
Perennial macrophytes	19.2	8.3
Ephemeral macroalgae	7.4	6.7
Periphytic algae	1.3	0.9
Phytoplankton	0.6	1.7
Total	28.5	17.6

two areas, their relative contribution to total plant biomass was higher in basin 2. Periphytic algae were more abundant than phytoplankton in the nutrient-poor area, while the opposite was observed in the nutrient-rich area. In both basins, periphyton and phytoplankton accounted for less than 10% of total plant biomass.

Table 2. Total annual primary production and the contribution of different groups of photoautotrophs in the nutrient-poor basin 1 and the nutrient-rich basin 2 in Roskilde Fjord.

	Primary production in gC m^{-2} y^{-1}	
	basin 1	basin 2
Perennial macrophytes	73	38
Ephemeral macroalgae	49	46
Periphytic algae	44	23
Phytoplankton	154	197
Total	320	304

3.3. Primary production.

In spite of major differences in nutrient richness, total annual production was approximately the same in the two areas (320 gC m^{-2} y^{-1} in basin 1 and 304 gC m^{-2} y^{-1} in basin 2, Table 2). However, the contribution of different groups of photoautotrophs changed markedly with nutrient richness. Phytoplankters were responsible for 48% of total primary production in basin 1 and 65% in basin 2. Nutrient enrichment also increased the relative contribution of ephemeral macroalgae, but the contributions of perennial macrophytes decreased from 23% in basin 1 to 12% in basin 2 and of periphytic microalgae from 14% in basin 1 to 8% in basin 2.

3.4. Diel oxygen changes.

The high biological activity of the shallow estuary resulted in substantial changes in oxygen concentrations over the diel cycle (Figure 2). The largest oxygen amplitudes were found during summer, but large changes also occurred during spring and autumn. The maximum amplitude was measured in shallow water (< 1 m). Differences between the two basins were small, but during summer oxygen depletion was more severe in the bottom waters of the most nutrient-rich basin, irrespective of the depth of the water column.

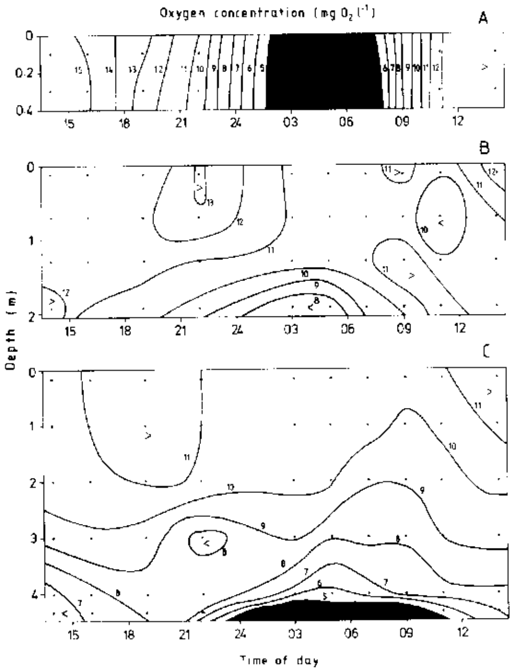

Figure 2. Depth-time diagrams of diurnal concentrations in oxygen at sampling stations of different maximum depth. Example from the nutrient-rich basin 2, June 9-10.

4. Discussion and conclusion.

In this study, we found that a doubling of nutrient richness between the two basins caused major changes in depth penetration of benthic macrophytes, biomass and production of the different groups of photoautotrophs, and a tendency towards more severe oxygen depletion. The effects of nutrient enrichment on marine photoautotrophic organisms have been the subject of numerous studies within the last two decades. However, most of this work has, by tradition, focused on phytoplankton responses. Nutrient enrichment is known to stimulate phytoplankton biomass and production, as was also observed here, and the occurrence of oxygen depletion has been interpreted as a direct result of the increase in primary production. Though such a simple scheme may adequately describe the situation in deep lakes and open marine areas, the integrated effect of nutrient enrichment in shallow lakes, estuaries and other coastal areas with a potential for significant benthic primary production is more complex. Under natural nutrient-poor conditions, slow-growing perennial macrophytes dominate because of their ability to accumulate nutrients over time and their ability to conserve nutrients within populations and individual plants (Borum _et al_., 1989; Pedersen _et al_., in prep.). With increasing nutrient richness phytoplankton biomass accumulates, light attenuation in the water column increases and the depth penetration of benthic flora is reduced. The stimulation of phytoplankton production, therefore, occurs at the expense of benthic production of macrophytes and periphyton. The production of ephemeral macroalgae was not stimulated in Roskilde Fjord, though this has been found in other shallow estuaries. Overall, nutrient enrichment reduced total biomass but did not change total primary production, and we believe this is a general response for all shallow water systems with a large benthic production of rooted macrophytes or perennial macroalgae. Accordingly, the more frequent occurrence of oxygen depletion is not related to a higher production of organic matter as such. With eutrophication, photosynthetic activity is restricted to fast-growing attached organisms and phytoplankton in the surface zone. The shift towards the more productive but also more readily metabolized ephemeral macroalgae increases diel oxygen amplitudes in shallow water. The increase in phytoplankton production in the surface zone of deeper water tends to spatially separate oxygen production in the upper part of the water column from oxygen consumption in the sediment and bottom water. Therefore, oxygen depletion occurs more frequently in nutrient-rich regions. It may be surprising that oxygen depletion is also observed in shallow water, but this only reflects that the high sediment respiration can easily strip oxygen from the small water volume, and that reaeration is to slow to compensate for the oxygen demand in these highly productive systems.

We conclude, that nutrient enrichment of macrophyte dominated, coastal areas leads to predictable reductions in the depth penetration of benthic macrophytes. Phytoplankters become more important and the turnover of plant biomass accellerates, but total primary production does not necessarily change and oxygen depletion may be caused by qualitative rather than quantitative changes in primary production. Ecologists and modellers working with shallow water systems should be aware, that nutrient enrichment may lead to increased, unchanged or even reduced total primary production depending on the potential for benthic production in the given area.

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Internal Loading in Shallow, Eutrophic Lakes

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INTERNAL LOADING IN SHALLOW, EUTROPHIC LAKES.

0. Summary

Internal P-loading delays lake recovery

Danish shallow, eutrophic lakes have a pronounced internal phosphorus (P) loading in summer which delays lake recovery from earlier external P-loading. The aim of the present project was to clarify the factors that control sediment P-release in these lakes. Due to their polymictic nature the study was focussed on P release from aerobic sediment surfaces.

Fe:P ratio in sediment indicates P efflux

In 15 studied lakes P release correlated negatively with the iron:P ratio of the surface sediment. We concluded that the possibility to control P efflux from aerobic sediments begin at iron:P ratios above 15 (by weight) - a finding that may be valuable for future lake management.

Importance of oxic surface layer

Temperature most important

Four lakes were studied intensively in 1987. Addition of sestonic matter, increased temperature and increased nitrate concentration changed the thickness of the oxidized surface layer and the P-release rates, in sediment cores incubated at "in situ" conditions. Lake water temperature seemed to be the most important "controlling factor" for the seasonal variation in P-release when compared with lake water pH and nitrate.

Dual effect of nitrate

High nitrate concentrations in lake water showed a dual effect on sediment P-release. In January, and to some extent in May, nitrate increased the thickness of the oxidized surface layer and reduced P-release, while in September raised nitrate concentrations enhanced the sediment P-release. This was probably due to a general stimulation of bacterial processes.

1. Introduction

In shallow temperate lakes, P released from the sediment often constitutes a major fraction of the total P-load to the trophogenic zone during summer (e.g. Andersen 1982, Ryding 1985). This P release is termed "internal P-loading" in contrast to the "external P-loading" which refers to P supplied with the inlet water. As P very often is the "growth limiting nutrient" for phytoplankton in lakes, an increased P supply to the lake water leads to eutrophication of the lake (e.g Wetzel 1983).

Therefore the internal P-loading is identified as an important mechanism in delaying recovery of shallow lakes after reduced external P-loading (Ryding 1985, Marsden 1989, Sas 1989), although homeostasis in the biological structure may be important too (e.g. Benndorf 1987, Jeppesen et al. 1990).

Few studies in shallow lakes

Most studies of internal loading have focussed on dimictic lakes where thermal stratification often creates anoxic conditions in the bottom water. In shallow, polymictic lakes oxygen is present at the sediment surface most of the season and this study therefore concentrates on phosphate release from oxidized sediment surfaces.

Aim of the study

As the shallow, eutrophic lake is the most common lake type in Denmark (e.g. Jensen et al. in press) and many of these lakes are in a state of recovery after reduced external P-loading, the aim of this study was to identify the factors controlling the internal P-loading in shallow lakes. Some of these mechanisms have been discussed in reviews by Boström et al. (1982, 1988) and Marsden (1989).

4 lakes studied intensively

Four shallow lakes with different contents of iron, calcium and phosphorus in the sediments, were studied during 1987 with respect to mass balances and sediment P-release. In a laboratory incubation system the effect of the chemical composition of lake water on sediment PO_4^{3-} release was studied by using undisturbed sediment cores. Additional 11 lakes were sampled in 1988 for a study of sediment characteristics and sediment PO_4^{3-} release.

2. Sampling sites and methods

2.1 Sampling sites

Relations between sediment characteristics and sediment P-release rates (section 3.9) were studied in 15 lakes (table 2.1). The fractional composition of sediment P-pools are shown in Fig. 2.1. Four lakes, number 13-16, (which will be referred to as "the NPo-lakes") were studied intensively during 1987 and 1988. Characteristics of these lakes are given in table 2.2.

Table 2.1: Characteristics of surface sediment (0-5 cm depth) in 15 lakes. Data set 12 and 13 represent two measurements in the same lake (May 87 and May 88). Abbreviations: %DM: Percent of dry material per wet unit; %LI: Percent loss on ignition per dry weight; Fe: Total iron; Ca: Total calcium; TP: Total phosphorus; all in mg gDW^{-1}.

Nr.	Lake	% DM	% LI	Fe	Ca	TP
1	Gudme Sø	2.6	54	5.7	85	3.2
2	Nørre Sø	4.3	40	8.9	117	1.3
3	Dallund Sø	10.3	19	4.5	221	0.93
4	Dons Nørre Sø	12.4	22	17	167	1.2
5	Engelsholm Sø	63.3	2.7	7.9	1.7	0.43
6	Skanderborg Lillesø	3.5	47	7.0	134	3.8
7	Dystrup Sø	4.2	57	5.1	16	1.6
8	Alling Sø	7.2	33	73	18	3.4
9	Hinge Sø	15.2	22	159	25	4.9
10	Almind Sø	10.6	23	78	8.3	1.2
11	Slåen Sø	10.0	27	98	8.2	1.2
12	Arreskov Sø, 88	5.9	35	14	70	1.3
13	Arreskov Sø, 87	5.9	35	14	70	1.3
14	Søbygård Sø	8.2	25	58	129	6.8
15	Kvind Sø	5.5	37	30	8.6	3.1
16	Væng Sø	5.5	35	64	10	4.6

Tabel 2.2 Characteristics of the four NPo-lakes.

	Area km^2	Mean-depth Z_m m	Max.-depth Z_{max} m	Water retension T_w month
Arreskov Sø	3.88	2.0	3.2	17
Kvind Sø	0.15	1.8	2.5	0.5
Søbygård Sø	0.39	1.0	2.0	0.9
Væng Sø	0.15	1.2	1.9	0.6

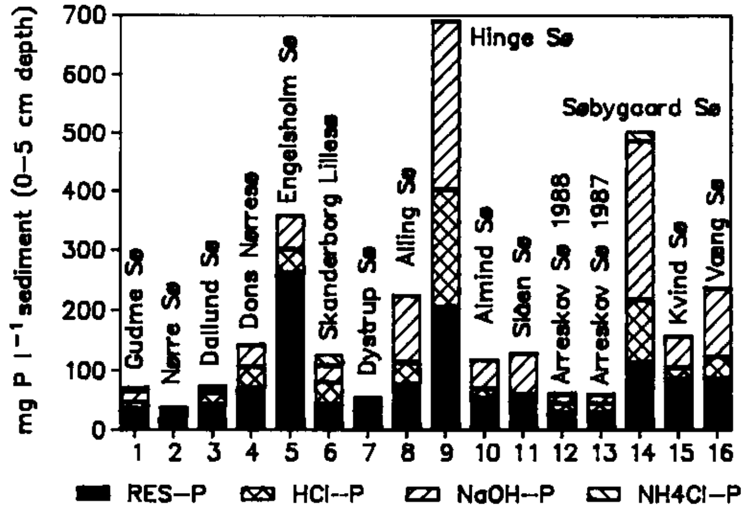


Figure 2.1
Phosphorus pools in surface (0-5) cm sediment of 15 Danish lakes (from Table 2.1) Fractionation of P-pools after Hieltjes & Lijklema (1980).

NH_4Cl-P: Loosely sorbed PO_4^{3-},
NaOH-P: Fe/Mn/Al bound PO_4^{3-}
HCl-P: Ca/Mg-bound P,
Res.-P: Organic P in various pools.

2.2 Methods

Sampling

Mass balances of P and N in the four NPo-lakes were measured in 1987 by sampling lake inlets and outlets at least 21 times. Lake water was sampled at a mid-lake station at the same times. Soluble reactive phosphorus (SRP), nitrate, ammonium, total phosphorus (TP) and total nitrogen (TN) were measured by Danish standard methods. Phytoplankton biomass and species composition were determined by microscopic methods.

Undisturbed sediment cores were sampled in perspex tubes (inner diameter 5.2 cm). For sediment characteristics 5 cores from each station were pooled before analysis.

Incubation

The cores were incubated in a continuous flow-through system for one day before measurements. Subsequently the flow was stopped and SRP flux was calculated from the change in SRP in the water during 24 hours. The measured rates are termed gross-release rates of SRP. When testing effects of changed conditions in the overlying water, fluxes were measured by this method twice a week during 3-4 weeks. Between the measurements, cores were conditioned by the flow through system. The water was always kept fully aerated. For SRP-flux measurements 5 individual cores were used at each of the incubation conditions.

Thickness of oxic surface layer

During long-term incubations the sediment cores evolved a distinct light surface layer, overlying a black layer of ferrous sulfide precipitations. The interface between the two layers was located at a redox potential (E_h) of 231 +/- 6 mV (based on 40 measurements) (Fig. 2.2). This potential is characteristic for the reduction of Fe^{3+} to Fe^{2+}. At the end of each experiment the thickness of the oxidized surface layer was measured and mean values for the five cores in each group were calculated.

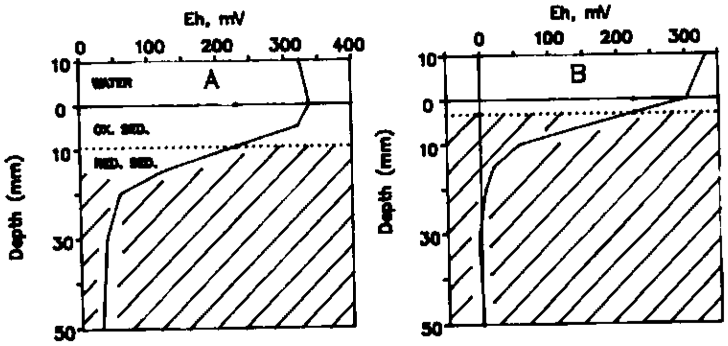


Figure 2.2
Redox profile and thickness of oxic surface layer in Lake Arreskov (means of 5 cores).
A: 7 °C.
B: 21 °C.

Sorption kinetics

Phosphate sorption kinetics of suspended surface sediments were measured in artificial lake water and the sorption characteristics were described by Langmuir sorption isotherms (Fig. 2.3) (Jensen et al. in press, Froelich 1988).

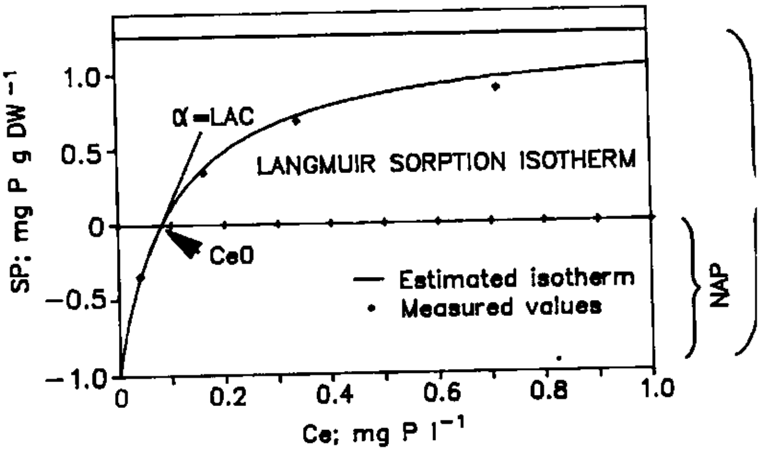


Figure 2.3
Example of a Langmuir sorption isotherm.
Ce: Equilibrium conc. of SRP
CeO: Cross-over concentration
SP: Sorbed SRP
NAP: Native Adsorbed SRP.
PSC: Phosphate Sorption Capacity LAC: Lineary Adsorption Capacity. (e.g. Froelich 1988).

3. Results and discussion

3.1 Mass balances and gross release of phosphate from the sediments

Reliable measurements of SRP-release

Measured rates of gross sediment P-release agreed well with the calculated net release rates in periods where no sedimentation took place (e.g. Fig. 3.6, Jensen & Andersen in press and Jensen & Andersen in prep.).

Seasonal variation explained by temperature

Gross SRP-release rates were measured 6 times in 1987 in the four NPo-lakes and, when comparing with lake water temperature, pH, nitrate and phosphate, only temperature correlated with the release rates ($r^2 > 0.7$; $p < 0.05$) in three of the lakes. In Lake Væng, pH showed the best correlation with the seasonal variation in sediment SRP-release ($r^2 = 0.61$; $p < 0.1$).

All four lakes showed pronounced internal P-loadings during summer and on an annual scale two of the lakes (Lake Arreskov and Lake Søbygård) showed a net release of P from the sediment (table 3.1).

Tabel 3.1

Area specific loading and retention of P and N in the four NPo-lakes before and after reduction in external loadings ($g/m^2*år$). (Including data from NERI and the counties of Fyn and Århus).

	Arreskov Sø		Kvind Sø		Søbygård Sø		Væng Sø	
	1974	1987	1978	1987	1978	1987	1978	1987
P-loading	0.40	0.22	12.3	10.7	27.7	3.28	5.5	1.5
P-retention	0.30	-0.34	0.87	0.13	15.9	-2.67	1.5	0.9
N-loading	9.34	5.83	273	331	125	118	84	71
N-retention	4.19	2.17	63.3	60	61.2	58.3	31	48

3.2 Temperature effect

In November 1987 sediment cores from the four lakes were incubated at 7, 14 and 21 °C in three parallel incubation systems. Oxygen uptake and SRP-release were measured during 25 days and cumulated values were calculated for each group as average of 5 cores (Fig. 3.1 and Fig. 3.2).

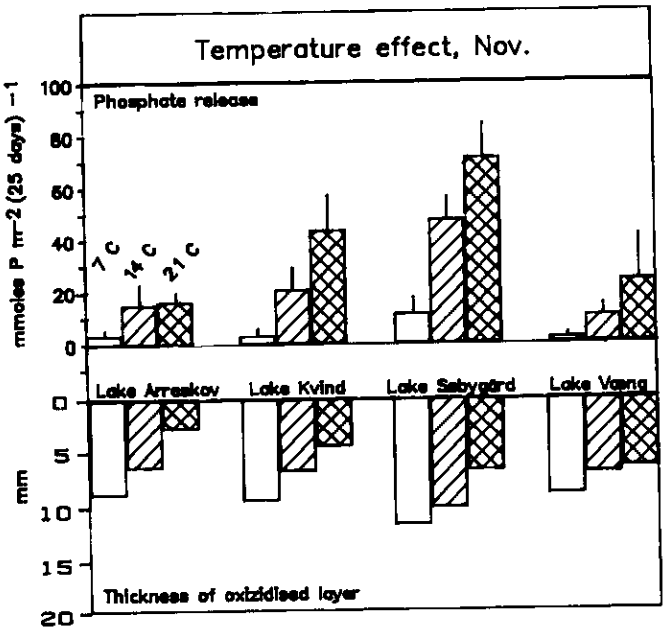


Figure 3.1
Temperature effect on SRP-release during 25 days (upper) and thickness of oxidized surface layer (lower) at the end of incubation.
Vertical lines indicate S.D. of five replicates.

P-release was affected more than oxygen uptake

SRP-release increased with temperature ($p<0.05$) (except in one case in Lake Arreskov). Also the oxygen uptake increased with temperature but, while sediment oxygen uptake showed a Q_{10}-value of 2.1 (as average for all four lakes and all temperature intervals) the corresponding value for SRP release was 4.9. This indicates that increased metabolism induced a mobilisation of minerogenic P-pools in the sediment, so that P-release exceeded the amount that could be expected from mineralisation of fresh organic matter (i.e. Redfield ratio) (Fig. 3.2).

As the oxidized surface layer decreased with increasing temperature (Fig. 3.1) it is reasonable to explain the additional SRP release by a mobilisation of iron-bound P. This also explains why Lake Arreskov, which had the lowest content of iron-bound P (Fig. 2.1), showed the smallest induced effect.

The classical iron theory

The "classical" theory of sediment uptake and release of PO_4^{3-} identifies reactive iron as a key-factor (Einsele 1936, Mortimer 1941, 1942), such that PO_4^{3-} is sorbed onto surfaces of iron oxyhydroxides at the oxic sediment surface and released as Fe^{3+} is reduced to Fe^{2+} when anoxic conditions prevails at the sediment surface.

Also valid for aerobic P-release

Several studies have confirmed that this mechanism is of major importance for P-release from anoxic sediments to the hypolimnion in stratified lakes (e.g. Nürnberg 1988). When the sediment surface is kept oxidized, as in this study, PO_4^{3-} diffusing from deeper anoxic sediment layers is more or less stopped by a narrow oxic surface layer (e.g. Tessenov 1972, Kamp-Nielsen 1974, Carlton & Wetzel 1988). Kamp-Nielsen (1974) found that there was no significant difference on aerobic and anaerobic SRP release rates in Lake Esrum when the temperature was raised to above 17 °C and he explained this result by the vanishing of the oxidized surface layer.

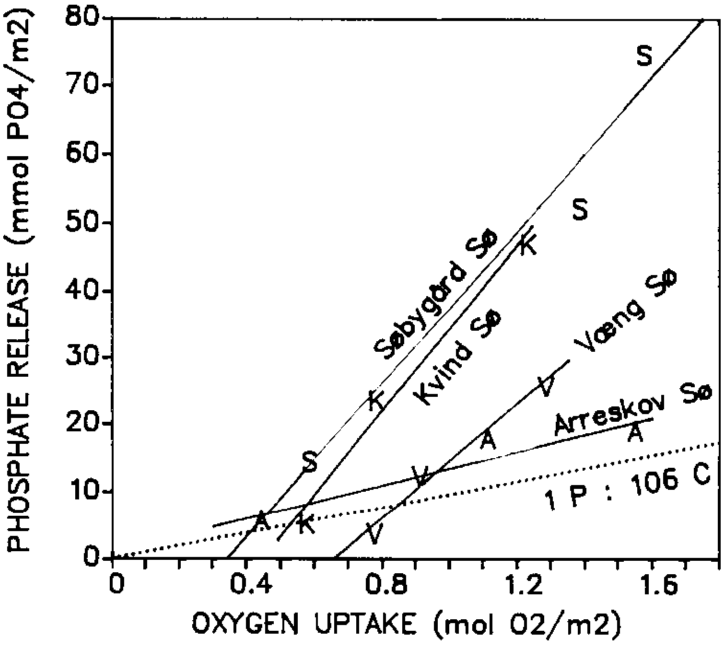


Figure 3.2
Cumulated Oxygen uptake and SRP release during 25 days in the temperature experiment. Each point indicates a mean of five replicates. Dotted line represents theoretical Redfield O_2/P-ratio.

3.3 pH-effect

The capability of the oxidized surface layer to sorb PO_4^{3-} is strongly related to the pool of reactive iron. It is thus reduced at raised pH values (e.g. Jacobsen 1977, Lijklema 1977). This mechanism is based on a simple ion exchange where OH^- occupies the sorption sites on the iron oxide/hydroxide surfaces. Raised pH in lake water has also been found to affect the aerobic SRP release from undisturbed lake sediments (e.g. Andersen 1975).

We studied the effect of pH on SRP release from undisturbed sediment cores twice in the four NPo-lakes (June 1987 and July 1988) and in May 1987 we studied the pH effect on phosphate sorption kinetics.

The sorption kinetic studies showed that pH values above 9 increased the cross-over concentration (CeO) and decreased the buffering capacity (LAC) of the sediment (see Fig. 2.3). This is shown for three of the lakes in Fig. 3.3. Lake Søbygård showed the same tendency but the phosphate release was so high that we could not form an estimate of the cross-over concentration.

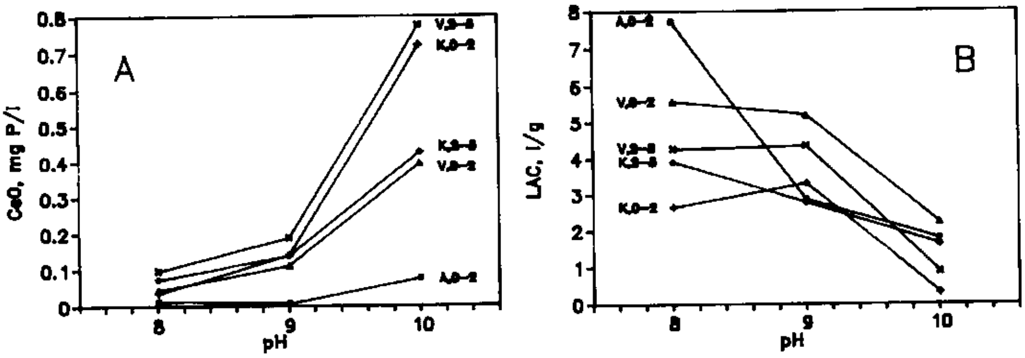


Figure 3.3
pH-induced changes in cross-over concentration (CeO) (A) and buffering capacity (LAC) (B) in 0-2 and 2-5 cm sediment depth in three NPo-lakes. See Fig. 2.3 for explanation of sorption kinetic terms.

pH was not as important as expected

The decrease in sorption capability with increasing pH explained the changes in SRP release in sediment cores from Lake Kvind and to a certain extent from Lake Søbygård (Fig. 3.4). In Lake Arreskov and Lake Væng, however, no significant changes in SRP-release were observed.

One reason for this could be that the experiments were carried out at conditions (summer, high temperature), where the iron-bound P pool already was exhausted due to low redox potentials in the sediment. If so, increased pH is probably only important for sediment P release in the spring (when the reactive iron is still oxidized), or at events of resuspension of surface sediment.

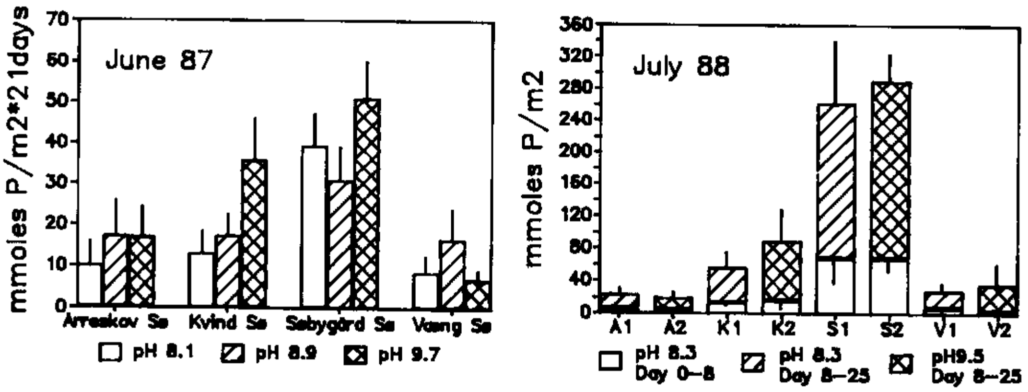


Figure 3.4
Effect of experimentally raised pH on sediment SRP release in the four NPo-lakes sampled in summer 1987 and 1988. Vertical lines indicate S.D. of five replicates.

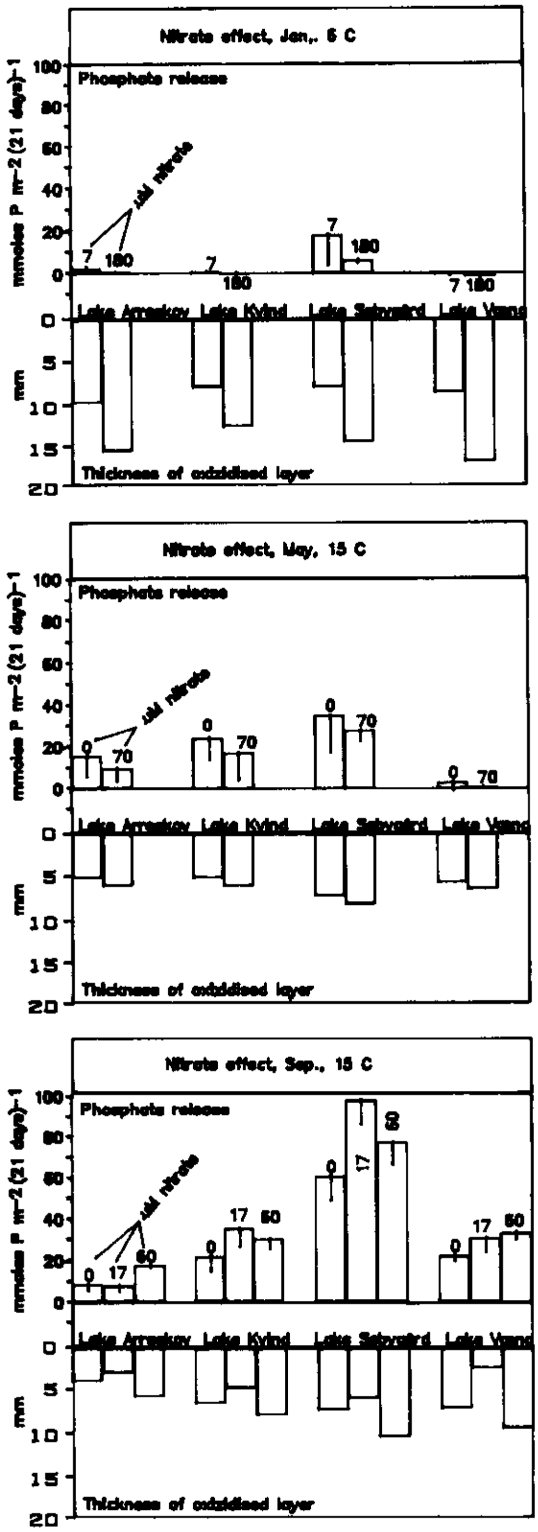
3.4 Nitrate effect

While it is generally accepted that a high nitrate concentration in lake water suppress phosphate release from anoxic sediments of dimictic lakes (Rippl & Lindmark 1978, Boström & Petterson 1982, Foy 1986), the reverse effect has been reported for oxic sediments from shallow, eutrophic lakes (e.g. Boström et al. 1988). In our study we tried to clarify the effect of nitrate on both gross internal loading (this section) and net internal loading (section 3.5) at different seasons, because lake water nitrate showed large seasonal variations in all four lakes.

Dual effect of nitrate

Results from three experiments with undisturbed sediment cores are shown in Fig. 3.5. In January, a high nitrate concentration of 2 mg N l^{-1} decreased the sediment P-release by increasing the thickness of the oxidized surface layer. This effect was less pronounced (and not significant) in May, and in September where lake water nitrate was low, the phosphate release was enhanced by increased nitrate concentration. Nitrate probably stimulated the mineralisation of organic matter in the sediment, as the oxidized surface layer decreased with moderate nitrate addition. At high nitrate addition the oxic surface increased again, but the oxidizing effect of nitrate could not outline the "stimulating effect" even at concentrations of 0.84 mg N/l.

Figure 3.5
Nitrate effect on sediment SRP-release and thickness of oxidized surface layer in the four NPo-lakes at three sampling dates in 1987.
Vertical lines indicate S.D. of five replicates. Nitrate concentrations in the water are also shown in the figure.



In an earlier study, with nitrogen poor sediment from Lake Arreskov, nitrate addition increased both sediment oxygen uptake and sulfate reduction (Jensen & Andersen 1982).

3.5 Impact of blue-green algae abundance on net internal P-loading

Meanwhile, Ryding (1985) found inverse relations between lake water TP and nitrate in shallow, Swedish lakes. In 22 shallow Danish lakes, Andersen (1982) observed no net internal P-loading when lake water nitrate was above 0.5 mg N l^{-1}. Time series of mean summer TP and nitrate in lake water from Lake Arreskov and Lake Kvind also showed inverse relations (Jensen & Andersen in press) and so did the seasonal variations in 1987 (Fig. 3.6).

Because blue-greens have low sinking rates, P accumulates in the lake water

However, the periods when nitrate was absent in the lake water in these two lakes were characterised by blooms of blue-green algae (mainly *Microcystis* spp.) and it was striking that peaks in blue-green algae biomass coincided with peak values of net internal P-loading. A probable explanation was that sedimentation rates were reduced by the positive buoyancy of the *Microcystis* colonies and with continued release of SRP from the sediments this will lead to accumulation of P in the lake water (Jensen & Andersen in press). The possibility that rising colonies of blue-green algae carry organic bound P from the sediment to the lake water also has to be considered (e.g. Trimbee & Harris 1984, Osgood 1988).

This mechanism has so far been overlooked in the discussion of "the nitrate effect" on sediment P-release in shallow lakes.

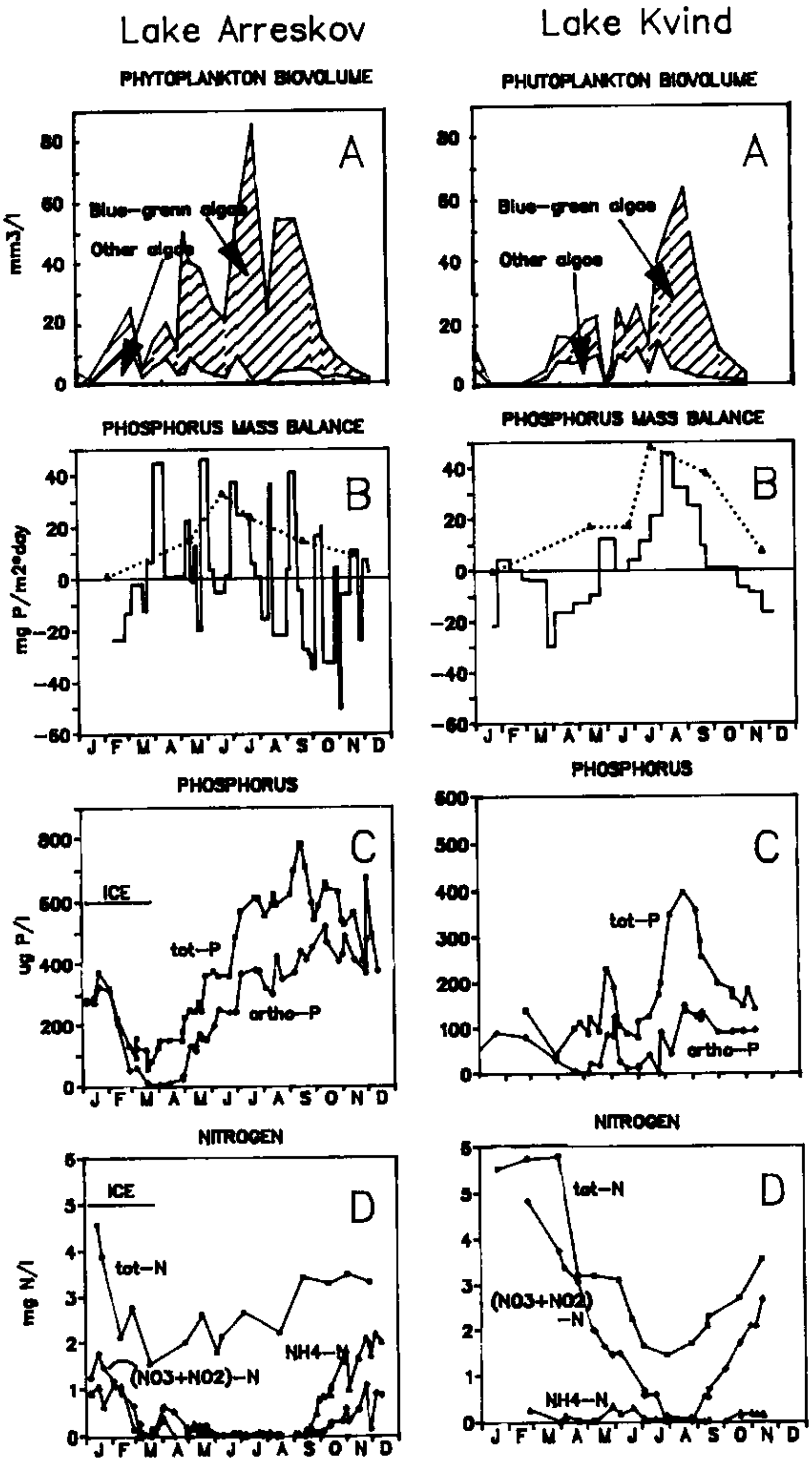


__Figure 3.6__ Phytoplankton biovolume (A), net internal P-loading (solid line) and measured sediment SRP release rates (dotted line) (B), lake water phosphate and total P (C) and lake water nitrate, ammonium and total N (D) in Lake Arreskov and Lake Kvind 1987.

3.6 Effect of "bulk addition" of phytoplankton on nutrient fluxes across the sediment water interface

Rapid changes in structure of planktonic communities followed by large sedimentation inputs to the sediment are common in shallow, eutrophic lakes. These events may cause increased nutrient efflux from the sediment (e.g. Søndergård et al. 1990). We simulated such events by adding known amounts of organic matter (lake water seston) to undisturbed sediment cores and following the nutrient fluxes for a one month period. This experiment was carried out both in august (18 °C) and in November (14 °C).

Elevated nutrient fluxes and sediment oxygen uptake persisted for 2-4 weeks when compared to controls and within this period nearly 75% of the organic matter was mineralized (based on the increased oxygen uptake). This rate of nutrient recovery was twice as high as the leaching rate (of N and P) in experiments without sediment and stresses the importance of sediments in nutrient regeneration in shallow lakes. An example from Lake Arreskov is shown in Fig. 3.7.

In all the lakes seston addition resulted in decreased thickness of the oxidized surface layer and accelerated SRP release was observed in three of the lakes. The effect was probably caused by release of iron-bound P.

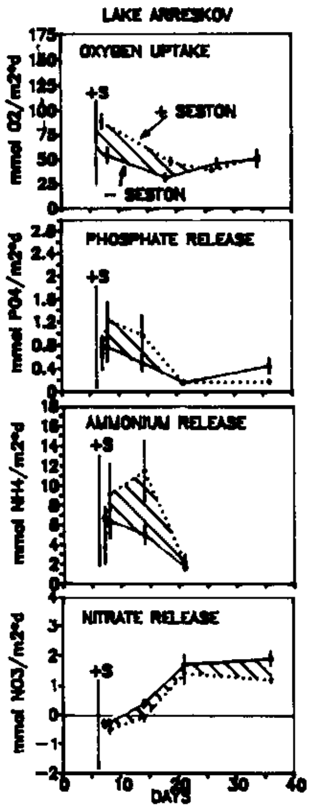


Figure 3.7
Oxygen and nutrient fluxes across the sediment surface during 30 days incubation in two parallel groups of cores from Lake Arreskov (Aug. 87, 18 °C). Effect of seston addition.

3.7 Influence of chironomids on decomposition of organic matter and nutrient exchange in the sediment

The density of chironomid larvae and oligochaetes is often high in soft bottom lake sediments. Thus, we found 4000 chironomids per m^2 in Lake Søbygård. A number of studies have shown an increased release of soluble inorganic compounds from the sediment due to chironomid bioturbation (e.g. Tessenow 1964, Granéli 1979, Gallep 1979, Holdren & Armstrong 1980). A review on bioturbation has been written by Petr (1977).

We studied the effects of chironomids on SRP exchange, oxygen uptake and exchange of N-salts, both summer and winter, in Lake Arreskov (Fig. 3.8).

Dual effect of chironomids on SRP-release

Sediment uptake of oxygen and nitrate was stimulated by increased density of chironomids both summer and winter. The fact that chironomids both accelerate the mineralisation and oxidize the sediment by their pumping activity probably explains the more scattering effect of increased density on phosphate and ammonium exchanges. In winter, and at the highest chironomid density in summer, chironomids decreased the SRP efflux, or even changed it to an influx. This is probably an effect of increased amounts of ferric iron, which enhances the sorption capacity of the sediment. At moderate high chironomid densities in the summer experiment we measured the highest release rates of both phosphate and ammonium (Andersen & Jensen in press).

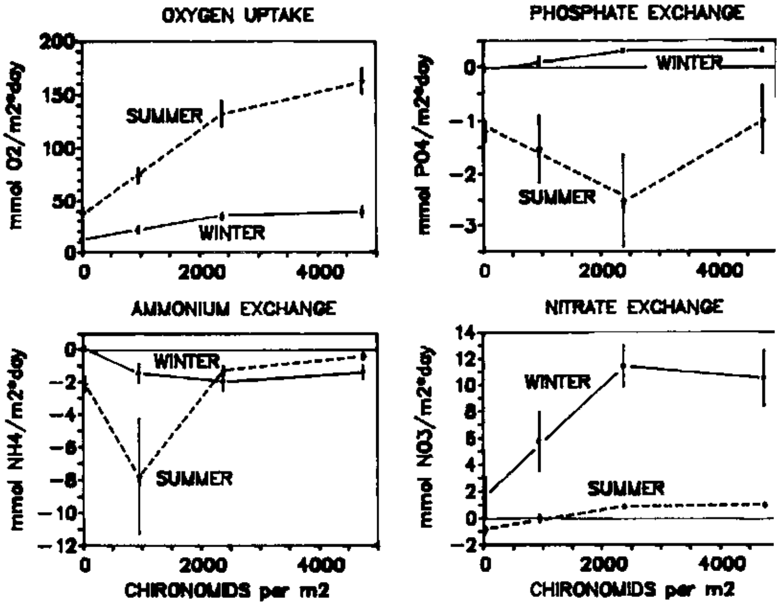


Figure 3.8
Effect of chironomid density on sediment oxygen uptake and nutrient fluxes winter and summer in Lake Arreskov. Vertical lines indicate S.D. of five replicates.

3.8 Concluding remarks on the study of the four NPo-lakes

Holdren & Armstrong (1980) concluded from a study of four North American lakes that chironomid density was more important for the magnitude of P-exchange than both temperature and pH in the lake water. We did not test chironomid effects on the other three lakes but we suppose that, in these more iron-rich lakes, the oxidizing effect of bioturbation would have a marked effect on phosphate sorption in the sediment.

Thus we believe that lake water temperature, in general, is the most important single factor in controlling the seasonal variation in P-release from shallow, eutrophic lake sediments. It is, however, obvious that sedimentation inputs of organic matter are necessary as catalyst for the release of minerogenic P-pools in the sediment.

The high release rates of SRP observed in summer had their origin in the large pools of iron-bound phosphorus in the sediment, and this study stresses the importance of the oxidized surface layer for retaining phosphate in the sediments.

3.9 Iron:phosphorus ratio in the surface sediment as an indicator of aerobic phosphate release

P in lake water of 101 lakes

Data on 101 Danish lakes from the Danish Lake Database (NPo-project C9) indicated an inverse relation between lake water TP and surface sediment iron:phosphorus ratio (Fe:P ratio). This relation was most significant for mean winter TP values (r^2=0.37) and there was no correlation between lake water TP and sediment TP or Ca (Jensen et al. submitted).

SRP-release in 15 lakes

On this background we initiated a study of the relation between sediment P release and sediment characteristics in 15 Danish lakes (table 2.1 and Fig. 2.1). We compared SRP release during 21 days with sediment P-pools, sed.Fe, sed.Ca, Fe:P ratio, Ca:P ratio in order to identify one or more conservative sediment characters that could predict rates of P-release from aerobic sediment surfaces.

Only the pool of loosely sorbed SRP and the Fe:P ratio correlated significantly (when discriminating at level: $p<0.05$) with the measured SRP release from undisturbed sediment cores (Fig. 3.9). Meanwhile, there was a high autocorrelation between the two parameters and therefore we suggest the latter, which is the most conservative of the two, to be used when forming an estimate of sediment SRP release from sediments of shallow, eutrophic lakes.

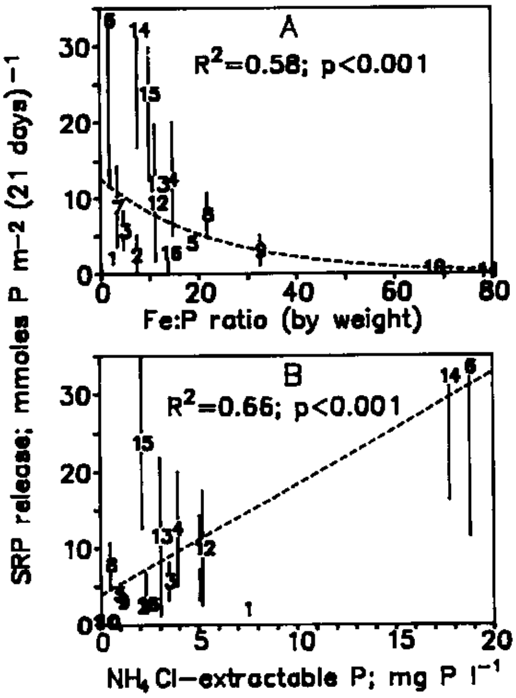
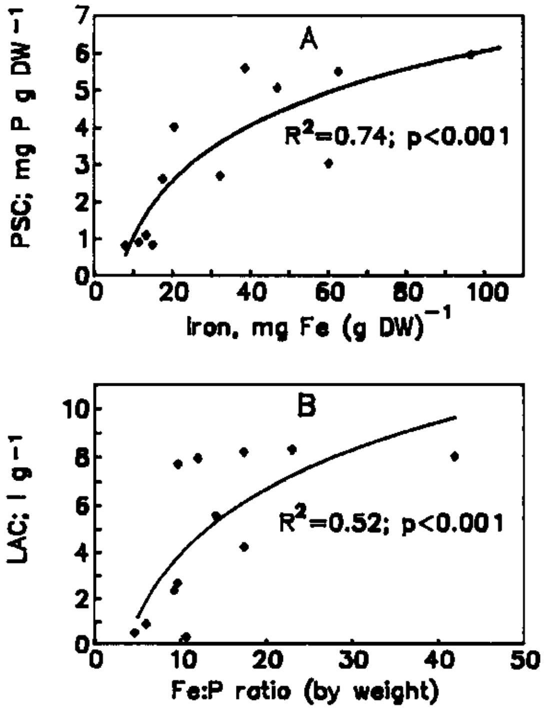


Figure 3.9
The cumulative SRP-release during 21 days from undisturbed sediment cores of 15 Danish lakes plotted against surface sediment Fe:P ratio (A) and the NH$_4$CL-extractable P-pool in the surface sediment. Vertical solid lines represent S.D. of five replicates. The numbers refer to the lake identification (Table 1).

Sorption kinetic in 12 lakes

The explanation of the relation between Fe:P ratio and P release is probably that the Fe:P ratio reflect the sediment capability to adsorb PO_4^{3-} in the oxidized surface layer. In accordance with this hypothesis we found that parameters such as sorption capacity (PSC) and buffering capacity (LAC) could be correlated with sediment iron and Fe:P ratio, respectively, in 12 Danish lake sediments (Fig. 3.10). Thus Fe:P ratio provides a measure of free sorption sites for phosphate in oxic sediments.

Figure 3.10
Parameters of Langmuir sorption isotherms plotted against surface sediment characteristics from 12 Danish lakes. Sorption capacity (PSC) versus total iron (A). Buffering capacity (LAC) versus Fe:P-ratio (B). See Fig. 2.3 for explanation of the terms. Data from 8 of the lakes are from Jacobsen (1977).



Application in lake management

The possibility to control internal P-loading by keeping the surface sediment oxidized seems to begin when the Fe:P ratio is above 15 (by weight). The Fe:P ratio is easy to measure and may be a valuable tool in the future management of shallow lakes.

Postscript

Results from NPo-project C4 are (or will be) used in five international publications. Three of these are indicated by (*) in the reference list below and two more are in preparation.

We are indebted to the NERI, Freshwater Division and the counties of Fyn and Århus for providing unpublished data for this project.

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Ministry of the Environment
National Agency of Environmental Protection
Denmark

The N, P and Organic Matter Research Program 1985-1990

Number C5

Carbon Dynamics in Coastal Marine Environments

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Dansk resume.

Det var projektets overordnede formål at integrere kemiske og biologiske parametre i en analyse af regulerende faktorer i stofomsætningen i kystnære marine områder. Det daværende videngrundlag var hovedsageligt bygget på traditionelle regressionanalyser af kvælstof og fosfor samt primærproduktion, og anvendelsen af disse informationer til at forudsige ændringer i de kystnære områder er meget dårlig. Vores resultater og konklusioner repræsenterer et første trin i en integreret analyse af de kystnære marine områders struktur og funktion. Vi har udarbejdet en række foreløbige kulstof budgetter og har kunnet vise, hvorledes tilsætning af næringsstoffer påvirker udvikling af biomasse og processer mellem autotrofe (planter) og heterotrofe (dyr og bakterier) organismer, og hvorledes biologiske processer som muslingers filtrering og zooplankton og fisks predation påvirker stofomsætningen. Eksempelvis viste det sig, at tilsætning af næringsstoffer øgede flowhastigheden af organisk stof igennem bakterier og heterotrofe flagellater og mindskede systemets stabilitet f.eks. ved pludselige ændringer i iltforbruget. Vi kunne sammenfattende vise, at balancen mellem tilsætning af næringsstoffer, planteplanktonets størrelsesfordeling, muslingernes fysiologiske tilstand og forekomst af fisk, der æder plankton, er afgørende for de kystnære marine områders struktur og funktion.

1. Resume.

The overall objective of this project was to examine important factors regulating carbon dynamics in coastal marine environments. The present understanding of the processes leading to deterioration of water quality is gene-

rally weak and often exclusively based on mass balances of the nitrogen and phosphorus cycles. Experience has shown that the predictive value of simple relationships between nitrogen and phosphorus is nevertheless limited. We tested the hypothesis that biological processes as well could have an impact on the production and decomposition of organic matter. While the project's results do not allow us to present a complete evaluation of factors regulating the carbon dynamics, we have demonstrated a provisional scenario of the carbon budget. We have also presented evidence concerning the effects of nutrient additions on the biomasses of and processes between autotrophs (plants) and heterotrophs (animals and bacteria) in the food web and the effects of biological mechanisms, such as filtration by mussels, grazing by zooplankton and predation by fish control on the structure and functions of the ecosystem. Moreover, we demonstrated how nutrient additions increased the flux of organic material through heterotrophic microorganisms (bacteria and heterotrophic flagellates) thereby decreasing the stability of the ecosystem. In summary, the project demonstrated that the balance between nutrient additions, the size distribution of phytoplankton, the physiological conditions of mussels, and the quantitative occurrence of planktivorous fish are important factors regulating fluxes of material through planktonic communities in coastal marine environments. We suggest that these factors be made an integral part of any evaluation of the structure and functioning of coastal marine ecosystems.

5. References

Part 1 of the list includes references produced only by the participants of the project. Part 2 includes other cited literature.

Part 1: Papers made by project participants

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Ministry of the Environment
National Agency of Environmental Protection
Denmark

The N, P and Organic Matter Research Program 1985-1990

Number C6

The Jutland Current
- a transport system from the German Bight to the Kattegat?

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Introduction

This report presents the results of a research project concerning the Jutland Current which has been carried out during 1986 and 1987 and has been coordinated by The Danish Institute for Fisheries and Marine Research. The entire project has been carried out in collaboration with The Marine Pollution Laboratory under the National Agency of Environmental Protection with funding from the National Agency of Environmental Protection's NPO Programme.

The final phase of the project (modelling of the Jutland Current) was carried out with two other collaborating institutes: The Institute for Hydrodynamics and Hydraulic Engineering at The Danish Technical University and The Danish Hydraulic Institute.

The Health Physics Department at Risø National Laboratory has also made a valuable contribution to the project by carrying out a study on the use of radionuclides in detecting the presence of water from the German Bight in the Kattegat.

The original goal of the project was to investigate water transport from the German Bight to the Kattegat. If it could be shown that the probability of transport of large quantities of water to the Kattegat via this route is small, then there would be no need to undertake more complicated studies aimed at quantifying the transport of nutrients and other non-conservative materials with the Jutland Current to the Kattegat.

The study has shown, however, that the possibility of considerable water transport from the German Bight to the Kattegat cannot be eliminated. By using radionuclides to trace water transport, it can be shown that much (possibly up to 1/2 or more) of the bottom water found in the Kattegat may come from the German Bight.

Evaluation of hydrographic data from the Jutland Current and Skagerrak

As a part of this project, hydrographic and chemical data have been assembled from a number of different laboratories that have carried out studies in the region affected by the Jutland Current. The purpose of assembling and analyzing these data was to try and identify seasonal variations or trends in the behavior of the Jutland Current. The data used is this analysis are listed in Table 4-1.

The Jutland Current can be defined as the transport of low salinity water from the southern part of the North Sea (German Bight) along the Danish West coast. The Jutland Current, itself, cannot be seen by examining the distribution of hydrographic parameters. However, the extent of low salinity water along the Danish west coast can be used to describe the area affected by the Jutland Cuurrent at any given period.

From hydrographic data and IR satellite images (which can be used to identify temperature differences in surface water) two extreme configurations of the Jutland Current can be described. In the first, low salinity water (<34 o/oo) extends from the German Bight along the entire Danish west coast to Skagen and, possibly, into the Kattegat. In the other extreme configuration, low salinity water is only found along the southern part of the west coast (south of the Limfjord).

There is no seasonal pattern in the occurrence of these two extremes. In fact, it can be shown that both configurations occur in all seasons and that hydrographic conditions along the west coast can change so quickly that both extremes can be seen within a period of 2-3 weeks. A possible explanation for these fast changes in hydrographic conditions may be that the presence of the Jutland Current is largely dependent upon wind conditions.

Concentrations of inorganic nutrient salts in the Jutland Current tend to be higher than those found in the open North Sea and, as a rule, these concentrations decrease from south to north (with increasing salinity) in the Jutland Current. In the winter (November to March) the observed fall in nutrient concentrations can be described as a function of dilution of the low salinity nutrient rich water in the Jutland Current with high salinity water of relative low nutrient content. During the rest of the year, nutrient turnover via biological activity makes it impossible to explain the decreasing concentrations of nutrients from south to north in the Jutland Current as a simple function of dilution.

Investigation of water transport using radiotracers

The study carried out by Risø was based on the use of radionuclides to trace water transport from the German Bight to the Kattegat. Their report concludes that dissolved material found at a given concentration in the German Bight will be found in the Kattegat ca. 1 year later at a concentration 3-4 x lower.

The most important assumption made in arriving at this conclusion is that all of the radionuclides originating from Cap de la Hague that are found in the Kattegat have passed through the German Bight. It is, however, very probable that this is the case.

The study treats bottom and surface water in the Kattegat together. We can say, roughly, that the concentration in bottom water will be approximately twice that in the surface. Thus, between 1/2 and 2/3 of the bottom water in the Kattegat may have originated in the German Bight.

This conclusion, however, must be viewed with caution as contamination from Cap de la Hague is considerably less than that from Sellafield and identification of contamination from Cap de la Hague can be difficult. In addition, contamination from the two plants varies from year to year and the reservoir effect associated with the North Sea should be taken into consideration in drawing conclusions using radionuclides as tracers.

Until a complete evaluation of this method and the associated uncertainties has been carried out, we must modify the conclusion contained in Risø's report as follows:

"The input of water to the Kattegat from the German Bight is <u>not</u> so insignificant that it can be neglected in an evaluation of water received by the Kattegat from adjacent water bodies. It is probable that up to 1/2 or more of the water entering the bottom water of the Kattegat has passed through the German Bight".

Hydrodynamic models of the Jutland Current

During the first phase of this project, three existing hydrodynamic models (Backhaus, 1985; Stigebrandt, 1984; Dansk Hydraualisk Institut, 1977; 1986) were investigated for their ability to produce a plausible estimate of how much water is transported from the German Bight to the Kattegat.

An acceptable model should also be able to describe

- variations with time in transport in the Jutland Current

- mixing processes occurring during transport in the Jutland Current

- other sources of water to the Kattegat

None of the models investigated could adequately describe transport and mixing proceses occurring in the Jutland Current. However, in 1986, Prof. Fl. Bo Pedersen proposed modifications to the model (System 21) developed by the Danish Hydraulic Institute which were designed to elucidate processes occurring in the Jutland Current.

By employing this modified model and applying different wind conditions, it has been possible to understand the two extreme configurations in salinity distribution which are observed along the Danish west coast.

These two situations (one in which low salinity water extends along the entire coast and one in which low salinity water is only found along the southernmost part of the coast) can be shown to be closely related to the windfield. With NW wind, the current is separated at ca. Hanstholm and south of this port, the current is southerly. E wind causes an increased outflow from the Skagerrak as well as transport away from the coast and can be said to "stop" the Jutland Current. W wind gives a stable northerly flowing Jutland Current and, thus, gives the best conditions for the transport of water from the German Bight to the Kattegat.

A theory has been developed to describe entrainment of water into the Jutland Current. With this theory applied to System 21, it is possible to describe transport and mixing in the Jutland Current from the German Bight to the Skagerrak. The model cannot, however, describe transport in the Skagerrak and therefore, is still unable to quantify transport from the German Bight to the Kattegat.

It has not been possible to implement the entrainment theory in System 21 within the scope of this project but it, hopefully, will be possible to do so in the near future.

Recommendations for further research

This report recommends that

- work on a mathematic model to describe the transport of water and dissolved materials from the North Sea to the Kattegat to be continued with the aim of understanding and quantifying both the transport and its variability in time.

- a project using Technetium-99 (from Cap de la Hague) as a tracer be established with the aim of quantifying mean transport from the German Bight to the Kattegat and, thus, serve as a check for the mathematic model.
- seasonal variations in the biologically mediated turnover of nutrients in the Jutland Current be investigated in order to be able to identify periods in which large quantities of inorganic N and P might be transported to the Kattegat.

Ministry of the Environment
National Agency of Environmental Protection
Denmark

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Number C7

Regulation of the Biomass of Benthic Algae in Small Streams

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REGULATION OF THE BIOMASS OF BENTHIC ALGAE IN SMALL STREAMS

Summary

Benthic algae are an important structural component of small streams. The regulation of algal biomass development and maximum biomass were examined by extensive and intensive field studies and field experiments in four small streams. Biomass regulation was found to be very complex and varied widely between streams, between sections of the same stream and even within the same section. Light availability, the type and stability of the substratum, the degree of invertebrate grazing and the phosphorus concentration were found to be important regulating factors.

1. Introduction

Benthic algae are an important structural component of many streams open to light. They are important for invertebrate scrapers (Cummins 1973) and the oxygen metabolism of large algal biomasses may deteriorate the environment for invertebrates and fish. Furthermore, stream denitrification is closely associated with the development of the algal biomass (Christensen et al. 1990).

One of the aims of this study was therefore to analyse the factors which regulate the development and maximum biomass of benthic algae in small streams.

The approach taken was a combination of

- field studies of algal biomass development in six stream sections for one-three years
- an intensive study of the algal spring bloom of one stream
- a field experiment with phosphorus enrichment.

Some of the main results are summarized below.

2. Description of the streams

The studies were undertaken using four 1st order streams, Skærbæk, Dalby Bæk, Ellerup Bæk and Gelbæk, located in the central part of Jutland, Denmark. A total of six stream sections of 170-250 m were studied. Mean stream width varied between 0.8-5.1 m, mean stream depth between 0.04-0.38 m and mean discharge between 19-109 l sec^{-1}. Annual variation in discharge was relatively low in Skærbæk and Dalby Bæk because ground water input was stable (min.:max = 1:2.4), was higher in Ellerup Bæk and was extremely high in Gelbæk (1:200).

Skærbæk was unshaded, whereas the bank vegetation in the other three streams caused varying degrees of shading in May-June. The stream sections in Skærbæk and Gelbæk were undisturbed, whereas the stream sections in Dalby Bæk and Ellerup Bæk were channelized, and managed.

Mean concentration of dissolved inorganic phosphorus during the growth season varied between 0.003 mg P l^{-1} in Skærbæk and 0.25 mg P l^{-1} in Gelbæk, whereas mean concentration of nitrate was high and only varied between 2.1-7.8 mg N l^{-1}. If nutrient limitations occurred, phosphorus was clearly the limiting nutrient.

3. Methods

The field studies included stratified random sampling of algal biomass in the upper 1 cm of the sediment and of macroinvertebrates, and determination of light availability at the sediment surface. On fine grained sediments core samplers were used, whereas on stony substrata 1-3 stones sampled with a net made up one sampling unit.

Algal biomass was estimated as chlorophyll by extraction in 96% ethanol according to Søndergaard and Riemann (1979). No attempt was made to correct for pheopigments. The invertebrates were assigned to functional feeding category (sensu

Cummins 1973) according to Merritt and Cummins (1978) and Iversen (1988). Some taxa were evenly shared by two functional feeding categories.

The intensive study of the algal spring bloom in Gelbæk in April-June 1988 included continuous measurement of water level and estimates of mean daily discharge. Daily mass balance of algae and particulate organic matter were determined from discharge and automatic diel sampling of stream water at the end points of the stream sections. Daily primary production and respiration were calculated from continuous free water measurements of oxygen and temperature with oxygen probes and thermistors.

The P-enrichment experiments were performed in six parallel 1.2 m long tiled channels. After colonisation for about a month under natural conditions, additional dissolved inorganic phosphorus was added continuously to three of the channels. The other three channels served as controls. Algal biomass and invertebrate density were determined at intervals for 1-3 months.

4. Results and discussion

4.1. The influence of light

Light availability was found to be a major factor regulating the seasonal development of the algal biomass on fine grained sediments. In the unshaded Skærbæk, in which the seasonal variation in light availability at the sediment surface only varied between 58-69% of the insolation in the open, maximum algal biomass occurred in the summer (Fig. 1A). Sediment instability modified

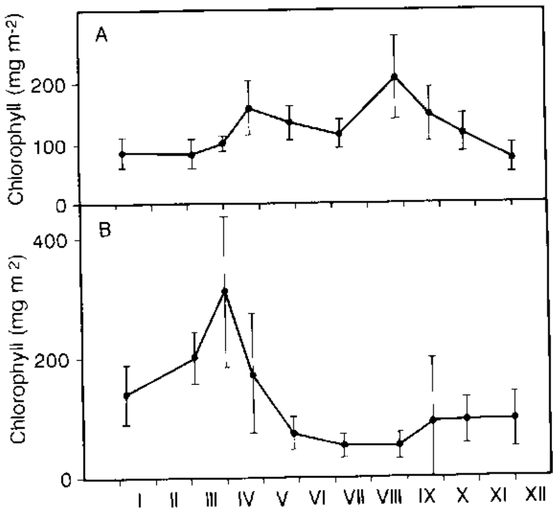

Fig. 1.
The seasonal variation of the benthic algal biomass on fine grained sediments (A) and stones (B) in Skærbæk 1987. Means ± 95% C.L. are given.

the pattern however. In Gelbæk, where shading due to the appearance of leaves on the alder trees and growth of herbaceous vegetation reduced light availability at the sediment surface in May-June to less than 5% of the insolation in the open, maximum algal biomass occurred in May (Fig. 2).

Algal primary production in Gelbæk was predominantly regulated by light. On the 19th of April and the 18th of May light availability at the sediment surface was similar (7 and 10 E m^{-2} d^{-1}, respectively) as was primary production (2 g O_2 m^{-2} d^{-1}), although algal biomass increased from 100 mg chl. m^{-2} in April to about 700 mg chl. m^{-2} in May.

The present finding that light is an important determinant of the seasonal development of benthic algae on fine grained sediments is in good agreement with Sand-Jensen et al. (1988) and

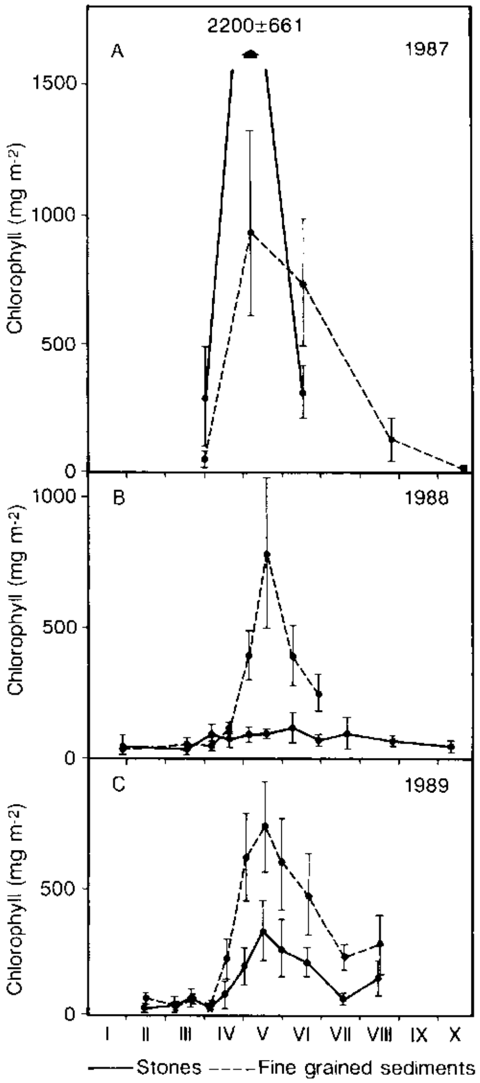

Fig. 2.
The seasonal variation of the benthic algal biomass on stones and fine grained sediments in Gelbæk 1987-1989. Means ± 95% C.L. are given.

Thyssen et al. (1990). Similar observations have been made using laboratory streams by McIntire et al. (1964) and in studies of algal biomass development on hard surfaces by Gregory (1980) and Munn et al. (1989). In contrast to fine grained sediments, the seasonal development of the algal biomass on stones was only occasionally regulated by light (Fig. 2A and C), whereas in most cases, other regulating factors dominated (Fig. 1B and 2B).

4.2. The influence of substratum

Biomass development and maximum biomass of benthic algae were significantly different on fine grained sediments and stones in Skærbæk (Fig. 1) and Gelbæk (Fig. 2), and this was also the case in Ellerup Bæk (Iversen et al. submitted) and Dalby Bæk. Significant differences in species composition on different substrata have been reported by many authors including Butcher (1940), Patrick (1948) and Reiter (1989). Tett et al. (1978) documented differences in algal biomass on different stream substrata. The present data are the first evidence that both seasonal biomass development and maximum biomass of different substrata in the same stream also differ significantly.

Whereas most studies on the regulation of benthic algae in streams have been made using artificial substrata or solid natural substrata such as stones or rocks, few have dealt with fine grained sediments. The present findings show that the regulating factors differ between substrata, and that obervations made on artificial substrates or stones/rocks cannot be transferred to fine grained sediments.

The stability of fine grained sediments varies with the underlying soil and the discharge regime. Although discharge varied only little in

Skærbæk, the sandy sediments were regularly rebedded by small increases in discharge. In winter algal biomass was evenly distributed in the upper 10 cm of the sediment. From March to June 1987 the amount of chlorophyll in sediments at a depth of 6-10 cm increased significantly from 2.5±0.7 μg chl. (g sand)$^{-1}$ to 4.2±1.7 μg chl. (g sand)$^{-1}$. This indicates that sediment instability in Skærbæk was an important factor, which modified the pattern of seasonal biomass development (Fig. 1A).

The two sections of Ellerup Bæk differed significantly as to sediment stability due to channelization. In Ellerup Bæk I, moving sand occurred midstream throughout the study period, whereas in Ellerup Bæk II, the fine grained sediment was stable. Although nutrient concentrations and light availability were approximately similar, mean algal biomass development differed significantly in the two sections, mean maximum biomass being 74±28 mg chl. m^{-2} and 160±41 mg chl. m^{-2}, respectively. The reason for the different patterns was differences in biomass distribution (Fig. 3). In Ellerup Bæk I the biomass development in the narrow zones with stable substratum along the banks was significantly higher than in the midstream zone with moving sand. In Ellerup Bæk II no such difference was observed.

The importance of sediment instability in Skærbæk has previously been suggested by Sode (1983), and Sand-Jensen et al. (1988) reported observations which indicate that sediment instability in the channelized River Suså was important in the early spring but not in late spring. As most Danish lowland streams are more or less channelized (Brookes 1984), the present results suggest that

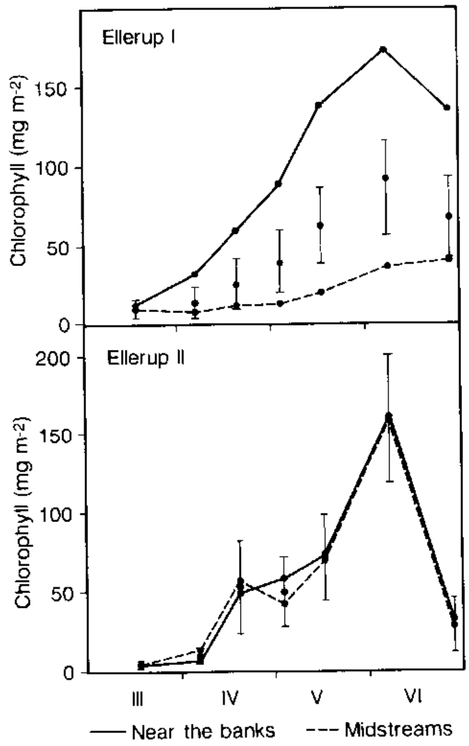

Fig. 3.
The seasonal variation of the benthic algal biomass on fined grained sediments near the banks and midstream in two sections of Ellerup Bæk 1988. In addition average algal biomass ± 95% C.L. are shown.

sediment instability might be important factor regulating biomass development and maximum biomass on fine grained sediments in Danish streams.

4.3. The influence of invertebrate grazing

Experimental studies on solid artificial and natural substrata have established that invertebrate grazing can regulate species composition of the algal community (Hart 1985; Lamberti et al. 1987) and algal biomass

(McAuliffe 1984; Jacoby 1985; Feminella et al. 1989).

In Gelbæk maximum algal biomass on stones in 1987, 1988 and 1989 was 2,200±661, 120±30 and 320±60 mg chl. m^{-2}, respectively (Fig. 2), even though the nutrient concentrations and light availability were approximately similar during the three growth seasons. Differences in the density of important invertebrate scrapers in January-March (Table 1) prior to algal biomass development strongly suggest that a high density of scrapers may totally prevent algal biomass development, such as was the case in 1988 (Fig. 2B).

	1987	1988	1989
Baetis rhodani Pict.	-	3,033± 798	2,047±318
Orthocladiinae	3,854±2,374	13,197±2,380	1,892±298
Lymnaea pereger Müller	-	10± 11	9± 7
Ancylus fluviatilis Müller	-	-	230± 79

Table 1.
The mean density ± 95% C.L. of insect and snail scrapers on stones in Gelbæk in January-March 1987-1989. Absence is shown by -.

In Skærbæk the average density of the total invertebrate community and of scrapers on stones was 22,000 ind. m^{-2} and 9,700 ind. m^{-2}, respectively, compared to 6,200 ind. m^{-2} and 1,100 ind. m^{-2}, respectively, on fine grained sediments. It is therefore likely that the seasonal pattern of algal biomass on stones (Fig. 1B) may be explained by invertebrate grazing, whereas other factors are more important on fine grained sediments. Total invertebrate fauna on stones in Dalby Bæk and Ellerup Bæk averaged 22,000-27,000 ind. m^{-2}, of which scrapers accounted for 8,500-8,900 ind. m^{-2}. These observations indicate that invertebrate grazing may be an important factor regulating algal biomass development and maximum biomass on stony stream substrata.

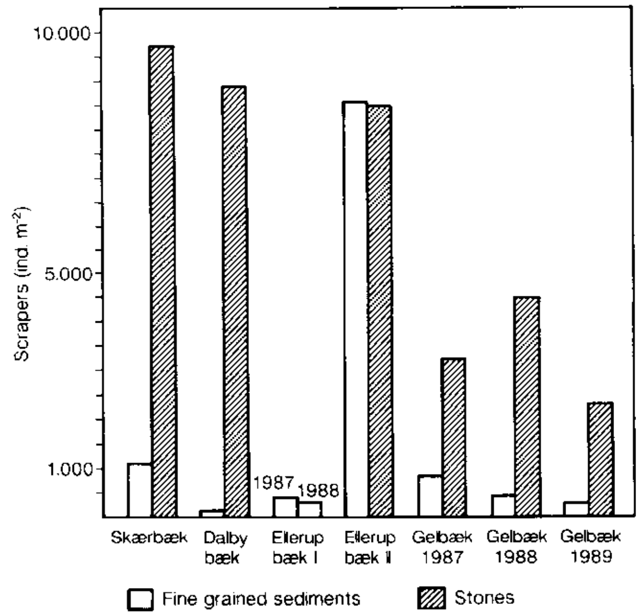

Fig. 4.
Mean density of insect and snail scrapers on stones and fine grained sediments in four small Danish streams.

The invertebrate community on fine grained sediment in Gelbæk 1987-1989 averaged 9,000-23,000 ind. m^{-2} and was dominated by Oligochaeta (64-74%), with scrapers only accounting for 290-920 ind. m^{-2} (Fig. 4). Similarly in Dalby Bæk and Ellerup Bæk I, of a total invertebrate density of 3,300-7,700 ind. m^{-2}, scrapers only made up 100-

400 ind. m^{-2}. In Ellerup Bæk II the total fauna averaged 19,600 ind. m^{-2}, the majority being Orthocladiinae (17,000±4,300 ind. m^{-2}). As these are known to be scrapers/gathering collectors, invertebrate grazing may have been an important determinant for algal biomass development. However, the invertebrate community structure and density in Skærbæk, Gelbæk, Dalby Bæk and Ellerup Bæk I suggest that invertebrate grazing is generally of little importantance for the regulation of the algal biomass on fine grained stream sediments.

4.4. The influence of nutrients

It is well established from experiments with artificial substrates and natural solid substrata that the availability of both nitrogen (Grimm & Fisher 1986) and phosphorus (Bothwell 1985; Freemann 1986; Biggs & Close 1989) can be limiting for algal biomass development. As the concentration of nitrate in Danish streams is generally greater than 1 mg N l^{-1} (Kristensen et al. 1990) and would thus be unlikely to be limiting, the present study concentrated on phosphorus.

Addition of 35-56 μg P l^{-1} to the experimental channels in Skærbæk significantly increased algal biomass development and the density of the dominating invertebrate taxa, thus confirming that nutrients may control the biological structure of small streams.

These experiments, however, do not present evidence as to whether or not phosphorus regulates maximum algal biomass. Recently Bothwell (1989) has shown that maximum algal biomass in experimental channels increased with increasing concentration of dissolved inorganic phosphorus up to about 0.1 mg P l^{-1}. Data from a number of studies on natural stream substrata (Fig. 5) have therefore been examined. On

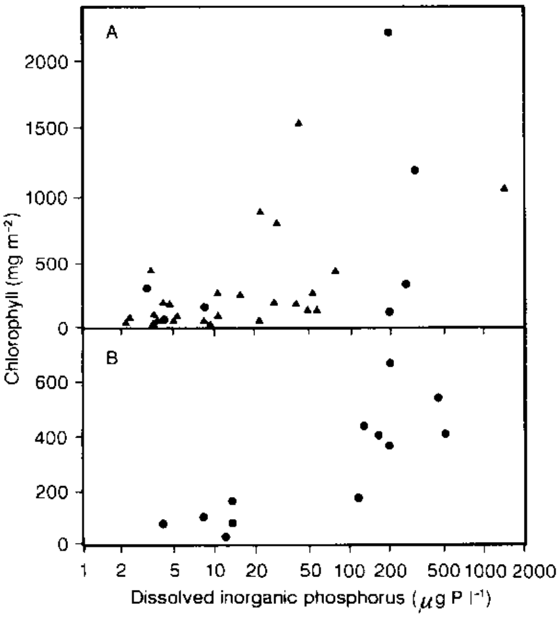

Fig. 5.
Maximum algal biomass on stones (A) and fine grained sediments (B) as a function of the concentration of dissolved inorganic phosphorus (log-scale). Data from 13 published and unpublished studies.

stones/rocks biomasses as high as 300-400 mg chl. m^{-2} have been reported at concentrations of 3-4 μg P l^{-1} (present study; Biggs & Close 1989), but generally maximum values increased to more than 2 g chl. m^{-2} at phosphorus concentrations of 0.1-0.2 mg P l^{-1} (Fig. 5A). However, the majority of streams had maximum algal biomasses of less than 300 mg chl. m^{-2} even at higher phosphorus concentrations.

Relatively few data are available from fine grained stream sediments, but the general pattern is the same as found for stones/rocks (Fig. 5B). The main differences appear to be that at a given phosphorus concentration the maximum algal biomass on fine grained sediments is lower than on stones/rocks and the variation is smaller. This may in part be explained by the absence of filamentous algae on fine grained sediments.

It can be concluded that stream water concentration of dissolved inorganic phosphorus determines the upper limit for the potential maximum algal biomass. Maximum biomass is reached at concentrations of 0.1-0.2 mg P l^{-1}, but in most streams other factors such as light availability, invertebrate grazing and substrate type and stability appear to be more important for the regulation of the maximum algal biomass.

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Number C8

Denitrification and Nitrogen Mineralization in Coastal Marine Sediments

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NUTRIENT CYCLING IN DANISH MARINE WATERS

Abstract

The nutrient uptake and remineralization, the relative importance of nitrogenous nutrients (ammonium, urea), and the influence of nitrogen and phosphorus on the primary production was investigated in the Kattegat proper and the Vejle Fjord. At both locations, ammonium was the most important nitrogenous nutrient. In the Kattegat, it was supplied by regenerative processes in the photic zone, while in the Vejle Fjord land run-off and benthic release were important as well. Nitrate had great significance in the spring in the Kattegat. Moreover, episodic erosion of the pycnocline resulted a in transient rise in the "new" (nitrate-based) production in the summer. In the Vejle Fjord, concomitant high ammonium concentrations reduced the importance of nitrate, and reduced compounds (ammonium, urea) were the most important at all occasions. Regression analyses showed highly significant linear relationships between the primary production and ambient nitrogen concentrations and less so to phosphate for the Kattegat proper. For Vejle Fjord, the correlation was significant for both nitrogen and phosphorus, but the analyses was based on very few measurements.

1. Introduction

A comprehensive knowledge of the abundance, distribution and behaviour of nutrients is fundamental to the assessment of effects of human-based eutrophication of the sea. The present project was initiated to improve our understanding of the cycling of nitrogen and phosphorus in the productive zone of the Kattegat area; an area which throughout the last decade, has been exposed to oxygen deficiency several times. Furthermore, studies were carried out in a Danish fjord to evaluate differences between the Kattegat proper and the coastal areas. The investigation included studies of nutrient uptake and remineralization, the relative importance of nitrogenous nutrients (nitrate, ammonium, urea) and the influence of nitrogen and phosphorus on the primary production.

The project was financially supported by the National Agency for Environmental Protection. Thanks are due to the staff at the National Environmental Research Institute for their technical assistance.

2. Material and methods

The studies were carried out in the Kattegat and the Vejle Fjord (Figure 1).

Depth profiles of salinity, temperature, light attenuation and fluorescense were obtained by CTD and in situ fluorometer. Water samples for analysis of nutrients, chl a, and particulate C, N and P were collected at one depth in (Vejle Fjord) and two to 7 depths in the Kattegat.

Uptake of CO_2, ammonium, nitrate, urea and phosphate was studied by isotope enrichment procedures. Two to 3 concentrations of the appropiate ^{15}N-labelled compound (99 % enriched) or $H_3{}^{32}PO_4$, and 5 µCi of $NaH^{14}CO_3$ were added to the water samples (1.5-2.5 l in polycarbonate bottles). Remineralization of ammonium and phosphate were determined as isotopic dilution in the same bottles as uptake and ^{14}C fixation. The samples were incubated for 2-5 h in deck incubators screened to simulate the light intensity at the sampling depths. The sample preparation followed Harrison and Harris (1986) for ^{32}P-samples and Kristiansen and Paasche (1982), Såby Johansen (1984) and Selmer and Sörensen (1986), for ^{15}N-samples.

The uptake rate of nitrate-N and urea-N was calculated according to Dugdale and Wilkerson (1986), while the uptake of ammonium-N and phosphate-P was carried out according to Laws (1985) taking isotopic dilution due to remineralization into account. Nitrogen uptake rates at ambient (i.e. unenriched) concentrations were calculated assuming the uptake rate vs. nutrient concentration followed Michalis-Menten kinetics.

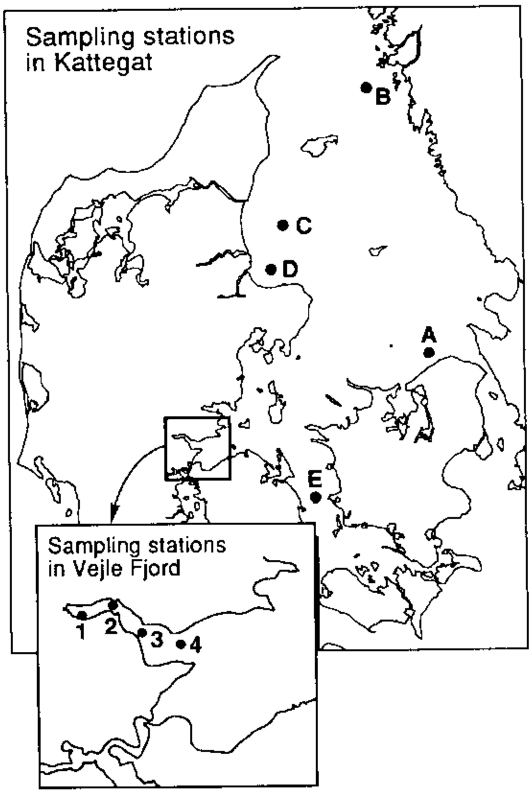


Figure 1
The position of the sampling stations in the Kattegat proper and the Vejle Fjord visited throughout 1988 and 1989. Station A was visited 18 times during 1988, while the other areas of the Kattegat were visited less regularly. The 4 sampling periods in Vejle Fjored covered the seasonal variation in 1989.

Data from the studies of diel and vertical variations in uptake were used to establish relations between light intensity and uptake of ^{14}C (PI curves), nitrogen species and phosphate (specific uptake rates, VI curves). Using these equations and monthly mean insolation values, the hourly uptake rates were converted to daily values.

The nutritional status of the phytoplankton assamblages was judged from the ratio of the maximal uptake rate (Vmax) to the uptake rate at ambient concentration (Vamb), derived from the Michaelis-Menten equation. Moreover, nutrient deficiency was studied at each sampling occasion by the Ammonium Enhancement Ratio (Yentsch et al. 1977), alkaline phosphatase activity (Healey and Hendzel 1979) and transient (2-5 h) photosynthetic responses to addition of nutrients in excess (Elrifi and Turpin 1985).

3. Results

Surface temperature, salinity, concentration of nutrients, chl a and PON at station A during 1988 are depicted in Figure 2. After the spring bloom in March/April nutrient concentrations were low until October. During this period the water column was strongly stratified with a primary pycnocline at 10-12 m depth. In May and June a secondary halocline was established at 4-6 m. Episodic erosion of the pycnocline occurred during periods with strong winds (e.g. in beginning of August). On Nov 1, the water column was mixed to the bottom (26 m) during a storm.

Primary production, uptake and regeneration of nutrients at St. A are shown in Figure 3. In general, high rates of primary production and nutrient uptake in surface waters were found in April, mid August and late October and November, while very low rates were found in June. The mean ratio of uptake of C, N and P (molar basis) are 65/11/1 (n=23), however with marked variations, especially in June. The remineralization of ammonium and phosphate generally followed the yearly variation in uptake rates. However, at several occasions the regenerative and assimilative fluxes were unbalanced. The nutrient uptake was generally higher at 2-4 m (the depth at which 25 % of the surface light penetrated) than at 4-10 m (10 % light). This was due to the light-dependent uptake of phosphate and nitrate.

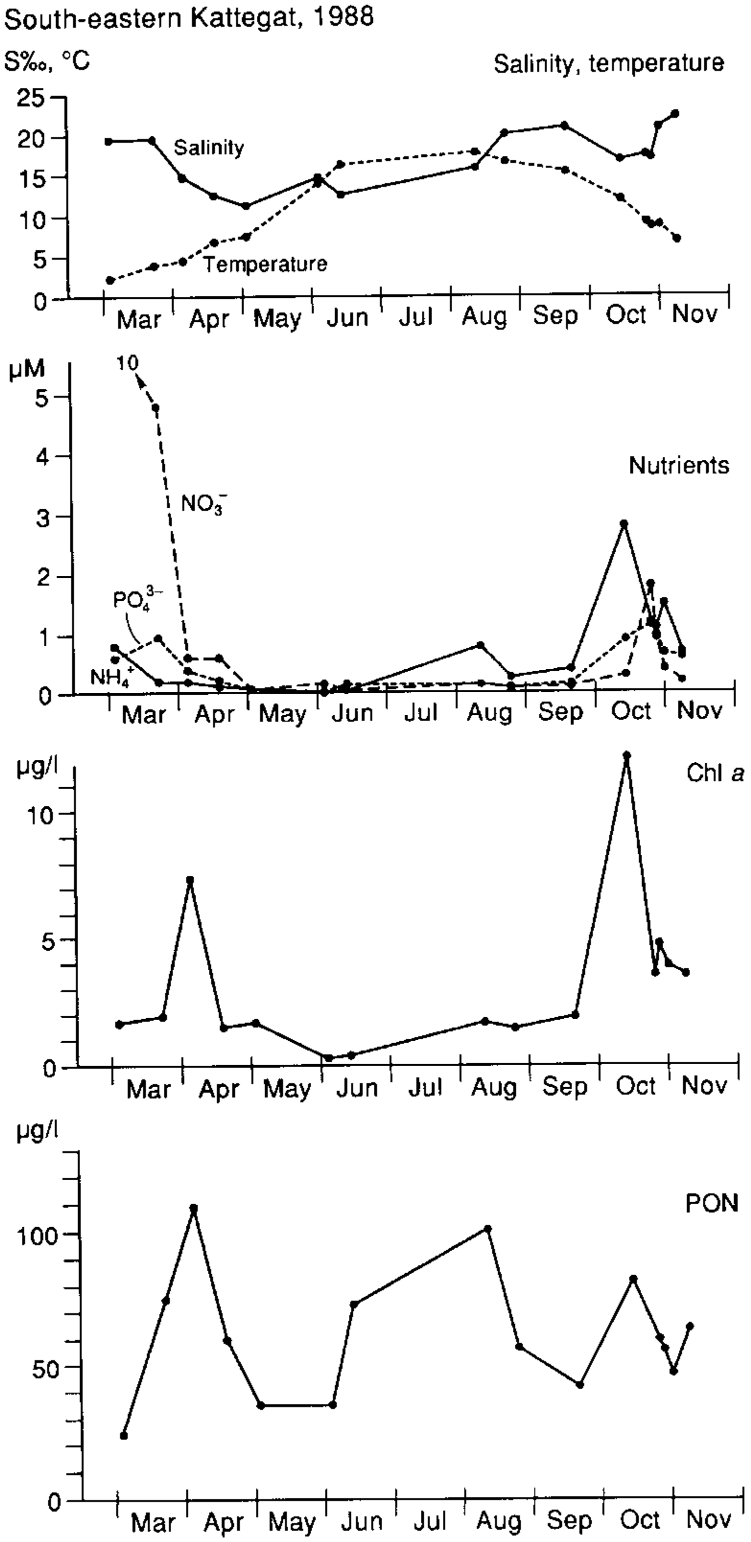


Figure 2
The seasonal variation at station A in the southeastern Kattegat in 1988 of.

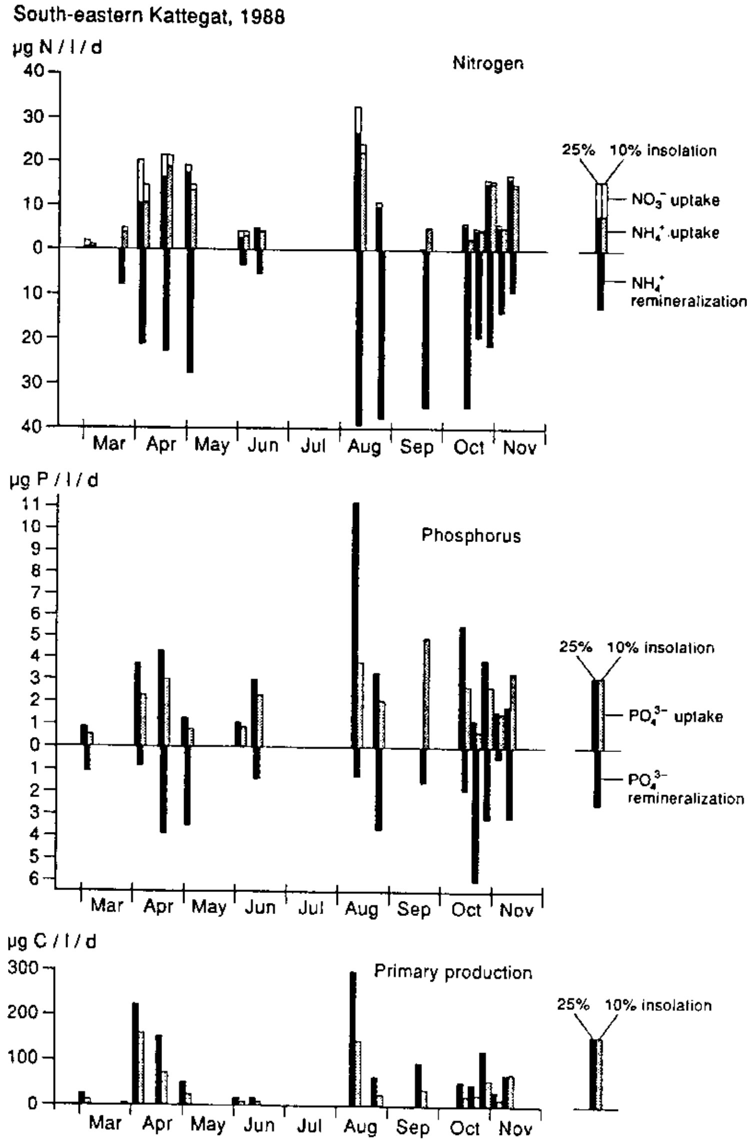
A. temperature and salinity.

B. concentration of nutrients.

C. concentration of chl a.

D. particulate nitrogen.

Figure 3
Primary production, nutrient uptake and remineralization at station A in 1988. Values are estimated for the 25% and 10% light-level. The remineralization was not light dependent and thus only one bar is given.



The F-ratio (Figure 4) was high during the spring bloom but declined strongly in April, and in the rest of the production period it was low varying between 0.02 and 0.2. During the study period the F-ratio calculated on the basis of absolute uptake rates was 0.18 and 0.11 at the two depths, respectively. More than 60 % of the "new" production occurred in March-April.

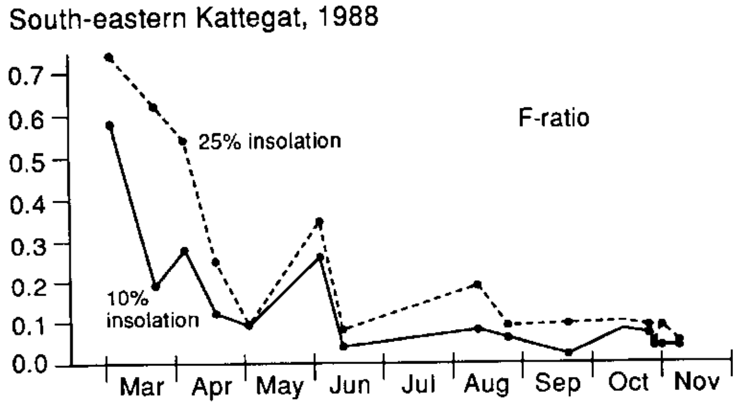
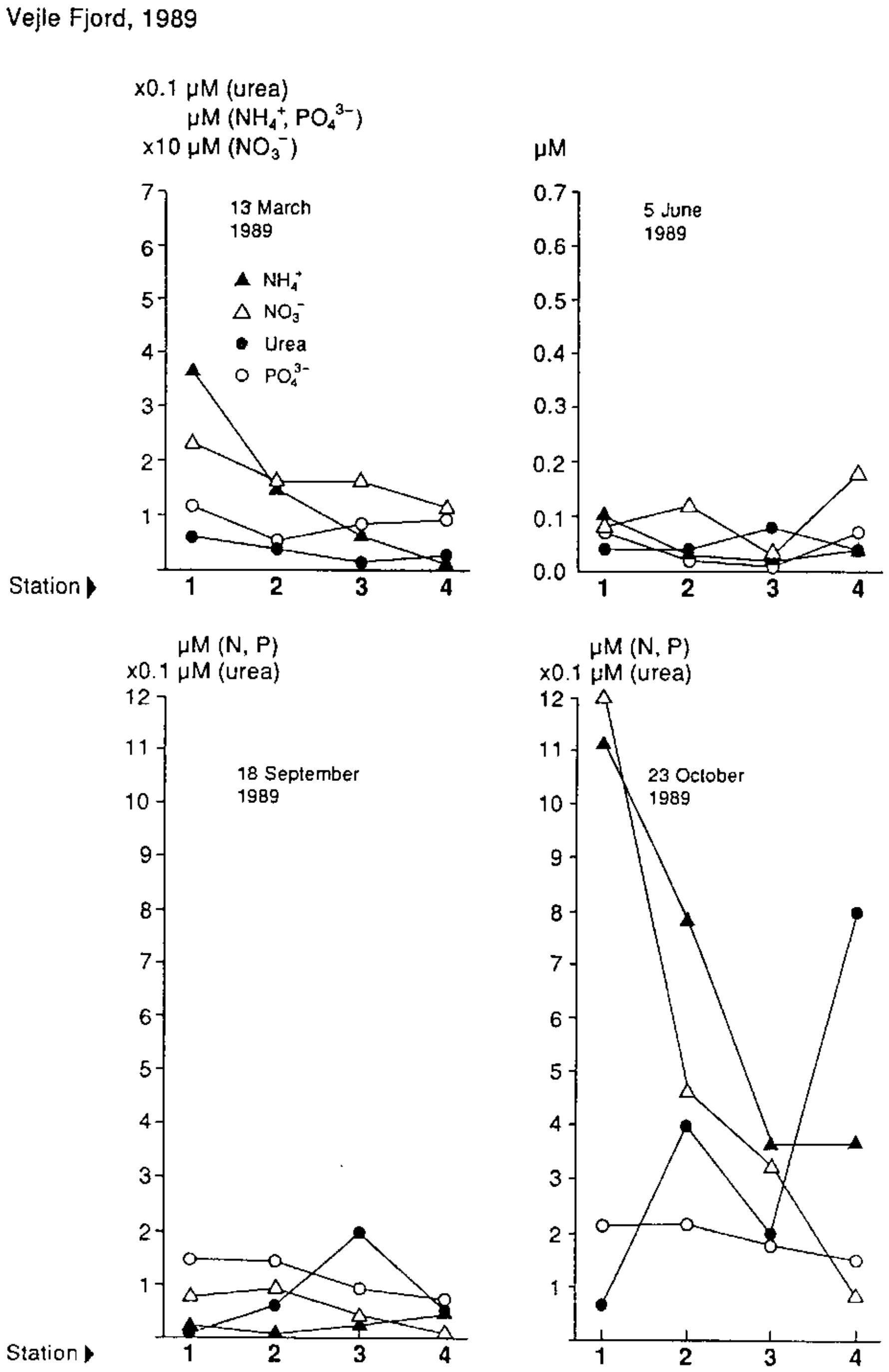


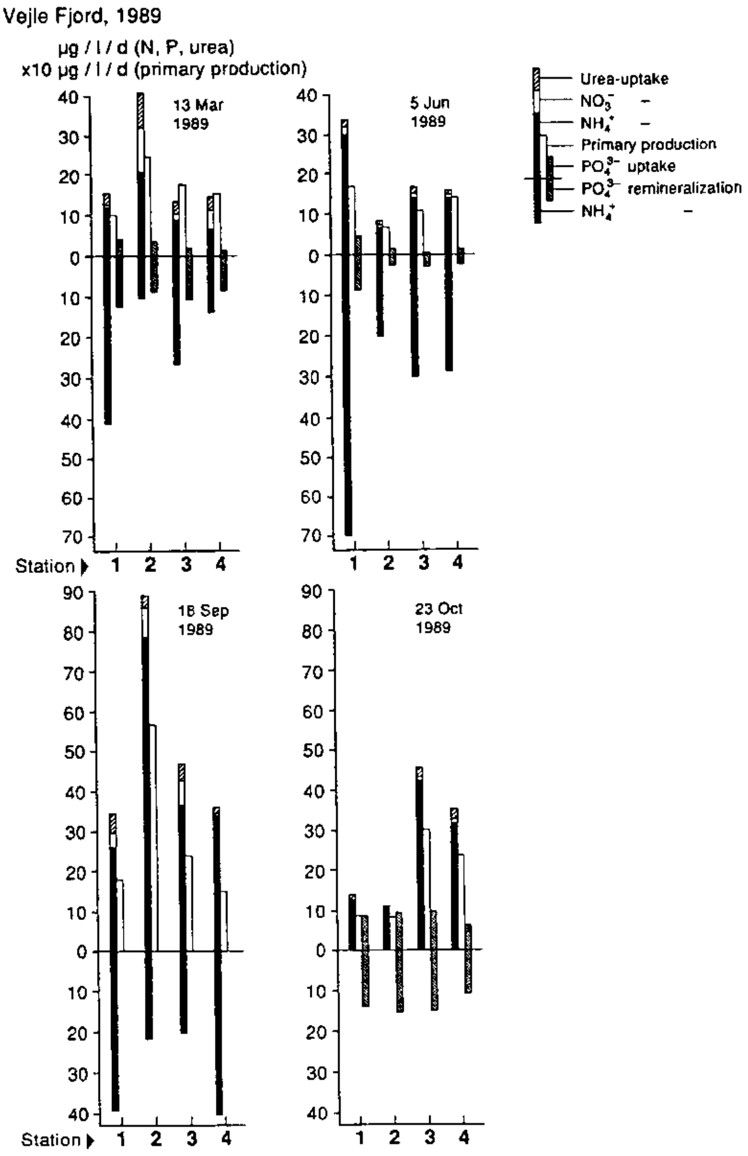
Figure 4
The relative importance of "new" (nitrate) in the nitrogenous nutrition of plankton algae at station A in 1988 expressed by the F-ratio (nitrate uptake / nitrate + ammonium uptake, Dugdale and Goering 1967).

In June special attention was applied to a subsurface bloom of Chrysochromulina polylepis. Nitrate was more important than ammonium for C. polylepis growth and in accordance with this, supersaturation (ca. 50 %) of oxygen was found at the depth of maximum primary productivity (not shown).

In Vejle Fjord the concentration of inorganic nitrogen generally showed a decline from the innermost station (St. 1) to the mouth of the fjord (Figure 5). The reduced nitrogen compounds constituted the largest part (63-98 %) of the total nitrogen uptake (Figure 6). The rates of remineralization were, with the exception of St. 2 and 3 in September when a bloom of Prorocentrum minimum occurred, larger or equivalent to the uptake rates.



<u>Figure 5</u>
The concentrations of nutrients of the four sampling stations in Vejle Fjord in March, June, September and October, 1989.



<u>Figure 6</u>
Primary production and nutrient uptake and remineralization of the surface layer (2.5m) of the four sampling stations in Vejle Fjord in Marts, June, September and October, 1989.

The nutritional status of phytoplankton as indicated by the short term response in nutrient uptake after enrichment indicated nitrogen deficiency from late May through September (Figure 7 and 8). In Vejle Fjord weak indications of a concomitant phosphate deficiency were found in June.

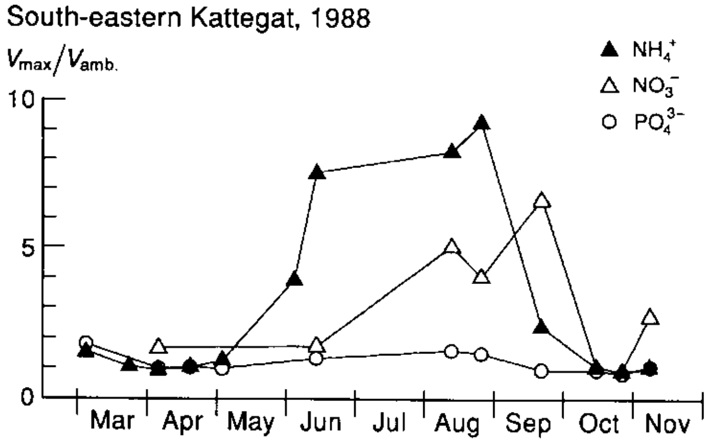


Figure 7
The nutritional status of the phytoplankton of station A in the southeastern Kattegat as indicated by the maximum uptake rate to the uptake rate at the ambient nutrient concentration (V_{max}/V_{amb}). Deficiency of a particular nutrient was assumed if the ratio was above 4-5 (Glibert 1988).

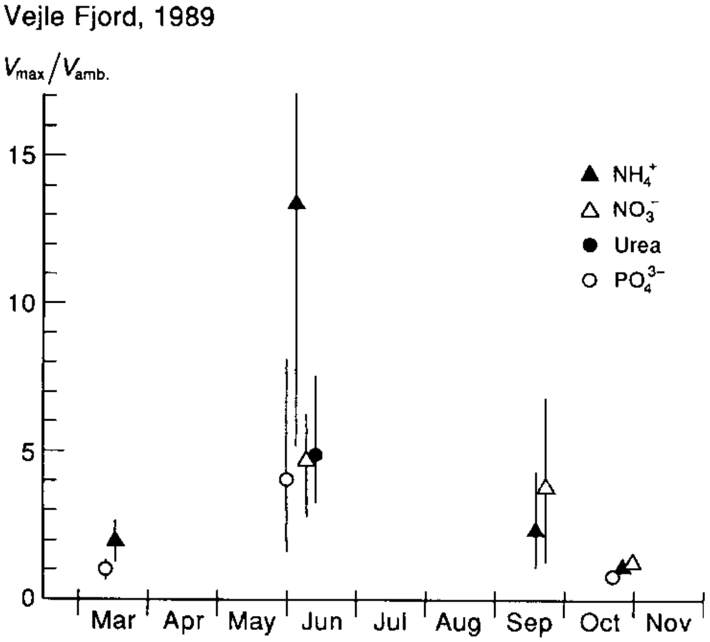


Figure 8
The nutritional status of the phytoplankton of Vejle Fjord as indicated by the ratio of maximum uptake rate to the uptake rate at ambient nutrient concentration (V_{max}/V_{amb}). Deficiency of a particular nutrient was assumed if the ratio was above 4-5 (Glibert 1988).

The nitrogen deficiency was not supported by the other dynamic parameters investigated (ammonium enhancement ratio, photosynthetic response to nutrient addition), The alkaline phosphatase activity was consistently low. During the period May-September the primary production (normalized to 100 PAR) in the Kattegat was highly correlated to the ambient concentration of inorganic nitrogen and less so to phosphate (Figure 9). For Vejle Fjord, the primary production in June and September was significantly (Spearman Rank Correlation, $p < 0.05$) correlated to both phosphate and inorganic nitrogen.

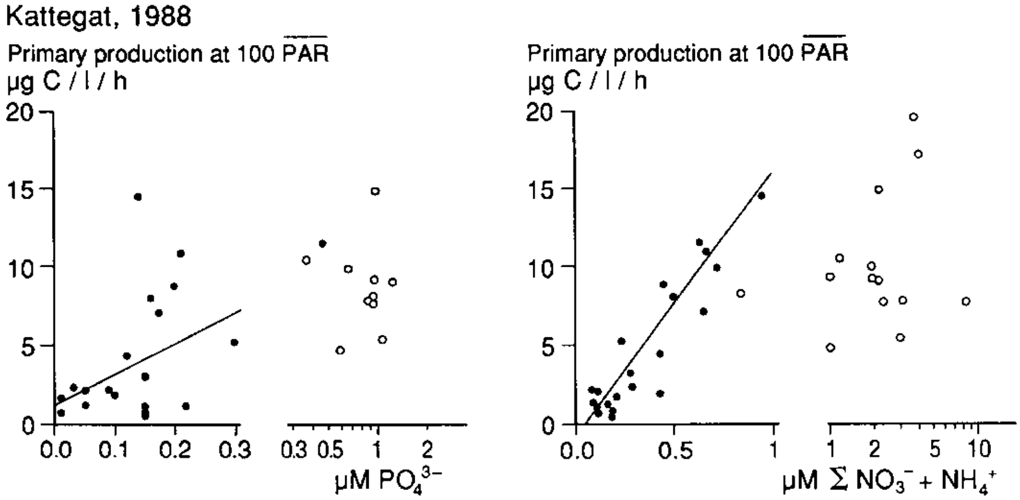


Figure 9

Primary production in the Kattegat as a function of the ambient phosphorus and nitrogen concentrations. The regression lines are based on values from May through September 1988, indicated by closed symbols. The open circles indicate values from March-April and October-November. (Spearman Rank Correlation coefficients N: 0.81, p = 0.004, P: 0.44, p = 0.055)

4. Discussion

The nutrient uptake of planktonic algae is influenced by a number of factors such as the biomass, the species composition and the growth rate of the algae, the light intensity including diel variation and the concentration of nutrients. In addition, the experimental approach applied will inevitably set limits on the validity of the results obtained, e.g. 24 h incubations, though optimal for obtaining daily values may, lead to nutrient (i.e. nitrate) exhaustion and induce bottle effects. Labelled nutrients once assimilated in the algae may reappear in the dissolved phase.

In this study we have been very cautious, e.g. we have used short term incubations and - established VI curves to arrive at daily values, and in addition taken isotopic dilution into account. In spite of this, the molar ratio of C,N and P uptake showed rather large variation. The deviation from the compositional ratio of algae (the Redfield ratio) was most prominent during periods of oligotrophy (May - June), which points to either heterotrophic uptake of nutrients (Wheeler and Kirchman 1986) or it may reflect "unbalanced" growth (Eppley 1981).

Data from the diel studies and the vertical profiles of nutrient uptake showed that the uptake of nitrate and phosphate was positively correlated to light intensity. The uptake of nitrate was in addition correlated to the concentration of nitrate but was found to be "inhibited" by ammonium concentrations exceeding ca. 1 µM, which is in accordance with numerous studies (e.g. Dortch 1990).

Nitrate that accumulated during the winter was the most important nitrogeneous nutrient during the spring bloom in Kattegat (Figure 5). Through out summer and autumn reduced nitrogen compounds (mainly ammonium) were by far the most important. Transient importance of nitrate in the summer was, however, demonstrated in August. "New" nutrients had probably been introduced to the surface waters prior to the sampling due to strong winds. In general, ammonium and phosphate were regenerated in the photic zone, in amounts sufficient to cover the assimilative needs. At several occasions, however, the assimilative and the regenerative fluxes were not in balance. Similar unbalance are recognized in other studies (e.g. Selmer 1988) and may be attributed to methological problems.

Over the year the "new" production (*sensu* Dugdale and Goering 1967) amounted to about 15 %, which is lower than the 25 % estimated for an adjacent area in the Kattegat (Sahlsten et al. 1988). The greater importance of nitrate relative to reduced compounds (urea included) in - their study may be due to: a) larger influences from land run-off (concentration of nitrate was higher in the Laholm Bay than at our Kattegat station A), 2) the ammonium uptake of the Laholm Bay may have been underestimated because remineralization was not accounted for, or 3) the difference may simply reflect year-to-year variations.

The sub-surface bloom of *Chrysochromulina polylepis* which occurred over a large area of the Kattegat and Skagerrak in May and June 1988 had deleterious effects on bottom fauna and caged fish. In the "stationary" or declining phase of the bloom in early June, nitrate still was the dominant nitrogen source for *C. polylepis* growth, and the pelagic remineralization was insignificant in accordance with the absence of heterotrophic organisms (Nielsen et al. 1990). Depth profiles of temperature and salinity and the large biomass of *C. polylepis* indicated that the water masses sustaining the bloom were intruding from outside the Kattegat.

In the Vejle Fjord, ammonium was consistently the most important nitrogen compound for algal growth (Figure 8). The original concept that nitrate uptake alone represents "new" production was clearly invalid during sampling in March and October as ammonium obviously was discharged into the inner part of the fjord (Figure 6). During a bloom of *Prorocentrum minimum* in September at Sts. 2 and 3, the ammonium uptake was much greater than the pelagic remineralization. The deficit seemed to be met by benthic ammonium release because advective transport of ammonium was insignificant (Figure 6).

The nutritional status of phytoplankton as reflected in the V_{max}/V_{amb} ratio indicated nitrogen deficiency in May through September in the Kattegat and in June and September in Vejle Fjord as the ratio exceeded 4-5. V_{max}/V_{amb} ratios exceeding 4-5 for ammonium have formerly been found in nitrogen limited algal cultures (McCarthy and Goldman 1979) and natural algal assemblages (Glibert and McCarthy 1984). The transient increase in V_{max} for nitrate might be less understandable as nitrate is usually supplied to the photic zone by hydrographic (i.e. unpredictable) events. However, our results indicate that the planktonic algae were able to exploit transient increases in nitrate concentrations. This is in accordance with some studies (e.g. Collos 1980) while others have failed to show a comparable enhancement in the uptake (Horrigan and McCarthy 1982).

The crucial role of nitrogen for algal production was substantiated by the significant relationships between primary productivity and inorganic nitrogen. During the summer months the ambient concentration of inorganic nitrogen could thus explain more than 80 % the observed variation in primary production in Kattegat. Unfortunately, we are not aware of comparable data in the literature.

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Number C9

Simple Empirical Lake Models

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1. Introduction

During the last two decades a large number of empirical models have been developed which describe the eutrophication level and biological structure of lakes in relation to nutrient loading (e.g. Vollenweider, 1976; OECD, 1982; McCauley and Kalff, 1981). Furthermore, models dealing with lake recovery and lakes as a nutrient sink have also appeared (Dillon and Rigler, 1974; Chapra, 1975; Benndorf and Pütz, 1987; Cullen and Forsberg, 1988; Marsden, 1989; Sas, 1989). These models were mainly developed using data from relatively deep and stratified lakes and may thus not be fully applicable to typical Danish lakes, which are extremely shallow (mean depth <3 m) and mostly unstratified.

The present study was initiated in order to test the validity of existing models and, where necessary, to establish new models for Danish lakes.

In this paper we focus on five topics relevant to eutrophication level and biological structure in relation to nutrient loading;

1. Phosphorus balance
2. Nitrogen balance
3. Secchi-depth versus algal biomass, suspended matter and P-concentration
4. Biological structure versus P-concentration
5. Lake recovery after P loading reduction.

Our main findings are briefly reviewed below.

2. Data and databases

During the project a database was established that contains information about lake morphometry, catchment utilization, chemical-, physical- and biological data from 28-600 Danish lakes, (the number depending on the variable concerned). The data were derived from the routine monitoring programmes undertaken in the 14 Danish counties and from various research investigations conducted by our Institute.

The Danish lakes are characterized by being small and shallow, and having a high flushing rate (fig. 1). Furthermore, they are highly eutrophic, mainly as a result of a high nutrient loading from domestic sewage and drainage water from agricultural land. Consequently phosphorus concentration (P-lake) and chlorophyll levels are high and transparency low (fig. 1).

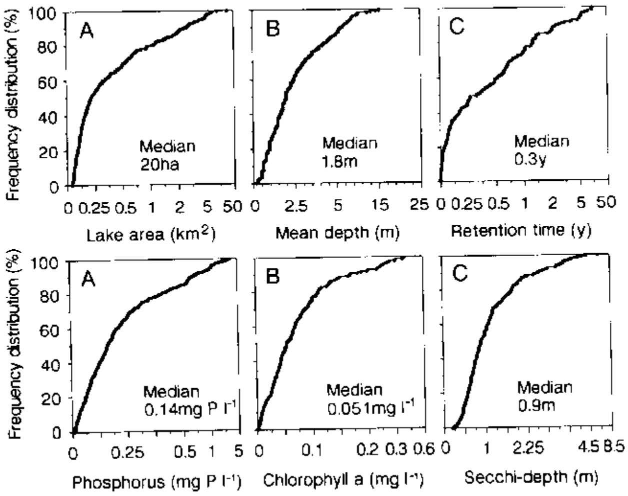


Figure 1: Frequency distribution of lake area, mean depth, hydraulic retention time, P-concentration in lake water, algal biomass and Secchi-depth in 220-450 Danish lakes larger than 5 ha. (From Jeppesen et al., submitted)

3. Results and discussion

3.1 Phosphorus balance

The P-loading of Danish lakes (131 lakes) was high, on average 13.1 g P m^{-2} y^{-1} (median = 2.5 g P m^{-2} y^{-1}), reflecting a high input concentration (median = 260 μg P l^{-1}) in combination with an overall low hydraulic retention time. 21 published loading-concentration models were tested on the data available from the 131 lakes using five test criteria. 4 of the models (i.e. Vollenweider, 1976; Canfield and Bachmann, 1981; Prairie, 1988 and 1989) predicted P-lake better than did the other 17. However, the mean difference (observed-predicted concentration) was rather high (52-74 μm P l^{-1} or 33-41%) although comparable to that found elsewhere (e.g. Canfield and Bachmann, 1981). It was not possible to improve these 4 models by alternative formulations.

tration (N-inlet) accounted for 50% of the variation (N-lake) increasing to 63% when the retention time and mean depth were included. These three models, together with three published models, were validated on an independent set of data from a further 98 lakes. Nitrogen loading in these lakes was based on export coefficients for the catchments involved (Jensen et al., in press, a).

	Calibration	Validation
N-lake = 0.45*N-inlet	$r^2=0.50$	0.33
N-lake = 0.42*N-inlet*tw**0.11	$r^2=0.61$	0.51
N-lake = 0.34*N-inlet*tw**(-0.16)*Z**0.17	$r^2=0.63$	0.60

Table 1: The three models relating total nitrogen in the lake water (N-lake) to inlet total nitrogen (N-inlet), hydraulic retention time (tw, y) and mean depth (Z, m). r^2 for calibration and validation on data from 69 lakes and 98 lakes, respectively is also shown. (From Jensen et al., in press b)

The three new models were found to be highly valid. The r^2-values were comparable to those obtained during calibration and the mean difference between observed and predicted concentration ranged from 2 to 8%. In contrast, mean difference ranged from 17 to 153% for the three published models (i.e. OECD, 1982; Bachmann, 1984; Lijklema et al., 1989). The difference between the validity of the prediction in our models, and that in the three published models may be related to differences in the data used for calibration and in the importance of nitrogen-fixing cyanobacteria (Jensen et al., in press a, b).

The 4 selected models deviated from the other models in that they predicted a relatively low P-retention. This may be interpreted as reflecting a high P-release from the sediment to the lake water due to the overall higher temperature and frequency of resuspension in shallow lakes in comparison with deep lakes. In addition, the high ortho-P concentration of Danish lakes may have further reduced net sedimentation.

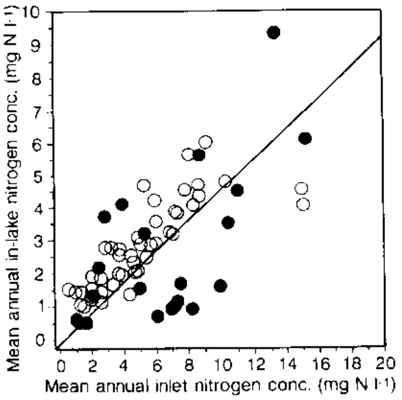


Figure 2: Relationships between mean lake concentration of total nitrogen and mean inlet concentration. ○: maximum depth <10 m, ● maximum depth >10 m. (From Jensen et al., in press, a).

3.2 Nitrogen balance

The N-loading of Danish lakes (69 lakes) was high, on average 142 g N m^{-2} y^{-1} (median = 52 g N m^{-2} y^{-1} corresponding to an average inlet concentration of 5 mg N l^{-1}). In spite of a large variation in loading, the N-loss was relatively constant, averaging 43% ± 6% (95% C.L.) of the loading. Denitrification accounted for 77% of the loss, while 23% was permanently burried.

In order to predict the in-lake nitrogen concentration (N-lake), three different models of increasing complexity were developed by regression analysis (fig. 2 and table 1). The inlet concen-

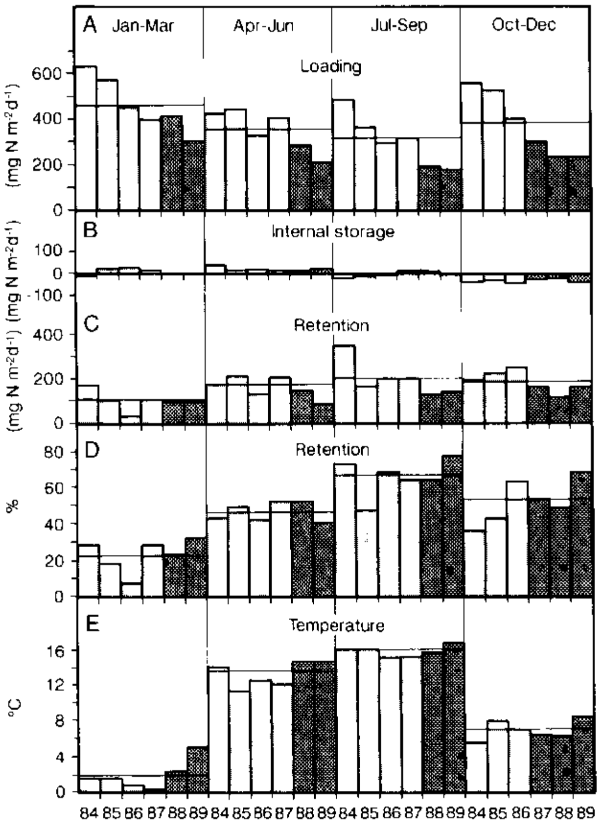


Figure 3:

Quarterly nitrogen budget of and temperature in Lake Søbygård during 1984-1989. A: External loading, B: Internal storage in the lake water, C. Net retention, D: Net retention in percentage of external loading, E: Mean water temperature. The horizontal line represent average values for each quarter during 1984-1989. (From Jensen et al., in press, b).

```
SD = 6.7*chla**(-0.47)
SD = 6.3*SM**(-0.66)
SD = 0.36*P-lake**(-0.29)*Z**0.51
```

Table 2: Three models relating Secchi-depth (SD, m) to chlorophyll a (chla, µg l^{-1}), suspended matter (SM, mg l^{-1}), total P in the lake water (P-lake mg P l^{-1}) and mean depth (Z, m).

A more detailed analysis was carried out in Lake Søbygård where a 36% reduction in N-loading occured during the study as a result of the closure of a local slaughterhouse. In spite of the short retention time (18-27 days) N-loss was high, ranging from 38 to 53% annually. Furthermore, N-loss was not affected by the reduction in N-loading, indicating that internal N-loading is not an important resilience factor in the recovery of this lake. There was marked seasonal variation in N-loss ranging from 23% of loading in the 1st quarter to 65% in the 3rd quarter (fig. 3). The seasonal variation in N-loss showed a hysteresis-like relationship to temperature, with a relatively high percentage in the 4th quarter. The latter probably reflects a high accumulation of organic matter in the sediment during summer and autumn, with resultant high denitrification in the 4th quarter (Jensen et al., in press, c).

It can be concluded that even shallow lakes with a short hydraulic retention time may contribute importantly to the reduction of nitrogen and thereby reduce N-loading, and hence eutrophication, of coastal areas.

3.3 Secchi-depth versus algal biomass, suspended matter and P-lake

That there is a close relationship between algal biomass, expressed as chlorophyll a, and the Secchi-depth is widely accepted (Lorenzen, 1980; Canfield & Bachmann, 1981). However for the shallow Danish lakes, the relationship was found to be weak (fig. 4 and table 2), partly due to differences in the impact of resuspension on different lakes. Hence, the best relationship was found to be between the concentration of suspended matter and Secchi-depth (fig. 4 and table 2).

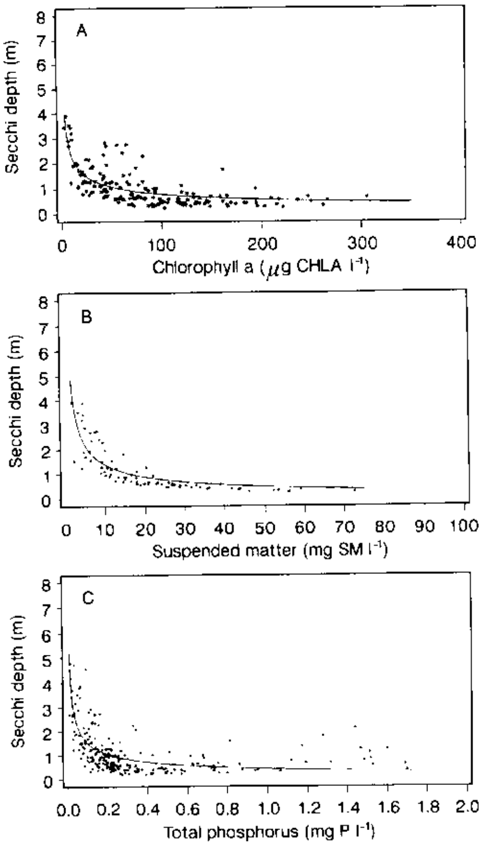


Figure 4: The relationship between Secchi-depth and algal biomass (A), suspended matter (B) and total phosphorus (C) in Danish lakes.

The impact of resuspension was confirmed by an extensive study in several lakes (Kristensen et al., submitted and unpubl.). For example, in Lake Arresø (surface area 41 km^2) a quantity of sediment corresponding to the top 2-3 mm of the sediment was resuspended on average every second day: resuspension per se reduced the Secchi-depth to an average of 0.5 m (Kristensen et al., submitted).

In order to predict changes in Secchi-depth in relation to changes in P-loading, we established a relationship between P-lake and Secchi-depth (fig. 4 and table 2). The model could be improved significantly by including mean depth. The Secchi-depth increased approximately with the square root of the mean depth, which is in accordance with the fact that resuspension per se decreases with depth (Kristensen et al., submitted). The relationship could not be improved by including N-lake to the model.

Resuspension may thus be important for the light climate in shallow lakes. Moreover, it may affect lake recovery following loading reduction because high organic matter accumulation during the period of high nutrient loading may lead to enhanced resuspension: This may in turn prevent improvement in the light climate, and hence recolonization by submerged macrophytes.

3.4 Biological structure versus P-lake

When P-concentration changes in shallow lakes, marked self-perpetuating changes in the biological structure often take place. Hence, when P-lake was low, the maximum depth distribution of submerged macrophytes, and the area covered by them, was found to be high (fig. 5C). The fish stock caught by survey gillnets of multiple mesh sizes was dominated by piscivores (pike and perch) (fig. 5D). The phytoplankton community (fig. 5B) was dominated by chrysophyceans and flagellates while the biomass was low (fig. 5A): consequently transparency was high (fig. 5E). The dominance of piscivorous fish, the low phytoplankton biomass and the high transparency is a pattern common in lakes in which P-concentration is low and in shallow lakes with a high coverage of submerged macrophytes (e.g. Moss, 1980; Timms and Moss, 1984; Jeppesen et al., 1990b, c). This pattern may be even more pronounced, among other reasons because young perch are better suited than roach to feed in amongst the vegetation (Winfield and Townsend, 1988) and because the number of pike increases (Grimm, 1989). This leads to a further reduction in planktivorous fish and consequently to reduced predation on zooplankton and hence increased grazing on phytoplankton. The reduced algal biomass and the sheltering effect of submerged macrophytes on resuspension should then result in improved light condition even when the nutrients concentrations are high. This was confirmed by the results in our study (fig. 6) in which lakes with a high macrophyte coverage had a higher transparency at a specific P-level than lakes with a low coverage (the data set was screened for lakes with nitrogen limitation).

Although homeostatic control of the biological structure by the submerged macrophytes (Moss, 1980; Scheffer, 1989) may prevent changes in the structure in response to increasing P-concentration, our results (fig. 5 and 6 and Jeppesen et al., 1990c and submitted) show that the homeostasis becomes ineffective at higher P-levels. The result is a reduction in the maximum depth colonization of submerged macrophytes, an increase in the density and relative importance of planktivorous fish followed by an increase in phytoplankton biomass and hence a reduction in transparency.

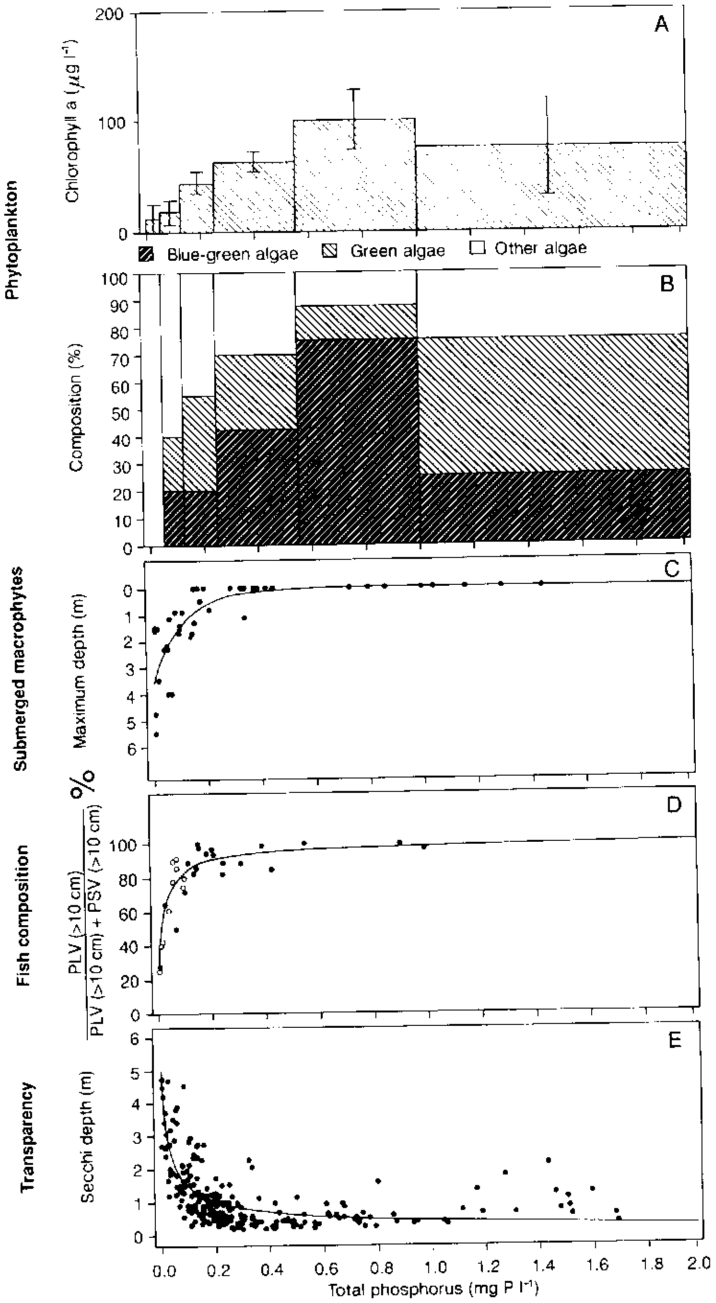


Figure 5:
The relationship between the concentration of total phosphorus in the lake water (May-Sept.) and A: algal biomass (July-Sept.), B: the percentage distribution of dominating species and genera among some phytoplankton classes (July-Sept.), C: maximum colonization of submerged macrophytes, D: ratio of numbers of planktivorous fish (PLV) to planktivorous and piscivorous (PSV), all larger than 10 cm caught during standardized test-fishing with survey gill nets of multiple mesh sizes.● = Danish lakes,○ = data from Lessmark, 1983, where piscivores included only perch and planktivores only roach, E: mean Secchi-depth (May-Sept.). (From Jeppesen et al., 1990b and submitted).

Our results also suggest that there are two thresholds for changes in the biological structure of shallow lakes (Jeppesen et al., submitted). The first threshold is at about 100 μg P l^{-1}: below this level the biological structure may change markedly with even minor changes in P-concentration (Fig. 5). The second threshold is at 1000 μg P l^{-1}. In between the two thresholds, cyanobacteria is the dominating phytoplankton class and the fish community is totally dominated by planktivores (Fig. 5). Cyanobacteria, by affecting the filter feeding zooplankton (and thereby the "top-down" control) negatively, and by their ability to store nutrients, may have homeostatic properties similar to those of the submerged macrophytes. Thus at P-concentrations between the two thresholds, changes in the biological structure, e.g. by biomanipulation, may be difficult to obtain (Jeppesen et al., 1990b, c).

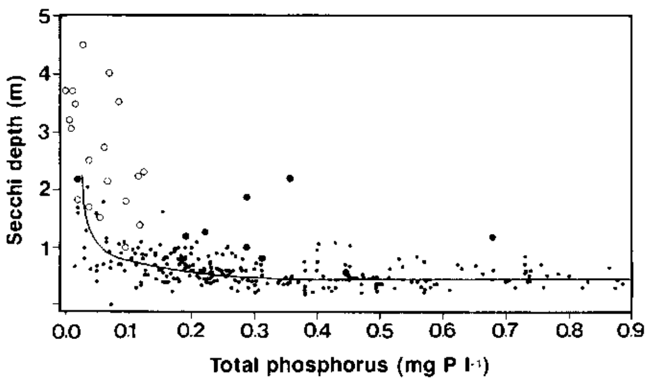


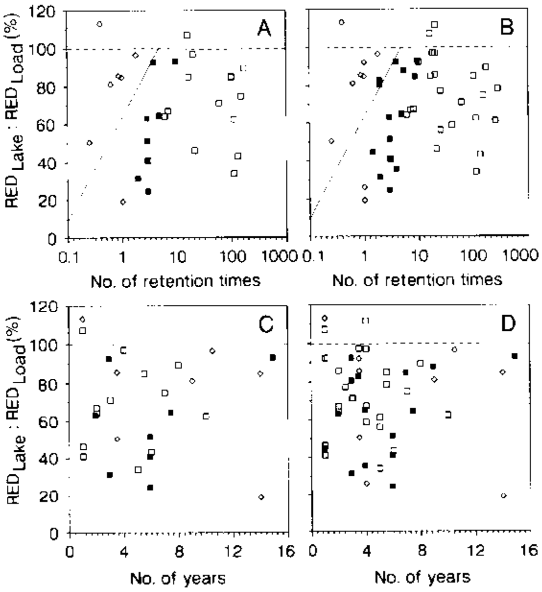
Figure 6:
Mean Secchi-depth in relation to mean lake water total-P (May-Sept.) for shallow (mean depth <3 m) Danish lakes. ○ = lakes with a high cover of submerged macrophytes and a surface area >3 ha; ● = lakes with a high cover of submerged macrophytes and a surface area <3 ha; and • = lakes with only minor unknown cover of submerged macrophytes. (From Jeppesen et al., 1990b).

At P-concentrations above about 1000 $\mu g\ P\ l^{-1}$, green algae and flagellates were the dominating phytoplankton groups. These lakes are further characterized by high biological instability, mainly due to large fluctuations in the fish stock (fish kill and shifting recruitments) with major "top-down" effects (Jeppesen et al., 1990a, b, c). During periods with low density of planktivorous fish, the density of filter feeding zooplankton is high, the phytoplankton biomass is low and often dominated by flagellates, and the transparency is high (Fig. 5). During high densities of in particular young fish, green algae dominate and the transparency is low. In this P-regime, changes in the biological structure e.g. by biomanipulation, may be possible but only short term effects would be expected (Jeppesen et al., 1990b, c).

3.5 Lake recovery

Resilience in the recovery of lakes after a reduction in P-loading is common (Cullen and Forsberg, 1988; Sas, 1989; Ryding, 1989; Marsden, 1989). We analysed the data from 27 Danish case-studies of recovery following reduction of P-loading. In general the percentage reduction in P-lake was less than or similar to the percentage reduction in loading, even 8-10 years after the loading reduction (fig. 7C).

In lakes with a high hydraulic retention time (tw), the percentage reduction in P-lake occurred faster than would by expected from simple dilution, suggesting that part of the surplus phosphorus pool in the lake water had settled and was immobilized in the sediment. In contrast, in most lakes with a low tw, a severe delay was observed (fig. 7) and the recovery period, expresses in tw, was 10-100 times larger than in lakes with a high tw. This difference could be partly explained by the fact that the P-pool accumulated during the period of high loading decreased with increasing tw (fig. 8). Thus during the recovery phase, lakes with a low tw then may suffer most from internal loading (Jeppesen et al., submitted). Besides the amount of P the sediment, however, the iron:P-ratio in the uppermost sediment was an important determining factor for the internal loading: the internal loading decreased with increasing iron:P-ratio, when this ratio was above 15 (Jensen et al., submitted).



<u>Figure 7:</u>
The ratio of the percentage reduction in lake P-concentration and P-loading versus hydraulic retention time (tw) (A, B) and the number of years (C, D) after the loading reduction, respectively. A and C are from Danish lakes and in B, D supplemented with data from Cullen and Forsberg (1988). □ tw ≤ 0.5 y, ■ tw>0.5 y, tw ≤2.5 y, ◇ tw>2.5 y. (From Jeppesen et al., submitted).

In summary it was concluded that the recovery of lakes with an extremely short hydraulic retention time may be long, and may even be just as long as for lakes with a high retention time (Jeppesen et al., submitted).

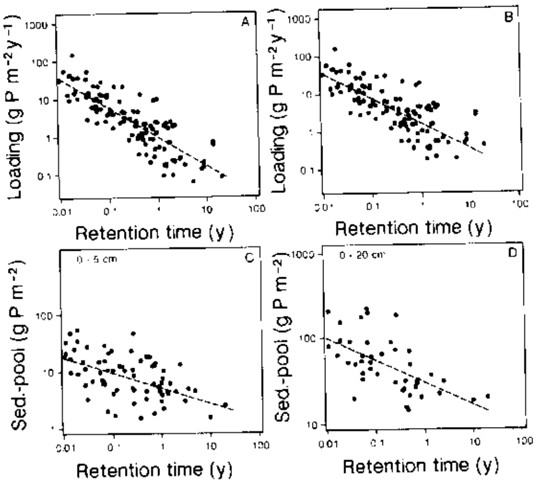


Figure 8: Relationships between hydraulic retention time and the P-loading (A, B) and the accumulated P in the sediment (C, D). A: P-loading, B: P-loading normalized by correcting for differences in P-retention according to the Vollenweider equation (Vollenweider, 1976), C: P-pool, 0-5 cm, D: P-pool, 0-20 cm (From Jeppesen et al., submitted).

In addition to internal loading, homeostasis in the biological structure may create resilience in the recovery process, as has been demonstrated in biomanipulation experiments in Lake Væng. There was no improvement in water quality of this lake during the first four years following loading reduction. However, a 50% removal of the planktivorous fish stock resulted in a self-perpetuating and long term shift in the biological structure with the consequence that the lake changed from a turbid water stage to a clear water stage, with subsequent reductions in N and P in the lake water (Søndergaard, et al., 1990; Jeppesen et al., 1990b and submitted).

Thus by using the empirical equations shown in fig. 5 we have defined threshold levels for the chance of successfully using fish manipulation as a supplement to loading reduction in lake restoration. It can be concluded that in shallow lakes (larger than 10 ha and not severely nitrogen limited) P-lake needs to be reduced to below 80-150 μg P l^{-1} before long term effects can be expected (Jeppesen et al., 1990b, c).

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Dynamic and Empirical Modelling of Biological Structure and Physical Factors in Lowland Streams

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DYNAMIC AND EMPIRICAL MODELLING OF BIOLOGICAL STRUCTURE AND PHYSICAL FACTORS IN LOWLAND STREAMS

Summary

The present study developed simple models that allow prediction of the seasonal biomass variation of rooted submerged macrophytes and microbenthic algae in small lowland streams. The models predict biomass variation as a function of regulating factors for which data can easily be obtained from permanent weather stations and routine water quality surveillance programmes. Models have also been developed that allow prediction of water temperature, velocity of flow and net accumulation of organic matter on the stream bottom. All models predict daily values for each dependent variable and require the input of daily values for air temperature, incident radiation, discharge of flow, and flow-related factors. Frequent data on water chemistry are also required for the application of some of the models. Given the present state of knowledge, however, only a few of the models are generally applicable. Thus suggestions for future stream research that will enable the models to be improved are also presented.

1. Introduction

The functioning of small lowland streams in temperate regions of the northern hemisphere is closely linked to human activity in the catchments and the resulting effects on the biological structure of the stream ecosystem.

In general, Danish streams are characterized by small size, mild slopes, low velocity of flow, heavy loading with nutrients and organic wastes from human activity in the catchment, and large accumulations of rooted macrophytes and microbenthic algal biomass. Streams with a high macrophyte biomass normally exhibit temporary accumulation of large amounts of oxygen-consuming organic matter on the stream bottom. Therefore, in order to avoid flooding of adjacent farmland, most streams are channelized and heavily managed. Management practices include weed cutting and sediment removal, leading to impoverishment of the stream as a fish habitat.

In order to be able to understand stream functioning and to be able to give more ecologically sound management advice, it is essential to understand the factors that regulate the development of the biological components, and how these components interact.

The aims of the present study were:

- to develop simple models for the seasonal development of important biological components of streams i.e. rooted macrophyte biomass and microbenthic algal biomass, and the accumulation of organic matter on the stream bottom

- to develop simple models for important regulating factors i.e. water temperature and velocity of flow.

A simple model is herein defined as a model that allows prediction of the seasonal variation of a particular response variable with a time step of one day, as a function of regulating factors for which data can easily be obtained from permanent weather stations or from routine stream water quality surveillance programmes.

2. Study sites

Model development was based on data collected from 9 small streams in Central Jutland (figure 1) that covered a relatively wide range of soil types, land use, physical characteristics, water chemistry and biological conditions.

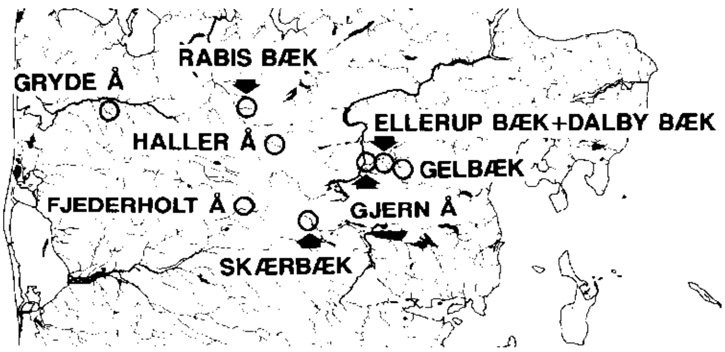

Figure 1
Study sites and their location.

Table 1 presents a summary of mean values for physical and chemical characteristics of the study streams during the entire study period. Some streams (Skærbæk and Rabis bæk) showed very little seasonal variation in discharge, indicating high infiltration rates and good contact

Stream	Rabis bæk	Haller å	Gelbæk	Gjern å	Skærbæk	Gryde å	Fjederholt å	Ellerup bæk	Dalby bæk
Discharge Q, $m^3\ s^{-1}$	0.123 (0.094-0.168)	0.190 (0.070-0.592)	0.053 (0.002-0.568)	0.723 (0.313-2.17)	0.091 (0.072-0.118)	0.386 (0.190-0.83)	0.311 (0.217-0.469)	0.024 (0.002-0.433)	0.101 (0.04-0.457)
Mean depth H, m	0.22 (0.11-0.54)	0.52 (0.23-0.79)	0.23 (0.03-0.58)	0.59 (0.31-1.20)	0.12 (0.10-0.15)	0.76 (0.39-1.12)	0.70 (0.43-0.96)	0.10 (0.04-0.35)	0.22 (0.14-0.29)
Mean water velocity V, $m\ s^{-1}$	0.32 (0.11-0.46)	0.19 (0.07-0.46)	0.14 (0.02-0.62)	0.36 (0.15-0.61)	0.38 (0.26-0.46)	0.20 (0.09-0.39)	0.18 (0.094-0.39)	0.21 (0.07-0.90)	0.45 (0.30-0.93)
Q_{max}/Q_{min}	1.8	8.5	284	6.9	1.6	4.4	2.2	217	11
Ammonia, $mg\ N\ l^{-1}$	N/A	0.053 0-0.843	0.361 (0.060-1.05)	0.214 (0.004-1.50)	0.015 (0-0.051)	0.063 (0-0.1.35)	0.118 (0.027-0.512)	0.370 (0.016-3.16)	0.256 (0.083-0.589)
Nitrate, $mg\ N\ l^{-1}$	1.79 (1.38-2.32)	3.04 (1.61-9.0)	7.30 (0.60-45.0)	3.31 (0.47-12.0)	2.81 (0.58-3.63)	2.13 (0.72-4.69)	3.49 (1.36-4.16)	5.41 (2.17-12.2)	2.70 (0.95-6.24)
Total phosphorus, $mg\ P\ l^{-1}$	0.095 (0.043-0.404)	0.095 (0.013-0.366)	0.398 (0.010-6.30)	0.192 (0.007-0.729)	0.014 (0-0.059)	0.062 (0.011-0.446)	N/A	0.130 (0.015-3.34)	0.149 (0.051-1.73)

Table 1: Physical and chemical characteristics of the study streams. For each variable the mean and range (in parenthesis) are given. N/A means no measurements available.

with the ground-water magazine whereas others, in particular Gelbæk and Ellerup bæk, showed extreme seasonal fluctuation in discharge emphasizing the importance of surface run-off and discharge of water through drainage tiles. Nutrient content was high in most streams. Nitrate concentration in particular was high due to leaching from agriculture and all streams, except Haller å, Skærbæk and Gryde å, which have limited agriculture in their catchments, also had a high concentration of phosphorus.

3. Methods

Field measurements were not undertaken specifically for the present project. The models have been developed from climatological, hydraulic, chemical and biological data obtained in earlier projects by the authors. Consequently it is beyond the scope of the present summary to describe the methods in detail. However, for the convenience of the reader, a summary of variables and parameters used in the models is presented in table 2.

Those with a special interest in the data acquisition, sampling strategy and analytical procedures on which this work is partially founded, are referred to the main report by Thyssen et al. (1990) and related papers (e.g. Thomsen & Thyssen, 1979; Thyssen, 1982; Erlandsen & Thyssen, 1983; Kelly et al., 1983; Iversen et al., 1984; Jeppesen et al., 1984; Rebsdorf et al., 1988 and Iversen et al., 1990).

4. Results and discussion

The 5 stream models that have been developed describe the seasonal variation of

a. water temperature
b. velocity of flow
c. biomass of rooted submerged macrophytes
d. biomass of microbenthic algae and
e. accumulation of organic matter on the stream bottom.

Each of these models will be presented separately, including a description of model structure, parameter estimates, model applicability and suggestions for future research that will enable the models to be improved.

Symbol	Meaning	Unit
a	Specific hydraulic loss of macrophyte biomass	day^{-1}
B	Biomass of rooted submerged macrophytes	g dw m^{-2}
Bio	Biomass of microbenthic algae	mg chl m^{-2}
Bio_{max}	Yearly maximum biomass of microbenthic algae	mg chl m^{-2}
C	Concentration of POM in the stream water	mg AFDW l^{-1}
d	Day number	
DIP	Concentration of dissolved inorganic P in the stream water	mg P l^{-1}
e_B	Light attenuation coefficient in the macrophytes	m^2 g^{-1} dw
e_w	Light attenuation in the stream water	m^{-1}
H	Daily mean depth	m
I_k	Half saturation constant for light	E m^{-2}
I_o	Daily integral of photosynthetic active radiation (400-700 nm) at the water surface	E m^{-2}
I_z	As I_o, but at depth z	E m^{-2}
K_s	Half saturation constant for DIP	mg P l^{-1}
Cumorg	Accumulated organic matter on the bottom	g AFDW m^{-2}
n	Mannings roughness coefficient	$m^{-\frac{1}{3}}s$
$p_{max}(20)$	Maximum specific rate of gross primary productivity at 20 °C	day^{-1}
Q	Daily mean discharge	m^3 s^{-1}
R(20)	Specific rate of respiration at 20 °C	day^{-1}
S	Slope of water table	m m^{-1}
S_i	Daily integral of global insolation	MJ m^{-2}
T_v	Daily mean water temperature	°C
T_l	Daily mean air temperature	°C
V	Daily mean velocity of flow	m s^{-1}
θ_p	Photosynthetic temperature coefficient	-
θ_R	Respiratory temperature coeffecient	-
μ	Shade factor from marginal vegetation	-
σ_w	System noise	day^{-1}
σ_v	Measurement noise	-

Table 2: Parameters and variables used in the models for water temperature, velocity of flow, macrophyte biomass, microbenthic algal biomass and accumulated organic matter.

a. Water temperature (T_v).

In order to describe daily mean water temperature in streams as a function of regulating factors for which data was easily obtainable, an empirical model with 4 parameters was formulated. In the model T_v is linearly related to global insolation S_i and logistically to daily mean air temperature T_l, as given by the equation:

$$T_v = a\,S_i + \frac{c}{1 + \exp(-b\,(T_l - m))} \qquad (1)$$

where a, b, c and m are the parameters. 487 corresponding measurements of T_v, T_l and S_i from 14 streams in Central Jutland were available for estimating the parameters. In order to differentiate streams according to the degree of shading by riparian vegetation, seasonal discharge variations and size (stream order sensu Strahler, 1957), 4 categories of stream were defined.

Category I: 1st order streams open to light (more than 75% of incident radiation reaching the water surface) with little seasonal variation in Q ($Q_{max}/Q_{min}<5$).

Category II: 2nd-3rd order streams open to light with moderate Q-variation ($5<Q_{max}/Q_{min}<25$).

Category III: 2nd-3rd order streams, moderately shaded (25-75% of incident radiation reaching the water surface) with moderate variations in Q.

Category IV: 1st order streams, heavily shaded, (less than 25% of the light reaching the water surface), with extreme Q fluctuations ($Q_{max}/Q_{min}>25$).

The parameter estimates determined by applying the model to the measurements are given for each category in table 3.

Category	a	b	c	number of localities	number of observations
I	0.046 (0.016)	0.124 (0.009)	16.0 (0.3)	2	61
II	0.123 (0.014)	0.165 (0.005)		2	52
III	0.030 (0.008)			4	238
IV	0			6	136

Table 3: Parameter estimates in the temperature models for 4 stream categories. Standard errors are given in parenthesis.

Category I streams predominantly conduct groundwater and their temperatures are little influenced by S_i and T_l. The influence of S_i in heavily shaded streams (category IV) was negligible ($a=0$), whereas streams in Category II and III only differ with respect to the shade factor, which appears as a difference in the value of parameter a.

These models can be applied to simulate the daily mean water temperatures of small streams ($Q<2$ m^3 s^{-1}) for which the degree of shading by riparian vegetation, the daily mean air temperature and the integral values of daily global insolation are known. The model can not be recommended for the prediction of temperature conditions in larger streams and lake outlets.

b. Velocity of flow (V).

Prediction of flow velocity, V, in streams is normally made using the Mannings equation, which relates V to the hydraulic radius, the slope of the energy line and n, a measure of the roughness of the stream channel. However, due to high seasonal variation in roughness of small lowland streams caused by submerged macrophytes, the application of Mannings equation with a constant value of n may lead to erroneous estimates of V.

Analysis of 449 corresponding measurements of V and the factors that are supposed to determine it, obtained from 8 streams, revealed that much better predictions could be obtained with an empirical 5-parameter model (equation 2) which includes physical characteristics of the stream channel (slope and meandering), discharge Q and biological structure in terms of biomass of submerged macrophytes B

$$V = 79.3\ Q^{0.47}\ B^{-0.058}\ S^{0.69}\ M^{-0.79} \qquad (2)$$

M is an arbitrary measure of meandering. If the channel is straight, M is 1. A natural meandering course is assigned an M of 3. Intermediates are assigned a value of 2. The model may be still be applied in cases where no macrophytes are present: B is then assigned a value of 1.

The relationsship between the measured and predicted values of V is shown in figure 2, and the residuals in figure 3. The model explained 76% of the variation in V and predicted velocity fairly well in small streams with discharges below 2 $m^3\ s^{-1}$, slopes below 7 o/oo and macrophyte biomasses below 500 g dw m^{-2}. The largest residual was 0.17 $m\ s^{-1}$, and the standard deviation was as low as 0.059 $m\ s^{-1}$.

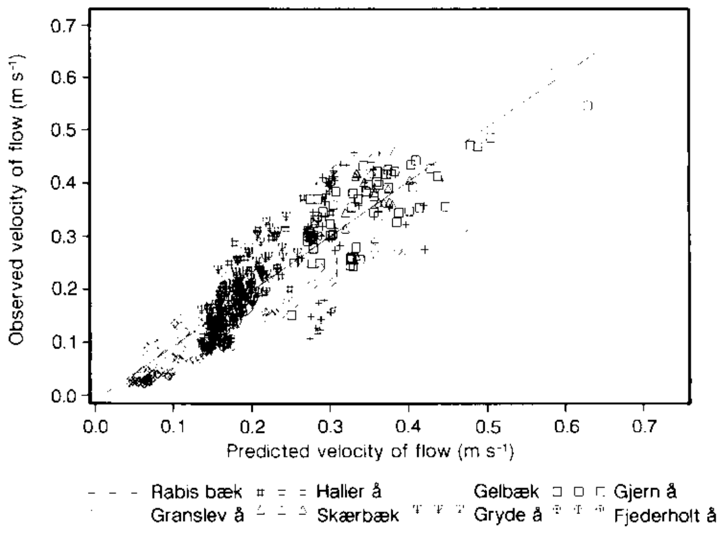

Figure 2
Observed velocity of flow versus velocity predicted by equation 2 in 8 small lowland streams.

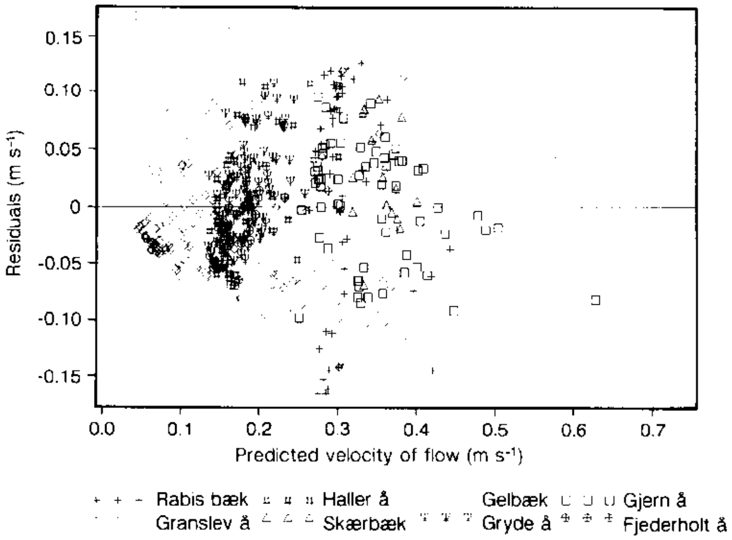

Figure 3
Residuals (observation-prediction) in the model for velocity of flow in lowland streams. Symbols are explained in figure 2.

In order to be able to further improve the model, more research is needed on the hydraulic roughness of different submerged and emergent macrophyte species.

c. Biomass of rooted submerged macrophytes (B).

Small macrophyte rich streams are a dominant feature of the open landscape in cultivated regions of northern Europe (Westlake, 1975). Their low flow velocity, fine grained sediment and the shallow depth provide a suitable environment for luxuriant growth of submerged macrophytes. The macrophytes play a key role in the ecology of these streams and are important primary producers. They modify velocity and sedimentation patterns and increase substratum available for microorganisms and invertebrates (Sand-Jensen et al., 1988). The suitability of a habitat for fish is directly proportional to the abundance of macrophytes (Mortensen, 1977) and macrophyte biomass has recently been shown to be a valuable food resource for higher trophic levels (Iversen et al., 1990).

For these reasons, and in order to be able to undertake ecologically sound stream management, it is important to know how macrophyte biomass is regulated, and how this knowledge can be used to develop models describing the seasonal variation of macrophyte biomass.
Two macrophyte biomass models have been developed. Model I describes the seasonal development of a mixed macrophyte community composed of 3 co-dominant species in a moderately shaded stream. Model II is a model for Ranunculus peltatus Schrank. in a stream open to light.

Both models have the same structure, which is a modification of a model developed by Titus et al. (1979) for Myriophyllum spicatum L. in lakes. The model is a 1st order differential equation of the form:

$$dB/dt=[f_p(S_i,H,T_v,\mu) - f_r(T_v) - f_t(Q)] * B \quad (3)$$

which describes seasonal macrophyte biomass change as the difference between the specific rate of production (f_p) and specific rates of loss through respiration (f_r) and non-respiratory hydraulic loss determined by storm flow events (f_t).

Production and loss functions are defined as follows:

Production:

$$f_p = p_{max} * [I_z/(I_k+I_z)]$$

$$p_{max}(T_v) = p_{max}(20°C)\ \theta_P^{(Tv-20)}$$

$$I_0 = [\mu + (1-\mu)/(1 + \exp(-b(d-m)))]*2.1*S_i$$

$b = -0.074$, $m = 194.5$ (May to September)
$b = 0.123$, $m = 321.2$ (October to December)

$$z = H/4$$

$$I_z = I_o * \exp(-e_w z - e_B B)$$

Respiration:

$$f_r = R(20)\ \theta_R^{(Tv-20)}$$

Hydraulic loss function:

$$f_t = a * \max\left(0, \frac{1}{Q}\frac{dQ}{dt}\right)$$

The model contains 11 parameters, of which are 5 biological, 4 physical and 2 statistical. Parameter estimates were obtained by fitting the model to time series of biomass observations from two streams with daily values of photosynthetically-active radiation at depth z, discharge and water temperature as forcing functions treating equation 3 as a stochastic differential equation with additional measurement noise. The resulting noise process contains two variance parameters, one for system noise and one for the measurement noise. Model parameters are shown in table 4 and the relationship between the measured and predicted values of B in the moderately shaded Gryde Å are shown on figure 4.

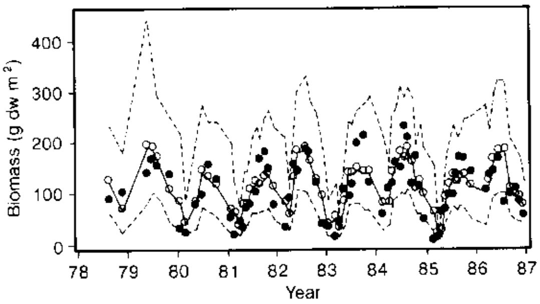

Figure 4
Seasonal and long term development of macrophyte biomass in Gryde Å. The points represent observations, the -○- curve model predictions and the dashed curves the upper and lower 95% confidence limits for the predictions.

The models were partly successfull when applied to 3 other streams with similar light conditions. The mixed community model predicted only partially correct seasonal biomass variation when applied to another stream (figure 5), whereas the monospecific model for Ranunculus peltatus was more accurate for streams not used for parameter estimation (figure 6).

Parameter	Value		Reference*
	Gryde å	Haller å	
p_{max}(20°C)	0.0666	0.0718	E
I_k	5.70	4.65	E
R(20°C)	0.0313	0.0200	E
θ_p	1.035	1.035	Madsen & Adams (1989)
θ_R	1.07	1.07	Erlandsen & Thyssen (1983)
a	0	0.10	E
μ	0.5	0.9	F
e_B	0.0099	0.0099	E
e_w	1.64	1.64	F
σ_w	0.0635	0.12	E
σ_v	0.135	-	E

*E: Estimation in model, F: Field measurement.

Table 4
Parameter estimates for two macrophyte models. For an explanation of the meaning of parameters refer to table 2.

In summary, causal and dynamic models have been developed that can predict seasonal variation of macrophyte biomass in streams solely on basis of daily measurements of incident radiation, water temperature and discharge. In order to improve the models specific models for each dominant macrophyte species should be developed. Such models should include species succession and descriptions of interspecific competition for light. For stream management purposes the models should include specific roughness estimates in order to be able to predict when stream management practices should be implemented so as to avoid flooding of adjacent farmland.

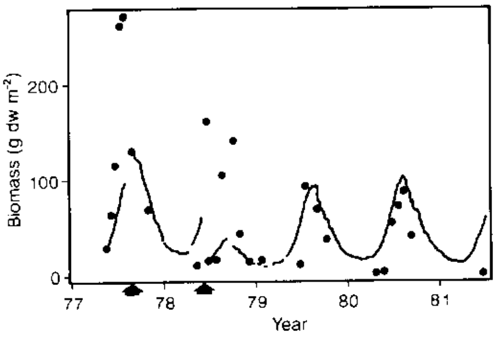

Figure 5
Example showing the application of the mixed community model to predict macrophyte biomass in Gjern Å. ● indicates observed biomass and the solid line model predictions. Arrows indicate cutting of macrophytes.

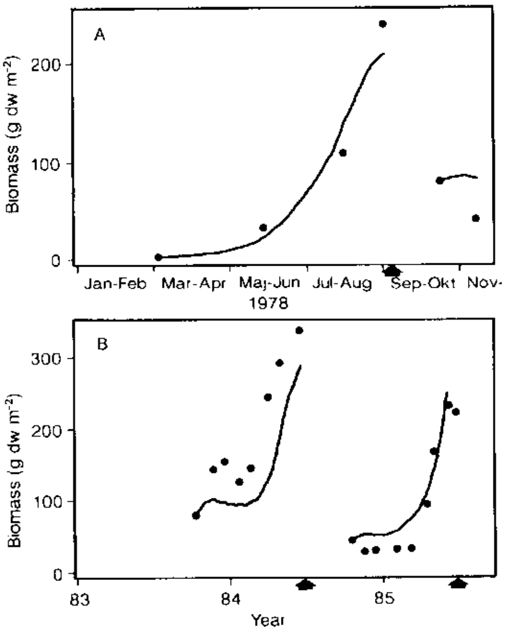

Figure 6
Application of the monospecific model to predict biomass development in Fjederholt Å (A) and Rabis Bæk (B). Symbols are explained in figure 5.

d. Biomass of microbenthic algae (Bio.).

Bottom dwelling microbenthic algae are a dominant feature of many shallow northern European lowland streams open to light. In addition to being an important food source for higher trophic levels, many important in-stream processes are closely linked to the benthic component (Sand-Jensen, 1983; Jeppesen et al., 1987; Iversen et al., 1990; Christensen et al., 1990). Regulation of algal growth has therefore received considerable attention in recent years. The model for microbenthic algae is the model developed for submerged macrophytes extended to include phosphorus limitation of growth. In order to estimate the parameters the model has been applied to 9 seasonal cycles of microbenthic algal biomass in 5 Danish streams. Light conditions at the bottom proved to be the main growth regulating factor. Although parameter estimates were realistic, they were very uncertain due to large measurement errors (heterogeneous distribution of the algae) and large system noise (variations in algal biomass not explained by the model). Thus a successful dynamic model of growth of microbenthic algae could not be developed. Instead we built an empirical model complex that allows prediction of growth initiation, algal biomass development, maximum algal biomass and its decay for streams with stable and unstable sediments, respectively. For both types of stream, growth was initiated when water temperature (10-day mean) exceeded 5.8 ^{o}C: the biomass then increased linearly with accumulated light reaching the stream bottom. For streams with stable substrates the relationship was:

$$BIO=0.52 * I_z(cum) \quad (n=29,\ r^2=0.94)$$

where BIO denotes the algal biomass t days after initiation of growth and I_z(cum) the accumulated photosynthetically active radiation at the bottom in the period from growth start to day t.

For streams with unstable sediments the relationship was:

$BIO=0.043 * I_z(cum)$ ($n=6$, $r^2=0.98$).

These equations are used until the biomass maximum is reached. The maximum biomass on fine grained sediments was determined by the concentration of dissolved inorganic phosphorus and could be described by Michaelis-Menten kinetics as:

$$BIO_{max} = 522 * DIP/(0.069 + DIP)$$

After the maximum had been reached, algal biomass was described according to Sand-Jensen et al. (1988) as:

$$BIO = BIO_{max} * \exp(-0.016*d)$$

where d is the day number after maximum has occurred.

At present we only recommend use of these relationships for prediction of the order of magnitude of algal development in streams. Model improvement requires more research, particularly on the factors that regulate maximum biomass (e.g. phosphorus, sediment stability, flow events, invertebrate grazing). Fundamental to the understanding of algal development in streams is an appropriate description of light conditions at the bottom. Such information is especially important for small streams lined by deciduous trees or herbaceous vegetation which can essentially prevent any light penetration to the autotrophic bottom components.

e. Accumulation of organic matter on the stream bottom.

Temporary retention of particulate matter on the stream bottom during summer may delay the transport of organically bound nutrients to that part of the year when downstream ecosystems are not nutrient limited (Jeppesen et al., 1987). Large accumulations of easily degradable organic matter on the stream bottom may, however, exhaust the oxygen content of the stream water and create poor living conditions for animal life. In the present study a dynamic model originally developed by Ariathuria and Krone (1976) and Iversen et al. (1984) has been modified to conditions in small streams. The model is a 1st order differential equation describing net accumulation of organic matter (CUMORG) as the difference between sedimentation and resuspension. Sedimentation is controlled by the concentration of particulate organic matter (POM) in the stream water (C), daily velocity gradients and resistance to flow caused by submerged macrophyte biomass. The sedimentation rate is given by:

$$d_{sed}/dt = (p_1+p_2*B)*(1-V_a^2/V_{a-1})*C$$

where V_a and V_{a-1} is the mean velocity of flow on day t and day (t-1), respectively. If $Va - V_{a-1}>0$ then $V_a^2/V_{a-1} = 1$. Resuspension is controlled by changes in velocity and the amount of accumulated organic matter on day t. Resuspension only occurs if $V_a>V_{a-1}$. If this condition is met then:

$$d_{resus}/dt = p_3*(V_a-V_{a-1})^2*CUMORG.$$

If $V_a<V_{a-1}$ then $V_a=V_{a-1}$, and the final model for net accumulation thus becomes:

$$d_{CUMORG}/dt = d_{sed}/dt - d_{resus}/dt$$

with 3 unknown parameters p_1-p_3 and 3 independent variables: velocity of flow, macrophyte biomass and concentration of POM. This model was fitted to observations on CUMORG from two streams in order to estimate the parameters. Figure 7 A and C present the fit for the two streams used for parameter estimation. On the right hand side of figure 7, the same parameters were used to simulate net accumulation outside the calibration period.

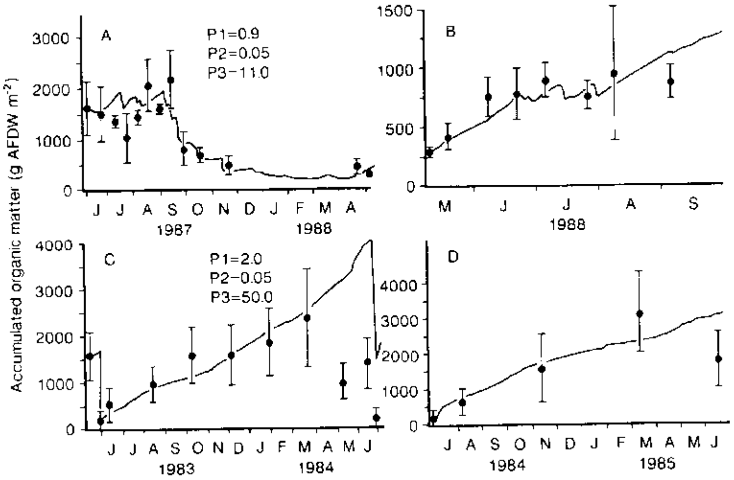

Figure 7
Calibration of the model for accumulated organic matter on the bottom of Gjern Å (A) and Rabis Bæk (C) (solid lines). ● indicate mean values and horizontal bars their standard errors. B and D show model predictions outside the calibration period for Gjern Å and Rabis Bæk, respectively.

Although model verification was quite successful in the studied streams, it should be noted that parameter values are stream specific, hence greaty reducing the general applicability of the model. More research is needed to identify and quantify the factors that regulate the dynamics of organic matter in streams. Such studies should be accompanied by frequent measurements of CUMORG in a number of streams that cover a larger variation in physical characteristics and macrophytic biomass then found in the present study.

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Erosion and Transport of Phosphorus to Streams and Lakes in Denmark

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EROSION AND TRANSPORT OF PHOSPHORUS TO STREAMS AND LAKES IN DENMARK

ABSTRACT

During the period from autumn 1987 to spring 1990 measurements of erosion and transport of phosphorus were undertaken from 2 catchment basins (Rabis and Langvad). In each basin also surface run-off from 2 plots of 0.5 ha was measured. Due to very mild and atypical winters without thaw events during the measuring period relatively small contributions of phosphorus transport and run-off were found.

From the Langvad basin (silt loam) transport of phosphorus was 35-73 kg total P/km^2/year and from the Rabis basin (sandy loam) 43-60 kg total P/km^2/year.

The surface total P run-off from the 4 plots was 3-51 g P/ha/year and largest in 1987/88. The highest run-off values were observed from silt loam soils with a 12% gradient.

Evaluating the results: Rabis (sandy soils) represents areas in Denmark having the lowest risk and rate of erosion, while Langvad (silt loam) represents areas in Denmark having a moderate rate of erosion.

1. INTRODUCTION AND PURPOSE

Until 1990 there were only a few rough estimates of the scale of soil erosion and its influence on transporting nutrients to streams and lakes in Denmark. Investigations here of transport in streams (Hasholt, 1983) indicate that some of the transported sediments originate from soil erosion. Hansen (1983) stated that water erosion from slopes could occur after rainstorms and during thaw events. Wind erosion is often observed in the sandy regions of Western Jutland (Kuhlman (1986).

On the basis of these studies a project was initiated with the purpose of evaluating the extent of erosion and transport of phosphorus in Denmark. The project was included in the NPo research programme running from 1985-90 and falls into three parts:

1. Transport of suspended sediment and phosphorus in the streams Rabis brook and Syv brook (Langvad) including observations of wind erosion.
2. Transport of sediment and phosphorus from 4 plots each of 0.5 ha.
3. To set measurements of erosion from 3 basins (Rabis, Syv and Gelbrook) into a country-wide

perspective by extrapolating the results by using a modified USLE (Universal Soil Loss Equation, Wishmeyer & Smith, 1978) adapted to the Danish soil data base.

The main purposes of the project were:

- to quantify the surface run-off of phosphorus in selected catchments.
- to account for factors controlling the transport of nutrients during surface run-off.
- to quantify the contribution of wind erosion on phosphorus transport.
- to evaluate the risk of soil erosion in Denmark.

The project was a joint venture between a group from the Institute of Geography, University of Copenhagen, (basin and river erosion), Danish Land Development Service (erosion from plots), and the Bureau of Land Data and Mapping (use of soil data base).

2. MATERIALS AND METHODS

The study basins are part of the larger NPo-Project and were selected partly because of earlier research and partly because of feasibility for studies of ground water pollution.

The two basins (approx. 10 km^2 each) represent "typical" Danish conditions. At Rabis the soils are loamy sands whereas the soils in the Langvad basin are silt loams.

Monitoring Stations

The purpose of the monitoring stations at the outlet of the basins was to measure the sediment transport very accurately, either continuously or at short time intervals so that erosion events in the basin might be related to the transport in the water course. In order to fullfil these requirements both manual and automatic sampling were carried out.

Manual water samples were collected weekly with a Swedish integrating sampler (Nilsson 1969).

There were two systems for measuring the sediment concentration. The first one based on automatic water-sampling by use of ISCO water samplers, the second was made indirectly by use of a Partech Suspended Solids Monitor with a single gap IR-sensor (Hasholt 1988).

Both measuring systems were placed in an insulated box to prevent freezing and growth of algae. The sampling frequency was twice a day, at 3 a.m. and 3 p.m. The samplers were emptied at least once a week.

Erosion from plots

The contribution of sediment to the water course from surface erosion was measured from 2 plots (0.5 ha) at selected sites in each drainage basin on slopes in the immediate vicinity of the stream. At the lower end of the slope two plots were placed on 6% slopes, loamy sand soils, in the basin of Rabis brook (Rabis 1 and 2) and the other two plots on 12% (Langvad 1) and 2% slopes (Langvad 2), silt loam soils, in the basin of Syv brook. Water and eroded soil were collected and led to the gauging system.

The gauging system consists of a 400 l tank where the water flow was recorded continuously, and flow-proportional samples were collected by a tipping bucket system (Hansen, 1989). The plots were weekly inspected.

Erosion in stream bed and banks

Both research basins were visited once a week. An inspection route along the main channel was followed by the observer who recorded all signs of erosion. The amount of sediment delivered to the water course was then computed by assuming that the observations at the selected stretches were representative of a certain reach of the water course.

Campaigns and supplementary investigations

During the winter season about four campaigns were carried out to study the spatial variations within the basin more closely and also the time variation in concentration within the different parts of the drainage system. Furthermore, land use was registered. The initiation of these campaigns was decided by use of the 5-day weather forecast in order to measure during rainstorms and/or thaw breaks as these are known to cause erosion.

Climatological parameters such as rainfall intensity, temperatures in air and at different levels in the soil as well as wind velocities were recorded within other parts of the NPo-Project and were thus available. Wind erosion events were also recorded.

Analyses and calculations of transport

The concentration of suspended solids was found by filtering through Whatman GF/F filters and subsequent drying at 60°C and weighing. Ignition loss was found by residue on ignition at 500°C. Content of phosphorus in filtered and unfiltered samples was found by use of a standard molybdate blue method and photometry. Dissolved solids were found by evaporation and drying.

The transport was computed by use of the concentration values and the corresponding discharge values.

3. RESULTS

The experimental period was from autumn 1987 to spring 1990 when the NPo research programme ended. Unfortunately, the climate was atypical, all three winters being very mild without any real frost/thaw events as in a typical Danish winter.

In the following, results from the 3 sub-investigations will be summarized.

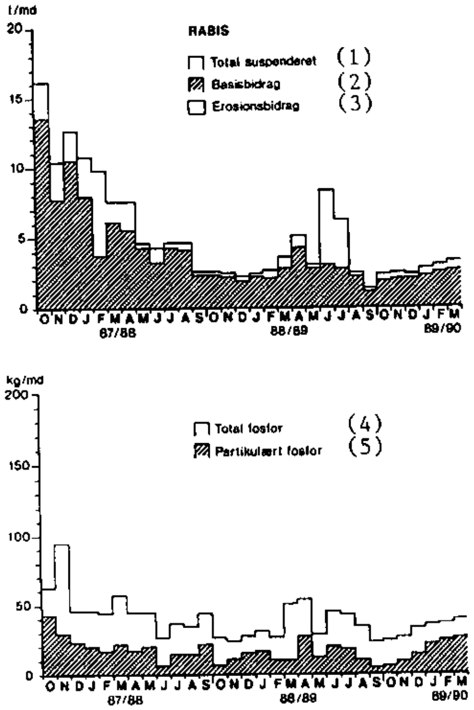
3.1 Transport of suspended sediment and phosphorus in Rabis brook and Langvad brook

The 24-hour transport values measured have been summed up, and monthly transport values are shown in Fig. 1 and Fig. 2. The same figures also show a grouping of the total transport in a base contribution and an erosion contribution. In table 1, transport and erosion contributions have been indicated for the "transport years" (01/10-30/09) 1987/88, 1988/89 and for the winter term of 1989/90.

The transport values measured indicate an absolute upper limit for the erosion-determined admission of sediment and phosphorus.

It has been found that erosion contributes to the transport of sediment and phosphorus in both water courses.

The erosion contribution was smallest from Rabis (loamy sand) and largest from Langvad (silt loams), but varies throughout the period due to varying precipitation. It was largest in 1987/88 and smallest in 1989/90.



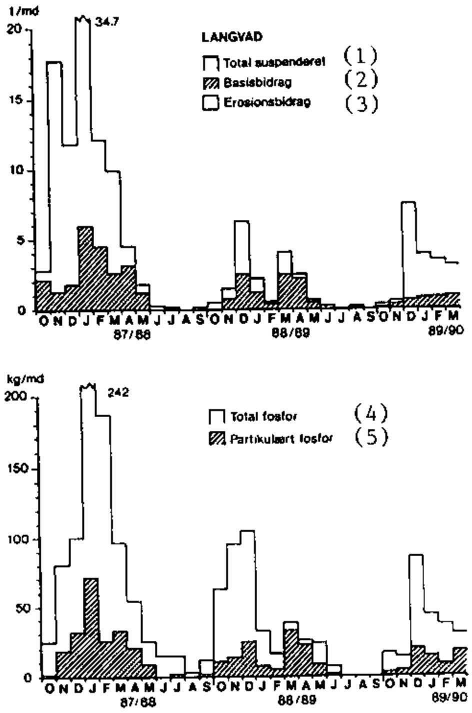


Fig. 1
Transport and
erosion.

Fig. 2
Transport and
erosion.

1. Total suspended load (t/month). 2. Part of transport not related to erosion events. 3. Transport related to erosion events. 4. Total phosphorus kg/month. 5. Particulate phosphorus kg/month.

Bank erosion contributes to the transport in both water courses. The relative share of the total transport was largest in 1988/89 and smallest in 1987/88.

Selective transport of fine-grained phosphorus sediment has been found.

Wind erosion only appeared to a limited extent during the measuring period, and the effect on the transport in the water courses is negligible.

Table 1

Transport amounts.

	Rabis			Langvad		
	87/88	88/89	89/90*	87/88	88/89	89/90*
Total susp. t	96	42	15	96	18	18
Total susp. t/km^2	9.9	4.3	1.6	8.2	1.5	1.6
Org. susp. t	25	20	9	27	6	5
Org. susp. t/km^2	2.5	2.0	0.9	2.3	0.5	0.4
Erosion % af total susp.	24	29	17	76	45	82
Basis % af total susp.	76	71	83	24	55	18
Total P kg	584	413	188	853	405	219
Total P kg/km^2	60	43	19	73	35	19
Part. P kg	250	160	49	214	119	63
Part. P kg/km^2	26	16	10	18	10	5

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*) Oktober-marts inkl.

3.2 Erosion and transport of phosphorus from plots

The run-off of surface water in mm/24 hours from the 4 plots during the measuring period is shown in Figs. 3 - 6. The measuring period was the hydrological years 1987/88, 1988/89 and the winter term of 1989/90 with insignificant thaw events.

During the measuring period the surface run-off was in general very small. It was shown that the surface run-off appears with great variations throughout the measuring period.

The largest run-off events took place from silt loam soils (Langvad). During the 3 largest run-off events from Langvad 1, 30% of one year's phosphorus contribution ran off.

It appears that the largest run-off event took place from silt loam soil with a 12% gradient (Langvad 1). From 2% gradient (Langvad 2) and grass-grown sandy soil with 4% gradient (Rabis 2) the run-off events were insignificant.

From contour-cultivated sandy soil with 4% gradient (Rabis 1) the run-off was smallest.

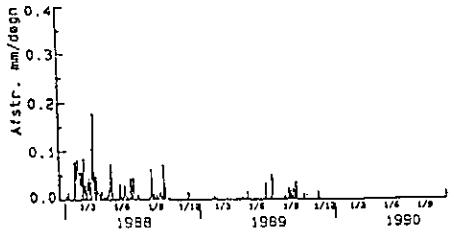

Fig. 3. Langvad 1

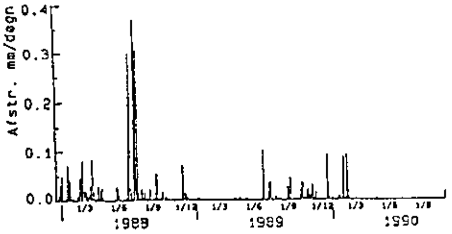

Fig. 4. Langvad 2

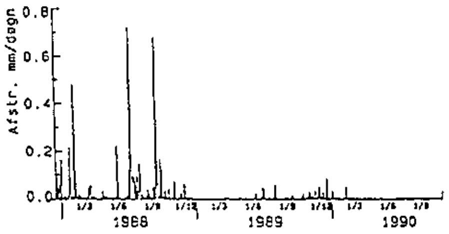

Fig. 5. Rabis 1

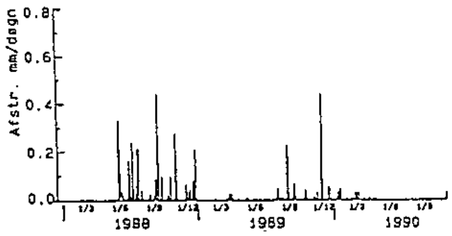

Fig. 6. Rabis 6

Figures 3, 4, 5 and 6. Surface run-off, mm/24 hours from 4 plots.

From the plots Rabis 1, Rabis 2, and Langvad 2 surface run-off will normally be of some importance only when the soil is frozen.

The total run-off was at its maximum during the hydrological year of 1987/88. Table 2 shows the accumulated losses of phosphorus in g/ha from each plot.

The calculation of losses of phosphorus indicates the maximum run-off losses from each plot through the experimental period.

After the larger run-off events selective transport of fine-grained sediment with large content of phosphorus was found.

Table 2. Loss of total phosphorus (g/ha) from each plot through the experimental period from autumn 1987 - spring 1990.

	Rabis 1	Rabis 2	Langvad 1	Langvad 2
1987/88	10.8	17.3	50.9	17.0
1988/89	3.1	4.1	17.2	26.5
1989/90	-	9.0	3.2	6.7

3.3 Evaluation of soil erosion in Denmark

3.3.1 Erosion in Denmark

The field measurements show that the erosion was insignificant in the experimental period from October 1987 to March 1990 inclusive.

Through the period diffuse surface run-off was found, but only in 1987/88 and 1989/90 rill erosion was found in the NPo areas.

In case of rill erosion, displacements of up to 2000 t/km^2 were observed in individual fields, and it was found that up to 14% of the material displaced reaches the water course.

A selective transport of phosphorus takes place in the way that the more fine-grained and phosphorus part of the erosion material is displaced further than the coarse-grained material, for which reason it more easily reaches the water course.

3.3.2 Evaluation of risk

An evaluation of risk of soil erosion in Denmark has been elaborated using USLE, the Universal Soil Loss Equation, modified for use of the Danish soil data base.

The evaluation shows that at least 3% of the area are potentially threatened with erosion, and at any rate those with a gradient above 6°. The threatened areas are particularly situated on silt loam soils in the young moraine formation, i.e. the eastern part of Denmark. Sandy soils in Western Jutland are not expected to be essentially exposed to water erosion.

Measurements show that erosion appears in areas with smaller gradient than 6°; there is thus a need for a possibility of differentiating areas with gradients between 0 - 6°.

Description has been made of a method for evaluation of the risk of admission of erosion material to water courses. It is, however, recommended to supplement with model calculations which include effect of freezing, melting and rill formation. The national survey can be used to point out those areas where model calculation can be carried out advantageously.

4. CONCLUSIONS

The transport of phosphorus from the basins was, due to the atypical mild winters, low in the period of investigations. 35-73 kg total P/km^2/year from Langvad and 43-60 kg total P/km^2/year from Rabis. The supply from wind erosion has during the period been at its minimum.

Types of soils and terrain are important factors. 6% slopes on loamy sand (Rabis) deliver less erosion than 12% slopes on silt loam (Langvad).

In comparison with other investigations intensive rain and mild winter climate give less erosion and transport of phosphorus than events of rainfall on frozen soils and thaw events.

The evaluation of the representative value of the 2 basins demonstrates that results from

Rabis are representative for areas in Denmark having the lowest rate of erosion, while a moderate rate of erosion is represented in the Langvad and Gelbrook basins.

The erosion measured from plots were relatively low but the same level of erosion was measured in Southern Sweden during 1987/88 (Ahlström & Bergman, 1988).

During the period the winter climate was very mild. A longer record of transport of total phosphorus at Gelbrook showed a transport of 234 kg P/km^2 in 1979, a cold and snowy winter. (Kronvang and Bruhn, 1990).

It is likely that this high value was due to erosion, but it is not possible to prove it because simultaneous observations of erosion were missing.

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Denitrification and other Nitrogen Cycling Processes in a Riparian Meadow

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1 Introduction

Riparian areas act as nutrient filters.

Nutrients in water from agricultural surface run off and drainage can be retained in riparian areas (Lowrance et al., 1983; Yates and Sheridan, 1983; Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985; Brüsch, 1990). Most of the work in riparian zones has focused on nitrogen removal by natural processes.

Denitrification removes nitrogen from ecosystems.

Bacterial reduction of NO_3^- to N_2, denitrification, is the most important process in nitrogen removal from ecosystems. Nitrate is used as the oxidizing agent when molecular oxygen becomes limited, e.g. in waterlogged soils. Alternatively, plant harvest could remove immobilized nitrogen from the system.

DNRA reduces nitrate to ammonium.

Dissimilatory nitrate reduction to ammonium (DNRA) is another anaerobic biological process reducing nitrate. The DNRA is favoured by strongly reduced conditions and will presumeably only take place when nitrate is introduced to strictly anaerobic environments. But knowledge about the DNRA process and its quantitative significance is very sparse.

Nitrate is also reduced during uptake by microorganisms and plants, but as in the case with DNRA, nitrogen is not removed from the ecosystem.

Drying/wetting cycles increases nitrogen removal.

Riparian areas are permanently or intermittently waterlogged. Drying/wetting cycles increases the nitrogen release compared to static conditions. During the dry aerobic periods mineralization and nitrification dominates and nitrate can accumulate in the soil. Flooding causes reduced conditions and the nitrogen will be removed through denitrification.

The purpose of this study was to

- quantify the natural denitrification in a riparian area and identify the limiting factors

- investigate how agricultural drainage affects the process and evaluate the effect on the amounts of nitrate reaching the stream

- determine other nitrogen cycling processes and assess the correlation between these processes and denitrification

2 Materials and methods

A riparian meadow irrigated by drain water.

The investigations were conducted in a 40 m wide meadow bordering on one side a small stream and on the other side agricultural land. The riparian vegetation was dominated by Carex species, and peat was found to at least 6 meter depth. Meadow and fields were seperated by a 2 meter high slope. At the lower edge of the slope drain pipes from the agricultural area fall into the meadow. Hydrophobic peat layers in 20 - 30 cm depths causes the drain water to move horisontally on the top or into the uppermost layer of the meadow.

Denitrification was measured in a control and an irrigated plot.

Denitrification was measured in a control plot and a plot irrigated with drain water from 6.8 ha agricultural land. The water was distributed evenly with a gutter over a 18 meter wide application front. When the experiments started in June 1988 the two plots were seperated by a ditch. This showed up to be insufficient and in January 1989 a sheet piling was put up to a depth of 50 cm confining the irrigated plot.

Denitrification was measured by the acetylene inhibition technique in undisturbed soil cores (Rice and Smith, 1982; Parkin et al. 1985) with 2 - 3 weeks intervals throughout 1.5 year. 15 cm soil cores were collected 1, 12, 23 and 34 meters from the stream. All sampling was done in triplicate. The cores were preincubated for 16 hours in a 10 % acetylene atmosphere to ensure an even distribution of acetylene in the soil volume.

Denitrifying enzyme activity (DEA) was measured in the 0-5 cm surface layer and in the top 5 cm of the saturated zone by the method of Smith and Tiedje (1979). Samples were collected in the same distances from the stream as described for the natural measurements, but only in the control plot. The measurements were performed with 3 month intervals throughout 1 year.

Soil slurries or intact cores were added chloramphenicol and amended with nitrate and glucose in four combinations. These incubations were performed several times throughout the study to evaluate substrate limitations.

Mineralization, immobilization, nitrification and nitrate reduction was determined by the isotope dilution technique (Koike and Hattori, 1978). ^{15}N labeled NO_3^- or NH_4^+ were added to undisturbed soil cores and after a 24 hours incubation period NO_3^- and NH_4^+ pool size and ^{15}N enrichment was measured. Gas samples were analyzed for ^{15}N.

3 Results and discussion

Soil properties. Soil pH, loss on ignition, and C and N content was 7.33, 52.7%, 26.8% and 2.0% respectively.

Gas exchange not affected by diffusional constraints.

Gas exchange between soil air and headspace during incubation. Acetylene concentrations in top and bottom of soil cores showed similar values after preincubation. Preliminary experiments showed a constant nitrous oxide accumulation rate in individual cores for at least 6 hours (R^2=0.94), indicating that gas exchange between soil core and headspace was not influenced by diffusional constraints. Furthermore, the amount of nitrous oxide dissolved in the soil core water, estimated by use of the Bunsen coefficient, approximated 90 % of the actual amount as measured by extraction of soil cores.

Vertical distribution of DEA shows a steep decline.

Sampling depth. In the 0-20 cm layer the DEA decreased sharply from 12.1 to 0.9 $\mu g\ N_2O\text{-}N\ g^{-1}\ h^{-1}$, beneath 20 cm depth the DEA was constant and low, about 0.16 $\mu g\ N_2O\text{-}N\ g^{-1}\ h^{-1}$ (Figure 3.1). 85 % of the activity in the 0-75 cm profile was located above 15 cm.

Thus, by sampling to 15 cm only a minor fraction of the activity will be lost. A similar steep vertical distribution of denitrifiyng enzymes was found in riparian forests in Georgia, USA, where 68 - 88 % of the enzymes in a 10 cm profile could be detected in the 0-2 cm surface soil (Ambus and Lowrance, 1990).

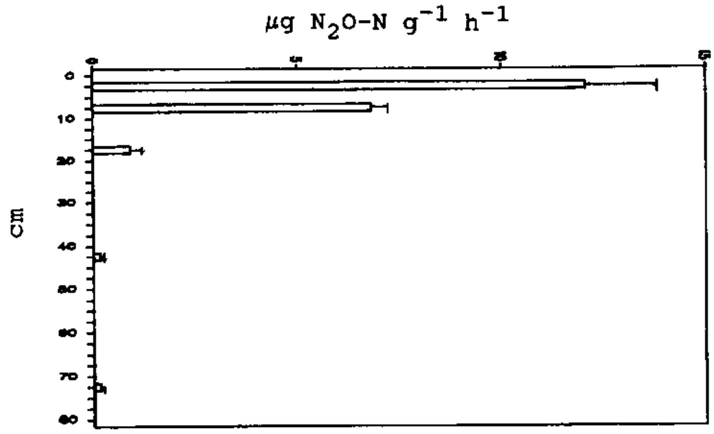

Figure 3.1
Denitrifying enzyme activity in a soil profile. Bars equal 1 standard deviation; n = 6.

Natural denitrification and effect of drainage.

Low constant denitrification in control plot and high variable activity in irrigated plot.

Denitrification activity in the control plot caused a nitrogen loss of about 3 mg N_2O-N m^{-2} day^{-1}. There was no spatial variability but slightly increased rates during autumn and winter (Figure 3.2 A). As opposed to this, denitrification in the irrigated plot showed a pronounced temporal a spatial variability. The activity was highest close to the drain outlet during autumn and winter (Figure 3.2 B).

High nitrate during summer, low during winter.

During summer and early autumn 1988 the nitrate concentration increased from 3-4 to 16-32 μg NO_3^--N g^{-1} peat. In October the concentration decreased to below 8 μg NO_3^--N g^{-1} peat and a relatively steady concentration throughout the winter indicated that nitrification and denitrifcation processes were balanced. The following summer nitrate increased again, but the concentration did not reach the same levels as during the preceding year. The meadow was partly or totally waterlogged during autumn, winter and early spring and more dry during the summer. There was no spatial variability in nitrate or water content within plots.

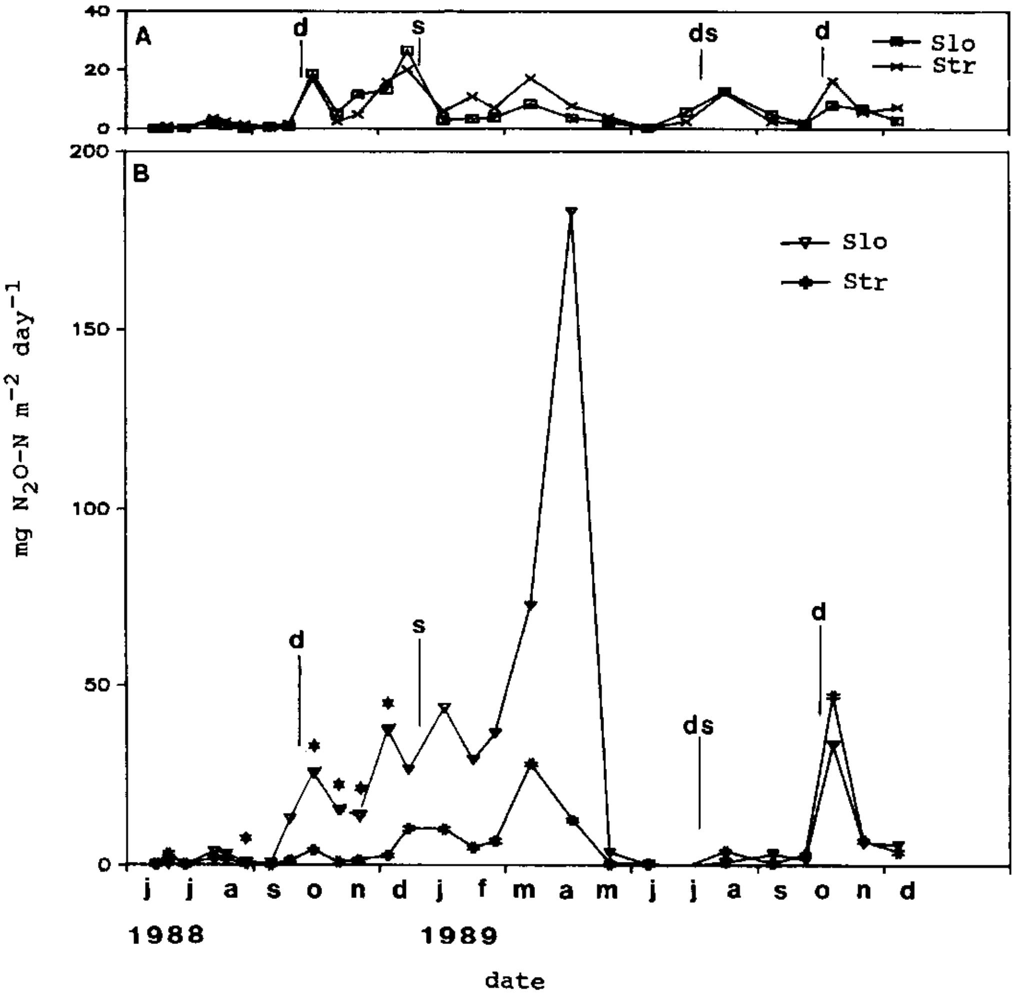

Figure 3.2

*Denitrification activity in control (A) - and irrigated (B) plots. Str = cores sampled in the stream-side (1 and 12 meters from the stream); Slo = cores sampled in the slope-side (23 and 34 meters from the stream). d = drainage starts; ds = drainage stops. s = sheet piling established. * = significant difference between denitrification in the stream-side and slope-side ($p < 0.05$). n = 6.*

Denitrification stimulated by drain water.

Drainage stimulated denitrification reaching the highest activity close to the outlet (Figure 3.2 B). Denitrification in the control area was not affected by drainage after an effective sheet piling was established.

No difference in soil water nitrate in the two plots.

Despite a daily supply of 492 g NO_3^--N in the drain water, porewater nitrate concentrations were similar in the two plots. The drain water ran on the soil surface and denitrification presumably reduced all the nitrate penetrating into the soil. Furthermore, nitrate was diluted by the relatively large volume of soil water compared to drain water.

Severe nitrate limitation in cores from control plot, more moderate in irrigated cores.

Nitrate addition increased denitrification in anaerobic cores from both plots (Table 3.1), but the highest response was in cores from the control area. The disappearing difference in denitrification between control and irrigated cores when nitrate was added clearly shows that the enhanced denitrification in irrigated soil was caused by a nitrate supply from the drain water. The results also show that denitrification capability in this type of meadow is very high.

Table 3.1
Denitrification in soil cores and slurries.
% wfp = 95-140. April 1989.

	ng N_2O-N g^{-1} h^{-1}		
	Irrigated	Control	
	anaerobic	anaerobic	aerobic
		Soil cores	
control	95 (110)*	17 (121)	14 (32)
+ N	462 (29)	489 (39)	528 (31)
+ N&C	832 (54)	717 (10)	877 (16)
		Slurries	
+ N&C	8100 (31)	7440 (10)	ND#
		Effect of treatment	
N/no	4.9	29.6	ND
N&C/N	1.8	1.5	ND
slur./core	9.7	10.4	ND

* means (% CV), n=4-16.
\# not determined

Carbon addition had only a minor effect, 1.5 - 1.8 times increase, which indicates that nitra-

Carbon had only a minor effect on denitrification in soil cores.
No oxygen inhibition.

te was the main regulator. Denitrification under aerobic conditions (atmospheric air) was similar to anaerobic activity which indicate, that oxygen did not inhibit denitrifying activity in this waterlogged soil. Diffusion of substrates inhibited denitrification in both soils since soil slurries denitrified 10 times faster than intact cores.

A multiple regression analysis on data from soil cores sampled close to the drain outlet during the two drainage periods explained 46 % of denitrifying variability ($p < 0.05$). Three parameters were included in the model,

$$y = 1.2656 \cdot N + 0.0136 \cdot V + 0.0014 \cdot NI - 0.8058,$$

	partial R^2
$y = \log_e$(denitrification rate)	
$N = \log_e$(nitrate concentration)	0.1664
V = % wfp	0.1048
NI = daily nitrate supply	0.1872

The daily nitrate supply was the strongest parameter explaining 19 %.

1 ha meadow can denitrify nitrate from 4.3 ha cultivated land.

If denitrifying activity in 1 ha irrigated meadow proceeded at a rate similar to the activity in the subarea close to the drain outlet, the nitrate in drain water from 4.3 ha cultivated land could be totally denitrified. Denitrification can presumably be even higher by increasing the retention time of the drain water or the meadow area in proportion to the water volume.

Denitrification in riparian forests differ.

During succession in riparian areas shrubs and trees will emerge. Denitrification in forested Sphagnum bogs as well as in danish Alder and Ash swamps was very low compared to the activity in two different meadows, even when the plots were substrate amended (Table 3.2). As opposed to this two forested ecosystems showed a denitrification in nitrate amended plots 3 - 44 times higher than the meadows, while nitrate reduction in a forest treated with sewage proceeded at a rate comparable to meadow denitrification (Table 3.2). Nitrate was the main regulator in all the areas.

Table 3.2
Denitrification in different riparian areas.

Type and locality	mg N m^{-2} day^{-1} a	b	c	Reference
Meadow, DK	16	441	835	this study
-	93	179	343	Ambus, 1990[@]
Alder stand, DK	0.5	7	13	Struwe and Kjøller, 1990
Ash stand, DK	<0.1	7	11	-
Forested Sphagnum bogs, USA	0.03	0.7		Urban et al., 1988
Forest, USA (loamy sand)	14	1260	1220	Ambus and Lowrance, 1990
- (fine loamy sand)	11	7931	11531	-
Forest, N.Z.			≈ 219[#]	Schipper et al., 1989

a unamended; b nitrate amended; c nitrate + glucose amended
nitrate reduced in a sewage irrigated forest.
@ unpublished.

Potential denitrification (DEA) and substrate limitations. DEA in the 0-5 cm surface varied between 5.2 and 17.9 µg N_2O-N g^{-1} h^{-1} (Figure 3.3) Spatial variability was negligible, while temporal variability was pronounced.

DEA variability explained 58 % by water.

The highest density of denitrifying enzymes was found when the soil was very wet or waterlogged and 58 % ($p < 0.01$) of the DEA variability could be explained by the soil water content.

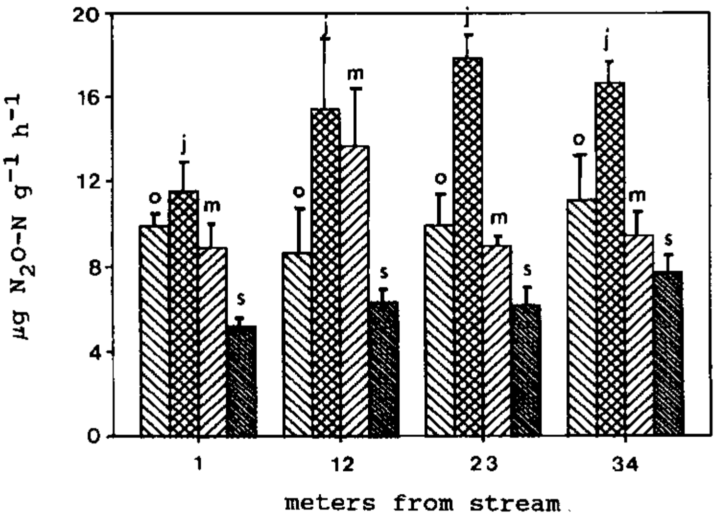

Figure 3.3
Potential denitrification at four distances from the stream in the control area. o = October, j = January, m = May and s = September. Bars equal 1 standard deviation; n = 3.

Available carbon limited enzyme activity.

Denitrifying activity in slurries increased 130 % on average if glucose was added, while nitrate increased the activity by 30 %. The results were independent of sampling site and date. Thus, available carbon was limiting denitrifying enzyme activity in this area. The nitrate limitation in cores, discussed above, was consequently rather caused by diffusional limitations than by a real nitrate lack in the environment.

Nitrate diffusion limited denitrification in intact cores.

In the saturated zone DEA was between 0.8 and 6.2 $\mu g\ N_2O\text{-}N\ g^{-1}\ h^{-1}$, on the average 5.1 times lower than at the surface. DEA in the deeper soil was always limited by available carbon, similar to the conditions at the surface.

Mineralization, immobilization, nitrification and nitrate reduction. ^{15}N was added to intact soil cores in November 1989. Soil water content was 107 % wfp. Nitrogen transformation rates were calculated for 0-5 cm and 5-10 cm soil depths (Table 3.3).

Positive net mineralization in surface soil.

Zero mineralization in deeper soil.

In the surface 0-5 cm a net mineralization produced 0.28 µg NH_4^+-N g^{-1} h^{-1} and nitrification approximately balanced the net mineralization. Nitrate reduction, 0.47 µg NO_3^--N g^{-1} h^{-1}, was twice as big as nitrification. In 5-10 cm net mineralization was zero and nitrate reduction 10 times faster than nitrification. Denitrification quantification failed, but ^{15}N-gas evolution could be detected.

Table 3.3
Nitrogen turnover in a riparian soil.

	Depth		µg N g^{-1} h^{-1}			Denitrif.
		Immobil.	Mineraliz.	Nitrific.	N-red.	
A	0-5 cm	0.86#	1.14	0.25	0.47	+*
	5-10 cm	0.42	0.42	0.05	0.58	
B	0-5 cm	ND¤	ND	0.66	0.40	+
	5-10 cm	ND	ND	0.15	0.46	

A $^{15}NH_4^+$ added for Immo and Mine measurements, $^{15}NO_3^-$ added for Nitr and N-red measurements.
B $^{15}NO_3^-$ + $^{14}NH_4^+$ added.
means of four cores, average CV = 54%.
* ^{15}N-gas in trace amounts.
¤ not determined.

Nitrification limited by substrate.

Nitrate assimilated at low ammonium.

The comparison of nitrate reduction and nitrification in experiment A is somewhat difficult since only $^{15}NO_3^-$ and not NH_4^+ was added. In experiment B where NH_4^+ - addition was included, nitrification increased and nitrate reduction decreased in both depths. This indicates that nitrification was limited by NH_4^+ which agrees with the findings in experiment A, where nitrification consumed all surplus ammonium. The decrease in nitrate reduction caused by ammonium reflects the microbial assimilation of nitrate.

The aerobic processes proceeded 1.5 - 5 times faster in the surface soil compared to the 5-10 cm depth, while nitrate reduction was largest in the deeper soil. This may be explained by a faciliated oxygen supply in the top soil compared to the deeper soil.

DNRA in deep soil only.

Assuming that nitrate assimilation was totally inhibited by ammonium addition, the average nitrate assimilation was 0.1 µg NO_3^--N g^{-1} h^{-1} while dissimilatory nitrate reduction (denitrification + DNRA) proceeded at a rate of 0.43 µg NO_3^--N g^{-1} h^{-1} on the average. In the top 5 cm DNRA did not take place since no ^{15}N ammonium was found after adding ^{15}N nitrate. But in the 5-10 cm depth DNRA proceeded resulting in 3.8 % of the recovered ^{15}N present in NH_4^+. This vertical distribution of DNRA agrees with the fact that DNRA is favoured by reduced conditions.

NO_3^--reduction and N_2O-production similar.

The similarity between nitrate reduction and N_2O-evolution rates measured in nitrate amended soil cores (Table 3.1), support the conclusion that denitrification is the prevalent nitrate reducing process (plant uptake not considered).

4 Conclusions

Denitrification in a riparian meadow was greatly enhanced by irrigation with nitrate rich drain water. At the conditions prevalent close to the drain outlet, denitrification in 1 ha meadow could totally reduce nitrate from 4.3 ha cultivated land, which is equivivalent to 30 mg NO_3^--N m^{-2} day^{-1}. Since denitrifying capability was much higher, a suitable handling of the drain water could probably increase the effectiveness. Low retention time and the area topography resulted in less than 5 % of the drain water nitrate being reduced in this study.

Denitrifying enzyme activity was highest during autumn and winter, and was always limited by available carbon. The increased denitrification following irrigation with nitrate rich drain water was therefore due to increased nitrate diffusion into peat layers where denitrifying conditions prevailed.

Under wet conditions nitrate consumption was limited by nitrification. Nitrate reduction processes listed in order of decreasing importance are denitrification, microbial assimilation, and DNRA in deeper soil. Plant uptake was not considered.

This study demonstrates, as found by others, that riparian areas act as nutrient filters. But to the authors opinion, establishment of riparian "buffer zones" should not be considered as the solution to nitrate pollution problems that might be better tackled at the source.

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Number C13 -b-

Water and Nutrien Balances for a Floded Riparian Wetland

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Introduction.

Several researchers (MacCrimmon, 1980, Peverly, 1982, Verry, 1982, Lowrance et al.,1983, Yeates & Sheridan, 1983, Jacobs & Gilliam, 1985, Brusch, 1990) have found that wetlands may act as nutrient sinks under natural conditions. Constructed wetlands (Gersberg et al., 1983, 1984) or management actions in natural wetlands (i.e. flooding/irrigation) have sometimes shown to be more effective, quantitatively, with respect to nutrient removal (Hoffmann, 1985).

In this study a riparian meadow was used to quantify the effiency of nitrogen removal when the meadow was irrigated/flooded with drain water from the neighbouring cultivated fields. Balance sheets for water and nutrients were drawn up. The flooded area of the meadow was compared to a non flooded area of the meadow with respect to concentration of nutrients in the rootzone. Ambus and Christensen (this issue) reported on denitrification and other nitrogen cycling processes from the same meadow.

Materials and methods.

The flooded part of the meadow was encircled by a sheet piling made of fibre glass (fig. 1), and going to a depth of 50 cm. Inflow of drain water and surface outflow was measured by use of electromagnetic flowmeters (Fischer & Porter). Rates of flow was logged every hour (Grant, Squirrel 1208) and totals continously registered. The inflowing drain water was distributed through a gutter.

Both meadow areas were equipped with pizometers to take soil water samples from two depths 0 - 15 cm and 15 - 30 cm. The piezometers were placed in four rows (seven in each row) in growing distance from the slope/inlet side (fig. 1 & 2).

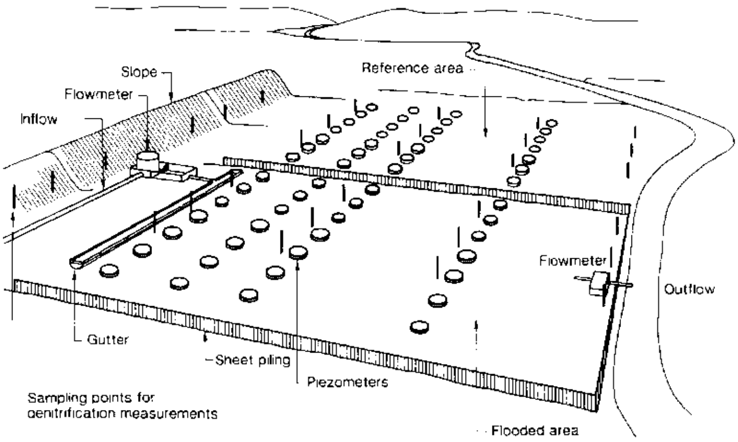

Figure 1.
Experimental design.

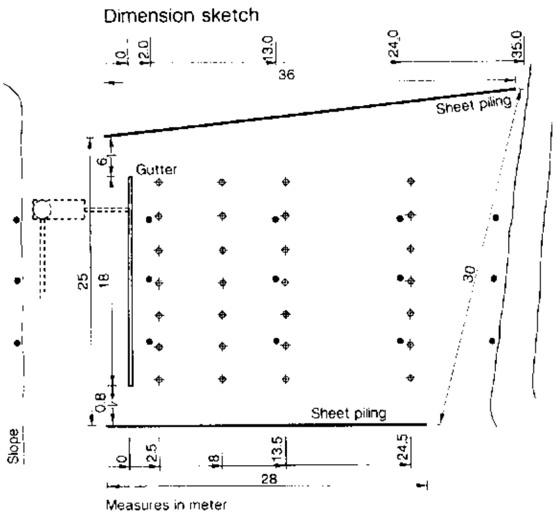

Figure 2.
Dimension sketch for the flooded meadow area.

All the paramaters in the water balance, equation shown below, were measured with the exception of leaching, which was calculated as the only unknown quantity.

Water balance eq.

LEACHING = (INFLOW + PRECIPITATION) - (OUTFLOW + EVAPOTRANSPIRATION)

Precipitation and evapotranspiration were measured on a nearby non flooded meadow area by Olsen (1990). Due to the flooded conditions the potential evapotranspiration was used in the calculation. This has shown to be a precise approach for a wet riparian meadow (Dahl 1990).

Water chemistry

Drain water and surface water (outflow) were analyzed for : NH4-N, NO3-N + NO2-N, total-N, PO4-P, total-P, SO4. Soil water was analyzed for the same elements except total- N & P.

Results.

Water balance for the flooded area.

Figure 3 shows the measured inflow and outflow rates and the calculated leaching. From a biological point of view the percolation of water through the rootzone (depending on infiltration capacity and hydraulic conductivity) is the most interesting, because removal of nutrients (immobilization, denitrification, sorption) takes place in the rootzone.

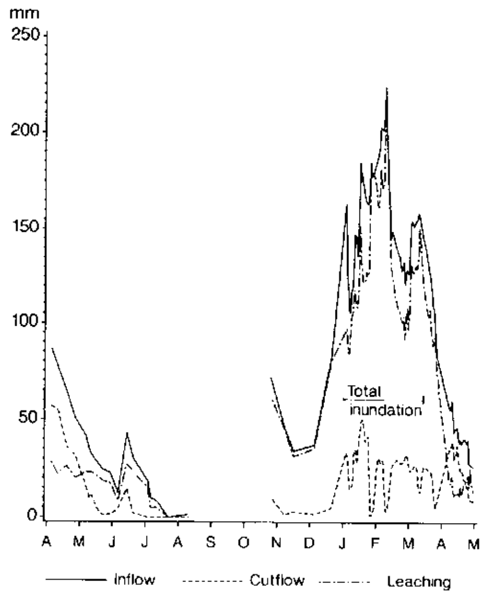

Figure 3.
Inflow, outflow and leaching for the flooded meadow, 1.4.89 - 30.4.90. From 18.12.89 to 13.3.90 leaching and outflow is incorrect, due to inundation of the river valley.
Unit : mm day^{-1}.

Leaching

The mean leaching in the spring/summer period 1989 was 19 l m^{-2} day^{-1}. There was no drain water flow from 23.7.89 to 23.10.89, except for two days in august. Surface outflow stopped about 1.7.89 and restarted 23.10.89. In the autumn 1989 the mean leaching was 50 l m^{-2} day^{-1}. In the spring 1990 the mean leaching was 47 l m^{-2} day^{-1}.

Investigation periods

The calculated water balance (table 1) includes the periods : 1.4.89 - 11.7.89, 24.10.89 - 21.12.89 and 13.3.90 - 30.4.89. From 22.12.89 to 3.1.90 the whole river valley was totally inundated due to heavy rainfall and no measurements were possible. From 4.1.90 to 13.3.90 the river valley was still inundated and the water balance for the flooded area could not be correct calculated, because run off of river water

Table 1.
The water balance for the flooded area.
Unit : m^3 water$\cdot$$ha^{-1}$

WATER	SEASON			TOTAL
	SPRING 89	AUTUMN 89	SPRING 90	
DAYS	102	58	48	208
INFLOW	37744	30457	33762	101964
OUTFLOW	15559	1743	10570	27872
LEACHING	19820	29279	22731	71830
PRECIP.	970	748	489	2207
EVATRANS	3336	183	950	4469

took place across the whole river valley, leaving the sheet piling partly out of function.

Concentration of nutrients.

Figure 4 shows the concentration of nutrients in the drain water (inflow) and the surface outflow, measured from january 89 to april 90. With a few exceptions ammonia-N and phosphate-P do not change much during the passage of the meadow. Nitrate-N and total-N are always lower in the outflow, most pronounced in the spring - summer periods. Sulphate shows the same pattern, but less distinct.

Soil water concentrations

In figures 5 and 6 the soil water concentrations are shown for both the reference area and the flooded area of the meadow. In the flooded area the soil water nitrate-N concentrations are mostly in the range 0 - 2 mg NO_3-N l^{-1}, and never exceed 4 mg NO_3-N l^{-1} (fig. 5, B). This is always markedly lower than the inflow concentrations, that varies between 4 - 9 mg NO_3-N l^{-1} (fig. 4, C). For both areas soil water ammonia-N is about 0.1 mg NH_4-N l^{-1} (fig. 5, C & D).
In the spring periods 89 & 90 the soil water sulphate concentration in the flooded area (fig. 6, D) is lower than in the inflow water (fig. 4, F) indicating that sulphate is reduced in the rootzone.

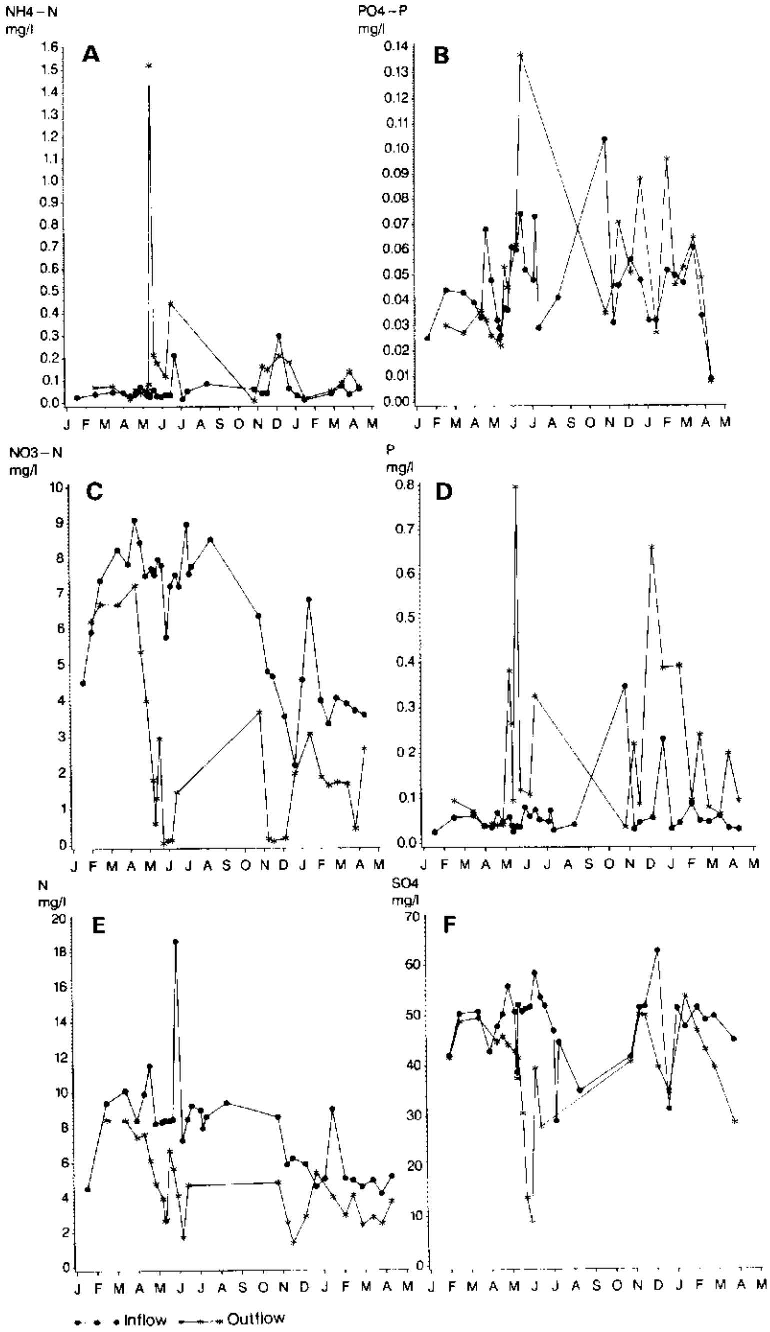

Figure 4.
Concentration of nutrients in inflow (drain water) and surface outflow from january 89 to april 90. A : Ammonia-N, B : Phosphate-P, c: nitrate-N, D : Total-N, E : total-P and F : Sulphate.

The concentrations of phosphate-P in the soil water in both areas are low, fluctuating around 0.05 mg PO_4-P l^{-1} (fig. 6, A & B).

The mean concentration of organic-N in the drain water (not shown) for the periods was 1.5, 1.9 and 1.0 mg N l^{-1}, while the corresponding outflow concentrations were 1.0, 1.7 and 1.2 mg N l^{-1}.

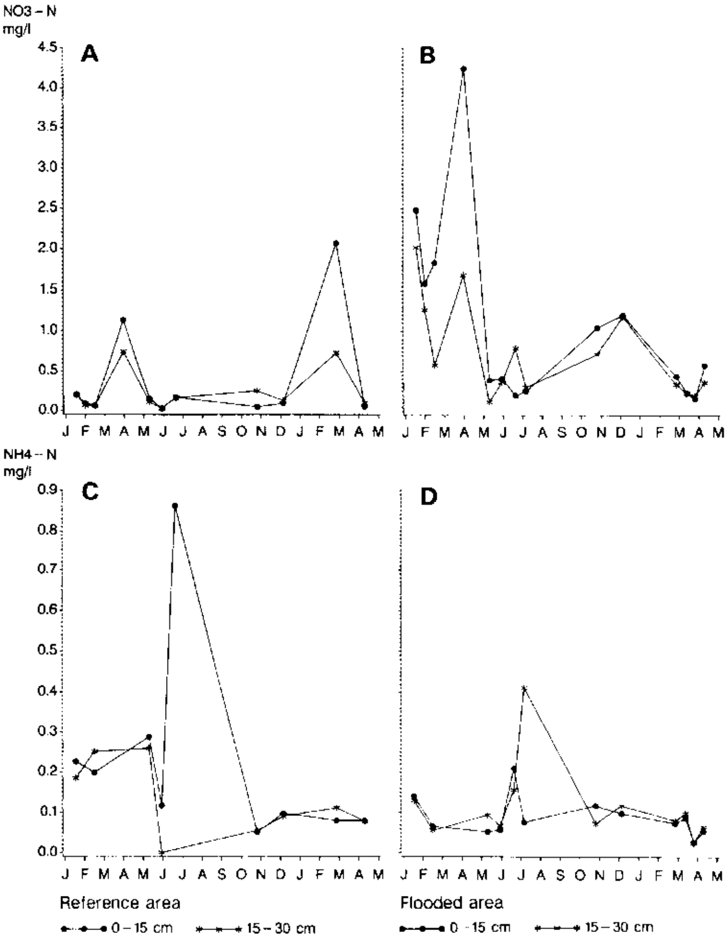

Figure 5.
The soil water concentrations of nitrate-N and ammonia-N in the reference and in the flooded area of the meadow.

In most cases the the total-P concentration i the drainwater was identical to the phosphate-P concentration. In the outflow water the organic-P (not shown) fraction was much more pronounced, about 50% of the total-P.

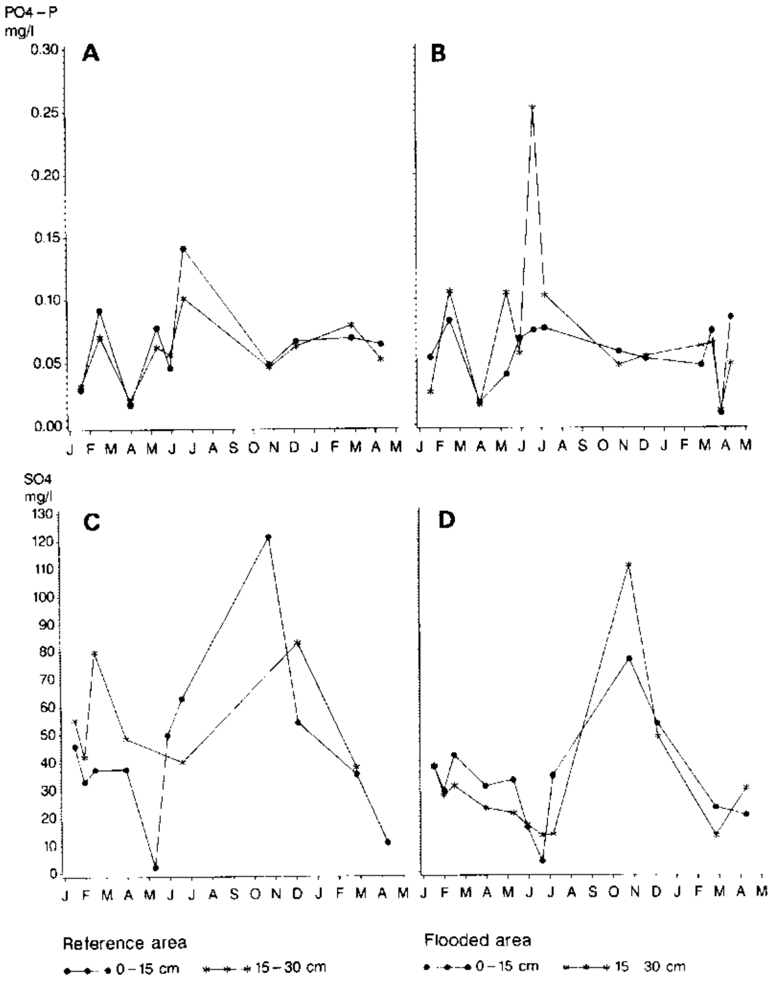

Figure 6.
The soil water concentrations of phosphate-P and sulphatein the reference and in the flooded area of the meadow.

Nutrient balances.

The balances are calculated on the basis of the water balance. The concentration of nutrients in the depth 15 - 30 cm are used in the leaching calculations. The results are made up for three periods : 1.4.89 - 11.7.89 (called spring 89), 24.10.89 - 21.12.89 (called autumn 89) and 13.3.90 - 30.4.90 (called spring 90). In between the periods drain water was not running or the river valley was totally inundated.

High nitrate removal

Table 2 shows the nitrate-N balance. The removal of nitrate in the three periods is high, 69, 71 and 77 %, respectively. This gives a mean dayly removal rate of 208, 124 and 198 mg NO_3-N d^{-1} m^{-2}.

Table 2.
Nitrate-N balance for the flooded meadow.
Unit : kg NO_3-N ha^{-1}.

NITRATE-N	SEASON			TOTAL KG/HA
	SPRING 89	AUTUMN 89	SPRING 90	
DAYS NO.	102	58	48	208
INFLOW	299	105	123	526
OUTFLOW	74	2	22	98
LEACHING	13	30	6	50
RETENT.	212	72	95	379

The ammonia-N balance, table 3, shows a minor outwash of ammonia. As shown in figure 4, A and figure 5, D, the soil water concentration of ammonia-N is 0.05 mg higher than the surface water concentration.

Table 3.
Ammonia-N balance for the flooded meadow.
Unit : kg NH_4-N ha^{-1}.

AMMONIA-N	SEASON			TOTAL KG/HA
	SPRING 89	AUTUMN 89	SPRING 90	
INFLOW	2	3	2	7
OUTFLOW	2	0	1	3
LEACHING	3	3	1	7
RETENT.	-2	0	-0	-3

Total-N balance

By the calculation of the total nitrogen balance, table 4, the total amount of inorganic-N is used to calculate the leaching of total-N, because organic-N was not analyzed in the soil water. The removal of total nitrogen in per cent during the three periods was 70, 76 and 73 %, respectively.

Table 4.
Total nitrogen balance for the flooded meadow.
Unit : kg N ha^{-1}.

TOTAL NITROGEN	SEASON			TOTAL KG/HA
	SPRING 89	AUTUMN 89	SPRING 90	
INFLOW	357	167	159	683
OUTFLOW	91	7	36	135
LEACHING	16	33	7	56
RETENT.	249	127	116	492

The phosphate-P balance, table 5, shows that input balances output.

Table 5.
the phosphate-P balance for the flooded meadow.
Unit : kg PO_4-P ha^{-1}.

PHOSPHAT P	SEASON			TOTAL KG/HA
	SPRING 89	AUTUMN 89	SPRING 90	
INFLOW	2	1	1	4
OUTFLOW	1	0	0	1
LEACHING	2	2	1	4
RETENT.	-1	-0	-0	-1

Total-P balance

The total phosphorus balance, table 6, is calculated in the same way as the total nitrogen balance, which means that leaching is based on the phosphate-P concentrations. During the autumn period the largest load of phosphorus is seen and about 50 % is retained.

Table 6.
The total phosphorus balance for the flooded meadow. Unit : kg P ha^{-1}.

TOTAL PHOSPHOR	SEASON			TOTAL KG/HA
	SPRING 89	AUTUMN 89	SPRING 90	
INFLOW	2	4	1	7
OUTFLOW	2	1	1	4
LEACHING	2	2	1	4
RETENT.	-2	2	-1	-1

There is net retention of sulphate in the spring 89 and 90, but in the autumn 89 the outwash of sulphate is pronounced (table 7). However, soil water sulphate was only analyzed a few times during the autumn 89 and the spring 90, making the balance somewhat uncertain.

Table 7.
the sulphate balance for the flooded meadow. Unit : kg SO_4 ha^{-1}.

SULPHATE	SEASON			TOTAL KG/HA
	SPRING 89	AUTUMN 89	SPRING 90	
INFLOW4	1873	1369	1519	4762
OUTFLOW	670	70	301	1042
LEACHING	368	2017	570	2955
RETENT.	835	-718	648	765

The atmospheric deposition is not included in all the above mentioned balances. Atmospheric

Atmospheric deposition

deposition was measured on a nearby meadow area, but stopped in the summer 89 (Grundahl & Hansen, 1990). From 1.6.88 to 1.6.89 the atmospheric deposition was : 7.0 kg NH_4-N ha^{-1}, 5.3 kg NO_3-N ha^{-1}, 9.7 kg SO_4-S ha^{-1}, while PO_4-P with some uncertainty was 0.08 kg PO_4-P ha^{-1} (Grundahl pers. comm.). The above mentioned figures for the atmospheric deposition leads to the conclusion, that only the ammonia balance is significantly changed, resulting in a net retention of ammonia of about 4 kg NH_4-N ha^{-1} y^{-1}

Discussion.

In previous investigations with irrigation/-flooding of wetlands with drain water the nitrate reduction has been of the same order of magnitude. Hoffmann (1985) found a net retention of nitrate, 300 kg NO_3-N ha^{-1}, at a hydraulic load of 46 l m^{-2} d^{-1}, in a lacustrine reedswamp at lake Glumsø, Zealand, Denmark. By increasing the hydraulic load to 57 l m^{-2} d^{-1} in the same reedswamp the retention of nitrate rose to 811 kg NO_3-N ha^{-1} y^{-1} (Jørgensen et al., 1990). In the river Stevns valley a riparian meadow was irrigated with 72 l m^{-2} d^{-1} of river water, equivalent to 0.3 g NO_3-N m^{-2} d^{-1}. The net retention was 1000 kg NO_3-N ha^{-1} y^{-1} (Dahl et al., 1990). In a riparian meadow, which was flooded in a natural way by upwelling ground water, Brusch and Nilsson (1990) found a net retention of nitrate, 390 kg NO_3-N ha^{-1} y^{-1}.

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Nitrient balance on Peat Soils

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1. Introduction

Background

Till now leaching from danish wet soils (Gley- and fluvisols) and deep peat soils (histosols), has only been sparsely investigated. Additional knowledge of this subject is essential in connection with a summarizing evaluation of the agricultural contribution to the NPo-strain.

Leaching from peat soils

The leaching from undrained peat soils left out of intensive cultivation is normally of minor importance. However, cultivation of peat soils often requires draining, leading to an increased mineralization of organic matter. This further leads to an increased mobilization of nutrients and an increased potential risk of leaching. In eutrofic peat soils with high pH, high CEC and low C/N-ratio the rate of mineralization will be relatively high, whereas oligotrofic peat soils have low pH-value, low CEC and higher C/N-ratio (Vedby, 1984).

2. Materials and methods

Localities in the research

Three areas, all drained 1-2 years before the start of the investigations plus two undrained neighbouring areas, serving as reference areas, have been investigated. For all three drained areas the drain water is collected in a well and pumped to a ditch. Table 1 displays a survey of the investigated areas.

Table 1. Survey of the investigated areas.

Locality	Year of draining		Area applicat.
Skovs-bjerggård	drained area	1986	rot. of crops
	reference	-	permanent grass
Volsted	drained area	1985	rot. of crops
		-	
Gøderup	drained area	1986	rot. of crops
	reference	-	grass & shrubs

Peat extension

At Skovsbjerggård (including reference area) and Volsted (all three situated in Jutland) the peat extend to a depth of at least 3-4 meters, at Gøderup (situated on Zealand) the peat extends to a depth of about 0,4 meters in both drained and reference area with clayey sand underneath. The cultivated areas have been supplied with normal amounts of fertilizer.

Soil water sampling

There has been used two different methods for sampling of soil water from the unsaturated zone above the groundwater level. At Gøderup water was sampled by use of porous cups (teflon), at Skovsbjerggård and Volsted soil samples were collected, and soil water was then extracted in the laboratory.

Groundwater sampling

The ground water samples were collected 0 - 20 cm below the groundwater level, at Gøderup only 0 - 10 cm below . This was done in order to secure, that it was the latest formed ground water, which was sampled. Furthermore it was a means to try to ensure that the ground water did not originate from neighbouring areas.

Number of samples

Sample depth

Soil and ground water were taken from 16 different points in the area in question and bulked before analysis. At Skovsbjerggård and Volsted soil samples were taken at four depths 0 - 25, 25 - 50, 50 - 75 and 75 - 100 cm in the drained areas, but only from two depths, 0 - 25 and 25 - 50 cm in the reference area due to the ground water level. At Gøderup soil water was sampled from 30 cm and 100 cm depth in the drained area, and from 30 cm depth in the reference area.

Water chemistry

Drain, ground and soil water were analyzed for NH4-N, NO3-N + NO2-N, PO4-P, total-N, total-P, Ca, Mg, K, Na, Cl, SO4, TOC/COD, total-Fe and pH and conductivity were measured.

The drain water run off was continously measured by use of flowmeters. Water samples were taken from the collecting wells.

Climatic data

Climatic data for use in the calculations of the water balance were acquired from Danish Meteorological Institute (Skovsbjerggård and Volsted) and from Olesen (1990, locality Gøderup).

Leaching computed

Leaching was calculated using the computer model simulation program, EVACROP, (Olesen and Heidmann, 1990).

Outwash of nutrients from the root zone was calculated by multipliing the computed leaching with the measured concentrations of nutrients in the soil water collected from the bottom sampling depth (100 cm depth in the drained areas and 25 - 50 cm depth in reference areas) in the unsaturated zone.

3. Results

3.1 Water balance.

Waterbalances

The calculated waterbalances of the investigated areas are displayed in table 2.

The actual evapotranspiration has been smaller and the leach larger on the drained areas than on the reference areas. In periods the drained areas were not cropped and the evaporation thus reduced. Due to a high groundwater level there may however in dry periods have been a capillary water rising which has not been taken into account in the calculations. This may indicate that the actual evapotranspiration is underestimated and therefore the leaching may have started earlier in the autumn than computed by EVACROP.

Table 2. Precipitation, potential and actual evapotranspiration, calculated leaching and measured drain water run off.

Locality	Year	Precipitation		Evapotranspiration				Leaching		Drain water	
				Potential		Actual					
		88/89	89/90	88/89	89/90	88/89	89/90	88/89	89/90	88/89	89/90
Skovsbjerggård	drained area	617	635	536	589	437	502	185	117	356	298
	reference	617	635	536	589	484	514	132	138		
Volsted	drained area	617	635	536	589	478	493	123	132	845	860
Gøderup	drained area	615	626	585	412	445	234	165	293	386	915
	reference	615	626	585	412	499	337	130	182		

Influx of ground water

The large drain water run off from the drained areas indicates that there has been a considerable influx of ground water from the neighbouring areas. At Skovsbjerggård, Volsted and Gøderup respectively only about 45%, 15% and 32% of the drainage water originates from the area itself. The rest is groundwater coming in from neighbouring areas, either laterally or vertically as artesian water.

Drain water

There has been drain water run off throughout the year from all three drained areas. According to the water balance calculations, there has only been leaching from the root zone during the wintertime. This shows, as mentioned above, that there is a considerable ground water influx to the drained areas (table 2).

3.2 Concentration of nutrients in soil, ground and drain water.

Table 3 shows the mean concentrations of nutrients in soil water, ground water and drain water. The annual fluctuations in nutrient concentrations in soil water in different depths of the unsaturated zone are shown together with concentrations of nutrients in ground and drain water for the drained area Skovsbjerggård in Jutland (fig 1) and the reference area belonging to it (fig 2). Figure 3 and 4 show the same for the drained and the reference area at Gøderup (Zealand).

High N- & P-concentrations

As shown in table 3 and in the figures 1 - 4 there has been found very high concentrations of all N- and P-fractions in the soil water. In most cases the highest concentrations have been found in the drained areas. The highest NH4-N concentrations in the soil water were however

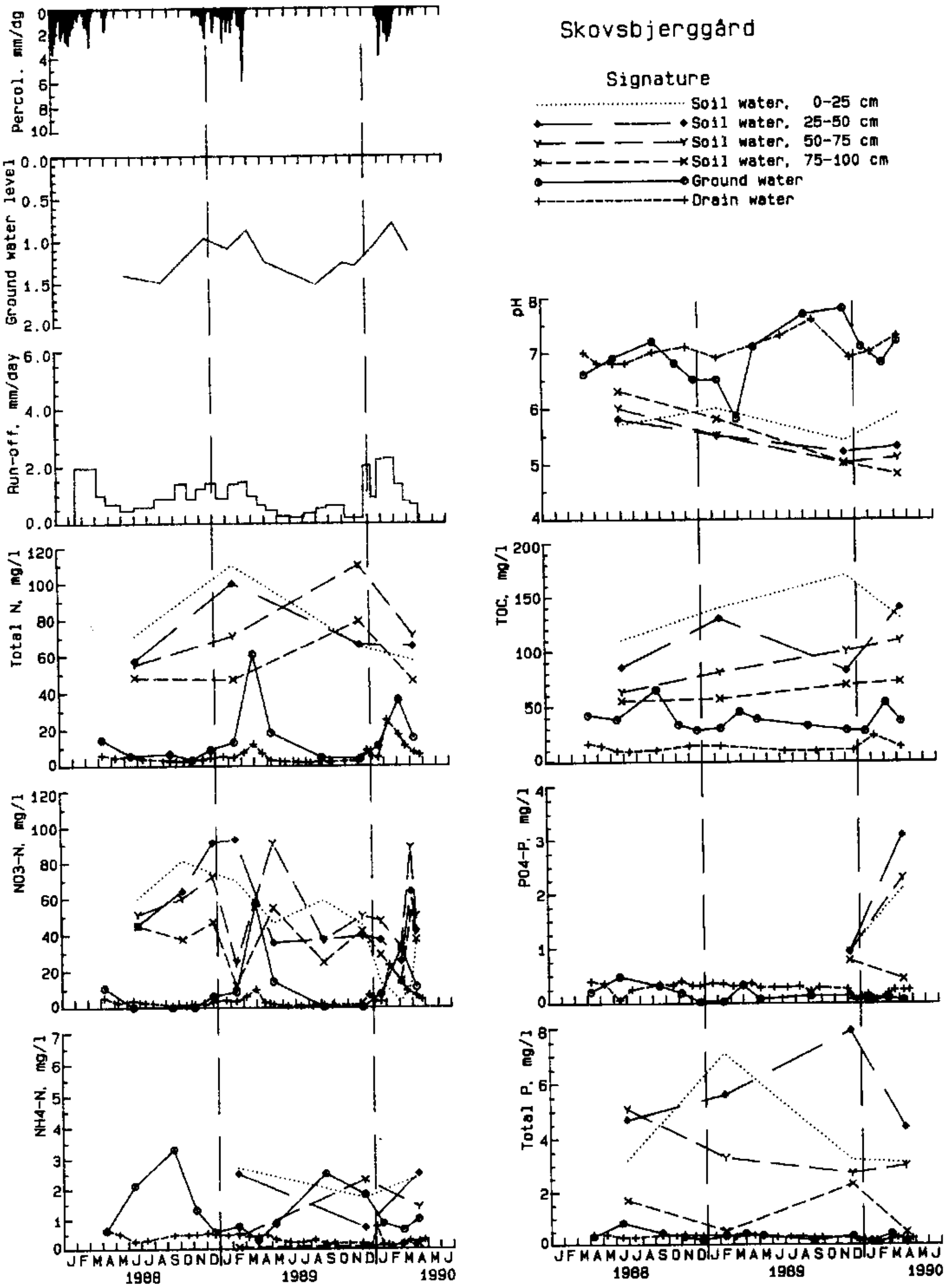

Fig.1. Drain water run off, leaching from the root zone and annual fluctuations of nutrients at Skovsbjerggård.

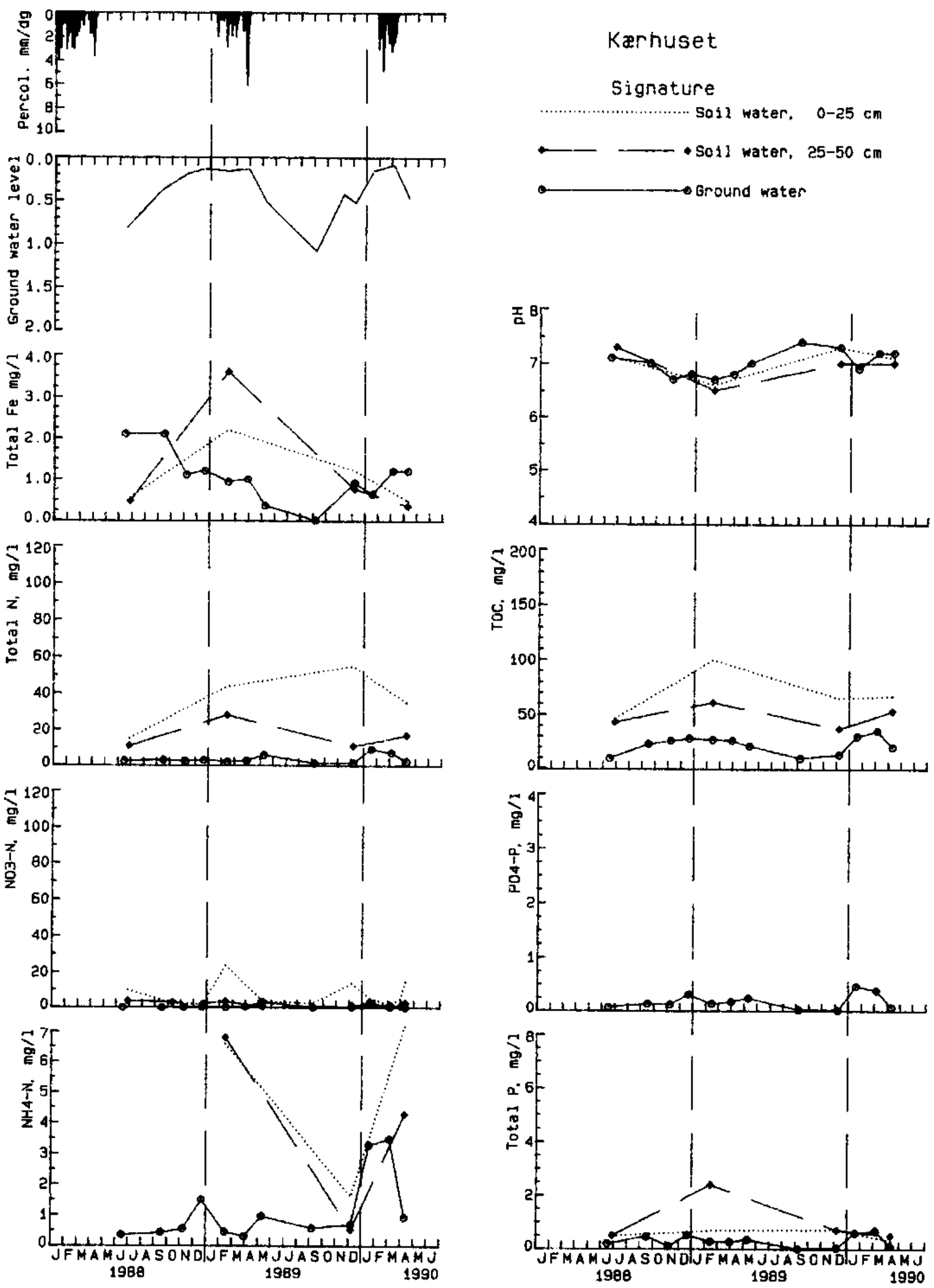

Fig.2. Leaching from the root zone and annual fluctuations of nutrients at the reference area at Skovbjerggård, named Kærhuset.

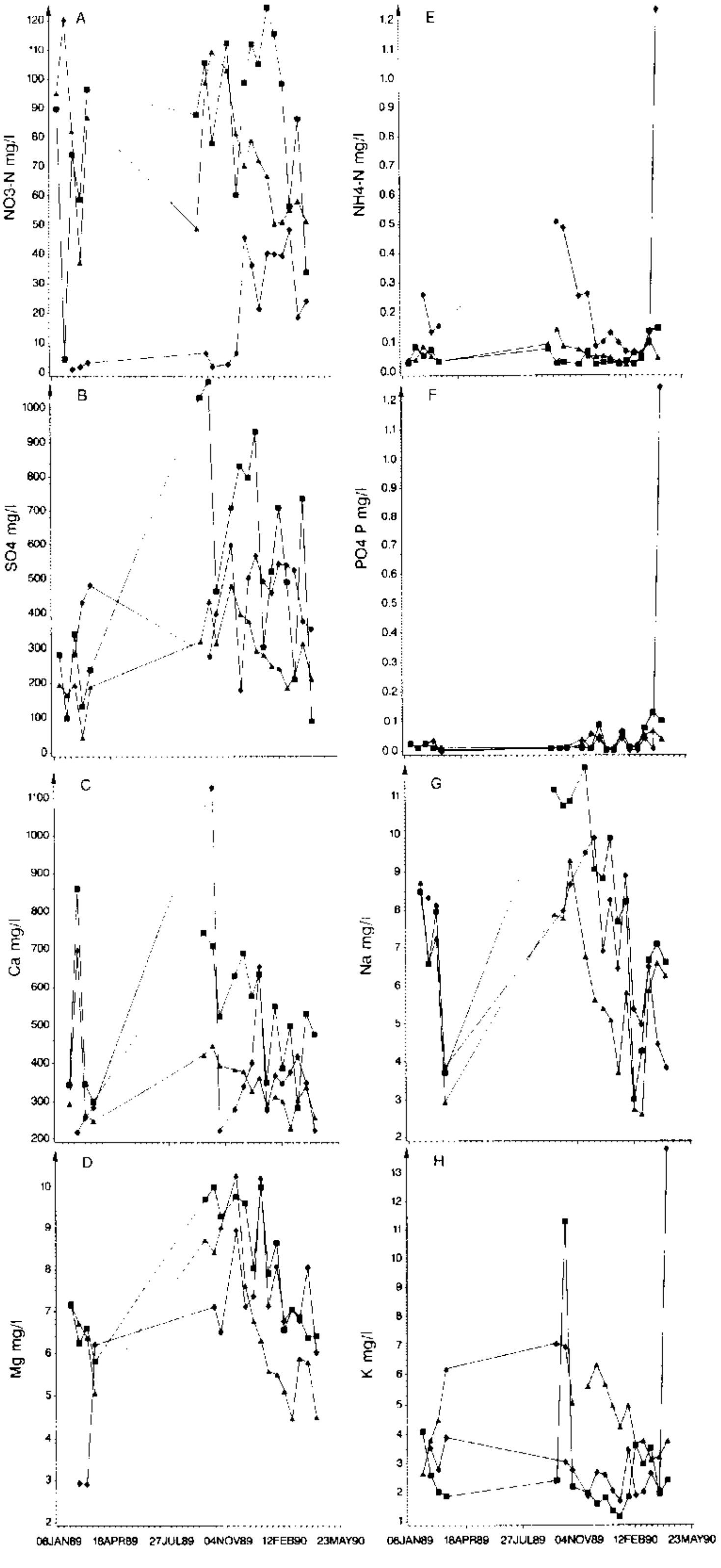

Fig. 3
Conc. of nutrients in soil and ground water in the drained area.

30 cm : ▲
100 cm : ■
200 cm : ◆

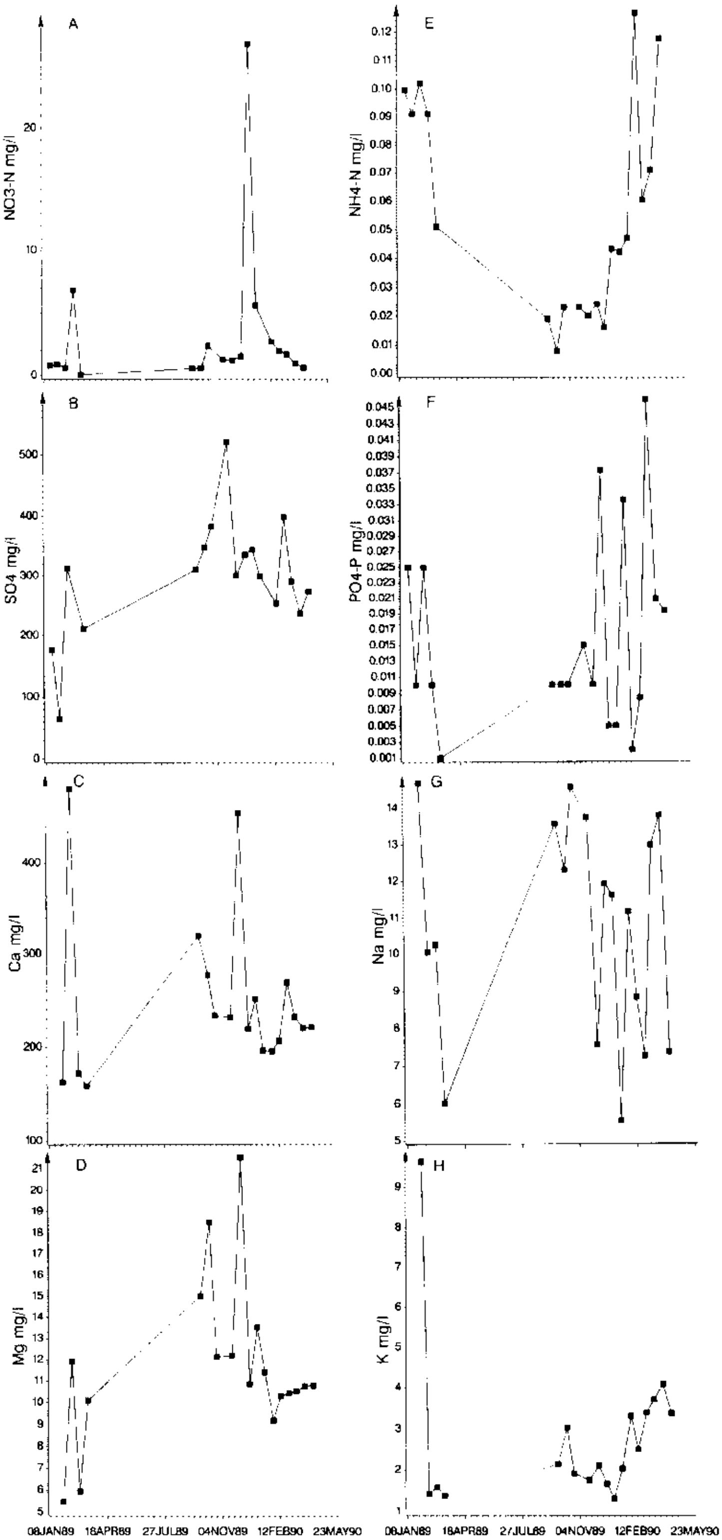

Fig. 4
Conc. of nutrients in ground water, 30 - 35 cm depth, in the reference area.

30 cm : ▲
100 cm : ■
200 cm : ◆

Table 3. Mean conc. of nutrients in soil water, ground water and drain water from all areas.

Depth cm	pH	TOC	NH_4-N	NO_3-N	Tot-N	PO_4-P mg/l	Tot-P	Ca	Mg	K	Na	Cl	SO_4	Fe
<u>Skovsbjerggård drained area</u>														
Soil water 0-25 cm	5.8	138	2.3	45.	75.	1.5	4.2	138	9	17	21	50	234	1.7
Soil water 25-50 cm	5.5	109	1.9	52.	72.	2.0	5.7	218	13	26	21	52	100	3.1
Soil water 50-75 cm	5.4	89	1.4	55.	77.	1.6	3.5	218	12	14	21	53	73	1.4
Soil water 75-100 cm	5.5	63	1.0	37.	55.	0.6	1.3	300	15	15	23	45	860	1.2
Ground water	6.9	38	1.3	11.3	15.5	0.15	0.33	310	26	10	21	35	473	2.0
Drain water	7.0	12	0.4	5.8	7.3	0.29	0.32	181	17	5	21	24	261	0.2
<u>Skovsbjerggård, reference area</u>														
Soil water 0-25 cm	7.0	70	5.1	8.8	37.	0.17	0.55	129	6	6	24	30	192	1.1
Soil water 25-50 cm	7.0	49	3.9	2.2	17.	0.30	1.03	117	7	5	23	38	239	1.3
Ground water	7.0	22	1.1	0.8	3.7	0.18	0.31	146	12	6	22	34	145	0.9
<u>Volsted, drained area</u>														
Soil water 0-25 cm	6.1	84	4.0	53.	87.	2.37	4.18	152	7	27	23	65	59	0.8
Soil water 25-50 cm	5.6	106	2.3	46.	69.	1.21	2.18	177	9	17	18	47	120	1.1
Soil water 50-75 cm	5.7	72	1.7	39.	50.	0.30	0.65	178	10	9	14	40	208	1.1
Soil water 75-100 cm	6.0	54	1.8	20.	24.	0.13	0.38	173	12	7	16	33	340	0.8
Ground water	6.9	45	7.5	8.7	17.3	0.43	0.68	159	18	6	17	28	191	1.2
Drain water	6.9	16	1.3	6.2	8.7	0.28	0.66	115	11	4	16	26	90	0.4
<u>Gøderup, drained area</u>														
Soil water 35 cm	7.2	.	0.07	71.	.	0.03	.	362	6	4	6	30	254	0.1
Soil water 100 cm	7.1	.	0.05	86.	.	0.04	.	546	8	3	8	36	481	0.1
Ground water	7.1	.	0.30	26.	28.	0.17	0.25	339	7	4	7	31	434	0.7
Drain water	7.0	.	0.11	14.	17.	0.02	0.10	284	7	3	8	31	366	0.5
<u>Gøderup, reference area</u>														
Ground water 35 cm	7.2	.	0.07	1.6	.	0.02	.	261	11	3	11	32	292	0.1

found in the reference area at Skovsbjerggård, where upto 7 mg NH4-N /l were measured (fig. 2, NH4). The concentrations in the ground water and in the drain water are mostly considerable lower than in the soil water. The ground water NH4-N concentration were however considerable higher than normally found for mineral soils.

Nitrate in groundwater

In periods without leaching the NO3-N concentrations in the groundwater at all areas were low, often less than 10 mg NO3-N/l. During periods with leaching the groundwater concentrations of NO3-N increased considerably, especially on the the drained areas.

TOC

Large concentrations of disssolved organic matter have been mesured, particularly in the soil water on the drained area at Skovsbjerggård (fig. 1, TOC). The concentrations decrease with depth.

pH drop

At Skovsbjerggård and Volsted the soil water pH in the deepest part of the root zone decreased considerably throug the period (fig. 1, pH).

3.3 Leaching of nutrients from the root zone and outflux of nutrients with drain water.

Table 4 shows the calculated leaching of water and nutrients from the root zone, the drain water run off together with the outflux of nutrients. At Skovsbjerggård and Volsted, with the exception of NO3-N, the leaching is calculated on the basis of only a few analysis, making the calculated values somewhat unsertain.

Drained areas show higher NO3 leaching

The calculations indicate however, that the leaching of Ca, Mg, SO4 and in particular NO3-N have been higher in the drained areas than in reference areas. Despite the large influx of ground water from the neighbouring areas, the amount of nitrogen in the drainage water have in several cases been lower than the leaching from the root zone of the drained areas. Even though the amount of nitrogen in the incoming ground water may be low, it indicates that besides dilution the turnover of N below the root zone at the drained areas may have some importance. On the other hand elevated nutrient concentrations in the ground water compared to soil water concentrations may originate from the influxed ground water.

Tabel 4. Yearly leaching of water and nutrients from the rootzone (1m depth) and runoff of water and nutrients from drainage pipes.

Area	Year	mm	TOC	NH_4-N	NO_3-N	Tot-N	PO_4-P	Tot-P	Ca	Mg	K	Na	Cl	SO_4	Fe
			<— kg/ha —>												
LEACHING FROM ROOTZONE															
Volsted	88/89	134	84	3.8	36	43	0.24	0 47	200	15	11	16	38	410	0.5
	89/90	147	68	2.1	42	50	0.16	0.55	300	23	6	15	39	625	1.5
Skovs-	88/89	185	98	1.0	51	84	1.3	1.2	550	24	40	44	76	910	4.2
bjerggd	89/90	117	83	1.5	23	68	0.65	1.3	410	20	12	25	51	810	1.0
Refe-	88/89	132	78	8.4	4	35	0.40	3.0	121	6	8	25	48	280	4.4
rence	89/90	138	64	3.8	2	20	0.41	0.8	194	12	4	30	57	420	0.8
Gøderup	89/90	293	.	0.09	289	.	0.10	.	1521	24	9	23	123	1824	0.2
Refe- rence	89/90	182	.	0.10	8	.	0.04	.	431	19	5	17	76	566	0.07
DRAIN WATER RUNOFF															
Volsted	88/89	845	136	15.0	26	45	2.5	7.0	860	84	36	128	200	573	2.9
	89/90	860	142	7.9	81	111	2.4	4.3	1107	99	37	140	248	957	3.5
Skovs-	88/89	356	40	1.5	15	21	1.1	1.2	514	50	15	70	85	738	0.4
bjerggd	89/90	298	38	1.0	23	27	0.8	0.9	673	59	19	65	90	972	0.4
Gøderup	89/90	915		1.1	173	230	0.4	1.2	2784	74	26	66	300	3728	1.8

4. Discussion and conclusion

Nitrate

All three drained areas show high NO_3-N concentrations in the upper 100 cm (20-120 mg NO_3-N /l). The values decline downwards and the concentrations of ground- and particularly drain water are somewhat lower. At Skovsbjerggård and Volsted it may be possible that the NO_3 in the soil water is denitrified / reduced, because of the large content of organic matter (available energy for the denitrification process) and there is still a high water content in the peat (poor oxygen conditions). The pH-value at both the mentioned localities decreases through the period of investigation. This may indicate that chemical denitrification plays a part in the reduction of nitrate, possibly in combination with oxidation of pyrites, both processes leading to a drop in the pH-value.

No nitrate reduction at Gøderup

At Gøderup the possibility of nitrate reduction does not exist, as the soil profile mainly consists of sand with only a small fraction of organic matter. Ochre and large amounts of SO_4 in the drain water indicates an oxidation of pyrites, which may result in nitrate reduction. At Gøderup no drop in pH is seen, and most of the SO_4 found in the water originates from dissolved $CaSO_4$.

However, with respect to all the drained areas, influx of ground water from the neighbouring areas have an important influence on the concentration of nutrients in the ground water and in particular in the drain water, especially at times without leaching.

Ammonia

The NH4 concentrations at Skovsbjerggård and Volsted are distinctively higher than the Gøderup values, probably caused by a better air change in the soil at Gøderup. This is suppurted by investigations of the mineralization process (Schou 1990) which showed that ammonificated nitrogen was nitrified immediately.

Phosphate

At Skovsbjerggård (0-100 cm) and at Volsted (0-50 cm) there has been found high concentrations of phosphorous and tot-P in the soil Water, while the concentrations of drain- and ground water are significantly lower. At Gøderup the P concentrations are very low through out the period of investigation. This may be due to better P-binding capacity in the Gøderup soil.

The drained area in gøderup showed significantly higher leaching of NO3-N, Ca and SO4 from the rootzone than the other drained areas. Skovsbjerggård had the highest leaching of PO4-P, total-P and total-Fe.

Leaching from cultivated areas

Compared to leaching results from cultivated areas (Christensen et al. 1990, Hansen 1981 & 1990, Pedersen 1983 & 1985 and Simmelsgård 1985) the results show that the leaching of NH4-N from the rootzone usually is highest in peat soils as compared to mineral soils. Leaching of NO3-N from the cultivated peat soils are at the same level as found for mineral soils (Skovsbjerggård & Volsted) or significantly higher (Gøderup).
The values varies however considerably, dependent on crops, fertilization etc. The leaching of N from uncultivated areas and apparently from areas permanently covered with an established crop are considerably smaller than leaching from areas with a shift between cropped and non cropped periods.

Phosphate leach

Generally the leaching of PO4-P from the rootzone appears to be larger in peat soils than in mineral soils. Leaching of phosphate from soils is however subject to some uncertainty due to methodical problems (Hansen et al. 1990).

The measured amounts of drain water from the three drained areas, showed together with the calculated leaching of surplus precipitation, that a significant influx of ground water took place from the surrounding areas. Most pronounced at Volsted, where 85% of the drain water was influxed ground water.

The amount of nutrients in drain water in relation to the drained peat soil area in question was in most cases considerably higher than seen in corresponding mineral soils. In mineral soils only a fraction of the soil water is leached as drain water, while peat soils have a certain influx of groundwater from the surrounding areas.

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Nitrate Transformation and Water Movement in a Wetland Area

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NITRATE TRANSFORMATION AND WATER MOVEMENT IN A WETLAND AREA.

Abstract

The NO_3^- transformation capacity of a riparian zone at Rabis stream, Denmark, was investigated for a period of 2 years. The riparian zone of 15-25 m received NO_3^--containing groundwater from the adjoining agricultural areas. The water flows as surface runoff along the surface of the wetland and in the root zone towards the stream. Changes in water chemistry, water balance and mass transport was investigated. The riparian zone acted as a buffer zone for NO_3^-, PO_4^{---} and dissolved Fe^{++}. The NO_3^--transformation capacity of the wetland was about 400 kg N/ha-/yr, but varied seasonally. A simple rearrangement of drain systems in wetland areas can reduce the content of NO_3^--N in the danish surface waters by 20.000 - 50.000 t N/yr.

Introduction

Increased nitrate content in groundwater, surface water and eutrophication of shallow marine waters has during the recent years led to an increased interest in nitrate transformation and denitrification in streams and wetland areas.

Denitrification under different climatic conditions and in different sediment types are described by many authors. Jacobs & Gilliam (1985) describe a 16 m wide riparian zone, which acts as a buffer zone for drain water containing nitrate. Other investigations of organogenic, fluviatile and lacustrine sediments show, that measured and calculated denitrification capacities vary between 20-1600 kg N/ha/yr (Hill et al., 1985; Terry et al., 1975; Gambrell et al., 1975; Lowrance et al., 1983; Rhodes et al., 1985; Nichols, 1983).

The purpose

The purpose of this investigation is to examine the nitrate transformation and -capacity of a riparian zone, and the flow pattern of ground- and surface water. The selected investigation field is situated in the NPO Research area at Rabis Bæk.

Hydrogeological setting.

The investigation field is composed of organic sediments with a significant lower hydraulic conductivity than the adjoining and underlying meltwater sand. The organic sediments act as a barrier, so that the groundwater is forced to the surface at the edge of the stream valley (fig. 1). From the valley edge the seepage water flows along the surface and seeps within the root zone towards the stream. Changes in water chemistry, mass- and water balance were investigated.

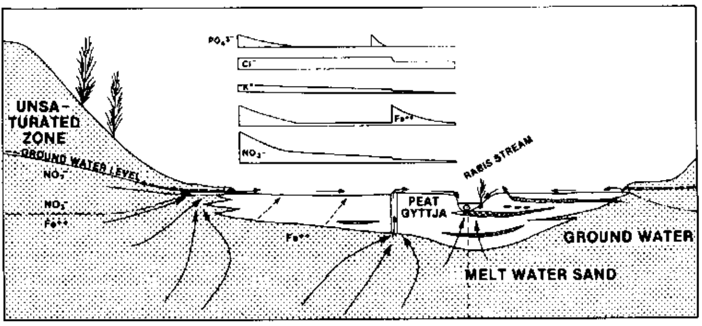


Fig. 1: Cross-section of the stream valley. The chemistry of the surface runoff is shown grafically. The spring near the stream supplies groundwater with a low content of Cl^- and NO_3^-.

Groundwater chemistry.

The groundwater in the catchment areas of Rabis Stream is formed by infiltration in agricultural, heath and coniferous forest areas. It is chemically characterized by the land use. The groundwater infiltrating in agricultural areas contains up to 200 mg NO_3^-/l and up to 150 mg Cl^-/l, while groundwater under the non-agricultural areas contain 1-5 mg NO_3^-/l and 15-20 mg Cl^-/l (Brüsch, 1987; Brüsch et al., 1987; Kristiansen et al., 1986). The selected riparian area is currently not utilized agriculturally.

The test field, 1600 m^2

The test field is 1600 m^2 (100 x 15-25 m), and composed of 2-3 m alternating layers of peat and gyttja. The thickness of the peat and gyttja deposits is greatest in the central part of the stream valley (fig. 1). The peat and gyttja can contain thin layers of sand.

Materials and methods

Surface runoff measurement

The surface runoff was measured by driving a sheet pile around the test field (fig. 2). In the sheet pile 9 outlet pipes were established (P1-9) connected to a discharge pipe in the stream. The discharge pipe was led to a Thompson weir, where the surface runoff was measured.

Sample points and driven wells

The test field was equipped with 16 fixed sample points (B1-16) in the test field surface for surface water collection, and 6 driven wells (R0-5) with a total of 10 screens, placed in the meltwater sand under the organic sediments. The groundwater seepage flux entering through the test field surface, and the piezometric head at the surface, was measured using seepage meters (fig. 3) (Lee, 1977; Lee 1978).

Seepage meters

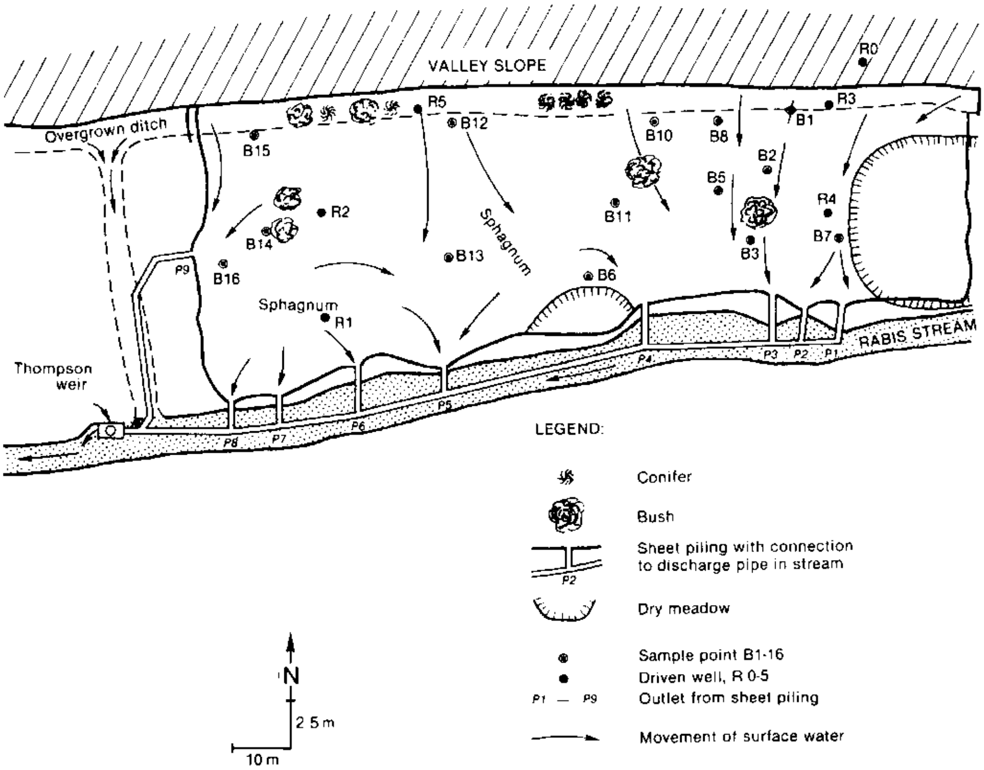
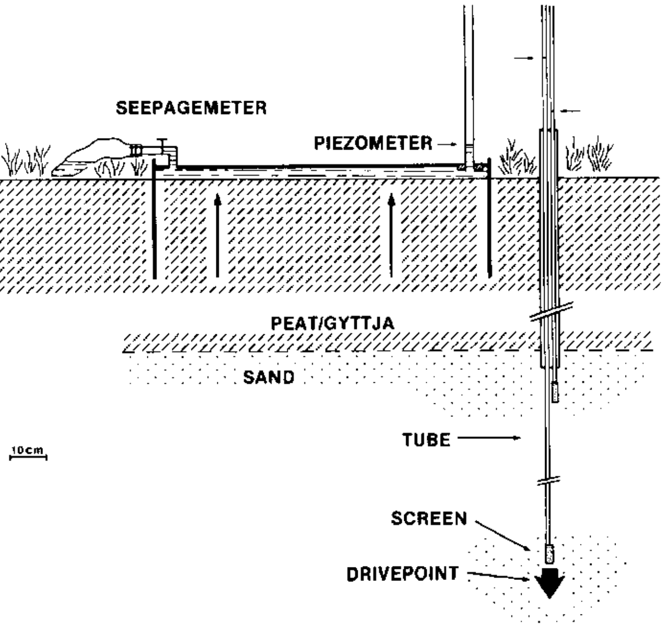


Fig. 2: The test field at Rabis stream, with the water collecting system.

Fig. 3: Seepage meter and driven well with two screens, for the measurement of groundwater seepage and piezometric heads.



Sample collection

Field measurements

Water samples were taken from outlets (P1-9), sample points (B1-16), wells (R0-5), the Thompson weir and for every 5 meters in the seepage front at the valley edge. 16 sample rounds were carried out. The collected samples were analyzed for NO_3^-, Cl^-, Fe^{++} and PO_4^{---}, while selected samples were analyzed for all macro ions. The NO_3^- content, temperature, pH and electric conductivity in the samples were also measured in the field and in up to 80 extra sample points in the test field.

Precipitation, air temperature, wind velocity, evapotranspiration and other parameters were measured at a climate station situated about 300 m NE of the test field.

Results

Surface runoff
1-1,5 m^3/h

The amount of surface runoff in the test field was measured for a period of 2 years. The discharge was about 1,5 m^3/h in the initial phase, but later decreased to less than 1 m^3/h (fig. 4). The surface runoff was about 7500 m^3 in

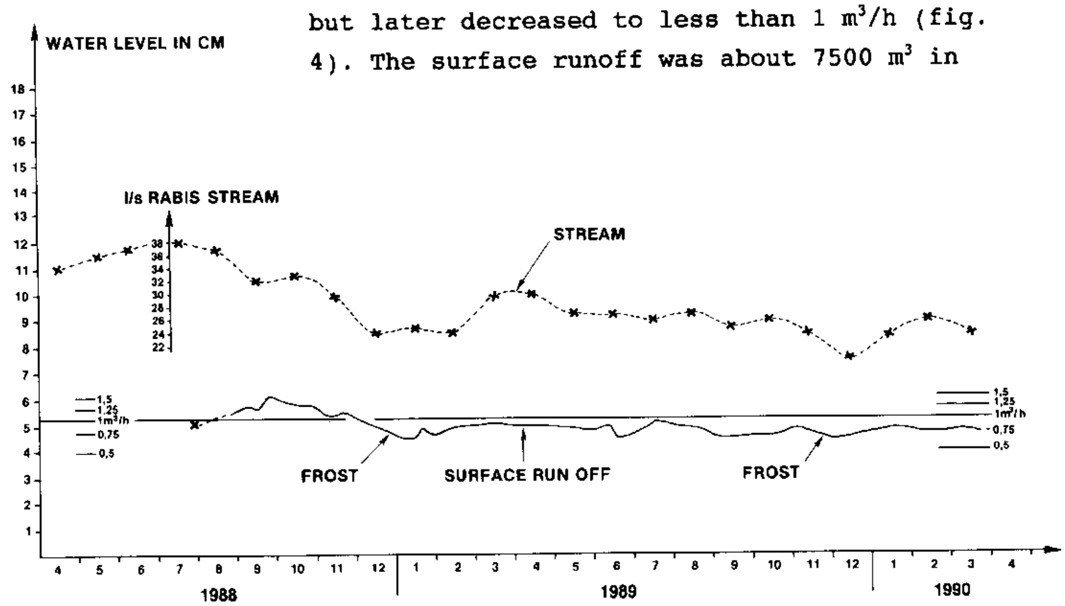


Fig. 4: The surface runoff measured at the Thompson weir, and the discharge of Rabis stream.

1989. Measurements of discharge from parts of the test field showed that the groundwater seepage was approximately the same per meter of seepage front.

Actual evapotranspiration 700 mm/yr = 1120 m^3/yr

Tracer test and residence time

The evapotranspiration in the summer was about 4-6 mm/day, and about 0,5-1 mm/day in the winter. The actual evapotranspiration was measured to about 700 mm in 1989, corresponding to 127 l/h in surface runoff (Brüsch & Nilsson, 1990). Other evapotranspiration measurements in wetlands and for root zone plants vary between 700 and 1500 mm/year (Løgstrup, 1986; Boelter et al., 1977; Gambrell et al., 1975). The residence time of the surface water runoff was determined by tracer tests, where NaCl was added to the seepage water, at the valley edge (fig. 5). The break through curve shows an increase after 3 hours in the electric conductivity at the sheet pile, followed by a decrease after 22 hours. The added amount of NaCl was transported over the test field in about 3 days.

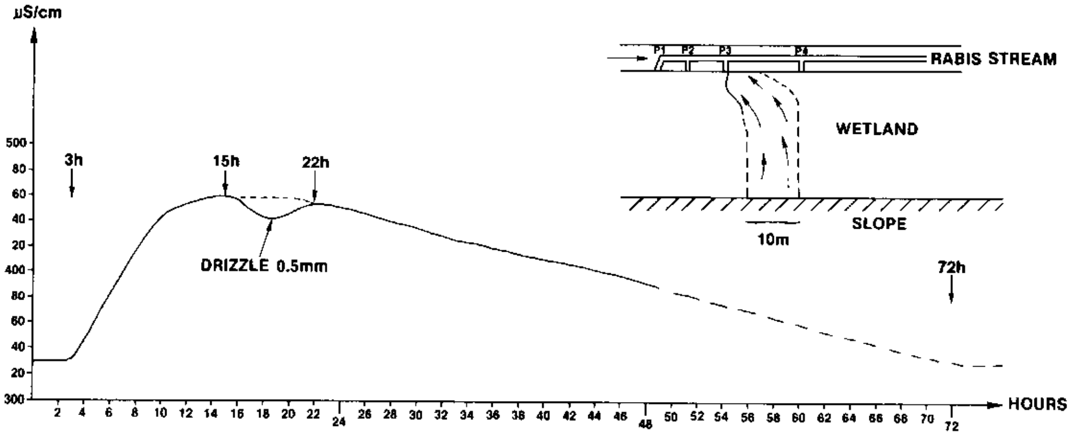


Fig. 5: NaCl-tracertest. The electric conductivity (µS/cm) measured in the test field, at the shet pile (P3).

The hydraulic conductivity of the organic sediments was measured in situ in the test field using seepage meters. The determined K-value

Groundwater flux in the surface - 3% of total surface runoff

was $0{,}9 \cdot 10^{-7}$ m/s, which corresponds to other investigations (Boelter et al., 1977; Vedby, 1984). The groundwater seepage flux in the surface of the test field varied between 0-65 l/h ($1600\ m^2$) and was on average about 35 l/h corresponding to 3% of the total surface runoff.

Non-agriculturally influenced groundwater

Agriculturally influenced groundwater

Fe^{++} and adsorption of PO_4^{---}

NO_3^--load in 1989: 707 kg NO_3^--N/ha

The groundwater below the peat and gyttja had a chemical composition similar to groundwater under the heath and forest areas: 16-20 mg Cl^-/l, 15-25 mg SO_4^{--}/l, 0 mg NO_3^-/l and 0,4-4,8 mg Fe^{++}/l. The groundwater under the central part of the stream valley contained 0,1-0,5 mg PO_4^{---}/l. The PO_4^{---} presumably originates from Tertiary PO_4^{---}-containing sediments underlying the Quaternary melt water sand. The younger and agriculturally influenced groundwater at the valley edge contained up to 160 mg NO_3^-/l, 40-120 mg Cl^-/l and 40-100 mg SO_4^{--}/l, while the reduced part of the young groundwater contained up to 1 mg Fe^{++}/l. The reduced nitrate free groundwater and the upper nitrate containing part of the groundwater were mixed during seepage. Ferrous Fe^{++} was oxidized and precipated at the seepage front, where PO_4^{---} was held back by adsorption on the precipitated iron hydroxide, where a Fe^{++}/PO_4^{---} weight ratio up to 6/1 was measured. The chemistry of the seepage water varied strongly along the seepage front. Field measurements showed two seepage zones with a low NO_3^--content (5-40 mg/l) and three zones with up to 150 mg NO_3^-/l (fig. 6). The average concentration of NO_3^- in the seepage water was constant through the investigation period, 56-58 mg NO_3^-/l, and the NO_3^--load of the test field was 501 kg NO_3^- in 1989 (707 kg NO_3^--N/ha) at a total groundwater seepage of 8620 m^3 (1 ha = 10.000 m^2).

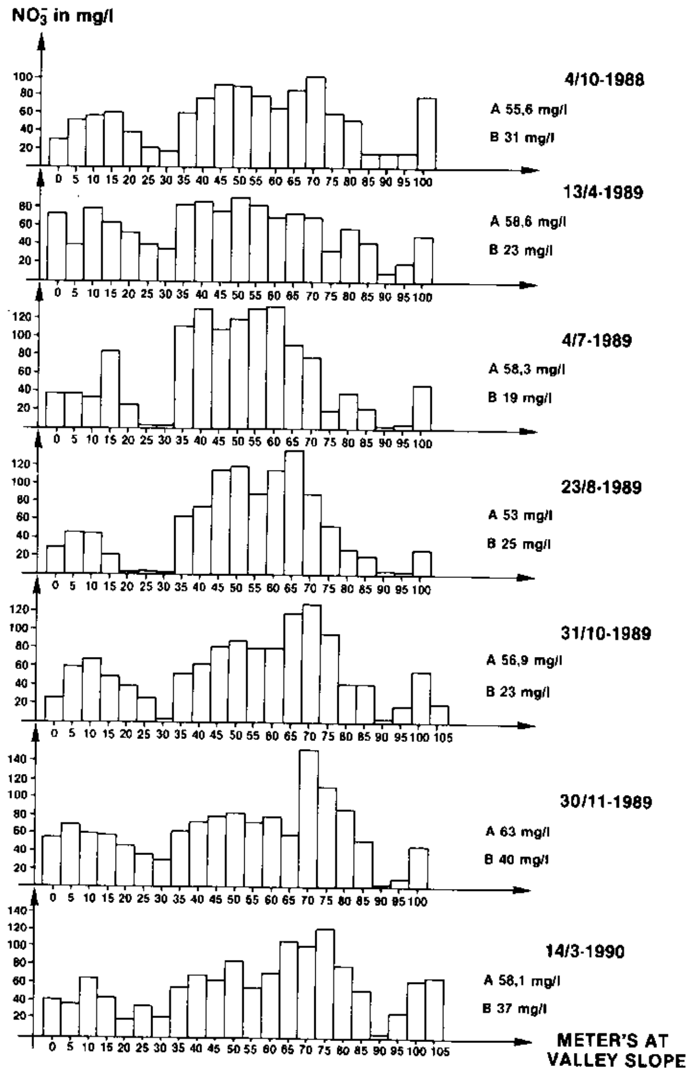


Fig. 6: Content of NO_3^- in groundwater seepage at the valley slope. Samples were taken for every 5 meters in the testfield as near as possible to the slope. A - Average concentration of NO_3^- in all seepage samples. B - Concentration of NO_3^- in surface runoff at Thompson weir.

Annual variation of NO_3^- in surface runoff

The content of NO_3^- in surface runoff showed a large variation on an annual basis (fig. 7). In the summer periods of 1988 and 1989 the content of NO_3^- was about 15 mg/l, while the content in the winter 1988-89 was about 50 mg/l. The nitrate content was 30-40 mg/l in the winter 1989-90, where the temperature was higher than in 1988-89. The chloride content was almost constant throughout the year, except in the summer, where the surface runoff was concentrated, as a result of evapotranspiration.

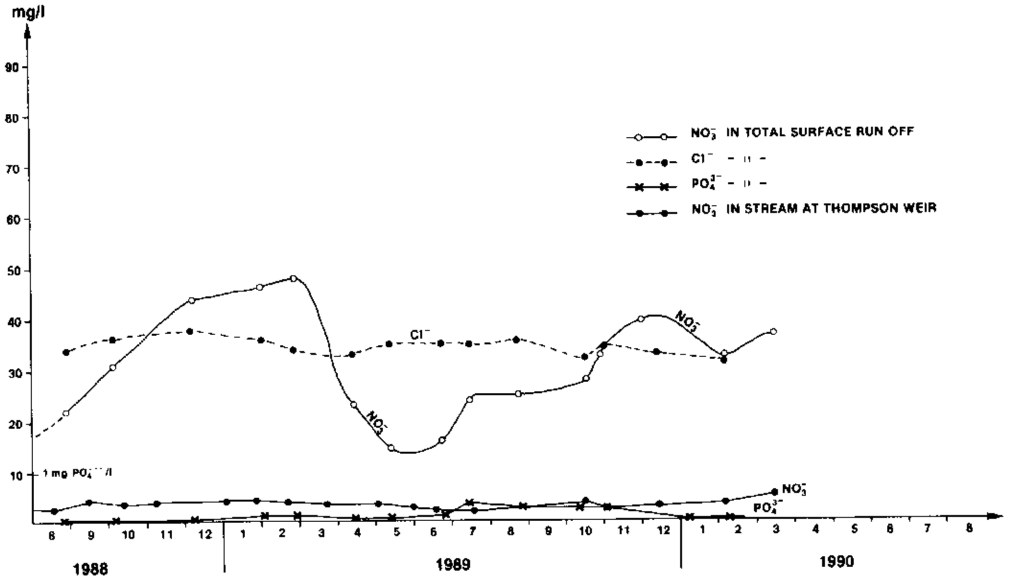


Fig. 7: Content of NO_3^-, Cl^- and PO_4^{---} in the surface runoff at the Thompson weir. 27.08.1988 - 14.03.1990.

Daily variation of NO_3^-

Continuous measurements of NO_3^- and frequent collection of water samples from the Thompson weir showed short term variations in the NO_3^--content, clearly related to the air temperature with an increase of NO_3^- occurring 3-4 hours after a decrease in air temperature (fig. 8).

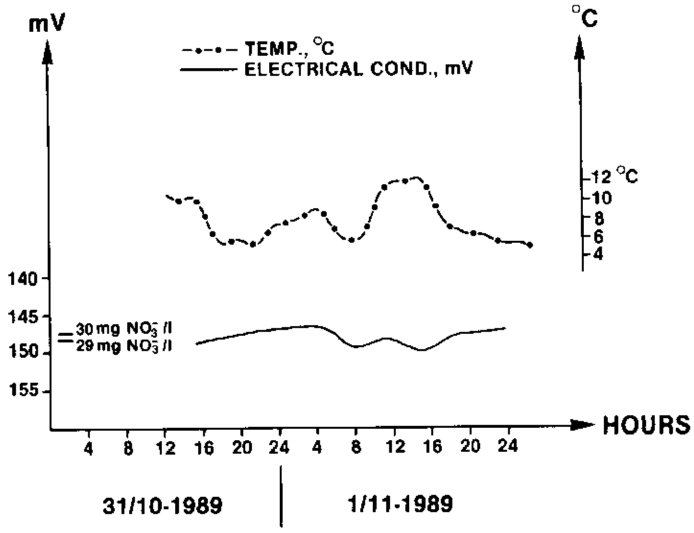


Fig. 8: Plot of a continuous measurement of NO_3^- in surface runoff at the Thompson weir, and the air-temperature. The NO_3^--content was measured using a membrane electrode.

The delay corresponds to the fastest water movement across the test field. In the summer the content of NO_3^- varied approximately 2 mg/l on a daily scale, corresponding to approximately 5-7% of the total nitrate transformation. A similarly low plant uptake has been measured in marshlands (Klopatek, 1978).

Plant uptake 5-7% of total nitrate transformation

Results from three sampling points B1, 2, 3 (fig. 9 & 2) on a flow line crossing the riparian zone illustrate how the chemistry varies seasonally in relation to the transport across the zone. Note that in the summer most of the NO_3^- is transformed within the first 2 m of the seepage front. The Cl^- curve shows how the water is concentrated in the summer on passing the test field.

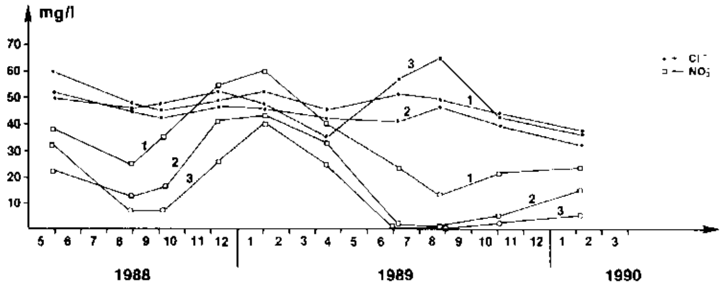


Fig. 9: Content of NO_3^- and, Cl^- in water samples from sample points B1, B2 and B3 (1, 2 and 3).

The distribution of the NO_3^--content in the test area (fig. 10) was partly caused by variations of NO_3^- in the seepage water along the valley edge and partly by the flow pattern of the surface runoff.

Discussion and conclusion

Capacity in 1989 390 kg NO_3^--N/ha

In 1989 the total NO_3^--N transformation was about 390 kg NO_3^--N/ha. Fig. 11 illustrates the seasonal capacity variation of the NO_3^- transformation and the variation of daily mean temperatures. The transformation was about 600 kg NO_3^--N/ha/yr (50 kg N/ha/month) in the summer,

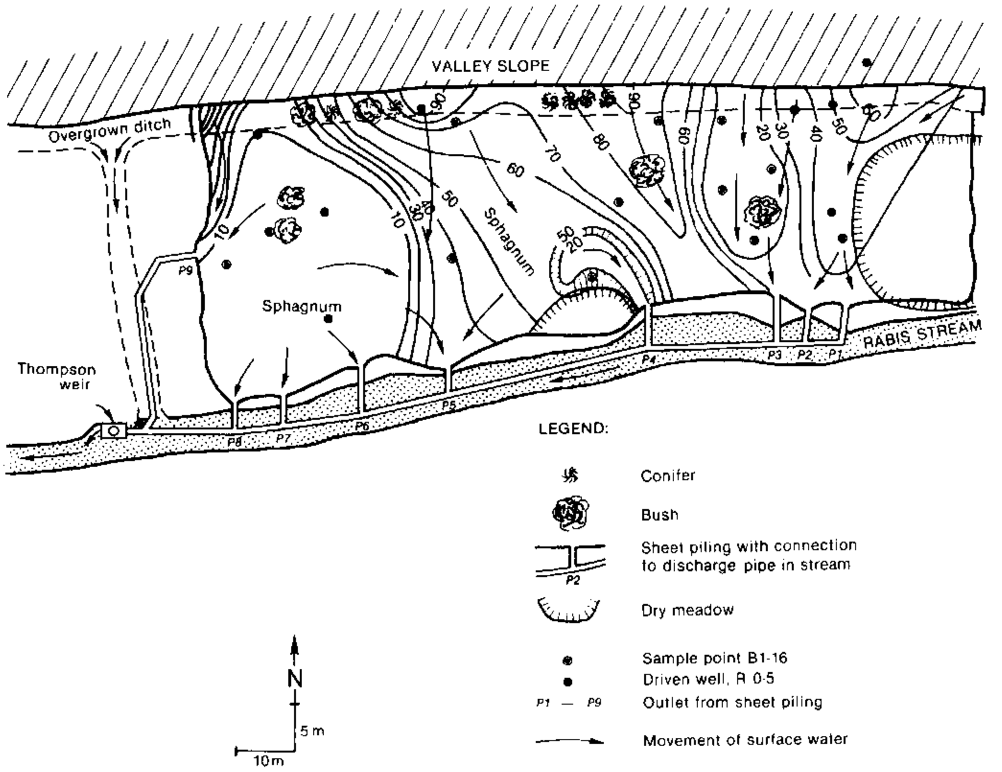


Fig. 10: Isoline-map of NO_3^--content in the surface water. 04.10.88. (87 measurements).

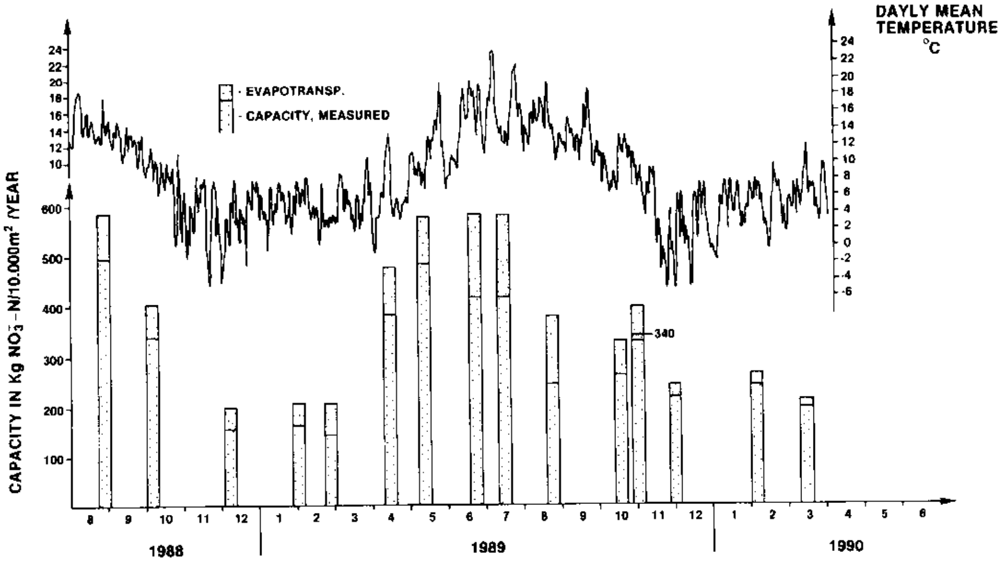


Fig. 11: The seasonal variation of denitrification capacity, and the dayly mean temperature. The NO_3^--transformation of 340 kg NO_3^--N/ha/yr in november 1989 was measured by acetylene blocking.

Capacity and air temperature

while the transformation dropped to 200-260 kg NO_3^--N/ha/yr (20 kg N/ha/month) in the winter. By comparing the daily mean air temperature with the calculated transformation capacities (fig. 12) it appears that the capacity is constant (200 kg N/ha/yr) at air temperatures below 5 °C, while it increases to 600 kg N/ha/yr at 10 °C. Above 10 °C the capacity is constant. The transformation capacity becomes constant below 5 °C due to a warming-up of the riparian zone by the 7-9 °C warm groundwater.

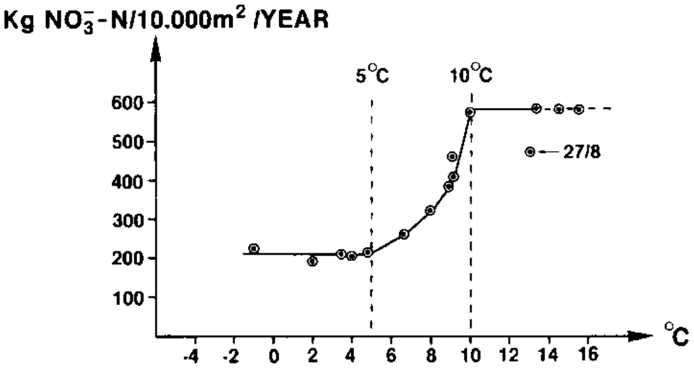


Fig. 12: NO_3^--transformation capacity and dayly mean air temperatures.

Controlling factors of nitrate transformation

The temperature of the air and of the riparian zone are controlling factors of nitrate transformation between 5 °C and 10 °C. Laboratory experiments show that bacterial denitrification increases with sample temperature. (Bremner et al., 1958; Terry et al., 1975; Reddy et al., 1980; Standford et al., 1975). The constant NO_3^--transformation of 600 kg NO_3^--N/ha/yr, at air temperatures above 10 °C, is probably due to lack of easy transferable carbon for the bacterial denitrification process. Adding glucose to the seepage water in a small area of the test field in autumn 1990 induced a drop in the NO_3^- content of the surface runoff from 40 mg/l to 20 mg/l. A similar increase is described in Ambus & Hoffman (1990) where the natural denitrification capacity in the test field at Rabis was measured to about 340 kg N/ha/yr (fig. 11), while the denitrification increased to about 1200 kg N/ha/yr after the addition of glucose.

Another limiting factor could be the slow NO_3^- diffusion, towards the active bacterial films in the riparian zone.

Wetland areas in eastern part of Denmark: Infiltration dominated

In the eastern part of Denmark, riparian zones are irrigated by stream water and drain water in a clayey catchment area. Investigations here show, that denitrification capacities range from 300-1000 kg N/ha/yr. The main part of the effluent drain water infiltrates, and the NO_3^- transformation occurs especially in the uppermost part of the peat (Hoffmann, 1985; Jørgensen et al., 1988; Ambus et al., 1990). The test area at Rabis stream differs from the above mentioned wetland areas partly due to the area being characterised by groundwater, and partly due to a piezometric head that prevents infiltration. The bacterial denitrification at Rabis stream probably takes place in thin films close to the surface of the riparian area.

Rabis: Groundwater dominated wetland

The riparian zone. A natural buffer

The wetland area at Rabis stream acted as a natural buffer zone for NO_3^- and dissolved Fe^{++} (fig. 1). The Fe^{++} was precipitated as ferrihydroxides in the riparian zone, accompanied by an adsorption of PO_4^{---} on the ferrihydroxides.

Future scopes

A significant reduction of the NO_3^- contents in streams and lakes would be obtained by a simple rearrangement of drain pipes and ditches in drained wetland areas. At least 250.000 ha of organogenic wetland areas occur in Denmark (Markmann et al., 1988; Madsen et al., 1987). Assuming that 20-50% of the wetland areas are drained, these would, after a rearranging of the drains and using a denitrification capacity of 400 kg N/ha/yr, reduce the NO_3^--N-supply to streams by 20.000-50.000 t N/yr. However, the capacity varies from area to area, as does the NO_3^- content in drain and groundwater.

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Performing organization(s):

Abstract:
Eutrophication related to contamination with nitrogen and phosphorus have been studied in the comprehensive NPo Research Programme. Abstracts of projects on eutrophication phenomena in streams, lakes and marine areas are presented. Denitrification rates have been estimated for wetlands and bottom sediments.

Terms:
erosion; transportation; biological models; monitoring; eutrophication; lakes; meadows; streams; marine areas; drains; transformation; nitrogen CAS 7727-37-9; phosphorus CAS 7723-14-0; oxygen CAS 7782-44-7

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Korte engelske sammendrag beskriver de enkelte forskningsprojekter.

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