BALTIC SEA ENVIRONMENT PROCEEDINGS
NO. 44

NITROGEN AND AGRICULTURE INTERNATIONAL WORKSHOP

9-12 April 1991
Schleswig, Germany

BALTIC MARINE ENVIRONMENT PROTECTION COMMISSION
– HELSINKI COMMISSION –

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VOLATILE AND LEACHING LOSSES FROM CONTEMPORARY AGRICULTURE CLEANER TECHNOLOGY AND MANAGEMENT PRACTICES

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Preface

Reports on the current ecological state of the Baltic Sea demonstrate environmental degradation over extensive areas. Recorded effects include elevated nutrient levels, increased primary production and increased frequency and severity of oxygen deficency in bottom waters as well as declining species diversity. The improvement of the state of the Baltic Sea and prevention of further impairment is a joint obligation of all Baltic Sea States. The Helsinki Commission is one fora which arrange cooperation on this matter.

Eutrophication has received particular attention in this work due to its grave impact on the ecology of the Baltic Sea. The single most important factor in the eutrophication of the Baltic Sea is the, since the 1960'ies, steadily increasing nitrogen load. The major anthropogenic source of nitrogen are the agricultural sectors of the Baltic Sea States. A common strategy to remedy this pollution is therefore necessary, not just to regenerate the Baltic Sea but also to conserve a resource vital to the Baltic Sea nations. The available technologies and means to abate nutrient discharges from agriculture were the topic of a workshop jointly arranged by Germany and Denmark, the results of which are reported in the present procceding.

The workshop was held in the town of Schleswig situated in Schleswig-Holstein, Germany. This publication includes a summary of the presentations and discussions made during the workshop and the expert communications submitted to the workshop. The publication and the discussions between scientists and officials formed the basis for seven Recommendations on measures to reduce nutrient inputs from agriculture. The Recommendations on measures to reduce nutrient inputs from agriculture. The Recommendations were submitted to the 13th meeting of the Helsinki Commission (February 1992, Helsinki) which adopted six of them as HELCOM Recommendations 13/7-13/12.

The editors, Mr. Torben A. Bonde and Mr. Uwe Schell, and the authors are considered to be responsible for the contents of the publication.

Summary

The eutrophication of the Baltic Sea is caused mainly by excessive nitrogen but also phosphorus inputs from both point and non-point sources including atmospheric deposition. The non-point or the diffuse sources are mainly of agricultural origin and contribute significantly to the overall load of the Baltic Sea. The variety of nutrient inputs from agriculture include ammonia volatilization, leaching, run-off and erosion losses of nitrogen and phosphorus, and farm waste discharges.

Ammonia is volatilized almost exclusively from animal manure with relatively small losses arising from commercial fertilizers, soils and plants. The ammonia losses from animal husbandry occur from animal housings, manure storages, field application of manure as well as from grazing animals.

Leaching, run-off and erosion losses of nitrogen from agricultural land are

inevitable in terrestrial ecosystems but can be reduced substantially as compared to losses caused by present farming practices. The nitrogen losses are mainly due to nitrate leaching, but also leaching losses of ammonia and organic nitrogen are significant. Run-off and erosion losses of nitrogen are most frequently a consequence of surface application of manure and the impact of rainfall and wind on soils, respectively.

Leaching, run-off and erosion losses of phosphorus are generally small from an agronomic point of view but may be critical to eutrophication of surface waters. Farm waste discharges or farm pollution is caused by the release of animal slurries, silage effluent, parleur water, and yard washings from farm premises into adjacent water courses.

Ammonia volatilization from animal housing:

Ammonia volatilization from animal manures commence immediately after excretion of the manure, i.e., losses occur from animal houses and during storage and field application of manure. The process is governed by many factors such as the ammonia concentration, the pH of the manure, air temperature, humidity, ventilation, management and construction of the housing systems etc.

Even if all these factors interact and condition the ammonia volatilization it was concluded at the workshop that the ammonia concentration and the surface area (the emitting surfaces) of the manure are the most critical factors for the losses. It was also stated very clearly that in advanced livestock breeding it is possible to reduce the overall excretion of nitrogen, mainly ammonia, by reducing the total amount of proteins in fodder, with a significant reduction of the ammonia volatilization from animal housing as one consequence. The reduction potential of the total nitrogen content in animal manure is up to 40-50%. This reduction can be accomplished without losses in weight gain and quality of meat by optimizing the qualtity and amino-acid composition of the fodder. Thus, it is important to set standards for commercially available fodder and fodder additives as well as standards for feeding in phases in order to utilize fodder efficiently and to reduce nitrogen in excretion.

On-farm produced fodder represents an undeclared product which makes it more difficult to optimize the use of this fodder. However, significant reductions of the manural nitrogen has been observed when on-farm produced fodder, such as wheat, was substituted by barley or oats. Further reductions were achieved by addition of specific amino-acids. It is therefore possible to reduce the excretion of manurial nitrogen also in the case where on-farm produced fodder is used.

In order to achieve a nation-wide reduction of ammonia losses from animal housing, feeding tables and standards (recommendations) should be set for fodder and feeding practises.

The reduced amount of ammonia in the animal manure has first of all implications for the volatile losses in animal housing, but also losses from storages and field application of manure may be reduced. A general principle which applies to all sources is the principle that the manure should be kept in a closed system as long as possible. In other words, the emitting surfaces must be reduced. It was additionally concluded by the experts that for many housing systems it is important to remove the manure from the animal housing to an outside storage as quickly as possible. When stored in an appropriate storage, ammonia losses are very small. Standards for frequent removal of manure should be developed.

A number of additional aspects such as temperature, ventilation rate, the density of animals may be considered. In the case of poultry manure, the drying of the manure as quickly as possible after excretion has been shown to result in a dramatic reduction of ammonia losses.

Biofilters was concluded to be a technology which locally could prevent smell and dust nuisance rather than a technology applicable on a national scale.

Ammonia volatiliza tion from storages:

The storage capacity of animal manures has implications for ammonia losses from manure when field applied, because losses depend on the climatic and soil conditions under which the manure is applied. Also for this reason it is important to establish sufficient capacity. The losses are further dependent on the type of manure, e.g., urine, slurry or solid manure.

Considering liquid manure the experts strongly recommended that:

1) Slurry should be stored in covered storages, either by floating covers, membranes, solid lids, or membranes in conjunction with retrieval of methane gasses. Further it is important to load the storages from the bottom in order to reduce the emitting surfaces also in this stage of the manure handling;
2) Urine should be stored in air-tight storages, i.e., storages should primarily be equipped with solid lids.

Slurry has a low **pH** and a high dry matter content relative to urine. This implies that the diffusion rate for ammonia to the surface layer is small, being the cause of a small volatilization rate. This is not the case for urine and an air-tight lid is necessary to prevent losses which otherwise are substantial (e.g. 50% of total N).

Solid manure contains a large fraction of organic nitrogen and a relatively small fraction of mineral N in the form of ammonia. However, when decomposing under aerobic conditions the organic N is mineralized with the potential for further losses. Losses of more than 50% of the total N are not unusual. It is important to avoid aerobic composting of solid manure and to store it under anaerobic conditions. To achieve this, bottom loading of manure heaps may be practised as may covers.

Ammonia volatiliza tion from field application of manure:

As a consequence of the general principle that manure should be handled in a closed system as long as possible, the liquid manure (slurry and urine) should be applied by means of either direct injection or trailing hoses depending on soil conditions and crops. The following conclusions were reached for the various types of manure:

1) Slurry should be directly incorporated when applied on bare soil, i.e., by means of direct injectors, trailing hoses equipped with units for injection, or immediate harrowing down of the manure.
   Slurry applied in crops should be directly incorporated or applied with trailing hoses (in crops with a dense canopy).
2) Urine should be directly incorporated or applied with trailing hoses or any comparable system both on bare soil and in crops.
3) Solid manure (only applied on bare soil) should immediately be plowed down.

In addition to these technological means of reducing losses, further reductions will be reached if application takes place during favourable whether conditions such as relatively cold and humid conditions.

Ammonia volatilization from plants:

Ammonia is volatilized from plants as from the soil itself in small quantities. The losses have been quantified in a few cases and found to be insignificant as compared to losses from animal manures. However, losses may be substantial when crops are fertilized with a surplus ofnitrogen. Also from this point of view over-fertilization should be avoided.

Nitrate leaching:

The experts agreed that the underlying cause of elevated nitrogen losses are the nitrogen surplus, i.e., the annual surpuls per ha agricultural land. The fundamental objective of abatement strategies should be to diminish or eliminate the nutrient surpluses. The surpluses and the associated losses are caused by inappropriate farming structures and practises such as high livestock denisties and excess fertilization. It was firmly concluded, that commercial fertilizers and animal manures should be applied based on the principle of replenishing nutrients removed by harvest. This statement implies that the utilization efficiency of both commercial fertilizers and animal manures must be optimized. To achieve this, crop farming and animal husbandry must be integrated and crop farming based on integrated production methods.

Over-fertilization with nitrogen can be reduced when appropriate prognosis tools for nitrogen are developed and available to farmers. Additional incentives such as compulsory fertilizer and crop-rotation schemes and nutrient budgets on a farm basis including control measures and fines are, however, needed if optimal fertilization is to be reached.

Economic incentives such as reduced agricultural subsidies have shown to be very efficient means to reduce losses through a general extensivation of the production. It was therefore generally agreed that ongoing negotiations in the GATT-organization and within the EC to reduce subsidies – if successful – would stimulate an environmentally sound development with major reductions in losses as one consequence hereof.

The utilization of manural nutrients:

The optimal utilization of nutrients in animal manures can be accomplished by the application of animal manure shortly before or during the early growing season. This implies that sufficient storage capacity should be available to farmers and it was concluded that a capacity of 6 – 10 months depending on farm practise will ensure a high utilization efficiency. However, it was also recognized that in order to reach the objective that all farmers have a storage capacity in the range indicated it is necessary to develop governmental programmes for subsidizing farmers investments.

Critical to the use of animal manures is the amount applied per ha. Based on the principle of replenishing nutrients a range of livestock units (LU) which correspond to maximum application rates were calculated, i.e., 1.5 – 2.0 LU for cattle, and 1.0 – 1.5 LU for pigs and 1.0 for poultry. For cattle grasslandfarming the upper limit is 2.5 LU.

Further increase of the utilization efficiency of commercial fertilizers can be accomplished by promotion of integrated plant production. Such farming practises optimize the efficiency of fertilizers through a precise determination of the optimal fertilization level and the use of appropriate crop rotations.

The allocation of animal manures among individual fields and farms is a key requisite if losses are to be reduced substantially. Further distribution of manures may be promoted by degassing of manure in large-scale biogas plants. This results in a stabilized and declared product collected in a omanure bank« which may be attractive also to crop-farmers. The use of lorries possibly in connection with a small application unit makes it possible to distribute the animal manure over larger areas.

The biogas plants reported on during the meeting receive large quantities of industrial waste which is mixed with the animal manure and degassed. This increases the recycling of nutrients in society and diminish the demand for land fills. City-dumps or land-fills are potential threats to ground- and surface waters and the use of the organic waste in biogas plants reduces this potential.

Phosphorus leaching and erosion:

Phosphorus losses from agricultural land make up a substantial contribution to the load of fresh-waters with phosphorous. Phosphorus is added to these waters in the form of particulate organic P mainly via soil erosion and in the form of orthophosphate mainly via leaching.

It is important not to add excess fertilizer or manurial P as is the case of nitrogen, but it is additionally important to consider tillage operations such as mouldboard plowing in an attempt to reduce losses. Further reductions are possible by increasing the fraction of the arable land with green fields, i.e., fields sown with catch crops or winter crops.

Sufficiently broad filter strips along watercourses (streams and creeks) should be considered as an additional measure, which would go well in hand with an increased permanent set-aside of agricultural land.

Farm waste:
Farms wastes are an important diffise source if farm buildings and facilities are not of sound construction and if liquids are not collected and diverted to storages. Farm waste has grave consequences for local surface-waters such as smaller lakes and streams, Consequently it has been relatively straightforward to motivate polluters to remedy this pollution in many countries. The experts recommended that standards should be set for farm-constructions holding animals or manure.

Wetland management for retention of nutrients:
The retention of nutrients in freshwater-systems was recognized during the workshop as an important additional measure to reduce the overall load of nutrients to the Baltic Sea. The processes include nitrogen removal by denitrification and nitrogen and phousphorus removal by assimilation. Watercourses should be allowed a natural meandering in order to increase the physical quality, the retention time of the water and thus the denitrification and the overall self-purification capacity of the watercourses.

Maintenance of water-courses should take into consideration the physical quality of the water-course (macrophytes may be cut less frequently, stones left. on the bottom etc.).

Drainage pipes in abandoned meadows should be plugged and the water allowed to infiltrate the meadow. This will in most cases increase denitrification and retention of phosphorus.

Ammonia emissions from animal houses

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Summary

Ammonia emissions from animal houses are discussed in conjunction with soil acidification, forest decline and input of nutrients into water surfaces. In systematic emission measurements in animal houses, the various parameters on ammonia emissions were determined, and the emissions from cattle, pigs and poultry (only gallinaceous birds) were compared.

Ammonia emissions increase with the size of the emitting surface in the animal house and the storage duration of slurry or manure. Temperature and exposure of the emitting surfaces to the airstream are affecting the emissions of ammonia.

Introduction

One part of the nitrogen input as intake of foodstuffs are animal products for the consumption. Losses in faeces are the correspondending other part. There are 3 sources for gaseous ammonia:

1. Slurry or manure in the stables
2. Emissions of storage systems
3. Application of slurry or manure in the field

The volatilization of ammonia in animal houses is a stepwise pathway. The uric acid input is an indirect function of the age of the animals and a function of the input-output-ratio of the foodstuffs. The uric acid input decides on the uric acid pool in litter or slurry. Depending on the pH and the temperature a bacterial ammonification from uric acid to ammonia nitrogen will take place. The relation between soluble and free ammonia depends on the pH and the temperature too (Elliot and Collins 1982).

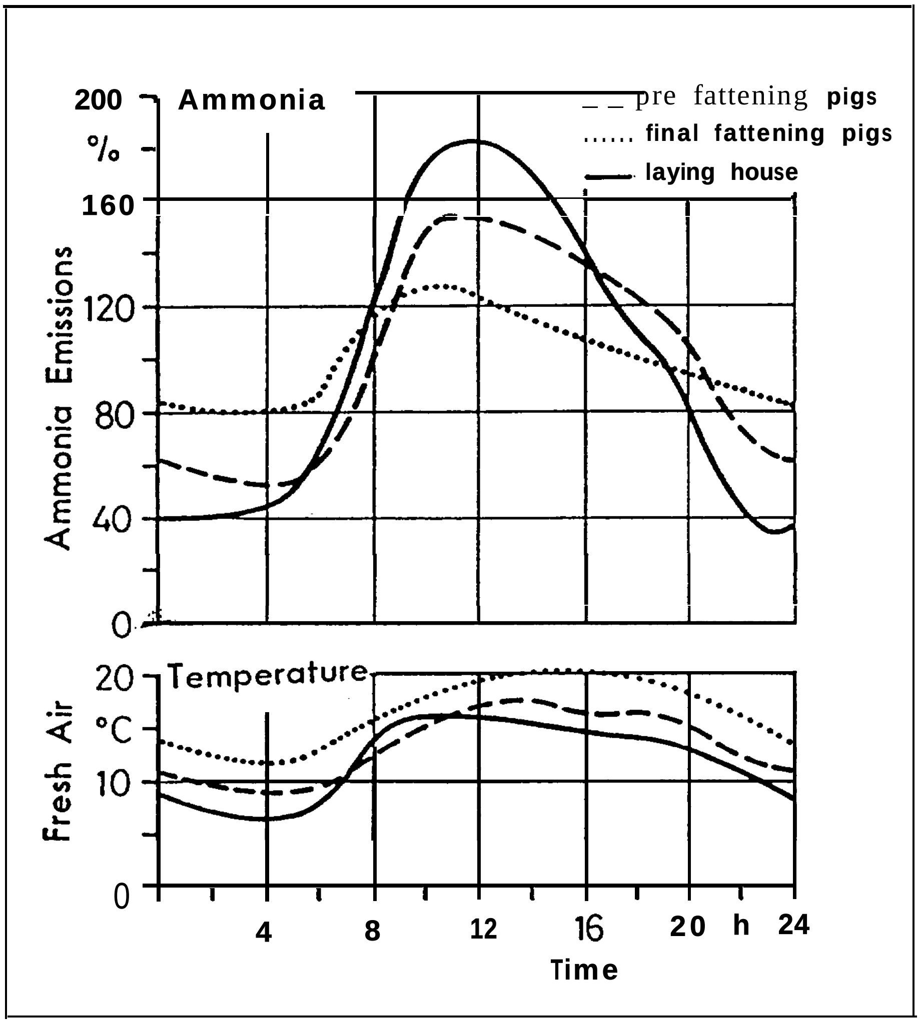
The mass transfer of ammonia from slurry or litter to the atmosphere is a function of the temperature and the concentration of ammonia in the atmosphere of the house near the slurry surface. Animal houses with slurry systems have two different atmospheres. One is the space between the slurry surface and the bottom of the slatted floor, the other is the factual room for the animals. The ambient ammonia level in this room is diluted continuously in dependence of the rate of ventilation. The ammonia emission of an animal house is the product of the concentration of ammonia in the exhausted air and the rate of ventilation.

Emission as a function of daytime

Figure 1 shows the ammonia emissions of three different housing systems over a period of 24 hours and corresponding fresh air temperature of the environment. The highest emissions from 10 to 12 a.m. are results of the high ventilation rates in all three housing systems. The high ventilation rate at this time is a function of the increasing fresh air temperature at this time and the high activity of the animals in dependence of the time of feeding and the light in the animal houses.

This observation is essential for the interpretation of results of measurements from day time. The daily mean for ammonia emissions is less than the result of one measurement at, for example, 12 o'clock.

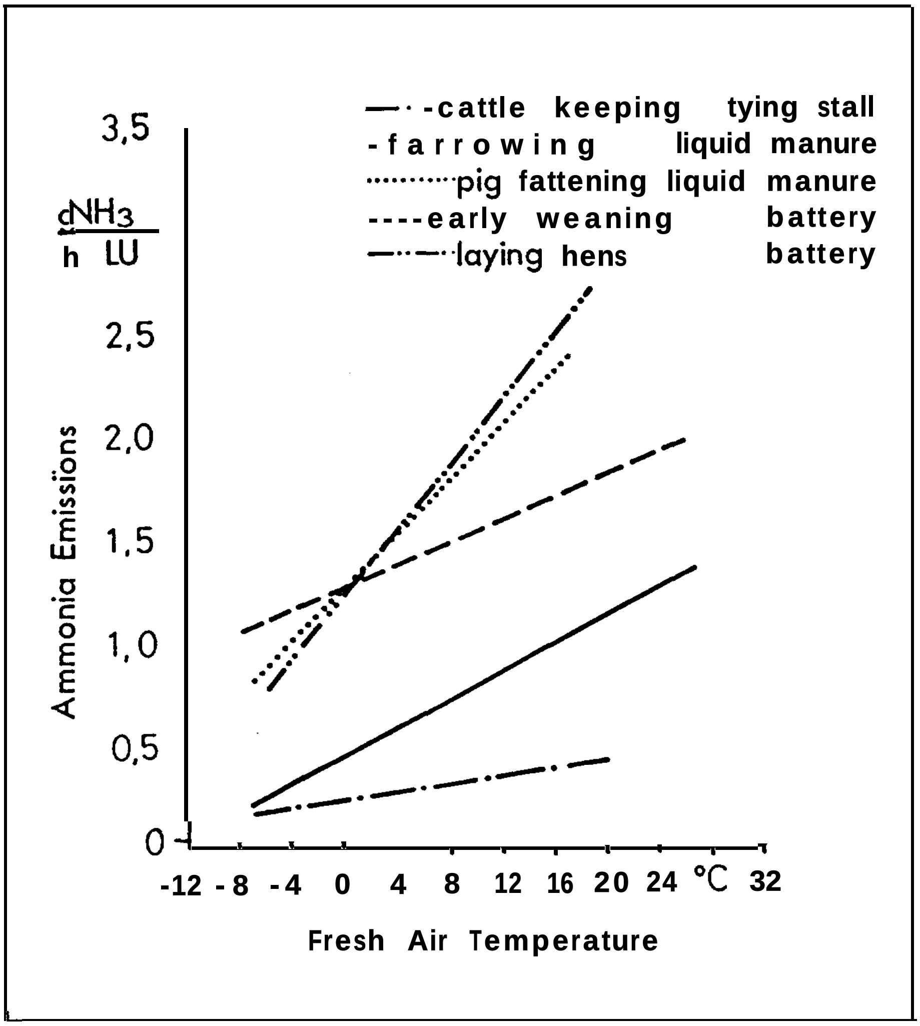
Fig. 1:
Standardized ammonia emissions over a period of 24 hours



Emission as a function of temperature

Figure 2 shows the ammonia emission of five different housing systems in relation to the fresh air temperature, especially in winter and summer. The increase of the ammonia emission is different in relation to the emission level of the housing system (the emission is showed as g ammonia per hour and livestock unit (500 kg bodyweight)). The summer emission of all systems is higher than the winter emission.

Fig. 2:
Ammonia** emissions **in relation to the fresh air temperature



Different housing systems

The average ammonia emissions of different housing systems on the basis of a yearly mean temperature of 7.9°Celsius shows fig. 3. This temperature is the mean for the city of Schleswig for a time of 40 years. On this basis one can compare the ammonia emissions of the analyzed housing systems.

Cattle keeping had with 0.24 g NH, $h^{-1} LU^{-1}$ the lowest emission of all analyzed housing systems. Cattle keeping needs the lowest rate of ventilation of 300 m^3 air per LU and h maximum. As said is emission the product **of** concentration and rate of ventilation.

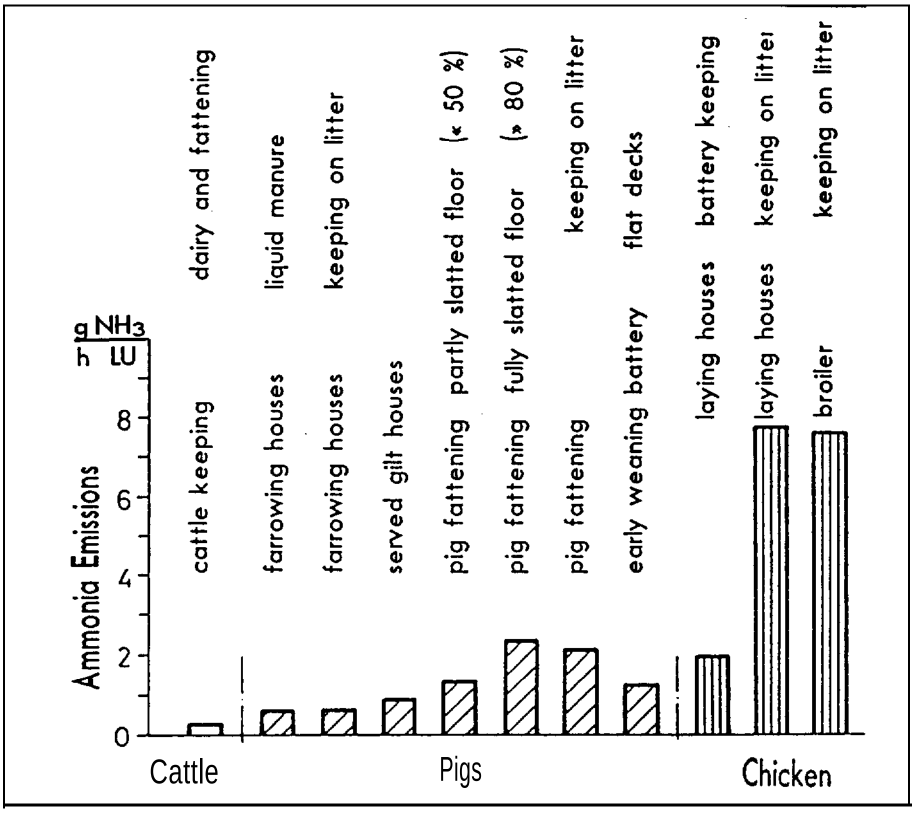
There are no differences between farrowing houses with liquid manure and farrowing houses with litter (0.65 g NH, $h^{-1} LU^{-1}$). The litter in far-rowing houses is taken away every day. The ammonia emission of served gilt houses for sows was a little bit higher (0.93 g NH, $h^{-1} LU^{-1}$).

There are significant differences between the ammonia emissions of pig fattening on partly slatted floor (1.35 g NH, $h^{-1} LU^{-1}$) and pig fattening on fully slatted floor (2.38 g NH, $h^{-1} LU^{-1}$). The ammonia emission of pig fattening houses with straw (litter) was 2.15 g NH, $h^{-1} LU^{-1}$. The emitting surface for ammonia is in the systems with pig fattening on fully slatted floor and keeping ***on*** litter higher than the emitting surface of pig fattening houses with partly slatted floor.

The early weaning batterys (flat decks) for small pigs were emitting 1.3 g

NH, $h^{-1}LU^{-1}$. The maximum ventilation rate for pigs is 600 $m^3 h^{-1} LU^{-1}$, the double of cattle keeping.

Fig. 3:
Average ammonia emissions of different housing systems on the basis of a yearly mean temperature of 7.9°C



The ammonia emission of laying hens in batteries was with 2.0 g NH, $h^{-1}LU^{-1}$ comparable to the emission of pig fattening systems. But the ventilation rate of chicken houses and for laying hens too is with nearly 1.200 $m^3 h^{-1} LU^{-1}$ maximum the highest rate for all animal housing systems.

Laying houses with hens on litter had the highest ammonia emission of all analyzed systems (7.9 g NH, $h^{-1}LU^{-1}$, 7.64 g NH, $h^{-1}LU^{-1}$ at young hen systems). In these two systems the litter is regularly stored from the first to the last day of chickens life (6 months in young hen systems, 1 year in laying houses with litter).

All in-all out systems: broiler farming

Broiler farms are the drastic example for the ammonia emissions of all in-all out housing systems. The broiler lifetime in Germany is very short. The German market needs chicken with 1.600 g bodyweight alive and 1.200 g dressed weight. For this reason the broilers will see the slaughterhouse after a lifetime of 35 days. The birds are growing very fast. At day one a bird has a bodyweight of 50 g, at day 35 1.600 g. This is a growing factor of more than 30.

Broilers are living on litter. The litter is fresh straw in the first days of the fattening period. After two weeks there are enough faeces for emitting ammonia.

Ammonia emissions of 30 g NH, $h^{-1}LU^{-1}$ maximum are resulting from temperatures near 30°C and a high rate of ventilation of 1.200 $m^3 h^{-1} LU^{-1}$ maximum. The product of growing of the animal mass and ammonia emission of one LU is like an explosion of emissions from the broiler plant.

Fig. 4:
Ammonia emissions from a broiler farm in relation to the fattening period.

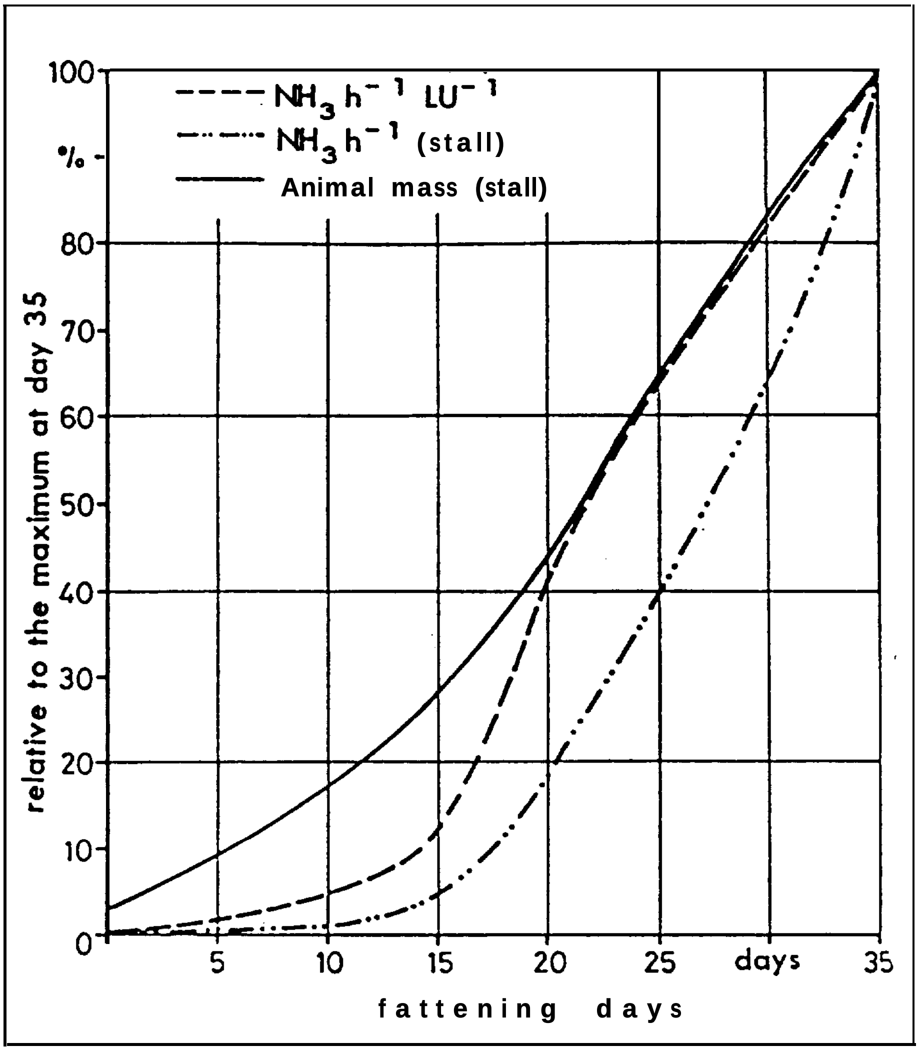
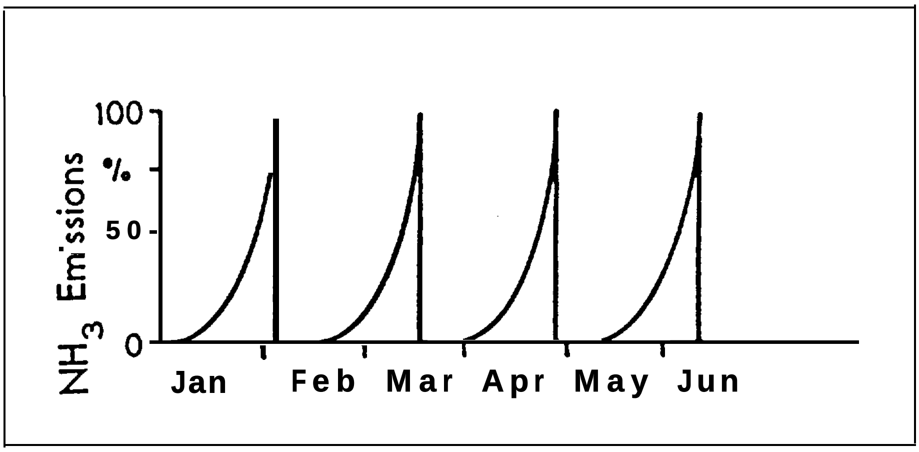


Figure 5 shows the emission of an all in-all out system during the year. All all in-all out systems, not only broiler houses, have no continuous emissions. This is a big problem for the calculation of air cleaning techniques.

Fig. 5:
Ammonia emissions from a broiler farm during a year



Protein reduced feeding

To reduce the consumption of animal products is one way for reducing ammonia emissions from animal houses. A second way is the use of cleaning techniques to reduce the existent ammonia emissions. There are more facilities:

The first result of new feeding strategies show a reduced ammonia emission from pig fattening houses during the final fattening period (60-l 00 kg bodyweight). The ammonia emission of the protein reduced feeding trial was 76% of the total ammonia emission of the normal feeding strategy. The volume of slurry was less than 60% of the normal feeding strategy. Slurry is the ammonia source during storage time and slurry application.

The foodstuffs for protein reduced feeding strategies are more expensive than normal foodstuffs by need of more synthetical amino acids.

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Ammonia volatilization from slurry during storage and in the field

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Summary

The emission of ammonia from manure represents the most important source of atmospheric ammonia in Europe. Ammonia losses related to livestock farming occur from animal housings, manure storages, spreading of manure, applied manure and from grazing animals. This report consider ammonia losses from stored slurry, during application and from surface applied slurry.

From stored slurry ammonia losses in winter-spring (7.3°C) was half the losses during summer (16.9°C), due to lower temperatures. Surface crustings reduced ammonia losses to 20% of the losses from slurry stirred weekly. In one experiment during a winter-spring period, it was shown that a layer of chopped straw (15 cm) could replace a surface crusting layer. Ammonia losses from slurry covered by rape oil, leca pebbles, sphagnum peat and floating foil was low, i.e. 2-50% of the losses from stirred slurry.

During application of slurry with conventional spreaders ammonia losses is low, but application with irrigation devices may cause losses of 10% of applied ammonium.

Ammonia loss from surface applied slurry is affected by climate, slurry composition, soil conditions and time from application until incorporation of the slurry. Generally no ammonia is lost from slurry injected into the soil, and after incorporation of slurry into the soil little ammonia is lost. Ammonia losses are exponentially related to air temperature during the first 6 hours after application and linearly related in following periods, and ammonia losses increase with wind speed. Rain or irrigation reduces ammonia losses from surface applied slurry. Ammonia losses increase with increasing slurry dry matter and pH or slurry alkalinity. These factors are not well described yet, and more knowledge about effects of soil conditions is needed. Therefore calculations of ammonia emission rates for countries or from regions remain uncertain, and it is impossible to calculate seasonal ammonia emissions.

Introduction

The emission of ammonia from manure represents the most important source of atmospheric ammonia in Europe (Buijsman *et al.*, 1987), while emission of ammonia from mineral fertilizers is of minor importance. In Denmark the estimated yearly emission of NO_x from combustion and NH, from fertilizers and manure was 76.000 Ton N and 75.000 Ton N, respectively, in 1988-l 989 (Miljostyrelsen, 1990). Agricultural ammonia losses may be somewhat higher as losses from stables seem to be underestimated.

Deposition of oxidized or reduced nitrogen can detrimentally affect nitrogen-limited natural ecosystems (Schulze *et al.,* 1989). The effect of deposited nitrogen on marine ecosystems is belived be high relative to the amount added, because the deposited nitrogen enriches the surface layers in which there is a high proportion of the marine biological activity.

Ammonia losses related to livestock farming occur from animal housings, manure storages, spreading of manure, applied manure and from grazing animals. An estimate of the ammonia emission from agriculture in UK showed that 44% was emitted from animal buildings and manure storages, 36% from manure spread on land and 25% from grazed grassland, only 5% was assumed to be lost from mineral fertilizers (Jarvis and Pain, 1990).

Jarvis and Pain (1990) stated that there are few dependable measurements of ammonia losses from stored manure. Most measurements have been made by mass balance studies, in which the losses have to be estimated as small differences in the total-N content of a very inhomogenous mass. At Askov we therefore measure the ammonia losses from stored slurry directly with a wind tunnel system.

Numerous studies have shown that ammonia losses from surface applied slurry are affected by climate, slurry characteristics and soil condition (Pain *et al.,* 1990a; Holzer *et al.,* 1988). Therefore ammonia losses from surface applied manure can vary considerably, and knowledge of the most important factors affecting the volatilization are essential when the ammonia emissions are calculated.

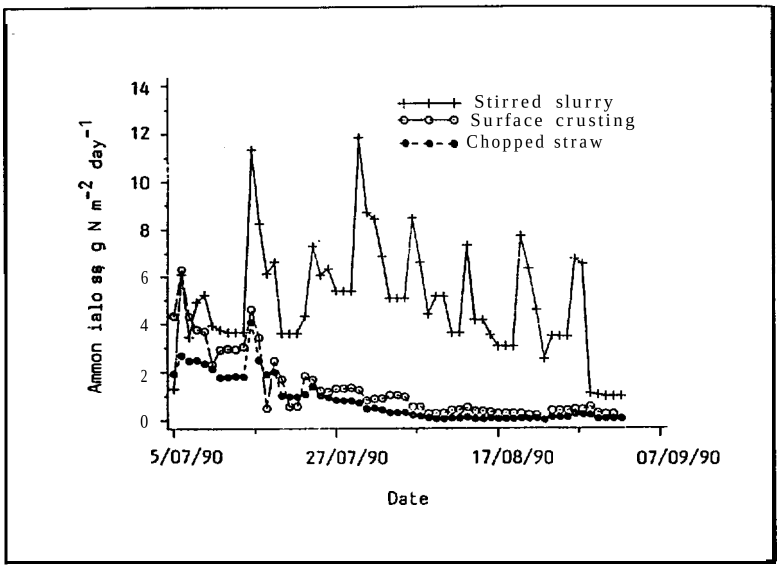
Ammonia losses from slurry storages

The rate of ammonia loss from stored slurry is correlated to the concentration of ammonia at the slurry surface, which is affected by total ammonia concentrations (TAN), pH and temperature. Ammonia in the slurry surface is in equilibrium with the ammonia in the immediate atmosphere, and the reduction of atmospheric ammonia due the convective transport will therefore increase ammonia losses.

Ammonia loss rate is therefore very high from stirred slurry as TAN readily is transferred by convection to the surface layers (Fig. 1). The ammonia loss rate was high on the day, when the slurry was stirred, but the loss rate decreased rapidly as the high losses resulted in a reduction of TAN concentrations in the surface layers. The experiments showed that ammonia losses from a slurry with no surface crusting can be as high as from a stirred slurry, probably because climatic induced convective transport increases TAN contents in the surface layers (Fig. 2).

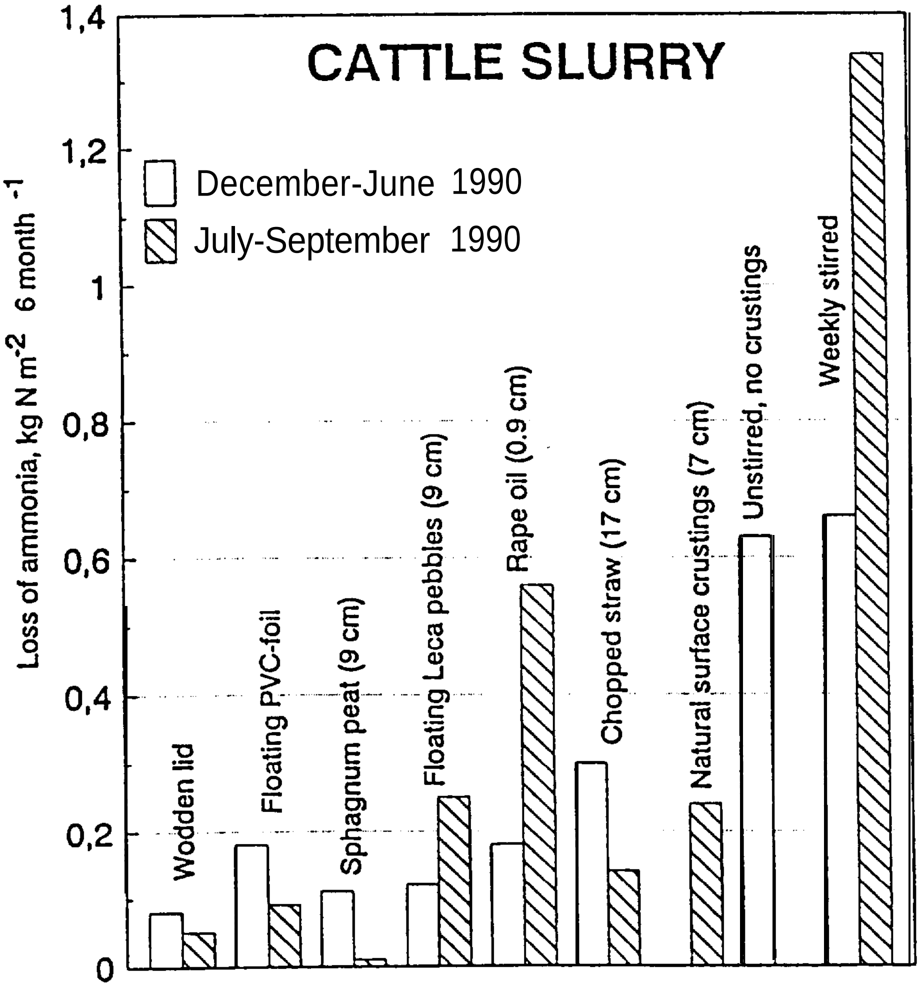
Ammonia loss from small scale slurry tanks containing cattle slurry showed a 6 month ammonia loss from the stirred slurry of 0.66 kg NH,-N m^{-2} and 1.34 kg NH,-N m^{-2} in a winter (7.3°C) and a summer (16.9°C) period, respectively (Fig. 2). The ammonia loss was measured with windtunnels in which the wind speed was adjusted to 3-4 m s^{-1}. In a Dutch wind-

Figure 1.
Daily ammonia loss rates from stored cattle slurry, stirred weekly, with natural surface crusting, and a surface straw layer (Sommer, 1991).



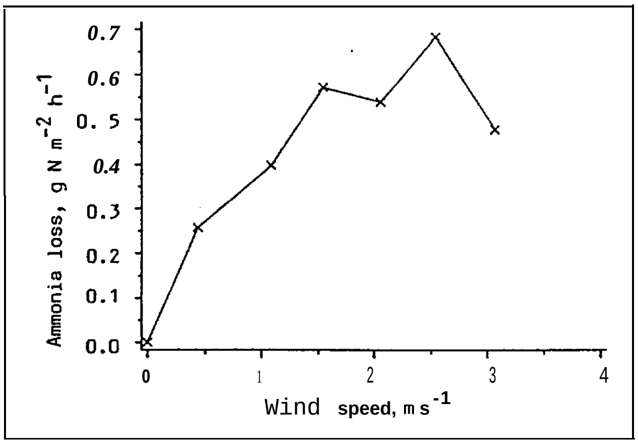
tunnel experiment *0.56* kg NH,-N m^{-2} and 1.30 kg NH,-N m^{-2} was lost during 6 month from a cattle slurry during a winter and a summer period, respectively' (Bode, 199 1). The wind speed was 1 m s^{-1}, and there were no surface crusting layer on **the** unstirred slurry. The high losses from slurry **without** surface crusting (Bode 199 1), was probably due to natural mixing in the slurry like it was seen in the Askov experiments (Fig. 2). The ammonia loss was two to three times higher from a pig than from a cattle slurry (Bode, 199 1), probably because the structural fibre precipitated in the pig slurry, reducing the viscosity, and increasing the potential for convective transport of ammonium.

Figure 2.
Ammonia losses from stored cattle slurry with eight different surface coverings (Sommer, ***1991).***



Surface crusting, sphagnum peat, copped straw and Leca pebbles create a stable air layer above the slurry surface, and the submerged part of the material will limit convetive transport in the surface layer of the slurry. Therefore ammonia has to be transported slowly by diffusion through the layers, whereby the liquid and air resistance will increase, and ammonia loss rate decrease (Muck and Steenhuis, 1982). In the Danish experiments the ammonia loss was reduced with more than 50% compared to the loss from the stirred slurry (Fig. 2). *In the* study of Bode 199 1 the ammonia loss from slurry with natural crust formation and with incorporated chopped straw was 27% and 6 1% lower than from slurry with no crust formation. A surface layer of rape oil reduced ammonia volatilization with 60 to 70%. TAN is insoluble in oil, whereby the transport of ammonia from the slurry to the air is restricted. Some ammonia was lost by gas bubbles penetrating the rape oil layer and because cracks developed in the oil layer as the oil was partly absorbed by the surface crusting and dried out. Very little ammonia was lost from the slurry covered with different surface materials, floating foil or a lid (Bode, 1991; Sommer, 1991).

*Figure 3. **Ammonia loss rates from stored pig slurry related to wind speed, determined with wind-tunnels during a winter period.***



In both studies the greatest reductions were found during the summer periods, where losses from the uncovered slurry storages were high.

Ammonia loss rates increase with increasing wind speeds, because of reduced gas phase resistance above the slurry surface. In the wind tunnel experiments at Askov (Fig. 3), the liquid phase resistance dominated at wind speeds higher than 5 m s^{-1} in the wind tunnels. Thus at higher wind speeds loss rate did not increase with wind speed. The relationship between wind speed and ammonia loss rates from slurry tanks seems to be logarithmic in accordance with the ammonia loss pattern from an ammonium carbonate solution, arising from hydrolyses of urea in aqueous sollution (Katyal and Carter, 1989). If the diffusive resistance was insignificant, the ammonia loss should be linearly related to wind speed (Vlek and Stumpe, 1978).

Ammonia volatilization during slurry spreading

Loss of ammonia during spreading with conventional slurry spreaders equipped with a central sprinkler-plate was less than 4% of applied ammonium in a study of Sommer (1989). Pain *et al.* (1989) found that the losses was less than 1%. There were no differences in ammonia losses, when the slurry was applied in a spreading fan of 11 m width and 10 m length or 6 m and 4 m (Sommer, 1989). This result is not supported by the observations reported by Boxberger and Gronauer (1990), who found that ammonia losses increased from 4% to 6.7% of applied ammonium when the spreading length increased from 1.5-10 m to 5-15 m. Spreading of slurry using an irrigation device having a spreading length of 25-30 m resulted in losses of 9.6% of the applied ammonium.

Volatilization from surface applied slurry

Most trials show that more than half of the total ammonia loss during a six day period takes place within the first day (Fig. 4; Thompson *et al.*, 1987; Pain *et al.*, 1989). Proton activity has a substantial influence on the flux of ammonia from surface applied slurry. After an initial period with high loss rate (Fig. 4), the ammonia volatilization potential will be reduced by acidification, decreasing concentrations of TAN and infiltration of TAN into the soil (Beauchamp *et al.*, 1978; Lauer *et al.*, 1976).

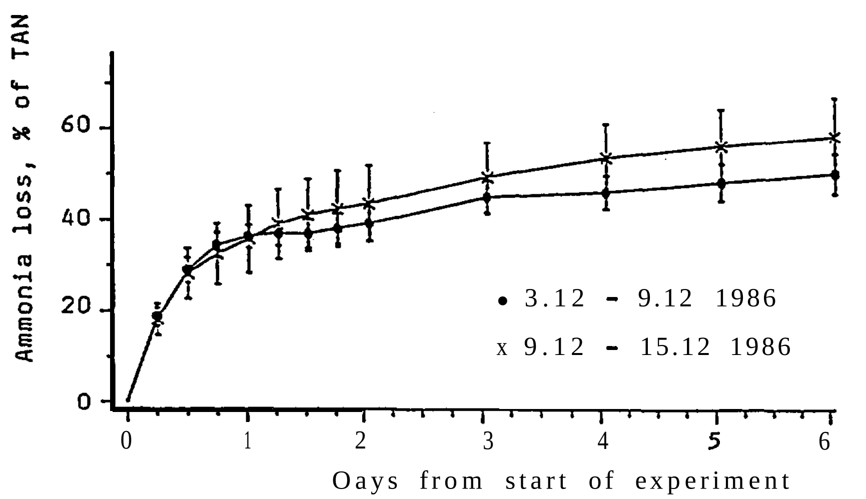


Figure 4. *Accumulated ammonia loss in percent of applied TAN in slurry (3 l/m²) for 2 trials (Sommer et* al., *1991). Mean* air *temperature was 3-6%, wind speed 4 m s^{-1}. Bars indicate ± S.E. of observations, n=3).*

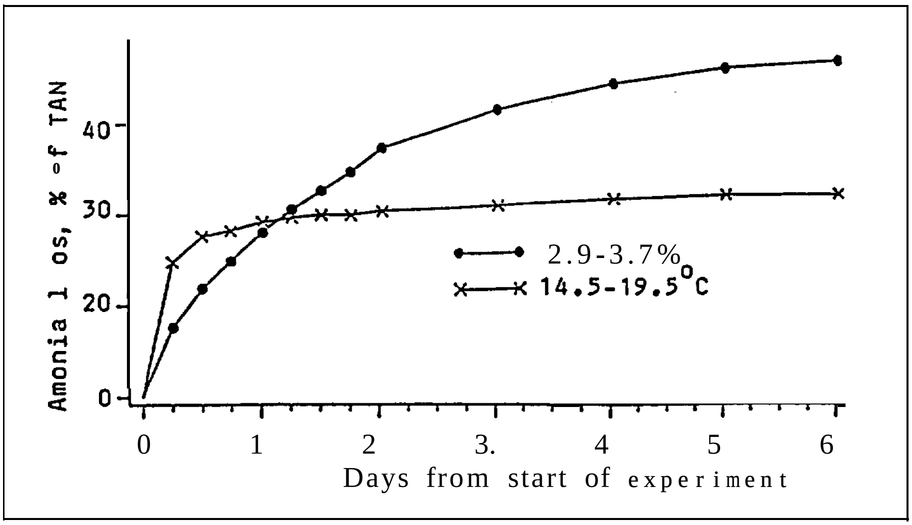
Factors affecting ammonia volatilization

Ammonia loss from different slurries after surface application to the soil depends on climate, slurry pH and slurry dry matter (DM), and time from application until the slurry is incorporated into the soil.

Climate

When temperatures were near zero, the rate of ammonia loss were generally low (Fig. 5). The accumulated loss over six days was high, however, due to a constant loss rate throughout the period. In the experiments the soil was saturated with water and partially frozen, factors which probably increases ammonia losses (Sommer *et al.*, 1991). Sustained ammonia losses from slurry during periods with near-zero temperatures have also been observed in a study of Thompson *et al.* (1987). At 19°C the initial loss rate were high but after 12 h almost no further loss occured (Fig. 5), probably due to surface crusting and rapid infiltration into the dry soil.

Figure 5.
Accumulated ammonia loss in percent of applied TAN in cattle slurry (Sommer et** al.,* ***1991). Slurry dry matter content was 7.5 %, pH 7.7 and TAN 2.6 g N/l.



The results of 20 wind tunnels-experiments with near identical cattle slurries were related to climatic conditions (Fig. 6). During the initial 6 h accumulated ammonia volatilization was exponential related to temperature. During the three succeding periods of 6-12 h, 12-24 h and 24 h-6 d the ammonia volatilization rate was low and only slightly related to temperature. In these periods ammonia loss pattern did shift to be linear related to temperature. Four equations describing the relationship between ammonia loss and temperature where determined for the four periods. For all experiments 46% of the measured loss over 6 days was accounted for with the models (Sommer *et al.*, 1991), showing that other factors like soil condition do affect the ammonia loss. The ammonia loss rate increased when wind speeds increased up to 1.75 m s^{-1} in the wind tunnel unit. No consistent increase in volatilization was found, when the wind speed increased from 1.75 to 2.8 m s^{-1}. This is similar to the relationship between ammonia loss rate and wind speed over the slurry tank.

Figure 6.
Relation between air temperature and accumulated ammonia loss during the periods (A) O-6 h, (B) 6-12 h, (C) 12-24 h and (D) 24 h-4 d in ***percent of TAN in the slurry at the beginning of the period (Sommer et*** al., ***1991).***

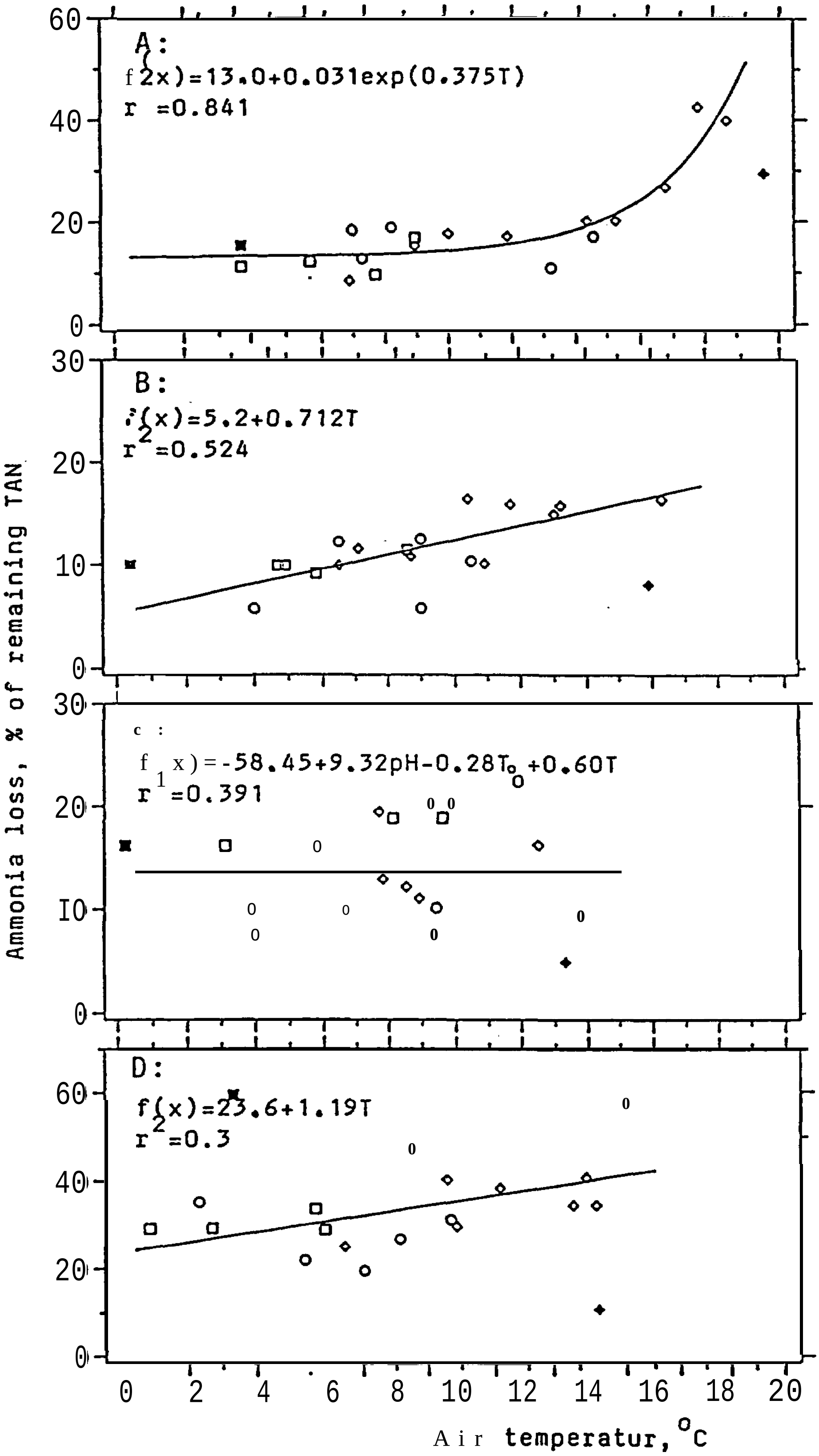
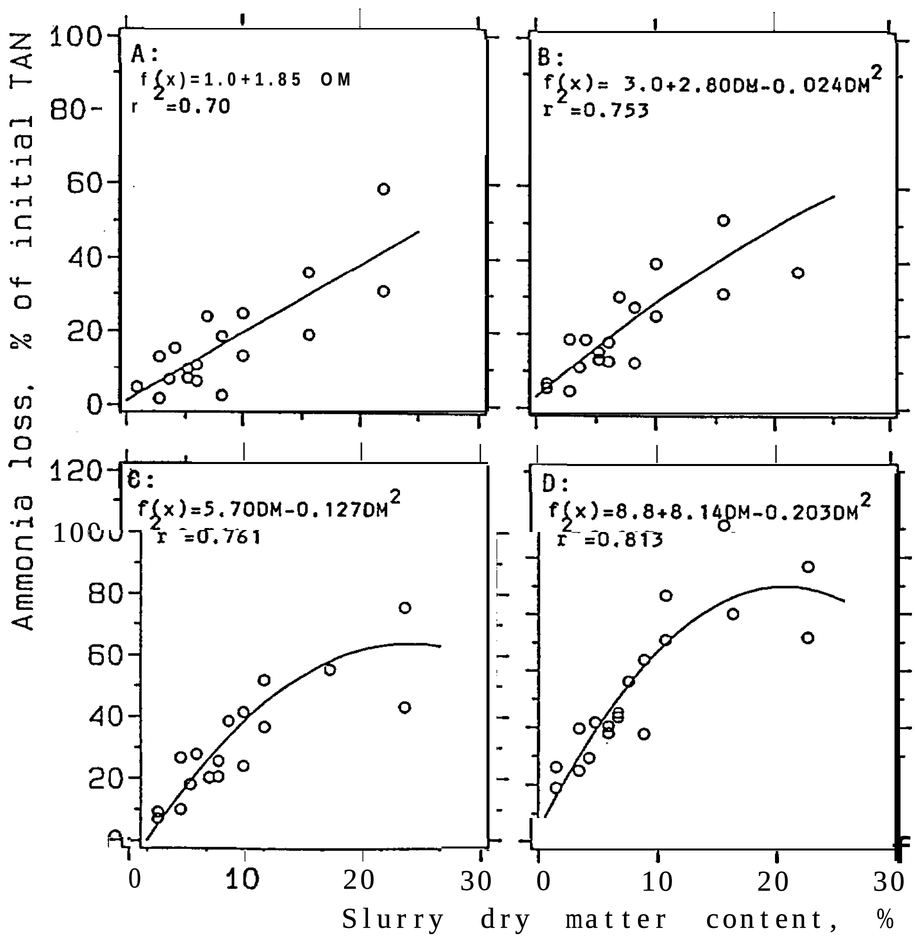


Figure 7.
Relationship between slurry dy matter content and accumulated ammonia loss during the periods (A) O-6 h, (B) 6-12 h, (C) 12-24 h and (D) 24 h-6 d, in percent of initial TAN at the beginning of the period (Sommer and* Olesen, *1991).

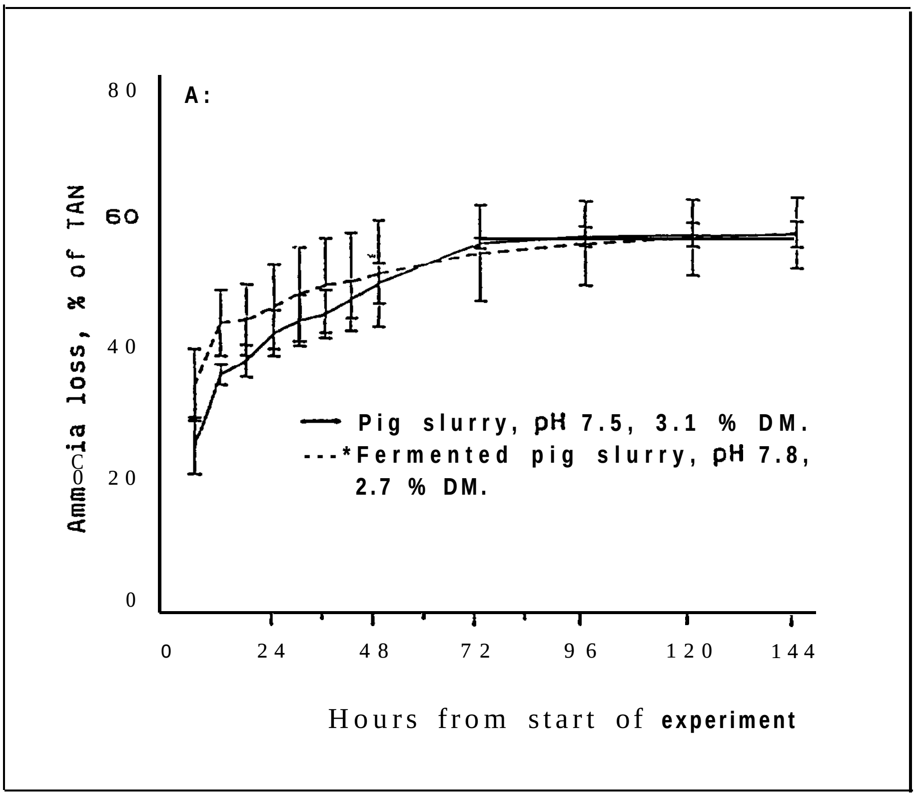


Effect of slurry composition

Ammonia loss was related to slurry dry matter in cattle slurry adjusted to different dry matter contents (Fig. *7).* The ammonia loss from 0 to 6 h was linearly related to dry matter content, while the relationship in the following periods was nonlinear. If the effect of pH and temperature was eliminated from the data the ammonia loss tended to be sigmoidally related to dry matter content in all four periods. This indicated that at low (<4%) and high contents (>12%) of dry matter, small changes in dry matter content have a limited influence on ammonia loss. If viscocity of the slurry is not correlated to dry matter content, differences in viscocity may be of importance as higher losses is seen from cattle slurry with an apparent high viscosity than from adjusted slurry with similar dry matter content. It was shown that interaction of temperature and dry matter content resulted in high losses from the fibrous fraction (20% DM) but not from the liquid fraction (1% DM) in winter experiments (Sommer and Christensen, 1990).

Fermentation of slurry had no effect on the ammonia loss following land-spreading (Fig. 8), this was also the conclusion of a study of Pain *et al.* (1990b). It was considered (Somrner & Christensen, 1990) that within the same category of manure (e.g. slurries, farmyard manure or urine), the loss of ammonia may be predicted from slurry dry matter content, pH and climatic conditions.

Figure 8.
Accumulated ammonia loss from pig slurry and fermented pig slurry (3 l/m²)* (Sommer *& Christensen, 1990).

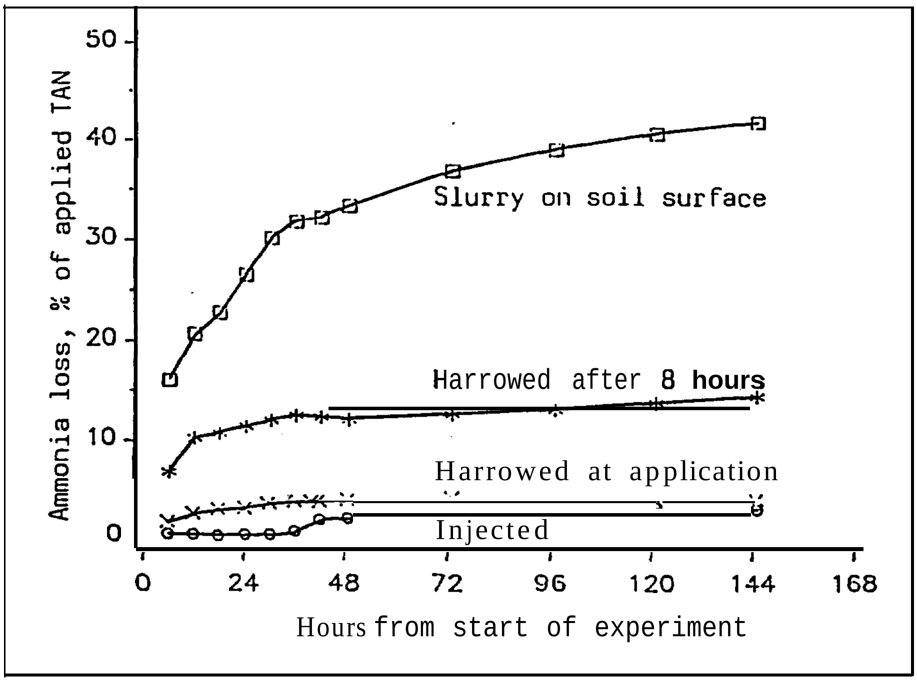


Increasing the initial pH of the slurry caused an increase in ammonia loss rates from the slurry (Sommer and Christensen, 1989). Recent studies has shown that ammonia loss is more strongly related to the alkalinity of the slurry than to pH (Husted *et al.*, 199 1). This could explain why ammonia loss from an anaerobic digested slurry with a higher pH than in an undigested slurry was identical (Pain *et al.*, 1990b).

Effect of incorporation

Injection of slurry or immediate incorporation into the soil reduced the ammonia loss to low value (Fig. 9). When incorporation of the surface applied slurry is delayed, ammonia loss is reduced from the time of incorporation. A high proportion of the ammonium was lost, when slurry was injected into a soil with high water content (Sommer and Christensen, 1990). Application of slurry on the soil between rowes of plants with hoses trailed on the soil behind the spreader can reduce ammonia losses (Bless etal., 1991). The effect is greatest, when the slurry is applied in a well developed plant canopy, due to a more effective wind reduction and a greater uptake of atmospheric ammonia by plant leaves. Loosening the soil surface before surface application of slurry reduces ammonia losses due to higher infiltration rate of slurry into soil (Horlacher and Marschner, 1990).

Figure 9.
Accumulated ammonia loss from fermented pig slurry (3 Urn"), injected, and rotor harrowed into the soil immediately after application and 8 hours after application (Sommer, not published).



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Ammonia emission after the application of manure in the field

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Introduction

During the last decade society has become more and more sensitive about environmental pollution. Also agriculture is requested to produce more ecologically. Intensive livestock farming and the utilization of liquid manure has already been criticized for a long time. Apart from the odour and the pollution of surface and groundwater by nitrate leaching, ammonia emission from manure is of public interest.

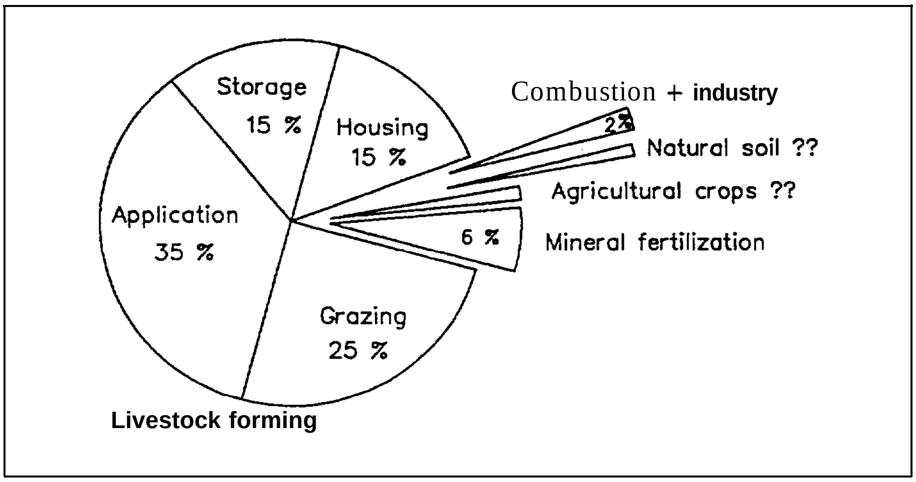
To quantify ammonia emission from solid and liquid manure after field application, extensive investigations were started in May 1989.

Source and effect of ammonia emission

Source

The predominant source of ammonia is agricultural. Apart from emissions from industrial fertilizers, livestock farming causes approximately 90%. During the microbial decomposition of animal waste ammonia (NH,+) is produced, which can volatilize as gaseous ammonia (NH,).

Fig. 1:
Sources of ammonia emission (Germany: 100% = 750.000 tons/year; estimation according to Isermann, 1990 and Möller and Schieferdeck, 1990)



About 15% of the ammonia already volatilizes from animal houses as well as during the storage of manure. Around 25% is produced during the grazing of animals. The major part of 35% of the total ammonia emission is lost when manure, mainly slurry, is applied in the field as an organic fertilizer (Fig. 1).

Effect

for agricultural crop management

For German conditions it is estimated that 260.000 tons NH,-N/year are emitted into the atmosphere after the application of manure in the field which should be used as an organic nitrogen fertiliier. These losses are equivalent to approx. 230,000 mil. DM (1 kg N as industrial fertilizer = 1 DM).

Table 1:
Some factors affecting ammonia volatilization (the influence of factors in bold letters were tested in experiments)

1. Management	2. soil	3. Atmosphere
Source of manure (animal type, housing type)	Infiltration Cation Exchange Capacity	Wind speed Temperature
Treatment of manure	**pH** Buffer capacity	Precipation Humidity
Storage of manure	**$CaCO_3$** content	Background concentration
Transport	Soil moisture	
Application technique	Temperature	
Type of industrial N-fertilizer	Urease activity Uptake by plants **Nitrification**	

The extent of ammonia losses depends on many factors (Tab. 1). As demonstrated by Aldag and Döhler (1989), ammonia emission varies between 5 and 95% of the ammonium nitrogen applied with the manure. For that reason it is very difficult for the farmer fertilizing with manure, to calculate the real nitrogen input required for optimum crop management, even when he is analysing the manure.

for the environment

Ammonia losses are not only losses of an expensive plant nutrient, but also damage terrestrial and aquatic ecosystems. Apart from a direct damage of vegetation near strong emission sources, the major part of the NH, volatilizing reacts with the SO, of the atmosphere. The product is ammonium sulphate, which can be transported long distance and reaches ground surface as wet or dry deposition. The atmospheric input of nitrogen amounts up to 30 kg/ha · y for Central European conditions, whereas more than 50% of the nitrogen is usually caused by ammonia emission (Blume et al., 1985; Isermann, 1990).

For oligotrophic ecosystems like forest, heap or the Baltic Sea nitrogen input of such an amount disturbs sensitive nutrient balances of the systems and is implicated in soil acidification by the nitrification of deposited NH, nitrogen (Ellenberg, 1990).

Description of site and methods

The objectives of our research work were to determine the ammonia losses the farmers have to calculate with, and the possibilities they have to reduce losses after the application of manure in the field, in order to practice economic as well as ecological farm management.

Site

Most of the experiments were carried out approx. 50 km east of Kiel (North Germany) near the Baltic Sea on typical luvisols (Ap horizon: sandy loam-loamy sand; pH 6.0-6.5; CEC 93-107 mval/kg). Ammonia emission from solid manure was carried out 40 km southeast of Kiel on cambisols (Ap horizon: loamy-silty sand; pH 4.9-5.5; CEC 60-80 mval/kg; Schleuß, pers. communication).

Methods

The ammonia emission was determined using micrometeorological methods.

The ammonia losses from slurry were calculated with the mass balance method (Denmead, 1983; Ryden and McNeill, 1984), which is based on the relationship

$$F = (1/x) \cdot \int_0^z (\bar{u} \cdot \bar{c} + \overline{u' \cdot c'}) \cdot dz \approx (1/x) \cdot \int_0^z (\overline{u \cdot c}) \cdot dz$$

where F ($\mu g \cdot m^{-2} \cdot s^{-1}$) is the ammonia flux density, x (m) is the radius of the circular application area, u ($m \cdot s^{-1}$) the wind speed, c ($\mu g \cdot m^{-3}$) the concentration of NH, and z (m) the height.

For the flux determination from the solid manure, the Bowen ratio method (energy balance) was used, and corrected with an aerodynamic model (Denmead, 1983).

For both methods, profiles of ammonia concentration were determined on a mast, placed in the centre of each circular plot. Air samples were collected at four heights (up to 4.0 m) with acid traps (80 ml of 2% boric acid) using a controlled air flow rate between 10– 16 1 min^{-1}, provided by a vacuum pump. Continuous measurements were carried out immediately after the application of the manure, for a duration of at least 80 hours. The trapped ammonia was analysed with a Technicon autoanalyser. Meteorological parameters like wind speed or temperature were determined at different heights on a central point of the whole experimental area.

The chemical composition of the manure (pH, dry matter, NH,-N, total N*) was determined in the laboratory (see foot note).

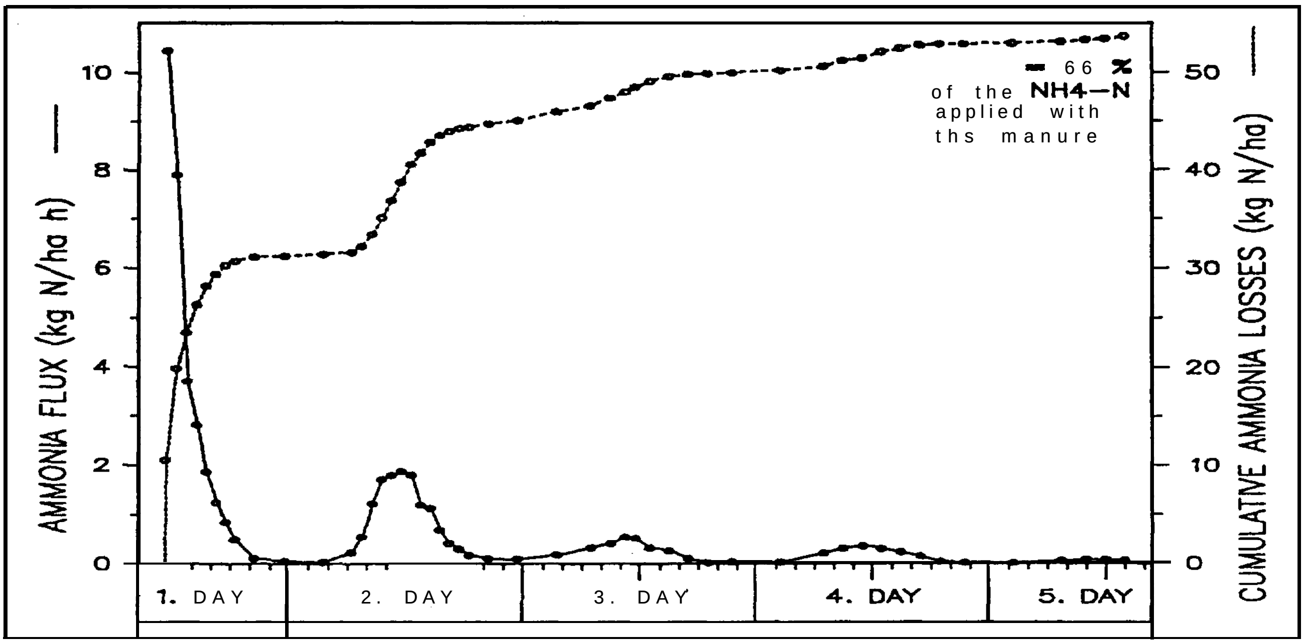
*) A detailed description of methods and results is given elsewhere (anol Bless, et al. 199 1)

Results and discussion

Time course of ammonia volatilization

As Fig. 2 demonstrates, ammonia flux was highest immediately after the application of manure. However the intensity of flux decreased continuously until the evening, because the CO, dissolved in the manure volatilized and the **pH** changed from neutral to alkaline. As a result, the equilibrium between ammonium (NH,-N) and ammonia (NH,) shifted to gaseous ammonia which volatilized. Atmospheric conditions affected this process. The higher the temperature and wind speed, the more CO, and, consequently NH,, was emitted.

Fig. 2:
Typical ***time course of ammonia volatilization after field application***



But as soon as the NH,-N in the liquid phase of the manure had contact to soil particles, it was absorbed by the cation exchange complexes, and the potential for NH, volatilization is decreased.

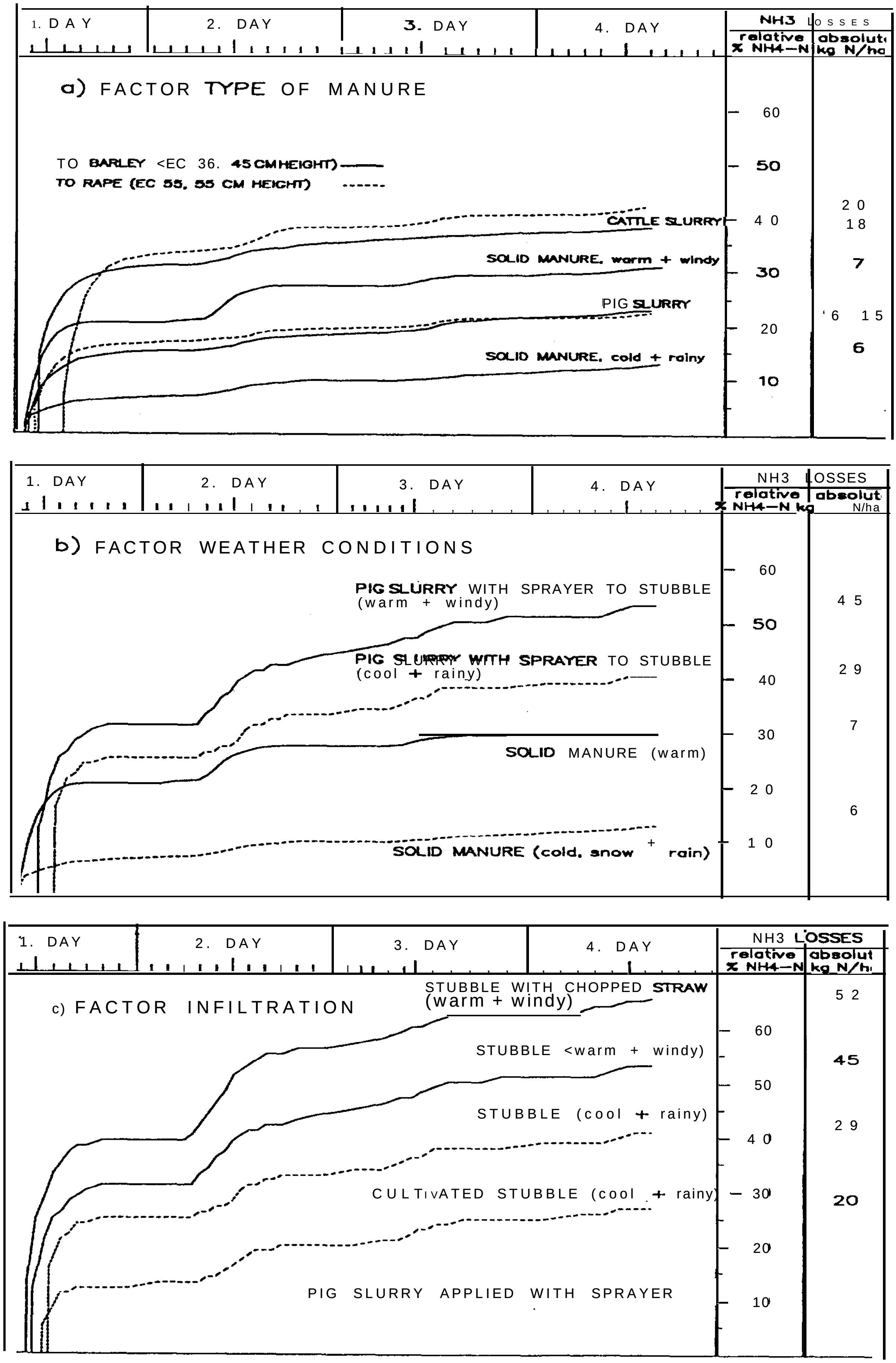
During the night ammonia losses decreased significantly. The next morning volatilization continued again. During the following period peaks occurred at midday, but the losses of ammonia became smaller with time, due to reduced potential of NH,-N in the manure.

Factors affecting ammonia emission

The results of 8 experiments each consisting of up to 3 different treatments are discussed here, according to the factors influencing ammonia volatilization.

The following **graphs** (Fig. 3-4) present the cumulative ammonia losses on the basis of the ammonium nitrogen applied with the manure in the field. These relative losses (% NH,-N) give the opportunity to compare manures of different type, composition and application quantity. Additionally the figures give the absolute losses of NH,-N / ha.

Fig. 3:
Influence of type of manure (a), weather (b) and infiltration (c) conditions on ammonia emission



Type of manure

The type of manure already affects the quantity of ammonia losses after field application. A comparison between cattle and pig slurry and solid manure is shown in Fig. 3a. Both types of slurry were applied with a sprayer systems, to barley during the shooting, and to rape before the flowering during cool weather conditions (8–10°C mean temperature). The relative losses were higher for the cattle slurry due to the higher viscosity and dry matter content. Compared to pig slurry cattle slurry covers the leaves of the crops for a long period and does not infiltrate into the soil as quickly. For that reason atmospheric conditions like temperature and wind increase the volatilization process and force the losses. From cattle slurry relatively more ammonium can be lost as ammonia, even when the absolute emission is not significantly higher compared to pig slurry.

As outlined in the graph, the range of losses of NH, nitrogen in solid manure was similar to slurry, but due to the small amount of ammonium in solid manure the absolute values were much lower.

Weather conditions

In general high losses of ammonia occur when manure is applied during warm and windy weather conditions. Application during a period with low temperature already reduces the emission.

Figure 3b demonstrates that from pig slurry applied with a sprayer more than 50% of the ammonium nitrogen of the slurry can volatilize when the weather was very warm and windy. On the other hand losses from a pig slurry of similar composition applied 2 weeks later on the same site during cool and rainy weather were significantly decreased. The most important factor for the difference described was the weather.

The same results were found for solid manure. Applied during a hot period with a maximum temperature of more than 25°C – but with low convection -the relative losses were much higher as applied on the same site during a cold period (0°C) with snow and rainfall.

Infiltration

The faster the liquid phase of manure where ammonium nitrogen is solved, enters the soil, the earlier the NH,+ anion is adsorbed by soil particles and the potential for further NH, volatilization is diminished.

A direct comparison of the emission from pig slurry applied to stubble with chopped straw, with a slurry treatment to stubble without straw residues, showed that for warm and windy conditions the ammonia losses were increased from 56 to 66% of the NH,-N due to a blocked infiltration of the slurry into the soil. On the other hand improved infiltration conditions as observed when the stubble is cultivated with a disc harrow before a slurry, decreased the ammonia losses compared to a non-cultivated stubble treatment.

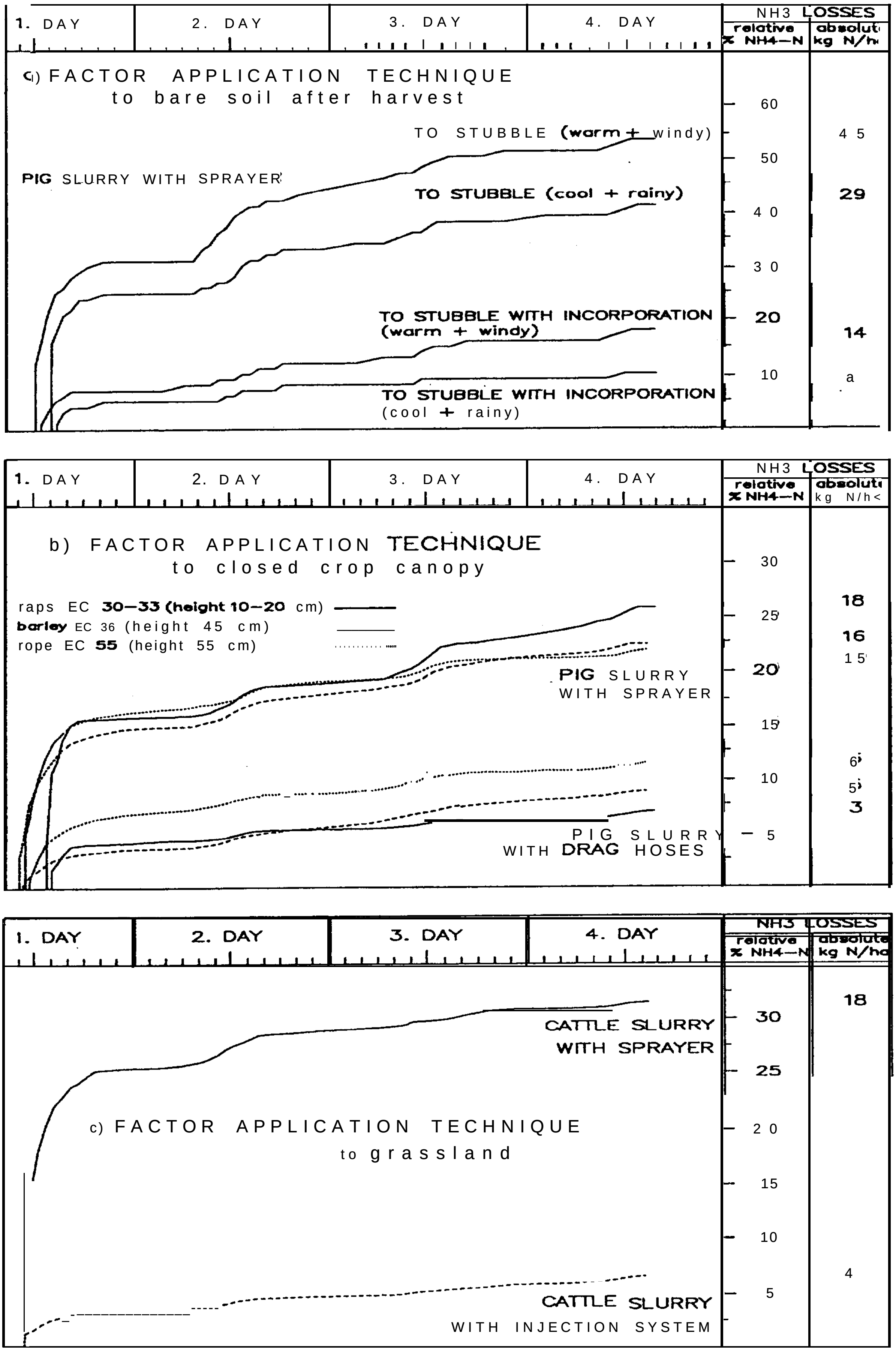
Application technique

to bare soil

Steps to support a fast contact of ammonium nitrogen of manure with the soil colloids reduce ammonia emission. Therefore on soils without standing crop manure can be incorporated.

Two experiments carried out during warm and windy, respectively cool and rainy weather, with pig slurry demonstrate that in general, ammonia

Fig. 4:
Inji'uence of the application technique to bare soil (a), to a closed crop community (b) and grassland (c) on ammonia emission.



emission after the application to stubble can be reduced to 70–80% of the losses when the slurry was ***immediately*** incorporated with a disc harrow (Fig. 4a).

Only an immediate incorporation is efficient to decrease ammonia losses, because all the experiments carried out have shown that 50% of the total losses found after four to five days occurred within the first three hours after application.

to closed crop community

Incorporatiny manure applied to a closed crop community is difficult or even impossible without causing damage. However the efficiency of the ammonium nitrogen supplied with the manure is higher because it can be used by the crops, which decreases the risk of nitrogen leaching after the nitrification. The preconsumption for a good fertilizing effect of manure is that the ammonium reaches the root zone and does not volatilize as ammonia.

Liquid manure is usually applied with conventional distribution system swhich spray the slurry on top of the crop leaves, whereas an alternative system consisting of drag hoses (distance of the hoses approx. 0.3 m) applies the slurry between the crops directly on the ground surface. The ammonium reaches the cation exchange complexes of the soil earlier and the crops protect the slurry from ammonia emission because the forcing atmospheric factors (temperature, wind) do not affect the volatilization process compared to a slurry covering a crop canopy with a crust.

The results from pig slurry applied to rape and barley during different stages of development with the two distribution systems demonstrate that ammonia losses were decreased by a factor of 2-3 when drag hoses were used (Fig. 4b).

to grassland

Apart from drag hoses a costly application technique is discussed for grassland which slits the grass sod and applies slurry directly to a depth of 5-10 cm. This so-called slurry injection system was compared with a conventional sprayer system. In spite of high rainfall 2 hours after the application, which certainly decreased the emission from the sprayer much more than from the injection treatment, the cumulative losses after 4 days show a reduction of ammonia emission by factor of 5 when cattle slurry was applied with the injection system (Fig. 4c).

Summary and conclusions

1. Ammonia emission is mainly caused by agriculture, especially livestock farming.
2. Ammonia emission is a volatilization loss of an expensive plant nutrient and an important stress factor for the environment.
3. The potential for ammonia is the ammonium in animal waste.
4. In the field the highest losses occur a few hours after the application of manure.
5. A large number of factors affect the extent of ammonia emission.
6. In general, ammonia losses after field application can be reduced by choosing favourable weather conditions (low temperature and wind speed, humid period).

7. Ammonia emission can be reduced by taking steps which support the fast contact of the ammonium nitrogen in the applied manure with the soil colloids. Recommended are:

- for the application to bare soil
  - *immediate* incorporation
  - slurry injection
- for the application to crops
  - drag hoses
- for the application to grassland
  - drag hoses
  - slurry injection

Acknowledgement

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A general code of practice to reduce ammonia volatilization from animal husbandry

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Introduction

Ammonia emissions from animal husbandry are recognized to contribute to the

- acidification of soils and surface waters,
- eutrophication of terrestrial and aquatic ecosystems and even
- climatic change

It is estimated that the world-wide ammonia emissions sum up to 28 to 45 million tons per year, and about 90% derives from animal production. The European emission rate is about 8 million tons/yr. In the Federal republic of Germany the calculations and up to about 400.000 to 800.000 t/yr.

In spite of the fact that the above figures are still estimations or rough calculations there is agreement that a reduction of ammonia is desirable not only for environmental reasons but also from an economical point of view.

Consequently, there is a need for concepts and strategies to prevent and control ammonia emissions. It seems useful to base such measures on the knowledge of the

- origin of the compound ammonia,
- physical and chemical nature of the compound,
- mechanisms which promote or prevent volatization, and of the
- main sources.

In the following an inventory is made on the sources of ammonia and on measures and opportunities which are available to reduce ammonia emission from animal production. The aim is to show the effectiveness and the practicability of the different measures.

Main sources of ammonia in animal production

Table 1 shows that there exist distinct differences between the various kinds of livestock with respect to ammonia emission. Cattle produce the greatest amount followed by pig and poultry. The other animals are of less importance. Table 1 further demonstrates the contributions of the different sections of animal production to the ammonia emission. It is estimated that field application contributes to more than 50% to the emission of ammonia from livestock production followed by buildings and stores (37%) and grazing (12%). A differentiation between buildings and stores is difficult because in many keeping systems the manure is stored inside the building.

The consequence of these figures is that reduction measures should concentrate on animal housing and field application.

Influence of feeding

Ammonia originates from the urine and the feces of the animals by bacterial decomposition of urea and uric acid which are nitrogen compounds. High protein supply in fodder increases the excretion of nitrogen compounds. Table 2 gives an example which shows that restricted protein feeding of fattening pigs has no negative influence on the gain.

Table 3 shows the effect of a stage feeding regime on gain and N-excretion in pigs. Best results are achieved with the three stage feeding plus synthetic amino acids. The nitrogen content in the slurry is reduced by more than 30%. However, the synthetic amino acids are expensive (about 10 DM per pig) (Spiekers and Pfeffer 1990). A practical approach is given in Table 4 which shows the reduction af the N-excretion in percent under different feeding practice.

Because of the typical physiology of digestion in cattle is it difficult to reduce the N-content in the feces. However, Table 5 shows that special feeding-rations can influence the pH-value of the manure and consequently the ammonia evolution rates (Kellems et al. 1979).

Influence of housing – temperature and ventilation rate

Bacterial activity depends on temperature. The lower the temperature the lower the activity of the bacteria and the lower the ammonia release. Figure 1 shows the influence of increasing air temperatures on the ammonia losses from buildings. Two mechanisms are responsible for the higher loss of ammonia. One is the higher ventilation rate because of the increased temperature (Figure 2). The second reason is the increased partial pressure of ammonia in the manure with increasing temperature.

The example from Gustafsson (1987) shows that the increase of the air exchange from 2 to 4 results in an increase of the ammonia emission from about 250 to 350 mg/75 kg pig an hour. That are – related to the pig population of the former F.R. of Germany* – about 20,000 t/yr. These figures fit well with former calculations on the ammonia emissions from piggeries (Hartung 1986).

Influence of manure handling and removal systems

Low pH-values and low temperatures prevent the release of ammonia from liquid manure (Figure 4). Up to now there is no common procedure available to adjust the pH-value of liquid manure under practical conditions. Good results are reported when pH-values less than 4.5 are used. Nitric acid is used as acidifier. Problems occur from insufficient mixing of manure, corrosion of materials and the land spreading of the nitrogen rich and acidic manure.

Kellems et al. (1979) found that the volatilization of ammonia from stored bovine urine was much higher than from the feces itself or a mixture of both and the addition of water helped to reduce the ammonia release (Table 5).

The use of bedding to reduce ammonia emission from buildings depends very much on the amount of bedding used and on the maintenance of the bedding. Table 6 shows that bedding can reduce the ammonia release by a factor of two for poultry and by a factor of 4 for pigs. However, these results are of limited value. It is assumed that solid manure will loose most of the ammonia during the process of mucking out and during the storage outside the building. Thus, the total ammonia loss from solid manure does not seem to be considerably lower than that from liquid manure. Vetter et al. (1989) describe the total loss of nitrogen from solid manure with about 20% and from liquid manure with about 15%. The ammonia emissions from liquid manure during storage depend on the duration of storage, the relation of manure volume and surface, temperature, air velocity along the surface, ammonium content, dry matter and floating covers.

The N-losses depend to a high degree on the keeping system and the manure removal. Table 7 gives values for poultry.

Influence of manure storage outside the building

The best way of reducing ammonia volatilization from outside manure stores is covering the surface with tents, solid covers or straw (4-7 kg/m^2) which forms floating covers (de Bode 1990). Reductions between 70 and 90% can be achieved. Lowest emissions are reported when solid manure is humid and anaerob.

Influence of land application

The highest ammonia losses occur during spreading of manure. The most important factors are the temperature and the type of slurry. Below 20°C no more than 30% of the ammonia present in pig slurry is released. From cattle slurry 30% of ammonia are released at 0°C, already. Working in the soil is the best way to avoid ammonia losses. Dilution of cattle slurry with water (1: 1) reduces ammonia losses by 50%. A similar reduction is achieved by separation (Döhler 1990). There exist a lot of technical equipments (tubes, low application device, soil injection) which can help to lower the ammonia emissions.

Other techniques

Schuchardt (1990) reports that there are losses between 3 and 75% during composting animal manure. Large C/N relations (>4 1) and lower temperatures (<50°C) reduce the N-losses. The technique is expensive.

The success of additives in feed and manure is still inconclusive. Numerous positive reports are in opposition to practical experiences. There is a lack of standardized test and evaluation procedures for these commercially offered products.

Discussion

This short overview shows that there is a variety of methods available to reduce ammonia volatilization from animal production. The highest emission come from cattle keeping (64%) and from land spreading (51%). Strategies to reduce the ammonia emission should concentrate on these areas. In Table 8 a draft synopsis is given on the ammonia emission amounts which can be reduced with the different measures and techniques.

It is obvious that all ammonia emissions must be considered losses of valuable fertilizer. Therefore it is necessary to minimize ammonia emissions not only for environmental reasons but also from an economical point of view. However, one must be aware of preventing other environmental problems to occur like nitrate leaching or N_20 emissions while reducing ammonia emissions. This comment is not only related to the emission of ammonia but on the emission of other gases like methane, nitrogen oxide and carbon dioxide which are considered to contribute to mechanisms related to the climate change problem.

All these measures and efforts must include the question of the necesseing to maintain and support the present form of animal production, which is likely to pollute air and water and contributes to the damage of forests and ecosystems. This should also be considered in relation to the present situation of overproduction on a large scale in animal production in Europe.

Grazed pastures
Ammonia emission from grazed pastures is relatively low compared to field emission after land spreading. Field measurements show that the rate of ammonia emission depends on the amorint of N-fertilizer gift as follows:

420 kg N/ha/yr: Emission 25.1 kg N/ha/yr (6.4% of input)
210 kg N/ha/yr: Emission 9.5 kg N/ha/yr (4.5% of input)
0 kg N/ha/yr: Emission 6.7 kg N/ha/yr (4.2% of input)

At a total N-input of 600 kg N/ha/yr approx6–19% of the input is lost due to ammonia emission.

Conclusions

1. Ammonia emissions from animal husbandry are part of the N-cycle in agriculture. 37% of all N-losses are lost as ammonia.
2. Animal production has a nitrogen efficiency of 17% only.
3. Ammonia emissions from buildings and stores amorint to $^1/_3$ of the total emission of ammonia from animal husbandry; about $^2/_3$ are deriving from application.
4. There are distinct differences among the animal species in respect to the emission rate per livestock unit (LU): Cattle is moderate, pig and poultry are high.

Measures to prevent ammonia losses from buildings are:

1. Low pH-value of the manure; lowered by acids
2. Low temperatures diminish decomposition of manure.
3. Bedding can decrease ammonia release if sufficiently used.
4. Removal of the urine from the building is more important than the feces.

5. Drying (ventilation) of poultry manure if applicable.
6. Restricted N-feeding.
7. Feed and slurry additives; its effect is still uncertain.
8. General hygienic rules (e.g. clean and dry pens, ventilation system according to ventilation norms, e.g. DIN-norm 189 10, relative humidity 60 to 80%, animal density not less than 3 m^3/fattening pig).

Future needs

Four typical areas can be identified (area building/stores)

1. General hygienic rules
   - clean and dry pens
   - ventilation system according DIN 189 10
   - relative humidity 60 - 80%
   - animal density (e.g. minimum 3 m^3/fattening pig)
2. Animal keeping systems with lov emission levels
   - drying the manure in the barn (manure removal belt)
   - removal of the manure as quick as possible to an outside store
3. N-restrictive feeding
4. Feed additives

- General aspects:
  - Is intensive animal production necessary in this form?
  - Polluter pays principle

Table 1:
Ammonia emissions in million (m))/kg/yr from different areas of animal ***keeping (6)***

Animals	stalls and storage		spreading		grazing		total	
	m kg	%	m kg	%	m kg	%	m kg	%
cattle	43	53	71	64	27	19	141	64
pigs	20	24	31	28	-	-	51	23
poultry	19	23	9	8	-	-	28	13
total kt	82	111	27	220				
percent %	37	51	12	100				

Table 2.
Comparison of the excretion of crude protein under different protein supply situations in fattening pigs (from Henkel 1989)

Example	protein minimum	DLG recommendation
feed kg	239	241
crude protein:		
intake kg	28.7	35.4
gain kg	13	13
excretion N kg	2.5	3.6

Table 3.
Influence of different feeding regimes (stage feeding, addition of amino acids) on gain and N-excretion (Spieker and Pfeffer 1990). AA = amino acids (Lys, Met, Thr, Trp). L W = live weight.

feeding system	stages			
	1	2	3	AA
20-50 kg LW crude protein (g/kg)	197	197	197	161
50-75 kg LW crude protein (g/kg)	197	182	182	142
75-100 kg LW crude protein (g/kg)	197	182	1264	118
N in feed (kg/pig)	6.6	6.2	6.0	4.6
N in gain (kg/pig)	1.8	1.8	1.8	1.8
N in slurry (kg/pig)	4.8	4.4	4.2	2.8

Table 4.
Reduction of N-excretion by the replacement of feed-stuff compounds for pigs (30-100 kg) in a wheat/bruised soya mixture (=100%) (Spieker and Pfeffer 1990)

feed	reduction (%)
replacement of wheat by oats	20
replacement of soya by peas and methionin	20
replacement of soya by beans and methionin	12
reduction of soya, addition of lysin and methionin	27
reduction of soya, addition of lysin, methionin, threonin and tryptophan	34

Table 5.
Volatilization of ammonia from stored bovine feces and urine (11)

composition of the manure			ammonia release
feces	urine	water	$\mu g NH_3/h$
100			3.15
	100%		426.0
50%	50%		120.0
75%	25%		16.0
75%		5%	3.4
50%		0%	6.6
25%		5%	9.7
5%		5%	2.2

Table 5a.
pH and NH, evolution rates from feces and urine mixtures collected from cattle receiving 75% corn, barley and milo rations

Ration	Avg pH[b]	Avg ammonia[b] (mg/hr)
Corn	7.21' ±.0634	2731.96 ± 304.72
Milo	6.78[d] ±.0839	2602.03 ± 302.56
Barley	7.65' ±.0871	3182.48 ± 241

[a] Samples composed ot 50 g feces +50 g urine from each of the respective groups
[b] Expressed as means of 28 observations ± standard error of mean.
[c, d, e] Means in the same column with different superscripts differ significantly (P<.05).

Figure 1:
Seasonal fluctuation at Barn 1. (A) Aerial temperature (measured at the University of Guelph Arboretum). (B) Atmosphereric NH,-N concentration in the barn. (C) NH,-N loss estimated by atmospheric sampling. (D) NH,-N loss estimated from the N mass budget.

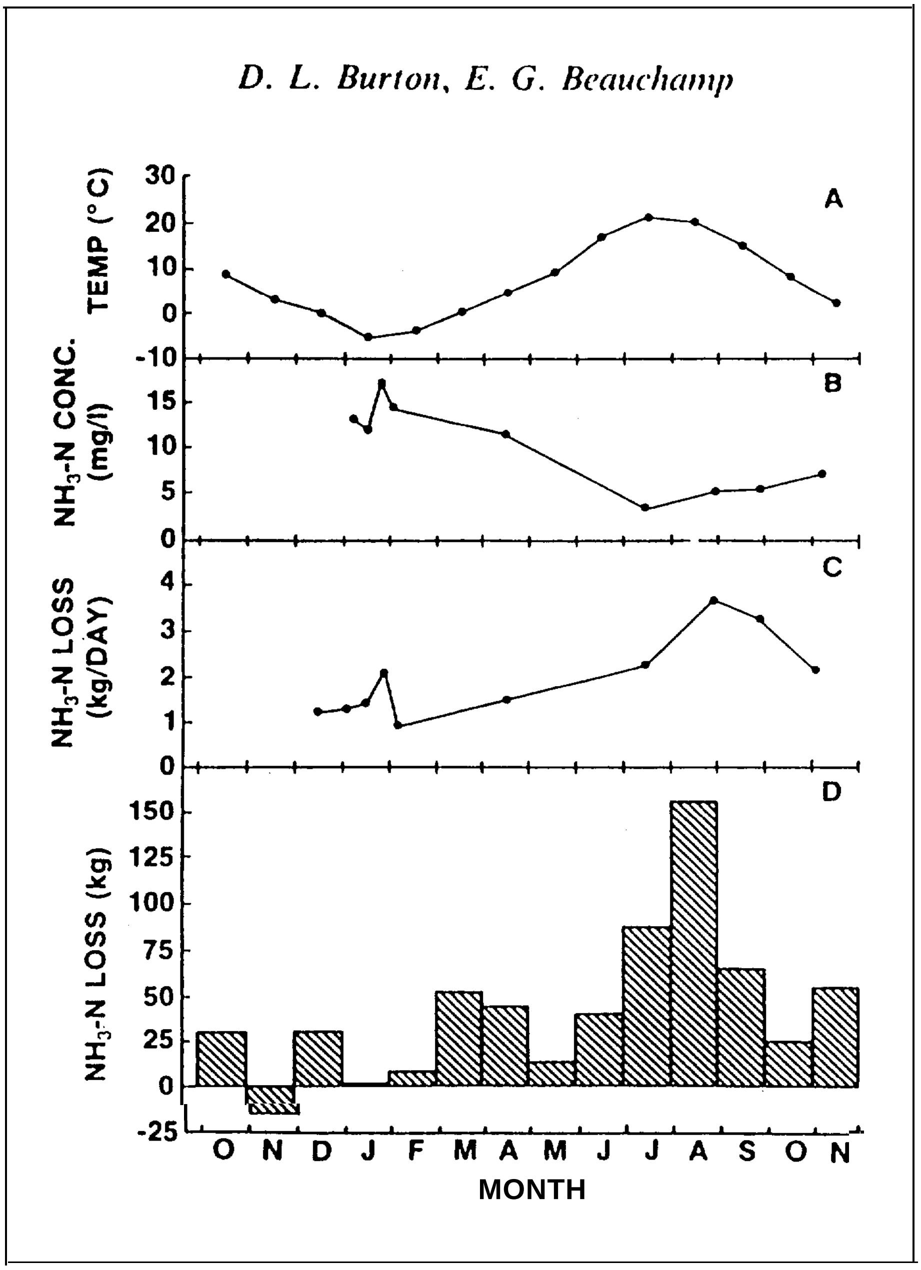

Figure 2:
Ammoniakverluste in Schweineställen in Abhängigkeit der Luftwechselrate und des Reingewichts (nach Gustafsson, 1987)

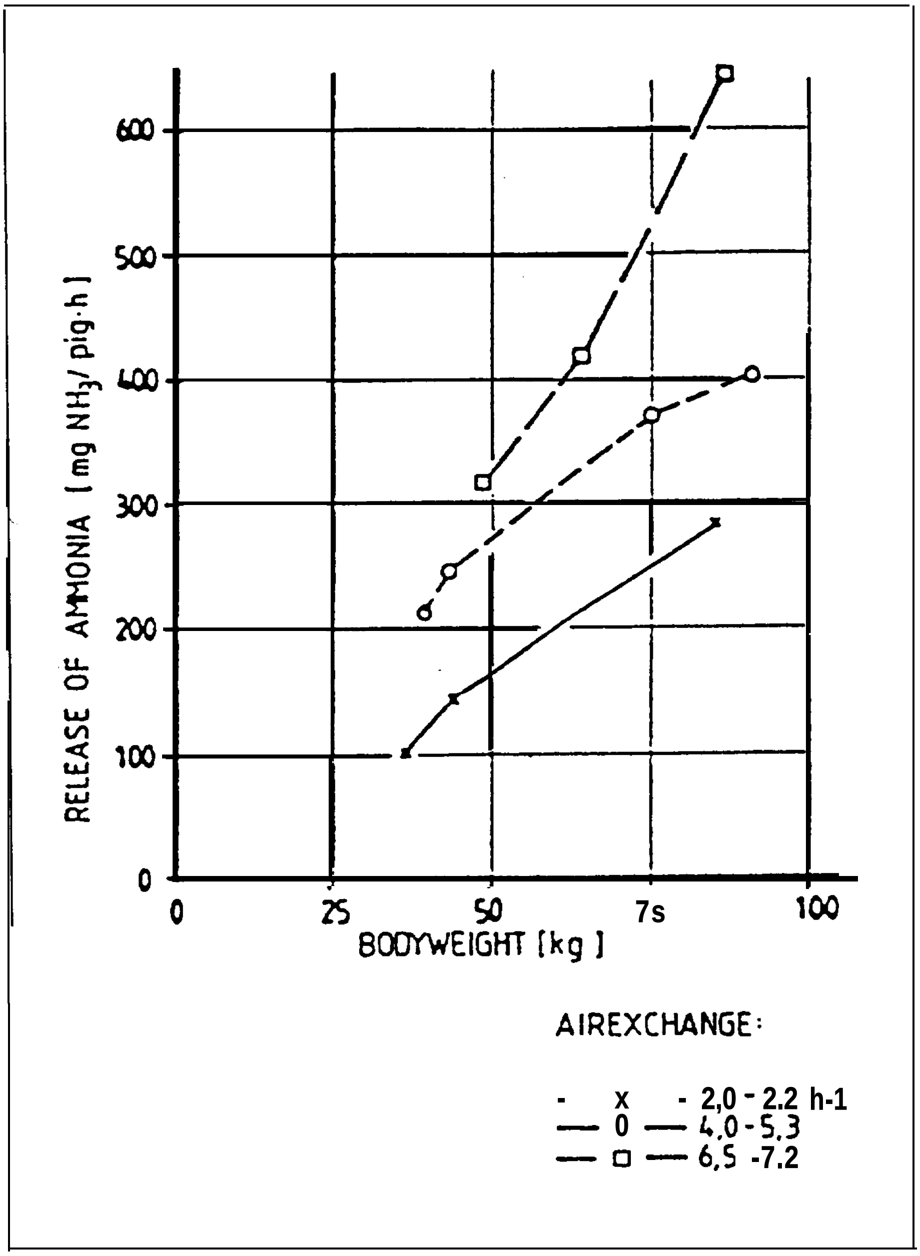

Figure 4:
Anteil des freien Ammoniak (Fraktion-NH$_4$) an der Ammoniak – Gesamtkonzentration in Gülle in Abhängigkeit von pH- Wert und Temperatur (°C) (Nach Miner 1974)

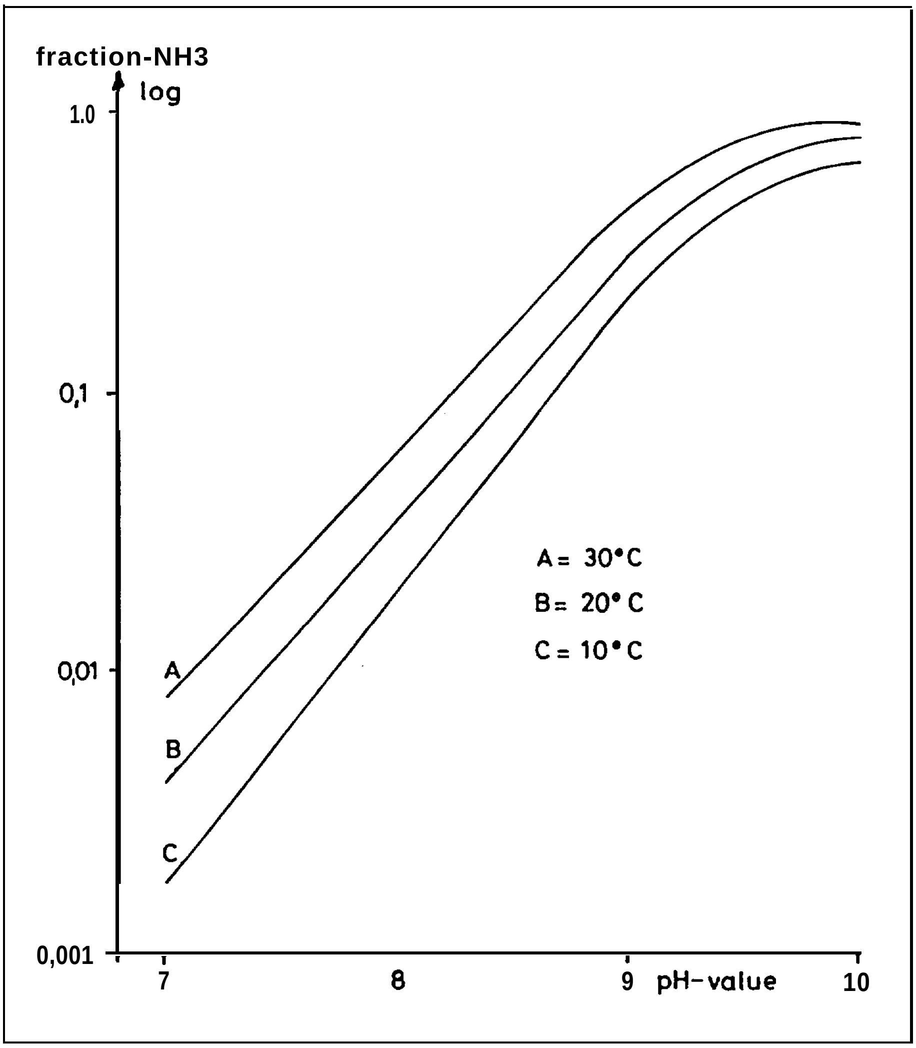

Table 6:
Ammonia emissions from different livestock and different keeping systems (BEF 1987)

livestock	keeping/manure system	ammonia emission g NH_3/h	kg NH_3/year
poultry	liquid manure	4.4	38.8
poultry	bedding/floor	1.9	16.7
pig	liquid manure	0.9	7.5
pig	bedding	0.2	1.7

Table 7:
N-losses in different keeping and manure handling systems in poultry (Priesmann et al. 1990)

System	manure storage time	N-loss (%)
floor with bedding	54 weeks	62
cages	7 days	13
cages with drying	7 days	1.3
liquid manure	6 months	29
broiler	37 days	28

Ta ble8:
A draft calculation of the possible reduction of ammonia emissions from different area of animal production

Restricted protein	feeding (pigs)		total
Total NH,	estimated	reduction	220 kg/yr
51 mkg	20–34%	10-17 m kg	- 15 m kg
land application			
71 kg cattle	50%	35 kg	- 35mkg
31 m kgpig	70%	22 m kg	- 22mkg
storage tank covers: efficiency 70–90%			
losses 3% poultry	about 0.6 m kg		- 0.5 m kg
losses 15% pig	about 2.4 m kg		- 2.0 m kg
manure drying under poultry cages			
about 10 m kg reduction 90%			- 9.0 m kg
		(total reduction about 38%)	83.5 m kg

Ammonia emissions from agriculture as a component of its nitrogen balance and some proposals for their adequate reduction

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Introduction

In the face of possible and actual environmental influences across national boundaries, the necessity for territorial, continental and global balancing of the nutrients carbon, nitrogen, sulfur and phosphorus and their environmentally relevant compounds already exists today but will become increasingly urgent in the future. As Table 1 illustrates, these cross-boundary environmental influences include

- long-range climate changes
- detrimental effects on sensitive terrestrial and aquatic ecosystems (such as acidification) and on buildings due to air pollutants (corrosion) (such as »acid rain«)
- eutrophication (hypertrophy) of sensitive terrestrial and aquatic systems.

The NH, emissions from agriculture of primary interest here, and only of secondary interest also those from other economic sectors (industry, transportation, energy), can very well influence landscapes and their users (forestry, nature preserves, water management areas, fisheries, tourist areas) even though they are far distant from the emission source. This is due to the direct and indirect effects of NH_3 or other subsequent N compounds (NH_4+, NO,-, NH_4NO_3, NH_4HSO_4, $(NH_4)_2SO_4$) arising from the long residence time just for NH,, for example, of 5 to 9 d (compared to the NO_x residence time of 1 d; Crutzen 1983, Wameck 1989). If these effects are adverse ones, however, they become a problem for both those affected and those causing them. As also can be seen in Table 1, such adverse effects stemming primarily from NH, emissions from agriculture are being debated in connection with:

a) the impairment of sensitive terrestrial (e.g. »new types« of forest damage) or aquatic ecosystems (e.g. acidification of soils or waters) (Isermann 1983, 1986, 1988; Nihlgard 1985)
b) building corrosion
nitrification of NH,+ n nitric acid along with NO_x and SO,) (see presentation by Sand, Hamburg)
c) the eutrophication of sensitive terrestrial and aquatic ecosystems (inland waters and especially N-limited coastal waters) (Isermann 1990a, b, c).

In the past agriculture generally took an economic view of the N losses from farmyard manures by determining degrees of utilization and then compensating by increased application rates. This accounts for such expressions as, for example in animal nutrition, »margins of safety«, or in plant nutrition (fertilization) »**mineral** fertilizer equivalentsa or »**active**« (e.g. ***only*** the NH,-N content of the liquid manure is counted) or »**inactive**« shares. The fact that these »**inactive**« N losses of agriculture actually do have an effect in other ecosystems or on lands managed for other purposes was first faced in the response of drinking water sources (nitrate problems), and more recently in that of forestry (»**new**« types of forest damage) and fisheries as well as the tourism sector (eutrophication of surface waters). In the future the response will come **from** the entire economy (changes in climate → greenhouse effect). Up until now the gaseous N losses from agriculture (ammonia losses and denitrification losses) served as an anonymous relief valve for the pollution of groundwater and springwater with nitrates. This situation now requires a different, interdisciplinary look, as well as integrated **problem-**solving proposals reflecting all the involved economic sectors and national economies. These proposals do offer an advantage to agriculture, because the avoidable N losses and especially the corresponding NH,-N losses in agriculture are also extremely significant economically, providing strong motivation **from** this viewpoint as well to promote prevention of such losses.

Ammonia emissions in different sectors of agriculture

NH, emissions in agriculture occur in the following sectors, listed in decreasing order of significance,

- livestock husbandry:
  - a) collection, storage, distribution of liquid and solid manures
  - b) pasture husbandry
- mineral fertilization
- fertilization with sewage sludge
- soils and beneficial plants

Ammonia emissions **from** livestock husbandry

With a view to the causes, the number and density of livestock populations but also their feeding influences the extent and density of the NH, emissions from the livestock sector. The manner in which liquid and solid manures are collected, stored and distributed **affects** these emissions only as a consequence.

Influence of feeding on ammonia emissions in livestock husbandry

The origin of NH, emissions in livestock husbandry is primarily the urea in the urine (cattle urine contains about 93% urea, while faecal matter contains only 25% soluble forms of N; Whitehead et al. 1986) or else the resulting NH,+ in solid and liquid farmyard manure. Excessively protein-rich feeding with feeds of too-high protein content (complete feeds) generally results in increased NH, emissions in the cases of feeder pigs above 80 kg (**Gädeken** 1986) and especially ruminants. In contradiction to the physiology of ruminants, protein-heavy feeding (with **3.5–4.0%** N instead of 1% N) results in N excretion in urine and faeces at a ratio of 4: 1, compared to a ratio of 1: 1 in the instance of the correct protein content but inadequate efficiency

(Whitehead et al. 1986). This is clearly demonstrated with the extensive and intensive pastures in Table 2, whereby pasture with 3.0% or 4.4% N produced urine/faeces ratios of 2.4 or 3.8 on an N basis.

Moreover, excessively protein-rich feeding effects an increase in the N level of the urea, which apparently has an even greater influence on the NH_3 emission than the urea volume (Vertvegt and Rutgers 1988). The immediate effect of protein feeding on the NH_3 emissions during the pasture season is shown in Table 3.

From excessive to intensive to extensive pasture management the animal-related NH_3-N emissions fall from 12.9 to 10.0 to 8.6 kg/cow, correlating with the N uptake via supplemental feed as well as with the N level of the pasture grasses of 4.4% to 3.5% to 3.0%. The ecologically much more relevant NH_3-N emission (density) per unit area likewise drops from 45 to 35 to 16 kg/ha. This level in the extensive system is brought about by halving the grazing density and consequently also the milk production per unit area. If the farm aims for the same level of livestock production under extensive management as under intensive management, however, the NH_3 emission on double the area will be just as high, but with the advantage of just half the emission density. The choice between the intensive or the extensive management systems will require not only an ecological evaluation, though, but also an economic one, with the result being a compromise. In the end the ecological demands must be borne by economic considerations on a farming operation. Under these extreme conditions, in trying to achieve maximum milk productivity (per cow and especially per ha farmland) the dairy cow as a ruminant (that actually should be largely making its »own« protein from carbohydrates with the aid of bacteria) is being fed almost as a pig (monogastric), to the extreme of feeding slaughter offal in Great Britain. The same is true regarding feeding with carbohydrates: the ruminant has the physiology to digest crude fiber, but not high-quality carbohydrates such as the starches in feed concentrates. This means not only the monogastric animals among the domestic animals are competing with humans for such high-quality carbohydrates, but now also the ruminants, making the conversion of feed to food increasingly inefficient.

Ammonia emissions during collection, storage and distribution of liquid and solid manures

Here a look at the influence of the different waste removal systems and hence of the different types of livestock keeping is necessary, including

- rotting manure (hot fermentation)
- deep stable manure (cold fermentation)
- slurries of varying levels
- pasture farming (see 1.1.2.4)

Contrary to rotting manure and slurry systems, deep stable manure systems collect and store manures on the same spot, namely in the stable. For this reason the NH_3 emissions in deep stables must be compared with the NH_3 emissions in the stable *plus* during storage (manure pile, slurry container) for rotting manure and slurry systems.

Sauerbeck (1985) conveys, independent of the waste removal system, an idea of the type of N losses that occur and their wide fluctuation (Table 4), though without singling out those just in the stable area. Also, it is doubtful whether it would be possible to completely avoid N losses (emissions) from farmyard manures during their distribution and in the soil, as implied in Table 4.

Ammonia emissions in the stable area

Studies by Flieg et al. (1939) already showed that in the rotting manure system the urine loses up to 20% of its total nitrogen during leaching and an additional 25% while traveling through the slurry gutter, depending on the time of year and the age of the straw mat. The situation must be similar for slurry collection in the stable area, but not à priori for the deep stable manure system: as results from Kirschmann and Witter (1989) in Table 5 show, the NH, losses during aerobic rotting-manure storage (in the stable and on manure heaps) are about 50 times higher (depending on the straw content) than during anaerobic deep stable manure storage. The advantages of deep manure storage in this respect compared to rotting manure storage become ever more pronounced the lower the content of straw.

Thus the old tenet continues to be true, that moist and tightly packed manure is the best manure. Because of the length of time the deep stable manure remains in the stable, however, in the end the NH, emission levels of all three manure systems (rotting manure, deep stable manure and slurry systems) are roughly equal. This conclusion is also reached in the study by the Bundesamt **für Ernährung** und Forstwirtschaft (German Ministry for Nutrition and **Forestry)/ABEF** (1989).

The differences are greater in the NH, emissions in the stable area, though, with respect to the types of animals involved, as is illustrated in Table 6.

As shown in Table 6, the average NH, emission in the stable for all manure systems and larger animal types is calculated at 8.0 **kg/unit.a.**

Ammonia emissions during manure storage and processing

Deep stable manure systems

In the case of deep stable manure systems, the NH, emissions listed above for the stable area include those for storage.

Rotting manure systems

Kowalewsky (198 1) and in Table 7 also Eerden et al (198 1) found that when solid and liquid manure is stored in the stable higher levels of NH, are released in the stable interior than during storage outside the stable. Drying the manure results in raised NH, emissions (ammonia stripping).

On the basis of the results given in Table 8 and Figure 3 (see Section **1.1.3.1),** the Ministexie van Landbouw en Visserij (1985) estimates the gaseous N losses during rotting of cattle manure to be **10–20%** (dense storage) and 14-23% (loose storage) depending on length of storage (2 to 6 months), relative to total N.

Slurry systems

Vetter et al., (1989) assign an overall amount of 20% to gaseous N losses during rotting manure storage, while in the case of the amount is 15% of the total N. The ABEF (1989) lists NH,-N losses of about 10% during lengthier slurry storage. The Ministerie van Landbouw en Visserij (1985) lists in Table 8 and Figure 3 gaseous N losses of only **5–8%** of the total N during slurry storage. These figures apply only for open slurry storage containers, however. The NH, emissions are practically zero in the case of closed or covered storage containers. With open storage containers the emissions are influenced by a number of factors, such as

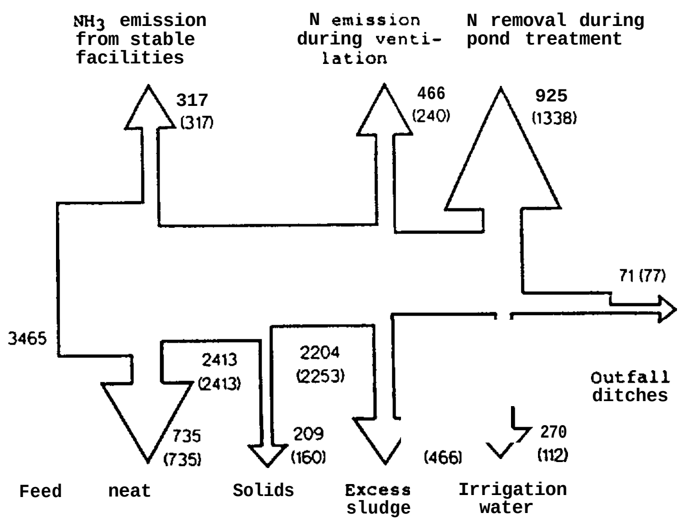
- length of slurry storage
- ratio of slurry volume to surface area
- temperature and wind conditions
- ammonia and dry matter content of the slurry
- slurry type (formation of swimming layers)

N losses of about 10% can occur during lengthier storage of slurry. These losses are to be classified as gaseous ammonia emissions (ABEF 1989). Furthermore, the statement of the ABEF (1989) can be inserted here:

»The ***processing*** of the slurry leads to increased ammonia emissions dependent on the degree of contact with air and warmth. The type of slurry involved plays an important role as well, however. If the slurry is aerated, additional gaseous N losses occur depending on the intensity of aeration. It hasn't yet been investigated whether higher ammonia emissions occur during storage of an anaerobically fermented slurry than during the storage of an unfermented slurry (Vetter 1986). In general, though, only relatively little ammonia escapes from unaerated slurry because of its high water content. This is also true for slurries used to produce biogas, even though here there is a particularly high percentage of the total nitrogen in the form of ammonium because of the biological fermentation process (Sauerbeck 1986).«

Emissions are especially high from »processing methods« that involve the ***submerged storage of slurry*** (lagoon system, pond treatment). In these cases slurry with or without previous aeration is left in submerged storage areas with or without liners often for over a year of free evaporation of water and NH,. This measure frequently even is considered by the farms to be environmentally supportive because the declared goal of reduced outfall pollution is indeed achieved; instead, however, the atmosphere is polluted with reactive N compounds (NH, NO_x, N_2O). This omethod for simplified storage of liquid organic manures« (Bölke 1990) is employed in places with industrialized livestock production without land restrictions, such as in the (ex-)GDR (pig fattening facilities in Neustadt/Orla or Eberswalde) or in Italy (Bonazzi 1987). The N balance of such a pig fattening operation (Neustadt/Orla) is presented in Table 9 and Figure 1.

Figure 1:
N flow in the feeder pig facility Neustadt/Orla in 1988 (1989); data in t/a; stock level: 175,026 (169,759) pigs (Kehr 1990)



According to this, 63%, or 69 kg N/livestock unit.a, of the N emissions are »disposed of« into the atmosphere, with »pond treatment« accounting for 37 kg N/unit of this. The environmental damage (damaged forests, polluted land and water) thus branches out into further spheres influenced by these disposal facilities. Likewise to be rejected are related recommendations for intensive aeration (Hansen and Isensee 1988), for drying the manure (Eerden et al. 1981; see Table 7) and even for oclean-up recommendations« by Baader (1989) to remove ammonia by blowing air into warmed or alkalinized (e.g. by adding lime) slurry, without any way of recovering it.

In summary for all types of manure disposal and types of animals and under consideration of the above results, the average NH,-N emission during the storage of liquid and solid manure is placed at 10 kg/unit.a.

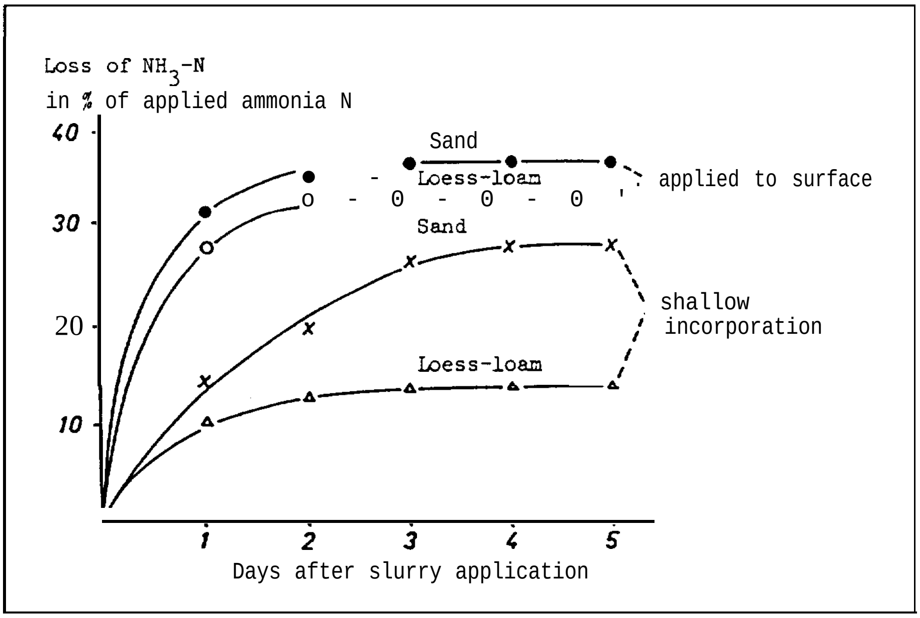
Ammonia emissions during the distribution of liquid and solid manure

Compared with the NH, losses during collection and storage of farmyard manures, those incurred during their distribution are the most significant. Table 10 reviews the extent of these NH, losses under the influence of various environmental factors and various distribution methods, fluctuating from ~ 1% (injected or drilled slurry) to < 100% (liquid manure onto straw) of the applied NH,-N.

In the following Table 11 especially those influential factors are emphasized again that are decisive for NH, losses during the application the slurries.

Special note should be made of those primary factors of influence which the farmer can utilize directly [time factor, dry matter content of the slurry (dilution), incorporation, infiltration capacity of the soil (work up the soil before application), avoiding straw and stubble residues] or indirectly (weather conditions) to minimize the NH, emissions as much as possible. Contrary to the view of Vetter et al. (1989) that the nitrogen losses during application onto the field need not be considered in determining the amount of nitrogen to be applied via farmyard manures, Horlacher and Marschner (1990) included them as part of the fertilization basis in the outline presented in Table 12. If the best possible N utilization of the farmyard manures is lower on the site in actual farm practice and the NH, losses larger, this must

***Figure** 2:*
Ammonia losses after applying cattle slurry on sand or loess-loam soils (Amberger et al. 1987)



be absorbed fully by the farmer and for economic as well as ecologic reasons may not be compensated for by corresponding increases in farmyard manure applications.

Because of the overriding importance of the factors time and incorporation, the extent of their influence on NH, emissions during the application of slurry is illustrated once again in Figure 2.

The results in Table 13 illustrate furthermore that especially in winter the NH, losses during the application of farmyard manure must be joined to the denitrification losses to give a true picture of overall gaseous N losses. This may also raise doubts about efforts made to limit NH, emissions by incorporating or injecting the slurry, since these measures may promote denitrification to such an extent that the emissions then equal the original NH, emissions with the restricted N_2-N/N_2O-N ratio (see Rheinbaben, VDLUFA-Kongress 1990; Confort et al. 1990, in Soil Sci.Soc.Am.J. 54, pp. 421-427).

So far, targeted measures to reduce NH, emissions during the application of farmyard manures have been the exception, not the rule in farm practice. Therefore, under consideration of the above results, an average NH,-N emission during the application of solid and liquid manure of 18 kg/unit.a is assumed here across all types of manure distribution and animals. This corresponds to about 37% of the applied NH,-N slurry equivalents (approx. 22% of the slurry's total N), keeping in mind that, for example in the Federal Republic of Germany, about 60% of the stable manure is applied as slurry, about 40% as solid manure (Krüll 1987).

Ammonia emissions during pasture husbandry

Pasture husbandry with ruminants takes center place here. The NH, emission arise

a) primarily from the animal excreta, but also
b) from the growing plants
c) from dying plant residues and
d) during the use of stable and mineral fertilizers

Ammonia emissions during pasture use

Table 14 gives a review of the NH, emissions from animal excreta during pasture grazing.

Between 75% and 95% of the N consumed via their fodder is excreted again by ruminants, whereby about 80% of the excreted N is in the urine, when the fodder contains 3.5% to 4% N (Ryder and McNeil1 1984, Whitehead et al. 1986). The risk of NH, emission is greater for urine than faeces: approx. 40% of the urine N and only 5% of the faecal N are subject to NH, emission (Buijsman et al. 1986). This underlines the important role of intensive pastures with regard to NH, emissions from livestock husbandry. As shown in Table 14, depending on N fertilization and management (compare perennial ryegrass with grass-clover) the NH, emission data fluctuate between 7% and 40% of the urine N and between 7.6% and 30% of urine and faecal N. If, as in Tables 3 and 12, it is assumed that only about 13% of the urine and faecal N are subject to NH,-N emission, the result is an NH,-N emission of 8.6 to 12.9 kg NH,-N / unit. 180 d., depending again on management (perennial ryegrass/grass-clover) and N fertilization. On top of this come NH, emissions from dead vegetation of about 3 kg NH_3-N/

ha. 180 d. or about l-2 kg NH,-N / unit. 180 d. (Vertregt and Rutgers 1988), so that pasture use produces an NH,-N emission of approximately 1 O-14 kg / unit. 180 d.

Ammonia emissions during the application of stable manures on pasture
Data on the application of slurry onto grassland is given in Table 10. If stable manure is also applied to pastures (farms with absolute grassland), assuming an average NH,-N emission during the application of about 50% of the 22 kg NH,-N / unit (^= 4 1 kg total N / unit) contained in the cattle slurry of 180 stable days, the result is an extra approx. 11 kg NH,-N emitted on the pasture. Under average conditions such as these, however, it can be assumed that at least 100 kg mineral fertilizer N are also applied, again subject to approx. 2% NH,-N emission (see Section 1.2), adding up to an additional NH,-N emission of about 2 kg NH,-N / livestock unit.

This means for cattle pasture (as portrayed in Table 15) the total NH, emission is approx. 23-28 kg/unit.a ifthe stable manure during the time the cattle are kept in the stable is returned fully to the pasture (absolute grassland management). It is approx. 14-l 9 kg/unit.a if the pasture is fertilized solely with about 200 kg mineral fertilizer N/ha.a. In this respect the NH, emission during pasture husbandry is almost on a par with that from year-round stable use.

Ammonia emissions from livestock husbandry in total

Animal-specific ammonia emissions
Cattle and pig husbandry largely determine the total extent of NH, emissions for Western European livestock husbandry. Figure 3 illustrates once more with its results the entire range ofvariation in the N losses and the N efficiency during the collection, storage and application of farmyard manures.

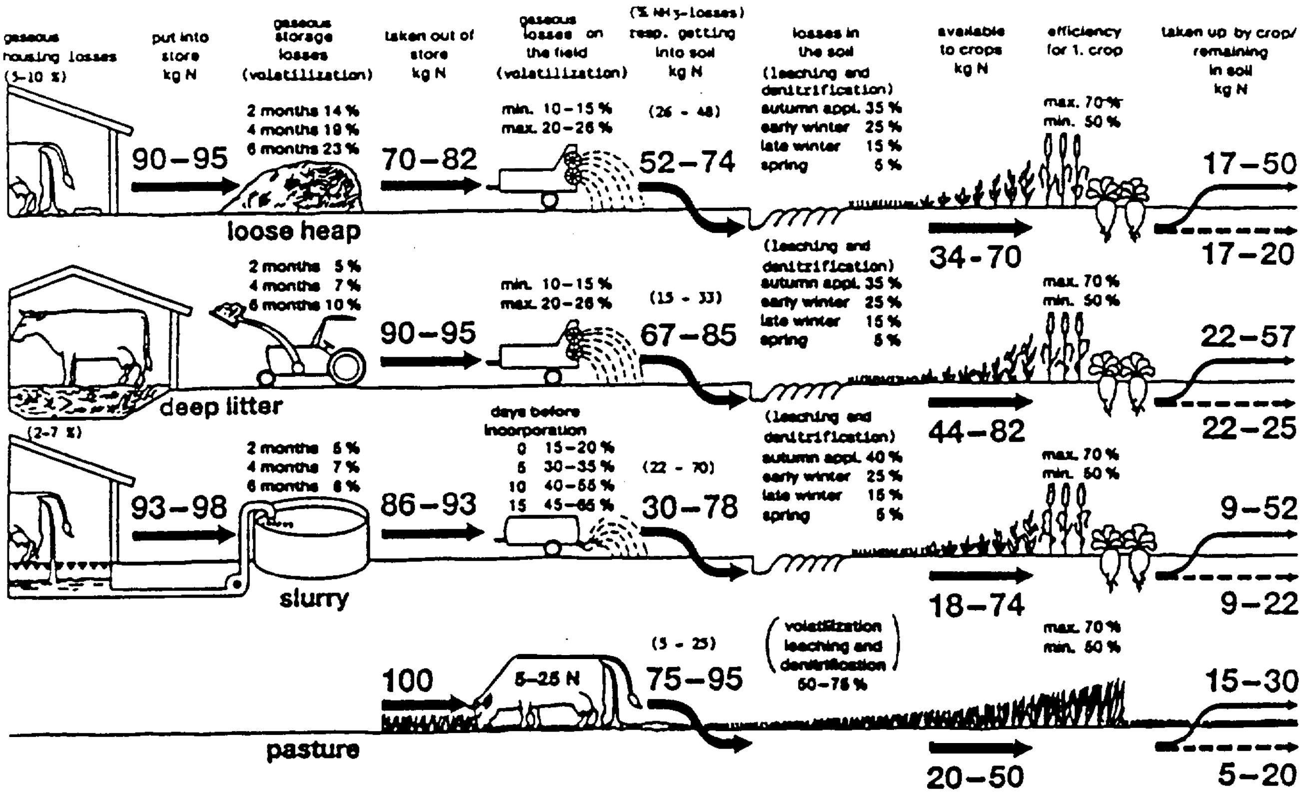


Figure 3:
Nitrogen ***efficiency of organic manures depending on losses during storage and application (per 100 kg N excreted by animals) (revised after Ministerie van Landbouw en Visserij 1985)***

According to the data in Table 16, again an overall NH,-N emission of 36 **kg/unit.a.** is calculated, with an N excretion of 100 kg **N/unit.a.**

Bach 1987) assumes the same animal-specific NH, emission when, **with** reference to Ruppert et al. **(1985),** he uses as a basis an N excretion for a cow of 90 to 120 kg N/a but calculates that only 64 kg **N/ha.a** actually end up on the field. Vetter and Steffens (1986) also estimate for a dairy cow with milk production of 4500 kg/a (ave. FRG in 1987: 4600 kg/a) an N excretion of 270 **g/animal.d =** 99 kg N/animal.a. In comparison, a dairy cow in Holland (ave. 1987: 5200 kg **milk/cow.a)** with the local feeding **practices** excretes 135 kg N/a **(Jongbloed** et al. 1985, **Lenis 1989),** with fluctuations just during the period in the pasture of 65 to 99 **kg/180** d depending on the intensity of pasture management (see Table 3). In the case of feeder pigs as well, **Lenis (1989),** for example, established an annual N excretion of 90 and Kehr (1990) one of 109 kg N/7 feeder pigs (see Table 9). The NH,-N emission listed above of 36 kg per 100 kg excreted N concurs with corresponding figures from the Netherlands of 32 kg NH,-N (see Table 16) or from Norway of approx. 58 kg gaseous N emissions from animal keeping per 100 kg excreted N **(Bøckman** et al. 1990). The animal-specific NH,-N emissions quoted here are about $^1/_3$ higher today than Buijsman et al. (1985) or Janssen (1985) said they were at the beginning of the **1980s,** although **Böttger** et al. (1978) and Cass et al. (1982) by then had already reported an annual NH,-N emission for cattle of 23-33 kg or 42 kg, resp. **Möller** (1990) also finds it necessary in her latest reports on NH,-N emissions from livestock husbandry in the (ex-)GDR to raise by emission factors of 1.5 and 2.0, resp., the **animal**-specific NH,-N emissions compared to the old data from Buijsman et al. (1985) for cattle and pigs. The same was done by **Asman** et al. **(1988),** who conclude that the reason for today's higher animal-specific emission levels is that the live weight of cattle has increased, the N content in the feed has increased, and storage conditions for farmyard manure have worsened.

Assuming the 36 kg NH,-N/unit.a given above, and assuming for example the maximum animal density permitted by agricultural legislation in the Federal Republic of Germany (1989) of 3.0 or (for cattle) 4.5 units/ha, we arrive at a maximum possible NH,-N emission density of approx. 110 to 160 NH,-N/ha farmland. The actual NH,-N emission densities primarily caused by livestock thus lie between 50 and 170 kg **NH_3-N/ha.a** in regions with pronounced livestock husbandry in Western Europa (e.g. de Brabander and Labeau 1987, **ApSimon** et al. 1987). Point sources (livestock facilities) achieve extreme emission densities of **>2000** kg **NH_3-N/ha.a (Möller** 1990). Measured emission densities on intensive pastures lie between 40 and 100 kg **NH_3-N/ha.a,** while on extensive pastures (unfertilized natural pastures) they are < 1 to 10 kg **NH_3-N/ha.a** (pure pasture operation) (Whitehead et al. 1986, Vertregt and Rutgers 1988).

Ammonia emissions from livestock husbandry as a whole in individual nations

Buijsman et al. (1985) were the first to see the need to collect such figures for individual countries in Europe, for the year 1982 (Table *17).*

The German Ministry of Agriculture (ABEF 1989) found a similar level of emissions in 1988 as had been found by Buijsman et al., since it used the same (low) animal-specific emission data. These animal-specific NH,-N emissions, however, not only were already underestimated by Buijsman et al. (1985) back in 1982, but they have increased even more in the meantime, so that livestock husbandry now faces a situation such as that described for several representative countries in Table 18.

Comparison of nitrous oxide and ammonia emissions from ***motor vehicles and livestock husbandry in the Federal Republic of Germany***

Table 19 presents such a comparison.

If it is assumed (Table 16) that each animal has an NH,-N emission of 36 kg/a, then such an animal emits about twice as much NH,-N as NO,-N is emitted by an average truck, or about $3^1/_2$ times as much as an average automobile. In total, approximately just as much NH,-N (500 000 t/a) is emitted into the atmosphere by agriculture's livestock as NO,-N by the country's trucks (548 000 t/a).

Ammonia emissions during the use of mineral fertilizer

As regards the NH, emissions stemming from fertilizer production. The UBA (cited in ABEF 1989) quotes a level of 4200 t NH,-N/a for the nitrogen fertilizer industry in the Federal Republic of Germany. Our own research reveals a reduction of these ammonia emissions in Germany's fertilizer production during 1989 and 1990 down to 4000 and 1900 t NH_3-N/a, resp., primarily due to the shutting down of production sites.

NH, emissions also occur, though, during the application of mineral fertilizers. These fluctuate widely depending on

- Type of fertilizer
- application method
- soil properties, especially pH, content of free lime, compaction, soil temperature, etc. (Kniippel 1988, Amberger 1990)

The following Table 20 gives a review of the ***average*** NH, emission rates as a function of the fertilizer type, that may fluctuate between >0% to 15% of the applied fertilizer N.

Accordingly, the emission rates for the individual fertilizers increase in the following order:

calcium ammonium nitrate (CAN) ~ multinutrient fertilizers (MNF) ~ diammonium phosphate (DAP) < urea (UR) -calcium cyanamide ($CaCN_2$) ~ ammonia gas (NH,-N < ammonium sulfate (AS).

The addition of nitrification inhibitors (such as DCD) may elevate even further the NH, emissions of alkaline fertilizers (such as urea) unless they are incorporated immediately after application (Rodgers 1983, Prakasa Rao and Puttanna 1987).

Overall in the Fed. Rep. of Germany, however, primarily those nitrogenous fertilizers are used (CAN, MNF) that exhibit a low NH, emission rate, as can be seen in Table 2 1. Table 2 1 tries as well to arrive at an average NH, emission rate for the area of the Federal Republic of Germany during application of all mineral fertilizers.

The average emission rates lie between 2.3 and 4.5 kg NH_3-N/ha.a, depending on author and year, with an average emission factor of approx. 1.9% or 3.4% of the total N.

Ammonia emissions **from** soils and plants

Ammonia emissions **from** soils

Little data is available on this. **Böttger** et al. (1978) estimate on the basis of what little data there is that ammonia production by natural ***unfertilized*** soil is 1 ug NH_3/m^2 per hour, which would amount to approx. 72 g *NH_3-N/ha.a.* ***The*** NH, emissions from plants are of more significance, especially those from plants with a high nitrogen supply.

Ammonia emissions from plants

Here again there are no data available on NH, emissions from unfertilized plants, so that no figures can be given in this respect for the ***natural ecosystem.*** As a result, then, no natural background level can be given. There is quite a lot of data, however, on ammonia emissions **from** useful plants, including under the influence of management measures.

Ammonia emissions from intact useful plants

Useful plants, and also weeds, emit considerable quantities of gaseous N compounds, of which approx. 8% to 28% are oxidized N forms (NO_x and N_2O) and approx. **3/4** are reduced N (NH,, volatile **amines**) (da Silva and Stutte 198 1). The release of NH, is a detoxification mechanism that works according to the following reaction scheme (especially when there is excess N) under the influence of high concentrations of NO and NO,, and during protein degradation (senescence) (Rowland et al. 1987).

nitrate nitrate **GS/GOGAT**

NO, → HNO, → HNO, → NH_{3n} → amino acids → proteins

reductase reductase (internal) ← ←

↓

NH_3

(external)

Table 22 reviews the net emissions of gaseous total-N and NH,-N for various intact useful plants and under the major influential factors.

These figures show that the net NH,-N emissions of these useful plants are approx. 1.6-15.5 kg NH,-N/ha per vegetation period. Assuming an emission of only 5 kg NH,-N/ha per vegetation period, the result would be an NH,-N emission of approx. 60,000 t/a for the agricultural land in the Fed. Rep. of Germany. Since, as mentioned above, there are no available natural NH,-N background values for corresponding ecosystems, however, we hesitate to include these NH, emissions by useful plants as a sort of extra emission in the calculation of the NH, emissions of agriculture.

Ammonia emissions from dying plant material
These NH, losses are important especially for grassland. Vertregt and Rutgers (1988) report an NH,-N emission of 3 kg NH_3-N/180 d and Whitehead and Lockyer (1989) report 0, 1.4 and 2.8 kg NH_3-N/180 d depending on the N content of the plants of 0.8%, 2.5% and 3.0%. Once again these NH,-N emissions are not taken into account in the overall review because of the lack of corresponding natural background values.

Ammonia emissions by agriculture and its share of territorial, continental and global ammonia emissions

Individual countries or continents

Even the older figures from Table 17 (reference year 1982) revealed that agriculture accounted for about 98% of the NH, emissions of (Western) Europe, meaning that the NH, emissions of industry played a minor role at about 2%. Within agriculture, the NH, emissions were caused to about 82% by livestock husbandry and to about 18% by the application of mineral fertilizers. These ratios have changed rather to the disadvantage of agriculture in the meantime, since today the NH, emissions from other sources (industry, energy, transportation, etc.) have more than likely decreased, while those from agriculture have theoretically and/or actually increased by about 2/3, according to the data in Table 23.

Global ammonia emissions

Total NH,-N emissions globally were listed at 22 to 35 million tons per year by Böttger et al. (1978). Those of natural origin are only 1 to 2 mio t (5%). Of the 2 1 to 33 mio t attributable to human activities, livestock husbandry is responsible for as much as 20 to 30 mio t (90%). Fossil fuels account for only 100,000 t, motor vehicles worldwide for 300,000 t of the NH, emissions. Table 24 gives the various data on this.

Wameck (1988) cited by Jenkinson (1990) arrives at similar ratios (see Figure 4) on the basis of higher estimates for global NH,-N emissions of 54 million t NH_3-N/a: 87% of the total NH,-N emissions can be traced to the complex of animal excreta, mineral fertilizers and soils (= 47 mio t NH_3-N/a).

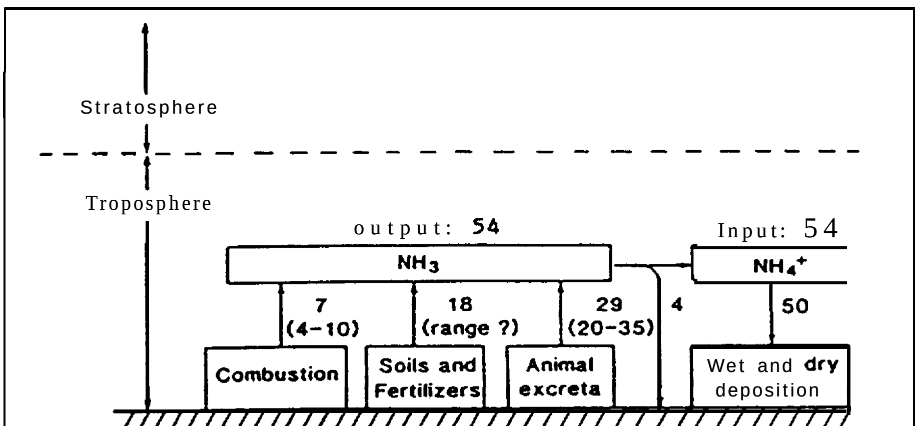


Figure 4:
Production and fate of NH, in the atmosphere; units are million tons N/a (Jenkinson 1990, from *Warneck* 1988)

The ammonia emissions by agriculture as a component of its nitrogen balance

In the following Table 25 the ammonia emissions of several West European countries from the agriculture sector are portrayed within the framework of their nitrogen balance.

The following can be drawn from these figures:

Within agriculture during the 1980s a total input of 152 (Norway) to 465 (Netherlands) kg N/ha.a contrasted with a net export via sales products of only 26 (Norway) to 98 (Netherlands) kg N/ha.a. If the nutrient efficiency is broken down according to plant and animal production, today plant production fares comparatively well at 59% (DK 1980) to 73% (FRG). The nutrient efficiency for livestock production, on the other hand, is considerably lower, at 13% (DK 1980) to 2 1% (NL). It is the nutrient efficiency of livestock production that primarily determines the total nutrient efficiency of agriculture, since on a nitrogen basis 83% (FRG) to 94% (NL, CH, Norway) of the total domestic plant production as well as the imported feeds are utilized for livestock production. This total nutrient efficiency ranges from 14% (DK 1980) to 28% (DK 1985/86) for the individual countries. Because of these relationships, for international comparisons the varying extent of the *N surplus* in the total N supply beyond net removal through sales products and the respective N emissions into atmosphere and hydrosphere is influenced primarily by the varying level of the total N supply and hardly at all by its varying N efficiency. (Compare Norway with Switzerland, Federal Republic of Germany, Denmark and the Netherlands). This statement is confirmed as well in a temporal comparison of the N balances of Denmark's agriculture in 1950 and then in 1980 and 1985/86.

With regard to the ***ammonia emissions*** of agriculture, in absolute terms they vary considerably from country to country, at 44 (DK 1985/86, FRG) to 99 (NL) kg NH,-N/ha farmland, but are extraordinarily significant at levels of 26% (FRG) to 30% (DK 1980) of the N surplus, with a low relative range of fluctuation. At 13-2 1% the rather well-balanced N efficiency within livestock production in the different countries had relatively little influence on the absolute extent of the NH,-N emissions; simply the level of livestock production was decisive.

Proposal for suffkient reduction of the ammonia emissions from the agriculture sector

Necessary level of reduction of (ammonia) N emissions

From the ecological viewpoint, it is the atmospheric (NH_x)-N emission into natural ecosystems wich is acceptable over the long-term, that serves as the guideline for the required reduction of pertinent N emissions. Atmospheric N emissions beyond this level (critical loads) pose a long-term threat to the (natural) regulatory functions of these sensitive ecosystems or soils (soil acidification, eutrophication or hypertrophy, nitrate accumulations in leached water, groundwater and surface water, etc.).

As shown in table 26, both agriculture and the engergy sector must realixe an approx. 50 % reduction (Nilsson and grennfelt 1988, Nordisc Ministerrad 1985, Bartnicki and Alcamo 1989) in the NH,-N and No_x-N emissions, in view of the acceptable N emission into natural ecosystems (including the North and Baltic Seas). In view of this exigency, only the maximum permissible N emissions into natural ecosystms (open land: 10 kg; woods: 20 kg; North/Baltic Seas: 5 kg N/ha.a) can serve as a basis, under consideration of what is attainable in the long term. Such levels are actually attainable only if (in addition to a real reduction in the No_x emissions of 63%) the NH, emissions are eventually halved, as called for also by the authors Bartnicki and Alxamo 1989, Liibkert et al. 1989. The laws to date have fallen far short of these demands regarding NH, emissions by agriculture. With this very background in mind, the Ministry of Environment in the Netherlands is demanding a 70 reduction in the NH, emissions from the livestock production sector by the year 2000 (see Table 26). If they were to be reduced only by 50 %, it should be noted, the subsequent NH_3 emission density would still be just as high as in the Federal Republic of Germany before any limitation of its NH, emissions.

Proposed **solutins** in agriculture for a **sufficient** reduction of its ammonia emissions

Appropriate solutions for sufficiently reduciong the NH, emissions can be divided into cause-oriented active measures or resultooriented passive measures on the one hand, and on the other hand nto the fundamental conditons required over the long term. The umbrella over all these efforts, however, must be the goal og the agricultural sector to ***simultaneously*** minimize *all* N losses (emissions) into the hydrosphere and atomsphere and to macimize the N efficiency for the best possible N balance (Isermann 1990a, b, c). In view of such diffuse N sources, a differentiated application of the leading principle of xomprehensive environmental protection becomes necessary, according to the risk of emissions on sites. This demand for somprehensive but differentiated environmental surface water protection (AG-Fließwasser 1990; Auerswald et al. 1990, as well as Werner 1989, Werner et al. 1989. Iserman 1990a, b,c, EC Directive 1988), but also as regards protection of the atmosphere from diffuse sources of ammonia.

Fundamental conditions necessary in the long term to reduce ammonia emissions

These are cause-oriented active measures that primarily minimize the NH, emissions from livestock husbandry by way of measures that directly influence them, i.e. measures to be taken in the sector responsible for about 80% of agriculture's NH, emissions.

Reduction of excessive livestock

Whereever the ***number of livestock is too high as*** measured by the risk of losses on the site the livestock density should be reduced. The Council of Experts for Environmental Issues 1985, Sauerbeck 1985, EC Directive 1988, AG Fließgewässer 1990, Auerswald et al. 1990, Werner et al. 1989 (also see cited authors in Isermann 1990a, b, c) set this threshold for excessive livestock at 2 units/ha suppliable farmland. In general then there is a demand for aligning the maximum livestock density on a particular site with the risk of losses on it. The guidelines of 2 units/ha suppliable farmland in the EC Directive for the Protection of Fresh, Coastal and Marine Waters (Draft 1988), the Danish Bulletin No. 668 by the Ministry of Agriculture dated October 14, '1987 (storage and distribution of farmyard manures equivalent to 1.7 to 2.3 units/ha.a) and the Slurry Guideline 1989 from Schleswig-Holstein (max. 2 units/ha farmland) all point in the same direction. The contrary direction, on the other hand, is faced in such official guidelines like the ones in the Federal Republic of Germany tolerating a maximum 3 to 4.5 units/ha farmland (Agrarstrukturgesetz 1989), in the Netherlands (on a I? basis) up to 5.7 units/ha farmland, or in Italy as much as 8 units/ha farmland, to which ABEF 1989 also draws attention.

Generally speaking it would make more sense to do away with the general concept of the livestock unit and replace it with the farm-specific and animal-specific nitrogen (nutrient) excretion rate. This rate can be found through nitrogen (nutrient) balances on farms (input by feed minus export via animals sold) for livestock production. In this way efforts would also be rewarded that elicit a lower animal-specific N (nutrient) excretion rate, such as changes in feeding programs, which would perhaps even result in higher acceptable livestock density levels.

Improved integration of livestock and crop production

In connection with the restructuring of livestock populations and densities, the integration of animal and crop production offers great possibilities for dealing directly with 'the causes in regions with excessive livestock husbandry, to reduce not only the NH, losses but also nutrient losses in general, as well as the related emission densities (also see ABEF 1989). This integration of livestock and crop production gives livestock husbandry the opportunity once more on farms with cereal crops (straw) to give preference to the ***deep stable manure system,*** that incurs less N loss compared to the slurry system (especially during collection and storage; Ministerie van Landbouw en Visserij 1985, Kirchmann and Witter 1989). Moreover, it is prerequisite to enabling increased use of farm-produced feed over imported feed, which in turn promotes farm-internal N recycling and N efficiency.

Changes in feeding programs

Important cause-oriented measures here include:

- the ***reduction of protein surpluses*** (ssafety margins«)
- ***Phase feeding*** for feeder pigs and poultry (Lenis 1989, Küther 1989)
- ***avoidance of excessively protein-rich ruminant feeding*** (esp. too high N dressings on pastures and meadows) (Whitehead et al. 1986, van Vuuren and Meijs 1987) or general ***replacement of feed concentrates by*** (actually more appropriate for ruminants) ***basic feed rations*** after the first third of lactation (Opitz von Boberfeld 1989)
- amino ***acid additives*** (lysine, tryptophan, methionine, cystine) to (cereal) ***feed rations forfeederpigs and poultry*** bring over the long term considerable improvements in protein utilization (N efficiency) along with approx. 40% reduction of N excretion but the same performance (Küther 1989, Franz and Salewski 1989, Lenis 1989).

These feeding adjustments thus create the conditions for being able to raise the maximum acceptable livestock densities again sometime in the future.

Short-term, supportive measures to reduce ammonia emissions

There are also measures that agriculture can take in the short term to reduce the NH_3 emissions and emission densities, and more generally all N losses. These are not sufficient by themselves, however, to effect a permanent decrease in NH_3 emissions or N losses to the desired extent, so that they are to be viewed only as supportive steps to the above longer-term comprehensive conditions (important but not adequate steps). In part they also lead up to these fundamental conditions. Oriented toward cause as well as effect, they begin with N fertilization, whereby for farmyard manures the circumstances of collection and storage are also to be considered.

General optimization of N fertilization through:

a) ***Farm-wide as well as single-field nitrogen balances*** oriented to the N removal with the harvest and including unavoidable and ecologically acceptable losses.

b) Consideration of the ***nitrogen supply level in the soil zone penetrated by roots*** (net mineralized N and potential N supply).

c) ***Optimization especially of the use offarmyard manures as regards quantity and timing, with maximum avoidance of NH_3 losses during their collection, storage and distribution.*** There is a great untrapped potential here for increasing the N efficiency. If, as in this case, it is brought about by reducing the losses, then the maximum livestock density to be expolited is to be lowered accordingly. In general, though, approx. $^1/_4$ to $^1/_3$ of the N fertilizer requirement should be set aside for mineral N fertilizers, so that the N fertilization can still be manipulated somewhat. This also increases the efficiency of the use of farmyard fertilizer N.

Targeted reduction of ammonia emissions by farmyard manures, mineral fertilizers and crop plants

Reduction of ammonia emissions during collection stable), storage, handling and use of farmyard manures

a) ***Collection of farmyard manures (stable area)***

Sequence of conditions favorable to NH_3 emissions (increasing NH_3 emissions)

Stable system:

Deep stable manure	<	Hot-fermented manure	=	Slurry
Danish stable setup	~	Fully slatted floor	< =	Semi-slatted floor
	with	<	without	

a) ventilation (bio-) filter
b) rinsing system (Hepag stable)

b) ***Storage and treatment of farmyard manures***

Here the sequence of conditions favorable to NH_3 emissions (increasing NH_3 emissions) can be assessed variably depending on the type of slurry treatment:

$$\text{Deep stable manure} \overset{>}{\underset{<}{=}} \text{slurry} \overset{>}{\underset{<}{=}} \text{hot-fermented manure (esp. dried)}$$

- losed container - added acid. or precip. treatment	<	- closed standard container - untreated, homognized	<	- open standard container - untreated	<	- open standard container - aerated or base added	<	- open pit »lagune« »pond treatment«

Pit-type storage sites should be forbidden or closed down because of their high NH_3 emissions (see Table 9 and Figure 1). The same applies for aeration measures and alkaline additives for slurries.

c) ***Distribution of farmyard manures***

A closer look at Tables 10, 11 and 12 are of assistance here. It is not so much the type (solid or liquid manure) as rather the applied quantity, the timing and the distribution (application method) as well as soil and weather conditions that determine the level of the NH_3 emissions. In practice this means consideration of:

- ***quantity:*** max. $^2/_3$ to $^3/_4$ of the N requirement of the crop based on the total N content of the farmyard manure;
- ***form:*** normal slurry with a high water content (not biogas slurry) and a high infiltration capacity (pig slurry better than cattle slurry) on light soils;
- ***timing:*** in general not during the vegetation period. Before the vegetation period begins at temperatures as low as possible and with minimum wind;
- ***distribution:***
  - immediate incorporation (max. after 1 hour), especially with slurries low in water and with heavy soils of low infiltration capacity;
    injection or slurry drill better than plow, which is better than disk-harrow or knife-covering, which is better than on the surface; hose better than deflectors
  - no application of slurry on stubble and/or straw without immediate incorporation; injection on grassland whenever possible
  - use of pastures with year-round stabling accompanied by optimized collection, storage and distribution of the farmyard manure is to be preferred to pasture use alone
  - no excessive N fertilization of pastures when using them

The utilization rates for farmyard manures today (relative to the total N content) are between 40% and 80%; that is, they approach the good utilization rates of mineral fertilizers only in extreme cases of optimal collection, storage and distribution (Bundesamt fiir Umweltschutz, Bern 1987; Bayerische Landesanstalt fir Bodenkultur und Planzenbau 1990). This is confirmed by Vetter et al. (1990) with the trial results presented in Table 27.

Vetter et al. (1990) come to the conclusion: »The maximum applications should be determined by the nutrient that is the first to supply the requirement of the plants. This would be potassium in the case of cattle slurry, phosphate in the case of chicken manure and chicken slurry, and phosphate and nitrogen in the case of pig slurry. Even with a very generous calculation of the fertilizer requirement, reasonable maximum levels per hectare for excess manure-producing operations do not exceed 40 m^3 cattle slurry, 30 m^3 pig slurry and 14 m^3 chicken slurry. In the long term the acceptable maximum applications are about 30 m^3 cattle slurry, 24 m^3 pig slurry and 11 m^3 chicken slurry per hectare« (which is equivalent to about 1.5 units/ha).

Reduction of the ammonia emissions during the use of mineral fertilizers
The following is necessary here:

a) No ***alkaline forms of N*** (such as urea, diammonium phosphate, UAN, lime nitrogen) ***on soils with:***
   - high pH level (>7.0) (no urea after grazing!)
   - high level of free lime
   - low cation exchange capacity (low-humus sand), but rather use of CAN or multinutrient fertilizers

b) ***If alkaline forms of N are used:*** immediate incorporation whenever possible, especially when nitrification inhibitors have been added.

Reduction of the ammonia emissions from crop plants
The only option in this case is to take care to avoid surplus fertilization of the plants in view of their NH, emission. Not enough is known about this yet, however.

A last comment here is that the required target of a 50% reduction in ammonia emissions is very well possible if the above long-term and short-term measures are taken. Beyond the primarily ecological base of this requirement, economics are also an issue here: considerable savings to agriculture would ensue as well.

Research needs with regard to ammonia emissions in agriculture

Table 28 summarizes the individual areas requiring more research to quantify and reduce the NH,-N emissions in the agricultural sector.

With respect to measures to reduce the environmentally relevant N losses in agriculture in general and the NH, emissions in particular, there is a low to medium research requirement, with the exception of some special areas (Isermann 1990c). Agricultural research (as well as interdisciplinary research over the past ten years) is able to offer agriculture a multitude of short-, medium- and long-term measures for the reduction of N emissions. These measures will produce a satisfactory level of success if they are employed conscientiously and widely according to local risk or potential risk of losses. It is not so much a research deficit that is the universal problem, but rather a deficit in the implementation of measures on the part of the farmer as **well** as the failure of agricultural policies to produce and enforce the proper legislation. The guideline for implementation should be a locally appropriate and hence environmentally safe land management in accordance with the risk or potential risk of losses (e.g. in soils). Comprehensive evaluation of this risk of loss or potential risk of loss does indeed mean there is a **sizeable** (longer term) need for research. But an extensive base has already been provided on the part of the soil sciences, geology, hydrology and atmospheric chemistry.

Summary

It is primarily the ecological reasons, along with the economic ones, that compel us to deal with the NH, emissions in agriculture and reduce them adequately. These NH, emissions are largely responsible for the direct impairment and eutrophication (hypertrophy) of sensitive terrestrial and aquatic ecosystems, such as in Western Europe. Almost as a postscript, their responsibility for the corrosion of buildings must also be noted.

NH, emissions in agriculture

In addition to type, number and density of the ***livestock,*** the level of the NH, emissions is strongly influenced already by the protein levels (excess, or incomplete utilization) in the feeds both with the monogastric animals (pigs) but especially with the ruminants. The NH, emissions come from the sectors:

Livestock husbandry (Federal Republic of Germany)

Based on numerous studies, the levels are as follows:

Stable area (production of farmyard manures)

An average emission of 8 kg NH,-N / unit.a is assumed across all types of stabling and manure removal, all types of animals.

(Open) storage of liquid and solid manure

The gaseous N losses are between about 10% and 23% of the total N with an extreme of 80% during the treatment and storage of slurry in pit-type containers. An average emission of 10 **kg/unit.a** is assumed across all types of manure removal and types of animals.

Application of liquid and solid manure

The NH, loss can range between **>** 1% and **<** 100% of the applied NH,-N under the influence of various factors (timing, characteristics of the manure, weather, soil properties, method of application). Across all types of animals and manure removal again, an average emission during the application of farmyard manures of 18 kg NH,-N / unit.a is assumed.

Pasture husbandry (cattle)

The emission as a whole was placed at 14 to 28 kg NH,-N / unit. 180 d, whereby 10 to 15 kg / unit. 180 d are in the pasture and the rest attributes to the release of NH, during use of farmyard and / or mineral fertilizers.

Livestock husbandry as a whole in the Federal Republic of Germany was responsible for an emission of 500,000 t NH_3-N / a, based on an average emission of 36 kg NH_3-N / unit.a for an average excretion of 100 kg N/unit.a. This places the specific NH_3-N emission of a larger livestock unit in Germany at approx. 2 to 3.5 times higher than the NO_x-N emission of an average truck (17.5 kg / a) or an average automobile (10.5 kg / a).

In areas of Western Europe with a high level of livestock husbandry, the emission densities lie between 50 and 170 kg NH_3-N/ha.a, while more extensive livestock production produces emission densities of only > 1 to 10 kg NH_3-N/ha.a (natural pastures).

Application of mineral fertilizers (Fed. Rep. Germany)
Their average emission was placed at 1.9% of the applied N, resulting in an annual total emission of approx. 28,000 t NH_3-N/a (≙ 2.3 kg NH_3-N / ha farmland) for 1986. The level has increased over the last decade because of the increasing share of urea in the fertilizers.

Useful plants
The net emission of beneficial plants of 1.9 to 1.5 kg NH_3-N/ha per vegetation period was not included in the calculation because of the lack of corresponding background figures for natural ecosystems.

Total agriculture in the Federal Republic of Germany
Total emissions add up to 528,000 t NH_3-N/ha.a (1986). Other authors refer to an approx. $^2/_3$ higher NH_3-N emission from agriculture in various countries in Western Europe (such as ex-GDR, the Netherlands, UK, Norway), based on the data by Buijsman et al. (1985) for 1982, plausible both mathematically as well as causally.

Agriculture's share of the NH_3 emissions from all sectors of the economy

From the territorial (individual countries of Western Europe), continental (Western Europe) and global standpoints, the total NH_3-N emissions of all economic sectors are attributable to about 90% to agriculture, and about 80% of this is attributable to livestock husbandry. The remainder can be traced to industry, transportation and the energy sector, as regards anthropogenic sources.

Ammonia emissions from agriculture as a component of its nitrogen balance

In agriculture in the 1980s a total deposition of 152 (Norway) to 465 (Netherlands) kg N/ha.a contrasted sharply with a net export via sales products of only 26 (Norway) to 98 (Netherlands) kg N/ha.a. If the nutrient efficiency is broken down into plant and livestock production, that of plant production at 59% (DK 1980) to 73% (FRG) fares quite well in the comparison. The nutrient efficiency in livestock production, on the other hand, is considerably lower, at 13% (DK 1980) to 21% (NL). It is the nutrient efficiency of livestock production that is the major determinant of the total nutrient efficiency of agriculture, since on an N basis 83% (FRG) to 94% (NL, CH, Norway) of the total domestic plant production is utilized for livestock production, along with the imported feedstuffs. This total nutrient efficiency ranges between 14% (DK 1980) and 28% (DK 1985/86). Because of these interrelationships, in international comparisons the varying extent of the N surplus in the total N supply via the net removal by the sales products and the related N emissions into atmosphere and hydrosphere is characterized primarily by the varying level of the total N supply and hardly at all by the varying N efficiences of the parts.

Turning then to the ***ammonia emissions*** from agriculture, they differ widely from country to country in absolute terms, at 44 (DK 1895–86, FRG) to 99 (NL) kg NH_3-N/ha farmland. But they are extremely significant, accounting for 26% (FRG) to 30% (DK 1980) of the N surplus, with a low relative range of fluctuation. The rather stable N efficiency for livestock production in different countries of 13% to 2 1% had little influence on the absolute extent of the NH,-N emissions; the major factor was quite generally the level of livestock production.

Proposal for sufficiently reducing the NH, emissions from agriculture

Ecological obligations towards affected sensitive terrestrial and aquatic ecosystems necessitate a 50% reduction of the atmospheric N emissions in Western Europe below the »critical load« and hence also of the respective NO_x and NH, emissions. This requirement can be fulfilled by agriculture in the long term, if it manages to implement the following corrective measures:

Long-Term measures (fundamental conditions)

Reduction of livestock numbers on the basis of environmentally safe maximum livestock densities of currently 2 units/ha suppliable farmland, in accordance with the demands of the Council of Experts for Environmental Issues (1985), Sauerbeck (1985), EC Directive (1988), AG Fließgewässer (1990), ditto Auerswald et al. (1990) and Werner et al. (1989). Future calculations of maximum livestock densities should be based on animal-specific (nutrient) N depositions within each farming operation and the risk of loss on each site.

Extensive integration oflivestock and plant production in areas with excessive emphasis on livestock husbandry

Changes in feeding programs, esp. correct protein nutrition

Short-term supportive measures

General optimization of N fertilization with farmyard and mineral fertilizers

Targeted reduction of NH, emissions during collection (stable), storage, treatment and application of farmyard manures

Targeted reduction of NH, emissions during the application of mineral fertilizers

The above proposals require not only an ecological assessment but also an economic one, since in the final analysis a compromise must be found that permits the ecological requirements to be carried by the economy of a particular area (farm).

Research needs

The necessity for research to quantify and reduce the NH, emissions from the agricultural sector is considered much smaller that the necessity for implementation of measures by the farmer as well as for the making and enforcing of (agricultural) policies on the basis of all the facts already available.

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Table 1:
Necessity *for territorial, continental and global balancing of the nutrients carbon, nitrogen, phosphorus and sulfur in view of (possible) cross-bounda y environmental damage, with special reference to agricultural interests*

Possible cross-bounda y environmental damage	*Nutrients and their environmentally relevant compounds*			
	C	**N**	S	**P**
1. Climate changes → greenhouse effect	$CH_4 \rightarrow CO \rightarrow CO_2$	N_2O	COS	–
2. Air pollution (e.g. »acid rain«) → damage to natural terrestrial and aquatic ecosystems (acidification) → building corrosion	CH, → 0, synthesis	NO_x=NO+NO_2 HNO, NO, NH_y=NH_3+NH_4 NH_4NO_3 NH_4HSO_4 and	H_2S,$(CH_3)_2S$, CS SO_2 H_2SO_3/H_2SO_4 SO_4^{2-} $(NH_4)_2SO_4$	–
3. Eutrophication (hypertrophy) 3.1 Terrestrial ecosystems (forests, heaths, highmoors, etc.)	CO_2	All metabolic N compounds listed in 1. and2., esp.: NH_y=NH_3+NH_4 NO,	–	(H_yPO_4)
3.2 Aquatic ecosystems (inland waters, oceans)	(CO_2)	NH_y=NH_3+NH_4^+ NO,-	SO_4 → P mobilization+ → inhibition of biolog. N fixation and NO,- assimilation	H_yPO_4

Table 2:
Increased N utilization and decreased N deposition on extensive and intensive dairy pastures over 180 days of grazing by cows (550 kg) with milk production of 20 kg/d (after Van Vuuren and Meijs 1987)

	N in grasses (g/kg DM)	*cows Per* ha	*N uptake* (kg)	*N excretion (kg)* Milk	Faeces	Urine*	*Urine N* / Faec. N
Extensive pasture (0 kg N/ ha.a)	**30**	**1**	91 (100%)	19 (21%)	(223%)	(556%)	2.4
					72 (79%)		
Intensive pasture (383 kg N/ ha.a.)	**44**	**1**	116 (100%)	19 (16%)	20 (17%)	77 (67%)	3.81
		3	348	57	60	231	
					291 (84%)		

* Urine consists a l: *92%* urea N, < 1% NH,-N, <1% org. N → NH,-N ↑

Table 3:
NH, emissions from pasture on 3 daiyfarms of va yingintensity during the pasture season of 180 days (Vertregt and Rutgers 1988)

	Farm type Excessive	Intensive (ave. 1986 Netherl.)	Extensive
N fertilization (kg/ha.a)	320-380	240-275	**0** (biol. N fixation)
Grazing density (cows/ha)	3.5	3.5	1.85
Grazing days (d/ha)	630	630	333
N level of grasses (g/kg DM)	44	35	30
Suppl. feed (g DM/cow)	2000	2000	470
N uptake via feed (kg/ha)	390	310	140
N in milk	42	42	19
N in urine and faeces			
a) kg/ha	348	268	121
b) kg/cow	99 (100)	77 (100)	65 (100)
NH,-H emission			
a) kg/ha	45	35	16
b) kg/cow	12.5 (13.0)	10.0 (13.0)	8.6 (13.2)

Table 4:
Nitrogen losses from farmyard manures (Sauerbeck 1985)

Phase:	***Collection*** Preparation Stabilization	***Distribution*** Spreading	In ***the soil*** Leaching Denintrification
% loss	30–90	0–50	0-50
Form	NH,, N_2	NH,	NO_3, N_2, N_2O

Table 5:
Influences of aerobic and anaerobic solid manure fermentation and the quantity of straw on the release of NH, mineralization* and *immobilization (Kirchmann and Witter 1989)

Solid manure	Added straw gg^{-1} init.N	Release of NH, mg NH,-N g^{-1}init.N	Ammonia concentration *mg NH, g^{-1} init.N* 0	201 days	Net mineral-ization mg N g^{-1} init.N	Immobili-zation mg N g^{-1} straw
Aerobic (fer-mented manure)						
C/N 18	20.7	437.4	5.8	11.6	443.2	11.2 (C/N 18-24)
C/N 24	41.4	190.9	0.6	33.7	224.9	
C/N 36	73.5	91.5	0.1	47.7	139.3	2.2 (C/N 24-36)
Anaerobic (deep stable manure)						
C/N 18	20.6	9.4	5.8	742.0	745.6	0
C/N 24	41.4	4.0	0.6	742.0	745.4	0
C/N 36	73.5	1.7	0.1	737.8	739.4	0

Table 6:
Ammonia emissions in the stable area as a function of animal type and type of manure removal

Type of animal	*Manure removal system*	*NH,-N emission per unit/a*	*Authors*
Cattle	not defined	4.1	SR-U (1985)
Dairy cow	not defined	7.2	KLARENBECK (1987)
Heifers under 2 yrs		4.6	
		4.7	
	Fully slatted floor	12.6	
	Half-slatted floor	12.6	
Feeder bulls	not defined	5.7-l 1.4 (Ø=8.6)	SR-U (1985) ABEF (1989)
Feeder pigs (7 feeder pig slots)	Danish system	9.6	EERDEN et al. (1981)
	Half-slatted floor	17.8	
	Fully slatted floor	9.8	
	not defined	12.8	KEHR (1990)
Chickens	Slurry 6.2 With litter 1.4 Slurry 31.8 Floor management	13.7	KOWALEWSKY (1981)
All types of animals	All manure removal systems	8.0	ISERMANN (here)

Table 7:
Ammonia emissions** in **laying hen housing systems (Eerden et al. 1981

Animal / Housing system	kg NH_3-N/ unit.a
Daily manure production	1.1
Manure storage..	7.2
Dry manure storage	23.7

Table 8:
Gaseous N losses during the storage of rotted cattle manure and cattle slurry as a function of storage time (Ministerie van Landbouw en Visserij 1985)

Storage time (months)	*Gaseous N losses in % of toral N*		
	Rotted manur		Slurry
	densely stored	loosely stored	
2	10	*14*	5
4	16	*19*	*7*
6	21	*23*	*8*
Compare VETTER et al. (1989)	-	**approx.20**	**approx. 15**

Table 9: N balance for the pig fattening facility Neustadt/Orla (ex-GDR) in the year 1988 (space for 175,026 pigs) (Isermann, after data from Kehr 1990)

Parameters for the balance	*t N/a*	*%*	*kg N* for each pig space	*kg N* for seven pig spaces (≙ 1 unit)
1. Input → **feed**	*3465*	**100**	**19.8**	**139**
2. output ...of which:	*3465*	**100**	**19.8**	**139**
2.1 Meat	*735*	**21**	**4.2**	**29**
2.2 **Excereta (wastes)**	*2730*	*79*	**15.6**	**109**
... of which:				
2.2.1 Recycled (solids, excess sludge, irrigation water)	*890*	*26*	**5.1**	**36**
2.2.2 Losses ... of which	*1840*	*53*	**10.5**	**74**
2.2.2.1 Gaseous	*1708*	*49*	**9.8**	**69**
a) NH,-N (stable + slurry vent.)	*783*	*23*	**4.5**	**32**
b) Pond treatment (N_2, N_2O, NO_x, NH,	*925*	*26*	**5.3**	**37**
2.2.2.2 Waters ... of which:	*132*	4	**0.7**	5
a) Ponds	*61*	2	**0.3**	2
b) Outfall ditches	*71*	2	**0.4**	3

The feeder pig facility operates at a N efficiency of 21% with:

a) Excretion of 2730 t N/a (= 109 kg N/unit.a), which is equivalent to the N amount produced by a large city with a population of 620,000 (Dresden + Jena)

b) whereby only 33% of the excreted N are recycled (farmyard manures) and

c) 63% of the excreted N escape into the atmosphere (69 kg N/unit.a)

Table 10:
Influence of various application methods and environmental factors on the loss of ammonia N during the spreading of liquid and solid manures

Authors	*Liquid or solid manure application* — Type CS = cattle or PS = pig slurry	Variants	*NH_3-N emission in % of the applied NH_4-N*
1. HOLZER et al. (1988)	CS		***64***
		20 loamy	***60***
		10 sand	***52***
		emp. (°C): 30	***43***
			60
		20 clayey	***55***
		10 loam	***46***
		30	***34***
		Quantity 35	***61***
		nanure: 70	***60***
		m^3/ha) 140	***51***
		Method: plastic	***86***
		surface	***60***
		incorp .	***2342***
		inject.	1
		On winther wheat:	
2. DÖHLER and ALDAG (1981)	CS	1) 5-leaf	23
	PS	2) tillering	24
	CS	3) beg.elongation	63
	CS	4) + knife-covered	54
		5) flag leaf	
		a) deflector	66
		b) hose	47
	CS	6) plowed soil	34
	PS	7) grassland (hose)	28
	c s	8) slurry drill in maize crop	15
	PS	9) prepared soil	36
	c s	10) prepared soil 54–59	
	c s	11) slurry drill in cultivated soil	1 B-2.2
	Stable manure	12) on straw stubble	59
3. DÖHLER et al. (1981)	CS	1) frozen soil	21
	CS	2) wet soil	25
	CS	3) plowed, harrowed	9
	CS	4) wheat stubble	95
	CS	5) wheat stubble + chopped straw, (cultiv., harrow)	10
	CS	6) Advance soil prep. (cultiv., harrow)	48
	CS	7) Grassland	37

Authors	*Liquid or solid manure application*		*NH,-N emission in % of the applied NH,-N*
	Type CS = cattle or PS = pig slurry	Variants	
4. DÖHLER and ALDAG (1990)	CS	1) surface / 20” C	40–60
	CS	2) surface / 0” C	30
	CS	3) frozen soil	20
	CS	4) surface / 20” C	15–20
	PS	5) sunny weather	35
	PS	6) on straw / 20” C	100
5. ASMUE (1990)	PS	bare soil	1520
6. AMBERGER (1989)	CS	1) general	40–60
		2) lightly	50
		incorporated	12
7. AMBERGER et al. (1987)	CS	1) sand	
		a) surface	38
		b) incorporated	28
		2) loess-loam	
		a) surface	32
		b) incorporated	14
		c) on stubble w/o straw	51
		d) on stubble with straw	69
8. HUBER and AMBERGER (1990)	CS	1) in cereal crop (elong.)	approx. 25
		2) on soil	40-52
	CS	1) on soil	35
		2) on maize straw	50
		3) on shallow/straw	60
	CS	1) on soil	33
		2) shallow incorp.	14
	CS	1) grassland – late summer	80
		2) grassland – spring	42
		3) grassland – winter	22
		4) soil – winter	16

Authors	Liquid or solid manure application Type CS = cattle or PS = pig slurry	Variants	NH,-N emission *in % of the applied NH,-N*
9. AMBERGEI (1990)	CS	1) 20" C; 8% DM	***43***
		2) 20" C; 6.4% DM	***38***
		3) 5" C; 8% DM32	
		4) 5" C; 6.4% DM	11
	PS	5) 20" C; 0.9% DM	17
		6) 5" C; 0.9% DM	4
	CS	1) sand not	45
		2) fsl incor-	40
		3) ul > porated	40
		4) sand incor-	28
		5) fsl por-	12
		6) ul > ated	7
	Stable manure	1) fresh manure	
		a) sand	19
		b) loess	20
		2) rotted manure	
		a) sand	45
		b) loess	36
10. RANK et al. (1988)	CS	Temp.: 5" C	15
		25" C	18
	PS	Temp.: 5" C	5
		25" C	17
	CS	on stubble and straw with	
		a) 5% dry matter	46
		b) 10% dry matter	73
	CS	l i g h t l y 8%	42
		incorporated DM	6
	PS	lightly 1%	18
		incorporated DM	2
11. HOR-LACHER and MAR-SCHNER (1990)	CS	1) on cereal stubble. incorp. after 5 h at 10-20" C	47-65
		2) on spring barley (10 cm)	55
		3) on maize stubble at O-5" C	35
		4) advance soil prep.	
		a) disk harrow	56
		b) plow	12

Authors	Liquid or solid manure application Type CS = cattle or PS = pig slurry	Variants	NH_4-N emission *in % of the applied NH_4-N*
12. STEVENS et al. (1989)	PS	on soil, added acid pH 7.0 pH 5.5	 *28* *3*
	CS	on soil, added acid pH 7.0 pH 5.5	 *28* *2*
13. MINISTERIE VAN LANDBOUW EN VISSERIJ (1985)	CS Slurry	surface-applied	*95*
	Stable manure	surface-applied Weather: a) severe frost b) rainy c) sunny and windy	*18* *15* % of total N *5* % of total N 26 % of total N
14. PAIN et al. (1988)	CS	surface-applied on grassland - winter - spring	 23-74 43-47
15. CHRISTENSEN (1988)	CS	on cereal stubble	95
	PS	on cereal stubble	80
	CS	on cereal stubble - 30 m^3/ha - 60 m^3/ha - 90 m^3/ha	 45 60 70
16. JARVIS et al. (1987)	CS	surface on grassland Timing: a) Nov. b) March a) Dec. b) April	 40 m^3/ha 50 80 m^3/ha 67 40 m^3/ha 34 80 m^3/ha 28 80 m^3/ha 64 80 m^3/ha 26
17. VAN DER MOLEN et al. (1989) (1990)	CS	on crop fields: surface-applied incorporated	20-70 (10-40 of total N) 11-16 (6-7 of total N)
18. VAN DONGEN et al. (1990)	CS	on fields: 1) surface 2) disk harrow 3) plow 4) injected	 30-45 2.5-9 0-2 0.1

Table 11:
Factors influencing NH, losses after application of slurries (revised after Horlacher and Marschner 1990)

Influential factors	low	← NH, losses →	high
1. Time**	short	⟷	long
(minutes, 1 h)	⟷	(low h or d)	
2. Manure properties			
a) Flowability	high	⟷	low
b) DM content**	low	⟷	high
c) NH_3 content	low	⟷	high
d) Quantity applied	low	⟷	high
	(high infilt.)		(low ingilt.)
3. Weather			
a) Sunshine*	low	⟷	high
b) Temperature*	low	⟷	high
c) Water vapor pressure	high	⟷	low
d) NH, concentration	high	⟷	low
e) Wind speed/profile*	low	⟷	high
f) Precipitation*			
- Quantity (distrib.)	high	⟷	low/none
- Type	slurry	⟷	on snow
4. Soil			
a) Infiltration**	high	⟷	low
b) CEC (soil type)	high (clay, loam)	⟷	low (sand)
c) pH level	low	⟷	high
d) Buffering capacity	low	⟷	high
e) Harvest residues**	low	⟷	high
(grass) stubble+straw	(none)		
f) Crop plants	present	⟷	none
5. Measures (farmer)			
a) Incorporation**	yes	⟷	no
b) Thinning**	yes	⟷	no
c) Soil preparation	aerobic	⟷	(anaerobic)
d) Additions	acid	⟷	base

* indirectly influenced primary factors in NH, emissions

** directly influenced primaty factors in HN, emissions

Table 12:
Outline for estimating NH, losses during application of cattle slurry and examples for calculating the losses (Horlacher and Marschner 1990)

°C	Potential total loss in %		
	Infiltration		
	low	medium	high
o- 5	30	22	5
5–10	45	35	5
10-15	70	55	0
15-20	90	75	5

↓

°C	Time factor											
	lh	2h	4h	h	12h	1d	2d	3d	4d	5d	8d	12d
o- 5	.04	.07	10	.15	.19	.25	.35	.45	.54	.60	.80	1.00
5-10	.06	.10	14	.20	.25	.35	.50	.65	.73	.85	1.00	
10–15	.15	.25	35	.50	.60	.73	.83	.92	1.00			
15-20	.20	.30	45	.65	.75	.85	.95	1.00				

↓

°C	Rain factor			
	Precipitation (mm)			
	0–2	2-5	5–10	0
o- 5	.30	.15	.05	0
5-10	.40	.20	.10	0
10-15	.60	.40	.20	0
15–20	.80	.50	.30	0

↓

Total loss

Table 13:
Influence of the timing of the applications of slurry onto winter cereals on thegaseous N emissions through denitrification and volatilization (Van den Abbeel et al. 1989)

Sluffy application		Gaseous N emissions (kg N/ha.period)		
Period	**kg total N per ha.period**	Denitrifi. (N_2, N_2O, NO_x)	**Volatiliz. (NH,)**	**Toral**
Nov.-Feb.	209 (100%)	21 (10%)	*48 (23%)*	69 (33%)
May-April	*202* (100%)	*3* (1%)	*78 (39%)*	81 (40%)

Table 14:
Ammonia emissions from animal excreta during pasture use

Type of animal	*Variant*	*Comparisonof NH, emissions*	*Authors*
Feeder bulls (1 y)	***Perennial ryegrass*** a) ***420*** kg N/ha.a b) ***200*** kg N/ha.a ***Grass-clover*** ***(0 kg*** N/ha.a)	7.8 5.6 3.7 } % of N up- take 12.1 11.2 9.8 } % of urine N	JARVIS et al. (1989a)
Feeder bulls (1 y)	***Perennial ryegrass*** a) ***420*** kg N/ha.a b) 200 kg N/ha.a Grass-glover (0 kg N/ha.a)	***12.5*** 5.0 7.0 } % of urine N	JARVIS et al. (1989b)
Dairy cows (1989)	***Perennial ryegrass*** a) 550 kg N/ha.a b) 250 kg N/ha.a	7.6-8.0 3.0 } % of urine + and faecal N	BUSSINK
Beef cattle	***Pasture***	***40%*** of urine N 30% of urine + faec. N	BUIJSMAN et al. (1986)
Dairy cows	***Perennial ryegrass*** a) 320-380 kg N/ha.a b) 240-275 kg N/ha.a Grass-clover (0 kg N/ha.a)	13.0 13.0 13.2 } % of urine + fae- cal N 12.9 10.0 8.6 } kg NH_3-N unit 180 d (see Table 3)	VERTREGT and RUTGERS (1988)
Beef cattle	***Pasture***	20–30% of urine N	STEELE & VALLIS (1988)
Beef cattle	***Pasture***	***8*** kg NH,-N/unit. 180d	JANSSEN (1985) ABEF (1989)

Table 15:
Ammonia emissions during cattle pasture husbandy

	*NH,-N emission (Kg/unit. 180 d)**	
1. *Pasture grazing*		**10–15**
a) from **excreta**	9-13	
b) from dying plant matter	l-2	
2. ***Fertilization***		13
a) Farmyard manure (5 0% of the **NH_3-N)**	11	
b) Mineral **fertilizer:** 100 kg **N/ha.a** (2% of the **N)**	2	
3. ***Pasture husbandy in total***		
a) with complete return of the farmyard manure (abosolute grassland operation)		**23–28**
b) without returning the **farmyard** manure (200 kg mineral **fertilizer/ha.a)**		14-19

* livestock density: 2 units/ha

Table 16:
Ammonia emissions per animal unit in livestock husbandy exemplified by beef and feeder pig husbandry (1986)

1 heifer (2 *yrs)*	***7 feeder pigs***	
1. Excreted N (kg **N/unit.a)**	100	100
2. NH,-N emission (kg NH,-N/unit.a)	**36***	36”
of which are attributable to farmyard manure:		
a) collection (stable)	8	***8***
b) storage	10	10
c) application	18	18
d) pasture husbandry	**14–28**	**–**

*) ***As comparison: livestock husbandry in the Netherlands 1985/86***

1. Excreted N:	594,000 t N/a (**JONGBLOOD** et al. 1985, **LENIS** 1989)
2. NH,-N emission:	192,000 t N/a (CCMRX 1988)
3. NH,-N emission in **%** of excreted N:	32% **=** 32 kg NH,-N/unit.a

Table 17:
Ammonia** emissions **and emission densities (1982) after Buijsman et al. (1985)

Country	*NH, emissions (in 1000 tons per year)*						
	Livestock nusbandry	**Min.fert.** applic.	Total	Emission density (kg/ha farmland)	Industry	Total NH, emission	Emission density **(kg/ha** land)
1. Netherlands	128	12	140	69.5	8	148	39.3
2. Belgium	74	4	78	54.6	4	82	25.8
3. Denmark	87	23	110	38.6	1	111	25.6
4. Norway	27	7	34	36.1	2	36	1.1
5. Ex-)GDR	159	42	201	32.1	6	207	18.7
6. Fed.Rep. Germany	329	35	364	30.1	6	270	14.1
7. France	569	130	699	22.1	9	708	12.8
8. Gr.Britain	307	90	397	21.2	7	404	16.3
9. Ireland	110	5	115	20.3	1	116	16.3
10. Italy	252	101	353	20.1	7	360	11.8
11. Greece	69	25	94	10.2	2	96	7.1
Countries l-l 1							
a) absolute	111	474	2585	23.9	53	2638	12.5
b) relative (%)	(80)	(18)	(98)	–	(2)	(100)	–
Europe:							
a) absolute	241	1091	*6332*	–	101	6432	8.1
b) relative (%)	(81)	(17)	(98)		(2)	(100)	

Table 18:
NH,-N emissions by livestock husbandy in certain Western European countries compared to the data from Buijsman et al. (1985) for 1982

Country	NH -N emissions by livestock husbandry (t/ha)			
	Buijsman et al. (1985) **Ref.yr.** 1982	Other authors: Quantity	Reference year	Authors
1. Fed.Rep. Germany	270 000 (236 000 for 1988 after ABEF 1989)	500 000	1986	ISERMANN (1990a,b,c
		400–500 000	1984	SR-U (1985)
2. Ex-GDR	130 000	167 000	1975189	MÖLLER (1990)
3. Holland	105 000	192 000	1986	CCMRX (1988)
4. Great Britain	252 000	536 000	1978	ROYAL SOC. (1983)
5. Norway	22 000	47 000 (gaseous N emiss.)	1980–89	BØCKMAN et al. (1990)

Table 19:
Comparison of the NO,-N emission from an average motor vehicle (truck or automobile) with the NH,-N emission by a larger livestock unit (e.g. 1 cow) (older than two years) in the Federal Republic of Germany

	Number (mio)	*NO, emissions*		
		NO_2 (t/a)	NO,-N (t/a)	kg/vehicle.a
Truck (1986)	31.4	1 800 000	548 000	17.5 a) (100)
Automobile	29.2	1 000 000	304 000	10.4 b) (100)

	Number (mio)	*NH, emissions*		
		NH_3 (t/a)	NH,-N (t/a)	kg/unit.a
Livestock units (1986)	13.9	607 000	500 000	36 a) (205) b) (350)

Table 20:
Average NH,-N emission rates during ***application of mineralfertilizers in % of total N***

Authors	*NH, emission rates of mineral fertiiizers in % of total N*						
	CAN	MNF	UR	AS	DAP	NH_3-N	$CaCN_2$
FLIEG et al. (1989) (% of N)	1.4	0.02	6.0	3.0	-	-	3.3
BUIJSMAN (1986) cited in ABEF (1989)	2	5	1015		10	10	
AMBERGER (1990) KNÜPPEL (1988)	1–5*	1–5*	10*	5–10*	1–3*	-	-

* Extreme rates om (loess) soils with pH 7.0 and share of free $CaCO_3$

Table 21:
NH,-N emission** rates **during the application of mineral fertilizers in the Federal Republic of Germany

A) **1987/88** (Bundesamt fiir **Ernährung** und **Fortswirtschaft/German** Ministry for Nutrition and Forestry = ABEF, 1989)

Fertilizer	Quantity **1987/88** in 1000 pure N	Emission factor in % of appl. N	NH,-N emission 1987188	
			in t	**kg/ha**
Calcium ammonium nitrate	1 100	2	***22 000***	-
Ammonium sulfate	4	15	***600***	-
Ammonium nitrate	2	10	***200***	-
Urea	110	10	11000	-
Calcium cyanamide	10	10	1 000	-
Ammonia gas	5	10	500	-
Multinutrient **fertilizer**	38	5”	19 150	
Total	1 614	3.37	54 450	***4.5***

* Extreme level for soils with **pH** 7.0 and share of free **$CaCO_3$**

B) 1986 (Isermann 1990)

Total	1 512	1.85	27 960	2.3

Table 22:
Net emission of gaseous total-N and of NH_3-N with intact beneficial plants

Plant	Period	Gaseous N emission (kg N/ha) Total N	NH_3-N	Authors
Wheat	Flowering-ripening	-	7.1	HAPER et al. (1987a)
Wheat	Pre-flowering Post-flovering	-	7.4 8.1 } 15.5	HARPER et al. cited in PARTON et al. (1988)
Wheat (winter)	Flowering-ripening	-	1.6	O'DEEN and PORTER (1986)
Wheat (spring)	Ear emerg.-ripening a) high } N b) moder. } supply		 4.4 2.8	PARTON et al. (1988)
Rice	100 d	14.1-15.2		DA SILVA and STUTTE (1981)
Rice	Vegetation period		1.9	NEUE et al. (1983)
Soybeans	Vegetation period	***45***		STUTTE et al (1979) STUTTE and WEILAND (1978)
Lucerne	Vegatation period	-	1.9	DABNEY and BOULDEN (1985)

Influential factors	low ←	NH_3 emission → high
N supply of the plant	moderate – low	high
Physiological age	young stage	senescence
Temperature	low	high
Transpiration	low	high

Table 23:
NH,-N emissions by agriculture in various Western European countries, in comparison to the data of BUIJSMAN et al. (1985) for 1992.

Country	NH,-N emissions from agriculture (t/a)			
	Bui jsman et al. (1985) Ref.yr. 1982	Other authors Quantity	Reference year	Authors
1. Fed.Rep. Germany	**298 000** (236 000 for	528 000	1986	ISERMANN (1990a,b,c)
2. Ex-GDR	165 000	215 000	1975189	MÖLLER (1990)
3. Netherlands	115 000	200 000	1986	CCMRX (1988)
4. Denmark	90 000	162 000	1980	SCHRODER. (1985)
5. Great Brit.	326 000	595 000	1978	ROYAL SOC. (1983)
6. Norway	29 000	47 ooo*	1980-89	BØCKMAN et al. (1990)
7. Switzerland		48 000	1987	STADELMANN (1988)

* gaseous total-N losses from livestock produktion

Table 24:
NH,-N emissions** in **the world (data in million t N/a) (Böttger et al. 1978)

Emission sources	*Northern hemisphere*	*Southern hemisphere*	*World*
Unfertilized soil	*0.6-l .3*	*0.148*	1.0–1.0
Miniral N fertilizer	*1.2-2.3*	*0.04407*	1.2-2.4
Livestock			
- cattle	11.8–17.5	2.6-3.8	14.4-21.3
- sheep	1.3–2-6	0.9-l .8	2.2-4.4
- pigs	1.2-1.7	0.1–0.2	1.3-1.9
- other	1 .4–2.2	0.4–0.6	1.8–2.8
Motor vehicles	0.2–0.3	0.1	0.2–0.3
Fossil fuels	0.1	0.1	0.1
Total emissions	18-28	4 7	*22-35*

Tabe125: The ammonia emmisions of agriculture as a component of its N balance

Country Reference year Author	Netherlands 1986 ISERMANN (1990)	Denmark 1950 SCHRØDER (1985)	 1980 SCHRØDER (1985)	 1958186 LAURSEN (1989).
1. Input (kg N/ha.a)	465	102	217	228
2. Output (kg N/ha.a)	465	102	217	228
2.1 Sales products	98	19	30	64
a) Crop production	14	7	10	28
b) Livestock prod.	84	12	20	36
2.2 Surplus (input – sales)	367 (100)	83 (100)	187 (100)	164 (100)
... of which NH,-N	99 (27)	25 (30)	57 (30)	44 (27)
3. N efficiency (%)				
3.1 Crop production	63	83	59	–
3.2 Livestock production	21	8	13	
3.3 Total agriculture	21	19	14	28
4. Share of domestic feed produktion in domestic crop production om a N basis (%)	94	84	87	–

(Table 25 continued)

Country Reference year Author	Fed. Rep. Germany 1986 ISERMANN (1990a-c)	Switzerland 1987 STADENMANN (1988)	Norway 1980–89 BØCKMANN et al. (1990)
1. Input (kg N/ha.a)	218	218	152
2. Output (kg N/ha.a)	218	218	152
2.1 Sales products	51	45	26
a) Crop production	23	10	6
b) Livestock prod.	28	35	20
2.2 Surplus (input – sales)	167 (100)	173 (100)	126 (100)
... of which NH3it0>-N	44 (26)	46 (27)	[50 (40)]*
3. N efficiency (%)			
3.1 Crop production	73		71
3.2 Livestock production	17	17	19
3.3 Total agriculture	23	21	17
4. Share of domestic feed produktion in domestic crop production om a N basis (%)	83	94	94

* gaseous total-N losses from livestock production

Table 26:
Actual and acceptable N emissions into natural ecosystems in Western Europe — Demands for the reduction of N emissions

Natural ecosystems	Atmospheric N emissions (kg/ha.a) Present situation	Atmospheric N emissions (kg/ha.a) Acceptable in the long term (critical loads)[1]
a) Open areas (nature preserves) such as high moors, heaths	<10–30	max. 10
b) Forests	10-200 (ø = 20-80)	max. 15-20
c) North/Baltic Seas	10	max. 5
... for coastal waters (southern)	9-15 (20)	max. *3-7*

[1]after NORDISC MINISTERRÅD (1985), NILSSON and GRENNFELT (1988); compare »naturel«: 4-5 kg N/ha.a (MENGEL, 1968: JOHNSTON et al. (1986) (open areas)

Table 27:
***Optimal applications of »slurry N« + »mineral fertilizer N«** in **kg/ha (Vetter et al.** 1990)*

	Humus sand		Slightly clayey silt		Range
	1988	1989	1988	1989	slurry N
Sugarbeets	0+60	100+60	150+0	140+30	O-150
Potatoes (starch)	1 10+60	100+60	150+30	110+0	100-150
Silage maize	90+30	80+60	140+0	80+30	80-140
Winter rye	120+30	140+60			120-140

Table 28:
Research needs for quantifying and reducing the ammonia emissions of agriculture

Sphere of N emission Quantity: FRG (1986)	Research needs
NH, losses 528,000 t/a ($54 44 kg/ha.a)	moderate: for specific processes; in general: implementation deficit greater than research deficit; NH, monitoring and N balancing in livestock husbandry to supervise implementation
1. Livestock husbandry (nutrition) ~ 500,000 t/a (≙ 42 kg/ha.a) collection storage application of farm yard manures pasture husbandry	large: measurement of the total N losses of diffenrent N removal systems in agricultural livestock production under comparable conditions (Figure 3) a) Farmyard manure management al) fermented manure (hot) a2) deep stable manure b) Straw composting with slurry c) Slurry management cl) untreated slurry c2) treated slurry c2.1) diluted slurry c2.2) aerobic (aerated) c2.3) anaerobic (biogas slurry) d) Pasture husbandry in comparison to meadow use with year-round stable use
2. Mineral **fertilizers** - production - storage - application (~ 28,000 t/a ≙ 2-3 kg/ha.a)	low research needs
3. Crops (NH, >> NO_x + N_2O) (?)	moderage: esp. NH, release after (too) high N fertilizationduring senescence (maturity), in legumes, in natural ecosystems

Do agricultural crops play a role in atmospheric ammonia pollution?

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Introduction

Numerous reports have documented that activities related to animal husbandry give rise to a considerable ammonia pollution of the atmosphere. The ammonia emission occurs from animal houses, manure storages as well as from animal manure applied to the fields. Similarly, several reports have shown that considerable amounts of ammonia may be lost from mineral fertilizers, in particular urea, lying on the soil surface.

Agricultural crops have not so far been considered as a source of atmospheric ammonia. There is, nevertheless, increasing evidence that agricultural plants growing under usual outdoor ambient NH_3 concentrations may give off ammonia to the atmosphere (Schjorring 1991). Ammonia emission is, however, not the sole role played by agricultural crops in atmospheric ammonia pollution. Young, vigorous plants growing close to a local source of ammonia like e.g. an animal house may absorb ammonia from the air instead of releasing it (Sommer and Jensen 1991). The role of agricultural crops in atmospheric ammonia pollution may thus be dual: in some cases the crop canopy may be a source of ammonia, in others a sink for ammonia.

The purpose of the present paper is to present some results on ammonia exchange between plants and the atmosphere. Furthermore, the paper briefly deals with the compensation point for ammonia in plants. This is done in order to give a basis for prediction of whether emission or absorption of ammonia will occur under a given set of conditions.

Ammonia volatilization from plants

Ammonia was for the first time identified as a volatile product given off by plants in 1928 (Klein and Steiner 1928). Since then several investigations have confirmed that plants are able to release significant amounts of ammonia from their leaves (Table 1).

From Table 1 it is seen that recorded rates of ammonia volatilization vary between 0 and 300 ng NH_3-N m^{-2} leaf surface s^{-1}. With a leaf area index of 5, as is typical for most agricultural crops, this roughly corresponds to a loss between 0 and 1 kg NH_3-N ha^{-1} day^{-1}. However, most of the results in Table 1 are obtained in short term studies under conditions not representative for

Table 1.
Rates of ammonia emission from plant leaves.

Plant species	*Flux* ng NH_3-N m^{-2} s^{-1}	*Reference*
Corn	0-10	Farquhar et al. 1980
Corn	30[c]	Weiland & Stutte 1979
Cotton	70[d]	Weiland & Stutte 1979
Rice	50-150[d]	Stutte & da Silva 1981; da Silva & Stutte 1981
Sorghum	50[d]	Weiland & Stutte 1979
Boybean	20–260[d]	Weiland & Stutte 1980; Stutte et al. 1979
Spring barley	0–40[b]	Schjorring 199 1
Spring wheat	20-300	Par-ton et al. 1988
Spring wheat	5–80	Morgan and Parton 1989
Weed species	50–160[de]	Weiland & Stutte 1979; Stutte & Weiland 1978
Winter wheat	<0–175[a]	Harper et al. 1987
Winter wheat	15–35[b]	O'Deen 1989
Winter wheat	0.4–1.5[c]	Hooker et al. 1980

a Field measurements during the course of one growing period
b Low light intensities
c No information on irradiance level
d The data include all volatile non-elemental nitrogen compounds. Emission of nitrogen oxides was usually less that 20% of that of reduced nitrogen forms. Fluxes were measured on single encased leaves with zero external partial pressure of non-elemental nitrogen compounds.
e Amarathus palmeri, Ipomoea hederacea, Datum stramonium, Xanthium pensylvanicum

a crop growing under normal field conditions. Accordingly, extrapolation to field conditions of the data in Table 1 has to be done with very great caution and with many reservations. Taking e.g. the results by Stutte and coworkers, they were obtained under conditions where the partial pressure of NH_3, NO_x, and of CO_2 were close to zero. This may have overestimated the volatilization. In contrast, experiments in closed chambers with a low rate of air renewal (e.g. O'Deen 1989) may have underestimated the ammonia volatilization because of relatively high ambient ammonia concentrations in the chamber.

The best basis for assessment of the role of agricultural crops in atmospheric ammonia pollution would of course be results from measurements carried out under field conditions. An additional requirement is that the results span the entire growing season and have been obtained with a method not influencing the microclimate in the crop, i.e. measurements carried out by use of a micrometeorological technique. Unfortunately, such measurements have until now only been carried out in very few cases.

Results from micrometeorological measurements of ammonia concentration profiles in two spring barley fields with low (40 kg N ha^{-1}) or high (160 kg N ha^{-1}) nitrogen status are shown in Table 2. The atmospheric ammonia concentration was higher in the barley field with high nitrogen status than in the field with low nitrogen status (Table 2). The higher ammonia concentration in the high-N field was due to ammonia emission directly from the plants because no ammonia was given off from the soil surface during the experimental period (Schjorring and Byskov-Nielsen 199 1).

Table 2.
Mean atmospheric ammonia concentrations in 2 heights above and 2 heights within the crop canopy on two spring barley fields. Mean values ± 95% confidence limits for the period between June 23rd and July 15th 1989 (62 to 85 days after seedling emergence). From Schjørring and Byskov-Nielsen (1991)

Height above soil surface* cm	Nitrogen application, kg n ha^{-1} 40	160
	Ammonia concentration, mg NH_3-N m^{-3} air	
250	1.5 ± 0.6**	***2.0 ± 0.7***
150	1.8 ± 0.6	2.0 ± 0.6
35	1.9 ± 0.6	2.8 ± 0.6
15	1.5 ± 0.7	3.9 ± 0.8

* Canopy height was appoxinately 70 cm.
** 95% confidence limits.

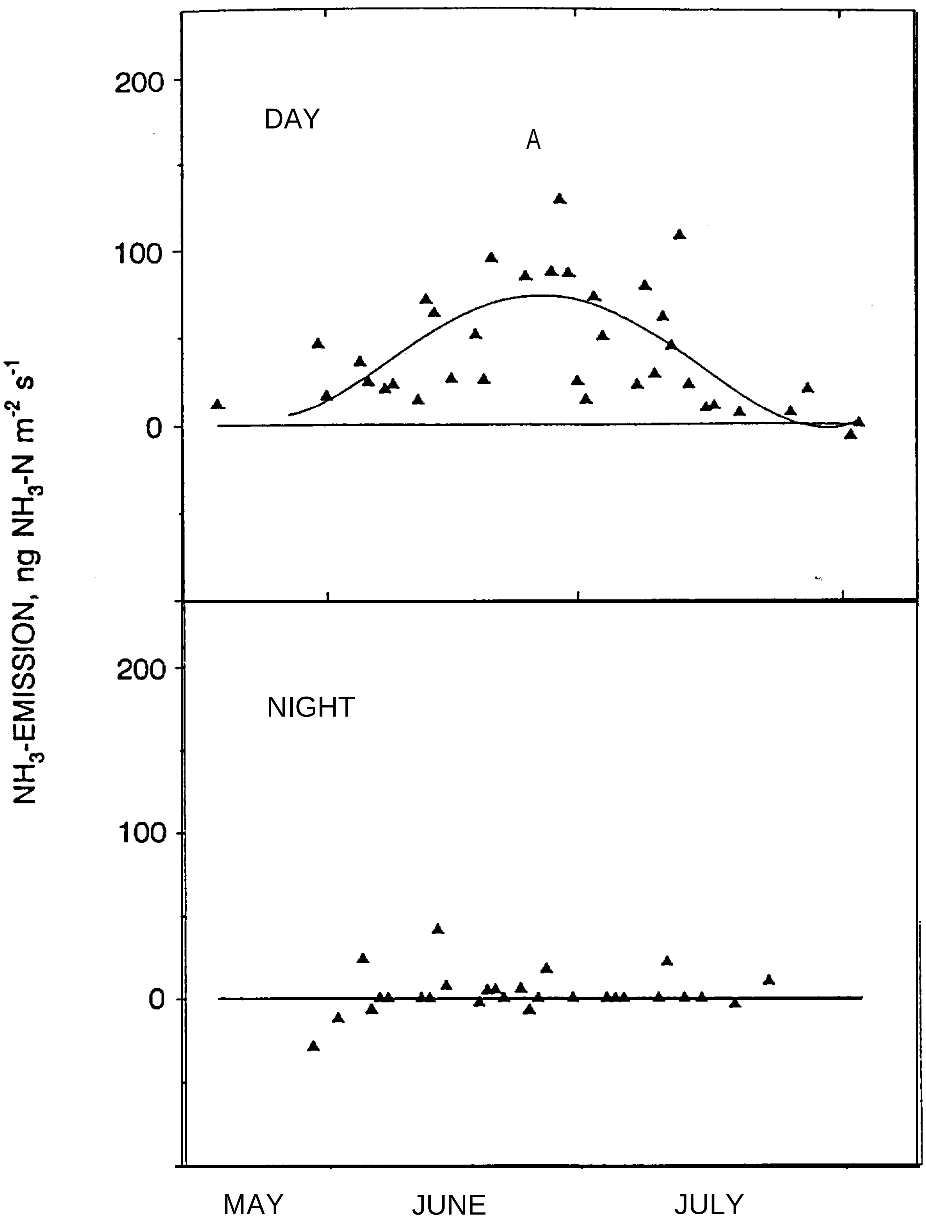
The time course of ammonia emission in June and July 1989 from a spring barley crop with high nitrogen status (applied 160 kg N ha^{-1}) is shown in Fig. 1. The flux of ammonia from the crop canopy to the atmosphere was relatively high in the last half of June and in the first half of July. This period coincides with ear emergence and accelerated senescence of the vegetative plant parts. The emission of ammonia was limited to the dayperiod (6 a.m. to 9 p.m.). The total loss of ammonia in the period amounted to 1.3 kg NH_3-N ha^{-1} in the field applied 160 kg N ha^{-1} (Table 3). Barley crops applied 40 or 120 kg N ha^{-1} lost smaller amounts of ammonia (Table 3).

Table 3.
Total ammonia loss, nitrogen harvest index, grain yield and grain protein content of spring barley applied 40, 120 and 160 kg N ha^{-1}, respectively, in 1989. Data for nitrogen harvest index, grain yield, and grain protein content are mean values ± 95% confidence limits ($n = 8$). From Schjørring and Byskov-Nielsen (1991)

	Nitrogen application (kg ha^{-1}) 40 N	120 N	160N
Ammonia loss (kg NH_3-N ha^{-1})	0.6	0.7	1.3
Nitrogen harvest index (%)	84 ± 2	80 ± 3	80 ± 3
Grain yield* (hkg ha^{-1})	64 ± 5	62 ± 7	63 ± 8
Grain protein content (%)	9.8 ± 0.5	12.6 ± 1.1	13.7 ± 0.8

* 15% water content

Figure 1.
Ammonia emission from the canopy of a spring barley crop during daytime (6 a.m. to 9 p.m.) and nighttime (9 p.m. to 6 a. m.) in June and July 1989. The ammonia emission is expressed in nanogram NH_3-N (ng = 10^{-9} gram) per square meter of the experimental area per second (Schjørring and Byskov-Nielsen 1991).



Compared to ammonia losses from manure, which on average in Denmark is estimated to be about 30 kg NH,-N per hectare of agricultural land, the measured ammonia loss from the spring barley crops was very small. The growth conditions in the experimental year (1989) were very favourable with high grain yields and high nitrogen harvest indexes (amount of nitrogen in grain relative to total amount of nitrogen in aerial plant parts; Table 3). This may have reduced the emission of ammonia. Schjorring et al. (1989) observed a loss of up to 40 kg nitrogen per hectare from barley plants with nitrogen harvest indexes below 0.63. A low nitrogen harvest index indicates that the capacity for nitrogen incorporation into the developing seeds during the reproductive growth phase has been restricted. This may have been caused by e.g. unfavourable climatic conditions. In such circumstances, the supply of nitrogen from the vegetative plant parts may be too high, giving the plant a ‘nitrogen hang-over’. The balance between the rate of nitrogen incorporation into reproductive plant parts and the rate of nitrogen supply from vegetative, senescing plant parts and roots may be particularly distorted in agricultural crops, because in agriculture it is a common practice to supply all fertilizer nitrogen at once, either at the time of sowing or to very young plants.

Harper et al. (1987) in a field **study** using a micro-meteorological technique found a total loss of about 15.5 kg NH,-N ha^{-1} from a wheat crop. The NH, loss **occured** as a post-anthesis loss of 7.4 kg NH,-N ha-‘, and a pre-anthesis loss of 8.1 kg NH,-N ha^{-1}. The rather large pre-anthesis losses took place in a period in which the wheat seedlings was absorbing fertilizer-N at high rates. In the same experiment, in a period with low soil-N availability prior to anthesis, wheat plants absorbed NH, from the atmosphere instead of released it (Harper et al. 1987). Plant N deficiencies can thus apparently result in NH, absorption instead of NH, emission.

Very little is still known about the influence of plant age, plant N status, climatic conditions, and edaphic conditions on the rate of ammonia **vola**-tilization from agricultural crops. The data *in* Table 3 indicate that NH, emission increases with the N status of the plant material. This is in accordance with growth chamber experiments with spring barley (**Schjør**-ring 199 1) and with spring wheat (Par-ton et al. 1988). Extrapolation of the data from Par-ton et al. (1988) to field conditions gives a loss of 2.8 and 4.4 kg NH,-N ha^{-1} for low-N and high-N plants, respectively, during the reproductive growth phase.

Ammonia absorption

Absorption of NH, by several different plant species has been found at high ambient NH, partial pressures. The absorption responds linearly to ambient NH, partial pressure in the range up to 700 nbar (Meyer 1973; van Hove et al. 1987; Whitehead & Lockyer 1987) and even from 1000 to 20000 nbar (Hutchinson et al. 1972). This indicates that ammonia, once past the stomata, quickly can be metabolized to amino acids and proteins. The high potential for ammonia absorption may be utilized for reducing ammonia losses under spreading of liquid animal manure. This can e.g. be done by application of manure in spring to the soil surface between the rows of autumn-sown cereal crops. In such cases, agricultural crops reduce the ammonia pollution of the atmosphere.

Data for winter wheat indicate that NH, emission from young plants during periods of rapid fertilizer N uptake may be followed by absorption of NH, in periods of low soil-N availability before **anthesis** (Harper et al. 1987). There is very little knowledge of the interaction between nitrogen uptake from the growth medium and exchange of gaseous nitrogen compounds between the leaves and the atmosphere. That there indeed is some interaction is shown by data for Italian **Ryegrass (Lolium** multiflorum): abundant supply of NOJ in the root medium decreased the amount of NH, absorbed by shoots growing in an atmosphere with NH, concentrations ranging from 14 to 709 μg NH, m^{-3} (Whitehead & Lockyer 1987). The proportion of plant nitrogen derived from atmospheric NH, increased from 4% at 14 μg NH, m^{-3} to 77% at 709 μg NH, m^{-3}.

Ammonia compensation point

Stomata represent the main pathway for gas exchange by terrestrial plants. If the partial pressure of ammonia in the atmosphere exceeds that in the substomata1 cavities, absorption will occur, while in the opposite case ammonia will be lost. This can be quantified by the following equation (Cowan 1977; Farquhar et al. 1980, 1983):

$$J = g(P_a-P_i)/P$$

where J is the molar flux density of NH, between plant leaves and atmosphere, g (mol $m^{-2} s^{-1}$) is the conductance to diffusion of NH, through the leaf surface, p_a (bar) is the ambient partial pressure of NH,, p_i (bar) is the partial pressure of NH_3 in the intercellular space (substomatal cavities) of the leaf, and P (bar) is the atmospheric pressure. The conductance (the inverse of resistance) includes stomata1 and cuticular conductances in parallel and usually also includes a boundary-layer resistance in series with the other terms.

According to Equation 1, the rate and direction of plant NH, fluxes is a function of the NH, partial pressure gradient between the stomata1 cavities and the atmosphere. Net emission of NH, will take place when the intercellular partial pressure of ammonia is greater than the ambient partial pressure. Any physiological process tending to raise p_i above p_a would lead to increased NH, efflux. When p_i exceeds p,, environmental factors and/or changes in leaf stmcture increasing the conductance to diffusion of NH, through the leaf surface would also increase NH, efflux. The internal partial pressure of ammonia probably varies with the ammonium concentration in the leaf apoplast, reflecting the balance between ammonium generating processes in the leaf (most importantly photorespiration, senescence induced protein degradation, and nitrate reduction) and ammonium utilizing processes (glutamine synthesis).

The atmospheric NH, concentration at which neither emission nor absorption of NH, takes place (no net exchange of NH,) is known as the NH, compensation point. No net exchange of NH, corresponds to the situation in which p_i equals p_a in Eq. 1. The existence of a NH, compensation point in plants was established by Farquhar et al. (1980). For several plant species they determined compensation points ranging between 1 and 3 µg NH,-N m^{-3} (1 µg NH, m^{-3} is equal to 1.43 nbar NH, at 20°C). Recently, Morgan and Par-ton (1989) have shown that the NH, compensation point may vary considerably during the life cycle of spring wheat plants: during early grain tilling, the compensation point was 13 µg NH,-N m^{-3}; it increased to 23 µg NH,- N m^{-3} during late grain filling, and was at the stiff dough stage higher than 25 µg NH,-N m^{-3}. Measurements of NH, concentration gradients in a winter wheat crop suggest a NH, compensation point around 15 µg NH,-N m^{-3} (Harper et al. 1987). Atmospheric NH, concentrations around 1 to 3 µg NH,-N m^{-3} have been measured in grass-clover canopies (Denmead et al. 1976), in canopies of Agropyron repens L. (Lemon and van Houtte 1980). in soybean canopies (Harper et al. 1989; Lemon and van Houtte 1980), and in spring barley canopies (Table 2).

In natural conditions, the compensation point for NH, is likely very close to the partial pressure of NH, in the substomatal cavities (p_i in Eq. 1). The existence of a NH, compensation point fairly close to natural ambient NH, levels (1 to 5 µg NH,-N m^{-3}) explains why NH, may be either absorbed' or emitted from the plant tissue under natural conditions.

Concluding remarks

Significant amounts of NH, can in some cases be lost from the foliage of growing plants. In other cases, particularly at high ambient ammonia concentrations, the crop may act as a sink for ammonia. The available data suggest that ammonia exchange over agricultural fields has significant relevance for N cycling in the soil-plant-atmosphere system. However, too few data are available to make a safe estimate of the importance of agricultural crops in atmospheric ammonia pollution.

Ammonia exchange takes place in response to a difference in the partial pressure of ammonia between the outside and inside of a plant leaf. Emission occurs when NH, releasing processes proceed more rapidly than NH, assimilating processes. The emission increases with plant age and nitrogen status. Furthermore, the rate of NH, emission depends on the nitrogen economy of the crop (nitrogen source-sink relationships). High levels of nitrogen application early in the growing period in combination with unfavourable growth conditions later on in the growing period can presumably give rise to large ammonia losses. The physiological and biochemical processes underlying plant NH, emission are still poorly understood. The same is the case for the influence of climatic and edaphic factors on plant NH, emission. Systematic investigations of the NH, emission rates under a variety of growth conditions are needed.

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The Danish monitoring programme on nutrient leaching from selected catchment basins

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Summary

The Danish Land Monitoring Programme started up in 1988/89. It deals with the NPo input from agriculture to the aquatic environment in six small catchment basins. The programme covers: i) a detailed soil profile description, ii) questionaire survey on land use and farming practices, iii) sampling stations -precipitation, soil water (root-zone), drainage water, upper ground water and streams.

Results from the first year of sampling, 1989, are presented. The precipitation was low, about 80% of the annual average for the years 1960-89. Nitrate leaching from the root-zone amounted to 78 kg N/ha/year for one sandy catchment basin and 42 kg N/ha/year for three loamy catchment basins. Nitrate leaching from drainage schemes on loamy soils were on average 12.9 kg N/ha/year. The nitrate leaching increased frem the East to the West of the country, coinciding with a change from heavier to ligther textured soils, and an increase in precopitation and livestock density. The export coefficient from the open land to the streams were measured at 10 kg N/ha/year for the sandy catchment basin and at 12.2 kg N/ha/year for the three loamy catchment basins.

Introduction

In 1987 the Danish Parliament adopted the »Plan of action against nutrient pollution of the Danish aquatic environment+. Over a 5 year period the plan calls for an investment of Dkr. 4.5 billion in agriculture, 1.5 billion in the industry and 6.0 billion in municipal sewage plants. The main objectives of the plan is to reduce nitrogen and phosphorus leaching by 50% and 80% respectively before 1994. To follow the input trends and to register changes in nutrient loadings arising from the Action Plan, a nationwide monitoring programme was initiated in 1988/89. The programme is unique in that it covers all environs, i.e. the atmosphere, the land, the ground water, the surface water and the near-shore and marine waters (Miljøstyrelsen, 1989) The Land Monitoring Programme deals with the nitogen, phosphorus and organic matter input from agriculture in selected catchment basins. The objectives are to follow the integrated impacts of farming on leaching of nutrients, particularly nitrogen through the soil to the surface water and ground water. The Monitoring Programme is described in this paper, and results from the first year (1989) are presented.

Description of the land monitoring programme

Description of the selected catchment basins

The Land Monitoring Programme has been established in 6 small well-defined catchment basins. Each catchment basin covers an area of 5-15 km² and includes 12-45 farms. The catchments have been selected to cover the major soil types, climatic variations and farming practices of Denmark. The locations are shown in Fig. 1, and some details are given in Table 1. It may be seen that 3 catchments have been selected to represent the loamy soils on the Islands and East Jutland and 3 catchments to represent the sandy soils in West and North Jutland. The catchments may be further characterized by a general increase in precipitation from the East to the West of the country; thus the annual rainfall for the years 1960-89 increased from 615 mm at catchment No 1 to about 980 mm at catchment No 5 and 6. Also the livestock density increases from East to West.

Organisation of the Monitoring Programme

The ***monitoring programme includes:***

- Soil survey
- Questionaire survey on land use and fertilization practise
- Sampling stations (see Appendix 1):
  Rain gauge st.
  Soil water st.
  Drainage st.
  Ground water st.
  Stream st.

The Danish Environmental Protection Agency has the overall responsibility for the monitoring programme whereas the National Environmental Research Institute (DMU) and the Danish Geological Survey (DGU) are responsible for the coordination and technical reporting of the programme. The regional authorities are responsible for carrying out the questionaire survey and for running the sampling stations.

Description of data sampling

Soil survey:
The soil survey has been carried out by the Bureau of Land Data (ADK), the Ministry of Agriculture. 10-l 1 profiles have been described and a large number of core samples taken in each catchment basin. Detailed soil maps have been produced (Jensen and Madsen, 1990).

Questionaire survey:

The questionaire survey are carried out once a year. All farmers in the catchment areas participate. The objectives of this component are to obtain a statistical estimate of the farming practices of the areas and to gather information which is needed for modelling nutrient leaching from individual fields.

The information of the questionaire survey covers:

- Fields: Soil type and drainage
- Landuse: Crops, yields, use of straw / crop residues, green fields, and exact times for ploughing, sowing and harvest
- Fertilization: Fertilizer/farm yard manure – type, amount of application, time of application, and time for ploughing in
- Livestock: Type, numbers, grazing, production of manure, storage capacity of slurry.

Soil water stations:

The objective of the soil water component is to calculate the nutrient leaching from the root-zone of the selected fields. The nutrient concentrations of the soil water are measured experimentally whereas the water flow from the root-zone is estimated by means of the Soil-Plant model, DAISY (Hansen et al., 1990).

In each catchment basins 6-8 soil water stations have been established. A soil water station consists of 10 tensiometer cells (teflon cells) for extraction of soil water. The cells are placed in a V-shaped pattern within an area of about 100 m^2. The cells are installed under the root-zone, at a depth of 1 00–200 cm (Blicher-Mathiesen et al., 1990). Soil water samples are extracted using a »continuous vacuum technique+ (Baiers et al., 1989). The initial vacuum being ~ 0.7 bar. Samples are taken for chemical analysis once a week in periods where run-off occurs (Appendix 1).

Drainage stations:

The drainage programme has been established in the three loamy catchment basins with 6-8 stations in each catchment. In two catchments (No 1 and 4) a large proportion of the land has been tile drained; here the drainage stations have been established on existing drainage schemes in connection with the soil water ***stations. In the*** third loamy catchment (No 3) the soils are mainly freely drained because of a sloping topography; drainage schemes in this catchment basin are rare, and the drainage stations are placed more randomly.

About one out of three drainage stations are automatic stations, i.e. with datalogger and 30” Thomson overflow for continuous measurements of the discharge. The other stations are manual stations, i.e. the discharge from the drains is measured manually once a week, and the total discharge from the scheme is calculated by correlation to the automatic stations. Samples are taken for chemical analysis once a week in periods where discharge occurs (Appendix 1).

Ground water stations:

This component deals with the quality of the upper ground water. In each catchment basin 21-25 stations have been established. Two stations are placed in connection with each soil water station and the remaining stations are distributed over the catchment area. A ground water station consists of

3 filters installed at 1.5, 3.0 and 5.0 m depth. The samples are extracted according to the Montejus-pump principle using nitrogen gas. The filters are clear-pumped for 2 days before the samples are taken for chemical analysis. Samples are taken 10 times a year. The samples are analyzed individually from the three depths (Appendix 1).

Stream stations:
This component deals with the quality of the stream water, and transport of nutrients by the stream. An export coefficient for nutrient leaching from the open land may be calculated.

In each catchment basin 2-5 stream stations have been established. The stations are set up with datalogger for automatic registration of the waterlevel of the stream; the flow is then calculated by means of calibrating waterflow measurements. Samples are taken for chemical analysis once a week in the period 15th Oct.- 15th April, and every second week in the period 15th April-15th Oct. The programme for chemical analysis corresponds to that for drainage water (Appendix 1).

A biological stream water programme is also carried out. It will not be dealth with here.

Results and discussion

The main results from the first year, 1989, are shown in Table 2 and 3. The monitoring programme for two sandy catchment basins (No 5 and 6) was initiated late in 1989; results from these two areas are therefore not presented.

Concentration levels

The average concentrations of nitrogen and phosphorus for 1989 are shown in Table 2 (Blicher-Mathiesen et al., 1990; Rasmussen and Gosk, 1990). On the three loamy areas the average concentrations of nitrate were 24.6 mg N/l in the soil water, 14.9 mg N/l in the drainage water and 6.2 mg N/l in the upper ground water. On the sandy area the concentrations were 39.6 mg N/l in the soil wafer and 22.7 mg N/l in the upper ground water. This implies that the nitrate concentrations generally are highest on the sandy areas, and that a larger proportion of the nitrate is leached into the ground water on the sandy areas.

The ammonium concentrations were low in the soil water and the drainage water but considerably larger in the upper ground water.

The phosphate concentrations in all the aquous environs are generally low but with some occasional large values.

Water flow and nitrate leaching

Precipitation, water flow and nitrate leaching measured at the sampling stations in 1989 are shown in Table 3 (Blicher-Mathiesen et al., 1990).

The precipitation in 1989 varied from 553 mm at catchment basin No 1 to about 620 mm for the other three catchments. This was on average 81% of the annual precipitation for the years 1960-89.

Water flow from the root-zone was estimated to be 120 mm at catchment No 1, and about 200 mm at catchment No 2, 3 and 4. Nitrate leaching from fields with sampling stations varied from 9-146 kg N/ha/year. The average leaching from the sandy catchment area was found to be 78 kg N/ha/year and for the three loamy catchment areas 4 1 kg N/ha/year. Within the loamy areas, the lowest leaching was found at catchment No 1 (20 kg N/ha) and the highest at catchment No 3 (7 1 kg N/ha). Several reasons may explain 'this large variation in nitrate leaching. With reference to the characteristics of the catchment basins and the actual climate, it will appear that nitrate leaching increases with the lighter soils, increasing precipitation and increasing number of livestock per unit land. A similar variation and level of nitrate leaching has been reported by Hansen (1990) for the years 1987-90. Furthermore, a nationwide modelling of nitrate leaching from the root-zone for the years 1986-90 was found to be 72 kg N/ha/year for sandy areas and 5 1 kg N/ha/year for loamy areas (Nielsen et al., 199 1).

Modelling of nitrate leaching from the root-zone at catchment level will be carried out at a later stage. The Soil-Plant model, DAISY, will be employed (Hansen et al., 1990) using the information from the questionaire survey, and climate and soil data as input data. The experimentally obtained values for nitrate leaching will be used for comparison and calibration of the model.

Discharge from the drainage schemes was low and variable in 1989 due to the low precipitation. Nitrate leaching from the drains was on average found to be 12.9 kg N/ha/year.

The water flow to the streams is shown in Table 3 as an average export coefficient from the open land. It may be seen that the flow to the streams varies considerably from the run-off from the root-zone. It is not expected that these two values should be identical for a single year. However, it may also be possible that the catchment areas for the ground water table is not fully known.

The average export coefficient for nitrogen in 1989 amounted to 10 kg N/ha/year for the sandy catchment basin and to 12.2 kg N/ha/year for the three loamy catchments. Within the loamy areas the lowest export coefficient was found for catchment No 1 (5.6 kg/ha) and the highest for catchment No 3 (17 kg/ha). In comparison, the Nationwide Stream Monitoring Programme for 1989 showed an average export coefficient for the open land of 13.2 kg N/ha/year, and it was the same for sandy and loamy catchments. In a year with »normal« precipitation, however, the coefficient is estimated to be 20-24 kg/ha for sandy areas and 25–30 kg N/ha for loamy areas (Kristensen et al., 1990).

The nitrogen leaching to the streams is a result of the water flow pattern and therefore much affected by the actual rainfall. On loamy areas the flow occurs mainly via drainage water and through the upper soil layers with large nitrate concentrations. On sandy areas, however, the flow occurs mainly via the ground water where nitrate concentrations are much lower.

Final remarks

Results from the first year of the Land Monitoring Programme has shown some trends in nitrate leaching. However, it is obvious that much longer time series of data are needed to evaluate the integrated effects of agriculture on nutrient inputs, leaching and distribution to the aquatic environments.

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Figure 1.
Locations of the selected catchment basins.

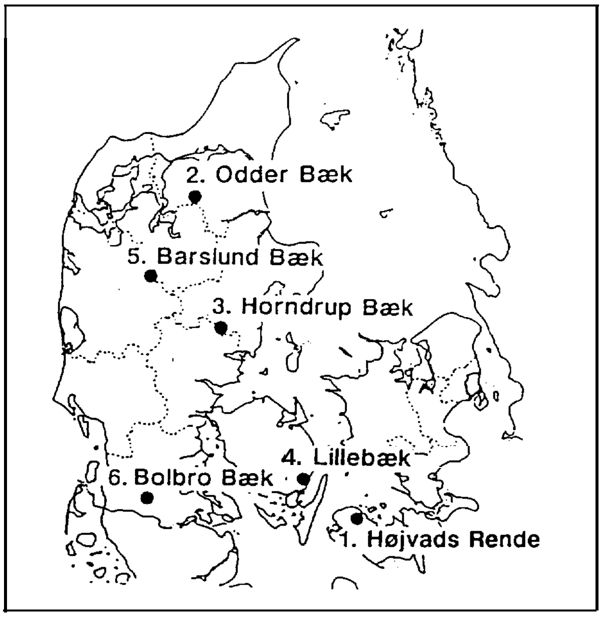

Table 1.
Characteristics of the six catchment basins.

Catchment	Area ha	Soil type	Soil % clay 10–50 cm	Annual[1] rain mm	Farming No of farms	Farming Av. farm size, ha	Farming Livestock unints/ha[2]
1	980	loamy	16	614	36	22	0.22
3	530	loamy	16	875	28	13	0.66
4	470	loamy	15	704	22	18	0.64
2	1140	sandy	5.4	794	45	24	1.63
5	1470	sandy	3.7	969	12	36	0.78
6	1330	sandy	5.8	993	29	26	1.29

[1] Average for the years 1960-89

[2] 1 livestock unit = 1 dairy cow based on a manure nitrogen content of approx. 100 kg N/year (Laursen, 1987)

Table 2.
Average nutrient concentrations in soil water, drainage water and upper ground water for four catchment basins in 1989.

Catchment	*Stations*	*pH*	NO_3-N mg/l	NH_4-N mg/l	PO_4-P mg/l
Loam 1	Soil water	*7.9*	18.8	0.015	0.105
	Drainage water	*7.5*	12.6	0.005	0.054
	Ground water		4.4	0.62	0.03
3	Soil water	*7.4*	37.3	0.032	0.052
	Drainage water	*7.3*	12.7	0.009	0.040
	Ground water		6.9	0.47	
4	Soil water	*7.8*	17.9	0.018	0.013
	Drainage water	*7.7*	22.3	0.027	0.040
	Ground water		7.3	2.33	co.02
average	Soil water	*7.7*	24.6	0.022	0.057
	Drainage water	*7.5*	14.9	0.014	0.045
	Ground water		6.2	1.14	
Sand 2	Soil water	*6.5*	39.6	0.025	0.002
	Ground water		22.0	0.62	0.02

Table 3.
Precipitation, water flow and nutrient leaching in 1989 for four catchment basins. The data represents soil water (root-zone leaching), drained areas and streams (given as the export coefficient for the open land og the catchment basin). Minimum and maximum values are shown in brackets.

Catchment	*Precipitat. mm*	*Stations*	*Water flow mm*	*Run-off kg N/ha*	
Loam 1	*553*	**Soil water**	**120**	**20**	**(943)**
		Drains	**9 2**	**13**	
		Stream	**73**	**5.6**	
3	*623*	**Soil water**	**183**	**71**	**(20-146)**
		Drain	**170**	**18**	
		Stream	**214**	**17**	
4	*634*	**Soil water**	**239**	**37**	**(21-78)**
		Drains	**35**	**7.8**	
		Stream	**142**	**14**	
average	*603*	**Soil water**	**181**	**42**	
		Drains	**99**	**13**	
		Streams	**143**	**12**	
Sand 2	**619**	**Soil water**	**180**	**78**	**(48-1 03)**
		Streams	**208**	**10**	

Appendix 1

Summary of sampling stations and chemical analysis carried out in the Land Monitoring Programme.

Sampling stations:

Type	No of *stations per catchment*	*Total no of stations*
Rain gauge sta.	1-2	10
Soil water sta.	6-a	40
Drainage sta.	6-a	21
Ground water sta.	21-25	136
Stream sta.	2-5	21

Chemical analysis:

Soil Water	*Drainage water*
(once a week in periods with run-off)	(once a week Oct.-Apr. once every 2 weeks Apr.-Oct.)
pH	pH
Nitrate-N	nitrate-N
ammonium-N	ammonium-N
phosphate-P	phosphate-P
	(total N)
	(total P)
	(potassium)
	(alkalinity)
	(conductivity)
	(COD)

Ground water	*Streams*
(10 times a year)	(once a week Oct.-Apr. once every 2 weeks Apr.-Oct.)
pH	pH
nitrate-N	nitrate-N
ammonium-N	ammonium-N
phosphate-N	phosphate-P
calcium	total N
magnesium	total P
potassium	alkalinity
sodium	conductivity
iron	COD
Chloride	
sulphate	
alkalinity	
acidity	
conductivity	

Nitrogen leaching from the North Poland river watersheds

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The leaching of nutrients from the agricultural watersheds conditioned by the hydrologic cycle is very unfavourable, both for the agriculture and for the quality of ground – and surface waters. The knowledge of conditions and mechanisms as well as the extent of nutrient runoff from the river watersheds of the Baltic drainage area is indispensable for taking proper water and economic decisions to counteract progressing degradation of the marine environment.

The above mentioned questions were the subject of many years investigations carried out by in the Institute of Meteorology and Water Management, Department of Water Protection in Gdahsk (Taylor 1984, Taylor et al. 1986, Taylor 1987).

The study was carried out in 1987-l 989 in watersheds, which are typical of the Middle Pomerania Lake Districts which have no significant point pollution sources. The selected watersheds are situated in the Vistula and Reda drainage basins and are utilized for agricultural purposes in 16 to 90%.

They are characterized by diversified surface sculpture and very easily permeable soils, on which agriculture of a rye-potato structure has been developed. The use of fertilizers during the investigation period ranged from 130 to 330 kg NPK/ha of arable land. Large resources of ground waters together with those retained in lakes cause the river alimentation to be relatively uniform and amplitudes of the water table to be small.

Annual loads of nutrients flowing off the watersheds were calculated on the basis of regressional daily loads.

These loads were obtained from measurements carried out every week in cross sections bounding the watersheds, a positive correlation being utilized between momentary loads and the corresponding momentary flows. (Zieba, 1980).

The basic material for estimation of nitrate nitrogen runoff dynamics was provided by daily measurements of its level in the water sampled at the cross section closing a small (6,3 km²), typically agricultural watershed the reaction of the river flow and nutrient concentration to the weather changes were there much faster and greater than in the bigger rivers.

Hydrological and meteorological data were prepared by specialized Department of the Institute.

The results of analogous investigations in the watershed selected in Great Poland Lake District in Odra drainage basin were used in this paper for the comparative purposes. The area at this watershed is rather plain, cut with riverine valleys.

It is covered with the moderately or poorly permeable, and fairly fertile soils. Poor resources of ground water caused rapid lowering of their levels and curtail runoff during dry season.

Results and discussion

Variations of nutrient concentrations and outflowing loads as compared **to** river hydrograph and environmental conditions were analyzed in details.

It has been proved, that dynamics and levels of nutrient concentrations in the river are closely related to the physicochemical properties of nutrients, being decisive of the routes of their transport from the watershed to river waters.

So, the amounts of nutrient runoff depend on current contribution of different sources to the total river alimentation, as well as on the volume of water taking part in the leaching processes i.e. on the meteorological and hydrological conditions.

Nitrates being well soluble and non absorbed on soil penetrate into river from underground waters, subsurface flow and overland flow.

The first source usually supplies a small amount of nitrate; namely, on their way to ground waters, nitrates undergo gradual denitrification, if they penetrate through the root system of plants. The subsurface flow, and water from the drainage pipes, which transport nitrates directly from upper layer of soil, are most abundant source of nitrate nitrogen in rivers. Nitrate concentration in the overland flow is variable, and according to our findings it greatly depends on the coincidence of the overland flow with the time of nitrogen fertilization.

So, the nitrate concentration in river water increases with flow i.e. with a rise of the contribution – to river alimentation – of more shallow ground waters, and then of the overland flow till the moment when the dilution processes begin to predominate over the washing off processes.

The NO,-N concentrations in the Pomeranian river waters where very low and ranged from 0.01 to 3.0 mg N/l. It resulted not only from the poorness of the soil, but first of all from specific water relations in the watersheds, i.e. the small amount of subsurface and surface flows because of very permeable soils. Quite different situation in the Great Poland watershed resulted in the large amplitude of nitrate concentration in the river water, i.e. 9.01 to 20.0 mg N/l. When the moisture conditions are good there, the groundwater table reaches nearly soil surface, and the contribution of the shallow soil waters to the river alimentation is great. Fig. 1 examplifies the annual (Oct. 82-Sept. 83) course of nutrient contents in the waters of Pomeranian watershed, as compared to the flow and meteorological conditions. It is evident, that there is a positive correlation between N-NO, level and the flow. This tendency is well illustrated by a plot of the relationship between nitrate concentration and river flow.

Fig. 2 shows this relation very clearly for the Great Poland watershed within the investigation cycle 1978/79. You can see rapid increase of the concentration curve with the rise of flow and its bent corresponding to the greatest flows during the intensive snow thaw. Naturally, the average nitrate content in river waters was dependent also on the utilization degree of the watershed by the agriculture. The waters of typical forest rivers were characterized by the lowest nitrate concentration (0.01 to 1.4 mg N/l).

Intensity of washing out the nutrients from the watershed is closely related to its hydrological conditions.

Therefore the amounts of nutrients discharged from the given catchment areas depend on the amount of water runoff. In the case of nitrates the regression curves based on the experimental data (momentary loads versus the flow) were of the power nature, while the correlation coefficients were greater, or very close to 0.9.

A similar positive correlation for the examined watersheds has been proved between the amount of nitrogen introduced with fertilizers, manure and precipitation and the nitrate nitrogen unit runoff.

Finally a statistic relation has been stated between the nitrogen unit runoff and two parameters which most of all influenced its value, i.e., the loading of watershed with nitrogen, and the sum of years precipitation. The six years experiments carried out in the Redda River drainage area constituted the basis for this relation. It is described by the equation:

$$Y = b + a\, x_L x_P$$

where

Y = Nitrogen runoff, kg/ha year
x_L = Watershed loading with nitrogen, kg/ha year
x_P = Sum of years precipitation, mm

and has been established by means of the multiply correlation method. Naturally the values of constants differ for the catchment areas of different physiography, the way of land use, as well as for some individual features.

As you can see in Table 2 the unit loads of nitrates and total nitrogen runoff from the Pomeranian watersheds ranged from 0.2 to 6.6 kg NO,-N/ha year and from 0.9 to 13.0 kg N/ha year respectively.

Its value was mainly dependent on the agricultural utilization degree and much less on the meteorological conditions.

The same parameters as in the case of nitrate concentration in river water for the least quantity of nutrients runoff from the forest area were characterized by 0.2-l .O kg NO,-N/ha year and 0.9-2.7 kg N/ha year respectively.

For all these reasons the runoff of nitrogen especially in the form of nitrates, from large area of Middle North Poland is relatively low and not highly variable. For the comparison, the unit nitrate and nitrogen runoff from the investigated Great Poland watershed ranged from 17.7 to 6 1.2 kg NO_3-N/ha and 18.5 to 64.6 kg N/ha year respectively. The great differentiation of years loads and predominant contribution of nitrates to the total nitrogen are evident here.

Dynamics of the nitrate runoff may be illustrated very well by means of the load cummulative curves. Fig. 3 shows such curves plotted for one of the Pomeranian agricultural watershed and for two investigation cycles (July 87-June 88).

In the year 1987/88 the total nitrate load was greater by 18% than in 1896/87, and it increased as much as the water runoff. During both investigation cycles the majority of the water – and nitrate runoff was connected with the snow cover thaw (95% and 68% respectively). During the all cold months (Nov.-March) nitrate runoff amounted to 96% and 88% respectively. The cases of even considerable increase of the nitrate concentration in the summer were of no importance for the total runoff from the watershed (high evapotranspiration).

The proportions of nitrate runoff from the Great Poland watershed in the cold and warm seasons were similar to those, from the Pomeranian watersheds (Fig. 4). But the one years loads differed very much. Intensive thaw of exceptionally thick snow cover in the year 1978/79 resulted in 3 times as large water runoff, and 3.5 times – nitrate runoff as the year 1977/78.

While in quite similar weather conditions water runoff from the Pomeranian watersheds increased only to 20% and nitrate runoff raised by 50%.

Watching figures 3 and 4 one can understand the difficulties created by the problem of nitrate leaching from the agricultural areas. The extent of nitrate transport depends mainly on the volume of water which can not be controlled in any significant degree.

The only solution is to continue the trend towards diminuation of nitrate concentrations in the water runoff and in the precipitation. It needs highly rational fertilizer – and manure economics in the agriculture, and on the other hand – the reduction of the nitrogen oxides in the air.

Conclusions

- The specific water relations prevailing in the Middle Pomerian watersheds of poor and permeable soils, as well as traditional agricultural economics determine the relatively low nitrate runoff from this area.
- The greatest nitrate concentrations in the water runoff from the agricultural watersheds arise during the periods of maximum soil saturation with the mosture during the cold seasons of the year as well as in the cases of overland flows, if they occur directly after nitrogen fertilization.
- The greatest nitrate loads in the water runoff from the agricultural watersheds occur during the periods of the intensive and prolonged rains in the conditions of soil saturation with water in the cold season of the year, as well as during intensive thaw of a thick snow cover.
- Diminuation of nitrate concentrations in water runoff can be achieved by highly rational fertilizer (in it manure) economics and the reduction of nitrogen oxides in the air.

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Table 2.

Potozenie zlewni	*Rodzaj zlewni*	*Runoff kg ha-' y-1* NO,-N	N_{tot}
Middle Pomorania	**Zlewnie rolnicze (57–75% of agr.area)**	**0.4-3.1**	**1.9-6.1**
Middle Pomorania	**Zlewnie Leśne (58–84% of forest area)**	**0.2-l .o**	**0.9-2.5**
Great Poland	**Zlewnie rolnicze (93% agr.area)**	**17.7-61.2**	**18.5-664.6**

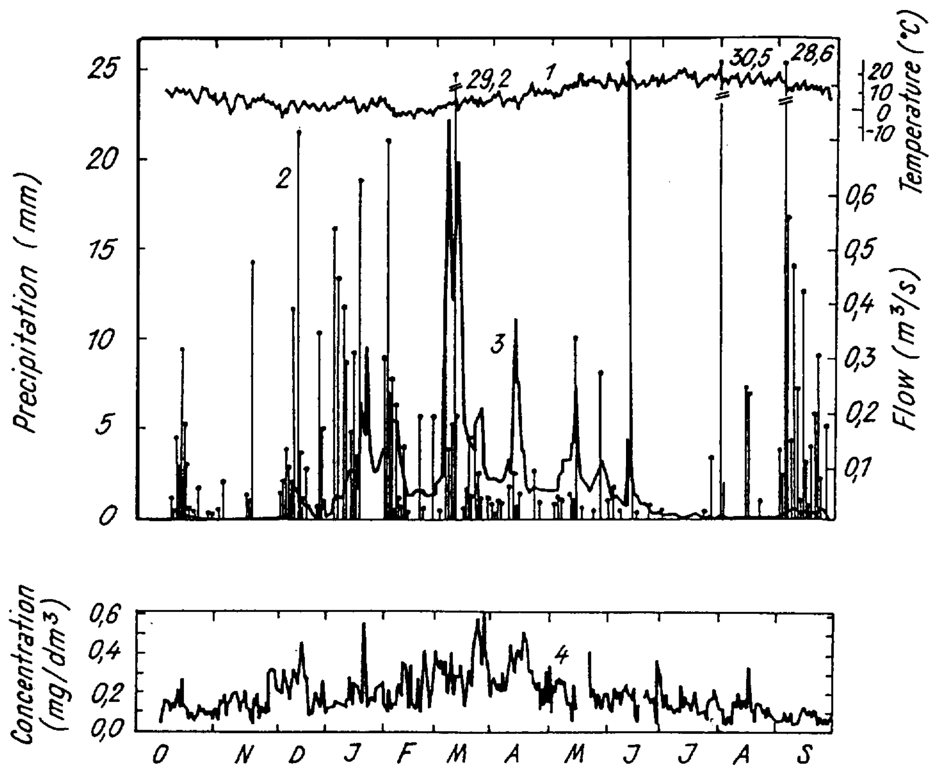


Fig. 1.
The effect of meteorological and hydrological conditions on formation of nutrient concentrations in Krzeszowska Struga River waters between Oct. 1982 and Sept. 1983.
1 – air temperature, 2 – precipitation, 3 – water flow, 4 – NO,-N

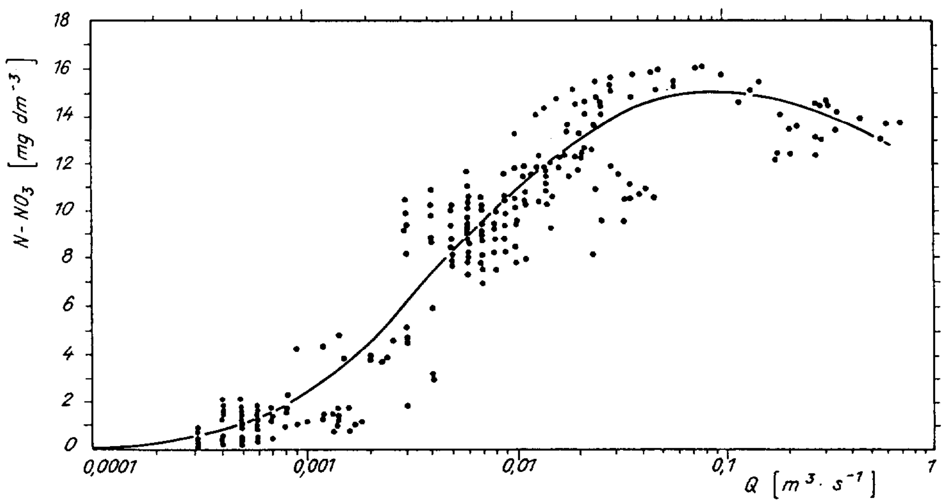


Fig. 2.
Relationship between flow volume and nitrate level in the Pomorka. Investigation period: November '78-October '79

Fig. 3.
Cumulative nitrate loads in Krzeszowska Struga River

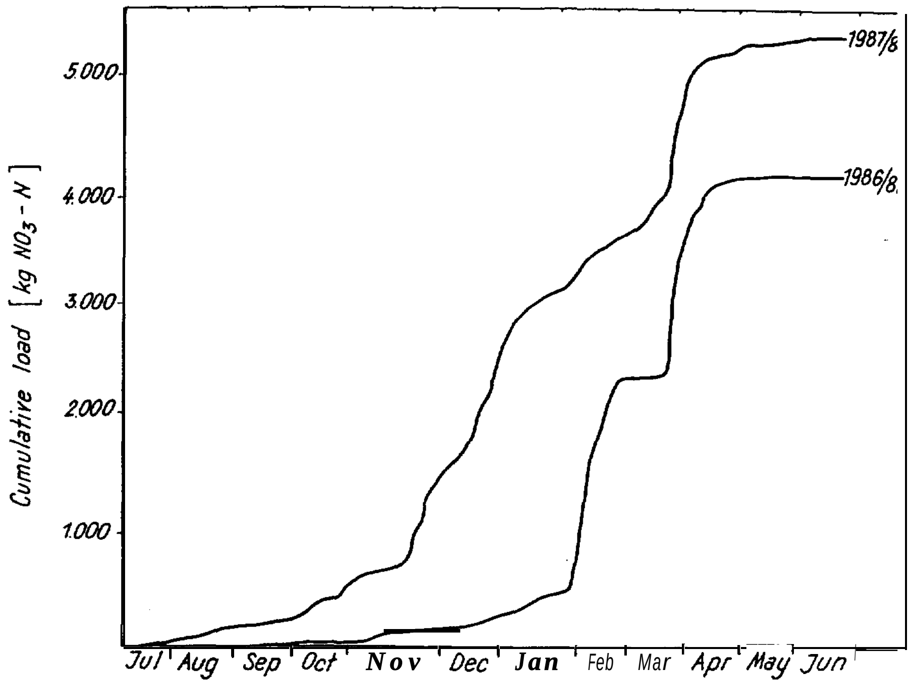
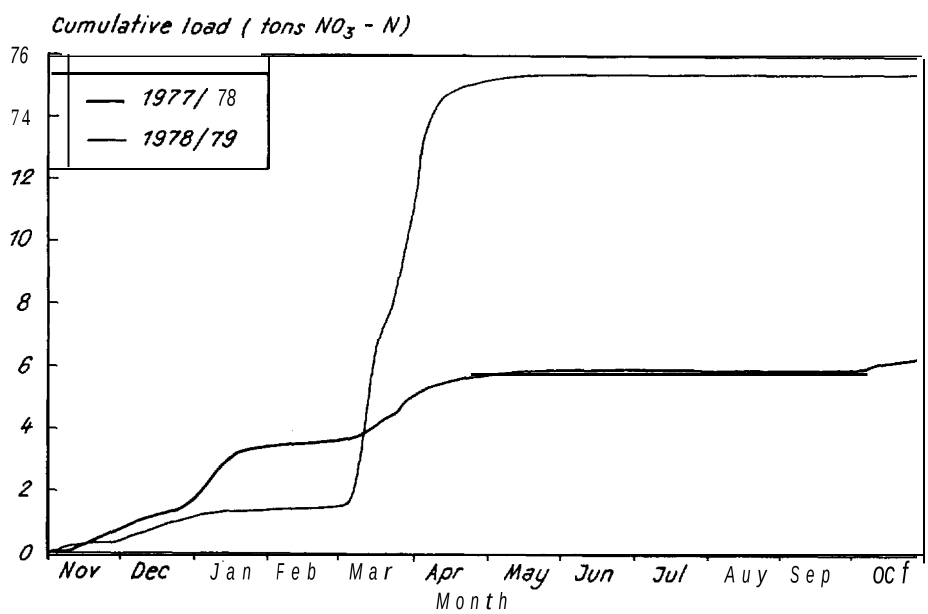


Fig. 4.
Cumulative nitrate loads in Pomorka River



Nitrogen in surface waters of Schleswig-Holstein

Dr. Ismo Bruhm – Umweltdiagnostik Kiel

Introduction

Schleswig-Holstein is characterized by an intensive agriculture. About 75% of the area is under agricultural management with grave consequences for the environmental quality. The ground and surface waters are loaded with substances which are used in agriculture. The near-surface ground water is polluted by nitrogen and pesticides, the surface water is characterized by high transport of nutrients like phosphorus and nitrogen causing eutrophication.

The paper report on a project on the Christian-Albrechts-University of Kiel concerning questions of the water quality of surface waters in Schleswig-Holstein. One aim of the project was to collect all datas about water-, soil- and atmospheric quality to get a survey of the actual situation in Schleswig-Holstein and to find which measuring networks have to be improved. All the facts and informations were handled and managed by use of a Geographical Information System.

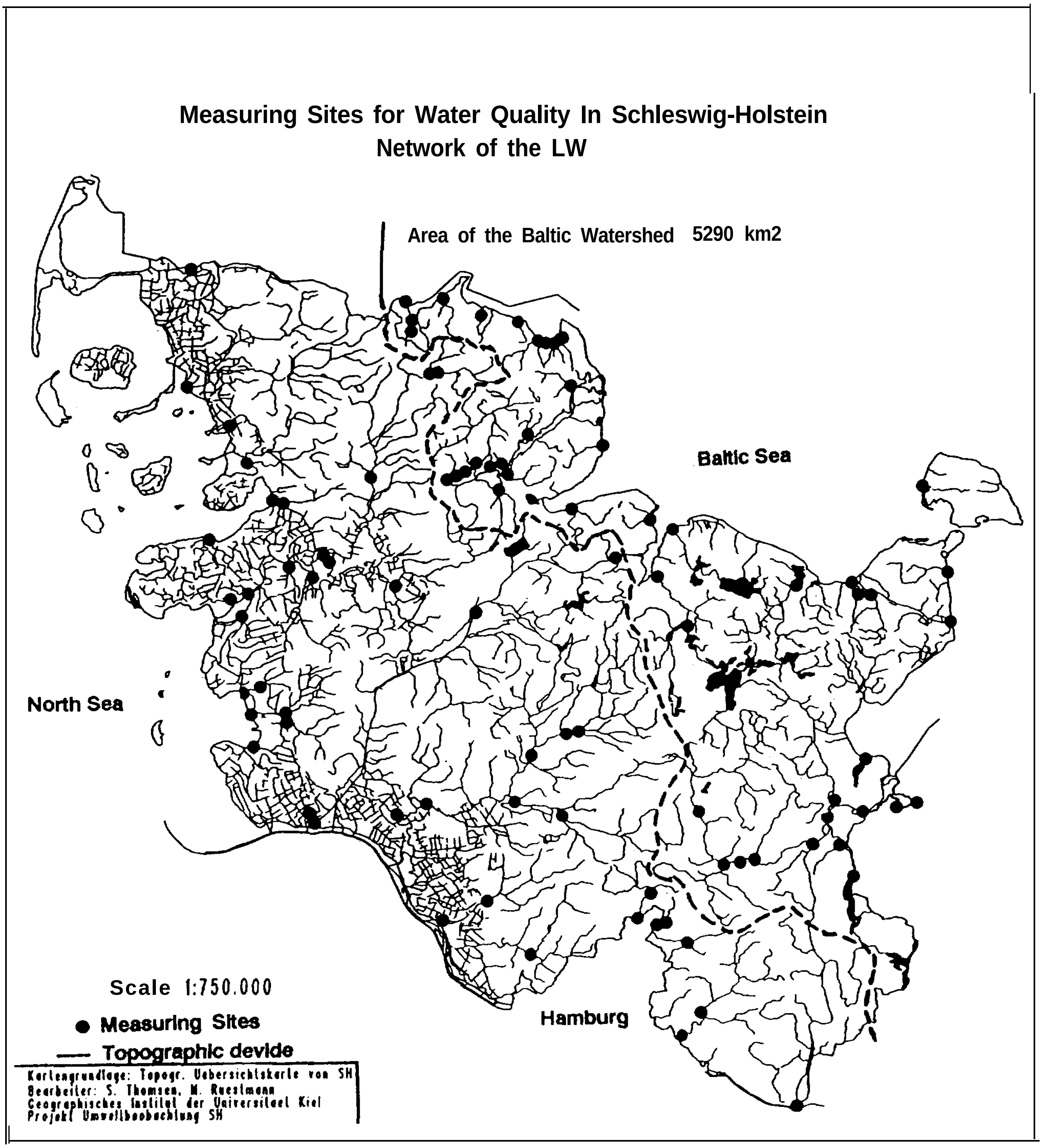
Surface water control in Schleswig-Holstein

The control of the surface waters in Schleswig-Holstein is part of the duties of the Federal State Office for Water Management (LW – Landesamt fir Wasserhaushalt und Küsten). Today nearly 100 measuring stations all over the country often placed near the sources are in operation. The following map illustrates the measuring sites which are marked with a black spot. The drawn in line illustrates the topographic devide between the North Sea-Watershed and the Baltic Sea-Watershed. The drainage area of the Baltic Sea is about 5290 km².

Unfortunately the locations for hydrophysical measurements (rate of runoff, velocity of surface waters) the called gauge-stations are not always on the same sites than those of the hydrochemical measurements. The result is, that the calculation of the nutrient transport into the Baltic Sea is difficult. Credible calculations on the nitrogen input into the Baltic Sea by surface water (Schleswig-Holstein) do not exist, because the nitrogen input is closely connected with the behaviour of the runoff. The water samples of the hydrochemical measuring network are taken monthly or 4 times a year. The analysed parameters are:

- Phosphorus (total, dissolved)
- Nitrogen (total, dissolved)
- Organic and inorganic Carbon
- Salinity, pH-value, dissolved Oxygen, BOD
- Heavy metals
- Organic hydrocarbons

Map 1:
Hydrochemical measuring sites of the L W in Schleswig-Holstein



In the following schedule the planning values for the water-quality are compiled.

Table 1:
Planning values for surface water quality of the government of Schleswig-Holstein

Parameter		***Water-quality Value (mg/l)***
Dissolved Oxygen	O_2	≥ 5.0
ph-Value		6.5-8.5
Ammonium	NH_4-N	≤ 0.5
Total Nitrogen	N_{tot}	≤ 10.0
BOD5	O_2	≤ 5.0
Total Phosphorus	P_{tot}	≤ 0.3
Organic Carbon	C	≤ 10.0
Chromium	Cr	≤ 0.050
Nickel	Ni	≤ 0.050
Cooper	Cu	≤ 0.010
Zinc	Zn	≤ 0.200
Cadmium	Cd	≤ 0.002
Iron	Fe	≤ 2.000
Mercury	Hg	≤ 0.001
Lead	Pb	≤ 0.050
Volat. chlorine Hydrocarbons		≤ 0.005
Organochlorine Pesticides		≤ 0.0001
Polychlorinated Biphenyls		≤ 0.0001

The aspired value for total nitrogen should be below 10 mg N/l. Fortunately most of the rivers are characterized by lower concentrations.

In Schleswig-Holstein the valuation of the water quality is based on an empirical developed index for the water quality. This index is calculated using the following parameters:

Total Nitrogen	N tot.
Ammonium	NH_4-N
Total Phosphorus	P tot.
Phosphate	PO_4-P
Organic Carbon	DOC + TOC

The classification runs from class 1 (unpolluted / unloaded) to class 4 (intense polluted). An evaluation of the indices of some selected measuring sites from 1979 to 1988 shows, that the index has decreased from 2.4 (distinct polluted) to 1.9 (moderate polluted).

A view on the listed values manifests that most of the concentrations decreased during the period of 10 years with the exception of nitrate. The decreasing of phosphorus, total nitrogen and ammonium is to be explained by the improvements of the waste-water management. The concentrations of nitrate are stagnant nearly on the same level during 10 years. The average nitrate concentration is about 3.5 mg N/l and of total nitrogen 4,5 mg N/l.

The larger rivers are characterized by lower concentrations caused by dilution (site No. 126029; fig. 2), the smaller rivers and brooks with an agricultural drainage area have concentrations sometimes above 10 mg N/l (site No. 126004; fig. 1) like the »Schmalenseefelder Au« (Dr. Reiche, this issue).

Tab& 2:
Yearly averages of 15 selected sites from** 1979-1988; **in mg/l

	NH_4-N	NO_3-N	N_{tot}	PO_4-P	P_{tot}	chem. Index
	ø	ø	ø	ø	ø	ø
1979	1.226	3.795	7.175	0.37 1	0.812	2.4
1980	1.029	3.947	6.742	0.306	0.629	2.4
1981	0.723	3.466	5.132	0.233	0.467	2.1
1982	0.863	3.079	4.836	0.261	0.495	2.1
1983	0.567	3.425	5.360	0.238	0.474	2.1
1984	0.531	3.618	5.531	0.188	0.438	2.1
1985	0.531	3.586	5.016	0.178	0.364	2.1
1986	0.593	3.714	4.753	0.193	0.388	2.1
1987	0.497	3.459	4.782	0.145	0.307	2.0
1988	0.380	3.510	4.561	0.119	0.205	1.9

Nitrogen in surface waters

In Schleswig-Holstein app. $^2/_3$ of the nitrogen freight in surface waters are caused by agriculture (agricultural waste water, surface runoff, drainage and/ or ground water) and app. $^1/_3$ by domestic and industrial waste water and atmospheric input.

The examples on the following pages (fig. l-3) show the development of nitrogen concentrations during 1979 and 1988. The concentrations are characterized by a typical and distinct seasonal pattern. The concentrations begin to rise in autumn (oct./nov.), reach the maximum in january or february and decrease to the minimum in july or august. The selected sites represent surface waters with an agricultural used catchment. After the harvest, when the biological uptake is fading, the nitrogen, especially the solulable nitrate is leached and washed out by precipitation and percolation. The highest concentrations of nitrogen appear similar to the highest rates of runoff.

The close connection between the concentrations of nitrogen and the rate of runoff is represented on figure 4 and 5.

The figures manifest that the calculation of the nitrogen transport depends on detailed informations about the rates of runoff.

The nitrate arrives the drainage channel either through subsoil pipes or by near surface groundwater flow. The input by erosion and surface runoff (overland flow) is of subordinated importance. In springs surrounded by agricultural land like the »Schmalenseefelder Au« the measured concentrations of nitrate are similar to those of the subsurface groundwater. This is a problem in some parts of Schleswig-Holstein because the water supply with acceptable drinking water is in danger.

The pollution of the ground- and surface waters with nitrate depends on the land-use, the use of fertilizers, the capacity of buffering and the permeability of the soil.

Possibilities and proposals for reducing the nitrogen transport in ground- and surface waters:

- The land-use in endangered areas (areas with sandy, permeable soils and/or intensive agricultural use) has to return to more ecological methods. Careful managing with reduced rates of fertilizers and liquid manure, which are adapted on the real biological need, has to take place.
- Areas with low economy have to be managed extensive (contracts with the government, subventions), noneconomic areas and/or those with a high ecological value have to be managed carefully or have to be purchased by the government.
- Planning and lay out of buffer-strips along the brooks and rivers to avoid erosion and direct input by surface runoff should be realized.
- To improve the water quality, the river basin has to be renaturated and sanitated. The monotonus piped and channeled course has to be diversified. The velocity of surface water has to be reduced, the time the water stays in the brooks and rivers has to be extended, to raise the possibility for a chemical and biological reduction of the nutrient transport.

Figure 1

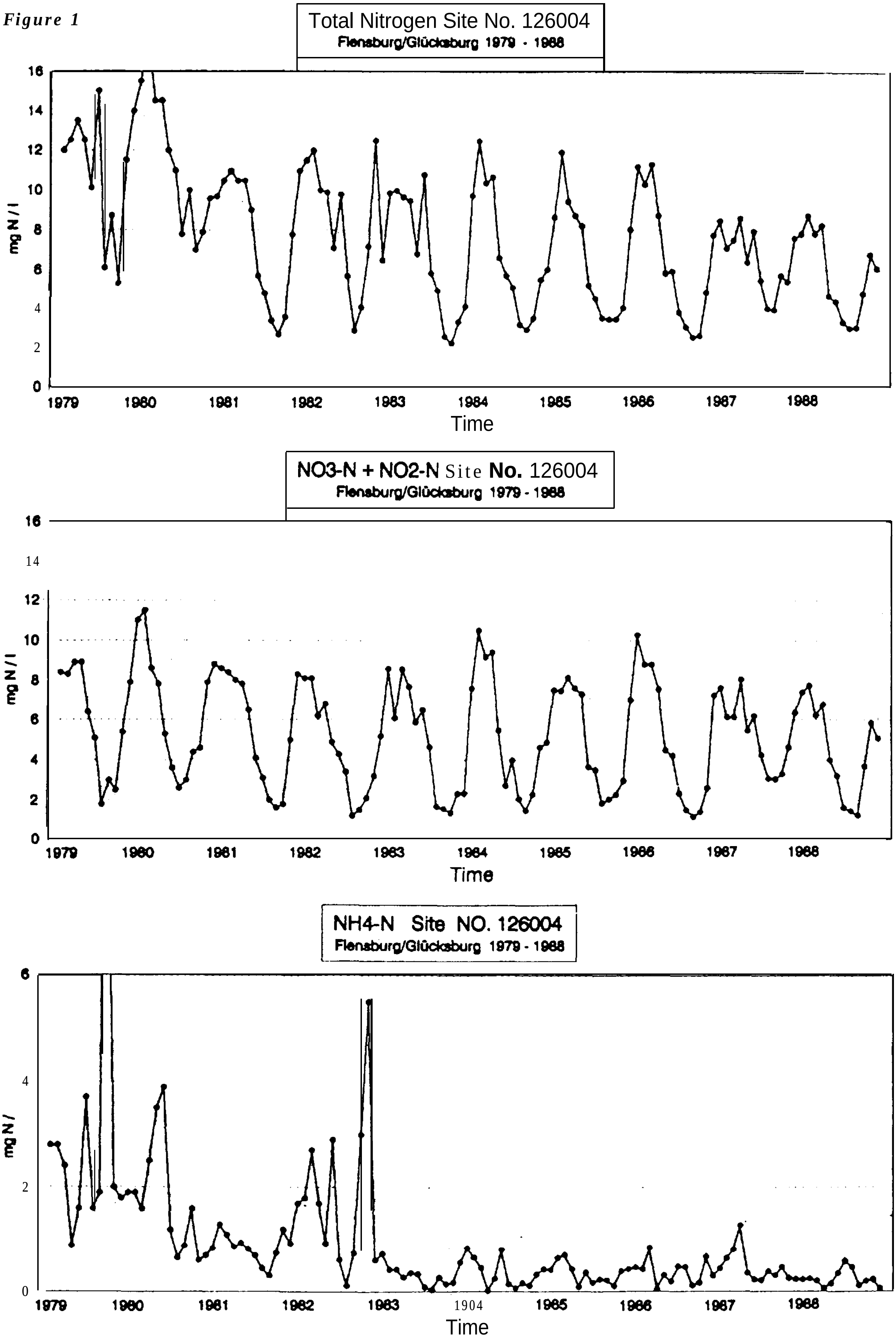
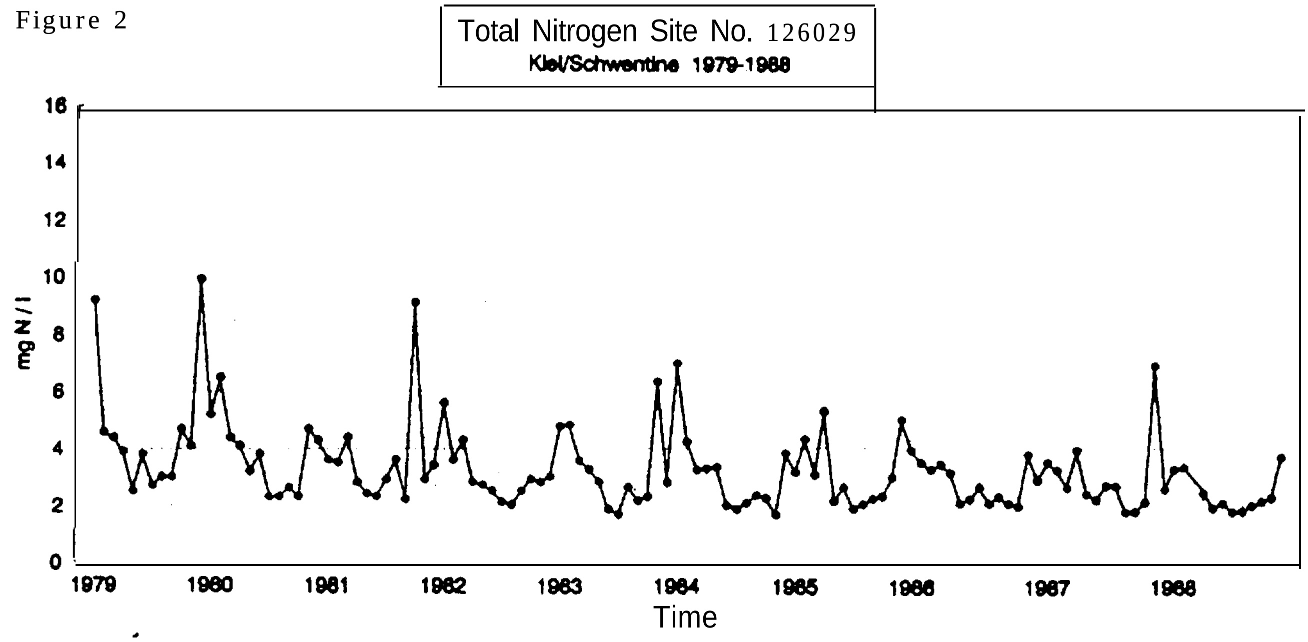
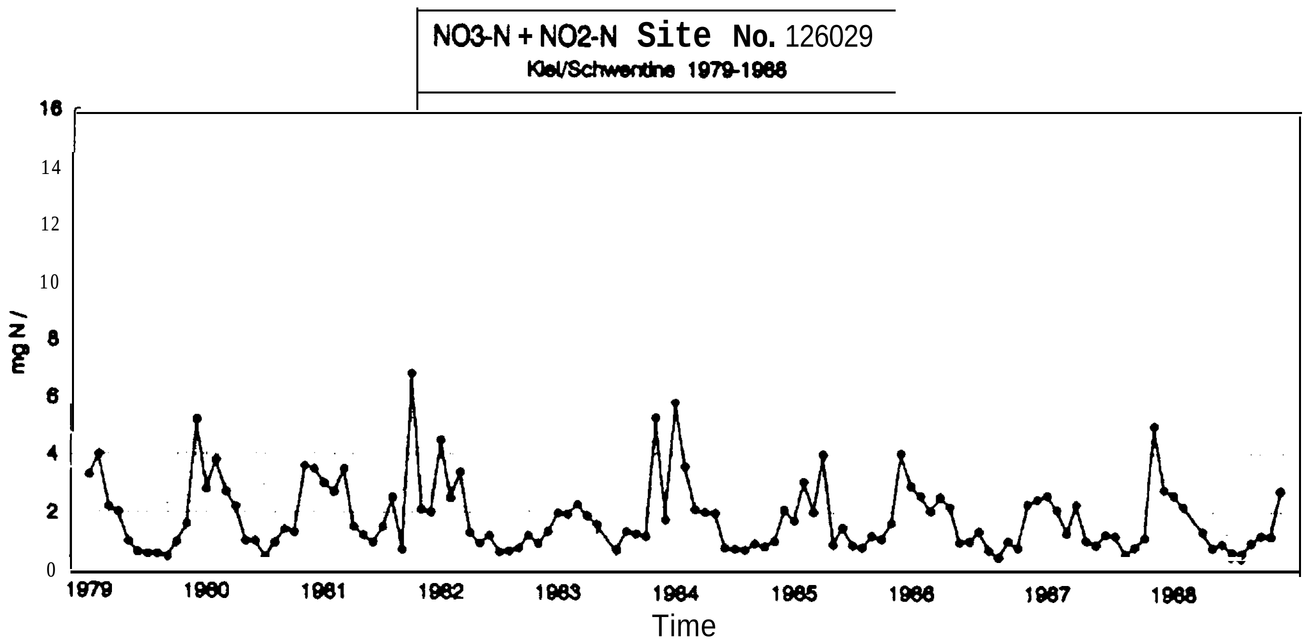

Total Nitrogen Site No. 126004
Flensburg/Glücksburg 1979 - 1988
mg N / l
16
14
12
10
8
6
4
2
0
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
Time
NO3-N + NO2-N Site No. 126004
Flensburg/Glücksburg 1979 - 1988
mg N / l
16
14
12
10
8
6
4
2
0
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
Time
NH4-N Site NO. 126004
Flensburg/Glücksburg 1979 - 1988
mg N /
6
4
2
0
1979
1980
1981
1982
1983
1904
1985
1986
1987
1988
Time

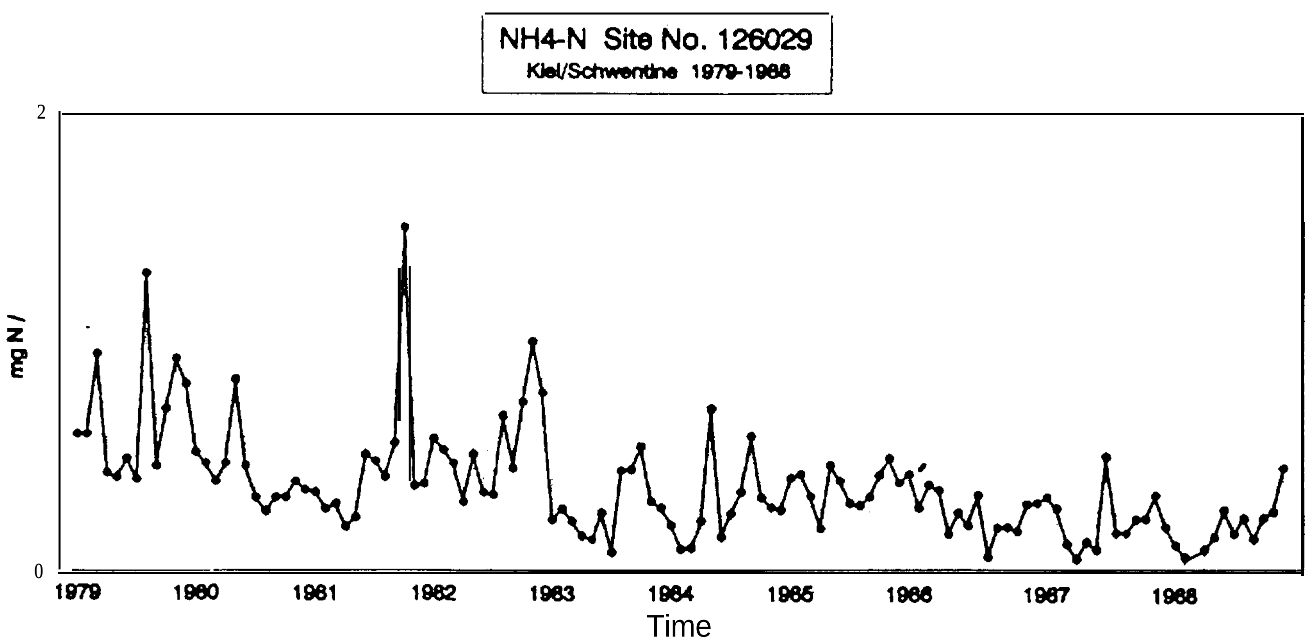
Figure 2



Total Nitrogen Site No. 126029
Kiel/Schwentine 1979-1988
mg N / l
0
2
4
6
8
10
12
14
16
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
Time

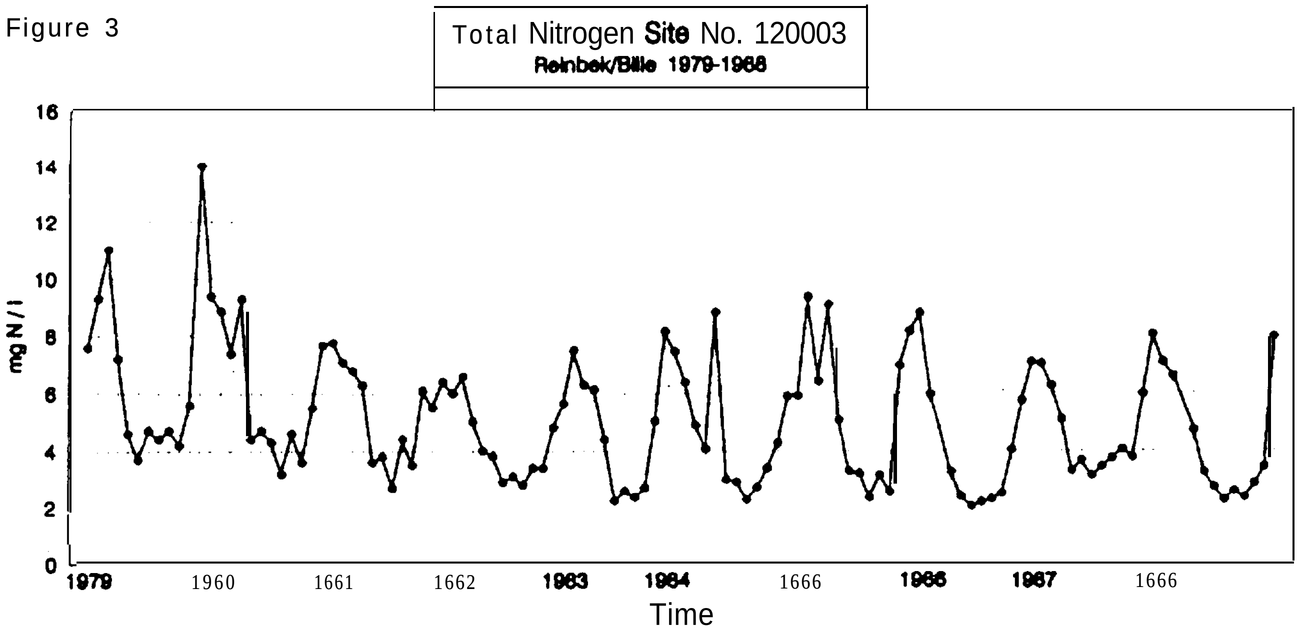


NO3-N + NO2-N Site No. 126029
Kiel/Schwentine 1979-1988
mg N /
0
2
4
6
8
10
12
14
16
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
Time



NH4-N Site No. 126029
Kiel/Schwentine 1979-1988
mg N /
0
2
1979
1980
1981
1982
1983
1984
1985
1986
1987
1988
Time

Figure 3


Total Nitrogen Site No. 120003
Reinbek/Bille 1979-1988
mg N / l
16
14
12
10
8
6
4
2
0
1979
1960
1661
1662
1983
1984
1666
1986
1987
1666
Time

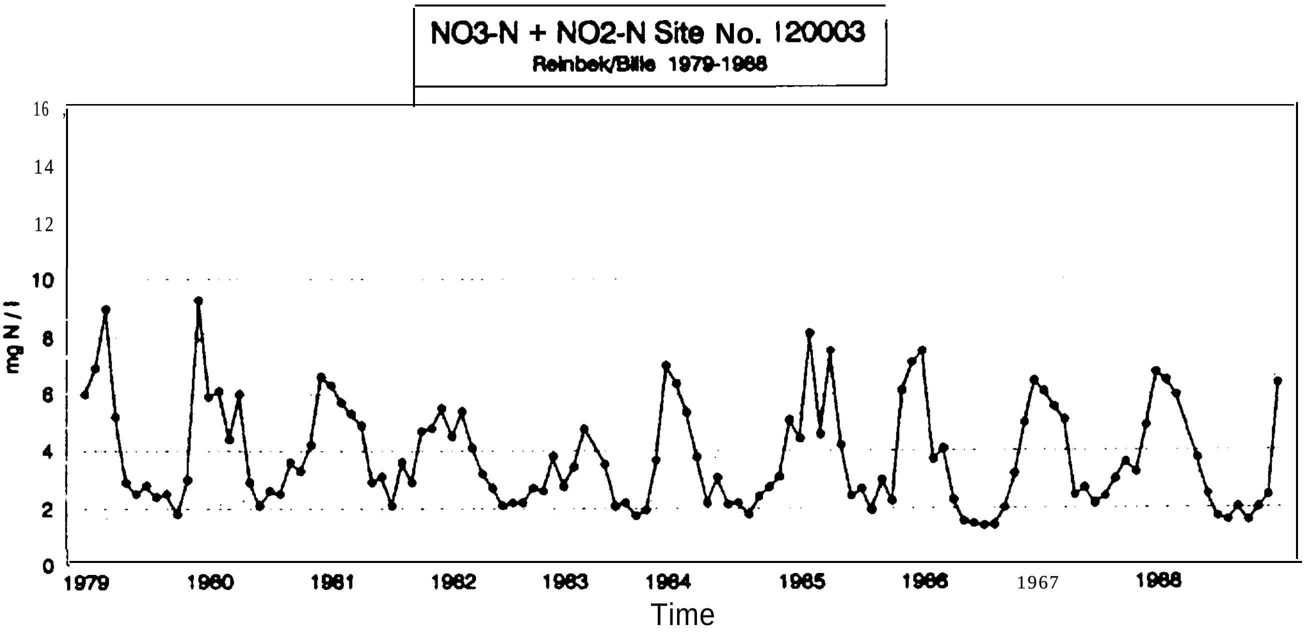

NO3-N + NO2-N Site No. 120003
Reinbek/Bille 1979-1988
mg N / l
16
14
12
10
8
6
4
2
0
1979
1980
1981
1982
1983
1984
1985
1986
1967
1988
Time

Figure 4

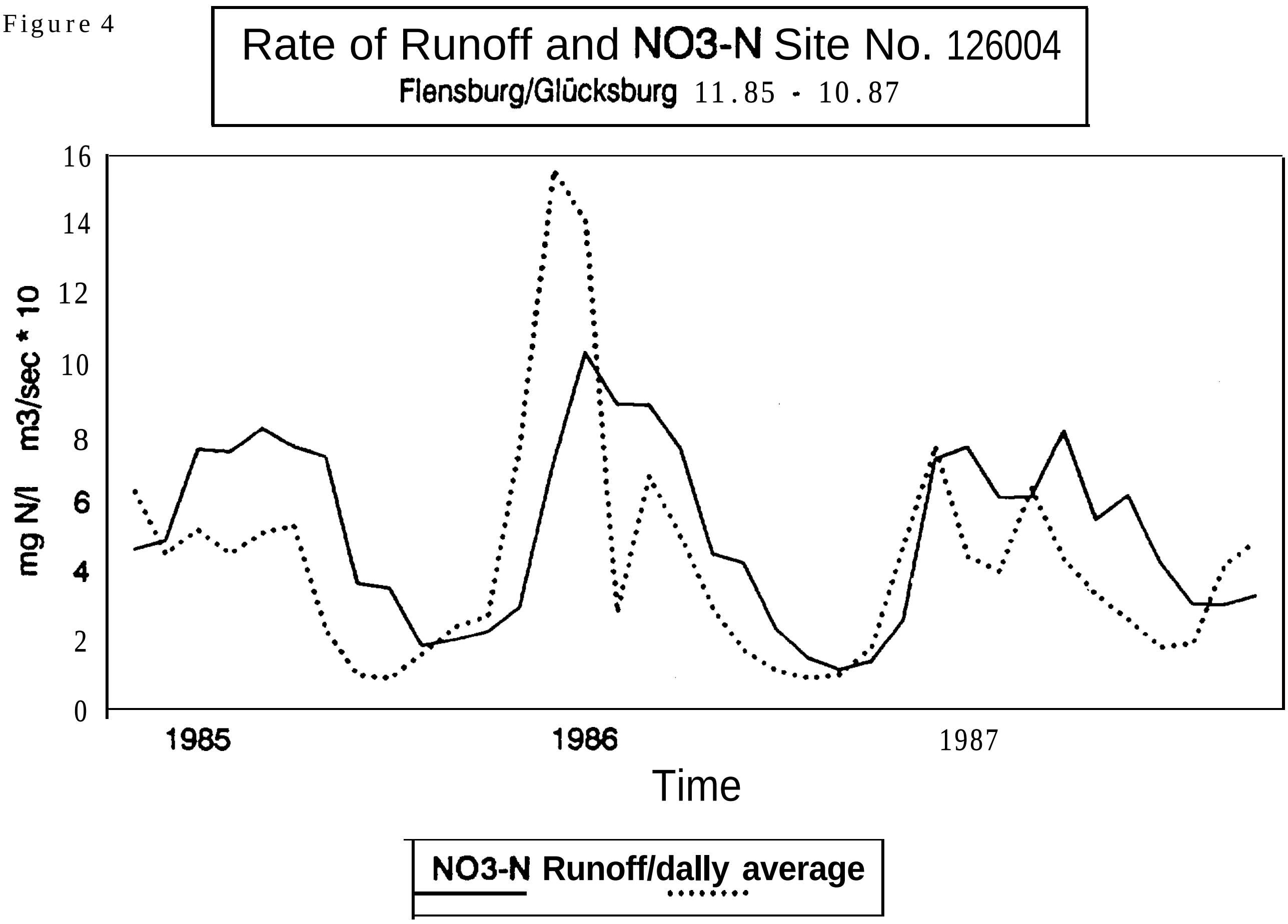
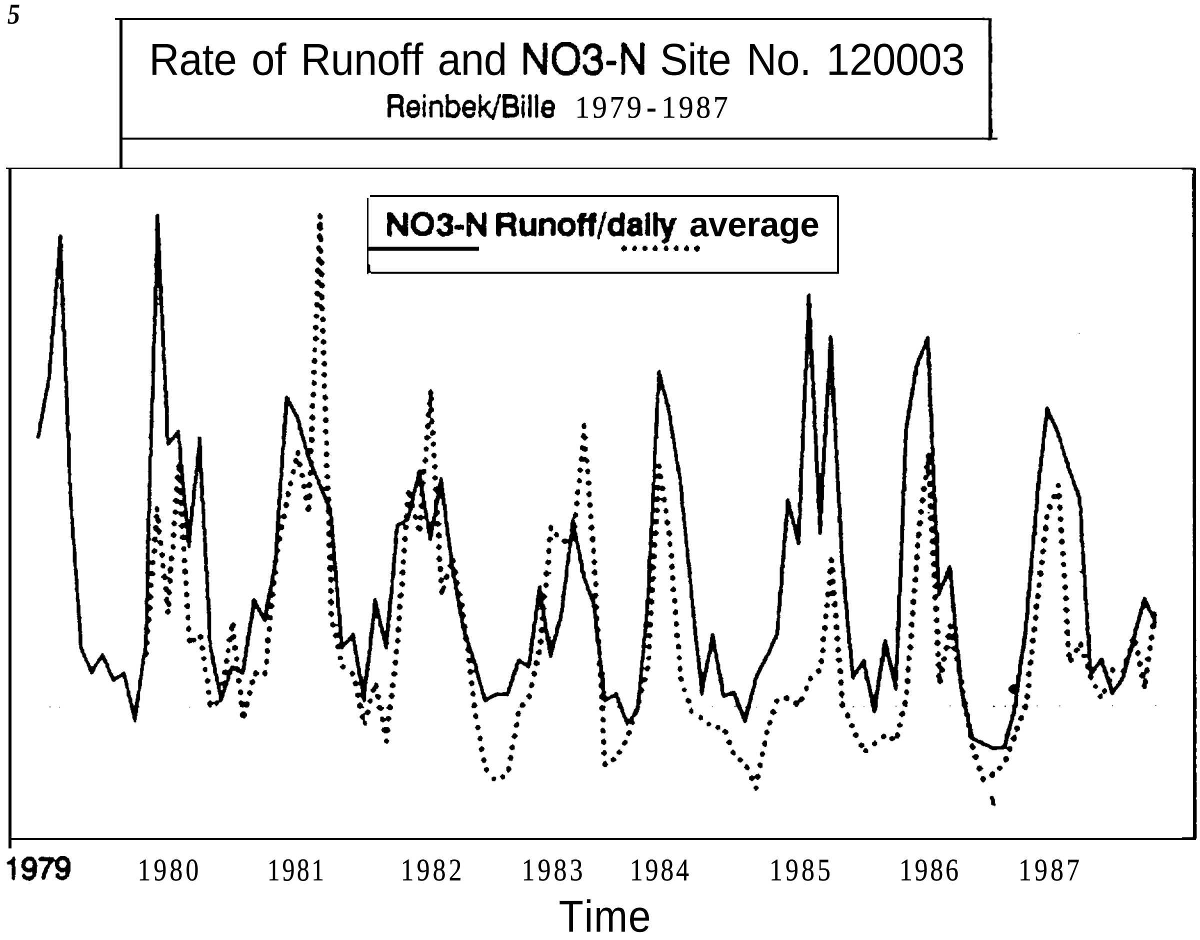


Figure 5



Agricultural nutrient loading of waters in Finland and measures to reduce it

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Agricultural nutrient loading of waters

The agricultural contribution of nutrient loading to water courses in Finland is presently greater than the nutrient load from industrial and municipal sources (Rekolainen 1989). The nutrient load, particularly from municipalities, has been significantly reduced during the last two decades by intensive construction of wastewater treatment plants. According to long-term monitoring results (Rekolainen 1989) the phosphorus load has been greater during the last years compared to the phosphorus load in the late sixties and seventies (Kauppi 1984), whereas the nitrogen load has been stable.

Two clear reasons can be found to explain the increase in phosphorus load: Firstly, phosphorus overfertilization, which has continued from the early fifties. Phosphorus input was for many years approximately 2.5 times higher than the output with yields. Secondly, the specialization of agriculture has lead to cereal production in southern Finland with decreasing grass cultivation, and this can be estimated to have increased erosion rates.

Balance calculations show that fertilizer input of nitrogen has recently been 2-2.5 times higher than output with yields. The reasons, why nitrogen losses have not increased, can be denitrification and accumulation in soil, because the nitrogen input with fertilizers is relatively low in Finland compared to central European countries.

Measures to reduce nutrient load to waters

In order to reduce phosphorus load from agricultural areas to waters, the most important measures are conservation tillage practices and vegetated filter strips for reducing erosion. In Finland the commonly used tillage implement is moldboard plow in fall, which leaves the soil surface without any vegetation (dead or alive). Using a reduced-tillage implement (e.g. cultivator, chisel plow, disk harrow), the mulch cover increases and reduces erosion and thus phosphorus losses. Recent model evaluations in Finland have shown that replacing moldboard plow in fall by, e.g., chisel plow reduces soil loss by 40–70% (Rekolainen and Posch 1991).

Conservation tillage practices hinder the detachment of soil particles caused by a falling raindrop, while vegetated filter strips mainly reduce the transport capacity of surface runoff. Presently, there are no model evaluations or experimental results from Finland to show the efficiency of filter strips to reduce phosphorus load, but results from many other countries indicate that they can significantly decrease soil loss.

The present policy for reducing the phosphorus **fertilization rate in** Finland is a step in the right direction. However, as long as the P-input is higher than the P-uptake by crops, phosphorus losses will not decrease. The present taxes on **P-fertilizers** should lead to further reductions, because at present prices the cost-optimal P-fertilization rate is lower than the present one.

Nitrogen losses to waters in Finland are lower than in many western European countries, because the amount of N-fertilizer (inorganic + organic) used is much lower. According to results from Sweden the N-input in Finland is presently approximately at the level, where the losses start to increase rapidly. However, the most beneficial N-input level for farmers is still (at present fertilizer prices) higher than the level presently used. Although the present policy in Finland concentrates on reducing **phosphorus** losses, nitrogen losses have to be reduced as well, because it has been recently found that nitrogen is the limiting factor for algal growth in coastal waters of the Baltic Sea.

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Rekolainen, S. 1989a. Phosphorus and nitrogen load from forest and agricultural areas in Finland. Aqua Fennica 19: 95–107.

Rekolainen, S. and Posch, M. 1991. Effects of conservation tillage techniques on erosion control in Finland – A model evaluation. Manuscript.

Nutrient loading of waters

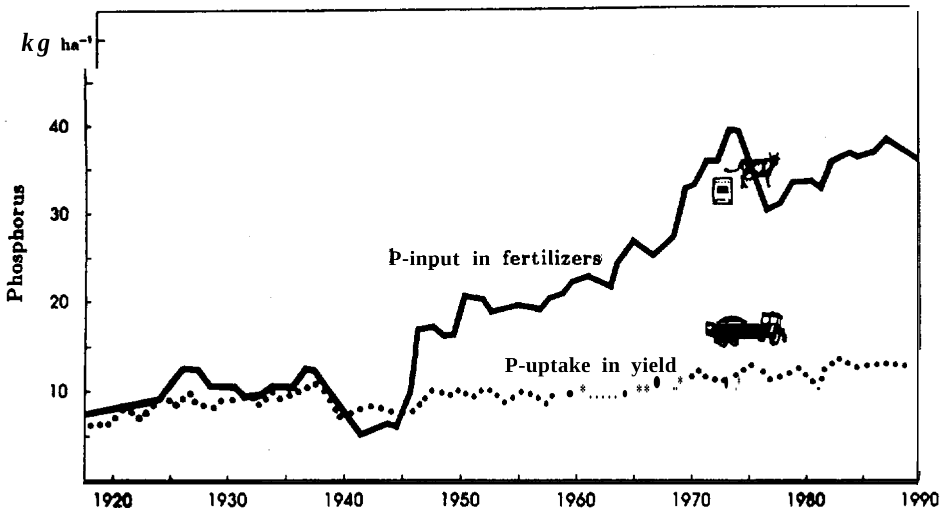
	Phosphorus t/a	Nitrogen t/a
Industry	800	7000
Municipalities	500	13000
Agriculture	2000–4000	20000–40000

Other non-point sources:

- forestry
- scattered population
- peat production
- land and water construction
- atmospheric deposition

- natural transport of nutrients

*Development of **P-fertilization** and P-uptake in **Findland***



SILLANPAA 1990, ELONEN 1990)

Measures to reduce **P** load

1. Reduce the fertilizing level

- **no quick effect on loading**
- **important: no increase in loading**

2. Reduced tillage techniques

- **any conservation tillage is** better **than moldboard plowing in autumn**

3. Green fallowing

- **fallowing is vastly used to reduce over-production. All fallowing should be green fallowing instead of traditional open land fallowing**

4. Filter strips

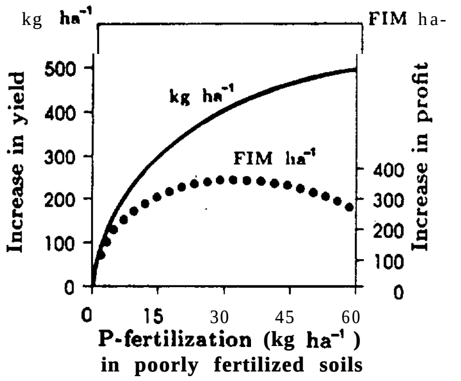
- **in • tsep slopes along the rivers and lakes filter strips should be used**

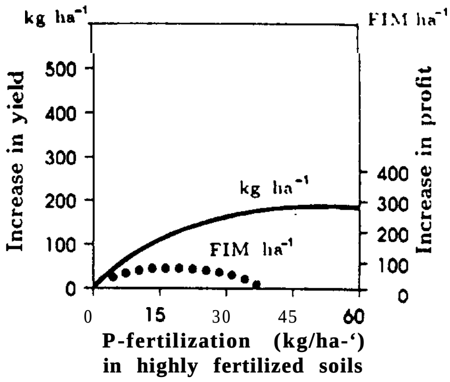
5. Storage of **manure**

- **storages in good** condition **(no leaching) and** the capacity **enough for at least 8 month storage (no application on •** now **cover or frozen soil)**

SR/vet 18.2.1991

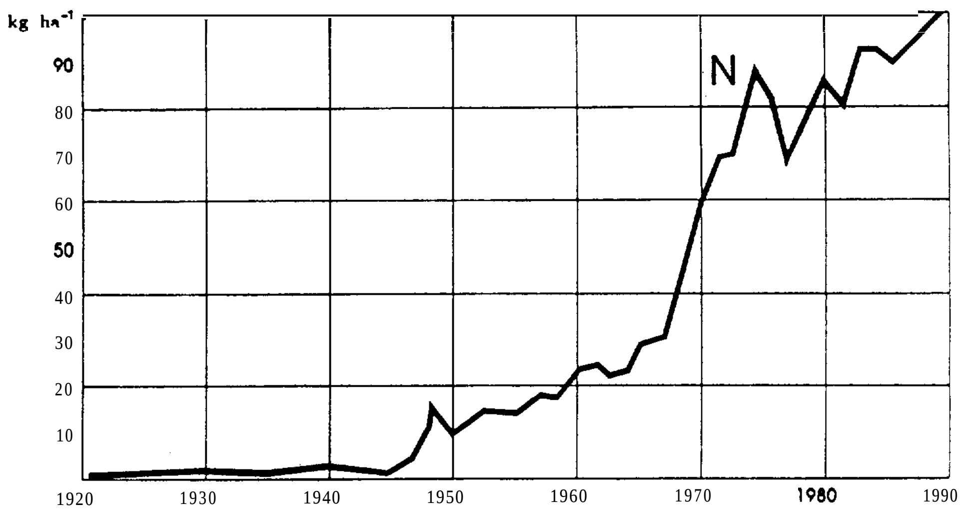
*Maximally profitable **P- fertilization***



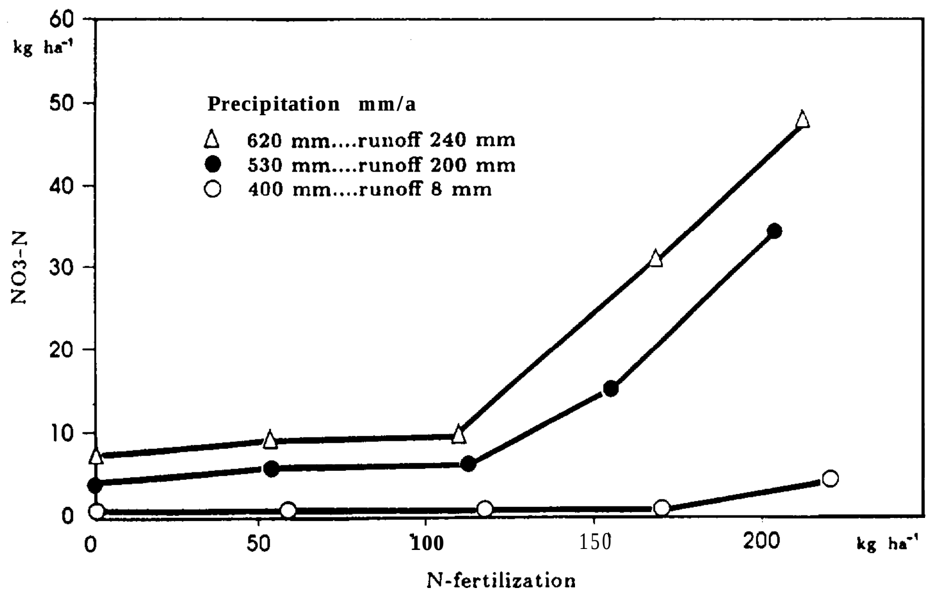


Saarela 1989

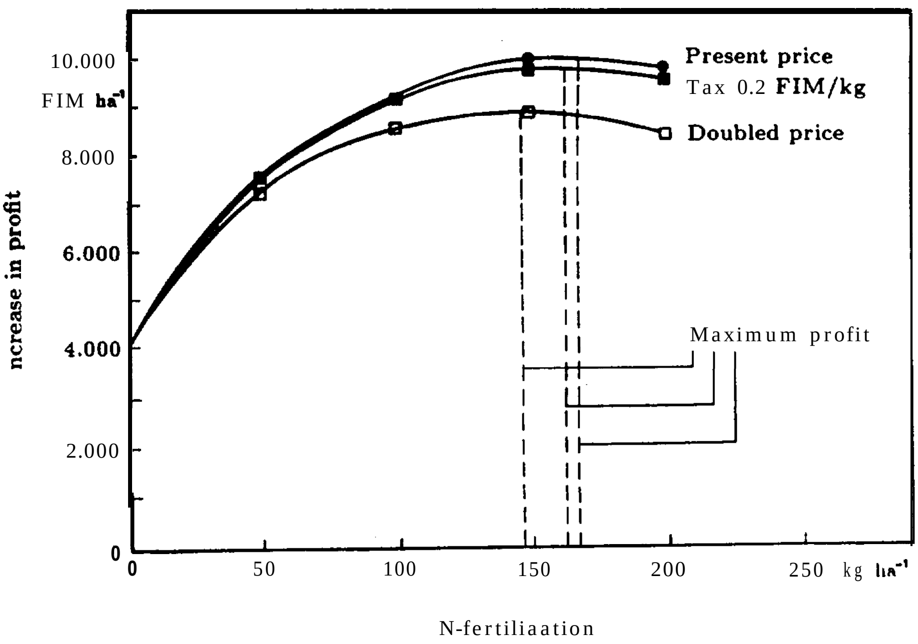
*Use of **artifical N-fertilizers** in **Findland** 1920-1989*



*Leaching of **nitrate-N from** Agricultural field*



BRINK 1984


10.000
FIM ha⁻¹
8.000
6.000
4.000
2.000
0
ncrease in profit
Present price
Tax 0.2 FIM/kg
Doubled price
Maximum profit
0
50
100
150
200
250 kg ha⁻¹
N-fertiliaation

Estimation of water and nitrogen dynamics at sites and in catchments by use of a computer model

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Schauenburgertr. 112, 2300 Kiel

Introduction

This paper explores a method of how to describe the water fluxes and the nitrogen dynamics in soils at sites and in catchments.

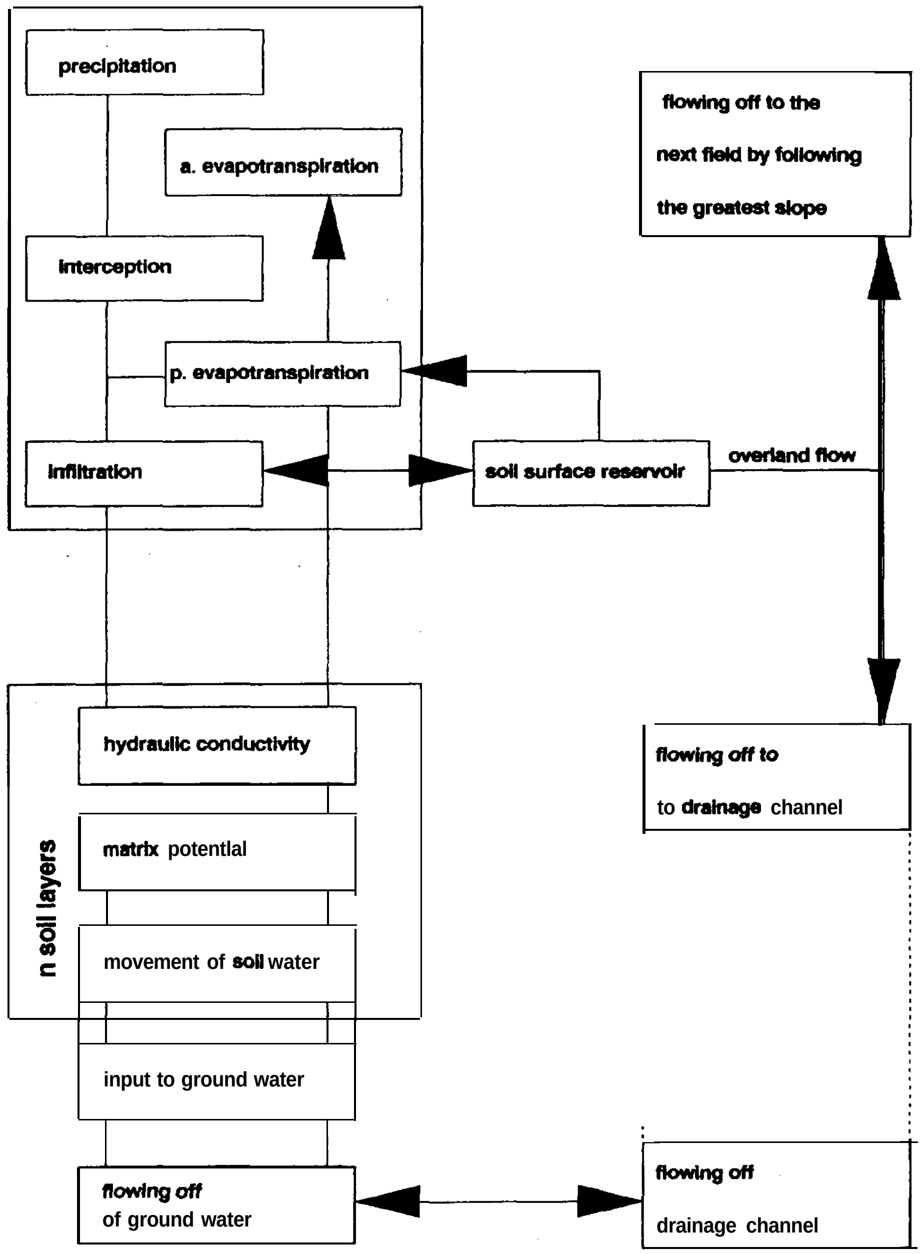
In the last few years several models have been developed in order to describe the dynamics of water and substances in soils. One of the typical objects of these models is the quantitative estimation of nitrogen losses to the ground water. This is caused by high fertilizer inputs. In several areas of Germany the nitrogen concentration exceeds the legal limit or the standard level. The development and the application of computer models in order to describe those processes is an important task of our research department »Ökosystemforschung im Bereich der Bornhöveder Seenkettee. One of the crucial aspects is to reduce the amount of input data to make the utilization as easy as possible. On the basis of established model-approaches to describe the soil water budget, the dynamics of soluble and adsorbed substances as well as the different processes of the soil nitrogen budget, the model system WASMOD&STOMOD gives the ability to carry out simulations for the area of small catchments. This paper gives a survey of the so-called model WASMOD&STOMOD. The procedures concerning the regional aspects will be dexcribed in detail. Therefore the combination with a geographical information system and the deduction of input values for complete areas are very important.

Finally an example is given to illustrate the effectiveness of the model.

Description of submodels

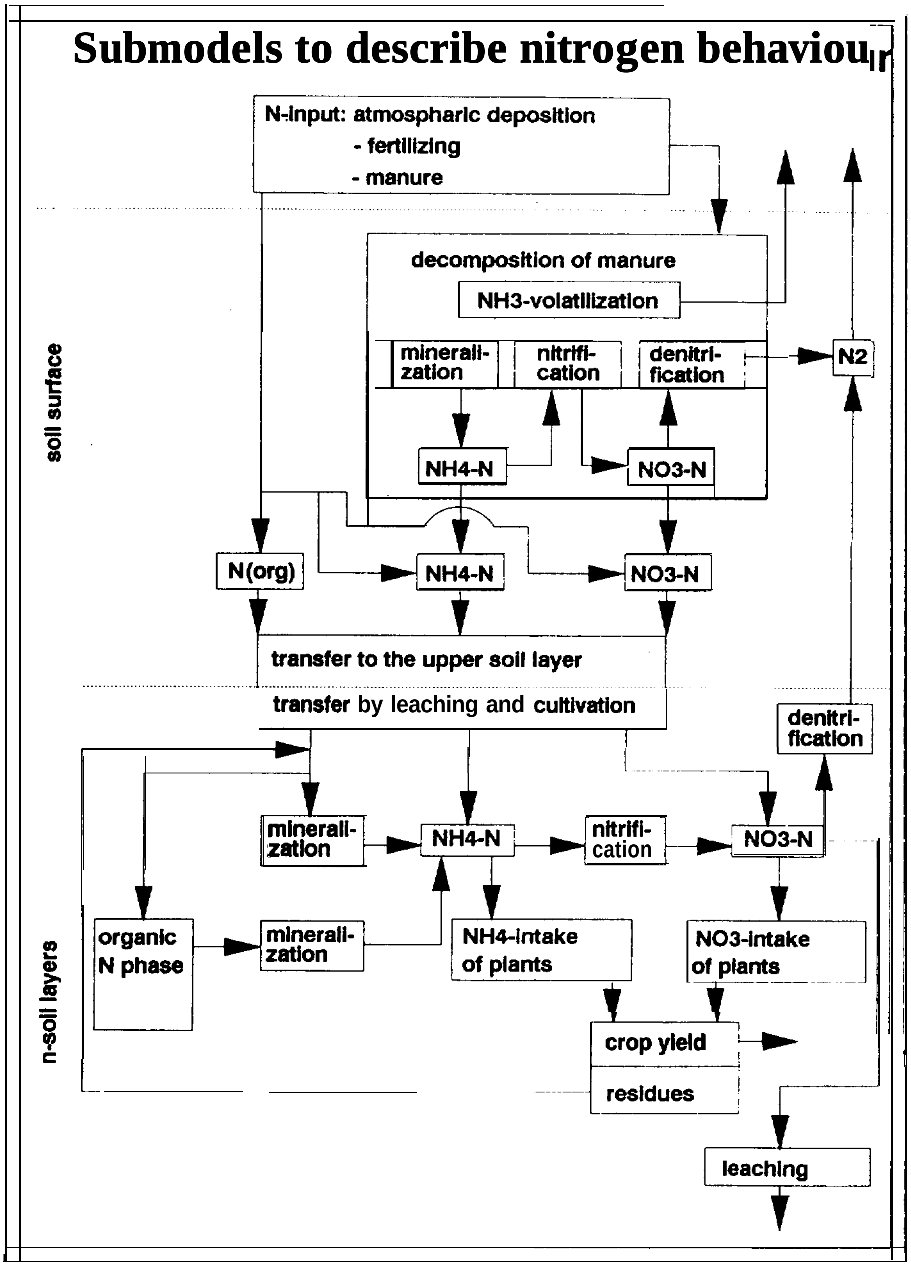
The vertical movement of soil water in the unsaturated zone is described according to Duynisveld (1984), Müller (1987), Fränzle et al. (1987) and they are based on the DARCYan law or the so-called Focker-Planck-equation. These equations are solved by the methods of »finite differences+. The vertical discretization of the soil columns can be freely chosen. The number of the per-day time steps depends on the calculated seepage water rates. The determination of the vertical range of the layer which covers the ground water table results from rates, which are calculated step by step for ground water input and output. The HAUDE-equation is integrated into the model to estimate the amounts of water losses by evaporation and transpiration, whereby the reduction from the potential to the actual amount is carried out by using plant-factors and by including the field capacity of each soil layer, the seasonal varying depth, and the distribution of roots. The calculation of water losses caused by interception is based on the precipita-

Figure 1:
Submodels to calculate water budget



tion rate and the foliage surface. The model includes methods for the calculation of the infiltration rate, of the amount of drainage water, and of overland flow. The approach used here to calculate the downward movement of nitrate includes the convective or mass flow term and the dispersion term. The differentiation between mobile and immobile pore water is considered as a function of the soilwater content and the distribution among different pore sizes (Müller 1987, Reiche 1990). In addition to the ion exchange processes described by the Freundlich type it is possible to simulate the ion transfer between different soil water fractions. The knowledge of the soil temperature is necessary, if one wants to describe the microbiological nitrogen metabolisms. The submodel »soil heat budget« describes the temperature distribution in soil as a function of air temperature (daily maximum and minimum), of soil water content, of heat capacity, and conductivity of organic and mineral constituents. In addition to the soil temperature and the soil water content the calculation of mineralization, nitrification and denitrification is based on data characterizing the acidity, the amount of organic matter and the C/N-ratio.

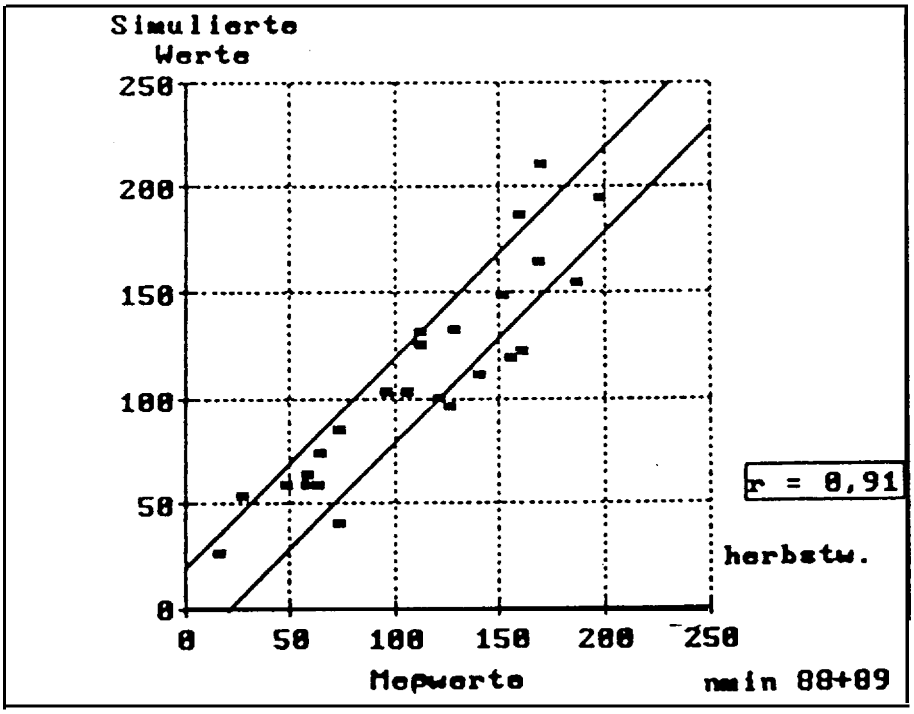
Figure 2:
Submodels to describe nitrogen behaviour



Submodels describing the characteristics of the behavior of nitrogen inputs by manure (like accelerated mineralization and ammonia volatilization) are integrated (cf. Hoffmann 1989). The input of nitrate and ammonia caused by atmospheric deposition is considered. The recent state of the model describes the plant's intake of nitrate and ammonia in a simplified way according to the water intake as mass flow. Parameters of high influence on plant's intake are those factors, which describe the root depth and distribution of particular plants.

Within the scope of the research project »environmental observation in Schleswig-Holstein« the nitrate and ammonia concentrations as well as the soil water contents of several fields were examined at monthly intervals (Reiche 1990, Branding 1990). The comparison of measured values with simulation results. This shows a good correspondence for soils of low permeability. This leads to the assumption, that the submodels describing the ammonia fixation and the denitrification are insufficient. Figure 3 shows measured and simulated amounts of nitrate and ammonia for 13 different soils (Nmin) determined for the period after harvest in autumn. The deviation exceeds in only 3 cases 30 kg N/hectar which is more than the error in measurement.

Figure 3:
Measured and stimulated amounts of nitrate and ammonia in soils after the harvest determinated for 13 different soils and 2 years (1988 and 1989)



Model expansion to make out balance sheets for water and nitrogen in catchments

The input parameters which describe the characteristics soil, relief, of vegetation, and the human utilization are stored by a geographical information system »ARC INFO«. The simulations run on the basis of small areas characterized by homogeneous parameter sets. The size of these areas is variable, depending on the variability of the catchment. Overland flow is simulated, if the infiltration capacity is exceeded because of heavy rainfalls and if a particular state of slope, soil surface, and vegetation occurs. In addition to these conditions the border between neighbouring fields must be in a state, which admits overland flow. Figure 4 presents a method applied for the estimation of overland flow. The simulation of the lateral flow of water in the saturated zone is based on the DARCY-equation. The model enables a calculation of daily input of water and nitrogen differentiated in parts coming as overland or groundwater flow for creeks and small rivers respectively for sections of them. So on the one hand the model shows the nitrogen losses caused by leaching for each single area respectively for the whole catchment, and on the other hand the nitrogen concentrations and loads which are carried by creeks and rivers to lakes and to the sea.

For running simulations calculating the water and nitrogen budget of large areas subdivided into some hundred parts, good data management based on a data base is very important. To start a simulation data must be stored in 4 different files disinguished climatic data, landuse data, data to characterize soil properties and topography, and data to initialize soil water

Tab. 1:
Cultivation of crops in the catchment of the Schmalenseefelder Au

crop	*fertilizer kg N/ha*	*manure kg N/ha*	*area in ha*	*sum kg N*
grassland	206.5	47,0	18,8	4766,0
rape	176,5	78,0	11,2	2850,4
wheat	170,0	89,6	2,04	519,2
beets	151,8	257,1	6,4	2616,9
barley	112,2	169,4	13,8	3686,1
rye	129,0	29,5	11,28	1787,9
oat	62,6	80,5	0,5	71,6

Tab. 2:
Water budget calculated for a part of the catchment of the Schmalenseefelder Au (south of the B430, period: 1.1.1988-31.12.1988)

	mm	in 100 1 f. 79,3 ha	in % des NDS
precipitation:	934,6	741574,5	100,0
interception:	123,1	97602,7	13,2
act. evapotranspiration:	409,2	324519,9	43,8
change of soil water amount:	45,8	36335,0	4,9
seepage water 100 cm U.S.:	343,4	27227,8	36,7
overland flow:	13,1	10349,3	1,4

Tab. 3:
Leaching of nitrate (100 cm below surface) calculated for a part of the catchment of the Schmalenseefelder Au (1988)

crop	*area*	*amount of seepage water*		*nitrate losses by leaching*		*N concentration in seepage water*
	in ha	mm	mm³*	kg N/ha	kg N*	in mg N/l
grassland	18,77	334,3	62741	31,9	598,8	9,5
rape	11,23	348,1	39095	66,4	745,2	19,1
beets	6,41	426,0	27306	109,0	698,7	25,6
wheat	2,04	320,0	6529	78,2	159,5	24,4
barley	13,81	405,4	55980	41,4	571,7	10,2
rye	11,28	363,4	40994	37,1	417,9	10,2
oat	0,5	395,o	1975	38,0	19,0	9,6
other	15,29	246,4	37679	18,6	284,4	7,5
sum	79,33		272279		3495,2	12,8

* calculated for areas simular crops

and nitrogen values. The climatic data set contains daily values of air temperature (maximum and minimum), precipitation, and humidity. The data set characterizing the soil properties and topography contains informations about the physical and chemical properties of soil layers and data to characterize the situation of the topography, field size, drainage, of the overland and groundwater flow, of the landuse and crop rotation for each site. Because soil maps exist for only a small part of Schleswig-Holstein, it was necessary, to install a module, which derivates the soil parameters from data of the German soil taxation (Bodenschätzung). These data are collected by a grid distance of 50 m. The landuse data set contains data to characterize the growth and phenological change of crops, and data about cultivation and fertilizing. An information pole was done with the farmers to get this data.

Application of the model WASMOD&STOMOD

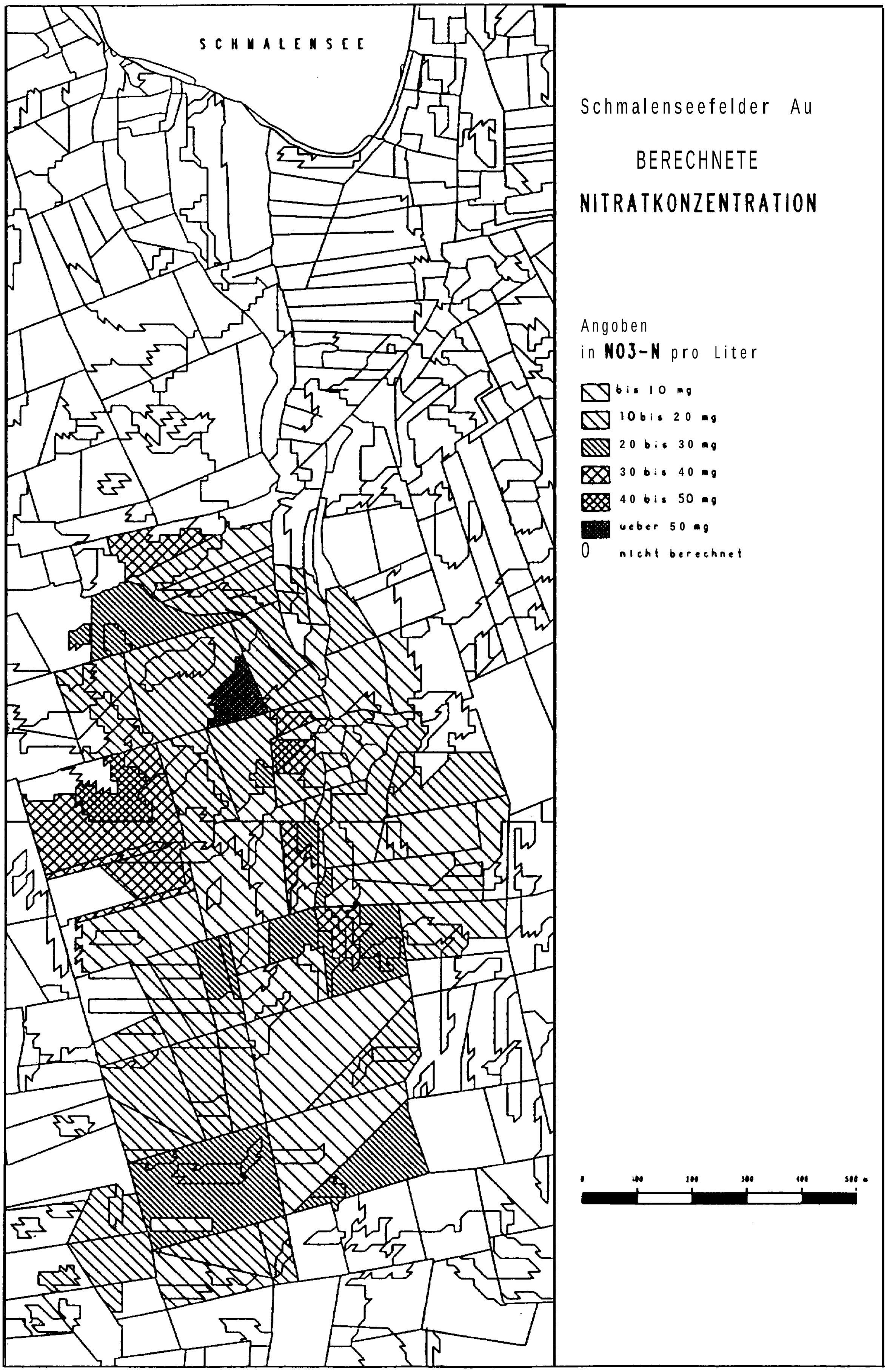
The Schmalenseefelder Au is a small creek situated in the south of the Schmalensee, one of the lakes of the Bornhöved lake district. The examination of water quality done by the office of water economy Schleswig-Holstein in 1979 and done by Bruhm in 1989 presents high values of nitrate (12 mg N/l). Even the water that had been taken near the spring contained high nitrogen amounts. The origin of these concentrations must be nitrate losses from soil surface caused by high fertilizer inputs. In order to discover fields with high nitrogen losses, the water and nitrogen budget of the catchment was calculated by simulation for a period of 6 years. The area was divided into 175 parts of homogeneous data sets. 56% of that area is used as arable land, 24% as grassland, the remaining 20 per cent as gardens, forests and settlements. There are, partly, sandy and loamy soils. Plane areas are characteristic for moorland. The crop rotation and data of fertilizer input originated from the information poll in 1988 (Fränzle et al. 1989) were used as data inputs for a simulation of the years 1983/84 – 1988/89. The annual amount of nitrogen input relating to the whole area runs up to 17,25 t. An amount of 18,6 kg N/ha of atmospheric deposition has been set in. Table 2 gives a survey of the calculated waterbudgets for the simulation period of 1988. The water losses by interception and evapotranspiration reach 57 per cent of the annual precipitation rate. Obvious differences occur inbetween single sorts of crops (rape 61,6%, beets 45,7%). The high rates of interception are caused by the fact, that wintercrops, rape and grassland represent a big part of the area.

Table 3 shows the nitrate losses by leaching and the nitrate concentration of soil water calculated for 1988. The highest amounts are calculated for beets, rape and wheat. In this case the nitrate concentration in drainage water exceeds the legal limit of 50 mg NO_3/l. With respect to the winter wheat field the high nitrate concentration is caused by a high rate of overland flow and a corresponding low level of drainage water.

The amount of nitrogen calculated for 1988 as load of the creek runs up to 3704 kg N, the average of nitrogen concentration is 13,1 mg N/l. The deviation to the average of measured values is less then 1 mg/l.

It was relevant to develop a suggestion for variational land use for parts of

Figure 4: Nitrate concentrations in seepage water calculated for a part of the catchment of the Schmalenseefelder Au



the catchment because of the above discussed simulation results. Subsequently, a second simulation was started. Changes were done in the first scenario for small parts of the area (16,5 ha). According to this design 8 ha of arable land were changed into extensive grassland, 5,1 ha into forest. In comparison to the simulation results based on the existing land use conditions, the effect is a reduction of 2,3 mg N/l calculated for the nitrate concentration of the Schmalenseefelder Au. To get a higher improvement of water quality a second landuse scenario was elaborated. In addition to the first changes, the amounts of fertilizer and manure inputs were reduced in those cases, where the nitrogen losses were calculated with the highest values (16,6 ha). The quantity decreases from 3 16,9 kg N/ha to 178,3 kg N/ha. The simulation calculates a change of the concentrations of nitrate in the groundwater belonging to the fields from 2 1,5 to 11.1 mg N/l. In the same period the calculated concentration of creek water goes down to 6,1 mg N/l.

The basic requirements of this second scenario are:

- the intensive advice to the farmers about fertilizing based on analysis of soil and manure
- the fully charge of nitrogen contents in manure
- the reduction of the amount of manure by reducing the number of livestock
- payment of economic deficits.

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Quantification of nationwide leaching losses from agricultural land using a soil-plant simulation model

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Abstract

A Danish simulation model for the soil-plant system was used to generate curves for the relation between the applied amounts of nitrogen fertilizer and the losses by leaching from the root zone. Based on information from a national sample inquiry about common farm management practies, figures from agricultural statistics, and the leaching-curves, the leaching losses of inorganic nitrogen from the arable land in five Danish regions were estimated. The simulation results showed very significant variation in the losses between the regions and between the individual years in the simulation period. The greatest losses were calculated from sandy soils in West- and North Jutland, with high input of manure and precipitation, and the smallest from Zealand, with clay soils and low input of farmyard manure and precipitation.

Introduction

In Denmark, as well as in many other West European countries, modem intensive agriculture with high rates of nitrogen application by fertilizers and animal manure to the crops have caused substantial leaching of nitrate from the soils. In many areas of Denmark with unprotected groundwater resources the concentrations of nitrate in the groundwater are well above the maximum level of 50 mg per liter for drinking water.

Nitrogen leaching to surface waters are often believed to controle the biological activity in streams, lakes, nearshore marine waters and has seriously added to the oxygen depletion in many places even in the Kattegat where e.x. lobsters have died in large numbers.

In order to assist decision makers in their evaluation of the Danish »Plan of action against nutrient pollution of the Danish aquatic environment« we have developed a new way to estimate the amount of inorganic nitrogen leached from agricultural soils on a regional scale.

We have used a simulation model »DAISY« developed by Hansen et al. (1990), and have established the information necessary to fulfill the models data demands with respect to soils, landuse, fertilization practices, meterological variables etc. on a statistically representative basis for each region of Denmark.

Description of the model calculations

Based on the different countries in the country, Denmark was divided in five regions as shown in Figure 1.

The regions are:

I Vestjylland (West Jutland)
II Nordjylland (North Jutland)
III Østjylland (East Jutland)
IV Fyn (Funen)
V Sjælland (Zealand)

One of the fundamental ideas of the calculations is to use the DAISY-simulation model to generate curves of the relation between the applied amounts of nitrogen fertilizer and manure, and the losses by leaching from the root zone. For each crop, soiltype and farm management practice with respect to fertilizer application etc. within the regions, we generate curves which relate these practices to the leaching losses from the soils.

The first step in this approach was to set up a scematic outline of the agricultural crop rotation for each region of Denmark.

Based on figures from the official agricultural statistics we were able to calculate how large a percentage of the area each crop occupied in the simulation period 1 January 1983 – 1 April 1990.

The DAISY model is developed to simulate crop production, leaching losses etc. for the following crops:

- Winter wheat.
- Spring barley.
- Sugar beets for fodder or
- sugar production.
- Potatoes.
- Spring rape.
- Grass.

A short summary of the DAISY soil plant system simulation model is given in the appendix.

In our simulations we assumed that winter wheat and winter barley behave similarly, and that oat behave like spring barley. After these approximations and dividing the area according to landuse by different farm types we made the crop rotation scheme for each region.

The four major farmtype were:

- Cattle farming.
- Cattle and pig farming.
- Pig farming.
- Crop farming (farms based on plant production).

Table 1 shows the percentages of the agricultural areas occupied by the different crops in the region Vestjylland. The percentages of arable land included in the simulation period are between 80 and 90. For the other regions these figure were much the same.

Table 2 shows the crop rotation scheme for the Vestjylland region. Each scheme contain 25 rotations which means that one cell represent four percent of the area in region. The 25 rotations were set up in such a way, that the block is balanced according to the figures from the agricultural statistics and so that the different crop rotation combinations are similar to common farming practice for each farm type in the region.

These crop rotation schemes are used to decide how to run the DAISY simulation model with respect to different agricultural crops in the region, which means that $25 \times 8 = 200$ different files are run per region in order to simulate crop productions.

Based on textural data from the Danish Soil Survey the area in each region was split between a sandy and a loamy soil type.

Figure 2 shows the soil water retention characteristics for three different depths of the two soil types. As the water flow in the unsaturated zone is simulated by a numerical solution to Richards equation in the DAISY model, we have given figures for the unsaturated hydraulic conductivity of the model. We have further assumed that the land use by the different crops are independent of the soil type within the regions.

With respect to soil type, Denmark may be devided in two major regions, North and West Jutland which have mainly sandy soils, and East Jutland and the Islands where the loamy and clay soils dominate. The Danish Soil Survey contributes with information of the areas of each soiltype in the regions.

Information on common farm management practices in Danish agriculture was obtained from a national sample inquiry among 925 farmers, out of which 687 gave very valuable information about how and when they applied fertilizer and manure to the crops, and how much they applied to each of the fields. From the sample survey this type of information was obtained for 7662 farm fields for the year 1989.

The nitrogen content of the manure was calculated from the total amount of manure applied in the field and standard values for total nitrogen concentration. Ammonium and dry matter contents in the different manure types were calculated in the same way.

Numbers from the agricultural statistics on livestock production in the regions were used as a check of the reliability of the information from the sample survey. Data from the producers of commercial fertilizers for the mean application of fertilizer per ha arable land in the regions was correspondely used as a check of the sample surveys fertilizer data.

In order to generate curves of the relations between application rates and leaching losses of inorganic nitrogen we decided that we as a minimum wanted four points on each curve. We would have preferred more points for the curve-drawing, but the time used to set up the nessesary files and run the computer reduced the number to four.

The total number of simulations per region were therefore:
$25 \times 8 \times 2 \times 4 = 1600$.

The field observations from the survey were ranked in four equal-sized groups according to the total amount of nitrogen applied to the different crops within each farm type. Based on information from the survey the area was further split up in one part where the crops recieved fertilizer alone and one where both fertilizer and farmyard manure were applied.

We calculated the mean rate of nitrogen application for each of the four groups, and used this in the curve generation. In order to copy the differences in fertilization strategies among the farmers the in the curve generationwe distributed the strategies in a representative way over the 25 crop rotations.

The simulations were carried out for each of these four different fertilization levels. By using these procedures »the residual effect« of manure and crops are simulated in a realistic way.

Required meteorological variables to run the simulation model are daily values of global radiation, air temperature, and precipitation. These data were obtained from the Meteorological Offices data base and from the Agricultural Meteorological Service.

We used area-weighted values for these variables in the curve simulation.

One of the key factors in the simulation model proper initialization of the model. We used a initialization period of four years in order to stabilize the more dynamic parts in DAISY on soil nitrogen and soil water.

The last step in calculating leaching losses in the regions was to use the individual observations in the sampling survey to relate the applied nitrogen rates to the leaching losses for each region by using the generated leaching curves.

We relate the fertilized observations area to these curve-types, and for the observations with both fertilizer and manure we used other curves.

We used linear interpolation between the individual points to draw the leaching-curves for the two fertilization strategies.

Results and discussion

The main results of these simulations are shown in Table 4. There are significant differences between the West- and East Denmark in leaching losses and there are very significant differences between the individual years as well.

North and West Jutland is capracteristic of having a large proportion of animal husbandry farms and a manure input of about 100 kg N per ha arable land on average, while it is only 85 to 50 kg per ha for the region East Jutland – Funen, and Zealand respectively. The input of chemical fertilizers are about 136 kg N per ha per year on average which holds for every region of Denmark.

Another factor to explain the variation between West- and East Denmark is the water dynamic. As illustrated in Table 3 the precipitation decrease from West to East-Denmark which in combination with land use and the soil type lead to some regional differences in the yearly water percolation from the arable land, as shown in Table 5.

Another way to evaluate our simulations is to make nitrogen balance-sheets for input and output.

Figures from such calculations based on informations from the Agricultural Statistics, Nielsen (1990), are shown in Table 6-9 in combination with figures from these simulations.

From these tables, one find that there are some differences between calculated and simulated values for the nitrogen contents in grass. These differences are partly explained by a too high production for the grazed pasture in the simulation as compared to the figures from the Agricultural Statistics and partly by use of a too low nitrogen concentration for calculating the nitrogen contents in the grass by Nielsen (1990).

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Tables for papers

Table 1.
The land use of agricultural crops in the region »Vestjylland« from 1983 until 1989. The area used by the listed crops is set equal to 100 percentages each year. Source: Agricultural Statistics, 1983-89.

	1989	1988	1987	1986	1985	1984	1983
Winter wheat & Winter barley	14.0	7.8	11.4	9.0	***9.6***	11.3	***5.4***
Spring barley & Oats	40.8	46.1	39.2	44.0	***43.8***	42.0	***50.2***
Sugar beets for fodder & sugar	6.6	6.6	6.9	6.9	***7.1***	6.8	***6.3***
Potatoes	2.2	1.8	2.0	1.9	***1.9***	1.8	***1.7***
Spring rape	5.0	8.0	n8.8	7.1	***5.7***	5.5	***3.4***
Cereals for fodder	3.7	3.2	3.5	3.4	***3.6***	3.7	***3.8***
Grass in rotation	16.5	15.9	17.0	17.2	***17.7***	18.4	***18.8***
Permanent grassland out of rotation	11.3	10.5	11.1	10.5	***10.6***	10.5	***10.6***
Total arable area in 1000 ha:	1067	1067	1072	1076	***1084***	1085	1088
Percentage of the arable area included in the simulacion:	***86.5***	87.4	81.9	85.5	***86.8***	90.2	***93.8***

Figure 1:

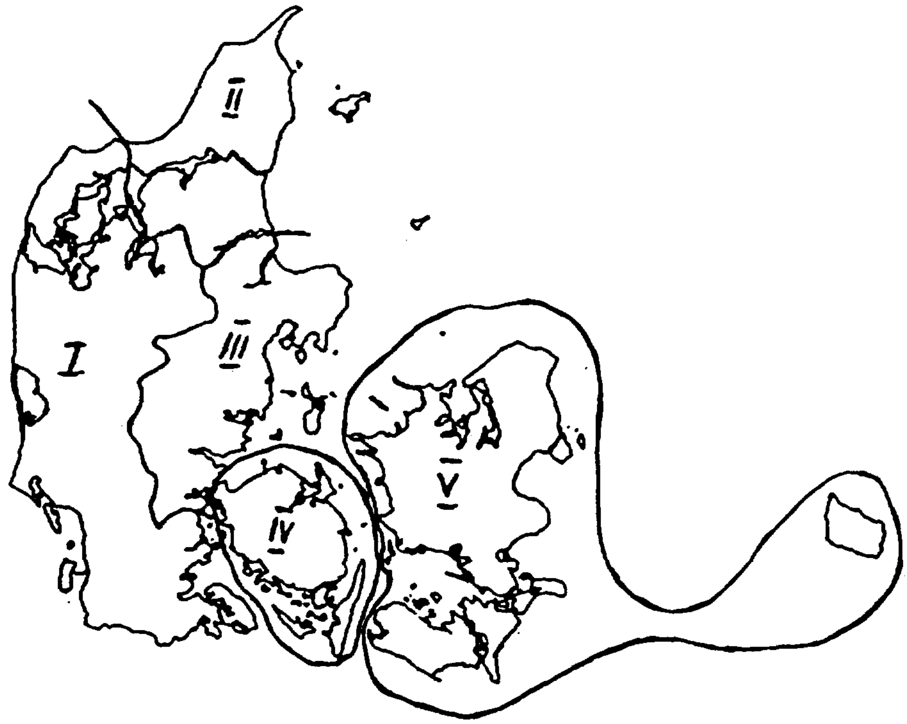

Figure 2:

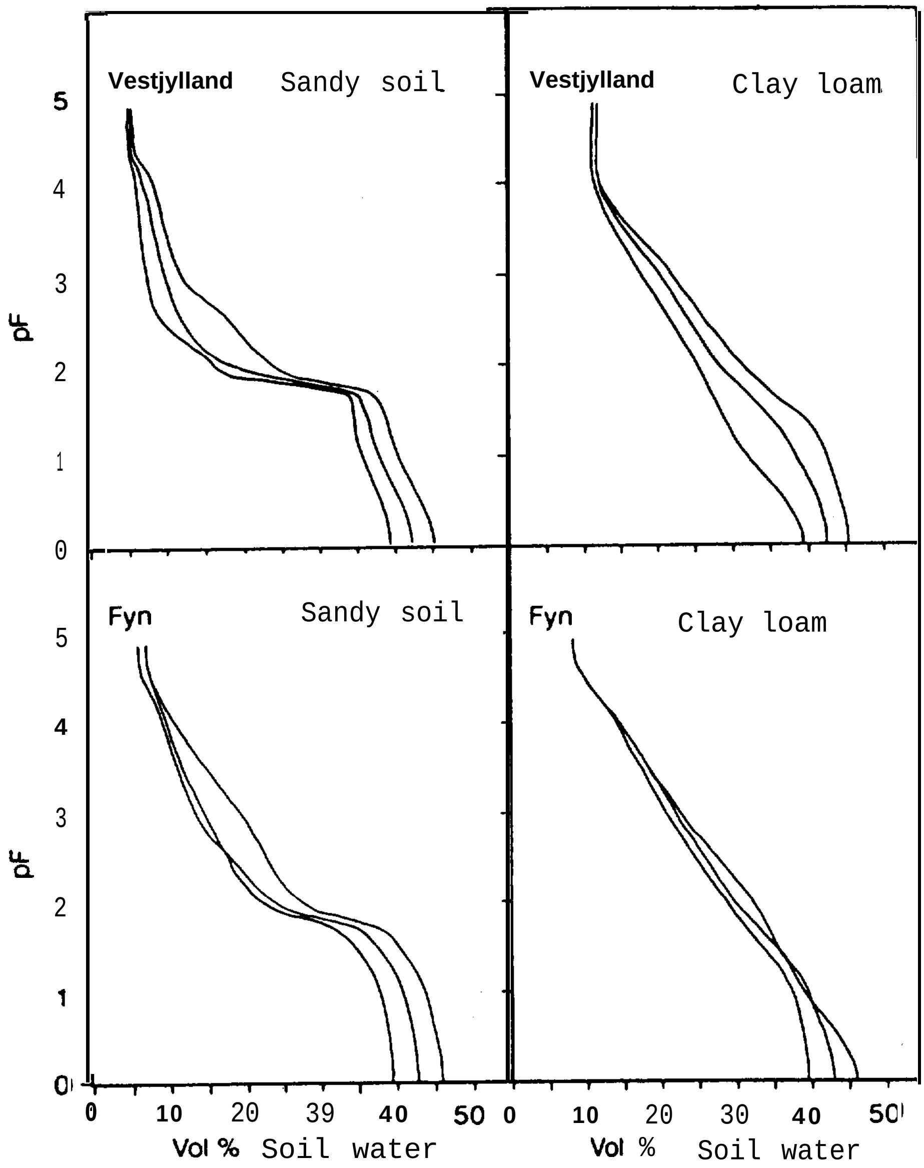
Vestjylland
Sandy soil
Vestjylland
Clay loam
Fyn
Sandy soil
Fyn
Clay loam
pF
pF
5
4
3
2
1
0
5
4
3
2
1
0
0
10
20
39
40
50
0
10
20
30
40
50
Vol % Soil water
Vol % Soil water

Table 2.
Schematic *outline of agricultural crop rotation in the region »Vestjylland« of Denmark.*
The arable land is devided according to land use by different farm type and agricultural crops within the farm types.
One cell represent four percent of the area ***included*** in ***the simulation. Bar. = Spring barley or oats,*** *Bar.+G=* ***Spring barley with undersown grass, GrassG = Grazed pasture, GrassH = Harvested grass, Cer. = Cereals for fodder, W. wheat = Winter wheat or W. barley,*** *S.rape* ***= Spring*** *rape,and Potat.* ***= Potatoes.***

No. *Year*	*9*	10	*11*	*12*	*13*	*14*	*15*	*16*	*17*	*8*
1990			GrassG	Cer.	Bar.+G	Beets	Bar.	Bar.	Bar.+G	GrassG
1989			GrassH	Bar.+G	Cer.	W.wheat	GrassG	Beets	Bar.	GrassH
1988	**Grass out of rot**	**Grass out of rot**	Cer.	Beets	Bar.	S.rape	GrassH	GrassG	Beets	Bar.+G
1987			Bar.+G	Bar.	GrassG	Cer.	Bar.+G	GrassH	Bar.	Beets
1986			Bar.	GrassG	Bar.+G	Bar.	Beets	Bar.+G	GrassG	Bar.
1985			W.wheat	GrassG	Cer.	Bar.	Bar.	Beets	GrassH	Bar.
1984			GrassG	GrassH	Bar.	Beets	W.wheat	Bar.	Bar.+G	GrassH
1983			GrassH	Bar.G.	Beets	Bar.	S.rape	GrassG	Bar.+G	

Cattle farming area

No. Year	1	*2*	*3*	*4*	*5*	*6*	*7*	*8*
1990	GrassG	GrassH	Bar.	Bar.+G	W.wheat			
1989	GrassH	Bar.+G	Beets	W.wheat	GrassG			
1988	Bar.+G	W.wheat	Bar.	GrassG	GrassH	Bar.		
1987	Beets	S.rape	W.wheat	GrassH	Bar.+G	Bar.		
1986	Bar.	Bar.	GrassG	Bar.+G	Cer.	W.wheat		
1985	W.wheat	GrassG	GrassH	Bar.	Bar.	S.rape	Beets	
1984	GrassG	GrassH	Bar.+G	Cer.	W.wheat	Bar.	Bar.	
1983	GrassH	Bar.+G	Bar.	Bar.	GrassG	Beets	GrassG	Cer.

Cattle and pig farming area

Year \ No.	18	19	20	21	22
1990	W.wheat	S.rape	Bar.	W.wheat	Bar.
1989	S.rape	Bar.	Bar.	Bar.	W.wheat
1988	Bar.+G	Bar.	Bar.	Bar.	S.rape
1987	Bar.	Bar.+G	W.wheat	Potat.	GrassH
1986	Bar.+G	Bar.	S.rape	Bar.	Bar.+G
1985	Bar.	W.wheat	Bar.	Bar.+G	Bar.
1984	Bar.	S.rape	Bar.+G	Bar.	Bar.
1983	Bar.	Bar.	Bar.	Bar.	W.wheat

Pig farming area

Year \ No.	23	24	25	7	6
1990	Bar.	W.wheat	Bar.	S.rape	W.wheat
1989	Bar.	Potat.	W.wheat	Bar.	Bar.
1988	Bar.	Bar.	Bar.	W.wheat	
1987	W.wheat	Bar.	Bar.	S.rape	
1986	S.rape	W.wheat	GrassH	Potat.	
1985	Bar.	S.Rape	Bar.+G		
1984	Potat.	Bar.	W.wheat		
1983	Bar.	Bar.	Bar.+G		

Agricultural farming.

Table 3.
Yearly percipitation for Denmark and five regions of Denmark. The periods starts 1 April first year and runs until 31 Mar. the following. The percipitatian are corrected to soil surface according to Allerup and Madsen (1979).

Region	Percipitation in mm per year 6/87	87/88	88/89	89/90	Mean 86-90
Vestjylland	865	1255	1011	861	***995***
Nordjylland	741	1095	854	780	***865***
Ostjylland	711	1097	749	751	***825***
Fyn	624	46	723	696	***742***
Sjælland	76	913	647	698	***732***
Denmark	61	1108	842	785	872

Table 4.
Annual means of simulated nitrogen leaching from the arable land in Denmark and five Danish regions. The results are given in kg inorganic N per ha and in tons N per region.

	Vestjylland	*Nordjylland*	*Østjylland*	*Fyn*	*Sjælland*	**Denmark**
Year 1986187						
Tons	72.000	24.000	28.000	9.000	12.000	144.000
Kg N/ha	67	59	56	37	20	51
Year 1987188						
Tons	87.000	40.000	42.000	16.000	3 1 .000	216.000
Kg N/ha	81	99	87	68	52	77
Year 1988/89						
Tons	68.000	24.000	2 1 .000	11 .000	12.000	136.000
Kg N/ha	64	58	45	46	19	49
Year 1989190						
Tons	76.000	30.000	29.000	16.000	29.000	181.000
Kg N/ha	71	76	61	70	49	65

Table 5.
Simulated yearly percolation at 1.3 m's depth from the'arable land in Denmark and in five regions of Denmark. The periods starts 1 April first year, and runs until 31 Mar. the following.

Region	Percolation in mm per year				
	6/87	87/88	88/89	89/90	Mean 86-90
Vestjylland	393	760	516	407	516
Nordjylland	276	614	346	323	387
Ostjylland	227	609	249	277	338
Fyn	168	30	194	210	246
Sjælland	57	398	130	184	215
Denmark	80	611	336	312	382

Table 6.
Nitrogen contents in livestock manure and the total nitrogen input into Danish agriculture 1980-88

	88/87	87/86	86/85	85/84	84/83	83/82	82/81	81/80
	Mill. Kg N							
Nitrogen contents in fodder and litter	423	***423***	427	***428***	424	440	461	457
Nitrogen contents in livestock products	86.3	***86.9***	87.9	***86.3***	84.4	85.0	81.3	81.3
Nitrogen contents in farmyard manure ex animal	337	336	***339***	***342***	340	355	380	376
Percentage of fodder nitrogen recovered in farmyard manure and straw	79.7	79.4	***79.4***	79.9	80.2	80.7	***82.4***	82.3
N in sludge and fruit spread on the farm land	4.0	4.0	4.0	4.0	4.0	4.0	***4.0***	4.0
Precipitation and dry depositing	28	28	28	28	29	29	***29***	29
Nitrogen fixation	40.9	42.2	41.9	31.7	29.4	28.5	***28.5***	30.2
Consumption of commercial nitrogen fertilizers*	357	371	372	388	402	381	***366***	364
Total nitrogen input	767	781	785	794	804	797	808	803
Nitrogen input of kg N per ha cultivated area	275	278	278	279	282	278	279	277

* 10 mill. kg N used for NH, treatment of straw have been deducted

Table 7.
Nitrogen contents* in *the* harvest products, which *have been removed from* the *fields 1980–88

	88/87	87/86	86/85	85/84	84/83	83/82	82/81	81/80
	Mill. Kg N							
Cereals	**141.0**	**140.0**	**146.8**	**158.8**	**144.5**	**132.2**	**140.0**	**131.2**
Pulses	**17.5**	**18.2**	**18.6**	**14.1**	**6.2**	**2.0**	**0.9**	**0.5**
Potatoes	**3.9**	**3.7**	**3.9**	**3.9**	**3.5**	**3.6**	**4.0**	**3.3**
Sugar Beets	**19.4**	**19.6**	**23.0**	**24.9**	**22.2**	**21.8**	**23.4**	**20.6**
Tops of sugar beets	**12.3**	**15.0**	**17.6**	**19.0**	**13.5**	**19.3**	**22.2**	**22.5**
Grass in and out of rotation	**85.2**	**79.8**	**84.1**	**94.1**	**93.0**	**95.5**	**104.8**	**103.8**
Seeds for manufacturing and for sowing	**20.1**	**22.1**	**19.5**	**17.2**	**11.5**	**12.4**	**11.2**	**8.6**
Maize and cereals for silage	**6.7**	**6.9**	**7.6**	**7.4**	**6.6**	**6.2**	**6.1**	4.8
Lucerne, after grass etc.	**10.2**	**9.0**	**10.4**	**11.6**	**11.6**	**12.3**	**14.2**	**14.9**
Straw removed from the fields*	**20.9**	**22.0**	**21.4**	**26.8**	**20.8**	**25.7**	**28.0**	**19.4**
Rape- and seed grass straw removed from the fields*	**3.2**	**3.6**	**3.3**	**3.1**	**2.4**	**2.7**	**2.5**	**2.0**
Vegetables	0.8**	**0.8**	**0.8**	**0.8**	**0.7**	**0.7**	**0.8**	**0.7**
Total	**341.2**	**340.7**	**357.0**	**381.7**	**336.5**	**334.4**	**358.1**	**332.3**

* Incl. straw burned down on the fields

** Estimate

Table 8.
Simulated annual amounts of nitrogen harvested in different agricultural crops. The amounts are calculated as means for the period of 1 April 1986 – 31 Mar. 1990 for Denmark and for five regions of Denmark.

Region	***Vestjylland***	***Nordjylland***	***Østjylland***	***Fyn***	***Sjælland***	***Demark***
Crop			***tons N***			
Grazed pasture	***60.754***	***30.064***	***26.303***	7.860	16.590	141.572
Harvested grass	***38.870***	***14.262***	11.307	1.804	4.117	70.359
Cereals for fodder	***2.871***	***943***	685	89	4.598	
Potatoes	***2.704***	***651***		479	3.833	
Beets (root)	***8.355***	***2.223***	1.954	3.054	7.023	22.609
Winter wheat & W.barley	10.669	***4.500***	12.748	8.918	20.249	57.085
Spring barley and oats	***22.545***	***7.014***	9.656	6.567	15.620	61.403
Barley w. grass	***9.100***	***3.153***	3.297	897	2.409	18.856
Undersown grass	***4.693***	***1.278***	789	290	331	9.321
Spring rape	***7.561***	***3.616***	5.135	1.612	3.953	21.877
Straw (cer.)	16.159	***4.739***	7.509	4.737	10.301	43.444
Straw (rape)	2.139	1.394	1.486	424	990	6.433
Beets (top)	***3.549***	1.199	949	47	2.796	8.540
Total	191.902	75.037	81.828	36.300	84.858	469.932

Table 9.
Nitrogen contents in the harvest production and nitrogen losses to the environment

	88/87	87/86	86/85	85/84	84/83	83/82	82/81	81/80
	Mill. Kg N							
Nitrogen contents in the harvest	341.2	340.7	357.0	381.7	336.5	334.4	358.1	332.8
Total N-input into agriculture	767	781	785	794	804	797	808	803
Nitrogen losses and accumulation in the soil	426	440	428	412	468	463	450	470
+N loss when burning down straw	5	5	5	9	6	9	9	9
Total N-losses to the environment	431	445	433	421	474	472	459	479
Total N-losses of kg pr.ha cultivated area	154	159	153	148	166	165	158	165
% of N-inputussed for harvest production (Utilization % for N-input)	44	44	45	48	42	42	44	41

Appendix 1

From Hansen et al. (1990)

Summary

Daisy is a mathematical model for simulation of crop production, soil water dynamics, and nitrogen dynamics in crop production at various agricultural management practices and strategies. The particular processes considered include transformation and transport processes involving water, heat, carbon, and nitrogen. For some of the processes theories and mechanisms are well established while for other processes existing knowledge is limited. The various processes considered have been described and modelled in accordance with existing knowledge.

The hydrological processes considered in the model include snow accumulation and melting, interception of precipitation by the crop canopy, evaporation from crop and soil surfaces, infiltration, water uptake by plant roots, transpiration, and vertical movement of water in the soil profile. In the model snow melting is influenced by incident radiation, and soil and air temperatures. Interception is determined either by precipitation or by the crop canopy. Description of evapotranspiration is based in a climatical determined potential evapotranspiration and the availability of water. Modelling of water uptake by plant roots is based on a quasi steady state solution of the differential equation for radial water flow to the root surfaces, and the plant root density in the soil profile. The vertical movement ofwater in the soil profile is modelled by means of a nummerical solution of the Richards equation.

Soil temperature is modelled by solving the heat flow equation taking into account heat transfer by conduction and convection, and changes in heat content by freezing and melting processes. The freezing process induces water flow in the soil as ice formation is assumed to take place in the large soil pores extracting water from small soil pores resulting in water flow towards the freezing zone.

Organic matter turnover is modelled by dividing the organic matter conceptually into three main pool viz. added organic matter which includes organic matter in plant residues and in manure, microbial biomass which includes organic matter in living microorganisms, and soil organic matter which includes non living native organic matter in the soil. Each main pool of organic matter is subdivided into two subpools each one being characterized by a particular carbon nitrogen ratio and by a particular turnover time. For each subpool of soil organic matter and added organic matter carbon turnover is modelled by applying first order kinetics assuming the rate coefficient to be influenced by soil temperature and soil water content. In the case of subpools of soil organic matter the rate coefficients are assumed also to be influenced by the clay content of the soil.

The biomass utilizes organic matter as substrate. Each subpool of biomass

is characterized by a substrate utilization efficiency, a maintenance respiration coefftcient, and a death rate coefftcient. The maintenance respiration and the death rate are assumed to be influenced by soil temperature and soil water content, and in the case of the resistant biomass subpool also by the clay content of the soil. Carbon is lost as carbon dioxide due to the biomass respiration processes.

During biomass growth and decay carbon is translocated between the various subpools of organic matter during which processes mineral nitrogen is released or immobilized depending on the carbon nitrogen ratio of the organic matter being utilized as substrate and the carbon nitrogen ratio in the microorganisms being synthesized. The overall result of all the organic matter turnover processes is net mineralization of nitrogen which may be positive in which case ammonium is released or negative in which case ammonium or nitrate is immobilized.

The soil mineral nitrogen processes in the model include nitrification, denitrification, uptake by plant roots, and vertical movement in the soil profile. Nitrification is simulated by applying first order kinetics assuming the rate coefficient to be influenced by soil temperature and soil water content.

Denitrification is simulated by defining a potential denitrification rate assumed to be related to the carbon dioxide evolution rate in the soil and the soil temperature. The potential denitrification rate is reduced according to the oxygen status of the soil expressed as a function of soil water content. The actual denitrification rate is either determined as the reduced potential denitrification rate or as the rate at which nitrate in soil is available for denitrification.

The nitrogen uptake model is based on the concept of a potential nitrogen demand simulated by the crop model, and the availability of nitrogen in the soil for plant uptake. Modelling of the uptake of nitrogen by plants is actually based on a quasi steady state solution of the differential equation for diffusive and convective radial transfer of nitrogen to the plant root surfaces, and the plant root density in the soil profile. Ammonium is taken up by plant roots in preference to nitrate.

The mobility of the ammonium in soil is considered less than that of nitrate due to adsorption of ammonium to soil Colloids which isdescribed by an adsorption desorption isotherm. The vertical movement of nitrogen is modelled by means of a nummerical solution of the convection dispersion equation for ammonium as well as for nitrate. The source sink term in the convection dispersion equation integrates the transformation processes in the case of ammonium as well as in the case of nitrate.

The crop model is based on the concept that the physiological age of a crop can be described as the thermal age in terms of a temperature sum. The crop canopy is described in terms of total crop area index and green crop area index simulated as functions of crop thermal age, and accumulated top dry matter.

Root penetration rate is simulated as a function of the soil temperature at the root tip, while root density is simulated as a function of the amount of accumulated root dry matter, and the rooting depth.

Simulation of crop dry matter production is based on calculation of canopy gross photosynthesis and partitioning of assimilates between the various parts of the crop. Canopy gross photosynthesis is determined by the amount of photosynthetically active radiation absorbed by the crop canopy and the efficiency by which the absorbed radiation is converted into carbohydrates. Partitioning of assimilates from gross photosynthesis between crop parts, i.c. top, roots, and for some crops storage organs, is simulated as a function of the crop thermal age. Respiration comprises growth respiration and a temperature dependent maintenance respiration.

The gross photosynthesis of the crop canopy may be limited due to water or nitrogen deficiency. In the case of water deficiency gross photosynthesis is reduced by the ratio of actual and potential transpiration. At high and extremely low nitrogen supply the nitrogen concentration in the dry matter is assumed to reach an upper and a lower limit, respectively, both of which are dependent of the crop thermal age. Between these limits a nitrogen concentration in dry matter a just ample nitrogen supply exists also depending on the crop thermal age. The limitation of gross photosynthesis due to nitrogen deficiency is assumed proportional to the deviation of the actual nitrogen concentration in dry matter from the nitrogen concentration in dry matter at just ample nitrogen supply.

The system management model allows for various agricultural management practice sand strategies including different crop systems, fertilization, irrigation, and soil tillage.

Required meteorological variables to run the simulation model are daily values of global radiation, air temperature, and precipitation. Furthermore a number of parameters characterizing the system is required.

Acknowledgement

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Evaluation of management strategies to combat nitrogen leaching

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Summary

A simulation model was used to evaluate different management strategies to combat nitrogen leaching from arable land in Western Jutland, Denmark. Leaching with actual management was estimated to 7 1 kg N/ha, with half of the manure applied in spring and half in autumn. Applying all manure in spring reduced the leaching to 67 kg N/ha. A further reduction in leaching to 59 kg N/ ha was obtained when all manure were applied in spring in combination with a reduction in application of fertilizer. The actual applied amount of fertilizer and manure to crops varied between different farmtypes and application strategies. The farm manure was only applied to half of the farms area giving very high nitrogen input to beets and spring rape, which respond with high leaching from these crops. An optimized application of manure to larger areas, where the nitrogen in the manure are accounted for giving nutrients to the crop production, may further reduce nitrogen leaching from arable land.

Introduction

In connection with me goals for the »Plan of action against nutrient pollution of the Danish aquatic environment« much attention has been paid to the effect of different management practice on reduction of nitrogen leaching from arable land. Information on actual management and leaching are known from quantification of nationwide leaching from agricultural land simulated with a soil-plant-model (Nielsen et al., 1992). A **national** survey sample in 1989 among 687 farmers, representing 7662 fields gave information on actual management of fertilizer and manure (Fredriksen, 1990). Type of fertilizer and manure, and amount of and month for application to different crops were important for realistic simulations of nitrogen leaching. In the sample half of the manure were added in spring and the other half in autumn. The objectives of this investigation were to estimate nitrogen leaching and yield with all manure applied in the spring and the effect on leaching with this management in combination with a reduction in applied fertilizer. In this paper results from Western Jutland, covering four counties with a cultivated area on 1.07 1 .OOO ha are presented.

Methods

Description of simulations by a soil-plant-model

The soil-plant-model simulate nitrogen leaching and crop production in two units: A waterbalance section and a section for carbon-nitrogen cycling (Hansen et al., 1990).

Driving variables for climate were based on data from stations covering the region. Two different soil types were used in the simulations: A sandy soil with 4.5% clay and 51.8% sand covering 67% of the area and a clay loam with 9.5% clay and 34.1% sand covering 25% of the area (Nielsen et al., 1992).

Modelsimulations were carried out from 1.4.1983 to 3 1.3.1990, with the period from 1983 to 1986 was as an initial1 period for the nutrient cycling in organic pools in soil. Reported values are annual means from 1986 to 1990 estimated from April 1[st] to March 3 1" .

Analysis of agricultural management and crop characteristics

Agricultural management and crop characteristics were analyzed for 254 farms in the region by classifying the farms into four types as follows: farms with no pigs and no cattle, but maybe other domestic animals; Cattle & pig farms with both pigs and cattle and maybe other domestic animals; Cattle farms with cattle and no pigs and maybe other domestic animals; and Pig farms with pigs and no cattle, but maybe other domestic animals.

To represent crops in the simulations, 25 crop rotations were generated based on the area of crops in the region as reported in the Agricultural Statistics 1983-1990.

Crop rotations were classified between the four types of farms in relation to crop area for these types. Four crop rotations were simulated with management practice for agricultural farming areas, 5 with management practice for cattle and pig farming areas, 10 with management practice for cattle farming areas and 5 with management practice for pig farming areas.

The model include crop parameters to simulate crop production for six crops: Winter wheat, spring barley, spring rape, fodder beets, potatoes and grass. In modelsimulations, winter wheat represents areas with both winter wheat and winter barley, spring barley represents areas with both spring barley and oats, and beet represents both sugar beets and fodder beets. With these crop parameters for the crop rotations the area of the crops in the model-simulations covered 85% of the cultivated area in Western Jutland.

For each farmtype, application strategies for fertilizer and manure to crops were divided into four groups: Fertilizer corresponds to application of commercial fertilizer alone, "Fertilizer & Manure" corresponds to application of both commercial fertilizer and manure, "Manure" corresponds to application of manure alone and "Nothing" corresponds to areas with no input.

Four application levels of total nitrogen were simulated for each crop rotation and soil type generating a dose-response curve application of total nitrogen and leaching. From 2930 fields in the survey sample the actual values of applied nitrogen were used to calculate leaching values by use of the dose-response curve.

Design for management strategies to combat nitrogen leaching

Analysis of the survey sample showed that in actual management half of the manure was added in spring and half in autumn. The effect of change in management of fertilizer and manure on leaching was simulated in two steps by the model. First, the effect on leaching was simulated when all the manure was added in spring just before date of sowing and other management practice such as type, amount and distribution of fertilizer and manure were kept at actual practice. Secondly, the effect on leaching was estimated when all manure were added in spring in combination with a reduction in application of fertilizer to 50 kg N/ha on these areas. On areas with application of fertilizer alone the amounts were reduced to recommended values.

Recomended values for application of fertilizer to different crops:

Grass	300 kg N/ha
Cereals for fodder	130 kg N/ha
Potatoes	150 kg N/ha
Sugar beets	120 kg N/ha
Fodder beets	160 kg N/ha
Winter wheat	170 kg N/ha
Spring barley	110 kg N/ha
Spring rape	170 kg N/ha

Furthermore, the effect of application of high and low amounts of manure on leaching and yield were estimated for four crops. All manure was applied in spring together with a reduction in application of fertilizer to 50 kg N/ha. On areas with application of fertilizer alone the amounts of fertilizer were reduced to recommended values.

Results and discussion

Analysis of agricultural management for application of fertilizer and manure showed that the farmers added manure alone or nothing at all to less than 3% of the area, all other areas were added fertilizer and manure (Table 1). Application of fertilizer alone was used on 76% of the areas with agricultural farming, while 24% of these areas were added both fertilizer and manure. For farms with animal husbandry the manure was applied only to half of the farm areas while fertilizer alone was applied to the other half.

The amounts of fertilizer and manure added to different crops were different between the types of farms. There were high applications to beet and rape for farms with animal husbandry (Table 2). For farms with cattle and pigs and with cattle alone beets were given very high amount of manure, with a mean application of respectively 555 and 660 kg N manure/ha on 25% of the areas cropped with beets. These amounts of manure were supplemented with a further application of 90-l 00 kg N/ha of fertilizer to the same areas. For pig farming areas and for cattle and pig farming areas spring rape was given high amounts of manure, respectively 7 16 and 45 1 kg manure N/ha as a mean for 25% of the areas cropped with rape for these two farm types.

The application of both manure and fertilizer showed, that many farmers do consider the nutrients in the manure to have little effect on crop production.

The reduction in leaching when all manure was added in spring, was only 4 kg N/ha (Table 3). Areas with application of manure were often given high amounts of total nitrogen. When manure was applied in autumn, leaching occurred through winter and the nutrients were not available for plant growth. When manure was applied in spring the nutrients were available for plant growth during spring and summer, but in many occations the plant-soil system was in excess of nitrogen because the input of nitrogen on areas applied with manure were high. The crop production on these areas was more limited by plant available water than nitrogen. Excess nitrogen on these areas was leached in later periods with run-off. A further reduction in leaching of 9 kg N/ha was obtained when the application of fertilizer was reduced to 50 kg N/ha on areas with application of manure and to recommended values for application of fertilizer alone. The differences with in yield between the three different management strategies were very little.

Leaching losses were very high from areas with high application of manure (tabe14). For areas cropped with beet and rape, application of respectively 487 and 331 kg manure N/ha gave rise to a leaching of 145 and 182 kg N/ha. A more optimal distribution of manure on larger areas in combination with a better utilization of the nitrogen in manure for crop production is bound to further reduce the nitrogen leaching from arable land.

. Final remarks

The simulation results presented must be interpreted with care. A number of uncertainties have been introduced by extending 25 crop rotations with four levels of nutrient application and in combination with two types of soil as representing agricultural management and leaching for a larger region. Nevertheless the analysis of actual management practice and simulation of nitrogen leaching with change in management gives useful information on the dynamics of nitrogen leaching in order to evaluate tools for regulation of farming management.

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Tables for paper

Table 1.
Different management strategies for application of fertilizer and manure to agricultural fields estimated in percent of cultivated area for 254 farmers classified into four farmtypes in Western Jutland, Denmark.

Farmtype	fertilizer	fertilizer & manure	manure	nothing
Agricultural farming (65)[1]	76	24	0	0
Cattle & pig farming (43)	49	48	3	0
Cattle farming (96)	45	53	1	1
Pig farming (59)	56	43	1	0

[1] numbers of farmers in investigation

Table 2.
Application of fertilizer and manure to springbarley, beets and springrape estimated as four 25% intervals for four farmtypes in Western Jutland, Denmark. F is application of commercial fertilizer alone and F & M are application of commercial fertilizer and manure. All data are reported in kg N/ha.

	S. BARLEY			*BEET*			S. *RAPE*		
	F	*F & M*		*F*	*F & M*		*F*	*F & M*	
Agricultural farming areas									
0- 25% of areas	90	96	113	-			90	122	111
25- 50% of areas	166	168	193	-			166	130	138
50- 75% af areas	128	82	189	-			128	82	195
75–100% of areas	148	163	196	-			148	152	247
Cattle & pig farming areas									
0- 25% of areas	80	85	87	-	107	*253*	148	104	119
25- 75% of areas	112	84	148	-	102	*379*	166	135	182
50- 75% of areas	121	96	172	-	109	*484*	178	135	250
75-100% of areas	136	87’	268	-	103	*660*	197	73	451
Cattle farming areas									
0- 25% of areas	78	78	86	-	99	*208*	126	104	111
25- 50% of areas	102	84	125	-	111	*320*	157	135	125
50- 75% of areas	119	96	152	-	111	*383*	171	135	195
75-100% of areas	146	105	220	-	91	*555*	183	73	233
Pig farming areas									
0- 25% of areas	101	96	89	-			121	125	139
25- 50% of areas	113	92	142	-			148	116	213
50- 75% of areas	120	86	195	-			163	123	255
75– 100% of areas	150	110	310	-			175	97	716

Table 3.
Effect of change in time of application of manure and in combination with a reduced application of fertilizer on leaching and yield for Western Jutland, Denmark. Results are annual mean for the period April 1st 1986 to March 31" 1990.

	Actual[1]	Spring[2]	Spring+reduction[3]
Fertihzer (kg N/ha)	143	143	111
Manure (kg N/ha)	88	88	88
Leaching (kg N/ha)	71	67	59
Yield			
kg N/ha	193	198	190
ton DM/ha	7,82	7,87	7,76

[1] Half of the manure is added in spring and half is added in autum.
[2] All manure are added in spring.
[3] As 2 in combination with a reduction in added fertilizer – see text.

Table 4.
The effect of different management strategies on leaching and yield for beets, spring-barley, winter-wheat and spring-rape in Western Jutland. F-alone is for areas with application of commercial fertilizer alone, FM-low is for areas with low input of manure and reduced input of fertilizer and FM-high is for areas with high inputs of manure and reduced input of fertilizer. The results are annual mean for the period April 1st 1986 to March 31th 1990.

	fertilizer (kg N/ha)	manure (kg N/ha)	leaching (kg N/ha)	yield (kg N/ha)	yield (ton DM/ha)
BEET					
F-alone					
FM-low (54,9)[1]	50	254	85	133	12,8
FM-high (45,1)	50	487	145	145	12,9
S.BARLEY					
F-alone (67,5)	104	0	67	72	3,3
FM-low (13,8)	50	88	99	95	4,2
FM-high (18,7)	50	197	130	101	4,3
W. WHEAT					
F-alone (53,2)	159	0	45	109	6,5
FM-low (2 1,4)	50	78	47	163	7,9
FM-high (25,4)	50	215	74	174	8,1
S.RAPE					
F-alone (33,7)	104	0	72	72	3,3
FM-low (29,2)	50	110	93	95	3,6
FM-high (37,1)	50	331	182	101	4,2

[1] Figures in brackets are percent area for the crop.

Nitrogen removal in freshwater ecosystems and coastal areas

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Introduction

The environmental impact of nutrients (e.g. nitrogen and phosphorus) has been throughly documented over the last decades. It has been shown that phosphorus generally is the prime limiting nutrient for primary production in freshwater ecosystems (Thyssen et al., 1990; Kristensen et al., 1990) while nitrogen may be limiting under certain conditions as in nutrient poor lakes in sandy areas (Schuurkes et al., 1986; 1987) or in lakes covered with submerged macrophytes (Ozimek et al., 1990; Jeppesen et al., submitted). In coastal areas both nutrients may be temporary limiting. In early spring phosphorus limits production (Borum et al., 199 1) and during summer and autumn nitrogen becomes limiting.

The regulation of primary production in the open marine areas has been much debated in recent years. Most research shows that nitrogen is the major limiting nutrient (Graneli, 1984; 1987; Richardson & Ærtebjerg, 199 1) although the relationship between nitrogen (N) and phosphorus (P), the N/P-quotient, is argued to determine the species composition of the phytoplankton community. High values of the N/P-quotient has been shown by Skjoldal & Aure (1989) and Edvardsen et al. (1990) to favour mass occurence of toxic algae (e.g. Prymnesium parvum and Chrysochromulina polylepis) in the seas around Denmark and Norway in 1989.

Decreasing the input of nitrogen to aquatic ecosystems or any process that removes or immobilizes nitrogen is therefore likely to improve the ecological conditions of the system.

Removal of nitrogen in aquatic ecosystems is mainly caused by three processes:

- removal of organically bound nitrogen
- sedimentation of organically bound nitrogen
- heterotrophic respiration of nitrate (denitrification)

In most systems, denitrification is by far the most important process regarding nitrogen removal.

In the present paper we summarize the present knowledge on aquatic denitrification: its magnitude in different environments, its regulation and propose management strategies to increase it.

Danish actions to reduce nitrogen in the aquatic environment

In order to reduce the risk of nutrient induced environmental catastrophies such as the oxygen depletion of bottom waters and associated fish kills the Danish Parliament in the 70ties and 80ties has passed several actions and initiatives to improve the environmental conditions of our waters including several research projects on the importance of denitrification for nitrate removal in freshwater and marine environments.

The cornerstone in this series of initiatives is the Action Plan on the Aquatic Environment which was passed by the Parliament in 1987. The main objectives of the plan were to reduce the input of nitrogen and phosphorus to the environment by 50 and 80%, respectively before 1994.

In order to control whether nutrient reductions would occur a nationwide monitoring programme was established not only to detect any trends in nutrient concentrations but also to gather information on changes and improvements in biological structure of the aquatic systems.

The fate of nitrate in a river system

Nitrate is the dominating nitrogen species in stream water. On the average, 83% of total nitrogen in were shown to be nitrate 130 Danish streams (Kristensen et al., 1990).

Nitrate concentrations in streams are predominantly determined by human activities in the catchments. Nitrates are not passively transported through stream systems, but can undergo a number of biologically mediated transformations of which uptake by growing plants and decomposers and denitrification within anaerobic layers of stream sediments are considered the most important. Although some studies (Howard-Williams et al., 1982; Cooper & Cooke, 1984) have attributed observed nitrogen losses in streams to macrophyte uptake most recent studies have identified denitrification as the major mechanism of nitrate removal.

Based on analysis of nitrate data obtained from the Danish Environmental Monitoring Programme in 1989, it was shown that small first to second order streams draining catchments dominated by agriculture had significantly higher nitrate concentrations (yearly mean = 5.9 mg nitrate-N l^{-1}, number of streams = 56) than streams draining catchments covered by natural vegetation or forest (mean = 1.35 mg l^{-1}, n = 7) Kristensen et al., 1990. On its course through the landscape, a stream continuously receives incoming water from various sources: tributaries, overland flow, tile drainage and groundwater in different proportions and with highly varying nitrate concentrations over season, modifying the initial upstream concentrations.

The spatial variation in nitrate concentrations in a river system is illustrated in figure 1 where data from 32 sampling stations on the largest Danish river, The Gudenå with a catchment area of 2580 km², which is located in central Jutland in a predominantly agricultural area, has been compiled and analyzed. Only major tributaries (catchment area > 10 km²) have been included in the analysis.

Generally, nitrate concentrations tend to be high in the upper, small streams and to decrease in the downstream direction. This tendency is particularly obvious for stream systems with inserted lakes like the rivers Hinge and Knud. On the contrary there is only insignificant spatial change in the River Lilleå which has no lakes included.

On a whole, 14 sampling stations are located in streams without upstream lakes and 18 including lakes. Mean nitrate concentrations were 5.0 mg nitrate-N l^{-1} for the former group and 2.1 mg l^{-1} for the latter, indicating that lakes are much more important sites for removal of nitrates than streams.

Based on data from the Environmental Monitoring Programme, it can be calculated that the total input of nitrogen to the Gudenå-system in 1989 was 5900 tonnes N of which only 2700 tonnes were transported to the sea at the downstream station. Thus the Gudenl-system with many lakes included removed approxinably 54% of the nitrogen input in 1989.

Denitrification

Denitrification is a heterotrophic respiration process in which organic carbon is oxidized with NO,. The process is thus parallel to the oxygen respiration process where oxygen is the electron acceptor:

Oxygen respiration: CH_2O + 0, → CO, + H_2O

Denitrification: CH_2O + NO, → CO, + H_2O + N_2

In the denitrification process, a plant nutrient (NO,) is thus transformed into the harmless nitrogen gas (N_2) simultaneously with the mineralization of organic carbon.

Denitrification seems only to be of ecological importance under anaerobic conditions (Christensen et al., 1989). Thus, three conditions must be obtained to make nitrogen removal by denitrification possible:

- availability of decomposable organic carbon
- availability of NO,
- anaerobic conditions.

In freshwater ecosystems, those demands are met in the sediment, which is the most important locality for the denitrification process. However, denitrification may also occur among the leaves of plants in the water column where anaerobic conditions occasionally may occur.

The nitrogen cycle in a sediment is schematically shown in Fig. 2. It is important to notice that the denitritication process may have two sources of nitrate: nitrate may diffuse into the sediment from the overlaying water column or it may be produced within the sediment due to the nitrification process in the aerobic surface layers. By use of microelectrodes (Revsbech et al., 1989), it has been shown that both the aerobic surface layer and the denitrification zone are limited to the upper few mm of the sediment. The penetration of 0, into the sediment is thus mostly limited to the upper mm and the denitrification zone, which is located immediately below the oxic surface layer, is under normal conditions only half a mm in extention in the depth (Christensen et al., 1989; Nielsen et al., 1990). Bioturbation or growth of macrophytes may thus greatly influence the nitrogen cycle by introducing 0, or NO, to deeper sediment layers.

Denitrification can be measured by different methods. A discussion of the methods and their applicability has been presented elsewhere (Christensen et al., 1991; Seitzinger et al., in press.).

Denitrification in streams

In Danish streams, a marked seasonal variation of the denitrification process has been demonstrated (Christensen and Sorensen, 1988, Christensen et al., 1990). During winter, the activity is almost absent. A maximum in activity (up to 1.4 mmol N $m^{-2} h^{-1}$) is found during spring (April-June) where benthic microalgae colonizing the sediment surface is an important carbon source and where the NO, concentration in the water column is still high (Christensen et al., 1990).

The acitivites given in Table 1 show the variations of nitrogen removal in different Danish streams. The activity is strictly dependent on the quality of the substrate. A stony or sandy sediment has a significant lower activity compared to a muddy sediment (Table 1). The yearly nitrogen removal varied from 5 to 700 kg N ha-' y^{-1} (Note that the values given for Gryde å and Suså are not yearly rates but only representing a mean value for the summer period: May-September). From the data in Table 1 and the general knowledge of Danish streams, it is possible to approximately estimate the maximum mean denitrification activity to 300 kg N ha-' y^{-1}.

The nitrogen removal by denitrification is thus lowest in the winter season where the nitrogen transport is at a maximum. As an annual mean, denitrification is only removing approximately 1–5% of the total nitrogen being transported in the streams. During the summer period, this percentage may be higher (30–40%; Jeppesen et al., 1987; Sorensen et al., 1990). The significance of rivers in removing nitrogen may be greater in bigger rivers. In River Scheldt, the nitrogen removal thus accounted for approximately 60% of the total nitrogen transport (Billen et al., 1986). Compared to Danish streams, this river is, however, tremendous large with a mean width of 260 m. The high nitrogen removal is caused by the large bottom areas and the fact that a major part of the river water is anoxic.

Denitrification in lakes

Several different factors will influence the ability of lakes to remove nitrogen by denitrification. Of importance is: the area and the depth of the lake, the retention time of the water and the location of the lake in the river system. Deeper lakes tend to show higher nitrogen removal compared to very shallow lakes. The ability to remove nitrogen also seems to decrease dramatically with low retention times of the water (below 6 month). Lakes located in the upper part of a river system tends to have a higher nitrogen removal potential compared to lakes located further downstream in the system, since the nitrogen input to the former group is mainly inorganic nitrogen compared to a major input of organic nitrogen to the latter group of lakes,

By use of a mass balance technique, Jensen et al. (1990) estimated denitrification in 58 shallow, Danish Lakes (Fig. 3). The lakes represent mainly first order lakes located in the upper part of the river system. As a mean, the lakes had an area of 330 ha, a depth of 4.1 m and a retention time of 1.2 year. Although some variation was found between the lakes, a remarkable constant percentage of the nitrogen input was removed in the lakes. Thus, the total nitrogen removal was 290 kg N ha^{-1} y^{-1} which corresponds to 43% of the total nitrogen loading. Denitrification accounted for 80% of the nitrogen removal and the denitrification process thus removed approximately 33% of the total nitrogen loading.

Denitrification in fjords

There are only a few good mass balances, which give an indication of the denitrification activity in the fjords. Mass balances of different Danish fjords indicate a nitrogen removal of 50-180 kg N ha^{-1} y^{-1} (Table 2).

Seitzinger (1988) has demonstrated that also in the fjords, a surprisingly constant percentage of the nitrogen loading is removed. Thus for 6 different American and European fjords, it was calculated that approximately 45% of the total nitrogen loading was removed.

In the understanding of the transformation and reduction of nitrogen in the different waters, we are in particular suffering from the lack of knowledge of the processes in fjords and coastal areas. Thus a useful method for measuring denitrification in those areas is still missing and more studies of the controlling factors are requested.

Nitrogen removal in relation to nitrogen transport

A value of 300 kg N ha^{-1} y^{-1} may be considered as the best estimate for a maximum of the mean area based nitrogen removal in streams and lakes. In Denmark the total area of streams and lakes are approximately 8.000 and 53.000 ha, respectively. Thus, on a yearly scale, the freshwater system removes some 2.400 + 16.000 tonnes nitrogen. A removal of 18.400 tonnes N y^{-1} is significant, compared to the cost of nitrogen removal in waste water treatment plants. However, compared to the total amount of 130.000 tonnes nitrogen which is transported to the sea through Danish streams during an average year (Kristensen et al., 1990), the nitrogen removal in the freshwater system is of minor importance since less than 15% of the nitrogen is removed before it is delivered to the sea.

In general, the Danish streams are simply too small and too short to have any influence on the transport of nitrogen to coastal areas. The lakes are responsible for the major part of the nitrogen removal. In this light, it may be considered to establish new lakes or wet lands in the river system in order to enhance the total nitrogen removal. The ecological effects of the entire river system should, however, be considered very carefully before establishing new lakes.

Enhancement of denitrification

Denitrification greatly seems to be stimulated by the occurence of aquatic macrophytes. In the stream Rabis Bæk, denitritication activity was significantly higher in a *Batrachium*– colonized sediment compared to a non-vegetated sediment (Fig. 4). Maximum activities in the vegetated sediment was found in the early summer just before the plants were cut and removed in order to enhance water flow in the stream (Christensen and Sorensen, 1988). In streams where growth of macrophytes is maintained throughout the growth season, higher acitivites may probably be obtained later in the summer.

Also in lakes, vegetated sediments have been shown to have a high denitrification potential (Christensen and Sorensen, 1986) In a sediment covered with *Littorella uniflora*, the rate of denitrification showed the same seasonal behavior as the biomass of the macrophytes and activity was found throughout the root zone (Fig. 5). In fact more than 70% of the total activity was found in the sediment depth between 1 and 10 cm. The deep located denitrification activity can only be explained by a close coupling of nitrification and denitrification in the rhizosphere of the plant, where oxygen release from the roots can support nitrification activity in the entire root zone.

Bioturbation of the sediment by aminals may also provide a route of oxygen or nitrate transport to deeper sediment layers by which denitrification can be stimulated significantly (Grundmanis and Murray, 1977; Chatarpaul et al., 1980).

Denitrification may similarly be stimulated within the plant canopy in the water column, where accumulation of organics, low oxygen concentrations and a high supply of nitrate may provide optimal conditions for denitrification. In the densely vegetated Suså, a major part of the NO, removal was thus attributed to activity at the leaf surfaces (Jeppesen et al., 1987). Similar canopy associated denitrification activity has been indicated from results obtained in the minor Danish Lake Væng. Here a removal of the phosphorus loading followed by a manipulation of the fish population provided improved light conditions at the sediment surface. As a result, a dense population of *Elodea* colonized the entire sediment surface within two years. Parallel to plant colonization, the nitrogen retention of the lake increased by a factor of two (Jeppesen et al., 199 1). The increased nitrogen retention could not be explained by nitrogen uptake by the plants but must be ascribed to denitrification within the plant canopy in which the oxygen concentration may show pronounced diel and seasonal variation.

To our knowledge there have been no reports on a stimulated denitrification in vegetated marine areas. It seems, however, reasonable to believe that macrophytes may enhance the nitrogen removal in those areas. A reduction in nitrogen loading will greatly improve the light conditions of shallow coastal waters and thereby increase the water depth to which macrophytes can penetrate (Borum et al., 1990).

Therefore, it is very likely that macrophytes can enhance nitrogen removal in both freshwater and marine systems. A stimulation of the denitrification in the different localities may be achieved by a less active removal of macrophytes in streams, a reduction in phosphorus loading of lakes and nitrogen loading of shallow coastal waters which in term may stimulate growth of macrophytes.

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Fig. 1:
Spatial variation of nitrate concentrations in Gudenå river system, Denmark. Lakes appear as boxes. Mean nitrate-N concentrations, mg N l^{-1} (upper cipher) of all available data (number in paranthesis) are shown for several locations in the main river and for major tributaries. The catchment size of individual tributaries is indicated below the name of the stream.

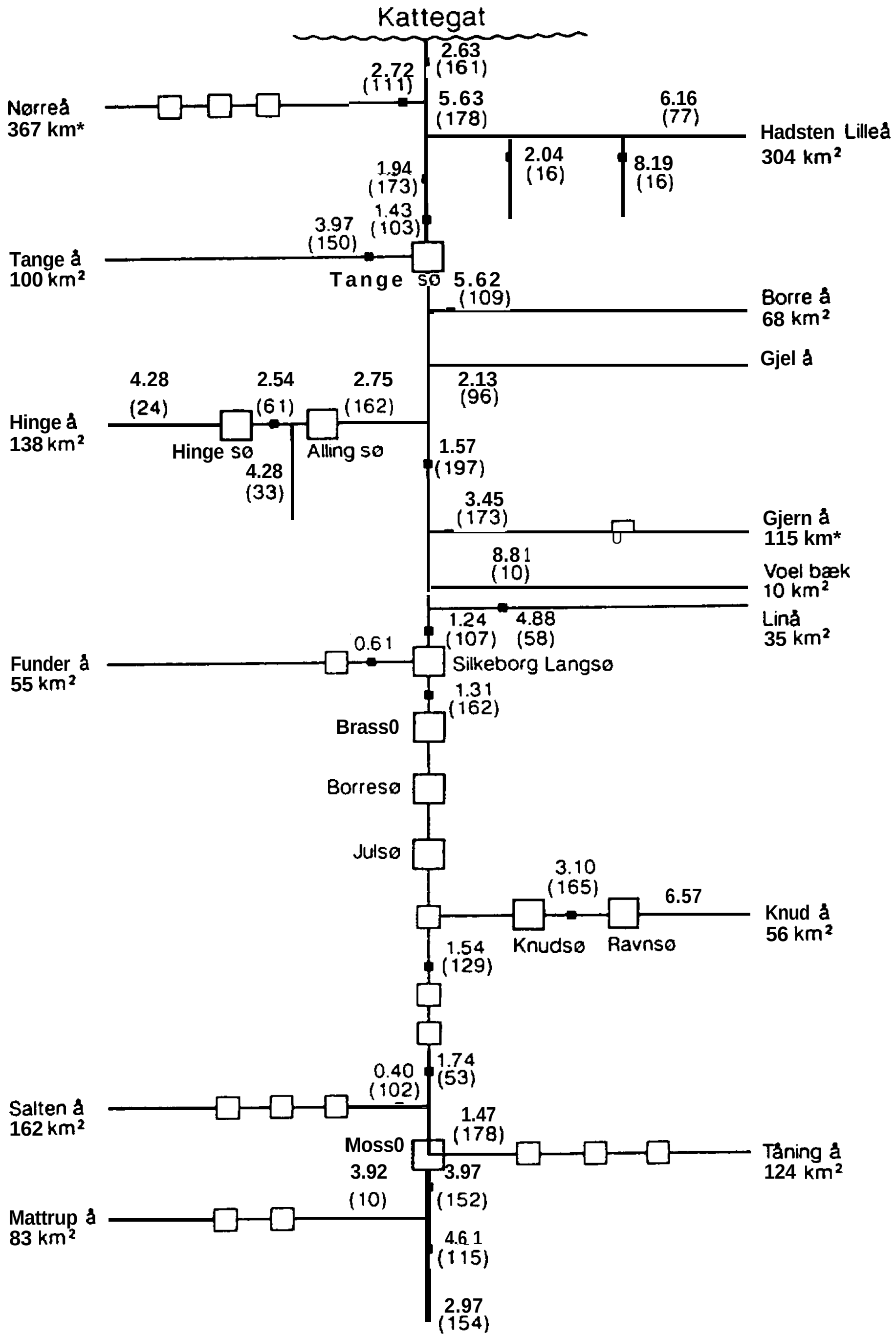

Fig. 2:
The nitrogen cycle at a sediment-water interphase.

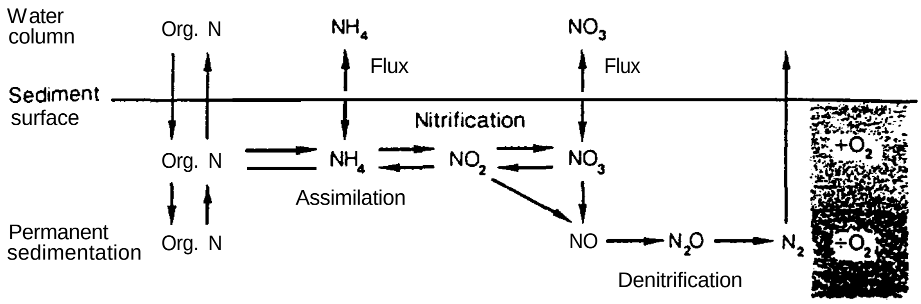

Fig. 3:
Denitrification related to nitrogen loading in 58 Danish lakes. Data from J.P. Jensen et al., 1990.

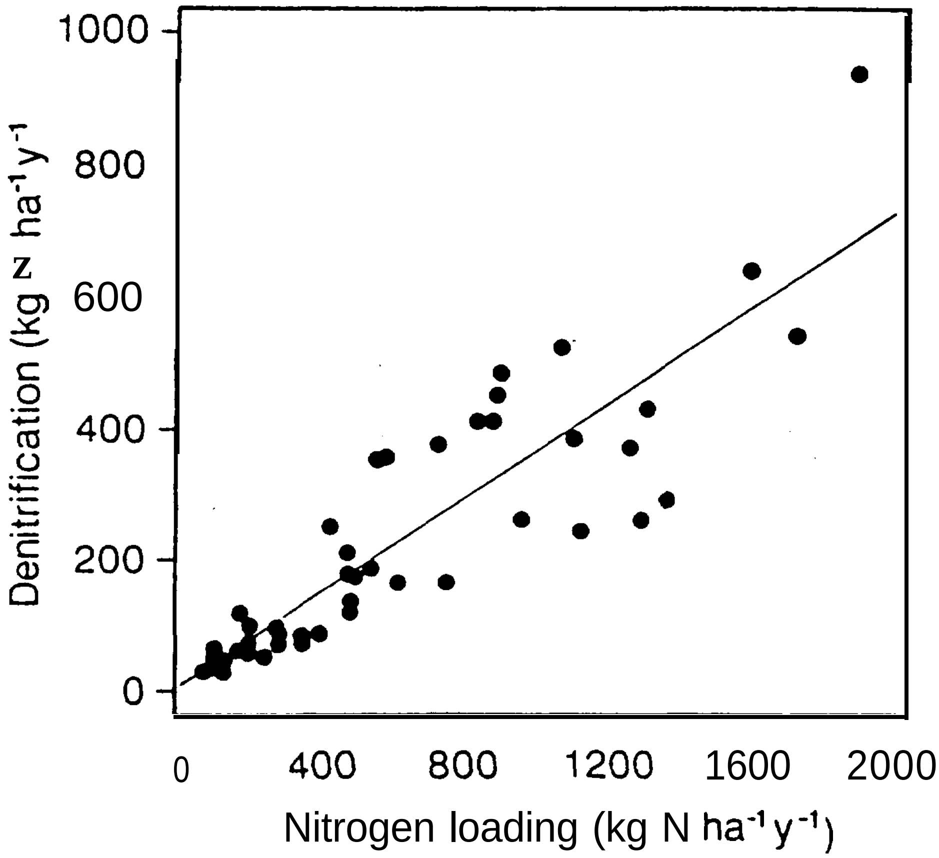

Fig. 4:
Seasonal variation** in denitrification **in Rabis bæk.
A): Activity measured in a** Batrachium – **vegetation sediment.
B): Activity measured in bare sediment. The arrow indicates the removal of the macrophytes. Data from Christensen and Sørensen, 1988.

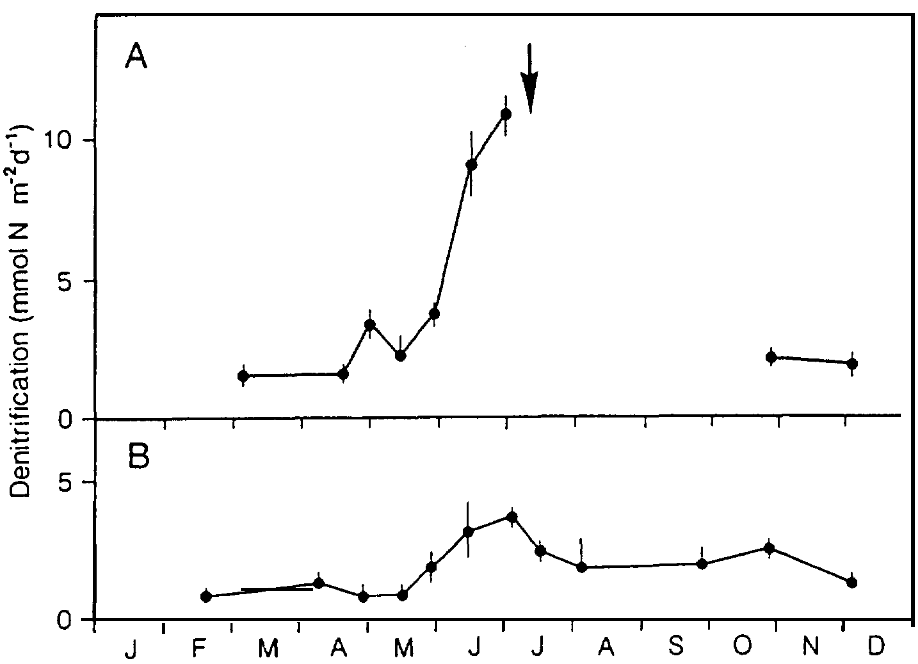

Fig. 5:
Seasonal variation of denitrification in a Littorella uniflora vegetated lake sediment. Data from Christensen and Sorensen, 1986.

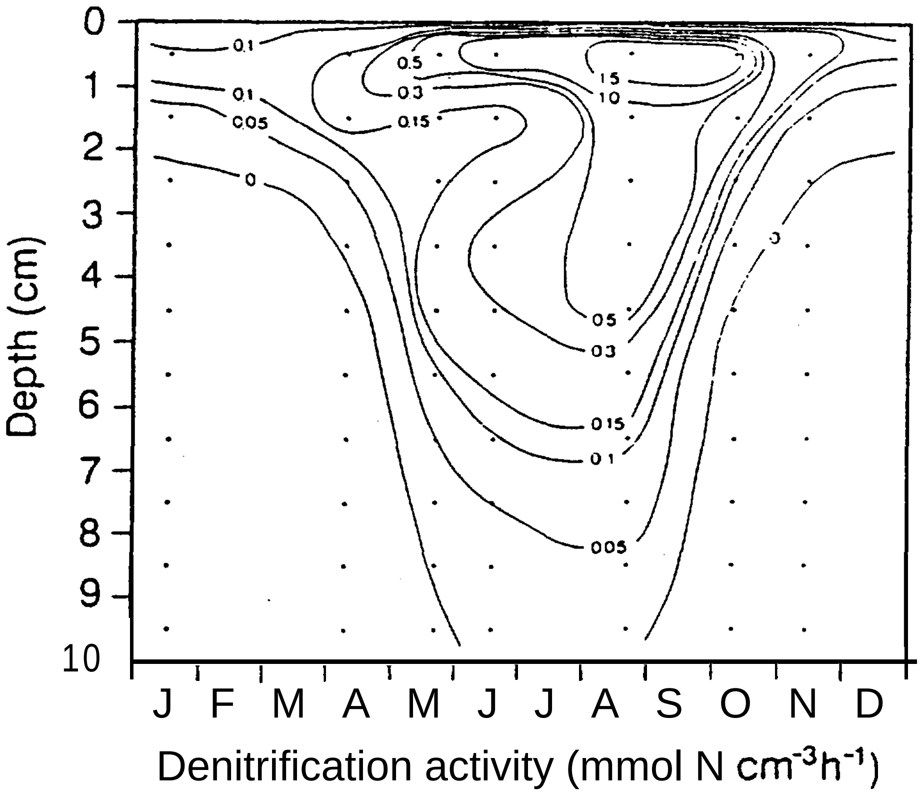

Table 1:
Nitrogen removal in danish streams. Data *from* : ***Christensen and Sorensen, 1988 (1), Sorensen et al., 1988 (2), Christensen et al., 1990 (3), Nielsen and Christensen, upubl. (4), Jeppesen et al., 1987 (5).***

Lokality	Substrate	N-removal (Kg N ha-' år-1)	Method	Ref.
Rabis Bæk	Sandy	100	C_2H_2-inhib.	1
"	Sandy with vegetation	*250*	"	1
"	stony	*5*	"	1
Gelbak	Stony	*20*	"	2
"	Muddy	*250*	"	1
"	Muddy	*700*	"	3
Dalby Bak	Sandy	*50*	"	4
Gryde Å	Sandy with vegetation	540*	NO,-uptake	5
Suså	Muddy with vegetation	1460*	"	5

*Note that the rate given is not a yearly rate but the nitrogen removal determined for the period: May-September.

Table 2:
Nitrogen removal in Danish fiords. Data from:
Finn Larsen, Viborg amt (1), Andersen et al., 1990 (2), Kamp-Nielsen, submitted (3), Carsten Jürgensen, Fyns amt (4), Jorgensen and Sørensen 1988 (5).

	N-removal	*Method* *(kgN ha^{-1} y^{-1})*	*Ref.*
Limfjorden	50	**Massba.**	1
Mariager Fjord	**160**	"	**2**
Roskilde Fjord	**175**	"	**3**
Odense Fjord	**180**	"	**4**
Norsminde Fjord	**200-450**	C_2H_2	**5**

Slurry handling in Schleswig-Holstein
- recommendations and regulations -

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In Schleswig-Holstein the so called »Gülleverordnung« – i.e. a special decree for handling slurry has been in operation since 1. August 1989. This decree was completed based on administration instructions from October 1989 for the local administrative authority.

Administration authority for controlling the slurry decree are the Ämter für Land- und Wasserwirtschaft, subordinated authorities of the ministry of agriculture. There are six such authorities in Schleswig-Holstein.

In this paper the most important regulations of the slurry decree are explained as well as governmental measures in Schleswig-Holstein with the aim to reduce problems resulting from the use of slurry. Practical experiences will also be mentioned.

Significant regulations of the slurry decree

- The slurry decree is in force for
  - all kinds of slurry and
  - poultry dung

  wich are used for fertilizing agricultural and horticultural soils.
- It is also in force for slurry which is prepared in special treatment plants. Problems in handling products made of slurry, as compost for example might occur, when these products are used out of farms. But no treatment plants are working there.
- The slurry decree is not in force for urine and dung. This is a great disadvantage, because there is no big difference between these substances and slurry when used for fertilizing or when contaminating water bodies. This was a political decision to reduce problems especially for smaller farms. In fact this is neither a helpful solution for agriculture nor for the environment.
- The unit of the slurry decree is the dung unit. The dung unit is defined as the amount of slurry being produced by a certain number of livestock during one year, which contains either 80 kg nitrogen or 60 kg phosphate. Livestock producing dung and urine only must not be considered when summing up the dung units.
- Differences to the numbers shown in the table are permitted, when

  1) special measures on the farm result in reduced nutrient contents of the slurry, as for instance by using protein- or phosphate-reduced feeding-stuffs, or when

  2) slurry or part of the slurry nutrients go out of the farm, for example by contracts with other farmers or by using special treatment plants.

These differences need a special permission from the responsible authority. Special feeding stuffs, reduced nutrients in the slurry and the amount of slurry taken out of the farm must be substantiated.

- Slurry has to be used in the way of good agricultural practice. This means that fertilization of crops has to consider
  - the required nutrients of the plants in quantity and time,
  - the available nutrients in the soil and
  - special circumstances of location.
- Slurry application in protected areas has to consider the specific regulations of these areas.
- Slurry has to be incorporated on not cultivated soils immediately after spreading, at least, however, the day after spreading.

  This is necessary to reduce ammonia volatilization as far as possible under practical given facts. More than this, however, it is the only way of minimizing the smell of slurry, too.

  Incorporation is possible only on not cultivated soils. So it only works before cultivation in spring and after harvest of crops in summer and autumn. Best efficiency of fertilization and most effective groundwater protection are achieved when slurry is applied short before or in the vegetation period. Application in autumn before cultivation of winter crops, only allows smaller quantities of slurry than in spring, because winter crops only need smaller amounts of nutrients till the end of the vegetation period.

  Therefore and because of the high part of winter crops in rotations most of the applied slurry can not be incorporated in Schleswig-Holstein.
- All over the country, no more than 2 dung units per hectar and year are allowed. Soils in water protection areas, strips along water bodies and set-aside areas must not be spread with slurry.

  Varying contents of nutrients in the slurry enable different quantities of slurry per hectar. The only limitation of slurry application are the 2 ***dung units*** per hectar and not the quantity of slurry.

  These 2 dung units were discussed heavily. In fact different dung units depending on different kinds of soils would be the better agricultural practice. 2 dung units may not be enough on heavy soils and could be too much on very light soils. But administration and controlling would nearly be impossible when different dung units were allowed. So practicability was the reason for not splitting the allowable units.
- When poultry slurry or poultry dung is used the allowed dung units must be reduced dependent on the content of phosphate in the soil. On soils with high phosphate contents only one dung unit is allowed, on soils with very high phosphate contents no poultry dung or poultry slurry are allowed.
- All farmers have to make notes about their livestock, the available acrage and about quantity, date and field of slurry application. These notes are controlled by the local authority in a rate of 10 till 20% of all farms per year.
- Normally slurry application is allowed from 1. March till 30. September. On grassland and on cultivated soils slurry application is allowed from 1. February till 15. October.

  Fixed dates are not the best way to regulate fertilization because of different climate conditions. But it is the only way to control slurry application effectively. The sooner in the year slurry is applied the lower are temperature, ammonia volatilization and smell-problems. On frozen soils damaging of soil-structure is prevented.

On the other hand, too early application on not cultivated light soils increases leaching. Application on deeply frozen soils means a higher risk of erosion especially in combination with heavy rain and inclined areas. Different application times for cultivated and not cultivated soils are justified, for cultivated soils need nutrients all over the vegetation period. The risk of too much nutrients in autumn and in early spring is small, because overfertilization is harmful to plants. So there is a kind of self-control system in the application time.

Normally in Schleswig-Holstein there is no vegetation during winter time, so there is no need of application. Uncultivated soils need nutrients shortly before cultivation and not slurry all over the winter as some farmers some times did before the slurry decree came in force. So the regulations for application times are necessary mainly under administration aspects but under aspects of experts, too. They are a bearable compromise.

- Beside these application times the slurry decree has special application prohibitions
  - to prevent water pollution by erosion and direct input,
  - to prevent ground- and drinking-water contamination by leaching,
  - to prevent ammonia volatilization and
  - to prevent input of other nutrients into ecosystems.

  So it is not allowed to apply slurry
  - to fallow land, because there is no need of nutrients
  - to forests, because of special fertilization problems and damages by ammonia volatilization,
  - to deeply frozen soils, because of the early mentioned problems,
  - to protected ecosystems, because of the not desired nutrient input and
  - to 5 meter wide strips along waters, because of the risk of direct input by spreading.

Altogether, the slurry decree is said to be an instrument which environment and most of the farmers are able to live with it.

Transitional regulations were in force till 1. Januar 199 1. The dung units have been reduced from 3 to 2 since then. Further exeptions are possible but they are handled as real exeptions.

Violations of the slurry decree effect fines from 100 till 1 .500 DM and more when violations are repeated. As mentioned before the slurry decree is a compromise which works quite well after more than one year experience. Improvements are possible and necessary, but this is not only a question to experts, but more or less a question to politicians too.

Governmental measures to reduce slurry problems in Schleswig-Holstein

Before the slurry decree came into operation investigations were made about the number of dung units in Schleswig-Holstein. The result were 1,3 dung units per hectar agricultural area on the average of Schleswig-Holstein. In this number all livestock are included, even those which do not produce slurry but dung and urine. A correct determination of animal housing systems was not possible.

There are only 2 regions in the southwest and in the northwest of Schleswig-Holstein were the average is little more than 2 dung units per hectar. In these mainly grassland regions and in some other smaller regions, too, problems were expected.

Therefore the government supported the installation of slurry agencies with nearly 900.000 DM. These slurry agencies are connected to the agricultural machinery agencies all over the country. They shall bring together farmers with too much slurry with others who need slurry. For this service the farmers have to pay a special amount of money per unit. It is likely, that only one or two of them will survive in a long run, because farmers have learnt to manage their slurry problems themselves.

Another governmental measure to reduce slurry problems was the financial support of 2 mobile slurry treatment plants with nearly 1.200.000 DM.

The most effective governmental measure was the so called agricultural environmental support. From 1974 till the end of 1990, 185 million DM were payed as subsidies for storage capacities.

Altogether 16.800 farmers got these subsidies.

9 million m^3 storage capacities for slurry and urine and 650.000 m^2 storage capacities for dung were built during this time. This means an average of more than 7 months storage capacity in Schleswig-Holstein. This has been the most effective help for making slurry problems smaller in Schleswig-Holstein.

The Danish action plan for optimal use of animal manure

by Senior Adviser Carl Age Pedersen
The Danish Agricultural Advisory Centre
Skejby – Denmark

Why an action plan?

New framework

From 1993 the Danish farmers must have established 9 months storage capacity for animal manure.

Today the farmers have already established storage capacity for more than 7 months production.

With an application period of 3 months (e.g. Marts, April and May) 9 months capacity enables the farmer to apply all the animal manure in the spring.

Demand for reducing nitrogen losses

A high utilization of nitrogen in animal manure ensure at the same time small losses of nitrogen to the surroundings.

The utilization pct of nitrogen in animal manure is defined as the amount of chemical fertilizer-N which can be replaced by 100 kg of total-N in animal manure.

Combination of environmental and agricultural goals

The farmers save money and less nitrogen is lost to the environment when the animal manure is well utilized.

The action plan was established by the advisory system in co-operation with the environmental authorities, the State Plant Science and Research Centre and the Agricultural Mashinery Association.

The goals

The main goal is to increase the first year utilization of animal manure from approximately 20 pct to approximately 40 pct of the total nitrogen content of the animal manure in Denmark.

The 40 pct of the total-N correspounds to 70 pct of the ammonia-N content in the manure or appr. 70 pct of the theoretical maximum utilization efficiency.

The maximum first year effect equals the ammonia-N content of the manure, and if the farmer is going to reach this high utilization, there must not be any losses of ammonia during and after the spreading.

An increase of 20 pct units in utilization equals 50.000 ton fertilizer-N per year.

How to do it?

Where is the ammonia-N?

The ammonia and nitrate losses depend on application time, rate - and - method, and are mostly related to the ammonia content of the manure.

Where is the ammonia then.

In Denmark

60 pct of the animal manure is slurry,
15 pct is urine and
25 pct is farmyard manure,

which means that

85–90 pct of the total ammonia is in the liquid manure slurry and urine.

Therefore most of the activities of the action plan have been related to slurry and urine.

Main problems

The main problems in relation to utilization of liquid manure are:

Application at inappropriate times
In Denmark one third of the liquid manure is applied during autumn.

Winter oil seed rape and some grasses need nitrogen application in the autumn. It is acceptable, that a minor part is brought out in the autumn, 10 pct or so.

Uncertainty
Most of the farmers do not know exactly what is the nitrogen content of the manure.

Therefore it is necessary to use a quick-test for determination of the ammonia content.

In addition, the spreading equipment is generally unable to eliminate the ammonia emission. As a consequence, the farmers don't know exactly how much nitrogen is left in the soil for fertilization.

Spreading equipment are common in practise, which:
- Distributes with variation-coefficients of 35 to 50 pct.,
- is unable to incorporate the slurry in the soil,

- is unable to place the slurry in bands on the soil between the growing plants in the spring,

Tools

Two years ago activities were launced to improve the farmers behaviour with respect to the mentioned three points by means of:

- field trials all over the country in co-operation with the local advisers. In 2 years we have made appr. 100 trials.
- writing articles in the agricultural papers and others, including spots in the TV.
- educating farmers (and advisers) at meetings and seminars.
- Alternative measures to bring about focus on better use of animal manure. Such as the special Stand at Agromek in Januar 199 1.

Results

It is a long process to change the behaviour of all the farmers and until now we have not noticed considerable reductions in the use of chemical fertilizerse. (When better used the animal manure will replace chemical fertilizers). When corrected for the crop distribution, the consumption of commersial fertilizers can be used to measure the effect of our efforts). However, new spreading equipment such as units with drag hoses is sold more frequently at present and it is expected that more manure is applied during springtime. We also know that there has been sold many instruments for quick-testing the ammonia content of slurry.

In the next 3 years a number of new demonstrations and field trials will be implemented. We have made agreements with 10 farmers, whose farms will function as demonstration farms. Most of the field trials will be located at these farms, and the economical and ecological consequences will be examined.

It is expected, that these measures will result in a significant reduction of losses.

Integrated plant production as a method for optimizing plant. production and nutrition Cycle – 5 years practical experiences

Dr. Cramer, Landwirtschafiskammer Schleswig-Holstein, Kiel

Agriculture in Schleswig-Holstein is characterized by high yields and high input systems.

Yields and N-supply in Schleswig-Holstein

	t/ha
Winter wheat	7,5–8
Winter barley	6,5–8,3
Winter rape	3,3–3,8

N-supply:	kg N/ha
from mineral fertilizer	175
from slurry, 50% effective	27
together	202

Using a high nitrogen input system for more than 20 years has increased the amount of N in the humus of the soils by about 2000 kg/ha, causing a high nitrogen release under favourable moisture and temperature conditions. The weather condition, even under the »moderate« climate of Schleswig-Holstein, is in the critical spring season so variable, that it is hard for practical agriculture and its advisory service to calculate the actual amount of nitrogen released from the soil and thus made available for plant nutrition.

Financially supported by the Ministry of Foods, Agriculture, Forestry and Fishery of Schleswig-Holstein, the Landwirtschaftskammer Schleswig-Holstein compared for 5 years a very high input system with the methods of »Integrated Plant Production«, each with about 40 ha on the farm Rosenkrantz.

Average Input of Fertilizer and Plant Protection, Rosenkrantz 85/86-89/90

System	Fertilizer DM/ha	rel.	Chem. Plant Protection DM/ha	rel.
Intensive	338	100	390	100
Integrated	257	76	236	61

Under the aspects of optimizing the nitrogen uptake by the plants, minimizing leaching and other N-losses, the main results are:

1. The economic optimum yield under 199 1 price-cost relations lies about 5–10% below the maximum yield.
2. Compared to the high input system it was possible to reduce the amount of N fertilized by about 20%.

3. A 5 years Nitrogen-Balance for the two different systems including the grown legumes shows a reduction of N-surplus by about 50%.

Fertilization and Uptake
by the Plants, Rosenkrantz **85/86-89/90** (in kg N/ha **/** year)

rotation	intensive 3 fields	integated 5 fields
fertilization	260	158
uptake by the plants	162	146
N collection legumes		36
balance/surplus	**+98**	**+47**

4. Practical agriculture in Schleswig-Holstein has already realized about half of the possible reduction in N-fertilization. It is a challenge for the farmers and their advisory service to realize even more, thus serving economy and ecology.

The nitrogen balance in an area of intensive farming

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Introduction

This paper aims to illustrate the nitrogen balance in an area of intensive farming:

- investigations were carried out between 1986 and 1989. The area was divided into 4 farms: 1 used 8,360 ha for crop production and the other used 20,630 ha mostly as grazing for 20,630 cattle (2,47 cattle / ha),
- Losten farm held a stock of between 80 and 80,000 pigs and 11,280 cattle on one site. The farm had at its disposal approx. 1,600 ha of land fitted with a sprinkler irrigation system to make use of liquid waste,
- average crop production between 1986 and 1989 amounted to 50,5 dt GE / ha LN.

Explanations of the nitrogen budget in the area are considered from two aspects:

- the nitrogen budget for animal husbandry considers the pig farm in particular because of its high concentration of nitrogen and the changes in its budget as a result of treating the manure. This includes the restructuring of nitrogen caused by the treatment process 'and the reduction in nitrogen due to ammonia emission,
- when illustrating the nitrogen budget for the crop-producing farm, the main focus is on the total budget of available nitrogen in the production process. The share of nitrogen from animal husbandry is also investigated and an estimate made of nitrogen input into the aquatic environment.

The nitrogen budget for the pig farm is based on long-term investigations carried out under production conditions by a research group from the University of Rostock.

Besides using our own results to calculate the nitrogen budget, figures from government offices relating to the use of mineral fertilizers and normative values for nitrogen from animal husbandry were also used. Point pollution sources are not considered, however. Statements should therefore be regarded as a much simplified illustration of the real process.

Manure treatment and the nitrogen balance in animal husbandry

Manure of Losten farm (high concentration of livestock, limited disposal area) must be treated. The treatment process is designed to dissipate part of the nitrogen in the manure and to fix the nutrients nitrogen and phosphorus.

This is done to make the use of mechanical processes possible when segregating the nutrients from the fluid phase. At the same time, a concentration of nitrogen and phosphorus is deduced in the solids from the separation process to justify their transport over large distances from an economical viewpoint.

The treatment process used is shown in Fig. I. It combines both mechanical and biological processes.

The manure is removed from the pigsties (1) and stored for up to 5 days (2). After being homogenized, the manure is fed into a separating apparatus (3) where the solids are separated using sieves. The solids are then transported to a storage site (4) and the slurry is fed into a tank (5). The liquid that drains from the solids at the storage site is also fed into the tank. From here the slurry is pumped into a biological treatment plant (6) where under aerobic conditions the dissolved organic substances (BSB,) and CBS are biodegraded up to 98 and 85% respectively. At the same time, ammonia nitrogen is partially nitrified and converted into nitrite and nitrate. This part of the treatment process for slurry lasts up to 12 days.

The treated slurry and surplus sludge are collected in another tank (7) where settling causes separation. Here the sludge develops a dried matter content of approx. 3%.

The treated slurry is now fed into a reservoir (9) large enough to hold over a year's slurry.

Using decanters (8), the surplus sludge is again treated mechanically so that it can later be transported in bulk. The resulting solids are transported to the dump and treated further if required. The slurry obtained in the process is fed into the reservoir (9) and stored until needed.

Nitrogen's material flow in this treatment process is shown in Fig. 2. The nitrogen present in newly dropped manure in the pigsties consists almost equally of ammonia nitrogen and organically fixed nitrogen. This ratio can change quickly depending on how manure is disposed of, its amount of dry solid matter, storage time, and especially the transmutation of urea to ammonia nitrogen. This causes the share of ammonia nitrogen in total nitrogen to increase to 70%. Transmutation facilitates the emission of large amounts of nitrogen in the pigsties.

Investigations by Schätzchen (1990) showed that great differences in nitrogen emissions exist depending on how the manure was disposed of: there was a reduction of a $^1/_3$ in emissions when manure was impounded in channels compared with flow channels. An important reason for this is the greater length of time which the manure spends in impound channels. The manure in the impound channel was found to have a reduced pH value caused, probably, by anaerobic processes. Conditions for ammonia emission were thus less favourable.

When the manure is sieved (3), only about 10% of the nitrogen in the manure is transferred to the dry matter when the separating efficiency for solid matter is 40%.

This is different when the slurry is treated biologically (6). As a result of the nitrification of the ammonia nitrogen and the following denitrification of the oxidized nitrogen, over 30% of the original nitrogen amount is emitted, most of it in the form of harmless molecular nitrogen. The amount of molecular nitrogen emitted was shown in the analytical results for oxidized nitrogen. During the treatment process in the same aeration tank, denitrification as well as nitrification occurs. The difference between the reduction in nitrogen as a whole and measured oxidizer nitrogen should thus be viewed as the upper limit value for ammonia emission at this stage in the treatment process. It is probable that a large amount of this nitrogen was also emitted in the form of molecular nitrogen.

An important result of treatment at this stage is also the transmutation of part of the ammonia nitrogen into an organic compound. This part of the nitrogen becomes a component of the biomass and thus part of the solid phase and can again be separated mechanically.

When the surplus sludge is separated mechanically using decanters (8), almost half the nitrogen still present in the treated manure after biological treatment can be separated with the solid phase. The net result is that only 43% of the nitrogen amount in the manure dropped in the pigsties remains in the manure after treatment.

Based on the results above, an attempt is made to show the nitrogen in animal waste which is used on the crop-producing land (Table 1). It should be stated that the large poultry stock held in addition to the cattle is not regarded as a source of nitrogen as its waste was not spread on the fields during the period under investigation.

Of the nitrogen present in the animal waste, only 55,5% is used in fluid or solid form in crop production (869 t/a). The pig farm accounts for only half of this because of the biological treatment of the slurry. The amount of nitrogen emitted mainly as ammonia as a result of decay is difficult to estimate. It was assumed when drawing up the nitrogen budget that the amount of nitrogen emitted in this area is effective. This means that on an area of 11,000 ha a nitrogen amount of approx. 43 kg/ha is emitted to the air.

Estimating the nitrogen budget in crop production

If the nitrogen amounts given above are applied to the crop-producing fields, this results in a nitrogen input from animal husbandry of 147 kg/ha·yr. Included in this is the nitrogen present in newly dropped manure and treated material. Nitrogen input from precipitation was also considered as an additional factor. If this amount is compared with the 160 kg/ha·yr given in the new Liquid Manure Ordinance in Schleswig-Holstein, it can be seen that the limit value would not be exceeded if the nutrients were spread across the whole of the crop-producing area. The total budget for nitrogen in the area is shown in Table 2.

With an average crop yield of 50,5 dt GE/ha, nitrogen uptake by harvest products including natural nitrogen loss amounts to 188 kg/ha·yr. The budget shows, however, that if input of nitrogen amounts to a total of 336 kg/ha·yr, nitrogen supply is excessive. Only about 55% of the nitrogen input is required for crop production. What is interesting is that the share of nitrogen from animal husbandry only totals 43,8% of nitrogen input. It would be possible to meet almost 80% of the 188 kg/ha·yr needed for crop-production with this amount.

The nitrogen surplus revealed in this budget poses a considerable threat to the environment, especially the aquatic one, but there are differences in the type of threat. Nitrogen leaching should be expected from mobile nitrogen compounds, in particular nitrate. For the purposes of this budget, ammonia nitrogen is also regarded as a mobile nitrogen compound, although slight differences exist between the two. Oxydized nitrogen is much more liable to leach as, unlike ammonia nitrogen, it cannot be fixed by minerals in the soil. It must be stated that any kind of nitrogen surplus is a source of threat to the water environment regardless of its initial compound type. Table 3 shows the initial potential threat posed by mobile nitrogen compounds. It shows that through organic fertilizers, **40,9%** of environmentally dangerous nitrogen components reaches the fields i.e. the lesser part. The pig farm's share of these compounds amounts to only 20%. This highlights the great threat which mineral fertilizers pose to the water environment, a fact that should certainly be borne in mind when considering the dangers posed by crop production and especially by waste from animal husbandry.

Finally some information confirming the partly calculated values for the area's nitrogen budget. Random sampling of outlet channels leading from the treated area showed monthly nitrogen input in the form of nitrate to amount between **6,8** kg/ha and **53,5** kg/ha and, as shown in the tables, to **1,4** kg/ha and **12,3** kg/ha for pure nitrogen. These are values similar to those measured in Schleswig-Holstein.

Summary

The nitrogen budget of a large-scale pig farm which treats its manure mechanically and biologically was the basis of calculations of the nitrogen budget of the area where the farm is situated. All nitrogen compounds used in the area were considered and an estimate made of the threat to the water environment posed by nitrogen processes from the various branches of agriculture on Losten farm.

Figure 1: The various stages of treatment for pig manure

Stages of treatment	Description	Material being treated	Aim of treatment
1	Pig farm	Pig manure	Pig production
2	Storage tank	Pig manure	Storage and homogenization of the manure
3	Separating apparatus	Pig manure	Sieving of manure
4	Storage site for solids	Solids	Storage and drainage of solids
5	Storage tank	Liquid manure	Storage of liquid manure
6	Biolocigal treatment plant	Liquid manure	Biological treatment of liquid manure
7	Sludge collector	Treated liquid manure and surplus sludge	Settling causes separation
8	Separating apparatus	Surplus sludge	Decanters used to separate sludge further
9	Reservoir	Treated liquid manure	Long-term storage of treated liquid manure

Figure 2:
Nitrogen's material flow at the various stages in the treatment process

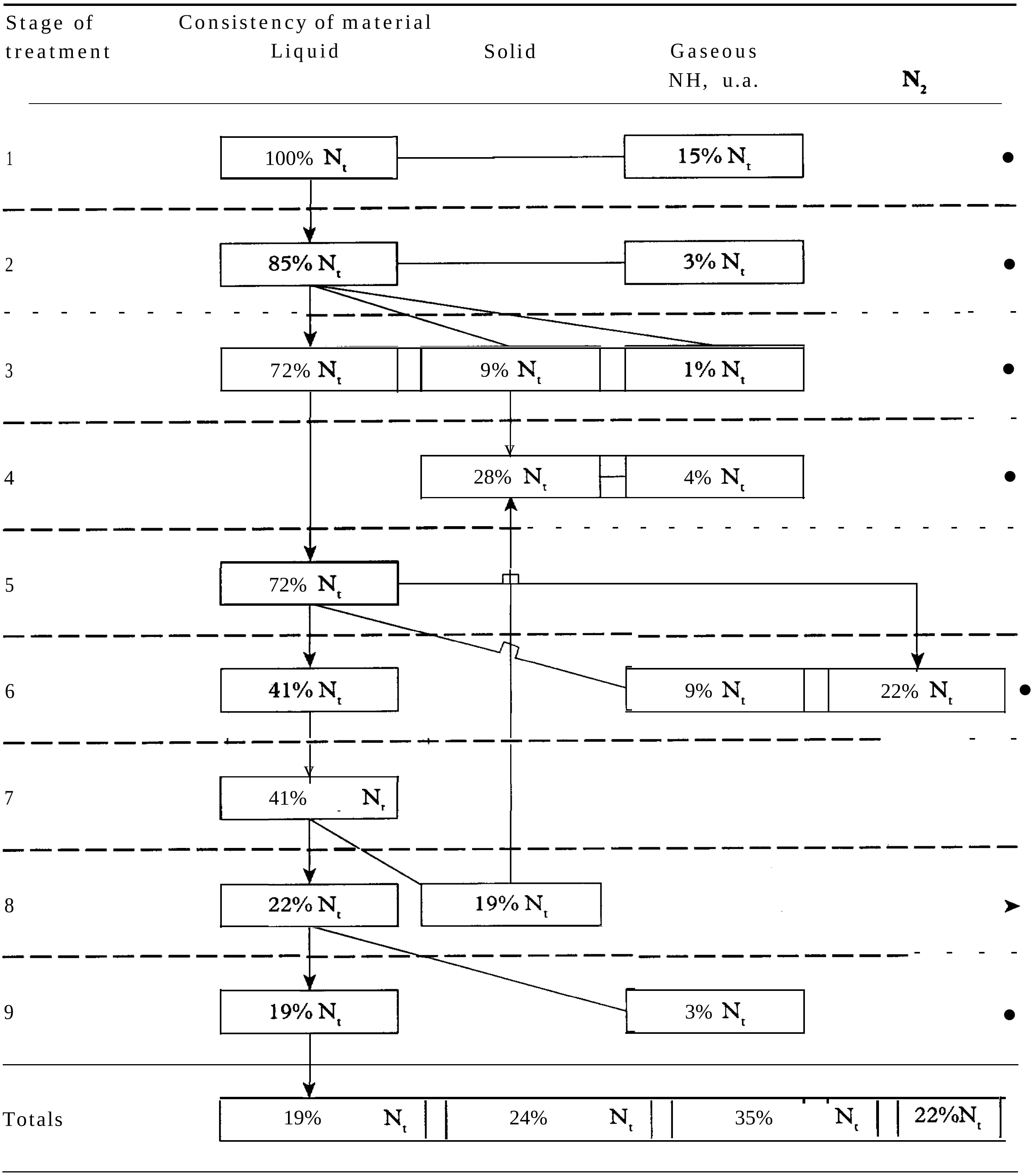


Table 1:
Nitrogen in animal waste around crop-producing farm in Mecklenburg village

	Amount of N		
	in total t/yr	on Lotser farm t/yr	Lotsen farm's percentage of total N
Total N (excluding poultry)	1566	1010	64,5
N in manure	1284	953	
N in farmyard manure	231	48	
N in black liquid	51	9	
Amount of N eliminated by biological treatment	220	220	
N losses through emission and digestion	477	354	
Amount of N available for use in crop production in solid and liquid form	869	436	50

Table 2:
Nitrogen balance calculated for crop-producing farm in Mecklenburg village

	Amounts of N		
	kg/ha·yr	t/yr	
1. Uptake – by crops	128	1065	
– by fixation, leaching etc.	60	499	
Total 100% requirement	188	1564	
2. Input precipitation	43	363	
– organic fertilizers	104	869	
– by bacterial absorbing N	38	318	
– seeds and plant material	3	25	
– mineral fertilizers	148	1240	
Total input	336	2815	
N surplus	148	1251	

Table 3:
Potential threat posed by mobile nitrogencompounds.

N input	*total N* t/yr	*of which mobile N* 5/yr	*percentage of mobile N* 0/0
Total input	**2815**	**2120**	**100**
of which:			
– precipitation	**363**	**363**	17,1
– organic fertilizers	**869**	**504**	23,8
– N-absorbing bacteria	**318**		
– seeds and plant material	**25**	**13**	0,6
– mineral fertilizers	**1240**	**1240**	58,5

The Danish programme for joint biogas plants

The Coordinations Committee for Joint Biogas Plants
The Biogas Secretariat

Background

The Action Programme for Joint Biogas Plants in Denmark was drawn up in 1986 by the ministries of Agriculture, Energy and the Environment with the purpose to clarify whether, through a technological development of joint biogas plants, financially competitive plants could be developed.

The Action Programme was initiated at the beginning of 1988 and is presently being concluded, with a final report due to be published May 199 1. A committee – the Coordination Committee of Joint Biogas Plants – was appointed as responsible body for the implementation of the programme. The committee is composed of representatives of central and local government authorities, supply organizations, agricultural associations, the biogas industry, research institutions and others, so that all affected parties would be involved.

As a secretariat to the Committee and the working group, the Biogas Secretariat was created at the Danish Energy Agency with the further responsibility of administering the Energy Agency's grants to the large-scale biogas plants.

Method of the Action Programme

The Action Programme has been carried out in three major steps:

1. Evaluation of potential biogas-plant projects and selection of the most appropriate plants to be included in the programme

9 biogas-plants were selected, 3 of which were already in operation at the beginning of the Action Programme. In order to gain a maximum of knowledge, this selection was based on the wish to test different biogas-plant concepts as regards biology, technology, biogas utilization, manure processing etc. The evaluation of the projects in question was based both on the experience from the biogas-plants already in operation as well as on the experts in the Technical Follow-Up Group, who functioned as a consultancy panel during this selction process, as to ensure a smooth technological development.

2. A 3-year data-collecting programme

Daily operational data from the 9 demonstration plants has been collected and analysed. The Secretariat has published a monthly bulletin containing these data as well as other relevant information from each plant. To evaluate the financial situation of the demonstration plants, the Danish Institute of Agricultural Economics has quarterly produced balance-sheets for each plant.

3. Analysis of the technical, financial, agricultural and environmental aspects of the demonstration plants on the basis of the collected data

Various detailed analyses have been carried out by the different research institutions represented in the Coordination Committee. The analyses have been performed within 5 different fields:

General description

Description of biogas plants in Denmark.

Biogas plant technology

Technical and financial analyses, transportation economics, the biogas process and the utilization of biogas.

Agricultural and environmental aspects

The impact of biogasplants on agriculture, fertilizer utilization and hygiene has been investigated in various research programmes. Furthermore the environmental impact on emissions of CO_2, NO_x, SO_2, ammoniaevaporation, methane-emissions, alternative treatments of organic waste and effect on leaching losses of nitrate has been investigated.

Financial analysis

Plant economy and socio-economic analysis.

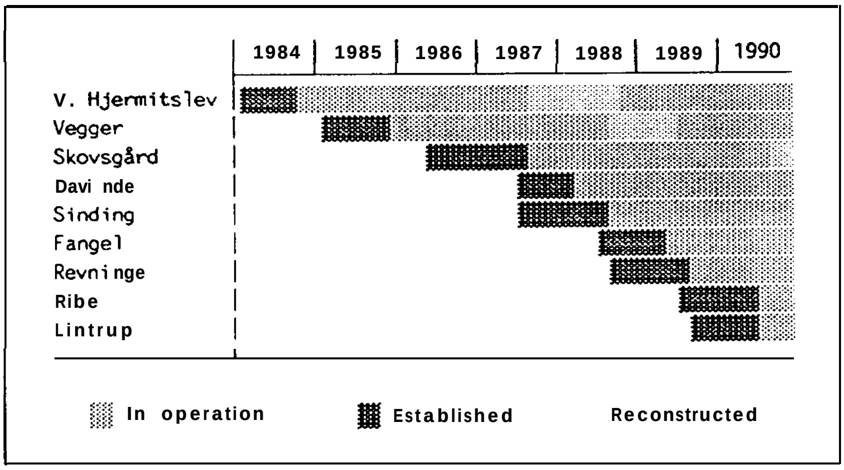
Dissemination prospects

Evaluation of the overall biogas potential in Denmark, dissemination scenarios, the biogas-plants of the future, identification of barriers as well as governmental instruments to stimulate continued dissemination.

The analyses has required extensive cooperation between all parties involved. The results are being published in 15 separate reports, which form the basis of the Main Report. An Intern Report was published in may 1989, and summaries of this report were published in English, German, French and Danish.

Other information activities in the course of the programme has included video-production, seminars, staff training courses, exhibitions, leaflets and the production of a quarterly magazine »Biogas News«.

Figure 1:
The demonstration plants under the Danish Biogas Programme



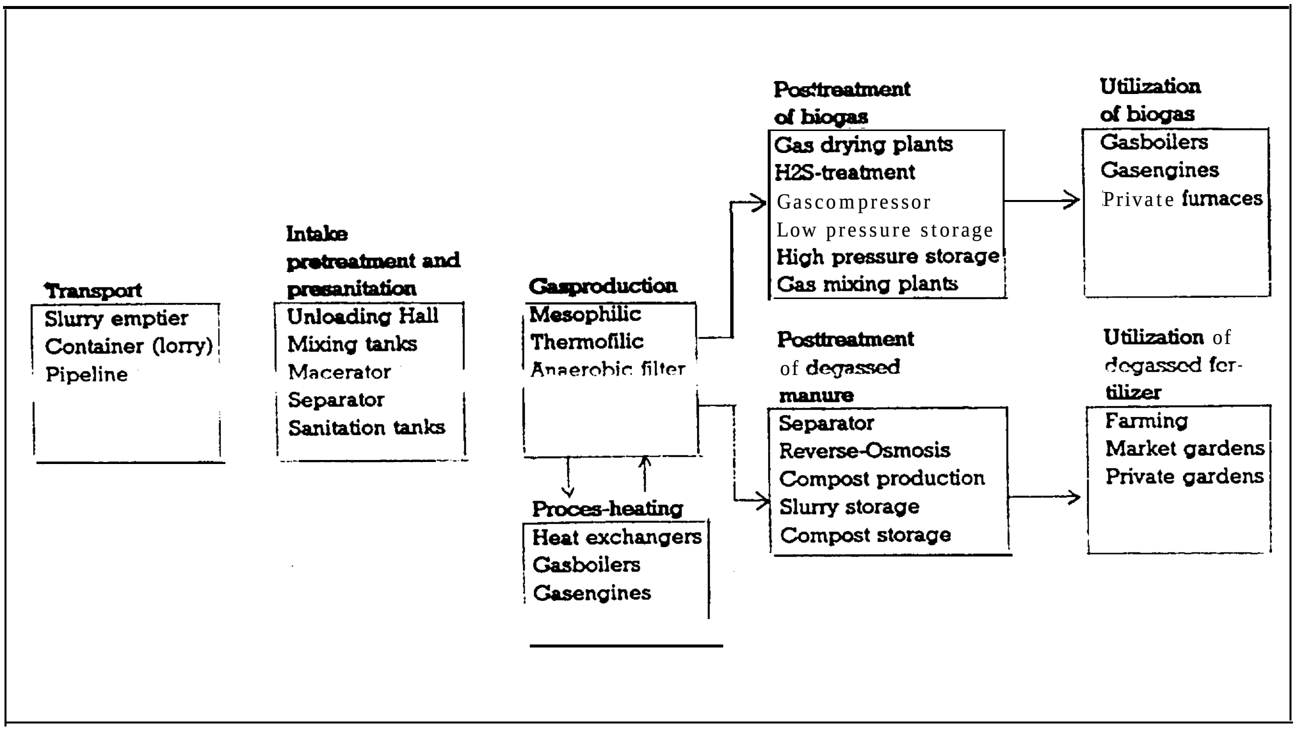
Results of the Action Programme

The Action Plan for Biogas Plants in Denmark analyses 9 plants which differ from one another in type, size, organizational structure and numbers of farmers attached; the plants are at Vester Hjermitslev, Skovsgaard and Vegger in Northern Jutland, Sinding in the municipality of Herning, Davinde, Fangel and Revninge on the island of Funen, and Ribe and Lintrup in Southern Jutland. Figure 1 shows when the plants were established and started up and when the three oldest plants were converted.

The system structure of the joint biogas plants

The term 'Joint Biogas Plants' covers the whole system from the collection of liquid manure from the individual farmer, to returning with fertilizer, storing, and spreading the degassed product on the soil. This delimination is important if the agricultural and environmental differences are to be evaluated in comparison with the way fertilizer is normally treated. The total system is illustrated in figure 2, and the location of the plants in figure 3.

Figure 2:
Systemdiagramme of a joint biogasplan t

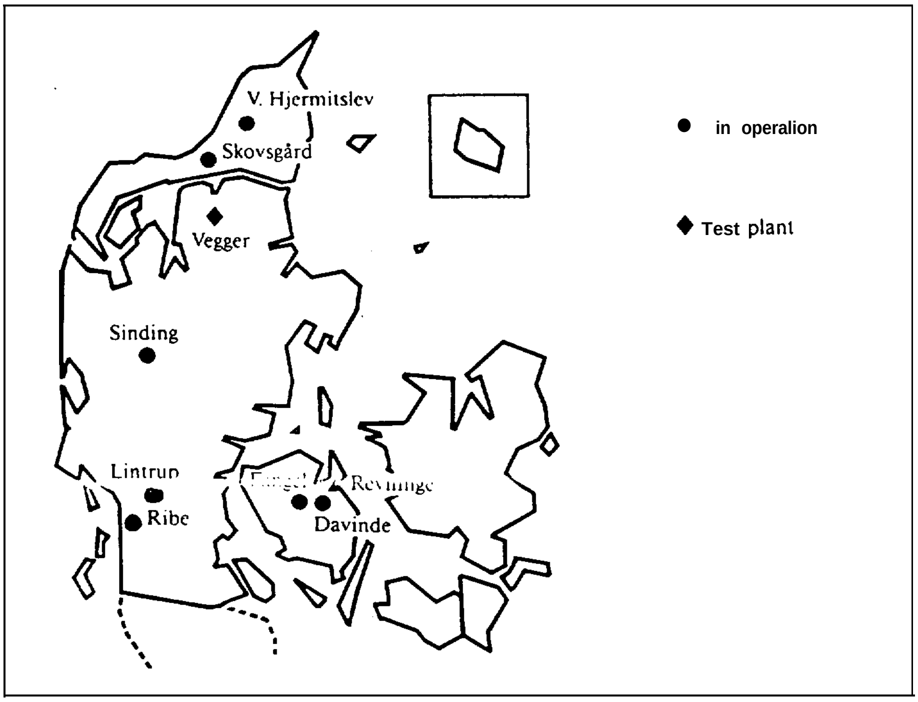


Biomass used in the plants

Although the joint biogas plants mainly use liquid manure to produce biogas, all plants supplement the slurry with small amounts of organic waste (1 O-l 5%). Many advantages are attached to adding other biomass than liquid manure to the plants. In many cases organic industrial waste is more easily convertible than liquid manure; it also greatly increass gas production without any significant rise in running costs. At the same time the plants earn money for receiving waste which would otherwise have to be dumped on a landfill or directly on farm land.

Finally, the fertilizer value of the waste is exploited in an optimal manner by returning it to farm land in a closed nutrient cycle. The plants are very careful that the organic waste meets the requirements as to heavy metals and hygiene thus ensuring that it can be soread onto farm land without any restrictions; this is a precondition for running the plants.

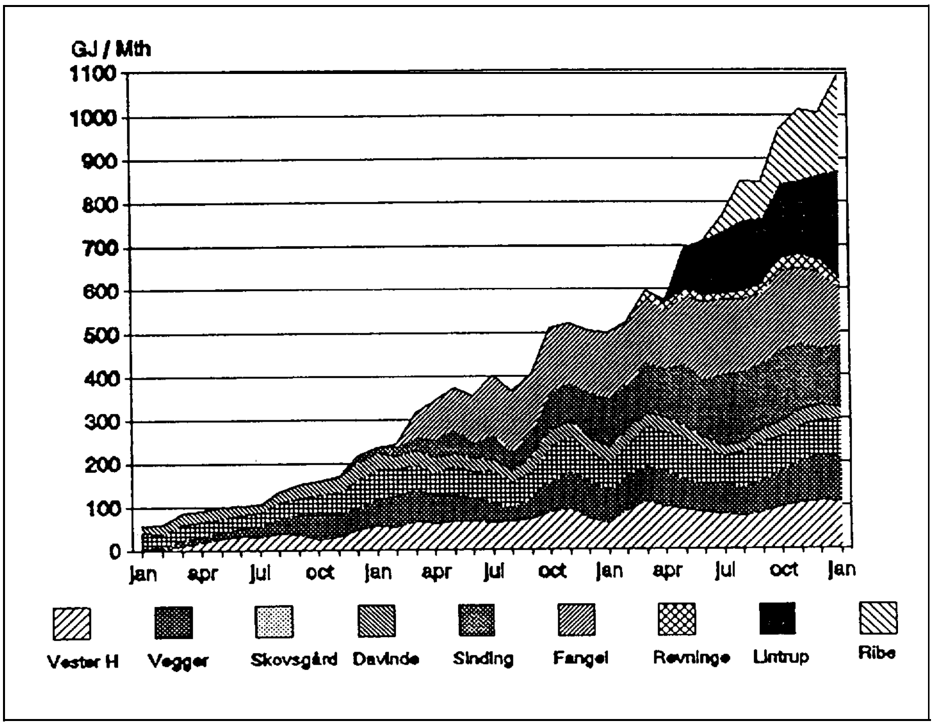
Figure 3: Geographical sitting of the demonstration plants and the test plant in Vegger



Technical aspects

Technology and gas production at the joint biogas plants have been greatly improved in the course of the last three years. Most of the plants are now operating reliably with a constantly high production of gas; this high gas

Figure 4: The gasproduction from the joint biogas plants 1988-1991



production is due both to the addition of'organic waste and to the fact that the plants have become more technically reliable. Figure *4* shows the month by month development in gas production at all of the joint biogas plants since January 1988.

It has proved to be the case that running a new joint biogas plant in, can take a long time from the points of view of technology and process; this is partly due to the desire to develop and test new technology at the new plants based on experience from the older plants. In other words, the blueprints of the joint biogas plants have not yet been standardized and it is still possible to improve the technology.

Finance

The economy of the first joint biogas plants, established during the first half of the 80s, was very bad for the first couple of years. The plants have been converted in step with experience and, as mentioned before, this has resulted in a high and stable production of gas. Moreover, the increased income from taking organic waste has helped to improve the plants' economy.

The preliminary results of the financial analyses point to the fact that it is possible to establish a plant, that is profitable from the point of view of company finance and social economy, provided that

1) the plant can sell the gas produced at a price that compares with, for example, the price of natural gas, and
2) income accrues from receiving organic industrial waste.

It seems to meet these conditions in several of the plants. However, the economy is very sensitive to increased running and maintenance costs and gas production that fails to materialize. In order to ensure contimuity in the technological development, future plants will need subsidies or the like for some time yet; this support will gradually be phased out over a short number of years.

Agricultural aspects

Part of the Biogas Action Plan consisted of farmers attached to the plants in the municipality of Herning and in Fangel being interviewed about the advantages and disadvantages of participating in the joint biogas plant.

The total average economic profit per farmer in the suppliers' association is an estimated approx. DKK 10,000 per year. This advantage can be divided up into savings in connection with:

Storage of fertilizer – in the form of cheaper storage tanks;

Spreading offertilizer – less use of labour as the fertilizer is stored near the fields in movable buffer stores;

The purchase of commercial fertilizer – partly because of the better distribution of manure from domestic animals i.a. to outlying fields where animal fertilizer has not been applied before, and partly because of the addition of nutrient-rich organic waste, and finally better utilization of nitrogen.

The analyses of the farmers in Fangel have also demonstrated the direct, economic profitability of being part of the circle of suppliers to the joint biogas plant.

A full-scale experimental installation has been constructed at the Lintrup plant where the degassed material is separated into a fibre fraction (c. 10%), a fertilizer concentrate (c. 35%), and water (c. 55%). After storage in lagoons, the water is piped to a river. There are important perspectives for fertilizer management in this filtering process. Apart from the potential management advantages and savings for farmers in spreading the fertilizer, the concentrate can potentially be upgraded to full fertilizer. A special 3 year experimental programme will verify or disprove the advantages and potentials. The plant has had running-in problems with its filter technique and is expected to be operating at full strength at some time during the summer of 199 1.

The following provisional conclusions can be drawn from the agricultural analyses:

Utilization of nitrogen

- Nitrogen is mineralized during digestion resulting in an NH, content of about 10% more than in non-digested slurry. Thus the N effect of the digested liquid manure can be expected to be better. However, it is a precondition for better utilization of the fertilizer that it is spread with a liquid manure applicator either injected into the ground or surfacelaid by tubes. And that this takes place during the growing season. When the liquid manure is spread by means of an ordinary spreader no improvement of the N effect can be seen.
- One experiment has demonstrated a significantly lower degree of denitrification when degassed as opposed to non-degassed liquid manure is applied, resulting in a lower discharge of nitrogen into the atmosphere. The probable reason is that the oxygen content of the soil falls more when non-degassed liquid is converted providing the (anaerobic) denitrification bacteria with better conditions.
- Being much more homogeneous than cattle manure in particular, degassed slurry is easier to dose evenly. In contrast to viscous and lumpy manure, there are no problems involved in using a manure injection applicator to spread degassed liquid manure.
- Sanitation of the liquid manure at the biogas plant allows it to be spread on grassy areas in the growing season resulting in a higher degree of N exploitation.
- Analyses of the degassed liquid manure provide farmers with a precise evaluation of nutrient content and enable them to utilize the manure from their domestic animals to the optimum through better fertilizer planning.
- The system of decentralized storage tanks has meant that apart from beibg able to spread slurry on fields that have never had domestic animal manure applied to them, the farmers have become more flexible with regard to spreading liquid manure at the most favourable time, thus securing a better N effect.
- Because of the lower viscosity, which means that degassed liquid manure does not adhere to leaves, ammonia evaporation is not greater when degassed rather than non-degassed liquid manure is spread by normal vacuum tank spreader despite the higher pH and the greater concentration of ammonia.
- The risk of increased ammonia evaporation from the storage tanks due to higher pH value of the degassed liquid manure, can be countered by covering the tanks with special membranes developed during the Action Plan. Additional advantages are, that the methane produced in the storage tanks can be utilized and that rainwater is kept out.

Utilizing phosphorous and potassium

- It has been proved that by mixing liquid manure from cattle and pigs phosphorous and potassium are more optimally distributed from a fertilizer point of view.
- As the biogas plant takes care of the transport and the farmer, in principle, can decide the manure is to be delivered, participating in a joint biogas plant provides the opportunity of applying animal manure to fields normally out of reach due to transport costs; this results in savings in the purchase of phosphorous and potassium.

Other effects

- A very important aspect is, that farmers with animal husbandry, usually are more than self-sufficient with regard to phosphorus, the most expensive fertilizer. To ensure optimal utilization of the nutrient salts in the manure (and thereby minimalize leaching losses) - include the N, **P** and K contained in the industrial and household waste – an export from farms with animal husbandry toward plant breeders is necessary. This will also improve soil quality. Joint biogas plants function as well-organized distribution systems, otherwise rarely available.
- At several joint biogas plants the system of accounting for the individual farmer's liquid manure is based on nutrient content and an assessment of the price of nitrogen, **P** and K. This has led to many farmers becoming more aware of the fertilizer value of liquid manure; they have accommodated domestic animal manure better into their fertilizer schedules with regard both to time and dosage.
- An important effect of joint biogas plants functioning as fertilizer stores is that the farmers who take part will be better able to adapt to structural developments. The joint biogas plant will easily be able to service fluctuations in the individual farmers's stocks of domestic animals and fertilizer demand because of the buffer capacity in the stores of liquid manure combined with flexibility of transport. Therefore by means of agreements with other suppliers it will be easy to deposit surplus liquid manure (thereby achieving environmental approval from the authorities) and equally simple to utilize surplus storage capacity.

Environmental Aspects

The advantages of joint biogas plants are both real and potential. The real advantages have been documented and the potential advantages, while probable, have not yet been more exactly quantified.

Real advantages

- The production of energy from domestic animal manure (and all other biomass) displaces fossil fuels and is CO,-neutral which means no net addition of the greenhouse gas, CO, to the atmosphere.
- Industrial waste and – in the longer term – household refuse will be done away with displacing dumping and incineration of waste, and the nutrients in the waste will be recycled to farming land achieving **sizeable** agricultural and environmental advantages.
- Degassed as opposed to non-degassed liquid manure is the source of far fewer unpleasant smells.

- Better distribution of N, P and K between the farms will lessen over-fertilization from farms with a surplus in relation to crop requirements.

Potential advantages

- The normal methane emission from liquid manure tanks on farms is reduced by degassing the slurry in a biogas plant. As methane is also a greenhouse gas this has a beneficial effect.
- By means of the increased N effect, greater mineralization of nitrogen during digestion and in the soil results in less nitrate leaching from farms making the best use of degassed liquid manure in relation to farms making the best use of raw liquid manure.
- Better utilization of fertilizer as a consequence of better distribution and fertilizer planning will mean less nitrate leaching.
- One of the effects of reducing denitrification will be better utilization of N resulting in less need for artificial fertilizer.
- An important potential aspect currently being investigated is the possibility that farms attached to a joint biogas plant may be able to considerably reduce nitrate leaching due to the spreading of manure from their domestic animals. As manure from great areas are collected and therefore concentrated at the joint biogas plant, various future manue treatment technologies can potentially be attached, such as:
  * Seperation of degassed material. By means of a sieve it is possible to separate the degassed slurry into a fibre fraction and a liquid fraction. A relatively large amount of the organically combined nitrogen in manure from domestic animals – that is the problematic nitrogen of which at least 50% is leached – is in the fibre fraction.
  * Reversed osmore. As earlier described, the full implications on leaching losses etc. by this technology, are presently being investigated in a follow-up programme attached to Lintrup Biogas Plant.
  * Other future, but not foreseeable technologies of manure treatment.

Potential disadvantages

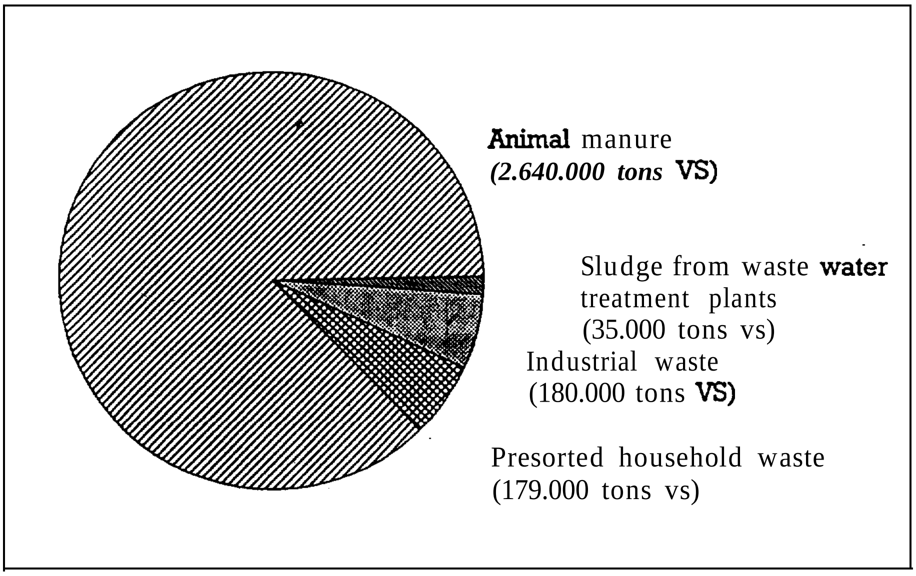
- Increased need for transport resulting in wear and tear on roads and inconvenience to traffic.

The potential of joint biogas plants in Denmark

The interim results of a specification of organic materials that can be treated in biogas plants can be seen from figure 5. The figure shows the maximum amounts of manure from domestic animals, source-sorted domestic waste, organic industrial waste and sludge from sewage treatment plants that can be added to biogas plants. Attention is drawn to the fact that the specification of organic industrial waste in particular is yet unsure.

As a measure of comparison, it can be mentioned that today the 9 plants receive about 1–2% of the maximum amount of waste specified here. The economy of the plants necessitates the addition of a certain amount of other organic waste as well as manure from domestic animals. On the basis of the distribution between domestic animal manure and other types of waste, it is estimated that it will be possible to exploit 20–30% of domestic animal manure in the long term.

Figure 5:
Maximum amounts of organic material usable in biogas** plants, **measured in tons VS (Volatile Solids)



International Workshop on Nitrogen and Agriculture in Schleswig 9 -12 April 1991

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International Workshop on Nitrogen and Agriculture in Schleswig 9 - 12 April 1991

List of oral presentations

Tuesday, 9 April 1991

9:oo -- 9:15	*Mr. Uwe Schell:* Opening of the meeting.
9:15 -- 10:45	*Dr.* Isermann: Ammonia Emission from Agriculture as a Component of its Nitrogen Balance and some Proposals for their adequate Reduction.
10:45 -- 11:00	Coffee break.
1 l:oo -- 13:oo	*Dr. Oldenburg:* Ammonia Emission from Animal Houses *Mr. S. Sommer:* Ammonia Volatilization from Slurry during Storage and in the Field. *Dr. Bless:* Ammonia Emission after the Application of Manure in the Field.
13:oo -- 14:30	Lunch break.
14:30 -- 16:00	*Dr. Kühl:* The Nitrogen Budget of an Area with intensive Agriculture and a high Livestock Production. Dr. *J. K. Schjörring:* Do agricultural crops play a role in atmosperic ammonia pollution? *Mr. C. Age Pedersen, Mr. J. Johnsen Hoy :* The Danish Action Plan for the Optimal use of Animal Manures.
16:00 -- 16:30	Coffee break.
16:30 -- 18:00	*Dr. Reiche :* Estimation of Water and Nitrogen Dynamics by the use of Computer Models.

Wednesday, 10 April 1991

9:oo -- 10:30	*Mr. Breinholt:* The Danish Programme on Methane fermentation of Animal Manures and Organic Wastes.	
	Dr. Hügle: Experiences with Mobile Slurry Treatment in Schleswig-Holstein.	
10:30 -- 1 l:oo	Coffee break	
1 l:oo -- 13:oo	*Dr. Mannebek:* Use of Biofilters to Reduce Ammonia Emissions. *Dr. Friedrich:* Reduction of Ammonia Emissions -- Technical and Legal Tools.	
13:oo -- 14:30	Lunch break	
14:30 -- 16:00	*Mr. N. Tyssen, Mr. l? B.Christensen, Mr. I? Kristensen:* Denitrification in Freshwater Ecosystems.	
	C. C. Hoffman: Wetland Management for Nutrients Retention. *Dr. Bruhm:* Nitrogen in Surface Waters in Schleswig-Holstein.	
16:00 -- 16:30	Coffee break	
16:3O -- 17:45	*Mr. Rekolainen:* Agricultural Nutrient loading of Surface Waters in Finland and Measures to reduce it.	
	Mrs. R. Taylor: Nitrogen Leaching from the North-Poland River Watersheds.	

Thursday, 11 April 1991

9:oo -- 10:30	*Dr. Cramer:* Integrated Plant Produktion as a Method for Optimizing Plant Production and Nutrition Cycle -- 5 years practical experiences. *Dr. Janßen:* Slurry Handling in Schleswig-Holstein - Recommendations and Regulations.
10:30 -- 11 :oo	Coffee break
1 l:oo -- 13:oo	*Ms R. Grant:* The Danish Monitoring Programme on Nutrient runoff from selected Catchment Basins.

	Ms. G. Blicher-Mathiesen, Mr. H. Nielsen, Mr. M. Erlandsen: 1. Quantification of Nationwide Nitrate leaching from Agricultural Land using a Soil-Plant Simulation Model 2. Evaluation of management strategies to combat nitrate leaching.
12:45 -- 14:00	Lunch break
14:oo -- 15:30	***Dr. Hartung :*** A general Code of Practice to Reduce Ammonia Volatilization. ***Mr. Lübbe:*** Good Agriculture Practice -- State of Art in the Federal Republic of Germany.
15:30 -- 16:00	Coffee break
16:00 -- 17:00	***Dr. Horstmann:*** Influence of Ammonia from Cattle Industry on the Nutrification Process in the Baltic.
17:oo -	***Dr. Torben A. Bonde:*** Closing of the Meeting.

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National Swedish Administration of Shipping and Navigation; 17-18 November 1986, Norrköping, Sweden
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