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Transport and Transformations of Nitrogen in an Arable Soil

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CONTENTS

PREFACE

This issue of "Ekohydrologi" consists of a Doctoral dissertation at the Department of Soil Sciences, Swedish University of Agricultural Sciences. The thesis, "Transport and transformations of nitrogen in an arable soil", consists of a summary and the following five papers:

I Bergström, L. 1986. Distribution and temporal changes of mineral nitrogen in soils supporting annual and perennial crops. Swedish J. Agric. Res. 16, 105-112.

II Bergström, L. 1987. Nitrate leaching and drainage from annual and perennial crops in tile-drained plots and lysimeters. J. Environ. Qual. 16, 11-18.

III Bergström, L. 1987. Leaching of 15-N-labeled nitrate fertilizer applied to barley and a grass ley. Acta Agric. Scand. (in press).

IV Johnsson, H., Bergström, L., Jansson, P.-E. & Paustian, K. 1987. Simulated nitrogen dynamics and losses in a layered agricultural soil. Agric. Ecosystems Environ. (in press).

V Bergström, L. & Johnsson, H. Simulated nitrogen dynamics and nitrate leaching in a perennial grass ley (submitted to Plant Soil).

The papers are referred to in the summary by their respective Roman numerals.

During the many years of work that it took to complete this thesis I have received help and support from a number of people, to whom I express my sincere gratitude.

Professor Nils Brink, my supervisor, introduced me to the field of water research, offered helpful guidance and critically reviewed the manuscripts included in this thesis.

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Stefan Ekberg, Annelie Mejbert, and Rose-Marie Niklasson helped with laboratory work and cheerfully discussed the local "Saturday nite action".

Barbro Boström skillfully typed my manuscripts.

Professor Thomas Rosswall effectively supervised the "Ecology of Arable Land" project, of which this study was one part, and devoted time to reading and constructively criticising my manuscripts.

Ruben Johansson helped with all kinds of field work; the careful way in which he collected field data was a major factor contributing to the progress made in this study.

Dr. David Tilles and Patricia Sweanor made excellent linguistic revisions, and even offered critical comments on the content of the manuscripts.

Britt-Marie Björkman, Kajsa Göransson, and Gudrun Sunnerstrand skillfully drew figures, and Peter Wigren made photographic reproductions.

Lennart Bergström, my father, proficiently drew illustrations.

The whole crowd within the "Ecology of Arable Land" project proved that working in a group creates an atmosphere fostering scientific achievement.

And last, but not least, my wife Aileen Bergström, showed incredible patience and understanding during completion of this thesis. She willingly read and commented on manuscripts dealing with leaching, barley, grass leys, and other problems far removed from her field of occupational therapy.

The work was carried out within the project "Ecology of Arable Land. The Role of organisms in Nitrogen Cycling". This project was funded by the Swedish Council for Planning and Coordination of Research, the Swedish Council for Forestry and Agricultural Research, the Swedish Natural Science Research Council, and the Swedish National Environment Protection Board.

Uppsala, March 30, 1987

Lars Bergström

TRANSPORT AND TRANSFORMATIONS OF NITROGEN IN AN ARABLE SOIL

Lars Bergström

Abstract. Nitrogen leaching from tile-drained plots and from three types of lysimeters was estimated during a 4-year period. Simultaneously, the mineral-N dynamics in the soil was investigated. The experimental field represented an arable soil located ca. 40 km north of Uppsala (Lat. $60^{o}10$'N, $17^{o}38$'E). The soil profile consisted of layers with variable texture and structure; this heterogeneity caused substantial differences in hydraulic properties between layers. Treatments included barley without and with addition of N-fertilizer, a N-fertilized grass ley, and a lucerne ley with no N-fertilization. Compared with tile-drained plots, lysimeters had generally higher drainage volumes and usually also higher leaching losses of nitrate. The maximum amount of nitrate leached from the fertilized barley treatment, measured from one of the lysimeter types, was 36 N kg/(ha yr). For unfertilized barley the corresponding amount was 5 N kg/(ha yr) during the same period. Nitrate leaching from the leys was mostly below 5 N kg/(ha yr), irrespective of the measuring method used. However, after the leys were ploughed, considerable leaching occurred, reaching 42 N kg/ha in the grass ley during the 4.5 months after ploughing. Simultaneously, the increase in soil mineral-N content in the grass ley reached 58 N kg/ha down to a depth of one meter. The mineral-N content of the soil was normally highest in the fertilized barley and lowest in the grass and lucerne leys.

An experiment using 15-N-labeled N-fertilizer, with enrichments ranging from ca. 1 to 99 % 15-N atom excess, showed that a maximum of 1.2 % of the labeled fertilizer was lost to drainage water during the 3 years after fertilization. Annual leaching of the labeled fertilizer never exceeded 1 N kg/ha from either the barley or the grass ley.

A soil nitrogen model, emphasizing mineral-N dynamics and losses, is presented and was used for a 3-year simulation in the barley and a 4-year simulation in the grass ley. Model predictions were compared with measurements of soil mineral-N content and nitrate leaching. Simulated values for mineral-N in the soil agreed fairly well with field data in both simulations. Discrepancies between simulated and measured soil mineral-N content in the topsoil were mainly related to plant uptake and mineralization. Discrepancies in deeper horizons as well as in the temporal distribution of simulated and measured nitrate leaching were mainly attributed to uncertainties in simulated water flows. The simulations indicated that measurements in tile-drained plots can considerably underestimate values for nitrate leaching.

INTRODUCTION

Several investigations have shown that the recovery of fertilizer-N in the field by a single harvested crop seldom exceeds 50 to 70 % of that applied and is often even less (Allison 1966; Hauck 1968). Evidently, considerable quantities of fertilizer-N are either immobilized by incorporation into organic-matter or are left in the soil in inorganic form. The latter often results in substantial leaching losses because the nitrogen commonly occurs as the mobile nitrate ion.

Concern over contamination of surface water and groundwater due to nitrate leaching has increased drastically during the last few decades. Along with this concern, there is a great demand for effective ways of reducing these leaching losses. Extensive work at the Division of Water Management at the Swedish University of Agricultural Sciences has resulted in a better understanding of the factors determining the magnitude of nitrate leaching under Swedish conditions (Brink 1982; Gustafson 1983; Bergström & Brink 1986). Thus a base of information, of value in designing proper counter-measures, has been created. Some factors, such as climate, hydrological conditions, and soil type, are site specific and hard to control. In contrast, N-fertilization rate and crop type are relatively easy to control for each individual farmer and therefore the type of factors that has been intensively tested in experiments focused on developing recommendations for reducing nitrate leaching.

Even though we basically understand how human disturbances and environmental factors affect nitrate leaching, there is still a problem in obtaining proper quantitative assessments. A variety of methods have been used for field measurements of leaching (e.g. Chichester 1977; Uhlen 1978; Cannell, Belford, Gales & Colin 1980; Bergström & Brink 1986). However, it is important to consider the principal differences between methods, especially when comparing quantitative results. Tile-drained field plots are often believed to give the most reliable leaching estimates, since they represent a suitable scale of arable lands with regular tile-drainage. However, lateral water movements and percolation to the groundwater may some times cause considerable underestimates of nitrate transport and drainage volumes from the soil (cf. Thomas & Barfield 1974; Hood 1977). Lysimeters, on the other hand, normally represent very small areal scales, often with disturbed soils. The percolating water at the bottom of the lysimeter can be considered as the sum of drainage water and water percolation to the groundwater. Thus, quantitative assessments of leaching and water movements in soils should be viewed with great caution.

To provide better guidelines for water and nitrogen management, mathematical simulation models have been developed and have received considerable attention in recent years. These models are often developed in connection to field measurement programs. Characteristic of most of these management models (e.g. Knisel 1980; Schaffer, Gupta, Linden, Molina, Clapp & Larson 1983) is the goal to enable widespread usage. Although many models dealing with soil nitrogen dynamics and nitrogen losses have the ultimate goal to be used for practical purposes, they also provide useful tools for organizing quantitative knowledge and for increasing our understanding of the complex nitrogen interactions in the soil-water-plant system. Data on soil water movements and leaching may be more thoroughly evaluated with the help of models. On the other hand, empirical field data are needed in the development process and for verification of mathematical models. Consequently, to minimize any potential problems associated with inefficient nitrogen usage and to avoid the degradation of water quality due to agricultural drainage, a good balance between models and field experiments is required .

PURPOSE OF THE STUDY AND METHODS USED

The objective of this study was to investigate how meteorological and hydrological conditions, soil properties, and cropping systems influence nitrate leaching and transformations of mineral nitrogen in an arable soil. The study was part of the project "Ecology of Arable Land. The Role of Organisms in Nitrogen Cycling", to which this study contributed quantitative data.

An intensive soil sampling program was performed to evaluate the seasonal distribution of mineral-N in the soil and its effect on the leaching pattern (I). Samples were collected down to a depth of 1 m.

Four different measuring methods were used for estimating nitrate leaching in the field (II), tile-drained plots and three types of lysimeters. The methods covered surface areas ranging from 0.36 ha (each individual large-plot) to 0.07 m^2 (each of the smallest lysimeters). In addition, lysimeters covering areas of 27 and 1.1 m^2 were used. To determine amounts of fertilizer-N in drainage water, 15-N-labeled fertilizer was applied on all lysimeter types at enrichments ranging from ca. 1 to 99 % 15-N atom excess (a.e.).

Piezometers were used to measure the hydrodynamic pressure of the experimental field (II). At each of three sites two piezometers, reaching two different depths, were installed so that the pressure gradient of the vertical groundwater flow could be determined.

To more thoroughly elucidate the causes of nitrate leaching and to determine water flow paths in the soil, a mathematical model was developed (Fig. 1) (IV). The model included the major processes of importance for inputs, transformations, and outputs of nitrogen in arable soils. It was first tested against measurements of soil mineral-N content and nitrate leaching in barley with and without N-fertilizer (IV). A second simulation included grass ley growth and ploughing of a grass ley (V).

THE EXPERIMENTAL FIELD

The experimental field, Kjettslinge, where the field work of this study was conducted, was situated ca. 40 km north of Uppsala (Lat. $60^{o}10$'N, $17^{o}38$'E). The field represents an arable soil that has been cultivated for a little more than 100 years. For many years prior to cultivation, the field was used for grazing and hay production. The climate in the region is cold-temperate and semi-humid. The investigated cropping systems were barley without and with N-fertilization (120 N kg/(ha yr), a N-fertilized grass ley (120+80 N kg/(ha yr), and a lucerne ley with no N-fertilization. The soil profile, down to a depth of 1 m, consists of four texturally and structurally different horizons: a topsoil layer (clay loam) down to a depth of 0.27 m, a sand layer down to ca. 0.5 m, and two clay layers of different structure. More details about the field are presented by Steen, Jansson & Persson (1984).

MODELLING APPROACH

The model included in this study (Fig. 1) was used to evaluate interactions, such as those between leaching and soil mineral-N content. Driving variables for the model are provided by a water and heat model (Jansson & Halldin 1979). The variables are as follows: surface runoff and infiltration, water flow between soil layers and flow to drainage tiles, unfrozen soil water content, and soil temperature.

The soil profile has a layered structure, and inorganic and organic pools of N and C are replicated in each layer. Ammonium and nitrate form the mineral-N pools, while organic-N pools consist of litter, faeces, and humus. Faeces, however, was not included in the simulations in this study. The litter pool consists of undecomposed material, while stabilized decomposition products form the humus pool. The division of soil organic-matter into two pools with different turnover rates seemed to be appropriate for simulation of N-release through mineralization (IV, V). The single plant component consists of nitrogen in above- and below-ground biomass.

In addition to the input of N-fertilizers (manure and industrial fertilizer) to the uppermost soil layer, atmospheric deposition is also considered. Outputs of N, via denitrification and leaching to drainage tiles, can occur from each soil layer.

Although the model was developed within a project where extensive information about nitrogen transformations and transports was available, the goal was to develop a model with a resolution compatible with information that is possible to obtain in agricultural field research. This considerably increases the number of possible applications.

DECISIVE FACTORS FOR NITRATE LEACHING

In this study it was possible to analyze the influence of some of the more important factors determining the rate of nitrate leaching. The influence of different crops on nitrogen dynamics in the soil and on associated processes was the overall objective for the "Ecology of Arable Land" project and the included cropping systems were shown to greatly influence nitrate leaching. Since the annual cropping systems were represented by barley with and without N-fertilization, it was possible to determine the rate of increase in leaching level due to input of N-fertilizer. The experimental period included years with both very dry and extremely wet weather which, in addition to the hydrological properties of the field and the mineral-N availability, defined the limits for the amounts of nitrate that could possibly leach during a certain period.

The following two sections briefly examine the influence of weather, hydrological conditions, and cropping systems on nitrate leaching.

Influence of weather and hydrological conditions

Under Swedish conditions, the influence of weather on nitrate leaching is primarily an effect of rainfall distribution. High rainfall during autumn, when soils are commonly bare, often results in substantial leaching of nitrate (e.g. Bergström & Brink 1986). Results from this study were no exception. Whenever high rainfall occurred and no crops were growing, such as during the autumns of 1981 and 1984, high leaching losses of nitrate occurred (II). In contrast, leaching was not observed during the dry autumns of 1982 and 1983. The low precipitation during summer/autumn 1982, when 15-N-labeled fertilizer was applied, resulted in extremely low leaching losses of the enriched fertilizer. Leaching of the 15-N-labeled fertilizer was not only low during 1982; it remained low throughout the entire 3-year period after fertilization (III). The relation between precipitation and leaching during spring was usually far less pronounced (II), owing to variation in factors such as frost depth and snow condition (cf. Jansson & Gustafson, in manuscript).

On a yearly basis, drainage volumes from arable lands not only depend on amounts of precipitation, but also on the relation to evapotranspiration, possible surface runoff, and groundwater conditions. When considering short periods, the soil's capacity to retain water is also of great importance. Thus, sandy soils usually have higher drainage volumes than clay soils, owing to the former's higher infiltration capacity and lesser ability to retain water in the soil. The soil profile represented in this study, with texturally and structurally very different horizons (I), made studies of water flow, and thus also of nitrate leaching, very complex. The upper ca. 0.5 m (topsoil and sand layer) had very high hydraulic conductivities while the fine textured layers below had considerably lower conductivities (Steen et al. 1984).

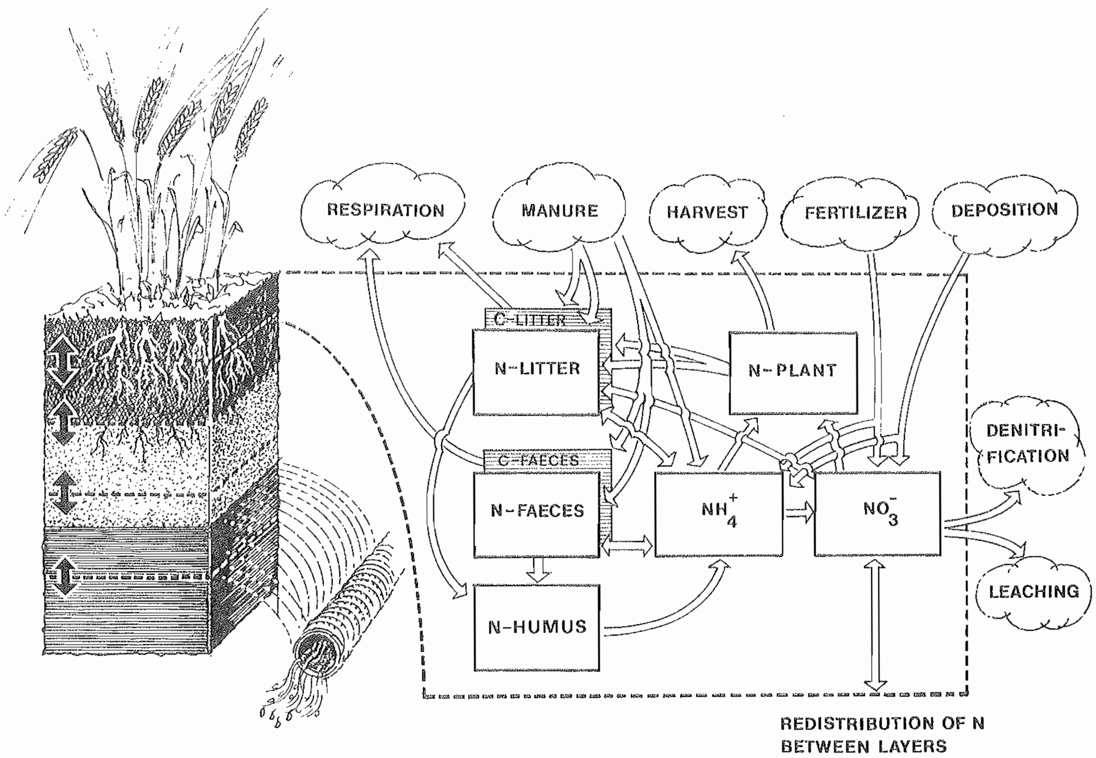

Fig. 1. Flow diagram showing the structure of the nitrogen model used in this study. Included within the dashed line are components representing the uppermost soil layer. Except for fertilization and deposition the subsurface layers have the same structure. (From paper IV).

Possibilities for studying water flow rates and flow paths in the field are often very limited, which was the case in this study. However, the differences in drainage volumes and discharge rates between the various measuring methods used (Fig. 2) (II), indicated that additional information on water movement through the soil was needed to fully understand the drainage and leaching pattern obtained. Piezometer readings implied that substantial tile-drainage bypass occurred following dry periods. In contrast, an inflow of groundwater, which contributed to the tile-drainage flow, was observed in certain parts of the field during wet periods (II). Both these conditions can explain much of the difference obtained between the tile-drained plots and the lysimeters concerning drainage volumes (Fig. 2) and amounts of nitrate leached.

The water and heat simulations used as input data to the nitrogen model (Fig. 1) provided possibilities to analyze water flow paths and also to explain the differences in drainage patterns between the tile-drained plots and the lysimeters. The most striking difference occurred during autumn 1984 (Fig. 2). The simulated drainage pattern corresponded fairly well with drainage from lysimeters, while the abrupt termination of the drainage flow from the tile-drained plots in October 1984 (Fig. 2) had little resemblance to simulated drainage. Water percolation to deeper groundwater, which was possible in the tile-drained plots, was certainly impossible in the lysimeters, and in the simulations it was also assumed that percolation to ground

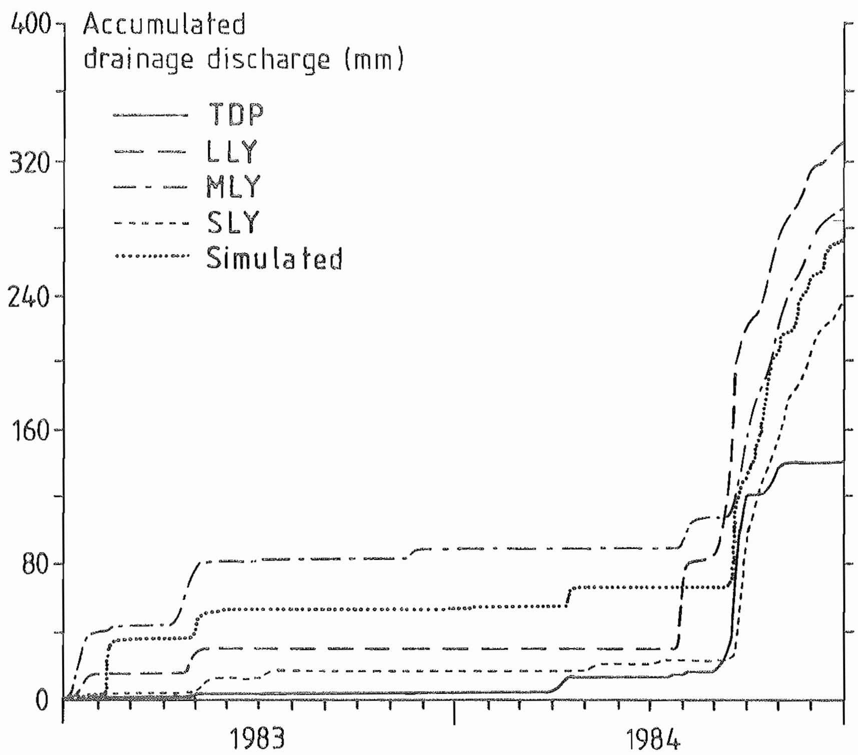

Fig. 2. Accumulated drainage discharge, simulated and measured with the different measuring methods, i.e., tile-drained plots (TDP), large lysimeters (LLY), medium lysimeters (MLY), and small lysimeters (SLY). The treatment was N-fertilized barley.

water was negligible (IV). Thus, the deviation between simulated and measured tile-drainage added further support to the idea that a tile-drainage bypass actually occurred. By using the model to analyze possible water flow paths in the heterogeneous soil profile, differences in flow rates between simulated drainage and tile - drainage during autumn 1984 were also successfully elucidated (V). A faster tile - drainage flow in the field might be explained in terms of either macro-pore flow or lateral water movement to drainage tiles. In terms of the grass ley simulation (V), the simulated water flow between the sand and the upper boundary of the clay during September 1984 coincided with the bulk of measured tile-drainage, which justifies the assumption of a lateral water movement in the sand.

Although no exact answers can be obtained with a mathematical model, such as the one used in this study, the examples above show that hypotheses can be tested and valuable information complementing the measurements can be offered. Not only does this hold true for questions related to drainage and leaching pattern, but it also applies to a variety of questions dealing with transformations and transport of nitrogen in agricultural soils.

Influence of cropping systems

When examining the influence of different cropping systems on the rate and extent of nitrate leaching, it is important to consider both the effects on the volume of water moving through the soil and the effects on nitrate concentrations in drainage water. In perennial crops, growing from early spring to late autumn, both drainage volumes and nitrate concentrations are usually reduced compared to annual crops with a relatively short growing season. Under Swedish conditions, the extended growing season, which is typical for most perennial crops, usually coincides with periods of high precipitation. Consequently, substantial reductions in nitrate leaching are often obtained by growing perennial crops (e.g. Gustafson 1983).

The perennial grass and lucerne leys included in this study had mean annual drainage volumes that were less than 50 % of the corresponding drainage volumes from the annual barley crops (II). This difference refers to measurements in the tile-drained plots. Also, nitrate concentrations in water draining from the perennial leys were mostly lower compared with the barley crops, irrespective of the measuring method used. Thus, the efficiency of a perennial crop to immobilize potentially leachable nitrate and thereby reduce leaching was well illustrated by results from this study (II). However, it was also demonstrated how drastically both nitrate leaching and mineral-N content of the soil increased after the leys were ploughed. Leaching and the increase in soil mineral-N content together accounted for up to ca. 100 N kg/ha during the 4.5 months after ploughing. Simulation results obtained with the nitrogen model (IV) suggested that this increase was mainly attributable to crop residue input and lack of a growing crop after ploughing, rather than to accumulation of litter during the experimental period (V).

Although the N-fertilized barley had mostly lower drainage volumes than the unfertilized barley (II), as a result of higher evapotranspiration, the overall difference was far less pronounced than the difference between the annual and the perennial crops. However, it did indicate that N-fertilization rate can strongly influence the water balance in the soil (cf. Jansson & Thoms-Hjärpe 1986).

As a strategy for reducing the problems associated with the accumulation of nitrate in the soil after harvest of an annual crop, the cultivation of catch crops has received considerable attention in recent years. Moreover, the results of attempts to reduce nitrate leaching with catch crops have been promising (e.g. Gustafson & Torstensson 1984; Kreuger & Brink 1984). When a catch crop is sown at the time of seedbed preparation in spring, nitrate leaching during autumn can be expected to be reduced to levels similar to those occurring in perennial leys (cf. I).

During spring there are also considerable differences between annual and perennial crops concerning the risks for high leaching losses of nitrate. An important factor in the crop response to fertilizer applications is the rate at which different crops accumulate N. Barley, for example, accumulates much of its nitrogen during the period starting about 1 month after emergence (Hansson, Pettersson & Paustian 1987). An established grass ley, on the other hand, begins taking up N immediately after fertilization. This fundamental difference in the timing of crop N-uptake between an annual crop, such as barley, and a perennial grass ley was well illustrated in this study by differences in soil mineral-N content following fertilization (I). The mineral-N content reached pre-fertilization levels within 1 month in the grass ley, while there was even a slight increase in soil mineral-N content in the barley during the month after fertilization. Loss of N through leaching during this period (cf. Schepers & Mielke 1983) can result in both environmental pollution (cf. Gustafson 1985) and reduced crop yield. To better coincide with the rapid uptake period, N-fertilizer should be applied 1 month after the sowing of barley. This would reduce the risks for nitrate leaching. Similarly, fertilizer placement to encourage root growth could lead to increased N-uptake by crops, which would also reduce the risks of nitrate being leached from the root zone. Split applications of N-fertilizer and use of slow-release fertilizers are also discussed as measures for counteracting nitrate leaching, although the positive effects on crop yield are more questionable.

When dealing with the risks for nitrate leaching, it is important to know how much of a single application of N-fertilizer is taken up by a crop or immobilized into soil organic-matter pools and the rates of these processes. Results obtained in the experiment using 15-N-labeled fertilizer (III) showed that high fertilizer-use efficiency by a crop can more or less eliminate a single N-fertilizer dose from being leached out during the following 3 years. Thus, not more than 1.2 % of the 15-N-labeled fertilizer applied to barley was recovered in drainage water during the 3 years after application. Even lower amounts of 15-N-labeled fertilizer were recovered in water draining from the grass leys. In contrast, 63 % of the labeled fertilizer was found in above-ground plant biomass (barley) during the first year (Lindberg, unpublished). Although these results indicate that only small amounts of nitrate in drainage water are derived from fertilizer, there is no doubt that N-fertilization, in the long run, substantially contributes to increased leaching of nitrate. This was well illustrated when comparing leaching in N-fertilized and unfertilized barley with the much higher losses in the fertilized treatment (II). From a long-term perspective there is reason to believe that leaching losses and N-losses to the atmosphere will

degree of enrichment needed for accurately quantifying the amounts of N-fertilizer in drainage water. This value is important to know owing to the high cost of 15-N-labeled fertilizer. The use of 15-N enrichments as high as 99 % a.e. is rare in field studies of leaching; we were more interested to know whether 1 % 15-N a.e. was sufficient. The results obtained in this study showed that the lowest enrichment used (ca. 1 % 15-N a.e.) seemed to be adequate for quantifying of fertilizer-N in drainage water. Such low enrichments enable extended usage, both in terms of area covered and number of replicates.

The depth of soil sampling was chosen so as to coincide with the drainage depth of the tile-drained plots. This enabled good possibilities to relate nitrate leaching through drainage tiles to the amounts of soil mineral-N potentially available for leaching. Especially in modelling work, measurements of mineral-N content of the soil are useful, since they reflect the actual nitrogen status of the soil also when no drainage is occurring. Thus, in both simulations the nitrogen model was adapted so as to obtain reasonable agreement between simulated and measured values for soil mineral-N content. The simulated values for nitrate leaching were arrived at independently. Consequently, the good agreement between simulated and measured values of leaching, that was possible to reach in this study, indicates that good leaching predictions can be obtained with this model if data on soil mineral-N content are available (cf. Jansson, Borg, Lundin & Lindén 1987).

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Distribution and Temporal Changes of Mineral Nitrogen in Soils Supporting Annual and Perennial Crops

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Abstract. The movements and distribution of mineral soil nitrogen (min-N) throughout the year were investigated in four cropping systems (unfertilized and fertilized barley, grass and lucerne leys). Soil samples were collected down to a depth of 1 m, six to nine times each year during the growing season and the vegetation-free period.

None of the cropping systems caused any major long-term accumulations of min-N during the experimental period. However, autumn amounts of as much as 70 N kg ha^{-1} down to a depth of 1 m were typical of the fertilized (120 N kg ha^{-1} yr^{-1}) barley treatment. The grass and lucerne leys commonly showed min-N levels below 35 N kg ha^{-1} which were usually lower than the corresponding levels in the unfertilized barley profiles. Substantial increases in min-N content, reaching 117 N kg ha^{-1} within three months, occurred after the leys were ploughed. In addition to type of cropping system, weather conditions also greatly influenced the amounts and distribution of min-N in the soil.

To reduce the min-N supply after harvesting fertilized barley, Italian ryegrass (*Lolium multiflorum*) was insown in the barley. A reduction of 23 N kg ha^{-1} in the top meter of soil was obtained with this treatment.

INTRODUCTION

During the last few decades, an increasing awareness of the low efficiency with which certain fertilized crops take up nitrogen has developed into a topic of major concern. A potential threat to environmental quality is posed by excessive N-fertilization and the costs of commercial N-fertilizers are increasing; thus their use needs to be restricted. Annual agricultural crops do not take up all of the N applied as fertilizer in a given season. Under Swedish conditions there is usually some portion of the applied fertilizer, beyond that lost by leaching and other N-loss mechanisms, that remains in the soil after the cropping season (Bergström & Brink, 1986). To reduce this residual N and N mineralized after the cropping season, thereby minimizing the risks of leaching, autumn-sown catch crops have been used successfully (Welch, 1974; Gustafson & Torstensson, 1984). However, unfavourable weather conditions can often prevent seeding and growth of these catch crops (Kreuger & Brink, 1984). Another possible approach would be to interseed a suitable crop with the main crop, thus reducing the risk of poor establishment. Perennial leys usually perform as optimal catch crops (MacLean, 1977; Gustafson, 1983; Bergström, in manuscript).

To follow the seasonal dynamics of mineral-N (min-N) in the soil and to clarify the causes of leaching, fairly intensive and well planned soil sampling programs are required. Large fluctuations in min-N content of the soil for various crops which were coupled to leaching of N were observed by Cameron et al. (1978*a*). Lindén (1981) conducted monthly sampling programs in central and southern Sweden and found similar fluctuations in min-N content when growing cereals. The additional uptake of N early and late in the growing season, which is typical of perennial leys, usually results in more stable conditions concerning the distribution and amounts of min-N in the soil. The design of proper counter-measures for reducing leaching of N requires that the dynamics of N-compound movements in the soil during the year be thoroughly understood.

The main objective of the present study was to use an intensive soil sampling program to evaluate the seasonal distribution of min-N in the soil in four cropping systems including annual as well as perennial crops. In addition, Italian ryegrass (*Lolium multiflorum*), insown in barley, was grown as a catch crop to test its capacity to reduce the min-N level of the soil.

MATERIALS AND METHODS

Experimental field

The experimental field, Kjettslinge, is situated in central Sweden, ca. 40 km north of Uppsala. The plots in which the soil samples were collected consist of two separate parts: four large-plots (each 0.36 ha) and sixteen small-plots (each 0.056 ha). Four small-plots were grouped in each of four

Table 1. *Sequence of crops, times for ploughing of leys and fertilization intensities (N kg ha^{-1}) in the large-plots and the small-plots in blocks A–D*

	B0		B120		GL		LL	
Year	Crop	N	Crop	N	Crop	N	Crop	N
Large-plots								
1980	Fallow	0	Fallow	0	Fallow[a]	0	Fallow[b]	0
1981	Barley	0	Barley	120	Grass	120+80	Lucerne	0
1982	Barley	0	Barley	120	Grass	120+80	Lucerne[c]	0
1983	Barley	0	Barley	120	Grass	120+80	Lucerne	0
1984	Barley	0	Barley	120	Grass	120+80	Barley[d]	120
1984 Aug.	–	–	–	–	Ploughed		–	–
Small-plots								
1980	Barley	0	Barley	120	Barley[e]	60	Barley[f]	30
1981	Barley	0	Barley	120	Grass	120+80	Lucerne	0
1982	Barley	0	Barley	120	Grass	120+80	Lucerne[g]	0
1983	Barley	0	Barley	120	Grass	120+80	Lucerne	0
1984	Barley	0	Barley	120	Grass	120+80	Lucerne	0
1984 Aug.	–	–	–	–	Ploughed[h]		Ploughed[h]	

[a] Grass sown in August. [b] Lucerne sown in August. [c] Ploughed and re-established in July. [d] Sown in May after ploughing the lucerne. [e] Grass insown. [f] Lucerne insown. [g] Lucerne small-plots in blocks C and D ploughed in July, thereafter excluded. [h] 50% of each plot was left unploughed for further soil sampling.

blocks (A–D). The large-plots and each block included four different cropping systems, i.e.:

1. B0, barley (*Hordeum distichum* L.) with no addition of N-fertilizer;
2. B120, barley with an annual N-fertilization rate of 120 N kg ha^{-1} (calcium nitrate);
3. GL, grass ley (*Festuca pratensis*) with an annual split N-fertilization rate of 120+80 N kg ha^{-1} (calcium nitrate); and
4. LL, lucerne ley (*Medicago sativa*) with no N-fertilization.

The sequence of crops, times for ploughing of leys and fertilizer application rates in the various plots are presented in Table 1. The barley stubble was ploughed down in the autumn of each year.

The large-plots were fallowed during the cropping season 1980 (Table 1) due to installation of tile-drainage systems for leaching measurements (Bergström, in manuscript). The LL large-plot and the LL small-plots in blocks C and D were treated with glyphosate, ploughed and re-established in July 1982 (Table 1) after a severe infestation of couch-grass (*Agropyron repens* L.). After 1 July 1982, the soil sampling was terminated in the LL small-plots in blocks C and D. The LL large-plot was furthermore ploughed in May 1984, three months earlier than initially planned, because of damage incurred during the preceding winter and was thereafter sown with barley fertilized with 120 N kg ha^{-1} (Table 1).

To follow the influence of interseeding, Italian ryegrass was insown in barley (B120) in May 1984, hereafter called the B120X-treatment. A small plot (4×4 m) with this treatment was situated in a corner of the B120 large-plot.

The experimental field has been described in detail by Steen et al. (1984).

Soil profiles

The soil down to a depth of 1.0 m consists of four distinct layers (Steen et al., 1984) corresponding to the divisions used for collecting soil samples:

(1) the topsoil, a clay loam with a mean thickness of 0.27 m;
(2) a fine sand layer of variable thickness. The mean thicknesses in the various large-plots were: 0.27 (B0), 0.17 (B120), 0.31 (GL) and 0.22 (LL) m. The corresponding thicknesses in the small-plots, calculated as mean values of the four replicates, were: 0.22 (B0), 0.12 (B120), 0.16 (GL) and 0.20 (LL) m.
(3) an oxidized clay layer, down to a depth of 0.75 m; and
(4) a non-oxidized clay layer, from 0.75 to 1.00 m.

In the following presentation of results all values of min-N refer to total contents in the top meter of soil.

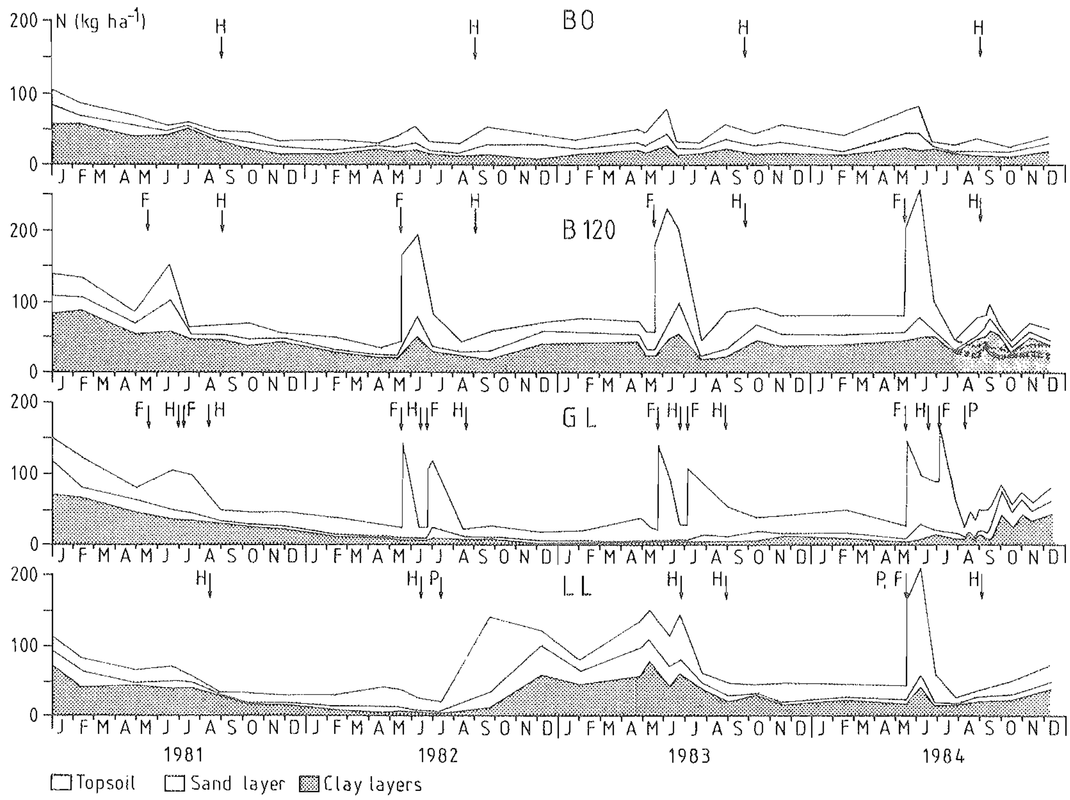

Fig. 1. Accumulated min-N contents in the top meter of the large-plot profiles (B0, B120, GL and LL). The dotted curves refer to the B120X-profile on top of which Italian ryegrass was insown. *F*, fertilization; *H*, harvest; *P*, ploughing.

The topsoil consists of 15–20% clay, 2.2% carbon and has a pH of 6.0–6.5.

Soil sampling

Soil samples from each plot were collected six to nine times each year, most frequently during the cropping season. Beginning in 1982, the samplings were timed so as to coincide with important phenological stages of the crops or to coincide with periods during which there was reason to expect essential changes in min-N content of the soil profile; i.e., periods following fertilization or periods of intensive mineralization. In addition, samples were also collected annually in early winter, mid-winter and immediately after the soil thaw in spring. During the autumn of 1984, all ley plots were sampled considerably more often in order to follow the fate of min-N after the leys were ploughed. More intensive sampling was also performed in the B120 large-plot and B120X-plot during the same period.

Tube drills were used for sampling. The sampling methods are described in detail by Lindén (1977). In each large-plot, twenty samples were taken from the topsoil, six from the sand layer and six from each of the clay layers. The larger number of samples was necessary in the topsoil because its N-content normally shows greater variation than in the lower soil layers (Smith, 1977). Four samples per small-plot in blocks A–D were taken from the topsoil while two each were taken from the sand and clay layers. Four samples from each layer were collected in the B120X-plot. The samples were mixed by layers into a pooled sample for each layer in each plot.

The difference in min-N content between ploughed and unploughed GL small-plots was compared using pair-wise *t*-test.

Soil analyses

Soil samples were deep-frozen on the day of collection to inhibit N-conversion. For analysis, 120 g of the thawed, moist soil was weighed and extracted

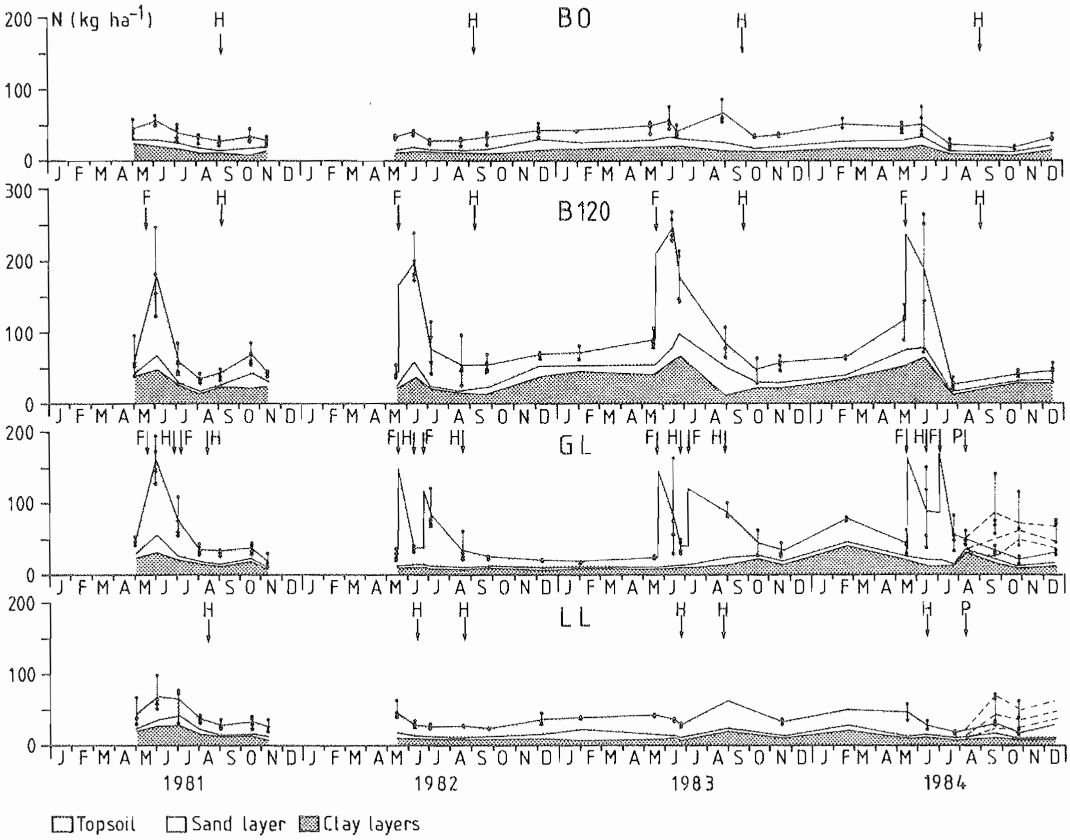

Fig. 2. Accumulated min-N contents in the top meter of soil in the small-plots. The curves represent mean values of the four replicates of each cropping system (B0, B120, GL and LL). The total min-N content (0–1 m) is shown for each small-plot (⊛); vertical line indicates the range. The dotted curves refer to the ploughed parts of the ley plots. *F*, fertilization; *H*, harvest; *P*, ploughing.

with 300 ml 2 M KCl. Ammonium and nitrate concentrations were determined in the extract using analytical methods in accordance with Swedish Standards (SIS, 1976), adjusted for an auto-analyser.

The analytical values were converted into N kg ha^{-1} using dry bulk densities of 1.45 kg dm^{-3} in the topsoil, 1.60 kg dm^{-3} in the sand layer and 1.25 kg dm^{-3} in the clay layers (Steen et al., 1984).

RESULTS AND DISCUSSION

Influence of cropping systems

The seasonal fluctuations of min-N in the soil in the four cropping systems are presented in Figs. 1 and 2. The relatively low variability in N levels between replicates in the various parts of the field allows a presentation that is valid for both the large-plots and the small-plots.

Nitrate usually constituted the major part of the two measured min-N fractions (NO_3–N and NH_4–N). Over the entire experimental period, the average relative distribution between nitrate and ammonium and the average quantities of ammonium and min-N in the large-plots were:

	B0	B120	GL	LL
NO_3–N (% of min-N)	74	86	66	70
NH_4–N (% of min-N)	26	14	34	30
NH_4–N (kg ha^{-1})	11	11	12	13
NH_4–N+NO_3–N (kg ha^{-1})	42	79	35	43

Thus, in proportion to total min-N content there was usually more nitrate in soils associated with barley than in soils associated with perennial leys. The quantities of ammonium on each sampling occasion were generally stable and less dependent of the cropping systems. Cameron et al. (1978*b*) also found very little change in ammonium concentra-

Table 2. *Min-N content in the soil (N kg ha^{-1}) before and after ploughing the leys (GL and LL) and comparison of min-N content (N kg ha^{-1}) between ploughed and unploughed small-plots. Standard errors are given within parenthesis (n=4)*

	Before plough.		After plough.			Plough.	Unplough.	
	July 1982	Aug. 1984	Sept. 1982	Dec. 1984	Diff.	Dec. 1984	Dec. 1984	Diff.
Large-plots								
GL	–	23	–	81	58	–	–	–
LL	20	–	137	–	117	–	–	–
Small-plots								
GL[a]	–	47 (±6)	–	64 (±7)	17	64 (±7)	28 (±2)	36**
LL[b]	–	15	–	60	45	60	27	33

[a] Blocks A–D. [b] Block A.
** Significant difference ($p<0.01$).

tions in the soil over time. A study carried out by Morrill & Dawson (1967) showed that low and rather constant levels of ammonium, as opposed to great nitrate fluctuations are to be expected for soils with chemical properties similar to those of the Kjettslinge field (Steen et al., 1984).

Most of the N fluctuations occurred in the topsoil and, to a lesser extent, in the clay layers. The sand layer showed very low amounts of N throughout the experimental period, reflecting its low capacity for storing of min-N which is typical of most coarse textured soils.

Crop type and N-fertilizer applications greatly influenced the temporal distribution of min-N in the soil. The two N-fertilized treatments (B120 and GL) showed fundamental differences in timing of their major uptake of applied fertilizer. The input of 120 N kg ha^{-1} in May each year was usually taken up within a month by the grass ley (GL), while no uptake of importance started until mid-June in the fertilized barley (B120). In fact, the B120-profile, as well as the BO-profile, showed increasing amounts of min-N during the month following fertilization, due to net mineralization.

Once the barley started to actively take up N the min-N supply in the soil decreased to pre-fertilization levels within about three weeks. Thus, the periods during which the main uptake of N occurred were usually short in both annual and perennial crops. These large, rapid min-N fluctuations emphasize the importance of frequent soil sampling to fully understand the seasonal dynamics of min-N in agricultural soils.

Spring and autumn increases of min-N were observed in the B0-profiles but were masked somewhat in the B120-treatment due to the high N-fertilization level. None of the cropping systems showed a major build-up of min-N during the experimental period. However, the B120-profiles reached levels high enough to justify reduced application rates of N-fertilizer.

Both perennial leys (GL and LL) were characterized by low contents of min-N in the soil, commonly below 35 N kg ha^{-1}. However, after the leys were ploughed considerable increases in soil min-N occurred (Figs. 1 and 2; Table 2). An increase by as much as 117 N kg ha^{-1} occurred during the three-month period following ploughing of the LL large-plot in July 1982. The amounts of min-N in the GL large-plot increased by 58 N kg ha^{-1} within a four-month period after ploughing in August 1984. For the GL and LL small-plots the average increases in min-N were 17 and 45 N kg ha^{-1}, respectively, during the same four-month period. Compared with the intact GL small-plots, the ploughed small-plots had significantly higher ($p<0.01$) amounts of min-N in the profile in December 1984, on average 36 N kg ha^{-1}. The corresponding difference for the LL small-plots was 33 N kg ha^{-1}.

Thus, large amounts of min-N were released following mineralization of the plant material incorporated into the soil when ploughing the leys. Only low amounts of this min-N usually remained in upper sections during the rainy autumn of 1984 (Figs. 1 and 2). This suggests that considerably elevated nitrate concentrations in drainage water, related to ploughing of leys, are also to be expected. Cameron & Wild (1984) estimated the amount of nitrate leached below 0.9 m to be 100 N kg ha^{-1} over two winters as a result of ploughing grassland.

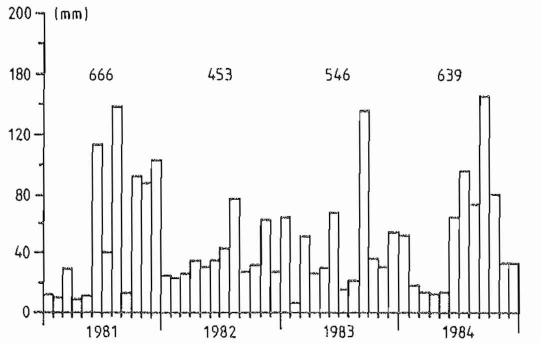

Fig. 3. Monthly precipitation during the experimental period. Annual precipitation is also given.

However, it is important to note that the more common practice is to sow winter rather than spring cereals after ploughing leys; this should reduce the supply of min-N in the soil during the autumn, thereby also reducing the risk for severe leaching.

Influence of weather conditions

Variations in the weather conditions, especially precipitation (Fig. 3), during the growing seasons as well as during the vegetation-free periods greatly influenced the amount and distribution of min-N in the soil.

In the barley treatments (B0 and B120), the min-N levels mostly depended on the amount of precipitation during autumn due to its crucial influence on N-movement in the profiles when no crop was growing. An autumn–spring relationship was observed: higher levels of min-N were found in the soil during the springs as compared with the preceding autumns, following the dry summer/autumns that occurred in 1982 and 1983. During these years (1 Aug. – 30 Apr.), the spring increases were 28 and 36 N kg ha^{-1} for the B120-treatment and 23 and 44 N kg ha^{-1} for the B0-treatment, all referring to the large-plots (Fig. 1 and Table 3). In contrast, following autumns with high precipitation, such as 1981 (Fig. 3), there was less min-N left in the soil during the spring. The amounts of min-N in the B120 and B0 large-plots decreased with 28 and 27 N kg ha^{-1}, respectively, from August 1981 to April 1982 (Fig. 1 and Table 3). Apparently, there were considerable vertical movements of min-N between the rainy autumn of 1981 and the following spring.

Ludwick et al. (1977) tried to correlate autumn nitrate contents with spring contents for a number of soils to quantify nitrate changes occurring during winter. An attempt was made by Bergström & Brink (1986) to correlate the residual amounts of min-N in the soil after harvest with autumn–winter leaching of nitrate. Both approaches yielded significant correlation coefficients (at the 0.01 and 0.0005 levels, respectively), suggesting that good estimates of nitrate contents prior to sowing and nitrate leaching can be obtained by results based on early autumn sampling of soils.

However, the results of the present study show that such relationships rarely exist, mostly due to the year-to-year fluctuations in amounts and distribution of rainfall. This, in turn, implies that fertilizer-N recommendations based on early autumn sampling can be very risky, which was also shown by Mattson & Brink (1982). A more sophisticated approach was developed by Burns & Greenwood (1982); they used a simple, quantitative model for estimating leaching under a range of conditions which was based on estimates of autumn nitrate residues and on meteorological data. They found that fluctuations in the residual amounts of nitrate leached by autumn–winter rains can cause the amounts of nitrate in the spring to vary between 9 and 100% of the autumn levels in the top meter of the soil. This casts further doubt on the belief that autumn and winter changes in the min-N supply in the soil can be predicted by amounts of N remaining after harvest.

The spring–summer rainfall distribution clearly

Table 3. *Precipitation (mm) and min-N content in large-plot profiles (B0 and B120) (N kg ha^{-1}) during autumn, spring and the difference between spring and preceding autumn content*

Year	Precip. July–Dec.	B0 Autumn	B0 Spring	B0 Diff.	B120 Autumn	B120 Spring	B120 Diff.
1981–82	477	60	33	−27	63	35	−28
1982–83	274	29	52	+23	44	72	+28
1983–84	296	30	74	+44	43	79	+36

affected min-N content in the grass ley treatment. As mentioned previously, the N-fertilizer applied in May was usually taken up rapidly by the grass ley. However, during springs characterized by low precipitation, such as 1984 (Fig. 3), considerable amounts of min-N was left in the soil at the time of the first cut (Figs. 1 and 2). After N-fertilization of the regrowth on 2 July 1984, heavy rainfall created more favourable moisture conditions in the soil for ley growth and the content of min-N in the large-plots as well as in the small-plots decreased by ca. 110 N kg ha^{-1} within three weeks (Figs. 1 and 2). In contrast, when the period subsequent to the first cut was dry, such as during 1983, the corresponding decrease was 20–25 N kg ha^{-1}.

Influence of interseeding

The Italian ryegrass, insown in the fertilized barley greatly reduced accumulations of min-N during the autumn of 1984 (Fig. 1). It was assumed that no decrease in min-N content due to interseeding of ryegrass occurred until the end of July, by which time the barley had most likely finished taking up N. In early December, the plot with ryegrass (B120X) had 23 N kg ha^{-1} less min-N than the B120 large-plot. Reductions by as much as 82% of the min-N supply following interseeding were observed by Andersson et al. (1984). Thus, it is reasonable to assume that considerable reductions in leaching of nitrate could be obtained by interseeding. Insown crops together with spring cereals constitutes a cropping system with a capacity to immobilize N during the autumn similar to that of perennial leys.

CONCLUSIONS

Results from this study show that frequent soil sampling is needed to properly investigate the seasonal dynamics of min-N in agricultural soils, especially during periods of active N-uptake resulting in sharp decreases in min-N content during short periods of time.

The temporal variation of min-N in the soil was mainly controlled by weather conditions and plant uptake.

The N-fertilizer applications of 120 N kg ha^{-1} in barley and the ploughing of perennial leys increased the min-N content during autumn/winter to levels clearly above the unfertilized barley. Interseeding, following harvest of the barley, showed ability to reduce the accumulation of min-N in the soil.

Simple methods used to predict spring levels of min-N based on autumn measurements are usually misleading, due to fluctuations in weather conditions from year-to-year. Simple models predicting spring time levels of min-N in the soil must take meteorological data into account.

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Nitrate Leaching and Drainage from Annual and Perennial Crops in Tile-drained Plots and Lysimeters[1]

LARS BERGSTRÖM[2]

ABSTRACT

Leaching of NO_3 with drainage water from tile-drained field plots and from three types of lysimeters was estimated during a 4-yr period. Treatments included barley (*Hordeum distichum* L.) with and without N-fertilizer, a grass ley (*Festuca pratensis*), and a lucerne ley (*Medicago sativa*) (i.e., 4-yr forage crops). The maximum amount of NO_3 leached was 36 kg N ha^{-1} yr^{-1} for barley fertilized with $Ca(NO_3)_2$ (120 kg N ha^{-1} yr^{-1}). For unfertilized barley the corresponding amount was 5 kg N ha^{-1} during the same period. The NO_3 fluxes from the grass and lucerne leys were mostly below 5 kg N ha^{-1} yr^{-1}. However, after the grass ley was plowed, considerable leaching occurred, reaching 42 kg N ha^{-1} during 20 weeks following plowing. Weather conditions had a strong influence on the temporal distribution of leaching losses. Lysimeters, compared with tile-drained plots, had generally higher drainage volumes. The slow dynamics of groundwater beneath the drainage-tiles can explain most of this difference. Lysimeters with disturbed soil profiles usually had higher drainage volumes than lysimeters with undisturbed profiles. Despite these differences, all methods consistently estimated the relative differences between the cropping systems concerning leaching of NO_3. The degree of variation in drainage flow between lysimeter replicates was also satisfactorily low.

Additional Index Words: **disturbed soil, groundwater, plowing of leys.**

Bergström, L. 1987. Nitrate leaching and drainage from annual and perennial crops in tile-drained plots and lysimeters. J. Environ. Qual. 16:11–18.

Enrichment of NO_3 in surface- and groundwater has become a common and increasingly serious problem. Fertilizer N is often considered to be the most important factor contributing to elevated NO_3 levels in water, primarily due to the large surpluses of N applied. In addition, climatic conditions, soil type, and type of crop are factors that influence the magnitude of leaching losses (Avnimelech and Raveh, 1976; Baker and Johnson, 1981; Gustafson, 1983).

Under Swedish conditions, leaching of NO_3 from arable lands occurs mainly during the autumn (September–November), which is characterized by high precipitation and low evapotranspiration (Bergström and Brink, 1986). During this period levels of inorganic N in the soil are often high, and the soils are commonly bare. A cover crop may significantly immobilize this supply of soluble N (MacLean, 1977; Andersson et al., 1984), and thereby reduce NO_3 leaching.

Field measurements of leaching have been performed using a variety of methods. However, the principal differences between methods and the conditions characterizing each method are important to consider, especially when comparing quantitative results. Uniform discharge rates were obtained by Bergström and Brink (1986) when using tile-drained field plots of 0.40-ha surface area. Lysimeters of 75-m^2 surface area were used by Uhlen (1978) to determine nutrient leaching and surface runoff from a cultivated soil. Many experiments with small-scale lysimeters, both with disturbed and undisturbed soil profiles, have been reported (Cassell et al., 1974; Chichester, 1977; Cannell et al., 1980). In this study, tile-drained plots and lysimeters were compared simultaneously on the same field site as well as lysimeters with disturbed and undisturbed soil cores.

The objective of this study was to evaluate the influence of different cropping systems and different measuring methods on leaching losses of inorganic N and on drainage discharge rates.

MATERIALS AND METHODS

Experimental Field

The experimental field was situated at Kjettslinge in central Sweden, ca. 40-km north of Uppsala (Steen et al., 1984). The four cropping systems included in this investigation were:

1. B0, barley (*Hordeum distichum* L.) with no addition of N-fertilizer;
2. B120, barley with an annual N-fertilization of 120 kg N ha^{-1} [calcium nitrate, $Ca(NO_3)_2$];
3. GL, grass ley (*Festuca pratensis*) with an annual split N-fertilization of 120 + 80 kg N ha^{-1} [$Ca(NO_3)_2$]; and
4. LL, lucerne ley (*Medicago sativa*) with no N-fertilization.

Both ley treatments (GL and LL), i.e., 4-yr forage crops, were established in 1980. The grass leys were plowed in August 1984. The lucerne leys were plowed in May1984 and thereafter sown with barley and fertilized with 120 kg N ha^{-1}. The LL large-plot (see below) was also plowed and reestablished in July 1982 owing to a severe weed infestation.

The soil down to a depth of 1.0 m consisted of four distinct layers of variable texture and structure:

(i) a layer of topsoil consisting of clay loam with a mean thickness of 0.27 m;
(ii) a fine sand layer of variable thickness. The mean thicknesses for the large-plots were: 0.27 (B0), 0.17 (B120), 0.31 (GL), and 0.22 (LL). The corresponding thicknesses in the plots in which the large lysimeters (see below) were installed were: 0.35 (B0), 0.16 (B120), 0.30 (GL), and 0.26 (LL);
(iii) an oxidized clay layer down to a depth of approximately 0.75 m; and
(iv) a nonoxidized clay layer at a depth of 0.75 to 1.00 m.

The topsoil consisted of 15 to 20% clay, 22 g organic C kg^{-1}, and had a pH of 6.0 to 6.5.

Drainage Water Measurements

The leaching measurements were started in May 1980 with four tile-drained large-plots (TDP), each covering an area of 0.36 ha (Fig. 1). Each plot represented one of the cropping systems. Six tile-drains were installed at 10-m spacing in each plot at a depth of ca. 1 m. The drainage flows from the individual plots converged at a central measuring station common to all plots, and thence directed to a collecting well with a pump (Fig. 1). Water samples were collected directly from the incoming pipes.

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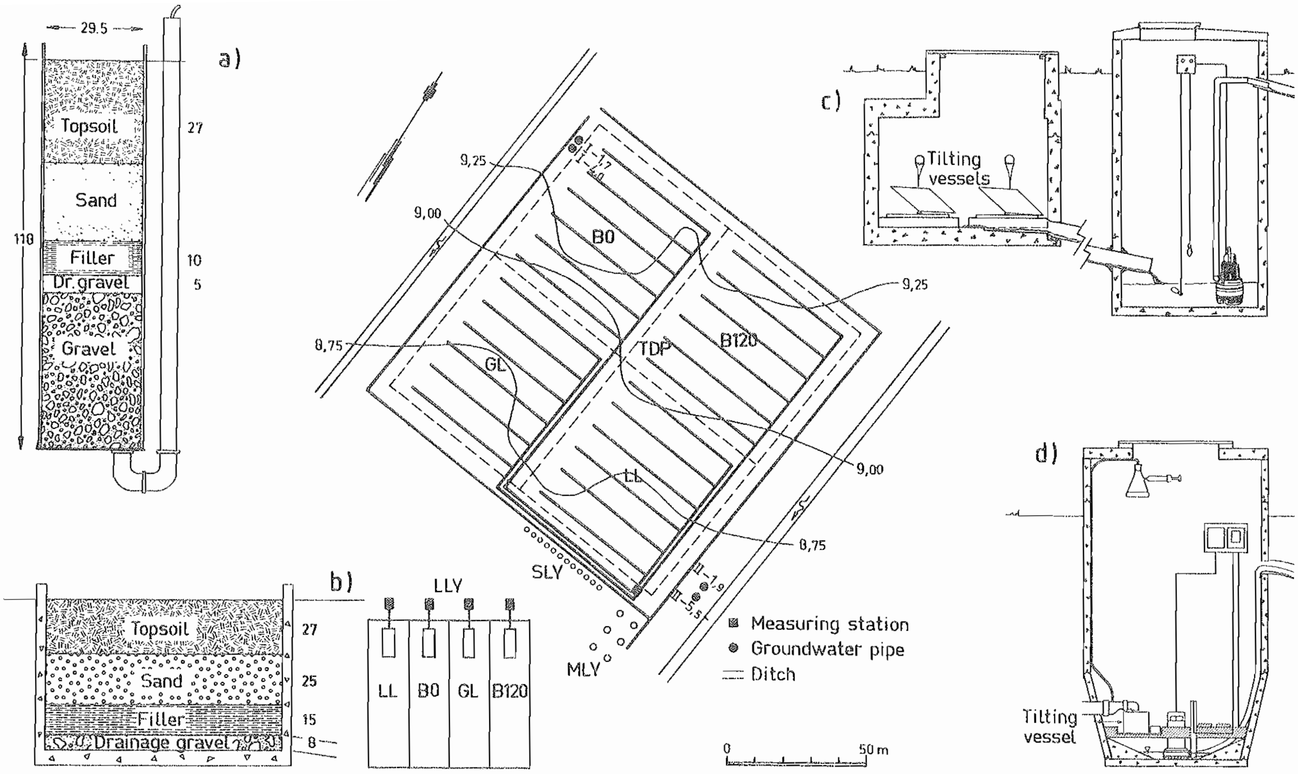

Fig. 1. Layout of the experimental field: (a) small lysimeter (SLY), (b) medium lysimeter (MLY), (c) measuring station for the large-plots (TDP), and (d) measuring stations for the large lysimeters (LLY). Dimensions of the lysimeters are given in centimeters.

In November 1979, four large lysimeters (LLY), similar to those used by Uhlen (1978), were installed, one in each cropping system. Measurements started in January 1981 after the measuring equipment was completed. These lysimeters, consisting of rubber sheets with vertical sides extending up to the topsoil, were placed in pits 9 by 3 by 1 m, i.e., covering a surface area of 27 m^2. A drainage pipe placed at the bottom of each lysimeter was connected via a plastic pipe to the individual measuring stations (Fig. 1). The soil was replaced in the lysimeters layer by layer, in accordance with the original stratification of the soil profile. Water was pumped out of the measuring stations using electric pumps controlled by electrodes. Water samples were collected with a vacuum handpump directly from the incoming pipes to avoid contamination.

The discharge rate from each individual large-plot and large lysimeter was measured with a tilting vessel (Brink, 1968). The number of emptyings was recorded by a mechanical counter, which was read three times a week.

Two additional small-scale lysimeter types were installed during the spring of 1982. Both lysimeter types contain a soil profile equivalent to the two uppermost layers of the Kjettslinge profile, the topsoil and the sand layer.

Six of the lysimeters, hereafter called medium lysimeters (MLY), had a diameter of 1.20 m and a depth of 0.75 m (Fig. 1). Four of these were assigned the GL-treatment, while the other two were assigned the B120-treatment. All six were filled with disturbed soil. A previously established grass turf was planted on each of the GL-lysimeters in April 1982. The walls and bottom of the lysimeters were made of concrete and sealed with rubber asphalt on the outside to prevent water movement through the concrete walls. A drainage pipe was placed at the bottom of each lysimeter and layers of drainage gravel (4–8 mm) and filler (0.02–0.2 mm) were placed on top of the pipe. The discharge was collected in plastic bottles located in the same pump well as that used for the large-plots. The amount of discharge was determined by weighing the bottles once a week when discharge occurred.

Small lysimeters (SLY) with undisturbed soil cores (Fig. 1) were also used in this study. Altogether, 12 lysimeters, nine with the GL-treatment and three with the B120-treatment, were installed. Three of the GL-lysimeters were never "plowed" during the experiment. The lysimeter casings had an i.d. of 0.295 m, a length of 1.18 m, and were made of polyvinyl chloride (PVC). The soil cores were collected using a drilling technique developed at the Dep. of Agricultural Hydrotechniques at the Swedish Univ. of Agricultural Sciences (L. Persson, 1982, personal communication). The drill consists of a steel cylinder with a sharp edge into which the empty plastic lysimeter casing is inserted. The drill-cylinder rotates around the plastic casing and carves out a soil core that is gently pushed into the casing. This technique allows less disturbance of the soil core than pressing down the casings. Before fitting the bottom plates of the lysimeters, layers of filler (0.02–0.2 mm), drainage gravel (4–8 mm), and gravel (8–12 mm) were placed between the subsoil and the bottom. The depth of the soil profile above the filler layer varied between 0.53 and 0.59 m. The variation was due to the actual variation in thickness of the sand layer occurring in the field where the soil cores were taken. In each bottom plate two holes were drilled and provided with plastic pipes through which water was pumped out with a vacuum handpump. The lysimeters were pumped dry weekly and the amount of discharge was determined by weighing the samples.

A surface pool of melted snow, which normally causes surface runoff, was pumped away from the medium and small lysimeters to better simulate field conditions. This was done once each spring (1983 and 1984) while there was still a thin layer of ice on the lysimeters.

Samples of drainage water from the large-plots and from all lysimeters were collected once a week when discharge occurred. All samples were immediately analyzed for NH_4, NO_3, NO_2, and total N content. Ammonium was analyzed with the phenate method (APHA, 1985). Nitrate was analyzed with the colorimetric Cd-reduction method (APHA, 1985); therefore, concentration data included NO_2. Nitrite concentrations were determined by excluding the Cd-column. To determine the total N concentrations, NH_4, NO_2, and organic N were oxidized by $K_2S_2O_8$ + NaOH to NO_3, which was analyzed as previously described. All analytical methods were

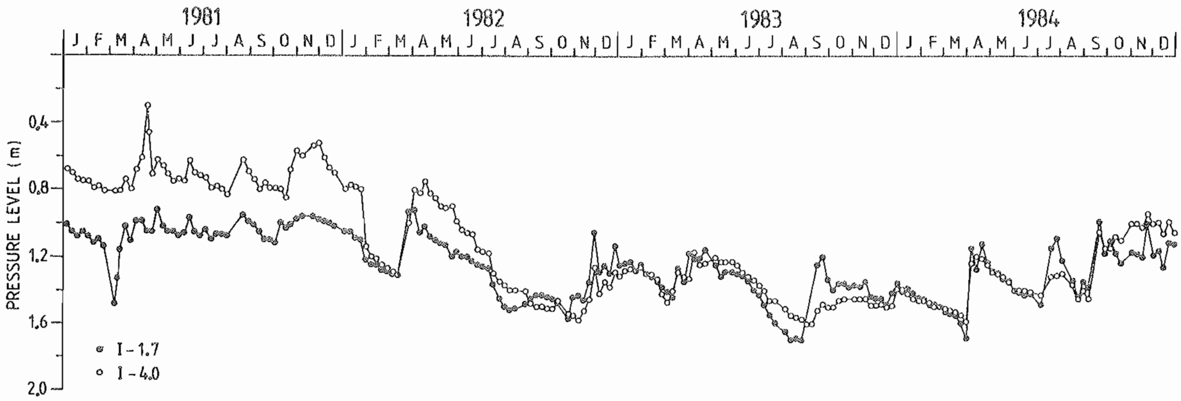

Fig. 2. Groundwater pressure represented by the pressure levels at Site I at two depths, 1.7 and 4.0 m, below the soil surface.

adjusted for an auto-analyzer. After 1 July 1983, only NO_3 was considered.

Groundwater Measurements

Groundwater pipes were installed at three sites in June 1980. They functioned as both sampling pipes and piezometers. Groundwater flowed into each pipe either through the open bottom or through a filter in the pipe wall. The depth was measured to the bottom of the pipe or to the upper edge of the filter. The length of the filter varied between 0.2 and 0.5 m. Each of the three sites was supplied with two pipes, one ending in the sedimentary layers at a depth of ca. 2 m and the other reaching the underlying moraine, ca. 5 m below the soil surface. Piezometer readings were performed with a plumb bob once a week. Groundwater samples for chemical analyses of N compounds (see above) were collected monthly with a vacuum handpump from September 1981 until June 1983.

RESULTS AND DISCUSSION

Groundwater Conditions

All three measurement sites were commonly characterized by an upward-directed gradient of groundwater pressures (Fig. 2), indicating that the field is part of a larger discharge area. It was clear that this up-pressure occasionally contributed substantially to the drainage discharge in certain parts of the field. During 1981, a strong up-pressure was observed at Site I (Fig. 2), located at the B0 large-plot; simultaneously, this plot had a 49% higher drainage volume compared with the B120 large-plot (Fig. 3). Although Jansson and Thoms (1986) estimated a difference of 30 mm in evapotranspiration between the two treatments during the 1981 growing season, an inflow of groundwater to the drainage-tiles must be considered. However, since the NO_3 concentrations in the groundwater were fairly low and stable (Table 1), only small contributions of NO_3 to the tile-effluent were to be expected. A drastic drop in the pressure level occurred at Site I in early 1982 (Fig. 2); this drop was mostly due to an increase in depth of the surrounding open ditches by ca. 0.5 m.

Pressure levels showed typical seasonal fluctuations, with peaks during the snowmelt periods and during autumn periods with high precipitation (Fig. 2). During dry periods, such as autumn 1982 and summer 1983, pressure levels dropped considerably and the continuous up-pressure was converted to down-pressure at all sites (Fig. 2), indicating that a downward flow occurred below the drainage-tiles.

Drainage Discharge—Weather and Crop Influence

Precipitation measured at Kjettslinge varied greatly during the experimental period (Fig. 3), ranging from considerably below the long-term average of 520 mm for the region (1982) to above average during 1981 and 1984 (Alvenäs et al., 1986). The distribution of rainfall over the year greatly influenced the timing of the discharges and their quantities (Fig. 3). Heavy rains during the autumn generally resulted in discharge peaks, as happened during 1981 and 1984. Rainfall during late spring or summer, when a soil-water deficit normally existed, had far less direct influence on the amounts of discharge. During the autumn of 1983 (September–December), low drainage flows ocurred, even though the amount of precipitation was almost as high (259 mm) as during the corresponding months in 1981 and 1984 (297 and 293 mm, respectively). The low amount of precipitation during the period preceding September 1983 caused a water deficit in the soil (Alvenäs et al., 1986) to the extent that most of the autumn precipitation was retained in the soil. Low drainage volumes were also favored by the frequent, moderate rains that prevented rapid flows through macropores from occurring.

Pronounced spring peaks in drainage flow, which are common following the snow melting period in central Sweden (Gustafson, 1983), occurred only during 1982 (Fig. 3). This peak was mainly due to a large accumulation of snow on the field in combination with a thin layer of soil frost (Alvenäs et al., 1986). Almost all of the melted snow percolated through the soil profile. During the other 3 yr, the ground remained frozen until relatively late, and most of the melted snow formed surface runoff in the surrounding, open ditches.

The influence of crop type on drainage volumes was considerable and mainly depended on the degree to which the soil-water deficit developed during the growing season. In general, the barley treatments (B0 and B120) had considerably higher drainage volumes than the treatments with perennial leys (GL and LL). This was obvious during 1982 when the barley large-plots discharged approximately twice as much drainage water as the corresponding ley plots (Fig. 3). Much of this difference

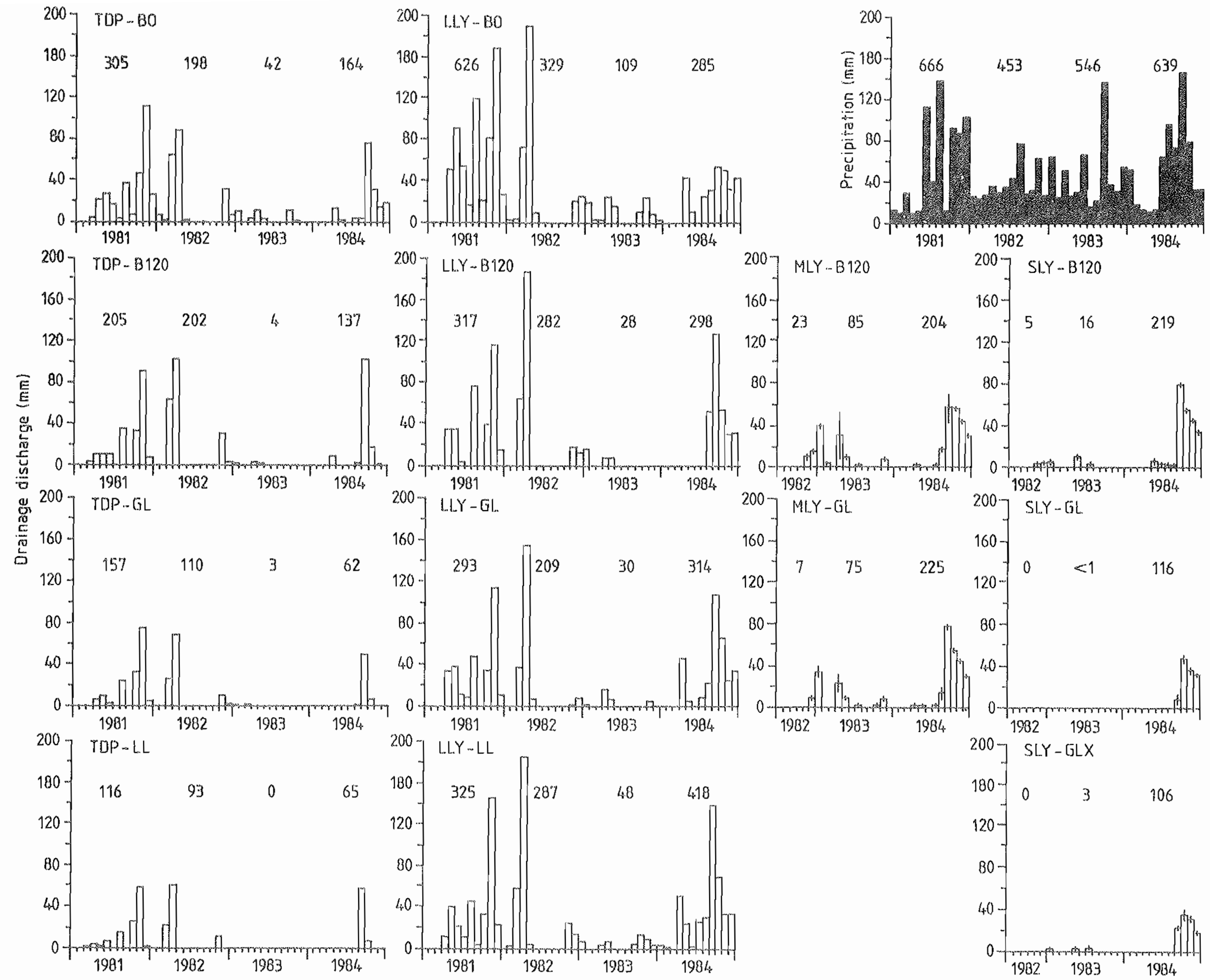

Fig. 3. Monthly precipitation (upper-right diagram) and drainage discharges from the large-plots (TDP) and from the various lysimeters (LLY, MLY, and SLY). The annual amounts of precipitation and discharges are shown in each diagram. Bars represent standard error.

was due to the extended growing season and the consequent uptake of water early and late in the season, which is typical for leys. Thies et al. (1978) found that the drainage volume from bare lysimeters was 26% higher than from lysimeters with a crop. Slightly higher drainage volumes from bare than from cropped soils were also noted by Hoyt et al. (1977). The fertilization intensity of the barley and the type of ley also had a certain, although less pronounced, influence on the amount of discharge.

Table 1. Estimates of N in groundwater at two depths for each of three sites. The concentrations are mean values for the period January 1981 to June 1983.

Site†	Pipe depth	NH_4-N	NO_3-N	Total N
	m		mg L^{-1}	
I	1.7	0.025	0.73	1.72
I	4.0	0.019	2.44	2.84
II	1.9	0.033	0.49	0.94
II	5.5	0.020	0.03	0.26
III	2.2	0.019	2.51	2.86
III	4.8	0.061	0.12	0.43

† Locations of Sites I and II are given in Fig. 1. Site III is located ca. 250-m southeast of the tile-drained large-plots (TDP).

The lower growth rate of the unfertilized barley (B0) and the associated lower transpiration rates commonly caused lower water tensions in the soil compared with the fertilized barley (B120) (Jansson and Thoms, 1986). Consequently, drainage volumes produced by the B0-treatment were often higher than those of the B120-treatment, such as during 1983 (Fig. 3).

Drainage Discharge—Influence of Measuring Method

Amounts of drainage discharge were compared between the tile-drained large-plots (TDP) and the large lysimeters (LLY) and between the medium lysimeters (MLY) with disturbed soil and the small lysimeters (SLY) with undisturbed soil. Since the medium and small lysimeters did not contain the clay layers, a comparison of all four measuring methods would have been less relevant.

Although the large-plots and the large lysimeters showed similar temporal patterns in variation when the drainage discharges were compared (Fig. 3), the lysimeters had considerably higher accumulated drainage volumes

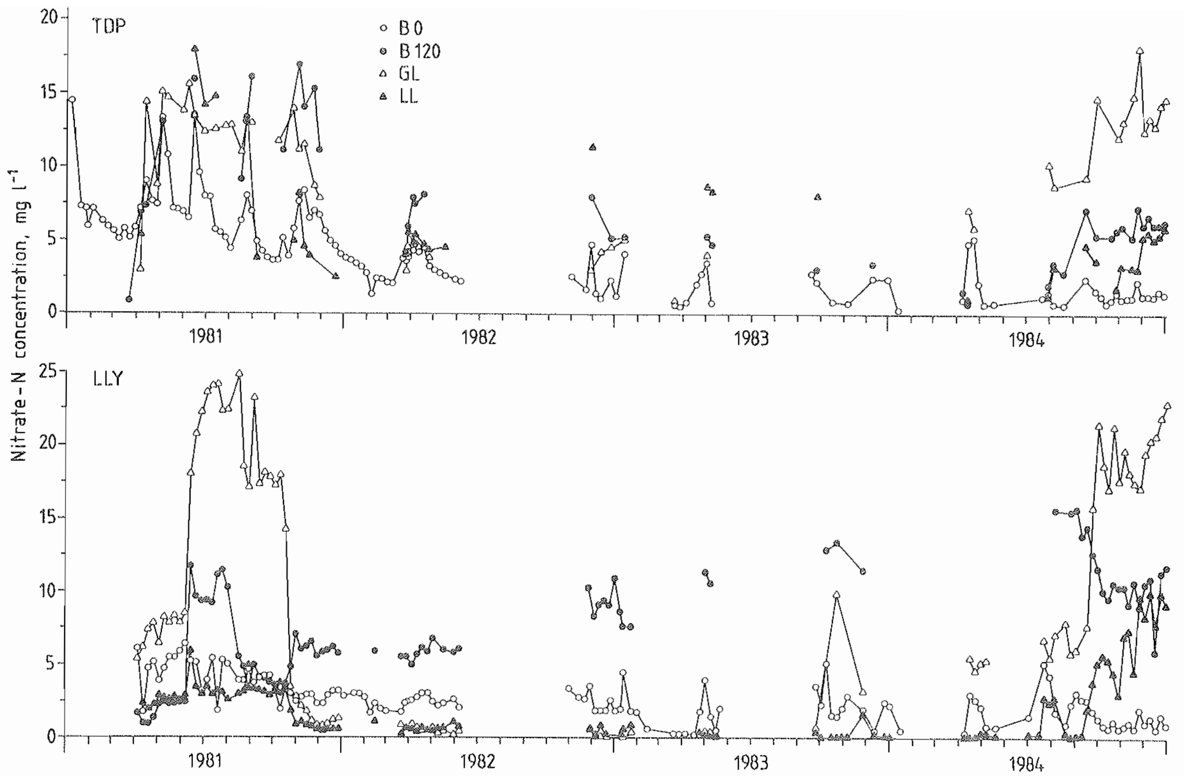

Fig. 4. Nitrate-N concentrations in drainage water collected from the large-plots (TDP) and from the large lysimeters (LLY).

each year than the corresponding large-plots. Much of this difference arose because all percolating water reaching the bottom of a lysimeter is measured, while only part of the inflow water is usually captured by a drainage system under natural conditions. The amount of discharge from tile-drains depends on, for example, groundwater level and possible lateral water movements. The low hydraulic conductivity in the clay limited the percolation rate substantially. This indicates that during periods of high precipitation, lateral water movements commonly occurred in the topsoil and sand layers with higher conductivity. The unreduced crop growth on the B120 large-plot during dry periods (R. Pettersson, unpublished data) suggests that water contributions from the subsoil also must be considered. Measurements of the soil-water tension and calculations of water flows indicated that a capillary rise occurred during the growing season (Jansson and Thoms, 1986). Such conditions also require that groundwater storage in the subsoil be restored following dry periods, resulting in substantially reduced tile-drainage flows. Hood (1977) found that only 20% of the rainfall was recovered from tile-drains under field conditions compared with 38% for lysimeters. Thomas and Barfield (1974) showed in two measurements that 11 and 37% of the total flow to an open ditch originated from tile-effluent. This suggests that measurements of tile-drainage volume in a field can give an unreliable picture of the total amount of water that actually moves in the profile. Drainage volumes measured from closed systems such as lysimeters may, in contrast, represent overestimations of actual drainage volumes in a field situation. Extrapolation of results from lysimeters to a field would therefore be inaccurate. Consequently, quantitative measurements of water transport in soils should be viewed with caution.

Drainage volumes collected from the medium lysimeters with a disturbed soil profile were usually higher than those collected from the small lysimeters with an undisturbed profile, especially during 1982 and 1983, when low drainage flows occurred (Fig. 3). During wet periods, such as the autumn of 1984, drainage water was collected from the medium lysimeters earlier than from the small lysimeters (Fig. 3), indicating faster water movement through the disturbed soil. This more rapid water movement can either be due to a higher hydraulic conductivity or a less developed water deficit prior to the infiltration. The difference in drainage volumes between the GL- and B120-treatments was greater in the small compared with the medium lysimeters (Fig. 3). This difference also remained in the small lysimeters after the grass leys were "plowed" (Fig. 3). Thus, considerable changes in unsaturated flow were caused by disrupting the soil during installation. Filling lysimeters with disturbed soil is certainly easier, but noticeable physical changes can be expected with uncertain consequences for water movement and root development. In a lysimeter experiment Cassel et al. (1974) found that disturbed soils retained more water than undisturbed soils for a wide range of soil-water pressures, indicating that less water would be expected to move through a disturbed profile than through an undisturbed profile, in contrast to the results from this study. However, it should be noted that the greatest relative differences in drainage volumes between the small and medium lysimeters occurred during periods when drainage volumes were low in all treatments. Thus, these results should be interpreted with great caution. Once

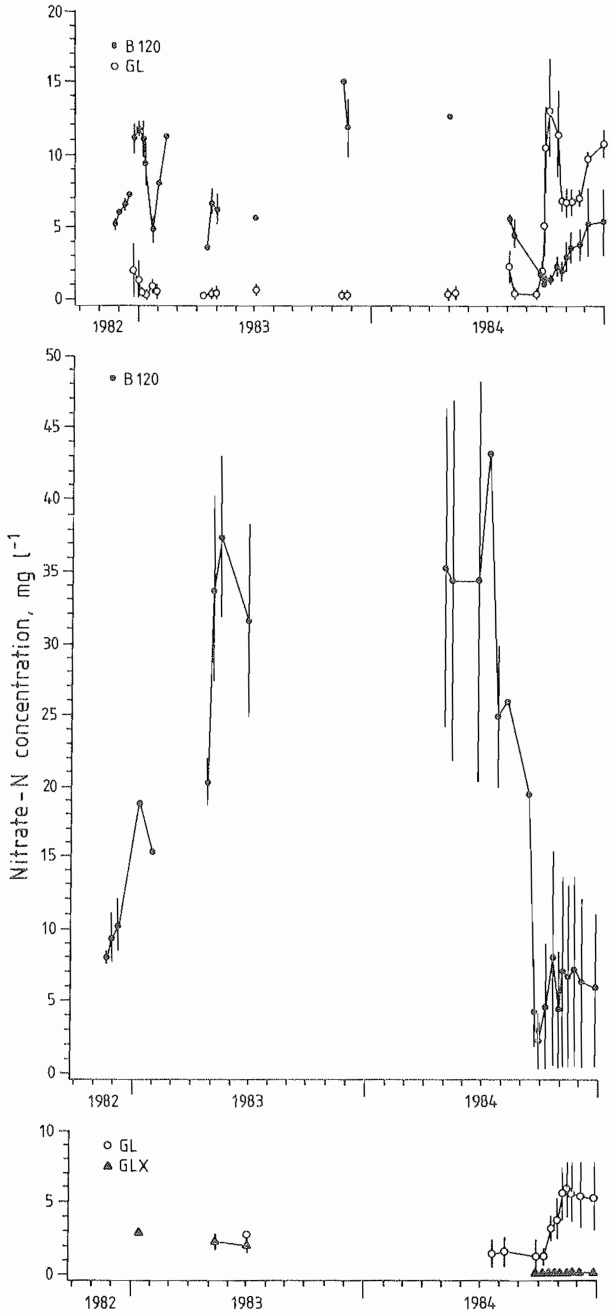

Fig. 5. Nitrate-N concentrations in drainage water collected from the medium lysimeters (MLY) (upper diagram) and the small lysimeters (SLY) (lower two diagrams). Bars represent standard error.

more frequent drainage flows started in the autumn of 1984, there was better agreement in drainage volumes between disturbed and undisturbed soil.

Plant growth in lysimeters in general and growth in disturbed and undisturbed soils may not be typical of growth in comparable fields for several reasons: root proliferation, aeration, temperature, and moisture conditions may be different in the middle of a lysimeter than adjacent to the lysimeter wall. These effects on drainage volumes are difficult to quantify, and such quantification was never attempted in this study.

Concentrations of NO_3 in Drainage Water

The overwhelming part of the N in drainage water occurred as NO_3, as is common from agricultural lands. On no occasion did the NH_4 or NO_2 concentrations exceed 0.1 mg N L^{-1}; neither did they seem to be affected by the different cropping systems. Consequently, the analyses of NH_4 and NO_2 were terminated on 1 July 1983 and are not discussed further.

Clear differences existed among the four cropping systems regarding the NO_3 concentrations and the seasonal variations of NO_3 in drainage water (Fig. 4). These differences were also apparent when studying the NO_3 content of the soil (Bergström, 1986). During 1981, the large lysimeters were still affected by previous installation work, which caused great variability in the NO_3 concentrations. During the following 2 yr, the NO_3 levels in water draining from these lysimeters were ordered hierarchically according to the cropping systems; their average NO_3 concentrations were 2.1 (B0), 8.0 (B120), 0.9 (GL), and 0.4 (LL) mg N L^{-1}, respectively. During 1982 and 1983, higher NO_3 concentrations were usually found in the drainage waters from the GL and LL large-plots compared with the corresponding large lysimeters. However, these values corresponded to overall low drainage volumes, especially during 1983 (Fig. 3) and are therefore fairly uncertain. The elevated NO_3 concentrations in the drainage water from the LL large-plot were also an effect of the plowing and reestablishment of the lucerne on this plot during July 1982.

The great difference in NO_3 concentrations between the B120- and the GL-treatments was further confirmed by results from the medium and small lysimeters (Fig. 5). The average NO_3 concentrations in drainage water for the B120-treatment were 7.9 (MLY) and 25.8 (SLY) mg N L^{-1}; for the GL-treatment average concentrations were 0.7 (MLY) and 2.2 (SLY) mg N L^{-1}. These figures represent the entire experimental period before the grass leys were plowed. The commonly lower drainage volumes and, therefore, less dilution of the drainage water from the small lysimeters compared with the medium lysimeters caused higher NO_3 concentrations in the former. However, the extremely low drainage volumes from the small lysimeters prior to September 1984 (Fig. 3) also made interpretation of the NO_3 concentrations uncertain in this situation. There were large differences in NO_3 concentrations between the SLY B120-replicates (Fig. 5). These differences were mainly due to variations in crop yields from these lysimeters (R. Pettersson, unpublished data). Differences up to 50% in yield among lysimeters were measured.

The stable conditions characterized by low NO_3 levels in the drainage water changed abruptly once the grass leys were plowed in August 1984. From levels commonly <1 mg N L^{-1}, the NO_3 concentrations increased to 10 to 20 mg N L^{-1} within a few months (Fig. 4 and 5). Except for the small lysimeters, the NO_3 concentrations in the drainage water from the GL-treatment even exceeded the concentrations of the corresponding B120-treatment. Considerable increases in NO_3 content of the soil due to autumn mineralization of ley crop residues also occurred (Bergström, 1986). Cameron and Wild (1984) found that about 100 kg N ha^{-1} of NO_3 had leached down to

Table 2. Transport of NO_3 by drainage water measured in full-scale, tile-drained plots (TDP) and with three types of lysimeters (LLY, MLY, and SLY). The estimates for the medium and small lysimeters (MLY and SLY) are mean values (±SE) for the replicates.

Method	Treatment	1981	1982	1983	1984
		kg N ha^{-1} yr^{-1}			
Large-plots (TDP)	B0	22.5	7.4	1.0	2.8
	B120	26.9	13.7	0.2	7.6
	GL	17.4	4.6	0.2	7.1
	LL	8.2	5.6	0.0	2.4†
Large lysimeters (LLY)	B0	22.9	8.1	2.2	4.8
	B120	14.8	16.2	2.8	36.4
	GL	23.4	1.4	0.4	44.9
	LL	6.1	1.6	0.1	14.4†
Medium lysimeters (MLY)	B120 (n = 2)	--	1.8 (±0.4)§	6.6 (±1.0)	6.0 (±1.7)
	GL (n = 4)	--	0.1 (±0.1)§	0.2 (±0.1)	15.4 (±2.0)
Small lysimeters (SLY)	B120 (n = 3)	--	0.5 (±0.3)§	4.4 (±1.4)	13.4 (±9.5)
	GL (n = 6)	--	0§	<0.1 (< ±0.1)	5.0 (±1.3)
	GLX‡ (n = 3)	--	0§	<0.1 (< ±0.1)	<0.1 (< ±0.1)

† Changed to the B120-treatment during May 1984.
‡ GLX = grass ley that remained unplowed during the experiment.
§ Values refer to the period 1 July to 31 Dec. 1982.

levels below 0.9 m over two winters as a result of plowing grassland, further supporting the belief that a large increase in mineralization is to be expected after plowing a ley.

Even though each cropping system apparently reached a state of balance after 1981, some seasonal fluctuations in NO_3 concentrations and fluctuations due to weather conditions still occurred (Fig. 4). The B0-treatment showed the greatest temporal variation in NO_3 concentrations of the drainage water. Some response to discharge peaks was observed. This response was characterized by elevated NO_3 concentrations during autumn and spring, although the drainage volumes during spring were mostly moderate. The spring peaks indicated that some mineralization activity also occurred during winter.

Fluxes of NO_3 in Drainage Water

The fluxes of NO_3 depend on drainage volume and NO_3 concentration; thus much of the previous discussion also applies to fluxes. Since the NO_3 concentraitons of drainage water were fairly stable, with characteristic levels for each cropping system, the variations in yearly fluxes mostly reflected varying drainage volumes. There are several other investigations showing a similarly strong dependence between mass emission and drainage volume (Bolton et al., 1970; Letey et al., 1977).

In general, the B120-treatment had the highest NO_3 losses, followed by the B0-treatment, while the GL- and LL-treatments had the lowest leaching losses (Table 2). These differences arose mainly because annual crops like barley usually leave considerable amounts of inorganic N in the soil at harvest in addition to the mineralization occurring during the autumn. These sources of N are usually very susceptible to leaching (Bergström and Brink, 1986). Several leaching studies comparing different cropping systems have come to a similar conclusion. Once a perennial ley is established, there is reason to expect a minimum of NO_3 leaching (Bolton et al., 1970; Kolenbrander, 1981; Gustafson, 1983). High losses occurred after the grass leys were plowed in August 1984 (Table 2). As much as 42 kg N ha^{-1} of NO_3 was leached during a period of about 20 weeks from the large lysimeters. High losses were also registered using the other measuring methods (TDP, MLY, and SLY) (Table 2).

CONCLUSIONS

This study shows that considerable amounts of water may percolate past the drainage system in tile-drained plots. Part of this water was used to compensate for the deficit caused by the upward capillary movement of water or by a slow lateral groundwater flow. Thus drainage volumes were much higher in lysimeters than in tile-drained plots.

There were noticeable differences in leaching loss estimates between the various measuring techniques, i.e., tile-drained plots vs. lysimeters, as well as between undisturbed and disturbed soils. However, the consistency that characterized each individual method makes each of them very useful for comparative studies of soil leaching.

This study suggests that even moderate N-fertilization rates applied to annual grain crops in central Sweden represent a risk for surface water pollution. Perennial leys acted more or less as optimal catch crops, mostly due to their extended growing season. Leaching rates of NO_3 from the grass and lucerne leys included in this study were even lower than the corresponding values for the unfertilized barley crop. However, plowing of the grass leys was followed by substantial leaching of NO_3. To minimize these losses it is recommended that winter rather than spring crops be sown as soon as possible after plowing of leys or that leys be plowed in the spring.

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Leaching of 15-N-Labeled Nitrate Fertilizer Applied to Barley and a Grass Ley

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Bergström, L. (Swedish University of Agricultural Sciences, Division Water Management, Department of Soil Sciences, P.O.B. 7072, S-750 07 Uppsala, Sweden). Leaching of 15-N-labeled nitrate fertilizer applied to barley and a grass ley. Received Sept. 24, 1986. Acta Agric. Scand. 37: 199–206, 1987.

A lysimeter experiment was conducted to determine the magnitude of fertilizer-N leaching over a three-year period. Three types of lysimeters and two treatments were included, i.e., barley and a grass ley fertilized with calcium nitrate at rates of 120 and 120+80 kg N ha^{-1} yr^{-1}, respectively. The N-fertilizer applied during the first year had 15-N enrichments ranging from ca. 1 to 99% atom excess (a.e.). 15-N-labeled fertilizer (ca. 99% 15-N a.e.) was also applied to three grass ley lysimeters (not fertilized with 15-N earlier) during the third year. Approximately 10% of the nitrate leached from lysimeters with barley during the first year originated from fertilizer. Dry weather during the first two years reduced N-leaching substantially. Leaching of 15-N-labeled fertilizer never exceeded 1 kg N ha^{-1} yr^{-1} from either the barley or the grass ley lysimeters. A maximum of 1.2% of the 15-N-labeled fertilizer applied to the barley was recovered in drainage water during the three years of the study. The grass ley generally had lower proportions of nitrate-N derived from fertilizer in drainage water compared with barley. The amounts of fertilizer-N appearing in drainage water were similar for each type of lysimeter and each level of 15-N enrichment. *Key word: Lysimeters.*

INTRODUCTION

Contamination of surface water and groundwater supplies by N-fertilizers has received considerable attention in recent years. A pronounced impact on drainage water quality due to excessive N-fertilization on arable lands has been found in several studies, both with industrial fertilizers and with manure (e.g., Baker & Johnson, 1981; Vetter & Steffens, 1981; Bergström & Brink, 1986). More quantitative information on the fate of fertilizers can be obtained with tracer methods. Use of 15-N-labeled fertilizer enables discrimination between fertilizer nitrogen and nitrogen liberated during decomposition of organic matter. Tracer methods using 15-N can also be used to determine temporal patterns and rates at which applied fertilizer is immobilized, and the influence of weather conditions, which are of importance for N-leaching.

Chichester & Smith (1978) found that ca. 30% of the 15-N-labeled fertilizer applied to lysimeters continuously cropped to corn was recoverd in percolation during the three years following fertilization. The greatest N-loss from these lysimeters was by leaching. Considerably less nitrogen, ca. 5% of a single application of N-fertilizer, was recovered in drainage water over a three-year period after fertilization in a Danish investigation with barley as the main crop (Kjellerup & Kofoed, 1983). The amounts of yearly discharge were similar in both investigations. Although many other factors were not equivalent, the great difference in leaching of N-fertilizer in these experiments reflects the complex nature of nitrogen transformations in soil-water-plant systems.

No quantitative data on fertilizer-N leaching from agricultural soils in Sweden have been published previously. Thus the objective of the present study was to determine amounts of

N-fertilizer in drainage water under field conditions. Three types of lysimeters, receiving N-fertilizer with different degrees of 15-N enrichment, were used for this purpose.

MATERIAL AND METHODS

Experimental field and design

The experimental field was situated at Kjettslinge in central Sweden, ca. 40 km north of Uppsala (Steen et al., 1984). The crops included in the present investigation were:

1. B 120, barley (*Hordeum distichum* L.) with an annual N-fertilization of 120 kg N ha^{-1}; and
2. GL, grass ley (*Festuca pratensis*) with an annual split N-fertilization of 120+80 kg N ha^{-1}.

Three types of lysimeters were used:

(i) Two large lysimeters (LLY), 9×3 m and with a depth of 1 m. One was assigned the B 120-treatment and the other the GL-treatment.

(ii) Six medium-sized lysimeters (MLY) with a diameter of 1.20 m and a depth of 0.75 m. Two were assigned the B 120-treatment and four the GL-treatment.

(iii) Nine small-sized lysimeters (SLY) with a diameter of 0.295 m and a depth of 1.18 m. Three were assigned the B 120-treatment and six the GL-treatment. During 1984, only one of the B 120 small lysimeters was used.

All lysimeters were designed for measuring drainage discharge. More detailed information on their construction and on the soil profiles has been presented by Bergström (1987, in press). Nitrogen was applied at 120 kg N ha^{-1} as calcium nitrate to all lysimeters in May 1982. In June 1982, a second dose of fertilizer (80 kg N ha^{-1}) was applied to the GL-lysimeters. The fertilizer was labeled with 15-N according to Table 1. Various levels of 15-N enrichment were used, depending on the type of experiment, other than leaching measurements, that was planned for each lysimeter, e.g., the high 15-N enrichments used in the small lysimeters (Table 1) were prompted by denitrification measurements. The 15-N-enriched fertilizer was dissolved in water and sprayed on to each lysimeter. The same fertilization rates were used during 1983 and 1984. No lysimeters received any 15-N-enriched fertilizer during 1983 and 1984 except for three of the GL small lysimeters, which received 15-N-enriched fertilizer along with the second fertilization in June 1984, hereafter called the GLY-treatment (Table 1).

The B 120 large lysimeter was ploughed in October each year. The GL large lysimeter was ploughed in August 1984. Simultaneously, the small- and medium-sized lysimeters were spaded to a depth of ca. 0.25 m, a practice simulating conventional ploughing.

Sampling and chemical analyses

The amount of drainage discharge was determined three times a week (LLY) and once weekly (MLY and SLY).

Drainage water for chemical analyses was collected weekly, starting immediately after fertilization in May 1982, when discharge occurred. All samples were immediately analysed for nitrate with the colorimetric Cd-reduction method (APHA, 1985); concentration data therefore included nitrite. The analytical method was adjusted for an auto-analyser.

Preparation for 15-N analysis was performed as follows: samples were evaporated to a final volume of 300 ml, heat-distilled in the presence of MgO and Devarda's alloy to reduce nitrate to ammonium, and then titrated. Ammonium was not determined separately since concentrations were overall very low, never exceeding 0.1 mg N l^{-1}. All quoted N-values refer to nitrate-N. When the amounts of nitrate were less than 0.7 mg N in the sample, 0.5 mg unlabeled ammonium-N was added to allow for isotope analysis. No samples with a nitrate concentration less than 0.1 mg N l^{-1} were analysed for 15-N. Also, certain samples were not analyzed for 15-N during the third year, when 15-N concentrations were low and stable, and at other times when drainage flows were extremely low. Determination of 15-N concentrations in water draining from the large lysimeters was terminated in Dec. 1983, when levels of 15-N-enriched fertilizer in the drainage water became too low to detect.

The isotopic composition of the titrated samples was determined with a mass spectrometer (Micromass 602) at the Department of Radioecology, Swedish Univ. of Agric. Sci.

The fraction of nitrate-N in drainage water derived from the 15-N-enriched fertilizer (NdfF) can be expressed by the equation:

$$\mathrm{NdfF} = \frac{\%\ \text{15-N a.e. in water sample}}{\%\ \text{15-N a.e. in fertilizer}}$$

RESULTS AND DISCUSSION

Weather conditions during the three years of the study clearly affected leaching of fertilizer-N. Low amounts of rainfall during the 1982 and 1983 growing seasons (Alvenäs et al., 1986) resulted in generally low drainage flows and leaching losses (Bergström, 1987, in press). In contrast, high drainage volumes occurred during 1984, which was a wet year. The first water draining from any of the lysimeters after fertilization with the 15-N-enriched fertilizer in May 1982 was obtained in Nov. 1982. Thus N-fertilizer was available for plant uptake and immobilization for several months without being transported down below the root zone. Relatively high soil water tensions in the topsoil during the 1982 growing season (Alvenäs et al., 1986) indicated that mineralization activity was markedly reduced (cf. Stanford & Epstein, 1974). This suggests that compared with a wet year, proportionally more of the total N needed by the crop had to come from fertilizer to provide the crops with sufficient N. The above-ground plant biomass of the barley was not reduced during 1982 compared with 1983 and 1984 (Pettersson, R., pers. comm.). Approximately 63% of the fertilizer-N applied in May 1982 to the medium-sized lysimeters with the B 120-treatment was recovered in above-ground plant biomass during the first year (Lindberg et al., unpublished), leaving relatively low amounts of fertilizer-N available for leaching later during autumn. Considerably lower fertilizer-use efficiencies were reported by Kowalenko & Cameron (1977) and Chichester & Smith (1978); i.e., about 27 and 23%, respectively, of the fertilizer-N was recovered from the first crop following fertilization.

The nitrate-N in water draining from the medium, small, and B 120 large lysimeters derived from fertilizer (NdfF) applied during the first year never exceeded 17% of the total

Table 1. *15-N enrichments (% a.e.) of applied N-fertilizer (calcium nitrate)*

	Large lys. (LLY)		Medium lys. (MLY)			Small lys. (SLY)		
Date	B 120	GL	B 120	GL	GLX[a]	B 120	GL	GLY[b]
May 1982	1.40	0.88	29.70	29.70	–	98.65	98.65	–
June 1982	–	1.01	–	–	18.24	–	–	–
June 1984	–	–	–	–	–	–	–	98.95

[a] Grass ley receiving 15-N-enriched fertilizer at the second fertilization in June 1982.
[b] Grass ley receiving 15-N-enriched fertilizer at the second fertilization in June 1984.

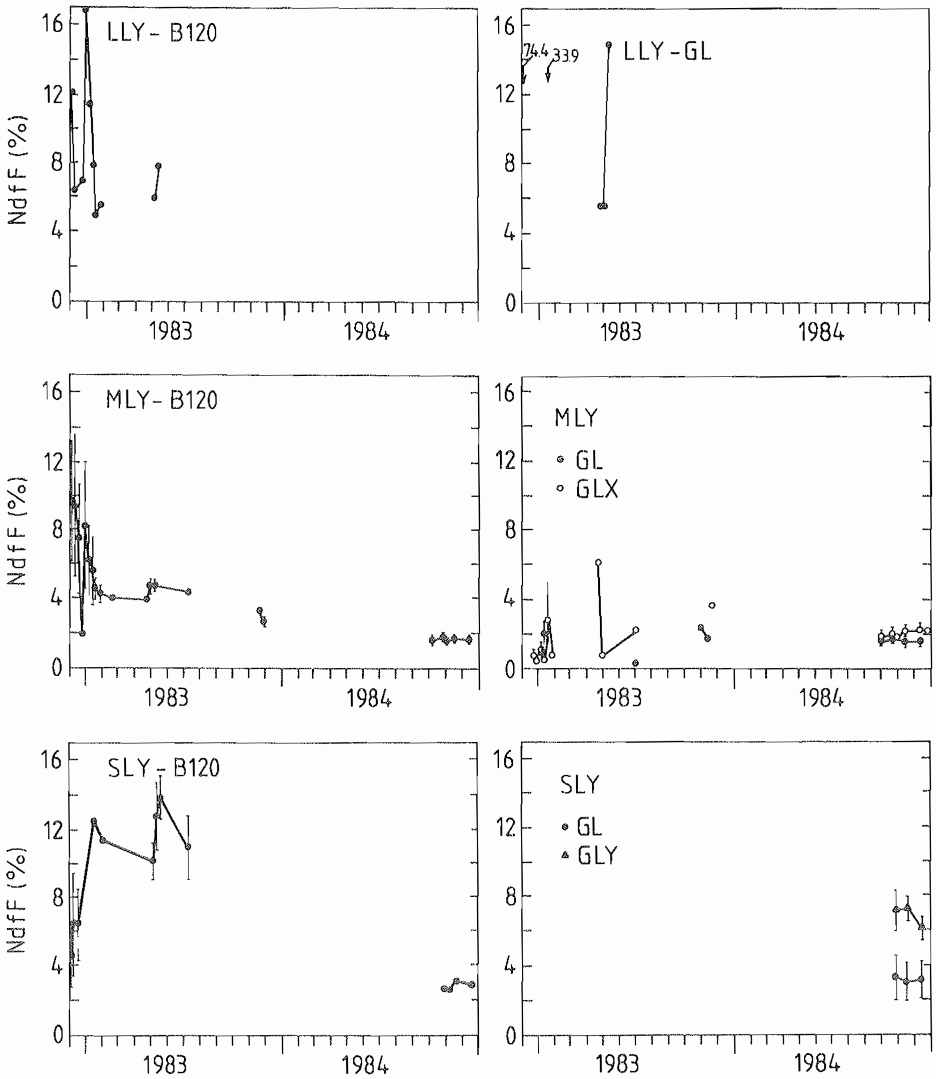

Fig. 1. Nitrate-N derived from 15-N-labeled fertilizer in water draining from the different lysimeters (LLY, MLY and SLY), expressed in percent of total nitrate-N content (% NdfF). Bars represent standard error. Sampling occasions when NdfF-values of the GL large lysimeter reached 74.4 and 39.9 percent are marked with arrows.

nitrate-N content (Fig. 1). Higher proportions of fertilizer-N were found leaching from the GL large lysimeter (Fig. 1). Large differences in NdfF between the B 120- and GL-treatments occurred during the first and second years. Except for the large lysimeters, barley usually had higher proportions of nitrate-N derived from fertilizer in drainage water compared with the grass ley. This difference reflects the relatively lower fertilizer-use efficiency of the barley (Bergström, 1986). Comparatively large fluctuations in NdfF occurred during the first two years (Fig. 1). During the third year after 15-N application, both crops had similarly low and stable NdfF-values, never exceeding 5% of the total nitrate-N content (Fig. 1). These low and stable NdfF-values in drainage water were

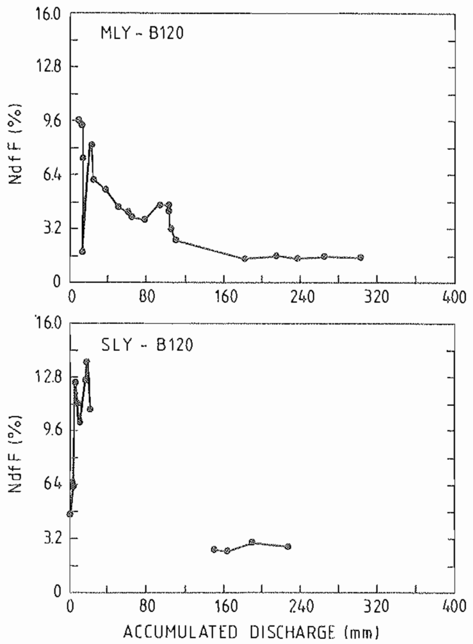

Fig. 2. The influence of drainage volume on the percentage of 15-N-labeled nitrate-N (% NdfF) in water draining from the B 120 medium and small lysimeters. Values of % NdfF and accumulated discharge represent means of replicates for each lysimeter type. Standard errors of the NdfF-values correspond to those shown in Fig. 1.

reached in the B 120 medium and small lysimeters after ca. 100 mm of accumulated drainage discharge (Fig. 2).

Addition of 15-N-enriched fertilizer to the small lysimeters with grass ley (GLY) in June 1984 doubled the NdfF-values in drainage water during autumn 1984 as compared with those from the GL small lysimeters with 15-N applied in May 1982 (Fig. 1). 15-N application to the grass ley regrowth (GLX) in the medium-sized lysimeters in June 1982 also increased NdfF in drainage to values above those of the corresponding GL-lysimeters receiving the earlier 15-N fertilization (Fig. 1). No marked changes in 15-N levels in drainage water were observed following ploughing of the grass leys.

All three types of B 120-lysimeters showed similar temporal patterns in variation when their NdfF-values were compared (Fig. 1). The NdfF-levels were also similar, ca. 10% at the beginning and 2–3% (MLY and SLY) at the end of the experimental period. Thus 15-N enrichments from ca. 1% to 99% atom excess (a.e.) of applied N-fertilizer seem to be appropriate for discrimination of fertilizer-N in drainage water. 15-N enrichments of 5–15% a.e. have been commonly used in experiments dealing with fertilizer-N leaching (e.g., Jones et al., 1977; Chichester & Smith, 1978; Dowdell & Webster, 1980).

The great temporal variations with which the nitrogen from a single application of fertilizer appeared in drainage water during the experimental period suggests that fertilizer-N not removed at harvest is immobilized in organic N pools subject to different turnover rates. Release of fertilizer-N included in easily decomposable organic compounds should cause the NdfF in drainage water to fluctuate considerably shortly after N-fertilization, as was observed for the B 120-treatment during the first year following 15-N application. In contrast, fertilizer-N included in more slowly cycling pools, composed of stabilized decomposition products, would probably result in low and stable NdfF-values, such as those observed during the final year of this study. The concept of multiple organic matter fractions in the soil is well established (e.g., McGill et al., 1981; Paul & Juma, 1981; Parton et al., 1983). Johnsson et al. (1987, in press) defined two organic pools with

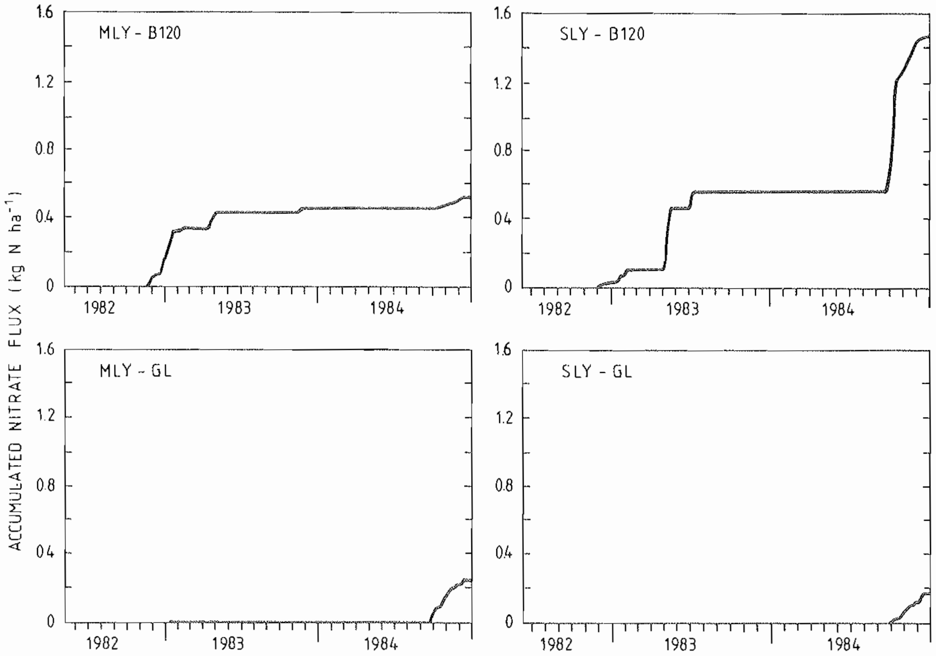

Fig. 3. Accumulated nitrate flux of 15-N-labeled fertilizer (kg N ha^{-1}) from the medium and small lysimeters (B 120 and GL). The fluxes are means of replicates for each lysimeter type.

Table 2. *Total nitrate-N transport by drainage water and transport of nitrate-N derived from the 15-N-labeled fertilizer (kg N ha $^{-1}$ yr $^{-1}$).*

Estimates for the medium and small lysimeters are mean values (± SE) for the replicates

Method and treatment	Leaching 1982[a]		Leaching 1983		Leaching 1984	
	Total	Fert.	Total	Fert.	Total	Fert.
Large lys.						
B 120	2.8	0.3	2.8	0.2	–	–
GL	<0.1	<0.1	0.4	<0.1	–	–
Medium lys.						
B 120[b]	1.8 (±0.4)	0.1 (±<0.1)	6.6 (±1.0)	0.3 (±<0.1)	6.0 (±1.7)	0.1 (±<0.1)
GL[b]	<0.1 (±<0.1)	<0.1 (±<0.1)	0.2 (±0.2)	<0.1 (±<0.1)	15.7 (±3.5)	0.2 (±<0.1)
GLX[b, c]	0.3 (±<0.1)	<0.1 (±<0.1)	0.3 (±0.1)	<0.1 (±<0.1)	14.9 (±3.6)	0.3 (±<0.1)
Small lys.						
B 120[d]	0.5 (±0.3)	<0.1 (±<0.1)	4.4 (±1.4)	0.6 (±0.2)	32.4[f]	0.9[f]
GL[d]	0	0	<0.1 (±<0.1)	<0.1 (±<0.1)	5.7 (±2.3)	0.2 (±<0.1)
GLY[d, e]	–	–	–	–	4.4 (±1.6)	0.3 (±0.1)

[a] Measurements started following fertilization in May 1982.
[b] n=2.
[c] Grass ley receiving 15-N-enriched fertilizer at the second fertilization in May 1982.
[d] n=3.
[e] Grass ley receiving 15-N-enriched fertilizer at the second fertilization in June 1984.
[f] Values refer to one lysimeter.

different turnover rates in a model considering leaching of N from agricultural soils. The model, based on the assumption of two organic N pools in the soil, showed reasonable agreement with measured leaching values, supporting the suggestion above. The addition of unlabeled N-fertilizer during the two years following 15-N application certainly also resulted in decreased NdfF-values in drainage water.

As indicated by the generally low leaching losses of N and the relatively low NdfF-values during 1982 and 1983, leaching of 15-N-labeled fertilizer never exceeded 1 kg N ha^{-1} yr^{-1} from either of the crops (Table 2). As long as the grass leys were established, practically no leaching of 15-N-labeled fertilizer occurred, while ca. 0.5 kg N ha^{-1} was leached from each of the B 120-lysimeters (LLY, MLY, and SLY) during the first two years (Fig. 3, Table 2). Although total N-fluxes during 1984 were considerably higher, the even lower NdfF-values in drainage water indicated that N-fluxes of 15-N-labeled fertilizer were also low during 1984 (Table 2). On average, ca. 0.4% (MLY) and 1.2% (SLY) of the 15-N-labeled N-fertilizer applied to the B 120-treatment in May 1982 were recovered in drainage water during the following three years. The corresponding figure for the GL-treatment was ca. 0.2% (MLY and SLY). During the two years following 15-N application, ca. 0.4% was recovered in water draining from the B 120 large lysimeter (LLY).

CONCLUSIONS

The results presented here illustrate how environmental factors, such as precipitation, can drastically change the N-balance within the soil. In this study, low precipitation led to extremely low leaching losses of fertilizer-N following a single application.

15-N enrichments as low as ca. 1% a.e. of N-fertilizers seem to be appropriate for quantification of fertilizer-N in drainage water. Such low enrichments enable repeated measurements over relatively large areas.

Perennial grass leys were shown to efficiently take up fertilizer-N, thereby minimizing leaching losses.

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Simulated Nitrogen Dynamics and Losses in a Layered Agricultural Soil

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ABSTRACT

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A soil nitrogen model emphasizing mineral nitrogen dynamics and losses is presented. The model has a one-dimensional layered structure and considers plant uptake, mineralization, immobilization, leaching and denitrification processes. Fertilization and manure additions are included as management inputs. A physically based soil-water and heat model provides daily values of temperature, unfrozen water content, water flow and drainage, at different depths in the profile. Input data requirements include standard meteorological variables, basic soil physical and biological properties and crop management characteristics.

Soil nitrogen dynamics were simulated for a 3-year period in N-fertilized and unfertilized barley. Model predictions were compared with measurements of nitrate leaching and mineral N content of the soil. Simulation of mineral N levels and leaching generally agreed with field data. Prediction of mineral N dynamics in both N-fertilized and unfertilized barley was better for surface layers than for deeper layers in the profile. Discrepancies between simulated and measured mineral N content in the topsoil were mainly related to mineralization and plant uptake. In deeper soil layers, differences between measured and predicted values were primarily related to the water flow and the drainage pathway. Simulated amounts of nitrate leached were close to measured values. Discrepancies in the temporal distribution of nitrate leached were mainly attributable to the simulated water flow. The model predicted that nitrate leaching to drainage tiles occurred mainly from the upper layers. Prediction of leaching depended as much on the simulation of drainage flow pathways and the vertical distribution of nitrate in the profile as on the simulation of water flow rates.

INTRODUCTION

The fate of mineral nitrogen, whether it is retained in the crop–soil system and contributes to productivity or is transported and lost from the system, is

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an important consideration in modern agriculture. Increasing the cost–benefit of fertilizer use, in an era of rising production costs, is a major challenge (Rosswall and Paustian, 1984). The enrichment of groundwater and aquatic systems resulting from inefficient nitrogen utilization represents a serious environmental impact in many regions, including Sweden (Gustafson, 1983). Modifying agricultural practices to solve these problems requires a better quantitative understanding of nitrogen cycling processes in agricultural soils.

Simulation models provide an effective means of organizing knowledge about the complex, interactive behaviour of nitrogen in soil (Frissel and van Veen, 1982) and a number of models have been developed for a variety of purposes (reviewed by Frissel and van Veen, 1981; Haith, 1982). Models for simulating crop nitrogen availability and nitrogen losses range from complex, multi-process models (e.g. Tanji and Mehran, 1979; van Veen and Frissel, 1981; Schaffer et al., 1983) to simpler, empirically-based formulations (e.g. Addiscott, 1977; Burns, 1980; Burns and Greenwood, 1982).

Among the models which have received the widest application are simple models of solute transport and nitrate leaching. The model of Burns (1974) has been used for estimating movement of fertilizer nitrogen in several field soils (Burns, 1976; Greenwood and Burns, 1979) and for making regional estimates of nitrate leaching (Burns and Greenwood, 1982). An independent test of three leaching models was carried out by Cameron and Wild (1982) using data on chloride transport.

A general characteristic of the more complex models developed as "research tools" is that they are often difficult to apply to many different sites, due in part to extensive information requirements (Frissel and van Veen, 1982). Only a few such models have been applied to different sites or data sets (e.g. van Veen and Frissel, 1981; Parton et al., 1983), although management models (Knisel, 1980; Schaffer et al., 1983) have been developed for widespread use.

We have incorporated existing theory into a comprehensive structure of the major processes involving nitrogen in soil, with emphasis on simulating mineral nitrogen dynamics and nitrogen losses. The focus of model development was to facilitate numerous applications, with a model resolution compatible with information generally available from agricultural field research. This paper describes the model and an application at the field level to simulate temporal dynamics of soil nitrogen. Nitrogen in N-fertilized and unfertilized barley was simulated and compared with measurements over a 3-year period for the field site of the "Ecology of Arable Land" project (Persson and Rosswall, 1983; Steen et al., 1984).

MODEL DESCRIPTION

Overview of model structure

The nitrogen model (Fig. 1) includes the major processes that determine inputs, transformations and outputs of nitrogen in agricultural soils. A water

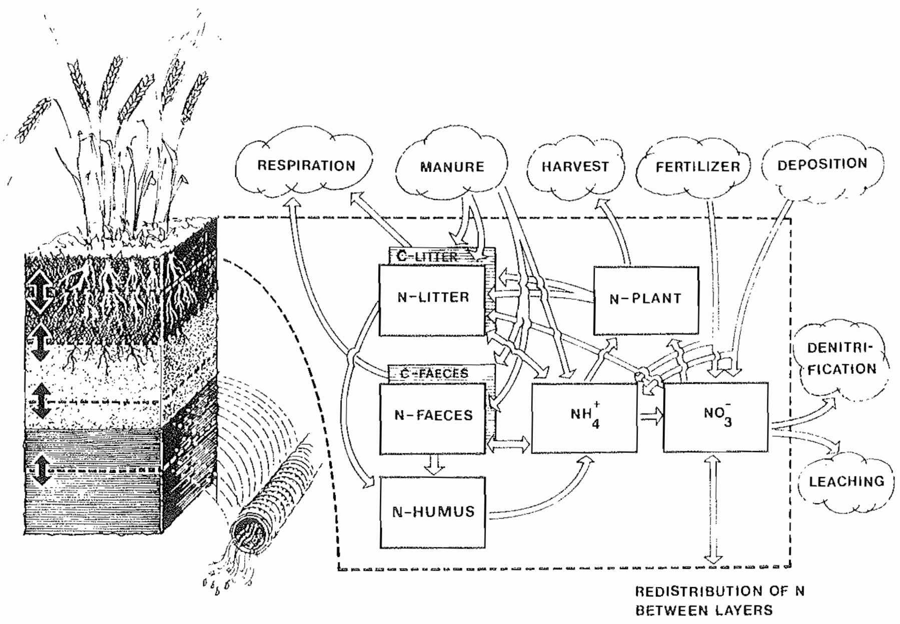

Fig. 1. The nitrogen model structure. Parts within the broken line represent the uppermost layer of the soil profile. Subsurface layers have the same structure as the surface layer but have no direct transfers from fertilizer and deposition.

and heat model (Jansson and Halldin, 1979) provides driving variables for the model, i.e. surface runoff and infiltration, water flow between soil layers and flow to drainage tiles, unfrozen soil water content and soil temperature.

Inorganic and organic pools are replicated for each soil layer. The soil profile is divided into layers (five in the present version) on the basis of physical and biological characteristics. Mineral N pools include ammonium and nitrate. Organic N is classified as litter, manure-derived faeces and humus. Carbon pools for litter and faeces are included for controlling nitrogen mineralization and immobilization rates. The litter component includes undecomposed material (e.g. crop residues, dead roots and microbial biomass), while stabilized decomposition products make up the humus component. The single plant component includes nitrogen in both above- and below-ground biomass. Root distributions within the soil profile are taken into account (see below).

Manure, inorganic fertilizer and atmospheric deposition are inputs to the uppermost soil layer (Fig. 1). Losses through denitrification and leaching of nitrate to drainage tiles can occur from each soil layer. Nitrate in solution can be transported between soil layers or to drainage tiles.

Water and heat flows

The water and heat model is based on two coupled, differential equations describing heat and water transport (derived from Fourier's and Darcy's laws, respectively) in a one-dimensional soil profile (Jansson and Halldin, 1979). Snow dynamics, frost, evapotranspiration, precipitation, groundwater flow, plant water uptake and drainage flow are included. The model predicts soil climate variables (e.g. soil temperature, water content, etc.), with a daily resolution at any level in the soil profile. A detailed technical description of the model is given by Jansson and Halldin (1980), but a short summary is given here.

The model has a one-dimensional vertical structure and therefore lateral flow is not explicitly calculated. Water flow to drainage tiles occurs when the simulated groundwater is above the level of the tiles, i.e. flow occurs directly from a layer to drainage tiles when water content is at saturation. The flow rate is proportional to the hydraulic gradient (estimated from the depth of the groundwater table and the density of the drainage tiles) and the thickness and saturated conductivity of each layer. Calculation of natural drainage is based on an empirical assumption for net horizontal groundwater flow.

Surface runoff can occur because of a limited infiltration capacity or a limited hydraulic capacity of the soil. Surface runoff is assumed not to transport nitrogen. Runoff from surface and drainage flows are combined to give total runoff.

The water and heat model uses standard daily meteorological input data, i.e. air temperature, humidity, precipitation, global radiation (or duration of sunshine or cloud cover) and wind speed. If necessary, precipitation, air temperature and a simple estimate of potential evapotranspiration can suffice as input data. Parameter values for hydraulic and thermal soil properties can be estimated from standard soil physical characteristics, or independent measurements can be used.

The model, originally developed for a forest soil, has been modified and tested for agricultural soils (Jansson and Thoms-Hjärpe, 1986) and for application to agricultural watersheds (Lundin, 1984).

Nitrogen inputs

In the process descriptions given below, the following conventions have been used:

(1) Pools (state variables) are denoted by capitals subscripted with name abbreviations.

(2) Flows are denoted by capital italics subscripted with the direction of the transfer. Layer is indicated by "z" in parentheses.

(3) Parameters are indicated by lower-case italic letters with appropriate

subscripts. A list of parameters used in the nitrogen model is given in Appendix 1.

External input of nitrogen includes atmospheric deposition, inorganic fertilization and manure. Atmospheric deposition includes wet deposition, calculated from amount of precipitation and a mean mineral N concentration, and dry deposition, which is given a constant daily rate. Nitrogen fertilizer is added to the topsoil layer and a dissolution rate constant (e.g. for pelleted fertilizer) controls its input to the soil mineral N pools. Added manure is incorporated into the two uppermost soil layers. Manure additions are split into three fractions: (1) bedding (i.e. straw, sawdust, etc.) which is shunted to the soil litter pool; (2) faeces; (3) ammonium (Fig. 1).

Incorporation of plant material into the litter component is simulated twice: at harvest and when the field is ploughed. At harvest the amount of root N going to litter is

$$N_{p\to l}(z) = f_r(z)\ (1 - f_{ar} - f_{lr} - f_{hp})\ N_p \tag{1}$$

where $f_r(z)$ is the fraction of roots in layer z, f_{ar} is the fraction of plant N remaining above ground after harvest, f_{lr} is the live root fraction of plant N after harvest, f_{hp} is the harvested fraction of plant and N_p is plant N prior to harvest. For an annual crop, f_{lr} would be 0 after harvest, while in a perennial crop, part or all of the root nitrogen could be excluded from transfer to the litter pool at harvest. Carbon flow to litter is proportional to nitrogen flow according to a constant C–N ratio for roots.

At ploughing, nitrogen in above-ground harvest residues (and remaining roots in a perennial crop) are incorporated into the plough layer.

$$N_{p\to l}(z) = (f_{ar} + f_{lr})\ N_p \tag{2}$$

The carbon incorporated is proportional to eqn. (2) according to a constant C–N ratio for above-ground harvest residues (and roots for a perennial crop).

Mineralization, immobilization and nitrification

The general concept of active and passive components of soil organic matter is well established (Jansson, 1958; Stanford and Smith, 1972; Paul, 1984). Many models address the heterogeneity of organic material in soil by simulating multiple organic matter fractions (e.g. Paul and Juma, 1981; McGill et al., 1981; van Veen and Frissel, 1981; Parton et al., 1983). We have defined two principle organic matter pools: a fast cycling pool (litter), representing an organic matter–microbial biomass complex which receives fresh organic material, and a slow cycling pool (humus) composed of stabilized decomposition products (Fig. 1). A separate component was included for manure-derived faeces because of the substantial differences in its chemical composition compared with plant material entering the litter pool.

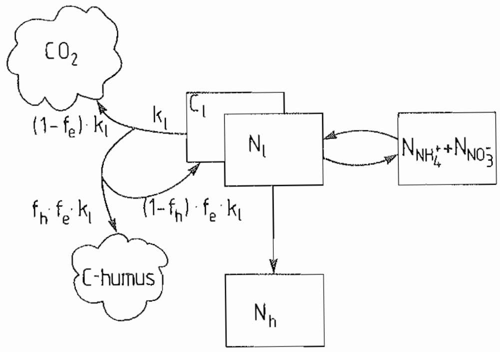

Fig. 2. Flow diagram showing carbon and nitrogen balance associated with litter (or faeces). Explanation of parameter symbols are given in the text.

Mineralization of humus ($N_h(z)$) is calculated as a first-order rate,

$$N_{h\rightarrow NH_4^+}(z) = k_h e_t(z)\ e_m(z)\ N_h(z) \tag{3}$$

where k_h is the specific mineralization constant and $e_t(z)$ and $e_m(z)$ are response functions for soil temperature and moisture, respectively (see below).

Decomposition in the two organic carbon pools (litter and faeces) are the main controls on N mineralization from these sources. Decomposition of soil litter carbon ($C_l(z)$)

$$C_{l(d)}(z) = k_l\ e_t(z)\ e_m(z)\ C_l(z) \tag{4}$$

is a function of a specific rate constant (k_l), temperature and moisture. The products of decomposition are CO_2, stabilized organic material (humus) and, conceptually, microbial biomass and metabolites (Fig. 2). Since a single pool (soil litter) represents the litter–decomposer complex, this synthesis of microbial biomass and metabolites constitutes an internal cycling (i.e. $C_{l\rightarrow l}(z)$, see below). The relative amounts of decomposition products formed,

$$C_{l\rightarrow CO_2}(z) = (1-f_e)\ C_{l(d)}(z) \tag{5}$$

$$C_{l\rightarrow h}(z) = f_e\ f_h\ C_{l(d)}(z) \tag{6}$$

and

$$C_{l\rightarrow l}(z) = f_e\ (1-f_h)\ C_{l(d)}(z) \tag{7}$$

are governed by a synthesis efficiency constant (f_e) and a humification fraction (f_h).

Mineralization or immobilization of nitrogen in the litter pool is governed by two assumptions: (1) that the internal cycling of carbon (eqn. 7) and the formation of humus (eqn. 6) have nitrogen demands determined by a constant C–N ratio of decomposer biomass and humification products (r_o); (2) that nitrogen is released by decomposition of soil litter carbon in proportion to the actual litter C–N ratio. From eqns. (4), (6) and (7), net mineralization or immobilization of nitrogen in litter ($N_l(z)$)

$$N_{l \leftrightarrow NH_4}(z) = \left[\frac{N_l(z)}{C_l(z)} - \frac{f_e}{r_o} \right] C_{l(d)}(z) \qquad (8)$$

is then determined by the balance between the release of nitrogen during decomposition and the nitrogen immobilized during microbial synthesis and humification. The switch between net immobilization and mineralization of N in litter occurs at the C–N ratio equal to r_o/f_e (cf. Parnas, 1975). A simplification is that the C–N ratio of organic matter humified is assumed to be the same as for microbial biomass and metabolites. Observed mean values of the C–N ratio of microorganisms and humus in agricultural soils are in the same range, normally between 5 and 15 (e.g. Parsons and Tinsley, 1975; Kowalenko, 1978; McGill et al., 1981).

When net immobilization occurs (i.e. $N_{l \leftrightarrow NH_4}(z) < 0$), immobilization will be reduced if sufficient mineral N is not available. The reduction in immobilization is determined in the same way as the reduction in plant uptake by roots (see eqn. 14), i.e. by assuming that a constant fraction of the total storage of mineral nitrogen is available. Both ammonium and nitrate can be immobilized, but with preference for available ammonium.

Nitrogen humified during litter decomposition is proportional to carbon humification (eqn. 6) according to the C–N ratio of microorganisms and humified products (r_o). Decomposition, mineralization/immobilization and humus formation for C and N in faeces are calculated in the same way as for litter.

Nitrification is not modelled explicitly as a microbial process, but the segregation of mineral N into ammonium and nitrate is made by assuming a nitrate–ammonium ratio characteristic for a particular soil. This simplification may be appropriate given rapid nitrification rates (e.g. Kowalenko, 1980; Berg and Rosswall, 1985) and the relatively small amount of ammonium often observed in agricultural soils (Cameron et al., 1978; Lindén, 1981). The transfer rate of ammonium to nitrate

$$N_{NH4 \rightarrow NO_3}(z) = k_n\, e_t(z)\, e_m(z) \left[N_{NH_4}(z) - \frac{N_{NO_3}(z)}{n_q} \right] \qquad (9)$$

depends on a potential rate (k_n) which is reduced as the nitrate–ammonium ratio (n_q) is approached. If $N_{NO3}(z)/N_{NH4}(z) > n_q$, no transfer of ammonium to nitrate occurs.

Abiotic response functions

Decomposition, mineralization and nitrification are regulated by the same abiotic response function involving soil temperature and moisture. A Q_{10} relationship (e.g. Bunnell et al., 1977) is used to express the effect of temperature,

$$e_t(z) = Q_{10}^{\left[\frac{T(z) - t_b}{10}\right]} \tag{10}$$

where T(z) is the soil temperature for the layer, t_b is the base temperature at which $e_t(z)$ equals 1 and Q_{10} is the factor change in rate with a 10-degree change in temperature.

The soil moisture factor decreases, on either side of an optimum level, in drier soil or in excessively moist soil (Jansson and Berg, 1985), i.e.

$$e_m(z) = e_s + (1 - e_s)\left[\frac{\theta_s(z) - \theta(z)}{\theta_s(z) - \theta_{ho}(z)}\right]^m \qquad \theta_s(z) \geq \theta(z) > \theta_{ho}(z) \tag{11a}$$

$$e_m(z) = 1 \qquad \theta_{ho}(z) \geq \theta(z) \geq \theta_{lo}(z) \tag{11b}$$

$$e_m(z) = \left[\frac{\theta(z) - \theta_w(z)}{\theta_{lo}(z) - \theta_w(z)}\right]^m \qquad \theta_{lo}(z) > \theta(z) \geq \theta_w(z) \tag{11c}$$

where $\theta_s(z)$ is the saturated water content, $\theta_{ho}(z)$ and $\theta_{lo}(z)$ are the high and low water contents, respectively, for which the soil moisture factor is optimal, and $\theta_w(z)$ is the minimum water content for process activity. A coefficient (e_s) defines the relative effect of moisture when the soil is completely saturated and m is an empirical constant. The two thresholds, defining the optimal range, are calculated as

$$\theta_{lo}(z) = \theta_w(z) + \Delta\theta_1 \tag{12a}$$

$$\theta_{ho}(z) = \theta_s(z) - \Delta\theta_2 \tag{12b}$$

where $\Delta\theta_1$ is the volumetric range of water content where the response increases and $\Delta\theta_2$ is the corresponding range where the response decreases.

Plant uptake of nitrogen

Plant uptake of N includes both nitrate and ammonium from each layer where roots are present. A logistic uptake curve (cf. Greenwood et al., 1974) is used to define the cumulative potential N demand during the growing season,

$$\int u(t)\,dt = \frac{u_a}{1 + \frac{u_a - u_b}{u_b} e^{-u_c t}} \tag{13}$$

where u_a is the potential annual N uptake, u_b and u_c are shape parameters and t is days after the start of the growing season. Diffusion and root absorption are assumed to be the main mechanisms behind nitrogen uptake by roots (Nye, 1977). Consequently, the uptake is not associated with plant water uptake. To limit uptake if soil mineral N concentration is low, a maximum availability fraction (f_{ma}) is assumed which is proportional to the total amount of ammonium and nitrate in each layer. Daily uptake of nitrate

$$N_{NO_3 \to p}(z) - MIN \left| \begin{array}{ll} f_r(z) \dfrac{N_{NO_3}(z)}{N_{NO_3}(z) + N_{NH_4}(z)} u & \text{(14a)} \\ f_{ma}\ N_{NO_3}(z) & \text{(14b)} \end{array} \right.$$

is then calculated from the relative root fraction in the layer ($f_r(z)$), the proportion of total mineral N as nitrate and the derivative of the growth curve (u). An empirical root distribution, which changes during the growing season, controls relative uptake rates from each layer. Actual uptake is taken as the minimum of the two functions. Ammonium uptake is calculated in the same way, but is proportional to the relative amount of ammonium in total mineral N. If actual uptake from a layer is below the potential rate, compensatory increases in uptake may occur from other layers (Drew and Saker, 1975). This is achieved by adding the difference between potential and actual uptake rates to the potential demand for the soil layer below.

Denitrification

Denitrification, the reduction of nitrate to gaseous nitrogen (N_2 and N_2O) products, is an anaerobic process and consequently is highly dependent on soil aeration. The model uses soil water content ($\theta(z)$) as an indirect expression of soil oxygen status, where the influence on the denitrification rate

$$e_{md}(z) = \left[\frac{\theta(z) - \theta_d(z)}{\theta_s(z) - \theta_d(z)} \right]^d \tag{15}$$

is expressed as a power function which increases from a threshold point ($\theta_d(z)$) and is maximum at saturation ($\theta_s(z)$), where d is an empirical constant. Below the threshold water-content no denitrification occurs. This expression is similar to the soil-water relationship developed by Rolston et al. (1984). The denitrification rate for each layer

$$N_{NO_3 \to}(z) = k_d(z)\ e_{md}(z)\ e_t(z) \left[\frac{[N_{NO_3}(z)]}{[N_{NO_3}(z)] + c_s} \right] \tag{16}$$

depends on a potential denitrification rate ($k_d(z)$), the soil water/aeration status ($e_{md}(z)$) and the same temperature factor ($e_t(z)$) used for the other biologically-controlled processes. The effect of nitrate concentration is controlled by the half-saturation constant, c_s (i.e. the concentration where the rate is 50% of the maximum, if all other conditions are optimal).

Nitrate transport and leaching

Nitrate is considered to be wholly in solution and movement between layers and leaching from the profile in drainage flow are determined by water flow rates. The nitrate transport is thus calculated as the product of the water flow and the nitrate concentration in the soil layer from which the water flow originates.

Ammonium is assumed to be immobile with respect to leaching (i.e. largely fixed to soil colloids) and is not transported by water flow (Krantz et al., 1943; Broadbent et al., 1958; Wiklander, 1977).

MODEL APPLICATION

Site description

Kjettslinge was the experimental site for the project "Ecology of Arable Land". In the project, data were collected on decomposition, denitrification, leaching, mineralization, mineral N profiles, primary production and soil abiotic conditions. A detailed description of the field site is found in Steen et al. (1984) and only a brief summary of site characteristics is given here.

The field site is situated in central Sweden, approximately 40 km north of Uppsala. The climate in the region is cold–temperate and semi-humid. Annual precipitation and mean annual temperatures at Ultuna, 10 km south of Uppsala, are 520 mm and 5.4°C (1931–60), respectively (Rodskjer and Tuvesson, 1975). The experimental field has two parts: the main field and four tile-drained plots used for leaching studies. The cropping systems chosen for the simulations were (1) B0, barley with no N fertilization, and (2) B120, barley receiving 120 kg N ha^{-1} $year^{-1}$ as calcium nitrate fertilizer.

Data on leaching (Bergström, 1987), soil mineral N profiles (Bergström, 1986) and crop production (R. Pettersson, personal communication, 1985) were taken from the tile-drained plots, which were established in 1981, following a fallow year in 1980.

The soil, down to a depth of 1.0 m, consists of four distinct layers of varying texture and structure: (1) topsoil, consisting of clay loam, with a mean thickness of 0.27 m; (2) a fine sand layer, with a thickness varying from 0 to 0.50 m; (3) an oxidized clay layer, down to a depth of approximately 0.75 m; (4) a non-oxidized clay layer, below the oxidized clay layer.

The topsoil consists of 15–20% clay, 2.2% organic carbon and 0.23% organic nitrogen and has a pH of 6.0–6.5.

The soil water and heat simulation

Simulation of soil water and heat processes, which are inputs to the nitrogen model, were based on a previous application to the field (Jansson and Thoms-Hjärpe, 1986).

The meteorological data used as driving variables were measured at the experimental field (Alvenäs et al., 1986). Missing data were substituted with

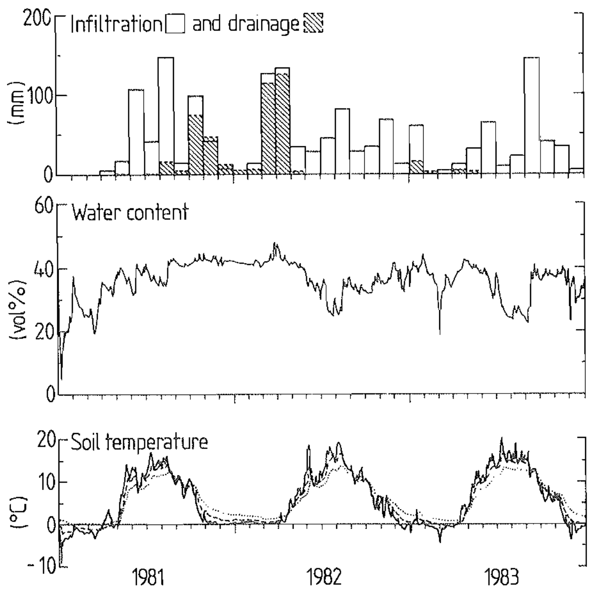

Fig. 3. Simulated abiotic conditions at Kjettslinge: monthly values of drainage, infiltration and daily mean values of soil water content in the topsoil and daily values of soil temperature at depths of 0.05, 0.36 and 0.85 m.

data from the meteorological stations at Marsta and at F16, both ca. 30 km south of the Kjettslinge field (Alvenäs et al., 1986; Jansson and Thoms-Hjärpe, 1986).

The soil profiles in both the soil-water and heat model and in the nitrogen model were divided into 5 layers according to the textural horizons in the field and numerical demands in the water and heat model (Appendix 1). A common set of soil physical parameters was used for the tile-drained plots, as opposed to the earlier application by Jansson and Thoms-Hjärpe (1986), which used separate parameter sets for each cropping system in the main field. Because of a lack of data on soil moisture dynamics in the tile-drained plots and since the model was concentrated on biological processes, this simplification was deemed appropriate.

Drainage was assumed to take place only through the drainage tiles. Measurements of groundwater pressure at the site showed a slight tendency for a positive pressure gradient from the subsoil (Bergström, 1987), indicating stable conditions or an inflow of water to the profile.

Simulated soil temperatures, infiltration, drainage and soil water content showed substantial differences between years (Fig. 3). Two summers, 1982 and 1983, were characterized by rather high temperatures with extended drought periods, whereas the summer/autumn of 1981 was comparatively wet.

A thick snow cover prevented the soil from becoming completely frozen during the winter of 1981/82.

The simulated drainage flow was adjusted, compared with that used by Jansson and Thoms-Hjärpe (1986), to achieve better agreement with drainage measurements from the B120 plot (Bergström, 1986a). The adjustment was done by changing evapotranspiration parameters controlling the water balance in the model.

Parameter derivation

The adaptation of the nitrogen model to the two barley treatments (B0 and B120) at the Kjettslinge field was done in two steps. First, parameter values were estimated from independent measurements at the field or estimated from the literature. In a second step, a few parameter values, which were found to strongly influence the results, were adjusted to improve the agreement between measured and simulated mineral N profiles. Adjusted parameters were the nitrogen availability fraction, litter and humus turnover rates, Q_{10}-value and saturation activity (Appendix 1). This second step was restricted to model and data comparisons for the B120-treatment. Subsequent simulations of the B0-treatment were not adjusted and the only difference was the lack of application of N-fertilizer. The parameter values chosen are summarized in Appendix 1.

Mean value for dry deposition of mineral nitrogen was set to 0.001 g N m^{-2} day^{-1} (Monitor, 1984). To simulate wet deposition a N concentration of 0.8 mg l^{-1} in precipitation was used (Monitor, 1984).

Parameter estimates for organic matter flows were based on data from the field site and from literature values. The microbial growth yield and microbial C–N ratio, parameters associated with the litter pool, were set to 0.5 (Kjøller and Struwe, 1982) and 10 (McGill et al., 1981), respectively. These together determine a critical C–N ratio for net N mineralization of 20. Litter decomposition rates were based on litter bag incubations (Andrén and Paustian, 1987) and we assumed that 10% of decomposing litter carbon became stabilized (cf. Jenkinson, 1977) and entered the humus pool.

A Q_{10}-value of 3 was used for the effect of temperature on decomposition. Reported values for Q_{10} generally range between 2 and 3 in temperate soil. For example, Campbell et al. (1984) found mean Q_{10} values, for mineralization, of 2.0, 2.18 and 2.75 for three major soil groups in Canada. In general, Q_{10} values tend to be larger for lower temperature intervals. In this study, with temperatures often around freezing during long periods, a relatively high Q_{10}-value was used to prevent unreasonably high activity rates at low temperatures.

The influence of soil moisture was expressed in straight-line segments as a function of unfrozen soil water content. The function had a value of 0 at the wilting point for each layer, increasing to 1 at a volumetric water content of

10% above the wilting point. For water contents greater than 8% below saturation, the value again declined to 0.6 at saturation. The soil moisture function also accounted for frost effects below 0°C, because of water freezing.

Simulated plant uptake started 2 weeks after sowing (20 May) and ended at harvest (25 August), with the maximum uptake rate in late June. The same dates for sowing plus fertilization and for harvest were used each year in the simulation and they represented means of the actual dates for the 3 years. Potential net nitrogen uptake was based on N-biomass measurements (including roots) from the field (Hansson et al., 1987; R. Pettersson, personal communication, 1985). Measurements of root production in N-fertilized and unfertilized barley indicated that 20–40% of plant nitrogen was found in the roots (Hansson et al., 1987), depending on the estimation method, stage of crop development and treatment. We assumed 30% of total plant nitrogen to be allocated to the roots. At harvest, 60% of nitrogen in plant biomass was removed and the 30% in the roots was transferred to the litter component. Harvest residues, 10% of total N, were incorporated into the litter pool in soil at ploughing. The C–N ratios assumed for roots (25) and for harvest residues (50) were based on mean values of chemical data for crop biomass during 1981 (Hansson et al., 1987).

For calculating nitrate uptake from different levels in the profile, changes in root depth distribution over time were based on field data (Hansson et al., 1987) for both crops. However, daily uptake from each layer was set to a maximum of 8% of the total mineral N content in the layer if potential uptake exceeded this value.

The potential rate of denitrification was estimated to be 0.1 g N m^{-2} day^{-1} (L. Klemedtsson and B. Svensson, personal communication, 1985). The actual rate was reduced to zero when the air-filled pore space exceeded a volumetric percentage of 10, but increased rapidly when the soil approached saturation (cf. Rolston et al., 1984).

RESULTS AND DISCUSSION

Variations in the soil mineral N content

The overall seasonal dynamics of simulated mineral N content of the soil and the differences between years, i.e. for different climatic conditions, agreed well with observed data (Fig. 4). For example, the model showed the decrease in nitrate content in the B120 profile during the wet autumn of 1981, as opposed to the increases in nitrate in the drier autumns of 1982 and 1983 (Fig. 4).

Agreement with observed values were better for surface horizons (0–27 and 27–45 cm) in both B120 and B0 (Fig. 4) than for deeper horizons. Simulated nitrate levels in the two clay layers were usually more static than the observed

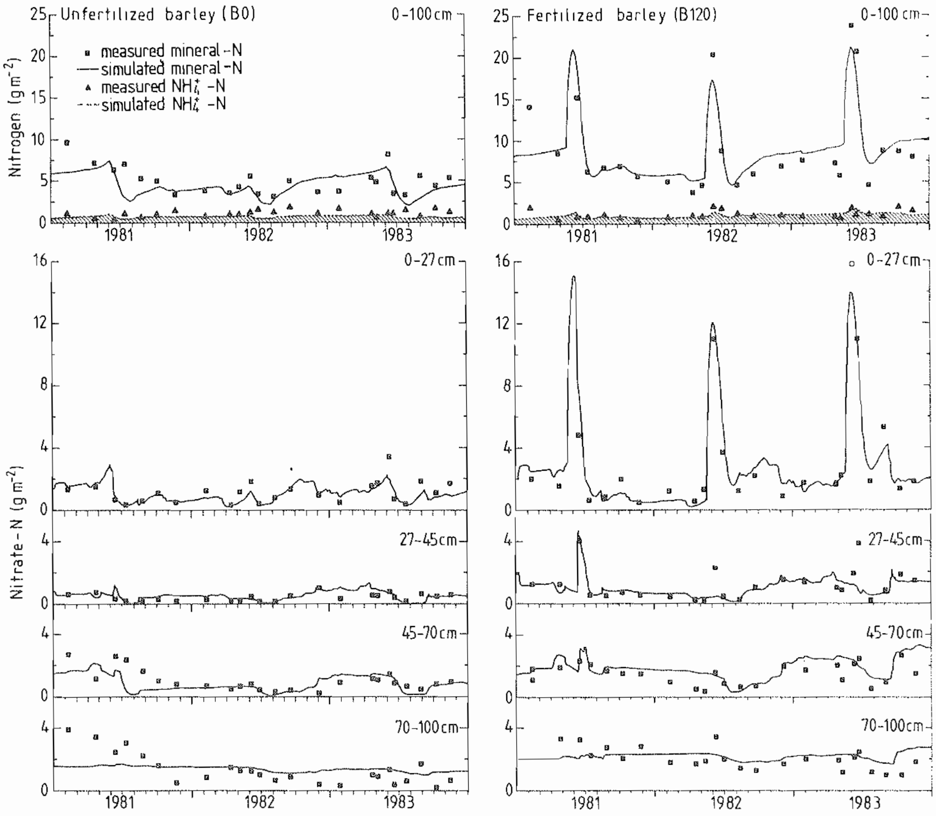

Fig. 4. Simulated and measured storage of mineral N, ammonium N and nitrate N in unfertilized barley and in N-fertilized barley at Kjettslinge.

levels. This appeared to be due to difficulties in simulating nitrogen transport and water flow between subsurface layers in combination with the flow to drainage tiles from different layers. For example, it was not possible to simulate the large decrease in nitrate in spring 1981, which had accumulated in the lower layers following the fallow in 1980. The initial nitrate levels were therefore set to the values obtained at the second sampling date (Fig. 4). An underestimation of water flow from the surface layer to the sand (25–45 cm) may also be the reason for the missing peaks in nitrate amounts found in the sand layer following fertilization (Fig. 4).

A considerable deviation from observed data occurred when the model did not show the sudden drop in nitrate content just prior to N-fertilization in May 1983 (Fig. 4). This decrease was observed in the two clay layers, but almost no leaching was observed in the drainage flow. However, leaching was simulated by the model and most of the water flow came from the sand layer. In

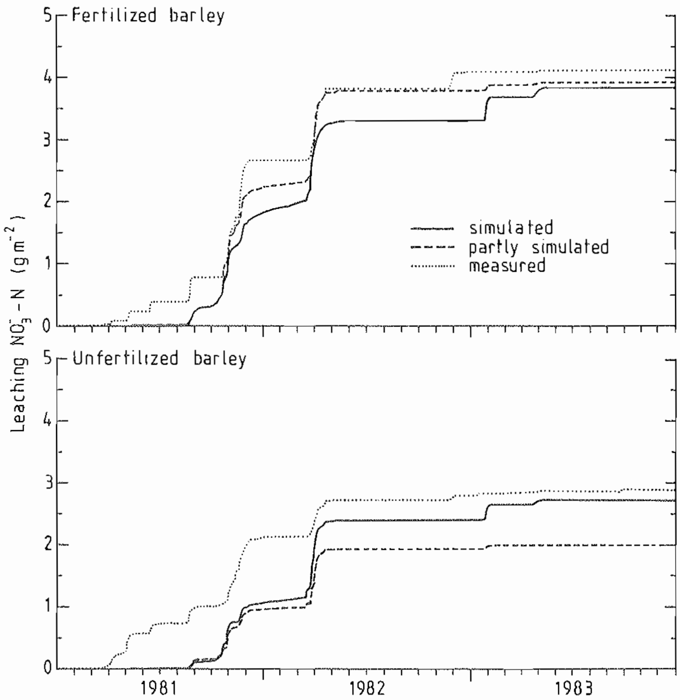

Fig. 5. Simulated, partly simulated (measured nitrate concentrations multiplied by simulated water flows from drainage tiles) and measured leaching from N-fertilized barley and unfertilized barley at Kjettslinge.

this case, measurements of groundwater pressure (Bergström, 1987) indicated that the groundwater table was below the drainage tiles during the spring of 1983, suggesting that percolating water reached the groundwater table rather than being discharged through the drainage system. The observed nitrate levels in the clay layers showed considerable dynamics. This indicated a greater degree of water movement in the lower profile than was predicted by the hydrological simulations.

Some of the deviations in mineral N levels for the two upper layers may be attributed to the simulation of mineralization. For example, mineralization appeared to be underestimated in the autumn of 1981 and 1983 in B120 and in late spring of 1982 and 1983, after fertilization, in B0 and B120 (Fig. 4). Simulated mineralization rates were higher in the fertilized barley because of greater crop residue incorporation (Table I). Simulated litter decomposition resulted in a net release of mineral N, although about 50% of litter N inputs to B120 and slightly less in B0 were stabilized in the humus fraction (Table I).

The empirical uptake equation was satisfactory in predicting the uptake of both crops (Table I; R. Pettersson, personal communication, 1985) in relation to mineral N availability. Availability of mineral N was the only factor limiting crop growth and thus causing differences between the B0- and the B120-treat-

TABLE I

Annual means of simulated nitrogen flows based on the 3-year simulations. Partial flows given in parentheses. All values in g N m^{-2} $year^{-1}$

Flow	N-fertilized barley (B120)	Unfertilized barley (B0)
Fertilization and deposition	12.8	0.8
Mineralization	8.0	6.9
Net litter mineralization	(3.4)	(2.3)
Humus mineralization	(4.6)	(4.6)
Humification	3.3	1.7
Plant uptake	18.3	6.8
Crop residues to litter	(7.3)	(2.7)
Harvest	(11.0)	(4.1)
Leaching	1.3	0.9
Denitrification	0.4	0.4

ment. Since crop uptake is normally the dominant N flow in cropped systems, small relative changes, especially in root distribution and nitrogen availability, in the simulation of crop uptake are critical for the prediction of soil nitrate levels.

The variation of mineral N in the soil was mainly due to nitrate dynamics. The ammonium level, simulated and measured, was quite stable and similar in both treatments.

Nitrogen losses

The simulation of nitrate leaching was dependent on two model outputs: soil nitrate concentration and drainage water flow. In addition to observed leaching calculated from data on drainage water flow and nitrate concentrations, time series calculated from measured nitrate concentrations and simulated drainage (i.e. "partly simulated", Fig. 5) were compared to model predictions (i.e. "simulated", Fig. 5). The partly simulated time series was included to isolate deviations in leaching patterns due to either poor prediction of drainage water flows or of nitrate levels.

Cumulative nitrate leaching was similar in all three time series (Fig. 5). Due to an underestimate in drainage water flow, the model failed to simulate the leaching which occurred in both treatments during the first half of 1981. After this time-period, drainage flows were well predicted by the model as indicated by the parallel courses of the partly simulated and measured time series.

The model overestimated nitrate concentrations in the drainage water during the spring of 1982 and 1983 in B0 and in spring 1983 in B120 (Fig. 5). In B0, simulated drainage from upper soil layers was too high and redistribution of water between layers was underestimated. In B120, leaching was observed during the previous autumn (Fig. 5) but was not simulated by the model, which may explain the overestimation of nitrate concentrations (and loss) for the following spring leaching period. The overall effect was a lag in the simulated leaching, but the cumulative amounts at the end of the period were similar. Overestimates of leaching on these occasions (Fig. 5) may indicate deviations from the assumed flow paths, e.g. percolation to groundwater instead of to drainage tiles (cf. Bergström, 1987).

Underestimations of simulated vs. observed nitrate concentrations were most evident in autumn 1981 in fertilized barley. Here the underestimation in mineralization rates, discussed above, may explain the differences.

Simulated N losses due to denitrification were less than one-half of leaching losses in both treatments (Table I). The yearly totals of denitrification were similar to preliminary estimates, based on field measurements at Kjettslinge, of ca. 0.5 g N m^{-2} $year^{-1}$ in B120 (Klemedtsson, 1986). This suggests that there were no large discrepancies, in other comparisons of data and model predictions, due to unknown losses via denitrification. An evaluation of the simulation of temporal dynamics of denitrification, in comparison with field measurements (Klemedtsson, 1986), will be presented elsewhere.

CONCLUSIONS

The present application showed generally good agreement between simulations and measurements. Both abiotic and biotic factors were found to be important when analyzing the dynamic behaviour of mineral N in the system. Disagreements between simulated and measured mineral N contents in the topsoil may be explained by biotic factors, whereas disagreements in deeper layers may be explained by abiotic factors.

The simulated nitrate dynamics in the topsoil showed patterns similar to the measured values in both the B0- and B120-treatments. However, the model often predicted a more dampened seasonal variation in nitrate levels, especially for the B0-treatment. This may be due to an inadequate description of temperature and moisture influences, including possible stimulus of activity from wet–dry and freeze–thaw cycles on decomposition and mineralization (Soulides and Allison, 1961). Also, to account for spring and autumn mineralization peaks, it may be necessary to include a labile pool of easily decomposable, nitrogen-rich material in crop residues (McGill et al., 1981).

Spatial relationships between root distribution and nitrate uptake from different levels of the profile are not well understood (Nye and Tinker, 1977). Our preliminary approach to modelling uptake showed a strong sensitivity to the chosen root distribution with depth. Static information on root biomass distributions may be of minor value because of compensatory growth and uptake by roots (Drew and Saker, 1975). A new formulation of this process in the model should more strongly emphasize the dynamic nature of the roots and root uptake activity.

The use of the fixed potential uptake curve to control uptake is a simple approach which mainly requires information on expected crop growth. However, it assumes that nitrogen availability is the only limiting factor for uptake and, currently, no consideration of other time-varying factors such as drought stress or meteorological elements are included. Also the definition of potential uptake is fairly specific to crop and site.

Leaching was simulated mainly from the sand layer and upper clay layer (45–70 cm). Measurements of mineral N in the two clay layers (45–100 cm) indicated that the main flow path was probably deeper. Both the water flow rates and the flow paths were shown to strongly influence the temporal patterns of leaching.

Leaching was not only dependent on the water flow rate but also on the flow path and the vertical distribution of nitrate in the soil profile. The measured and simulated depth distributions of nitrate showed that a layered model structure improves simulation of nitrate leaching. The simplifying assumption that nitrate was only associated with mass flow, i.e. diffusion and dispersion could be ignored, appeared reasonable.

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APPENDIX 1

Parameters in the nitrogen model and values used for the application

Parameter definition	Equation	Symbol	Unit	Value
External inputs				
N-fertilization			g N m^{-2} $year^{-1}$	0/12[1]
Dry deposition			g N m^{-2} day^{-1}	0.001
Wet deposition, concentration			mg l^{-1}	0.8
Fertilizer specific dissolution rate			day^{-1}	0.15
Crop and management				
Fertilization date				20 May
Harvest date				25 Aug.
Ploughing date	(1)	f_{hp}	—	16 Oct.
Harvested fraction	(1)	f_{ar}	—	0.6
Above ground residue fraction	(1)	f_{lr}	—	0.1
Live root fraction				0
C–N ratio of above-ground residues			—	50
C–N ratio of roots			—	25
Ploughing depth			m	0.27
Mineralization and immobilization				
Humus specific mineralization rate	(3)	k_h	day^{-1}	7.0×10^{-5}
Litter specific decomposition rate	(4)	k_l	day^{-1}	0.035

Parameter definition	Equation	Symbol	Unit	Value
Efficiency	(5)	f_e	—	0.5
Carbon humification fraction	(6)	f_h	—	0.2
C–N ratio of microorganisms and humified products	(8)	r_o	—	10
Fraction of available mineral N			day^{-1}	0.08
Specific nitrification rate	(9)	k_n	day^{-1}	0.2
Nitrate–ammonium ratio	(9)	n_q	—	8
Soil moisture response				
Saturation activity	(11)	e_s	—	0.6
Increasing interval	(12)	$\Delta\theta_1$	%	10
Decreasing interval	(12)	$\Delta\theta_2$	%	8
Porosity	(11)	$\theta_s(z)$	%	45/45/40/53/53[2]
Wilting point	(11)	$\theta_w(z)$	%	15/15/4/27/27[2]
Coefficient	(11)	m	—	1
Soil temperature response				
Response to a 10°C change	(10)	Q_{10}	—	3
Optimal temperature	(10)	t_b	°C	20

Plant uptake				
Start of plant uptake				3 June
Potential nitrogen uptake	(13)	u_a	g N m^{-2} $year^{-1}$	20
Coefficient	(13)	u_b	—	1
Coefficient	(13)	u_c	day^{-1}	0.12
Fraction of available mineral N	(14)	f_{ma}	day^{-1}	0.08
First date of roots in layer z				3/12/20/29/-June[2]
Fraction of roots in layer z (when fully developed)			—	0.6/0.28/0.08/0.04/0[2]
Denitrification				
Potential rate	(16)	$k_d(z)$	g N m^{-2} day^{-1}	0.1
Half-saturation constant	(16)	c_s	mg l^{-1}	10
Soil moisture effect:				
Activity range			%	10
Coefficient	(15)	d	—	2
Structural				
Thickness of soil layer z			m	0.10/0.17/0.18/0.25/0.30[2]

[1]Unfertilized and fertilized barley.
[2]For layers 1–5.

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SIMULATED NITROGEN DYNAMICS AND NITRATE LEACHING IN A PERENNIAL GRASS LEY

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Key words Mineralization, Nitrogen model, N-uptake, Ploughing of leys, Water flow paths

Summary A soil nitrogen model was used for a 4-year simulation of nitrogen dynamics and nitrate leaching, both during grass ley growth and after ploughing a grass ley. Model results were compared with field measurements of soil mineral-N status and leaching. A soil water and heat model provided daily values for abiotic conditions, which were used as driving variables in the nitrogen simulation.

Simulated values for mineral-N levels in the soil agreed well with field data for the first 3 years of the simulation. During the final year the model predicted considerably higher levels of soil mineral-N content compared with measurements. To reach the mineral-N level measured at the time of ploughing the ley, the simulated N-uptake by plants had to be increased by 8 g N m^{-2}.

Simulations of nitrate leaching suggested that estimates of leaching based on measurements in tile-drained plots can be considerably underestimated. Accurate quantification of leaching in tile-drained plots often requires additional information on water-flow paths.

A substantial increase in simulated and measured values for the mineral-N content of the soil occurred after ploughing the ley. In the simulation, most of the increase was due to a high crop residue input and absence of a growing crop after ploughing. Litter accumulations in the soil during the 4-year period contributed little to the increase in soil mineral-N.

INTRODUCTION

Perennial grasslands range in quality from low productivity rangelands to high-producing, heavily fertilized grass leys used as forage crops. The potential for water pollution from moderately productive systems is likely to be minimal owing to the generally long growing period typical for grass leys and usually associated with a nitrogen deficient status (e.g., Gustafson, 1983; Bergström, 1987). Although the rapid N-uptake by grass during periods of favourable growth conditions minimizes leaching, considerable leaching can occur during periods of poor ley growth if weather conditions are unfavourable. Henkens (1977) estimated annual leaching losses to be as high as 44 kg N ha^{-1} from grassland receiving an annual dose of 400 kg N ha^{-1} as fertilizer. High leaching losses are usually also a result of the intensive mineralization associated with ploughing of grass leys (Cameron & Wild, 1984; Bergström, 1987). However, the rates at which such nitrogen may be released by mineralization after ploughing are difficult to determine, owing in great part to the complex nature of soil organic matter.

The general concept that soil organic matter consists of active and passive components is well established (Jansson, 1958; Paul, 1984). Several models also deal with the decomposition of soil organic matter using the assumption of active and passive components (e.g., Paul & Juma, 1981; McGill et al., 1981). A valuable way to test the validity of this kind of model would be to use it to describe soil nitrogen dynamics following ploughing of grass leys, since such ploughing drastically changes the composition of organic matter in the soil by incorporating large amounts of crop residues.

This paper describes a simulation of nitrogen dynamics in the soil and nitrate leaching during growth and after ploughing of a grass ley. The aim of this study was to test the applicability of the model for studying a perennial crop and to analyze the deviations between simulated and measured values for soil mineral-N content and nitrate leaching. Simulated results were compared with measurements obtained within the project "Ecology of Arable Land" (Persson & Rosswall, 1983) over a 4-year period. The model used (Johnsson et al., 1987) includes two organic matter pools, one representing fast-decomposing, fresh organic material and the other representing stabilized decomposition products.

MODEL DESCRIPTION

The nitrogen model (Fig. 1) includes the major processes of importance for inputs, transformations, and outputs of nitrogen in arable soils. A detailed description of the model is given by Johnsson et al. (1987), and only a brief overview of the model structure is presented here.

Driving variables for the model, i.e., surface runoff and infiltration, water flow between soil layers and to tile-drains, unfrozen soil water content, and soil temperature, are provided from a water and heat model (Jansson & Halldin, 1979) modified by Jansson & Thoms-Hjärpe (1986) for arable land.

A division of the soil profile into layers is based on physical and biological properties. Biological N-transformations are replicated for each layer. Mineral-N pools include ammonium and nitrate. Organic-N is classified as litter, faeces, and humus. The carbon pools for litter and faeces are included to control nitrogen mineralization and immobilization rates. Undecomposed material (e.g., crop residues, dead roots, microbial biomass) constitute the litter component, while the humus component consists of stabilized decomposition products. The plant component includes nitrogen in both above- and below-ground biomass, but root distributions are also calculated.

Manure, inorganic fertilizer, and atmospheric deposition are inputs to the uppermost soil layer. However, manure is not included in the present simulation. Losses due to denitrification and nitrate leaching to tile-drains can occur in each soil layer. Nitrate in solution can move between soil layers or to tile-drains.

SITE DESCRIPTION

The Kjettslinge field was the experimental site for the project "Ecology of Arable Land". The field is located in central Sweden, ca. 40 km north of Uppsala. The climate in the region is cold-temperate and semi-humid. Mean annual precipitation and mean annual temperature at Ultuna, ca. 10 km south of Uppsala, are 520 mm and 5.4 ^{o}C (1931-60), respectively (Rodskjer & Tuvesson, 1975). In the project, data were collected on most of the major processes concerning nitrogen occurring in arable soils. A detailed description of the field is given in Steen et al. (1984).

The data on nitrate leaching (Bergström, 1987) and mineral-N content of the soil (Bergström, 1986) used in the present study were taken from one of four tile-drained plots (0.36 ha each). The crop grown on this plot was a grass ley (GL) (Festuca pratensis) receiving an annual split N-fertilization of 120+80 kg N ha^{-1} as $Ca(NO_3)_2$. The grass ley was established in 1981 after a fallow year in 1980 and was ploughed in August 1984.

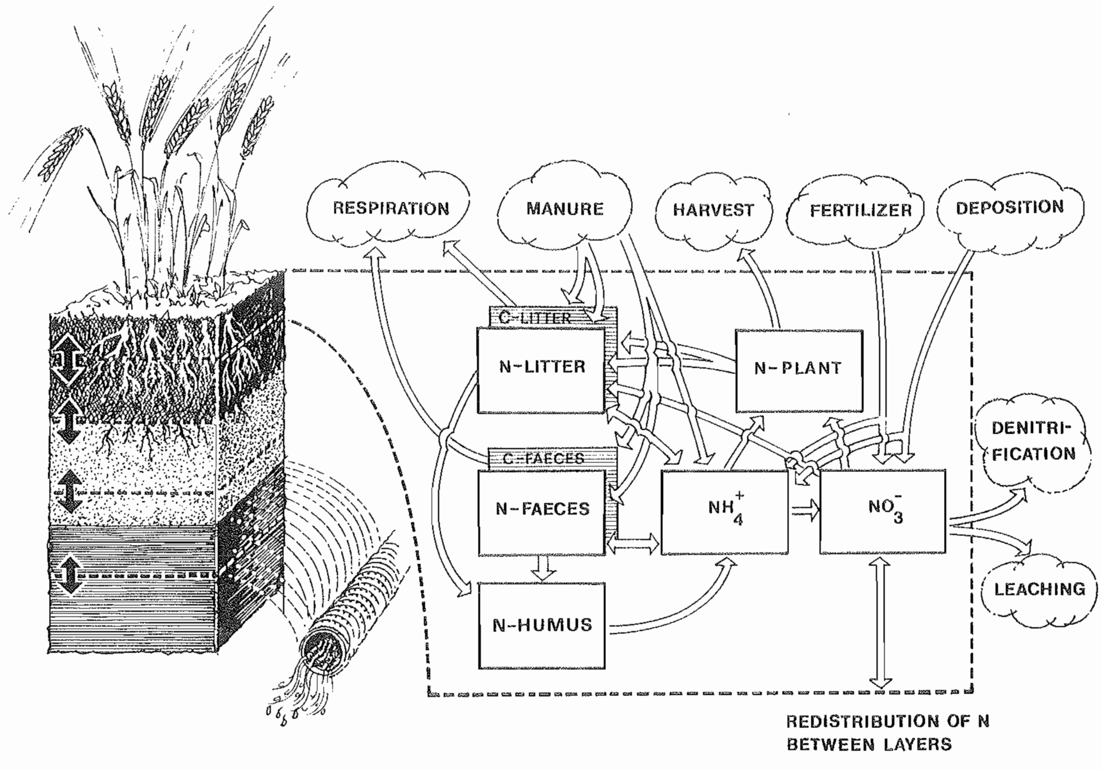

Fig. 1. Flow diagram showing the structure of the nitrogen model. Parts within the dashed line represent the uppermost layer of the soil profile. Subsurface layers have the same structure as the surface layer but receive no direct transfers from fertilizer and deposition. (From Johnsson et al., 1987).

The soil down to a depth of 1.0 m consists of four layers of variable texture and structure (Steen et al., 1984):

i) topsoil, consisting of clay loam, with a mean thickness of 0.27 m;
ii) a fine sand layer with a thickness varying from 0 to 0.50 m;
iii) an oxidized clay layer down to a depth of ca. 0.75 m; and
iv) a non-oxidized clay layer below the oxidized layer.

The topsoil consisted of 15-20 % clay, 2.2 % organic-C, 0.23 % organic-N, and had a pH of 6.0 to 6.5.

MODEL APPLICATION

The adaption of the model to the grass ley was based on parameter values used by Johnsson et al. (1987) for a previous application with barley at the Kjettslinge field. However, some parameter values were changed to more accurately reflect conditions in the grass ley treatment.

Soil water and heat simulation

The simulation of soil water and heat processes, which are inputs to the nitrogen model, is described in Jansson et al. (1987), and only a brief summary is given here.

The meteorological data used as driving variables were measured at the experimental field (Alvenäs et al., 1986). Missing data were substituted with data from the meteorological stations at Marsta and F16, both ca. 30 km south of the Kjettslinge field (Alvenäs et al., 1986; Jansson & Thoms-Hjärpe, 1986).

The soil profile in the soil water and heat model and in the nitrogen model was divided into 5 layers in accordance with the textural horizons in the field and the numerical demands in the water and heat model. The thicknesses were 0.10, 0.17, 0.20, 0.28, and 0.25 m for the upper and lower topsoil, the sand layer, and the two clay layers, respectively.

Drainage was assumed to occur only through the drainage tiles. Measurements of groundwater pressure at the site showed a slight tendency towards a positive gradient upwards from lower layers of the profile (Bergström, 1987), indicating stable conditions or occasionally an inflow of groundwater to layers above the drainage tiles.

Simulated estimates of soil temperatures and soil water content showed clear differences between years (Fig. 2). Two summers, 1982 and 1983, were characterized by dry weather and high temperatures. In contrast, the summers/autumns 1981 and 1984 were comparatively wet. A thick snow cover during the winter of 1981/82 prevented the soil from becoming completely frozen. Detailed descriptions of weather conditions during the experimental period are given in Alvenäs et al. (1986) and Jansson et al. (1987).

Nitrogen simulation

Simulation of crop N-uptake was divided into three periods according to the management schedule practiced in the field:

i) the first period, starting April 1 and ending around June 22 - when the first harvest and the second fertilization were conducted; the first fertilization occurred around May 18;

ii) the second period, between first and second harvest - around August 20; and

iii) the third period, between second harvest and the end of the growing season - October 31.

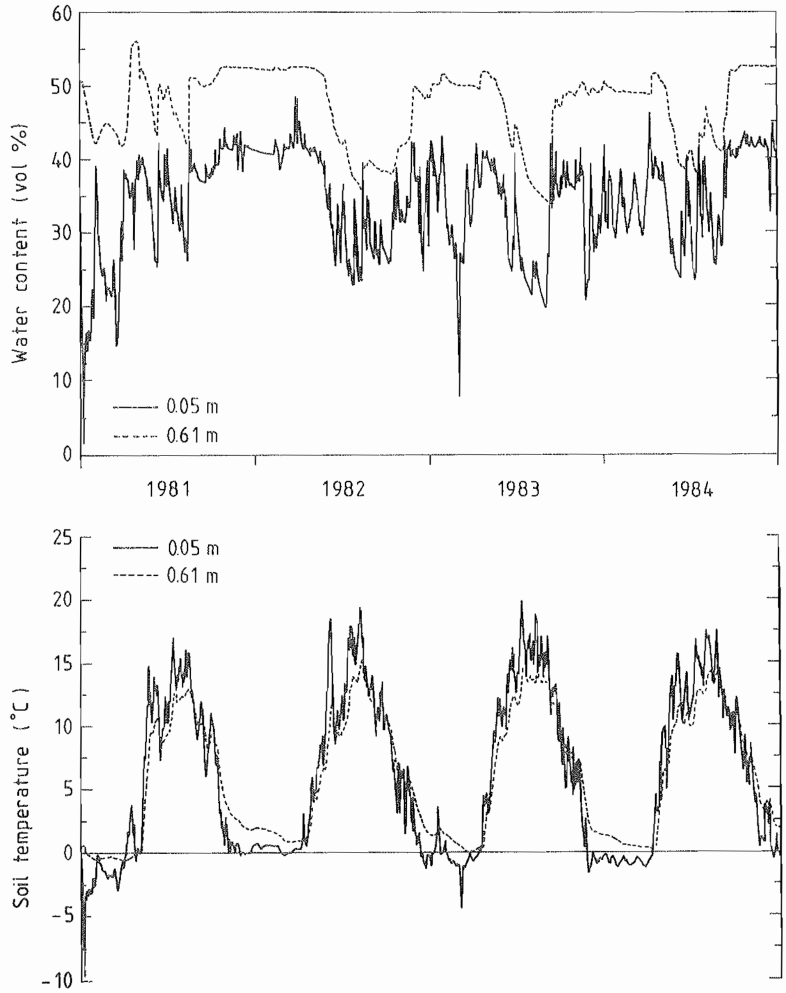

Fig. 2. Simulated daily mean values for soil water content and soil temperatures at depths of 0.05 m and 0.61 m.

Table 1. Simulated yearly values for N-uptake by plants and harvested-N, both based on measurements, and the resulting annual amounts of N in crop residues (g N m^{-2})

	1981	1982	1983	1984	
Total plant uptake	40.0	41.0	37.3	25.6	33.1[a]
Total harvest	17.6	23.6	24.6	20.0	22.9[a]
Crop residues (above- and below-ground)	14.7	14.0	10.3	18.9	23.7[a]

[a] With increased N-uptakes to achieve agreement between simulated and measured soil mineral-N content at the time of ploughing the ley.

Since this study was not aimed at simulating crop growth, and since measurements of above- and below-ground N-uptake were available from the experimental field (Hansson & Pettersson, in manuscript; Pettersson, unpubl.), we decided to use estimates of N-uptake and harvested-N, both based on measurements, as target values for the simulation. The estimates of yearly N-uptake were the sum of measured above-ground N-uptake for each year and estimated below-ground N-uptake, 8 g N m^{-2}, based on measurements from 1982. The simulated yearly uptake of N and harvested-N was modified until an agreement within 0.5 g N m^{-2} was achieved with the estimates (Table 1.). Plant-N at harvest was divided into three fractions: harvested-N, N in above- and below-ground crop residues, and N in living roots. Estimates of these fractions for each growth period were based on measurements (Hansson & Pettersson, in manuscript; Pettersson, unpubl). Considering the whole experimental period, the fractions at harvest had mean values of 0.46, 0.11, and 0.43, respectively. The C/N ratio of above- and below-ground crop residues was set at 25 in accordance with measurements (Hansson & Andrén, 1986; Hansson & Pettersson, in manuscript; Pettersson & Hansson, in manuscript). Primary production data were not available for 1981. Thus, for 1981 the values for parameters determining potential N-uptake were chosen so as to obtain a reasonable agreement between measured and simulated mineral-N profiles at the end of 1981.

To provide the simulated crop with sufficient N, as given by the measurements, the rate constant for humus mineralization was increased by a factor 2.9 compared with the previous barley simulation (see below). At the same time measurements of the mineral-N content of the soil indicated that simulated mineralization during winter was initially too high, which made us change the temperature-response function for microbial activity. At temperatures below 0 ^{o}C, simulated microbial activity was 0; at temperatures between 0-5 ^{o}C there was a linear relationship, while for temperatures above 5 ^{o}C the response was identical to that used for the previous application, with a Q_{10}-value of 3. Field measurements of litter decomposition (Andrén & Paustian, 1987) also indicated that microbial activity decreased substantially as temperatures dropped below 5 ^{o}C.

The soil moisture-response function for microbial activity used in the barley simulation was also used for the grass ley simulation, i.e., the function increased from 0 at the wilting point to an optimal rate at a water content of 10 % above the wilting point, and decreased linearly from 8 % below saturation down to 0.6 at saturation.

The nitrate/ammonium ratio in the soil was found to be considerably lower in the grass ley soil than in the barley soil (Bergström, 1986), which motivated a lowering of the parameter determining this ratio to 1.

The root distribution in the nitrogen simulation was 60, 25, 10, 3, and 2 % of roots in the respective layers downward from the topsoil. This was based on field estimates of roots reported by Hansson & Andrén (1986). The simulated distribution of roots was kept unchanged throughout the growing season.

RESULTS AND DISCUSSION

Mineral-N dynamics in soil

With the exception of the results from 1984, there was good agreement between simulated and observed mineral-N dynamics in the soil (Fig. 3). The differences in soil mineral-N content between years, caused by the varying weather conditions (cf., Fig. 2), were well illustrated by the model. Since the effects of crop uptake on the N-level in soil were taken into account by adjusting simulated uptake so as to reflect actual measurements, the good agreement indicated that the assumed response of mineralization to changes in temperature and moisture were reasonable. However, to supply the crop with sufficient N, it was necessary to increase N-mineralization (Table 2) compared with the previous barley simulation. The humus-N pool decreased by 19 g N m^{-2} during the simulated period (1981-84) (Fig. 4). Simultaneously, litter-N increased by 7 g N m^{-2} (Fig. 4). Of the N taken up by the crop, on average 60 % was harvested and 40 % was returned to the soil in crop residues (Table 1). These results suggest that a perennial grass ley cause a decrease of N in soil organic matter, at least from a short-term perspective. Results compiled from a number of field trials in the United States show that soils in a fertilized grass ley can have a negative N-balance as large as 2-4 g N m^{-2} yr^{-1} (Alexander, 1977). N-inputs from sources not considered in the present version of the model, e.g., non-symbiotic N-fixation, could only substitute a minor part of the crop's nitrogen needs which had to be provided through mineralization in this simulation.

A substantial deviation occurred between simulated and observed values of mineral-N content of the soil during the 1984 growing season (Fig. 3). The simulated value for mineral-N in the soil at the time of ploughing the ley was ca. 8 g N m^{-2} higher than the measured value. Measurements of soil mineral-N content were very frequent after ploughing the ley and revealed a continuous increase in mineral-N content (Fig. 3). Thus, it is not likely that the discrepancy between simulated and measured values was due to incorrect determinations of the soil mineral-N content. Nitrogen losses due to denitrification and N-leaching are generally low in grass leys

Table 2. Yearly litter- and humus-N mineralization (g N m^{-2}).

N-source	1981	1982	1983	1984		
Litter	2.8	8.1	7.8	3.9^a	3.9^b	4.0^c
Humus	9.3	11.2	11.0	6.4^a	5.0^b	5.0^c

[a] Before ploughing. [b] After ploughing. [c] After ploughing, with increased plant uptake during the 1984 growing season.

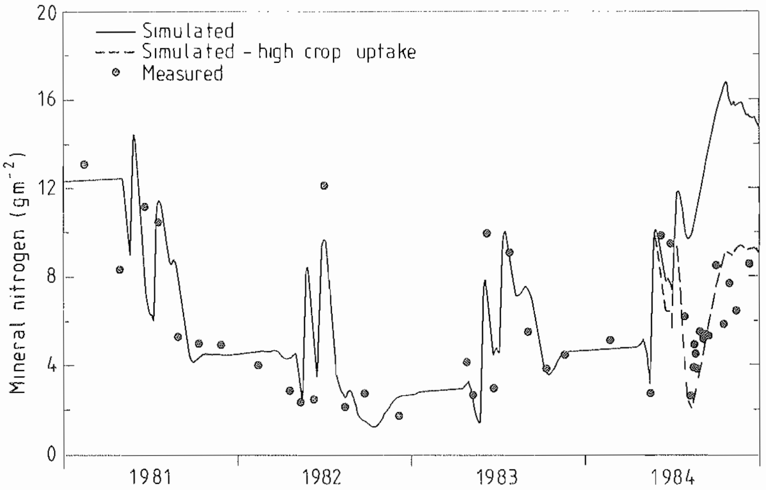

Fig. 3. Simulated and measured mineral-N contents in the soil down to 1 m depth.

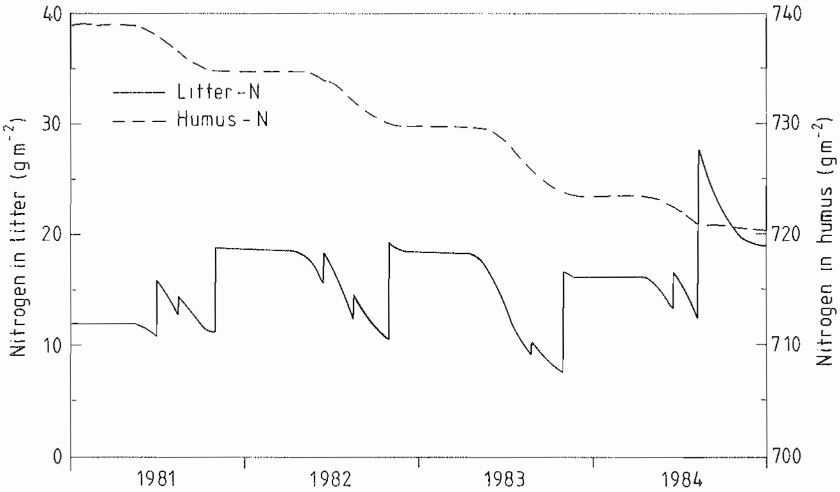

Fig. 4. Simulated seasonal variations in the litter- and humus-N pools in the soil.

(e.g., Klemedtsson, 1986; Bergström, 1987), and even if these N-sinks were underestimated in the simulation, it is difficult to believe that denitrification and leaching could together account for an additional loss of 8 g N m^{-2} during the 1984 growing season, since both losses were overall low during the other years. Before ploughing the ley, simulated denitrification and leaching did not exceed 0.3 and 0.8 g N m^{-2} yr^{-1}, respectively. Neither could the value for mineralization during 1984 be decreased by as much as 8 g N m^{-2}. This would result in a total simulated mineralization of less than 2.5 g N m^{-2} for 1984, up until the ley was ploughed. Consequently, it seemed as if the simulated value for total crop uptake of N was substantially underestimated during 1984. It should be kept in mind that below-ground N-uptake was based on measurements only from 1982 (see above). Thus, part of the underestimated N-uptake during 1984 could possibly be explained by variation in below-ground N-uptake between years. To obtain a reasonable agreement between simulated and measured values of mineral-N content of the soil an additional simulation was performed where the simulated crop uptake was increased by 8 g N m^{-2} during 1984 (Fig. 3) (cf., Table 1).

Nitrate leaching

Clear differences between simulated and measured nitrate leaching occurred during the autumn/winter of 1981/82 and after the ley was ploughed in 1984 (Fig. 5). In the former case, the deviation was mainly due to higher measured nitrate concentrations and, to some extent, it was also due to an underestimate of the drainage-water flow in the simulation. In the latter case, a considerable overestimation of the simulated drainage-water flow resulted in an overestimated value for nitrate leaching in the simulation. Simulated values for nitrate leaching were higher than measured values during the spring of 1982 (Fig. 5), mostly as a result of overestimated drainage-water flow.

Simulated drainage-water flow from the clay layers was limited to only 10 % of the total simulated drainage volume during autumn 1981 (Jansson et al., 1987). The major part of the soil nitrate occurred in the clay, according to both the simulation and field measurements. However, while the simulated nitrate distribution in the soil was fairly static, field measurements showed that there was an overall loss of nitrate from the profile, especially from the clay layers. This indicated that the main flow path was probably deeper than assumed in the simulation. It also explained why the measured nitrate concentrations in drainage water were higher than the simulated concentrations. Indications in the previous barley simulation (Johnsson et al., 1987) also suggested that the flow path was deeper.

A factor contributing to the lack of agreement between simulated and measured nitrate leaching during the autumn/winter of 1981/1982 was the deepening (ca. 0.5 m) of the open ditches surrounding the experimental field, which changed the drainage conditions (Bergström, 1987). An additional complication was the difficulty in simulating drainage behaviour in partly frozen soil (Jansson et al., 1987).

Two dry years, 1982 and 1983 (Alvenäs et al., 1986), preceded the ploughing of the ley in 1984, resulting in the absence of both simulated and measured leaching

(Fig. 5). After the ley was ploughed in August, with wetter conditions prevailing, drainage-water flows and leaching started earlier than predicted by the model but stopped abruptly in October (Figs. 5 and 6). Once drainage-water flows and leaching started in the simulation they continued throughout December (Figs. 5 and 6). On the other hand, the simulated drainage pattern agreed well with drainage measurements from lysimeters at the Kjettslinge field (Bergström, 1987).

There are two reasonable explanations for the drainage pattern observed during 1984. If macro-pores were created in the clay as a result of the dry weather prior to ploughing, it is plausible that the first water reaching the drainage tiles came through these pores. A subsequent swelling of the clay, which should have occurred after the rainy period, should have halted the flow through macro-pores, resulting in a vertical matrix flow. It is also conceivable that the structurally different soil horizons, with substantial differences in hydraulic conductivity, could have created water-flow barriers in the soil. After ploughing the ley, the first rains should have rapidly percolated through the topsoil and sand layer, both with high conductivity. Since the clay had a considerably lower conductivity (Steen et al., 1984), a lateral movement of water to the drainage tiles should have then preceded any pronounced matrix flow. The bulk of simulated water flow in September between the sand layer and the upper clay layer roughly coincided with the measured peak in drainage-water flow (Fig. 6). Once the vertical matrix flow became dominant in the field, the groundwater storage in the subsoil had to be restored before any tile-drainage could be expected, explaining the absence of measurable nitrate leaching and measurable drainage-water flow during November and December 1984 (Figs. 5 and 6). Since neither flow through macro-pores in dry soil directly to drainage tiles nor lateral water flow above the groundwater table were considered in the simulation, the two hypotheses may explain the discrepancy obtained between simulated and measured nitrate leaching during 1984.

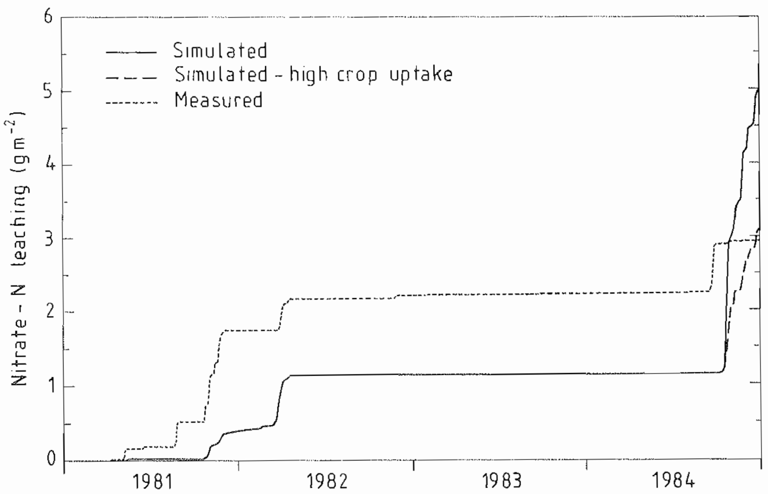

Fig. 5. Simulated and measured cumulative nitrate leaching.

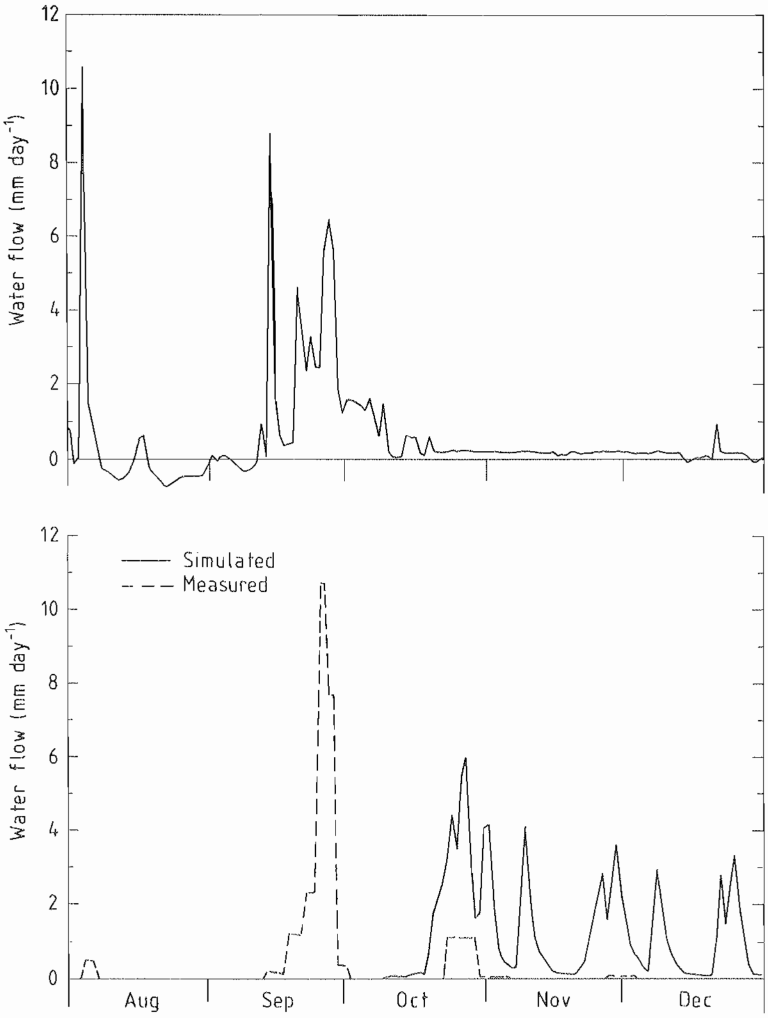

Fig. 6. Simulated water flow between the sand layer and the upper clay layer (upper figure), and simulated and measured water flows through the drainage tiles (lower figure). Both figures refer to the period Aug.-Dec. 1984.

Effects of ploughing the ley

A substantial increase in the mineral-N content of the soil occurred after ploughing the ley (Fig. 7). During the first month after ploughing, the soil mineral-N content in this simulation and in the measured data increased at nearly identical rates (Fig. 7, curve a). Since the litter-N buildup was less than 1 g N m^{-2} during the experimental period, up until the ley was ploughed, it contributed little to the large increase in soil mineral-N content that occurred after ploughing the ley. Instead, the main reasons for this increase were the high input of crop residues of relatively high N-content, the absence of a growing crop after ploughing, and early ploughing at a time when soil temperatures and soil moisture conditions were favourable for mineralization. To come to this conclusion a simulation was performed without ploughing and with continuing crop growth (Fig. 7, curve c). Furthermore, to determine the effects of crop residue input on the soil mineral-N content after ploughing, a simulation was made assuming the absence of ploughed-in crop residues (Fig. 7, curve b). We were unable to determine whether or not a litter-N buildup occurred in the field based on the available measurements.

CONCLUSIONS

The total yearly N-uptake by the grass ley was about 40 g N m^{-2}; simulated mineralization was about 20 g N m^{-2} yr^{-1}, and leaching was about 1 g N m^{-2} yr^{-1}. Thus, small variations in N-uptake can easily overshadow the changes in soil mineral-N content caused by yearly N-leaching. Consequently, the accurate simulation of N-uptake by plants is vital for determining the soil mineral-N status and all N-flows influenced by the mineral-N content of the soil.

The simulation indicated that measurements in tile-drained plots can considerably underestimate values for nitrate leaching. To properly quantify leaching losses in the field, additional information on water-flow paths in the soil is needed.

The large increase in simulated soil mineral-N content after ploughing the ley was mainly due to a high crop residue input and no N-uptake after ploughing. In contrast, the small litter-N buildup had little effect on the soil mineral-N content.

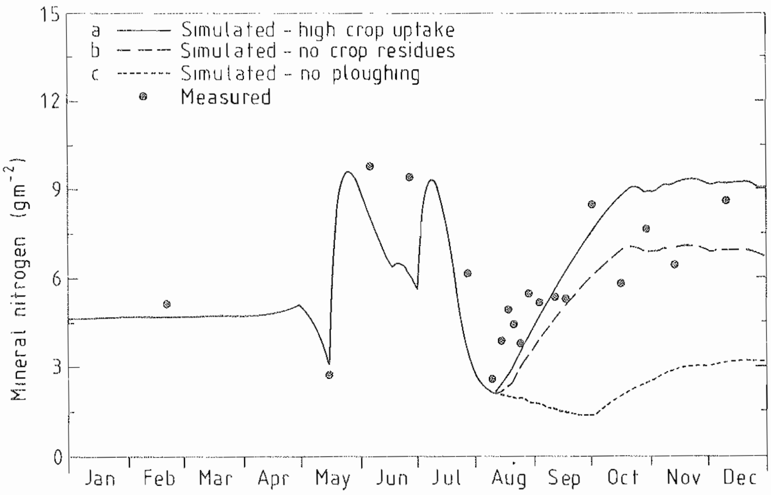

Fig. 7. Simulated and measured soil mineral-N contents during 1984 down to a depth of 1 m. After ploughing in August the curves represent (a) input of crop residues (b) no input of crop residues (c) continuing crop growth.

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