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N management in agrosystems in relation to the Water Framework Directive

Proceedings of the 14th N Workshop
October 2005, Maastricht, the Netherlands

J.J. Schröder & J.J. Neeteson, editors

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Preface

'Nitrogen management in agrosystems in relation to the Water Framework Directive' was the overall theme for the 14th Nitrogen Workshop, held from 24 to 26 October 2005 in Maastricht, the Netherlands. The workshop was organised by the business unit Agrosystems Research of Plant Research International of Wageningen University and Research Centre, Wageningen, the Netherlands and the Congress Office of the University of Maastricht, the Netherlands.

Previous workshops were held biannually since 1982. About 20 delegates, predominantly from the UK, participated to the first workshop. However, since 1992 the workshops have attracted an increasingly international audience. The number of delegates in the latest workshop in Maastricht exceeded 200 from 28 countries and five continents.

This report is a synthesis of the contributions from the 14th Nitrogen Workshop, drawn from oral and poster presentations and from the discussions in the various working groups. All contributions to this report have been refereed by two independent scientists.
We are most grateful to the reviewers for providing this service:
Frans Aarts, Hein ten Berge, Jules Bos, Sjaak Conijn, Wim Corré, Tom Dueck, Frits van Evert, Huib Hengsdijk, Don Jansen, Herman van Keulen, Annette Pronk, Frank de Ruiter, Bert Smit, Jan Verhagen, Koos Verloop and Adrie van der Werf.

This book is divided in eight main sections:

1. N flows at the regional level: policy implications of the Water Framework Directive
2. N flows at the farm level: indicators and tools for improved N management
3. Manure quality: can it be manipulated and what are the effects on whole-farm N efficiency?
4. Grassland renovation: prudent or risky?
5. Buffer strips and catch crops: cosmetic or beneficial?
6. Crop-related indicators: is the crop able to tell farmers what to do?
7. Soil-related indicators: ex ante hints or ex-post evaluation?
8. N turnover and losses at the plot scale: are detailed studies still informative?

The organising committee of Agrosystems Research:

Jacques Neeteson (Convenor)
Irene Gosselink (Coordinator)
Jaap Schröder
Hein Korevaar
Pieter van de Sanden

Working group 1

N flows at the regional level: policy implications of the Water Framework Directive

Report of Working Group 1

Nitrogen flows at the regional level: policy implications of the Water Framework Directive

Report by Berge, H.F.M. ten[1*] & Bolt, F.J.E. van der[2]

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Which measures are best suited to achieve the ultimate Water Framework Directive (WFD) goal: to assure that all water bodies are in good ecological status by 2015? This is the issue that agricultural and environmental research must address. How much have we progressed on the road to providing answers, and what should further research be focussed on?

The working group meeting was structured around the steps we consider essential in the process of identifying the most effective measures for achieving the WFD goals:
(a) regional assessment of current water quality and nitrogen(N)-load;
(b) assessment of desired quality and corresponding load;
(c) inventory of potential measures; and
(d) evaluation of impact.

Participants were asked to ponder about setting the research agenda for the next few years, to fulfil our obligations as research community in the light of the WFD. Several speakers gave introductory presentations to start off the discussions on the respective aspects listed. Based on submitted abstracts, there appears to be a strong emphasis now on Step (a). Is there a need to shift emphasis? Which elements are missing? Do we need to reconsider methodologies? Do we really need to know processes in such detail? These were the issues discussed; the results are outlined below.

As stated, research so far appears to be largely focussed on the regional assessment of current water quality and N-load. Also, there is quite some activity in modelling to draw up potential measures (c) and to make ex-ante assessments of their expected impact (d). It need not surprise that the material available at this stage regarding measures and their impact, is largely from modelling studies and that experimental or monitoring results are scarce. Afterall, the WFD has been announced recently. Legislative measures may be costly for stakeholders, and so must be taken with greatest care. Next, because many actors may be involved, they must pass the political process which is time consuming. Lastly, once in place, measures may perhaps take effect only slowly and the impact on target variables will show in the course of years.

Apart from the observation that data material on the impact of measures is still very limited, it was felt that there is insufficient attention, at the moment, for defining the desired water quality, at least within the community represented at the meeting. Some researchers argued that there is a large gap between the 'agro-world' and the 'water-world'. Workers in the water-world do give much attention to defining the desired quality, but they seem to be little connected with those working on mitigating diffuse emissions from agriculture.
Another contrast identified is that between scales. Much work seems to be conducted at the farm scale, little work at the regional scale. Some of the modelling studies were focussed at the regional scale, and these were highlighted in the meeting. An examplary study was presented by Merete Styczen from Denmark (see contribution in this book), where entire catchments were modelled including the impact on the water quality in the downstream Odense and

Ringkobing fjords. In the subsequent discussions, this served as a frame to highlight various viewpoints by the scientists present. Key points are summarised here:

- The design of monitoring programs must fulfil the requirement that results will expose the impact of policy.
- Models are very effective tools in 'cleaning up' data sets; all data sets from monitoring contain errors and some of the most obvious can be identified with the help of models; models are also considered indispensable in the interpretation of monitoring results.
- Surface water systems are interconnected and form one widespread continuous system. Systems analysis and geohydrological modelling ('particle tracking') are key tools to relate observed water quality to the sources associated; without models this is hardly possible unless monitoring density is very high..
- This also helps to implement measures only where they matter most, thus alleviating pressure on the larger farming community; efforts must be targetted at sensitive zones, usually zones with little natural capacity to eliminate undesired pollutants such as areas where water flows directly to surface waters without passing the redox cline.
- We should be aware of non-agricultural (often point) sources; now there is little attention for their relative importance.
- As for nitrogen, more attention must be given to organic N-species. These turn out to represent an important fraction of N found in surface waters.

As for the WFD, ecological objectives for surface water quality will likely be more restrictive than nitrate emissions via groundwater. In addition, phosphorus is highly relevant for surface water whereas it received limited attention so far, when research was more concerned with groundwater and, hence, nitrogen.

As for farming, there will be an increased public pressure to produce with lower emissions. Research must facilitate the development of more efficient techniques and sustainable systems. This is the well known track of current work. Within this domain, the question was raised why livestock-systems seem to respond more directly to a reduction of inputs than arable systems. Pools of organic N and their turnover must be better understood, including the role of tillage.

It was recognised that the nature of crop and animal growth comes with inherent inefficiencies which cannot be overcome unless we switch to systems now regarded as very or too innovative, that is, using entirely different organisms to generate our food.

Keeping with the same crops and animals, it is not obvious beforehand that 'extensive systems' are more nutrient-efficient. They are, for sure, not more land-efficient as they require more surface area per unit produce. Rather than just aiming to reduce nutrient use in agriculture, we should be more aware of options available to achieve the same goals at catchment scale by segregation of agriculture and 'nature'. This requires (rules for) land use planning, allowing rather intensive agriculture in some parts, at the expense of permissible activities in other parts.

In areas where we will, indeed, continue to farm in a way roughly comparable to current ways, only better optimised within the natural limitations of the production species, better use must be made of options to reduce nutrient loads via retention in surface water systems ('effect-measures'). Examples are buffer zones, zones with specialised catch crops, riparian purification zones. There is a need for systematic comparison and classification of wetland bodies, with respect to their potential for water purification. A typology must be set up for this purpose. Then, given the characteristics of specific areas, options for water management must be explored, and the impact of measures on ecology specified.

In specific cases, as the Dutch peat district, it is obvious that only drastic changes in hydrology management can stem the release of nutrients from the original parent materials. Drained peat releases nutrients irrespective of agricultural land use, and (partial) flooding might prove, in the end, the only solution to reduce nutrient loads substantially.

Poster Presentations

Cost-Cube: A measure centric model for characterisation of diffuse pollution

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Abstract

A new conceptual modelling framework that explicitly represents the mode of action of mitigation measures for reducing diffuse pollution is presented. The parameterisation and application of this 'measure-centric' framework is given, as well as its ultimate deliverable, a cost-curve analysis of mitigation measures. The framework centres on the Cost-Cube, an explicit representation of the proportions of the total pollutant loss due to aspects of the source, mobilisation and transport dimensions of diffuse pollution. The three dimensions of the Cost-Cube are sub-divided into three aspects of diffuse pollution, representing type of source, process of mobilisation, and path of transport. The volume of the Cost-Cube is in proportion to the total loss for a specific crop-soil-climate condition. The model is parameterised for a reference set of conditions and a vector model used to interpolate to other environmental and soil conditions. The model has been applied to six different farm systems. For each farm system, a list of applicable pollutant control measures was produced, and the annual cost of implementation calculated in proportion to land area, livestock numbers and quantity of managed manure. A mathematical model was then used to calculate the best order of implementation of the applicable measures to optimise benefit over cost.

Keywords: ammonium, BOD_5, cost curve, modelling, nitrite, pathogens

Background and objectives

The Water Framework Directive requires implementation of land management options that reduce the totality of diffuse pollutant losses, including transfers of nutrients, sediment, organic matter (giving rise to an oxygen demand in the watercourse), agrochemicals and pathogenic organisms. To do this effectively, we require an understanding of the key behavioural characteristics of diffuse pollutant groups and the potential for targeting of measures. As a first step in this direction, we present a new conceptual modelling framework that explicitly represents the mode of action of mitigation measures for reducing diffuse pollution (including various forms of inorganic nitrogen). The parameterisation and application of this 'measure-centric' framework is given, as well as its ultimate deliverable, a cost-curve analysis of mitigation measures.

Materials and methods

The model framework centres on the Cost-Cube, an explicit representation of the proportions of the total pollutant loss due to aspects of the source, mobilisation and transport dimensions of diffuse pollution (Figure 1). The three dimensions of the Cost-Cube are sub-divided into three aspects of diffuse pollution, representing type of source,

process of mobilisation, and path of transport. The volume of the Cost-Cube is in proportion to the total loss for a specific crop-soil-climate condition.
A measure that reduces pollutant losses can be visualised as targeting one or more of the cube dimensions or aspects, reducing their volume and hence the magnitude of total pollutant loss. A farm system can be represented by one or more Cost-Cubes, making explicit the total pollutant loss and the relative importance of the pollutant pathways.

Figure 1. Dimensions and aspects of the Cost-Cube model.

The model was parameterised for a reference environment condition by the questioning of diffuse pollution experts on their opinion of the relative importance of each pollution pathway. A simple mathematical model (the Cost Cube Vector model) of environmental vectors (the mobilisation or transport mechanisms, such as over land flow) was then used to interpolate the expert evaluation to other environmental conditions. The model can be used to investigate the relative importance of different aspects of the source, mobilisation and transport dimensions under a wide range of environment conditions.

Cube Name:	Ammonium		
Cube Description:	Ammonium Loss at [Heavy : Arable : Dry : Drained]		
Cube Export Coefficient:	1.00		
Active:	TRUE		
Cost-Cube Dimensions / Aspects (Relative Proportions)			
SOURCE	**Internal**	**External**	**Recycled**
	0.10	0.45	0.45
MOBILISATION	**Internal**	**External**	**Recycled**
Detachment	0.05	0.00	0.00
Solubilisation	0.95	0.10	0.10
Contingent	0.00	0.90	0.90
PATHWAY	**Detachment**	**Solubilisation**	**Contingent**
Surface	0.23	0.20	0.20
Preferential	0.77	0.75	0.80
Through	0.00	0.05	0.00

Figure 2. Model screen for the expert assessment of the contribution of each Cost-Cube dimension and aspect to total loss of ammonium under the reference condition.

Figure 2 shows the results of the expert parameterisation for the reference condition for ammonium loss. This exercise was also conducted for nitrite, BOD_5 and pathogens. For each pollutant, the principal sources were fertilisers (external aspect) and manure (recycled aspect), and the principal mechanisms of mobilisation were solubilisation and contingent or incidental losses. The dominant transport pathway was preferential flow in the cracks and tile drainage.

Six representative farm systems were defined (in terms of total livestock numbers, crop areas and manure production) for application of the Cost-Cube model. For each farm system, the Cost-Cube Vector model was used to calculate total pollutant losses for combinations of climate, soil drainage and soil texture typical of England and Wales.

For each farm system, a list of applicable pollutant control measures was produced, and the annual cost of implementation calculated in proportion to land area, livestock numbers and quantity of managed manure. The efficiency of each mitigation measure was measured on a simple indicator scale and is specific to each pollutant source, mobilisation and delivery pathway. The cost-curve calculation therefore embraces the conceptual model of measure types portrayed in the measures matrix and Cost-Cube framework.

Results and discussion

Finally, a mathematical model (the Cost-Cube Cost-Curve model) was used to calculate the best order of implementation of the applicable measures, to achieve maximum pollution reduction for minimum cost. Table 1 lists a sample set of results for the control of ammonium losses from the model dairy system. The table shows the generic form of a cost-curve, with increasing ratio of cost to benefit as more measures are implemented on the farm. In this example, money is initially saved by taking advantage of the nutrient content of manure to reduce fertiliser applications. Measures controlling the timing and placement of manure are next, with a modest but cheap reduction in pollutant loss. The most expensive measures, such as reducing stock account (and hence farm revenue) are implemented last. They apparently have minimal effect in reducing pollutant loss because the preceding measures have already significantly reduced losses.

Table 1. Modelled cost-curve for the control of ammonium losses from the model dairy system averaged across each of the climate, drainage and soil texture Cost-Cube scenarios.

Cost-step	Measure Description	Pollutant loss (%)	Total Annual Cost (£)
0	Baseline	100	0
1	Integrate Fertiliser with Manure	77	-8,467
2	Introduce Clover to Grassland System	72	-10,782
3	Do Not Apply Slurry to Well Connected Areas	62	-10,499
4	Avoid Grazing High Risk Fields when Wet	58	-10,349
5	Establish Artificial Wetland	44	-8,515
6	Change Fertiliser Type	34	-6,622
7	Allow Drainage to Deteriorate	30	-5,122
8	Avoid Slurry Spreading at Times of High Risk	25	-1,816
9	Reduce Dietary Nitrogen Intake	17	9,014
10	Do Not Apply Fertiliser to Well Connected Areas	16	10,949
11	Export 50% of Slurry	14	16,593
12	Aeration of Slurry	12	22,994
13	Install Hedges and Reduce Field Size	11	33,288
14	Establish Riparian Strip	10	38,054
15	Reduce Stock Count	8	88,663
16	Avoid Fertiliser Spreading at Times of High Risk	8	103,663
17	Reduce Field Stock Rates when Wet	8	111,983
18	Batch Storage of Slurry	8	132,093
19	Use Slowly Available Nitrogen Fertiliser	8	187,098

Conclusions

The framework is highly flexible, is able to take account of mutually exclusive measure options, and can be easily edited to analyse additional measures.

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Soil and Groundwater Nitrate Responses to Dairy Farming Practice

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Abstract

A three-year investigation undertook to measure nitrate concentrations in subsoil and groundwater under a grassland dairy farm managed at a stocking rate of 2.4 LU ha^{-1} (LU = livestock units). The hydrogeological regime is designated as 'extremely vulnerable' with soil cover less than 3 m over karstic, fissured limestone aquifer. From borehole piezometers and ceramic suction cups in soil, average annual nitrate concentration in the subsoil was 31 mg l^{-1} NO_3 and in groundwater was 58 mg l^{-1} NO_3, exceeding specified target levels. Although highly variable, nitrate concentrations were directly correlated with surface nitrogen loading, corresponding to different dairy farming agricultural management practices (grazing only, grazing with dirty water application, grazing with cutting for silage). Dual flow mechanisms – soil matrix and preferential/bypass flow were found to occur resulting in minimum travel times to groundwater, at 25m depth, of 18 days. Principal driving mechanism for nitrate response was organic N loading at surface but coupled with relevant hydraulic loading (rainfall and dirty water). Results confirmed assessed contamination risk to groundwater under extremely vulnerable conditions.

Keywords: dairy farming, groundwater, nitrate, preferential flow

Background and objectives

A farm-scale hydrogeological investigation was established on a 50 ha dairy farm in north Cork, the south west of Ireland (national grid co-ordinates R813 008). This grassland farm is characterised by freely draining sandstone till, which forms the subsoil overlying a karstified-limestone bedrock aquifer. Groundwater flow in this limestone bedrock unit is characterised by fissure flow, that is, predominantly through interconnected, solutionally enlarged fracture zones. The average combined soil and subsoil thickness was 2.5 m, but it ranges from 0 to 4.5 m, conforming to the karst terrain. These characteristics of the subsoils in combination with the nature of the karstic aquifer create a groundwater environment vulnerable to potential contamination. The stocking density on the farm was ~2.4 LU ha^{-1}, which suggests an organic loading rate of 204 kg N ha^{-1} yr^{-1} in Ireland. All farm fields received 300 kg N ha^{-1} yr^{-1}, on average, as inorganic nitrogen fertiliser. The farm, in accordance with common Irish dairy farming practice, has dedicated management zones for grazing, grazing with dirty water application, grazing with one cut of silage and with two cuts of silage. Curtin's farm is an ideal study site because it is located on a plateau, close to a groundwater divide. Therefore, it was determined that nitrate concentrations in the underlying groundwater mainly reflected the influence of recharge percolating vertically through the subsoils of the farm – and are not influenced by the surrounding groundwater regime. The aim of the research was to define and measure subsoil and groundwater-nitrate response to meteorological and agronomic loadings, at the farm-scale.

Materials and methods

The investigative approach involved quantification of all loadings, both meteorological and agronomic, and monitoring of the response of the subsoil and groundwater system at numerous locations on the farm. Meteorological loadings were calculated using daily measured meteorological data, derived estimates of

evapotranspiration using the Penman-Monteith approach (FAO, 1998) and soil moisture deficit accounting (Aslyng, 1965). All nitrogen loadings (inorganic fertiliser, grazing animal's depositions, applied slurries and dirty water) were recorded daily for each field in each management zone. Nine monitoring boreholes were drilled and monitored for water level and hydrochemical response to dairy farming practice. Boreholes were instrumented with piezometers to a depth of 27-30 m below ground level (bgl) and 5-10 m below the water table. The groundwater sampling zone was isolated from direct contamination from the land surface by a cement and bentonite grout seal. Groundwater levels were recorded in each piezometer using a manual dip-meter each week and before all sampling events. Each piezometer was purged before sampling by removing three times the casing volume (i.e. the volume of standing water in the piezometer). Sampling frequency was twice monthly throughout the first year of the study but the summer sampling frequency was reduced in the second year, to once a month. Duplicate samples were collected from each piezometer and analysed in the laboratory. In addition to the suite of nutrients, a suite of ions was analysed for each sampling event in order to define the natural hydrochemical regime and the agricultural signal in the groundwater. The subsoil (at 1 m depth) in each of the four management zones was instrumented with ceramic cups. The free-draining subsoil is never saturated; there is no perched water table within the subsoil. Three replicate fields in each management zone contained eight ceramic cups each, resulting in a total of 96 cups. Subsoil porewater samples were extracted, under a 50 kPa suction, each week for the duration of the 'drainage season', which was typically from October to April or May of each year. A tracing experiment was conducted to investigate the rate of vertical migration of surface applied water and solutes through the unsaturated zone to groundwater.

Results

Annual rainfall was 1071 mm, on average, from which effective rainfall available for drainage, through the soil and subsoil, to groundwater was calculated to be 560 mm, on average (Table 1). Nitrogen loadings analysis showed that there was large spatial variation in organic nitrogen loading rates in the different dairy management zones (calculated range of 165 kg N ha^{-1} yr^{-1} in the grazing only zone and 471 kg N ha^{-1} yr^{-1} in the grazing with dirty water irrigation zone). The water table was 25 m bgl, on average, with a maximum annual range of 15 m, typical of karstic limestone. Measured groundwater nitrate concentrations in the groundwater ranged from 4 – 136 mg l^{-1} NO_3 over the entire study period and showed large variation both spatially and temporally. The 3-year mean groundwater nitrate concentration was 58 mg l^{-1} NO_3. The associated 3-year mean subsoil porewater concentration for all treatments was 31 mg l^{-1} NO_3. Annual average results are shown in Table 1. On an annual scale, groundwater nitrate concentrations were positively correlated (R^2 = 0.95) with grazing activity at the field scale (Figure 1). Results from the bromide tracing experiments in this freely draining soil, indicated that contaminant transport to groundwater was by a dual flow mechanism, i.e., by matrix as well as preferential flow. Bromide was transported to groundwater, by preferential flow, in 18 days under spring recharge of 50mm and a single dirty water irrigation rate of 16mm. Results suggest that delivery to groundwater, by matrix pulse, was in the following year's recharge season.

Table 1. Meteorological loadings and average nitrate concentrations (mg l^{1} NO_3)

Study Year	Rainfall (mm)	Effective Rainfall (mm)	Subsoil (mg l^{-1} NO_3)	Groundwater (mg $l^{-1}NO_3$)
2001-2002	1163	679	35	72
2002-2003	995	464	19	55
2003-2004	1055	537	40	48

Figure 1. Relationship between grazing activity in 2001 and groundwater nitrate concentrations in the 2001-2002 'drainage season' (the trend line is indicated); BHC.1 = groundwater piezometer number 1, etc.

Conclusions

The results from this three-year study indicate that nitrate nitrogen concentrations in groundwater arising from current farming practices on an intensive dairy farm, located in an area zoned as 'extremely vulnerable' in North Cork, do not meet groundwater quality targets, as currently specified. Rainfall and agricultural management practices that increase either the hydraulic or nitrogen loading, or both, were identified as important contributing factors. Hydraulic loading was shown to be a critical driver of nitrate responses. Spikes in groundwater concentrations of phosphorus, potassium, ammonium and nitrite were observed in response to certain recharge events, again suggesting a vulnerable hydrogeological setting and the influence of preferential flow. Nitrate concentrations recorded from both the subsoil ceramic cups and groundwater piezometers were positively correlated with the different management practices in zones at the surface. However, the ceramic cups, in the subsoil, generally underestimated the impact of agricultural practice on groundwater nitrate concentrations. This may be due to the ceramic cup sampling methodology under grazing conditions. The management practices that resulted in the highest nitrate concentrations in both subsoil and groundwater included dirty water irrigation and animal grazing intensity. Results suggested that the organic loading rate dictated nitrate loss responses. The results indicate the potential for changes in management to achieve groundwater quality targets. The designated hydrogeological vulnerability category was shown to be a good indicator of risk and groundwater vulnerability.

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Nitrogen loading of surface water in a polder used for dairy farming

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Abstract

In agriculturally used peat land areas standards for nitrogen concentrations are frequently violated. In this study nitrogen flows of an intensively managed grassland on peat soil were analysed. The large magnitude of hard to manage N fluxes like soil N mineralization and field denitrification complicated the relation between agricultural activities at field level and nitrogen loading of surface water. Also, in the surface water additional sources and processes further veil the direct relation between nitrogen inputs at field level and nitrogen discharge of the ditch.

Keywords: dairy farming, nitrogen, peat, surface water

Background and objectives

In agriculturally used peat land areas in the Netherlands, surface water quality standards for nitrogen (N) are frequently violated. Nitrogen loads from agricultural fields to surface water are typically 40-85 kg N $ha^{-1}y^{-1}$ (RIVM, 2002) and originate from fertilizers and manure, atmospheric deposition, seepage (if present) and net mineralization of peat soil. The objective of this paper was to quantify the contribution of these N sources in a polder used for dairy farming and to evaluate current measures to reduce N inputs from fertilizers and manure to agricultural fields to improve surface water quality.

Materials and methods

The present study was carried out in the 'Vlietpolder' (52°10' N; 4°36' E). One field (40 x 400 m) was selected for intensive monitoring and was surrounded by ditches at 3 sides. In one dead-end ditch, called 'experimental ditch' a flow meter for discharge proportional sampling was placed (Figure 1). Monitoring data included detailed farm management information, hydrologic inventories, soil water composition, meteorological data, field denitrification, ditch denitrification and mineralization of soil N. Briefly, denitrification measurements were performed using acetylene inhibition in undisturbed soil samples (field denitrification) and in tubes placed in the ditch sediment (ditch denitrification). The tubes were 1 m and about 20 cm elevated above the surface water (headspace). The composition of the soil solution was measured every two weeks using ceramic suction cups in a transect perpendicular to the ditch (6 distances x 6 depths). Soil N mineralization was determined by so-called zero-N plots. A more elaborate description of methods used for analyzing N fluxes in the Vlietpolder can be found in.

Figure 1. Location of the Vlietpolder (left) and of experimental field in the Vlietpolder (right). During periods of water shortage water is let in at the inlet station and flows through the ditches (indicated by lines, middle). During periods of excess water (mainly winter) water is pumped out at the pumping station to the regional surface water.

Results and discussion

On average the soil N balance had a surplus of 134 kg $ha^{-1}y^{1}$ (Table 1), but when ranges in estimations due to temporal and spatial variations were taken into account, input and output fluxes partly overlapped (not shown).

Table 1. Average soil N balance of the experimental field (kg $ha^{-1}y^{1}$) for the period 2001-2003.

Input		Output	
Fertilizers, manure and cattle droppings	452	Harvested and grazed grass	387
Soil N mineralization	244	Field denitrification	168
Atmospheric deposition	31	Leaching from fields	38
Sum	727		593

The composition of the shallow soil solution showed a remarkable accumulation of N just below the experimental ditch. Most probably, this local accumulation of N just below the ditch reaches the ditch water by convection and/or diffusion. An estimation of this additional N load to the surface water yielded on average 8 kg N $ha^{-1}y^{-1}$, and was founded by Cl mass balances (van Beek *et al.*, 2004). When this input term, which is mainly caused by anaerobic mineralization, is included in the soil N balance, the surplus decreases to 19-228 kg N $ha^{-1}y^{1}$. However, N mineralization rates from soil are likely to be overestimated, because of prolonged N release from past fertilization (no clover was present). Also, field denitrification rates are likely to be underestimated because of methodological problems. An improved surplus was achieved by using minimum measured annual rates for mineralization rates and maximum measured annual fluxes for denitrification. The improved N surplus equaled 13-42 kg N $ha^{-1}y^{1}$, which is in the same range as losses via NH_3-volatilization (not measured).

Similarly, the ditch N balance was set-up. The average N balance of the experimental ditch showed a surplus of 8 kg N $ha^{-1}y^{-1}$ (Table 2). This surplus could be caused by biases and uncertainties in flux quantifications, but could also be stored in the ditch sediment.

Table 2. Average N balance of the experimental ditch (kg $ha^{-1}y^{-1}$) for the period 2001-2003.

Input		Output	
Inlet water	2	Denitrification	3
Atmospheric deposition	10	Groundwater recharge	<1
Seepage	0	Slush application	15
Accumulation of N below the ditch	8	Discharge at flow meter	32
Leaching from fields	38		
Sum	58		50

We used several methods to calculate the contribution from the use of fertilizers and manure to the N loading of the surface water, which together ranged between 20 and 50% of the total loading (Van Beek *et al.*, 2004). Another major input was mineralization of soil N and a major output was field denitrification (Table 1). Due to the magnitude of these two hard to manage fluxes, the contribution of agricultural inputs in the soil N balance is diluted. Additionally, leached N was diluted with more than 50% by other N sources in surface water (Table 2). Hence, the direct effect of reducing fertilizer inputs to improve surface water quality in the Vlietpolder will only result in minor improvements, because of 1) dilution of fertilizer- N with other, often hard to manage, N sources in the field and 2) dilution of leached N with other N sources in the surface water. Also, field denitrification exceeded leaching with about a factor of about 4.5, indicating that the majority of the reduction of N inputs will result in decreased field denitrification rates.

Conclusions

In the Vlietpolder several sources of N contributed to the N loading of the surface water. Reducing one source of N to improve surface water quality will only result in modest improvements. This is caused by the large size of hard to manage N fluxes like soil N mineralization and field denitrification. Also, in the surface water additional sources and processes further veil the direct relation between N inputs at field level and N discharge at the flow meter.

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Sensitivity of using different soil type representations for field and regional simulation of N leaching

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Abstract

A method for simulating N leaching on a regional scale based on field scale simulation of water and N balances and crop production is presented. The method is used in regional analysis in a western region of Denmark (Ringkøbing region). The analysis includes 7000 farm observations, 108.000 fields and 362.000 hectares of arable land. Farm specific cropping systems and N fertilisation schemes were set up from national registers on crops and actual N use. Eleven years of simulation using either detailed soil data or only the dominating soil types was conducted. The uncertainty introduced by using only the dominating soil type was analysed based on analysis of the residuals. The results were summarised and analysed at different scales: field, farm, and four grid scales (1x1 km 2x2 km, 5x5 km and 10x10 km). The study showed that if analysis is conducted on a 2x2km grid scale level or lower scale, we recommend using detailed soil data in the field scale simulations. On a higher scale the dominating soil types can be used without seriously increasing uncertainty. We found that simulated N leaching was more sensitive to the soil type selection than was N harvest. With respect to simulated percolation and evapotranspiration the sensitivity was at the same level.

Keywords: nitrogen leaching, Regional analysis, soil type, uncertainty

Background and objectives

Simulation models are valuable tools in assessing the environmental impact of changes in agricultural management e.g. effect of new legislations. One problem related to modelling commercial farms is lack of sufficiently detailed data for agricultural practice and soil type distribution. Often we have large spatial soil variability between farms, within a farm, and even within a field. In modelling at field scale level often only one soil type is selected to represent the whole field. However, the soil type and especially the hydraulic properties are very important for the level of the simulated N leaching. The aims of this paper are: 1. To present an approach for simulation of N leaching on a regional scale, e.g. watershed, county or country, and 2. To assess the sensitivity of different methods for soil type selection in field and regional scale simulations. The approach is used in a regional study.

Materials and methods

The method for simulating N leaching on a regional scale is presented in Figure 1. The approach consists of four simulation steps. First basic field-scale simulations of N and water balance and crop yields are created using the DAISY-model (Abrahamsen and Hansen, 2000) for different representative combinations of farm types, N-fertilisation schedules, soil types, irrigation, and climate. Six dominating soils were used in the simulations and were described based on a statistical analysis of soil data from the region. Soil hydraulic properties were obtained using pedotransfer functions (Børgesen and Schaap, 2005). The basic DAISY simulation results are stored in the DAISY database (Figure 1). Secondly a crop-rotation and N fertilization model (CropFert-model) is run generating farm crop rotation systems and N fertilization schemes. The input data to the CropFert-model are based on national databases on crops, fertilization schemes, soil types, climate, irrigation, and farm type. The third part uses a statistical 'General N Leaching' model (GNL model). Based on representative DAISY simulation results regarding crop rotation, soil type, irrigation, year, and N fertilization schedule the GNL-model simulate the annual water balance (evapotranspiration,

percolation) and the N balances components (N leaching, denitrification, N harvest and the change in soil N pools) and crop yields on field scale level. The fourth part is an up-scaling procedure used to aggregate the field results to farm, grid and regional.

The model approach was used in the catchment of Ringkøbing Fjord and in the rest of the county of Ringkøbing (Denmark). Based on collected farm and field data (7.000 farm observations, 108.000 field observations covering 362.000 hectares of arable land), farm-specific cropping systems and N fertilization schemes were set up using the CropFert-model.

The field simulation results were summarised at farm scale and aggregated to a 1x1 km, 2x2 km, 5x5 km and 10x10 km square grid level.

The sensitivity of using two methods to include soil data in the GNL simulations was tested. As standard in the GNL model all soil types classified within the field is used and simulations are conducted for every soil type and then summarised to field scale. However, for large areas this method can be slow due to the many single simulations. An alternative method is to use only one dominating soil type on field scale. The effect of using one dominating soil versus using all soil types represented within the field was analysed based on the residuals between the two methods. The results were analysed on the different scales considered. The uncertainty introduced by using only one dominating soil was evaluated by calculating the coefficient of variance (CV) and the 95% confidence interval of the simulations.

Figure 1. Model approach used in regional modelling of N balances, water balances, crop yields and N leaching.

Results and discussion

The mean results (average of 11 years of simulations) by using the detailed soil data are presented in Table 1 along with CV and the 95%-conf. int. On regional scale the deviation in N leaching and N harvest introduced by using only the dominating soil types was less than 1 kg N $ha^{-1}y^{-1}$ and for evaporation and percolation the deviation was less than 1 mm/year.

The simulated nitrogen leaching for the whole region averaged 87 kg N $ha^{-1}y^{-1}$ obtained as average of the 11 years of simulations. The leaching varied especially with soil types, types of crop rotations and N fertilization level. The CV decreased from 0.041 obtained on field scale to 0.036 on farm scale and reduced further to 0.009 at 10x10 km grid. The 95% conf. int. was reduced from ±6.9 kg N $ha^{-1}y^{-1}$ on single predictions at field scale to ±1.7 kg N $ha^{-1}y^{-1}$ at 10x10 km grid scale.

The annual leaching obtained on field scale for the each of the 11 years of simulations varied from 42 kg N $ha^{-1}y^{-1}$ to 140 kg N $ha^{-1}y^{-1}$. The CV obtained on annual field scale results (not shown) varied from 0.003 to 0.115. We found that the soil effect was dependent of the climate of the single years and on average CV was higher (0.056) than the average results obtained from the 11 years of simulations (0.041). The nitrogen harvested with crops for the whole region averaged 115 kg N/ha and varied also with soil types, types of crop rotations and N fertilization level. For N harvest the CV decreased from 0.023 at field scale to 0.022 at farm scale and reduced further to 0.005 at 10x10 km grid scale. The 95% conf. int. was reduced from ±5.1 kg N $ha^{-1}y^{-1}$ on single predictions at field scale to ±1.2 kg N $ha^{-1}y^{-1}$ at 10x10 km grid scale.

The simulated evapotranspiration averaged for the whole region 483 mm y^{-1} and the percolation averaged 513 mm y. Between the years the evapotranspiration varied between 428 mm y^{-1} and 547 mm y^{-1} and the percolation varied between 282 mm y^{-1} and 734 mm y^{-1}. For the evapotranspiration the CV decreased from 0.023 obtained at field scale to 0.020 at farm scale and reduced further to 0.007 at 10x10 km grid scale. The same trends but at a lower level was found for percolation.

Table 1. Mean results of N-leaching and N harvest [kg N $ha^{-1}y^{-1}$], and evapotranspiration and percolation [mm y^{-1}] obtained from 11 years of simulations. Coefficient of variance (CV) and 95% confidence intervals (95%-conf.) of the simulations by using only simple soil classification in field scale simulations.

Mean result	N leaching		N harvest		Percolation		Evapotranspiration	
Scale	87 kg N $ha^{-1}y^{-1}$		115 kg N $ha^{-1}y^{-1}$		513 mm y^{-1}		483 mm y^{-1}	
	CV	95%-conf.	CV	95%-conf.	CV	95%-conf.	CV	95%-conf.
Field	0.041	±6.9	0.023	±5.1	0.021	±21	0.023	±22
Farm	0.036	±6.1	0.022	±4.9	0.018	±18	0.020	±19
1 x 1 km	0.030	±5.1	0.017	±4.0	0.018	±17	0.018	±17
2 x 2 km	0.022	±3.7	0.009	±3.0	0.012	±12	0.013	±13
5 x 5 km	0.013	±2.3	0.008	±1.8	0.007	±7	0.008	±8
10 x 10 km	0.009	±1.7	0.005	±1.2	0.006	±6	0.007	±6

Conclusions

Dependent on the scale of analysis and simulation certainty, different Conclusions can be drawn. This study showed that if analysis is conducted on field, farm, 1x1 km or 2x2 km grid scale we would recommend to use detailed soil data in the field scale simulations. On a higher scale the dominating soils only can be used in the basic field scale simulations without introducing a larger error. We found that N leaching was more sensitive towards the soil type representation than was the N harvest. With respect to percolation and evapotranspiration the sensitivity was at the same level. We found from the annual results that the uncertainty introduced by using only the dominating soil type varied between the years and on average was higher than found for the average results of the 11 years of simulations

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Nitrate leaching monitoring from cropping systems in Lombardy (North Italy)

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Abstract

Current environmental and agricultural policies of the European Union (e.g.: Dir. 91/676) are aimed at the development of 'best management practices' leading to a more sustainable agricultural production. Therefore decision makers need more accurate information about the real contribution of cropping systems to water contamination by nitrate in different pedoclimatic conditions to effectively plan and regulate the fertilizers use in agriculture.
The paper provides a survey of objectives, methodologies and first results of the project ARMOSA developed by the Region Lombardy in order to set up a soil monitoring network to measure the nitrate leaching.

Keywords: leaching, management, monitoring, nitrate, Nitrate Directive, nitrogen, soil

Background and Aims

Agriculture is often pointed out as one of the main causes of surface and groundwater nitrate pollution. Agriculture causes pollution with nitrate because it uses large amounts of organic and inorganic fertilizers, and because inappropriate agronomic practices are often used. In the frame of the European Directive 91/676 and the following Dir. 2000/60, the Lombardy Region developed a programme where 'vulnerable' areas to nitrogen losses from agriculture are identified, and sustainable crop management practices are promoted to preserve and restore the quality of water resources.

Material and methods

With respect to the nitrate leaching risk two main tasks are provided by the programme:

1. setting up a soil monitoring network to measure the nitrate leaching from different cropping systems in different pedoclimatic conditions;
2. identifying alternative and sustainable nitrogen management strategies, at farming level, both aware from the environmental point of view and economically acceptable by farmers.

To achieve the first goal, a project called ARMOSA and carried out by ERSAF with the partnership of the Universities of Milan and Naples and the National Research Council, has been formulated. Ten soil monitoring sites representative of the pedoclimatic conditions and cropping systems of the region have been selected on the Lombardy plain. Soil profile description and both hydrological and micromorphological characterization and analysis of soil horizons have been measured. Each site is equipped a with meteorological station, TDR probes, tensiometers, suction cups and soil-temperature probes at various depths to 150 cm.-
Data collected will allow the calibration and validation of mathematical environmental models simulating the N-cycle for the Lombardy environment and cropping conditions. Once calibrated, these models will be used to compare different scenarios and to extrapolate the results of the monitoring to the entire region.

Results and discussion

Here we discuss two monitoring sites where maize was cultivated. Crop management practices at these sites are shown in Table 1.

Table 1. Management practices in the monitoring sites in the 2003.

Soil	Site 1: Typic Calciustept fine (S.T., USDA, 2003)		Site 2: Ultic Haplustalf coarse loamy (S.T., USDA 2003)	
Crop	maize		maize	
Practices	plowing (35-40 cm)		minimum tillage	
Irrigation	sprinkler 7 irrigations (~ 97mm each one)		surface 5 irrigations (~ 200mm each one)	
Nutrient Management	4 applications: cattle and pig manure and inorganic fertilizer		2 applications: cattle manure, sewage sludge	
N input		Kg ha^{-1}		Kg ha^{-1}
	urea	191,4	sewage sludge	215,6
	manure	325	manure	26,05
	Tot	~ 515	Tot	~ 242

Nitrate leaching was 116 kg $N\text{-}NO_3$ ha^{-1} $year^{-1}$ on the coarse textured soil and 53 kg $N\text{-}NO_3$ ha^{-1} $year^{-1}$ on the fine textured soil.
Leaching occurred primarily in summer (July and August) and after harvesting in the fall (October and November). Leaching in summer is correlated with irrigation, while leaching in winter is correlated with rain.

Figure 1. Nitrate content in different soil layers in the 2003 (site 2: Ultic Haplustalf coarse loamy).

For monitoring site 2, Figure 1 shows nitrate content in various soil layers and Figure 2 shows nitrate leaching. Nitrate content decreases rapidly during the irrigation period (from June to the half of July), suggesting that leaching is occurring, even though plant nitrogen uptake is taking place during this period. High soil nitrate contents are observed in the fall after harvesting. These are possible because mineralization of organic matter continues and plant nitrogen uptake ceases; nitrate content decreases dramatically and suddenly after a large rainfall event.
These first results suggest the importance that should be attached to a more efficient irrigation scheme and the use of 'cover' or 'catch' crops in the fall and in winter. The measurements so far have shown that sites with different

pedoclimatic conditions display different nitrogen dynamics. This necessitates a monitoring network covering a large area as planned in this project.

Figure 2. Nitrate leaching in the 2003 in the monitoring site 2 (Ultic Haplustalf coarse loamy).

Conclusions

Instruments and methodologies show their capabilities to monitor precisely water and nitrate movements in the soils, but it is clear that this detailed information cannot be directly extrapolated to a large geographical area. Thus, it was planned to support the monitoring with modelling tools specifically calibrated and validated. Tested models (ANIMO, LeachN, CropSyst) showed difficulties in simulating the multi-layers experimental data, indicating the needs of more in depth analysis of the hydrological and microbiological processes involved.
The project ARMOSA is planned to be running for two years yet. More information can be found on the web site http://www.ersaf.lombardia.it.

Acknowledgments

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Agricultural practice and nitrogen leaching at the watershed scale: Risk assessment using the Indigo indicator

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Abstract

The objective of this work was to study the farming practices within a watershed in order to evaluate the risks of nitrate pollution they induce. 420 fields corresponding to 10% of the total area of the watershed were surveyed to determine not only farming practices but also soil and climatic conditions. Cropping systems were described taking into account variability in practices depending on the crop grown or on the rotation, and corresponding risks were calculated by the use of the indicator Indigo. Risks were then extrapolated to the watershed scale. Wheat, grain and maize-silage did not differ significantly, but were significantly different from winter barley. The least predictable crop in terms of risk seemed to be winter wheat. Analysis of the indicator's sensitivity was also carried out: the difference between nitrogen applied and the crop's needs seems to be the most important for explaining risk, followed by soil preparation practices, and thirdly by management of the period between two crops.

Keywords: cropping system, indicator, Indigo, leaching, management, nitrogen

Background and objectives

Decision-making relating to the protection of the water quality is very frequently undertaken at a large scale. The key for the decisions-makers is to better understand the relationships between agricultural impacts, the physical environment and water quality. However, modelling the impact of non-point source pollution in catchments is a complex problem. The assessment of the effects of agricultural activities on rivers quality involves a representation of the processes of water transfer and transformation of the elements that it transports.

When constructing a decision-making aid at the watershed scale it is important to precisely analyse this variability in farming practices on a sample of fields. This paper presents a nitrogen loss risk assessment at the cropping system scale; risks are classified taking into account agricultural, meteorological and soil factors.

Materials and Methods

Farmers on a 385 km² agricultural watershed (La Moine – Pays de la Loire, France) were questioned in 2002 in order to characterise their farming practices. Data about farming practice, climate and pedology from a sample of 420 fields, (*i.e.* representative of 10% of the total surface of the watershed) were used as input variables for the agri-environmental indicator Indigo (Bockstaller and Girardin, 2001; Figure 1).

Using this method, nitrogen losses are evaluated chronologically throughout the cropping system from the beginning of winter of the previous year to the beginning of winter of the year of calculation (corresponding approximately to the start of drainage). The indicator value $INO_3 \leq 7$ corresponds to a benchmark indicating a very low risk of leaching which corresponds to recommended levels of N application. For the indicator value $INO_3 > 7$, one point corresponds to leaching risk of 30 kg N/ha.

The risk estimated at the watershed level results from the weighting of the plot's leaching risk by the corresponding area.

Figure 1. Overview of the indicator Indigo (after Bockstaller and Girardin, 2001) assessing nitrogen losses (SMN = soil mineral nitrogen).

Lastly, the factors responsible for the nitrogen pollution risk are arranged hierarchically, according to the share of risk that they explain. In order to achieve this, sensitivity tests are carried out. For each factor, only one input variable of the Indigo indicator varies at a time, the others being maintained unchanged. Then, for each factor tested and each field, the difference between INO_3 scores of the most contrasted situations are calculated. A principal component analysis made it possible to compare and relate the influence of different explanatory factors.

Results and discussion

Leaching risks according to crop

For wheat, maize-silage and maize, no significant difference in nitrogen leaching risks was noted (Figure 2). INO_3 scores for winter barley were significantly different (Kruskall & Wallis, $p < 5\%$) from those of the other crops. The risk difference between winter barley and wheat crops is more particularly surprising since these two crops are managed similarly.

Figure 2. Boxplots of INO_3 scores by crop.

Figure 3. Boxplots of Indigo scores by succession.

Leaching risks according to crop succession

There are important interactions between two successive crops. Thus, the year n+1 crop will absorb a variable fraction of the nitrogen given to the year n crop (in particular at the beginning of the winter) and thus will decrease the quantity of nitrogen lost after the year n harvest. The statistical results indicate that there are significant differences (Kruskall & Wallis, $p < 5\%$) between nitrogen leaching risk scores of the various crop successions (Figure 3).

More precisely, three groups of homogeneous successions were defined.

Group 1: winter cereal - spring crop or winter cereal – winter cereal

Group 2: spring crop - winter cereal or spring crop – spring crop

Group 3: winter cereal - grassland or spring crop - grassland

For each succession, the practices corresponding to the highest risk scores (top 20%) and the lowest risk scores (bottom 20%) were used to set the parameters of the hydro-agrological model SWAT at the watershed scale (Arnold *et al.*, 1998).

Analysis of the Indigo indicator sensitivity
There were significant differences between the factors (Friedman, $p < 5\%$) regulating nitrogen leaching but only a small number are controllable (Figure 4).

Figure 4. Difference between INO_3 scores by factor regulating nitrogen leaching.

The difference between crop needs and nitrogen supply seems to be the most important factor : it is the main factor determining the potential quantity of nitrogen leaching. Secondly, soil preparation practices: when soil has been ploughed, water infiltration is slower. The quantity of nitrate captured by water is therefore more important (ITCF, 1995). Thirdly, management of the period in-between two crops: when the following crop is a winter crop, its sowing date can have a strong positive effect on nitrate transfer dynamics. In the case of a following spring crop, establishment of a crop in-between can lead to significant reduction of nitrate losses. The latter effect is more difficult to test statistically because this situation occurred less frequently on our sample plots. In any case, the longer the soil remains bare during winter, the lower the risk of leaching.
Our results concord with those of Meynard *et al.* (1997) who found that correspondence between plant needs and nitrogen supply, as well as soil exposure are important factors in the regulation of nitrate losses. As regards soil preparation, studies also show that this can have a significant role, though its effect is less obvious. According to ITCF (1995), management of the period between two crops has a greater effect on nitrate leaching than simplification of soil preparation. The effects of both vary according to different factors: climate, soil, crops, crop residue management.

Conclusions

Finally, if one considers the factors responsible for nitrate leaching in terms of cropping system upon which a decision-maker can act, rather than in terms of factors beyond control such as climatic conditions, this study shows that winter crops seem to be the least predictable in terms of nitrogen pollution risk compared to that of maize which seems easier to predict. Adjustment of nitrogen supply to plant needs and management of the period in-between two crops to avoid exposing the soil are also essential factors.

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Use of Agricultural Nitrogen balances to asses N pressure on ground and surface waters for Municipalities in the Basque Country

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Abstract

The N surplus balance provides an indication of potential water pollution and identifies those agricultural areas with very high nutrient loads calculating the balance between N added to an agricultural system and N removed from the system per hectare of agricultural land. The N balance has been calculated per municipality for the Basque Country, establishing an N surplus target level according to the climatic conditions, the denitrification capacity of Basque Country soils and the maximum permissible concentration of 11.3 mg NO_3-N L^{-1} imposed in the drinking water Directive (80/778/EEC) and in the EC Nitrates Directive (91/676/EEC). In the Basque Country the average N surplus was 101 kg N ha^{-1}. The municipalities with higher N surpluses are those with greater cattle stocking numbers, while the regions with mainly market crop farms are characterised by lower surpluses. The municipalities susceptible to reach the 11.3 mg NO3-N L^{-1} in leaching waters are located in the South, because of the lower rainfall surplus and the higher evapotranspiration.

Keywords: nitrogen, nutrient balance, Water Framework Directive

Background and objectives

For the implementation of the Water Framework Directive, the spatially differentiated evaluation of water eutrophication caused by non-point source N input constitutes a central issue. The N balance surplus is a used indicator for identifying nutrient pollution vulnerable areas. The WFD recommends a catchment area of 10 km^2 for river basin management plans, thus it is also appropriate to calculate the N balance with a comparable spatial resolution (Bach and Frede, 2005). The objectives are the calculation of N surplus for agriculture at municipality level and the establishment of a N reference level in order to asses the N pressure on ground and surface waters.

Material and methods

The N balance was calculated as the difference between the total N inputs entering, and the total N outputs leaving the soil over one year (OECD 2001a). The annual total N inputs considered the following factors: chemical fertiliser, biological fixation, atmospheric deposition, N contained in seeds and planting materials and net livestock manure production. This last factor was calculated multiplying the number of animals by the N coefficient excretion, subtracting gaseous N losses during the housing, storage, application and grazing. The annual N outputs were based on harvested production of crops multiplied by the N content obtained from NEIKER unpublished data and published literature. The calculation of N surplus was based on data from the Basque Country Agricultural Census for the municipalities. (EUSTAT, 2004). The N nutrient balance database was linked with a geographic information system for mapping N soil surface balance surpluses for all municipalities (Figure 1).

Figure 1. N balance surplus for municipalities in Basque Country per hectare of agricultural area.

The N surplus reference level has been established according to the climatic conditions, the denitrification capacity of soils and the maximum permissible concentration of 11.3 mg NO_3-N L^{-1} in drinking water (80/778/EEC) and in the EC Nitrates Directive (91/676/EEC). The following assumptions have been made to estimate the N surplus reference level: rainfall surplus (L/ha) x (11.3 mg NO_3-N L^{-1}) x (100 kg N surplus/50 kg leaching).

The rainfall surplus has been obtained calculating the difference between the rainfall and the potential evapotranspiration. The Basque Country is not an homogenous climatic region, three zones can be distinguished in broad outline: the Atlantic zone in the North, the Mediterranean continental zone in the South and a transition zone between the Atlantic and the Mediterranean zone, where the Atlantic characteristics predominates. Due to the low importance of the transition zone, only two climatic areas have been taking into account: the North and the South, with an average of 800 and 300 mm of rainfall surplus respectively.

It was assumed that 50% of the nitrate would be denitrified in soil and groundwater and 50% is lost by leaching (Scholefield *et al,* 1991). To meet the legal requirement of the 11.3 mg NO_3-N L^{-1}, N surplus must not exceed the 70 kg N/ha in the South and 180 kg N/ha in the North (Figure 2).

Figure 2. Basque Country areas where the N surplus exceed the Directive 80/778/ECC guide (11.3 mg NO_3-N $L^{-}1$) and compulsory (5.65 mg NO_3-N $L^{-}1$) levels.

Results and conclusions

The total N surplus of the Basque Country was about 22,000 tons N, corresponding to an average N surplus for all municipalities of 101 kg N/ha of agricultural area, whereas the variation was from 23 to 245 kg N/ha of agricultural area. As expected, the regions with higher N surpluses appeared to be those with greater cattle stocking numbers, while the regions with mainly market crop farms are characterised by lower surpluses. An annual surplus of 70 kg N/ha in the South and 180 kg N/ha in the North, resulted as a reference level in order to asses the risk for ground and surface waters contamination. The reference level is lower in the South due to the lower precipitation and the higher evapotranspiration. In spite of their lower N surpluses, the municipalities in the South are more susceptible to reach the 11.3 mg NO_3-N $L^{-}1$ in leaching waters compared to the North, because of the lower reference level.

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Development of nitrogen fertilisers used in agricultural production

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Abstract

In many European countries there has been a substantial increase in the use of nitrogen fertiliser from 1950 to the 80's. Many countries supported the use of industrial fertilisers and pesticides the first three decades. With rising surpluses of agricultural products and more focus on the environmental effects in the 80's, in many countries agricultural prices were reduced and farmers were recommended to reduce the use of inputs and to use fertilising plans.

For most of the countries studied, lower prices for wheat, as an indicator for agricultural plant product prices, leaded to less application of nitrogen fertiliser because the application was less profitable. The influence prices for nitrogen fertiliser were not as clear as wheat prices and differed for different countries.

Keywords: Europe, fertiliser, nutrient budget, yield

Background and objectives

Since 1945, while many European countries still imported more food than they exported, the use of industrial fertilisers and pesticides was supported in many European countries by the governments. With rising surpluses of agricultural products and more focus on the environmental effects of a intense use of industrial inputs in agriculture, the European Union introduced production reducing means in the 80's. Some examples are the milk quota system, set aside programs for agricultural land, and reduced prices for many agricultural products. Simultaneously, farmers were recommended to reduce the use of inputs, and to use fertilising plans.

The objective of this study is to analyse how the use of nitrogen (N) fertiliser has developed in different countries inside and outside the European Union under different economic conditions the last 40 years, and what impact the changes in N fertiliser use had on the efficiency of plant production.

Material and methods

Records of agricultural area (only area for conventional farming), amounts of N fertilisers used, yields and prices were collected for selected European countries; mainly by means from FAOSTAT (2005) and EUROSTAT (2005). The aim was to test whether the national fertiliser use could be explained statistically by the cost of fertiliser and price of grain. The countries included in this study were Denmark, Finland, Germany, the Netherlands, Norway, Sweden, and the United Kingdom.

Provided that a farmer wants to maximize the expected profit of N fertiliser used, the profit maximum for the use of a factor is determined by the production function for the special plant, the price of N fertiliser, and price of the plant product (neoclassical approach). The production function describes the relationship between levels of nitrogen input and yields. An increasing input of nitrogen leads to increasing yields, but at diminishing rates. At the profit maximum for the use of nitrogen, the marginal product equals the relative price ratio between price of nitrogen and price of product. All other factors are assumed not to influence the optimal N fertiliser use. Hence it is expected that the use of fertiliser depends both on the price of fertiliser itself and the price of product for a given production function. To test this hypothesis, simple linear regression was used.

Results and discussions

A crude comparison of the use of N fertiliser between different countries is of limited value, because the fertilisation level reflects differences in edaphic and climatic conditions, cropping patterns, manure access, relative prices etc. (Vedeld, 1997).

For all years since 1950, the use of nitrogen fertiliser (kg N per hectare) was highest in the Netherlands, more than twice the level for the other countries from the European Union (Figure 1). From 1950 to the early 80's the average use increased from about 70 to 250 kg, and was reduced to about 150 kg from the year 2000. Other countries where the farmers on average used more than 100 kg N fertiliser per hectare and year from the 70's to the late 80's were Denmark, Germany, and Norway. The more N fertiliser farmers in these countries applied in this period, the more they reduced the application from the late 80's. In contrast to the Netherlands, Denmark and Germany, the farmers in Norway reduced the application of N fertiliser just little.

Figure 1. Development of nitrogen fertiliser applied per hectare agricultural area

The amount of land allocated to different types of crops has varied significantly within time in Europe. For different types of crops, the recommended levels of N fertilisation differ. Hence, a change of crop will influence the application of fertilisers.

As a rough approach for the productivity of nitrogen fertiliser for each country and year, the national N fertiliser use per hectare was related to wheat yield. Wheat yields and prices was used as an indicator for grain and because of good data availability. However the share of agricultural area with wheat was just from 1% to 12% in 1961, and increased to the range of 6% to 23% in 2002 in the studied countries, so the results have to be used with care. While the use of N fertiliser increased for all countries in the 60's and for the most part of the 70's, the productivity of nitrogen fertiliser decreased. Farmers in countries with high use of nitrogen fertiliser in the early 80's, as the Netherlands, Denmark, and Germany, reduced the use of nitrogen fertiliser in the 80's and 90's without reducing yields to a great extend.

Both natural conditions, traditional practises, and different prices result in different nutrient balances for different countries. Andrews (2001) estimated soil surface balances (including atmospheric deposition, livestock manure, biological nitrogen fixation, and mineral fertiliser) for 15 countries of the European Union in 1995. For all countries, the N budget was positive. For Poland, Sweden, and the United Kingdom the balance was between 20 and 40 kg N ha^{-1}. For Finland, France, and Denmark it was between 50 and 75 kg N ha^{-1}. For Germany it was about 100 and for the Netherlands about 210 kg N ha^{-1}. Rystad (1990) claims that only 60% of the nitrogen in grains was removed as yield from the fields in Norway. For all agricultural land in Norway Dahl (1990) estimates that some 130 million kg (about 125 kg N ha^{-1}) of nitrogen is left unaccounted for. Some of the nitrogen surplus will be stored in the ground, some runs off to watersheds, and some diffuses as ammonium gas.

A linear regression for the time series for the countries included in this study showed that the increases in annual wheat price increased the application of N fertiliser. This relation was significant at the 5% level, except for France and Finland. Because of this correlation in combination with relatively high prices for plant products in Norway, it seems to be a challenge to fulfil the commitment of the North Sea states to reduce the inputs of nutrients by 50%

(Ministry of the Environment, 1999). The influence of the price for N fertiliser on the N fertiliser use was not as clear as the wheat price. As expected, a higher N fertiliser price was negatively correlated to the nitrogen fertiliser consumption for Denmark, Finland, and Sweden (significance level < 10%). For France and the United Kingdom, a higher N fertiliser price was positively correlated to the N fertiliser use.

Conclusions

The objective of this study was to study how the application of N fertiliser on agricultural area has developed in different European countries. There was a substantial increase in the use of N fertiliser from 1950 to the 80's for all countries examined. Although wheat was grown on just 6% to 23% of the agricultural area in the studied countries in 2002, it was shown that the annual wheat price positively correlated to the use of N fertiliser.

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A method to compare the accuracy of indicators of water pollution by nitrates

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Abstract

Numerous indicators of different levels of complexity were developed to assess the risk of water pollution by nitrates, but the accuracy of these indicators is not well known. The purpose of this paper is to describe and apply a simple statistical method for evaluating the sensitivity and specificity of indicators of risk of water pollution by nitrates. This method can be used to help decision makers to select an efficient indicator of risk of pollution and to determine an optimal decision threshold value. A case study is presented in which seven indicators, based on field characteristics and/or nitrogen balance, are evaluated using 89 soil mineral nitrogen measurements obtained at harvest during nine years in winter wheat (*Triticum aestivum L*) fields located in France. The results showed that the best indicators were those using fertilizer dose and crop yield as input variables. The use of an early measurement of soil nitrogen did not improve the accuracy of the indicators significantly.

Keywords: indicator, sensibility, specificity, water pollution

Background and objectives

Numerous indicators were developed to assess the risk of water pollution by nitrates at the regional scale (Meynard *et al.*, 2002). These indicators are based on direct field measurements, on nitrogen balances, or on models simulating nitrate leaching. All these indicators are subject to uncertainties (Oenema *et al.*, 2003) and it is thus important to evaluate and compare their performances. The purpose of this paper is to describe and apply a simple statistical method for evaluating the sensitivity and specificity of indicators of risk of pollution. A case study is presented in which the performances of seven indicators are evaluated using 89 soil mineral nitrogen measurements obtained at harvest in a watershed located in northern France (Picardie).

Material and methods

Murtaugh (1996) and Makowski *et al.* (2005) used a statistical method based on Receiver Operating Characteristic (ROC) curves to assess the usefulness of ecological indicators. The ROC curve analysis is a classical method for evaluating the classification accuracy of medical diagnostic tests. In this paper, we demonstrate the interest of this method for evaluating indicators of pollution by nitrates. Consider a dataset including n measurements of soil nitrogen content or nitrate concentration in soil solution, and suppose that each measurement (noted Y) was obtained in a field chosen in a region of interest. The dataset is divided into two subgroups depending on whether the measurement Y is above a predefined pollution threshold Y_t or less than this threshold. The output I of each indicator is then determined for each field in both subgroups, and each value is compared to a second threshold (a decision threshold) noted I_t. The results are used to calculate the true positive proportion noted TPP (number of field with I higher than I_t in the subgroup of fields with Y higher than Y_t divided by the total number of fields in this subgroup) and the true negative proportion noted TNP (number of fields with I lower than I_t in the subgroup of fields with Y lower than Y_t divided by the total number of fields in this subgroup). TPP is referred to as sensitivity and TNP is referred to as specificity. The ROC curve of an indicator is a graphical plot of sensitivity against (1-specificity)

obtained for a given value of Y_t the value of TPP and TNP being calculated by allowing the decision threshold I_t to vary over the whole range of values taken by the indicator. This procedure is repeated for all the indicators and for different thresholds Y_t. Each ROC curve is then summarized by its area under the curve (AUC) and statistical tests are performed to compare the AUC obtained for different indicators.

Here, this method was used to evaluate seven indicators including different input variables and noted I_1-I_7 (Table 1). A ROC curve analysis was performed for three different thresholds of soil mineral nitrogen at harvest, Y_t= 30 kg.ha^{-1}, Y_t= 40 kg.ha^{-1}, and Y_t= 50 kg.ha^{-1}. Sensibility and specificity values were estimated from a dataset of n=89 soil mineral nitrogen measurements obtained at harvest between 1991 and 1999 in different winter wheat fields located in a French watershed (basin of Bruyères, Picardie) (Beaudoin *et al.*, in press). The AUC was estimated for each threshold Y_t and for each indicator.

Table 1. Characteristics of the seven indicators of pollution

Indicator	Formula
I_1	I_1=amount of applied nitrogen
I_2	I_2=applied nitrogen + soil mineral nitrogen at winter
I_3	I_3=applied nitrogen – recommended fertiliser dose
I_4	I_4=applied nitrogen / grain yield
I_5	I_5=[applied nitrogen + soil mineral nitrogen at winter] / grain yield
I_6 (Equif)	I_6=soil nitrogen + apparent recovery *applied nitrogen – nitrogen requirement *grain yield
I_7 (Corpen)	I_7=applied nitrogen – nitrogen content in grain *grain yield

Results and discussion

Figure 1 shows examples of ROC curves obtained for two indicators, I_6 and I_7, and for one pollution threshold, Y_t= 50 kg.ha^{-1}. The two curves were not very different and this result reveals that I_6 and I_7 have similar performances. Such curves are useful to determine a decision threshold value (I_t) leading to high values of sensibility and specificity. For example, the ROC curve obtained for the indicator I_7 (Corpen, 1988) showed that it is possible to obtain sensibility=0.66 and specificity=0.7 when the decision threshold was I_t=-30 kg.ha^{-1}. Note that this value is much lower that the current decision threshold value used by the French ministry of Agriculture (I_t=0 kg.ha^{-1}).

Figure 1. ROC curves obtained for indicators I_6 and I_7 and for Y_t = 50 kg.ha^{-1}.

Table 2. Values of AUC (area under the ROC curve) for the seven indicators and the three different thresholds of soil mineral nitrogen at harvest Y_t.

Threshold Y_t	AUC estimated for the indicator						
	I_1	I_2	I_3	I_4	I_5	I_6	I_7
30 kg.ha^{-1}	0.58	0.59	0.58 a	0.62 a	0.65 a	0.64 a	0.64 a
40 kg.ha^{-1}	0.57 a	0.58	0.54	0.62 a	0.66 a	0.64 a	0.63 a
50 kg.ha^{-1}	0.58	0.55 a	0.56 a	0.62 a	0.64 a	0.64 a	0.65 a

Same letter for a given threshold: AUC of the indicators are not significantly different P≤0.05

Values of areas under the ROC curves (AUC) are displayed in Table 2. The AUC estimated for I_1, I_2, and I_3 were close to 0.5 and this is typical of non-informative indicators (*i.e.* indicators that do not perform better than random decisions). The AUC values obtained for the indicators I_4, I_5, and I_6 were in the range 0.62-0.66 and, so, are closer to 1 (AUC value of a perfect indicator). This result shows that the indicators including the variable 'grain yield' performed better. On the other hand, the use of an early measurement of soil mineral nitrogen did not increase the AUC value and, so, did not increase the sensibility and specificity of the indicators.

The AUC values obtained for the seven indicators were all relatively low compared to the values reported in past studies for other agro-ecological indicators. For example, the AUC value estimated by Makowski *et al.* (2005) for an indicator of risk of crop disease occurrence was higher than 0.8.

Conclusions

A methodological framework was described in this paper to evaluate indicators of risk of water pollution. This framework is useful to help decision makers to select an efficient indicator of risk of pollution by nitrates and to determine an optimal decision threshold value. In our case study, seven indicators were evaluated from a dataset including nine years of soil mineral nitrogen measurements obtained in a high number of winter wheat fields. Most of these indicators were based on a simple calculation of nitrogen balance. This case study could be extended in two ways. First, it would be useful to perform similar calculations for other crops like sugar beet, oilseed rape and potatoes. Second, it would be interesting to consider more complex indicators based on static or dynamic models.

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Nitrogen flow analysis in the Spanish agricultural and food production system

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Abstract

Substance flow analysis has been applied in order to assess N flows in the Spanish agri-food production system, identifying and quantifying the principal pathways in the period 1996-2000 with the best available data in the statistics and literature. The main flows are fertilizer and manure input to soil, supported by a high import of fertilizers and raw materials for feeding livestock. The economic subsystem exports nitrogen to water (through wastewater and leaching) and to atmosphere (mainly as ammonia from agriculture).

Keywords: management , nitrogen, nutrient balance, substance flow analysis

Background and objectives

Nitrogen is an essential element in human, animal and plant nutrition, but the presence of large quantities of reactive nitrogen in the biosphere implies environmental problems at global, national and regional levels. The aim of this work is to identify and quantify nitrogen sources, processes and sinks between the economic, societal and environmental compartments involved in the Spanish agricultural and food production system, using substance flow analysis methodology.

Material and methods

The system has been defined as the agricultural and food production system in the Spanish territory, including the domestic consumption, the management of wastes and the environmental subsystems (water and atmosphere). The main compartments of the system and flows between them have been defined and identified. Average flows of nitrogen in the period 1996-2000 have been calculated using the best available statistical data of economic activity (i.e. Ministries of Agriculture and Environment, Fertilizer Industry, Statistics Institute, FAO, European Pollutant Emission Register, EMEP programme, etc.) and average N content of products (i.e. food and feed composition tables, excretion factors, nutrient content of sludge, water and wastewater, etc.). Literature values have been also used for calculations (i.e. N in waste and compost, N lost in leaching and composting processes, etc.). EMEP/ CORINAIR methodology has been used to calculate gaseous emissions from soil, livestock housing and manure storage and waste management (EEA, 2004). The bookkeeping account methodology has been used, registering and calculating the flows that enter or exit every compartment, and the mass balance principle has been applied. Several assumptions have been taken into account across all calculations.

Results and discussion

Eleven compartments have been defined in the system (Figure 1): fertilizer industry, soil, plant production, livestock production, food and feed industry, domestic consumption, waste management, wastewater management, water, atmosphere and imports and exports. Forty flows of nitrogen have been identified and quantified between those compartments, but only those higher than 50 Gg N y^{-1} have been represented in Figure 1. The most important flow is the input of fertilizers in soils. Industrial ammonia fixation (426 Gg N y^{-1}) and importation of ammonia and N

fertilizers (989 Gg N y^{-1}) support fertilizer production, nevertheless an important part of N is also exported from industry compartment (265 Gg N y^{-1}). Fertilizers and manure represent more than 80% of total N input to soil (Table 1). With regard to manure and slurry production, cattle livestock accounts for 30% and poultry for 23% of total N excreted. Other important inputs in soils are legume fixing crops, atmospheric deposition, irrigation water and organic wastes. The main outputs from soil are crop products, forage, straw and grazed pastures and grasslands, but ammonia emissions and nitrate leaching are also important outflows to atmosphere and water, respectively. So there is a great surplus in this compartment and the nitrogen utilization efficiency is about 35%, less than the value reported by Bleken and Bakken (1997) for crops at global level (i.e. 50%). The eco-efficiency (ratio of N emitted and N in useful products) is 0.4 kg N kg^{-1} N. A part of the N included in the term inputs minus outputs may be stored in soil, but another part may be lost to the environment. Leaching is clearly underestimated in this study, where only irrigated crops have been considered and some other sources of N have not been accounted for.

A: Industrial fixation of ammonia. B: Import of ammonia and fertilizers. C: Fertilizers applied to soil. D: Export of fertilizers. E: Crop yields. F: Legume fixing crops. G: Atmospheric deposition. H: Emissions from soil. I: Irrigation water. J: Leaching from soil. K: Forages, straw and grazing. L: Raw plant products. M: Animal feeds. N: Animal products. O: Manure and slurry. P: Emissions from manure management. Q: Import of raw materials, food and feed. R: Human food consumption. S: Export of raw materials, food and feed. T: Industrial wastewater to municipal treatment plants. U: Municipal solid waste. V: Domestic wastewater. W: Emissions from wastewater treatment plants. X: Treated and untreated wastewater.

Figure 1. Representation of compartments and flows of N in the Spanish agri-food system.

Table 1. Nitrogen balance in soil compartment.

	N flow (Gg N y^{-1})	N rate (kg N ha^{-1})	Percentage of total
Total inputs	2,443	95.6	100
Fertilizers	1,172	45.9	48.0
Manure	817	32.0	33.5
Legume fixing crops	118	4.6	4.8
Atmospheric deposition	176	6.9	7.2
MSW compost + sludge + industrial wastes	49	1.9	2
Seeds	25	1.0	1.0
Irrigation water + re-used treated wastewater	87	3.4	3.5
Total outputs	1,165	46	100
Harvested crops	547	21.4	46.9
Forages, straw and grazing	299	11.7	25.7
Ammonia emissions	166	6.5	14.2
N_2O + NO emissions	48	1.9	4.1
Leaching	106	4.2	9.1
Inputs - Outputs	1,278	50.0	52.3% of inputs

Harvested crops are used as raw materials for food and feed production. This industry is also supplied by a big import of raw commodities (mainly soya products and cereals). Even though Spain has a great agricultural sector, it is a net importer of N in food and feed. In this study, about 19 kg N $person^{-1}$ y^{-1} is imported; whereas the amount of N exported in agricultural products is 3 kg N $person^{-1}$ y^{-1}; these values are higher than the average for the European Union (Bleken and Bakken, 1997). The use of feeds in livestock production accounts 724 Gg N y^{-1}. Animals excrete N in urine and faeces and emissions of NH_3 and N_2O occur from livestock buildings and manure storage facilities (105 Gg N y^{-1}). So, the efficiency of livestock compartment is ca. 15%, higher than the global value of 10.5% given by Van der Hoek (1998), and the eco-efficiency is 5.8 kg N kg^{-1} N.

Food industry also produces food for human consumption (291 Gg N y^{-1}). This represents an intake of 80 g protein $person^{-1}$ day^{-1}, more than recommended dietary protein. Moreover, there is a high proportion of animal protein in the diet (55% versus 45% from plant products). Non-edible parts of food, table waste and storage losses are lost as municipal solid wastes (60 Gg N y^{-1}) and a large proportion of the N intake is excreted (12.3 g N $person^{-1}$ day^{-1}) to wastewater. Some nitrogen included in solid waste is recycled to land application, but a big proportion is accumulated in landfills (95 Gg N y^{-1}), and gaseous compounds are lost in composting, incineration and disposal facilities (6 Gg N y^{-1}). Circa 42% of the N in wastewater is eliminated by different treatments and it flows in sludge or as gaseous N compounds. A part of the N in sludge and wastewater is re-used in soils, but a great proportion is discharged to water in treated and untreated wastewater (182 Gg N y^{-1}).
The environmental compartments receive nitrogen from the economic subsystem. Thus, atmosphere has a net import of reactive N (250 Gg N y^{-1}), where agriculture contributes with a 75%, mainly as ammonia from manure and fertilizers. The food subsystem also exports nitrogen to water through wastewater spillage; this N comes from human consumption of over-recommended quantities of proteins from animals feeded with a large proportion of imported commodities. Water compartment also receives nitrogen from leaching of agricultural soils, due to the surplus in the soil N balance.

Conclusions

The main N flows in the Spanish food production system are inputs in soil of fertilizers and manure. Although outputs in plant production are also high, the result is a nitrogen surplus in agricultural soils. The system is a net importer of N in fertilizers and raw materials (soya products and cereals). These goods and inner agricultural production feed

livestock that produce big quantities of manure and gaseous emissions. Economic subsystem exports N to atmosphere (mainly as ammonia from manure and fertilizers) and to water (from wastewater and leaching). This framework shows us the main nitrogen pathways where environmental indicators development and nutrient policy-making should focus in the near future.

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Experiences with integrated catchment scale modelling of nitrate loads to coastal waters under Danish conditions

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Abstract

Catchment and fjord-models were implemented for Odense and Ringkoebing Fjord in order to analyse N-transport and transformation. In the fjords, effects of N and P on the ecological system were analysed, for Ringkobing Fjord in combination with salinity. Scenarios including a number of N-control measures such as catch crops, different restrictions on manure, implementation of forest, decrease in the number of animal units and increase of wetland areas were analysed. The goals of good ecological quality (Odense Fjord) or 2 m transparency (Ringkoebing Fjord) could not be reached with any combination of the tested measures.

Background and objectives

The implementation of WFD calls for approaches and tools capable of dealing with surface and sub-surface water bodies in an integrated context. In connection with preparation for the 3rd Danish water environment plan and WFD, comprehensive modeling studies were carried out for the catchments of Odense and Ringkoebing Fjord (Nielsen *et al.*, 2003, Jacobsen and Gjertz, 2005). Decades of intensive agricultural production have led to increased nitrate loads to groundwater and surface water, resulting in poor ecological quality of the downstram fjords. Various measures taken towards reduction of nutrient load during the past 20 years have been deemed insufficient and further actions are required to obtain satisfactory environmental conditions.

The aim of the studies was to describe the spatially distributed nitrate leaching from the root zone and the transport and transformations through groundwater and wetlands to the stream and to evaluate the effect of different nutrient loads on the ecological conditions in the fjords.

Materials and methods

The models applied are Daisy (Abrahamsen and Hansen, 2000) for the root zone, MIKE SHE (Abbott *et al.*, 1986) for the groundwater zone, MIKE11 for streams and small lakes, and MIKE 3 for the fjord.

The catchments are situated on moraine clay and moorland plain/hill island deposits respectively. The soil types are primarily sandy loam on the moraine and sand on the moorland deposits. The moraine clay catchment is intensively drained. Overall land use information stems from the 'area information system' (National Environmental Research Institute) supplemented with information from the counties. Information concerning farm types, crop distribution and use of fertilizer and manure came from databases administered by the Ministry of Food, Agriculture and Fisheries. Rainfall data stems from 10-km climate grids while one representative temperature and radiation record was used for each area. The geological descriptions were in both cases based on earlier work. For the Odense Fjord catchment, the location of the redoxcline was determined from database information (230 boreholes) as described in Nielsen *et al.* (2003). For Ringkoebing, the redox cline has been mapped through a systematic analysis of more than 2000 boreholes and the discrete point data set was used to interpolate a continuous map of depth to the redox cline for the entire basin.

Measures investigated to reduce the nitrate load were improved fodder efficiency, stricter norms for application of organic manure, catch crops, increase of wetland areas, introduction of forest and removal of animal units.

Results and conclusions

The measures taken with respect to manure application showed that higher fodder efficiency did not change leaching much because the reduction in nutrient content is substituted with artificial fertilizer according to the Danish legislation. However, a combination of that and a higher efficiency demand on manure (the fraction of N in manure that is expected to be used in the crop, and for which substitution by artificial fertilizer is not allowed) did reduce the removal from the root zone significantly in the Odense Fjord catchment. Further establishment of catch crops on all areas, where this is possible, reduced the load by a similar amount. The scenario with a large reduction in animal units showed a very delayed effect – such farm areas generally have high amounts of organic matter in the soils and after a change in land use, leaching remains high for a long period of time. In the longer run, however, a significant reduction must be observable.

In Odense catchment the reduction zone removes about half of the leached nitrate and in Ringkoebing about 70%. This also means that the effect of general measures implemented all over the catchment will be reduced by a similar factor. For the measure 'forest for protection of groundwater resources', the reduction factor in the Odense catchment was about 2/3.

It is possible to identify areas where the effect of groundwater reduction is least and the effect of implemented measures highest through a combination of hydrological modeling and particle tracking. This was done through an analysis of which areas that deliver water to the stream system that has not passed a redoxcline. In such areas, particularly if they are drained, implemented measures are expected to give maximum effect on the stream system. In the Ringkobing catchment, these areas are generally associated with stream valleys, while the pattern for the Odense Fjord catchment is more complicated.

While the Odense Fjord catchment contained relatively small areas of wetland along several stream stretches, considerable wetland areas have been made along the main stream in the Ringkobing Fjord catchment. Here, the methodology for simulating nitrate removal in wetlands has been subject to particular attention. Measurement of denitrification in different types of wetlands carried out by the Danish EPA has formed the basis for the model conceptualisation. Four types have been introduced, which are characterised by:

- Type 1: Infiltration – mean denitrification potential of 2500 kg N/ha/year.
- Type 2: Irrigation - mean denitrification potential of 600 kg N/ha/year.
- Type 3: Occasionally flooded areas - mean denitrification potential of 450 kg N/ha/year. Maximum 60% of nitrate mass flux to the wetland removed.
- Type 4: Shallow lakes with low residence time - mean denitrification potential of 250 kg N/ha/year. Maximum 10% of nitrate mass flux to the wetland removed.

The available recorded data on the 3,300 ha wetlands within the basin did not allow subdivision into the respective wetland classes in all cases and assumptions were necessary to assign appropriate types. Wetlands received runoff in terms of drainage, stream or groundwater discharges depending on the local conditions. Associated nitrate loads were subject to a temperature dependent denitrification in each time step of the simulation model (on hourly basis in the surface water component). The removal is, of course, limited by the actual amount of nitrate entering the wetland. At present, the wetland areas are estimated to remove about 16% of the leached nitrate in Ringkoebing, but only 7-8% in the Odense catchment. In the best wetland scenario, wetlands could remove about a third of the N in the stream system of Ringkoebing catchment. Inclusion of wetlands may be an important measure, but the marginal return of more wetlands generally decline as the amount within the catchment increases. In addition, the effect of measures implemented is not additive. A reduction in the nitrate load arriving in a wetland also leads to less denitrification in the wetland area. As with respect to the redoxcline, it is extremely important to locate actions at the sites, where the largest effects are generated.

The work showed that integrated modelling using separate models is possible and that the elements required for a full description exists. However, a number of problems were highlighted through the studies, particularly at the boundary between different domains. Simulations were of good quality for sandy areas, while in the moraine area the boundary condition between the root zone and the upper groundwater is inadequately described. This influences the

temporal dynamics and the flux to the fjord was about two weeks delayed. As the ecological model for Odense Fjord required a very precise simulation of the temporal distribution of the N-load due to the short residence time for water in the Fjord, a very precise temporal distribution of the load was required. The redox zone in the groundwater cannot be viewed as a continuous layer, particularly in areas with shallow groundwater, and its presence should be associated with a probability. For wetlands, scale problems were obvious – the 500 m grids used in the catchment model only allowed an empiric description of nitrate removal in wetlands.

For each of the fjords it was attempted to calculate the requirements to reach good ecological status (Odense) or a transparency of 2 m during summer of 2 m (Ringkoebing, DHI, 2003) as a function of N and P-load. For Odense, the present (year 2000) load is 2244 T N and 50 T P/year, and for N the reductions required to reach something within 50% of 'natural conditions' with respect to macro-algae density and other biological indicators are in the order of 75-80%. No scenarios were able to reduce the load to the 'natural' level. Most scenarios, combining a range of measures produced a load in the range of 1200-1400 T N/year. To reach the goal for Ringkobing Fjord, a load reduction of 50-75% for N is required, or alternatively a reduction in both N and P loads of 50%. None of the scenarios for the Ringkoebing catchment were able to produce a reduction of more than 33%.

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The position of dairy farming amidst N-sensitive ecosystems

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Abstract

The Water Framework Directive (WFD) requires adjustments in Dutch land use. A modelling study was carried out to find out how a specific N concentration in water can be achieved through a combination of land use and farm management. The study indicates that farms with a high production level per ha, may only have a future if they recycle N efficiently. Even then they should be surrounded by a substantial amount of conserved or rehabilitated natural areas.

Keywords: ammonia, dairy farming, eutrophication, land use, nature, nitrate

Background and objectives

According to the EU WFD (Anonymous, 2000) member states must promote the ecological quality of waters bodies and hence take measures to reduce the concentration of various compounds among which nitrogen (N). A good ecological quality may require a N concentration < 5 ppm N. Agriculture is the major source of N. As targets of the WFD refer to the spatial scale of river basins, the cumulative effect of all farms together as well as surrounding non-agricultural land use must be considered. However, farming in general and intensive dairy farming in particular is the dominant form of land use in many parts of the Netherlands. Adjustments of the dairy industry may thus be needed.

The N load from farms into the environment equates the product of input inefficiency and input level, implying that even efficient systems can be environmentally questionable whenever their intensity i.e. input level is high. Conversely, low input systems are not 'clean' by definition. The N efficiency of a livestock farm can be increased by avoiding luxury protein consumption, thus reducing manure production, and abating N losses associated with manure recycling (Schröder, 2005). Existing variation justifies to treat the (in)efficiency level together with production intensity as variables rather than as fixed points of departure. Production intensity also determines evapotranspiration and thus the precipitation surplus from agricultural land, being the other determinant of the N concentration.

Natural areas can have a diluting effect on the ultimate N concentration. However, their diluting effect becomes disproportionally smaller the larger the relative share of agriculture, as ammonia immissions from agriculture into natural areas wear off their diluting effect. Land use and production intensity further interact as a targeted regional milk quota can be produced on either a large number of extensively managed hectares leaving relatively little room for undisturbed nature but yet fostering farmland nature, or on a small number of intensively managed hectares poor in wild life themselves but amidst saved 'undisturbed' nature. These feedbacks require a regional analysis of the combined effects of land use and management within farm boundaries on milk production, on nature and on water quality. The present study explores how much non-agricultural land is needed as a function of the targeted N concentration in receiving water bodies and the characteristics of farming and nature, through a modelling approach.

Materials and methods

In our approach we adopted only two types of land use: 'nature' and 'dairy farming'. We combined simple models of the water balance and N flows of both types of land use. In-farm N fluxes and conversion efficiencies and consequent N loads into soil and air were based on Schröder *et al.* (2003). Precipitation surplus (PS, mm) in agriculture was described as a function of crop N yield (NY, kg N per ha.yr) according to PS = *MIN*(450; 450 –

1.33 * (NY – 200)) for NY < 275 and PS = 350 for NY > 275). N load to the soil under nature was set equal to the product of volatilized ammonia-N per ha agriculture and the land use fraction agriculture, reflecting our simplification that all ammonia-N volatilized from agriculture was supposedly deposited within regional boundaries. Assumed precipitation surplus in natural areas was fixed at either 200 (pine forest) or 300 mm (reed) reflecting the differences of rainfall interception in leaf canopies (Bouten *et al.*, 1992; Roth & Mellert, 2004). We assumed that in agriculture on average 50% of the soil N surplus denitrified, whereas in natural areas we allowed this value to range from 75% (reed) to 25% (pine forest) (Schröder *et al.*, 2005). Variables and parameter settings are presented in Table 1.

Table 1. Variables and parameter settings in scenario studies.

Land fraction	Variable		Low	High
Water	Targeted N concentration		1 ppm	12 ppm
Farm	Milk production		9,000 l per ha.yr	13,000 l per ha.yr
	Dietary protein		15% in DM	20% in DM
	Manure management quality	Gaseous N losses from housing and stores	30% of the TAN* excreted	20% of the TAN* excreted
		Gaseous N losses from land spreading	50% of the TAN* applied	10% of the TAN* applied
		Mineral soil-N recovered by crops	50-70%	60-80%
			Wet: 'reed like'	Dry: 'pine forest like'
Nature	'Nature' of nature	Precipitation surplus	300	200
		Fraction of N soil surplus which is denitrified	75%	25%

* *TAN = total ammonia-N*

Results and discussion

Acknowledging that 1) N and water fluxes may cross regional boundaries, 2) in real life there are more forms of land use than just dairy farming and nature, and 3) ecological quality is not merely determined by N nor is it just needed in receiving water bodies, our explorations yet reveal that there are numerous pathways to similar destinations in terms of milk production, nature and environmental quality. Obviously, the need for non-agricultural land use increases the stricter the targeted N concentration (Figure 1A). Our analysis shows that non-agricultural land use could be considerably smaller to attain a specific N concentration, if the dietary protein content would be reduced from the common 20% to 15%, or if manure management would be adjusted to ambitious but attainable levels (Bannink *et al.*, 2005; Schröder, 2005). Figure 1B shows that the type of nature has a considerable effect on the need for non-agricultural land use. The diluting effect of 'pine forest'-like nature is smaller than that of 'reed'-like nature. More agricultural land use would be possible with lower levels of milk production per unit area. Note that the regional milk production is slightly larger with many low productive hectares than with fewer high productive hectares. This is due to the disproportional rise in ammonia emission into natural areas when the fraction of agricultural land use increases (Figure 1C).

Figure 1. Allowable fraction of agricultural land use, as affected by the targeted N concentration and (A) manure management and dietary protein content, (B) the type of nature, and (C) milk quota per ha agriculture and dietary protein content (in each graph the non-varied factors were fixed at intermediate levels).

Conclusions

The WFD does not necessarily put an end to intensive dairy farms provided that these farms drastically improve the use efficiency of N, and save or rehabilitate sufficient hectares for nature through permanent set-aside. Arbitrary but legitimate choices concerning ones definition of nature (i.e. farmland nature *vs.* saved wilderness), appear to determine whether regional targets of milk production and water quality should be achieved by a small number of hectares with intensive farms or by a large number of hectares with extensified farms.

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Analysis of Nitrogen Transport in a Catchment Area with Agricultural Land Use

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Abstract

The research was carried out in the Grabia lowland catchment area located in Central Poland. Numerical simulations were run for the catchment area using GIS software. Geological structure as well as hydro–geological conditions were taken into account.

Nitrogen surplus is the determining factor when considering the amount of nitrogen infiltrating groundwater. It is the structure and character of agricultural use in each of the municipalities of the Grabia catchment area that defined this factor. Nitrogen load, which might flow to the groundwater, was 10 $kg \cdot ha^{-1} \cdot year^{-1}$ in the upper, less intensively used part of the Grabia river catchment area, and 40 $kg \cdot ha^{-1} \cdot year^{-1}$ in the lower part, which was intensively used by both agriculture and settlement.

Research demonstrated that the influence of ecotone zones on groundwater quality is dependent on the geological and flownet structure. Adopted variable values for effective infiltration q reflected the difference between precipitation and evapotranspiration, the latter being a factor characteristic of ecotone functions. The ecotone showed greatest effectiveness at evapotranspiration and when polluted water flowed though its root zone. This was the case when the contamination source was situated near the ecotone.

Keywords: ecotone zone, groundwater, nitrogen dynamics, soil properties

Background and objectives

Groundwater is the main road for nitrogen migration from the catchment area into the river. In catchment areas where agricultural use dominates, the factor determining groundwater quality is intensity of agricultural production. Nitrogen load transported to the river with the groundwater may, in certain cases, be diminished by riparian ecotone zones.

The objectives of the study were:-to evaluate the effects of agricultural production in the research area on nitrate nitrogen concentration in groundwater, taking into consideration geological conditions and meteorological factors. Model analyses included vertical (from the surface to groundwater) and horizontal (including groundwater into the river) flow; -to assess the role of ecotones in nitrate nitrogen reduction from groundwater flowing into the river.

Material and methods

The research was carried out in the Grabia lowland catchment area located in Central Poland.

The Grabia River is a tributary of the Widawka River. The former covers a part of the Piotrków Upland and the Lask Upland and has an area of 819.5 km^2.

Numerical simulations for the catchment were run using GIS software. It was assumed that the transport of nitrogen to the rivers is the result of groundwater flow, both vertical and horizontal.

An assessment was made of the supply of groundwater. The volume of water flowing to the groundwater was calculated as the difference between the multi–year (1951–1980) average atmospheric precipitation and evapotranspiration. The ABIMO program was used for calculations. Groundwater deposited nitrogen load calculations took into account the quantity of nitrate nitrogen washed out of the soil root zone into the zone saturated by groundwater, utilizing balance total nitrogen surplus on the soil surface and including a determination of

microbiological denitrification. Nitrogen surplus was calculated for the municipalities of the Grabia catchment area on the basis of simplified nitrogen balance totals. On the inflow side, was mineral and organic fertilization, while in the case of the outflow side, nitrogen from agricultural lands in the form of crop yields (Kleinhanss, 1994). The balance total did not take into account atmospheric nitrogen fixing nor its emission into the atmosphere. Calculations were conducted using data from 1989, when the use of mineral fertilizers in Poland was at its highest level, reaching an average of 90 $kg \cdot ha^{-1} \cdot year^{-1}$. The nitrate load flowing to the rivers with groundwater was also subject to assessment. Calculations of horizontal nitrate nitrogen transport were based on the WEKU model (Kunkel, Wendland, 1997), applying the assumptions of the Darcy and Dupuit filtration theory.
A nitrogen transport simulation was conducted in order to define the role of the ecotone in the process of purifying groundwater supplying the river; it assumed an advective–dispersive contaminant transport in porous media using the FLOTRANS model.
A site in the Grabia catchment area, where a narrow river valley built of sands is isolated from the village by boulder clay forming an upland, was selected for the modeling. The village is at a distance of 50 m from the river and is the source of contaminants finding their way into the groundwater of the valley. Assumed edge conditions were H = 3 m and q = 0 for the three impermeable boundaries. The upper boundary, which is a depression line (groundwater level), is described by the variable q. The assumed effective infiltration variable values q reflect the differences between precipitation and evapotranspiration (characteristic for the operation of the ecotone). The starting point was q = 0.002 $m^3 m^{-2} d^{-1}$ (the ABIMO program calculated), Another variable was the nitrate concentration C_0, which equaled 2 $mgdm^{-3}$ and 15 $mgdm^{-3}$. The area subject to modeling was subdivided into four zones (transect length): q1C1 – 10 m of the ecotone; q2C2 – 10 m section preceding the ecotone, where major rainfall may result in the accumulation and infiltration of water; q3C3 – 20 m consisting of a meadow serving as a pasture; q4C4 – 10 m zone of flow from the uplands, including contaminated water.

Results and discussion

In order to define nitrate nitrogen load reaching the groundwater, the ABIMO program was used to calculate effective infiltration as the most important factor responsible for the movement of nitrates to the groundwater. The basic input data nitrogen surplus, which is a reflection of agricultural production in the given area. The total nitrogen surplus in for the whole catchment area amounted to 69.5 $kg \cdot ha^{-1}$, whereas number for agricultural areas was 91.4 $kg \cdot ha^{-1}$. These values decrease as a result of denitrification in the upper soil layers, reaching 57.3 $kg \cdot ha^{-1}$ in for the whole catchment area and 75.2 $kg \cdot ha^{-1}$ with respect to agricultural areas. This is the load of nitrate nitrogen washed out of the root zone. This load is subjected to further decrease, and thus the supply of this component to the groundwater amounts to an average value of 55.9 $kg \cdot ha^{-1}$ for the whole catchment area and 73.3 $kg \cdot ha^{-1}$ for agricultural areas (Behrendt, Dannowski, 2005; Zdanowicz, 2001) The risk potential of the permeation of nitrates into the groundwater has been expressed using such factors as travelling velocity. It is contained in the range from 1 to 20 $dm \cdot year^{-1}$. In areas where the supply of groundwater exceeds 200 $mm \cdot year^{-1}$, the transport speed is high, from 5 to 20 $dm \cdot year^{-1}$, while transport time short, from 1 to 10 years, where a large load of nitrate nitrogen reaches groundwater, up to 45 $kg \cdot ha^{-1} \cdot year^{-1}$. This is seen in the central part of the Grabia catchment area. The upper area—the eastern section of the catchment area—has a lower risk potential; there, travelling velocity falls below 2 $dm \cdot year^{-1}$, and travelling time increases to 40 years, while the supply of groundwater does not exceed 50 $mm \cdot year^{-1}$, respectively. This being the case, the nitrate nitrogen load is relatively smaller (approximately 10 $kg \cdot ha^{-1} \cdot year^{-1}$). It has been assumed that 50% of the nitrate nitrogen leaving the root zone following denitrification in the upper soil layers is subsequently vertically transported with the infiltrating water to groundwater and is the load reaching the river network.
Simulations using the FLOTRANS program have demonstrated that the possibility of influence on the ecotone zone is, to a great extent, dependent on the current lines—i.e. the direction of water flow in the aeration zone. In options that assumed uniform supply throughout the whole valley area (q = constant), the q1C1 ecotone zone does not 'work' and does not differ from neighbouring sections in terms of evapotranspiration value. This also approximates winter conditions when there is no evapotranspiration and the uptake of water by the plants of the ecotone is weak. In cases where the evapotranspiration significantly exceeds precipitation (assumption q1 = -0.002), what happens in increased uptake of water by the plant life of the ecotone. This has a major impact on a change in the hydrological regimen in the area of ecotone influence (Figure1). The direction of groundwater flow is subject to change, where it

does not flow into the river, but is uptaken by the plant life. When the flow of contaminated water is just prior to the ecotone zone, the greater part of the contaminants is uptaken by the plant life with the water and does not reach the watercourse because the front of the contaminated water passes through the root zone of the ecotone (Figure 1C,D). In such a case the impact of the ecotone may be significant. However, in cases in which the source of contamination is found at a significant distance from the ecotone, the contaminated water flows below the ecotone root zone, permeating into the depth of the soil profile (Figure 1A,B).

Figure 3. The effect of ecotone on purification of ground water feeding the river in relation to the distance from the source of pollution; - current and equipotential lines with marked flow zone of pollutants infiltrating to groundwater: A- near the upland, C- in front of the ecotone; - distribution of nitrate-nitrigen concentrations in groundwaters 1000 days after their introduction: B- near the upland, D- in front of the ecotone.

Conclusions

Of greatest threat from the flow of nitrate nitrogen to groundwater are areas of intense agricultural use (the greatest surplus of nitrogen) subject to high atmospheric precipitation (large volumes, in excess of 200 mm·year^{-1}, of precipitation infiltrating groundwater). The nitrate nitrogen load flowing to the aquifer layers is dependent on type of soil to a lesser extent.

The flow of contaminated groundwater to the river occurs at various depths of the aquifer layer, depending on geological structure and location of the source of contamination. In cases in which the contaminated water zone reaching the river bed is below the root layer, for example, the effect of the ecotone may be insignificant. The impact of the ecotone zone on the quality of groundwater reaching the watercourse is great when the contaminated water flows through the root layer of the ecotone.

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Working group 2

N flows at the farm level: indicators and tools for improved N management

Report of Working Group 2

N flows at the farm level: how to uniform balance terms and the use of indicators for improved N management

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Introduction

Nutrient balances and balance-derived nutrient use efficiencies and nutrient surpluses have become key issues in debates among farmers, scientists and policy makers. Currently, balances are increasingly used to set legal standards and to evaluate and compare (groups of) farms, regions or countries as a whole.
However, without a consensus on definitions and methods, evaluations and comparisons are ambiguous at best and can be misleading. Pitfalls are the lack of uniformity in defining balance terms (deposition, biological N fixation, pool changes) and insufficient awareness of the principal differences between e.g. arable farms and livestock farms, or between self-sufficient farms and farms relying on other actors.
The working group at the 14th Nitrogen Workshop at Maastricht facilitated a lively discussion on the use and misuse of nutrient balance approaches. The theme was introduced by C. Grignani (University di Torino, It), R. Lambert (UCL, B), L. Guichard (INRA, Fr) and K.B Zwart (Alterra, NL). In their presentations they were focusing on individual balance terms (and the associated pitfalls), as well as on indicators (for efficiency, management strategies, environmental losses, etc.), based on balance approaches.

Background

Nutrient balance calculations are primarily intended to provide information on the environmental impact of a farm expressed per unit area. Results are also used for comparison between farms and to evaluate a farmer's 'environmental performance' (Schröder *et al*, 2003). For example, on a dairy farm schematically four major compartments can be distinguished, i.e. animals, manure, soil and crop (Aarts, 2000; Van Keulen *et al.*, 2000; Schröder *et al.,* 2003). Nutrients cycle through these compartments, but there are 'costs' associated with the transfer of matter between compartments (Figure 1).

Figure 1. Schematic representation of N-flows on a dairy farm and derived Farm Balance and Crop Balance.

The different balance terms of a Farm Balance (FB) and a Crop Balance (CB) are indicated in Figure 1. The relations between the two balance calculations are:
input – output = surplus
soil surplus (2) + ammonia volatilization (3) = farm surplus (1),
so,
farm surplus (1) – soil surplus (2) – ammonia volatilization (3) = 0

In this scheme, the process of nitrogen mineralization from soil organic matter is not included. Usually, when calculating balances, the assumption is made that mineralization and the reverse process immobilisation are in equilibrium. In the long term and in sustainable systems that might be a valid assumption, but in the short term and/or at smaller scale large differences may occur between the two processes, e.g. following grassland renovation. This implies that the assumption of equilibrium is more likely for a farm balance than at field/crop scale. In a steady-state situation (no accumulation/depletion) the soil/crop surplus represents losses to the environment (leaching and denitrification). In addition, in the farm surplus, gaseous losses are included.

Comparing farms

Nutrient balances for (groups) of farms may be calculated for various reasons. Balance-derived indicators help to understand and interpret complex systems. By synthesizing data and communicating the current status to users, effective management decisions can be taken aimed at realizing specific objectives. Improvements in management can not only be derived by examining the individual terms (input as well as output) of the balance in more detail, but also from balance-derived indicators. This holds especially if the values of such indicators from similar farms can be compared.
To evaluate efficiency and management of different farms, reference values are needed to determine whether a specific value for a balance is ‘good’. However, how should such a reference value be defined when farms differ in certain aspects? Is it possible to compare the ‘sustainability’ of different farming systems? Both, Lambert and Grignani indicated in their presentations that a reference value can be established for each (type of) farm by taking into account its own

characteristics, which means *amount and type of production* (dairy cows, breeding pig, laying hens, etc.), *land area, stocking rate, cropping pattern* and *manure production.*
Especially when comparing farms of different production type (arable farming, dairy farming, pig fattening or mixed farming) this is of major importance, because N-use efficiency, when calculated according to a standard procedure, may vary strongly. Based on each of the farms' specific characteristics, a reference value for each of the input and output terms is calculated. This yields a reference value for surplus; if a farms' surplus is lower, it may be concluded that its N-use efficiency is high and that more organic N is exported than necessary. If, on the other hand, the surplus is higher than the reference surplus, the scope for improvement becomes apparent.
Grignani calculated Crop balance surplus and Farm balance surplus for four different types of farms (beef cattle, dairy cows, suckler cows, pig breeding). Pig breeding farms showed the largest farm surplus, but on the other hand, this type of farm showed the highest nitrogen use efficiency when calculated as N_{sold}/N_{input}. These results illustrate the relative value of different indicators.
For the same farm types, crop balance surpluses were calculated. In theory (see Figure 1), the difference between Crop balance and Farm balance should represent gaseous losses. This turned out to be the case for three farm types, but not for the pig breeding farms. It was concluded that farm sustainability could be adequately evaluated through the combination of Farm and Crop/Soil balances

Balance-derived indicators and emissions

Often, balances and balance-related indicators are calculated to establish relations with emissions (e.g. air and (ground)water quality). The background is that policies might be formulated on the basis of these indicators.
In the presentation of Guichard, the values of 6 indicators were compared for 4 cropping systems ('productive', 'integrated', 'organic' and 'no tillage'). A distinction was made between indicators based on direct measurement (e.g. soil Nmin at harvest), indicators based on field nutrient balances (e.g. applied N – N–yield of grain) and indicators based on a model-predicted nitrate losses. It was shown, that classification of cropping systems on the basis of the risk of nitrate losses, was strongly dependent on the indicator used. Classification based on different indicators may vary due to weather conditions, boundaries of the system, the hydrological situation.
It was concluded that indicators that combine scale (field, farm, watershed), timing and user (scientists, policymakers, extension services) are promising.

Difficulties associated with individual balance terms

Apart from the fact that proper recording and registration of data may be wrought with difficulties, specific balance terms are often difficult to quantify. Especially biological N-fixation, N-mineralization, N-immobilization and denitrification are highly variable balance terms that are difficult to measure accurately. As already mentioned, N-mineralization is often assumed to be in equilibrium with immobilization (assuming organic matter content and quality of the soil to be more or less constant). In such situations quantification of immobilization and mineralization is not needed.
Usually, farm balances are easier to calculate and more reliable than crop balances. To calculate crop balances, additional information is needed on internal flows on the farm (e.g. forage production, N-content of manure and crops, manure application, crop rotation).
In both balances, often difficulties are encountered in reliably assessing N-contents of products and crops, variations in stocks (crops, animals), legume components in grasslands, atmospheric depositions (N and P), and export of manure from farms.

The use of farm balances at regional or national scale

In a Danish approach (S. Kristensen) an existing farm typology is combined with the corresponding farm gate balances and statistical data on total number of farms in each of the farm types to scale up surpluses to national level. For calibration purposes the aggregated results are compared to national statistics on imports and exports. Following satisfactory agreement, the model system has then been used to explore the possible impact of policy measures through scenario analyses (cf. Jensen *et al.*, 2002).

Concluding remarks

The presentations as well as the discussion have clearly illustrated the complete lack of consensus on nutrient balance calculations, definition of terms and interpretations. This situation is understandable considering the widely varying backgrounds of scientists working on nutrient balances, their varying objectives, and the variation in farming systems to which they are applied. But it makes comparative analyses extremely difficult. There is little chance that standardization in calculation and quantification of nutrient balances will be attained (for most scientists the adagium remains 'anything you can do, I can do better'). Nevertheless, it would be advisable to compile a comprehensible summary of the different approaches and on that basis standardize a number of definitions that unequivocally express the most important notions in this field, that would then be accepted by the major players. However, considering the history of science, even that is probably trying to reach utopia.

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Oral presentation

Farm-N: An internet-based tool for calculating the likely effect of intensification on losses of nitrogen

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Abstract

In order to achieve an easy coherent feed and nutrient balance between herd, field and farm an Internet prototype tool for calculating static balances has been developed. The tool has been developed on a basis of measurements on pilot farms and on average findings on feeding and yields in fields. On basis on reliable farmgate balances the N-surplus is distributed between aerial (ammonia and denitrifications) and nitrate leaching after adjusting for soil-N changes. The distribution is calculated on basis on Danish agreed standards. Examples of different crop rotations illustrated that soil-N changes differs considerable according to the proportion of grass/clover in crop rotation and making influence of the calculated leaching.

Keywords: losses, nitrogen, nutrient balances, soil

Background and objectives

Agricultural losses of nitrate, ammonia and nitrous oxide in Denmark are scrutinised by national and international environmental authorities. This scrutiny is accompanied by increasing restrictions on farm management. This paper describes a decision support tool (Farm-N) that allows environmental regulators and farmers/farm advisors to assess the losses of nitrogen from a farm, using a range of user-defined scenarios. The tool is also aimed at providing a means for communication between these groups.

Material and methods

The Internet was chosen because it offers the future possibility of linking to national databases of area use, stock number, climate and soil, and in order to ease updating. An overview of the tool prototype is given on Internet: www.Farm-N.dk > Internet tool. The user describes first the current and then the proposed farm structures and management, in terms of the number and type of cattle and pig livestock, the animal housing and manure storage facilities, the land available, the crop rotation and the type of manure spreading equipment to be used. For ruminant livestock farms, the production of milk and livestock has to be consistent with the animal feeding practice, the choice and productivity of the crops chosen, the sale of crop products and the amount of additional animal feed imported. To ensure that these conditions hold, the tool contains two sub-models. The first is based on a series of decision rules and constructs agronomically sensible crop rotations (Detlefsen, 2004). The second sub-model predicts N excretion from livestock and N losses from housing and storage. In situations where cattle feed includes homegrown products, there is a closed cycle between crop production, protein content, N excretion by the livestock, N losses in housing, storage and during manure spreading and the N available to the crop. After calculating the amount of manure to be applied to fields, the import of mineral N fertilizer allowed by Danish regulations is calculated. This means that the tool has to be used iteratively on ruminant farms, until a consistent feed and N flow is achieved.

The tool calculates a farm gate N surplus and uses relatively simple models to estimate ammonia emissions from housing, storage and field application, nitrous oxide and dinitrogen emissions (see the Web-documentation for DK coefficients) and changes in the soil N (Petersen, 2005). The remainder of the surplus is assumed to be lost via nitrate leaching.
For illustration of the farm level dynamics of the model a dairy, an arable and a pig farm on irrigated sandy soil has been chosen. The farms are dominating types of farm in Denmark (for details see Kristensen *et al.*, 2005 and Kristensen, 2005a & b).
On the dairy farm two alternatives are calculated:
Alternative 1 High proportion of grass/clover uptake in herd and
Alternative 2 High proportion of silage maize uptake in herd.

In order to ease comparison, the 'average' farms have been simplified to a single 10-field crop rotation.
In Table 1 the idealized farms are dairy (column 1), arable (4) and pig (5).
Conventional dairy farms in Denmark grow c. 30% of the area with grass/clover fertilized with c. 130 kg artificial fertilizer N per ha plus 123 kg N in slurry and 115 kg excreted N per ha, according to Nielsen and Kristensen (2005). Maize and whole crop barley for silage satisfy the roughage demand of the herd. Animal and cash crop yields are from 272 farms in 1999-2002, yields for year 2002 see Kristensen, 2005a & b. Roughage yields are from pilot farms studies on 5-20 farms per year since 1989. Standard feed and protein requirements are based on Poulsen and Kristensen (1997). User should have some knowledge of how to balance the roughage production in relation to herd demand.

Table 1. N balances for simplified dairy, arable and pig farming systems on irrigated sandy soil in Denmark, year 2002. Data sources, Kristensen (2005a & b).

System	DK dairy		Alt.1 high grass		Alt.2 high maize		DK arable		DK pigs	
Stocking rate [LSU[a)] ha^{-1}]	1.94		1.94		1.94		0.5		1.4	
Milk + meat prod. [kg FPCM[b)] + meat ha^{-1}]	9161+500		9161+500		9161+500		+1200		+3400	
Idealized farm area use [% of farm area]										
Permanent grassland	30		70		30					
Maize + whole crop	20 + 20		0 + 14		0 + 70					
Rape for seed + grain	0 + 30		0 + 16		0 + 0		10 + 90		10 + 90	
N import (farm gate) [kg N ha^{-1}]										
Mineral fertilizer	56		80		62		118		54	
Biological N_2 fixation[c)]	30		71		29		0		0	
Atmospheric N deposition and irrigation	18		18		18		18		18	
Feed	104		57		114		86		245	
N export (farm gate) [kg N ha^{-1}]										
Milk	48		48		48		0		0	
Livestock	13		13		13		31		89	
Cash crops	0		0		0		91		91	
Field N surplus[d)] [kg N ha^{-1}] (field efficiency %)	121	(57)	142	(59)	156	(49)	90	(51)	110	(46)
Herd N surplus[e)] [kg N LSU^{-1}] (herd eff. %)	200	(23)	200	(23)	200	(23)	55	(36)	156	(36)
Farm-gate surplus [kg N ha^{-1}] (farm eff. %)	148	(29)	165	(27)	162	(27)	100	(55)	137	(57)
N losses [kg N ha^{-1}]										
Ammonia volatilization	36		34		37		16		38	
Denitrification	14		13		14		6		10	
Changes in soil-N with soil-C/N=15 (C/N = 12)	55	(43)	71	(61)	36	(26)	23	(26)	38	(26)
NO_3 leaching, difference with soil-C/N=15 (12)	43	(54)	47	(57)	75	(86)	55	(63)	51	(63)

a) *LSU = Livestock unit, 1 heave dairy cow = 0.85 LSU, 1 sow = 0.23 LSU.*

b) *Fat protein corrected milk.*

c) *Fixation assumed constant = 103 kg N per ha grass/clover.*

d) *Field N surplus = (mineral N fertilizer + N_2 fixation + atmospheric deposition + seeds + manure + excreta) – (cash crops + fodder crops), yield of straw is not included in yield or efficiency.*

e) *Herd N surplus (amount of manure produced) = (Fodder crops + concentrates + imported livestock) – (milk + exported livestock).*

Results and discussion

On average dairy farms, 60% of the N-uptake is from homegrown feed (75% of feed-uptake). On the arable area, this equates to 178 kg N ha^{-1} yr^{-1} in animal manure and 56 kg N ha^{-1} yr^{-1} in allowed N fertilizer according to Danish legislation. When N fixation in grass/clover is assumed to equal the average from the pilot farms (103 kg N $ha^{-1}yr^{-1}$), the farm gate surplus is around 150 kg N ha^{-1} yr^{-1}, equal to average farms (Kristensen, 2005a). With standard Danish ammonia and denitrification emissions of 50 kg N ha^{-1} yr^{-1} the changes in soil-N and leaching are around 100 kg N ha^{-1} yr^{-1}, with 43 kg N-leaching. At a high level of grass/clover of rotation (column 2), the soil-N is calculated to be 30% higher. With high maize proportion of rotation, the soil-N accumulation is 33% lower than the shown DK dairy farm, due to 44% lower plant residues after maize compared to grass/clover, and the leaching increases to 75 kg N ha^{-1}, assuming the same feed- and N-utilization in herd. Other alternatives are calculated very easily, and can give the farmer an impression of the consequences of alternative management scenarios.

Discussions with target users are considered vital. During the developing process the potential users indicated that they are satisfied with the technical aspects of the tool but disagree amongst themselves concerning the role of soil-N in the future scenarios. Under Danish conditions, both models and monitoring suggest that there is an accumulation of N in

the soil of grassland farms and depletion on many other farm types. Most Danish farms were mixed livestock/arable until 20-40 years ago but have become increasingly specialised since then. Grassland inputs higher amounts of organic matter to the soil than other. The soil organic model used here suggests that this accumulation and depletion could still continue for many decades, slowly decreasing as the soil moves toward a new steady state. The disagreements relate to whether accumulated N should be considered semi-permanent storage or a large source of N that eventually could be lost to the environment.

Conclusions

This tool provides a consistent method for calculating farm gate N balances. Relying only on field N balances may give a skewed picture, as this method neglects the vital balancing between the field production and the livestock requirement. Discussions with target users were considered vital. However, whilst the potential users indicated that they are satisfied with the technical aspects of the tool, they disagreed amongst themselves concerning the role of soil-N in assessing the environmental impact of developments in farm structure and management.

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Poster presentations

Productive efficiency and nitrogen abatement in the intensive livestock sector

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Abstract

Defining an efficient level of N pollution requires the determination of abatement priorities according to the impact that each form of N has in the environment. This study used Cost-Benefit Analysis (CBA) and Multi Criteria Analysis (MCA) frameworks to weight different impacts of N in order to define productively efficient N abatement strategies at a case-study poultry farm.

The identification of optimal abatement strategies is complicated because of the existence of trade-offs between emissions of different N forms. This indicates the importance of prioritising the reduction of specific forms of N emissions. The CBA and MCA approaches defined two different abatement strategies, highlighting the importance of the judgements being made as to the relative significance of different types of environmental impacts and the potential contributions from alternative approaches.

Keywords: Cost-Benefit Analysis, livestock production, Multi-Criteria Analysis

Background and objectives

In order to use resources in a manner that is most beneficial for society, it is important to achieve productive efficiency, where this refers to producing a given level of output at the lowest attainable cost. This is a pressing issue in the intensive livestock sector, where there is clear evidence that production strategies lead to environmental impacts (Angus *et al.*, 2003) and increasing pressures to reduce them. These environmental impacts have costs to society that should be internalised, i.e. taken account of by the producer, in order to promote productive efficiency.

To date legislation has been fragmented, regulating single forms of N in isolation without consideration of the pollution swapping effects this may promote (Smith *et al.*, 2000). There is currently a lack of knowledge as to what strategies would constitute efficient integrated N abatement. This reflects the limited attention that has been given to the question and the fact that it is difficult to compare the relative impact that each form of N has in the environment. Therefore, this study aims to examine the effect of applying different valuations to each form of N emission in a linear optimisation framework in order to determine how these valuations affect the production strategies at a case-study poultry farm in the UK.

Materials and Methods

A case study approach was taken as spatial impacts of N are important in a policy context and intensive livestock farms represent a major source sector. In order to focus the issues, a representative poultry installation in the UK was selected. This installation is described in Angus *et al.* (2005), with the addition that manure from the unit can also be spread on agricultural land surrounding the installation, or sent to a manure incinerating power station.

Linear Programming (LP; Boehlje and Eidman, 1984) and Goal Programming (GP; Bouzaher *et al.*, 1987) models were constructed to represent production at the case study installation. In the LP model, N emissions were priced according to the values given by Pretty *et al.* (2000; given in Table 1). The objective of this model was then to maximise net margin from the production of poultry. The GP model instead set constraints on the amount of N emitted from the installation and attempted to meet these restraints at minimum cost. The GP model used the Likert scale scores (Table 1) to weight N emissions. This scale was derived in a Delphi study of experts' views of the relative importance of different types of emissions (Angus *et al.* 2003). These weights put the emissions and impacts of N on to common scales, allowing the models to define an abatement strategy that maximises the net income of poultry production at the case study installation.

Table1. Summary of the weights used on different forms of N in the Linear and Goal Programming models to represent their environmental impacts

Form of N release	Weight used in the LP model using monetary weights	Type of N impact	Weight used in the GP model using Delphi weights
Emission of NH_3-N kg year^{-1}	141	deposition of NH_3-N on nearby nature reserve kg ha^{-1} year^{-1}	4.22
Emission of N_2O-N kg year^{-1}	4,792	Emission of N_2O-N kg year^{-1}	1.67
Emission of NO_3^--N kg year^{-1}	3,420 (estimate from Borjesson, 2000)	Concentration of NO_3-N mg litre^{-1}	
In groundwater	3.78		
Emission of NO_2-N kg year^{-1}	190 (estimate from Borjesson, 2000)	Reduction in net margin	10

Results and discussion

The LP model incorporating the monetary valuations maintained baseline production levels (the production strategy determined by the model, where no N constraints were placed on production).
The abatement strategy adopted was to produce birds on a Poultry Integrated Management System, where dietary controls reduce NH_3 and N_2O emissions by 9% of the level achieved by deep litter systems. Equal amounts of manure were disposed of by sending it to a poultry litter incinerating power station and agricultural fields. The GP model using Delphi weights, selected strategies where bird production was reduced by 43% of the baseline level. The production strategy was to stop broiler production in units that were less than 0.5 km from a nearby nature reserve, so as to reduce NH_3 deposition on a sensitive habitat, and use a dietary control on protein intake.

A sensitivity analysis of the Delphi GP model results indicated that the production strategy at the installation would be dependent on the abatement priority selected. The modelling approach showed that the potential to reduce NH_3, N_2O and NO_3^- leaching simultaneously is lower, or more expensive than a given reduction of each emission individually. This is because of the limited availability of relatively cost-effective (at current prices and costs) options for reducing the different forms of N simultaneously, and in particular of reducing NH_3 emissions in combination with other forms of N.

Conclusions

The analysis suggests that poultry installations in areas close to habitats sensitive to N deposition would have to reduce NH_3-N emissions as a priority in order to achieve productive efficiency. However, where an installation is sited in a non-sensitive area, cost-effective production would be achieved by giving a greater emphasis to limiting N_2O-N emissions.

Thus, the absence of technology that can reduce all forms of N emission simultaneously at a relatively low cost necessitates a ranking of N abatement priorities. These rankings will tend to be site specific meaning that there will be a range of optimal abatement strategies across different landscapes.

The two systems of valuation adopted here, one undertaken under a Cost-Benefit Analysis framework and the other undertaken under a Multi-Criteria Analysis framework, have highlighted two different abatement strategies. This indicates that care is required when developing valuations of the environment. The Cost-Benefit Analysis framework may be seen as providing more robust estimates of the broad social costs of N, drawing on previous economic analysis, while the GP approach was more interactive and transparent, allowing participants in the Delphi exercise to take account of local factors in framing their responses. Neither may be seen as offering the correct answer and there will be scope for developing more reliable approaches by including elements of both approaches in decision-making.

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Comparing indicators of nitrate leaching in various cropping systems

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Abstract

A wide range of indicators is available to assess the risk of nitrate leaching to groundwater. They are based on field measurements, nutrient balances at plot scale, or are composite indicators or models. But few studies have compared the various indicators. Do they all give similar results? Do they lead to the same classification of cropping systems? The aim of our work is to compare the outputs of six different indicators obtained for a given data set and to discuss their relevance and complementarities. This comparison is performed using a long-term experiment on a deep loamy soil in France, where four experimental cropping systems have been studied since 1997. The systems tested are 'productive' (or 'conventional'), 'integrated', 'organic farming' and 'permanent ground cover and zero tillage'. The results showed that cropping systems differ in the risk of nitrate pollution, but according to the indicator used, ranking of cropping systems is different. Further work is needed on the evaluation of existing indicators, and on the development of methodologies for comparing the accuracy of different indicators.

Keywords: cropping system, indicator, nitrate, risk

Background and objectives

It is now obvious that agriculture contributes to nitrate enrichment of ground and surface waters and there is a need of evaluation of this contribution. A wide range of tools is available to assess the risk of nitrate emission to groundwater (Ten Berge, 2002). They are frequently implemented in France (but also in Europe) by different stakeholders (extension services, farmers, etc.). They are either based on field measurements (mineral soil N, N fertilisation applied), nutrient balances at plot scale, are composite indicators or models. Widely used, they differ in their construction and application. Nevertheless, few studies were conducted to compare them: Do they all give similar results? Do they lead to the same classification of cropping situations? The aim of our work is to compare their outputs from a given data set, and to discuss their relevance and complementarities. The work is undertaken at the cropping system scale.

Material and methods

A long term experiment has been carried out since 1997 (Bertrand *et al.*, 2005), in the Paris area to study four cropping systems in a deep loamy soil. The four systems tested are (a) 'productive' (or 'conventional'), (b) 'integrated', (c) 'organic farming' and (d) 'permanent ground cover and zero tillage' (Table 1). A set of decision rules was defined for each system for various agronomical and environmental objectives and constraints. 'Productive' and 'integrated' systems were both implemented in a wheat/oilseed rape/wheat/spring-sown pea rotation and were characterised by different levels of nitrogen and pesticides use. The 'organic farming' system was defined according to the French organic farming frame (no chemical fertilisers and pesticides), and was implemented in a rotation including wheat one year over two. The system 'permanent ground cover' was implemented in a wheat/maize/wheat/ spring-sown pea rotation and was characterized by no tillage, direct sowing and the presence of a permanent cover crop (in association or in between two crops). Yields, N management practices (rate and date), plant N-status, and mineral N in soil at harvest and after winter were measured yearly until 2003 in all these cropping systems.

Six indicators commonly implemented by extension services were used to evaluate the risk of nitrogen pollution associated to the four cropping systems. Two indicators are based on direct measurements (Total amount of N applied and Mineral soil N at harvest). Two others, Corpen and Equif, are nutrient field balances (Corpen, 1988; Aimon-Marié *et al.*, 2001) with differences in the way these balances are calculated. The Corpen balance is based upon differences between imports and exports at the field level. Soil is considered as a black-box. Equif is expressed as a balance based on crop N requirements and N supply from soil and fertilisation. At last two indicators, Indigo and Deac (Bockstaller *et al.,* 2002; Cariolle, 2002) are based on a more detailed approach on mechanism which allows focusing on the important fluxes or pools. Finally, Indigo and Deac allow an estimation of nitrate losses and nitrate concentration in leaching water according to the sequence of techniques applied.

Table 1. Main characteristics of the four cropping systems.

	Productive	Integrated	No tillage	Organic
Soil tillage regime	Ploughing 3 years/4	Ploughing 1 year/2	No ploughing Direct sowing	Ploughing each year
Yield objective	+++	++	++(+)	+
Fertilisers use	+++	++	++	0
Pesticides use	+++	++	++(+)	0

Results and discussion

The indicator values differ across cropping systems, showing contrasted risks of water pollution by nitrates (Figure 1). But the classification of the 4 cropping systems depends strongly on the indicator used for the evaluation.

According to the indicator 'Total amount of N applied', the productive system unsurprisingly leads to the highest level of risk, whereas the organic farming system presents no risk. The indicator 'Mineral soil N at harvest' does not discriminate between the cropping systems. Despite slightly lower values in the organic system compared to the others, the levels remain always low, close to the minimum value commonly observed in the Paris area.

The use of indicators taking into account the yield level for the calculation of nutrient balances leads to another ranking: with the 'Equif balance', the results for cropping systems *a, b* and *c* are equivalent and express lower surplus than cropping system *d*. With the 'Corpen balance', the cropping system *a* expresses a higher surplus, equivalent to the cropping system *d*. With this indicator, the organic cropping system is the only one with no risk, as far as there is no N supplied. Finally, the use of indicators based on an estimation of fluxes in soil and water (Indigo, Deac) leads to a third ranking: the indicators Indigo and Deac are equivalent and rank the organic cropping system as intermediate between the no tillage cropping system (which expresses the lowest risk) and the productive and integrated systems. This different ranking is explained by the fact that Indigo and Deac are not only based on nitrogen balances but also take into account nitrogen absorbed by the crop during the in-between two crops period.

This study has shown that the scales of comparison (rotation, succession, year) were also important (data not shown) on the result of ranking according to the indicators adopted.

Figure 1. Classification of cropping systems according to the risk of nitrate losses, depending on the indicator used. 1 = low risk, 2 = intermediate risk, 3 = high risk.

Conclusions

The indicators do not integrate the same mechanisms in their calculation, and consequently quantitative values can't be compared. But their ranking should be similar. Results showed great differences which point out the issue of indicator's choice. In conclusion, further work is needed on the evaluation of existing indicators (Bockstaller *et al.*, 2003; Schröder *et al.,* 2003), and on the development of methodologies for comparing the accuracy of different indicators (Makowski *et al.,* 2005). Additional research in this field could help users to take care of what they can conclude or not, depending on the indicator chosen. Our results also emphasised the importance of the threshold values defined for each indicator.

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Evaluation of two simulation models as an environmental management tool for diffuse nitrate pollution

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Abstract

In order to reduce the diffuse nitrate pollution risks to groundwater, agricultural development agencies are encouraging collective advisory activities with farmers. Could mathematical simulation models of water and nitrogen fluxes assist these people to prioritise changes in nitrogen management practices and to estimate their impact on the nitrate content of groundwater recharge waters? Here we present the results obtained during the evaluation of the ability of two models (STICS and AGRIFLUX) to simulate various agricultural scenarios of changes in land use and/or farming practices. These simulations confirm, *a posteriori*, the environmental value of an advisory operation and demonstrate the value of the models as an environmental management tool. Despite several differences, the data obtained by simulation with the two models are in quite good agreement.

Keywords: environmental management, fertilizer, modelling, nitrate, nitrogen, water pollution

Background and objectives

In a large number of French regions the agricultural development agencies are putting into place agro-environmental operations to reduce the risks of nitrate leaching into groundwaters. Not having suitable tools enabling them to prioritise changes in farming practices to implement for the local conditions, they base their technical advice on knowledge often acquired on the regional scale. They cannot estimate the impact of these changes on the nitrate content of groundwater recharge water, i.e. the water which percolates down through the soil profile, below the rooting zone. We believe that biophysical simulation models of water and nitrogen balances could be useful tools to help in the environmental management of collective advisory work.

With the help of observed data recorded at various study sites (sites with porous cups, lysimeter boxes and spring catchment area), we have evaluated the properties of two mathematical models based on different formalisms, the agronomic model STICS 4.0 (Brisson *et al*, 1998) and the environmental model AGRIFLUX 2.0 (Banton *et al.*, 1996). Here we present the results of the test on the scale of the catchment area. We have tested the ability of STICS and AGRIFLUX to simulate various agricultural scenarios relating to changes in land use and/or farming practices.

Material and methods

The plateaux of Haut-Saintois, situated 50 km to the south of Nancy, are calcareous aquifer formations whose groundwaters are captured for the supply of drinking water to the surrounding villages. Our study sites include, on one of these plateaux, two plots equipped with porous cups and a 30 ha catchment area of a small dammed spring. The land use in this basin is made up of 10% woodland, 39% ploughed land and 51% permanent grass.

An agro-environmental operation 'FERTI-MIEUX' was started in 1992 on the Haut-Saintois with the objective of restoring the quality of the water, whose nitrate content was above the drinking water standard of 50 mg L^{-1}. The organisors of the operation have thus helped nearly 40 livestock breeders to alter the organic manuring practices which were the cause. These changes have resulted in the introduction of collective composting of manures, the reduction of the tonnages spread and the frequency of application, in particular to maize, and lastly the introduction of nitrate-trapping catch crops (NTCC).

The adoption by the breeders of these new practices, called 'FERTI-MIEUX Practices' i.e. best nitrogen management practices, has resulted in a general lowering of the nitrate contents by 25% in the groundwaters of Haut-Saintois. Using simulation we compare the impact of these actual FM practices (FM in Table 1) and four virtual scenarios for effects of changes in farming practices on the nitrate content of the groundwater recharge water. The simulation of FM practices was done from our knowledge of crop successions and cultural practices used by the farmers cultivating this small basin. Scenario 1 corresponds to the absence of implementation of FM practices (reversion to practices observed before 1992). Scenario 2 corresponds to a total ban on organic matter application and scenario 3 to the implementation of FM practices with planting throughout with NTCC. Finally scenario 4 is to replant the basin with grassland.

Results and discussion

Without the introduction of FERTI-MIEUX practices (Sc. 1), the nitrate content of the groundwater recharge water remained at a high level (+ 83% or + 111% on average for the total period according to the model), incompatible with the production of quality water (Table 1).

Table 1. Annual mean nitrate contents in the water below the rooting zone, simulated with the STICS and AGRIFLUX models for the years 1992 - 2003.

	STICS [NO_3^-] mg L^{-1}					AGRIFLUX [NO_3^-] mg L^{-1}				
	FM	Sc. 1	Sc. 2	Sc. 3	Sc. 4	FM	Sc. 1	Sc. 2	Sc. 3	Sc. 4
1992	118	139	111	112	57	97	112	74	96	60
1993	72	89	66	67	37	147	155	72	109	72
1994	132	142	130	112	56	137	158	79	126	71
1995	61	89	60	53	29	105	157	75	107	47
1996	90	145	84	79	43	60	155	48	54	19
1997	53	95	53	51	37	35	125	26	33	14
1998	53	131	45	52	25	29	130	25	31	13
1999	51	119	47	50	24	48	107	16	28	11
2000	65	129	58	61	30	70	116	15	33	9
2001	50	127	44	48	22	68	123	30	47	11
2002	50	121	42	46	27	54	115	15	46	8
2003	79	145	68	75	22	87	168	25	76	10
Δ * (FM – Sc. i)		+ 83%	- 8%	- 7%	- 52%		+ 111%	- 46%	- 16%	- 66%

* *Mean difference between the FM scenario and the various scenarios over the whole of the period (%).*

The results obtained for scenario 2 show notable differences which could be due to a different way of taking account of the after-effect of applications of organic matter by the two models. The environmental impact simulated with STICS is distinctly smaller than that obtained with AGRIFLUX. Note that the FERTI-MIEUX practices already involve very modest applications of organic matter with a recommendation of 15 T ha^{-1} 3 years^{-1} of composted manure. The difference between FM and Sc2 is thus quite small in terms of changes in practice. Scenario 3 would have an environmental impact only slightly greater than that of FM practices as it would induce a mean fall in nitrate content of 7% or 16%,

depending on the model used. The NTCC are present in the FM practices but only between winter and spring crops. In the case of scenario 3 we have added to this NTCC between two spring crops (ryegrass between successive maize crops). It is therefore possible that this small environmental impact is due to bad parameterisation of this crop succession, which could not be validated on the scale of the plot.

Scenario 4 (reversion to pasture) would allow nitrate contents to be considerably reduced compared with the FERTI-MIEUX practices (-52% and -66% depending on the model), but would necessitate a profound change in the farming systems of the Haut-Saintois which would be incompatible with an advisory operation which depends on the cooperation of the farmers.

Conclusions

This first approach to the STICS and AGRIFLUX models for simulations on the scale of a small catchment offers interesting prospects. This work has already given us a glimpse of the value which the use of a biophysical model might offer as an aid to guiding agro-environmental operations. This is particularly true for the sites where the absence of experimental equipment prevents any precise measurement of the environmental impact of projects to change practices. Despite certain differences, the simulation results given by the two models are fairly similar.
Also, the simulations done from practices recorded on this site and the virtual scenarios confirm *a posteriori* the environmental value of the advisory activity of Haut-Saintois.

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PLANET - a nutrient management decision support system

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Abstract

The PLANET (Planning Land Applications of Nutrients for Efficiency and the Environment) nutrient management decision support system was developed as a computerised version of Defra's industry standard reference publication, 'Fertiliser Recommendations for Agricultural and Horticultural Crops (RB209)'. RB209 provides nitrogen recommendations for a crop and also covers other major nutrients and lime. PLANET provides a tool for farmers and their advisers to adopt best management practice in the use of organic manures and fertiliser. Fertiliser recommendations for a field are calculated based on previous field cropping, fertiliser and organic manure applications. Records of actual applications may be kept and a number of reports can be generated. To encourage maximum uptake by the farming community, the logic to generate a fertiliser recommendation based on input data was developed into an ActiveX DLL and made available to commercial agricultural software developers for integration within their crop recording systems. The PLANET standalone version of the software, incorporating this DLL, was developed for farmers who do not already possess commercial farm recording software.

Keywords: decision support system, fertiliser, management

Background and objectives

In response to the EC Nitrates (91/676/EEC) and Water Framework (2000/60/EC) Directives, UK Government policy needs to help the agricultural community to reduce diffuse nutrient pollution by ensuring that production methods are based on environmentally sustainable approaches. To achieve this, Government needs to take proactive measures so that advice delivered to farmers is adopted effectively and helps to maximise profits by avoiding waste. Effective nutrient management planning contributes to meeting these objectives. A nutrient management decision support system (DSS) was proposed to encourage the adoption of best management practice in the use of fertilisers and organic manures.

Material and methods

The PLANET (Planning Land Applications of Nutrients for Efficiency and Environment) software concept was developed as a computerised version of the industry standard 'Fertiliser Recommendations (RB209)' publication (Anon., 2000). It was designed to provide farmers and advisers with a quick and easy way of obtaining RB209 nutrient recommendations for arable, horticultural and grassland crops for each field, each year, taking account of the crop nutrient requirements as well as the nutrients supplied from organic manures, soil and fertilisers. It needed to allow the user to develop a nutrient and manure management plan for a group of fields covering the use of nitrogen (N), phosphate (P_2O_5), potash (K_2O), magnesium (as MgO), sulphur (as SO_3), sodium (as Na_2O) and lime. This plan needed to allow modification during the season and actual applications to be recorded. The previous year's field records together with other information would then be used to generate the RB209 recommendations for the following year.

It was recognized that the agricultural industry generally is skeptical about the value of decision support tools though evidence suggests that where the tools address a particular need they can deliver benefits and adoption will occur. It was decided to involve the agricultural software industry early in the PLANET project planning, as they were in a position to generate rapid uptake through their existing user base, as well as having valuable experience of deploying computer systems in agriculture.

All software development on the PLANET project was undertaken using Microsoft Visual Basic 6.

Results and discussion

The PLANET nutrient management planning tool was developed as two components. The first component was comprised of a Dynamic link library (DLL) which interpreted the information present in RB209. The DLL provided the logic and functionality to provide recommendations for the major nutrients (nitrogen, phosphorus, potassium, magnesium, sulphur, sodium) and lime at the field level. The nutrient recommendations are accompanied by a series of structured text based comments which provide qualifications and advice in addition to the numerical recommendations. The DLL was tested by expert agronomists who validated the output data before its release to software developers. This DLL was then made available under a no-cost licence to commercial agricultural software developers for integration within their crop recording systems.

The second component was the development of a Windows® based Decision Support System (DSS) as stand-alone software for those farmers and advisers who do not already have a field recording system. The stand-alone software also incorporated the PLANET DLL. The broad objectives of the system were that it should provide a practical tool for farmers and their advisers in England and Wales to optimise nutrient management, provide a means of keeping the necessary records required by the NVZ Action Programme and to encourage speedy and effective adoption of improved nutrient management practices.

The PLANET standalone DSS was divided into four areas:

- Farm Details which allows entry of some simple farm and field information during the initial set-up of a farm.
- A records module which stores previous cropping and other relevant nutrient management information for individual fields on the farm for an infinite number of years. These records are used to generate RB209 recommendations for the current or next crop, including allowance for the rolling balance of phosphate and potash.
- A recommendations module which allows entry of cropping, soil analysis and organic manure application details for the current or next crop in individual fields. Using this information, and information from the records module, RB209 recommendations for each field can be generated for selected nutrients or lime. The 27 year average annual rainfall for the farm is used when calculating nitrogen recommendations. In arable cropping situations, if excess winter rainfall is known, this is used in preference to the average annual rainfall when calculating the Soil Nitrogen Supply Index and nitrogen recommendations.
- A reports section where field recommendations or records may be printed or exported as files in PDF format.

Throughout the software, information must be entered accurately in order for each module to function properly and to generate correct recommendations. The user is prompted if information is entered incorrectly or if essential information is missing. Table 1 shows default values required for an RB209 recommendation.

Table 1. Default values used for an RB209 recommendation.

Parameter	Default value if no user input
Soil pH	6.5
Soil P Index	Index 2 (3 if the previous crop is a vegetable)
Soil K Index	Index 2- (2+ if the previous crop is a vegetable)
Soil Mg Index	Index 2
K releasing clay	No (Yes if soil type is a Deep Clay soil)
S deficient	Yes
SNS status (grassland)	Moderate
Organic manures – rate of application	Depending on the manure type
Organic manures – incorporation 12-24 hours	(Arable & Horticulture Surface applied (Grassland)
Organic manures – nutrient content	As in RB209, section 3 (Anon, 2000)

Validation of the output from the PLANET standalone DSS was provided by expert agronomists before the software was released to the farming community.

Conclusions

Correct nutrient planning for each field, each year, can be a lengthy and complicated process. Using the PLANET stand-alone, or commercial software incorporating the PLANET DLL, will make this easier for farmers or advisers to achieve knowing that PLANET recommendations reflect RB209 recommendations. The PLANET stand-alone software was released in January 2005 and, in the first 3 months over 3,000 copies were distributed to the agricultural industry. A commercial software project incorporating the PLANET DLL was released in the summer of 2005 and other commercial software houses are currently incorporating the PLANET DLL into their crop recording systems.

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A nitrogen turn-over model for sub-Saharan climate and soil conditions under smallholder farm management

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Abstract

The AfricaNUANCES (**N**utrient **U**se in **A**nimal and **C**ropping Systems: **E**fficiencies and **S**cales) project aims to increase our understanding of the spatial and temporal dynamics of rural livelihoods in sub-Saharan Africa (SSA). To analyse nutrient tradeoffs a descriptive farm model will be developed. Corresponding response functions will be generated using process based models. One module of the farm model will be used to simulate soil nutrient dynamics. In this study the process based nitrogen turnover model SUNDIAL was applied to an African data set and exemplary response functions were calculated using a Monte Carlo approach. First results showed satisfying agreement with soil N measurements which supports the assumption that the model is applicable for African conditions. The exemplary response function shows a RSME with the SUNDIAL results of 0.36 and 0.38 which suggests the general applicability of this approach.

Keywords: management, modelling, nitrogen, nitrogen dynamics, Sub-Saharan-Africa, use efficiency

Background and objectives

AfricaNUANCES is a project that aims to increase our understanding of the spatial and temporal dynamics of rural livelihoods and their relationship to food security, sustainability and resilience of natural resources in sub-Saharan Africa (SSA). One of the technological objectives is to develop an integrated dynamic modelling tool to analyse African mixed crop/livestock farming systems (Giller *et al.*, 2005). To keep the farm model simple in terms of low data demand and adaptable to many different situations a descriptive modelling approach is used. Detailed process based models are used to derive descriptive functions for the summary crop growth model. The latter includes several soil sub-modules dealing with the soil carbon, nitrogen (N), phosphorus (P) and potassium dynamics, water balance and soil erosion (Tittonell *et al.*, 2005). To derive descriptive functions for these sub-modules, an existing carbon and N turn-over model (SUNDIAL) is being used, adapted and evaluated against crop and soil farm data sets under SSA conditions. SUNDIAL was developed, parameterized and evaluated in western Europe under temperate climate conditions (Bradbury *et al.*, 1993; Smith *et al.*, 1996). However, African arable systems are more constrained in different ways than those in Europe, and different processes affect the dynamics of N:

- P is a major limiting factor due to insufficient fertilizer availability, low native soil P and high P fixation rates of tropical soils (e.g. Kwabiah, 2003), so N x P interactions need to be considered.
- Uni- or bimodal weather patterns in SSA with distinct dry and wet seasons and higher temperatures result in increased rates of decomposition compared to European soils (Tiessen *et al.*, 1998). At the beginning of the rains and after a long dry season a N flush can be observed in the soil. Both processes lead to substantial N losses via leaching and erosion.

In the first stage of this study, the unmodified SUNDIAL model was used to simulate a trial in Zimbabwe and the performance of the model was evaluated.

Materials and methods

SUNDIAL is a process-based multi-compartment model that simulates N and carbon flows between the crop and soil at point scale. The structure of the model is fully described in Bradbury *et al.* (1993), Wu *et al.* (1998) and Gabriele *et al.* (2002). A data set from an experimental site in Zimbabwe (Chikowo, 2004) was simulated using SUNDIAL with no specific modification for African conditions, and the initial performance of the model was evaluated. The data set describes maize growth and N dynamics on a Lixisol with no applications of N fertilizer during two consecutive years (October 2000 to April 2002). Maize was cultivated during the rainy season. During the dry season the field was left fallow. Weather data were derived from a global monthly climatology data set (Mitchell *et al.*, 2004). Model simulations of N content in the soil profile were tested against measurements at several times during the growing season. Monte Carlo simulations with varying temperature (+/-3 °C), precipitation (+/-10%) and residue amounts of previous yield (1-5 t ha^{-1}) were carried out to derive a response curve for seasonal soil N mineralization. The data of the experiment in Zimbabwe were used as base values.

Results and discussion

Comparison of measured and simulated nitrate-N in the soil profile (0 – 1.2 m) during the growing season (figure below), suggest satisfactory model performance. The measured values indicate a pulse of nitrate-N moving down the profile: the simulated values show a similar process although this occurs earlier in the season. Simulated values were on average higher than measured. Similar results were obtained for the second season.

The measured nitrate- and ammonium-N totalled over the whole profile at each sampling date shows an average difference of 20 kg nitrate-N ha^{-1} and 50 ammonium-N kg ha^{-1}, respectively. Expressed on a mass basis the average difference between measured and simulated values are between 0.28 – 2.79 mg N kg $soil^{-1}$. This compares well to the least significant difference (LSD) ($P<0.05$) of the measured values, which are between 0.5 and 1.0 mg N kg $soil^{-1}$ (Chikowo, 2003). The difference between simulated and measured values can be attributed to the use of default soil parameters, which were derived for soils in the UK and have a relatively high soil organic matter content; UK crop parameters, which differ in variety to the African crops; and weather data interpolated from monthly values.

Since these first, exploratory results showed good agreement with the measured values, a response function for seasonal mineralized N was generated based on a 500 run Monte Carlo simulation. A 3 dimensional function was fitted and the coincidence was tested using the root mean square error (RSME = 0.36 in the season 1; RMSE = 0.38 in the season 2). The resulting response function for net mineralized N was:

$$\text{Net mineralized N} = 512 \times \text{temperature} + 3.22 \times \text{precipitation} + 1.28 \times \text{amount of}$$

where the units of the above terms are net N mineralized (kg N ha^{-1}), temperature (°C), precipitation (mm), and amount of residue (t N ha^{-1}).

Conclusions

The first evaluations of the SUNDIAL model in SSA show promising results. SUNDIAL simulates N turnover processes adequately, but needs to be adjusted to the specific soil and crop conditions in SSA. The model may be of some use in the evaluation of soil management alternatives for resource-poor farmers in SSA, the reduction of N losses and the improvement of N use efficiency. Further data sets will be analysed in the near future to verify these Conclusions in a wide range of conditions in SSA.

Nitrate-N [mg N kg^{-1} soil] in the first growing season (Oct. 2000 – April 2001).

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Nutrients Waterproof: Post harvest measures and treatment of drainage water to meet water quality targets

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Abstract

The EU Nitrate and Water Framework directives impose stringent demands on groundwater and surface water quality. In many cases, arable and horticultural farms in the Netherlands are unable to meet water quality targets set by these directives. In the project Nutrients Waterproof, arable and horticultural farming systems are developed in order to comply with the water quality targets set for nitrate leaching. In these systems, efficient fertilization strategies, post harvest measures, such as catch crops and removal of crop residues, are implemented, as well as treatment of leached nutrients in constructed wetlands. First calculations show that the objectives are still not met after taking all feasible fertilization and post harvest measures. To reach the objectives, removal of nutrients in drainage water using constructed wetlands may be an effective but expensive option. Combination with other functions of wetlands may enhance their feasibility. From this year onwards, field measurements have to demonstrate whether expectations are fulfilled and whether changes are needed for further reduction of nitrate leaching.

Keywords: catch crop, crop residues, drain water, fertiliser management, water pollution

Background and objectives

The EU Nitrate and Water Framework directives impose stringent demands on groundwater and surface water quality. In many cases, arable and horticultural farms in the Netherlands are unable to meet water quality targets set by these directives. Only about 30% of the arable farms on sandy soils meet the EU-norm of 50 mg nitrate per litre in groundwater. Between 1998 and 2003, no decline in leaching occurred and fertilizer inputs remained constant (Milieu en Natuurplanbureau RIVM, 2004). Field and model studies show that reduction of fertilizer inputs results to only a small decline in nitrogen leaching, while increasing risks of yield reduction. The studies also show that growing catch crops and especially the removal of nutrient-rich crop residues may reduce nitrate leaching more than further reductions in fertilizer input (Assinck and Willigen, 2004, Smit *et al.*, 2005). Removal or retention of leached nutrients in buffer strips and constructed wetlands may further reduce nutrient export to surface waters (Carpenter *et al.*, 1998).

Post harvest measures and treatment of drainage water are explicitly taken into account in the project 'Nutrients Waterproof'. In this project, arable and horticultural farming systems are developed which comply with the EU Nitrate and Water Framework directives. The systems were developed in 2004. From 2005 onwards, the systems are tested and monitored on the experimental farm Vredepeel in the S.E. of the Netherlands. 'Nutrients Waterproof' builds on results from earlier work in the project 'Farming for a future' (Langeveld *et al.*, 2005, Zwart *et al.*, 2005). This paper describes the design and ex-ante assessment of the systems in 'Nutrients Waterproof'.

Materials and methods

The prototyping method was used to develop farming systems (Vereijken, 1999; Haan and Garcia Diaz, 2002). Two integrated systems differing in mineralization capacity have been developed. Both systems have a six-year crop rotation with potato, triticale, lily, peas, leek, maize and sugar beet. Cropping and fertilization strategies have been developed for both systems in order to reduce leaching, while maintaining high yield levels. Catch crops are grown after triticale

and maize. Crop residues of triticale, leek, sugar beet and catch crop are removed in the system with a low mineralization capacity only. The effects of the cropping strategies on leaching are calculated with a simple spreadsheet model (XCLNCE, Zwart *et al.*, 2001).

An assessment of economic perspectives of the system was made by comparing the fertilization related costs of the designed systems with those of conventional strategies.

A system was designed to collect and treat drain water in three different types of constructed wetlands: a surface flow system which is easily combined with nature development, and two (with or without common reed) horizontal subsurface flow systems. The medium of the wetland without reed consists of a mixture of sand and straw. This system is a new cheap concept for arable farming. The horizontal subsurface flow system designed according to conventional guidelines, uses the smallest surface area, but is the most expensive.

Results and discussion

Calculations shows that the adopted fertilization strategy results in a 50% lower nitrogen input from fertilizers and manure compared to conventional fertilizer recommendations (Table 1). Calculated N-surpluses are low compared to conventional systems, as expected yield levels and removal of N are at least equal to regional averages. The removal of crop residues reduces N-surpluses even further in the system with low mineralization capacity. First results of cropping season 2005 confirm that yield levels can be maintained, with the low input of fertilisers.

Nitrate leaching, calculated with XCLNCE, is expected to exceed the EU-limit of 50 mg/l for ground water in both systems. In view of earlier results with XCLNCE, nitrate leaching may be somewhat overestimated. Differences in nitrate leaching between the two systems are small. For that reason, the systems have been slightly altered to increase differences in mineralization rates.

In both systems, costs of fertilization related measures are lower than conventional strategies. Direct fertilization costs are lower because of lower fertilizer input. Post harvest measures are more expensive since these measures are maximally implemented in the systems whereas they are almost absent in conventional systems in the region.

Table 1. Calculated N-surplus, N-input and N-leaching and costs (costs for constructed wetlands excluded).

	Available N-input from fertilizers and manure (% of conventional recommendations)	N-surplus (kg N ha^{-1})	Nitrate leaching (mg NO_3 l^{-1})	Costs fertilization (€ ha^{-1})	Costs post harvest measures (€ ha^{-1})	Difference in costs fertilization related measures compared to conventional strategies (€ ha^{-1})
High mineralization capacity	51	77	88	301	73	-21
Low mineralization capacity	43	21	85	270	49	-78

All three types of constructed wetlands are expected to have the same efficiency: it is expected that 80% of the incoming N is removed. About 6-8% of the cropping area is needed for the water reservoir and wetlands. The costs of the constructed wetlands are estimated to be between € 10,000 – 20,000 per ha of arable land. The cost-effectiveness is about € 15 – 20 kg^{-1} N removed. Constructed wetlands are effective but expensive. Combination with water storage and nature development increases perspectives (green/blue services).
From 2005 onwards, measurements in the systems have to demonstrate whether nitrate leaching, N-input, N-surplus, yield levels and N-removal in wetlands meet expectations.

Conclusions

Model calculations using XCLNCE do not confirm the hypothesis that integrated farming systems with efficient nutrient management strategies and maximum use of post harvest measures have low nitrate leaching. This is not in line with earlier field experiments and model calculations in 'Farming for a future'. Fertilization related costs of these systems are comparable to conventional systems; however, risks of yield reduction are higher.

When after implementation of all feasible on field measures nitrogen leaching is still too high, the removal of nutrients in buffer strips or wetlands might be an effective but expensive option. Combination with other functions may enhance its feasibility.

From 2005 onwards, measurements have to demonstrate whether expectations are fulfilled and whether adjustments are needed to reduce nitrate leaching further.

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Nitrate leaching from different tillage systems in winter cereals

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Abstract

Non-inversion tillage may have a potential to reduce nitrate-N leaching. In order to examine the effect of soil tillage intensity and crop rotation on nitrate leaching, experiments were established in autumn 2002 on a loamy sand at Foulum and a sandy loam at Flakkebjerg, Denmark. The tillage treatments were P: Ploughing; H_{8-10}: Harrowing, 8-10 cm; H_{3-4}: Harrowing, 3-4 cm and D: Direct drilling. Three rotations were included in the experiments: R2, R3 and R4. In the first two nitrate measuring years, the crops in rotation R2 were winter barley (*Hordeum vulgare* L.), winter rape (*Brassica napus* L.). The crops in R3 and R4 were winter wheat (*Triticum aestivum* L.)/catch crop and spring barley (*Hordeum vulgare* L.)/catch crop. In R3 straw was removed while straw was cut and left in R2 and R4. At Foulum and in crop rotations with catch crops there were no significant differences in leaching between different soil tillage systems. At Flakkebjerg non-inversion tillage resulted in poor crop establishment of winter rape, which increased leaching, especially in H_{3-4} and D, while leaching from H_{8-10} was not significantly different from P. The risk of poor establishment of a crop must be taken into consideration when environmental effects of non-inversion tillage are evaluated.

Keywords: crop establishment, direct drilling, tillage

Background and objectives

The importance of non-inversion tillage has increased in Denmark in recent years, especially because it has the potential of reducing labour and machinery costs. Non-inversion tillage may also reduce nitrate-N leaching (Hansen and Djurhuus, 1997; Stenberg *et al.*, 1999), although this has not been investigated under Danish conditions in winter cereal-based rotations. The objective of the research presented here was to study the effect of soil tillage intensity and crop rotation on nitrate leaching.

Materials and methods

The experiments were established on a loamy sand at Foulum and a sandy loamy at Flakkebjerg, Denmark. The clay (<2 µm), silt (2-20 µm), fine sand (20-200 µm) and coarse sand (200-2000 µm) contents of the soil (0-25 cm) at Foulum were 92, 126, 444, 307 g kg^{-1}, respectively, and at Flakkebjerg 147, 137, 426 and 270 g kg^{-1}, respectively. The organic carbon content at Foulum was 18 g kg^{-1} and at Flakkebjerg 12 g kg^{-1}. The experiment was established in autumn 2002 as part of a larger experiment. The actual design was a split-plot design in four replications with two factors: crop rotations (R2-R4) as main plots and soil tillage as sub-plots. In the first two years, when nitrate leaching was measured, the crops were winter barley and winter rape in R2 and winter wheat/catch crop, spring barley/catch crop in R3 and R4. In R3 straw was removed while straw was cut and left in R2 and R4. The tillage treatments were P: Ploughing to 20 cm, H_{8-10}: Harrowing to 8-10 cm, H_{3-4}: Harrowing to 3-4 cm and D: Direct drilling. The catch crop consisted of perennial ryegrass (*Lolium perenne* L.) undersown in the main crop. At Flakkebjerg the soil was ploughed in late autumn. At Foulum the catch crop was killed by herbicides at approximately 1 November and the soil was ploughed in spring. All crops were fertilized with 100 kg NH_4-N ha^{-1} in pig slurry and the remaining amount from the fertilizer recommendation was applied as mineral fertilizer. At Flakkebjerg 30 kg N ha^{-1} of the fertilizer recommendation for winter rape was applied in autumn. All other crops received the total amount of manure/fertilizer in spring.

For the calculation of nitrate leaching, soil water samples were taken using porous ceramic cups which were permanently installed in the autumn of 2002 at a depth of 1 m. The sampling system consisted of suction cups (655x01-B1M1, 1 bar, standard, Soilmoisture Equipment Corporation, Goleta, CA) mounted on PVC pipes (Hansen *et al.*, 2000). Two samplers were installed per plot (i.e., eight per treatment giving a total of 96 samplers per location). A suction of approximately 70-80 kPa was imposed 2 to 3 days before sampling. During this period the suction decreased as a result of water sampling. The soil water samples from each plot were bulked before analysis, frozen within a few hours and later analyzed for nitrate N (Best, 1976). Generally, sampling was carried out once every other week, except in periods of drought or frost. Percolation was calculated using the model Evacrop (Olesen and Heidmann, 1990).

Results and discussion

A significant interaction between soil tillage and crop rotation was found at Flakkebjerg, while this was not the case at Foulum. No significant effect of crop rotation was found at any of the locations. Leaching from the three rotations at Foulum during the period from March 2003 to November 2004 showed no significant differences between different soil tillage systems (Table 1). In R3 and R4 a catch crop was grown in autumn/winter 2003-2004 and in R2 winter rape was grown. These crops have taken up nitrogen during the autumn and thereby reduced leaching to a level that was not significantly different for the different tillage treatments.

At Flakkebjerg there was a significant effect on leaching in R2, where winter rape was grown in 2003-2004 (Table 2). Leaching in H_{3-4} and D (33 and 38 kg N ha^{-1}) was significantly higher than in P (12 kg N ha^{-1}). As leaching only took place from December 2003 to May 2004, the extra leaching with non-inversion tillage is probably a result of a poor establishment of winter rape in the dry autumn of 2003, especially in H_{3-4} and D. In P, H_{8-10}, H_{3-4} and in D there were 100, 79, 17 and 5 plants m^{-2}, respectively. Also at Foulum the establishment of winter rape with non-inversion tillage was poor, even though the number of plants was higher at 141, 86, 85 and 60 plants m^{-2}, respectively. Fertilization in the autumn with 30 kg N ha^{-1} at Flakkebjerg most likely increased leaching in the non-inverted plots.

Table 1. Foulum. Nitrogen leaching (kg N ha^{-1}) from 12 March 2003 to 15 November 2004.

Crop rotation	R2	R3	R4
Crops	Winter barley – winter rape	Winter wheat /CC – Spring barley/CC	Winter wheat /CC – spring barley/CC
Straw	Left	Removed	Left
Treatment			
P	22	40	31
H_{8-10}	18	36	33
H_{3-4}	19	39	38
D	33	39	40
$LSD_{.95}$	ns	ns	ns

Table 2. Flakkebjerg. Nitrogen leaching (kg N ha^{-1}) from 12 March 2003 to 15 November 2004.

Crop rotation	R2	R3	R4
Crops	Winter barley – winter rape	Winter wheat /CC – spring barley/CC	Winter wheat /CC – spring barley/CC
Straw	Left	Removed	Left
Treatment			
P	12^{b}	30	33
$H_{8\text{-}10}$	24^{ab}	28	38
$H_{3\text{-}4}$	33^{a}	23	28
D	38^{a}	20	32
$LSD_{.95}$	15	ns	ns

Values within a column followed by the same letter are not significantly different.

The suction cups were installed permanently in autumn/winter 2002-2003 in such a way that soil tillage, including ploughing, could be carried out without disturbing the equipment. However, the installation disturbed the soil to a depth of 40 cm, which means, that the results of the first year are uncertain with respect to evaluating the effect of soil tillage on leaching. Anyway, an increase in leaching with non-inversion tillage (R2, Flakkebjerg) after establishment of the second crop seems to be due to effects of tillage on crop N uptake.

Conclusions

At Foulum and in crop rotations with catch crops there were no significant differences in leaching between different soil tillage systems. At Flakkebjerg non-inversion tillage resulted in poor crop establishment of winter rape, which increased leaching, especially in $H_{3\text{-}4}$ and D, while leaching from $H_{8\text{-}10}$ was not significantly different from P. The risk of poor establishment of a crop must be taken into consideration when environmental effects of non-inversion tillage are evaluated.

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Evaluation of farm gate nitrogen balance. What is reachable considering nitrogen use efficiency?

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Abstract

The farm gate nitrogen balances (FGNB) are widely used methods to asses sustainability or pollution risk. The FGNB is also compulsory in some countries for the application of the Nitrate Directive. A method of FGNB calculation and evaluation of the N surplus based on managed outputs and N use efficiency has been developed in Walloon Region. The evaluation is made by the comparison between the N surplus of the FGNB and a N surplus reference value. This reference value is calculated for each farm. It is the N surplus of a theoretical FGNB where the N input is calculated from N outputs in farm products considering a reference N use efficiency for each production step. The difference between the N surplus of the farm and the N surplus reference value provides an estimation of room for improvement.

The method was applied on a set of 18 farms very different in production types. One half of the farms had a N surplus lower than the reference value indicating that nitrogen management can be improved quite easily. It was also showed that some farms with a high N surplus had sometimes few possibilities to diminish it because their N use efficiency was already good considering their specialization.

Keywords: nitrogen efficiency, nutrienet balance,

Background and objectives

The farm gate nitrogen balance calculates the balance between nitrogen (N) added to an agricultural system and N removed from it. It is a widely used indicator to asses sustainability or pollution risk. In Wallonie, the FGNB and the 'farm soil surface balance (FSSB)' are compulsory for farmers engaged in 'the quality approach' (QA). The QA allows the farmer to use more organic nitrogen on his farm. On arable land the maximum quantity allowed to be spread is 80 kg organic $N.ha^{-1}$ in vulnerable areas and 120 kg organic $N.ha^{-1}$ elsewhere. On grassland, this amount is limited to 210 kg $N.ha^{-1}$ everywhere including direct restitutions by grazing animals. In the QA, these maximums are 130 kg organic $N.ha^{-1}$ on arable land and 250 kg organic $N.ha^{-1}$ on grasslands.

For this purpose, a method of FGNB and FSSB was set up by a team of scientists and approved by the Government (Moniteur Belge, 2004). Despite of the close link between FGNB and FSSB, this paper concern only the evaluation of the FGNB.

According to the law, farmers engaged in the QA have to obtain a satisfying value of FGNB and FSSB. But, what is a satisfying value of FGNB?

As shown in the literature, it's difficult to fix reference values because big differences of nitrogen surplus are found between farms of different specialization or production systems (Simon *et al.*, 2000; Öborn *et al.*, 2003). In particular, it is difficult to set one reference value for farms with more than one animal species or more than one specialization (e.g. milk and meat).

An evaluation method of the N surplus of the FGNB has been developed to overcome this problem. This method permits to establish a reference value for each farm taking into account its own characteristics. The difference between the N surplus of the farm and the N surplus reference value provides an estimation of room for improvement of the nitrogen

efficiency. In the QA, complementary to the FGNB, the FSSB and its reference value are used as indicators of the pollution risk.

Material and methods

N surplus of the FGNB is the difference between N inputs (as fertilizer, feedstuffs, symbiotic and asymbiotic fixation, atmospheric deposition, purchased animals) and N outputs in sold products (milk, meat, eggs, crop…) and other products moved off farm (e.g. manure exportation). Stock variation, increase and decrease are taken into account respectively as output or input. N surplus represents both an environmental loss and possibly a storage in the soil.
The N surplus reference value is calculated for each farm individually from outputs as follows (Figure 1):

1. N requirement in feedstuffs is calculated by dividing N amount in animal products by its N use efficiency value. N use efficiency of each animal production type has been established considering a typical herd including young animals for replacement and unproductive period (N use efficiencies :20% for dairy cows herd, 9% for suckler cows herd, 19% for fattening beefs, 17% for sows and piglets, 30% for fattening pigs, 50% for broilers and 35% for laying hens)
2. N amount in feedstuffs to be produced in the farm is the N requirement in feedstuffs minus N amount in bought feedstuffs. If purchased feedstuffs exceed this requirement, it is considered that purchases must be limited to the amount required.
3. N amount in crops is the sum of N in sold crops and in feedstuffs to be produced.
4. N fertilizer requirement for crops is calculated by dividing the N amount in crops by a fertilizer N use efficiency of 70%.
5. N available from farm manure is the sum, for each animal category, of product of animal number, mean organic N production and N efficiency coefficient. Regulations limit organic N spreading capacity and the amount in excess needs to be exported out of the farm.
6. Mineral N fertilizer to buy is calculated as the difference between N requirement for crops and N available from farm manure.

Figure 1. Steps for the calculation of the 'N surplus reference value'.

Conventionally, the N surplus of the FGNB is expressed in kg $N.ha^{-1}$.
The difference between real N surplus and N surplus reference value provides an estimation of possible improvement.

Results and discussion

The method was tested on 18 farms with different productions (arable, dairy cattle, suckler cows, fattening beefs, sows and piglets, fattening pigs, broilers and laying hens). Eight farms had more than two different productions. The evaluation method needs to separate produced and bought feedstuff for each specialization to use the right N use efficiency coefficient. This separation is sometimes difficult particularly when feedstuff are the same for different production (e.g; milk or meat cattle) or when animals are moving from one specialization to another inside the farm (e.g. piglets become fattening pigs, calves of suckler cows become fattening beefs).

Data acquisition on farms is often difficult and time consuming. Sometimes, data accuracy can be questioned principally for N amount in sold products, manure nitrogen content and amount, stock variation…

A significant correlation was found between the N surplus reference value and the N surplus of the farm (N surplus ref. = 0.8932 Nsurplus + 19.264; r=0.69). The nitrogen surplus was lower than the reference value for one half of the farms. This indicates a good result for nitrogen management. Bad results and thus possible improvement in nitrogen management are not linked to a high nitrogen surplus. On the other hand, good nitrogen efficiency was also observed in farms with a high nitrogen surplus.

For farms with a nitrogen surplus higher than the reference value, progresses in nitrogen use efficiency are required. This could be obtained thanks to balanced diet and nitrogen fertilization taking better account of nutrients in farm manure and soil nitrate residues.

Conclusion

This method of evaluation seems interesting to fix realistic target values for individual farms taking into account multispecialization of the farm and to evaluate room for improvement. Its practicability needs to be improved for a wider use. Also coaching of farmers is necessary to improve accuracy of data collection by good book keeping of purchased and sold products.

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Nitrogen balances and nitrate leaching of conventional and organic crop rotations under Northern German conditions

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Abstract

Various studies, based on either small-scale plot experiments or modelling approaches, indicate a lower risk of nitrate leaching in organic than in conventional farming systems. In this study, an N-intensive conventional all-arable crop rotation and three organic rotations were compared over a three-year period on the experiment farm Lindhof in Northern Germany. In our study, the higher N inputs and higher N surplus in the conventional system did not lead to significantly higher nitrate leaching than in the organic all-arable crop rotations. Comparison of an organic all-arable crop rotation with the corresponding mixed farming system showed significantly higher nitrate leaching in the all-arable system. Management of the grass/clover (mulching versus feeding) had the strongest influence on nitrate leaching in the organic systems. Since farm type and farming practices had a decisive impact on nitrate leaching, we conclude that a comprehensive assessment of land use systems at both the regional and the farm scale is needed to legitimize incentive payments for the adoption of organic farming standards.

Keywords: clover, conventional farming, grassland, leaching, nitrate, organic farming

Background and objectives

Nitrate leaching from agricultural soils significantly contributes to nitrate enrichment of groundwater and eutrophication of natural ecosystems. Various studies, based on either small-scale plot experiments or modelling approaches, have indicated lower risks of nitrate leaching from organic than from conventional farming systems (Hansen *et al.,* 2000; Haas *et al.*, 2002). However, some studies on farm nitrogen budgets indicate a substantial risk of nitrate leaching on organic farms (e.g., Scheringer and Isselstein, 2001). Comparisons of organic and conventional arable cropping systems under favourable soil and climatic conditions are rare. Because of lack of farm-scale data, N-fluxes of different conventional and organic cropping systems were compared in a study on highly productive arable soils at Kiel University's experimental farm Lindhof in Northern Germany that has been sub-divided in an organic and a conventional farm unit.

Materials and methods

During the conversion from conventional to organic farming, different crop rotations were implemented at the field scale at Lindhof, an arable farm of 150 hectares located on fertile loamy soils in Northern Germany close to the shoreline of the Baltic Sea (mean annual air temperature 8,7 °C, mean annual precipitation 774 mm). Over a three-year period (1999/2000- 2001/2002), organic and conventional crop rotations were analyzed for productivity, nitrogen balances and nitrate leaching. The conventional crop rotation (system 1) was oilseed rape – winter wheat – sugar beet – winter wheat. Average annual N input was 186 kg ha^{-1}. Three organic crop rotations represented different farming systems with regard to (i) intensity of N input (50% legumes (system 2) vs. 33% legumes (system 3) in a rotation) and (ii) farm type (all-arable farming system with grass/clover green manure (system 2) vs. a mixed farming system with livestock production, harvesting of grass/clover and manure application to non-legume crops (system 4), both with 50% legumes). Each crop rotation was carried out on four fields, representing four 'replicates'. Only the harvest years 1999-2001 were considered, to minimize the risk that organically managed fields that had been converted to organic standards in 1994, were still affected by residual effects of conventional practices, such as high levels of soil nutrient supply. The farm scale of the experiment ensured that crops were managed as on commercial farms, and that yields

were comparable to practical conditions. To compare yields of the different crops, it was necessary to transform them to comparable standards. The yields of the all-arable crop rotations were transformed to grain equivalents (GE) using values for standardised fresh matter contents from the official tables of the German Federal Agency for Agriculture and Nutrition (BLE, 2004). To compare yields of the all-arable crop rotation with those of the mixed farm system (Table 2), yields were transformed into metabolisable energy (ME) using data of the official German feedstuff evaluation tables (Anonymous, 1997). Leaching of nitrate was determined with ceramic suction cups, of which 300 had been installed on the farm area. Leachate was sampled weekly during the three winters and analyzed for NO_3. The volume of drainage water was calculated by a general water balance model. Nitrogen fixation was estimated on subplots as difference of the absolute measured
N-amounts of crop and crop residues (root, stubble and litter) between the considered legume and a non-N-fixing reference crop. Reference crop for peas was oats and for grass/clover pure perennial ryegrass grown on subplots under a similar management system.

Results and discussion

Some agronomic and environmental characteristics of the analyzed all-arable farming systems are given in Table 1. Yields (in GE) in the conventional system were much higher than in the organic all-arable systems. This may be attributed to a higher nutrient input, a target-oriented use of plant protecting agents, and the absence of a (non-yielding) green manure crop in the conventional system. In spite of the significantly higher N input and N surplus of the conventional system, nitrate leaching did not differ significantly from the organic crop rotations. The observed range in nitrogen leaching was from 20.1 to 23.6 kg NO_3-N ha^{-1}. Related to the average drainage (253 mm in three winters), NO_3-N loads were below the EU threshold value of 50 ppm NO_3 in drinking water, which is equivalent to the leaching of 28.6 kg N ha^{-1}. The relatively high N losses via leaching in the organic all-arable systems were due to inefficient utilization of mineralized N from the grass/clover mulch.

Table 1. Yield of grain equivalents (GE), N input, N balance, N leached, fossil energy input, and energy efficiency of all-arable farming systems during the experimental period 1999/2000-2001/2002 (means of entire crop rotations, all values are averages per year).

Farming system	Crop rotation	Yield [GE ha^{-1}]	N input [kg ha^{-1}]	N balance [kg ha^{-1}]	Leached NO_3-N [kg ha^{-1}]	Energy input [GJ ha^{-1}]	Energy efficiency [GE GJ^{-1}]
1. Conventional	1.1 Sugar beet						
all-arable farm	1.2 Winter wheat	107.5 a[1)]	186.0	47.5	23.6 a	15.57 a	6.65 a
	1.3 Winter oilseed rape	(100%)	(100%)	(100%)	(100%)	(100%)	(100%)
	1.4 Winter wheat						
2. Organic	2.1 Grass/clover mulched						
all-arable farm	2.2 Oats	31.8 b	88.5	12.1	21.2 a	6.07 b	5.28 b
50% legumes	2.3 Grain legume	(30%)	(48%)	(25%)	(98%)	(39%)	(79%)
	2.4 Winter wheat/potato						
3. Organic	3.1 Grass/clover mulched						
all-arable farm	3.2 Oats	29.8 b	67.0	17.5	20.1 a	4.50 c	6.58 a
33% legumes	3.3 Winter rye	(28%)	(36%)	(37%)	(85%)	(29%)	(99%)

[1)] *Same letters in one column are not significantly different $P \leq 0.05$.*

Furthermore, the relatively high average input of mineral fertilizer-N of 186 kg ha^{-1} into the conventional system was the main reason for the much higher input of fossil energy compared to the organic systems. As productivity in the conventional system was also much higher, energy *efficiency* was not lower. Table 2 shows the same characteristics for the all-arable and mixed organic farming systems with 50% legumes. Utilisation of grass/clover herbage in animal

production and higher yields of non-leguminous crops due to the application of manure led to 50% higher energy yields and 30% higher energy efficiency in the mixed farming system. Nitrate leaching was significantly lower in the organic mixed farming system than in the all-arable system even though total N input was higher. Harvesting the grass/clover herbage resulted in higher nitrogen fixation and lower leaching losses in the following winter. Feeding the grass/clover herbage produced manure which resulted in better N-utilization within the entire rotation.

Table 2. Metabolisable energy yield, N input, N balance, N leached, fossil energy input, and energy efficiency of organic all-arable and mixed farming systems during the period of 1999/2000-2001/2002 (means of entire crop rotations, all values are averages per year).

Farming system	Crop rotation	Yield [GJ ME ha^{-1}]	N input [kg ha^{-1}]	N balance [kg ha^{-1}]	Leached NO_3-N [kg ha^{-1}]	Energy Input [GJ ha^{-1}]	Energy efficiency [GJ GJ^{-1}]
2. Organic all-arable farm 50% legumes	2.1 Grass/clover mulched 2.2 Oats 2.3 Grain legume 2.4 Winter wheat/potato	36.3 b[1)] (100%)	88.5 (100%)	12.1 (100%)	18.5 a (100%)	6.07 b (100%)	5.90 b (100%)
4. Organic mixed farm 50% legumes	4.1 Grass/clover harvested 4.2 Oats 4.3 Grain legume 4.4 Winter wheat/potato	55.4 a (153%)	137.2 (155%)	11.1 (92%)	11.4 b (62%)	6.96 a (115%)	7.87 a (133%)

[1)] *Same letters in one column are not significantly different P≤0.05.*

Conclusions

Farming system (specialised arable versus mixed farming) had a decisive impact on agronomic and environmental performance. Under favourable growth conditions, organic farming was not advantageous in terms of nitrate leaching and fossil energy efficiency. We conclude that a comprehensive assessment of land use systems at both the regional and the farm scale is needed to legitimize incentive payments for the adoption of organic farming standards.

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Climate-rainfall changes and N-fertilization effects on triticale yield

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Abstract

Agroecological quality has a well established dependence on climate-rainfall changes because the water problems are pressing. The droughts and floods that were experienced in Hungary in the early 1980's have drawn renewed attention to the analysis of these problems. New research on climate change-soil-plant systems is focused on yield. This paper reports the climate-rainfall changes x soil (acidic sandy brown forest) x mineral N-fertilisation x plant interactions on triticale yield in a long term field experiment set up at Nyírlugos in north-eastern Hungary under temperate climate conditions in 1962.

The main conclusions of the research are:

1. On the basis of „general' and triticale-specific rainfall deficiency values 1991, 1995 and 2000 could be classified as average, 1993 as dry, 1992, 1994 and 1996 as droughty, 1997, 1998 and 2001 as wet and 1999 as extremely wet.
2. In average years the yield of the control plots stabilised at 1.4 t ha^{-1}. N fertilisation resulted in an increase from 0 to 0.85 t ha^{-1} in the grain yield as compared to the control. The triticale yields could only be enhanced economically by full treatment with NPK, NPKCa-, NPKMg- or NPKCaMg.
3. Without fertilisation the yield in the dry and drought years was 14% and 36% respectively lower than the yield in average years. With nitrogen fertilizer application the yield decreased by 32% and 36%, respectively.
4. In the wet years on the unfertilised plots the yield declined 14% and in the case of the nitrogen fertilisation the yield was not different from the yield in average years. In the extremely wet year the yields did not differ from the yields obtained in average years.
5. The relationships between rainfall during the vegetation period and the effect of fertilizer application on yield could be characterised by a second-degree correlation depending on the level of fertilisation. The maximum triticale yield (5.0-6.0 t ha^{-1}) was obtained with 550-600 mm rainfall during the growth period.

Keywords: climate, fertilizer, nitrogen, triticale, yield

Background and objectives

Climate-rainfall changes are recognized as a serious environmental issue (Johnston, 2000). Presently the build up of greenhouse gases in the atmosphere and the inertia in trends in emissions may cause significant climate changes in the 21th century (Márton, 2001a., b). There is a need to understand the effect climate change may have on agricultural production. Among the natural catastrophes, drought and flooding currently cause the greatest problems in field crop production (José *et al.*, 2001). Weather-rainfall changes at Hungary started already in 1850 (Márton, 2002a, b, c). Triticale is the most important arable crop in many countries (Márton and Pekli, 2003) but sofar, little research in the field of climate-rainfall change impact assessment has been undertaken. Since triticale is sensitive to the prevailing weather-rainfall conditions it is important to evaluate the effects of anthropogenic climate-rainfall change on its production. Having a particularly high requirement for soil nitrogen, phosphorus, potassium, calcium and magnesium, triticale is an indicator crop for the soil nutrient status. Our research objective was to estimate and describe the rainfall change and fertilizer-N effects on triticale yield on an acidic sandy brown forest soil in a long term experiment under temperate climate conditions at Hungary from 1990 to 2001.

Material and methods

The effect of rainfall quantity and distribution on the yield of triticale dependent on fertilisation level was studied in a long-term field experiment on a acidic sandy brown forest soil in North-Eastern Hungary. Agrochemical characteristics of the soil were: pH (H_2O) 5.8, pH (KCl) 4.6, hydrolytic acidity 8.1, hy_1 0.3, humus 0.6%, CEC 5-10 mgeq 100 g^{-1}, total N 33 mg kg^{-1}, AL-P_2O_5 43 mg kg^{-1}, AL-K_2O 52 mg kg^{-1}. The experiment consisted of 32x4=128 plots in a split-split plot and factorial random block design. The gross plot size was 10x5=50 m^2. The nitrogen fertiliser rates in kg ha^{-1} $year^{-1}$ were 0, 50, 100, 150. The average phosphorus, potassium, calcium and magnesium application rates were 90 P_2O_5 ha^{-1}, 90 kg K_2O ha^{-1}, 175 kg Ca ha^{-1} and 40 Mg ha^{-1}, respectively. The fertilisers were applied in the form of calcium ammonium nitrate (25%), superphosphate (18%), potassium chloride (40%), calcium carbonate and magnesium sulphate. The groundwater table was at 2-3 m depth. Rainfall and experimental data were analysed according to Hungarian traditional (Harnos, 1993) and RISSAC-HAS (Márton, 2002c) standards, MANOVA (SPSS) and regression analysis (SPSS).

Results and discussion

On the basis of „general' (Harnos, 1993) and triticale-specific rainfall deficiency values (Márton, 2002c) the years 1991, 1995, 2000 could be classified as average, 1993 as dry, 1992, 1994, 1996 as droughty, 1997, 1998, 2001 as wet and 1999 as extremely wet. In average years the yield of the control plots stabilised at the level of 1.4 t ha^{-1}. N fertilisation resulted in a grain yield increase of 0 to 0.85 t ha^{-1} as compared to the control. The yield increase of triticale as a result of fertiliser application was only profitable when NPK, NPKCa, NPKMg or NPKCaMg was applied. Grain yields up to 3.9 t ha^{-1} were then obtained. Without fertilisation the yield in the dry and drought years was 14% and 36% lower than the yield in an average year. With nitrogen fertiliser application the yield in the dry and drought years was 32% and 36% lower than in an average year. In the wet years on the unfertilised plots the yield declined 14% and in the case of nitrogen fertilisation the yield was not different from the yield in an average year. In the extremely wet year similar yields were obtained as in an average year. The relationships between rainfall during the vegetation period, N application and application of combinations of N with P, K, Ca and Mg, and yield were characterised by a second-degree correlation depending on the level of fertiliser application. The maximum yield (5.0-6.0 t ha^{-1}) was obtained with 550-600 mm rainfall during the growth period.

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Can field-grown nursery stock meet the EU Nitrate Directive?

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Abstract

Field-grown nursery stock on dry sandy soils has to comply with the EU nitrate limit of 11.3 mg nitrate-N L^{-1}. A maximum soil N-surplus of 76 kg N ha^{-1} complies with this limit. The soil N-surpluses of five commercial nursery stock companies exceeded the maximum N-surplus in almost all years. The measured nitrate N-concentrations on these nurseries ranged from 25 to 50 mg L^{-1} exceeding the EU nitrate-limit in all years. Measures identified in a desk study to reduce the N-surplus were banning slurry products and use organic materials low on effective N, postpone slurry applications to the second or following growing season, and the use of winter hardy, annual catch crops.

Background and objectives

In the Netherlands, field grown nursery stock is located for the larger part on dry, sandy soils. On these soils, Schröder *et al.* (2004) estimated that a maximum surplus of 76 kg nitrogen (N) ha^{-1} $year^{-1}$ on the soil N-budget complies with the EC nitrate limit of 11.3 g nitrate-N L^{-1} in the shallow groundwater given the boundary condition of a steady state. Taking into account climatic conditions and soil characteristics it is estimated that a farm gate N-surplus (based on total N) of 90 kg N ha^{-1} will also comply with the EC nitrate limit. But, is the nitrate-N concentration on nursery stock companies too high? In addition, can either of these N-surpluses be met? These questions were addressed in a study which focused on 1: the farm gate and the soil N-budgets of commercial nursery stock companies, 2: the accompanying nitrate-N concentration in the shallow ground water and 3: on developing cost effective packages to meet the predefined N-surplus.

To investigate items 1 and 2, an on-farm project was started with 33 participating commercial arable and horticultural farms. In this project, called 'Farming for the Future', five commercial ornamental nursery stock companies participated. To investigate item 3, a desk study was performed to develop cost-effective packages of measures to improve N-utilization and meet the maximum N-surplus.

Materials and methods

Commercial nursery stock companies: In the on-farm project two shrubs and hedges nurseries (B1= 20 ha; B2 = 0.84 ha), two rose nursery (root stocks and bush roses, B3 = 25 ha, B4 = 26 ha) with arable crops (barley, sugar beets) and one street tree nursery (B5 = 18 ha) participated. Annual N-budgets were calculated for each nursery based on registered fertilization practices from 2000 to 2003 as the N-application + aerial deposition - N-crop removal. The farm gate N-budget was formulated with the total N applied whereas the soil N budget was formulated with the effective N from applied organic products. The aerial deposition was 43 kg N ha^{-1}. N-crop removal values, specified for each crop and crop age (from other research) were used. In addition, nitrate-N levels in the shallow groundwater were measured in the spring at each nursery from 2002 to 2004 by the National Institute for Public Health and the Environment (RIVM).

Nursery stock models: In the desk study three model nurseries were designed resembling the nurseries in the 'Farming for the Future' project, a shrubs and hedges nursery, a rose nursery and a street tree nursery (Van Der Sluis *et al.*, 2004). For each nursery a N-fertilization strategy was assumed following the Dutch fertilization guidelines. A nursery specific organic matter application strategy was based on local customs. Measures identified in research and which reduce the N-surplus were:

1. the replacement of slurry applications by organic products containing less effective N in the year of application, like compost;
2. to balance effective N application rates with N uptake in the year of application;
3. to limit application rates of organic products to 2 tons of effective organic matter (meaning that 2 tons of organic matter remain in the soil one year after application);
4. growing winter hardy, annual catch crops sown in August in the year of planting;
5. delaying slurry applications to the second and following growing seasons of perennial street trees.

These measures did not reduce the available N to below the requirements for optimal dry mass production, according to the Dutch fertilization guidelines. The effect of these measures on the N-budgets and nitrate-N concentrations were evaluated.

Results and discussion

Commercial nursery stock companies: The average N removal by crops was 64 kg N ha^{-1} and was relatively low compared to arable crops (Table 1). The application of N with organic products was 93 kg N ha^{-1}. The application of fertilizer N was low, 32 kg N ha^{-1} on average. The average farm gate N-surplus of all five nurseries was 104 kg ha^{-1}, exceeding the farm gate N-target value. In some years, B4 approached the 90 kg N ha^{-1} target value but in those years trees were planted on ploughed grassland. The average soil N-surplus was much smaller than the farm gate N-surplus and nursery B1 and B4 met the predefined target of 76 kg N ha^{-1}.

Table 1. The total N applied with organic products, the mineral N from deposition, fertilizers and organic products, the crop removal and the calculated farm gate N-surplus and soil N-surplus (average of four years) of the five commercial nursery stock companies. (All units in kg ha^{-1}).

	total N-applied	Mineral N			Removal	N-surplus	
	organic	deposition	fertilizer	organic		farm gate	soil N
B1	88	43	26	44	57	100	54
B2	44	43	67	32	60	94	82
B3	134	43	24	101	73	128	95
B4	131	43	0	102	78	96	67
B5	68	43	41	60	50	102	94

The measured N concentrations ranged from 25.0 to 50.2 mg L^{-1} (mean = 37.6), exceeding the EU nitrate-limit (Figure 1). Neither the farm gate N-surplus nor the soil N-surplus showed a relationship with the measured N-concentrations. A possible reason might be that these nurseries have not reached a steady state (yet) and that high N-concentrations are a result of large application rates of previously applied organic products.

Nursery stock models: Three measures were identified in the desk study: 1) ban slurry products and use organic materials low on effective N, like compost; 2) postpone slurry applications to the second or following growing season; 3) winter hardy, annual catch crops sown in August in the year of planting. The fertilizer application in the second growing season can be reduced as the catch crop residue N mineralizes but growing catch crops increased cost by 100€ ha^{-1}. However, few measures were identified which complied with the demand for the organic matter and the farm gate N-surplus target value. The combination of catch crops grown in the year of planting of the commercial crop, combined with the use of low compost applications were identified to comply with the demand for organic matter as well as the farm gate N-surplus target value.

Figure 1. The average nursery stock nitrate concentrations in the shallow groundwater measured by the RIVM of the five commercial nursery stock companies.

Conclusions

In conclusion, nursery stock companies will not easily comply with the EU Nitrate Directive on sandy soils due to relatively low crop N-uptake and high N-application rates with organic products. Measures to reduce the N-surplus may conflict with the (assumed) need for organic matter applications. Fertilizer applications are low and therefore hard to reduce. Additional cropping techniques need to be explored and implemented.

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Increased plant density to increase nitrogen use efficiency and reduce nitrate losses

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Abstract

A possibility to reduce N-losses might be to increase N-uptake through increased plant densities. A combined model consisting of the conifer-growth model and the soil water and nitrogen balance model was used to test if increased plant densities reduce nitrate-N concentrations to meet the nitrate directive for white cedar. Plant densities ranged from 1 plant per m^2 to 4 plants per m^2 (common practice) to finally 25 plants per m^2. Nitrate-N concentrations were not reduced to the target value in the first growing season whereas nitrate-N levels were reduced sufficiently when plant densities were increased to 6.25 plants per m^2 in the second growing season.

Background and objectives

Ornamental conifers are commonly grown on dry, sandy soils. As plant densities and N-uptake of ornamentals are generally low, N-losses may easily pollute shallow groundwater to above the target value of the European Union (EU) of 11.3 mg NO_3-N L^{-1}. One possible solution to reduce N-losses might be to increase N-uptake by increasing plant densities. Current practice uses plant densities of 4, 5 or 6.6 plants m^{-2} (0.5 m inter-row and 0.5, 0.4 or 0.3 m intra-row distances, respectively). Row width is linked to the mechanical handling but an intra-row width of 0.3 m still produces marketable plants. When row width is reduced to 0.3 m, the plant density increases to 11.1 plants m^{-2}. But will this reduce the N-losses to below the target value? If not, will a further increase of plant density suffice? In this study model calculations show whether increased plant densities due to decreased row widths are able to reduce N-losses below the EU target value.

Materials and methods

A combined model consisting of the conifer-growth model CONGRO and the soil water and nitrogen balance model FUSSIM2 (Heinen and De Willigen, 1998; Pronk, 2004) was used to explore effects of increased plant densities of white cedar (*Thuja occidentalis*) on dry mass production, N-losses and available nitrogen use efficiency (N_{aus}). N-losses are calculated as the total amount of nitrogen that is lost over the bottom boundary of the system. The N_{aus} is calculated as the total amount of mineral nitrogen that was available for uptake during the growing season. CONGRO simulates a demand for water and nitrogen which is then imposed on FUSSIM2. If the demand is not met transpiration and N-uptake are reduced to meet the available sources. Dry mass production is reduced accordingly. Effects of increasing plant density on N-uptake and nitrate-N concentrations in percolating soil solution at 1 m depth were also investigated. The Dutch standard fertilization recommendations for conifers of 50 kg N ha^{-1} in the first growing season mid May and at the end of June, and 60 kg N ha^{-1} in the second growing season mid May and at the end of June were used. The available nitrogen in the top soil (0-0.3 m depth) was subtracted from all application rates. Plant densities were the standard practice of 4 (S) plants m^{-2} (0.5 to 0.5 m) and a range of different plant densities: 1.0 (P1), 6.25 (P2), 11.1 (P3) and 25 (P4) plants m^{-2} (1.0, 0.4, 0.3 and 0.2 m inter-row and intra-row distances). Although P1 and P4 are never used in practice, these treatments were included for the theoretical exploration of the reduction of N-losses through increased plant density. Fertilisers were applied broadcast. Drip irrigation of 0.5 L per plant was scheduled when 250 hPa was reached in the root ball to optimize growth (Pronk *et al.*, 2005). Runs were made for 30 stands, starting in 1970 and using weather data from Wageningen, the Netherlands.

Results and discussion

Dry mass production per m^2 increased with increasing plant densities, although total dry mass of individual plants decreased from 109 g for P1 to 64 g for P4 in the first growing season and from 1091 g for P1 to 193 g for P4 in the second growing season (Table 1). N-uptake and N_{aus} increased also with increasing plant densities but N-losses hardly decreased. Where N-uptake of S was more than doubled (first growing season) N-losses were only reduced by 29%. The N-loss decreased from 74 kg N ha^{-1} for S to 62 kg N ha^{-1} for P4 in the first growing season. In the second growing season the N-loss decreased from 74 kg N ha^{-1} for S to 49 kg N ha^{-1} for P4. N-losses occurred at all times during the cultivation period: the first growing seasons (April-October), the winter period (November-March) and the second growing season (April-October). Fertiliser application rates slightly increased due to increased plant densities.

In the first growing season, the NO_3-N concentration at 1 m depth varied between 28 (P1) to 21 mg L^{-1} (P3) (Figure 1). In the second growing season the concentration for P1 and S exceeded the EU target value whereas the concentrations of the increased plant densities were in agreement with the target value.

Table 1. Simulated dry mass, nitrogen uptake (N-uptake), nitrogen losses (N-loss) and the nitrogen use efficiency (N_{aus}) of white cedar in the first and second growing season.

	Dry mass				N-uptake		N-loss		N_{aus}	
	g m^{-2}		g $plant^{-1}$		kg ha^{-1}		kg ha^{-1}		%	
Treatment	first	second	first	second	first	second	first	second	first	second
S	109	1091	102	645	29	122	80	86	8	37
P1	407	2580	109	1091	8	51	74	74	28	71
P2	598	3163	96	506	41	143	71	66	39	79
P3	968	3967	87	357	57	167	64	54	55	86
P4	1598	4816	64	193	66	174	62	49	64	89

Figure 1. The simulated nitrate-N concentration in the first and second growing season with increasing plant densities and the nitrate N-target value of the European Union.

Conclusions

This study showed that in the first growing season increased plant densities will not reduce nitrate concentrations to the EU target value for white cedar on sandy soils. However, in the second growing season a small modification of the planting distance to 0.4 to 0.4 m reduced the nitrate-N concentration to below the target value.

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Nitrogen and phosphate surpluses after the introduction of a new agricultural policy in Switzerland

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Abstract

In 1993 a new agricultural policy was introduced in Switzerland with the aim of reducing the negative environmental impacts of agricultural production. As a major constraint of the new policy farmers have to comply with a balanced nitrogen (N) and phosphate (P_2O_5) supply at the farm level (10% supply restriction) as revealed by the official Swiss nutrient balance reference method 'Suisse-Bilanz'. The aim of our study was to test the effectiveness of the new legislation and to estimate how well the official 10% surplus restriction rules for N and P_2O_5 are met in practice. We therefore calculated national nutrient balances according to the official reference method for 1990 and 2000. The results show a significant reduction of the nutrient surpluses for N and P_2O_5 from 28.9 to 4.8 kt N and from 39.5 to 5.4 kt P_2O_5. Given in percent of the nutrient requirements, this was a reduction from 31% to 6% for N and from 68% to 10% for P_2O_5. The strong surplus reductions demonstrate the effectiveness of the introduced legislation and indicate that the official 10% surplus restriction rule is respected by most farmers.

Keywords: manure, nitrogen, nutrient balance, phosphate

Background and objectives

In 1993 a new agricultural policy was introduced in Switzerland with the aim of reducing the negative environmental impacts of agricultural production. The new strategy primarily aimed at uncoupling the traditional link between the agricultural price policy and farmer's income by replacing production subsidies with a direct payment system for 'special ecological contributions'. In 2003, 87% of the Swiss farms participated in this basically voluntary program and over 95% of the agricultural area was cultivated according to the program guidelines (Flury 2005). To benefit from payments, farmers have to act in accordance with around 40 different criteria related to crop rotation, nitrate leaching, soil erosion, pesticide use, animal welfare and biodiversity. In addition, as a major constraint of the program the participants have to comply with a balanced nitrogen (N) and phosphate (P_2O_5) supply at the farm level. Using the official Swiss nutrient balance reference method 'Suisse-Bilanz' (Amaudruz *et al.* 2003) the participants have to demonstrate that the calculated N and P_2O_5 inputs to crop production in the form of manure and organic and mineral fertilisers do not exceed the crop requirements by more than 10% at the whole farm level. The nutrient balance has to be calculated on a yearly basis either by the farmer or an extension officer and is subject to regular controls by local or national authorities.

The aim of our study was to test the effectiveness of the new agricultural policy by the calculation of nutrient balances at the national level for 1990 and 2000 using the 'Suisse-Bilanz' methodology. The approach should also allow estimating how well the official 10% surplus restriction rules for N and P_2O_5 are met in practice.

Material and methods

The N and P_2O_5 balance calculation were done with the Excel-based 'Suisse-Bilanz' (LBL 2004). The method balances the nutrient requirements of crops and grassland according to official guidelines with nutrient inputs in the form of manure and organic and mineral fertilisers (Flisch *et al.* 2001). For the nutrient requirements of arable crops, non-yield dependent fixed values are used for N, while P_2O_5 requirements are linearly adjusted to yield. The yield-dependent

requirements of grassland are calculated via the roughage consumption of the livestock held on the farm. The nutrients in manure are calculated on the basis of guide values on excretions of different livestock categories. While 100% of the P_2O_5 excretion is taken into account, unavoidable gaseous N losses in houses and during manure storage of 15 to 40% (depending on livestock category and housing system) are accounted for. Of the total N applied in manure an availability of 60% for grassland and 45% for arable crops is considered, assuming good or even best farming practice. The activity data used for the calculations was based on official statistics (animal numbers, areas of crops and grasslands, mineral fertiliser use, compost and sewage sludge application and national importations on roughage) (BFS 1992/2001, SBV 1990/2000). For farm management parameters (type of housing systems, grazing practice, manure management, feeding strategies, etc.) the calculations were based on a representative survey conducted in the year 2002 (Reidy *et al.* 2004).

Results and discussion

The nutrient balance calculated with the official 'Suisse-Bilanz' methodology for 1990 revealed a national surplus of 28.9 kt for N and 39.5 kt for P_2O_5 (Table 1). The N and P_2O_5 inputs through manure and organic and mineral fertilisers thus exceeded the recommended nutrient requirements of crops and grassland by approximately 31% for N and 68% for P_2O_5. Several years after the introduction of the new agricultural policy both surpluses have drastically decreased. For the year 2000 the calculated national nutrient surplus was 4.8 kt for N and 5.4 kt for P_2O_5. Compared to 1990 this was a decrease of 83% for N and 86% for P_2O_5.

Table 1. National nutrient balance for nitrogen and phosphate in 1990 and 2000 calculated with the official Swiss reference method (Amaudruz et al. 2003).

	1990		2000	
	N	P_2O_5	N	P_2O_5
	Kt		kt	
Nutrients from livestock manure	49.2	51.9	37.7	42.2
Nutrients from mineral and organic fertilisers	71.5	46.0	53.3	17.7
Total available nutrients	120.8	97.9	91.0	59.9
Nutrient requirement crop production	91.9	58.4	86.2	54.5
Surplus	28.9	39.5	4.8	5.4
Surplus in percent of requirements	31%	68%	6%	10%

This significant reduction of the surplus can be interpreted as a direct consequence of the new legislation and especially of the introduction of the nutrient balance restriction. It led not only to a strong decrease of the organic and mineral fertiliser input (-25% for N and -62% for P_2O_5). In parallel, the amount of nutrients available from livestock manure decreased by -23% for N and -19% for P_2O_5. These results show that farmers adapted relatively quickly to the new restrictions by reducing mineral fertilizer use, reducing livestock numbers or changing management practices related to the nutrient excretion of the animals. Since 1990 livestock numbers of cattle and pigs therefore decreased by about 15% and the utilisation of feed with low protein or/and phosphorus content in pig production is widespread (Reidy and Menzi 2005). Only a small reduction can be observed between 1990 and 2000 in the nutrient requirement of arable crops and grassland because the agricultural cultivated area remained constant.

Calculating national input/output balances for nitrogen and phosphorus from Swiss agriculture Spiess (2005) found a national surplus of 62 kt N and 44 kt P_2O_5 for 1990 and 51 kt N and 14 kt P_2O_5 for 2000, respectively. The contrasting results of the two balancing methods can be primarily explained by methodological differences. For N the two methods differ strongly with respect to the extend gaseous losses and other N sources (e.g. atmospheric deposition, biological nitrogen fixation) are taken into account. They are therefore hardly comparable. Although the methodological

differences are less pronounced for P_2O_5 they may still explain the differences to a large extend. A detailed comparison of the two balances revealed that the different results can be primarily attributed to the utilisation of different assumptions on the feeding strategy of farmers and thus of livestock excretions. With respect to the comparison of different balancing methods, similar Conclusions were drawn by Schuepbach (2002). By comparing different balancing approaches he concluded that different methods are only comparable to a limited extent and a correct interpretation of element balances is only possible if the methodology is well known (assumptions used, aims and consequences of the balance) and taken into account.

Conclusions

Calculation of a national nutrient balance according to the Swiss nutrient balance reference method 'Suisse-Bilanz' revealed a strong decrease of the N and P_2O_5 surpluses from 1990 to 2000. This reduction can primarily be attributed to the introduction of a new legislation limiting the utilisation of N and P_2O_5 at the farm level. However despite the strong decrease of the surplus the inputs in 2000 still exceeded the recommended nutrient requirements of arable crops and grassland by approximately 6% and 10% for N and P_2O_5, respectively. Although these results may differ within a range of $\pm$5-10%, depending on the assumptions of key input variables, they indicate that at the national level the official 10% surplus restriction rules are respected by most farmers. The 10% safety margin allowed is possibly sometimes exhausted but not generally exceeded. Taking into account the relatively low surpluses and the given uncertainty of several input variables, it can be concluded that the participating farms meet the official guidelines.

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A biophysical and economic-GIS modelling approach for livestock manure management in the humid tropics

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Abstract

In Malaysia, large amounts of manure are being excreted daily in intensive pig production units. This manure is poorly handled in pond systems and under-utilised as a fertiliser resource. Applying manure as an organic substitute for mineral fertiliser in annual or perennial cropping systems could provide a viable solution to the problem. Optimising application and distribution of manure in neighbouring cropping systems would minimise environmental pollution loading and contribute to improvement of farm economics. Managing excessive amounts of manure and in consequence their nutrient load, in particular N, could be assisted by using a biophysical, economic and spatially explicit modelling system.

Background and objectives

In Malaysia, large amounts of manure are being excreted daily from intensive pig production units. This manure is poorly handled in pond systems and under-utilised as a fertiliser resource. As a result, it has become an environmental hazard that contributes to greenhouse gas emissions and waterways pollution. Applying manure as an organic substitute for mineral fertiliser in neighbouring annual or perennial cropping systems could provide a viable solution to the problem. Optimising application and distribution of manure in neighbouring cropping systems would minimise environmental pollution loading and contribute to improvement of farm economics. Managing excessive amounts of manure and in consequence their nutrient load, in particular N, could be assisted by using a biophysical, economic and spatially explicit modelling system. Several approaches to model manure application on agricultural land in respect of nutrient N cycling have been developed, particularly for temperate regions. Few attempts, however, have been made to design such modelling systems for tropical areas. Models could provide information not only on availability of manure and optimising application rate on crop land, but also on- and off-farm nitrogen losses and the likelihood of their spatial distribution. This study aims at developing a GIS based biophysical and economic modelling system on nitrogen cycling to assess (i) efficacy of pig manure application on neighbouring perennial cropping systems, and (ii) the impact of pig manure on the surrounding environment in humid tropical conditions.

Material and methods

Site specific biophysical and economic models at farm level and their databases were integrated into ArcGIS 8.2 through the share coupling method (Brandmeyer and Karimi, 2000). The integration enabled scaling up from single farm level to all pig farm units in the study area at watershed level. The biophysical models consisting of a Pig Production/Manure Model (=Model 1; Tee *et al.*, 2004a), a Soil N Cycle and Plant N Uptake Model (=Model 2; Tee *et al.*, 2004b) and an Economic Model that evaluated biophysical processes and cost effectiveness of manure applications in comparison to commercial fertiliser. The joined model databases and GIS themes were used to perform spatial analysis, e.g. density distribution and interpolation methods. Initial biophysical and economic model simulations were based on a 50-sow pig farm and a matured (5 -15 years old) oil palm plantation as an example to assess the interaction of N cycling from pig farm to soil, plant uptake, environmental pollution and cost efficiency of manure utilisation. The model simulation considered three different types of manure inputs (fresh = FM), liquid = LM and compost = CM) that

might contribute to a better manure management with respect to nutrient carrying capacity, minimising nutrient N losses and manure application cost effectiveness in relation to distance from the pig farm. The results were evaluated for environmental and economic aspects.

Simulation results

Table 1 summarises the simulation results of biophysical and economic models. Model 1 estimated that about 13.5 tonne of manure N are annually excreted from the model farm, of which about 23, 54, 59% N were lost for FM, LM and CM respectively, during waste handling. This indicated that using FM directly can significantly reduce N losses and conserve nutrient N for utilisation in oil palm plantations. Results of Model 2 indicated that the optimum amounts of manure to satisfy 90% oil palm yield were 111, 213, and 269 tonne/yr for FM, LM and CM, respectively. The total N losses in the field were 255, 231 and 199 kg /ha/yr. The inverse results of amount N required and N losses for FM, LM and CM in the field were due to the different nutrient contents among these manure sources. The simulation results also indicated that if the available amount of manure N after losses (10, 6 and 5.5 tonne for FM, LM, and CM) were applied onto oil palm plantations as a substitute for mineral N fertilisers, FM could supply the N amount required for 17 hectares followed by 9 and 7 hectares for LM and CM.

Table 1. Simulation results generated by the biophysical and economic models for a 50-sow farm and manure application in a matured (5 – 15 years) oil palm plantation.

	FM	LM	CM
Fertiliser aspects			
Manure N excreted (kg N/yr)	13512	-	-
Amount of manure N after losses (kg N/yr) [1]	10364	6206	5555
Amount of manure generated (tonne/yr) [2]	1919	1939	1916
Optimum amount of manure per ha (tonne/ha/yr) [3]	111	213	269
Land area required (ha)	17	9	7
Environmental aspects:			
Total N losses at farm (kg N/yr)	3148 (23)[7]	7306 (54)	7957 (59)
Total N losses at field per ha (kg N/ha/yr)	255	231	199
Total N losses from farm to field (kg N/yr)	7483	9385	9350
Economic aspects:			
Manure application cost per ha[4] (RM[5]/ha)	78	112	131
Crop production cost (RM/ha)	2218	2252	2271
Net income (RM/ha)	2298	2264	2245
Net profit (RM/ha) [6]	465	431	412

[1] *Calculated from model 1 = 1919 tonne/yr/farm;*
[2] *Total N in unit kg/tonne: FM – 5.4 kg/tonne; LM – 3.2 kg/tonne and CM – 2.9 kg/tonne (Model 1);*
[3] *Amount of manure needed to satisfy 24 tonne Fresh Fruit Bunches (tFFB)/ha yield;*
[4] *Hauling distance = 15 km;*
[5] *RM = Malaysian Ringgit, e.g. 1 Euro = RM 4.50;*
[6] *Offset commercial fertiliser net income=1833 RM [this is not quite clear is it the additional (e.g. difference) to commercial fertilizer?];*
[7] *Percentage.*

Economically, manure application showed a good net income potential to oil palm farmers. If the manure was given free, the oil palm farm could earn about RM 465/ha, RM 431/ha and RM 412/ha, respectively for FM, LM and CM by replacing inorganic fertilizer with manure. In the cost effectiveness evaluation of pig manure utilisation, FM had the lowest manure application and hauling costs. This was due to a higher nutrient N content and hence lower amount of FM

manure required for application. If all the FM was applied on 17 hectares of oil palm, the total net profit of FM was RM 7914, which was 50% and 60% more than LM and CM, respectively.

Figure 1. Distribution of NH_3 of all pig farms in the study area.

Figure 2. Distance of pig farms to surrounding oil palm plantations.

Figures 1 and 2 illustrate the GIS-biophysical and economic integration results of some of the spatial analysis. Figure 1 indicates the hot spots of NH_3 emission and the distribution rate of NH_3 from the entire pig farm units over the study area. Figure 2 shows the distance of pig farms to surrounding oil palm plantations and which determines the possibility and cost effectiveness of delivering manure to these sites. These scenario results could provide strategic information on manure management to the potential users. The visualised results could assist in evaluating and monitoring environmental impact of N loading and provide strategies for distributing the manure into neighbouring fields.

Conclusions

The biophysical and economic models showed an ability to evaluate the efficacy of manure application for both environmental and economic aspects. The models indicated that fresh manure was the best option when compared to LM and CM. It had the lowest total N losses from farm to field of a 50-sow farm (7483 kg N/yr), it supplied N to double the land area compared to LM and CM, and it had the lowest manure application cost with the highest net profit. Integration of biophysical and economic models with GIS allowed us to scale up from farm to landscape level and provided visual information and scenarios for farmers, regional planners and policy makers on environmental impact and cost effectiveness of alternative manure utilisations.

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Integrated spatial modelling of nitrogen pollutants at a landscape scale

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Abstract

A modelling framework, LANAS (Landscape Analysis of Nitrogen and Abatement Strategies), has been constructed to simulate the spatial and temporal flows of nitrogen (N) within a rural landscape. Five models of N flow (covering the principal systems: grazed land, arable land, farmyards, freshwater and the atmosphere) were linked together to simulate the flows and interactions of N for a rural study area in eastern England. This work highlights the spatial aspects of N management, i.e. the distances at which the impacts occur depend on the type of N pollution and where the sources and sinks (e.g. farms and nature reserves) are in relation to each other. For instance, impacts of NH_3 emission and NO_3^- leaching tend to occur close to the source (e.g. within the catchment), while N_2O emissions have impacts at a global scale. The LANAS modelling framework provides a tool for investigating the spatial and temporal flows of N and can also be used to simulate pollution abatement scenarios and the trade-offs between different methods of abatement.

Keywords: landscape, modelling, nitrogen, spatial

Background and objectives

Successful nitrogen (N) management at a landscape scale must take many factors into account to protect the environment while keeping farms viable. Losses of N from agricultural systems can take many forms and can impact on terrestrial and freshwater ecosystems (ammonia (NH_3) and nitrate (NO_3^-)) as well as contribute to climate change (nitrous oxide (N_2O)). Ideally N management should minimise the losses of all N pollutants, but due to the nature of the systems this is not always possible. Trade-offs between reductions of different pollutants have to be considered when formulating a management strategy to avoid swapping one problem for another. Ammonia is emitted into the atmosphere principally from farming activities such as livestock housing and the spreading of manures and slurries onto agricultural land (Misselbrook *et al.*, 2000). The re-deposition of NH_3 close to the source can cause eutrophication and acidification of sensitive ecosystems resulting in loss of plant species diversity (Bobbink *et al.*, 1998). Nitrous oxide is predominantly emitted into the atmosphere from agricultural soils and has a high radiative-forcing potential and thus contributes to global warming (IPPC, 2001). Nitrate is leached from agricultural soils, predominantly as a result of fertilisation by organic and inorganic manures, and increased groundwater NO_3^- concentrations can affect freshwater quality leading to impacts on freshwater ecology and even toxicity to humans (Dunn *et al.*, 2004). Without considering the spatial and temporal flows of N pollutants it is difficult to understand these systems fully and therefore more difficult to devise management practices to reduce impacts on the environment. To model the system effectively, an integrated modelling framework, LANAS (Landscape Analysis of Nitrogen and Abatement Strategies), has been constructed which incorporates five N flow models. This has been applied to a real landscape - study area in rural eastern England

(Norfolk) - which improves the understanding of N flows and interactions and allows the investigation of abatement scenarios.

Materials and methods

Details of the modelling framework are reported elsewhere (Theobald *et al.*, 2004). It consists of the following five models, simulating N flows in the atmosphere, grazed fields, arable fields, farmyards and river catchments:

LADD	Atmospheric dispersion and deposition	(Dragosits *et al.*, 2002)
SUNDIAL	N cycling in arable systems	(Smith *et al.*, 1996)
FYNE	NH_3 and N_2O emissions from farmyards	(Theobald *et al.*, 2004)
NGauge	N flows of grazed systems	(Brown *et al.*, 2005)
INCA	Catchment flows of N	(Whitehead *et al.*, 1998)

The models are linked as shown in Figure 1. The input data for the models are land cover maps (derived from aerial photographs), farm management data (collected through farm surveys) and meteorological data. The modelling framework runs at a monthly time-step.

Figure 1. Linkages between the component models of the LANAS integrated model and the flows of nitrogen pollutants between them.

Results and discussion

Output from the LANAS framework is in the form of a spatial database from which maps of the major N flows can be produced. Figure 2 shows an example map of atmospheric N deposition to the study area. The region of large deposition rates to the east is from a large poultry farm with multiple units. Deposition rates are largest close to the poultry units as well as in the southeast corner of the study area where there is extensive semi-natural vegetation. Similar maps were produced for NO_3^- leaching and the atmospheric concentrations of NH_3 and N_2O.

Figure 2. An example map of total nitrogen deposition to the study area.

Conclusions

This work has highlighted the spatial aspects of N flows, i.e. the distance at which the impacts occur depends on the type of N pollution and where the sources and sinks are in relation to each other. For instance, impacts of NH_3 deposition and NO_3^- leaching tend to occur locally or regionally, while N_2O impacts at the global scale. This integrated modelling approach has allowed the investigation of the spatial and temporal interactions between the pollutants. LANAS also provides a tool for devising and testing abatement options for farms and can help assess which N pollutants should be prioritised for abatement at any particular location.

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Tradeoffs in the allocation of nutrient (N), cash and labour resources within smallholder African farms

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Abstract

Resource allocation decisions have implications for the fertility of the soils. Efficient use of N and its retention in the farm system are keys to ensure short and long-term food production, by impacting on soil organic matter (SOM). Focusing on a single season, our objective was to analyse tradeoffs between resource allocation strategies to improve the N capture efficiency (NCE) within the system, while ensuring food production. We used inverse modelling techniques, linking a dynamic simulation model for N dynamics at plot scale with a non-linear optimisation shell, and applied it to a case-study farm with heterogeneous soil qualities in the East African highlands. Maximising food production (maize) and minimising N losses were the main objectives at farm scale, with constraints set by labour and cash availability. The NCE varied widely (0.1-0.8), and at farm scale, the highest NCE's were achieved at intermediate maize yield levels (c. 2 t ha^{-1}); higher and specially lower yields led to greater N losses by erosion and leaching. High yielding strategies associated to high N rates were achieved when enough labour was hired to ensure good agronomic practices (e.g. weeding).

Keywords: erosion, Inverse modelling, leaching, losses, nitrogen, Optimisation models, soil, use efficiency

Background and objectives

Operational, day-to-day decisions made by farmers in allocating resources have implications for the future fertility of their fields. Increasing productivity and ensuring sustainability of smallholder farms depends on understanding the tradeoffs between immediate concerns such as generating food and cash, and longer-term goals such as maintaining/improving soil fertility, which is closely related to soil organic matter (SOM) content. N inputs sufficient to increase biomass production and thereby SOM are unlikely to be justified by immediate returns, unless the N 'capture' efficiency (NCE) is increased by optimising management and resource allocation across the farm. Since farms are spatially heterogeneous, the best measures to ensure high NCE's will vary for different fields, posing trade-off questions to farmers – i.e. allocation of scarce resources (N, cash, labour) to different fields/activities. Focusing on short-term (operational) decisions, our objective was to analyse tradeoffs between different strategies for resource allocation to improve the NCE within the system, by minimising losses, while ensuring food production.

Materials and methods

The tradeoffs analysis was done by linking a dynamic simulation model to a non-linear optimisation tool, using inverse modelling techniques. We linked DYNBAL (DYNamic simulation of Nutrient BALances), which was tested and used in western Kenya (Tittonell, 2003), to MOSCEM (Multi-Objective Shuffled Complex Evolution Metropolis) (Van Wijk and Vrugt, 2005). We focused on a highly populated region of East Africa (average farm size is 1.7 ha) that is representative of most tropical highlands for its high agricultural potential (good soils – Dystro mollic Nitosols – and favourable climate). Maize is the main food crop grown in the region, where the average annual rainfall reaches 2000 mm. For a case-study farm, we combined different scenarios of financial liquidity to allocate to competing farm management activities (parameters) and accounting for differences in soil quality between the various fields of the farm (Table 1). The case study farm was one of relatively high resource endowment, to support the assumptions of high investments in inputs and in hiring labour.

Table 1. Key biophysical parameters used to characterise the different soil quality units of the case-study farm

Soil quality class*	Area per land class (ha)	Clay content (%)	Soil organic C (g kg^{-1})	Total soil N (g kg^{-1})	Bulk density (Mg m^{-3})	Slope length (m)	Slope steepness (%)	Field capacity (% v/v)
Fertile	0.5	39	21.6	2.3	1.26	25	2.1	42
Average	1.3	33	17.9	1.6	1.24	38	9.6	38
Poor	0.4	21	14.4	1.7	1.29	39	22.7	35

* *As recognised by farmers.*

Having maximising food production and minimising N losses as farm-scale objectives, and setting the total amount of capital for investment in agricultural production as main constraint (low = 5000 or high = 10000 KSh, where 75 KSh = 1 us$), we ran DYNBAL as many times as necessary to optimise a set of management parameters (e.g. fertiliser use, allocation of hired labour to different activities) for each field within the farm, considering different indicators of success (soil erosion, maize grain yield, N balance, income). The assumption was made of no nutrient limitations other than N.

Results and discussion

The underlying soil quality of the different fields of a farm, together with the operational management decisions, affected the NCE (roughly varying from <0.1 to >0.8) and thence the results of the optimisations across the farm. Threshold yields were identified for certain fields, above which soil (N) losses by erosion increased abruptly. Their value depended on the soil quality of each field (3.7 to 1.3 t ha^{-1} from close to remote fields). At farm scale, yields could be increased from 0 to almost 2 t without a substantial increase in the amount of N lost from the system (Figure 1 A); for this case study farm (2.2 ha), this represents an average yield of less than 1 t ha^{-1}, often considered as the regional average (e.g. Shepherd *et al.*, 1997).

Figure 1. (A) Relation between farm-scale grain production and N losses by erosion and leaching during one growing season for two different scenarios of farmers' investment capacity in agricultural production (N inputs, hired labour).

(B) Competing allocation of available capital between buying N fertiliser and hiring labour for weeding (the points plotted correspond only to farm scale yields above 5000 kg farm^{-1}).

When more N was applied to improve yields under the high investment capacity scenarios N losses increased abruptly, basically due to the application of N fertilisers to the poor fields; i.e. strongly sloped fields prone to large erosion losses (cf. Table 1). Subsequently, larger yields (> 3 t per farm) and more labour allocated to soil ridging led to better soil cover and N capture efficiency (larger root biomass) thereby reducing N losses by erosion and leaching. The larger NCE's (smaller N losses) were achieved within the range of total farm yields of 3.5 to 5 t, which corresponds to average yields of 1.6 to 2.3 t ha^{-1}, when farmers had a high investment capacity. Good yields with less associated N losses were obtained under the low investment scenario, suggesting that the current low-input practices on these naturally fertile soils (cf. Table 1) have higher NCE's. However, the maximum total farm yields to be achieved were below 7 t vs. almost 8 t for the high investment scenario (3.2 vs. 4.0 t ha^{-1}, respectively). The fact that these soils can produce relatively good yields without fertilisers has been previously documented (Tittonell *et al.*, 2005). Indeed, when only those points for which the farm scale yields were above 6.5 or 7.0 t (average yields > 2.7 or 3.2 t ha^{-1}, in the low and high investment scenarios, respectively) were considered, the allocation of cash under both scenarios of liquidity favoured investments in labour for weeding over investments in N fertilisers (Figure 1 B). Good agronomic management practices, such as weeding on time or proper planting densities, are crucial for improving NCE. Only when enough capital has been invested in labour for e.g. weeding, then it becomes profitable to invest a larger fraction of that capital in N fertilisers (high investments scenario).

However, for a comprehensive evaluation of resource allocation activities not only the operational (seasonal) but also the strategic time horizon should be considered. Sustained large yields over the long term (i.e. several seasons) represent larger additions of C to the soil that favour the build-up of soil organic matter, consequently improving soil quality. In such respect, Giller *et al.* (1997) indicated that any long-term strategy for building-up the N capital in the system and increase its use efficiency needs to be coupled with one for stabilisation of organic C in the soil.

Conclusions

The use of mineral N fertilisers may improve land and labour productivity at farm scale provided that simultaneous measures are taken to improve the NCE within the system; however, such measures are not always feasible for resource-poor farmers, due to labour and financial limitations. For the analysed scenarios of soil qualities, labour and capital availabilities, the overall N capture efficiency at farm scale improved for intermediate grain yield levels (around 2 t ha^{-1}), below and above which a large part of the applied N was lost from the system. This (non-linear) optimisation technique showed good potential for a truly integrated analysis of the biophysical and socioeconomic aspects determining the efficiency of N use within smallholder systems.

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Nitrogen surpluses of Flemish dairy, beef and arable farms: changes between 1992 and 2000

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Abstract

The farm level N-surplus of Flemish dairy, beef and arable farms were calculated for the period 1992-2000. In 2000 the dairy farms showed the highest N-surplus (262 kg N ha^{-1}) followed by the beef (195 kg N ha^{-1}) and arable farms (139 kg N ha^{-1}). The decreased surplus was mainly caused by a strong reduction of mineral fertilizer use. A better use of organic manure due to a better application strategy and the introduction of clover in the grassland are the most important reasons for the mineral fertilizer reduction on the cattle farms.

Keywords: arable farming, beef farming, dairy farming, nitrogen, nutrient balance

Background and objectives

Nitrogen surplus is a widely used indicator for assessing the environmental impact of agricultural production systems. There is no Flemish legal obligation to calculate farm N budgets, but some financial benefits from the government made that some farms use them as a tool for their management to optimize their nutrient use. It is also an opportunity to be better prepared to stricter legislation rules of the manure decree (Anonymous, 1991 and Anonymous, 2000) e.g. introduction of vulnerable zones.

Materials and methods

The Flemish Farm Accountancy Data Network (FADN) is a database of technical and economic data from a representative set of Flemish farms. We extracted the entries of the specialized dairy, arable and beef farms for the years 1992 to 2000 and calculated the nitrogen budgets on farm and on soil level. The considered N-inputs on farm level were mineral fertilizer, concentrates and by-products (e.g. sugar beet pulp, brewery grain), straw, purchased roughage, imported manure (e.g. from pig farms), deposition (48 kg N ha^{-1}) and N-fixation. The outputs were milk, sold animals and arable crops. The yields of the arable crops are based on the FADN data. Also stock differences were taken into account. We calculated the N-surplus as total N-input – total N-output; N-efficiency was defined as: (total N-output/total N-input) x 100 (Nevens *et al.*, 2005). In the same way we calculated the N-surplus and N-efficiency on soil level for the dairy and beef farms. On arable farms, N-surpluses on both levels are identical. The considered components of the N-input on soil level were mineral fertilizer, manure, deposition and N-fixation; N-output was N-yield by crops (forage and arable).

Results and discussion

Figure 1 shows that the farm N-surplus decreased for all types of farms during the period 1992-2000. Dairy farms showed the highest N-surplus compared to beef and arable farms; the N-surplus of the arable farms was the lowest and did not change during the period 1992-2000. Also Simon *et al.* (2000) found the highest farm N-surplus for dairy farms and the lowest for arable farms. Arable farms had no ammonia losses due to housing and storage losses of organic manure which resulted in a lower surplus than cattle farms.

Figure 1. Evolution of the farm N-surplus of different Flemish farm types during the period 1992-2000.

The average farm surplus decreased for all types of farms, but the reductions were stronger for dairy (57 kg N ha^{-1}) and beef farms (98 kg N ha^{-1}) than for arable farms (19 kg N ha^{-1}) (Table 1). The decreased surplus was mainly caused by a strong reduction in mineral fertilizer use (25% on all farm types). A better use of organic manure by a better application strategy (obligation by law) and the introduction of clover in the grassland were the most important reasons for the mineral fertilizer reduction on the cattle farms. The decrease in crop production in 2000 is not necessarily due to a lower fertilization level, but weather conditions are probably more important. The reduction of the N-surplus on the beef farms was higher due to a strong decrease of concentrate and byproduct use. The total N-input (concentrates + byproducts) decreased with 8% on dairy farms and 41% on beef farms. The preferential track for achieving a further decrease of the farm N-surplus is an even higher reduction of mineral fertilizer-N use (Verbruggen *et al.*, 2004).

Table 1. Farm characteristics of specialized dairy, beef and arable farms in 1992 and 2000.

		Dairy farms		Beef farms		Arable farms	
Year		1992	2000	1992	2000	1992	2000
Number of farms		129	79	14	24	13	11
Farm area	ha	27.3	32.1	27.4	39.4	48.9	34.4
Mineral fertilizer	kg N ha^{-1}	203	150	146	110	195	147
Import organic manure	kg N ha^{-1}	25	30	17	-4	67	90
N-fixation	kg N ha^{-1}	2	5	0	4	0	0
N-input by concentrates	kg N ha^{-1}	91	85	88	54	-	-
N-input by byproducts	kg N ha^{-1}	19	16	17	8	-	-
N-output by animal products	kg N ha	69	60	30	32	0	0
N-output by vegetable products	kg N ha	2	4	10	11	152	146
Farm N-surplus	kg N ha^{-1}	319	262	293	195	158	139
Stdev. farm N-surplus *	kg N ha^{-1}	93	87	111	84	39	91
Farm N-efficiency	%	18	22	11	19	49	51
Soil N-surplus	kg N ha^{-1}	280	223	252	147	158	139
Soil N-efficiency	%	48	54	42	64	49	51

* *Stdev.: standard deviation.*

The farm N-efficiency increase was smaller as expected by the decrease of the mineral fertilizer. This is due to a decrease of the N-outputs by animals or vegetable crops on dairy and arable farms. On the other hand, no reduce of N-outputs was found on the beef farms, resulting in an increased farm N-efficiency.

The soil N-surplus of dairy farms was higher than the soil N-surplus of beef and arable farms. This was due to a higher N-input of mineral fertilizer and organic manure. However, the soil N-efficiency of dairy farms and arable farms did not differ because the N-output by forage crops on dairy farms was much larger than the N-output by crops on arable farms. In 2000 beef farms exported organic manure which resulted in a reduced soil N-input and an increased soil N-efficiency (64%).

Verbruggen *et al.* (2004) assumed that a soil surplus of 110 kg N ha^{-1} and a corresponding farm surplus of 150 kg N ha^{-1} (for cattle farms) are realistic and practically feasible goals for sustainable farming in Flanders (i.e. compliant with nitrates directive). The found average values for each farm type are still above this goal. But as the standard deviation showed there was a large variation between farms and there were examples of farms actually attaining these goals. Those farms could play an exemplary function to other farms.

Conclusions

Dairy farms showed the highest N-surplus compared to beef and arable farms. The farm N-surplus of the arable farms was the lowest. The reductions of the farm N-surplus were the biggest for the beef farms followed by the dairy farms. The decreased surplus was mainly caused by a strong reduction in mineral fertilizer use. A better use of organic manure and the introduction of clover in the grassland are the most important reasons for the mineral fertilizer reduction on the cattle farms. Also a reduction of the concentrate and byproduct use resulted in a stronger decrease on the beef farms. The best farms realize soil surpluses below 110 kg N ha^{-1} which seems an achievable goal for a sustainable farm in Flanders.

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Minimising N losses in organic greenhouse horticulture using model-based fertiliser strategies

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Abstract

The large supply of N to greenhouse crops can lead to considerable loss of N, especially in organic horticulture where only organic fertilisers are used. A simulation model may be of help when optimising fertiliser strategy. To reach this objective, a mechanistic soil/plant model was parameterised. We used data from greenhouse trials with the crops chrysanthemum and sweet pepper. In both trials a number of organic fertilisers were applied with base and/or side dressings. Modelled mineralisation rate, parameterised on soil column data, slightly overestimated mineralisation in the trials. The fit between observations and simulation results was close for soil water status and satisfactory for crop N uptake and soil mineral N content. N losses in the greenhouse were mainly attributed to denitrification, as found in measurements and simulation. Losses by leaching were negligible due to an adequate irrigation strategy. The model seems robust in the calculation of annual N budgets for the soil/plant system, enabling it to be used into planning organic fertiliser strategies in greenhouse horticulture.

Background and objectives

Greenhouse crops require large amounts of N to realize their high yields. The high supply levels of N can result in considerable N losses, especially in organic horticulture where only organic fertilizers are used (Voogt, 1999). An additional challenge to organic growers is posed by the EU regulation that restricts applications of animal manure to 170 kg N ha^{-1} y^{-1}, urging growers to seek for alternative N sources. A simulation model may help commercial growers to optimise their fertiliser strategy to minimise N loss without loss of production (De Visser *et al.*, 2004). The simulation should incorporate the processes involved in N inputs and outputs at the field level, i.e. mineralisation, root uptake, storage, leaching and denitrification of N. The aim is to integrate the model in a decision support tool for growers.

Material and methods

Greenhouse trials

The N dynamics were studied in two organic fertiliser trials, in a year-round chrysanthemum crop in 2002 and in a sweet pepper crop in 2003 in a greenhouse in Naaldwijk, NL. In both trials, N supply of organic fertilisers was equal to a practice-based estimate of crop N demand. In the chrysanthemum trial, six treatments were carried out: application as an annual base dressing (compost, lucerne), as base dressing per harvest (blood meal, maltaflor) and as frequent side dressing (fontana, feather meal). In the sweet pepper trial, seven treatments were carried out: no fertiliser, base dressing (two types of mushroom compost, lucerne), side dressing (feather meal, lucerne, wulpack). In both trials, a base dressing of farm yard manure (FYM) of 170 kg ha^{-1} was given (EU maximum). Treatments were duplicated (chrysanthemum) or triplicated (sweet pepper). The response of crop growth to N supply and climatic conditions was quantified by determination of the biomass at the final harvest in four consecutive production cycles of ca. 80 days each (chrysanthemum) or one cycle of 266 days (sweet pepper). In the soil, water content and temperature were recorded hourly, and inorganic N measured bi-weekly. Greenhouse climate was measured at hourly intervals. The decomposition rate of C and release of N from 12 organic fertilisers was determined during lab incubation for 94 days at 20°C at 60% of saturated water content. Denitrification was measured according to the acetylene inhibition methods

(Tiedje, 1982). Potential denitrification was determined on intact soil cores by saturation with water and nitrate at 20 ^{0}C. Actual denitrification rate was measured at three depths in four treatments on three occasions during crop growth.

Modelling

The soil model consisted of a mechanistic 2-D model, containing modules on mineralisation (MOTOR), denitrification (DENIT), and water and N transport. For the MOTOR model (Whitmore *et al.*, 1997), decay rates of stable and labile organic matter pools were calibrated on basis of the incubation results. The DENIT model calculates actual denitrification from potential denitrification, soil water content, nitrate content, and temperature. The observed denitrification rates were used to calibrate the reduction functions for nitrate content, water-filled pore space and temperature (Heinen, 2005). Calibration of soil hydraulic parameters was based on measured soil water contents of the chrysanthemum trial. Transpiration parameters were tuned to the results of a fertigation model that has shown good fits with observations for a large set of greenhouse situations (W. Voogt, pers.comm.). Photosynthesis parameters were calibrated by fitting simulated to observed crop biomass. For this, data on crop growth and N uptake from the feather meal treatment were used, showing a high, non-limiting N nutrition. The mechanistic plant model was coupled to the soil model for hourly exchange of data on demand/supply of water and N, on root mass and on nitrate/ammonium preference.

Results and discussion

The organic fertiliser applications resulted in strong differences in inorganic N content in the soil during crop development. Yet, N uptake of both crops was of the same order of magnitude in all treatments (ca. 460 kg N ha^{-1}) except for treatments with high C/N ratio organic fertilisers ((mushroom)composts) or with reduced input (i.e. 170 kg N ha^{-1} with FYM only).

Figure 1. Inorganic N content in soil (0-25 cm) of compost (left) and blood meal (right) treatment in chrysanthemum. Please note the difference in scale. Lines, simulation.

The MOTOR module on decomposition and N release could be adequately parameterized with the incubation data ($R^2 \geq 0.90$). Calculation rules were quantified for denitrification by using a combination of measured potential denitrification and parameterized reduction functions. The soil/plant model predicted inorganic N content in the soil reasonably (Figure1), but for some fertilisers an overestimation occurred at the end of the examined period. Despite these differences, the simulated N uptake in the crop agreed with observed uptake for most treatments ($R^2 = 0.94$). Denitrification was notably high in the first months after base dressing application, with highest values in the lucerne treatment both in chrysanthemum (measured and simulated) and in sweet pepper (solely simulated). This denitrification loss was caused by moist, warm soils in combination with a high soil nitrate content.

Having all the relevant N fluxes of the soil/plant system available, an N balance was calculated for the complete growth period (Table 1).

Table 1. Simulated N fluxes in some of the treatments (kg N ha^{-1}). The listed organic fertilisers were applied as base dressing. A total input of 600 kg N in chrysanthemum and 830 kg N in sweet pepper was given, including 170 kg N ha^{-1} from FYM. Soil depth is 90 cm. SOM, soil organic matter.

	Chrysanthemum		Sweet pepper		
	Compost	Lucerne	-none-	Mushroom Compost	Lucerne
Anorganic org.fertiliser	55	48	4	23	11
Mineralized org.fertiliser	37	257	335	506	598
Mineralized SOM	227	227			
Sum In	**319**	**524**	**339**	**530**	**608**
Uptake	369	493	231	441	455
Denitrification	19	97	137	126	158
Leaching	5	5	12	12	11
Sum Out	**394**	**595**	**379**	**579**	**625**
In – Out	**-74**	**-70**	**-39**	**-49**	**-16**

The pool of soil inorganic N was reduced in all treatments, as was indicated by simulation (Table 1) and observations in soil layers 0-25 (Figure 1) and 50-55 cm (results not shown). The reduction resulted from either higher uptake/mineralisation ratios (compost, lucerne) or high denitrification losses due to low uptake/mineralisation ratios. Only minor amounts of water and associated nitrate were calculated to leach from the root zone.

Conclusions

The model is able to realistically simulate soil N dynamics as affected by fertiliser applications. Parameterisation for a specific situation is laborious. As a generic model, the soil/plant model can be used as a tool to evaluate organic fertiliser strategies in horticultural practice for their impact on crop production and on the environment.

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Reduction of nitrate leaching from intensive arable cropping by specific crop management

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Abstract

The nitrogen fertilization of an arable farming system which is common for the South-eastern part of the Netherlands could be reduced by 25-50% compared to the generally accepted advisory amounts, without affecting crop yields and crop quality negatively. Groundwater nitrate concentrations were reduced from ca 100 to ca 50 mg l^{-1} simultaneously. Key factors to obtain these results were: (i) careful tuning of expected N-demand and N-supply, (ii) accounting for N from deposition and for N from mineralization and (iii) using catch crops as far as possible.

Background and objectives

Nitrate leaching from agriculture is a serious problem in the Netherlands. Nitrate concentrations in shallow groundwater exceed the EU-Nitrate Directive level of 50 mg l^{-1} in large areas of the country. Nitrate leaching is, at least partly, the result of excess nitrogen inputs in agriculture. Dutch arable agriculture is characterized by a high level of crop production and quality, partly the result of high nutrient application rates. Under arable and horticultural production fields, the nitrate concentrations in shallow groundwater are the highest in the Southeastern part of the Netherlands. The objective of the present study was to decrease nitrate leaching and simultaneously maintain high production rates and high crop quality. In this paper we describe the results of a number of different measures to reduce nitrate leaching in arable and horticultural farming.

Materials and methods

An experiment was performed during three years on experimental farm 'Vredepeel', where the arable rotation included a number of horticultural crops and maize. The soil is a reclaimed peat soil (mesic typic Haplaquod) with on average 3.8% organic matter in the upper 30 cm and less than 0.5% below. The phosphate level is high (water soluble is 40 mg phosphate l^{-1} soil). The average highest ground water level is at 90 cm below the surface. Two systems were studied, the first of which very much resembled the current situation of the farms in the local area (integrated farming system IFS), while the second one (experimental farming system EFS) was developed to meet the environmental conditions regarding nitrate in groundwater. The crop rotation included potato, sugar beet, carrot, triticale or spring barley, maize, and fresh peas followed in the same year by fresh beans (the latter crop was replaced by a catch crop in EFS). Fertilization for each crop was based on the Dutch fertilization recommendation scheme and was similar for EFS and IFS except for nitrogen fertilization. Measures to reduce nitrate leaching included the application of fertilizer N only instead of a mixture of animal slurry and fertilizer N; split application of N, using catch crops where possible, accounting for atmospheric N deposition and from N mineralization and removing crop residues (final year only). N mineralization was estimated using a simple spreadsheet model (Zwart, 2001). In addition, a crop of peas followed by beans was replaced by peas followed by a catch crop. Crop production and nitrogen uptake was established for each crop and groundwater nitrate concentrations were measured frequently.

Results and conclusions

On average, nitrogen input, nitrogen uptake and crop production were 67%, 92% and 98%, respectively in EFS compared to IFS (Figure 1). In the final year N input was even reduced to 50% in EFS (Figure 2) without affecting crop production negatively. In the final year N from mineralization during the crop uptake period was estimated using XCLNCE. In previous years the results of this model with respect to inorganic N in the soil, corresponded fairly well with measured data (Figure 3). The average groundwater nitrate concentration was 50 mg l^{-1} in EFS and 98 mg l^{-1} in IFS (Figure 1), with a high variation during the year and between crops.

Figure 1. Average N input, N-uptake and yield in EFS as % of IFS and average groundwater nitrate concentration in IFS and EFS (mg l^{-1}).

Figure 2. Average N-input in IFS (2001-2003) and EFS as % of IFS

Figure 3. Linear regression between measured and XCLNCE calculated inorganic N (N-min, kg ha^{-1}) in the upper 30 cm of arable soils at experimental farm Vredepeel.

We concluded that (i) it was possible to reduce nitrate leaching without affecting crop production, (ii) application of a catch crop and accounting for N from mineralization was a critical factor, (iii) Dutch N fertilization recommendations need a revision and (iv) even more stringent management measures are needed to reduce nitrate leaching further.

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Working group 3

Manure quality: can it be manipulated and what are the effects on whole-farm N efficiency?

Report of Working Group 3

Manure quality: can it be manipulated and what are the effects on whole-farm N efficiency?

Report by Bannink, A.[1*] & Bos, J.F.F.P.[2]

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This working group addressed the implications of manure quality for a judicious use of nitrogen (N). An attempt was made to evaluate the influence of manure quality on N efficiency for several aspects of farm management, as well as on whole-farming N efficiency. The following aspects of manure-N management can be distinguished:

- feeding strategy and manure composition;
- N losses during manure storage;
- fertilizer value of manure.

These aspects can be represented as separate subsystems of whole farming systems according to Figure 1. The implications of management practices are not necessarily restricted to a single subsystem, but, in most cases, involve several subsystems simultaneously. This implies that changes in the effect of altered management on the efficiencies of N use within the subsystem are mutually dependent. This needs to be recognized when evaluating the effectiveness of specific management strategies for whole farm N efficiency.

Figure 1. Simplified representation of a farm system with its subsystems.

Feeding strategy and manure composition

The potential to manipulate the characteristics of cattle manure by nutritional measures appears to be high. Several measures may be considered:

- shifting from cutting grass in an early stage of maturity to a later stage;
- reducing fertilization rates;
- introducing maize silage, straw and energy-rich by-products in the ration;
- changing the level of concentrate feeding;
- feeding animals closer to or even below protein recommendations;
- altering protein degradation characteristics of feed N.

Modelling studies indicate that, depending on which of these measures are taken, N excretion may vary from 100 to 200 kg N per cow per year and ammonium content in slurry from one third to two thirds of total-N (Figure 2). When expressing the effect of the above nutritional measures per unit of animal product, N excretion is expected to range from less than 15 to more than 22 g N per kg milk. Due to the variation in ammonium-N content of slurry, ammonia emission also varies substantially, i.e. from 4.6 to 1.5 g per kg milk. Values for the lower end of the calculated ranges are confirmed in a case study for a dairy farm with atypical management practices (Bouma *et al.*, 2006), such as feeding of a ration with an estimated crude protein content of only 13.4%, resulting in less than 40% ammonium-N in slurry (Sonneveld and Bouma, 2005).

Particularly, too high a level of protein feeding should be avoided because this will not result in extra animal product and increase ammonium-N excretion and N losses from the farming system. Also, the characteristics of the protein may be manipulated by nutritional means. These characteristics differ between seasons and among plant species (e.g. legumes versus grasses) or even breeds. Protein characteristics influence the nutritional value of the feed and digestion of protein and, hence, also the ratios of ammonium-N and organic N in excreted manure, its fertiliser value and N losses.

Figure 2. Distribution of excreted N over ammonium-N (Nm), easily degradable organic N (Ne) and resistant organic N (Nr) as influenced by feeding strategy (Reijs et al., 2006). HFEC=high fertilization, early cut grass. LFLC+MSIL=low fertilization, late cut grass, supplemented with maize silage.

Reductions in ratios of ammonium-N and organic N in excreted slurry may result in a more than 20% reduction of its short term N fertilizer value and a doubling of the C:N ratio. Such changes in manure characteristics may have substantial effects on the applicability of manure as a fertilizer and the development of soil fertility in the long term. Feeding strategy not only affects the fertilizer value of manure. There are clear indications that the feeding strategy is also an important determinant of both type and amount of gaseous N losses from soils after slurry application. The quantity and the quality of C appears to be relevant and particularly the availability of water soluble C (volatile fatty acids) seems crucial in this respect.

Current results indicate that nutritional measures are extremely powerful to reduce N excretion by animals and ammonia emission rates from manure. Manipulation of the diet of animals may also have a profound impact on the fertiliser value of manure and seems a powerful tool to control (gaseous) nutrient losses from soils. It seems

worthwhile to further investigate these effects and to include them in an analysis of the consequences of nutritional strategies for N efficiency and performance of the whole farming system.

N losses during manure storage

Besides the animal subsystem, an important subsystem of farming is that of manure handling and storage. At the moment, fixed emission factors are being used in surveys of the effect of animal husbandry on ammonia and greenhouse gas emission. However, there may be several measures that have a potential for mitigating these emissions.

Tests with farmyard manure (FYM) of beef cattle indicated that the combination of compaction and covering reduces microbial activity in manure and emissions of ammonia and nitrous oxide. Separate case studies with beef cattle FYM and pig FYM both indicated the effect of covering on ammonia emissions (reductions from 8 to 3% and from 16 to 3% of available N, respectively), whereas the effect on nitrous oxide emission remained marginal (3% of N). Differences in figures for separate case studies or duration of FYM storage relate to the differences in FYM composition (pigs versus cattle) and storage conditions. Also, a large proportion of more than 20% of N in FYM was unaccounted for which may be associated with the loss of di-nitrogen. Similar findings have been reported in literature, but the size and the source of the unaccounted N are not conclusive (see Figure 3).

Processes

		Measurements
Ammonia volatilisation	physico chemical	NH_3
Nitrification	biological	N_2O
Denitrification	biological (chemo)	N_2O.....N_2
Mineralisation	biological	-
Immobilisation	biological	-
Leaching	physical (biological)	NH_4^+, NO_3^-, organic N

Figure 3. Processes acting in solid manure heaps (Source: Chadwick et al., 2006).

During composting of FYM, or the solid fraction of cattle slurry, the ammonium concentration rises with microbial activity and temperature during the thermophilic phase of composting. After this phase, the concentration of ammonia drops whereas that of nitrate rises. It seems that dry matter content of compost and the addition of straw both result in lower concentrations of ammonium during the thermophilic phase. During composting substantial changes occur in manure organic matter, e.g. with the C:N ratio decreasing from about 35 to about 15. This means that C emission proceeds at a much higher rate than that of N.

Manure characteristics (composition) as well as specific measures concerning the storage and handling of manure (compaction, duration of storage, covering, turning) have important effects on the type and the amount of gaseous N emissions and on manure quality. More insight in the effectiveness of measures and the implications of manure characteristics seems necessary to be able to control emissions and the effective application of stored or composted manure.

Fertiliser value of manure

Manure is not an easy to use source of N. Extensive research confirms that the N fertiliser value (NFV) of manure ('N equivalency') depends on numerous factors such as the manure composition, the rate, timing and method of its application, the crop type, the field history and soil and weather conditions. Hence, not surprisingly, researchers and extension services across Europe have different opinions about the N equivalency of manure, even when controlling factors seem more or less equal. Consequently, recommendations on the necessary mineral supplements vary strongly as well.

Organic fertilisers have a residual nitrogen effect after the year of their application as the decomposition of organic material usually takes more than a year. When organic fertilisers are used repeatedly, residual effects accumulate and increase the availability of N (Figure 4). In practice and as part of a risk-avoiding strategy, estimates of the N contribution of organic fertilisers are generally conservative, as they are based only on the N available in the first season following application, or on short-term field trials at best (Schröder and Stevens, 2004). Better estimates of the NFV of organic fertilisers, taking into account long term effects, may result in reduced mineral N inputs without compromising yields.

Proper definitions are a prerequisite for the assessment of the NFV. NFV in terms of N equivalents can only be properly assessed in experiments including mineral fertiliser treatments next to manured treatments. Response curves, constructed from data for the mineral N fertiliser treatments, provide the basis for the assessment of the NFV. From the perspective of reducing unnecessary N inputs, the relative NFV (RNFV) of manure is ideally calculated as the ratio of the recovery of manure-N and the recovery of mineral fertiliser-N. Recoveries can be determined by the difference method and by isotope dilution techniques, each method with its own artefacts (for which see Schröder and Stevens, 2004).

Figure 4. Modelled accumulation of residual N effects (solid line) of ten consecutive annual manure applications of 100 kg organic N per ha and their extinction when manuring is interrupted (dotted lines) (Schröder, 2005).

Sørensen (2006) presented the results of a field experiment in which the short term (2-3 years) NFV of cattle slurry in barley was assessed, using isotope dilution techniques. The experiment revealed that a significant proportion of ammonium-N in cattle slurry is immobilised within a few weeks after application, due to the presence of easily decomposable organic matter in the manure. Moreover, the remobilisation rate of this N was slow: even after three years, still about 40-45% of the initially (subsurface-)applied ammonium-N was present in the soil in organic form. For mineral fertiliser-N (applied as $(NH_4)_2SO_4$), this proportion was much lower, i.e. only 24%. The higher proportion of slurry-derived ammonium-N that is immobilised caused its RNFV in the first crop to be of the order of 70-80%. Nevertheless, the RNFV on the basis of total-N in the slurry (ca. 55%) was about as if the RNFV of the ammonium-fraction had been 1. This was due to mineralisation of the organic N applied in the slurry during the growing season. The residual RNFV of cattle slurry in the first year after the year of application was equivalent to only about 3% of total-N applied.

Schröder *et al.* (2005a) used the results of long-term experiments with maize to calibrate and validate a soil-crop model, and subsequently quantified short-term and long-term RNFVs of cattle slurry. Model calculations indicated that the RNFV of cattle slurry increases from 55-60% when manure is first applied to approximately 80% after 6 and 8 years, using annual relative decomposition rates (RDRs) of organic N in cattle slurry of 33 and 25%, respectively. The substantial difference between short-term and long-term RNFV shows the importance of taking into account residual effects after repeated slurry applications. The existing literature on RDRs of manures, however, is anything but consistent, even for similar manure types. While this may be true, the lack of consistency does not question the difference between short-term and long-term RNFV of manure. RDRs do, however, affect the number of years needed to attain equilibrium between annual inputs of organic N and annual decomposition of organic N. For example, using a RDR of 10% in the model calculations, instead of 33 and 25%, would have resulted in a long-term RNFV of 80% after 20 years, instead of 6 and 8 years.
The plea to take better account of long-term RNFVs of manures, if applicable, has yet another implication. This relates to the implicit incorporation of a certain soil N supply in N recommendations. When N recommendations for a crop are predominantly based on trials that were carried out on sites to which manures have been amply applied before the experiments started, the true requirements of that crop may be underestimated, and vice versa. This aspect of residual effects deserves attention as well.

Conclusions

There is no perfect model or best method to predict or study the consequences of farming practices on N efficiencies in subsystems or the farming system as a whole. On the one hand, detailed approaches are being followed to study the effects of single aspects of farm management, such as strategies for feeding livestock, handling and storage of manure, application of manure, optimisation of animal manure use next to artificial fertiliser, and crop harvest. On the other hand, more general analyses at the level of whole farming systems are needed to understand how the different subsystems interact and contribute to the whole farm outcome. Awareness of these interactions is extremely relevant as measures are not necessarily propagated throughout all consecutive subsystems and may sometimes even be countered, so that the overall effect at the whole farm level is smaller than suggested by the effect at the subsystem level. Schröder *et al.* (2005b) explored these kinds of interactions with a simple whole farm model.

Applying a combination of both detailed approaches at the scale of subsystems and more general approaches at the farm scale is probably the best way of ensuring that results can be related to the diversity in farm management encountered in practice. Detailed approaches need to be scaled up to the level of whole farming systems to be able to indicate the implications for whole farm management. More general approaches at the level of the whole farm need to be diversified to be able to predict accurately the consequences that must be expected for a specific farmer

or under a specific set of farming conditions. Depending on the specific aim of the study, a choice may be made for either a more general or a more detailed approach.

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Poster presentations

Evaluation of milk nitrogen efficiency in commercial dairy farms from the Basque Country

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Abstract

Basque commercial dairy farms have intensified dairy production lately, arising environmental concern by excessive slurry accumulation in some regions. Improving the efficiency of feed N utilization is considered the most effective means to reduce N losses from dairy farms. The present study was designed to evaluate the N efficiency on 44 commercial dairy farms.

A confidential survey, which included information on dairy herd ration and milk production for the sampling period, was conducted. N intake was estimated using the ration information together with collected samples of each ingredient analyzed for DM and CP. Milk N was estimated considering the mean milk yield and the mean milk protein content of the tank. Average N efficiency by lactating cows in sampled farms was 23.9% (SD=4.4). Rations close to Broderick´s (2003) recommendations on N ingestion (591g/day) achieved the highest milk N efficiencies (31.7%) mainly due to increasing milk yields (34.9 litres/day). Besides, results show that dairy cows with a high milk production excreted less N to the environment per litre milk produced. We conclude that feeding protein closer to recommendations and increasing production, improve the efficiency of feed N utilization and reduce N losses from dairy farms.

Keywords: dairy farming, losses, milk yield, nitrogen

Background and objectives

Dairy farming is the second most important economic activity in the primary sector of the Basque Country (Eustat, 2005), having undergone important transformations for the last years. Farms have been hardly intensified and thus most dairy production systems are actually characterized by increasing dairy cattle sizes and milk yields and by being based largely on imported feeds. Concerns about excessive nutrient accumulation on intensive farms stem from farm imports of elemental nutrients in purchased feeds. In this sense, nutrient losses to ground and surface waters from manure can affect water quality (Van Horn *et al.* 1996). Manure is considered one of the main sources of reactive N in the environment (Spears *et al.*, 2003), as more than 70% of the N consumed is excreted in feces and urine (Tamminga and Verstegen, 1996). N efficiency is currently calculated as the ratio of protein N in milk produced and total N consumed. Jonker *et al.* (2002) assured that improving the efficiency of feed N utilization by dairy cattle is the most effective means to reduce nutrient losses from dairy farms. In this sense, feed N utilization efficiency could be enhanced by feeding protein closer to recommendations and increasing milk yield using strategies such as three-times daily milking, use of BST or photoperiod manipulation.

The present study was designed to evaluate the N efficiency in a selection of commercial farms in the Basque Country and to evaluate the opportunities to decrease N losses.

Materials and methods

A confidential survey was conducted from March to November 2003 on 44 commercial dairy farms in the Basque Country. These farms had been previously selected according to the advisory centres located in each region with the aim of covering a wide range of milk production (5,591 to 11,800 L cow^{-1} $year^{-1}$) and representing the different feeding systems used for dairy cow nutrition (TMR, commercial blends, rations based on grass silage and concentrate).The survey included information on dairy herd characteristics, ration and herd mean milk production with regard to the period of sampling. Herd daily N intake was estimated using the ration information together with collected samples of each ingredient, which was analyzed for dry matter (DM) and crude protein (CP). Milk samples were taken from the tank and analyzed for fat and protein. Milk N was estimated considering the herd mean milk yield and its protein content. Fecal and urinary samples representing at least 5% of sampled herd were collected and every sample was analyzed for N. The Cornell Net Carbohydrate and Protein System for Dairy Cattle 5.0 model was used to estimate daily fecal and urinary excretion volumes.

Results and discussion

Observed mean DM and N intake were 22.8 kg/day (SD = 1.9) and 589 g/day (SD = 69.9) per head, respectively. Mean milk yield was 28 litres/day (SD = 4.8) while milk protein content was 3.2% (SD = 0.1). The average milk yield for these sampled farms was higher than the average milk production in the Basque Country (23.4 litres/day, FEPLAC 2003). Average efficiency of feed N utilization for milk production by lactating cows in sampled farms was 23.9%. However, N efficiency ranged from 15.9 to 31.7% and according to Jonker *et al.* (2002), higher milk yields improved milk N efficiency (Figure 1.).

Figure 1. Herd milk yield (litre/day) and N efficiency (%).

Similar milk yields presented a great variability on N efficiency values and this fact might point at an inefficient protein feeding management in some farms. Feeding systems did not show any significant influence on N efficiency although TMR based feeding systems had a better response on milk yield and milk N. The highest milk N efficiency (31.7%) was achieved by feeding rations close to Broderick´s (2003) recommendations in protein content, with 22.7 kg DM/day and 591 g/day ingested N.

With respect to N losses, urinary and fecal N excretion increased with higher protein intakes. However, when N excretion was related to milk yields, higher yielding cows excreted less N to the environment per litre milk produced (Figure 2). According to Kuipers *et al.* (1999) these results suggest that in a European milk quota system a better environmental N efficiency could be achieved improving mean milk yields which could minimize the N emissions associated with manure.

Figure 2. Excreted N/Milk yield (g/l) and Milk yield (l/day).

Conclusions

Results of this study confirm that feeding protein closer to recommendations and increasing cattle milk yield per head, improve the efficiency of feed N utilization and reduce N losses from dairy farms. Thus, dairy farmers could concomitantly improve the whole-farm N balance and their income.

Acknowledgements

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Nitrogen flows in a grazing dairy system in Galicia (north-western Spain)

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Abstract

In the northwest area of Spain (Galicia), dairy industry provides in 30 % of agricultural income. To develop economically and environmentally sustainable farming practices it is necessary to monitor nutrient flows (input/output balances). The objective of the study was monitoring nitrogen (N) fluxes of a dairy herd and those of the associated fields.

In a grazing system where inputs of concentrates were increased to improve milk production farm gate balance calculation for N resulted in a low surplus. However, soil mineral N contents at several dates during winter indicated high nitrate losses by leaching in the areas that were re-sown in autumn. Consequently, when N leaching has to be estimated, it is necessary to include information about N fluxes at field level, to complete the information given by farm gate balance.

Keywords: dairy farming, leaching, nitrogen, nutrient balance

Background and objectives

The dairy industry represents 30 % of the total agricultural income in Galicia and produces about 2.2 million tonnes of milk per year (Xunta de Galicia, 2003). From economic and environmentally points of view a sustainable future depends on the efficiency of the farming system. Monitoring the nutrient flows (input/output balances) and the storage of nutrients in the soil (soil testing) can be used to find ways to decrease nutrient losses to the atmosphere and to the ground water (Van Beek *et al.*, 2003).

In October 2003 the ´Green Dairy´ project started in several countries in the Atlantic area. One of the aims was to monitor nutrient flows on nine experimental dairy stations. This paper presents part of the results obtained for 2004 at the Agricultural Research Centre of Mabegondo (north-western of Spain). The main objectives were to determine N fluxes at herd level, to calculate N efficiency in milk production, to estimate farm gate surplus for N, and to investigate the risk of nitrate leaching in the paddocks associated to the herd management.

Materials and methods

The study was carried out in the area used for dairy herd at the Agricultural Research Centre of Mabegondo located about 97 m above sea level with annual average temperature of 12.7 ºC and annual average rainfall of 1128 mm. Soil texture was silty loam. Three groups (A, B and C, Table 1) of lactating Holstein Friesian cows were formed. Main difference between the groups was the amount of feed additional to the intake of grass and white clover by grazing. During the year the herd management had two grazing periods (in autumn and spring); mild winters made it possible to extend grazing season in late autumn and early spring. Dry matter intake during grazing was determined by pre and post grazing sampling. Grass samples, forages and concentrates were analyzed for crude protein contents and daily milk yields per cow were recorded and analysed for crude protein.

Soil mineral N (SMN, NH_4^+- N + NO_3^-- N) was determined in the 0-10, 10-30, 30-60 and 60-90 cm layers of the soil profile in nine grassland fields, which represented in management and fertilization the variability found in total area. The first soil sample was taken before the beginning of the drainage period (27 September), the subsequent ones during winter (30 November, 2 February), and the last one at the end of rainy period (5 May).

Table 1. Dairy herd in Mabegondo experimental farm (January 2004- December 2004).

Herd	A	B	C	Total	LSU 12 month
Lactating cows	28	22	34	84	84
Rest of cows				55	55
Heifers 1-2 yr				11	7.7
Heifers <1 yr				25	10
TOTAL				175	156.7

Results and discussion

Table 2. Feeding regimes and milk yield for the groups of lactating cows in 2004.

	A	B	C
FEED			
Concentrates, kg DM cow^{-1}	882	1420	2733
Grass silage, TDM cow^{-1}	0.9	1.5	0
Maize silage, TDM cow^{-1}	0.7	0	1.5
MILK			
Yield, L cow^{-1}	6621	7154	8549
Protein %	3.1	3.3	3.1
Nitrogen efficiency, %	26.3	27.2	32.1

Table 2 shows the diet composition, milk yield and average protein content for the three groups in 2004. N efficiency was calculated as the ratio of milk-N (based on the protein level) and the feed-N (as protein content of the ration). There was an increase of N efficiency in milk production from 26.3 to 32.1 % when milk yield increased from 6621 L cow^{-1} in the group A to 8549 L cow^{-1} in group C, mainly as a result of an increased intake of concentrates. Farm gate balance for N taking into account concentrates, manure and fertilizer as inputs and the sales of milk and meet as outputs resulted in a surplus of only 49 kg N ha^{-1}.

Table 3. Field management (G = grazing, S = cutting for silage), fertilization, area, SMN and nitrate residues at the first soil sampling date.

Field	Management/N source	Ha	SMN kg.N ha^{-1}	NO_3^-N kg.N ha^{-1}
1	G / Mineral N		19.4	10.0
2	G / Mineral N		15.1	8.7
3	G / Mineral N	36.3	18.9	11.8
4	G or G+S /Mineral N+ Organic N	10.6	9.2	2.1
5	G or G+S/Mineral N	21.9	14.7	5.0
6	newly sowm grassland	3.8	43.4	33.7
7	newly sown grassland	4.0	68.3	40.9
8	newly sown grassland	2.1	244.9	234.7
9	newly sown grassland	5.5	433.7	415.3

Table 3 shows the management (grazing or grazing and silage production, the type of N fertilization), SMN and nitrate contents at the first soil sampling date for each of nine representative field. Rainfall from September to April accounted 669 mm and the beginning of drainage was estimated between 4 and 15 October when soil water content reached field capacity. In fields number 1, 2, 3, 4 and 5 SMN was lower than 20 kg N ha^{-1} and nitrate represented about half of the SMN contents. In the other fields a high percentage of SMN content was found in the form of nitrate, especially in those fields that had been ploughed in spring and were sown in September (fields 8 and 9). In the case of fields 6 and 7 the old grassland was ploughed in autumn and immediately the new crop was sown. Such situation explains lower N mineralization than in fields 8 and 9, and consequently lower SMN residues in the soil in September. Taking into account the results of successive sample dates during the rainy period the displacement of nitrate along the soil layers identified fields 6, 7, 8 and 9 as the most risky regarding nitrate losses to ground water (Figure 1).

Figure 1. Soil mineral N at several dates from September 04 to May 05.

Conclusions

In a grazing system where conversion of N in feed into N in milk was improved, farm gate balance calculations show a low surplus for N. However, measurements of SMN contents at several moments during winter indicate high nitrate losses on farm areas were the grassland is re-sown in autumn. Consequently, when N leaching has to be estimates it is necessary to include knowledge about N fluxes at field level, to complete the information presented by a farm gate balance.

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Fate of N during composting of the solid fraction of dairy cattle slurry

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Abstract

Screw pressed dairy cattle slurry solid fraction (CSSF) was collected from two dairy farms with 30% and 22% dry matter (DM) content and the effects of straw addition and of turning frequency on the fate of N were examined. Thermophilic temperatures were attained soon after separation of CSSF with 30% DM. In contrast, temperatures were much cooler in CSSF with 22% DM without straw. The pH ranged from 8 to 9 and C/N ratio was similar amongst treatments on each sampling occasion. Mineral N production was characterized by high NH_4^+ and low NO_3^- contents during the thermophilic phase followed by a decrease of NH_4^+ and an increase of NO_3^- towards the end of composting. Higher DM and straw addition were both associated with lower NH_4^+ compost concentrations. Agronomically suitable compost can be obtained with CSSF, but future research is required to minimize N loss as NH_3 gas given that pH was alkaline during the thermophilic phase of composting.

Key-words: Ammonium-N, compost, C/N ratio, nitrate-N, pH

Background and objectives

Manure surpluses in dairy farms can be reduced by separation of the solid fraction from slurry (Ford and Fleming, 2002). This fraction can be exported to other farms with a high demand for organic amendments after composting to produce a uniform stabilized product that can be land applied and marketed as a soil amendment, which is easy to handle and has little or no odour, and is sanitised to destroy pathogens and weed seeds. Frequent turning may ensure positive N fertilizer values of dairy waste composts (Shi *et al.*, 1999) but may increase NH_3 emissions and reduce the agronomic value of the final product (Hao and Chang, 2001). However, N loss during composting may be controlled by increasing the C/N ratio to enhance N immobilization, and by lowering compost pH (Raviv *et al.*, 2004). At a low pH, the balance between NH_4^+ and NH_3 is shifted towards the NH_4^+, thus reducing NH_3 volatilization during the thermophilic stage of composting. Here, we studied the fate of N during composting of CSSF with straw and with different turning frequencies.

Materials and methods

Screw pressed cattle slurry solid fraction (CSSF) from two dairy farms located in NW Portugal with 30% and 22% dry matter (DM) content was collected during winter 2004. The CSSF1 with 30% DM was collected at a rate of 1 m^3 h^{-1} and the material with 22% DM (CSSF2) was collected at the rate of 4 m^3 h^{-1}. Oat straw (cut in 5 cm length) was added to the CSSF1 at the rate of 11% (w/w) DM. Piles of 15 m^3 were constructed and sampled (n=5) over 15 weeks for chemical analysis. Over this period, piles were turned with a tractor-mounted front-end loader at a frequency of 5 and 10 times at the first farm and 4 and 8 times at the second farm. Compost DM, pH, electrical conductivity, organic matter, and Kjeldahl N were determined by standard procedures (CEN, 1999) and compost mineral N was analysed with a molecular absorption spectrophotometer, after extraction with 2 M KCl. Compost temperature was monitored automatically with a thermistor positioned in the centre of each pile (Delta-T Devices).

Results and discussion

Thermophilic temperatures were attained soon after separation of CSSF1 with 30% DM (Figure 1), particularly when mixed with straw. In contrast, temperatures were much cooler in CSSF2 with 22% DM without straw. Moreover, solids with initial 30% DM had a faster rate of composting and took shorter time to reach maturity than solids with initial 22% DM. The pH was alkaline and ranged from 8 to 9 during the monitoring period (Table 1). The C/N ratio was similar amongst treatments on each sampling occasion and decreased from over 40 initially to a value of 14 towards the end of composting period (Table 1) indicating an advanced degree of stabilization by Zucconi and de Bertoldi (1987).

Mineral N production was characterized by high NH_4^+ and low NO_3^- contents during the thermophilic phase followed by a decrease of NH_4^+ and an increase of NO_3^- towards the end of composting (Figure 2). Higher DM and straw addition were both associated with lower NH_4^+ compost concentrations.

Table 1. pH and C/N ratio during composting of cattle slurry solid fraction with initial 30% DM, without (CS1) and with straw (CS1S) and with initial 22% DM without straw (CS2), ($\bar{x} \pm SE$, n=5).

	CS1		CS1S		CS2		CS1		CS1S		CS2	
Turnings	5	10	5	10	4	8	5	10	5	10	4	8
Time (d)	pH						C/N					
7	9,2±0,1	9,1±0,1	8,7±0,4	8,8±0,2	8,7±0,1	8,5±0,1	34±2	34±3	36±2	36±3	37±1	37±2
14	9,1±0,1	9,0±0,1	8,8±0,3	8,8±0,3	8,2±0,6	8,6±0,1	34±2	36±2	30±1	35±1	38±1	31±1
28	9,0±0,2	8,7±0,1	8,8±0,4	9,0±0,2	8,6±0,1	8,7±0,1	26±2	21±2	27±1	26±1	25±6	27±1
42	9,2±0,1	9,0±0,1	8,9±0,1	8,9±0,1	9,1±0,1	8,9±0,1	27±6	18±1	17±4	21±5	19±2	19±4
63	8,5±0,2	8,5±0,3	8,9±0,1	8,6±0,1	9,0±0,1	8,8±0,2	20±4	15±1	16±1	15±1	20±2	17±1
91	8,2±0,1	7,9±0,1	8,2±0,3	7,9±0,2	9,2±0,1	9,1±0,1	14±1	15±1	14±1	15±1	17±4	17±1
105	8,0±0,1	7,8±0,1	7,6±0,2	7,6±0,1	8,8±0,1	9,0±0,1	14±1	15±1	13±1	14±1	14±1	16±1

Figure 1. Evolution of the temperature during composting of cattle slurry solid fraction with initial 30% DM, without (CS1) and with straw (CS1S) and with initial 22% DM without straw (CS2). T indicates turning times.

Figure 2. Evolution of ammonium and nitrate N (mg kg-1) during composting of CSSF with initial 30% DM, without (CS1) and with straw (CS1S) and with initial 22% DM without straw (CS2). T indicates turning times.

Since higher DM content was important to increase compost temperature, to ensure effective destruction of pathogens and viable weed seeds, reducing the screw press yield rate is recommended. This also had the benefit of reducing ammonia gas production. Although chemical evolution based on parameters such as pH, and C/N ratio provided substantial evidence that agronomically suitable compost can be obtained, optimizing the process to minimize the loss of nitrogen in the form of ammonia gas is needed given that pH ranged between 8 and 9 throughout the composting process.

Conclusions

In contrast to CSSF with 30% DM temperatures were much cooler in CSSF with 22% DM without straw, and higher DM and straw addition were both associated with decreased NH_4^+ compost concentration. Therefore, increasing DM by slowing the rate of the screw dewatering mechanism and the addition of straw reduced N losses and improved composting efficiency as indicated by higher pile temperatures. Future research includes composting to minimize N loss as NH_3 given that pH was alkaline during the thermophilic phase of composting.

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Do nitrogen balances for solid farmyard manure suggest di-nitrogen (N_2) emission?

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Abstract

Solid manure is stored in heaps for varying periods of time before spreading on the land. During storage nitrogen (N) is lost to the environment via leaching (organic N, NH_4, NO_3), ammonia (NH_3) volatilisation and nitrous oxide (N_2O) emissions. Where studies have measured these losses as well as the N content of the manures heaps before and after storage, it has been possible to construct N balances. This paper reports on N balances from two studies where pig and beef cattle farmyard manure (FYM) has been stored for up to 6 months. The results demonstrate that not all of the N loss measured by difference can be accounted for by the measured losses of N leached and emissions of N_2O and NH_3. Up to 25% of the N loss cannot be accounted for. Some of this unaccounted for loss could be due to errors associated with obtaining representative sub-samples of manure at the start and end of storage, as well as errors associated with the loss measurements. However, since conditions are favourable for denitrification and emissions of N_2O, some of this unaccounted for N loss could be due to N_2 emissions that were not measured.

Keywords: ammonia, cattle, manure, N_2, N_2O, nitrogen, nutrient balance, pig, storage

Background and objectives

Several studies have been conducted to determine changes in the nitrogen (N) content of solid manure during storage. Where studies have measured N losses by leaching (organic N, NH_4, NO_3), ammonia (NH_3) volatilisation and nitrous oxide (N_2O) emissions, it has been possible to determine N balances, which suggest that additional N is lost that has not been accounted for.

$$\% \text{ N loss unaccounted for} = ((\text{Heap } N_{initial} - \text{Heap } N_{final}) - (\text{N-}NH_3 + \text{N-}N_2O + \text{N-leached})) / \text{Heap } N_{initial} * 100$$

Where, Heap $N_{initial}$ and HeapN_{final} are the mass of N (kg/heap) at the start and end of manure storage, and N-NH_3 + N-N_2O + N-leached are the mass (kg/heap) of N lost by these processes.

In this paper, we present information from two studies on N losses and balances from straw-based pig and cattle farmyard manures during storage, and demonstrate how the N balance is affected by heap management.

Materials and methods

In study 1, replicate (three) solid cattle and pig farmyard manure (FYM) heaps were established (*c.* 4-5 tonnes) on drained concrete bases and either covered with plastic sheeting (SHEET) or left open to the elements (OPEN). Representative manure samples were taken at the start and at the end of the storage period, 6 months later. Ten samples (*c.* 0.5 kg each) were taken from throughout each heap as it was established and combined to generate one sample per heap, which was sub-sampled in the laboratory for analysis. During the storage period, NH_3 volatilisation and N leaching losses were measured. A portable emission hood system, based on that used by Chadwick (2005) used fan ventilation to generate airflow over each heap. Ammonia emissions were measured from the heap

by trapping a known proportion of the inlet and outlet air in 0.02M orthophosphoric acid. The acid traps were connected to pumps that drew air through the traps. The pumps were fitted with gas meters to measure the flow rate and total volume of air flowing through the traps. After exposure, the orthophosphoric acid was analysed for ammonium-N colorimetrically (MAFF, 1986) and the amount of ammonium-N collected (inlet subtracted from outlet) was combined with airflow through the polytunnel to calculate the ammonia emission for that period. Leachate from the heaps was collected at the lowest corner of each bunker and piped under gravity into collection tanks. When leachate was produced the volume collected was measured at weekly intervals and samples were taken from the collected bulk of leachate every 50-mm of rainfall. Leachate samples were analysed for total-N, nitrate-N and ammonium-N (MAFF, 1986).

In study 2 (Chadwick, 2005), which compared N balances between conventionally stored (OPEN) and compacted, covered (COMP/SHEET) cattle FYM heaps, N_2O emissions were also measured. The portable emission hood system was also used to quantify the N_2O emission from the heap. Air samples were taken from the inlet and outlet air flows and analysed by gas chromatography. The concentration data was combined with data for the airflow through the emission hoods to generate fluxes. Measurements were made from each of the heaps over a 1-2 h period and at appropriate intervals over the storage period. N loss data and differences between the initial and final N content of the manure heaps were used to construct N balances and to determine the quantity of N loss not accounted for.

Results and discussion

In study 1, losses from the cattle FYM heaps via NH_3 volatilisation accounted for 3% and 8% of the initial heap N content for the SHEET and OPEN treatments, respectively. For the pig FYM, the equivalent values were 3% and 16% (Table 1). The greater % N lost as NH_3 from the OPEN pig FYM heaps compared to the OPEN cattle FYM heaps was probably related to the higher total N and ammonium N content, being 8.5 kg/t and 2.3 kg/t respectively for the pig FYM and 4.6 kg/t and 1.3 kg/t for the cattle FYM. Leachate losses accounted for between 1 and 4% of initial heap N contents for both manure types and storage treatments. Losses of N_2O were not measured, but the maximum N_2O emission from cattle FYM heaps measured in study 2 represented <3% of initial heap N contents.

Table 1. N losses from pig FYM heaps stored for 6 months in study 1. N=3. Values in parentheses represent 1 standard error of the mean.

Storage treatment	Initial N (kg/heap)	Final N (kg/heap)	N-NH_3 (kg/heap)	N-N_2O (kg/heap)	N-leached (kg/heap)	N loss unaccounted for (%)
OPEN	40.9 (4.8)	21.6 (1.9)	6.7 (0.6)	1.2†	1.1 (0.2)	25
SHEET	35.0 (7.2)	24.3 (6.0)	1.0 (0.1)	1.0†	1.3 (0.4)	21

† *N_2O was not measured. Estimate based on measurements made by Chadwick (2005).*

In study 1, total N losses from the OPEN pig FYM heaps were 47%, and 30% from the SHEET heaps. Total N losses from the cattle heaps were smaller at 18% and 3% for the OPEN and SHEET storage treatments, respectively. The total N losses from the cattle manure heaps were lower than the 30% loss measured from OPEN cattle FYM treatment in study 2 (Table 2). In study 1, the % of initial heap N that was unaccounted for (assuming 3% loss of N via N_2O) ranged from 21-25% for the pig FYM heaps, depending on the storage treatment. However, in study 1, there were only low levels (<5%) of unaccounted for N losses from the OPEN cattle FYM heaps. Yet in study 2, *c.* 30% of the initial N from cattle FYM heaps was lost and not accounted for. The reasons for this difference are not yet clear.

Table 2. N losses from cattle FYM heaps stored for 6 months from study 2 (Chadwick, 2005). N=3. Values in parentheses represent 1 standard error of the mean.

	Storage treatment	Initial N (kg/heap)	Final N (kg/heap)	N-NH_3 (kg/heap)	N-N_2O (kg/heap)	N-leached (kg/heap)	N loss unaccounted for (%)
Period 1	OPEN	25.5 (1.3)	16.5 (1.1)	1.2 (0.2)	0.6 (0.0)	0.1	28
	COMP/SHEET	27.4 (2.0)	16.0 (1.5)	0.1 (0.0)	0.2 (0.0)	0.2	40
Period 2	OPEN	42.9 (0.3)	30.3 (1.9)	0.1 (0.1)	0.0 (0.0)	nd	29
	COMP/SHEET	42.2 (7.2)	34.6 (1.6)	0.0 (0.0)	0.8 (0.2)	nd	16

nd = not determined (excessive rainfall overloaded the sampling equipment such that accurate assessment of volumes, and hence mass of leached N, was not possible).

This 'not accounted' for N could partially be explained by errors associated with obtaining representative FYM samples and also with errors associated with the sampling of each of the N loss pathways. The reliability of sub sampling solid manure may be checked by calculating the phosphorus (P) balance since P is not volatilised and little is leached. The total P content of the manure heaps is not available for study 1. However, in study 2 (Chadwick, 2005), the percentage recoveries of initial heap P for the OPEN treatment were 88% and 97% for the two storage periods shown in Table 2. Hence, reliable balances were made from those two storage periods which showed *c.* 30% of the N loss was not accounted for.

Conclusions

Unaccounted for losses of N from manure heaps may be associated with errors in obtaining representative manure samples at the beginning and end of the storage period as well as errors associated with the various loss measurements. However, as solid manure heaps are sources of N_2O, they also provide conditions favourable for denitrification, so some of this lost N that could not be accounted for may have been emitted as di-nitrogen (N_2). Further research is required to better quantify rates of N transformations within solid manure heaps, including the measurements of N_2 emissions. This could be achieved by adapting the acetylene inhibition technique (Ryden *et al.*, 1987) or by using a direct measurement technique such as that described by Cardenas *et al.* (2003).

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Nitrate leaching under a soil treated with urea-DMPP

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Abstract

Nitrification inhibitors control nitrate leaching by prolonging the time soil nitrogen (N) remains in its ammonia form. 3,4-Dimethyl-pirazole phosphate (DMPP) only inhibits the first step of nitrification. The aim of this work was to evaluate the effects of applying DMPP added to urea (U) in a 1% ratio (w/w) at two N doses (U and U+) on soil N and groundwater nitrate leaching, and on the yield of an irrigated corn crop grown in Mediterranean conditions.
The inhibitory effect of DMPP on the nitrification process led to a build up in NH_4^+ over 50 days. In contrast, NO_3^- levels increased significantly within the first 15-30 days after fertilization in the treatments lacking DMPP, while treatments including DMPP showed similar NO_3^- concentrations to unfertilised control plots. The use of DMPP reduced nitrate leaching: 76, 59, 44, 31 and 17 kg N ha^{-1} were lost for the treatments U+, DMPP-U+, U, DMPP-U and C, respectively. Although higher values of dry matter, grain yield and plant N uptake were observed for the urea + DMPP treatments compared to the urea alone treatments, significant differences (P<0.05) were detected only between the unfertilised control treatment and the remaining treatments.

Keywords: DMPP, inhibitor, leaching, nitrate, nitrification

Background and objectives

The goal of a nitrification inhibitor is to fix N in the clay-humic complex and thus maintain soil N in the ammonia form for a longer time, increasing the efficiency of applied nitrogen (N). A new nitrification inhibitor developed by BASF (3,4-dimethlpirazole phosphate, DMPP) inhibits only the first step of nitrification and, in doing so, diminishes the activity of *Nitrosomonas* bacteria in the soil over a period of time. DMPP is highly specific at inhibiting nitrification for several weeks when applied at a ratio as low as 0.8% (w/w) with the ammonium or ureic N in a fertilizer or slurry. The action of DMPP is linearly related to thermal time. Besides ammonium nutrition, plants can use other pathways to build up their biomass. Reduced nitrate levels in vegetables (such as spinach and lettuce) are further benefits of the addition of DMPP to fertilizers. According to Trenkel (1997), the benefits of nitrification inhibitors are limited to coarse-texture soils and situations in which excessive water leads to heavy N leaching. The aim of this work was to evaluate the effects of applying DMPP added to urea in a 1% ratio (w/w) at two N doses (optimal and excessive), on soil and groundwater nitrate leaching and on the yield of an irrigated corn crop grown in Mediterranean conditions.

Materials and methods

Experimental plots were established at La Poveda Field Station in Arganda del Rey (Madrid) (40º 19'N, 3º 19'W) in the middle of the Jarama river basin. The *Typic Xerofluvent* soil (Soil Survey Staff 1998) is a sandy-loam that becomes progressively sandier at increasing depth and has a gravel layer at 1.5-2.2 m. Soil samples were taken for pH (8.1), organic matter (14.0 g kg^{-1}) and carbonate determinations (34.0 g kg^{-1}) (AFNOR 1987). N (EUF-N both fractions 8.37 mg N $100g^{-1}$), P (EUF-P 20ºC 1.48 mg P $100g^{-1}$), K (EUF-K 20ºC 12.25 mg K $100g^{-1}$) were estimated using the electroultrafiltration (EUF) technique (Nemeth, 1979). Total N was determined in EUF extracts (EUF-N) of soil samples by digestion with UV radiation and subsequent oxidation with potassium persulphate in alkaline medium. Phosphorus was also determined colorimetrically using ammonium molybdate as a reagent (AOAC 1990). Potassium and Ca were determined by flame emission photometry. Soil bulk density was 1.47 t m^{-3}. The position of the water

table was 4-4.5 m below the soil surface. Average rainfall for this area is 460 mm yr^{-1}. Four treatments were applied to triplicate plots (9.9 x 11.1m): an optimal rate of urea (U) and the same dose plus DMPP (U-DMPP), an optimal N rate of urea plus 40 kg N ha^{-1} (U+), the U+ dose plus DMPP (U-DMPP+), and a control with no N fertiliser (C). Based on EUF soil analysis and the criteria established by Sánchez *et al.* (1998), optimal N rates for corn in these conditions are 220 kg N ha^{-1}. Corn *cv Tector 700* was sown at the end of April at a plant density of 90.000 plants ha^{-1}. An overhead mobile-line sprinkler system was used to irrigate the crops. Four of the fifteen plots corresponding to different treatments were fitted with EnviroSCAN semi-permanent, multisensor, capacitance probes at a 150 cm depth for monitoring soil water-content in real time. The sensors inside the probes were positioned at depths of 10, 40, 70, 120 and 150 cm to determine drainage (151 mm during the crop cycle) and evapotranspiration (618 mm). Mean quality components of the irrigation water were: NO_3^-, 5.1 ± 0.5 mg N L^{-1}; Na, 90 ± 16 mg L^{-1}. Two ceramic cups were used in every plot to obtain soil solution samples at a depth of 1.4 m (Díez *et al.* 2000). During the experimental period, water sampling was performed 12 times to determine NO_3^- concentration. A vacuum of –80 kPa was applied to the tubes and maintained for a period of 7 to 10 days. After this period, water samples were extracted using air pressure, and NO_3^-, Na^+ concentration, and EC were determined. Any water reaching a depth of 1.4 m near the gravel layer was considered to have leached to the groundwater (average depth of 4 m) because of the high hydraulic conductivity (Smith *et al.* 1991). During drainage periods and during the crop cycle, NO_3^- leaching was calculated weekly by multiplying its weekly drainage by the corresponding NO_3^- concentration at 1.4 m at each sampling event (Díez *et al.* 1997).

Corn plants were harvested from the central five meters of the rows of each plot and aboveground biomass determined. Ten of the harvested plants were randomly selected to separately weigh the different parts of the plants (stalks, leaves, bracts, cob and grain). Weights were obtained before and after oven-drying for 24 h at 60ºC followed by a further 2 h at 80 ºC to determine dry matter (DM). The harvest index (HI) was calculated as grain weight over aboveground biomass (percentage). Grain yield kg ha^{-1} was calculated and plant N contents determined (Díez *et al.*, 2000).

Results and discussion

The application of DMPP increased the NH_4^+ content of the soil; the inhibitory effect of DMPP on the nitrification process leading to the build up of NH_4^+ over a 50-day period. In contrast, NO_3^- levels increased significantly within the first 15-30 days of fertilization in the DMPP-free treatments, while treatments including DMPP gave rise to similar NO_3^- concentrations to the control unfertilised plots.

Groundwater contamination was evaluated using nitrate concentration data for the soil solution at 1.4 m. Cumulative NO_3^- discharge at 1.4 m was found to depend mainly on the irrigation water applied and the fertilizer treatment. As observed by Díez *et al.* (2000), total leaching was mainly related to drainage and less to variations in NO_3^- concentration at the percolation depth. The use of N-containing fertilizers plus DMPP reduced nitrate leaching. Significant differences in the levels of NO_3^- leached were observed among treatments, with losses of 76(d), 59(cd), 44(bc), 31(ab) and 17(a) kg N ha^{-1} recorded for the treatments U+, UDMPP+, U, UDMPP and C, respectively (different letters in brackets indicate significant differences between treatments (Duncan Test) (Figure 1). Our findings suggest a beneficial effect in decreasing nitrate pollution. The treatments including DMPP (DMPP-U and DMPP-U+) were able to preserve higher N levels in the soil, especially in the soil volume occupied by roots, for longer than the urea (U and U+) treatments, and also reduced N losses through NO_3^- leaching.

There was no response to the surplus of N since the optimised doses used adequately satisfied the nutritional needs of corn. Some authors (Pasda *et al.* 2001), nevertheless, have observed increased crop yields in several agricultural and horticultural crops, corn included. Good results were obtained in terms of dry matter and grain yield with the fertilizer applied as topdress. Although higher values of dry matter, grain yield and plant N uptake were observed in the UDMPP and UDMPP+ plots compared to the U and U+ plots, respectively, significant differences ($P<0.05$) only emerged between the control unfertilised treatment and the remaining treatments.

Figure 1. Nitrate leaching during a corn crop cycle according to the treatment applied (C, UDMPP, U, UDMPP+ and U+). Values are means of four replicate drainage determinations and five replicate nitrate concentration determinations.

Conclusion

DMPP successfully inhibited the nitrification of urea fertilizer. Consequently, nitrate leaching was reduced and the amount of N available to corn was increased.

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Effects of nitrification inhibitors in cattle slurry on annual ryegrass yield and soil mineral N dynamics

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Abstract

Slurry spreading at sowing of the ryegrass (*Lolium multiflorum Lam*) can induce important N losses by nitrate leaching to groundwater as well as ammonia and nitrous oxide emissions to the atmosphere resulting in groundwater pollution and a low N use efficiency. The use of nitrification inhibitors (NIs) allows maintaining the mineral N in soil as ammonium and consequently preventing nitrate leaching and nitrous oxide emissions. In the present work, the effect of two NIs, DCD (Dicyandiamide) and DMPP (3,4-dimethyl pyrazole phosphate) applied at two rates to cattle slurry, on soil mineral profiles, annual ryegrass yield and N uptake has been studied. It appears that both NIs induced a higher content of $N\text{-}NH_4^+$ in the soil. However, DMPP seems to be more efficient considering in that it leads to higher levels of $N\text{-}NH_4^+$ and that it effects lasted longer in time. Furthermore, the use of NIs generated higher dry matter yields with the highest value achieved when slurry was amended with the higher rate of DMPP. Similar results were obtained for forage N removal.

Keywords: cattle, DCD, DMPP, inhibitor, leaching, manure, nitrate, nitrification

Background and objectives

Animal feeding on intensive dairy farms in the 'Entre-Douro e Minho' (NorthWest Portugal) is based on a double-cropping forage system, maize and annual ryegrass (*Lolium multiflorum Lam*), with high rates of slurry being applied to both crops. In autumn, after slurry spreading at sowing of the ryegrass, important N losses are expected to occur due to nitrate leaching as well as ammonia and nitrous oxide emissions. These losses result in air and groundwater pollution and a low N use efficiency. A solution to prevent nitrate leaching consists in maintaining the soil mineral N as ammonium by using nitrification inhibitors (NIs). DCD (Dicyandiamide) is one of the most common NIs and, more recently, DMPP (3,4-dimethyl pyrazole phosphate), a new chemical compound shows high potential as NI.
The aim of this work was to compare the effect of DCD or DMPP applied at two rates to cattle slurry on soil mineral N profiles, annual ryegrass yield and N uptake and assess the effectiveness of both NIs.

Materials and methods

A field experiment was carried out at Braga in the NW region of Portugal between November 2003 and April 2004. The soil was a deep well-drained sandy loam derived from granite and classified as dystric cambisol. The trial was laid out with a randomised block design with three replicates and six fertilization treatments: T0, a control not fertilized; T1, 50 m^3 of cattle slurry ha^{-1}; T2, 50 m^3 of cattle slurry ha^{-1} + 10 kg DCD ha^{-1}; T3, 50 m^3 of cattle slurry ha^{-1} + 20 kg DCD ha^{-1}; T4, 50 m^3 of cattle slurry ha^{-1} + 4 L of 25% DMPP solution ha^{-1}; T5, 50 m^3 of cattle slurry ha^{-1} + 8 L of 25% DMPP solution ha^{-1}. The rate of cattle slurry applied corresponded to approximately 98 kg total N ha^{-1}. NIs were previously mixed with the slurry in the vacuum tank spreader. Slurry applications were all performed before planting and followed by superficial soil tillage in order to reduce ammonia volatilisation. Ryegrass was sown on November 11 and harvested on April 26. Typical rainfall pattern is shown in Figure 1. The measured parameters were soil $N\text{-}NO_3^-$ and $N\text{-}NH_4^+$ profiles during crop growth period, biomass production and plant N removal.

Figure 1. Accumulated rainfall between soil sampling dates.

Results and discussion

Figure 2 shows the time course of the amount of N-NH_4^+ and N-NO_3^- in the 0-10 cm and 10-30 cm soil layers observed in the different treatments. It appears that the content of N-NH_4^+ in soil was higher when slurry was mixed with DCD or DMPP (treatments T2 to T5), than when slurry was not treated (treatment T1). However, DMPP seems to be more efficient considering the observed higher levels of N-NH_4^+. Furthermore, DMPP effects lasted longer in time, since, 39 days after seeding, treatments with both NIs showed higher -N-NH_4^+ amounts in soil, while, after 70 days, this parameter only differed in DMPP treatments. Similar results were obtained in a field experiment of Wissemeier *et al.* (2001) who concluded that DMPP had a longer longevity and was 15 to 30 times more efficient than DCD.

Figure 2. Time course of the amount of N-NH_4^+ (Am) and N-NO_3^- (Ni) in the 0-10 cm and 10-30 cm soil layers observed on the different treatments.

Considering the rate of NIs used, no significant differences appeared between treatments T2 and T3, nor between treatments T4 and T5. Therefore, it may be concluded that the concentration of the inhibitors, in the ranges used in the present work, had little influence on the amount of $N\text{-}NO_3^-$ and $N\text{-}NH_4^+$ in soils.

As shown in Table 1, dry matter (DM) yields were affected by the addition of the NIs to the slurry. Indeed, the use of both NIs resulted in higher DM yields with the highest value being reached when slurry was amended with the higher rate of DMPP (8.8 ton ha^{-1} with treatment T5 against more or less 7 ton ha^{-1} with treatments T2, T3 and T4). Similar results were obtained for forage N removal.

Table 1. Effect of treatments under study on forage DM yield and forage N removal.

Treatment	DM yield (kg N ha^{-1})	Forage N removal (kg N ha^{-1})
T0 (control)	4,711 a[1)]	40 d
T1 (slurry)	5,162 a	38 d
T2 (slurry + 10 kg DCD ha^{-1})	6,844 b	54 e
T3 (slurry + 25 kg DCD ha^{-1})	7,049 b	53 e
T4 (slurry + 4 L DMPP 25% ha^{-1})	7,292 b	59 e
T5 (slurry + 8L DMPP 25% ha^{-1})	8,798 c	65 f

[1)] *Data followed by the same letters do not differ at $p<0.05$ level, Tukey test.*

Conclusions

This study shows that the addition of DMPP to cattle slurry just before sowing an Italian ryegrass winter crop, induced a delay on the transformation of ammonium to nitrate leading to higher DM yields and possible environmental benefits (reduction of leaching and N_2O emissions).

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Evaluation of the use of nitrification inhibitor DMPP on the risk of nitrate leaching in different crop systems in Spain

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Abstract

Leaching is the main pathway of nitrogen loss in the Iberian Peninsula agriculture. One of the common methods to reduce this loss is the use of nitrification inhibitors (NI), nowadays the most common is 3-4 dimethylpyrazole phosphate (DMPP).

Results of different trials in extensive and intensive crops with the nitrification inhibitor DMPP are reviewed and discussed. Two years of maize (*Zea mays*) trials performed in the Spanish central area showed a significant reduction (15 to 40%) of nitrate leaching when urea + DMPP are used as nitrogen source. In an experiment carried out in broccoli, nitrate content in soil solution at 30 cm depth in the treatment with conventional fertigation was 18% higher than in the treatment with fertigation + DMPP. In a citrus pot trial the use of ammonium sulphate with and without DMPP was compared, applying the same nitrogen dose in both treatments. Results show that DMPP use reduced the nitrogen losses as nitrate with 50%.

Keywords: 3-4 dimethylpyrazole phosphate, DMPP, inhibitor, leaching, nitrate, nitrification, Spain

Background and objectives

In the Iberian Peninsula weather and soil conditions leaching is the main pathway of the nitrogen losses from the agricultural system. As a consequence 12% of the agricultural land has been declared as vulnerable areas to groundwater nitrate pollution according to the 91/676/CEE legislation. Spanish growers have to improve their fertilization practices with the aim of decreasing nitrogen losses. Among different possibilities growers can use stabilized fertilizers with nitrification inhibitors. These compounds can be added to any type of fertilizer and delay the bacterial oxidation of NH_4^+ to NO_2^- (Prasad and Power, 1995), due to the competition for the active site in the AMO (McCarty, 1999). Nitrification inhibitors increase ammonium-N and decrease nitric-N in soils, as a result they also decrease N losses through nitrate leaching (Scheffer, 1994, Fettweis *et al.*, 2001). N use efficiency is improved and N doses and number of applications can be reduced, resulting in economical and environmental benefit (Trenkel, 1997).

In Europe the most commonly used nitrification inhibitor is 3,4-dimethylpyrazol phosphate (DMPP), marketed under the brand of ENTEC®, and developed by BASF. DMPP is a highly efficient molecule, with a soil optimal activity at 1 $kg \cdot ha^{-1}$, and without negative effects on the crops and the environment (Zerulla *et al*, 2001).

This work shows a review of the different research experiments carried out in several Research Institutes in Spain, with the aim of evaluating the mineral soil nitrogen dynamics and nitrogen leaching in different crops and irrigation systems.

Material and methods

In the trials included in this study the conventional fertilization practices are compared with similar nitrogen rates mixed with the nitrification inhibitor 3,4-dimethylpirazole phosphate (DMPP). Treatments are detailed for each trial in the results section.

Results and discussion

TRIALS IN EXTENSIVE CROPS: A maize trial was performed by the Spanish Superior Council of Scientific Research in Madrid (central area of Spain) during 2002 and 2003, comparing two doses of urea with and without DMPP applied in a single application. NH_4^+ accumulation in DMPP treatments was significant the first 60 days after fertilizer application (data not shown). For high and low doses of urea nitrate leaching was lower when DMPP was incorporated into the soil with the fertilizer, reducing accumulated nitrogen leaching with 15 to 40%, depending on the dose of urea (Figure 1).

Figure 1. Accumulated leaching of N-nitrate during two crop cycles. Trial performed in the Spanish Superior Council of Scientific Research.

The Agronomic Technical Institute of Albacete (ITAP) carried out a maize trial in south-western Spain during 2003 and 2004, comparing a conventional fertilization of 320 kg N ha^{-1} in two applications with only one application of DMPP-containing fertilizers at 260 kg N ha^{-1}. Results show a significance increase of NH_4^+-N in V12 maize stage in DMPP treatments and a higher mineral nitrogen (+ 90 kg ha^{-1}) in conventional treatment at harvest. This mineral-N accumulation at the end of the crop means an important risk of nitrate water contamination, when the autumn rains leach the nitrate present in the surface layer to deeper soil layers. Crop yield was similar in both treatments.

TRIALS IN INTENSIVE CROPS: In 2003 the CIFACITA research centre (Murcia, south Spain) developed an experiment in broccoli (*Brassica oleracea*) where different fertilization strategies were compared. An initial base fertilization with manure or mineral fertilization produced more nitrate in soil depth layers (over 30 cm) than only fertigation (Table 1).

In all fertilization strategies DMPP use reduced nitrate in the soil, reducing the nitrate pollution risk. As example the content in the soil solution at 30 cm depth in the treatment with mineral base fertilization and conventional fertigation was 18% higher than the treatment with fertigation with soluble ENTEC fertilizers.

In orange trees (*Citrus sinensis*) a pot trial was carried out in 2002 comparing the use of ammonium sulphate (12 g N·pot^{-1} in six fractions every 20 days) with and without DMPP, applying the same nitrogen dose in both treatments. Results show that DMPP use reduced the nitrogen losses as nitrate with 50% (Figure 2).

Table 1. Nitrate content (expressed in mM) in soil at 30 cm depth.

Fecha	Manure + Fertigation [2]	Manure + DMPP Fertigation [3]	Mineral + Fertigation [4]	Mineral + DMPP Fertigation [5]	Fertigation [6]	DMPP Fertigation [7]
07/03	9,52 ab [1]	3,36 a	12,68 b	11,54 b	7,09 ab	5,26 ab
28/03	0,36	0,20	0,49	0,22	0,25	0,21
24/04	0,03 a	0,01 a	0,73 b	0,03 a	0,11 a	0,06 a

[1] *Each data is the average of three measures. Different letters in the same row means significant differences at P≤ 0,10. Letter Absence means no statistical differences between treatments at P≤ 0,10.*
[2] *Base fertilization: chicken manure (300 kg N·ha^{-1}) + fertigation (120 kg N·ha^{-1}) with conventional fertilizers.*
[3] *Base fertilization: chicken manure (300 kg N·ha^{-1}) + fertigation (109 kg N·ha^{-1}) with DMPP fertilizers.*
[4] *Base fertilization: mineral fertilizers (108 kg N·ha^{-1}) + fertigation (67 kg N·ha^{-1}) with conventional fertilizers.*
[5] *Base fertilization: mineral fertilizers (108 kg N·ha^{-1}) + fertigation (47 kg N·ha^{-1}) with DMPP fertilizers.*
[6] *Fertigation (141 kg N·ha^{-1}) with conventional fertilizers.*
[7] *Fertigation (121 kg N·ha^{-1}) with DMPP fertilizers.*

Figure 2. Losses of nitrogen (mg N/pot) in drainage water. Values are means of four replicates.

Drainage water recovered from fertilized pots contained elevated NO_3^- levels and very low NH_4^+ contents. The total amount of NO_3^- leached from the ASN+DMPP treatment was much lower than that found in the ASN treatment. In contrast, the amount of NH_4^+ in drainage water was higher in the treatment with ASN+DMPP than with ASN alone.

Conclusions

The use of fertilizers with DMPP, both in fertigation and ground fertilization, reduced nitrate content in the soils and, as a consequence, nitrate leaching was decreased. Also higher amounts of ammonium nitrogen remained in the soil, assuring optimum yields and decreasing the risk of nitrate accumulation in plants. The fertilization with DMPP-containing fertilizers can contribute to decrease nitrate pollution of ground and surface waters, so this is a good method to fertilize crops in vulnerable zones.

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Nitrification inhibitors addition to summer and autumn-applied pig slurry: effects on soil, water and atmosphere

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Abstract

The environmental effects due to the addition of two nitrification inhibitors (NI: nitrapyrin and 3,4-dimethylpyrazole phosphate) to pig slurry, when used in summer and autumn applications, were verified on a silty clay loam and moderately calcareous soil covered with wheat stubble and left uncropped from September to April.

The two NI produced comparable effects. Analysis on soil water revealed that when the slurry+NI were applied in late summer the nitrification process was not significantly influenced whilst on plots treated in autumn the use of NI appeared to limit N-losses during the rainy month of November.

Nitrous oxide losses to the atmosphere, measured some days after the distribution, were reduced by the use of chemicals, in both spreading periods.

The use of both nitrification inhibitors in slurry landspreading seems to give the best results when carried out within 3-5 weeks before soil nitrification is reduced to a minimal level by temperature lowering, consequently delaying most of the nitrate production to after the autumn rainy periods and possibly till the following spring and subsequent crop uptake.

Keywords: ceramic suction cups, inhibitor, leaching, losses, manure, nitrification, nitrogen, pig

Background and objectives

To empty the slurry stores before the winter period, when landspreading is forbidden, farmers in the Po valley are forced to use liquid manure even on bare soil during summer and autumn. At this time of year, soil temperature and moisture favour the microbial process of nitrification. As there is no crop uptake, nitrate build-ups in the soil solution and leaching can occur, leading to groundwater nitrate pollution (Mantovi *et al.*, 2005).

Nitrification inhibitors, which slow down the conversion of NH_4^+ to NO_2^- in soil by acting selectively on *Nitrosomonas* spp., have been used especially with mineral fertilizers to improve nitrogen utilization by crops, reducing losses to both the groundwater and the atmosphere (Mc Carty, 1999; Weiske *et al.*, 2001).

The objective of this study was to assess the environmental effects due to the addition of nitrification inhibitors to pig slurry when used in summer and autumn applications.

Material and methods

The effects on soil, water and air of nitrapyrin and 3,4-dimethylpyrazole phosphate added to pig slurry were tested on a soil classified as *fine silty, mixed, superactive, mesic*, *Udifluventic Haplustepts* covered with wheat stubble. The liquid manure was applied in late summer (06-Sep-2004) or autumn (05-Oct-2004) at a rate of about 170 kg N ha^{-1}, with and without the two chemicals, in accordance with the manufacturer's recommended amounts (3,4-DMPP 25%: 4.00 kg ha^{-1}; Nitrapyrin 22.2%: 4.68 l ha^{-1} = 2 quart $acre^{-1}$). The main chemical characteristics of the soil and pig slurry are reported in Table 1.

Table 1. Main chemical characteristics of soil and pig slurry.

Soil (two depths)	0-30 cm	30-60 cm	Pig slurry (two application periods)	Summer application	Autumn application
Sand (%)	8.3	6.3	Dry matter (% fm)	4.22	3.87
Silt (%)	59.3	58.7	Volatile solids (% fm)	2.67	2.36
Clay (%)	32.3	35.0	pH	7.9	8.0
pH (in H_2O)	7.7	7.9	Total organic carbon (% dm)	40.9	37.4
Total $CaCO_3$ (%)	4.7	6.3	TKN (‰ fm)	3.13	3.09
Active $CaCO_3$ (%)	3.7	3.7	Ammonia N (% TKN)	57.7	64.8
Organic matter (%)	2.4	1.4	Total P (‰ fm)	0.84	1.22
TKN (‰)	1.8	1.2	Total K (‰ fm)	3.03	3.21
CEC (meq/100g)	24.4	21.2	Total Cu (mg/kg dm)	352	469
Salinity (meq/100g)	1.4	1.3	Total Zn (mg/kg dm)	986	1094

fm: fresh matter; dm: dry matter

The experimental plots were equipped with ceramic suction cups to collect soil water (at depths of 0.3 and 0.6 m). Three replicates were carried out for each treatment, arranged in a split-plot design. The soil was hoed immediately after landspreading for slurry incorporation and was left uncropped.

Soil and soil water were sampled from September to April at periods ranging from 10 to 20 days (soil water) and from 1 to 3 months (soil) to determine NO_3^- and NH_4^+ concentrations.

Ammonia and nitrous oxide losses to the atmosphere were measured respectively using the wind-tunnel method (at landspreadings) and the chamber method connected with a photoacoustic gas analyser (approximately once a month).

Results and discussion

The two substances produced comparable effects. On plots treated in late summer the nitrification process was not sufficiently reduced, resulting in NO_3^- leaching during the autumn rainy period, whilst on plots where the two nitrification inhibitors were used in autumn the NO_3^- soil water concentrations during the dry winter period were maintained on average 30-35% higher than in other treatments (Figure 1), probably due to reduced leaching during some weeks after landspreading and to a continuation of the nitrification process (Thompson, 1989). Analysis of NH_4^+ in soil and soil water resulted in very slight concentrations with respect to NO_3^-.

Ammonia emissions at spreading were not affected by treatments, whilst nitrous oxide losses to the atmosphere, measured some days after the distribution, were reduced by the use of chemicals, in both spreading periods.

Samplings and measurements should have been further moved closer in the early months of trial, when the nitrification process was more intense and influenced by NI. The abundant rainfall in September and November partially concealed the N-mineralization phenomena, generating significant nitrogen leaching.

Conclusions

The use of both nitrification inhibitors in slurry landspreading seems to give the best results when carried out within 3-5 weeks before soil nitrification is reduced to a minimal level by temperature lowering, consequently delaying most of the nitrate production after the autumn rainy periods and possibly till the following spring (as in this year, when a dry winter occurred), and subsequent crop uptake.

Additional studies are required to test the efficacy of this practice in reducing overall leaching losses, i.e. verifying if the N uptake by a catch crop or a successive crop is raised. If so, nitrification inhibitors could be considered as an alternative solution to the unaffordable costs of high capacity storages.

Figure 1. Meteorological conditions (A) and NO_3-N relative concentration in soil water at 30 cm (B) and 60 cm (C) after the autumn landspreading.

Note: In (A), the daily rainfall and daily average environmental temperature are presented. In (B) and (C) the concentrations are expressed as differences in respect to the values measured before the landspreading. For each sampling data, the ovals group values significantly different at $P \leq 0.05$, according to the SNK's test.

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Nitrogen pig slurry effects in a dry land cereal system

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Abstract

Catalonia, north-eastern Spain, has large areas of dryland cereal ($200*10^3$ ha) and it accounts for 26% of Spanish pig production. Nevertheless, there is a lack of field experiments (at medium and long term) related to pig slurry fertilization. In order to find an optimal strategy for N fertilization of dryland cereals with pig slurry, and to quantify residual effects of organic N in pig slurry, according to plant N uptake and yields, field experiments where conducted over a 4-year period (2000-2004).

The production system highly depends on water availability, which makes it difficult to quantify pig slurry N residual effects. For N uptake, total mineral N equivalence of residual effects, at the end of the study, ranges from 35 to 54% of the organic N fraction applied, that is, from 33 to 73 kg N ha^{-1}. Similar results have been observed for grain yield biomass. However, the mineral N equivalence is higher, between 70 – 94% of organic N fraction applied, which represents 43 to 82 kg N ha^{-1}.

Keywords: nitrogen, production curve, residual effect, use efficiency

Background and objectives

Pig slurry is potentially an important organic fertilizer in north-eastern Spain. However, problems linked to environmental risks, often to water quality, raise the question about its agronomic efficiency. A review from European research of environmental impacts and fertilizer value of organic fertilizers has already been made by Brogan (1981), yet an approach onto a dryland cropping system was still missing. Nevertheless UdL staff (Boixadera, Bosch, Sió and Teira) have summarized (Pam/Pnue-Car/PP, 2004) how to improve agronomic efficiency of organic and mineral fertilizers in Mediterranean countries. The main objective is to find an optimal strategy for N fertilization, using pig slurry, in dryland cereal production. The specific objective is to quantify residual effects of the organic N fraction (Norg) in pig slurry according to plant N uptake and yields.

Materials and methods

Field experiments were conducted over a 4-year period (2000 – 2004). Experiments took place in a dryland cultivated area in Lleida (North-eastern Spain), where the climate is defined as 'Continental Mediterranean' according to Papadakis classification. Soil is classified as Typic Xerofluvent, fine silty, mixed (calcareous), mesic (SSS,1998). Crop rotation was: barley-barley-wheat-barley. Mineral and organic fertilizer (pig slurry) treatments were carried out with three replicates (Table 1). When pig slurry is not used at sowing and in AN treatments, 96 kg P_2O_5 ha^{-1} and 107 kg K_2O ha^{-1} are added. Mineral treatments were set up to fit cereal production curves. Soil nitrate-N content was analysed during the cropping season. At maturity, plants were harvested, total and fractioned biomass (straw and grain) were quantified and analysed for their N content.

Recovery of slurry treatment N fertilizer value (NFV) of the slurry applied each year was calculated as: (Nyield biomass of the slurry treatments – Nyield biomass of mineral N treatment (in which, as in slurry treatments, 60 kg mineral N has been applied))/(total N applied as slurry). Residual effects for organic N pig slurry fraction were derived from a mineral N response function (mineral N equivalent values), fitted to either grain yield or N uptake.

Table 1. Pig slurry (m^3 ha^{-1}) treatments at sowing[(1)] and mineral treatments (kg N ha^{-1}) at sowing and at tillering as ammonium nitrate (AN) in the 4 years considered (2000/01-2003/04).

	2000-2001		2001-2202		2002-2003		2003-2004	
Treatment	Sowing	Tillering	Sowing	Tillering	Sowing	Tillering	Sowing	Tillering
Mineral								
N0	0	0	0	0	0	0	0	0
N1	0	30	0	30	0	30	0	30
N2	30	30	30	30	30	30	30	30
N3	30	60	30	60	30	60	30	60
N4	30	120	30	120	30	120	30	120
Pig slurry								
P11	0	0	0	0	0	0	0	0
P12	0	0	0	0	20(105/66)	60	0	0
P13	0	0	0	0	40(210/132)	60	0	0
P14	0	0	0	0	80(420/264)	60	0	0
P21	0	0	20(88/31)	60	0	0	0	0
P22	20(163/61)	60	20(88/31)	60	20(105/66)	60	20(60/31)	60
P23	20(163/61)	60	0	0	0	0	0	0
P31	0	0	40(182/57)	60	0	0	0	0
P32	40(319/117)	60	40(182/57)	60	40(210/132)	60	40(119/62)	60
P33	40(319/117)	60	0	0	0	0	0	0
P41	0	0	80(330/108)	60	0	0	0	0
P42	80(581/210)	60	80(330/108)	60	80(420/264)	60	80(238/124)	60
P43	80(581/210)	60	0	0	0	0	0	0

[(1)] *Figures in brackets are the N applied (kg N ha^{-1}) as ammonium nitrogen and organic nitrogen applied as pig slurry.*

Results and discussion

Applied N rates from pig slurry fertilization are shown in Table 1.

Quadratic regressions for yields and N uptake have been computed for each cropping season.

Potential production curves for pig slurry treatments at sowing, combined with mineral N at tillering, are higher than for mineral fertilizer N treatments alone (data not shown).

The NFV value (data not shown) is influenced by rainfall during the cropping season but, as an average, does not exceed 30%. No statistical differences were found for treatments where slurry was used for the first time and where it was used repeatedly or between treatments. Some tendencies need further experimental design adjustments. Residual effects, as fertilizer N equivalent values with respect to yield, show a high annual variability (data not shown). Similar trends are observed respect to N uptake (data not shown). Fertilizer N equivalent values based on N uptake suggest that the amount of the organic N fraction recovered by barley and wheat, over three years after application, can vary between 33 to 54% and attains a maximum of 73 kg N ha^{-1} when 210 kg of Norg are applied. Similar results have been observed for grain yield biomass; however, the mineral N equivalence is higher, between 70 – 94% of organic N fraction applied, which represents 43 to 82 kg N ha^{-1}.

For the soil nitrate content, the residual effects at sowing are only discernable at the highest slurry rate (Table 2). At the highest application rate, higher soil nitrate content at sowing can explain some of the observed residual effects.

Table 2. Probability level (Pr>F) of Anova analysis and mean N-NO3- content [(1)] (kg ha^{-1}) before sowing (October) between 0-90 cm depth for each cropping season and treatment (Treat.).

2000-2001		2001-2002 Pr > F= 0.0232 R^2 = 70.43				2002-2003 [(2)] Pr > F= 0.0247 R^2 = 66.62				2003-2004 Pr > F= 0.0008 R^2 = 69.55			
Treat.	Slurry (m^3 ha^{-1})	Treat.	Slurry (m^3 ha^{-1})		Mean	Treat.	Slurry (m^3 ha^{-1})		Mean	Treat.	Slurry (m^3 ha^{-1})		Mean
P42	80	P42	80	96	A	P42	80	199	A	P42	80	281	A
P43	80	P43	0	93	A	P32	40	109	B	P14	0	186	B
P33	40	P33	0	64	AB	P41	0	104	B	P32	40	119	BC
P32	40	P32	40	60	AB	P31	0	81	B	P43	0	101	BC
P23	20	P23	0	49	B	P22	20	75	B	P22	20	99	BC
P11	0	P11	0	48	B	P21	0	74	B	P12	0	92	C
P22	20	P22	20	34	B	P11	0	65	B	P23	0	88	C
						P23	0	57	B	P13	0	87	C
						P43	0	55	B	P21	0	84	C
						P33	0	42	B	P31	0	80	C
						P12	20	-	-	P11	0	71	C
						P13	40	-	-	P41	0	67	C
						P14	80	-	-	P33	0	66	C

(1) Multiple means comparison performed by Duncan's multiple-range test at α=0.05 probability level.

(2) Plots P12, P13, P14 (12x15)m^2 were built up by splitting up in four quarters plot P11 (12x60)m^2. Treatments (m^3 pig slurry ha^{-1}) are applied at sowing. When pig slurry is applied, 60kg N ha^{-1} as ammonium nitrate are used at tillering

Conclusions

Nitrogen fertilizer value (NSV) of the slurry does not exceed 30% of total N applied in pig slurry. Production curves, both grain yield and biomass, obtained for pig slurry treatments, show higher production than the ones for mineral fertilizers only. Residual effects from the organic nitrogen fraction of pig slurry can be observed in grain yields and N uptake although there is an annual variability linked to water availability.

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Leaching and crop offtake as related to the application rate of mineral nitrogen fertilizer in crop rotations dominated by cereals in Denmark

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Abstract

Nitrogen (N) balances are often used in an assessment of the impact of N use in agriculture on the environment, but investigations of how N-balance components change with changes in nitrogen fertilization rates are lacking. The aim was to estimate the long-term change in crop N offtake, nitrate-N leaching, and the depletion rate of soil-N with changes in the fertilizer-N application rate in order to identify any consistency in the expected relationship. We examined a management system equivalent to a crop rotation dominated by cereal crops, with N applied exclusively as mineral fertilizer. We collated results from mainly long-term lysimeter and field experiments all characterized by more than one application rate of mineral-N. Estimates are in kg N ha^{-1} (kg N ha^{-1} applied)$^{-1}$. The change in nitrate-N leaching was estimated at 0.30 for a typical cereal crop, and we found the change in the total N offtake to be within the range 0.45-0.65. The change in the soil-N depletion rate was estimated at 0.08-0.14, and the change in denitrification at 0.05. We conclude that the change in the outputs (crop offtake, leaching, denitrification and change in the soil-N depletion rate) reflects the change in the fertilizer-N application rate.

Keywords: leaching, N-uptake, nitrate, nitrogen, nutrient balance, soil N

Background and objectives

Use of nitrogen (N) balances may have different purposes and are used at varying scale and degree of detail. N balances are often used in an assessment of the impact of agriculture on the environment, but investigations of how N balance components change with a change in nitrogen fertilization rates are lacking. However, since 1998 the Danish legislation on N norms for crops restricts the N application rate to 90% of the economic optimum to reduce the environmental impact of N fertilization (Mikkelsen *et al.*, 2005), causing much debate between farmers and environmentalist. The aim of this desk-top research was to estimate long-term changes in crop N offtake, nitrate-N leaching, and the depletion rate of soil N with a change in the fertilizer N application rate in order to identify any consistency in the expected relationship. This presentation is based on the report by Petersen and Djurhuus (2004).

Materials and methods

We examined a management system equivalent to a crop rotation dominated by cereal crops, with N applied exclusively as mineral fertilizer. We collated results from lysimeter and field experiments all characterized by more than one rate of mineral N. The maximum N rate in the experiments was not subjected to legislative N norms. All field experiments had been running for more than 30 years and lysimeter experiments for up to eight years.

The change in crop N offtake was estimated using linear or quadratic regressions on the application rate for grain and straw separately. Changes were estimated using data from two long-term experiments: The long-term experiments at Askov Experimental Station, described by Christensen *et al.* (1994) and the Swedish Bördighetsförsök recently described by Carlgren and Mattsson (2001). In addition, one-year on-farm experiments during 1993-2001 by local agricultural advisers were used.

Based on field experiments, Simmelsgaard and Djurhuus (1998) proposed a simple model to estimate the relative nitrate-N leaching: $L_r = L_x/L_1 = \exp(0.71((N_x/N_1)-1))$, where L_x and L_1 constitute nitrate-N leaching at a given fertilizer rate (N_x) and the normal fertilizer rate (N_1). Leaching data from lysimeter experiments at Askov Experimental Station, Denmark, were compared with the relative nitrate-N leaching model. Differentiating the model ($dL_x / dN = L_1 (0.71/N_1) \exp(0.71((N_x/N_1)-1))$) give us the change in nitrate-N leaching by a change in the application rate.

The long-term experiments at Askov Experimental Station and the Swedish Bördighetförsök were used to estimate the change in the soil-N depletion rate with changes in the fertilizer N application rate. All calculated estimates express the annual change in kg N ha^{-1} for a change in the application rate of 1 kg N ha^{-1} y^{-1}.

Results and discussion

The number of sources able to provide data is limited as only few long-term experiments include increasing rates of mineral N fertilizer exclusively. The change in crop N offtake and the change in the soil-N depletion rate are both estimated using long-term experiments lasting more than 30 years, but as long-term leaching experiments do not exist, the change in nitrate-N leaching was estimated using nitrate-N leaching experiments that have run for up to eight years.

A typical N-balance in a crop rotation dominated by cereal crops may be written as:

$$N_{applied} + N_{deposition} - N_{offtake(grain+straw)} - N_{leaching} - N_{loss\ from\ crop} - N_{denitrification} = \Delta N_{soil}$$

The question is how do the posts change, and are the changes consistent:

$$\Delta N_{applied} + \Delta N_{deposition} - \Delta N_{offtake(grain+straw)} - \Delta N_{leaching} - \Delta N_{loss\ from\ crop} - \Delta N_{denitrification} = \Delta(\Delta N_{soil})$$

Based on data from long-term experiments, the change in crop N offtake ($\Delta N_{offtake}$) was estimated at 0.50, whereas the calculated slope for the one-year experiments was lower at 0.30 for spring barley and 0.40 for winter wheat. Based on the long-term experiments, the N offtake in straw was estimated at 0.15. Thus, we fund a change in the total N offtake within the range 0.45-0.65 kg N ha^{-1} for every 1 kg N ha^{-1} change in N application rate.

The calculated relative nitrate-N leaching was graphically compared with the model proposed by Simmelsgaard and Djurhuus (1998). The empirical model describes the experimental data well, whether the experiments are from the first or last year in an eight-year experimental term. Using the empiric model for Danish soils, the change in N leaching ($\Delta N_{leaching}$) was estimated at 0.25-0.35 for a typical cereal crop at normal N fertilizer rate of about 100 kg N ha^{-1}. A mean value of 0.30 was used for further calculations.

Assuming that the N deposition is independent of application rate, the change in deposition ($\Delta N_{deposition}$) is set to nil. The change in the denitrification rate may be estimated at 0.05 (Vinther, pers. comm.). Schjørring and Mattsson (2001) estimated an N loss of 0-5 kg ha^{-1} due to ammonia volatilization from the canopy and Petersen and Sørensen (2004) estimated an N loss of 2-7 kg ha^{-1} due to pollination. However the changes in these N losses ($\Delta N_{loss\ from\ crop}$) were not estimated, and are assumed to be nil.

Thus, the change in the N balance may be reduced to:

$$1 + 0 - (0.30+0.15) - 0.30 - 0 - 0.05 = \Delta(\Delta N_{soil})$$
$$1 + 0 - (0.45+0.15) - 0.30 - 0 - 0.05 = \Delta(\Delta N_{soil})$$

for low and high crop N offtake respectively, or rewritten as an inequality:

$$0 < \Delta(\Delta N_{soil}) < 0.20$$

Among 45 long-term experiments, only three experiments include more than one rate of mineral N fertilizer (Glendining and Powlson, 1995). This exclusive group comprises the Broadbalk experiment at Rothamsted and the above-mentioned experiments at Askov and in Sweden. However, these experiments may be used for estimating $\Delta(\Delta N_{plough\ layer})$ rather than $\Delta(\Delta N_{soil})$, as soil samples represent the plough layer only. Assuming that changes in the subsoil N-pool are constant and independent of N fertilizer rate, *i.e.* $\Delta(\Delta N_{subsoil}) = 0$, then $\Delta(\Delta N_{soil})$ is equal to $\Delta(\Delta N_{plough\ layer})$.

A negative estimate of –1.6 10^{-6} %-point y^{-1} (kg N $ha^{-1})^{-1}$ for $\Delta(\Delta N_{soil})$ in the Broadbalk experiment (calculated from a graph in Glendining *et al.* (1996)) is interpreted as a higher depletion rate of the soil N pool when more mineral fertilizer N is applied. However, this is not common sense and the effect could be due to variability within the field as the treatments were neither replicated nor randomized. Therefore, using an estimate for $\Delta(\Delta N_{soil})$ of 3-4·10^{-6} %-point y^{-1} (kg N $ha^{-1})^{-1}$ obtained from the two Scandinavian long-term experiments, a plough layer depth of 20-25 cm and a density of 1.3-1.4 g cm^{-3}, the mean depletion rate of about 20-40 kg N ha^{-1} y^{-1} changes by 0.08-0.14 kg N ha^{-1} (kg N $ha^{-1})^{-1}$. This estimate is within the limits of the inequality.

Conclusion

We conclude that the change in output components of the N balance (crop offtake, leaching, denitrification and change in the soil N depletion rate) reflects the change in the fertilizer-N application rate. Furthermore, when a complete dataset with synchronous recordings is not available, the collating data from different sources was shown to be a valuable tool for estimating the N balance components and valuating the consistencies of N balances.

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Modelling the impact of dietary adjustments on slurry composition of stall-fed dairy cows

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Abstract

A dynamic rumen function model was expanded with static equations to predict faecal and urinary Organic Matter (OM), carbon (C) and nitrogen (N) output, classified in six different components. With this model, forty different diets for stall-fed dairy cows, covering a wide range of ration composition, were evaluated. N excretion was partitioned in three fractions, N_M, N_E and N_R, based on the C:N ratio of individual components, varying in plant availability. Simulation results show that nutrition can have a substantial effect on total N excretion and excreta composition, as a result of absolute and relative differences in N_M and differences in the C:N ratio of N_R. Furthermore, substantial differences in OM degradability among different diets were simulated. The simulated differences will probably have a large effect on ammonia emission from housing and plant availability of N. From the selected strategies, adjustment of grass silage type and inclusion of maize silage showed the largest effects on excreta composition.

Keywords: excreta composition, losses, nitrogen, nutrition, plant availability

Background and objectives

Worldwide, cattle husbandry is the major source of anthropogenic ammonia emissions. Nutrition management has often been suggested as a means to reduce emissions from dairy systems and it has been shown that ammonia emissions can be reduced by dietary adjustments (Monteny, 2001). However, diet composition not only influences emissions, but also affects Organic Matter (OM) and nitrogen (N) excretion by the cow and their distribution over different components in faeces and urine. The mixture of faeces and urine, slurry, is an important organic fertilizer in crop (fodder) production and it has been shown that slurry N availability is affected by diet composition (Sørensen *et al.*, 2003).

To quantify the effect of differences in nutrition on excreta composition, a dynamic, mechanistic rumen function model (Dijkstra *et al.*, 1992) was extended with static equations that describe intestinal digestion. This model was applied to evaluate differences in excreta composition for 40 different diets. The effect of these differences on N availability and potential ammonia losses is discussed.

Materials and methods

An extant dynamic model representing microbial fermentation in the rumen (Dijkstra *et al.*, 1992) was expanded with static equations on digestion processes in the small and large intestine and excretion of individual urinary components derived from a review of literature. The model predicts OM, carbon (C) and N output for four faecal and two urinary components (Table 1). Forty different diets were explored, representing stall-fed situations and including four different grass silage types resulting from variation in fertilisation (HF = 350 kg N ha^{-1} yr^{-1}, LF = 175 kg N ha^{-1} yr^{-1}) and cutting regime (EC = 3000 kg DM ha^{-1}, LC = 4500 kg DM ha^{-1}), five grass silage replacement types (no replacement, 50% maize silage, 15% straw, 15% pressed sugar beet pulp or 15% potatoes) and two levels of concentrates (20% and 40%). Concentrate composition was assumed to be constant. Chemical composition and rumen degradation characteristics of individual feedstuffs were derived from literature. Dry Matter Intake (DMI) of the complete rations was estimated using the 'Cow Model' (Zom *et al.*, 2002). Milk production (FPCM) was assumed to

be the lowest of the simulated potential milk productions based on available energy and lipogenic, glucogenic and aminogenic nutrients (Dijkstra *et al.*, 1996).

Table 1. Individual compounds of modelled output for faecal and urinary components.

Component	*Category*	*Individual compounds*
Urinary components		
UUC	Urea-like	urea, uric acid, allantoin
UNUC	Non-urea-like	hippuric acid, creatine, creatinine, xanthine, amino acids
Faecal components		
FMC	Microbial	undigested rumen microbial biomass, microbial biomass produced in LI
FEC	Endogenous	epithelial cells, digestive enzymes, bile, mucus
FFFC	Feed Fibre	undigested NDF (rumen digestible and indigestible), rumen indigestible protein
FOFC	Other Feed	undigested non-NDF (starch, fat, rumen digestible protein), VFA produced in LI

LI = Large Intestine; NDF = Neutral Detergent Fibre; VFA = Volatile Fatty Acids.

Results and discussion

DMI, crude protein (CP) and net energy content (NEL) of the simulated diets ranged from 16.0 to 22.4 kg d^{-1}, 108 to 209 g kg^{-1} DM and 5.71 to 6.76 MJ kg^{-1} DM, respectively, representing a wide range in ration composition. For all strategies, milk production (FPCM) was limited by energy availability and ranged from 19.1 to 33.8 kg d^{-1}. Despite the low protein contents in some diets, aminogenic nutrients were never in short supply.

Table 2. Mean values and range of faecal and urinary OM and N excretion and C:N ratio in faeces, urine and total excretion for 40 different diets for stall-fed dairy cows.

	OM excretion (g d^{-1})		N excretion (g d^{-1})		C:N ratio	
	mean	range	mean	range	mean	range
Urine	649	275 – 1151	231	94 - 414	0.9	0.9 - 1.0
Faeces	4028	2633 – 5375	138	118 - 157	13.2	9.3 - 16.8
Total excretion	4678	3381 – 5864	370	218 - 564	5.9	3.0 - 9.7

Table 2 shows that for all strategies, faecal N excretion is relatively constant with a variable C:N ratio, while urinary N excretion shows a large variation with a fixed C:N ratio. Total N excretion varied by a factor of 2.5 between diets with a highly variable C:N ratio. As ammonia emission from dairy houses is related to total N excretion (Monteny, 2001) these differences will have major effects on total ammoniacal N-emissions.

Table 3. Mean values and range of three different N fractions following partitioning of faecal and urinary components based on C:N ratio of the individual components.

Distribution of N excretion (%)				C:N ratio	
Type	Origin[1]	Mean	Range	Mean	Range
Immediately available (N_M)	UUC	50 ± 8	33 – 63	0.5	0.5 – 0.5
Easily decomposable (N_E)	UNUC, FEC, FMC	33 ± 5	25 – 44	3.4	3.0 – 3.8
Resistant (N_R)	FOFC, FFFC	18 ± 3	12 – 24	31.5	20.0 – 43.4

[1] *For explanation of acronyms, see Table 1.*

To estimate plant available N, manure N is often partitioned in three fractions (N_M, N_E and N_R, Sluijsmans & Kolenbrander, 1977) and N availability of manure components has been shown to be related to their C:N ratio (Van Kessel *et al.*, 2000). In Table 3, N excretion is partitioned in three different fractions, on the basis of the C:N ratio of individual components. The N_M fraction shows a considerable range in excreted N (33 – 63%), with an average of 50% (Table 3). Total N excreted in N_E and N_R was relatively constant, but their fractions increased with decreasing UUC (N_M) excretion. The N_R fraction showed a considerable variability in C:N ratio, which may have a strong effect on N availability of all fractions due to immobilization. N availability is also related to OM degradability in soils. The fraction of OM excreted as NDF varied between 47 and 77%, of which 15 – 58% was potentially rumen–degradable, hence different diets result in considerable differences in OM degradability.
Figure 1 shows that adjustment of silage type had the largest effect on both, total N excretion and the distribution of N over the different fractions. Both, reduced fertilisation and delayed cutting, resulted in a strong absolute and relative decrease in N_M excretion. Inclusion of maize silage and straw had a similar effect. Inclusion of maize silage and the use of LC grass silage, with a lower digestibility, increased the C:N ratio of the N_R fraction. Inclusion of beet pulp and potato resulted in a slight increase in the N_E fraction.

Figure 1. Average N excretion in N_M, N_E and N_R for different diets based on four different grass silage types (HFEC = high fertilisation and early cut, HFLC = high fertilisation and late cut, LFEC = low fertilisation and early cut, LFLC = low fertilisation and late cut) and five types of grass silage replacements (NO=no replacement, MSIL= 50% Maize silage, PBP=15% beet pulp, POT=15% potatoes, STR=15% straw).

Conclusions

The simulation results show that nutrition can have a substantial effect on total N excretion and excreta composition, as a result of absolute and relative differences in N_M and differences in the C:N ratio of N_R. Furthermore, substantial differences in OM degradability between diets were simulated. From the selected strategies, adjustment of silage type and inclusion of maize silage showed the largest effects on excreta composition.

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Utilization of nitrogen from cattle slurry applied to grassland as affected by diet composition

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Abstract

Slurry composition of non-lactating dairy cows was manipulated by feeding diets with extreme high and low levels of dietary protein and energy. Differences in fertilizing value of the slurries that showed a wide range in C:N ratio were evaluated in a one-year grassland experiment and by long-term modelling. Average first year availability of slurry-N compared to fertilizer (MFE), was negatively related to its C:N ratio. From model simulations it is concluded that with the use of 180 kg slurry N ha^{-1} in a situation with slit injection, the reduced first year N availability of slurry with a high C:N ratio will only be compensated for by soil N mineralization at the very long term.

Keywords: C:N ratio, cattle, diet composition, fertilizing value, grassland, manure, nitrogen, use efficiency

Background and objectives

Increasing the efficiency of nitrogen (N) use in dairy cow feeding is an important tool for decreasing environmental pollution. It has been shown that a reduction of the protein content of dairy cow diets reduces slurry N volatilization (Paul *et al.*, 1998; Külling *et al.*, 2001). At the same time, Sørensen *et al.* (2003) showed that the short-term fertilizer N value of slurry was negatively related to its C:N ratio. On the long term, application of dairy cattle slurry will result in a significant amount of residual organic N, increasing the long-term fertilizer N value. As organic matter and organic N accumulation are determined by the input of C rather than the input of N (Hassink, 1994), slurry C:N ratio can also affect its long term fertilizer N value.

This paper investigates whether there are differences in fertilizer N value of slurries from non-lactating dairy cows, that have been fed diets with combinations of extremely high and low protein and energy levels, when applied to grassland in a one-year experiment. In addition, the effect of long-term application of cattle slurries varying in C:N ratio on soil N mineralization was estimated using a simulation model.

Material & Methods

Eight pairs of non-lactating dairy cows were fed diets with extremely high and low levels of dietary energy and protein (Table 1). The produced slurries (n=8) and slurries from commercial farms (n=4), were slit injected on two grassland fields on the same sandy soil type with differences in age to evaluate their short-term fertilizer N value. Slurry was applied in spring (100 kg N ha^{-1}) and after the first cut (80 kg N ha^{-1}) while in total four cuts were harvested. Inorganic fertilizer N treatments were included in the experiment to calculate the Mineral Fertilizer Equivalent (MFE) of slurry N.

In addition, long-term effects of slurry application on the time course of soil N and C content of the top 25 cm and subsequent mineralization were simulated for the experimental slurries with highest and lowest C/N ratio (GYH and GOL) for both fields (OLD and NEW) using equations described by Groot *et al.* (2003).

Table 1. Selection of dietary and slurry characteristics of the slurries used in the grassland experiment.

Slurry	Forage type *	*Diets (kg^{-1} dm)* NEL** (kJ)	CP (g)	*Slurry characteristics* NH_4-N (g kg^{-1} dm)	C:N ratio	C:Norg ratio
GMH	60% MS & 40% HDGS	6700	182	48	5.9	19.1
GYH	100% HDGS	6679	200	45	5.1	12.4
GOH	100% LDGS	5513	185	38	6.4	17.8
SH	100% STR	5327	187	48	6.8	25.2
GML	60% MS & 40% HDGS	6693	115	27	8.4	18.7
ML	100% MS	6700	101	28	9.6	23.8
GOL	100% LDGS	5327	110	17	11.4	22.9
SL	100% STR	5251	105	26	10.1	27.0
Range of commercial slurries used (n=4)				13-34	7.2-11.2	16.7-18.6

* *MS=maize silage; HDGS=high digestible grass silage; LDGS=low digestible grass silage; STR=straw.*
** *According to Dutch standards for net energy lactation.*

Results & Discussion

The OLD field showed a higher total N uptake whereas DM yields were similar for the two fields. Total N uptake of the fertilized plots was linearly related to N application with artificial fertilizer for both fields, which allowed for the calculation of the MFE. Ranges in slurry MFE were large for both the NEW (37-78%) and the OLD field (36-65%). Average MFE of the slurries on the OLD field (47%) was lower than on the NEW field (56%), probably as a result of denitrification of slurry N during wet conditions in spring. Slurries from high crude protein diets showed a significantly higher MFE ($p<0.05$) compared to low crude protein diets. No significant differences in MFE were observed between slurries from high and low energy diets. On both fields, MFE appeared to be positively and linearly related to the ammonium content (in DM) ($P<0.001$) and negatively related to the C:N ratio of the slurry ($P=0.001$) (Figure 1). These relations show great similarity with findings of Sørensen *et al.* (2003).

Figure 1. Relation between the average mineral fertilizer equivalent (MFE) of experimental (▲ NEW and Δ OLD) and commercial (● NEW and ○ OLD) cattle slurries on a NEW and OLD grassland and A) the C:N ratio of the slurry and B) the ammonium (N-NH4) content in DM of the slurry.

Simulation of the effect of long-term annual application of 180 kg N ha^{-1} with highest and lowest C:N ratio suggested that both slurries would lead to an increase in annual soil N mineralization. Both, soil N mineralization and soil organic C are substantially higher in equilibrium state for the slurry with the highest C:N ratio. The lower N availability

of slurry GOL will be compensated by the increase of soil N mineralization only at the very long term (Figure 2). These results imply that, if ammonia losses can be limited, slurries with a low C:N ratio will give higher yields and a higher N utilization in the short and medium term. However, possible beneficial effects of C addition on soil properties and biological soil activity from a slurry with a high C:N ratio were not taken into account in the current model evaluation.

Figure 2 *Simulated time course of annual availability of slurry and soil N after long-term annual application of 180 kg N ha^{1} from slurry with a low (GYH) and a high (GOL) C:N ratio on a NEW and an OLD grassland.*

Conclusions

The average first year availability of N from dairy slurries with a wide range in C:N ratio resulting from dietary differences, as determined in a grassland experiment through comparison with fertilizer N (MFE), was negatively related to the slurry C:N ratio. From model simulations it is concluded that the reduced first year N availability of slurries with a high C:N ratio, applied as 180 kg slit-injected slurry-N ha^{-1}, will only be compensated for by soil N mineralization at the very long term.

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Long term N fertilizer value of cattle slurry applied to maize

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Abstract

Results of long-term field experiments were used to calibrate and validate a soil-crop model describing the accumulation of the residual nitrogen (N) effect of cattle slurry. Subsequent calculations indicated that the N Fertilizer Value (NFV) of cattle slurry applied to maize rises from 55-60% when manure is applied for the first time to 80% after 6-8 years. Such more realistic estimates of the NFV could help to save mineral N fertilizer.

Keywords: decomposition, maize, manure, nitrogen, residual effect

Background and objectives

The Nitrogen Fertilizer Value (NFV) of manures is often derived from one-year (or at best short-term) field trials by comparison with (a) mineral fertilizer nitrogen (N) treatment(s). This design may overlook the cumulative residual N effects of manures (Figure 1). NFV's can hence be underestimated (Schröder & Stevens, 2004).

The N requirement of a crop (Nopt) is generally based on multi-year response trials with mineral fertilizer N. These trials are often carried out on fields to which manures have been applied in preceding years. Initially, crops in these trials benefit from residual N effects. Nopt's are hence underestimated when the eventual recommendations take no account of the N supplying power of a soil resulting from these earlier manure inputs (Schröder, 2005; Schröder *et al.*, 2000).

These combined biases of NFV's and Nopt's will compensate each other somehow in our advices to farmers. However, site specific N management requires a better disentanglement of our imperfect recommendations on NFV and Nopt. We have attempted to better quantify the residual N effect of manures, and thus NFV's, through a combination of experiments and modelling. The focus of this work was on cattle slurry, the most common manure type on dairy farms in the Netherlands.

Material and methods

From 1997 to 2003 we conducted a field experiment (Experiment 1) on a sandy soil in the Netherlands. Treatments comprised different time series of spring-injected cattle slurry applied to silage maize at rates ranging from 0 to 220 kg total N ha^{-1} yr^{-1}, whilst compensating for differences in available phosphorus and potassium. Results were used to calibrate a simple model reflecting the N flows between soils and crops (Schröder *et al.*, 2005a). Subsequently, the model was tested with an independent data set derived from a long-term (1988-2002) field experiment (Experiment 2) on another sandy soil, comprising repeated annual spring-injected cattle slurry applications to silage maize at rates ranging from 0 to 170 kg total N ha^{-1} yr^{-1}, with and without mineral fertilizer N supplementations (Schröder *et al.*, 2005b).

Figure 1. Hypothetical accumulation of residual N effects resulting from repeated manure applications (—●—) and the residual N effect in the first (——), second (— —), third (——) and fourth (— —) year after manuring has stopped.

Results and discussion

Dry matter and N yields of silage maize responded positively (P<0.05) to both current cattle slurry applications and applications in previous years (Experiment 1, Table 1). N yields could be satisfactorily predicted with a simple N model by adopting an annual relative decomposition rate (RDR) of the organic N in cattle slurry of 25-33% and a RDR of 3% of the native organic N pool in the upper 30 cm soil layer (Experiment 1, Figure 2). The parameter setting did not need any adjustments to achieve a satisfactory match between observed and simulated N yields in the test (Experiment 2, Figure 3). Subsequent model calculations indicated that the relative N fertilizer value (RNFV) of cattle slurry increases from approximately 55-60% when manure is first applied, to approximately 80% after 6 and 8 years for RDR's of 33% and 25%, respectively. Of course, outcomes can be different for other manure types, soil types, application techniques and crops (Schröder *et al.*, 2005a).

Table 1. Total aboveground silage maize yield (tonnes DM ha^{-1}), as affected by cattle slurry application rates (Schröder et al., 2005a).

Year(s):	Average cattle slurry rate (kg total N ha^{-1} yr^{-1}) in consecutive periods:								
1997-1999	0	0	0	106	106	106	212	212	212
2000-2002	0	111	221	0	111	221	0	111	221
2003	0	96	191	0	96	191	0	0	0
1997	12.3 a[a]				14.2 b				15.1 b
1998	7.6 a				11.1 b				12.3 b
1999	8.9 a				13.7 b				15.8 c
2000	7.2 a	11.7 c	14.3 d	9.5 b	12.9 cd	14.3 d	11.3 c	13.9 d	14.2 d
2001	7.5 a	11.5 c	13.3 d	9.6 b	12.8 cd	12.4 cd	9.6 b	12.6 cd	15.1 d
2002	9.7 a	15.1 c	16.4 d	11.8 b	16.0 d	16.8 d	12.9 b	16.6 d	16.8 d
2003	8.9 a	13.3 de	14.2 ef	10.4 ab	13.5 de	15.5 f	11.3 bc	12.1 cd	12.2 cd

[a] *Different letters within rows denote significant (P<0.05) differences.*

Figure 2. Observed N yields of silage maize versus simulated N yields, when relative decomposition rates of 25% yr^{-1} for the pool of organic manure-N and of 3% yr^{-1} for the soil organic N pool had been adopted (Experiment 1, Schröder et al., 2005a).

Figure 3. Observed N yields of silage maize versus simulated N yields, when relative decomposition rates of 25% yr^{-1} for the pool of organic manure-N and of 3% yr^{-1} for the soil organic N pool were imposed (Experiment 2, Schröder et al., 2005b).

Conclusions

The long manuring history of most agricultural systems, justifies adoption of less conservative estimates of the NFV of manures which take better account of the cumulative residual N effects of repeated manure applications. Such *a priori* estimates of the 'native soil fertility' may save mineral fertilizer N and may be more reliable than estimates based on soil tests.

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Effects of nitrogen fertilizer and compost on nitrogen utilization by wheat in four Japanese soils

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Abstract

A 3-year field experiment was conducted on four soils to investigate the effects of N fertilizer and compost on N use efficiency in wheat (*Triticum aestivum* L.). Mean grain yield (GY) in no N treatment was 2193 kg ha^{-1} and 4859 kg ha^{-1} in no compost and composted plots, respectively. Nitrogen fertilization increased GY up to 5831 and 6460 kg ha^{-1} in no compost and composted plots, respectively. GY was significantly correlated with N uptake (Nup). In no compost plots, N fertilizer rates (Nf) in GY of 5000 kg ha^{-1} were comparable, and were higher or lower than Nup, depending on the soil type. In composted plots, GY in two of four soils were more than 5000 kg ha^{-1} even in no N treatment. Utilization efficiency decreased with Nup and increase in Nf and compost application reduced agronomic efficiency. Apparent N recovery of fertilizer N decreased in composted plots compared with no compost plots especially at higher Nf. Compost application increased N mineralization potential (NMP). The simple methods used in estimating NMP were not as reliable as aerobic incubation method. Appropriate N management would bridge the gap between Nup and Nf. Long-term compost application contributes to reduction of chemical N fertilizer use.

Keywords: compost, mineralization, N use efficiency, nitrogen, nutrient balance, wheat

Background and objectives

Japan produces only 13% of the country's wheat requirement. The challenge is to increase domestic production and improve its quality as well. Wheat production can be influenced by N availability and supply (Whitfield and Smith, 1992; Bar-Tal *et al.*, 2004). Applying composts to soil increases soil fertility. Organic matter content and net N mineralization increased in compost-treated soil with time (Bar-Tal *et al.*, 2004). Nitrogen use efficiency is vital in crop management system (Mahler *et al.*, 1994). Unaccounted portions of N could pose negative impacts on the environment (i.e. leaching, denitrification). For cereals such as wheat, N use efficiency is low (Raun and Johnson, 1999). Therefore, it is important to optimize N use efficiency while increasing wheat GY for sustainable production. This study was conducted to investigate wheat response to N fertilizer and compost and N mineralization potential in soil.

Material and methods

Field experiments were conducted at NARC, Tsukuba, Japan (36º01' N, 140º01' E) on four soils (artificially placed up to 80 cm depth with concrete enclosures of 20 m x 25 m, without bottom) with and without annual plant compost application (20 t-fm ha^{-1} yr^{-1}, *ca.* 220 kg-N ha^{-1} yr^{-1}) since 1983. The soils are 1) Cumulic Andosol (CA, Pachic Melanudands), 2) Low-humic Andosol (LHA, Typic Hapludands), 3) Yellow soil (YS, Typic Paleudults) and 4) Gray Lowland soil (GLS, Typic Hydraquents). Winter wheat (cv. Ayahikari) was grown for three seasons (2001/2002, 2002/2003 and 2003/2004). Four N fertilizer treatments (N0, N1, N2 and N3, kg ha^{-1}) were set up in each season (Table 1) and equal amounts of N were applied at two timing (basal and stem elongation stage). The GY (kg ha^{-1} at 125 g kg^{-1} moisture content) and Nup (kg ha^{-1}) were measured at maturity.

The N use efficiency parameters were calculated as follows using terminology according to Good *et al.* (2004):
Apparent N recovery (ANR) of fertilizer N (%) = 100 x (Nup in N treatment – Nup in no N treatment)/Nf.
Agronomic efficiency = (GY in N treatment - GY in no N treatment)/Nf.

Utilization efficiency = GY/Nup.
ANR of compost N (%) in no N treatment was calculated as 100 x (Nup in composted plot – Nup in no compost plot)/compost N applied.

Table 1. Application rates of N fertilizer (kg ha^{-1}).

Treatments	CA	LHA		YS		GLS	
	01[1)]-03	01	02-03	01	02-03	01	02-03
N0	0	0	0	0	0	0	0
N1	40	40	80	60	80	30	40
N2	80	80	160	120	160	60	80
N3	160	160	240	240	240	120	160

[1)] *01, 02 and 03 mean 2001/2002, 2002/2003 and 2003/2004 seasons, respectively.*

Soil sampling was conducted in July 2004 at 0-15 and 15-30 cm depths. Twelve sub-samples were collected in each plot and mixed to form a composite sample from that plot. The samples were air-dried, passed through a 2-mm sieve and used for the determination of organic C, total N and NMP. The NMP was measured by aerobic incubation method (30°C, 28d), hot KCl extraction method (Gianello & Bremner, 1986), neutral phosphate buffer method (Matsumoto *et al.*, 2000) and Illinois soil N test method (Khan *et al.*, 2001).

Results and discussion

Mean GY of four soils in no N treatments was 2193 kg ha^{-1} in no compost plots and 4859 kg ha^{-1} in composted plots (Table 2). In N fertilization treatments, mean GY increased up to 5831 in no compost plots and 6460 kg ha^{-1} in composted plots. The GY of wheat in Japan is about 4000 kg ha^{-1} at present and is expected to be more than 5000 kg ha^{-1}. In composted plots, GY in LHA and GLS was over 5000 kg ha^{-1} even in no N treatment and there was no significant correlation between GY and Nf in both soils.

Table 2. Grain yield and N uptake by wheat (average of 3-seasons).

Soil	Grain yield (kg ha^{-1})				N uptake (kg ha^{-1})			
	No N treatment		N treatments[1)]		No N treatment		N treatments[2)]	
	-comp.[3)]	+comp.	-comp.	+comp.	-comp.	+comp.	-comp.	+comp.
CA	2113	4227	5330	5757	36	85 (22)[4)]	135	148
LHA	2613	5997	6070	6880	43	131 (40)	157	203
YS	1033	3560	5493	6393	21	67 (21)	144	163
GLS	3013	5653	6430	6827	54	121 (30)	149	162
Mean	2193	4859	5831	6464	38	101 (29)	146	169

[1)] *Average of the highest yield in each year.*
[2)] *Average of Nup at the highest yield in each year.*
[3)] *-comp., no compost plot; +comp., composted plot.*
[4)] *Value in parenthesis is ANR of compost N.*

Mean Nup in all soils in no N treatment was 38 and 101 kg ha^{-1} in no compost and composted plots, respectively (Table 2). The GY was strongly correlated with Nup ($GY = -0.175Nup^2+66.4Nup$, r = 0.97, n = 48) in no compost plots. This relationship was weaker in composted plots ($GY = -0.115Nup^2+57.6Nup$, r = 0.79, n = 48). These equations suggested that utilization efficiency decreases linearly with Nup and utilization efficiency is greater in no compost plots than in composted plots when Nup is less than 146 kg ha^{-1}. Assuming GY of 5000 kg ha^{-1}, the estimated Nup in no compost plots is 104 kg ha^{-1}. Based on the linear regression analysis (Nup = aNf + b), the Nf required to obtain this level of Nup was almost comparable with the Nup in CA and LHA, while the Nf required was much more than Nup in YS and less in GLS. The estimated value of Nup (134 kg ha^{-1}) in composted plots was similar to Nup in no N treatment in LHA and GLS. ANR of fertilizer N was significantly correlated with agronomic efficiency in both no compost (r = 0.94, n = 36) and composted plots (r = 0.77, n = 36). The increase in Nf and compost application reduced agronomic efficiency substantially (Table 3). Mean ANR of fertilizer N in all soils was lower in composted plots than no compost plots (62 vs 50%). However, this difference was small at lower Nf level (73 vs. 67% at N1). Mean ANR of compost N was 29% (Table 2). Wheat response to compost and N fertilizer varied depending on soil types. The results suggested that better understanding of soil N status is essential to maintain and/or improve N use efficiency in composted soil.

Table 3. Apparent N recovery of fertilizer N and agronomic efficiency.

Treatments	CA		LHA		YS		GLS	
	-comp.	+comp.	-comp.	+comp.	-comp.	+comp.	-comp.	+comp.
				ANR of fertilizer N				
N1	66	77	60	43	58	65	106	83
N2	66	63	60	51	60	49	87	65
N3	47	27	41	15	37	29	50	27
				Agronomic efficiency				
N1	33	24	29	6	33	25	48	23
N2	26	16	22	7	27	16	38	7
N3	20	8	17	1	17	11	24	6

-comp., no compost plot; +comp., composted plot.

There was significant correlation between NMP by aerobic incubation and NMP by other methods (r =0.93 for hot KCl method, r = 0.57 for P-buffer method, r = 0.76 for ISNT method, n = 16). Compost application increased NMP in all soils. Correlation between Nup in no N treatment and NMP in soil at 0-15 cm depth was significant in aerobic incubation (r = 0.89, n = 8) and hot KCl methods (r = 0.71), and insignificant in other methods (r = 0.16 for P-buffer method, r = 0.32 for ISNT method).

Conclusions

Appropriate N management is required to bridge the gap between N uptake and the amount of fertiliser-N applied. Long-term compost application contributes to a reduction in chemical N fertilizer use. Further improvement of the simple methods for NMP is needed to have a near-actual estimate of the soil N supply capacity across various types of soil. Hot KCl extraction was a relatively simple yet excellent method to evaluate soil N supply capacity, but the aerobic method was more reliable.

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Nitrogen oxides emission from a soil fertilised with treated pig slurries

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Abstract

Organic and mineral fertilizers are known to be key variables in the regulation of nitrous oxide (N_2O) and nitric oxide (NO) emissions from soils. Denitrification, N_2O and NO emissions from a soil amended with untreated and treated pig slurries were quantified in a field experiment covering the growth season of an irrigated potato crop (*Solanum tuberosum)*. Anaerobically digested pig slurry, composted pig manure (CP) and untreated pig slurry (IPS) with and without a nitrification inhibitor dicyandiamide (DCD) were compared with urea and a Control treatment without any N fertilization. Digested pig slurry mitigated the total N_2O emission and denitrification (by 15% and 24% respectively) when compared with IPS but NO emission was unchanged. CP treatment increased the N_2O emission by 14% and NO emission by 66% but reduced the denitrification losses (by 24%) relative to untreated pig slurry. DCD partially inhibited nitrification during the first 20 days and reduced N_2O and NO emissions from pig slurry by at least 29 and 74%, respectively.

Keywords: denitrification, N_2O, manure, nitric oxide, organic fertilizer, pig

Background and objectives

Pig slurry usually contains high concentrations of NH_4^+-N, which is rapidly nitrified when mixed with aerated soils. Slurries also supply easily decomposable organic C that can both sustain denitrification and induce anaerobiosis by stimulating biological O_2 demand (Rochette *et al.*, 2000). Several authors have reported increases in N_2O and NO emission following application of slurry to soils (Chadwick *et al*, 2000; Vallejo *et al.*, 2005). The land application of livestock manures increases the risk to animal and human health because of the diffusion of pathogens in the soil and the air. Anaerobic digestion and composting are treatments that reduce the number of pathogens in manures (Burton and Turner, 2003) and also the odour compounds - another important problem associated with the application of manures. These treatments change its chemical composition, especially with respect to the degradable C fraction. Little information exists on the influence of anaerobically digested pig slurry (Petersen, 1999) and composted pig manure applied in the field on the N_2O and NO emissions.

Dicyandiamide (DCD), a nitrification inhibitor, mixed with slurries fertilizers could be efficient in mitigating N_2O and NO emission from soils (de Klein *et al.*, 1996; Vallejo *et al.*, 2005), although more studies on its efficiency of reducing gas emissions are necessary to study its effect on the composition of emitted gases. The aims of this study were: 1) to quantify N_2O and NO emission from an irrigated crop in a Mediterranean climate; 2) to compare the effects of the treated pig slurries with untreated pig slurry on the N emission, and 3) to evaluate the effect of the DCD nitrification inhibitor to reduce N_2O and NO emissions.

Materials and methods

The field experiment was carried out at 'El Encín' field station (40º 32'N; 3º 17' W) on a potato crop in 2004. The soil was a *Haploxeralf* and has a clay-loam texture (281±3 g kg^{-1} clay) and pH=7.6. Eighteen plots (8 m x 5 m) were selected in the experimental field and on May 16th six fertilizer treatments were applied: digested pig slurry (An-PS), composted solid fraction of pig slurry (CP), immediately incorporated pig slurry (IPS), incorporated pig slurry + dicyandiamide (IPS+DCD), urea (U) and a Control treatment (Control) without any fertilizer. The application of organic fertilizers was adjusted to provide 175 kg available N ha^{-1}. On June 7th the potato crop (*Solanum tuberosum*, cv Desirée) was sown in rows with 75 cm spacing. The potatoes were harvested on October 15th. Watering was by a sprinkler system and a total of 14 irrigation sessions took place from June 16th to September 15th at a frequency of once per week. The N_2O and NO were sampled using closed chambers, 35 cm in diameter and 23 cm in height, inserted into the soil to a depth of 3 cm. NO concentration was analysed using a chemiluminescence detector (Environment AC31 M), and N_2O concentration by gas chromatography (HP6890), using a ^{63}Ni electron-capture detector. Denitrification was also estimated in the field by a core incubation method in the presence of acetylene (C_2H_2). Gas samples were taken once a day for 10 days after the fertilizer application, once or twice per week during July and August and fortnightly from September to November.

Results and discussion

Table 1. Cumulative denitrification losses, N_2O and NO emission during the potato growing season.

Treatments	Denitrification losses (kg N ha^{-1})	N_2O emission (kg N ha^{-1})	NO emission (kg N ha^{-1})
Control	8.0±0.10 a[1]	3.69±0.36 a	0.005±0.004 a
Untreated pig slurry	28.5±0.89 b	5.62±0.23 bc	0.103±0.023 ab
Digested pig slurry	21.7±0.06 ab	4.69±0.60 ab	0.101±0.025 ab
Pig slurry+ DCD	28.4±0.87 b	4.01±1.42 ab	0.027±0.016 a
Composting solid fraction[2]	21.6±0.11 ab	6.41±0.87 c	0.171±0.077 bc
Urea	23.8±1.70 b	7.31±1.39 c	0.240±0.185 c

[1] *Mean value of accumulated N_2O emission from three plots ±standard deviation. Different letters within each column indicate significant differences between fertilizer treatment (P<0.05) according to LSD test.*

[2] *Composting solid fraction of pig slurry.*

Digested pig slurry reduced denitrification losses and N_2O emission by 24% and 14% respectively, relative to untreated pig slurry (Table 1), but had no effect on NO emission. Petersen (1999) also observed a reduction of N_2O emissions by 20 to 40% from digested slurry applied to a Danish soil. CP treatment increased N_2O (by 14%) and NO (by 66%) emission compared with IPS, in spite of reducing denitrification losses (by 24%). The mineral fertilizer (Urea) had the highest N_2O and NO emission, although it also reduced denitrification.

DCD partially inhibited nitrification during the first 20 days and mitigated N_2O and NO emissions from pig slurry by at least 29 and 74%, respectively, but had no effect on denitrification losses. Vallejo *et al.* (2005) also observed a reduction of nitrogen oxides when DCD was mixed with pig slurry applied to an irrigated grassland soil.

Williams *et al.* (1998) used the relative emission of NO and N_2O as a potential method to distinguish between nitrification and denitrification of soil in situ. With the criteria of these authors, it can be said that denitrification rather than nitrification was the dominant process in the period before irrigation because the average of the molar ratio of NO-N to N_2O-N ranged from 0.003 to 0.053. Urea and CP had the highest values, whereas Urea+DCD had the lowest.

Conclusions

In soils with a low level of organic C, the addition of organic fertilizers mitigated N_2O and NO emissions relative to a mineral fertilizer (Urea). Anaerobic digestion treatment improved the quality of pig slurry as fertilizer and is an option to mitigate denitrification losses and N_2O emission, although no effect was observed on NO fluxes. The use of dicyandiamide mixed with untreated pig slurry is another option to mitigate N_2O and especially NO, although denitrification was not affected.

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Working group 4

Grassland renovation: prudent or risky?

Report of Working Group 4

Grassland renovation: prudent or risky?

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Introduction

This working group gathered to discuss aspects of grassland renovation in relation to nitrogen (N) emissions and the Water Framework Directive. Five participants had been asked to give a presentation of their work (E.M. Hansen, G.L. Velthof, J.G. Conijn, L. Bommelé and F. Vertès). After each presentation a short discussion followed and at the end of all presentations a more general discussion was held. The working group focussed on two important questions: (1) under which conditions (climate, soils, species) is grassland renovation an appropriate (or prudent) option for farmers and (2) which measures are effective in reducing the risk of N emissions after grassland ploughing. Before the five presentations started, the theme was briefly introduced by presenting some general background information on grassland renovation and its relation with N emissions.

Why grassland renovation?

For most dairy systems across the European Union (EU) home-grown feed is an indispensable resource. Especially in intensive dairy farms with high-yielding cows productive swards and arable crops, e.g. maize, are needed to produce at low costs. Farmers will therefore aim at sustaining the productivity of their fields at a high level suited for their farm situation. However, crop productivity may decline due to various reasons, some of which can hardly be influenced by the farmer. Swards may deteriorate by adverse weather conditions (frost or drought) and soil organic matter level on arable land may become too low for sufficient crop production. In addition, a farmer may want to (re)introduce clover into his grassland or remove persistent weeds or pests (Conijn *et al.*, 2002). Grassland renovation, in this report used for both grass-to-grass reseeding and grass-arable rotations, is often practised by farmers to overcome these situations and mostly includes ploughing of grassland before sowing grass or an arable crop. Crop productivity can be improved e.g. by using the best grass varieties at grass reseeding and by amelioration of the soil conditions during the grass phase of a grass-arable rotation system. If crop productivity is improved, N use efficiency, defined here as N output / N input, will likely to be improved as well from which not only the farmer but also the environment may benefit.
The extent at which grassland renovation is practised by farmers has been reported in Conijn *et al.* (2002) for North-west European countries and there it ranges from 2 to 10% per year of the total grassland area on a (sub)national scale and to even 20% per year of the grassland area on a regional or local scale (figures mainly based on grass seed sales).

Is there a problem?

Grassland renovation is one of the many management activities farmers perform in the execution of their profession and before starting a discussion on the relation between grassland renovation and N emissions, as in this working group, one would like to know whether there is a problem or not? What makes it worthwhile to discuss grassland renovation in the context of the Water Framework Directive? Most grassland renovation involves ploughing of

grassland, which usually causes an accumulation of inorganic N in the soil, because the release of N from fresh and old organic matter tends to exceed the uptake potential of the newly sown crop for some time after killing of the 'old' sward (e.g. Velthof and Hoving, 2004). This high N availability may be lost from the soil depending on the susceptibility of the soil to N loss and on weather conditions (such as a high or low precipitation surplus) and may then contaminate the atmosphere with N_2O or surface and ground waters with NO_3. Large N losses have indeed been reported after grassland ploughing (e.g. Adams and Jan, 1999; Shepherd *et al.*, 2001 and Springob, 2004). Combined with the extent at which grassland renovation is practised, there may be a problem with N emissions from ploughed grassland to the environment.

Society is concerned about N emissions to the environment, because it may threaten other functions of rural areas. In the EU this has lead to the definition of water quality goals (e.g. Nitrate Directive, Water Framework Directive). Legislation in EU countries has been developed to comply with these water quality goals and grassland renovation (among others) has drawn the specific attention as being a potential risk. In order to limit the N emissions, regulations have already been formed that restrict farmers' practice of grassland renovation in a number of countries. Examples are: N fertilization of arable crops on ploughed grassland should be lowered compared to the same crops grown on arable land (e.g. in Denmark) and grassland ploughing on sandy soils is only allowed during spring (e.g. in the Netherlands). These examples illustrate that grassland renovation is considered to give environmental problems if not regulated properly and that investigating the effects of grassland renovation on N emissions is important in relation with the Water Framework Directive.

Prudent or risky?

The question whether grassland renovation is prudent or risky with respect to N emissions, can not simply be answered uniformly for all situations. This has two main reasons: (1) in general, we face both a positive (higher N use efficiency) and a negative (higher inorganic N level in the soil) effect of grassland renovation on N emissions and (2) management choices related to grassland renovation have a large influence on the risk of N emissions. An example was given by Nevens and Reheul (2004) who concluded that a grass-maize rotation could save mineral N fertiliser and that growing fodder beet in the first year after ploughing could prevent excessive high residual soil N levels, unlike the situation with silage maize. Figure 1 illustrates a working hypothesis on the positive and negative effects that may occur around grassland renovation (Conijn & Taube, 2004). The overall net effect on yield and nutrient losses depends on soil type, climate conditions and management, and the consequence of this is that we have to look carefully at various situations of grassland renovation and define for each situation the conditions in terms of soil, climate and management whether renovation is prudent or risky. It is then important to have a long term view instead of focussing on one or two years after grassland ploughing. In many situations risks are highest on the short term, while advantages work at the long term, which means that a complete balance can only be made after analysis of a whole grass-to-grass reseeding cycle or grass-arable rotation in a farm context. In the five presentations of this working group various topics and measures were highlighted that are relevant for the evaluation of grassland renovation in relation with N emissions.

Figure 1. Hypothesised development of yield (continuous line) and nutrient losses (dotted line) before and after renovation of old grassland.

For grass yields: decreasing yields with grassland age, loss of production in the year of ploughing, increased yields in the first year(s) after resowing followed by higher production levels of the new sward relative to the old sward. Yield may be expressed in terms of dry matter, nitrogen, protein or metabolic energy and refers to net yield or net intake.

For nutrient losses: increasing losses with grass sward ageing, a high risk of losing nutrients shortly after ploughing, lower emissions in the first years after resowing followed by lower levels of nutrient losses relative to the old sward.

Presentations and discussion

In the first presentation Hansen (Hansen *et al.*, 2006) dealt with the effects of cultivation of grazed grass clover swards on coarse sandy soils on N leaching. They tested the effectiveness of an early catch crop (Italian ryegrass) undersown into a barley crop used as a green crop and cut twice for forage production during late summer/autumn. The control treatment was a barley crop used as a mature crop and followed by mechanical weed control in late summer/ autumn. Leaching losses of N were measured during the following winter using ceramic suction cups. The catch crop treatments resulted in low N losses (7 - 9 kg N ha^{-1}) while the control treatments caused very high losses ranging from 174 to 316 kg N ha^{-1}. It was concluded that catch crops established via undersowing are a very effective measure in order to reduce N losses following reseeding of short or mid term clover grass swards. Additional forage production as well as carbon (C) storage are positive trade off effects.
In the second paper Velthof presented results of an experiment dealing with the effects of grassland resowing procedures on gross productivity and on N losses via leaching and via nitrous oxide emission due to soil type and time of cultivation. The hypothesis of increasing dry matter yields following grassland resowing was not confirmed by the presented multi-site experiment. The results showed that risk of N leaching is much higher when grassland is renovated in autumn than in spring. Risk of nitrous oxide emission was high both at spring and at autumn renovation. The intensity of soil cultivation (direct drilling without tillage versus resowing after ploughing) had a minor effect on N losses. It was concluded that the relevance of both investigated pathways of N losses have to be taken into consideration in order to evaluate the effects of soil type and time of resowing on negative environmental consequences in a well balanced way.
The model N*fate* was presented by Conijn (Conijn, 2006) aiming to calculate the short and long term effects of grassland resowing in Dutch agriculture. N*fate* is a dynamic model using yearly time steps to calculate N yield, N losses and the change of N in the soil/plant system as a function of N inputs, soil/climate characteristics, crop species and management. The model was calibrated with short term data. Simulation showed a reliable prediction of N losses due to grassland renovation in autumn and spring via leaching and highlighted the differences between

short and long term effects. Due to the year step structure of the model it can be used to simulate the whole cycle from resowing to resowing over a long term.
Bommelé (Bommelé *et al.*, 2006) highlighted the effects of rotocultivation of young and old grassland on N delivery in the succeeding crop (potatoes). Average additional net N from mineralization following cultivation of both grassland types compared to an arable rotation ranged between 232 kg ha^{-1} in the first succeeding crop and 144 kg ha^{-1} in the second succeeding crop. This indicates that net mineralization following cultivation of grassland is much higher than often documented in the literature, especially in the second year after cultivation. Bommelé concluded from their experiment that growing potatoes after grassland caused high soil mineral N residues in autumn, even in the non-fertilized treatment.
Finally, Vertès (Vertès *et al.*, 2006) focussed on the long term effect of fodder crop rotations on soil organic matter quality using a long term data set covering 30 years of measurements. Six rotations covering a range of grass/maize ratios were compared indicating that the grass/maize ratio was a powerful driving force in order to understand soil C and N dynamics in a long term. Organic N content of the soil was only remaining constant when at least three years of grass were combined with one year of maize, even if organic inputs were taken into consideration. However, soil organic N content did not fully explain the differences in N mineralization rates between the rotations. The ratio N mineralization rate/total N content increased with the grass/maize ratio indicating that changes in soil organic matter quality, viz. distribution of C and N among various soil organic matter fractions, also influenced the mineralization rate.

The general discussion was highlighting the links between the different topics presented in this working group. It was evident that grassland cultivation and grass-arable rotations cause different results in terms of N release and N losses related to arable rotations without grass crops due to a wide range of accumulation of C and N in the soil under grassland and ley systems as well. The huge variation in N losses due to grassland cultivation is due to a wide range of soil properties, weather conditions and management options covered by the presented experiments. In order to generalize the presented results methods were discussed allowing a prediction of C and N fluxes following grassland renovation. The group agreed that multi-site experiments with a common protocol would be a powerful tool in order to calibrate dynamic models simulating C and N fluxes following grassland renovation.

Conclusions

Grassland renovation and grass-arable rotations are of major concern regarding the economic benefits of dairy farms as well as regarding the consequences for the environment. Time of grassland ploughing and choice of the following crop(s) are effective ways to manipulate nutrient losses. The hypothesis of increasing grass yields following grassland reseeding was not confirmed by the results presented in the working group, which is in line with results from the literature. As a consequence the focus should be switched to measures maintaining permanent grassland performance without killing of the grass sward. On the other hand grass-arable rotations were identified as a promising production system, but questions still remain to be resolved with respect to the nutrient use efficiency of the whole system in relation to soil and climatical conditions. The ratio of grass in grass-arable rotations is a key issue in order to maintain soil fertility from which the arable crop may benefit. Dynamic models are a powerful tool in order to simulate consequences of grassland renovation and grass-arable rotations at different sites and due to different management options, but more data are needed for calibrating such models.

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Poster presentations

Different methods for quantifying actual denitrification for a permanent and temporary grassland

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Abstract

Nitrogen emissions from agricultural soils to the surrounding environment are controlled by field management and physical conditions in the soil. Measuring denitrification under field conditions is difficult and expensive. This study compared several methods to quantify denitrification for a wet and a dry sandy soil with grass: a balance method, a (complex) deterministic model, a simplified process model and measurements. The first three methods were either used with generic data or with plot-specific data. Most methods show that denitrification is highest in the permanent grassland with shallow groundwater levels (wet field). Taking into account plot-specific circumstances (weather, N surpluses) results in different estimates for denitrification in comparison to the generic input data. For the dry field, temporary grassland with deep groundwater levels, the differences between the estimated denitrification were the largest. The best suitable method probably depends on the available data, but should be as plot-specific as possible.

Keywords: denitrification, grassland, leaching, modelling, nitrate

Background and objectives

Nitrogen (N) emissions from agricultural soils to the surrounding environment are controlled by N application rates, field management and physical conditions in the soil. Measuring denitrification under field conditions is difficult and expensive due to temporal and spatial variations and methodological difficulties. The objective of this study is to compare different methods to quantify total denitrification under field conditions. The difference between the total amount of N emission and the total denitrification is an estimate for nitrate leaching, as there is a general trade-off between N losses.

Material and methods

We present the results of a desk study focusing on different modelling approaches, supplemented with a variety of field and laboratory measurements as input to different models. The field measurements were carried out on two fields on the experimental farm 'De Marke' in the Netherlands. Field A, a permanent grassland, was situated on a sandy soil with shallow groundwater levels during winter (Mean Highest Groundwater level (MHG) between 25 and 40 cm below soil surface (-ss)). Field B, a temporary grassland (ley), was situated on a dry sandy soil (MHG below 140 cm -ss) and is considered to be vulnerable for nitrate leaching.

During the winter of 2004-2005 soil samples were taken from six layers of each field plot. The soil samples were used to determine potential denitrification (Van Beek *et al.*, 2004), actual denitrification using isotope pairing (Arah, 1992), mineral nitrogen content, bulk density and volumetric water content. During this period groundwater levels, precipitation, and air temperature were monitored. Nitrate concentrations in groundwater were measured once.

The used methods are described in Table 1. They were applied to calculate the total denitrification for each plot (period March 2004 - March 2005; soil layer 0 - 1 m -ss).

Table 1. Description of the seven methods used to compute denitrification.

Method	Description
1	The balance method: denitrification and nitrate leaching were calculated using representative values for nitrogen surpluses, leaching fractions and precipitation surpluses following Schröder *et al.* (2004).
2	The plot-specific counterpart of (1) uses the actual nitrogen and precipitation surpluses and measured nitrate concentrations (in spring).
3	Denitrification was taken from the Dutch national scale model STONE (Wolf *et al.,* 2003) for two STONE plots resembling the two fields of 'De Marke'.
4	With SWAP (Kroes and Van Dam, 2003) and ANIMO (Groenendijk and Kroes, 1999) (both incorporated in STONE) plot-specific denitrification was computed by adapting input of (3) to site-specific conditions of the two plots.
5	A widely used simplified denitrification process model (Heinen, 2005a,b) was used to calculate actual denitrification based on potential denitrification and three reduction functions for nitrate content, degree of water saturation and soil temperature. The required time series (per decade) for nitrate content, degree of saturation and temperature came from method (4).
6	Instead of using a standard parameter valueset (as in (5)), plot-specific parameter values were used in the simplified denitrification process model. The plot-specific parameter values were estimated from the measurements in the soil samples.
7	The last method calculates the total denitrification by integrating the measured actual denitrification over depth and time.

Results and discussion

The calculated denitrification per plot, using methods (1) - (7), is presented in Figure 1.

Figure 1. Total denitrification in kg N ha^{-1} for the permanent (A) and temporary (B) grassland using different methods. (1) balance method based on Schröder et al. (2004), (2) plot-specific balance method, (3) STONE model for representative plots, (4) STONE model with plot-specific input, (5) simplified denitrification process model with representative parameter values, (6) simplified denitrification process model with plot-specific parameter values, (7) integration of denitrification measurements over time and depth. (*) Method (7) is applied over a shorter period, i.e. 120 days instead of 365 days.

Results of the first balance method (1), both STONE methods (3,4), and both simplified process models (5,6) show that denitrification is highest in field A. Field A is the wettest field of both fields considered, which is advantageous for denitrification to occur. Furthermore, field A is a permanent grassland, which generally implies higher N applications and higher amounts of N stored in the soil. The plot-specific balance method (2) indicates that denitrification in field B is comparable or even higher than in field A. This can be explained by the difference in N surplus in the considered period. The N surplus of field B is almost 30 kg ha^{-1} higher than the N surplus of field A. Furthermore, March 2004 – March 2005 was a relatively dry period with 719 mm precipitation (the 30-year average precipitation for this region is 750 - 775 mm).

The differences between the calculated denitrification of the generic methods (1) and (3) and the plot-specific methods (2) and (4) are mainly caused by drier than average circumstances and by the differences in N surpluses. The used plot-specific parameter values in (6) are the result of a poor fit, so results of this method should be judged with care.

The measured denitrification of field A is the lowest. However, the measured period is a third of a year. We expect more denitrification to occur in field A than in field B during the remainder of the considered year. This expectation is confirmed by modelling results of method (3) and (4). We could not find a good explanation for the low measured denitrification of field A. The samples of field A were usually a bit warmer (except last sampling) and had a higher moisture content. The nitrate contents in the first two layers were highest in field B. In the other layers nitrate contents were highest in the samples of field A.

Conclusions

The average denitrification of methods (1) - (6) for field A and B is 99 (± 27) and 39 (± 37) kg N ha^{-1}, respectively. The variation between the different methods is high. Most methods show that denitrification is highest in the wet permanent grass field A despite the lower N surplus with respect to the dry temporary grass field B. Taking into account plot-specific circumstances (weather, N surpluses) results in different estimates of denitrification. The best suitable method probably depends on the available data, but should be as plot-specific as possible.

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Influence of preceding crop on the clover content in mixed swards under cutting management

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Abstract

The soil N-supplying capacity influences the establishment of ryegrass-white clover mixtures (Loiseau *et al.*, 2001). Our experiment focuses on the clover content in a newly established grass-white clover sward (simulated grazing management). Swards were installed in 2002 after different preceding crops: 35 year old rotocultivated permanent arable land (PA) and grassland (PG), temporary arable land (TA, 35 year-rotation of 3 years arable land-3 years grassland) and temporary grassland (TG, rotation as TA).

Dry matter (DM) yield and clover content of the new, unfertilised swards were determined in each cut during 2002-2004 and related to the net N from mineralisation. The N mineralisation of soil organic matter was studied on fallow plots. After arable land the net N mineralisation was less than 60 kg N ha^{-1}; after TG it was 169 and 145 kg N ha^{-1} in 2002 and 2003, respectively, and after PG it was 283 and 226 kg N ha^{-1}. The clover content declined from 77% (2002) to 65% (2003) after arable land; it increased from 31% (2002) to 51% (2003) after grassland. The N uptake was higher after arable land than after grassland. Residual soil N of all swards was extremely low (<25 kg NO_3-N ha^{-1}). Clover establishment and persistence was higher after arable land.

Keywords: clover, grassland, mineralization, nitrogen, nitrogen uptake, preceding crop

Background and objectives

According to Loiseau *et al.* (2001), the risk of clover extinction may be avoided by installing grass-clover swards on soils with a poor N supplying capacity. Our experiment focuses on the effect of different preceding crops on the N supplying capacity of the soil, the yield, and the performance and evolution of the clover content in newly established ryegrass-white clover swards.

Material and methods

The experiment was conducted on a sandy loam soil at the experimental farm of Ghent University in Melle (Belgium, 11 m asl). During spring 2002 a mixture of perennial ryegrass (*Lolium perenne* L., cvs. 'Plenty' and 'Roy', 40 kg ha^{-1}) and white clover (*Trifolium repens* L., cv. 'Huia', 4 kg ha^{-1}) was sown in both rotocultivated arable land and in grassland. Part of the arable land had been arable land for 35 years (PA), part of it had been temporary arable land (TA) during 35 years in a rotation: 3 years grassland-3 years arable land. Part of the grassland had been permanent grassland for 35 years (PG) and part of it had been in a 35-year rotation of 3 years arable land-3 years grassland (TG). The swards were cut according to a simulated grazing management. During 2002, 2003 and 2004, DM yield and clover content were determined in each cut. Part of the seed bed was left uncropped in order to study the N mineralisation of the soil organic matter: N mineralisation was estimated as the difference between the soil N content (0-90 cm profile) at the beginning and the end of the growing season. DM yield and clover content were related to the mineralised N.

Results and discussion

Dry matter yield (DMY) was significantly lower after TG during 2002 and 2003 (Table 1). PA significantly outyielded the other preceding crops in 2003. During the period 2002-2004, the overall lowest yields were obtained after TG. If only the two full harvest years 2003 and 2004 are considered, the same trend can be seen: TG has the lowest total yield during these two years. In 2002 and 2003, N uptake was significantly lower on TG swards than on PA swards (Table 1).

The net N mineralisation, measured on fallow plots, was 43, 249 and 253 kg N ha^{-1} in 2002 and 60, 182 and 226 kg N ha^{-1} in 2003 on PA, TG and PG, respectively. The extra net N mineralisation on TG and PG compared to PA, was 206 and 210 kg N ha^{-1}, respectively, in 2002 and 122 and 166 kg N ha^{-1} in 2003.

On the one hand, TG yields less and has a low N uptake, on the other hand it has a high extra net N mineralisation in comparison to PA. This is expected to result in a very high residual soil N content, which is confirmed by our observations (Table 1).

Annual mean clover content was significantly higher on PA swards than on TG or PG in all three years (Table 1). The evolution of the clover content is shown in Figure 1. During the year of establishment, PA and TA had the significantly highest clover content, probably due to the low net N mineralisation, favouring clover development. Clover content continued to be highest in PA during 2002 and 2003. Clover content after grassland (both TG and PG) increased during the first full harvest year. All clover contents declined during the second full harvest year.

Table 1. Grass-clover swards with different preceding crops (PA: permanent arable land, TG: temporary grassland, PG: permanent grassland, TA: temporary arable land): annual DM yield (DMY, kg ha^{-1} y^{-1}), annual N uptake (kg ha^{-1} y^{-1}), residual soil N (kg ha^{-1}) and annual mean clover content (MCC, % on dry matter basis) in the period 2002-2004. Duncan letters indicate significant differences within columns.

Preceding crop	2002	2003	2004	Sum (2003+2004)	Sum (3 years)
	DMY (kg ha^{-1})				
PA	7981^{b}	12998^{c}	10980^{a}	23978	31959
TG	6289^{a}	9824^{a}	10977^{a}	20801	27090
PG	9191^{b}	11430^{b}	10351^{a}	21781	30972
TA	8175^{b}	11909^{b}	11222^{a}	23131	31306
	N uptake (kg ha^{-1})				
PA	303.3^{b}	476.3^{c}	295.3^{a}	771.6	1074.9
TG	170.5^{a}	324.5^{a}	288.3^{a}	612.8	783.3
PG	308.0^{b}	383.0^{b}	311.5^{a}	694.5	1002.5
TA	299.3^{b}	414.8^{b}	314.8^{a}	729.6	795.1
	residual soil N (kg ha^{-1})				
PA	27.3	65.8	56.4		
TG	180.6	203.8	94.0		
PG	297.0	234.9	73.1		
TA	19.4	93.0	52.2		
	MCC (%)				
PA	77.4^{c}	74.0^{b}	37.2^{b}		
TG	20.2^{a}	50.0^{a}	24.1^{a}		
PG	41.3^{b}	52.9^{a}	23.5^{a}		
TA	76.7^{c}	55.9^{a}	25.8^{a}		

Figure 1. Evolution of the clover content (% in DM) (PA: permanent arable land, TG: temporary grassland, PG: permanent grassland, TA: temporary arable land).

Conclusions

Newly installed grass-clover swards perform best when installed in arable land: they had the highest yield and the highest clover content and persistence. Furthermore, swards on arable land had the lowest residual soil N values. Because of the low yields combined with the high soil N residues, installing grass-clover after temporary grassland is not recommended.

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Nitrogen delivery after rotocultivated old and young grassland

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Abstract

Rotocultivated young grassland releases high amounts of nitrogen (Nevens and Reheul, 2002). Due to the increasing potato production in Belgium and the Netherlands, we wondered whether potato is recommendable as a first crop following grassland. Nitrogen delivery by permanent arable land (PA) and converted grassland was estimated and related to potato yield, N-uptake and residual soil N. Potato yield (fresh and DM yield) was determined at three N dressings on plots with PA, permanent (PG) and temporary grassland (TG) as preceding crops. Yield on unfertilised PA was significantly lower during two subsequent years. N mineralisation was studied on fallow plots. The additional net N from mineralisation compared to PA in the first and second year after TG, was 206 and 122 kg N ha^{-1}. Following conversion of 35- (PG) and 36-year old (PG*) permanent grassland, the additional net N was 210 and 307 kg N ha^{-1} respectively. The second year after PG the extra net mineralisation was 166 kg N ha^{-1}. On PA plots, the N-uptake by potatoes exceeded the net N mineralisation; plots after grassland showed the inverse, resulting in a higher residual soil N as compared to PA.

Background and objectives

Three-year old rotocultivated grassland releases high amounts of nitrogen (Nevens and Reheul, 2002) and the best crop to capture this released nitrogen is a long growing crop as fodder beet. Because of the increasing area of potato production in Belgium and the Netherlands, the question rises whether potato could also be recommended as a first crop following the destruction of grassland.

Material and methods

The experiment was established on a sandy loam soil at the experimental farm of Ghent University in Melle (Belgium, 50°59' N, 03°49' E, 11 m asl).

In spring 2002, potatoes were planted after forage maize grown in permanent arable land (PA) and in rotocultivated grassland (G). Part of the grassland was 35-year old grassland (PG), the other part had been in a rotation of 3 years arable land-3 years grassland (TG). A part of the seed bed was left uncultivated to study the N mineralisation of the soil organic matter. The net N mineralisation was calculated as the difference between the soil mineral N content of the fallow plots at the beginning and the end of the growing season.

In spring 2003, potatoes were planted on the same plots as in 2002 (this means two subsequent potato crops in the same field) and in rotocultivated 36-year old permanent grassland (PG*). Fresh tuber yield was determined at the end of the growing season at three different N dressings (0, 75 and 200 kg N ha^{-1}). This allowed the calculation of the yield response by Mitscherlich curves. At the beginning (April) and the end (October) of the growing season the mineral N content of the soil profile (0-60 cm) was determined in the unfertilised potato plots and in the fallow plots. Dry matter- and N content (Dumas method) and quality of the potatoes were determined by estimating the scab infestation and the under water weight (UWW). The net N-uptake by the potato tubers was estimated by multiplying potato DM yield by the N content.

To determine the residual mineral soil N, soil samples (sampled monthly until November) were extracted with KCl: ammonia and nitrate were determined colometrically using a continuous flow analyser.

Results and discussion

The soil mineral N content (0-60 cm) of fallow and 0N plots in 2002 increased from April till October and decreased later on. In 2003 N dynamics differed between fallow and 0N plots. On PG* plots for example, the mineral N content of fallow plots increased during May-August 2003 and decreased again from October while the N content of unfertilised PG* plots increased during July-November.

The net N mineralisation during the period April-October 2002 was 43, 249 and 253 kg N ha^{-1} on PA, TG and PG respectively; in 2003 values were 60, 182 and 226 kg N ha^{-1} respectively and on PG* it was 367 kg N ha^{-1}. Compared to PA, the extra net N from mineralisation in the first and second year after TG was 206 and 122 kg N ha^{-1} respectively; in the first year after PG and PG* it was 210 and 307 kg N ha^{-1} respectively while the second year after PG the extra net mineralisation was 166 kg N ha^{-1}.

The fresh tuber yield and the dry matter yield on plots with different preceding crop and N dressings, are given in Table 1.

Table 1. Fresh tuber yield and dry matter yield of the potato crop on plots with different history; (PA: permanent arable land, TG: temporary grassland, PG: permanent grassland (35 y.), PG: permanent grassland (36 y.). Duncan-letters indicate significant differences within columns.*

Preceding crop	Fresh tuber yield (Mg ha^{-1})						DM yield (Mg ha^{-1})					
	0 kg N ha^{-1}		75 kg N ha^{-1}		200 kg N ha^{-1}		0 kg N ha^{-1}		75 kg N ha^{-1}		200 kg N ha^{-1}	
	2002	2003	2002	2003	2002	2003	2002	2003	2002	2003	2002	2003
PA	30.8^{a}	22.4^{a}	53.3^{a}	43.0^{a}	65.9^{b}	56.6^{b}	6.9^{a}	4.6^{a}	11.0^{a}	9.4^{a}	13.1^{b}	12.4^{b}
TG	60.0^{b}	36.4^{b}	60.1^{a}	45.6^{a}	66.8^{b}	59.3^{b}	13.0^{b}	8.3^{b}	12.9^{b}	10.4^{a}	13.0^{ab}	13.4^{b}
PG	53.1^{b}	38.1^{b}	60.3^{a}	43.9^{a}	55.0^{a}	46.8^{a}	11.3^{b}	8.7^{b}	12.4^{ab}	10.1^{a}	11.3^{a}	10.7^{a}
PG*	/	55.4^{c}	/	56.7^{b}	/	55.1^{b}	/	12.5^{c}	/	13.2^{b}	/	12.8^{b}

First year after G: a nitrogen dressing after G did not increase potato yield. Potatoes grown in G outyielded potatoes grown in PA; yield differences in TG and PG did not differ significantly. The advantage of G disappeared at 200N. During the second year, G continued to outyield PA at 0N but differences between G and PA were no more significant at 75N. G amended with 0 or 75N could not substitute the effect of 200N on PA.

The influence of preceding crop and N dressing on potato quality was compared by means of the scab index (mean percentual tuber coverage by scab) and the under water weight (UWW). The N dressing did not influence scab infestation. In 2002 there was significantly more scab between TG or PG and PA: potatoes grown after grassland were significantly more infested by scab: PG>TG>PA. The UWW on PA and TG decreased significantly with increasing N dressing in 2002 and UWW on PA was significantly higher than on PG (no significant difference in 2003).

The N-uptake by the potato crop is given in Table 2. In 2002, N-uptake on PA was significantly lower than on TG and PG. In 2003 overall N-uptake was lower than in 2002; in 2003 the highest N-uptake at each N dressing was found on PG*. On PA plots, the N-uptake by potatoes exceeded the net N mineralisation; on plots after grassland the inverse was seen, resulting in a higher residual soil N as compared to permanent arable land.

Table 2. Net N uptake by potato crop on fields with different preceding crop and N dressing (0, 75 and 200 kg N ha^{-1}) in the first and second year after planting. (PA: permanent arable land, TG: temporary grassland, PG: permanent grassland (35 y.), PG: permanent grassland (36 y.)*

Preceding crop	0 kg N ha^{-1}		75 kg N ha^{-1}		200 kg N ha^{-1}	
	2002	2003	2002	2003	2002	2003
PA	51^{a}	48^{a}	111^{a}	103^{a}	199^{a}	200^{b}
TG	147^{b}	79^{ab}	183^{b}	128^{ab}	258^{c}	229^{c}
PG	157^{b}	103^{b}	205^{b}	136^{b}	230^{b}	173^{a}
PG*	/	169^{c}	/	212^{c}	/	235^{c}

The residual soil mineral N is given in Table 3.

Table 3. Residual soil nitrate-N (kg ha^{-1}, profile 0-90 cm) at the end of the growing season (2002 and 2003) after harvest of potato crop, grown on plots with different history (PA: permanent arable land, TG: temporary grassland, PG: permanent grassland (35 y.), PG: permanent grassland (36 y.) and fertilised with three different N dressings (kg N ha^{-1}). Duncan letters indicate significant differences within columns.*

Preceding crop	2002			2003		
	0 kg N ha^{-1}	75 kg N ha^{-1}	200 kg N ha^{-1}	0 kg N ha^{-1}	75 kg N ha^{-1}	200 kg N ha^{-1}
PA	16^{a}	25^{a}	24^{a}	36^{a}	53^{a}	73^{a}
TG	49^{a}	62^{ab}	102^{b}	96^{b}	75^{a}	122^{b}
PG	77^{b}	90^{b}	153^{b}	129^{b}	112^{b}	113^{b}
PG*	/	/	/	107^{b}	93^{ab}	165^{c}

In 2002, residual soil N in the 0-90 cm profile was significantly higher on PG than on PA. In 2003 soil residual N is significantly lower on PA than on PG or PG* plots.

Conclusions

Nitrogen dressing in potatoes planted in ploughed down grassland does not increase potato yield. Growing potatoes after grassland is not recommended, due to high scab incidence and high soil mineral N residues.

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The nitrogen and non-nitrogen-contribution effect of ploughed grass leys on the following arable forage crops: determination and optimum use. European Journal of Agronomy, 16, 57-74.

Simulated short and long term effects of grassland reseeding on nitrate leaching

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Abstract

Intensively used grassland in the Netherlands is frequently reseeded by farmers to improve sward productivity, which may increase the risk of nitrate leaching. A simulation study has been conducted to determine both short and long term effects of grassland renewing on nitrate leaching. With the model N*fate* three 'treatments' have been simulated: no, spring (April) and autumn (September) reseeding and all major N fluxes at field level have been calculated for each year in a reseeding cycle of seven years. During the first year of the reseeding cycle the N leaching at autumn reseeding is strongly increased, but averaged over seven years N leaching is nearly equal to that of no reseeding, because lower leaching losses during the remaining six years of the reseeding cycle compensate for the higher losses of the first year. At spring reseeding N leaching losses are always below that of no reseeding, even during the first year, accumulating on average in a 15% lower N loss by leaching. Results make clear that short and long term effects of the whole reseeding cycle should be evaluated together.

Keywords: grassland, leaching, modelling, nitrate, nitrogen uptake, reseeding

Background and objectives

Intensively used grassland in the Netherlands is frequently reseeded by farmers to improve sward productivity (Conijn *et al.*, 2002), which often involves ploughing up grassland before resowing. In this situation the risk of nitrate leaching is enhanced, because a surplus of inorganic N usually occurs in the soil during the period between killing of the old sward and full establishment of the new sward. On the other hand, the increased productivity of the new sward increases the nitrogen use efficiency and may therefore reduce nitrate leaching. A simulation study has been conducted to determine the overall effect by calculating both short and long term effects of grassland renewing on nitrate leaching, including the influence of the time of ploughing.

Material and methods

The model N*fate* (*N*utrient *f*luxes in *a*gricultural soils and *t*o the *e*nvironment) has been used for this study. N*fate* calculates N yield, N losses and changes in the amount of N in the soil-crop system as a function of N inputs, soil/climate characteristics, crop species and management. The in/outputs of the model apply to the field level and refer to a whole year. N*fate* is a dynamic model, which means that the amount of N in the system may change after one time step (i.e. a year), which has consequences for the calculations in the next time step. In the model three N pools are distinguished (Table 1); Figure 1 pictures the major N fluxes that are used/calculated by the model. An outline of the model is given in Conijn (2004).

Table 1. Description of the N pools in Nfate (see Figure 1).

Name	Description	Details	Unit
NorgSoil	Amount of organic nitrogen in the soil organic matter	Age > 1 year, soil layer: 0-20 cm	kg N ha^{-1}
NminSoil	Amount of inorganic nitrogen in the soil	Rooted soil layer, measured in early spring	kg N ha^{-1}
Nplant	Amount of nitrogen in the plant	Below- and aboveground living plant tissue	kg N ha^{-1}

Figure 1. N fluxes and N pools in the soil-crop system of Nfate. Res is used as abbreviation for plant residues, which includes dead plant parts and field harvesting losses.

In this study the model is first used to calculate the equilibrium situation of permanent grassland on a dry sandy soil in the Netherlands with a mixed use of grazing and cutting. Level of N inputs has been derived from the proposed Dutch legislation for dairy farms in 2009. Three 'treatments' have been simulated with N*fate*: no, spring (April) and autumn (September) reseeding, where reseeding occurs once every seven years. Reseeding includes killing of the 'old' sward, which sets *Nplant* to zero by distributing the N content of *Nplant* among *NminSoil* and *NorgSoil*, and a gradual built-up of *Nplant* again up to the level of the 'old' sward during the years after reseeding.

Results and discussion

In the equilibrium situation the N pool sizes amount to circa 4700, 30 and 170 kg N ha^{-1} for *NorgSoil*, *NminSoil* and *Nplant*, respectively. Organic matter content in the upper 20 cm of the soil is 4.6%. Net grass yields equal 9.9 tonne dry matter ha^{-1} y^{-1} and 289 kg N ha^{-1} y^{-1}. Total N input and loss are 428 and 138 kg N ha^{-1} y^{-1}, of which 48 kg N is lost by leaching, resulting in a nitrate concentration of 59 mg l^{-1} in the groundwater. Results compare reasonably

well with those from Schröder *et al.* (2005), who determined a yield of 285 kg N ha^{-1} y^{-1} at a total input of 404 kg N ha^{-1} y^{-1}, as average values of (sub)optimal conditions.

In Figure 2 the net N yield and amount of leached N is given for each year during the reseeding cycle, including the average over the whole period for the three reseeding 'treatments'. In the first year, both spring and autumn reseeding cause a drop in N yield relative to no reseeding, because it takes some time before a new sward is productive. Thereafter, the reseeded swards show higher net N yields (at equal N inputs) due to lower losses. Overall, the N yields of the reseeded swards, averaged for the whole 7-year period, differ only slightly from the non-reseeded sward. The amount of N leached during the first year at autumn reseeding is remarkably higher compared to the two other 'treatments'. It illustrates the mismatch between the release of N (partly from the 'old' sward) and the N uptake by the new sward during a period with a high precipitation surplus. On the other hand, spring reseeding is usually followed by a period of precipitation shortage together with a higher N uptake capacity by the new sward, which leads to lower N leaching losses. These results agree with the experimental findings of Velthof and Hoving (2004). After the first year calculated leaching levels of reseeded swards are lower than those from the non-reseeded sward because (a) N is immobilized by uptake into the roots and stubbles of the new sward and (b) more N is removed from the reseeded swards. Overall, averaged over seven years, levels of N leaching are similar for no reseeding and autumn reseeding, whereas it has decreased with 15% after spring reseeding.

Figure 2. Calculated net N yield and N leaching during a reseeding cycle of seven years. Ploughing and reseeding takes place in April or September of year 1. N fluxes refer to the period between 1 March of year x until 1 March of year x+1. Bars at the right hand side (beyond year 7) represent the average values for the 7-year period.

Conclusions

The model N*fate* predicts for grasslands on dry sandy soils in the Netherlands that during the first year of the reseeding cycle spring reseeding causes no increase in N leaching, but autumn reseeding more than doubles the loss of N by leaching compared to no reseeding. However, averaged over the whole reseeding cycle, the results indicate that N leaching is reduced at spring reseeding, whereas at autumn reseeding it is nearly equal to that of no reseeding, despite the relative high losses in the first year. These results make clear that a whole cycle (from reseeding to reseeding) should be evaluated for an adequate estimation of the effect of grassland reseeding on total nitrate leaching.

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Simulation of nitrate loss by denitrification and leaching from grassland under cutting

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Abstract

High nitrogen (N) fertilisation rates are applied to grassland of dairy farms in the Netherlands, which causes high N emissions to the environment. Effective ways of controlling these emissions were investigated with a detailed simulation model for grassland management, water and soil C and N dynamics. Various N fertilisation strategies have been explored for a specific sandy soil in the Netherlands, including the effects of irrigation on nitrate leaching, grass yields and denitrification. Lowering the mineral N fertilisation in the non-irrigated situation was effective: grass dry matter yield was reduced by 5% only, whereas the nitrate concentration dropped by 43% which was needed to realise the target of 50 mg nitrate l^{-1} on average. Irrigation in this study also had a large decreasing effect on nitrate leaching and can be very helpful in controlling nitrate leaching on intensively used grasslands that (occasionally) suffer from drought conditions. Denitrification was responsible for most (> 70%) of the nitrate loss by denitrification and leaching together. Denitrification was strongly reduced when mineral N fertilisation was lowered, but increased if irrigation was applied. N_2O emission from the soil surface was positively related to mineral N fertilisation and it was concluded that the selected measures for controlling nitrate leaching in this study are also very effective in decreasing N_2O emission from sandy soils.

Keywords: denitrification, grassland, leaching, modelling, N_2O, nitrogen, yield,

Background and objectives

Most of the grassland on dairy farms in the Netherlands is used intensively, for which high N fertilisation rates are needed. Besides high grass yields, this causes high emissions of N to the environment (atmosphere, surface water and ground water). Related problems due to these high emissions are e.g. the eutrophication of natural waters, increased greenhouse gas effects and a poor drinking water quality. Society is concerned about these effects and seeks effective ways to control the emissions. Lowering the N input level is one of the strategies to reduce emissions to acceptable levels. A simulation study for a specific soil type has been conducted to answer the following questions:

1) how to fertilise grassland in order to limit the N concentration at a specified depth to 50 mg nitrate l^{-1},
2) what are the consequences of less N fertiliser for grass yields and
3) what is the effect on denitrification and the N_2O emission to the atmosphere?

Material and methods

Detailed dynamic models for grassland management, water and soil C and N dynamics (*CNGRAS*, see Conijn, 2005 and *FUSSIM2*, see Heinen and De Willigen, 2001) have been coupled to explore various fertilisation strategies and their effects on grass yield, nitrate leaching and N_2O emission. For the simulation study a sandy soil has been selected with an organic matter content of 4.1% in the top soil (0-20 cm). The soil can be characterised as moderately moist with mean highest and lowest groundwater level of -10 cm (winter) and -125 cm (summer), respectively. Only vertical movement of water and solutes through the soil has been simulated in this study. The grass sward (100% *Lolium perenne* L.) was cut approx. 5 times per year (no grazing). Total N fertilisation consisted

of a basic application of slurry (total N of 245 kg ha^{-1} y^{-1}) and a variable amount of mineral fertiliser (0 – 220 kg N ha^{-1} y^{-1}), split into 3-5 applications per year. Two choices were made with respect to irrigation: zero irrigation and optimal irrigation to prevent growth reduction due to drought stress. Other input parameters apply to average farm conditions in the Netherlands (Conijn and Henstra, 2003). Calculations were performed for a period of one year using 15 separate years with different weather conditions to determine mean yearly values. Calculated N leaching and nitrate concentration apply to the net fluxes of water and N across the soil boundary at 100 cm depth. Denitrification is given as total N loss via N_2 and N_2O from the whole soil profile (0–350 cm) and N_2O emission refers to the net emission from the soil surface into the atmosphere.

Results and discussion

Results of this simulation study have been compared with data from a number of experiments. Figure 1 shows the comparison with an experiment from a similar site, but with slightly different management and soil/weather conditions. The dotted line has been derived from the data of the experiment, averaged over a period of five years (Van der Meer, 2000), and the data points represent the average results of the non-irrigated model calculations for 15 years. All data points lie close to the experimental line, which gives confidence in the model ourcome.

Figure 1. N leaching as function of N fertilisation. Results of this study (markers: 'Model') are plotted together with results of an experiment conducted in 1980-1984 (dotted line: 'Experiment').

Table 1 contains a summary of the calculated results on agronomic and environmental key data of cut grassland of this sandy soil in the Netherlands (more results can be found in Conijn and Henstra, 2003). In the non-irrigated situation and with the recommended N fertilisation level ('economic optimum'), the average nitrate concentration in the percolation water at 100 cm soil depth exceeded the target of 50 mg nitrate l^{-1} by 74%. By lowering the mineral N fertilisation by circa 110 kg N ha^{-1} y^{-1}, the target could be realised, at least on average. Total N loss (via denitrification and leaching) then dropped from 132 to 85 kg N ha^{-1} y^{-1} and the N_2O emission was reduced by 40%. Calculated reductions of grass yields were relatively small (-5% for dry matter and -11% for nitrogen), which is consistent with the results of Aarts *et al.* (2005). In the irrigated option irrigation was only needed in 10 out of 15 years with a mean application of 57 mm y^{-1}. Irrigation was very effective in reducing nitrate leaching: the nitrate concentration exceeded the target by only 11% if the recommended fertilisation was combined with optimal irrigation. To bring the nitrate concentration further down to 50 mg l^{-1}, the N fertilisation should be reduced by 57 kg ha^{-1} y^{-1} (approximately half of the reduction that was necessary in the non-irrigated situation). In this situation the loss of N by denitrification was enhanced (+ 21 kg N ha^{-1} y^{-1} compared to the non-irrigated situation), partly because of the irrigation itself but mostly due to the higher (allowed) N input level. In the irrigated situation the N_2O emission dropped by 18% (cf. 40% in the non-irrigated situation).

Table 1. Calculated grass yields and nitrogen dynamics in cut grassland on a specific sandy soil in the Netherlands. Each value represents the mean of 15 individual model runs using the years from the period 1971-1985.

	Non-irrigated		Irrigated	
	A	B	A	B
N-fertilisation (kg effective N ha^{-1} y^{-1})	355	242	355	298
Dry matter yield (kg ha^{-1} y^{-1})	10900	10400	12000	11600
Nitrogen yield (kg ha^{-1} y^{-1})	353	316	377	351
Denitrification (kg N ha^{-1} y^{-1})	92	61	100	82
Leaching (kg N ha^{-1} y^{-1})	40	24	27	25
Nitrate (mg l^{-1})	87	*50*	56	*50*
N_2O emission (kg N ha^{-1} y^{-1})	6.5	3.8	5.0	4.1

A: Fertilisation level, recommended to farmers as economic optimum in the non-irrigated situation.
B: Fertilisation level, corresponding to a mean of 50 mg nitrate l^{-1} in the percolation water at -100 cm.

Conclusions

From this study can be concluded that for this specific sandy soil in the Netherlands a reduction in mineral N fertilisation is an effective way to bring down the nitrate concentration to 50 mg l^{-1} because the large relative change in N loss is accompanied by much smaller relative changes in grass yields. It can also be concluded that irrigation is a very helpful measure to control nitrate concentration in percolation water which illustrates the importance of an adequate water supply in relation to N leaching on intensively used grasslands.
Denitrification was responsible for a large part of the calculated total loss of N by denitrification and leaching from the selected sandy soil: only 29% (non-irrigated) or 22% (irrigated) of the total loss was leached below 100 cm soil depth, whereas the remainder (> 70%) had been denitrified mainly into N_2 and N_2O. Measures that were taken to reduce the nitrate concentration also caused a decrease in N_2O emission. If a decrease of e.g. 6% in N_2O emission is pursued (cf. Kyoto climate conference in 1997), no additional measures were needed in this study after realisation of the water quality target of 50 mg l^{-1}.

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Apparent recovery of urine-N in grassland on sandy soils

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Abstract

In order to answer the question whether a ban on grazing in autumn could substantially decrease the nitrate leaching from dairy farming on sandy soils in the Netherlands, a field experiment and a lysimeter experiment were established. The main measurements included the apparent nitrogen recovery in the crop and the nitrate leaching, measured as the nitrate content of the upper groundwater in the field and the leaching at 60 cm depth in the lysimeters. Nitrate leaching was increased in urine patches, but the effect of the date of application on the nitrate leaching was relatively small. Hence, also a policy of no grazing in autumn will probably only have a limited effect on nitrate leaching.

Keywords: grassland, grazing, leaching, nitrate, nitrogen, sandy soil, urine

Background and objectives

Fulfilling the Nitrate Directive of maximum 50 mg per litre in the upper groundwater is a persistent environmental problem in dairy farming on sandy soils in the Netherlands. Limited grazing is seen as a promising option to reduce nitrate leaching under grassland while maintaining the production level. In urine patches locally large amounts of N are deposited on grassland. The utilisation of this N is limited, especially at the end of the growing season. Hence, urinary N is an important source of soil mineral N, vulnerable to leaching. However, to what extent grazing might cause enlarged nitrate leaching has not yet been measured in the Netherlands. The supposed relation between grazing and nitrate leaching is not based on measurements of nitrate leaching, but on measurements of soil mineral N in autumn (Vellinga *et al.*, 2001). Furthermore, in many experiments with urine patches appreciable balance deficits are found, i.e. a large part of the N deposited is not accounted for when the apparent N recovery is calculated, and most experiments were established at higher N fertilisation rates than currently used in practise. The effects of application date on N utilisation, nitrate leaching and N balance (apparent nitrogen recovery, ANR) were studied in a field experiment and in a lysimeter experiment with artificial urine patches.

Materials and methods

In the field experiment artificial urine with 80% of the N as urea was applied in a quantity of 400 kg N ha^{-1} on 10 m^2 plots on six dates between 31 May and 22 October in permanent grassland on a light sandy soil at the experimental farm 'Cranendonck'. Measurements included apparent nitrogen recovery (ANR) in the crop, development of mineral nitrogen contents of the soil and nitrate contents of the ground water after winter. The results were compared with calculations with the model 'NURP' (Vellinga *et al.*, 2001). In this model the nitrate leaching is calculated on the basis soil mineral N in autumn. The basis of the calculation of nitrate leaching is an exponential relationship between date of urine deposition and soil mineral N and a linear relationship between soil mineral N and nitrate leaching.
For the lysimeter experiment (depth 60 cm, diameter 20 cm) undisturbed soil was taken from permanent grasslands on sandy soils at the experimental farms 'Cranendonck' and 'De Marke'. Artificial urine was applied in a quantity of 400 kg N ha^{-1} on three dates in autumn. Measurements included volume and nitrate contents of the leachate, ANR in the crop, stubble and roots and soil mineral and total N. In the lysimeter experiment ^{15}N labelled urea was used, results on this topic are described in Van Groenigen *et al.*, 2005.

Figure 1. Nitrate leaching from urine patches, data from the field experiment and model calculation (NURP).

Results and discussion

In the field experiment urine patches showed increased nitrate leaching, resulting in a positive effect of limitation of grazing on nitrate leaching. The date of application of urine, however, had only a limited effect on nitrate leaching in comparison to model calculations, as shown in Figure 1. Overall, nitrate leaching and N utilisation in the crop were small, resulting in large balance deficits, as shown in Figure 2. These deficits, actually N not accounted for, were including ammonia volatilisation, which can be estimated at 10% on average for Netherlands conditions. The balance deficits did originate in the period shortly after application, a period in which neither leaching nor denitrification was likely to be of significance.

Figure 2. Balance for urine-N in urine patches in a field experiment on a sandy soil in spring 2003, applied at different dates in 2002. Dates: 31 May, 24 June, 22 July, 22 August, 19 September, 22 October.

The results of the lysimeter experiment are shown in Figure 3. Compared to the field experiment, the ANR in the crop reached the same level and the nitrate leaching was increased. This resulted in smaller but still significant balance deficits.

Figure 3. Balance for urine-N in a lysimeter experiment with undisturbed grassland soils from two sandy soils in spring 2005, applied to lysimeters at different dates in 2004. Soils: C = Cranendonck; M = De Marke. Dates: 6 September, 4 October, 1 November.

The difference in leaching level between the two experiments might be caused by uncertainty of the representativeness of the sampling of the groundwater in the field experiment or by processes decreasing the nitrate content of percolation water between a depth of 60 cm and the groundwater level.
The relatively limited effect of the date of application of urine patches on nitrate leaching indicates that other management practices in dairy farming will probably have a larger effect on nitrate leaching than a ban on grazing in autumn. The most effective practice is probably a combination of moderate N fertilisation and limitation of grazing to the day period with supplemental feeding of roughage with a high energy/protein ratio, like silage maize. Furthermore, grazing must be proportional to grass production. When grass intake is decreased due to a too high stocking rate to production ratio, more urine patches will be present compared with the calculations, with an increased nitrate leaching as a result.

Conclusions

Urine application increased nitrate leaching, but the effect of the date of application was limited. Hence, also a policy of no grazing in autumn will probably only have a limited effect on nitrate leaching.

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Nitrate leaching following cultivation of grazed grass-clover on coarse sandy soil

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Abstract

When grass-clover is ploughed in there is a high risk of nitrate leaching. In order to try to minimize leaching an experiment was established on two fields on a commercial organic farm with a coarse sandy soil. A 3-year-old and a 5-year-old grass-clover were ploughed in spring 2003, and the following treatments were established: 1) spring barley (*Hordeum vulgare* L.), harvested at maturity ('Mature') and 2) spring barley harvested early as a green crop for silage with an undersown catch crop of Italian ryegrass (*Lolium multiflorum* Lam.), ('Green'). The treatments were fertilized with 0, 60 or 120 kg ammonium-N ha^{-1} in cattle slurry. Nitrate leaching was measured in 0 and 120 kg N ha^{-1}. The experiments showed that 'Green' treatments could reduce leaching by 166-309 kg N ha^{-1}, corresponding to 95-98% in comparison with 'Mature' treatments. In addition to nitrate, 10-30 kg N ha^{-1} was leached as other N-containing compounds. In contrast to 'Mature' treatments, leaching from 'Green' treatments did not differ from each other, irrespective of whether manure was applied or not. A high production of roughage was possible and the difficulties with clover fatigue experienced by many Danish farmers were avoided.

Keywords: catch crop, field experiment, organic N

Background and objectives

When grass-clover is ploughed there is a high risk of nitrate leaching (e.g. Djurhuus and Olsen, 1997; Eriksen *et al.*, 1999). The ploughing of grass-clover in spring (Djurhuus and Olsen, 1997) and the undersowing of a catch crop like perennial ryegrass (*Lolium perenne* L.) in the following spring cereal is recognized as a successful way of reducing nitrate leaching the following autumn and winter (Djurhuus, 1992; Thomsen and Christensen, 1999). However, sometimes the effect of the catch crop is less than expected, probably due to delayed harvest of the main crop and heavy rainfall after harvest, which may leach nitrate below the root zone of the catch crop (Hansen and Djurhuus, 1997). The objective of this study was to examine the effectiveness of an earlier catch crop than perennial ryegrass in reducing nitrogen leaching from a coarse sandy soil. A barley silage crop was undersown with Italian ryegrass in spring and harvested at the beginning of heading, and the Italian ryegrass was subsequently used for roughage production in the autumn.

Material and methods

Experiments were established in spring 2003 on a commercial organic farm with a coarse sandy soil. Two fields with grass-clover were ploughed. A 3-year-old grass-clover field had formed part of a crop rotation dominated by cereals, and another 5-year-old grass-clover field was part of a grass-intensive rotation grazed by dairy cows. After ploughing the grass-clover, the following treatments were established in each of the two fields: 1) spring barley, harvested at maturity and subjected to mechanical weed control in the autumn ('Mature') and 2) spring barley harvested early as a green crop for silage with an undersown catch crop of Italian ryegrass ('Green'), which was mowed twice in the autumn. The treatments were fertilized with 0, 60 or 120 kg ammonium-N ha^{-1} in cattle slurry, injected in the spring following ploughing (Table 1).

Table 1. Treatments, seeds, and treatments including (+) determination of N leaching.

Treatment[1]	Seeds, spring barley[2]	Seeds, Italian ryegrass[3]	Leaching
Mature-0N	350	0	+
Mature-60N	350	0	
Mature-120N	350	0	+
Green-0N	200	25	+
Green-60N	200	25	
Green-120N	200	25	+

[1] *0N, 60N and 120 N indicate kg ammonium-N in cattle slurry injected before sowing.*
[2] *Number of seeds per m^2 capable of germinating. Variety mixture: Cisero, Punto og Otira.*
[3] *Kg per ha of tetraploid Italian ryegrass, Ajax.*

For calculation of nitrate leaching, soil water isolates were taken using porous ceramic cup samplers installed below the root zone in spring 2003 in selected treatments (Table 1), (Djurhuus and Jacobsen, 1995). Two samplers were installed per plot (i.e. eight per treatment). A suction of approximately 70-80 kPa was imposed 2 to 3 days before sampling. During this period the suction decreased as a result of sampling the water. The soil water isolates from each replication were bulked before analysis, frozen within a few hours and later analyzed for nitrate N (Best, 1976). Total N was analysed as described by Cabrera *et al.* (1993) and dissolved organic N calculated as the difference between total N and nitrate N. Generally, sampling was carried out once every other week, except in periods of drought or frost. Percolation was calculated using the model Evacrop (Olesen and Heidmann, 1990).

Results and discussion

Nitrate leaching after Mature-0N was 174 and 240 kg N ha^{-1} in the 3-year-old and the 5-year-old grass-clover, respectively, when the soil was kept bare by rotovating twice during the autumn (Table 2). In Mature-120N leaching was 302 and 316 kg N ha^{-1}. In Green-0N and Green-120N leaching was only 7-9 kg N ha^{-1}. This means that the 'Green' treatments reduced leaching by 166-309 kg N ha^{-1}, corresponding to 95-98% (Table 2). In addition to nitrate leaching, 10-29 kg N ha^{-1} was leached as other N-containing compounds with the highest amounts from Mature-0N and Mature-120N after the 5-year-old grass-clover (Table 2).

Table 2. Leaching of nitrate and dissolved organic N (in brackets), kg N ha^{-1}, from 21 May 2003 to 12 May 2004 following ploughing-in of grass-clover in spring 2003.

Treatment	Following 3-yr-old grass-clover		Following 5-yr-old grass-clover	
		Reduction, %[1]		Reduction, % [1]
Mature-0N	174^b (13)	-	240^a (23)	-
Mature-120N	302^a (19)	-	316^a (29)	-
Green-0N	8^c (10)	95	7^b (16)	97
Green-120N	9^c (10)	97	7^b (12)	98

Values followed by different letters are significantly different from each other according to Duncan-test.

[1] *Reduction in nitrate leaching in comparison with Mature-0N or Mature-120N.*

Yields harvested in Mature-0N were 3.4 and 3.9 Mg dry matter ha^{-1}, and total yield harvested in Green-0N were 6.5 and 9.7 Mg dry matter ha^{-1} green crop and grass. A comparison of the treatments Mature-0N and Manure-120N showed additional nitrate leaching of on average 102 kg N ha^{-1} when 120 kg ammonium-N ha^{-1} was applied (Table 2). This is matched by a corresponding lack in yield increase when applying 120 kg N ha^{-1} (Table 3). In contrast to 'Mature' treatments, leaching from 'Green' treatments did not differ, irrespective of whether manure was applied or not (Table 2). This can be explained by an additional N uptake of on average 127 kg N ha^{-1} in Green-120N compared with Green-0N (data not shown). So in the 'Green' treatments most of the manure N was taken up by the ryegrass instead of being leached.

Table 3. Yield of grain, straw and green crop (Mg DM ha^{-1}) in treatments with spring barley and green crop grown after ploughing-in of 3- and 5-yr-old grass-clover in spring 2003.

	Following 3-yr-old grass-clover				Following 5-yr-old-grass-clover			
	Grain	Straw	Green crop	Grass[1] 15/8+ 20/10	Grain	Straw	Green crop	Grass[1] 15/8+ 20/10
Fertilization, kg N ha^{-1}	Mature		Green		Mature		Green	
0	3.4^{b}	2.6^{b}	2.9^{b}	3.6^{b}	3.9^{a}	4.3^{b}	4.6^{b}	5.1^{b}
60	3.9^{ab}	3.8^{a}	4.8^{a}	3.8^{b}	4.1^{a}	$4,8^{b}$	6.1^{a}	6.2^{a}
120	4.1^{a}	4.1^{a}	5.7^{a}	4.7^{a}	3.9^{a}	$5,6^{a}$	6.6^{a}	6.4^{a}

Values followed by different letters are significantly different from each other according to Duncan-test.

[1] *Grass cut on 15 August and 20 October 2003.*

Conclusion

The experiments showed that a barley silage crop undersown with Italian ryegrass could reduce leaching to a minimum. This offers advantages not only for the environment but also for farmers, as a high production of roughage was possible and the increasing difficulties with clover fatigue experienced by Danish farmers could be avoided.

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Nitrogen utilization under different tillage systems

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Abstract

Crops in non-inversion tillage systems may require larger N inputs than with conventional tillage. In order to examine N utilization under different tillage intensities, an experiment was carried out on a loamy sand in Denmark. Two crops were grown in 2002-03, winter barley (*Hordeum vulgare* L.) and winter wheat (*Triticum aestivum* L.). Tillage treatments were P: Ploughing; H_{8-10}: Harrowing, 8-10 cm; H_{3-4}: Harrowing, 3-4 cm and D: Direct drilling. Six different N strategies were carried out in each crop and tillage treatment. No significant interaction was found between soil tillage and N treatment. There was some growth reduction in especially H_{3-4} and D, which could not be offset by N fertilization. Notably, both crops were characterized by poor growth in certain patches, possibly caused by poor soil structure. Increasing fertilization up to 125% of recommended levels could not increase yields with non-inversion tillage to the same level as with ploughing. Yields and N uptake after application of 100 kg ha^{-1} NH_4-N in pig slurry was not significantly different from an application of 100 kg ha^{-1} N in mineral fertilizer. Autumn application of 15 kg N ha^{-1} of the total amount of N cannot be recommended for the stimulation of growth of winter cereals.

Keywords: direct drilling, fertilizer, nitrogen, tillage

Background and objectives

A decrease in nitrogen (N) mineralization has often been reported for soils in non-inversion tillage systems compared with conventionally tilled soils (e.g. Nyborg and Malhi, 1989). This suggests that crops in non-inversion tillage systems may require larger N inputs than in conventional tilling systems. However, the crop response to fertilization in non-inversion tillage systems has received little attention in Scandinavia (Rasmussen, 1999). The objective of the experiments presented here was to evaluate N utilization under different soil tillage intensities when growing winter barley and winter wheat as the first crop after initial tillage on previously ploughed soil.

Materials and methods

The experiment was carried out in winter barley and winter wheat/catch crop on a coarse sandy loam at Foulum, Denmark. The two cereals were grown in 2002-03. The clay (<2 µm), silt (2-20 µm), fine sand (20-200 µm) and coars e sand (200-2000 µm) content of the soil at 0-25 cm was 92, 126, 444, 307 g kg^{-1}, respectively. The organic carbon content was 18 g kg^{-1}. The experiment was established in autumn 2002 as part of a larger experiment. The actual design was a split-plot design in four replications with two factors: soil tillage as main plot and N rates as sub-plots. The tillage treatments were as follows: P: Ploughing to 20 cm; H_{8-10}: Harrowing to 8-10 cm; H_{3-4}: Harrowing to 3-4 cm and D: Direct drilling. Each tillage plot was 72.2 m long, which allowed two rows of five different sub-plots to be placed within the design. The gross area of each sub-plot was 13.70 m x 3.00 m and the net area for yield determination was 1.58 m x 10.00 m.

Four different N rates were applied to sub-plots in the two grain crops: 50 (0.50N), 75 (0.75N), 100 (1.00N) and 125 (1.25N) % of the recommended N rate. In winter barley and winter wheat, 1.00N received 139 and 172 kg N ha^{-1}, respectively. An additional treatment consisted of fertilization at 1.00N, but with application of 15 kg ha^{-1} of the total amount in autumn ($1.00N_{Aut}$), while all other plots were fertilized in spring. In a reference treatment, 100 kg NH_4-N ha^{-1} was applied in pig slurry, while the rest of the fertilizer recommendation was supplied as mineral fertilizer. The

slurry was applied with trailing hoses. In both crop rotations straw was cut and retained after harvest. The catch crop in winter wheat was perennial ryegrass (*Lolium perenne* L.), which was undersown in spring.
Each plot was harvested for grain yield with a plot combine. The dry matter content was determined by a near-infrared spectroscopy analyser (Infratec™ 1241 Grain Analyzer, Foss A/S; Buchmann *et al.*, 2001) on which also the protein content in the cereals was determined. The nitrogen content in grain was calculated using a protein factor of 5.70 for wheat and 6.25 for barley. Analysis of variance was carried out using the SAS general linear models procedure GLM (SAS Institute 1996).

Results and discussion

The provisional results show that yield and N uptake in grain generally increased when crops were fertilized with up to 1.00N, although the increase was not significant in all cases (Tables 1 and 2). Another general trend was for yield to decrease with decreasing tillage intensity, but this was only significant in winter wheat ($1.00N_{Ref}$, 1.00N and 1.25N). N uptake in grain was not significantly affected by tillage in either of the crops.

Table 1. Grain yields (85% DM) and N uptake in grain in winter barley at different N rates.

	Grain yield, Mg ha^{-1}					N uptake in grain, kg N ha^{-1}				
N rates	P	H_{8-9}	H_{3-4}	D	$LSD_{.95}$	P	H_{8-9}	H_{3-4}	D	$LSD_{.95}$
0.50N	4.6^{d}	4.4^{b}	4.8	4.6	ns	66^{d}	61^{b}	68^{d}	66^{c}	ns
0.75N	5.0^{cd}	4.8^{b}	4.9	4.8	ns	76^{c}	69^{b}	72^{cd}	76^{bc}	ns
1.00N	5.6^{ab}	5.7^{a}	6.0	5.3	ns	92^{b}	93^{a}	99^{ab}	86^{ab}	ns
$1.00N_{Ref}$[1]	5.5^{abc}	5.7^{a}	5.5	4.8	ns[3]	91^{b}	93^{a}	92^{ab}	80^{bc}	ns
$1.00N_{Aut}$[2]	5.3^{bc}	4.8^{b}	5.4	4.9	ns	86^{b}	74^{b}	87^{bc}	82^{abc}	ns
1.25N	6.0^{a}	6.2^{a}	5.9	5.3	ns[3]	106^{a}	107^{a}	107^{a}	99^{a}	ns
$LSD_{.95}$	0.54	0.85	ns[3)]	ns		8.9	15.7	17.0	17.5	

Values within a column followed by the same letter are not significantly different.

[1] *100 kg NH_4-N ha^{-1} in pig slurry and the rest of the fertilizer recommendation in mineral fertilizer.*

[2] *Supplied with 15 kg N ha^{-1} of the fertilizer recommendation in autumn.*

[3] *Significant at 10% level of significance.*

Table 2. Grain yields (85% DM) and N uptake in grain in winter wheat at different N rates.

	Grain yield, Mg ha^{-1}					N uptake in grain, kg N ha^{-1}				
N rates	P	H_{8-9}	H_{3-4}	D	$LSD_{.95}$	P	H_{8-9}	H_{3-4}	D	$LSD_{.95}$
0.50N	6.9	6.4^{b}	6.0	5.8	ns	86^{c}	85^{b}	78^{b}	84^{c}	ns
0.75N	7.3	6.7^{ab}	5.9	5.7	ns[3)]	106^{b}	95^{b}	86^{b}	90^{bc}	ns
1.00N	7.5_{A}	7.0^{a}_{AB}	6.7_{B}	6.4_{B}	0.59	121^{a}	112^{a}	109^{a}	110^{a}	ns
$1.00N_{Ref}$[1]	7.2_{A}	6.9^{a}_{AB}	6.3_{B}	6.2_{B}	0.82	114^{ab}	113^{a}	101^{a}	107^{ab}	ns
$1.00N_{Aut}$[2]	7.5	7.1^{a}	6.5	5.2	ns[3)]	118^{ab}	113^{a}	104^{a}	89^{c}	ns
1.25N	7.0_{A}	6.4^{b}_{B}	6.3_{B}	6.3_{B}	0.51	118^{ab}	110^{a}	109^{a}	114^{a}	ns
$LSD_{.95}$	ns	0.56	ns	ns		13.7	10.0	13.1	18.0	

Values within a column followed by the same lower-case letter are not significantly different and values within a row followed by the same capital letter are not significantly different.

[1] *100 kg NH4-N ha-1 in pig slurry and the rest of the fertilizer recommendation in mineral fertilizer.*

[2] *Supplied with15 kg N ha-1 of the fertilizer recommendation in autumn.*

[3] *Significant at 10% level of significance.*

No significant interaction between soil tillage and N rate was observed in any of the crops.

There was some growth reduction in especially H_{3-4} and D, which could not be offset by N fertilization. Notably, both crops were characterized by poor growth in certain patches (assessed visually). As an average for winter wheat 1, 9, 17 and 36% of each plot exhibited poor growth in P, H_{8-10}, H_{3-4} and D, respectively. In winter barley, the comparable figures were 20, 13, 30 and 37%. Generally, the plants in areas with poor growth had no recognisable attack of pests, diseases or nutrient deficiencies, but the plants were much smaller than adjacent plants in areas without poor growth. Plants with poor growth resembled bonsai plants, which are plants growing in small containers. Passioura (2002) also refers to the semblance with bonsai plants and points out that roots emerging from germinating seeds in non-inverted soil may have access to a very small volume of disturbed soil. Visual assessment of root growth beneath the plants showed that roots were concentrated in the upper few centimetres of the soil due to a more compact soil below. This is in agreement with results by Munkholm *et al.* (2003), who concluded that soil compaction is a critical factor when adopting direct drilling on sandy loam in a moist and cool Danish climate. Already after the first year of direct drilling, soil strength was substantially higher than in the ploughed soil (Munkholm *et al.*, 2003).

In the $1.00N_{Aut}$ treatment, yields and N uptake in grain tended to be less than in 1.00N in most crop and tillage treatments (Tables 1 and 2). The effect was significant in winter barley grain yield (H_{8-10}) and in N uptake in winter barley (H_{8-10}) and winter wheat (D).

Yields and N uptake in grain in $1.00N_{Ref}$ (with 100 kg NH_4-N ha^{-1} in slurry) and 1.00N was in neither case significantly different from each other. This seems to indicate that any NH_3 volatilization must have been compensated by mineralization of organic matter applied with the slurry.

Conclusions

The results show that increasing fertilization up to 125% of the fertilizer recommendation could not increase yields with non-inversion tillage to the same level as with ploughing. Yields and N uptake after application of 100 kg ha^{-1} NH_4-N was not significantly different from application of 100 kg ha^{-1} N in mineral fertilizer. Autumn application of 15 kg N ha^{-1} of the total N fertilizer recommendation cannot be recommended for the stimulation of growth of winter cereals.

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The effect of urine application on NO_3-N loads leached from grassland lysimeters

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Abstract

Nitrate leaching from no fertiliser, fertiliser only and fertiliser + cow urine at different times in the summer and autumn was assessed on large undisturbed monolith lysimeters (0.8 m diameter x 1 m deep) containing 3 different soil types. Over the first winter drainage period mean NO_3-N from the control (0N) and fertiliser only were low (1.5-23.7 and 6.2-40.4 kg/ha, respectively). Mean NO_3-N leaching from the fertiliser+urine treatments were high ranging 48-170, 204-243 and 221-343 kg/ha for applications in July, August and September, respectively. Urine application in autumn, on all soil types, clearly increased the quantity of N leached from the grass lysimeters. Soil type greatly influenced the loads of NO_3-N leached in this experiment, the poorly drained gley soil (Rathangan) had consistently lower losses than the brown earth and brown podzolic soils due to the greater potential denitrification. The potential impact of autumn grazing on NO_3-N leaching is highlighted.

Keywords: dairy farming, grazing, leaching, nitrate, urine

Background and objectives

The quantity of nitrate leached from grazed grassland is a function of fertiliser rate, animal grazing and mineralisation rate. The quantity of nitrate leached from grassland is often associated with the intensity of grazing (Cuttle, 1992) associated with spatial distribution of locally high loads of nitrogen (N) deposited in urine patches. Jarvis and Pain (1990) report urine patch N loads to range from 400 to 1200 kg/ha. Grass has a limited ability to utilise N deposited as urine and this increases the potential for losses of N to environment to occur especially when urine is deposited during the autumn/winter drainage season. The objective of this study was to examine the relationship between urine application and soil type on nitrate leaching.

Materials and methods

The study was carried out at a field lysimeter facility at Johnstown Castle, Wexford. In 2003, 75 undisturbed monolith lysimeters (0.8 diameter and 1 m deep) were sampled using the method described by Cameron *et al.* (1992). The lysimeters had petrolatum injected between the monolith and the lysimeter casing to prevent edge flow. The lysimeters represent 3 Irish soils types of differing drainage class, Rathangan (poorly drained gley), Elton (moderately well drained brown earth) and Clonakilty (well drained brown podzolic). A urine timing experiment commenced in July 2004. A total of five treatments per soil type (Table 1) were established in a randomised block design with three replicate lysimeters per treatment. Urine was collected, on the day of application, from dairy cows at milking in July, August and September. Urine total N was determined and 3 l of urine applied to each of the urine treatments.

Table 1. Urine treatments, inorganic fertiliser, urine N application rates (kg/ha) and urine N concentration (mg/l).

Treatment	Fertiliser N (kg/ha)	Urine N	Urine N mg/l
1. Control	0	0	-
2. Fertiliser only	132	0	-
3. Urine July	132	494	8,226
4. Urine August	132	572	9,530
5. Urine September	132	437	7,284

Lysimeter drainage was quantified and sub-samples were analysed for nitrate-N, ammonium-N, phosphorus and chloride on a Konelab colorimeter. Herbage was harvested on a 28 rotation, DM and % N were determined in the laboratory.

Results and discussion

Mean drainage volumes, between 21/09/04 and 02/08/05, for the Clonakilty, Elton and Rathangan soil types were 462, 433 and 362 mm, respectively. Urine N content varied between months (Table 1) and the highest application was in August. The mean load of NO_3-N leached from each treatment is presented in Figure 1.

Figure 1. Mean NO_3-N leaching (kg/ha) from grassland lysimeters receiving a range of treatments, drainage collected between 21/09/04 and 02/08/05.

With the exception of the Clonakilty soil, NO_3-N leaching from control treatments on all soil types was low (1.5, 1.7, 23.7 kg/ha). Mean NO_3-N leached from the fertiliser treatment ranged 6.2 to 40.4 kg/ha. The application of urine (July) in addition to fertiliser increased NO_3-N loads leached to between 48 and 171 kg/ha. Mean NO_3-N leaching increased with applications later in the growing season for all soil types. For example the Clonakilty soil type NO_3-N leaching from urine applied in July, August and September was 171, 220 and 332 kg/ha.

Leaching losses were consistently low on the Rathangan (gley) soil type, although high leaching loads were observed from urine application in August and September. Lower NO3-N leaching from the Rathangan soil was most likely due to denitrification. In March 2005 elevated nitrous oxide emissions were observed from Rathangan lysimeters that received urine in November 2004 The NO_3-N leaching losses from the Clonakilty and Elton soil types appear to be similar except from urine in July when there was 123 kg/ha more leached from the Clonakilty soil. On certain sampling dates maximum NO_3-N concentrations of up to 200 mg/l were observed from the urine treatments.

Data collection for the experiment is still continuing, when complete the data generated will be used for modelling of grazed grassland to identify optimal management practices for sustainable grassland management. The experiment is being repeated 2005/06 to examine leaching from urine with the same urine N content applied in spring, summer and autumn.

Conclusions

The experiment highlights the importance of urine patches and urine timing on nitrate leaching on all soil types. Soil type was also important on NO_3-N losses from all treatments although losses were still high on the Rathangan soil receiving urine in autumn. To reduce nitrate leaching from grazed grasslands careful grazing management is needed to address the intensity and duration of grazing at higher risk times such as in the autumn.

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Long term effect of the length of the grass period in ley-arable rotations on the quality of soil organic matter

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Abstract

Crop rotations affect soil organic matter (SOM) dynamics and thus soil quality. Long term effect of the length of the grass period in ley-arable rotations on SOM was studied in a long term experiment (28 years), where grass/maize (G/M) ratio varied from null to 6. SOM decrease was strongly linked to G/M ratio, and 3 types of rotations differed significantly for most of the measured parameters: annual rotation (G/M = 0 or 1), biennial rotation (1<G/M≤3) or long duration rotation (G/M= 6 and permanent grass). C and N mineralisation rates measured at the end of the trial increased with G/M ratio, more than proportionately to soil C or N content, what indicates changes in SOM quality (regarding mineralization). Those changes concerned C and N distribution in size fractions and probably the proportion of active SOM.

Keywords: Ley-arable rotations, long term experiment, soil organic matter mineralisation,

Background and objectives

To achieve sustainable fodder systems, yield has to be considered as well as soil quality and nitrogen loss risks. Soil organic matter (SOM) content is a key attribute of soil quality, as it affects physical, biological and chemical properties of soils. In ley-arable rotations, the nitrogen and carbon storage in soils depends on rotation management, among which the proportion of grass and the frequency of tillage (Haynes, 1999; Velthof *et al.*, 2001). The long-term evolution of SOM quantity is well known (Johnston *et al.*, 1986; Conijn *et al.*, 2002), the changes of SOM quality and the processes involved are less understood. This paper deals with the long term effect of the ratio grass/maize on SOM dynamics.

Material and methods

A long term experiment was settled in Quimper between 1978 and 2005 to study the effect of fodder rotations on crops yields and soil quality (Simon *et al.*, 1992). This trial compared the effects of 3 annual, 5 biennials and 3 long duration crop rotations, fertilized with cattle slurry. Among the 11 rotations, 6 are compared here: maize monoculture (rotation B), maize + 6 (C), 12 (E) or 18 (D) months *Lolium multiflorum*, maize + 3 years *Lolium perenne* (J) and permanent *Lolium perenne* (I). The grass/maize (G/M) ratio, calculated as relative duration of both crops, varies from 0 to 6.The mean crop yields, as silage maize and/or cut grass, were 15.6 (C) > 13.9 (D) > 13.1 (B) > 12.7 (E) > 11.5 (J) > 11.4 t DM.ha^{-1}.yr^{-1} (I).

The soil is a loamy-sandy soil (45.2% sand, 38.4% silt, 16.3% clay), pH =5.8. The initial C and N content were respectively 2.93 and 0.24 g kg^{-1}dry soil. The top soils (0-25 cm) were sampled 7 times during the 28 years, and analysed for total N (Kjeldahl) and C (Dumas). Detailed investigations on possible changes in the active part of SOM were achieved at the end of the experiment (February 2005), with 3 (I, J) or 4 replicates (B,C,D,E) per treatment:

- measurement of potential C and N mineralisation. Soil samples (0-25 cm), sieved at 2 mm to remove fragments of plant residues, were incubated at 15°C and at constant soil water content (90% field capacity) for 220 days. Mineralised C was continuously monitored by CO_2 trapping. Mineralised N was determined at regular intervals on soil samples.

- SOM was separated in 3 pools: < 50 µm, 50-200 µm and 200-2000 µm according to the physical fractionation method proposed by Balabane and Balesdent (1992). The two sand-size fractions were analysed for total C and N content, data for the finest fractions being calculated by difference.

Results and discussion

- *Long term SOM evolution* : after nearly 30 years of constant practices, organic N and C decreases vary between -5% (permanent grass) and –30% (maize monoculture). Those decreases are strongly correlated with the G/M ratio, as shown on Figure 1. Total SOM content remained nearly constant when G/M ≥ 6 (I and J). Three classes of rotations are distinguished (Table 1), according to G/M ratio, and corresponding to annual, biennial and long term rotations. The two compartments model of Hénin & Dupuis (1945) was tested to simulate long term SOM evolution, according to the equation: SOM (t) = k_1m/k_2 + (SOMinit - k_1m/k_2) $\exp^{(-k2t)}$. The factor k_1m was calculated from known plant residues and manure inputs, with isohumic coefficients from the literature. The coefficient of annual SOM mineralization k_2 was optimized using the Excel solver tool. As indicated in the Table 1, this coefficient decreases significantly when grass duration increases. Extrapolation of SOM kinetics on a very long period gives asymptotic values for final SOM content about 1.9 (B), 2.5 (C), 3.4 (D) and 2.9% (E).

Table 1. Main characteristics of SOM and C and N mineralisation in incubated soils, for the 3 classes of G/M ratio (letters: significant differences P ≥ 0.95).

G/M ratio	Norg final %	Corg final %	k_1m* t.ha^{-1}.yr^{-1}	k_2 moy yr^{-1}	Nf-Ninit/ Ninit %	C min. rate N min. rate kg C (N).ha^{-1}.day^{-1}		N min rate/ Norg final
G/M ≤ 1 (B-C)	0.171^a	2.10^a	1.14	0.021	- 28.5^a	3.68^a	0.283^a	1.65^a
1<G/M≤3 (D-E)	0.188^b	2.31^b	1.28	0.017	- 21.2^b	4.26^b	0.327^b	1.74^b
G/M > 3 (I-J)	0.226^c	2.71^c			- 6^c	5.54^c	0.461^c	2.04^c

* *k_1m calculated from known plant residues and manure inputs, with isohumic coefficients from the literature.*

- *Short term SOM mineralisation*: initial rates of C and N mineralization (first 28 days) differ significantly among the 3 groups, increasing with the G/M ratio (Table 1). Net N mineralisation after 2 months were respectively 15, 17 and 23.5 g N.kg dry soil^{-1}, the last value being intermediate of those observed by Accoe *et al.,* (2004) on 14 and 50 years old grasslands. The Nmin-to-Cmin ratio varies between 0.077-0.088 (28 days) and 0.090-0.098 (203 days), slightly higher for grass-based rotations. As the ratio N(C) mineralisation rate/total N(C) content increases with the G/M ratio, soil organic N(C) content does not fully explains the differences in

mineralisation rates: high proportion of grass in rotations leads to higher C and N mineralisations, possibly linked to changes in the most labile SOM pool.

Results of particle-size fractionation are shown by Table 2. Recovery of mass ranged from 98.5 to 100%, without any significant differences on the relative part of fractions between treatments.

Table 2. SOM fractions characteristics for the 3 classes of G/M ratio (letters indicate significant differences P ≥ 0.95).

G/M ratio	C%Ctot	N%Ntot	C/N	C%Ctot	N%Ntot	C/N	C%Ctot	N%Ntot	C/N
Size of fractions	200-2000µ			50-200µ			< 50µ		
G/M≤1 (B-C)	4.1 ^{ns}	3.24 ^{a}	26.2 ^{a}	8.6 ^{a}	11.1 ^{a}	13.9 ^{a}	87.3 ^{a}	91.5 ^{a}	11.8 ^{ns}
1<G/M≤3 (D-E)	3.81 ^{ns}	3.32 ^{a}	26.3 ^{a}	9.0 ^{a}	13.3 ^{b}	15.3 ^{b}	87.2 ^{a}	91.0 ^{a}	11.8 ^{ns}
G/M > 3 (I-J)	4.22 ^{ns}	5.19 ^{b}	21.6 ^{b}	12. ^{b}	24.6 ^{c}	16.1 ^{c}	82.9 ^{b}	86.5 ^{b}	11.5 ^{ns}

Expressed on a whole soil basis, the amounts of organic C and N increased with decreasing particle size, and C/N ratio decreased with decreasing particle size, as observed in the literature. Significant differences were observed for nearly all data between long rotations (I and J) versus short rotations, even when dominated by grass (G/M =1.5 or 3). The largest relative increase in C and especially N content were found in the intermediate (50-200µm) and large fractions (200-2000 µm) under grass based rotations compared to crop rotations, while relatively less C and N was associated to smallest SOM fraction. This result is consistent with the hypothesis of the labile SOM pool increase as observed by Haynes (1999) and Accoe *et al.,* (2004).

Strong and highly significant positive correlations are calculated between C and N mineralisations (after 28 and 203 days) and all C and N contents (not shown), while negative correlations are found with C-to-N ratio of the 2 sand-size fractions, but not with Ctot-to-Ntot nor with the < 50µm fraction. Nevertheless linear regression model between C(N) mineralisation and Ctot or Ntot over-estimates mineralisation for rotations with G/M ratio < 3 and under-estimates it for grass-based rotations soils, that is also consistent with the hypothesis of an increase in the proportion of labile soil organic matter as suggested by Accoe *et al.*, (2004).

Conclusions

Ley-arable rotations usually bring an advantage in terms of higher dry matter and nutrient yields, relative to monoculture, that could include a more efficient use of nitrogen and decrease the risk of N leaching. We actually observed higher SOM decrease rates from permanent grass to silage maize monoculture, which corresponded to lower plant residues restitution to soils, but these were not linked to average crop yields. Large spatial variability of SOM in soils explains that the differences on total organic C and N content between treatments are significant only after about 20 years. SOM decrease was strongly linked to G/M ratio, and 3 types of rotations differed significantly for most of the measured parameters: annual rotation (G/M = 0 or 1), biennal (1<G/M≤3) or long duration rotation (G/M= 6 and permanent grass). C and N mineralisation rates increased with G/M ratio more than proportionally to soil C or N content, indicating changes in SOM quality (regarding mineralization) and were associated to higher proportion of C and N in the SOM coarser fractions.

More investigation is necessary to determine how fast SOM fractions differ, and how far some pools indicate potential N and C mineralisation rates. Some progress should also be made in the understanding of SOM dynamics by linking size aggregate in fractions and in stability tests, in order to determine simple indicators to propose diagnostic tools for SOM quality.

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Working group 5

Buffer strips and catch crops: cosmetic or beneficial?

Report of Working Group 5

Managing N by catch crops and buffer strips

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Catch crops

The use of catch or cover crops to control leaching and pass nutrients on to subsequent crops is already known for at least a century (Fream, 1905, cited in Vos and van der Putten, 1997). The use of a catch crop for the sake of N preservation became obsolete because of the introduction of cheap nitrogen fertilizers. Since the 1990s the awareness grew of environmental problems arising from N emitted from agriculture into natural terrestrial and aquatic environments. Since then a lot of research has been done. The state-of-the-art can be summarized by stating that sufficient knowledge has been acquired on the use of catch crops as a crucial tool to reduce nutrient losses and achieve compliance with the nitrate emission norm of the EU (e.g. Shepherd, 1996; Shepherd and Lord, 1999; Vos and van der Putten, 2004). However, to farmers it is difficult to Iperceive the benefits that catch crops offer in terms of reduction of N leaching. So, extension should be intensified to get catch crops grown across the entire EU as a standard element of Good Agricultural Practice.
Some aspects of catch crops could be elucidated by future research:

1. integrated effects of various catch crops within a crop rotation on the long term (modelling seems most appropriate);
2. better transfer of knowledge on catch cropping;
3. in general, uniform norms and procedures to monitor compliance with environmental norms throughout Europe.

Buffer strips

Buffer strips are notably used to control run-off. A big drawback is their limited capacity to store nutrients and to cope with large volumes of run-off (Kessel, see proceedings). A solution for limited storage capacity would be to periodically mow and harvest grass strips, and of course broadening the strip. In case of large volumes of water running-off across steep slopes, an extra measure could be to make a ditch or even a constructed wetland below the strips. Helophytic species such as Phragmites australis (common reed) could be used to recover nutrients or facilitate denitrification (Harrington *et al*, 2005).
Buffer strips are also used to accommodate indigenous flora and fauna including benificial insects. Up till now, the function of buffer strips as natural habitats and as run-off control do not seem to be integrated. This integration requires the solving of conflicting measures, but may help to promote the region-wide use of bufferstrips, needed to really make a difference in control of nutrient losses.

Future research

The participants of this working group generally agreed, that future research should scale-up from field and farm to (sub-)catchment. This requires cooperation with hydrologists, who can help to allocate and design cover crops, buffer strips, watercourses and constructed wetlands to store and recycle nutrients, adapted to the local precipitation patterns.

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Poster presentations

Vegetated buffer strips on sandy soil to reduce nitrate leaching to surface water

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Abstract

During two consecutive leaching seasons the nitrate (NO_3) concentrations in the upper groundwater of an arable field were measured as a function of distance to a ditch. A non-fertilised grass buffer strip of 3.5 m was located between the arable field and the ditch. The NO_3 concentrations decreased towards the ditch, and were significantly lower in the buffer strip than in the arable field. Clear evidence for denitrification was obtained from the Cl/NO_3 ratios (both seasons) and from delta ^{15}N data (second season only). Due to the phenomenological character of this research we were unable to establish the effectiveness of the buffer strip, since we could not elucidate non-buffer strip related effects. However, the observed decrease in NO_3 concentrations is promising and justifies the demand for additional research under Dutch conditions.

Keywords: ^{15}N, buffer strip, Cl/NO_3 ratio, leaching, nitrate

Background and objectives

Buffer strips are widely used to reduce nutrient leaching to surface water. However, the efficiency of buffer strips in reducing nutrient loads is highly variable as a result of differences in hydrological and soil conditions. On an arable farm on sandy soil the performance of an unfertilised grass buffer strip was assessed during two consecutive leaching seasons. This paper presents some highlights reported by van Beek *et al.* (2005).

Materials and methods

The arable farm 'Vredepeel' is situated in the southern part of the Netherlands on a sandy soil. A compacted peat layer is present at approximately 2.5 to 3 m soil depth. A field (100 x 70 m) with an initially three-year old buffer strip (3.5 m wide) adjacent to a ditch was selected for monitoring. The buffer strip mainly consisted of Red fescue, which was cut annually coinciding with a removal of about 50 kg N $ha^{-1}y^{-1}$. Hydraulic heads, groundwater levels, nitrate-N (NO_3-N) and chloride (Cl) concentrations in groundwater and (in the second season) delta ^{15}N values of groundwater were measured at about every 30 mm of precipitation surplus (which equalled eight to ten sampling rounds per leaching season). Altman and Parizek (1995) and Mengis *et al.* (1999) stated that the ratio of an inert tracer, here Cl, and NO_3 may increase under a buffer strip, if NO_3 removing processes, such as uptake and denitrification, occur. When delta ^{15}N increases under the buffer strip this is a qualitative indication that denitrification does occur (Mayer *et al.*, 2002). During the growing season measurements were interrupted because of farm activities.

Results and discussion

Although the hydrology at the farm 'Vredepeel' is complex, the water movement was mainly directed towards the ditch and was restricted to a shallow layer of soil of 2.5 to 3 m below soil surface. There was a small downwards leakage across the peat layer. Averaged over both seasons, the NO_3-N concentrations under the buffer strip were significantly lower than in the remainder of the field: 11 mg N L^{-1} versus 20 mg L^{-1}, respectively (Figures 1 and 2).

Figure 1. Average NO_3-N concentrations in upper ground water as a function of distance from the ditch for both leaching seasons. The vertical broken line represents the boundary between buffer strip and arable field.

Figure 2. Average NO_3-N concentrations in upper ground water in buffer strip and arable field as a function of time.

However, the EU NO_3-N target level for ground water of 11.3 mg N L^{-1} was exceeded frequently in the buffer strip, especially during the second season. Within the buffer strip the concentrations decreased in the direction of the ditch.

Close to the boundary between buffer strip and arable field the concentrations were highest. This was most likely caused by soil compaction due to turning of agricultural machineries resulting in less growth and thus less N uptake, and presumably by spilling of fertilizer. At the end of the first leaching season the NO_3-N concentration in the buffer strip was higher than in the arable field. Apparently, a concentration peak originating from the arable field reached the buffer strip, while in the field itself the NO_3 concentrations decreased in time.
Since we have measured concentrations near the ditch, it seems worthwhile to estimate the NO_3-N load towards the ditch. The water flow towards the ditch was estimated from a classical drainage theory (Hooghoudt, 1937, 1940) based on measured ground water levels in the middle of the field and an assumed hydraulic conductivity at

saturation of 2 m d^{-1}. The estimated loads during both leaching seasons were 5 kg N ha^{-1} and 7 kg N ha^{-1}, respectively. For a whole year the loads will be somewhat higher, and then will be close to the Dutch Inventory of MINAS and the Environment of 25 kg ha^{-1} (RIVM, 2002).

Ratios of Cl/NO_3 increased towards the ditch, indicating that NO_3 consumption processes do occur under the buffer strip. Uptake by the crop at greater depths is unlikely, since the grass roots are mainly present in the top 10 to 20 cm. In the second season at a few occasions we found a clear increase in delta ^{15}N under the buffer strip. An increase in delta ^{15}N values is qualitative evidence of denitrification. At the end of the second season we determined TOC in the groundwater samples, with an average of about 40 mg L^{-1}, with no trend with distance to the ditch. So, even at greater depths, there is organic matter present, a prerequisite for denitrification to occur.

Conclusions

Due to the phenomenological character of this research we cannot exactly assess the effectiveness of this buffer strip. We have shown that NO_3-N concentrations decreased within the buffer strip and that denitrification occurred in the buffer strip. Because of the observed trend of decreasing NO_3 concentrations in the upper groundwater under the buffer strip, in a forthcoming project the efficiency of unfertilised buffer strips will be quantified in more detail by comparing N loads exported to the ditch for fields with and without a buffer strip.

Acknowledgements

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Potentially mineralizable nitrogen in soil after biocidal green manure incorporation

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Abstract

Biofumigant crops used as green manures, in addition to producing a biocidal effect on some soilborne pathogens and pests, may represent a way of integrating N fertiliser in agricultural soils. In this work we compared potentially mineralizable N in soil added either with glucosinolate-containing (GLS+) and non-containing (GLS–) green manures, or with metam sodium, which is a chemical treatment widely used as an alternative to methyl bromide. Nitrogen mineralization in soil added with *Brassica juncea* and *Eruca sativa* green manures was faster (1.10 and 0.74 mg N kg^{-1} soil d^{-1}, respectively) than that with winter wheat (*Triticum aestivum* L.) (0.29 mg N kg^{-1} soil d^{-1}). As biocidal green manures are often incorporated into the soil a short time before the sowing of the following crop, these results suggest that inorganic N released in soil due to biocidal green manure mineralization is more likely to become available to the following crop when the crop needs it. Metam sodium, while showing a remarkable nitrification inhibition activity, greatly stimulated the soil indigenous-N mineralization.

Keywords: Brassicaceae, green manures, metam sodium, methyl bromide, soil fertility

Background and objectives

The cultivation of sulphur-rich crucifer crops and their incorporation into the soil is frequently used method for the biological control of soil-borne pathogens and pests (Brown and Morra, 1997). The technique of biocidal green manures represents an interesting alternative to the use of methyl bromide, which has been recently subject to severe restrictions and is in phase-out from 2005 (Lazzeri *et al.*, 2004).

A renewed interest in green manure as source of plant nutrients also exists, due to the recently increased EU attention towards techniques capable of favouring environmental sustainability and biological crop production. The organic N added to soil at green manure incorporation can become available for the nutrition of the following crop (Aulakh *et al.*, 2001). The amount of N and the timing of N availability to crops both depend on the mineralization rate of the green manure organic N. In the hypothesis that this rate is linked not only to pedoclimatic conditions of green manure cultivation and incorporation, but also to the plant species (Thorup-Kristensen, 1993), the aim of this work was to compare the N mineralization potential of soils added with glucosinolate-containing (GLS+) and non-containing (GLS–) green manures.

Material and methods

Potentially mineralizable N in soil after green manuring with selected GLS+ (*Brassica juncea* and *Eruca sativa*) and GLS– (winter wheat, *Triticum aestivum* L.) crop species was evaluated in a laboratory experiment, according to Drinkwater *et al.* (1996). The results were compared with those obtained for an untreated soil and for a soil fumigated with methyl isothiocyanate (metam sodium), which is a chemical treatment widely used as an alternative to methyl bromide. The soil used in the experiment is a La Boaria silty clay (fine, mixed, mesic Udertic Haplustepts). The amounts of green-manure biomass and of metam sodium added to soil (Table 1) were comparable to those

applied at field scale. After green manure or metam sodium incorporation the soil was left for one week at room temperature (22 °C) and then incubated for three months at 30 °C, in a completely randomised block design with 3 replications for each treatment. Throughout the incubation period the soil moisture was maintained at 75% of the plant available water content.

Table 1. Selected plant and soil data values, at the beginning of the experiment.

Treatment	Plant Kjeldahl N	Plant C to N ratio	Amount of plant DM added to soil	Kjeldahl N in soil after incorporation
	g N kg^{-1} DM		g DM kg^{-1} of dry soil	g N kg^{-1} of dry soil
Brassica juncea	22.0	18	23.9	2.18
Eruca sativa	14.8	27	13.4	1.71
Winter wheat	13.3	31	26.8	1.80
Metam sodium			0.3	1.65
Control			0.0	1.58

Results and discussion

One week after green manure and metam sodium incorporation (time 0 of the incubation period) all treatments, except *E. sativa*, contained significantly higher inorganic N amounts than the untreated control (N_{ini}, Table 2). The ammonium N content was higher in soil treated with winter wheat and metam sodium, whereas nitrate N was more abundant in the biocidal-manured soils and in the control (data not shown). After the start of the incubation period the nitrate N levels increased, except in metam sodium, after a lag phase that varied from a minimum of 10 days, for *B. juncea*, to a maximum of 45 days, for wheat (Figure 1a). The ammonium N content decreased and, at the end of the incubation period, it was lower than 1 mg kg^{-1} for all treatments except for metam sodium (Figure 1b). The nitrate N content at the end of the incubation period was higher than that of the control, for *B. juncea* and *E. sativa*, whereas it was lower for winter wheat (Table 2). These results are in agreement with those of other authors. It is well known that low C to N ratios favour the mineralization process, and the two biocidal green manures, which have shown a higher N mineralization potential, had also a C to N ratio lower than that of wheat (Table 1). It is also known that the addition of winter-wheat straw to soil directs the microbial transformations preferentially toward inorganic N immobilization. At the end of the incubation period the soil with metam sodium had accumulated more inorganic N than that of all the other treatments. Ninety per cent of it was NH_4–N, which confirmed the nitrification inhibition activity of metam sodium (Welsh *et al.*, 1996). The accumulation ratio (Table 2), that is the amount of inorganic N formed per unit of total (Kjeldahl) N in soil at the beginning of the experiment, in soil with metam sodium was also the highest. In the absence of green manuring, the ammonium N found in the soil added with metam sodium was necessarily produced at the expense of the soil organic-N reserve. Therefore metam sodium seemed to stress soil N mineralization.

Table 2. Potentially mineralizable N in soil added with glucosinolate-containing and non-containing green manures, compared with metam sodium and an untreated control.

Treatment	Inorganic N content (NO_3–N+NH_4–N) in soil			Mean daily net N mineralization[2]	Accumulation ratio[3]
	1 week after incorporation (N_{ini})	at the end of the incubation time (N_{end})	accumulated inorganic N[1]		
	mg N kg^{-1} dry soil			mg N kg^{-1} dry soil d^{-1}	%
Brassica juncea	41.9a	142.8a	101.0	1.10	4.63
Eruca sativa	8.7c	76.7b	68.1	0.74	3.98
Winter wheat	17.9bc	44.6c	26.7	0.29	1.48
Metam sodium	26.7b	138.4a	111.7	1.21	6.77
Control	15.8bc	67.0bc	51.2	0.56	3.24

[1] *Accumulated inorganic N = $N_{end} - N_{ini}$.*
[2] *Mean daily net N mineralization = accumulated inorganic N/incubation time (92 days).*
[3] *Accumulation ratio = accumulated inorganic N/initial soil Kjeldahl N, from Table 1×100.*

Figure 1. Net N mineralization for the control (C) and for the soil added with B. juncea (BJ), E. sativa (ES), winter wheat (WW) and metam sodium (MS). a) NO_3–N, and b) NH_4–N evolution during the incubation time. Bars show standard deviations (n=3).

Conclusions

These first results point out the great influence the plant species can exert on green manure N mineralization. In particular, biocidal green manures showed a higher mineralization rate when compared to winter wheat. As biocidal green manures are often incorporated into the soil a short time before the sowing of the following crop, our results would suggest that inorganic N released in soil from biocidal green manure mineralization is likely to become available to the following crop just when the crop needs it.

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Proposal of an indicator for the environmental sustainability of animal breeding for land planning purposes

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Abstract

Indicators on the environmental sustainability of animal breeding are needed as tools to help regional policy-makers in the application of the EU Directives for environmental quality protection. The ratio of N actually produced with animal waste (N load) to the amount of N that can be removed by crops is here proposed as an indicator of animal breeding sustainability. The goal of this proposal was to conciliate the care for the environmental problems with the need to make the most of animal waste as fertilizer for crops. An example of application of the indicator to agricultural areas of Modena province, in the Po Valley (Northern Italy), is illustrated.

Keywords: crop fertilization, indicator[s?], livestock, nitrogen load, water vulnerability

Background and objectives

The European Union pursues the general goals of safeguarding and improving environmental quality by means of different directives, the Water Framework Directive (2000/60/EC) and the Strategic Environmental Assessment Directive (SEA, 2001/42/EC) being the most recently issued, following the Nitrate Directive (1991/676/CEE). At present, environmental indicators are being evaluated as potential tools for helping policy-makers in the application of these directives (OECD, 2002).

In the Modena province (Northern Italy, Lower Po Valley) numerous livestock production units exert remarkable environmental pressure, due to land spreading of animal waste. A frequently used pressure indicator is the N load from animal waste. This indicator takes only into account the pollution effect of animal-waste land spreading. However, environmental sustainability of animal breeding derives not only from the pollution reduction, but also from the possibility of recycling animal waste as a nutrient source for crops. Within the framework of a Government-granted project for the SEA application in Modena province, an indicator of animal breeding environmental sustainability (Ca_r) was therefore included. The aim of this work has been to provide Modena Province's policy-makers of the urban and land planning, and agricultural sectors with a procedure for the estimate of Ca_r. An indicator of animal-waste land spreading potential (N_{crit}) was also proposed as ancillary to the Ca_r indicator.

Material and methods

For each of the 47 municipalities of Modena province, belonging to 6 agricultural regions, the total amount of N deriving from animal waste (Nr_{eff}) was estimated and the agricultural areas of interest for the calculation of the indicators (S_{ind}) were defined. The Nr_{eff} value was calculated on the basis of the livestock number of heads, for each livestock class (from veterinarian registries), and on the manure-N content of each livestock class (CRPA, 2001). For each municipality's S_{ind}, the total amount of N removed by the crop economic production (Nc), and the maximum amount of manure N that can be distributed on a given land area, according to the current environmental laws (Nr_{max}), were estimated. The amount of N admitted to land spreading by law equals 170 kg N ha^{-1}, in vulnerable areas (ZV), and 340 kg N ha^{-1}, in the remaining areas. Municipality areas where animal-waste land spreading is forbidden (AD) were removed from the calculation. All these estimates were based on GIS- and Government

statistics-supported data. We assumed crop N requirements to be covered solely by the N supplied by animal wastes, and leguminous species to be N consumers rather than N fixing. In fact it has been reported that leguminous species remarkably reduce their N fixing activity and in preference take up N from the soil when it contains large amounts of available N (Peterson and Russelle, 1991).

The Ca_r value was expressed as the Nr_{eff} to Nc ratio. The N_{crit} value was calculated as the Nr_{eff} to Nr_{max} ratio. The higher the Ca_r value, the higher the possibility for land spread animal waste not to be taken up by crops and to be a cause of pollution. The higher the N_{crit} value, the higher the excess of N from animal waste not allowed for land spreading, according to local law restrictions.

Due to the lack of precise and complete information about the N mineralization potential of the soils of Modena province, we could not include in our procedure the evaluation of the N amount derived from soil organic matter mineralization.

Results and discussion

Whereas municipalities with the highest N load (Nr_{eff}) were located in the Modena and Carpi plains, municipalities with high Ca_r and N_{crit} values, that is, at risk of N pollution, were mainly located in the Modena Plain Region (Formigine and Spilamberto municipalities, in Figure 1 and 2). This region lies in the piedmont conoid areas, characterised by the highest water vulnerability to nitrate pollution (high ZV values, Table 1). It was also possible to ascertain that municipalities with large areas suitable for manure land spreading (high Nr_{max} value) have a low livestock concentration (low Nr_{eff} value).

Table 1. Values of some items used as entries for the calculation of the Ca_r and N_{crit} indicators. S_{ind}, agricultural area of interest for the calculation of the indicators; ZV, vulnerable area; AD, land area excluded from animal waste land-spreading; Nr_{max}, maximum amount of N allowed in the S_{ind}, by law; Nr_{eff}, N produced by livestock; Nc, N removed by crops.

Agricultural region[1]	Area	S_{ind}	ZV	AD	Nr_{max}	Nr_{eff}	Nc
	km^2				Millions of kg		
Modenese Lowlands	480	444	0	38	13.8	1.5	7.1
Carpi Plain	386	347	5	14	11.3	2.0	5.1
Modena Plain	413	332	195	30	7.3	2.8	5.4
Modena Hills	463	314	65	46	8.3	1.6	4.0
Upper Panaro River	692	288	39	64	7.2	1.2	4.3
Dragone and Rossena Valleys	256	117	15	29	2.8	0.4	1.8
Total	2688	1842	319	219	50.8	9.5	27.8

[1] *Municipalities in the agricultural regions: Modenese Lowlands: Camposanto, Cavezzo, Concordia s/Secchia, Finale Emilia, Medolla, Mirandola, Novi, San Felice s/Panaro, San Possidonio; Carpi Plain: Bastiglia, Bomporto, Campogalliano, Carpi, Nonantola, Ravarino, San Prospero, Soliera; Modena Plain: Castelfranco Emilia, Castelnuovo Rangone, Formigine, Modena, San Cesario s/Panaro, Spilamberto; Modena Hills: Castelvetro, Fiorano, Guiglia, Maranello, Marano s/Panaro, Prignano s/Secchia, Sassuolo, Savignano s/Panaro, Serramazzoni, Vignola; Upper Panaro River: Fanano, Fiumalbo, Lama Mocogno, Montecreto, Montese, Pavullo nel Frignano, Pievepelago, Riolunato, Sestola, Zocca; Dragone and Rossena Valleys: Frassinoro, Montefiorino, Palagano, Polinago.*

Figure 1. Classes of the Ca_r indicator values, in the municipalities of Modena province. Ca_r = N load from animal waste/N removed by the crops.

Figure 2. Classes of the values of animal-waste land-spreading potential (N_{crit}), in the municipalities of Modena province. N_{crit} = N load from animal waste/maximum admitted N load.

Conclusions

Local policy-makers may take advantage of the proposed procedure to evaluate the environmental sustainability of building new dairy farms in a given province's area, or to give farmers direction on the best way for animal-waste land-disposal. The proposed procedure, being based on information easily available to policy-makers in charge of the EU Directives application, and even subject to frequent updating, is suitable for indicator time-trend evaluations.

Acknowledgements

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Nitrogen Losses in Hillside Cropping Systems of Northeast Thailand as Affected by Soil Conservation Measures

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Abstract

The effects of four soil conservation measures and fertiliser application on runoff, soil loss, erosion and leaching losses of nitrogen (N) were investigated in hillside cropping systems of Northeast Thailand. Soil loss and runoff were measured by using a collecting system at the lower end of each plot. N leaching was determined with the resin core method. Data recorded during 2003 and 2004 showed a significant reduction of soil loss, runoff, and N losses with all soil conservation methods but particularly with papaya-grass hedge. However, as trade-off higher N losses via leaching were observed in the papaya-grass treatment. We attributed the higher leaching losses to an increased infiltration rate and an increase in macropores.

Keywords: Erosion, leaching, maize, runoff, soil loss, vegetation barriers

Background and objectives

In Northeast Thailand, soil erosion is a severe problem in uplands (Kunaporn *et al.*, 1999). It causes soil nutrient depletion, leading to decreases in soil fertility and crop productivity in the course of time. Nutrient losses by runoff contribute to water body eutrophication and environmental stresses. On the other side, high amounts of fertilizer are required to compensate these nutrient losses and to mitigate soil degradation (Kongkaew, 2000). Both, integrating soil conservation measures in annual cropping systems as well as studying the dynamics of N losses are needed to provide guidelines for sustainable land-use systems. The objectives of this study were (i) to monitor N losses by erosion and leaching in hillside cropping systems of Northeast Thailand and (ii) to assess effects of soil conservation measures on these losses.

Material and methods

A field trial was carried out on a clayey, kaolinitic, typic Haplustalf at Ban Bo Muang Noi, Loei province in NE Thailand (17°33' N and 101°1' E; 572 m a.m.s.l.) in 2003 and 2004 to measure soil erosion and nutrient leaching. Slope gradients ranged from 21-28%. The soil at the experimental site had a silty clay loam texture, a pH (H_2O) of 6 and an organic matter content of 3.5%. Available P (P-Bray) and exchangeable K contents were 14 mg kg^{-1} and 200 mg kg^{-1}, respectively. The site has a tropical savannah climate with a mean annual rainfall of 1280 mm, falling from May to October. Mean annual temperature was 26°C.

Plots of 4m x 18m with a collection system for runoff water and eroded soil at the lower end of each plot were established in a split plot design with two replicates and two fertilizer levels (no fertilizer and 61 kg ha^{-1} of N plus 14 kg ha^{-1} of P) as main plots and five soil conservation measures with maize (*Zea mays*) as subplots. Subplot treatments were farmers' practice as control (T1), vetiver (*Vetiveria zizanioides*) grass strips (T2), mango (*Mangifera indica*)-grass strips (T3), leucaena (*Leucaena leucocephala*) hedges (T4) and papaya (*Carica papaya*)-grass strips (T5). Soil loss and runoff were measured after each erosion event and representative samples were collected to determine their N content. Soil and water samples were analyzed according to the methods described by Kongkaew (2000).

Nutrient leaching was determined by using the resin core method (Schroth and Sinclair, 2002). Statistical analysis was done with SPSS version 11.5 by using the procedure for split-plot design. Least significant differences were estimated by Tukey's test at the 5% level.

Results and discussion

Fertilizer application significantly reduced runoff in both years and increased N losses through leaching significantly in 2003 (Table 1). In both years, soil conservation measures showed a significant decrease of soil loss, runoff, and N losses through erosion, but led to higher N leaching losses. The interaction between fertilizer application and soil conservation measures showed significant differences for soil loss and N losses through leaching in 2003 while in 2004 only runoff differed significantly.

Runoff, soil loss, N losses through erosion were always highest in the control treatment, indicating the high erosion potential of maize cropped under farmer's practice. The four soil conservation measures showed positive effects in controlling erosion compared to the control, reducing runoff, soil loss and N losses through erosion up to 26%, 79% and 73%, respectively. Differences were always significant, except for runoff in the mango-grass strip treatment during 2003. In both years, the lowest runoff, soil loss and N losses through erosion were observed in the papaya-grass strip treatment. Similar results have been reported by Lal (1995) where hedgerows and grass strips also effectively reduced soil loss, runoff and N losses through erosion. These positive effects can be attributed to a reduced runoff speed associated with a higher soil infiltration and a lower sediment load of the runoff water in cropping systems with vegetation barriers.

The amount of soil losses observed in the farmer's practice was much lower compared to data reported by Pimentel and Kounang (1998) for various agro-ecosystems in Asia, Africa and South America (19-22 t ha^{-1} *vs.* 30-40 t ha^{-1}). Soil losses of this magnitude, however, are still far away from the threshold which can be tolerated. According to Lal (1985), permissible values are 5 to 12 t ha^{-1} for shallow to deep soils which were met in T3 and T5 in 2003 and in 2004 in all soil conservation treatments. N losses through erosion showed a similar trend as soil losses when compared with other studies. They were also much lower than those reported by Fagerström *et al.* (2002) who found N losses of 150 kg ha^{1} in continuous maize production systems on hillsides in Vietnam with slope gradients of 20-22%. Total N losses of the economically most promising treatment with papaya-grass strips were of the same magnitude as those of the control, whereas all other treatments with soil conservation had lower total losses than the control in both years. Main pathway of N losses, however, was leaching in all treatments. Leaching losses of N were 68% of the total N losses in the control and accounted for 86-91% of the total N losses in treatments with soil conservation. On average, we observed 12% higher N losses through leaching in conservation treatments as compared with farmer practice. However, losses were higher when fertilizer was applied. The main reason for higher leaching losses of N is a surplus on the water balance combined with a surplus on the N balance in treatments with soil conservation as the speed of the runoff water was reduced by the vegetation barriers. But it may also be attributed to the amount of rainfall infiltration and the density of active plant roots (Lal, 1995) since vegetation barriers increase infiltration and root growth in hedges and induce increased macropores (Rowe *et al.*, 2005). The N losses through leaching were lower in 2004 than those of 2003 which was due to a lower precipitation in the second year of observation.

Table 1. Runoff, soil loss and N losses through erosion and leaching as affected by fertilizer application (F) and soil conservation measures (SC). Data were collected at Ban Bo Muang Noi, Loei province in NE Thailand.

	Runoff		Soil loss		N losses through erosion		N leaching losses	
	2003	2004	2003	2004	2003	2004	2003	2004
Fertilizer application (F)								
+F	698b [1]	441b	12	7	16	9	71a	55
- F	778a	686a	12	9	15	9	54b	44
Soil conservation measures (SC)								
Farmer's practice	834a	694a	22a	19a	29a	19a	53b	47b
Vertiver grass strip	695b	519b	11b	8b	13b	6b	62b	51b
Mango-grass strip	745ab	536b	10bc	4b	12b	5b	61b	38b
Leucaena hedge	727b	555b	12b	6b	12b	7b	61b	51b
Papaya-grass strip	690b	513b	7c	4b	8b	5b	74a	60a
F x SC								
+F								
Farmer's practice	802	648ab	20a	20	30	18	55b	43
Vertiver grass strip	661	333c	13b	5	14	6	67ab	59
Mango-grass strip	717	527abc	11bc	4	13	5	70ab	41
Leucaena hedge	699	398bc	12bc	4	12	6	75ab	53
Papaya-grass strip	611	298c	7c	4	9	5	87a	78
-F								
Farmer's practice	866	739a	25a	18	28	20	50b	51
Vertiver grass strip	730	705a	10bc	11	12	6	57ab	44
Mango-grass strip	774	546abc	8bc	4	11	5	52b	37
Leucaena hedge	755	712a	12b	8	12	7	48b	49
Papaya-grass strip	769	728a	7c	5	7	5	61a	41
Analysis of variance								
F	* [2]	**	ns	ns	ns	ns	**	ns
SC	**	*	**	**	**	**	**	*
F × SC	ns	*	*	ns	ns	ns	*	ns

[1] *Values followed by the same letter are not significantly different at P≤0.05 (Tukey's test).*
[2] *ns = not significant.*
* *= P ≤ 0.05.*
** *= P ≤ 0.01.*

Conclusions

Soil conservation by using papaya-grass barriers controlled soil loss, runoff and N loss through erosion most effectively, but as a trade-off led to higher N losses through leaching due to an increased infiltration rate. The results indicate possible pathways for a sustainable land-use on the highly soil erosion prone areas of Northeast Thailand.

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Soil nitrogen leaching from tree row strips and grassed alleys in a pear orchard

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Abstract

A study was carried out to evaluate nitrogen losses between October 2004 and February 2005 in a pear (*Pyrus communis* cv. Abbé Fètel) orchard, in the eastern Po Valley (Northern Italy). NO_3^- concentration was determined in the soil solution extracted with suction lysimeters in the tree rows and in the grassed alleys. Nitrate losses were calculated multiplying N concentration in the soil solution by the amount of water drainage out of the soil profile, estimated by a meteorological approach. Two treatments (with or without N supply – 48 kg N ha^{-1} - by fertigation) were compared. Nitrate-N concentration in soil solution and N losses were highest in the tree row strips of fertilized plots and lowest in the grassed alleys. Results indicate that localised addition of nitrogen through the irrigation system at rates used in this experiment increased the N losses through leaching by 3.8 kg N ha^{-1}, accounting for about 8% of fertilizer N. Negligible amounts of nitrate-N were lost from grassed alleys, suggesting a beneficial role of the extension of grass presence along tree row strip during autumn-winter.

Keywords: grasses, leaching, nitrate, orchard, pear

Background and objectives

Excess of fertilizer supply to orchard may result in nitrogen losses by leaching, mainly in the form of nitrate (NO_3^-), that can easily dissolve in drainage water. Nitrate losses from agricultural ecosystems can reach ground and surface water, with problems regarding public health, because of increasing risk of drinking water contamination. Nitrogen pollution of water is also of environmental concern, because of lake acidification and eutrophication of costal marine environments (Vitouosek *et al.*, 1997). Sustainable management of fruit trees should aim to minimise the losses of nutrients from the orchard soil. Decisions on the type of soil management, the type and the amounts of nutrient supply are important tools to reduce nutrient losses by leaching (Tagliavini *et al.*, 1996).

Material and methods

The study was carried out between October 2004 and February 2005 in a pear (*Pyrus communis*) orchard located in the eastern Po Valley (Northern Italy). Trees (cv. Abbé Fètel grafted on quince C rootstock) were planted in 2002/2003 winter and planting density was 2924 tree ha^{-1}. Soil management included herbicided tree row strips of 1.0 m width and grassed alleys (2.8 m width). Soil texture was silty clay loam. Drip irrigation was provided with drippers at 0.9 m along tree row. The plot was divided into 6 blocks, and within each block, a set of four adjacent trees received no fertilization (control plot), while another set of four trees received 48 kg urea-N ha^{-1} by fertigation, split into 22 supplies from April to the beginning of September (fertilized plot).

Soil moisture was determined at the beginning of the study to calculate initial soil deficit in the soil profile (0-70 cm). Rain, reference evapotranspiration and crop coefficient data were used to calculate when soil moisture deficit reached zero (Leach *et al.*, 2004). From this moment on, the amount of water drainage out of the 0-70 cm soil profile was determined. Suction lysimeters with a ceramic cup at 70 cm depth were placed in the soil in the tree rows within each plot (12 in total) at 0.2 m from the drippers, as well as in the centre of the grassed alleys

(6 in total). Samplings of soil solution were collected every two weeks from each lysimeter, and nitrate and ammonium concentration was determined with an autoanalyser (AxFlow Bran+Luebbe, Germany). The amount of nitrogen losses was than calculated multiplying N concentration by the volume of drainage water.

Figure 1. Precipitation and drainage fluxes (from tree row areas) from a 70 cm soil profile.

Figure 2. $N\text{-}NO_3$ concentration in soil solution under the tree rows and the grassed alleys. Vertical bars represent standard error.

Results and conclusions

In the October-December period, rainfall amounted to 355 mm, while from January to April it was only 159 mm. Soil water holding capacity was reached by mid October (Figure 1) and total water drainage from the 0-70 cm soil layer amounted in the whole period to 286 mm. Nitrate- N concentration in the soil solution extracted in the tree row strips of fertilized plots at 70 cm depth ranged from 6.5 to 9.0 mg l^{-1} and was significantly higher than that recorded in the unfertilized plots (1.5-4 mg l^{-1}) (Figure 2). Nitrate-N concentration was lowest in the soil solution extracted from the grassed alleys (generally less than 0.5 mg l^{-1}) throughout the whole period. The concentration of ammonium-N was always negligible. Differential nitrate-N concentration caused significant differences in the amounts of N drained out of the soil profile (Figure 3). The total amounts of nitrate-N leached out per square meter of soil were: 2.12 g in the fertilised row strip, 0.68 g in the unfertilized tree row strip and 0.07 g in the grassed alleys. Results indicate that localised addition of nitrogen through the irrigation system at rates used in this experiment increased the N losses through leaching (as compared to unfertilized control) by 3.8 kg N ha^{-1}, accounting for about 8% of fertilizer-N. These amounts are considered relatively low as compared to several literature data (Haynes and Goh, 1980; Ramos *et al.*, 1994; Stevenson and Nielsen, 1989); however, strategies like reducing fertiliser-N inputs and/or avoiding N

supply in late summer-early autumn should be applied to enhance the recovery of fertilizer-N in the system. The low amounts of nitrate-N lost from grassed alleys are likely due to higher N uptake by grasses than by trees in autumn (Stork and Jerie, 2003); drainage fluxes were in fact almost the same in the row and interrow areas. A beneficial role of the extension of grass presence along tree row strips during autumn-winter is suggested.

Figure 3. Daily N losses from the tree rows and the grassed alleys.

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Working group 6

Crop-related indicators: is the crop able to tell farmers what to do?

Report of Working Group 6

Crop related indicators: Is the crop able to tell the farmers what to do?

Report by Radersma, S.[1] & Evert, F.K. van[2*]

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The question in the title can be split in two parts:
1. which crop-indicator tells what about the crop and
2. how is this indicator-measurement translated into N-application rate for the farmer.

What can a crop indicator tell?

Crop indicators are used to fine-tune nitrogen (N) fertilization, either in N-deficient situations (Alam *et al.*, 2005) or in N-excess situations (Radersma *et al.*, 2006). Well-known crop indicators for nitrogen nutrition are N-concentration of tissue, N-concentration of sap (in petioles, midribs of leaves), leaf colour and crop light-reflection. Less well-known crop indicators are polyphenolics content (Cartelat *et al.*, 2005) and remote sensing, although the latter is in effect a crop light reflection method at a large scale.
Due to the fact that leaf colour (and probably crop reflection) is generally changing more strongly in N-deficient tissue, crop indicators based on leaf colour (or light reflection) may function best in N-deficient situations. On the other hand, in situations where N-excess needs to be reduced to avoid unnecessary losses to the environment, N-concentration may be the best crop indicator. In N-excess situations, N-concentration increases rapidly once maximum biomass production is reached and luxury consumption of N occurs. In N-deficient situations, each increase in N-uptake is accompanied by an increase in biomass production, so that N-concentration increases little even when total N-uptake increases. These responses are shown in Figure 1. However, sap N-concentration has a clear relation with N-nutrition in both the N-deficient and N-excess situations.
The applicability of any crop indicator does not only depend on N-excess or N-deficiency, but also on crop characteristics. For instance, leaf colour measurements were effective in tomato but not in olive trees (A. Stellaci, pers. comm.).

Figure 1. Relationship between relative dry matter (DM) yield and N concentration (g N/kg DM) in silage maize. Reproduced from Herrman and Taube, 2005.

How to translate the indicator measurement into a N-application rate for the farmer

To be of practical use to the farmers, indicators need to be cheap, easy to use, and sufficiently precise.

At the start of crop growth in the field, N-application rate can best be based on soil N supply indicators and the expected initial N-uptake of the crop, rather than on crop indicators, because the coverage of the crop is too low for reflection measurements and leaf-colour and leaf N content mainly indicate seed N-reserves, and not the amount of N in the soil that is available to the plant. In later crop-growth stages, crop indicators often prove to be more reliable than soil-indicators for the determination of N-application. Soil N measurements are very variable in time and space and crop N-demand is high.

The translation of a crop indicator measurement into a N-application rate can be done in two ways. The first is an absolute method, the second relative. When used in an absolute manner, the crop-indicator measurement and its relation with the nitrogen application need of one specific crop is calibrated in a sufficiently large series of experiments or existing crop fields. After this development of the relation between the indicator measurement and one specific crop (or variety), the crop indicator measurement can be directly translated into a decision of N-application rate in the specific crop. However, this absolute relation between indicator-measurement and crop-N-need is often limited to the environment where the calibration is done, or to closely similar environments. Moreover, it is limited to the specific crop and sometimes to the specific crop-variety for which the relation is made.

When used in a relative manner, crop-indicators are used in combination with crop-windows. Crop-windows used in combination with crop indicators are small plots in the field which have received a higher N-application and which serve as reference plots to decide on additional N-fertilization in the main field. If the crop indicator measurement indicates a lower N-status in the main field as compared to the crop-window, N-application is necessary and the rate may be determined according to the extent of the difference between window and main field. This relative use of crop-indicators combined with reference windows is suitable for crops/varieties for which an extensive calibration between indicator measurement and crop-N-need is not yet done. It is also suitable to situations with high variability between environments and seasons, but it is less suitable for heterogeneous fields, because a window may occupy a non-representative spot. The risk of using crop indicators in combination with crop windows is that a difference between main field and window may be detected too late for the crop to recover up to the level of the window.

Conclusion

Crop indicators used in N-fertilization can tell farmers what to do, better than standard N-recommendations per crop. This is due to the fact that standard crop recommendations for N-fertilization are based on many seasons/climates and fields/soils taking values which assure that in e.g. 95 % of all situations maximum production is reached. Proper use of crop-indicators to determine N-application rate takes into account differences in fields and seasons, and thus N-application rates become similar to or less than the recommendation in 95% of the situations and increase compared to the recommendation in less than 5 % of the situations.
Variation in N-application between seasons or fields (in the same environment) according to the working group members are shown in Table 1.

Table 1. Rough estimates of seasonal and spatial variation in crop-N-indicator derived N-fertilization on different crops in different environments.

Crop / country	Seasonal variation in N-application (kg ha-1)	Variation in N-application between fields (kg ha-1)	Reference
Silage maize / Northern Germany	little	150-200 kg N/ha (75% organic, 25% mineral)	Herrmann, pers. comm.
Maize / Canada	0 – 100		N. Tremblay, pers. comm.
Oilseed rape / France	0 – 200	0 - 200	Reau, pers. comm.
Wheat / Navarra	0-200	0-200	Arregui *et al.*, 2005
Leek / Netherlands	100 – 200	100 - 250	Radersma *et al.*, 2006
Romanesco/ Spain	100-200	100-200	Rodrigo *et al.*, 2006

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Poster presentations

Chlorophyll content as an indicator of nutritional status of sugar beet crop

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Abstract

Nitrogen (N) management in sugar beet crop (*Beta vulgaris* L.) should be based in reliable indicators, in order to complain with agronomic, industrial and environmental issues. Leaf chlorophyll content was evaluated as an indicator of nutritional status of sugar beet crop in a trial located in the Duero Basin (Spain). Four increasing rates of N were applied in order to obtain a wide range of N supply. Chlorophyll content was measured along the crop growth. Although absolute values of chlorophyll content index (CCI) changed in all treatments with crop age, the indicator was responsive to differences in N availability and it was able to discriminate between treatments It was also highly correlated with sugar yield and root quality.

Keywords: chlorophyll meter, fertiliser, nitrogen, sugar beet, yield

Background and objectives

Nitrogen (N) is an essential factor for yield and quality of sugar beet crop, so its management should be based on crop based indicators that take into account field and year specific characteristics, complementing soil based indicators (Schröder *et al.*, 2000). These indicators should discriminate between crop N availability, should be correlated with yield and quality and they should be based on reliable, fast, easy and practical methods. The aim of this work is to evaluate the leaf chlorophyll content of sugar beet crop as an indicator of nutritional status and thus as a potential fertilization decision support tool.

Material and methods

A trial was carried out in 2003 on a commercial field located in the Duero Basin (Spain). Cultivar Fresca was sown on 20 March. At harvest plant density reached 100,000 plant ha^{-1}. Irrigation scheduling was made with tensiometers as described by Arroyo (2002). The trial layout was randomized blocks with five replicates and four treatments of N application: 0, 90, 180 and 270 (over-fertilized) kg N ha^{-1}. The treatment of 180 kg N ha^{-1} was the rate recommended by the advisors in the area. Thus, a sufficient range of available N was achieved. These rates were split in three equal parts: one at preplanting (with urea fertilizer) and two sidedressings (with ammonium nitrate-sulphate fertilizer). This distribution of the fertilizer is the usual practice in this cropping area, as N splitting is recommended due to the length of the crop growth. Plot size was 36 m^2.

Leaf samples were taken during the crop growth at eight times: first and second sidedress (4-6 and 10-12 true leaves stages, respectively), two and four weeks later and four other times with a frequency of one month until harvest. Twenty young fully expanded leaves were sampled in each plot. Chlorophyll content index (CCI) of each leaf was measured with a CCM-200 Chlorophyll Content Meter (OPTI-SCIENCES) in the distal left part of the leaf, and an average value was calculated for each plot. A relative value was calculated for each treatment as the ratio of the plot value and the value of the over-fertilized treatment. On 7 November, a surface area of 10 m^2 of each plot was

harvested and crop yield (fresh root weight, sugar content and sugar yield) and quality parameters (invert sugars, α-amino N, sodium and potassium content and quality index) were measured.

Results and discussion

Absolute values of chlorophyll content index (CCI) changed in all treatments with the crop age, as described previously for a sugar beet crop by Wiesler *et al.* (2002). The relative values of CCI (Figure 1) showed differences between treatments during the crop growth, except in the first sampling date. Crop-colour based indicators fail to reflect different N supplies at earlier growth stages (Schröder *et al.*, 2000). CCI relative values of treatments 0 and 90 kg N ha^{-1} showed a decreasing trend during the crop growth, and these were lower than values of the over-fertilized treatment. Chlorophyll contents corresponding to treatment 180 kg N ha^{-1} showed a stable trend. So, this indicator was responsive to differences in N availability and it was able to discriminate between treatments as found by Wiesler *et al.* (2002) in sugar beet crop, and other crops (e.g. Blackmer *et al.*, 1996).

Figure 1. Trend over the crop growth of the relative CCI value.

There were statistical differences in yield and quality parameters between treatments (doses, Table 1), as described previously in other work (Gordo, 2003). Root yield increased with a higher amount of N applied, so the over-fertilized treatment reached the maximum yield, although this was not statistically different from the yield at 180 kg N ha^{-1}. However, sugar content showed a significantly decreasing trend as fertilizer rates increased. Thus, sugar yield was maximized at a rate of 180 kg N ha^{-1}. This yield was the same for the over-fertilizer treatment and higher than the sugar yielded at the lower N rates. With respect to crop quality, α-amino N and sodium contents were significantly higher at high N rates, so quality index decreased in those treatments, showing the worst quality in the over-fertilizer treatment.

Table 1. Crop yield and crop quality parameters.

	Crop yield			Crop quality		
Rate (kg N ha^{-1})	RY [1)] (t ha^{-1})	SC [2)] (%)	SY [3)] (t ha^{-1})	α-N [4)] (mmol 100g^{-1})	Na [5)] (mmol 100g^{-1})	QI [6)] (%)
0	116.0 b	19.2 a	22.3 b	0.98 a	0.60 a	88.3 a
90	120.4 b	18.8 b	22.6 b	1.22 b	0.64 a	87.7 a
180	132.6 a	18.1 c	24.1 a	1.68 c	0.72 b	86.4 b
270	136.5 a	17.6 d	24.1 a	2.30 d	0.86 c	84.5 c

Different letters within one parameter denote significant differences P≤0.05.
[1)] RY: root yield. [3)] SY: sugar yield. [5)] Na: sodium content.
[2)] SC: Sugar content. [4)] α -N: α -amino nitrogen. [6)] QI: quality index.

The correlation between relative values of the chlorophyll content and sugar yield was investigated. This was done with CCI values of the first four sampling dates. This is the period during which decisions on N sidedressing must be made. Good and positive correlation was found, as earlier reported by Wiesler *et al.* (2002). The best agreements in our experiments were found using quadratic functions. This means that sugar yield increases with chlorophyll contents and there is an optimum CCI value for maximising yield at each sampling date. However, precise optimum values should be obtained with a greater number of experiments.
In the same way, the correlation between indicator values and quality index was also examined. A negative and linear correlation was found between those parameters, so the root quality decreased as chlorophyll contents increased.

It seems that there is not a unique value of CCI which indicates a sufficient N supply during crop growth. Furthermore, chlorophyll contents are also determined by other sources of variation (i.e. cultivar, location and year, crop water status, point of measurement, meter type, meter batch, etc). The accuracy of readings was high in this experiment. The standard error of mean showed values between 2 and 4 units and variation coefficient was around 12%. Another experiment focusing on the effects of the position within the leaf on measurements (Arroyo-Sanz and Soler-Rovira, unpublished results), indicated that the variation of CCI values between leaves was larger than the variation within the same leaf.

Conclusions

Chlorophyll content performed well as crop nutrient status indicator, because it was able to discriminate between differences in N availability in the period when decisions on N sidedressing must be made. It was also highly correlated with sugar yield and root quality. This indicator, as compared with other plant-based indicators, is reliable, quick, easy and non-invasive. However, there is not a unique value of CCI that indicates a sufficient N supply during crop growth, and consequently an over-fertilized reference plot is necessary. Thus, the potential use as fertilization decision support tool is promising but further research is needed.

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Nitrogen fertilizer management based on chlorophyll measurement

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Keywords: chlorophyll meter, fertiliser, N-Tester, nitrogen fertilizer, SPAD,

Abstract

Plant analysis is a widely established tool for the identification of nutritional disorders in crops and thus can be used for fertilizer management. Optimum plant growth is closely related to optimum chlorophyll content, which in turn depends on sufficient supply with plant nutrients. This paper summarizes how in-field chlorophyll measurements can be used to assess the nutritional status of crops and the suitability for fertilizer management. The chlorophyll meters Minolta SPAD 502 and Yara N-Tester proved to be suitable tools for nutrient management by providing a rapid, in-field diagnostic method. Since chlorophyll is most sensitive to nitrogen it was possible to conclude from chlorophyll meter readings on the nitrogen fertilizer demand. Different options to use in-field chlorophyll measurement for fertilizer management are possible. The most simple use is to monitor the chlorophyll content of the crops over time in order to be able to detect latent nutrient deficiencies early without knowing the exact reason for decreasing chlorophyll concentrations. For many crops, however, a close relationship between leave chlorophyll content and N concentration has already been established. In this case, in-field chlorophyll meter readings could replace lab analysis on N. Based on extensive calibration work it is even possible to correlate the chlorophyll meter measurements to absolute rates of required N fertilizer. In that case, the chlorophyll meter is calibrated for given growth stages and defined leaf positions. Since the chlorophyll content is genetically affected, the recommendation has to be variety specific. In addition water stress and severe deficiencies of other nutrients, in particular of sulphur influence the chlorophyll meter reading and thus should be avoided.

Background and objectives

The use of plant analysis (covering a range of procedures from quantitative laboratory analysis to semi-quantitative 'quick' tests carried out in the field) is based on the idea that the plant itself is the best indicator for the nitrogen supply from the soil within the growth period. The actual N level in the plant is the result of the interaction between N from soil N mineralization, from pre-crop residues, water status, root growth, nitrogen uptake efficiency, etc. that influence nitrogen availability and uptake.

The simplest (and oldest) method to use the plant itself as an indicator for the assessment of the N requirement is a visual judgment based upon crop colour and crop density. Researchers tried to get some kind of standardization for assessment of the greenness and to turn the qualitative approach into a more quantitative one by using, for example, in-field chlorophyll measurement with chlorophyll meters such as the Minolta SPAD 502 or the Yara N-Tester. This article gives an overview of the possibilities to use in-field chlorophyll measurements for nitrogen fertilizer management.

Material and methods

The chlorophyll meter SPAD 502 (Minolta Corp., Japan) is a small handy device, which measures light transmittance at red (650 nm, chlorophyll absorption) and near-infrared (960 nm) wavelength. The additional measurement of the absorbed near-infrared light is a reference measurement to correct the chlorophyll measurement for external

impacts such as leaf thickness. The Yara N-Tester (Yara International ASA, Norway) is technically based on the SPAD 502 but was slightly modified and adopted to practical use by farmers (Neukirchen and Lammel, 2002).

Results and discussion

The strong correlation of chlorophyll concentration and SPAD/N-Tester value was validated for many crops such as wheat, rice, potato, maize and tobacco (Wood *et al.*, 1993; Castelli *et al.*, 1996; Vos and Bom, 1993).

Figure 1. Correlation between chlorophyll content and N-Tester readings of tobacco leaves.

Leaf N content and chlorophyll concentration are also strongly correlated (Schröder *et al.*, 2000; Wood *et al.*, 1993), which is the prerequisite to use chlorophyll measurements for nitrogen fertilizer management decisions. Although the leaf N concentration has the strongest effect on the chlorophyll meter readings also other factors may impact the measurement. Besides nitrogen, sulfur deficiency shows the strongest influence on the leaf chlorophyll concentration (Wells *et al.*, 1992). Thus, to derive a reliable N fertilizer recommendation from the chlorophyll measurement it is important to ensure sufficient sulfur supply of the crops. Under drought stress plants suffer from water deficiency and the chlorophyll concentration tends to increase (Ommen *et al.*, 1999), without representing a better nutritional status. Furthermore, the crop variety and the growth stage of the crop (Neukirchen and Lammel, 2002), the age of the leaf (Boquet *et al.*, 1999) and the point of measurement on the leaf itself (Hoel, 1998) impact the chlorophyll reading and, thus, have to be clearly defined, if the values are used for N fertilizer recommendations. Taking into account these possible sources of variation the use of chlorophyll measurement for N fertilizer management needs 'a strict sampling protocol' (Schröder *et al.*, 2000) in order to avoid misleading interpretation of the results.

Different options to use in-field chlorophyll measurement for fertilizer management are possible. The most simple use is to monitor the chlorophyll content of the crops over time in order to be able to detect latent nutrient deficiencies early without knowing the exact reason for decreasing chlorophyll concentrations. Since for many crops nitrogen fertilizer recommendations based on leaf N concentrations already exist (Shaahan, 1999), it is possible to use the good correlation between leaf N concentration, chlorophyll concentration and thus, also chlorophyll meter readings in-field chlorophyll measurements in order to replace time-consuming and costly lab analyses.

It is also possible to develop a nitrogen fertilizer recommendation that gives concrete N rates for chlorophyll meter readings. Defined sampling protocols together with concrete N fertilizer recommendations based on chlorophyll meter readings can either be based on a relative approach, e.g. on the ratio of chlorophyll meter readings in an over-fertilized reference plot compared to the rest of the field (Denuit *et al.*, 2002), or as an absolute recommendation scheme using variety-specific correction factors for the readings (Neukirchen and Lammel, 2002). In Germany over 100 field trials with winter cereals have been conducted since 1992 in order to establish N-Tester based recommendations for N topdressing.

Conclusions

Independent of the fact which plant indicator is measured in the field, each method of plant analysis has to be calibrated under field conditions. The real challenge of any plant analysis method is to transfer the obtained results into a fertilizer recommendation. In principle the result of a plant analysis can be used to decide about the necessity, about the optimum timing and about the optimum rate of a N fertilizer application. Such calibration algorithms have been developed for chlorophyll meter measurements based either on N response trials, comparative measurements on reference-plots or a mixture of both. The in-field chlorophyll measurement enables the farmer to match N supply with variations in crop N demand both spatially (i.e. between fields) and temporally (i.e. within and between seasons). This is an important component for maintaining optimum yields while improving the environmental performance of agricultural systems (Cassman, 1999).

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Field scale N-management: soil and plant diagnostic tools applied within the Agricultural Surface Survey (Belgium)

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Abstract

Within the PGDA farm framework, decision support tools for N-fertilization were tested: Azobil (INRA, Laon, France), N-TESTER (chlorophyll meter, Yara) and GPN (reflectance meter, Grande-paroisse). Three questions were addressed:

1. Could a digitalized pedological map adressing the spatial heterogeneity of SMN be of help for the PGDA sampling strategy? Not currently considered in the PGDA protocol, topographical and pedological parameters were chosen as additional factors. They showed significantly different levels of SMN (Soil Mineral Nitrogen).
2. Could residual SMN in autumn be used to qualify the whole crop N management? For sugar beet, response is moderated by the fact that, without any problematic increases of SMN, higher N-fertilization could lead to a significant increase of N in leaves, making higher mineralization and SMN levels likely at a later stage.
3. Do the GPN and N-TESTER tools improve the standard of N-fertilization monitoring in wheat crops? GPN seems a more effective tool for cereals.

Keywords: GIS, GPN, management, nitrogen, N-tester, pedology, soil map

Background and objectives

In Wallonia, the PGDA (Nitrogen Sustainable Management Programme in Agriculture) implements the Nitrate Directive CEE 91/676 (Vandenberghe *et al.*, 2005). Within PGDA farm framework, decision support tools for N-fertilization were tested to improve economic and environmental performance: N-TESTER (Bond, 2002) and GPN (reflectance-meter, Grande-Paroisse) were chosen given that levels of autumnal SMN are among the main PGDA requirements. Called OGRFGC, the project has to specify what seems relevant concerning fertilization monitoring and to feed it back to authorities, research structures and farmers. Spring advices are elaborated with Azobil (fertilization software, INRA Laon, France).

Three questions are addressed here:

1. Given the spatial heterogeneity of SMN, could a digitalized topo-pedological map be of use? It could be of importance given that PGDA requirements are essentially based on expected levels of SMN (according to the crop concerned).
2. Considering the prominence of residual SMN, how far can this parameter be used to qualify whole crop N-management?
3. GPN and N-TESTER are two reflectance-meters proposed by fertilizer producers (Grande Paroisse and Yara). Do they equally improve the N-fertilization monitoring of cereals ?

Materials and methods

Question 1. Not currently considered in the PGDA protocol (Vandenberghe & Marcoen, 2004), topographical and pedological parameters were introduced. Far from trying to check their influence on SMN, the operation was instead aimed at verifying whether they can help '*ex-ante*' to improve the sampling strategy. The tools used were numerical topographical (IGN, Belgium); geo-referenced parcel plan; and pedological (PCNSW, 2000) maps, displayed using the 'ArcView' software (ESRI). On that basis, 5 winter wheat parcels (main Walloon crop) in 2004 on 4 farms were subdivided by taking those parameters into account. SMN profiles were checked after harvest and in the first decades of October, November and December, as well as before spring application and after harvest of the following crop.
More precisely, the new parameters considered were (topography) top/terrace, slope and bottom, with a break of slope of more than 2.5% between each; and (pedology) differences in parent rock and natural drainage, which are distinguishable at a glance on cited maps. For drainage, a minimum gap of two classes was considered necessary to be taken into account.

Question 2. N rates applied to sugar beet (usual SMN at harvest non-problematic) or potato (highly problematic SMN at harvest) by farmers were compared to Azobil recommendations. Sugar beet data originated from a survey conducted in 1998/1999 in 5 sites Belgian loamy region (Renard, 2001). SMN levels were checked as well as leaf N concentrations.
Potato data were derived from split application experiments (4 parcels, 2 agricultural regions) in which an early 70% N rate (Azobil-based) was complemented by a second dressing if needed, as indicated by N-TESTER results (Olivier, Goffart, 2002). Post-harvest and haulm stripping SMN levels were checked. The objective was to determine whether these two types of SMN were any different, thus having consequences on the farmer's responsibility and the reliability of residual SMN, and, secondly, to verify the influence of that method on SMN.

Question 3. Appraisals of the N-status (leaves) of wheat/barley crops using the GPN and N-TESTER were carried out at Zadoks stages 37 and 47 from 3 different sites, 4 parcels (Cuvelier, 2005), and compared with a reference method (N total NIR analysis). Yields and SMN profiles were also recorded for 2 parcels. Factors considered were mineral fertilization (2 doses), seedling density (one usual and one 2/5 less than usual), and pedology.

Results and conclusions

Question 1. Table 1 shows the 2004 and 2005 SMN results for each subparcel. Parcels were situated in four villages (two agricultural regions), with more than 25 km between each other.

Table 1. SMN (kg ha^{-1}) for parcels (wint. wheat in 2004) with topo-pedological segmentation.

Parcels	Subparcels		2004 Post-h.1	Cover crop	Oct.	Nov.	Dec.	2005 Spring	Post-h.2
3	A	d B[1]	53	Y	MD	86	33	208	89
	G	b S[1]	33	Y	MD	42	55	301	72
4	A	d B[1]	49	Y	78	91	84	56	NYA
	G	b P[1]	58	Y	61	78	60	79	NYA
	A	b S[1]	35	Y	43	67	57	53	NYA
8	A	d[1]	18	Y	61	41	50	28	NYA
	G	b[1]	27	Y	71	64	22	30	NYA
20		d[1]	17	N	MD	81	51	78	28
		b[1]	26	N	MD	73	80	72	46
35		b S[1]	29	N	MD	49	38	MD	15
		d B[1]	104	N	MD	123	110	MD	108

NYA: Not yet available. Y: Yes.
MD: Missing data. N: No. All data in kg ha^{-1}; depth = 90 cm.

[1] *Parameters characterizing subparcels (cited if different): parent rock/texture ('A' = loamy soils or 'G' = pebbly soils, i.e. if > 5% pebbles); soil natural drainage ('a' = excessive, 'b' = favorable, 'c' = moderate, 'd' = imperfect); and topography (T/P = Terrace, S = Slope, B = Bottom).*

Topo-pedological subdivisions showed significantly different levels of SMN. When available, the sign of the difference between subparcels is the same for both years. For subparcels that differ for at least two parameters, the SMN levels are almost always significantly different, even if the sign of the difference may vary (possible explanation: cover or winter crops grow more where SMN levels are high, bringing the levels down afterwards).
Needing no field observation, the digital maps improved sampling precision, even if modalities, causal links, thresholds (here empirical) must be better thought through. More than just easily improving accuracy (interesting for farmers), these maps could also allow reliable 'remote' sampling organization (i.e. in case of broad SMN measures to control legal thresholds), and so be advantageously added to the PGDA protocol.

Question 2. Farmers' doses often exceed Azobil advice by 30% (Table 2). A higher fertilization dose could lead to a significant increase of N in leaves without any problematic increase of SMN (sites 3, 4, 5 in '98, 2 and 5 in '99). In such cases, increased mineralization after harvest from leaf residue is likely. For potato, this year's data will indicate whether the proposed alternative brings about improvement in terms of SMN residuals.

Table 2. N in leaves and soil after harvest of sugar beet (two years, two N mineral doses).

Sites		1998 N-min F	N in leaves	SMN (soil)	1999 N-min F	N in leaves	SMN (soil)
1	Azobil dose	105	157	74	120	121	23
	Farmer dose	150	160	140	180	110	20
2	Azobil dose	136	169	53	80	115	13
	Farmer dose	175	182	89	145	126	11
3	Azobil dose	80	87	13	175	90	23
	Farmer dose	140	113	10	140	72	22
4	Azobil dose	0	91	11	135	91	18
	Farmer dose	90	154	19	150	84	14
5	0	not done	not done	not done	0	117	22
	Azobil	55	140	20	100	145	25
	Farmer dose	150	212	31	150	162	35

N-min F: applied mineral N fertilizer; N in leaves: micro-Dumas method; all units in kg ha^{-1}.

Question 3. Because GPN measures reflectance upon the crop and not for single leaves, it seems (Table 3) a more effective tool for cereals, given links between biomass, tillering and seedling. Correlations between GPN /N-TESTER results, yield and N soil residuals tend also to show clearer relationships between yield and GPN.

Table 3. Correlations between GPN, N-TESTER and NIR N-analysis (winter cereal leaves).

Factor modulated	Fertilization modalities				Seedling density			
Site	1		2		1		2	
Zadocks stage	37	47	37	47	37	47	37	47
GPN/NIR	0.73	not done	0.82	0.83	0.73	-0.14	0.75	0.75
N-Tester/NIR	-0.24	0.32	0.86	0.91	-0.49	-0.10	0.94	0.73

All data are correlation coefficients.

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Nitrogen fertilization as strategy to improve salinity tolerance in citrus trees

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Abstract

On the Mediterranean coast of Spain, about eighty per cent of citrus commercial orchards are irrigated with aquifer water; many of them are saline. Previous studies have reported that N fertilization could reduce salt damage in citrus plants. The aim of this work was to evaluate the effect of different nitrogen (N) rates (5 and 8 mM) and sodium chloride (NaCl) levels (0, 20 and 40 mM) on N and Cl uptake, biomass yield, and some soil parameters. Plants of Nules clementine (*C. Clementine* Hort. Ex Tan) grafted on Cleopatra mandarin (*C. reticulata* Blanco) and Carrizo citrange (*C. sinensis x P. trifoliata*) rootstocks were used. Labelled potassium and calcium nitrate (5% ^{15}N enriched), and ammonium nitrate were used as N source. The trees treated with 20 and 40 mM NaCl had significantly lower biomass yields. The N absorbed from fertilizer decreased in the 20 and 40 mM NaCl treatments, in comparison with 0 mM NaCl treatment. N uptake was significantly increased at 8mM N rate, (at 20 and 40 NaCl mM.). Salt treatments caused chloride to accumulate in the whole tree of both rootstocks. Increasing N rate resulted in a significantly lower chloride accumulation in Carrizo citrange at 20 mM NaCl. The 8 mM N rate resulted in a significant decrease in the sodium adsorption ratio (SAR) values. The exchangeable sodium percentage (ESP) increased when enhancing the NaCl rates.

Keywords: ^{15}N, biomass, N and Cl uptake, NaCl, rootstock, soil properties

Background and objectives

On the Mediterranean coast of Spain, about eighty per cent of citrus commercial orchards are irrigated with aquifer water; many of them are salinized (from 20 to 40 mM), which represents an increasingly common problem. Previous studies have reported that N fertilization could reduce salt damage in citrus plants. The aim of this work was to evaluate the effect of different N rates and NaCl levels on N and Cl uptake of young citrus trees. Also yield biomass and soil physico-chemical parameters were studied. This information is necessary to develop Best Management Practices for citrus crops in areas with salt water irrigation.

Materials and methods

Plants of Nules clementine (*C. Clementine* Hort. Ex Tan) grafted on Cleopatra mandarin (*C. reticulata* Blanco) and citrange Carrizo (*C. sinensis x P. trifoliata*) rootstocks were grown outdoors in pots filled with 25 kg of a sand-loamy soil. The rootstocks differ in salt tolerance, the Carrizo being less tolerant. Twelve treatments were applied (2 rootstocks, 2 N levels, 3 salt levels) through a drip irrigation system. The N levels were 5 and 8 mM (N1, N2) and the salt levels were 0, 20 and 40 mM (NaCl-0, NaCl-20 and NaCl-40). Four trees were used for each treatment. Labelled potassium and calcium nitrate (5% ^{15}N enriched) and ammonium nitrate were used as N source. The trees were harvested and fractioned in several organs at the end of the vegetative cycle (November). Statistical analysis was performed by analysis of variance (ANOVA), the statistical significance of differences among means was determined using LSD-Fisher ($P \leq 0.05$) test.

Results and discussion

Tree biomass was reduced by salt application (NaCl-20 and NaCl-40); no differences due to rootstock or N rate were found (Figure 1A). Bañuls *et al.* (1990) also reported that citrus growth was reduced when applying different NaCl rates to these rootstocks. The highest N rate resulted in a slight increase in tree biomass, not significant at P≤0.05 but significant with an 84% of probability. In this way, Iglesias *et al.* (2004) found a biomass increase when supplementing nitrate to salinized citrus seedlings.

Rootstock	NS	RxNaCl	NS
NaCl	***	RxN	NS
N	NS^	NaClxN	NS

Figure 1. *Effect of NaCl (NaCl-0, NaCl-20, NaCl-40= 0, 20, 40 mM respectively), N (N1= 5 mM, N2=8 mM) and rootstock (ClM: Cleopatra mandarin, CC: Carrizo citrange) on the whole tree biomass (A) and on the N absorbed by the whole tree (B). Significant effects of treatments are given at P≤0.001 level (***) and at P>0.5 no significant (NS). ^Significant at P=0,16. Same letters are not significantly different at 5% level (LSD-Fisher test).*

The N absorbed from fertilizer by the whole tree decreased significantly for NaCl-20 and NaCl-40 in comparison with NaCl-0 (Figure 1B); the reduction ranged from 40 to 60% respectively. Cerezo *et al.* (1999) also reported a reduction in N absorption of the same rootstocks. On the other hand, N uptake was significantly increased at N2, at NaCl-20 and NaCl-40, up to 30%.

Salt treatments caused chloride to accumulate in the whole tree of both rootstocks. Figure 2A shows that all trees grafted on ClM (salt tolerant) accumulated less Cl in the tree top than those grafted on CC (salt sensitive). The same results were found for the root system (Figure 2B). These results are consistent with those by Moya *et al.* (2003) who found that the tolerant genotype Cleopatra 'excluded' more chloride than Carrizo, as it absorbed lower amounts of chloride per volume of water, which seemed to be due to a less efficient root system. Increasing N rate (N2) resulted in a significantly lower chloride accumulation in CC at NaCl-20, both in the tree top and root system. These findings are in line with the report of Iglesias *et al.* (2004) who also found that nitrate supplementation significantly inhibited chloride accumulation in roots of salt-treated Navelina orange grafted on Carrizo citrange, but not Cleopatra mandarin. Nitrogen concentrations N1 and N2 had no effect on chloride accumulation in ClM.
The sodium absorption ratio (SAR) and the exchangeable sodium percentage (ESP) increased with increasing NaCl applications (Table 1). SAR was lower at N2 than at N1 and no effect of N on ESP was found. The cation exchange capacity of the soil ranged from 23.3 to 25.0 meq Na/100 g soil and no differences were found.

Table 1. Effect of NaCl (NaCl-0, NaCl-20, NaCl-40= 0, 20, 40 mM respectively), N (N1= 5 mM, N2=8 mM) and rootstock (Cleopatra Mandarin, Carrizo citrange) on sodium absorption ratio (SAR) and exchangeable sodium percentage (ESP). Same letters are not significantly different at 5% level (LSD-Fisher test).

	SAR	ESP
Carrizo citrange		
NaCl-0+N1	0.22a	3.14a
NaCl-0+N2	0.26a	3.66a
NaCl-20+N1	0.60c	8.22b
NaCl-20+N2	0.51b	7.16b
NaCl-40+N1	0.92e	12.39c
NaCl-40+N2	0.89d	12.06c
Cleopatra mandarin		
NaCl-0+N1	0.23a	3.13a
NaCl-0+N2	0.25a	3.55a
NaCl-20+N1	0.64c	8.52b
NaCl-20+N2	0.50b	6.92b
NaCl-40+N1	0.90e	11.70c
NaCl-40+N2	0.88d	11.93c

Figure 3. Effect of NaCl (NaCl-0, NaCl-20, NaCl-40= 0, 20, 40 mM respectively), N (N1= 5 mM, N2=8 mM) and rootstock (CIM: Cleopatra Mandarin, CC: Carrizo citrange) on the Cl concentration in the tree top (A) and in the root system (B). Same letters are not significantly different at 5% level (LSD-Fisher test).

Conclusions

The present experiment showed that an increased N rate (from 5 to 8 mM) resulted in lower chloride uptake by trees grafted on the salt sensitive root stock Carrizo citrange at 20 mM NaCl. The additional N fertilization also increased the N absorbed at saline conditions (20 to 40 mM NaCl) by Carrizo citrange and Cleopatra mandarin rootstocks. In addition, N supplementation is able to diminish SAR rate, improving the soil permeability. In conclusion, moderate increases of the conventional N fertilization could minimize the damages originated in citrus plants grafted on salt sensible rootstocks when using salty water.

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Decision support systems for nitrogen fertilization to maintain production and reduce potential N-losses: Main questions and answers in the Netherlands

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Abstract

High fertilizer rates in the Netherlands lead to relatively high nutrient losses to the environment. The aim of this research was to develop decision support systems which achieve maximal crop production at reduced nitrogen losses. The decision support systems intend to synchronize nitrogen supply and nitrogen demand for crop x soil combinations which are susceptible to nitrogen losses: Potato (*Solanum tuberosum* L.), leek (*Allium porrum* L.), and hyacinth (*Hyacinthus orientalis* L.) on sandy soils.
Three main N-application questions were based at the synchronization of supply and demand.
The first question 'when / how much' was answered by different split nitrogen application (SNA) systems based on soil or crop measurements: SNA-soil, SNA-Cropscan and SNA-petiole reduced potential N-loss by 10-35 kg N ha^{-1}, dependent on crop and the standard system with which it was compared. Answering the second question 'where' by bed-application rather than broadcast N-application, resulted in a reduction of potential N loss with 32 kg N ha^{-1}in hyacinth. The third question 'how / what' answered by using fertigation or Entec (a slow release fertilizer) resulted in no effect or an increase in potential N-loss. Yield levels remained stable.

Background and objectives

Fertilizer use is high in countries where its cost is low compared to returns in crop production. In Europe high nitrate levels in drinking water, and deterioration of ecosystems urged the European community to set the maximum allowable level of nitrate in groundwater at 50 mg L^{-1}. The nitrate level in groundwater under agriculture on dry sandy soils in the Netherlands varies between 0 and 500 mg L^{-1} with an average of about 80 mg L^{-1} (Hack ten Broeke *et al.*, 2004). These high nitrate levels are largely due to long term net nitrogen (N) input of agriculture of ca. 280 kg N ha^{-1} yr^{-1} (OECD, 2001). Farmers are urged to decrease fertilizer use, but fear a decrease in crop production and quality, due to suboptimal N fertilization, while their position in the international market is vulnerable. The objective of this research was therefore to design and test fertilization systems which increase synchronization of N-application and N-demand of the crops and thus maintain crop production at lower N input levels, with a concomitant decrease in N-loss to the environment.

Materials and methods

During 2002 and 2003 experiments were conducted on sandy soils in the Netherlands in several crops with low N-efficiency: Potato, leek and hyacinth. In each of these crops the standard method of fertilization, the control treatment, was compared to alternative fertilization systems. Criteria were quantity and quality of crop production, and potential N-loss. In potato (var. Seresta) the standard N-fertilization system (the control) was one basic application of 225 kg N ha^{-1}at planting. The alternative fertilization systems had decreased N-applications at planting by 1/3 – 1/2. Supplementary N-applications (SNA) were determined twice during the growing season a) by analysis of mineral N in the soil and crop-N-demand according to a standard N-uptake curve (= treatment SNA-soil), b) by analysis of crop light-reflection, derived crop N-content and further crop N-demand dependent on present and

expected biomass production (= treatment SNA-Cropscan) (Booij *et al.*, 2001), and c) measurement of nitrate in sap of petioles and comparison with norm nitrate contents at the particular growth stage (treatment SNA-petiole).
In leek (autumn) the standard N-fertilization system was: at planting 120 kg N ha^{-1} minus the mineral N-stock in the top 0-30 cm soil, after 6 weeks 75 kg N ha^{-1} and after 12 weeks 75 kg N ha^{-1}. The alternative fertilization systems received 35 kg N ha^{-1} less at planting and the rate of the other two N applications was determined by a) SNA-soil and b) SNA-Cropscan, c) SNA-soil but with the usual fertilizer Calcium Ammonium Nitrate (CAN) in the second and third applications replaced by Entec (a fertilizer with increased ammonium content and a nitrification-inhibitor).
In hyacinth the standard N-fertilization system was SNA-soil (as described for potato) with monthly N applications from February till end of May. The first two alternative fertilization systems kept SNA-soil, but in a) the N-rate was reduced by 30%, by fertilizing only the flower-beds and not the paths which make up 30% of the field area (= treatment bed application), and in b) the N was applied dissolved in water through perforated drip irrigation tape permanently positioned along the plant rows (= treatment fertigation). The third alternative was SNA-Cropscan instead of SNA-soil, in which small plots with 1.3 x SNA-soil were used as a reference to determine whether N application would be necessary or not. In the fourth alternative within SNA soil, the usual fertilizer CAN/CN was replaced by Entec.

Results and discussion

Alternative fertilization systems did not show significant yield decreases as compared to standard fertilization (Table 1). On the other hand potential N losses to the environment were clearly reduced when using most of the alternatives. Increased potential N loss by SNA-soil in potato was mainly due to the erratic N uptake curve used for potato in SNA-soil. Use of the alternative fertilizer Entec, meant to be less susceptible to leaching, did not decrease but even increased potential N-loss.

Although originally the experiments were meant to compare the different fertilization systems, Figure 1 shows why comparison was not always appropriate. Comparing different fertilization systems is only appropriate if two systems attempt to answer the same N-

Table 1. Yield and potential N-loss differences between standard N-fertilization and alternative N-fertilization systems for potato, leek and hyacinth, and the question of the decision support system answered.

Crop	Alternative fertilization system	Application question answered by alternative	Difference yield	Difference potential N-loss (kg N ha^{-1})
Potato	SNA-soil	when / how much	ns	+ 39
	SNA-Cropscan	when / how much	ns	- 32
	SNA-petiole	when / how much	ns	- 30
Leek	SNA-soil	when / how much	ns	- 31
	SNA-Cropscan	when / how much	ns	- 35
	Entec	what / how	ns	0
Hyacinth	SNA-Cropscan	when / how much	ns	- 11
	Bed-application	where	ns	- 32
	Fertigation	what / how	+	- 2
	Entec	what / how	ns	+ 15

Potential N-loss calculated by: N application - (N in exported product from alternative fertilization system - N in exported product from 0-N-plot), SNA: supplementary nitrogen application, ns: non significant difference.

Figure 1. Decision support system for N-application: three main questions and their presence / ability to reduce N-loss in the standard and alternative fertilization systems.

Application question: 1) When and how much?, 2) Where? and 3) How/What?. If two fertilization systems answer different questions they can be combined: Thus bed-application (answering the question 'Where?') can be combined with *either* SNA-Cropscan *or* SNA-soil in Hyacinth (both answering the question 'When/how much'?). If the standard fertilization system already answered one of these questions, the improvement an alternative system could make was smaller than if the standard system did not yet attempt to answer the question: SNA-Cropscan in hyacinth reduced the potential N loss less than SNA-Cropscan in potato, partly because in hyacinth the standard system answers the question 'when / how much' already. Comparison of different fertilization systems answering the same question, like SNA-soil and SNA-Cropscan in potato, revealed that the whole fertilization system is as strong as the weakest link. The main weak link in SNA-soil for potato was an erratic N-uptake curve. Once a weak link is improved (i.e. improving the N-uptake-curve for potato) a system may function satisfactorily.

Conclusions

Decision support systems to reduce N-fertilization and losses while maintaining yield have to ask three main questions with respect to N-application: 1) When how much, 2) Where and 3) How/What. Fertilization systems answering different questions can be combined. In potato and leek answering the first question by SNA-soil, SNA-Cropscan or SNA-petiole led to the largest reductions in potential N-loss of 30-40 kg N ha^{-1} at stable yield levels. In hyacinth answering the second question by bed-application of fertilizer led to the largest reductions in potential N-loss of 32 kg N ha^{-1} at stable yield level. Answering the third question by fertigation in hyacinth or the use of Entec in leek and hyacinth did not reduce or even increased potential N-loss. Comparison of fertilization systems is inappropriate if a) two systems answer a different N-application question and b) a system contains flaws like the erratic N-uptake curve for potato in SNA-soil.

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In situ evaluation of nitrogen status of *Eucalyptus globulus* Labill. stock plants using a chlorophyll meter

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Abstract

With the aim of evaluating the use of the portable chlorophyll meter 'Minolta SPAD 502' readings (SPAD readings) as an indicator of the nitrogen status of *Eucalyptus globulus* Labill. hedge stock plants, a nitrogen fertilization experiment was performed. During 3 years plants were fertilized, periodically, with complete nutrient solutions containing 50, 100, 200, 400 and 800 mg N L^{-1}. Increasing levels of applied nitrogen till 400 mg N L^{-1}, led to an increase of total biomass, a higher production of stem cuttings and an improvement of the rooting ability of the cuttings.

A quadratic and plateau model was fitted to the relationship 'SPAD readings *vs* relative production' and the R^2 values found (between 77 and 85%) were similar to those obtained for 'leaf nitrogen content *vs* relative production'(78 to 92%). The relationship between SPAD readings and leaf nitrogen contents was linear, positive and the slope was significantly different from zero. Results showed that SPAD readings can be used to evaluate nitrogen status of *E. globulus* hedge stock plants with a diagnostic quality similar to leaf nitrogen contents. SPAD readings between 42 and 46 represent leaf nitrogen contents ranging from 25 to 30 g N kg^{-1} and were considered adequate for the production of high quality cuttings.

Keywords: *Eucalyptus globulus*; nitrogen, SPAD, vegetative propagation,

Background and objectives

Eucalyptus globulus Labill. is a forest tree widely grown in Mediterranean climates. Nowadays, traditional propagation of *E. globulus* by seed is being replaced by vegetative propagation by stem cuttings. As a consequence, nurseries with a high density of hedge stock plants are being established in several countries (Ribeiro, 2004). If stock plants are maintained as hedges, information is needed regarding hedge culture and management (Wilson, 1999). However, for *E. globulus*, there is no sufficient information related to the management of stock plant N fertilization and to the use of N status indicators. This aspect is of most importance as N affects both shoot production and adventitious rooting (Rowe *et al.*, 2002). The main purpose of this study is to evaluate the use of the portable chlorophyll meter 'Minolta SPAD 502' readings (SPAD readings) as an indicator of the nitrogen status of *E. globulus* hedge stock plants.

Materials and methods

A nitrogen fertilization experiment was performed with two clones (HD 161 and CN5) of *E. globulus* hedged stock plants grown in an open-air nursery in 12 L pots. During 3 years plants were fertilized, periodically, with complete nutrient solutions containing 50, 100, 200, 400 and 800 mg N L^{-1}. The annual amount of applied nitrogen was 0.5, 1.0, 2.0, 4.0 and 8.0 g N $plant^{-1}$ in the first year, and 3.5, 7.0, 14.0, 28.0 and 56.0 g N $plant^{-1}$ in the second and third years. In 1999, stock plants were subjected to formative pruning. In 2000 and 2001, biomass, number of cuttings produced by the stock plants and rooting ability of the cuttings were evaluated. During this period nitrogen content and SPAD readings of the youngest fully expanded leaves were measured at 5 different sampling dates. Data on cutting production and rooting ability were converted to 'relative production' expressed as a percentage of the maximum cutting production or rooting ability obtained at each sampling date.

Results and discussion

Increasing levels of applied nitrogen led to: i) an increase of total biomass ii) a higher production of stem cuttings iii) an improved rooting ability of the cuttings (Table 1). Results show that optimum N level for the production of cuttings (400 mg L^{-1}) also had the highest rooting ability. Leaf nitrogen content and SPAD readings increased asymptotically with increasing levels of applied nitrogen. At each sampling date, coefficients of determination higher than 90% were obtained when the Mitscherlich modified model ($y= a (1-be^{-cx})$) was fitted to the data. This asymptotic response of SPAD readings is similar to the relationship between nutrient concentration in the plant and nutrient availability in the soil, that generally follows an asymptotic curve (Mengel and Kirkby, 2001).

Table 1. Effect of nitrogen concentration on biomass production (g plant^{-1}), number of stem cuttings produced (nº cuttings plant^{-1}) and rooting ability of cuttings (%).

Clone	Nitrogen (mg L^{-1})	Biomass		Cuttings		Rooting	
		year 2000	year 2001	year 2000	year 2001	year 2000	year 2001
	50	119.2 ab [1)]	159.4 b	15.4 a	35.4 b	50.0 dc	69.1 bcd
	100	240.1 c	272.5 c	25.0 c	57.8 c	55.2 ed	63.7 b
HD161	200	357.5 d	381.8 e	40.9 d	87.9 e	65.0 e	78.9 de
	400	464.6 e	592.6 g	51.8 f	123.1 hg	79.4 f	92.5 f
	800	489.8 e	600.0 g	48.2 ef	132.0 h	78.1 f	89.3 ef
	50	63.4 a	98.6 a	12.3 a	24.1 a	27.8 a	37.2 a
	100	164.0 b	192.9 b	20.4 b	44.1 b	34.5 ab	46.8 a
CN5	200	262.2 c	324.0 d	27.5 c	69.7 d	42.6 bc	58.4 b
	400	476.0 e	509.7 f	44.4 de	105.4 f	54.8 de	76.4 cd
	800	485.3 e	541.5 f	48.1 ef	113.6 gf	60.5 e	66.2 bc

[1)] *In each column, values with the same letter are not significantly different at $p \leq 0.05$.*

A quadratic and plateau model was fitted to the data: 'SPAD readings *vs* relative cutting production' and 'SPAD readings *vs* relative rooting ability' (data of individual clones and grouped data of both clones). R^2 obtained varied between 77 and 85%, with 78 and 79% for the grouped data of both clones (Figure 1). These values were similar to those obtained when the model was fitted to the relationship 'leaf nitrogen *vs* relative cutting production' and 'leaf nitrogen *vs* relative rooting ability' (78 to 92%, and 80% for the grouped data of both clones).

Figure 1. Relationship between SPAD readings and 'relative production of cuttings' and 'relative rooting ability'.

The slope of the linear regression (95% confidence interval of 0.704 to 0.795) relating SPAD readings (*y*) to leaf nitrogen content (*x*, g kg^{-1}) was significant larger than 0 and smaller than 1. In addition, no differences between clones were found (Figure 2). Similar linear trends have been found by several authors for different species (Shaahan *et al.*, 1999; Chang and Robinson, 2003). In fact, proteins of Calvin cycle and thylakoids represent the majority of

leaf nitrogen, and thylakoid nitrogen is proportional to chlorophyll contents. Therefore, within species there is a strong linear relationship between nitrogen and chlorophyll content (Evans, 1989).

Figure 2. Linear relationship between leaf nitrogen contents (g N kg $^{-1}$) and SPAD readings on the youngest fully developed leave of E. globulus stock plants. n=150.

According to Ribeiro (2004), leaf nitrogen contents between 25 and 30 g N kg^{-1} can be considered as the adequate range for *E. globulus* stock plants, which indicates that SPAD readings between 42 and 46 are also adequate for optimal production of high quality cuttings.

Conclusions

Results showed that SPAD readings can be used to evaluate nitrogen status of *E. globulus* mother plants with a diagnostic 'quality' similar to leaf nitrogen contents. The relationship between SPAD readings and leaf nitrogen contents was linear, positive and significant. SPAD readings between 42 and 46 represent leaf nitrogen contents ranging from 25 to 30 g N kg^{-1} and were considered adequate for the production of high quality cuttings.

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Quick sap tests for nitrate content in romanesco plants, and its variation with time and soil mineral nitrogen

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Abstract

Petiole sap nitrate concentration in recently matured leaves was tested as an indicator of crop nitrogen status in romanesco as a tool to improve nitrogen fertilization. Two rapid methods of sap analysis were assessed: a selective ion meter, Cardy Meter® and a hand-held reflectometer RQflex©. A trial was conducted at an experimental field, in L'Horta de Valencia (Spain), with four levels of available soil mineral nitrogen in a completely randomised block design with three replicates. Results obtained with the rapid methods were correlated with those obtained by a laboratory method and a correlation coefficient (r) of 0.90 was obtained for the Cardy Meter® and 0.94 for the RQFlex2©. Sap nitrate concentration increased with the available mineral N in soil and varied with time: increased after fertilization, and then tended to stabilize. The two quick methods for sap analysis gave values that were correlated with the N application rate and available soil mineral N. The critical nitrate concentration in sap at mid season (66 days after planting) was about 2800 mg NO_3^- l^{-1}.

Keywords: nitrate, nutrition, plant sap analysis

Background and objectives

In some parts of Europe, agriculture is responsible for more than 80% of nitrogen emissions to aquatic ecosystems (EPI, 2001). In Spain there are many agricultural zones with groundwater polluted with nitrate, especially in the Valencian Community (European Commission, 1999; ITGME 1998). Increasing health and environmental concerns now demand effective systems of fertilizer recommendation. Plant nutritional measurements have been used as a guide for N fertilisation in many crop plants (Matthäus and Gysy, 2001; Villeneuve *et al.*, 2002). An alternative to soil and plant standard analysis has been petiole sap quick tests. Several methods and equipments as hand-held reflectometers or selective ion meters have been developed and tested in different species with satisfactory results (Hartz *et al.*, 1994; Hochmuth, 1994; Williams and Maier, 2002). Although some studies have been done with broccoli (Kubota *et al.*, 1997; Villeneuve *et al.*, 2002) no data are available for romanesco in our conditions. Romanesco is prone to contribute to groundwater nitrate pollution due to its high nitrogen requirement. The aim of this study was to assess different methods of nitrate sap analysis in romanesco plants and their use as a tool for nitrogen fertilizer recommendation under the Valencian Community conditions.

Material and methods

Romanesco (*Brassica oleracea* var *botrytis* L.) cv. Veronica plants were grown in an experimental field in 'L'Horta de Valencia' (Spain) and subjected to four N fertilisation treatments to obtain four levels of available soil mineral N: N0 (170 kg N·ha^{-1}), N1 (230 kg N·ha^{-1}), N2 (290 kg N·ha^{-1}) and N3 (350 kg N·ha^{-1}). The experimental design was a complete block design with four N treatments and three replicates. The fertilizer was applied in one single application at 55 DAP (days after planting). At three times during the growing cycle, 15 recently matured leaves from each experimental unit were excised early in the morning, placed in plastic bags and stored in a cooler until processed. In the laboratory, the midribs were cut and squeezed in a stainless steel garlic crusher and NO_3^- concentration in the sap was determined by three methods: a portable selective ion meter Cardy Meter® (Horiba, Kyoto, Japan), a hand-held reflectometer RQflex© (Merck, Darmstadt, Germany) as rapid on-farm methods, and a

laboratory method (Sempere *et al.*, 1993). Soil mineral nitrogen from 0 to 90 cm. depth was determined before transplanting, at mid development stage and after harvest.

Results and discussion

Petiole sap nitrate concentration measured by the rapid methods was highly correlated with the laboratory method (Figure 1). They were fit with linear relationships (r = 0.90 and r = 0.94 for the selective ion meter and for the reflectometer, respectively) across a wide range of nitrate concentrations. The effects of method on nitrate nitrogen determination are estimated by differences in the intercept. No significant differences were found between intercepts in both methods.

Sap nitrate concentration increased with the N application rate and varied with time: there was a sharp increase after fertilization (66 DAP), afterwards tended to stabilize or slightly increased at 15 cm diameter curd stage (91 DAP) (Figure 2). As Vitosh and Silva (1996) observed in potato, at higher N rates sap nitrate tended to remain steady or showed only a small decrease with time. Petiole sap nitrate was very sensitive to the different nitrogen treatments with both portable devices. The lowest N min available treatment (N0) had lower sap nitrate content during the growing period.

The two quick methods for sap analysis gave values that were correlated with soil available mineral N. They were both fit with quadratic relationships with r=0.98 (Figure 3). Sap nitrate concentration increases as soil mineral N did until 280 kg N·ha^{-1}. In addition, greater yields (32 t·ha^{-1}) were reached with N2 treatment, higher amounts of fertilizer (N3) did not increased yield (data not presented). Therefore it could be deduced that at 280 kg N·ha^{-1} the critical concentration in sap a mid season (66 DAP) was about 2,800 mg NO_3^- ·l^{-1}.

Figure 1. Sap nitrate concentration (mg NO_3^-·l^{-1}) measured by Cardy Meter® and hand-held reflectometer RQFlex© compared to the laboratory method (Spectrophotometer UV).

Figure 2. Sap nitrate concentration (mg NO_3^-·l^{-1}) evolution during the growing cycle measured by Cardy Meter® and hand-held reflectometer RQFlex©, respectively. Treatments: N0 (170 kg N·ha^{-1}), N1 (230 kg N·ha^{-1}), N2 (290 kg N·ha^{-1}) and N3 (350 kg N·ha^{-1}).

Figure 3. Effect of soil available mineral nitrogen on sap nitrate concentration (mg NO_3^-·l^{-1}) at mid season measured by hand-held reflectometer RQFlex© and Spectrophotometer.

Conclusions

The quick methods for nitrate sap analysis gave results highly correlated with the laboratory method, and nitrate levels in plant sap varied with different levels of nitrogen supply, and with time during the different stages of the romanesco growing period. Sap nitrate analysis has shown to be a sensitive indicator of crop nutrient status. Petiole sap nitrate determined by either the Cardy Meter® or the RQflex© can be a quick and useful tool to assess romanesco nitrogen status.

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Effect of different nitrogen sources on growth and yield of durum wheat in Mediterranean climate

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Abstract

Nitrogen is one of the most important nutrients for plant growth, but its irrational and excessive use can cause serious problems for environment and human health. Nitrogen recovery can be optimised by correctly defining the rate and timing of application and the source of fertiliser. In order to compare the effect of different nitrogen sources (traditional, organic, organic-mineral, slow and controlled release fertilisers) on the growth and yield of a herbaceous crop in a Mediterranean climate a trial has been carried out in pots on durum wheat (*Triticum turgidum* L.) in Southern Italy. With the same amount of N applied (75 kg N ha-1), eight fertilisation treatments were compared to an unfertilised control. During the crop cycle soil mineral nitrogen content was monitored, SPAD [explain] values were measured and N use efficiency was determined. The results showed that, despite the initial high soil fertility, the highest yield and N uptake efficiency were obtained via split application, but above all by supplying the plants with a source of readily available N during the maximum uptake period. Good results were also shown by the two coated and the stabilised fertilisers, but further investigations in poorer soils are needed.

Keywords: durum wheat, fertilizer, nitrogen, use efficiency

Background and objectives

Nitrogen is one of the most important nutrients for plant growth, but its irrational and excessive use can cause losses through leaching and volatilisation, accumulation as nitrate in the edible parts of the plants and, consequently, serious problems for environment and human health. Nitrogen recovery can be optimised by correctly defining the rate and timing of application and the source of fertiliser that sometime is of critical importance. In the last decades several commercial products have been synthesised which, through different mechanisms, are able to release nutrients gradually and therefore supply the plants throughout the crop cycle. Due to their high cost, many of these fertilisers have been used only for high value crops; moreover time and modality of nutrient release, which depend on many factors such as soil moisture, temperature and microbial activity, are unfortunately not easily predictable, so that the slow release not always matches crop demand. In order to compare the effect of different fertiliser sources on the growth and yield of a herbaceous crop in a Mediterranean climate a trial has been carried out on durum wheat (*Triticum turgidum* L.) in Southern Italy.

Material and methods

The trial has been carried out in pots at the experimental station of Bari University. The pots were filled with a clay-loamy soil characterised by high chemical fertility, with mean content of total N and organic matter of 1.46 and 26.1 g kg^{-1}, respectively, and an initial mineral N content of 23 mg kg^{-1}. An unfertilised control was compared to eight fertilisation treatments, obtained by applying the same N rate (75 kg N ha^{-1}, corresponding to 0.661 g pot^{-1}) in form of: traditional (NH4NO3 – treatment 1; (NH4)2SO4 and NH4NO3 – treatment 2), organic-mineral (product containing peat and leather - treatment 3), organic (product containing hoofs and horns - treatment 4), slow and controlled release (one condensed, two coated and one stabilised – treatments 5-8), fertilisers. In particular, the condensed fertiliser contained ureic-N (8.34%) and ureaformaldehyde (19.66%); the two coated products, controlled N release through a multilayer and a polymer-sulphur coating, respectively; the stabilised fertiliser contained N in ammonium and nitric form and an inhibitor of the nitrification process (3.4 DMPP).

A randomised block design with four replicates was used. The wheat was sown on December 2004 and harvested on June 2005. The traditional fertilisers were applied: totally at side-dressing as NH4NO3 (tr. 1); split in two rates, 20% before sowing as (NH4)2SO4 and the remainder at side-dressing as NH4NO3 (tr. 2). All the other fertilisers were incorporated into the soil before sowing. During the crop cycle soil mineral nitrogen content was monitored. Moreover, SPAD values were measured every 15 days after the tillering stage. In order to evaluate the amount of N effectively available for the crop by applying different fertilisers, N use efficiency was determined. Therefore, at harvesting, dry matter of grain, culms and leaves was measured and, on two replicates, N content of the single plant parts was determined. Nitrogen use, uptake and utilisation efficiencies were calculated according to Olsen (1993) and Janssen (1998). Statistical analysis of experimental data was carried out.

Results

Soil mineral nitrogen content decreased considerably during the crop cycle from average values of 8 mg kg^{-1} at the end of tillering to values of 1 mg kg^{-1} at mealy ripening, showing the rooting system ability of the crops of all compared treatments to absorb the N available in the soil. The values were highly variable in the first sampling and constant in the last; however, no significant differences were detected among treatments.
After the tillering stage differences in growth became evident among treatments and the highest SPAD values were observed in the crops fertilised at side-dressing (tr. 2 and 1) and in the treatments 7 and 8.
The best yield results, significantly different from the control, were obtained in treatment 2. The total rate distributed at side-dressing (tr. 1), the two coated fertilisers (tr. 6 and 7) and the stabilised one (tr. 8) showed intermediate behaviour, whereas the poorest performance was obtained in the other treatments, which however were significantly different from the control. The lowest percentage of non-vitreous kernels was observed in tr. 1 soon followed by tr. 2.
Grain protein content, and therefore technological quality, was not significantly affected by the treatments, but the highest values were obtained in treatments 2, 1, 7 and 8 which showed an increase of 23.8, 21.6, 19,9 and 16,3% compared to the unfertilised treatment; the grain protein content resulted also highly positively correlated to the percentage of vitreous kernels (R^2=0.81). N content of culms and leaves showed the same behaviour of grain protein content but the differences were statistically significant.
The different fertilisation treatments determined differences in total N uptake, which reached the highest values in tr. 2 and the lowest in the organic fertilisation and in the unfertilised treatment, as shown in Table 1.
Finally, no significant differences were found in relation to N use and N utilisation efficiency, while N uptake efficiency was affected by the kind of fertiliser applied. In particular, the highest fertiliser recoveries were obtained in tr. 2 and the values ranged from 73.5% (tr. 2) to 27.5% (tr. 4); the highest value of N use efficiency was found in tr. 2, and was equal to 29.05 g g^{-1} (tab. 1).

Discussion and Conclusions

The crop was very efficient in absorbing all the N present in the soil (no statistical differences were detected among the fertilisation treatments), so the NUE and in particular the fertilisation recovery became the only way to estimate the N made available by the fertilisers during the crop cycle and to compare the treatments.
No significant differences were found in N use efficiency, which reflects the ability of the crop to use applied N for grain production, while significant differences were found in relation to the fertiliser recovery, which represents the efficiency of the whole plant in absorbing applied N; this shows that a lower amount of N was made available by some of the fertilisers for crop uptake. Grain protein content and therefore technological quality was not significantly affected by the treatments, though the highest values were obtained in treatments 2, 1, 7 and 8. Both these behaviours could indicate that, though the treatments 2, 1, 7 and 8 appeared in general more efficient in supplying N to the crop, the nutrient availability across the crop cycle was not optimal to achieve the maximum yield and grain recovery. Maybe an anticipated availability at tillering stage, to allow more tillers to be formed, and a delayed one later in the season, in order to allow more N to go in the kernels, could give the best results. Of course, such a scheduling of fertiliser application can't be realised with a slow or controlled release fertiliser, applied only once during the crop cycle and which release time is difficulty predictable, while could be possible by splitting soluble fertilisers.

Fertilisation with traditional product allowed to obtain the highest N recovery, while the other fertilisers, and in particular the organic-mineral and the organic left a higher residue in the soil, potential source of pollution.
Despite the initial high soil fertility, which almost certainly reduced the differences among the treatments, the highest yield and N uptake efficiency were obtained by splitting the fertiliser in two rates (tr. 2), but above all by supplying the plants with a source of readily available N in the maximum uptake period (tr. 2 and 1). Good results were also shown by the two coated and the stabilised fertilisers, but further investigations in poorer soils are needed.

Table. 1. Grain yield, total aboveground dry matter, total N uptake and N use and uptake efficiencies measured and calculated for the fertilisation treatments compared.

Treatment	Fertiliser category	Grain yield (g pot^{-1})	Total aboveground dry matter (g pot^{-1})	Total aboveground N uptake (g N pot^{-1})	N use efficiency (g g^{-1})	N uptake efficiency (%)
1	Traditional	49.42 a	95.22 ab	0.852 ab	23.82 a	58.5 ab
2	Traditional	52.35 a	99.42 a	0.949 a	29.05 a	73.5 a
3	Organic-mineral	46.5 a	89.13 b	0.681 b	19.06 a	33.0 b
4	Organic	43.8 a	82.98 b	0.644 b	14.06 a	27.5 b
5	Condensed	43.5 a	94.20 b	0.719 b	20.43 a	38.5 ab
6	Coated	49.05 a	95.37 ab	0.740 ab	24.06 a	42.0 ab
7	Coated	47.0 a	92.40 ab	0.842 ab	17.25 a	57.0 ab
8	Stabilised	46.8 a	93.81 ab	0.778 ab	20.87 a	47.5 ab
9	Unfertilised control	34.05 b	70.32 c	0.464 c		
X		45.83	90.32	0.741		
		**	*	**	n.s.	*

n.s. = not significant
** = significant at P<0.05*
*** = significant at P<0.01*
different letters indicate significant differences at P<0.05 according to SNK test

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Strategies for environmentally responsible N management using state-of-the-art crop sensing tools

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Abstract

Diagnosing the N status of crops during the growing season with the help of sensing tools may enable managers to minimize N losses without incurring yield loss by adapting fertilizer doses. Three instruments that are available to perform N status diagnosis on corn were compared as well as N sources to establish well-fertilized reference plots. These instruments were the chlorophyll meter, the Greenseeker and the Dualex. Dualex readings reacted faster to N conditions than chlorophyll meter and revealed potentially toxic effects of N applications. The Greenseeker showed a lower discrimination potential than the two other technologies. There were differences in the behaviour in time of saturation index depending on N sources. This suggests that selection source for reference plot establishment should take into account the growth stage at which the diagnosis is targeted.

Keywords: chlorophyll meter, Dualex, Greenseeker, reference plots, saturation index, SPAD

Background and objectives

Relying on averages for N fertilizer application is unsatisfactory because of the unpredictable nature of weather conditions which have in turn a major impact on crop uptake, N mineralization, and N losses. Sensing tools can help in diagnosing the N status of crops so that managers can react accordingly. A sound strategy for the N fertilization of the corn crop should focus on: 1) lowering N in the starter; 2) establish well-fertilized reference plots; 3) perform a SI calculation on crop N status at topdressing. Sensing a crop for nitrogen (N) status is possible with a variety of instruments (Tremblay 2004. Data can be gathered either on a convoluted biomass and chlorophyll status (Greenseeker), chlorophyll (chlorophyll meter), or polyphenolics levels (Dualex; Goulas *et al.*, 2004), which are related to N conditions. It is also pertinent to install in each field a well-fertilized reference plot in order to alleviate the influence of factors unrelated to N requirements (Tremblay 2004). Sensor measurements from the rest of the field are then compared to the reference plot in the form of a saturation index (SI). SI is seen as an indicator of potential response to fertilizer N. However, the optimal source of N and mode of application for these plots have not been fully established. The objectives of this paper are: 1) to evaluate the potential of 3 crop sensing technologies in assessing the N status of a corn crop; 2) to compare N sources and application methods for their value in maintaining the highest N status possible in reference plots.

Material and methods

The experimental corn cultivar Pioneer 38A24 was sown on May 18, 2005 (day of year 138, or D138) in a completely randomized block design with 14 N treatments (Table 1) and 4 blocks. Chlorophyllmeter readings were taken 10 times at 3 to 9 days intervals from D158 to D194. GreenSeeker readings were obtained 5 times between D164 to D181. Dualex readings were taken on D189 and D194 and results were presented as averages of the abaxial and adaxial faces of leaves. The Dualex was unavailable before D189. Chlorophyllmeter and Greenseeker readings were also expressed relative to fully-fertilized reference plots, which were installed at sowing time by adding 180 kg N/ha to a 45 kg N/ha CAN (calcium ammonium nitrate) application. For treatment 02 the supplement was added as CAN in a broadcast fashion (CANB). For treatments 11, 12, 13 and 14, the supplement was banded along the rows in the form of CAN, N32, CPU (coated polymer urea), or urea, respectively. Saturation index (SI) was calculated by dividing the reading from each plot by the reading from the reference plot of the corresponding block

and expressing the result as a percentage. Nitrogen applications for treatments 3 to 6 were similar until topdressing (D185) (Table 1). Sensor measurements from these four treatments were therefore averaged per block, representing the 45 kg N/ha starter dressing. Similarly, sensor measurement from treatments 7 to 10 were also averaged by block to produce a mean for plants grown with 0 kg N/ha in the starter dressing. On D185, topdressing of N was performed. Treatments 3 to 6 received 45, 90, 135 or 180 kg/ha of N as N28 (UAN solution), respectively. Treatments 7 to 10 were treated the same way. In Quebec, growers would have normally performed tropdressing between D170 and 175.

Table 1. Description of nitrogen fertilization treatments applied to the corn crop and results of chlorophyll meter and Dualex measurements performed on D189 and D194.

Treatment number	kg N/ha at sowing	kg N/ha at topdressing	D189		D194	
			Chlmeter	Dualex	Chlmeter	Dualex
1	0	0	37.6c	0.913b	36.8c	0.855ab
2	45+180 CAN broadcast (CANB)	0	49.1a	0.726ef	52.1a	0.681efg
3	45	45	44.0b	0.810cd	44.0b	0.745d
4	45	90	44.0b	0.801cd	43.9b	0.736def
5	45	135	43.2b	0.830c	43.8b	0.788cd
6	45	180	43.6b	0.812cd	43.5b	0.740ef
7	0	45	36.2c	0.959ab	34.2cd	0.878ab
8	0	90	37.1c	0.931ab	35.6cd	0.867ab
9	0	135	35.9c	0.945ab	35.5cd	0.838bc
10	0	180	35.1c	0.978a	33.5d	0.910a
11	45+180 CAN row (CANR)	0	48.9a	0.710f	53.7a	0.648g
12	45+180 N32	0	49.6a	0.769de	52.4a	0.637g
13	45+180 CPU	0	48.2a	0.716ef	52.8a	0.673g
14	45+180 Urea	0	48.4a	0.743ef	52.3a	0.677fg

Same letters within columns refer to numbers not significantly different (LSD, $P \leq 0.05$).

Results and discussion

Chlorophyllmeter raw readings (Figure 1a) and SI (Figure 1c) were significantly different with 0 vs 45 kg N/ha starter dose at D158 and from D178 to 185. For D164 to 171, differences could be detected but only at the $P \leq 0.10$ level. There was clearly an effect of starter N on the diagnostic parameter trends in time. For the 0 kg N/ha at sowing treatment, the trend in chlorophyll meter reading from D158 to 185 was quadratic but it was curvilinear for the 45 kg N/ha at sowing treatment. CANBSI, CANRSI, N32SI decreases in time (Figure 1c) were curvilinear while CPUSI and UreaSI were only linear, regardless of starter N. Only CPUSI was affected also by a cubic component at starter = 45 kg N/ha. The initial N fertility conditions (0 vs 45 kg N/ha) could be distinguished at all stages at the $P \leq 0.15$ level. Chlorophyll based diagnosis of N condition of corn crops did not increase in sensitivity with time after sowing (data not shown). Hence, even if the 45 kg N/ha starter dose lead to lower SI than the 0 kg N/ha dose, there was no evidence that the starter dose delayed the growth stage at which a reliable diagnosis could be made.

Figure 1. *Results of three diagnosis methods for evaluation the N status of a corn crop as a function of day of year. a) chlorophyll meter; b) Greenseeker NDVI; c) chlorophyll meter results rationed to five types of well-fertilized N reference plots (saturation indexes, or SI).*

The Greenseeker NDVI (Figure 1b) and SI (not shown) were able to discriminate the 0 from the 45 kg N/ha starter dose at D167, 178 and 181. An exception was CANRSI which was only successful on D178. On D167, N32SI and CPUSI also failed to meet the $P \leq 0.05$ criteria. With starter dose 0 kg N/ha, only N32SI and UreaSI exhibited a trend (curvilinear) in time. With starter dose 45 kg N/ha, CPUSI presented a linear component while UreaSI presented a quadratic component. Starter dose of N did not significantly change the trends in time of any of the Greenseeker based SI. Trends in time of Greenseeker NDVI (Figure 1b) were significant in the linear, quadratic and cubic components for both starter dose 0 or 45 kg N/ha.

Both chlorophyll meter and Dualex revealed globally highly significant effects of treatments ($P \leq 0.001$) on D189 and D194. Dualex vas inversely related to chlorophyll meter readings (Table 1). On D189, as early as 4 days after topdressing, the Dualex suggested a toxic effect of the 180 kg N application on plots having received only 0 kg N/ha since then (treatment 10 vs 1). These high Dualex readings on treatment 10 were maintained on D194.

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Working group 7

Soil-related indicators: ex ante hints or ex-post evaluation?

Report of Working Group 7

Soil related indicators: ex ante hints or ex post evaluation?

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In this working group we discussed two aspects of soil N processes: soil N mineralization which on the one hand makes more N available for plant production, but also may result in abundant N that the crop is unable to use and which is thus a potential danger for nitrate leaching (Figures 1 and 2).

Figure 1. N processes in the soil, potential indicators for mineralization and nitrate leaching.

Figure 2. Importance of knowing the level of N mineralization.

Significance of knowing N mineralization

In the EU Water Framework Directive intends to ensure the quality of ground and surface waters. N losses from soils used for agricultural production should be reduced and farm management must be changed to meet the eventual targets. N-related soil indicators are of great importance for monitoring the impact of adaptations in management on water quality. Indicators of N mineralization in the soil may help farmers to adjust their N fertilization strategies (Figure 2). Indicators that are related to N losses may be used as a basis and/or justification for action programs developed by policy makers.
We further discussed a number of questions which arise in the context of the Water Framework Directive:

- Can indicators give us an answer or help us to provide relevant additional information?
- What makes an indicator adequate?
- What is the impact on soil use and nitrogen management of farms?
- Which criteria are important to judge indicator methods of nitrate leaching (cheap, quick, easy, reliable, …)?

Indicators for mineralization

The session started with two presentations. In the first presentation Valé described research on arable soils in France to measure and explain *in situ* N mineralization. Variability was only reasonably predicted by means of statistical models using a range of soil and crop characteristics (Valé *et al.*, 2005). O'Connell presented results of soil organic matter N uptake by permanent grassland on Irish dairy farms. Soil organic matter content is a major source for net N mineralization, which is influenced by weather and soil conditions (O'Connell *et al.*, 2005).

In the discussion attention was given to the questions:

- How relevant and reliable are estimations of the mineralization?
  - for fine-tuning N fertilization for the crop;
  - to prevent excessive residual N levels at the end of the season.
- Which criteria are important to judge indicator methods of mineralization (cheap, quick, easy, reliable, …)?
- At what scale should they be applied (field, farm, region)?

The group concluded that the most relevant indicators mentioned were:

- Local soil characteristics (texture, organic C, soil depth, drainage, cropping history, tillage).
- Climate (rainfall, temperature).
- Reliable tests (*in vitro, in situ*).

Indicators for nitrate leaching

In his presentation Haberle showed that there was a positive relationship between autumn Nmin and the decrease of Nmin during winter. His results support the use of Nmin in late autumn as an indicator for the risk of N losses in upland arable regions of the Czech Republic (Haberle *et al.*, 2005). In the Netherlands Smit *et al.* (2006) developed regression models to predict nitrate concentrations at levels of the field, farm and region using different indicators, the most important ones were: soil type, groundwater regime and crop. The percentage of variance for the nitrate concentration explained by the model is poor. The models can predict nitrate concentration at regional scale and farm scale better than at crop scale.

The discussion focused on:

- Can soil indicators help us to prevent excessive residual N concentrations at the end of the season?
- At what scale should they be applied (field, farm, region)?

Potential indicators for the risk of nitrate leaching are:

- Mineral N at the end of the season.
- Previous crop and management (manuring, fertilizers, grazing, catch crop, etc.).
- 'Permanent factors' (drainage, texture, organic C, etc.).
- Recent weather conditions (previous season).

Although these are fairly simple conclusions we were pleased that the group was able to rapidly reach consensus on the importance of mineral N at the end of the season, previous crop, permanent soil factors and weather conditions as indicators for nitrate leaching.

Further research

There are many questions still open. For instance what are the differences between potential and actual nitrate leaching, and under what conditions are these important. Denitrification, the use of catch crops, immobilization of N, the degree of drainage, mineralization during mild days in winter etc. are some of the processes which have an impact at the actual annual leaching. It is a high priority to test these factors in future research. We suggest working together in a coordinated programme across Europe to evaluate effects of these processes in the different climates of Europe in a monitoring programme. We know that the variability of mineralization and nitrate leaching is high due to permanent factors (particularly soil type and potential drainage), previous crops, field management and weather conditions. We expect to be able to explain a greater proportion of the variance when we study these processes over a greater range of these factors which is possible when we include soils, weather conditions, crops and management throughout Europe.

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Poster presentations

Nitrate concentration in upper groundwater shows large temporal variations

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Abstract

We measured detailed profiles of the nitrate concentration immediately below the groundwater table. The measurements were performed at a fixed depth below the groundwater table, and therefore at a variable depth below the soil surface. The nitrate concentration showed a temporal variation from 10 to170 mg NO_3 L^{-1}, and that the preferred sampling time is in the autumn. Moreover, we were able to show that the loading of groundwater with nitrate is irregular in time.

Keywords: groundwater, leaching, nitrate, temporal variation

Background and objectives

According to the Nitrate Directive of the European Union the concentration of nitrate in groundwater should not exceed the drinking water limit of 50 mg NO_3 L^{-1}. However, in the directive no specification has been made about the method of monitoring the nitrate concentration. In the Netherlands the upper meter of the groundwater is used as a reference, but the time of sampling has not been specified. Remarkably, sofar few research has been directed to the temporal variation of the nitrate concentration in the upper groundwater. In this paper we present observed (Van Beek, 2004) and simulated data on this topic (Roelsma, 2002).

Materials and Methods

In order to obtain information about the temporal variation of the nitrate concentration in the upper groundwater, a well, screened over its whole length with horizontal slots, was installed to a depth of 1 m below the groundwater level. In this way the groundwater level always intersects the well screen. In the well a multilayer sampler (MLS) was placed consisting of 50 dialysis cups over a length of 1,5 m. The dialysis cups were individually separated by rubber sleeves. Krajenbrink *et al.* (1989) describe this method in greater detail. At the start of each measurement the center of the MLS was positioned at the groundwater table. In this way the groundwater level always intersected the MLS. After equilibration for 4 weeks, the dialysis cups were collected and the depth of the groundwater table was recorded. Only the dialysis cups below the groundwater table were chemically analyzed. The experiments were carried out in old agricultural soils (so called 'enk' soils), consisting to a great depth of fine sand. The field was used as grassland by a dairy farm. In order not to disturb normal agricultural practice, and to be sure that the results were indeed representative, the wells have been cut of at a depth op about 20 cm below surface. The top of the well was covered by a lid and a pavement tile, which was buried below topsoil and grass-sods.
For a similar situation the nitrate concentrations have been simulated with the help of the ANIMO model. The ANIMO model is a one-dimensional process-oriented model which combines fertilization level, soil management and leaching

of nutrients for a wide range of soil types and different hydrological conditions (Groenendijk and Kroes, 1999, Kroes and Roelsma, 1998).

Results and discussion

Figure 1 shows the concentration of nitrate in course of time. The depth of the groundwater table was measured every 4 weeks, and the chemical composition every 8 weeks. The results of January 2003 are missing, as the soil was frozen. In August and in October 2003 only the top 27 cm of the chemical composition could be measured as the groundwater table dropped to a very low level.
From figure 1 it is clear that the concentration of nitrate varies between 10 to 170 mg NO_3 L^{-1}. The greatest changes in concentration coincide with the greatest changes in depth of the groundwater table. The dichotomy in nitrate concentration in February 2004 is striking: about 15 mg L^{-1} in the upper 18 cm as opposed to about 90 mg L^{-1} directly below that layer. This behavior can only be explained by the complete leaching of nitrate by the first winter rains. As nitrate has been about completely leached, further rain cannot leach nitrate anymore. Such a zone of low nitrate concentration will be absent in dry winters and be extended in wet winters.

Figure 1. Observed concentration of nitrate in various layers of groundwater and depth of groundwater table.

This course of nitrate concentration has been confirmed by model simulations of a similar situation. In Figure 2 the results of the simulated concentration of nitrate as a function of depth and time are presented. Figure 2 also shows that large temporal variations in nitrate concentration occur in the upper groundwater.

Conclusions

The concentration of nitrate in the uppermost groundwater shows large variations over time, from below to far above the limit value of 50 mg/l NO_3. The largest changes in concentration of nitrate coincide with the largest changes in depth of the groundwater table. The depth of the groundwater table changes least during summer and autumn. The preferred moment of sampling should be in autumn after harvest before the winter rains.
An additional conclusion in this research is the complete leaching of nitrate in the fine sandy soils of the experimental area by the first winter rains.

Figure 2. Simulated concentration of nitrate, in mg $NO_3 L^{-1}$ down from the soil surface, in mbs, and the depth of the groundwater table (line).

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Nitrogen characterization and distribution in northern Tunisian soils

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Abstract

Studies of soil organic matter (OM) characterization and its nitrogen (N) distribution, in the arid and semi-arid regions are rare and topical. In our research we have focused on OM distribution in selected soils of northern Tunisia as well as N distribution during humification. The samples studied are selected following a broad E-W direction. Soil fractions < 50µm were analysed for total organic nitrogen (TN), using an Ultrasonic Generator. Humic and fulvic acid (HA and FA) quantification were determinated following an alkaline extraction method. Our OM N results show that more than 70% of TN in the soil was located in the fine fraction (FF) of the soil, especially within the deep horizons of all the soil types. The ratio between TN in FF and TN in other soil fractions was highest in the fine textured soil. It was lowest in the surface horizons of the humin-rich rendzina and the leached brown fersiallitic soil. The humin (H) fractionation analyses indicate that 60% of FF N content is mainly located within the H.

Key words: nitrogen, organic matter, soil

Background and objectives

N is one of the most important elements of soil organic matter (OM) which in turn is an essential component of living matter. The study of such element offers a double interest displayed on the economic and environmental fields. From an economic point of view, the study of N distribution between the various compartments of a specific soil, allows a better quantification of the required amounts for an optimum fertilization. In addition, this assessment prevents possible N losses through leaching and therefore possible groundwater and hydro-graphic network contaminations. This study aims at a better characterization and identification of the N distribution in Tunisian soils under different climatical conditions.

Materials and methods

Figure 1. Sites localization according to bioclimatic stages.

The study focuses on five soil types, belonging to different ecological areas of Tunisia, which can be summarized as follows: a brown leached soil (P1) and a brown leached fersiallitic soil (P2) which are sampled on two naturally well-exposed soils located in the Aïn Draham and the Tabarka rainy areas, respectively (Figure 1). They are covered by a forest vegetation and are characterized by a highly humid bioclimate with a temperate winter. The third soil (P3) is an encrusted fersiallitic one, whereas the fourth (P4) is a vertisoil. Both soils are located in the Béja area (Figure 1), ranging from a sub-humid mediterranean bioclimate with a mild winter to a middle semi-arid bioclimate. The fifth soil (P5) is a humin-rich rendzina located at the Sakiet Sidi Youssef area in the uppermost-part of north-western Tunisia (Figure 1). The climate of this zone varies from semi-arid to highly humid with generally a mild winter (Figure 1).
The procedure of granulometric fractionation is as follows: 50 g of 2 mm sized ground soil are added to 250 ml of water, later submitted to 5 minutes ultrasound agitation to ensure its homogenisation and dispersion (Balesdent *et al.*, 1991). The suspension is then filtered with 50µm under a distilled water jet. The fractions below 50 µm are recovered in distilled water. Each fraction is later dried in 50°C oven to reach a constant weight. The different fractions are finely crushed to be analysed for total organic N (TN). Part of the fulvic acid (FA) and humic acid (HA) is extracted from the fine fraction of the soil (FF) using $Na_4P_2O_7$ 0.1 *M* (pH of 9.8). The remaining residue is mixed with NaOH 0.1 *M* (pH of 12) in order to recuperate all the acids. The totality of the FA and the HA are finally separated by acidification under a pH of 1.5. This procedure of alkaline extraction is described by Dabin (1976) and Ben Aïssa (1993).

Results and discussions

The obtained results show variable N ratios $N_{<50\mu m}/N_T$ in the FF. The ratios are high especially within the deep horizons of almost soil types, probably related to the proportion of the FF. On the other hand, the lowest ratios occur in the surface horizons of the humin-rich rendzina and the brown leached fersiallitic soils, notwithstanding that these soils contain high percentages of total OM. The proportion of N in the fraction < 50 µm of these two soils was 36% and 40%, respectively. This can be explained by the fact that this fresh OM probably mainly originates from leaves of coniferous trees which are resistant to the biological breakdown. The high ratios $N_{<50\mu m}/N_T$ in deep horizons, the proportion of $N_{<50\mu m}$ exceeds 60%, can be attributed to the fact that the OM in the deeper soil layers is entirely of a humific nature. Moreover, the highest values are recorded in the finest textured soils of the deep horizons, such as the vertisoil and the encrusted fersiallitic soil. These observations allowed us to deduct for all soils studied a correlation between the N contents of the FF with either the clay contents (A) or the finest silt contents (FS). We found a positive correlation between the highest A and FS contents and N contents which as has been previously advanced, indicates that such soil textures enhance N incorporation into OM, which in turn is mainly associated to the finest fractions of any soil (Elliott, 1986; Jocteur Monrosier *et al.*, 1991).
Analyses of N in the different separated humic compounds show that most of the N is located in the H fraction, because this fraction constitutes the most significant component of the FF. In addition, this enrichment can be the result of a preferential accumulation of microbial proteinic materials in the FF on their high specific surface (Barriuso Benito, 1985). The analyses also show that soil enrichment in H takes place from the profile top to the bottom. In fact, in the deepest and enriched horizons, the N tends to be located mainly in H, where it constitutes up to 90% of TN of the FF. In the impoverished horizons of the brown leached soil and the brown leached fersiallitic soil, only 30% of TN is present in the FF. It should be noted that the N extraction rate, of the two latter soils, is low and that it further decreases with increasing depth. Indeed, the proportion of TN in both HA and FA as compared to TN in the FF decreased with depth from 47% to 22% for the brown leached soil and from 56% to 31% for the brown leached fersiallitic soil. The total organic carbon concentration in the HA and the FA remained very high with increasing depth. We can deduce then that the HA and FA compounds have little evolved, present a low degree of polymerisation, and are impoverished in aromatic structures. Regarding, the encrusted fersiallitic soil, the ratios C_H/N_H are low from the profile top to the bottom except for the encrusted horizon where it reaches the value of 20. This enables us to suppose that this horizon, contrary to the others, contains inherited and not very advanced humins. The study of the humic compounds evolution shows that the obtained extracts are richer in N than in C. This is probably related to a preponderance of HA compared to FA contrary to the case of soils of medium acidity. The vertisoil shows a relatively homogeneous OM content throughout the profile. In addition, the $N_H/N_{<50\mu m}$ ratios are almost all above 0.9. In the case of the humin-rich rendzina, the ratios C_H/N_H are relatively low, but they reach a maximum at the surface horizon. The $N_{AS}/N_{<50\mu m}$ ratios decrease from top to bottom, and by consequence, the

C_{AS}/N_{AS} ratios present a mirror image reaching a maximum of 16.7 at the deepest horizon C_{ca}. Again, this indicates the small degree of evolution of the HA and FA, probably blocked by a $CaCO_3$ crystalline coating. However, the total de-calcinisation of the soils limits the action of Ca^{2+} on the humic compounds (Schnitzer, 1991). The latter compounds, mainly the condensed FA and HA, will interact with the other mineral components via cations such as Fe, Al or Mg which will take part in their stabilization and their resistance to the alkaline reagents. These newly formed organomineral complexes are likely to migrate in-depth. Thus, the obtained values for the C_{AS}/N_{AS} of carbonate soils, testify the presence of rather advanced extractable compounds with low content of aliphatic chain, as compared to those obtained on the level of the non-carbonate soils.

Conclusions

Based on our results, we can conclude that the northern Tunisian soils are well characterized by a polymerised OM, very rich in N. This N is incorporated into the soil FF, and mainly within the H. Therefore, the N is in a stable organic form, unfortunately unusable by plants as it cannot be mineralised easily. Consequently, manure application to such soils is crucial. However, such solution may lead to contamination of surface and subsoil waters. To help to minimize energy cost and especially the risk of contamination without any remarkable fall of the output, it is necessary to establish an assessment of the required N on Tunisian soils. An accurate quantification of the available N must be started, taking into account the mineralization kinetics of the existing organic N of the Tunisian soils.

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Regional analysis of potential N uptake of catch crops at different sowing terms in the Czech Republic

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Abstract

Delayed sowing of catch crops results in the poor establishment of crop stand and slow growth that diminishes their positive effects. Aboveground biomass of winter oilseed rape (*Brassica napus)* and sunflower (*Helianthus annuus)* was simulated with daily meteorological data (1961-2000) from twelve sites in the Czech Republic. The probability that the biomass reaches a specific limit was calculated for successive sowing or emergence terms, from August to October. The data were upscaled to the regional level using the regression of average climatic characteristics on the probability of reaching the specific biomass.

Keywords: biomass, catch crop, climate, GIS, modelling, WOFOST

Background and objectives

The Czech Code of Good Agricultural Practices includes catch crops as an effective measure to reduce the nitrate content of soils in autumn and to lower the risk of erosion. To increase the area of catch crops, a subsidy (150 euro per ha) is granted to farmers who register in the system and will grow them regularly (total 200,000 ha in 2004). Twenty crops, both over-wintering and frost-susceptible species are entitled to this subsidy. Catch crops must be sown before 10th October and must be left at the field to at least the 15th February. Farmers are inclined to postpone sowing until the end of September. The aim is to have less crop residues since they complicate soil tillage and the emergence of spring crops. Of course, the date of sowing varies also due to the harvest date of pre-crops and due to actual weather conditions.

The depletion of excess nitrate by catch crops depends on adequate growth that generates demand for nitrogen. Under the Czech transition climate conditions, the establishment of crop stand and growth of catch crops are limited in some years by a short growth period (low sum of temperature, early start of winter) and/or due to water shortage. Especially, in the upland potato growing region, with a shorter growing period, late sowing increases the risk that the catch crop will not deplete a reasonable amount of residual nitrogen before winter.

Through a modeling study we intended to estimate the risk that a catch crop does not reach sufficient biomass under delayed sowing terms.

Materials and methods

Daily meteorological data from twelve locations collected from 1961-2000, together with digital maps on climate and altitude in the Czech Republic were used. Selected locations represent diverse conditions ranging from warm lowlands to cold highland regions with higher precipitation. The following climate characteristics of the sites were used: average yearly and monthly temperatures, the sums of effective temperatures (above 0° and 5° C), the start, end and the length of period with average temperature above 0° C and 5° C, the sums of yearly and monthly precipitation.

Water-limited and potential growth of winter rape and sunflower were simulated with the model WOFOST (Diepen *et al.*, 1989). The simulated growth is not limited by other biotic or abiotic factors besides weather. The simulation of numerous combinations of years, sites, sowing terms and other plant or soil input parameters can be easily performed within the system PERUN, combining the WOFOST model and a weather generator (Dubrovský *et al.*, 2004).

Winter rape and sunflower were selected as models for overwintering and frost susceptible crops, respectively. Winter rape is not listed among the catch crops granted subsidy, but there are other Brassica species (spring rape, yellow mustard, fodder radish). We assume that winter rape is a convenient model crop due to its vigorous growth in autumn that may represent the upper limit of production for other overwintering crops. Simulations of total above-ground biomass (TAGP) using successive dates of sowing or emergence from August (sunflower) or from September (rape) to mid October were performed. The probability that the TAGP of the crop reaches specific limit, viz. 1.5, 2.0 or 2.5 t of dry matter per hectare per year was calculated for successive emergence terms, using the available weather data base of forty years. We estimate that the production of about 2 t TAGP per ha ensures a reasonable potential demand for 40 kg of N ha^{-1} and more. The data were upscaled to the regional level using a regression of temperature and other climatic indicators on the biomass produced or the probability of reaching a specific biomass.

Results and discussion

There was a great variability of the simulated TAGP across the separate forty years. For example, the TAGP of winter rape with emergence on day 274 (1st October) ranged from 0.5 to 2.5 t ha^{-1} in the warm region, and from 0.2 to 1.8 t ha^{-1} in the upland region. Soil types and initial conditions, using realistic ranges, had little effect on the outputs of simulation, as the growth was predominantly limited by temperature and radiation. The model was not calibrated, input parameters supplied with the model were used. The comparison of simulated TAGP's with yields of several catch crops in three years and 6 sites (Procházka, Vach-unpublished) showed that the maximum biomass observed was about 50-75% of the simulated one. There was a small difference between potential and water-limited biomass, except when a coarse soil and very low initial available water content was used as input. The results are in some contradiction with farm experience where emergence is often delayed and/or irregular due to the desiccation of superficial soil layers.

There was a good relation between average climatic characteristics of the locations and TAGP or the probability of reaching a specific biomass (Table 1). The best correlations were found for average temperature, sum of temperature (Figure 1) and the end of period with average temperature above 0° C and 5° C. There was a strong relationship between the climatic characteristics and altitude, but the relationship of altitude and the simulated probabilities was weaker than for temperature. There was a poor correlation between precipitation and simulation outputs.

Table 1. The correlation coefficients from linear regression of climatic characteristics on the probability that TAGP of rape (emergence on day 265) and sunflower (emergence on day 235) reaches 2.5 or 2.0 t ha^{-1}, respectively.

	Average temperature		Sum of effective temperatures			End of average temperature		Altitude
	Year	Aug-Sept	Year	Aug-Nov	Sept-Dec	> 0° C	> 5° C	
Winter rape	0.91	0.91	0.91	0.92	0.92	0.91	0.88	- 0.82
Sunflower	0.88	0.87	0.88	0.88	0.89	0.86	0.87	- 0.84

Maps showing the risk of not reaching a specific biomass of winter rape and sunflower at different dates of emergence were produced. The simulations suggest that the emergence of rape and sunflower after mid September and mid August, respectively, rapidly decreases the probability of reaching 2 tons of dry mass per hectare, except for the warmest regions of the Czech Republic.

Figure 1. The relationship between probability of reaching 2.5 t ha^{-1} of biomass of winter rape (emergence on day 265, 22nd September) and climatic characteristics of twelve sites.

Conclusions

The maps may be used for the identification of agricultural regions where sowing (emergence) of catch crops after a specific date is not effective, i.e. where the growth is not sufficient in most years. Further steps are needed to improve reliability of the findings. Model outputs will be validated with experimental and farm data from sites with various soil-climate conditions. Climatic data sets from other stations (total 45) will be used for the simulation and regression analysis. Input parameters for other catch crops will be compiled.

Map 1. The simulated probability of reaching 2.5 t ha^{-1} of TAGP of winter rape with emergence on day 265, 22nd September.

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Apparent losses of soil mineral nitrogen between autumn and spring on farms in the Czech Republic

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Abstract

Apparent loss of N_{min} from the 0-60 cm soil zone during four winters was calculated as the difference between N_{min} contents in late autumn and early spring. In most sites and years a strong or moderate positive relationship between N_{min} in autumn and the apparent loss was observed. Simulated losses of N_{min} agreed well with observations on farms. The results support the use of N_{min} in autumn as a reliable indicator of the risk of N leaching from the topsoil and the shallow subsoil in upland regions of the Czech Republic.

Keywords: leaching, modelling, Nitrate Directive, nitrogen, precipitation

Background and objectives

In 2003 the Nitrates Directive (91/676/EEC) was adopted into Czech legislation. Forty-three % of agricultural land was designated as vulnerable zones. The Czech Code of Good Agricultural Practices (Dostál *et al.*, 2004) and Action Programme do not specify any limit of the nitrate content in the soil. Restrictions on the use of fertilizer, manure and slurry and on crop rotations, aim at decreasing high nitrate levels in autumn and at reducing the risk of leaching. To get data from farms, the content of mineral nitrogen (N_{min}) in the top- and subsoil was determined at several sites, mostly in upland regions, in the autumn and spring from 2001 onwards. The observed decrease of N_{min} during winter was directly related to N_{min} in autumn in three seasons (Svoboda *et al.*, 2004). The N_{min} method is a simple tool for the evaluation of the nitrate leaching risk before winter but it has some shortcoming arising from the intricate turnover of nitrogen. Mathematical models may contribute to interpreting ambiguous results (Johanovský *et al.*, 2002)

The objective of our study was to analyse the effect of N_{min} in autumn, soil cover, soil type and weather conditions on the apparent mineral nitrogen losses during winter and to compare simulated changes of N_{min} with field observations from four years.

Materials and methods

From 2001 to 2005, fields on eight farms under different soil-climate conditions were sampled for the content of mineral N (N_{min}=NO_3-N+NH_4-N). At each field soil cores were taken with gouge auger at 16 locations at cca 25 x 25 grid from the topsoil (0-30 cm) and the subsoil (30-60 cm). Care was taken to take samples at least 50 m from field borders. The locations of sampling points were recorded by GPS to sample the same locations in autumn and spring. Soils were sampled before the start of winter (end of November) and in early spring (March). The fields included various combinations of preceding crops, winter crops or fallow soil; in some fields manure or slurry was applied in autumn. Different fields were sampled in experimental years. The processing of samples followed standard procedures; samples were kept in a cooler during sampling and kept in refrigerator to be processed the next day. Soil was shaken in 2% K_2SO_4, 1:5 soil:solution ratio for 1 hour, NO_3 and NH_4 were determined by colorimetry.

The model CANDY (Franko *et al.*, 1995) was used to simulate N_{min} changes in a soil during winter. Common crop rotations, yields and other inputs values were used to overcome the lack of exact data from farms. N_{min} contents in the range of the observed values at autumn sampling term were used as input. Daily meteorological data from nearby stations were used. The indicator 'Category of infiltration capacity' (CIC) for sampled fields was compared

with the apparent loss data. The indicator (1-5) is calculated from the soil type, depth, texture, stone share, slope and climate region (Kvítek T., Lechner P., pers.comm).

Results and discussion

The amount of N_{min} in the 0-60 cm soil zone in autumn observed in farms moved in a wide range, up to 300 kg N ha^{-1}. Cumulative distributions of observed N_{min} contents in three farms (Figure 1) suggest that in three out of four years about 50% of the fields had N_{min} content higher than 100-120 kg N ha^{-1}, and 27-13% of fields had over 150 kg N ha^{-1}. The lower N_{min} content in 2002 probably resulted from leaching caused by extreme rainfalls in late summer. The selected farms and fields are representative for the regions. Therefore it may be deduced that similar range of autumn N_{min} contents would be found in other farms, as well.

Figure 1. The cumulative distributions of autumn N_{min} contents in the 0-60 cm layer in three farms.

Most of the sites and years showed a strong or moderate positive relationship between N_{min} in autumn and the apparent loss of N_{min} from the 0-60 cm zone during winter. Further, outlying data can be mostly explained by organic fertilization applied closely before sampling. We observed lower apparent losses of N_{min} in the season 2004/2005 due to the combination of a warm winter and a moderate precipitation. The nitrogen (mostly as nitrate) leached below 60 cm, may be depleted by the roots of many spring and winter crops. However, the nitrate in the deep subsoil (often a light or coarse soil) is prone to leaching from the root zone by spring rainfalls as the upper layers have reached field capacity by then.

The relationship between the CIC and apparent N losses was weak. The relationships between N_{min} in autumn and apparent N losses during winter derived from simulations were similar to the observed ones (Figure 2), but the fit was weaker in the season 2004/2005. The simulations showed that leaching (determined by precipitation) was the dominating factor; the preceding crop and soil cover. Fertilization had a weaker effect on apparent losses in most years. The simulated losses were sensitive to the date of spring sampling. It is in agreement with observed fluctuations of precipitation and topsoil temperature in early spring which determine mineralization of organic matter and the losses of N.

Figure 2. *The comparison of simulated (full circles) and observed (empty symbols) apparent losses of N_{min} from the 0-60 cm soil zone in three farms in the season 2001/2002. The simulated losses are calculated on March 4 (lefthand picture) and on March 23 (righthand picture).*

Conclusions

The results of the study support the use of N_{min} in autumn as a reliable indicator of the risk of N leaching from the topsoil and the shallow subsoil in upland regions of the Czech Republic.

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Accuracy of the nitrate sampling scheme, proposed in Walloon area, for grazed grasslands

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Abstract

In grazed agro-ecosystems, urine patches act as a point source for nitrate leaching with high local concentrations. These grasslands require attention and need to be sampled in the right way to obtain a correct picture of the situation. The aim of this work is to evaluate the precision of the sampling scheme applied in the Walloon area i.e. the sampling of 30 soil cores taken from the 0-30 cm profile and mixed before analysis.
After having demonstrated that such a scheme leads to normally distributed means, we assessed the accuracy of this methodology in 25 grasslands grazed by dairy cows under contrasting stocking rates. To do so 5 samples, each integrating 30 soil profiles, were sampled in each grassland.
The results underlined that one sample, composed of 30 profiles, allowed to reach an accuracy of 10 kg NO_3-N ha^{-1} in 60% of the sampled fields. Such an accuracy was reached in all but one fields having a mean NO_3-N content lower than 15 kg NO_3-N ha^{-1}. For the other fields (mean NO_3-N content = 23 kg ha^{-1}), 2 to 4, and in one case 10 mean samples, would have been necessary. Improvements of the sampling scheme are discussed.

Keywords : grazing, nitrate, Nitrate Directive, sampling scheme, Standard Deviation

Background and objectives

The application of the Nitrate Directive in the Walloon region of Belgium has led to the definition of a 'soil linkage' coefficient that is the ratio between the organic nitrogen produced by farm livestock and the quantity of organic nitrogen that could be spread in view of the available land area. If this ratio exceeds a value of 1, the farmer can adopt a 'quality approach' in which he accepts, among other measures, the sampling of soil profiles in 4-5 of his fields to quantify nitrate leaching risks.
In systems based on grassland valorisation, grazing management plays an important role in the environmental pressure. Indeed urine excretion in grazed grasslands acts as a point source for nitrate leaching, distributed in an heterogeneous way, with local concentrations as high as 1000 kg N ha^{-1}. So these grasslands require a better control and need to be sampled in the right way to obtain a correct picture of the situation. Today, the sampling scheme includes the collection of 30 soil samples from the 0-30 cm profile that are mixed before analysis.
The 0-30 cm profile is a good indicator of the 0-90 NO_3-N load, however the relation between both has to be updated each autumn [in 2001: NO_3-N [0-90] = 6.1+1.2* NO_3-N [0-30] (N = 22; R^2 = 0.810***)]. The objective of our work was to define the precision of such a sampling scheme.

Materials and methods

A first experimental was set up to define the distribution of the NO_3-N content in grazed grassland. With this purpose 47 to 50 samples of the 0-30 cm soil profile, were individually collected from two separate grazed grasslands and analysed for their NO_3-N content. Both grasslands received around 755 BLU (Big Livestock Unit) grazing days during the entire grazing season. Based on this distribution, we were able to determine the distribution of population

means, each mean including 30 samples. On the basis of the Standard Deviation (SD) associated with the population means distribution we could thus determine the level of confidence associated with the sampling scheme proposed by the quality control service.
This relation was applied on the data of a second experiment in which a series of 25 meadows, from 25 dairy farms, was selected on the basis of the number of BLU grazing days per year, ranging from 240 to 960, and of the number of BLU grazing days from the 1st of September to the end of October, ranging from 50 to more than 400 BLU grazing days ha^{-1}, in order to obtain a gradient of environmental pressure (Table 1).
In each of these fields, 5 samples, each composed of 30 0-30 cm soil cores, were collected from mid-October till mid-November, and analysed for their NO_3-N content. The SD and mean values were thereafter calculated for each of these 25 fields in order to determine the number of composite samples, each including 30 soil samples, that would have been necessary to reach an accuracy level of 10 kg NO_3-N ha^{-1}. This level of precision was chosen in view of the repeatability of the analysis chain.

Table 1. Number of the sampled grasslands as related to the number of BLU grazing days observed during all or during the end of the grazing season.

Nbr of grazing days during all the season	Nbr of grazing days since the 1st September 0-99	100-199	200-299	300-399	400 and more	Total
300-499	1	4				5
500-699	5	6	2		1	14
700-899		2	2	1		5
900 and more	1					1
Total	7	12	4	1	1	25

Results and discussion

In the 1st experiment, the mean (±SD) NO_3-N contents were 32.1±49.9 and 16.7±12.9 kg ha^{-1} in the two grasslands The distributions were log-normal ($p>0.10$, Shapiro-Wilk normality test after a logarithmic transformation of the NO_3-N content). As a log-normal distribution of means including more than 30 points has a normal distribution (Figure 1), we can now define the number of samples (N), each integrating 30 profiles, necessary to obtain a fixed level of confidence around this mean through the relation : $N = (SD/\text{level of confidence})^2$.
In the second data set, observed NO_3-N contents ranged from 4.8 to 31.8 kg ha^{-1}. As the SD was directly linked to the mean [SD=0.612*Mean–0.085 (R^2=0.502***; N=25)], the number of composed samples necessary to obtain a 10 kg NO_3-N ha^{-1} level of confidence, also increases with the mean.
Nevertheless, one sample, composed of 30 profiles, allowed us to reach this level of confidence in 60% of the sampled fields. Such an accuracy was reached in all, but one fields having a mean NO_3-N content lower than 15 kg ha^{-1}. For the other fields, having a mean NO_3-N content ranging between 19 and 32 kg ha^{-1}, 2 (24% of the sampled fields) to 4, and in one case 10 samples, each integrating 30 0-30 cm soil profiles, would have been necessary to reach such an accuracy of 10 kg NO_3-N ha^{-1}.

Figure 1. Distribution of the 50 means obtained through a re-sampling of the initial profiles. Each mean integrates a random selection of 30 values out of the 47/50 initial ones. The same value could be retained more than once.

Conclusions

The increase of the SD of observed NO3-N contents with their mean, as found in both experiments, confirms the complexity of sampling in grazed grasslands. The results of our approaches allow to refine the sampling scheme applied by the quality control service in the Walloon region.

Indeed, it could be interesting to act in a dichotomous way by adjusting the sampling scheme to the observed environmental pressure. If the analysis of one sample, composed of 30 profiles, points at a NO_3-N content higher than a threshold value, which is to be defined each year in reference fields (15 kg ha^{-1} in 2004 ≈ the mean of the 25 sampled fields = 16 kg ha^{-1}), then two additional samples of 30 profiles have to be taken to precise the observed content.

N mineralization in soils using an extraction - incubation method

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Abstract

A study was conducted to develop an improved method for measuring organic N (net) mineralization in which chemical extraction takes place in combination with suspension incubation in ammonia-absorbing membrane bottles. To obtain direct evidence of the extent to which extracted organic N is mineralizable, the extraction suspension was further incubated immediately after the extraction procedure with mild and selective extractants.

Keywords: extraction, incubation, nitrogen, phosphate, pyrophosphate

Background and objectives

A number of studies have been published on laboratory methods for estimating N mineralization to improve N fertilization recommendations (Bremner, 1965; Keeney, 1982). Standard incubations are commonly used. However, even the most intensified methods require an incubation period of at least a week (Stenberg *et al.,*1997). Therefore, chemical extraction methods have been developed. With these, part of the total soil organic N is extracted on the basis of specific biological or physical characteristics. However, even a high correlation between the extractable N fraction and estimated net N mineralization is no proof that the 'active organic N' has been extracted.

Most enzymes and other N-containing organic compounds in the soil tend to be associated to a various extent with organic and mineral soil constituents (Boyd and Mortland, 1990; Stevenson 1982). Therefore, it was hypothesized that releasing enzymes and substrates from soil with phosphates accelerates N mineralization and phosphate soil suspension incubation could offer a proper method for determination of soil mineralizable N.

In this 'extraction-incubation' method, extraction continues during the incubation but only relatively easily mineralizable organic matter is released. Standard anaerobic incubation is usually carried out in sealed N_2–flushed bottles. However, when phosphate or pyrophosphate soil suspensions are incubated, mineralization is much higher than in soil water suspensions. Further, accumulation of ammonia + (ammonium) and other gases, can affect the reaction rate and final reaction equilibrium in the sealed incubation flask. It was to avoid these effects that the membrane method was developed. With this procedure, the flask is closed with an ammonia-absorbing membrane permeable to other gases.

Material and methods

Five gram samples of ground air dry soil was extracted with 50 ml of 1) deionized water, 2) 1/15M phosphate buffer (pH 7.0) consisting of Na_2HPO_4 $12H_2O$ (14.6 g l^{-1}) and KH_2PO_4 (3.5 g l^{-1}) solution and with 3) 0,05 M $Na_4P_2O_7$ solution by shaking 1 hour on a rotary shaker in 100 ml glass jars. Each suspension was further incubated immediately after the extraction procedure at 37º C in sealed bottles (SB), sealed N_2 gas flushed bottles (SBN2) and in bottles with ammonia trapping filters (MB) for 0, 2, 5 and 10 days.

Figure 1. *Cumulative mineralized NH_4-N during 10 days' incubation. Error bars indicate differences between means of different incubation methods at each sampling date. Water suspension (a), phosphate suspension (b), and pyrophosphate suspension (c). MB = membrane bottle, SB = sealed bottle and SBN_2 = sealed N_2 gas-flushed bottle.*

Results and discussion

The maximum amount of NH_4^+-N released during 10 days' incubation was 133.0 mg kg^{-1} in the water, 208.0 mg kg^{-1} in the phosphate and 454.1 mg kg^{-1} in the pyrophosphate suspension (soil total C content 6.2% and N 0.25%). During incubation in phosphate and pyrophosphate suspensions, the mobilization was nearly linear in membrane bottles. The linearity in membrane bottles suggests that the accumulation of reaction products did not cause deceleration of the reaction rate and there were sufficient substrates for enzyme reactions to take place at constant rate in phosphate and pyrophosphate suspensions. The variation between replicates was also smallest in these bottles. It was concluded that membrane bottles were best suited to incubation when mobilization reactions were accelerated with phosphate or pyrophosphate extractants. The method was easy to perform and gave results with good replicability.

Conclusions

Net N mineralization can be accelerated considerably by releasing enzymes and substrates from soil mineral particles with phosphate extraction. It was concluded that membrane bottles were best suited to incubation when mobilization reactions were accelerated with phosphate or pyrophosphate extractants. The work is continued 1) studying the developed method with more soils, 2) studying the extraction power of water, neutral phosphate and pyrophosphate 3) studying the C and N changes during incubation 4) studying the accuracy and applicability of an extraction-incubation method for organic N net mineralization test 5) studying the influence of manure application on test values.

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N mineralization of soil organic matter extracted by phosphate buffers

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Abstract

Dynamics of soil organic N mineralization was studied with three soils using an 'extraction-incubation' method. Also the extraction power and biodegradability of the extracted organic matter were determined. Mineralization was always considerably higher in phosphate and pyrophosphate suspensions than in water suspension irrespective of soil. Pyrophosphate was much stronger extractant than phosphate buffer or water. Biodegradability of the extracted organic matter with different extraction solutions was soil dependent. In mineral soils biodegradability did not depend on extraction solution whereas in peat soil water extracted organic matter was more easily biodegradable than phosphate or pyrophosphate extracted organic matter.

Keywords: extraction, incubation, nitrogen, phosphate, pyrophosphate

Background and objectives

Dynamics of soil organic nitrogen mineralization has been studied with a number of different ways. One has been characterising of soil organic N by fractionating it to recalcitrant and labile pools. One possibility to study the labile pools of organic N is to use enzymes. Phosphate-releasing enzymes were successfully used to investigate hydrolyzable organic phosphorus in either animal manure or soils (He *et al.*, 2004). On a preliminary study commercial enzymes were used to hydrolyze soil organic N, but soil inherent enzymes easily decomposed added enzymes showing the aggressiveness of soil enzymes. As a result NH_4-N formed in hydrolysis of added enzyme can not be distinguished from that NH_4-N formed in soil organic N hydrolysis. In consequence, it was more reasonable to use soil own enzymes. Easily mineralizable chemical constituents, such as proteins and amino-sugars, may be physiochemically protected against microbial attack by binding to soil inorganic components. Therefore, it was hypothesized that releasing enzymes and substrates from soil with phosphates accelerates N mineralization and phosphate soil suspension incubation could offer a proper method for determination of soil mineralizable N. To avoid accumulation of reaction products in sealed incubation flasks, the membrane method was used (Kokkonen *et al.*, 2005). In previous study (Kokkonen *et al.*, 2005) focus was in bottle method development. In this current study the objective was to study 1) dynamics of soil org. N mineralization as a function of time with different types of soils using an 'extraction-incubatio' method (Kokkonen *et al.*, 2005) 2) to study extraction power of H_2O, neutral phosphate buffer and $Na_4P_2O_7$ and to study incubation of supernatant separated after extraction from soil solid particles and compare to the mineralization rate in soil suspensions 3) to study applicability of 'extraction-incubation' method for organic N net mineralization test and the influence of manure application on test values.

Material and methods

Dynamics of soil org. N mineralization was studied with three soils. Clay, loam and peat soil were extracted with a) a very mild solubilizing agent (H_2O); b) a neutral phosphate buffer solution with a relatively higher affinity for solid particles and c) a stronger solubilizing agent (0.05 M $Na_4P_2O_7$) by shaking for 1h. Mineralizable N was determined by further incubating of suspensions in flasks closed with an ammonia-absorbing membrane permeable to other gases at 37˚C for 0, 1, 2, 3, 5, 7, 10 and 15 days.

Extraction power of water, phosphate and pyrophosphate was studied by determining the total organic C and N content of supernatant solutions separated from soil solid particles by centrifuge after extraction (extraction as above, but extraction time 2h). Incubation of supernatants was done as above.

Method applicability for organic N net mineralization test was studied by calculating correlation between developed phosphate and pyrophosphate 'extraction-incubation' methods (Kokkonen *et al.*, 2005) with water suspension incubation and with plant N uptake. Also the effect of manure application on test values were studied with field experiments.

Results and discussion

Figure 1. Net NH_4^+-N mineralization as a function of time in water (a), phosphate (b) and pyrophosphate (c) suspensions studied by loam, clay and peat soils. Error bars indicate differences between means of different incubation methods at each sampling date.

Net ammonium mineralization as a function of time is presented in Figure 1. The amount of NH_4^+-N released during 15 d' incubation was 8, 112 and 184 mg kg^{-1} in water, phosphate and pyrophosphate suspensions (0.5, 7.0 and 11.0% of soil total org. N) in loam soil. In clay suspensions mineralization was 75, 126, 244 mg kg^{-1} (2.7, 4.5 and 8.7% of soil total org. N) and in peat 167, 414 and 389 mg kg^{-1} (1.2, 3.0 and 2.8% of soil total org. N), respectively.

Extraction power of water, phosphate buffer and pyrophosphate and biodegradability of extracted organic matter are presented in Table 1. As expected water was much weaker extractant than phosphate buffer or pyrophosphate, ca. 1% of the total SOC and SON pools were water extractable in clay and loam soils. In peat soil only 0.5% respectively. Phosphate buffer extracted on average 3.3% of the total SOC and 2.6% of the SON pools in studied three soils. Pyrophosphate was the most strongest extractant and extracted on average 27.5% of the SOC pool and 19.1% of the SON pool respectively.

Table 1. Total organic C and N content of supernatant solutions, net mineralized N during 15 days incubation of supernatants and its ratio to the organic N of supernatant in percentage.

Extraction solution	Soil	Extracted C, %	Extracted N, %	Mineralized NH_4+NO_3 mg/kg	Mineralized/ Norg extracted %
Water	Clay	1.2	1.0	6.1	25.0
Phosphate	Clay	3.6	3.4	20.6	24.9
$Na_4P_2O_7$	Clay	20.2	11.4	66.3	22.6
Water	Loam	1.3	1.3	4.4	27.4
Phosphate	Loam	3.9	4.3	11.4	23.9
$Na_4P_2O_7$	Loam	33.1	24.6	76.5	24.5
Water	Peat	0.4	0.5	23.7	39.0
Phosphate	Peat	2.4	3.1	73.9	20.0
$Na_4P_2O_7$	Peat	29.6	21.2	321.4	12.5

Considerably less NH_4^+-N was released in supernatants than in suspensions. This was especially true for water and phosphate supernatants. Thus, the percent of N mineralized of solutions' total organic N was higher in supernatants than in suspensions. In clay and loam soil phosphate and pyrophosphate extracted org. N was not less biodegradable than water extracted org. N. In peat soil water extractable org. N was more biodegradable than org. N extracted with phosphate or pyrophosphate and phosphate extractable org. N more biodegradable than pyrophosphate extractable org. N.

Correlation between NH_4^+-N released during 15 d' incubation in phosphate and water suspension was 0.93 and between water and pyrophosphate 0.92 (n=20). Comparison between methods showed that correlation between plant N uptake and phosphate and pyrophosphate incubation was significantly higher than between N uptake and water incubation.

The mean values of net mineralized N during 5 days incubation were 25.6, 77.0 and 131.7 mg kg^{-1} for water, phosphate and pyrophosphate suspension incubation in non-manured field plots. Respectively, for manured field plots the mean values were 30.2, 79.5 and 141.0 7 mg kg^{-1}. Manure application can be seen on test values responding irrespective of 'extraction incubation method.

Conclusion

The results obtained suggest that developed phosphate 'extraction-incubation' method can be applied to study dynamics of organic N mineralization to approximate the amount of mineralizable N, rate of mineralization and pools of mineralizable org. N.

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NIMF, an N-reduction project in the province of Gelderland, the Netherlands

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Abstract

Fifty four dairy farms in the province Gelderland participated in the NIMF project. Every year the surplus on the farm N balance (MINAS) was calculated, mineral N content of the soils was measured and on some farms also the nitrate concentration of the upper groundwater was sampled. Despite farmers reducing the MINAS surplus and 80% of them achieving the 2003-norm in 2002 already, the nitrate concentration did not significantly decrease and remained about twice the target of 50 mg l^{-1}.

Background and objectives

From 1999-2003 several 'Nitrate-projects' were carried out in the Netherlands. Besides a broad variety of farm level projects, also a number of regional projects were started. In the province of Gelderland the goals of the NIMF-project were:

- to stimulate farms on dry sandy soils to achieve the targets of the Dutch MINeral Accounting System (MINAS) at least one year earlier than the statutory regulations;
- to reduce the nitrate concentration of groundwater.

NIMF was an initiative of the province to stimulate the development and transfer of knowledge about nutrient management in the eastern part of Gelderland. The project was set up with the regional farmer's organization (GLTO) and the drinking water company (Vitens).

Table 1. Farm data and mineral balance for three regions (Nieuwenhuis, 2003).

Group	Hengelo ('t Klooster)		Neede Borculo		Varsseveld-Veluwe	
Number of farms	11		23		20	
Year	1999	2002	1999	2002	2001	2002
Grassland ha	23.9	24.2	25.4	30.6	26.0	29.8
Maize land ha	6.9	9.4	8.5	9.2	9.0	10.5
N application grassland kg ha^{-1}	377	303	402	292	318	296
Number of dairy cows	53	54	59	68		64
Milk kg cow^{-1}	7,418	7,652	7,8555	7,973	7,791	7,866
N input kg ha^{-1}	**438**	**314**	**489**	**338**	**315**	**292**
- concentrates	216	160	235	203	138	154
- fertilizers	199	138	216	113	156	123
N output kg ha^{-1}	**170**	**142**	**191**	**161**	**99**	**125**
- cattle	56	37	64	48	19	
- milk	70	68	77	77	75	
- manure	42	26	50	32	4	11
N surplus	**268**	**172**	**298**	**177**	**216**	**167**
Correction for inevitable NH_3 losses from animal houses'	51	39	52	45	37	34
MINAS surplus kg N ha^{-1}	**217**	**134**	**246**	**132**	**179**	**133**
2003-norm calculated including dry sandy soils		**152**		**158**		**153**

NIMF consisted of seven sub-projects stimulating adaptations in farm management, monitoring the N and P utilization, the financial results of the farms and the reduction of nitrate leaching to the groundwater. Also the effects of the use of slurry additives were monitored and the impact was studied of far-going adaptations in water catchment areas to extensify dairy farms.

Material and methods

Thirty four farms participated in the project during 3 years (2000-2002). An extra 20 farms participated in the final year 2002. Average farm size was 39 ha (approx. 29 ha grassland and 10 ha silage maize) with 64 dairy cows. Average milk production per cow was 7900 kg $year^{-1}$. For every farm the MINAS surplus (Neeteson *et al.*, 2001) was calculated every year. In October 2002 about 400 soil samples were taken at the farms and on a group of comparable farms in the province Overijssel to analyze the mineral N content in the upper 60 cm of the soil. Nitrate concentration in the upper groundwater of 50 fields from 11 farms in Hengelo ('t Klooster) was measured in the autumns of 2000-2003. Eight bore holes per field were sampled and 20 permanent wells in a neighboring pine forest as a reference.

Table 2. Mineral N content in soils under grassland, grass-clover, silage maize and other crops, autumn 2002 (Brouwer, 2003).

Crop	Number of samples	Mineral N kg ha^{-1}		
		25% lowest	average	25% highest
Grassland	227	18	45	87
Grass-clover	34	14	36	65
Silage maize	110	35	71	119
Other	13	20	108	224

Remarks: grass-clover swards contain at least 20% clover.

Results and discussion

The MINAS-surplus at the 34 farms in Hengelo and Neede Borculo decreased from 237 kg N in 1999 to 133 kg N ha^{-1} in 2002 (Table 1). It coincided with a decrease in N application rate on grassland (fertilizer N and effective N from slurry) from 394 to 296 kg ha^{-1}. There was also a reduction in the use of N in concentrates. Under intensive coaching the 20 farms in Varsseveld-Veluwe in 2002 reduced their MINAS surplus in one year from 179 to 133 kg N ha^{-1}. In 2002 the average MINAS surplus in every group was lower than the norm for 2003 on dry sandy soils and 80% of the farmers achieved the 2003-norm (Nieuwenhuis, 2003).

Figure 1. Relation of N fertilization and mineral N in autumn (Brouwer, 2003).

Autumn 2002, the mineral N content in the soil on maize land was higher than on grassland and grass-clover fields (Table 2). Highest mineral N content on grassland was measured under fields which were reseeded in autumn 2002. Other crops were mostly grain. Some of these fields got a slurry application in August or September prior to the sampling; this causes the highest mineral N values.

N-fertilization level did not relate to Nmin in autumn (Figure 1), neither with grazing pressure, but there was a positive relation between Nmin and organic matter content of the soil (Brouwer, 2003).

The nitrate concentration in the upper groundwater of 50 fields was measured in autumns of 2000-2003). The variation between years on the same field was large and showed no pattern (Van Beek *et al.*, 2004).

Table 3. Nitrate in upper groundwater in 't Klooster in 2000-2003 under farmland and pine forest, with additional data for 1989 and 1993 (Van Beek, 2003; 2004b).

Year	1989	1993	2000	2001	2002	2003
Farmland						
Number of fields	25	24	50	49	51	49
NO_3 mg l^{-1}	155.3	165.2	99.5	103.5	93.4	102.3
Nature (forest, mostly pine trees)						
Number of wells			17	16	14	
NO_3 mg l^{-1}			24.4	36.5	37.7	

The average nitrate concentration under farmland varied between 93.4 and 109.3 mg NO_3 l^{-1}, the concentration showed no significant decrease despite the reduction of the MINAS-surplus in this period (Table 3). The difference in nitrate concentration under grassland and arable fields in 2003 was 96.4 versus 120.6 mg NO_3 l^{-1} (Van Beek, 2004a).

Conclusions

Despite their efforts to reduce the MINAS surplus and although 80% of the farmers in 2002 achieved the 2003-norm, the nitrate concentration showed during these years no significant decrease. The nitrate concentration under farmland is still about twice the target of 50 mg l^{-1}.

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Multifunctional land use and its impact on the nitrate concentration in groundwater under grassland

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Abstract

During three years data of six grassland types were collected on 14 farms. Grass production, number of plant species, N fertilization, Nmin in autumn and nitrate leaching in following spring showed a large variation. There was no relation between N-fertilization level and Nmin, neither between Nmin in autumn and nitrate concentration in the following spring.
Multifunctional land use, including diversification and extensification of grassland, can help to meet the target of 50 mg nitrate per l in groundwater.

Keywords: DM production, fertilizer, grassland, leaching, nitrate, nitrogen

Background and objectives

The objective of multifunctional land use is to combine different functions (e.g. food production, nature conservation, environmental protection and recreation) within one area. Multifunctional land use is considered as an option to enlarge the economical and environmental sustainability of an area. In 1998 a large program on sustainable land use started in the Winterswijk area. The area consists of a small-scale landscape with a dominant role for agriculture. Substantial parts of the area are covered by nature, forests, recreation areas and campsites. The focus of the program is:

- to create more variation in grasslands and arable crops, than only perennial ryegrass swards and silage maize
- to enlarge the ecological values of farmland
- to make the area more attractive for recreation and tourism
- to reduce environmental losses like nitrate leaching.

Material and methods

From 2002 till 2004 an intensive monitoring programme was carried out on 14 farms covering a broad range in farming types. Relations between fertilization, production, farm income, biodiversity and environmental issues were analysed thoroughly.
In this paper we present results of different grassland management types on dry matter production, number of plant species, N-fertilization (fertilizer N and effective N from slurry), N harvested in the crop, residual Nmin in autumn and nitrate in upper groundwater in the following spring.

Results and conclusions

The grasslands were grouped into the six types (Table 1). Total N-fertilization (mainly cattle slurry) on grass-clover swards was less than half the amount of fertilized ryegrass swards (which were regarded as a kind of control for the ordinary farming system in that region), but the average dry matter production, N-uptake and nitrate concentration on grass-clover swards were similar. The average clover content (expressed as ground cover by clover leaves) on the grass-clover fields was 33%.

Fertilization levels on the fertilized grass mixtures (of native grasses) and species-rich grasslands (grassland with wild flowers) were low. Nitrate concentration in groundwater under grass mixtures and species-rich grasslands (fertilized and unfertilized fields) was far below the EU-limit.

Table 1. Number of plant species, dry matter yield, N-fertilization, N-uptake, N-min in autumn and NO_3^- concentration in groundwater in following spring on different multifunction grassland types in 2002 to 2004.

Grassland type	n	Number of plant species on 100 m^2	DM production t ha^{-1}	N-fertilization kg ha^{-1}	N-uptake by the grass kg ha^{-1}	N-min autumn (layer 0-60 cm) kg ha^{-1}	NO_3 in upper ground-water mg l^{-1}	NO_3 % of fields <50 mg l^{-1}
Ryegrass; fertilized	9	11	12.1	191	370	59	92	33
Grass-clover	7	8	11.9	84	384	36	95	43
Grass-mixtures; fertilized	5	18	9.3	57	239	29	18	80
Grass-mixtures; unfertilized	6	17	8.1	0	153	25	18	100
Species-rich; fertilized	8	29	9.8	62	235	17	25	75
Species-rich; unfertilized	9	21	5.2	0	83	8	5	100

Figure 1. Relation of N fertilization and mineral N in autumn.

Figure 2. Relation of mineral N in autumn and nitrate in upper groundwater in the following spring.

There was no relation between N-fertilization level and Nmin (Figure 1), neither between Nmin in autumn and nitrate concentration in the following spring (Figure 2).

Figure 3. Relation of N fertilization and nitrate in upper groundwater in the following spring.

The relation between N fertilization and nitrate is a complicated one (Figure 3). There were two subgroups in the data. The ryegrass and grass-clover swards had a high N input by fertilization, fixation by clover and excretion by cattle and most of them (10 of the 16) were located on soils with a good water management (low groundwater table). The 28 grass-mixtures and species-rich grasslands had a much lower N input (all < 200 kg N ha^{-1}) and almost all these fields (15 of the 28) were situated on wet soils. These fields were not of only a part of the season grazed and cut once or twice a year for hay or silage. On wet soils more denitrification may be expected. So the lower nitrate concentration under grass-mixtures and species-rich grasslands is explained as a combination of a lower N input, less grazing and more denitrification.

Grass-mixtures and species-rich grasslands are more common on soils with a high groundwater table, for instance along brooklets, than grass-clover and fertilized ryegrass fields. A low fertilization with a maximum of 100 kg N ha^{-1} stimulates grass production to a level that still a high biodiversity is achievable and the nitrate leaching stays at an acceptable level.

Conclusions

Multifunctional land use, more precisely diversification and extensification of grassland, in most cases brings by less grass production, but more plant species and more often meeting the target of 50 mg nitrate per l in groundwater. A low total N-fertilisation level (on grass-clover swards) however is no guarantee to meet the NO_3 target.

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Quantifying the contribution of net mineralised soil organic matter N to uptake by permanent pasture

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Abstract

In permanent grassland not containing clover, background (non-fertiliser) N is mainly made available to plant roots in the soil from net mineralisation of soil organic matter N. The objective of this study was to measure the rate of supply of background N from the soil under permanent grassland throughout the growing season. The study was undertaken at two sites with contrasting soil types, over two years, on both cut swards and swards grazed by dairy cows. Over the two years of the study background N supply averaged 127 kg ha^{-1} yr^{-1}. The rate of supply of background N on a weekly basis was influenced by soil and climatic conditions ranging from circa 0.35 kg N ha^{-1} d^{-1} in February/March to circa 0.77 kg N ha^{-1} d^{-1} in May then declining to circa 0.45 kg ha^{-1} d^{-1} in October (mean of treatments, sites and years). Knowledge of the supply of background N during the growing season allows fertiliser N applications to be refined to more accurately meet sward requirements. This lowers the quantity of N that can be lost to water and air.

Keywords: fertiliser, grassland, mineralization, nitrogen

Background and objectives

Permanent grassland accounts for over 90% of agricultural land area in Ireland. Dairy farms make most intensive use of grassland and average fertiliser N use on Irish dairy farms in 2000 was 176 kg ha^{-1} yr^{-1}(Coulter *et al.,* 2002). There is pressure to lower N losses from intensively managed grassland. Legislation such as the Nitrates Directive (91/676/EEC), the Water Framework Directive (2000/60/EC) and the National Emission Ceilings Directive (2001/81/EC) is in place to reduce losses to water and air. Aiming to maximise N-use efficiency of grassland is an important component of reducing N losses from farms. In the absence of clover the main supply of background N (non-fertiliser N) is net mineralised soil organic matter N. Other sources are inorganic N in the soil profile in spring and atmospheric deposition. The supply of background N under permanent grassland can be substantial and can affect the requirement for fertiliser N (Hassink, 1995). The quantity of background N which becomes available for plant uptake during the growing season depends on soil characteristics and factors such as rainfall, soil temperature, depth of topsoil, soil organic matter (SOM) content, etc. (Herlihy, 1979; Hatch *et al.,* 1991; Gill *et al.,* 1995). As a result the rate of supply of background N is variable during the growing season. This study aimed to measure the rate of supply of background N from soil under permanent pasture on a weekly basis throughout the growing season. This information underpins the development of more efficient fertiliser N application strategies for grassland.

Materials and methods

The experiment was carried out during 2001 and 2002, at two sites with contrasting soil types. Moorepark, (52°09'N, 08°15'W) was a free draining acid brown earth with sandy loam texture, 0.45 m depth of topsoil and 80 g kg^{-1} organic matter in the upper 0.20 m. Solohead, (52°51' N, 08°21'W) was a mixture of grey-brown podzolic and gley with impeded drainage and texturally a clay loam. Depth of topsoil was 0.22 m and 130 g kg^{-1} organic matter in the upper 0.2 m. Seven-year average annual rainfall was 1028 mm at Moorepark and 1063 mm at Solohead.

The supply of background N was assessed retrospectively by measuring the amount of N in grass harvested at weekly intervals from unfertilised grassland without clover (Brockman, 1969; Hassink, 1995). Treatments were: (1) swards not receiving fertiliser N and not grazed (C) and (2) swards not receiving fertiliser N and grazed (G). To measure the rate of soil N supply the experiment was arranged as outlined by Corrall and Fenlon (1978). Each treatment consisted of a series of five plots and grass was harvested from one of the plots each week. Hence there was five weeks grass growth on each plot at harvest. The five plots of both treatments were arranged in a randomised block design. Each block was replicated three times at both sites. Grass samples were taken to 50 mm above ground level. After harvest, pre-conditioned non-lactating dairy cows were assigned to the grazing plots on a grass allowance basis and grazed for 24-hour periods. Grass N concentrations were determined after oven drying at 40°C for 48 hours using a LECO 528 auto-analyser.

Results and discussion

There was no significant difference in annual grass N uptake between the two treatments, nor was there any significant interaction between treatment, site and year. This indicates that any impact of N recycled by the grazing cows on plant-available soil N supply was not detected in this experiment. Grass N uptake was significantly ($P<0.05$) higher from the clay loam than from the sandy loam being 134 kg ha^{-1} and 119 kg ha^{-1}, respectively (mean of treatments and years; s.e.m. = 3.4). Mean grass N uptake was significantly ($P<0.001$) higher in 2002 (140 kg ha^{-1}) than in 2001 (114 kg ha^{-1}) (mean of treatments and sites; s.e.m. = 3.4). Differences between sites were attributed, in part, to the higher SOM content in the clay loam than in the sandy loam soil. Weather conditions contributed to differences in grass N uptake between the two years. Rainfall in 2001 was 829 mm and 1175 mm in 2002 (mean of two sites). It appears that moister (greater wetting and drying) soil conditions during 2002 enhanced the mineralisation of SOM-N compared with 2001 (Herlihy, 1979).

Figure 1. *Supply of background N (kg ha^{-1} d^{-1}) on cut (C) and grazed (G) permanent grassland not receiving any fertiliser N, at Moorepark (M) and Solohead (S) during 2001 and 2002:*
*□ = C M 2001; ■ = G M 2001; ▲ = C S 2001; Δ =G S 2001; ○ =C M 2002; ● = G M 2002; + = C S 2002; * = G S 2002.*
Data are mean of three replications per site: Year x treatment x week & site x treatment x week: P < 0.001; s.e.m. = 0.03, I = ± s.e.m.

The rate of supply (week by week) of background N was affected by significant ($P<0.001$) interactions between treatment, week and year and between treatment, week and site (Figure 1). It varied during the growing season, ranging from circa 0.35 kg N ha^{-1} d^{-1} in February to circa 0.77 kg N ha^{-1} d^{-1} in May then declining to circa 0.45 kg ha^{-1} d^{-1} in October (mean of treatments, sites and years).

Mean background N supply of 127 kg ha^{-1} yr^{-1} (mean of treatments, sites and years) can amount to a significant proportion of the swards' requirement for N. For example, grassland producing 10 t DM ha^{-1} yr^{-1} and containing an

average N concentration of 30 g kg^{-1} in the grass DM requires the uptake of 300 kg ha^{-1} yr^{-1} of N from the soil. Background supply of 127 kg ha^{-1} yr^{-1} amounts to 0.42 of these requirements and is obviously an important source of N for pasture production.

Knowledge of the rate of background N supply provides for the development of fertiliser N application strategies that take into account this important source of N in the soil lowering reliance on fertiliser N, thus improving overall efficiency of fertiliser N use.

Conclusions

There is scope to improve efficiency of fertiliser N use by permanent grassland by developing application strategies tailored to the pattern of supply of background N during the growing season.

Acknowledgements

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Nitrate in groundwater of farms and relationship with fertilisation, soil type and groundwater table

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Abstract

Intensification of agriculture from 1950 onwards has led to increased emissions of nitrogen and phosphorus to the environment. European legislation is formulated to achieve 'good water quality for all purposes'. To achieve this, knowledge is required on the relationship between N management and nitrate concentration in groundwater. On thirty-four farms, nitrate in shallow groundwater was measured. Nitrate concentrations in groundwater were strongly influenced by local circumstances such as soil type, the occurrence of peat layers and the depth of the groundwater table. The effect of fertilisation variables and soil mineral nitrogen in autumn on nitrate in groundwater was studied in a subset of sandy soil without peat layers. Differences in nitrate concentration in groundwater were better described by SMN than by N-surplus or N-input. The relationships increased when farm averages were used instead of individual sampling data, and when averages over several years were used. This is explained by variability caused by changes in the pool of organic N in the soil, dilution in the precipitation surplus or the time of vertical transport between soil surface and groundwater. Therefore, to show effects of changing fertilisation on nitrate concentration in groundwater, it is required that sampling is carried out over several years and at least includes all crops of a rotation cycle. At some farms net mineralization from the soil N pool will have occurred as measured nitrate concentrations were higher than those calculated from N-surplus or SMN and an average precipitation surplus of 350 mm. Net mineralization complicates achievement of a nitrate concentration below 50 mg/l, especially on well-drained sandy soils where denitrification is low. Extrapolation of the regression equation data showed that nitrate concentrations in groundwater of 50 mg/l are achieved with a surplus of 80 kg/ha for poorly drained soils. For well drained soils, a negative surplus of 40 kg/ha is required.

Keywords: leaching, nitrogen, nutrient balance, soil

Background and objectives

The intensification of agriculture from 1950 onwards has led to increased emissions of nitrogen and phosphorus to groundwater and surface waters (Matson, 1997; Smith, 2003). To protect groundwater and reduce or prevent eutrophication of surface waters, member states of the European Communities are obliged to establish national action plans to reduce nutrient emissions in order to achieve 'good water quality for all purposes' (EC, 1991; EC, 2000). To determine whether nitrate loss is sufficiently reduced to achieve good water quality, knowledge is required on the relationship between N management and nitrate concentration in groundwater.

Material and methods

Part of the Dutch action program was the project 'Farming with a future'. The project focussed on arable land that comprised agriculture, horticulture, bulb cultivation and tree cultivation (nursery stock). Thirty-four farms cultivating a large variety of crops on different soil types participated in this project (Figure 1, Table 1). Four of these farms were experimental farms at which different farming systems were designed and tested on their capability to achieve desirable water quality.

Fertilisation, other N inputs and crop yields were registered by the farmers from 2000 through 2003. These data were complemented with data from other sources to calculate nitrogen balances. Every autumn, soil mineral nitrogen (SMN) in the profile (0-90 cm) was measured at all fields. In three successive years (2002-2004) the nitrate

concentration in shallow groundwater was measured at 16 to 48 sampling sites per farm. Regression analysis explained variation in nitrate concentration in groundwater using the variables groundwater table, SMN, N-surplus, mineral-N-surplus, total N-input and mineral N-input.

Figure 1. Location of the farms.

Table 1. Soil types of the groups of farms from Figure 1.

Group	Soil type
Ar-sw	clay
Ar-ne	sand: reclaimed peat, relatively high OM-content
Ar-se	sand
Ho-cb	sand
Ho-se	sand
Bulb	coarse dune sand, low OM-content, high groundwater table
Nur	sand

Results and discussion

Soil type strongly affected nitrate concentration in shallow groundwater. Low nitrate concentrations in groundwater were found in dune sand with high groundwater tables (4 mg/l) and in clay soil (18 mg/l). In sandy soils, nitrate concentrations were higher and varied between groundwater table classes. Average nitrate concentration was 158 mg/l in dry, well drained soils and 77 mg/l in poorly drained soils. Presence of a peat layer of at least five cm thickness reduced the nitrate concentration in sandy soil by about 80 mg/l. Peat layers occurred in eight percent of the sampling sites in sandy soil.

The effect of fertilisation variables was studied in a subset of sandy soil without peat layers. In sandy soil without peat layers, differences in nitrate concentration in groundwater were better described by SMN than by N-surplus or N-input. The relationships increased when farm averages were used instead of individual sampling data, and when averages over several years were used. This is explained by reduced variability in linking N management and nitrate

concentration in groundwater. Variability is caused by changes in the pool of organic N in the soil, dilution in the precipitation surplus or the time of vertical transport between soil surface and groundwater. Averaging may reduce variability as overestimations of the effect of a variable on nitrate in groundwater with one crop or year can be counterbalanced by an underestimation with another crop or year (Van Beek *et al.*, 2003; Oenema *et al.*, 2003). Therefore, to show effects of changing fertilisation on nitrate concentration in groundwater, it is required that sampling is carried out over several years and at least includes all crops of a rotation cycle.
At some farms net mineralization from the soil N pool will have occurred as measured nitrate concentrations were higher than those calculated from N-surplus or SMN and an average precipitation surplus of 350 mm. Net mineralization complicates achievement of a nitrate concentration below 50 mg/l, especially on well-drained sandy soils where denitrification is low. Extrapolation of the regression equation data showed that nitrate concentrations in groundwater of 50 mg/l are achieved with a surplus of 80 kg/ha for poorly drained soils. For well drained soils, a negative surplus of 40 kg/ha is required.

Conclusions

- On well-drained sandy soil without peat layers, nitrate concentrations below 50 mg/l are difficult to achieve.
- Sampling should be carried out over several years and at least include all crops in a rotation cycle to show effects of changing fertilisation on nitrate concentration in groundwater.

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The use of an indicator for nitrate concentrations at different scale levels

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Abstract

The project 'Focus on Nitrate (2000-2004)' aims to establish and test indicators for NO_3^- concentrations in groundwater (regression models) in dairy and arable farms. In this paper we show the models resulting from the analyses and discuss the use of the models at different scale levels.

Background and objectives

Leaching of nitrate from agricultural lands to groundwater is a problem in the Netherlands, as well as in other regions of Europe. While action programs are being implemented to reduce NO_3^- emissions, there is a need to monitor progress at farm level as well as regional or national scales. Monitoring of nitrate concentrations is labor-intensive and expensive. Therefore there is a need for indicators.

Materials and methods

The Focus on Nitrate study was based on a procedure to acquire an even representation of soil types, groundwater regimes, and crops prevailing in the sand districts of the Netherlands. About 60 different combinations of soil-crop-groundwater regime were identified, and are referred to as 'land use units' (LUU). Observations refer to $20m^2$ 'spots' distributed among these LUU's. At each spot soil mineral N was measured at three depths (0-30, 30-60 and 60-90 cm) in autumn, the nitrate concentration in the top 1m groundwater was measured in spring. In addition many other observations were made (Hack-ten Broeke *et al.,* 2004). The search for indicators and establishment of regression models was performed on 478 spots, distributed among 34 farms. The models were tested on accuracy and applicability for different scale levels. For testing data from LUU's distributed among 19 farms were used. (Smit *et al.*, 2004) In addition LUU's in three regions, were sampled (about 300 spots) (Roelsma *et al.*, 2003)

Results and discussion

The results from the model development show that for all soil-crop-groundwater regime combinations the best performance was obtained if we used the NO_3^- part of N_{min}, denoted as $N_{min,NO3}$, summed over 90 cm depth. Differences between land use and ground water regime were only expressed by different intercepts (Figure 1). The explained variance of the simplest models ranged from 21% for grass, 24% for maize to 36% for arable crops, while the standard errors amounted respectively 49.8 mg/l, 65.6 mg/l and 59.6 mg/l. The performance could be slightly increased by adding other parameters like cumulative precipitation during summer or winter, or N-rate; although the explained variances increased only by a few percent and the decrease of the standard errors were 5 mg/l at most. However, testing showed that the more complicated models were less stable. Aggregation to farm levels and regional scale level brings much improvement.

Figure 1. Response of NO_3^- concentration in groundwater to residual mineral soil N (Nmin-NO3) according to the regression model obtained from data for grass, maize and arable crops. Based on pooled 2000/2001, 2001/2002 and 2002/2003 data.

Predictions of NO_3^--concentrations at the whole farm level are composed from the individual LUU-nitrate values, given a specified farm composition (the respective areas of LUU's). The nitrate concentrations at whole-farm level were predicted better than the predictions at the individual 'spots'. Linear regression analysis of these results showed a significant relation, although at higher values (> 68.7 mg.l^{-1}) the predicted nitrate concentrations are under estimated (Figure 2).

Figure 2. Predicted and measured nitrate concentrations (mg.l-1) at 'spot-level' (upper graphs), at farm-level (lower left graph) and regional scale (lower right graph). Based on 2003/2004 test data.

Predictions of NO_3^--concentrations for LUU at the regional level also appeared to be rather good. Using de LUU per region an average value per region could be composed. These results indicate that predicting the nitrate concentrations in the upper groundwater

When using these models for monitoring, it should be kept in mind that the models are only applicable at the current range of nitrate concentrations. Decreasing N rates in order to diminish the mineral N in soil and eventually the nitrate concentrations may also mean that the models become less useful. Therefore, monitoring nitrate concentrations by measuring mineral N in soil and predicting nitrate concentrations in groundwater should be combined with measurements in groundwater. Only in this way, the models can be adjusted to new situations.

Conclusions

The use of the soil-crop-groundwater regime combinations or LUU's appeared to be a very useful approach. The data collected at the relatively small spots could be aggregated to different larger scale levels, varying from farm level tot regional level. The accuracy of the predictions increased at larger scales. The models can be used for monitoring nitrate concentrations, but they should be updated frequently as N rates in Dutch agriculture are continuously subject to change.

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N mineralization and denitrification in irrigated and warm conditions

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Abstract

In agricultural systems, denitrification is often considered to be a minor cause of N loss compared to nitrate leaching and ammonia volatilisation. However, high levels of irrigation in summer could create favourable conditions for denitrification. Nitrogen mineralization and denitrification were studied in a covered field plot submitted to different irrigation treatments, using simple dynamic models and ^{15}N isotope techniques. The average ^{15}N recovering rates ranged from 66 to 76%. As *i)* no nitrate leaching was calculated by a N balance method and corroboratively no ^{15}N was measured in the deeper layer at 60-90 cm and *ii)* less than 10% of ^{15}N was immobilized, the major part of the N loss may be attributed to denitrification. Soil mineral-N dynamics over the two months of the experiment were well simulated by the LIXIM program when denitrification was taken into account. The simulations made with the NEMIS model confirmed the assumption of denitrification based on ^{15}N balance results. We suggest that denitrification can occur just after irrigation and could be a major cause of N loss under warm conditions when the topsoil contains high nitrate-N levels (i.e. after fertilizer-N application).

Keywords: ^{15}N, denitrification, LIXIM model, nitrogen mineralization, recovery

Background and objectives

In agricultural systems, denitrification is often considered to be a minor cause of N loss compared to nitrate leaching and ammonia volatilisation (Zhang *et al.*, 2004). Studies dealing with denitrification have mostly been carried out on water saturated soils during winter and have focused on N_2O production (Teepe *et al.*, 2004). However, we hypothesized that denitrification can also occur in warmer conditions with massive water additions due to irrigation. These conditions, through drying / rewetting cycles, may also induce a flush of soil nitrogen mineralization (Agarwal *et al.*, 1971).
Our objective was to investigate the effect of irrigation on the N dynamics in field conditions, by focusing on i) denitrification and ii) a possible occurrence of a N mineralization flush.

Material and methods

A field experiment (clayed loamy soil) was carried out in semi-controlled conditions from August to September 2003 at the INRA Toulouse station (South-West France). A covered field plot was used to control the water budget (rainfall and irrigation). The soil was maintained bare in order to avoid errors in estimating N mineralization due to errors in plant N uptake measurements. Four treatments were applied in PVC cylinders: 1) no irrigation, 2) moderate irrigation every week, 3) high irrigation every two weeks, and 4) moderate irrigation every two weeks. Soil moisture and temperature were monitored continuously. Soil mineral N was measured weekly in four layers up to 90 cm depth. N leaching below 90 cm was calculated using the LIXIM program in calculation mode (Mary *et al.*, 1999). Denitrification was estimated based on the recovery of ^{15}N labelled KNO_3 added in the irrigation water every two weeks. A small amount of tracer was applied in order to minimise changes in mineral N in soil, i. e. 5 kg ha^{-1} as NO_3-N. ^{15}N was measured four times during the eight weeks of the experiment (once every two weeks). Denitrification was also simulated using the NEMIS model (Henault and Germon, 2000).

Top soil (0-30 cm) was incubated for four months in conditions similar to those encountered in the field experiment: warm (28°C) and dry (30% water filled pore space). This incubation study was carried out to calculate the potential N mineralization rate of the soil, using temperature and moisture functions calibrated in another incubation study (results not presented here). The potential N mineralization rate is expressed in normalized days (ND), a hydrothermal time which takes into account the effect of soil moisture and temperature on nitrogen mineralization (e.g. Mary *et al.*, 1999). The calculated potential mineralization and denitrification rate were imposed in the LIXIM program so that N dynamics could be simulated using the aforementioned temperature and moisture functions (LIXIM in simulation mode). The goodness of the simulation was evaluated with model efficiency criteria (Smith *et al.*, 1996).

Results and discussion

In the Toulouse experiment, the average ^{15}N recovering rates (means of all sampling dates) were 66%, 68% and 76% in treatments 2, 3 and 4, respectively. According to the LIXIM program, there was no nitrate leaching for any treatment. Less than 10% of ^{15}N was immobilized in the soil organic fraction, probably because ^{15}N was applied as nitrate which induces less immobilisation than other sources of labelled N (Malhi *et al.*, 1996). Total ^{15}N contents in the deepest layers were not significantly different from the natural ^{15}N abundance measured in treatment 1 which did not receive labelled KNO_3 (no irrigation). The conclusion is that there was no ^{15}N leaching below 30 cm. Based on these results, the major part of N losses could be attributed to denitrification. To test this assumption, the NEMIS model was run. The denitrification simulated with the NEMIS model was in good agreement with the supposed ^{15}N loss for treatment 2 (irrigated every week) but was not as good for treatments 3 and 4 (irrigated every two weeks) (Figure 1). The model gave lower values of denitrification than indicated by unaccounted for ^{15}N. However, this confirmed that the measured N losses were mainly due to denitrification.

Figure 1. Total denitrification calculated from ^{15}N recovery (black blocs with standard error bars) and using the NEMIS model (grey blocs) for each date of measurement for treatment 2 (A) and for the whole period of experimentation for all irrigated treatments (B).

The potential N mineralization rate calculated in the incubation study (0.3 kg N ha^{-1} ND^{-1}) was used to parameterize the LIXIM program. When no irrigation was applied (treatment 1) the LIXIM model simulated water and soil mineral-N dynamics well (Figure 2). Model efficiency was 0.95 for water and 0.81 for nitrate. In the case of irrigated treatments (2, 3 and 4), nitrate contents were overestimated in the top layers. However, soil mineral-N dynamics were well simulated with LIXIM for those treatments when denitrification calculated on the basis of ^{15}N balance was taken into account (Figure 2).

Figure 2. Measured soil mineral-N dynamics in the two top layers for treatments 1 and 3 (symbols with errors bars) and simulated results with LIXIM without (dashed lines) and with (solid lines) denitrification.

Moreover, no flush of net N mineralization was observed in our field conditions, as soil mineral-N dynamics in irrigated treatments were well simulated with a single potential mineralization rate, despite the four drying-rewetting cycles (Figure 2).

Conclusions

We suggest that denitrification can occur just after irrigation and could be a major cause of N loss under warm conditions when the topsoil contains high nitrate-N levels. Consequently, these results point out the need for integrated management of irrigation and fertilisation practices in order to avoid N losses due to denitrification.

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Quantification and prediction of *in situ* nitrogen mineralization in various French arable soils

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Abstract

Simple predictive models are needed to estimate N mineralization with a reasonable accuracy for a large range of cropping systems and pedoclimatic conditions. *In situ* N mineralization was investigated in 55 arable soils spread over France. The soil was fallowed for one year and water and mineral-N were measured monthly. On the basis of these measured data, *in situ* nitrogen mineralization rate, expressed in normalized days (defined on the basis of temperature and moisture effects on soil mineralization) was calculated with the LIXIM program. For each site, soil characteristics (e.g. texture, C and N contents, CaCO3, pH, CEC, …), biological indicators (e.g. microbial biomass, …) and cropping system history were recorded. Descriptive models were formulated with 2 statistical methods: i) stepwise multiple linear regression (MLR) and ii) Partial Least Square regression (PLS). The predictive quality of the models was evaluated using cross-validation criteria. The statistical model based on the MLR method explained 75% of the *in situ* N mineralization variability. Better results were obtained with the PLS method (R^2 = 81%), because qualitative data (that could not be taken into account in the MLR-method) significantly contributed to the predictive quality and PLS allowed inclusion of all sites, even if there were missing values (for 8 of the 55 sites).

Keywords: LIXIM model, mineralization, Partial Least Square regression

Background and objectives

Improved nitrogen fertilization management requires better knowledge of soil nitrogen mineralization. Many studies have demonstrated the effect of environmental factors on N mineralization, the most important being soil temperature and moisture content (Cassman and Munns, 1980). Soil properties are also known to have a strong influence on soil N mineralization (Jarvis *et al.*, 1996). The objective of our study was to measure and analyse N mineralization variability within a selection of French arable soils. Permanent soil properties were selected in order to explain and predict N mineralization variability. Temperature and moisture functions were calibrated in a previous study, to calculate normalized days ND (e.g. Mary *et al.*, 1999). This concept allows standardization of *in situ* N mineralization rates to a constant soil temperature and moisture value. Then we can compare *in situ* N mineralization rates observed in field experiments under varying and variable climatic conditions. Our final objective is to propose a predictive model of soil nitrogen mineralization, based on permanent soil characteristics and qualitative information on the cropping system.

Material and methods

In situ N mineralization was investigated in 55 arable soils, representing most of the French arable and pedoclimatic conditions. Field experiments were carried out under bare soil conditions, following various crops. Soil mineral N and water content were measured monthly for one year over 3 or 4 layers up to 120 cm depth. Crop residues were exported to avoid net nitrogen immobilization. Nitrate leaching was calculated using the LIXIM program (Mary *et al.*, 1999). Net N mineralization was deduced from soil water and N budgets and expressed in normalized days ND (expressed at 15 °C and soil moisture at field capacity). Soil N and C mineralization were also measured under

controlled conditions (incubation study for 6 months at 21°C). These *in vitro* results were also used to estimate mineralization rates of labile C and N pools using exponential functions derived from Stanford and Smith (1972). Several soil characteristics were measured on the 55 sites: soil texture, contents of $CaCO_3$, organic C and N, P-Olsen, pH, CEC, water content at field capacity and wilting point. Biological characteristics measured were microbial biomass and metabolites and organic matter fractions (>50 µm). Cropping system history, such as preceding crops and rotation, was also recorded.

In situ N mineralization was predicted using statistical analyses. First, stepwise multiple linear regression (MLR) using the Backward method was used to analyse N mineralization variability. However, this classical statistical tool has limitations: it is only suitable for quantitative data and does not take into account samples with missing values which was the case for 8 of the 55 sites of our database. We also wanted to test a method using qualitative information. For that purpose, the Partial Least Squares regression (PLS) seemed suitable (Wold *et al.*, 1983). All data were mean-centred before analysis to allow model comparison. The predictive quality of the models was evaluated using cross-validation.

Results and discussion

In situ N mineralization rate varied from 0.2 to 1.6 kg $N.ha^{-1}.ND^{-1}$. A similar range was obtained in the incubation study (0.1 to 1.5 kg $N.ha^{-1}.ND^{-1}$). However, the significant correlation (R^2 = 0.44) between these *in situ* and *in vitro* N mineralization is not as strong as expected (Figure 1). Surprisingly, potential mineralization recorded in the incubation study was lower than that calculated with the LIXIM program for most of the sites. This could be due to errors in water and N measurements in the field studies and/or budget calculations. Parameters used to calculate Normalised Days could also be inadequate for some pedoclimatic conditions.

Figure 1. Relation between in situ N mineralization rate calculated with the LIXIM program and measured in vitro rate in the incubation study, expressed in kg $N.ha^{-1}.ND^{-1}$.

The variability in *in situ* N mineralization was investigated by taking into account all available information, including eventually *in vitro* N mineralization rate. Using multiple linear regression (MLR), the model had 9 significant variables and explained 75% of the *in situ* N mineralization (Figure 2A). The main parameter selected was soil organic N content ($t.ha^{-1}$ in 30 cm depth). Other easily available parameters, such as clay, organic carbon, sand, loam and phosphorus contents, pH and soil moisture at wilting point, had a significant contribution to the model. Surprisingly, biological indicators and $CaCO_3$ content did not improve the predictive quality of the MLR model, as expected on the basis of literature (e.g. Jarvis *et al.*, 1996).

Figure 2. In situ measured vs predicted N mineralization rate A) MLR, B) PLS methods.

Using Partial Least Squares regression (PLS), the model had 4 significant principal components, corresponding to 12 variables and explained 81% of *in situ* N mineralization (Figure 2B). In contrast to the MLR model, the main contributor was the labile C pool mineralization rate, though soil organic N content was still important. Other biological indicators (microbial biomass, metabolites and organic matter fractions >50 µm) also had a significant influence in the PLS model. Some qualitative variables (types of the two preceding crops and crop rotation) explained a significant part of the variability in *in situ* mineralization. The discrepancy with the MLR model could be explained by the fact that biological indicators appeared strongly correlated to soil C and N contents. Now the MLR method can not select variables that are strongly correlated, in contrast to the PLS method (Wold *et al.*, 1983). Hence, the predictive quality of the model based on PLS was better than that of the MLR model, as indicated by the cross-validation criteria.

Conclusion

This study showed that *in situ* N mineralization could be reasonably predicted using permanent soil characteristics and qualitative information for a wide range of pedoclimatic conditions and arable cropping systems. The PLS method appeared well-suited for the purpose and permits selection of the statistically best predictive model.

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Implementation of the Nitrate Directive in Belgium: the Agricultural Surface Survey

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Abstract

In the Walloon Region of Belgium, the EEC 91/676 European Directive is implemented through the Programme for Nitrogen Sustainable Management in Agriculture ('Programme de Gestion Durable de l'Azote en agriculture', P.G.D.A.). This P.G.D.A. limits the application of organic N on grasslands and crop lands. A derogation of these limits can be granted if the farmers undertake to follow a 'Quality Approach' (Démarche Qualité - DQ) with very close surveillance and the implementation of specific nitrogen-control techniques. Such evidence of good nitrogen management in their farming practice is obtained by means of the soil nitrate profiles established at the beginning of the leaching period and compared with standards. In order to determine the annual standards for these nitric nitrogen concentration profiles, a network of 25 pilot farms is monitored by GRENeRA – Gembloux Agricultural University and its partner, the Laboratory of Grassland Ecology - Catholic University of Louvain. These 25 pilot farms constitute the referential for the 'agricultural surface survey' (Survey Surfaces Agricoles - SSA). 2004 was the first year of using the standards to evaluate DQ farmers. Only 44% of them had a good score.

Keywords: management, nitrate, nitrogen

Background and objectives

In the Walloon Region of Belgium, the EEC 91/676 European Directive is implemented since 2000 through the Programme for Nitrogen Sustainable Management in Agriculture ('Programme de Gestion Durable de l'Azote en agriculture', P.G.D.A.) (Anonymous, 2002). This P.G.D.A. limits the application of organic N to 210 kg ha^{-1} on grasslands and 120 kg ha^{-1} units on crop lands (80 kg N ha^{-1} within vulnerable zones). A derogation of these limits (210 becomes 250 and 120 or 80 becomes 130) can be granted if the farmers undertake to follow a 'Quality Approach' (Démarche Qualité - DQ) with very close surveillance and the implementation of specific nitrogen-control techniques (Delloye *et al.*, 2003). Such evidence of good nitrogen management in their farming practice is obtained by means of the soil nitrate profiles established at the beginning of the leaching period and compared with standards. GRENeRA – Gembloux Agricultural University and its partner, the Laboratory of Grassland Ecology (ECOP) - Catholic University of Louvain, have been put in charge of fixing these annual standards by the Walloon Government. They are established by means of pilot farms which constitute the referential for the 'agricultural surface survey' (Survey Surfaces Agricoles - SSA).

Material and methods

In order to fix these annual standards, a 25 pilot farms network (part of Agricultural Surface Survey) distributed throughout the Walloon Region is monitored since 2002 by two scientific teams (GRENeRA and ECOP). Within these farms, more than 200 fields have been selected to fit (Vandenberghe and Marcoen, 2004):

- soil types: the soils of the selected fields are representative of soils of the areas where they belong to,
- type of culture: only cultures accounting for more than 5% of the total used agricultural area of the region are taken into consideration.

Results and discussion

For three years, the nitrate profiles have been measured in these fields spread over the pilot farms. In order to minimise nitrate residue in the soil during the leaching period, nitrogen fertilization advices following calculation of required amounts are given to those farmers in spring. After the harvest, the evolution of the nitrate residue in the soil is monthly monitored from October to December. The samples are taken and analysed according to the strict technical specifications defined by GRENeRA (Anonymous, 2004). Fields are split into 3 classes: from low nitrate residue expected (sugar beet, cereals with catch crop, ...) to high nitrate residue expected (potato, maize, ...). The results are used to establish annual standards. To take into account the meteorological conditions and the date of the sampling, results are present by graphs (Figure 1).

Figure 1. Evolution of nitric nitrogen concentration profiles after cereals and catch crop.

Figure 2. Valuation scale for farms in DQ.

Over three years of measures, results (Vandenberghe *et al.*, 2005) show that the monitoring must be:

- monthly conducted to take account of the intra annual nitrate residue evolution in the soil (Figure 1) from October to December and
- annually repeated to take account of the inter annual meteorological conditions which also influence the results (Figure 3).

Figure 3. Evolution of nitrate nitrogen concentration profiles through 3 years monitoring.

DQ farms need to show each year, in order to keep DQ status, that their nitrate profiles (sampled in five randomly chosen fields) are not widely greater than these standards. The DQ farms can be assessed according to the valuation scale (Figure 2) on the basis of the gap between the sample measured and the annual standard. Whenever, for a given culture the nitrate profile of a farm using the DQ exceeds widely the reference value, the DQ farmer must take measures to improve his fertilization practice as well as post harvesting management.
By the end of a 4-years period, an evaluation will be drawn. As a result of this evaluation, farmers are allowed or are no longer allowed to apply the 'Quality Approach'.
In 2004, about 250 farmers have followed the DQ. It was for them the first year of evaluation. Only 44% of them had a global (mean of the five randomly chosen fields) good score (a null or positive quote in Figure 2).

Conclusion

The network of pilot farms allows verifying that the nitrogen management within the D.Q. farms fit with a sustainable agriculture.
Due to annual meteorological conditions, the standards have varied through the 3 years of monitoring. This enforces annual measures of nitrate residue in the soil to establish the standards.
More than a half of the farms using the DQ need to be advised in their nitrogen management. This is the role of Nitrawal, a group of advisers created and financed by the Walloon region and trained by GRENeRA and ECOP.
In 2005, more than 500 farms are using the DQ. By the Nitrawal's advices, it is expected that these farms would rapidly (before 4 years) have a good score.

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Working group 8

N turnover and losses at the plot scale: are detailed studies still informative?

Report of Working Group 8

Nitrogen turnover and losses at the pot scale: are detailed studies still informative?

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Introduction

The first function of plot research is the explanation of the functioning of systems by research on the functioning of the underlying subsystems. When relationships between and within subsystems are explored and described in detail, the description becomes a mechanistic model. Developing mathematical solutions for the systems behaviour without knowing or exploring the subsystems relationships leads to empirical models. For nitrogen(N) budgets most often flows and losses can only be understood by knowledge of the subsystems which are usually the processes in the N cycle. The processes can only be measured and understood in small scale experiments with a large degree of control. But processes may also be treated as black boxes and then there is less need to unravel all detailed aspects of these processes to understand N budgets. There is still not much understanding of larger scale budgets on the basis of integration of small scale results. But do mechanistic models provide better predictions than empirical models?

Another function of plot research is the investigation of the effects of measures concerning agricultural practices on yields and environmental performance.

The Water Framework Directive applies largely to the watershed level whereas agriculture operates at the farm level. Hence, one might ask whether research on the plot level is still informative. Does it really contribute to a better understanding of the possibilities for improving the water quality by mitigation of agricultural practices?

Plot scales

The plot scale does not have dimensions defined by nature. Plot size can vary between a minimum of may be less than 1 cm^2 and a maximum of even 100 m^2, as long as the size is smaller than a complete field and large enough to represent a system, and not, for example, only an organism. Large plots are supposed to be representative for the field scale and used to study the effects of a single factor or a combination of factors on one or more phenomena. The phenomena investigated deal with agriculture or the environment, like for example the effects of fertiliser type and level on yield and nitrate leaching. Smaller or 'micro' plots are no longer representative for the field scale, mostly because of boundary effects or because of the plots have been taken out of their natural environment. Depending on the problem studied and the methods used, the plot size can still vary considerably.

As an illustration of plot research on different scales, three examples are given in the text box.

Three examples of plot research

1) Nitrate leaching from intensive horticultural systems (Thompson)
Example of small plot research. Nitrate leaching was measured from isolated plots, filled with growth medium or repacked soil and grown with several crops. The research objective was to investigate the contribution of different production systems to the regional problem of very high ground water nitrate contents.

2) Manipulating the N release from crop residues (Chaves)
Example of plots representative for the field scale. Objective was the investigation of the possibilities of decreasing the N mineralisation from crop residues in autumn by adding C rich organic matter and increasing the N mineralisation in spring by adding organic wastes. The plot size was large enough to represent field conditions and small enough to handle several treatments and replicates.

3) Nitrate leaching from urine patches (Corré)
Example of research on a problem with a natural small size appearance. Natural urine patches cover 0.5 to 0.7 m^2. Plot size was determined by the research methods: 10 m^2 in a field experiment to make representative sampling of the ground water possible and 3 dm^2 in a lysimeter experiment for reasons of the costs of ^{15}N labeled urea and the possibility of taking undisturbed soil columns.

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Advantages of plot scale research

'Possibility of comparison of treatments with replicates' and 'easier management and lower costs' came forward from the plenary discussion as the main advantages of plot research. These advantages apply particularly to larger plots, as well as the advantage 'combines field level with low heterogeneity'. More specific for smaller plots are the advantages 'possibility for understanding and quantifying processes' and 'providing detailed data for modelling processes on a larger scale'.

Disadvantages of plot scale research

As main disadvantages 'scaling problems', in both spatial and temporal scaling, and 'applicability' were recognised. A general applicability of the results can be hampered by the limited range of soil and weather conditions of experiments or by the unnatural conditions under which a small scale experiment is done. Another disadvantage can be the 'focus on one process while the other processes remain a black box'. This makes it hard to integrate the different processes into a complete system.
Furthermore it was pointed out that small scale plot research or process oriented research mostly needs years to provide information which might be used on a larger scale. Sometimes answers, important in policy implementation, need to be available much faster.

Discussion

Since the informative value of the larger scale 'field plots' was broadly accepted, the discussion focussed on small size 'micro plots'. Research using this small plot scale is mostly not possible on a larger scale. Restrictions to the plot size can be the necessity of an isolated system, e.g. the measurements of gaseous emissions, the costs, e.g. when stable isotopes are used, or an improved homogeneity, which makes better comparisons possible. Furthermore, studied objects can have a small size by nature, like urine patches, and a small size can be used as a pilot before investing much more effort in a larger scale field experiment.

Experiments on small plots have provided massive information on the different processes of the N cycle, but in many experiments still large amounts of N 'not accounted for' are found which cannot be explained. For denitrification, for example, the process and the qualitative effects of agricultural practises and environmental factors on the process rate are clear, but a quantitative estimate of the N loss by denitrification in a practical situation is still very hard to make, if possible at all. The same holds for immobilisation of N in the soil. In many studies a clear immobilisation of ^{15}N labelled N is measured. Still, it is not yet clear whether this represents a net accumulation of N or only a substitution of the mineralisation of predominantly ^{14}N, with no net change in soil organic N as a result.

With respect to scaling problems some researchers perform excellent on the plot scale, developing mechanistic models, and others perform very well on the larger scale, developing empirical models. Integrating mechanistic models to larger scales however, is a difficult task with many pitfalls and is therefore less practiced. Hence, a better cooperation between small and large scale researchers could very well improve the informative value of small scale experiments.

Concluding remarks

As a general conclusion from the working group discussion it can be distilled that whether an experiment with small plots can be informative or not firstly depends first of all on the subject studied and the methods used, and not on the plot size. In other words, the question to be answered determines the appropriate plot size that gives the best chances on informative research results on time. Whenever the use of a small plot size is unavoidable for technical reasons, a thorough evaluation of the consequences for the applicability of the results is necessary. In this respect more focus is needed on the suitability of the results for upscaling from plot to field and regional level, and less on their meaning at the plot scale itself.

Oral presentation

Nitrate concentration in soils and subsoils as affected by farming practices in intensive agricultural areas

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Introduction

The nitrate concentration in ground and surface waters is increasing in a large part of intensive agricultural areas in Europe. This is mainly due to the complexity of interactions between social, economic and biophysical processes (Letcher and Giupponi, 2005). Moreover, administrative or economic rules are established at regional or farm scale whereas biophysical processes are site specific. The relevant spatial scale to assess the efficiency of theses rules is the watershed, but the field scale is essential to understand the effect of cropping practices on N leaching. The rotation period is the minimal temporal scale to integrate the effects of agricultural practices on N leaching, because the global N cycle is affected by all practices interacting within the cropping system.
We examine the impact of various cropping systems and farming practices (mineral and organic fertilisation, tillage, grassland destruction, catch crops) on N leached and nitrate concentration on crop rotation scale. The emphasis is put on long term experiments (> 5 years) in order to take into account the cumulative effects of these practices.

Nitrate behaviour in soil as affected by single practices

The amount of nitrate leached is strongly linked to the water balance (rainfall - evapotranspiration), N inputs (inorganic and organic), N transformations in soil (mineralisation, immobilisation, absorption, symbiotic fixation, gaseous losses). The intensity of these processes varies widely with pedoclimatic conditions and cropping systems. The nitrate leached and the nitrate concentration in drained water are affected by fertilisation (mineral and organic), catch crop, straw disposal, soil type and rotation. Leaching can be due to over fertilisation in situations with a large surplus of nitrogen or difficulties in controlling N in shallow soils. However, in many cases, it does not originate directly from the fertiliser as shown by experiments conducted with ^{15}N labelled fertilisers in fields or lysimeters. The fertiliser enters into the soil humus pool which has a slow turnover time and mainly contributes to N pollution in the long-term. Reducing N fertilisation rate below the optimum rate leads to a moderate reduction in N leaching, depending on pedoclimatic conditions. The reduction varies from 14% at Thibie with a 35% N reduction (Mary *et al.*, 2002) to 27% at Jyndevad (Hansen and Djurhuus, 1996) and 33% in Lincolnshire (Johnson *et al.*, 2002) for a 50% fertilizer N reduction.
At a similar N rate organic waste application often results in a higher N leaching compared to using mineral fertilisers. A literature review shows that the increase of NO_3^- in drained water due to manure application may vary from 0 to more 27 mg $NO_3\,L^{-1}$ according to climate, land occupation, date of application and nature of the organic wastes (Simmelsgaard, 1998; Foissy and Blouet, 2003; Olesen *et al.*, 2003; Van Dijk *et al.*, 2003; Chambaut *et al.*, 2004; Bakhsh *et al.*, 2005).
Catch crops (CC) can have a very significant effect on N leaching. The observed reduction is 59% at Thibie (Mary *et al.*, 2002) and 47% at Jyndevad (Hansen and Djurhuus, 1996) when CC are grown every year. Olesen *et al.* (2004) found a 31% reduction in a rotation including a CC every two years. The reduction is 23% in farming conditions in France with a CC grown every 3 years (Beaudoin *et al.*, 2005).

Straw management may affect soil N mineralisation in the autumn and therefore N leaching. Net mineralisation from August to December is decreased by 20-24 kg N ha^{-1} when cereal straw is incorporated *vs.* removed (Mary *et al.*, 1996, 1999; Beaudoin *et al.*, 2005). These authors and others (Nicholson *et al.*, 1997) found that the annual reduction of leaching varied from 5 to 30 kg N ha^{-1}, according to the weather conditions in winter.
Grassland destruction is known to enhance soil mineralization which can be increased from 150 to 420 kg N ha^{-1} during the following year (Vertès *et al.*, 2002). Mineralisation is favoured in grazed (vs. cut) grasslands, in mixed stands (vs. pure gramineous) and in old pastures (Bommelé *et al.*, 2005). As expected, N leaching is higher when grassland is destroyed in autumn than in spring (Vertès *et al.*, 2002; Conijn, 2005).

Nitrate behaviour in soil as affected by cropping system

The mean concentration also depends on crop rotation (Hansen *et al.*, 2001; Hall *et al.*, 2001). The 'good agricultural practices' (GAP), recommended by the EU, combine several of these improved practices. They aim at reducing inorganic nitrogen content before the beginning of drainage while maintaining the farmer's income. GAP can abate the nitrate concentration by about 50% compared to conventional management (Hansen and Djurhuus, 1996; Mary *et al.*, 2002; Johnson *et al.*, 2002). Application of GAP by farmers during 8 years shows that the nitrate concentration of drained water below the rooting zone can respect the EU standard in deep loamy soils (mean concentration = 31 mg $NO_3^- L^{-1}$) but not for shallow or sandy soils (86 mg $NO_3^- L^{-1}$) in Northern France (Beaudoin *et al.*, 2005). Other studies show variable impacts of GAP in shallow or sandy soils: the concentration varies between 36 and 131 mg $NO_3^- L^{-1}$ according to the importance of water dilution, the N input level and the frequency of CC (Johnson *et al.*, 2002; Webster *et al.*, 2003; Olesen *et al.*, 2004).
The nitrate leached on the long term will depend on N immobilisation that will occur along this time course. The N organic storage can be partially related to the N balance (N inputs – N exported by crops). On the long term, the N balance represents the sum of leaching, gaseous losses (volatilisation, denitrification) and N immobilisation. In situations without a large N surplus, no correlation is found between leaching losses and N balance (Mary *et al.*, 2002; Van Dijk *et al.*, 2003). This shows that the gaseous losses and the immobilisation processes vary differently according to cropping practices. In situations with a high N surplus (with organic manure applications), N immobilisation in soil organic matter is the main sink for the excess N (Wolf *et al.*, 2005). Quantifying the effect of GAP on N losses towards the atmosphere (NH_3, N_2, N_2O) requires very short-term investigations. Conversely, the understanding of mineralisation/immobilisation processes needs long term studies since the turnover time of organic matter takes place over decades.

Nitrate fate and transfer within geological layers

Others key points of the response of hydrological systems to the GAP concern the time of transport and the fate of N leached within the vadose zone. The transport time can vary from a few months in the case of tile-drained soils to several decades for deep aquifers. The water transfer rate in the unsaturated zone of the chalk aquifers in Northern France has been evaluated using the natural tritium tracer between 0.4 and 0.7 m y^{-1}, depending on the water balance (Normand *et al.*, 2004). The response time of a 30 m deep aquifer is then greater than 45 years. This explains the long delay which may exist between the adoption of improved practices and the effect on the water quality in the aquifer: a 12 years time was found to observe a decrease in nitrate concentration after the initiation of GAP application at Bruyères catchment (unpublished data).
The fate of leached N may be affected by the hydrological regimes within the landscape. Three main cases can be distinguished along a topographic cross section: 1) the upland and the hillslope top where the drainage is vertical and the subsoil oxidised, except in the case of perched groundwater, 2) the hillslope bottom where the drainage has a lateral component and the subsoil is reduced 3) the wetland in which drainage is lateral and soil is alternatively or permanently under anaerobic conditions. The nitrate concentration observed in zone 1 is often higher than that found in zone 2 or 3. The downward transfer within zone 1 seems to be conservative: we compiled two studies indicating that the nitrate concentrations in drained water flowing out of the rooting zone are close to those found in the subsoil (between 2 and 10 m depth). In zone 2, Cambardella *et al.* (1999) found that the nitrate concentrations at the outlet of a tile drain (42 mg $NO_3^- L^{-1}$) were higher than those measured at 3-10 m depth (19 and 8 mg $NO_3^- L^{-1}$) and attributed the difference to denitrification. In zone 3, denitrification can also occur due to interactions between

soil and groundwater (Michelin, 2001; Oehler *et al.*, 2005). Finally, the water originating from agricultural soils can be diluted by non-polluted water coming from forest or grassland zones. Such complexity means that spatial and temporal integration is needed to predict the final water quality.

Conclusion

Farmers and decision makers ask for quantitative estimates of the impacts of GAP on water quality. We have a rather good knowledge on the effect of the main cropping practices onto nitrate leaching which is embedded in simulation models allowing to make scenario simulations. However there is still a lack of understanding of the effect of farming practices and pedoclimatic conditions on the whole N balance, including N storage and gaseous losses, particularly on the long term. Improving this knowledge is essential in order to optimize the whole N cycle and avoid substituting NH_3 or N_2O emissions for NO_3^- leaching.

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Poster presentations

Nitrogen runoff and leaching losses on beef production systems in Southern Chile

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Abstract

The Lakes Region of Chile produces 60% of the country's milk and 45% of its meat. Production is expected to increase in the near future, yet there are few data on the environmental impact of local livestock systems. A field experiment was carried out during 2004 to measure nitrogen (N) runoff and leaching losses in beef cattle-grazing production systems. Two immediate stocking rates were tested (63 and 191 steers ha^{-1} d^{-1}). Grazing was carried out with Holstein-Friesian steers (212 kg initial live weight), with a stocking rate of 3.5 steers ha^{-1}. Water samples were collected with surface lysimeters and ceramic cups, and were analyzed for available N and dissolved organic N (DON). Total losses were calculated as the product of N concentration and accumulated drainage at each sampling date. No significant difference in the total N loss was found between treatments (2 and 5 kg N ha^{-1}, respectively; $P>0.05$). The main pathway of N loss was leaching, accounting for 99% of the total N loss. Ammonium and DON were the main forms of N loss in runoff (45% each). Nitrate was the main form of N loss by leaching (99%). Low N losses could be related to low N inputs in fertilizers and to physical and chemical properties of volcanic soils.

Keywords: ammonium, andisol, DON, livestock production, nitrate

Background and objective

The Lake Region of southern Chile has suitable climatic conditions and soil types for cattle production. Consequently, 56% of the national cattle herd is concentrated in this maritime temperate climatic region, grazing on natural and improved pastures. It is expected that the production of this sector will increase because of the commercial trade agreements between Chile and the European Union, the United States of America and South Korea.

In developed countries the environmental impact of livestock systems has been widely studied, because of the important role of this activity on water, soil and air pollution (Jarvis, 1993). The amount of nitrate leached below a grass sward grazed by cattle has been reported to be 5.6 times greater than that leached below a comparable cut sward (Ryden *et al.,* 1984), because ruminants are inefficient in converting ingested N into milk or protein and live weight gain. The excess N is excreted in dung and urine and is returned directly to the pasture during grazing (Jarvis, 1993).

Up to now Chilean beef and dairy production has been focussed in increasing yields, so that there is little information about the contribution of N to water sources resulting from this activity. Some lysimeter studies have been carried out under field conditions (Salazar, 2002) to evaluate N leaching losses after pasture cutting, but this does not mimic exactly the conditions created by grazing.

We quantified N losses in runoff and leaching in beef production systems affected by the immediate stocking rate.

Materials and methods

The experiment was carried out at the National Institute for Agricultural Research (INIA), Remehue Research Centre (40º35'S, 73º12'W). The drainage season lasted between 12th April and 31st October 2004. The soil at the site is an Andisol of the Osorno soil series, which has 6% slope, more than 1 m depth, 17% organic matter and high available phosphorus content (Olsen P). The average rainfall for the area is 1260 mm yr^{-1} (30 years average).
In this study two immediate stocking rates (nº animals/area unit/day) were tested (63 and 191 steers ha^{-1} d^{-1}). Grazing was carried out with Holstein-Friesian steers (initial live weight of 212 ± 9.6 kg), which were managed under rotational grazing on a permanent pasture (2 ha per farmlet). Grazing strips were 1254 m^2 d^{-1} and 418 m^2 d^{-1} per treatment, respectively, and the stocking rate used in both treatments was 3.5 steers ha^{-1}. Treatments were fertilized in autumn 2004 (27th April) with 45 kg N ha^{-1} (urea, 46% N) and in the spring of 2004 (10th of September), with 25 kg N ha^{-1} (sodium nitrate, 16% N) and 29 kg P ha^{-1} (triple superphosfate, 46% P_2O_5).
To quantify N losses in surface runoff, three surface lysimeters (5 x 5m) were established in each treatment, according to the methodology described by Scholefield and Stone (1995), and surface runoff was collected three times per week. The accumulated surface runoff was measured at each sampling date with the use of a graduated collector. Leaching losses at 60 cm depth were estimated through the use of ceramic cups (three cups per lysimeter, n=9 per treatment). Samples were collected fortnightly and N losses for the period were calculated according to Lord and Shepherd (1993). All individual collections were chemically analyzed and no filtering was needed. Runoff and leaching samples were frozen until analysis for available N ($N\text{-}NO_3^-$ and $N\text{-}NH_4^+$). Nitrate was measured using the salicylic acid method, ammonium was determined through the indophenol methodology. Runoff samples were also analysed for total N by the methodology Nº 10071 of the test'N Tube (®Hatch). Dissolved organic N concentrations were calculated as the difference between total N and available N in the samples. Total N losses for the period were calculated as the sum of N losses in runoff and N losses by leaching. Analysis of variance (ANOVA) was used to compare N concentrations and losses between treatments.

Results and discussion

Total rainfall for the experimental period was 868 mm and total drainage (surface runoff plus leaching) during the same period was 634 mm in both treatments. Of total drainage, 99% was due to water moving down the soil profile. No difference in the contribution of surface runoff to the total drainage was found between treatments (P>0.05; Table 1).
Total N losses obtained were low compared with those reported by Scholefield *et al.* (1993) for beef cattle production systems in south West England (38 kg N ha^{-1} yr^{-1}). This could be explained by the low N inputs in fertilizer in the present study (70 kg N) in relation to the previous study (200 kg N ha^{-1} yr^{-1}). No significant differences were found between treatments (P>0.05; Table 1). This can be related to the high organic matter content of the soil (17%), so that the soil biomass activity and the natural soil N mineralization processes could be the key factors controlling N losses, regardless the animal regime imposed. Water pollution observed in livestock systems of developed countries is, nevertheless, an attention call for the intensification of current Chilean beef cattle production systems.
The main pathway for inorganic N losses was leaching, which represented 99% of the total N lost. The proportion between ammonia and nitrate in the losses varied according to depth. At the ground level, ammonia losses represented on average 43 ± 3% of the total N lost in runoff, while at 60 cm, ammonia losses were on average 6 ± 1% of inorganic N lost by leaching. Total DON losses in surface runoff were equivalent to those measured as nitrate. Nitrate concentration in water samples collected at 60 cm depth was below the ECC directive for drinking water (*i.e.* European Union, 91/676/EEC). Nitrate concentration in the surface runoff samples was higher than the ECC limit and ammonium concentration in these samples was over the Chilean Directive for quality of surface continental waters (max. 1 mg L^{-1}). This situation represents a risk for incidental N surface water pollution of water bodies located closed by grazing areas in Southern Chile.

Table 1. Drainage (% of total drainage), average N concentrations in surface runoff and leaching samples (mg L^{-1}) and N losses (kg N ha^{-1}) in treatments grazed by 63 and 191 steers ha^{-1} d^{-1}, in southern Chile (± standard error of the mean).

Variables	Immediate stocking rate 63 steers ha^{-1} d^{-1}	191 steers ha^{-1} d^{-1}
Total drainage collected (mm)	633.8	634.0
Surface runoff	1% a	1% a
Leaching (> 60 cm)	99% a	99% a
Mean surface runoff concentrations and range (mg L^{-1})		
$N\text{-}NH_4^+$	27±11.3 a (1–76)	30±13.9 a (0.3–89)
$N\text{-}NO_3^-$	26±13.6 b (1–66)	61±23.9 a (1–98)
DON	14±8.2 a (0.3–286)	16±5.4 a (0.3–260)
Mean leachate concentrations and range (mg L^{-1})		
$N\text{-}NH_4^+$	0.03±0.006 a (0.02–0.05)	0.04±0.006 a (0.02–0.04)
$N\text{-}NO_3^-$	2±0.6 a (0.7–2.7)	5±2.0 a (2.2–9.1)
N losses (kg ha^{-1})		
$N\text{-}NH_4^+$ in surface runoff	0.01±0.001 a	0.03±0.003 a
$N\text{-}NO_3^-$ in surface runoff	0.003±0.0016 a	0.004±0.0022 a
DON in surface runoff	0.01±0.0008 a	0.03±0.002 a
$N\text{-}NH_4^+$ in leaching	0.2±0.04 a	0.2±0.04 a
$N\text{-}NO_3^-$ in leaching	2±0.6 a	5±2.0 a
Total N losses	2.2±0.62 a	5.3±2.02a

Different letters indicate significant differences between treatments ($P \leq 0.05$).

Conclusions

Nitrogen losses were less than 5.5 kg N ha^{-1} yr^{-1} and did not differ between treatments. Results suggest that the high organic matter content of the soil had an important effect in controlling N losses under grazing. A second drainage season will provide a better understanding of the processes involved in N losses under Chilean conditions.

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Nitrogen runoff and leaching losses on beef production systems as affected by the field slope

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Abstract

The Lakes Region of Chile produces 45% of the country's meat. Production is based on grazing, yet there are few data on the environmental impact of this activity. A field experiment was carried out during 2004 to measure nitrogen (N) runoff and leaching losses in beef cattle-grazing systems with different field slopes (3 and 12%). Pasture was grazed with Holstein-Friesian steers (212 kg initial live weight), with a stocking rate of 3.5 steers ha^{-1}. Water samples were collected with surface lysimeters and ceramic cups, and were analyzed for available N and dissolved organic N (DON). Total losses were calculated as the product of N concentration and accumulated drainage for each sampling date. No significant difference in the total N loss was found between treatments (2 and 1 kg N ha^{-1}, respectively; $P>0.05$). The main pathway of N loss was leaching, which contributed with 99% and 84% of the loss in each treatment, respectively. Ammonium and DON were the main forms of N loss in runoff in the 3% slope treatment (45% each). With 12% slope, DON was 79% of the total N lost in runoff. Nitrate was the main form of N loss by leaching (87%). Nitrogen losses quantified were low, which could be related to low N inputs in fertilizers and to physical and chemical properties of volcanic soils.

Keywords: ammonium, andisol, DON, livestock production, nitrate

Background and objective

The Lake Region of southern Chile has suitable climatic conditions and soil types for cattle production. It is expected that the production of this sector will increase because of the commercial trade agreements between Chile and the European Union and South Korea.

In developed countries the environmental impact of livestock systems has been widely studied, because of the important role of this activity on water, soil and air pollution (Jarvis, 1993). It has been estimated that agriculture contributes 37% to 82% of the nitrogen (N) input into surface waters of Western Europe (Isermann, 1990).

Despite the importance of livestock production in southern Chile and that most of this production is generated in sloppy soils (>5%), there is little information about the contribution of N to water sources from beef production systems in the region and there are no studies on the effect of the slope of the soil on N losses, under grazing conditions.

The objective of this experiment was to quantify N losses in runoff and leaching in beef production systems as affected by the field slope.

Materials and methods

The experiment was carried out at the National Institute for Agricultural Research (INIA), Remehue Research Centre (40º35'S, 73º12'W). The drainage season lasted between 12th April and 31st October 2004. The soil at the site is an Andisol of the Osorno soil series, *c.* 1 m depth and 17% organic matter. The average rainfall for the area is 1260 mm yr^{-1} (30 years average).

Grazing was carried out with Holstein-Friesian steers (initial live weight of 212 ± 9.6 kg), which were managed under rotational grazing on a permanent pasture (2 ha per farmlet), with a stocking rate of 3.5 steers ha^{-1} in two different soil slopes, 3 and 12%. Treatments were managed as indicated by Alfaro *et al.* (2005).

To quantify N losses in surface runoff, three surface lysimeters (5 x 5m) were established in each treatment, according to Scholefield & Stone (1995), and surface runoff was collected three times per week. The accumulated surface runoff was measured at each sampling date with the use of a graduated collector. Leaching losses at 60 cm depth were estimated using ceramic cups (three cups per surface lysimeter, n=9 per treatment). Samples were collected fortnightly and N losses for the period was calculated according to Lord and Shepherd (1993). All individual collections were chemically analyzed and no filtering was needed. Runoff and leaching samples were frozen until analysis for available N ($N\text{-}NO_3^-$ and $N\text{-}NH_4^+$). Runoff samples were also analysed for total N (methodology Nº 10071 test'N tube by ®Hatch). Dissolved organic N was estimated following the methodology indicated by Alfaro *et al.* (2005). Nitrogen losses were calculated as the product of drainage and N concentration in the respective samples. Total N losses for the period were calculated as the sum of N losses in runoff and N losses by leaching. Analysis of variance (ANOVA) was used to compare N concentrations and losses between treatments.

Results and discussion

Total rainfall for the experimental period was 868 mm and total drainage (surface runoff plus leaching) during the same period was 634 mm, in both treatments.

Total N losses obtained were low compared with those reported by Scholefield *et al.* (1993) for beef cattle production systems in south West England (38 kg N ha^{-1} yr^{-1}) (Table 1). This could be explained by the low N inputs in fertilizer in the present study (70 kg N) in relation to Scholefield's study (200 kg N ha^{-1} yr^{-1}). No significant differences were found between treatments ($P>0.05$; Table 1), which could have been related to the high organic matter content of the soil, so that the soil biomass activity and the natural soil N mineralization processes could be the key factors controlling N losses, acting as a buffer.

The main pathway for inorganic N losses was leaching, but while in the 3% slope treatment this represented 99% of the total N lost, in the 12% slope treatment this represented only 84% of the total N lost.

Total losses in runoff were nine times greater in the 12 than in the 3% slope treatment, associated to a greater nitrate and DON concentrations in these samples ($P\leq0.05$; Table 1). In the 3% slope treatment ammonia and DON losses represented on average 43 ± 3% each of the total N lost in runoff. In the 12% slope treatment, DON represented 79% of the total N lost in this pathway ($P\leq0.05$); this difference was probably related to the greater molecular weight of DON, so that with greater slope these molecules are carried easier and faster by water running off down the slope, than those of lower molecular weight. At 60 cm, ammonia losses were 14 ± 4% of inorganic N lost by leaching, on average of both treatments, being twice the value in the 12% slope treatment. This can be related to the lower ammonia loss in runoff.

Nitrate concentration in leachate samples was below the ECC directive for drinking water (*i.e.* European Union, 91/676/EEC). Nitrate concentration in surface runoff samples was higher than the EU limit and ammonium concentration in these samples was over the Chilean Directive for quality of surface continental waters (max. 1 mg L^{-1}). This situation represents a risk for incidental N surface water pollution of water bodies located closed by grazing areas at times of heavy rainfall or after N fertiliser applications.

Conclusions

Leaching was the main pathway for N losses in beef cattle production systems of southern Chile. Nevertheless, and increase in four times the slope gradient resulted in nine times greater total N losses in runoff. Runoff samples collected in the 12% slope treatment had twice the nitrate concentration and almost four times the DON concentration of runoff samples collected in the 3% slope treatment. Results suggest that soil chemical properties had an important effect in controlling N losses under grazing, independently of the site slope. A second drainage season will provide a better understanding of the processes involved in N losses under Chilean conditions.

Table 1. Drainage (% of total drainage), average N concentrations in surface runoff and leaching samples (mg L^{-1}) and N losses (kg N ha^{-1}) in treatments with 3 and 12% slope, in southern Chile (± standard error of the mean).

Variables	3% slope	12% slope
Total drainage collected (mm)	633.8	633.8
Surface runoff	1% a	1% a
Leaching (> 60 cm)	99% a	99% a
Mean surface runoff concentrations and range (mg L^{-1})		
$N\text{-}NH_4^+$	27±11.3 a	43±15.7 a
	(1–76)	(6–95)
$N\text{-}NO_3^-$	26±13.6 b	52±19.9 a
	(1–66)	(2–121)
DON	14±8.2 b	52±27.2 a
	(0.3–286)	(0.1–526)
Mean leachate concentrations and range (mg L^{-1})		
$N\text{-}NH_4^+$	0.03±0.006 a	0.04±0.001 a
	(0.02–0.05)	(0.03–0.04)
$N\text{-}NO_3^-$	2±0.6 a	1±0.1 a
	(0.7–2.7)	(0.7–0.8)
N losses (kg ha^{-1})		
$N\text{-}NH_4^+$ in surface runoff	0.01±0.001 b	0.04±0.005 a
$N\text{-}NO_3^-$ in surface runoff	0.003±0.0016 b	0.006±0.0047 a
DON in surface runoff	0.01±0.0008 b	0.17±0.02 a
$N\text{-}NH_4^+$ in leaching	0.2±0.04 a	0.2±0.01 a
$N\text{-}NO_3^-$ in leaching	2±0.6 a	0.9±0.14 a
Total N losses	2.2±0.62 a	1.3±0.14 a

Different letters indicate significant differences between treatments ($P \leq 0.05$).

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Nitrate leaching from grazed grassland lisimeters: effects of fertilizer input, field drainage, age of sward and patterns of weather. *Journal of Soil Science* 44, 601-613.

Drainage and nitrate leaching in winter cereals under humid Mediterranean conditions

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Abstract

Quantification of nitrate leaching from cropping systems is necessary to optimize N-fertilizer application and determine the impact on groundwater quality. The objectives of this work were i) to quantify drainage and nitrate leaching in a winter cereal cropping system under humid Mediterranean conditions, and ii) to analyze the effect of N-fertilizer rate on nitrate leaching. A factorial (5 N-fertilizer levels) experiment with winter cereals was conducted during two cropping seasons in Navarra (Spain). The results indicate that there were three phases: i) from planting in the end of October or the beginning of November to the end of January, with a high risk of drainage and nitrate leaching, ii) from February to mid spring, with low drainage and leaching, and iii) from mid spring to harvest in July, when no drainage or nitrate leaching took place. Nitrate leaching was predominantly related to soil nitrate at sowing and to rainfall from planting to mid spring. In the second season, differences in nitrate leaching were only observed in the overfertilized treatment of the previous year.

Keywords: barley, drain water, leaching, nitrate, wheat

Background and objectives

Aquifer contamination from nitrate leaching receives special attention in European Union legislation (EC, 2000). Quantification of nitrate leaching from cropping systems is necessary to optimize N-fertilizer application so as to minimize the impact on groundwater quality. Since drainage and nitrate leaching in agricultural soils are difficult to measure, indirect methods are commonly used for their estimation (Webster *et al.*, 1993). A detailed knowledge of soil water dynamics in cropping systems is essential to improve estimations of drainage and nitrate leaching. The objectives of this study are i) to quantify drainage and nitrate leaching in a winter cereal cropping system under humid Mediterranean conditions, and ii) to analyze the effect of N-fertilizer rate on nitrate leaching.

Materials and Methods

A factorial experiment (5 N rates) with winter cereals was conducted during two cropping seasons (2002-2003 and 2003-2004) at the Experimental Field Station of ETSIA (Pamplona, Spain). The soil is classified as Calcixerollic Xerochrept, and has a silt-clay loam texture in the upper 0.6 m and a silt loam in the 0.6-1.0 m layer. The upper 0.3 m has a pH (1 g soil: 2 mL H_2O) of 7.8, and contains 14.5 g C kg^{-1} and 1.6 g N kg^{-1}.
The experiment was sown with winter wheat (*Triticum aestivum* L.; cv. Soissons) on 28 October 2002, and with barley (*Hordeum vulgare* L., cv. Hispanic) on 6 November 2003. There were four replicate plots (5x8 m^2) for each of five N fertilizer rates in a completely randomised block design. Nitrogen rates ranged from 0 to non-limiting and were applied as ammonium nitrate-sulfate. Crop N requirement was estimated as the product of the expected yield (6 t ha^{-1}) and an N extraction coefficient (wheat = 30 kg N t^{-1}; barley = 26 kg N t^{-1}). Fertilizer was manually broadcasted to the plots at the beginning of tillering (GS-21; Zadoks *et al.*, 1974) and at the beginning of stem elongation (GS-30). In July, a 1.5 m wide central strip was harvested from each plot for the determination of yield

and biomass. Crop N uptake was calculated as the product of biomass and N content in above–ground plant material.
Soil mineral-N content was determined in soil samples taken before sowing, before fertilizer application, and after harvest at 0.3 m intervals to a depth of 0.9 m. Nitrate and ammonium concentration were determined following extraction with 1 M KCl.
Drainage at 1 m depth (D) was determined by applying a simplified one-dimensional (vertical) water balance equation (Allen *et al.*, 1998) at each position of measurement:

$$D = P - ETc \pm VR$$

where P is precipitation (mm), ETc is crop evapotranspiration (mm) and VR is variation of soil water reserve. The ETc was calculated daily by multiplying the crop coefficient by the reference evapotranspiration calculated by the FAO Penman-Monteith method (Allen *et al.*, 1998). The VR was based on daily soil volumetric water content (θv) measurements in eight plots. To monitor θv, EnviroScan® sensors (Sentek Pty. Ltd., Australia) based on frequency domain reflectometry, were installed in the soil at 0.20 m-depth intervals to 1 m depth. To determine the end of the drainage period we adapted the concept of fraction of field capacity (FCC), where FCC is the soil water content of a layer as related to its water content at field capacity (Williams, 1991). The end of the drainage period was the day in which FCC of the deepest layer was equal to 1, and the FCC of the previous layer was less than 1. Drainage was calculated weekly and summed to obtain cumulative drainage.
Nitrate leached to 1 m depth was calculated by multiplying drainage volume by nitrate concentration of the soil solution at that depth. Nitrate concentration was measured in soil solution samples collected weekly in two ceramics cups installed at 1-m soil depth in each plot.

Results

In the cropping season 2002-2003 drainage commenced on 7 November and continued until 1 April. The total amount of rainfall during this period was 492 mm (Figure 1). In the cropping season 2003-2004 drainage began on 10 November and continued until 22 April. The total amount of rainfall was 347 mm. Cumulative drainage was 266 mm in 2002-2003, and 144 mm in 2003-2004. In both cropping seasons ETc followed a similar temporal distribution; cumulative values were 537 mm in 2002-2003, and 494 mm in 2003-2004. The differences in drainage between seasons are mostly attributable to the higher rainfall during the first season.
Before sowing in the 2002-2003 growing season, the soil mineral-N content (0-0.6 m) was 252 kg N ha^{-1}, decreasing to 146 kg N ha^{-1} just before fertilizer application, and to values of 76 to 96 kg N ha^{-1}, depending on fertilizer rate, at harvest. In the 2003-2004 season, the soil mineral-N content depended on fertilizer treatment. Before sowing, values were 116 to 170 kg N ha^{-1}, before fertilizer application they were 62 to 79 kg N ha^{-1}, and at harvest they were 54 to 77 kg N ha^{-1}. Crop N uptake increased with fertilizer application, ranging from 104 to 168 kg N ha^{-1} in 2003-2004, and from 45 to 49 kg N ha^{-1} in 2003-2004.
Nitrate leaching was much larger during the 2002-2003 than during the 2003-2004 cropping season. The total nitrate-leaching loss was predominantly determined by cumulative drainage and the soil mineral-N content (Figure 1). In 2002-2003, there were no statistically significant differences in nitrate leaching among the treatments. In 2003-2004 there were statistically significant differences between the highest N rate and all of other the N rates (Figure 1). In the second season, differences in nitrate leaching losses were observed before fertilizer application; therefore, the excessive N fertilization of the previous year was a contributing factor.

Figure 1. Cumulative drainage for all treatments and nitrate leaching in various fertilizer treatments throughout the growing seasons 2002-2003 and 2003-2004. X was obtained by correcting crop N requirement by soil mineral-N content at presidedress. A = 40 kg N ha^{-1}.

Conclusions

Our results indicate that there were three different crop phases related to nitrate leaching:

I) From sowing to the end of January: high rainfall and low ETc led to considerable drainage. Crop N demand was very low and most of the mineral N present in the soil before sowing was lost by nitrate leaching.

II) From February to mid spring: crop growth, ETc and N uptake were appreciable and consequently drainage and nitrate leaching were very low.

III) From mid spring to harvest: no drainage or leaching took place.

Nitrate leaching, during phase I, was the most important N loss process from this crop-soil system. The loss, during this period, represented 32 % of soil mineral-N at sowing. During phase II, nitrate leaching had very low values, and no significant differences were observed between treatments. There where no differences among treatments in the soil mineral-N content after harvest, suggesting appreciable gaseous N losses at the higher N-fertilizer rates.

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Relationship between respiration and dehydrogenase activity in a soil amended with organic wastes

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Abstract

Aerobic incubations were conducted using the upper layer of a dystric Cambisol amended with an organic material very rich in nitrogen - meat/bone meal - mixed with 5 other organic materials (straw, sawdust, elder tree berry, grape stalks and olive oil mill waste), to obtain different C/N ratios. The objective of the study was to compare CO_2-C release and dehydrogenase activity, both considered as good potential biochemical indicators of soil microbial activity. Evolved CO_2 from different mixtures was trapped in 1M NaOH and determined titrimetrically and periodically during the incubation. Soil samples were taken and dehydrogenase activity was measured according to TTC method at day 7, 49 and 84. The addition to soil of different organic materials led to a significant increase of both soil respiration and dehydrogenase activity, the maximum values being achieved during the first 7 days of the incubation. At day 7 dehydrogenase activity was well related with soil respiration with a determination coefficient (R^2) 0.71. This correlation fell along the incubation period with an 'R^2' value of 0.22 at day 84. Both parameters decreased along the incubation, although proportionally soil respiration revealed a higher decrease. Results obtained did not allow drawing any Conclusions concerning which parameter shows a greater reliability to evaluate microbial activity.

Keywords: dehydrogenase activity, microbial activity, organic residues, soil respiration

Background and objectives

Biological oxidation of organic compounds is generally a dehydrogenation process. As a matter of fact, the dehydrogenase enzyme systems are considered to play an essential role in the initial stages of the oxidation of soil organic matter as they transfer hydrogen and electrons from substrates to acceptors. Many different specific dehydrogenase systems are involved in the dehydrogenase of soils being these systems integral part of the microorganisms. Therefore, the result of the assay of dehydrogenase activity would show the average activity of the active microbial population (Skujins, 1967). Consequently, dehydrogenase activity can be considered as a good index of oxidative activities in the soil as well as a general indicator of soil microbial activity. This is supported by several authors (Casida, 1977, Tabatabai, 1982, Trevors, 1984, Camiña *et al*, 1998), who found a good correlation between dehydrogenase activity and microbial respiration when exogenous carbon sources are added to soils. In fact, microbial respiration is stimulated by the addition of easily biodegradable organic substrates to soils which act as an energy source to microorganisms. However, because the dehydrogenase activity depends on the total metabolic activity of soil microorganisms, its value in different soils containing different populations does not always reflect the total numbers of viable microorganisms isolated on a particular medium (Skujins, 1967).
The main objective of this study is to compare CO_2-C release and dehydrogenase activity, both considered as good potential biochemical indicators of soil microbial activity, in a soil amended with an organic material very rich in nitrogen - meat/bone meal - mixed with several organic wastes in order to obtain different C/N ratios, since C/N ratio is usually pointed out as one of the most important characteristics regulating N availability from organic residues.

Materials and methods

Aerobic incubations were carried out using the upper layer of a dystric Cambisol which was air dried at room temperature for several days, and sieved to pass through a 2 mm mesh. The soil was amended with meat/bone meal in a dose equivalent to 75 mg N kg^{-1} soil, alone and in combination with 3 different doses of straw and sawdust in order to obtain C/N ratio of 10, 20 and 30, and with 3 doses of elder tree berry, grape stalks and olive oil mill waste to obtain C/N ratio of 7.5, 10 and 12.5. Soil and residue mixtures were wetted with distilled water enough to obtain 60% water holding capacity (WHC) and maintained at 25 ºC in an incubation chamber. Treatments were replicated three times. A control treatment - soil alone - was also carried out. All the organic wastes were previously freeze-dried and ground to pass a 2 mm sieve. The mixtures added to the moist soil (50g oven dry equivalent) were placed in 100 mL vessels in a 1 L air tight glass jar. The glass jars were opened weekly in order to ensure aerobic conditions. Evolved CO_2 was trapped in 1M NaOH and determined titrimetrically (Zibilske, 1994) periodically during the incubation period (182 days). Soil samples were taken and dehydrogenase activity was determined at day 7, 49 and 84 according to TTC method which involves colorimetric determination of 2,3,5 - triphenyl formazan (TPF) produced by the reduction of 2,3,5 – triphenyltetrazolium choride (TTC) by soil microorganisms (Öhlinger, 1995). Both parameters, soil respiration and dehydrogenase activity, are expressed per gram of carbon added trough organic residues.

Results and discussion

As expected the addition of the different organic materials to soil led to a significant increase of both soil respiration and dehydrogenase activity.

During the incubation period, the maximum respiration activity for every mixture was achieved during the first 7 days of incubation due to the presence of high amounts of easily degradable organic carbon which stimulates soil microbial population activity. The lowest value was observed for the treatment with a C/N ratio = 30, corresponding to soil amended with the mixture meat/bone meal + sawdust – 6.41 mg CO_2-C g^{-1} C_{added} day^{-1} This value is due to the small amount of N that becomes available through mineralisation and to the presence of a higher proportion of recalcitrant organic forms. The highest value was obtained for the treatment corresponding to soil amended with meat/bone meal (34.34 mg CO_2-C g^{-1} C_{added} day^{-1}). After the initial high respiration rate a gradual decrease was observed for all treatments, becoming fairly constant after day 63. At day 84, evolved CO_2-C ranged from 1.27 to 6.60 mg CO_2-C g^{-1} C $_{added}$ day^{-1}.

Dehydrogenase activity was higher at day 7, varying in a range from 1.53 (soil amended with the mixture meat/bone meal + sawdust with a C/N ratio = 30) to 7.72 mg TPF g^{-1} C 16 h^{-1} in the treatment correspondent to soil amended with meat/bone meal.

At day 7 dehydrogenase activity was well related with soil respiration with a determination coefficient (R^2) 0.71 (Figure 1). However, this correlation fell along the incubation period with an 'R^2' value of 0.22 at day 84 (Figure 1).

Figure 1. Soil respiration vs. dehydrogenase activity.

Conclusions

The addition to soil of the different organic materials assayed led to a significant increase of soil respiration and soil dehydrogenase activity, the maximum values being achieved during the first 7 days of incubation.
Both parameters soil respiration and soil dehydrogenase activity, decreased along the incubation period although proportionally soil respiration revealed a higher decrease. However, results obtained did not allow drawing any Conclusions concerning which parameter shows a greater reliability to evaluate microbial population activity.

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Mineralization of oat residues, and microbial abundance and activity in sandy soil affected by tillage

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Abstract

Mineral N derived from surface residues is more easily immobilized and thus less prone to leaching than when residues are incorporated into the soil. Essays under controlled conditions may indicate the potential net N mineralization in the field. Soil microbial biomass responds more rapidly than soil organic matter to changes in management. This work was carried out to study the effects of oat residues and soil practices (till (CT), no till (NT)) and depth (0-7.5, 7.5-15 cm) on N mineralization and microbial abundance and activity under controlled conditions. Randomized block designs with three replicates were laid out in an oat/oat rotation in Portugal. Aerobic incubations at 18/19 ºC and 50% soil water content were run for 5, 21, 42 and 84 days, using undisturbed soil cores. Immobilized N was more pronounced under NT than CT. No differences for immobilized N were found with depth. Mineralizable N depended on the incubation period, with the greatest values at 21 and 42 days and was related to dehydrogenase activity and tillage. This activity was greater at the surface layer and after 21 days. In contrast, bacterial number was lower at this period. No effects were detected for fungi number.

Keywords: bacteria, fungi, incubation, soil depth, undisturbed soil cores

Background and objectives

Depending on tillage practice, residues can be removed from the soil, incorporated by cultivation, or left on the soil surface. In the long-term, residue conservation in the soils will increase the quantity of organic matter, with potentially beneficial effects such as soil protection against erosion, and increases in water retention and plant nutrients. Tillage increases the breakdown of residues and microbial cells, promoting the release and subsequent degradation of previously protected organic matter, raising the level of plant available nutrients. Besides the tillage system, tillage depth also play a decisive role on soil microorganisms, namely fungi. Straw residues can have a detrimental effect on the growth of the subsequent crop due to their relatively high C/N ratio (60-80) promoting immobilization of soil mineral N as most microbial populations are unable to satisfy their N demand from such carbonaceous substrates (Fox *et al.*, 1990). However, because of this mechanism, straw residues can also be advantageous to reduce susceptibility to nitrate leaching from the soil.

Soil enzymes are predominantly of microbial origin and are closely related to microbial abundance and activity. Numerous enzymes have been tested on their suitability for soil research, although a major weakness of enzyme tests is that actual microbial activity of a soil is not well reflected (Insam, 2001). Little is known about the short-term effects of tillage on mineralization of N and microbial activity.

The present work had two main objectives: *i)* to evaluate the effects of soil management practices (conventional till (CT), no till (NT)) and depth (0-7.5, 7.5-15 cm) on residues decomposition and soil biomass, under controlled conditions, using undisturbed soil cores; *ii)* to measure the effects of different incubation periods and oat residues on soil N net mineralization/immobilization and nitrification, and on microbial activity and abundance.

Material and methods

The study was conducted in two sequential seasons (2002/2003 and 2003/2004) with *Avena sativa* L. (oat after oat), on a Haplic Podzol (Portugal), under two tillage systems: NT and CT repeated three times in randomized designs. The soil was sandy, with a low cation exchange capacity (0.88 $cmol_{(+)}$ kg^{-1}), 7 g organic C kg^{-1} and 0.6 g Kjeldahl N kg^{-1}.

About 9 t $ha^{-1}$of above- and below-ground oat biomass were harvested at crop maturity respectively both CT and NT plots and returned immediately to areas very close to their origin. Several undisturbed soil cores containing oat residues were taken on day zero (t_0) (immediately after rotavation in CT plots) from each plot using PVC cylinders. Tubes were incubated at 18/19 ºC for 5, 21, 42 and 84 d. Soil water content in each soil core was adjusted to 50% of the water field capacity. After each incubation period, mineral N, and organic C and N contents were determined from each soil core divided into 0-7.5 and 7.5-15 cm sections.

Results and discussion

In Table 1, some characteristics of oat crop (*Avena sativa* L.) at maturity are shown. Oat straw and roots presented values of C/N>20 but lower than the reference (Fox *et al.*, 1990) values (60-80). Lignin/N ratios in the oat organs were greater than a critical level (10.4) reported by Fox *et al.* (1990) for barley straw. Straw and roots also presented low proteolytic activities (13.2 U and 1.2 U, respectively). These results seem to indicate the potential of these residues for an initial slow net mineralization rate, or even for N immobilization. Principal Component Analysis showed a strong inverse relationship between dehydrogenase activity and C in the soil (Figure 1). The enzymatic activity was positively related to fungi number which in turn was strongly negatively related to soil N concentration. Both bacteria and fungi abundances were strongly related to mineral N in the soil at the end of incubations. Bacteria were still related to mineralized and nitrified N. An equation was fitted for the enzymatic activity (Y) as follows:

$$Y=3.080-0.116t_i + 0.140orgC_{(ti)} - 0.139D, (R^2=0.48, p<0.001)$$

where, $t_i$represents the incubation periods, $orgC_{(ti)}$is the amount of C at t_i, and D is the soil depth (0=0-7.5, 1=7.5-15 cm).

Bacteria number differed significantly in each incubation period (Figure 2). Greater values were measured for 42 and 84 d. An opposite pattern was observed for dehydrogenase activity. Microbial biomass decreased significantly with depth, from 3.9 to 3.6 CFU g^{-1} dry soil.

The incubation period also significantly affected the potential net N mineralization (Figure 2) with greater values for 21 and 42 d. N immobilization occurred at 5 d (-3.40 mg N kg^{-1} dry soil) and 84 d (-1.30 mg N kg^{-1} dry soil). The best fit equation for net N mineralization (*Nmin*) showed that N assimilation by microbes reduced available soil N, especially in NT plots:

$$Nmin=13.175-3.861*x-1.896T, (R^2=0.29, p<0.01)$$

where x represents (log (dehydrogenase activity+10)) and T refers to tillage (0=NT, 1=CT).

Table 1. Some chemical characteristics of oat (Avena sativa L.) residues, at maturity.

	C (g kg^{-1} DM)	N (g kg^{-1} DM)	C/N	Lignin (g kg^{-1} DM) Soluble	Insoluble	Total	Lignin/N	Proteol. activ.[a] (U)
Straw	477	8.3	57.5	37	177	213	25.7	13.2
Roots	214	6.2	34.5	24	235	265	42.7	1.2

[a] = *Proteolytic activity.*

Figure 1. Principal Component Analysis for the significant variables measured in the soil cores with oat residues.

Figure 2. Mineralizable-N (mg N kg^{-1} soil), dehydrogenase activity (log(dehyd.activ.+10)) and bacteria abundance (log(bact.abund.+10)) after four incubation periods (1=5 days, 2=21 days, 3=42 days, 4=84 days) considering the mean effects of tillage and depth. Equal letters for the same variables are not significantly different (p<0.05).

Conclusions

Results clearly show the negative priming effect during the first five days after residues placement in the soil, with N assimilation by soil microbes. After this N immobilization, a mineralization took place and lasted 37 days until labile N was depleted and a new smaller N immobilization occurred. A succeeding crop included in the rotation must be sown within one and a half months after oat residues are returned to soil in order to use the mineral N produced from residues, before immobilization or leaching occur. The dehydrogenase activity was a good indicator for soil N immobilization.

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The changes in microbial biomass nitrogen in a long-term field experiment with crop rotation

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Abstract

Microbial biomass nitrogen was studied in long-term field experiments with crop rotation. Higher contents of microbial biomass nitrogen were found in treatments with organic fertilisers. The highest contents of microbial biomass N were in the plots treated with sewage sludge. The average content of microbial biomass N in this treatment was 48% to 59% higher compared with the control. A positive influence on the microbial biomass N content came from application of manure. The average content of microbial biomass N was estimated to be 35% higher than in the control. In treatments with mineral N-fertilisers, there was a tendency towards a lower content of microbial biomass, compared with the plots without fertilisers. The average content of microbial biomass N in variant with N-fertilizers reached 83% of that in the control.

Background and objectives

Management practices such as crop rotation, residue management and fertilization affect the quality and quantity of soil organic matter. Microbial biomass and microbial activity are closely related to soil organic matter content. They are positively influenced by organic amendments such as crop residues and animal manures (Anderson and Domsh, 1989, Schnürer *et al.,* 1985). Microbial biomass measurements can detect fertilization and crop rotation effects on soil earlier than total organic C or N measurements in soil (Powlson *et al.,* 1987) and therefore they may be an indicator of potential C sequestration. In this context, microbial biomass can be a valuable tool for understanding changes in soil properties and in the degree of soil degradation or soil quality (Smith and Paul, 1990, Sparling, 1997). The objective of this study was to evaluate the effect of long-term crop rotations and organic and mineral fertilization on microbial biomass N under a different soil and climatic conditions.

Material and methods

The field experiments with crop rotation were established in 1996 in five different soils and climatic regions of the Czech Republic. There is a rotation of three crops in order – potato, winter wheat, spring barley. The experiment consisted of seven treatments: 1) control, 2) farmyard manure, 3) farmyard manure + N-fertilizer, 4) sewage sludge I, 5) sewage sludge III, 6) N-fertilizer, 7) N-fertilizer + straw. Organic fertilizers (sewage sludge, manure and straw) were applied for potato fertilising in autumn (November) and mineral N-fertilizers were applied in spring (March) for wheat and barley fertilising. The size of experimental plot was 60 m^2. Soil samples were collected from the soil profile 0 - 30 cm in autumn (September) every year in period 1997 to 2004. Microbial biomas N was estimated by the fumigation - extraction method after preextraction (Brookes *et al.,* 1985, Mueller *et al.,* 1992). Microbial biomass N was calculated as a difference in N content in fumigated and nonfumigated sample (E_N) using coefficient k_{EN} (microbial biomass N = $E_N : k_{EN}$). The value k_{EN} = 0.54 was used to calculate microbial biomass N (Brookes *et al.,* 1985, Jenkinson, 1988).

Results and discussion

The content of microbial biomass N depended on the treatment, soil and climatic region and year and fluctuated between 7.3 mg N kg^{-1} to 35.8 mg N kg^{-1}. Higher contents were found in treatments with organic fertilisers. The highest contents of microbial biomass N were in this case estimated in the plots treated with sewage sludge (12.2 mg N kg^{-1} to 35.8 mg N kg^{-1}). The average content of microbial biomass N in treatment sevage sludge I was 48% higher compared with the control and in treatment sewage sludge III it was 59%. A positive influence on the microbial biomass N content came from application of manure. The average content of microbial biomass N was estimated to be 35% higher than in the control. The highest contents of microbial biomass nitrogen in plots with organic fertilizers were found in samples collected the first year after organic fertilizers application on plots with potato. At the samples from winter wheat and barley plots (samplings the second and the third year after application of organic fertilizers) the content of microbial biomass nitrogen had declined to about half of its size first year after organic fertilisers addition. This was still ca 15% more than in the unfertilised control, but differences were not significant.

In treatments with mineral N-fertilisers, there were a tendency towards a lower contents of microbial biomass, compared with the plots without fertilisers. There were differences between soil conditions and duration of experiment. First six years duration of the experiment there were no differences between control and N-fertilisers treatments. The last two years of duration of the experiment the average content of microbial biomass N was decreased by 17% in variant with N-fertilizers on experimental sites with sandy-loam soil. There were no differences between control and N-fertilisers treatments on experimental sites with clay-loam soils. There were no significant differences between control and N-fertilisers + straw treatments. However, the addition of straw caused an increase in the content of microbial biomass N by 26% compared with N-fertilisers treatment.

Table 1. The average contents of microbial biomass nitrogen (1997 - 2004)-potato plots.

Treatments	Control	Sew. sl. I	Sew. sl. III	Manure	Manure + N	N fertil.	N + straw
mg Nkg^{-1}	12.6	18.3	20.0	17.3	15.0	12.3	15.5

Figure 1. The average contents of microbial biomass nitrogen in plots with potato.

Conclusions

The contents of microbial biomass depended on the treatment and year of sampling. Higher contents of microbial biomass were found in treatments with organic fertilisers compared with the control. In treatments with mineral N fertilisers there was a tendency towards a lower contents of microbial biomass, compared with the control.

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Soil tillage practice affected CO_2 and N_2O emissions in a newly initiated experiment in Denmark

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Abstract

The environmental effect of reduced soil tillage intensity was studied in an experiment established in 2002 in Denmark on a loamy sand soil. Three treatments were included, namely conventional tillage (CT), reduced tillage (RT) and direct drilling (DD). Measurements of CO_2 and N_2O emissions by static chambers within whole vegetation periods of spring barley, winter oilseed rape and winter wheat during 2003-2005 showed a decreasing CO_2 emission in the order CT>RT>DD. Increased soil tillage intensity in the RT and CT treatments increased N_2O emissions compared to the DD on a short term time scale for all vegetation periods. For the winter cereals, RT gave the highest N_2O emissions, but the cumulated emissions of N_2O were much lower for winter oilseed rape and winter wheat compared to the spring barley. Neither soil total C and N, mineral N measured in spring barley after tillage, nor soil microbiological parameters in winter oilseed rape could explain the difference in emissions between tillage treatments on the time scale studied.

Keywords: green house gases, spring barley, tillage, wheat, winter rape

Background and objectives

Reduced tillage practices are commonly in use as conservation tillage in America, Australia and South Africa. These soil tillage practices are much less common in Europe, and their environmental side-effects on different soils and climates in the short and long term are poorly documented and should be quantified (Olesen, 2005).

The objective of this study was to assess the effects of different soil tillage intensities on the emissions of CO_2 and N_2O from arable cropping systems over the whole vegetation period. The measurements were conducted in a tillage experiment, which is initiated in 2002.

Materials and methods

The measurements were conducted in 2004 in spring barley undersown with perennial ryegrass and in 2003 to 2005 in winter oilseed rape followed by winter wheat. The experiment was located at Foulum (56.50ºN, 09.57ºE) on a loamy sand soil (9% clay, 4% organic matter) with a mean annual precipitation of 704 mm and a mean temperature of 7.3°C. The measurements were taken in three soil tillage treatments, represented by 30 m^2 experimental plots in four replicates in a completely randomised block design: CT - conventional tillage with rotovation to 8-10 cm and ploughing to 20 cm followed by rolling before sowing; RT - reduced tillage with rotovation to 8-10 cm; and DD - direct drilling.

The soil surface fluxes of CO_2 and N_2O were measured by chambers (Petersen, 1999) during a period of 110, 201 and 121 days in spring barley, winter oilseed rape and winter wheat, respectively. To correct the measurements for the diurnal variation in emissions, measurements of the fluxes were taken four times during the day for a period of three days. Additional measurements of soil water content (TDR, 30 cm depth) and temperature were taken.

Results and discussion

Compared with CT the cumulated CO_2 emissions for the period March-May were lower for the reduced tillage treatments in 2004 for spring barley and winter oilseed rape and in 2005 for winter wheat (Table 1).

Table 1. Relative cumulated CO_2 (March-May) and N_2O (whole vegetation) emissions from studied treatments (in % of emissions from CT).

Crop	Treatment		
	CT	RT	DD
CO_2 emissions			
Spring barley (2004)	100	67	65
Winter oilseed rape (2004)	100	77	70
Winter wheat (2005)	100	74	65
N_2O emissions			
Spring barley (2004)	100	81	54
Winter oilseed rape (2003)*	100	267	67
Winter wheat (2004-05)	100	117	89

* *Cumulated N_2O emissions are presented only for Autumn 2003.*

The cumulated N_2O emission from spring barley (estimated from the observed period of 110 days) was 46% higher from CT than from DD. The emission from CT was significantly higher even before tillage and the difference increased after tillage, but decreased after fertilization (Figure 1). For winter oilseed rape and winter wheat the differences in N_2O emissions between the treatments were smaller, which may partly be due to fewer days of measurements, so that periods of large differences may have been missed in the measurements. Nevertheless, soil under winter cereals emitted lower N_2O amounts from DD, whereas emissions from RT were comparable to those from CT or even higher, as for winter oilseed rape in the Autumn 2003 (Table 1). Both daily N_2O fluxes and cumulated emissions of N_2O were much lower for the winter cereals as compared to the spring barley.

Generally, the increased soil tillage intensity in the RT and CT treatments increased both CO_2 and N_2O emissions compared to the DD on a short-term scale. However, neither soil total C and N, mineral N measured in spring barley after tillage, nor soil microbiological parameters in winter oilseed rape could explain the differences in emissions between the treatments.

Figure 1. Measured N_2O fluxes from sandy loam soil under CT, RT and DD tillage treatments during cropping with spring barley in 2004 and winter wheat in 2004-05; bars indicate S.E. (n=4); arrows indicate time of harrowing (HA), ploughing (PL), sowing of spring barley (B), winter wheat (W) and catch crop (G) and N applications (N) with doses in kg ha^{-1}.

Conclusions

Reduction in tillage intensity is able to reduce both CO_2 and N_2O emissions in the field. As we found no biochemical explanation for the differences in emissions between the treatments on the short-term time scale they probably could be explained by changing soil properties, including diffusivity. The N_2O emissions from spring cereals may be reduced by increasing efficiency of N uptake in the spring. Rotation with winter cereals decreased daily fluxes of N_2O and cumulated emissions as compared to spring cereal, but increased risk of N leaching in the autumn and emissions from RT.

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Manipulating the N release from crop residues by using organic wastes: a field study

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Abstract

The potential to immobilize N released from crop residues of green waste compost and straw and the potential to remineralize the immobilized N of malting sludge and vinasses were tested in a field experiment on a loamy sand. The compost was not able to immobilize the released N in the microbial biomass despite its high C:N ratio, while straw showed a good N immobilization potential. Possible reasons could be the high lignin content of the compost, and cold temperatures and intensive rainfall after the incorporation of the compost compared to straw. Neither malting sludge nor vinasses were able to stimulate a remineralization of immobilized N, probably due to an unsuitable chemical composition.

Keywords: crop residues, immobilization, nitrogen, organic fertilizer, priming effect

Background and objectives

Upon mineralization of vegetables crop residues, large amounts of mineral N are released in soil, resulting in high nitrate leaching risks. Organic wastes like straw, paper sludge and green waste compost have shown to possess an N immobilization potential. Both intensity of immobilization and the time at which remineralization occurs, seems to be manageable by the right choice of waste (De Neve *et al.*, 2004). Especially wastes with a low lignin content and high C:N ratio seem to have a potential to immobilize N (Chaves *et al.*, 2005). It also seems possible to remineralize immobilized N by incorporating other organic wastes, like molasses (De Neve *et al.*, 2004) and vinasses (Chaves *et al.*, 2005). The aim of this study was to test the effect of organic wastes on the N mineralization-immobilization turnover of crop residues under field conditions (2 seasons).

Materials and methods

A field trial was set up on a loamy sand. Following materials were incorporated: cauliflower residues as N-rich crop residues and green waste compost as N immobilizer (4 November 2003), malting sludge as remineralizer of immobilized N (14 April 2004) and leek residues as second N-rich crop residues and straw as second N immobilizer (12 October 2004). At regular times soil samples were taken with an auger to a depth of 90 cm in 4 layers, and analysed for their mineral N and microbial N content. During the first spring, leek was grown and during the second spring, lettuce was grown.

Table 4. The (bio)chemical composition of the crop residues and organic wastes (FM_{ap}: applied fresh matter; N_{ap}: applied total N; DM: dry matter; OM: organic matter; N_{tot}: total N content; WS: water soluble compounds; H: hemicellulose; Cel: cellulose; L: lignin).

	FM_a	N_{ap}	DM	OM	N_{tot}	C:N	WS	H	Cel	L
	kg ha^{-1}	kg ha^{-1}	%	%	%		% OM			
Cauliflower leaves	53.3	275	14.1	62.8	3.67	8.48	30.5	35.7	10.7	21.6
Compost	22.7	124	82.4	94.4	0.66	69.4	2.7	25.0	26.2	46.1
Malting sludge	44.4	62	5.02	56.9	2.75	10.4	65.6	20.6	0.0	13.8
Leek leaves	53.3	137	7.16	81.9	3.58	10.9	28.5	36.9	15.7	18.9
Cereal straw	12.8	50	90.8	93.9	0.44	105.4	6.2	33.0	24.1	36.7
Vinasses	6.1	216	60.0	81.0	5.89	6.96	100.0	0.0	0.0	0.0

Results and discussion

During the first winter, the cauliflower residues released up to 93 kg N ha^{-1} in 0-15 cm layer (Figure 1) and increased the microbial N content with 53.3 kg N ha^{-1} (Figure 2) compared to the unamended soil. However, the compost was not able to immobilize the released N in the microbial biomass despite its high C:N ratio. A possible reason could be the coarse and woody structure of the compost and its high lignin content, what may have slowed down its decomposition and N immobilization. Also cold temperatures and intensive rainfall may have reduced the decomposition of the compost, increased the denitrification and reduced N immobilization. Adding malting sludge at the start of the first spring did not significantly increase the dry matter yield or N uptake of leek (Table 1). Hence, remineralization to an extent that a crop could benefit from it, did not occur after incorporation of malting sludge what could be due to a low amount of immobilized N during winter or an unsuitable chemical composition of malting sludge to induce remineralization.

During the second winter, the leek leaves released up to 35 kg N ha^{-1} in 0-15 cm layer (Figure 1) and increased the microbial N content with 32.3 kg N ha^{-1} (Figure 2). When the leek residues were mixed with straw, the mineral N content in soil was significantly lower ($P < 0.05$) (Figure 1) and the microbial N content was significantly higher ($P < 0.05$) (Figure 2) than in the leek only treatment, indicating that straw did immobilize leek-N into the microbial biomass. The better N immobilization potential of straw compared to the green waste compost could be due to its higher C:N ratio (105) and lower lignin content (36.7% on OM). Furthermore, straw was incorporated almost one month earlier than the compost, so that the average temperature during first 30 days after the incorporation was higher after straw incorporation (on avg. 10.2°C) than after compost incorporation (on avg. 8.0°C). This may have lead to an enhanced decomposition of straw, and hence, to a better N immobilization. Adding vinasses at the start of the second spring, again did not significantly increase the dry matter yield or N uptake of lettuce (Table 2). Hence, also vinasses was not able to induce remineralization to an extent that a crop could benefit from it possibly due to an unsuitable an unsuitable chemical composition.

Figure 1. Mineral N content in the top 15 cm; error bars are standard deviation.

Figure 2. Microbial N in the top 15 cm; error bars are standard deviation.

Table 2. Dry matter yield (DM t ha^{-1}) and N uptake (kg N ha^{-1}) of leek (first winter-spring) and lettuce (second winter-spring) cropped on the loamy sand (between brackets standard deviations; different letters indicate significant differences (P< 0.05) in a column)

	Leek		Lettuce	
Treatment	DM *	N	DM	N
Unamended soil	5.3 (0.6)^{a}	162.3 (23.7)^{a}	4.1 (0.4)^{a}	171.6 (12.5)^{ab}
Cauliflower/leek only	8.1 (0.9)^{b}	231.0 (16.8)^{c}	5.2 (0.4)^{b}	216.2 (18.9)^{b}
Compost/straw	6.5 (0.5)^{a}	193.4 (17.4)^{ab}	4.5 (0.5)^{ab}	185.6 (21.5)^{ab}
Soil + malting sludge/vinasses	6.3 (0.6)^{a}	196.2 (11.2)^{ab}	3.8 (0.4)^{ab}	165.8 (20.5)^{a}
Cauliflower/leek + malting sludge/vinasses	6.6 (0.6)^{a}	190.0 (10.5)^{ab}	4.4 (0.7)^{ab}	183.2 (32.2)^{ab}
Compost/straw + malting sludge/vinasses	6.3 (3.5)^{a}	213.7 (36.7)^{bc}	5.0 (0.4)^{a}	205.9 (20.7)^{ab}

* *Leek leaves were not removed before weighing the plants.*

Conclusions

In order to have a good N immobilization potential, organic wastes should not only have a large C:N ratio, but they should be readily decomposable in soil (i.e. low lignin content, sufficiently high temperatures). Stimulating remineralization of immobilized N seems not easy to accomplish.

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Variability of the Denitrifying Enzyme Activity at the field scale

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Abstract

We examined spatial and temporal variability of Denitrifying Enzyme Activity (DEA) in an arable soil. Samples were collected from both undisturbed (soil cores) and mixed topsoil samples, at both winter and summer sampling dates, on two distinct sampling grids (25m and 10cm). DEA measured on soil cores ranged from 0.007 to 0.419 mg N_2O N $kg^{-1}d^{-1}$. DEA measured on mixed soil samples ranged from 0.646 to 3.246 mg N_2O N $kg^{-1}d^{-1}$. The DEA of soil cores is significantly lower than the DEA of mixed soil samples. Coefficients of variations are higher for soil cores than for mixed soil samples. CV are as high at a small scale sampling (10cm) than at 25m grid sampling. A nugget value of 0.38 was found. This represents an undetectable variability of more than 30% at the scale of sampling.

Keywords: denitrification, spatial and temporal variability

Background and objectives

Denitrification is known to be one of the most variable soil processes (Parkin *et al.*, 1987). Important spatial and temporal variability (coefficient of variation (CV) of 100-200%) have been reported for N_2O emission rates (Röver *et al.*, 1997). This makes it difficult to study denitrification in the field (Simek *et al.*, 2000). Groffman and Tiedje (1989) and Watson *et al.* (1994) found evidences that a laboratory technique measuring the activity of existing denitrifying enzyme (DEA) could be directly related to annual field denitrification rates. The seasonal and spatial variability of DEA correspond to CV of 5 to 35% (Ambus, 1993; Simek *et al.*, 2000; Dhondt *et al.*, 2004). Ambus and Christensen observed that activities from amended soil slurries were 13 times higher than the activities in soil cores. The cores exhibited CV of 96-149% due to seasonal variation and CV of 64-314% due to spatial variation in the denitrification activity. Hotspots of denitrification are responsible for the high CV and Röver *et al.* (1999) noticed that it contributed to more than 50% of total N_2O emissions. Thus, collecting soil samples instead of measuring field rate emissions failed in lowering observed variability. This stresses the question of an appropriate sampling strategy for DEA measurements. We examined spatial and temporal variability of Denitrifying Enzyme Activity (DEA) in an arable soil. Samples were collected from both undisturbed (soil cores) and mixed topsoil samples, at both winter and summer sampling dates, on two distinct sampling grids (25m and 10cm).

Material and methods

The study site is a cropped field located in Southern Belgium, 10 km north-west of the city of Arlon. The mean annual rainfall is 1031 mm. Soil is a sandy silt to clayey sandy silt. Soil sampling was realised in triplicate and achieved on 2 sampling grids of 3 rows and 3 columns, one with a sampling interval of 25 m, the other with a sampling interval of 10 cm. Soil cores (5cm i.d. by 5.1 cm long) were collected in October 2002 (25m grid) and in July 2003 (10 cm grid). Soil sampling was achieved in February 2004 (10cm grid) and in July 2004 (25m grid). Nitrous oxide emissions rates were measured according to the Denitrifier Enzyme Activity (DEA) method (Tiedje, 1994). An immediate incubation was performed to prevent the analysis from any storage disturbance. Subsamples (25 g) of mixed soil were made into slurries with addition of 20 ml of a 0.1g l^{-1} chloramphenicol, 0.3g l^{-1} glucose and 0.05 g l^{-1} KNO_3 solution. Soil cores were saturated with the same solution. Soil samples are incubated in a silicone capped glass bottle which allows gas tight syringe sampling (500 µl) of the headspace. Anaerobiosis was imposed by a 10min N_2 flush. 1-10% volume of acetylene is added to block the transformation of N_2O to N_2. A one-way vent

ensures pressure balance with local atmospheric pressure. Soil slurries were continually agitated on an orbital shaker. Analysis of N_2O is performed on a gas chromatograph (HP6890) equipped with an Electron Capture Detector (ECD). N_2O content of each bottle is analysed every 15 minutes, for 90 minutes. The production rate of N_2O was calculated on the basis of the linear regression slope of the evolution of N_2O concentration vs. time. All results are expressed on a dry weight basis. Means, standard deviations and coefficients of variation were calculated using UMVUE/LAND estimators (LSTAT, Parkin, 1991). Anova analysis were realised in Statistica (StatSoft Inc.). Semivariograms were computed with Geo-EAS (EPA, Environmental Monitoring Systems Laboratory).

Results and discussion

DEA measured on soil cores ranged from 0.007 to 0.419 mg N_2O N $kg^{-1}d^{-1}$. DEA measured on mixed soil samples ranged from 0.646 to 3.246 mg N_2O N $kg^{-1}d^{-1}$. The DEA of soil cores is significantly lower than the DEA of mixed soil samples ($p<0.0001$, Table 1). Mixing the soil sample into a slurry increases the DEA by 10-40 times in comparison of incubating an undisturbed soil sample. In an agitated soil slurry, there is a constant feeding of the denitrification process through permanent contact between nutrients and denitrifiers. In soil cores, even water and nutrients saturated, denitrification is limited. We hypothesized this is due to differences in the diffusion of both nutrients to the active denitrifying microsites and N_2O from these sites to the headspace of the vials.

Table 1. Descriptive statistics of Denitrifying Enzyme Activity.

Grid	Date	Mean (mg N_2O N $kg^{-1}d^{-1}$)	Std deviation	CV (%)	Sampling technique
25 m	Oct2002	0.149	0.127	85.38	cores
10 cm	July2003	0.056	0.049	88.89	cores
10 cm	Feb2004	2.826	0.351	12.41	mixed soil
25 m	July2004	1.519	0.623	40.99	mixed soil

Coefficients of variations are higher for soil cores than for mixed soil samples. CV are as high at a small scale sampling (10cm) than at 25m grid sampling. This is consistent with the results of Ambus and Christensen (1993) who observed that the variability of denitrification between static soil cores replicates was as high as the variability along a 40m transect. Undisturbed soil samples are more closely representing natural denitrification hotspots and hence spatial variability of denitrification. Mixing soil samples and incubating agitated soil slurries reduces the CV by half to four by giving an optimal expression of DEA.

Figure 1. DEA of soil cores (A) and of soil slurries (B) in the sampling grid of 25m.
Each bar is the mean of 3 replicates. Grey bars on the A charts are for the 10cm grid.

Our data set was analyzed by ANOVA (n=30). Explained variations of 6.43, 8.74 and 75.16% were found for the grid, season and sampling technique respectively ($p<0.05$). The season effect is larger than the grid effect in our site. The seasonal variation of control parameters overflows any spatial dependence. This trend can partly be related to the water content fluctuations. Consistently with observations by Röver *et al.* (1999), no spatial relationship was found. A nugget value (semivariance at lag distance 0) of 0.38 was found on the variogram computed (sill 0.7). This represents an undetectable variability of more than 30% at the scale of sampling. Patterns for DEA variance in soil cores seems similar to the small scale spatial variability of microbial carbon and nitrogen. Stark *et al.* (2004) identified a nugget effect : the observed variance was not spatially structured at the 10-40cm scale.

Conclusions

We conclude that homogenization of the soil sample by mixing allows a larger expression of the DEA. It reduced the coefficient of variation of DEA. A nugget effect and a small scale variability is identified. Soil sampling should take the centimetric scale into account.
As the seasonal variation was important, soil sampling should occur at different contrasting seasons.

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Nitrate fluxes and nitrate removal in the shallow aquifer below an Andisol field

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Abstract

Quantitative information on denitrification in groundwater is indispensable for predicting diffuse pollution by nitrate. However, such data are scarce, mainly due to the difficulties in determining *in situ* groundwater flow. The objective of this study was to determine field-scale water and nitrate fluxes and nitrate removal in an unconfined aquifer below a cropped Andisol. Spatial distributions of groundwater levels, depths to the low-permeability layer underlying the aquifer and nitrate concentrations were measured for determining horizontal discharge fluxes of water and nitrate, based on numerical simulation. Mass balances of water and nitrate in the subsoil showed that vertical discharge through the low-permeability layer was the major exit for water and nitrate. The average nitrate-nitrogen removal rate was 5.2 kg N ha^{-1} yr^{-1}. The half-life of nitrate (5.3 yr), based on first-order kinetics, was approximately twice as high as the average residence time of nitrate in the aquifer. This indicates that the contribution of the denitrification process to mitigation of nitrate pollution in this aquifer is limited and that a significant proportion of the nitrate leached from the crop root zone discharges through the low-permeabitlity layer towards the underlying deeper aquifers.

Keywords: anion adsorption, denitrification, groundwater, half-life

Background and objectives

Quantitative information on denitrification in groundwater, such as the degradation rate constant of first-order kinetics for nitrate (NO_3^-), is indispensable for predicting and mitigating diffuse pollution by NO_3^-. Several studies have attempted to determine NO_3^- loss in aquifers through the mass balance method (Jacobs and Gilliam, 1985; William *et al.*, 1995), but the degradation rate constant of NO_3^- in aquifers has not been obtained under field conditions. This is mainly due to the lack of information on the spatial distribution and/or the temporal variation in groundwater levels, NO_3^- concentrations and the depth of the low-permeability layer, underlying the aquifer. In this study, we focused on accurately determining field-scale water and solute fluxes through the shallow aquifer overlying a low-permeability layer. The objective of this study was to determine field-scale water and NO_3^- fluxes and NO_3^- removal in an unconfined aquifer below a cropped Andisol.

Materials and methods

The study was conducted in a field (24.5 × 38.5 m) located on Joso Upland, Tsukuba, Japan. Annual average precipitation and temperature are 1,250 mm and 13.4 ˚C, respectively. The soil is an Andisol (Hydric Hapludand) with a texture of light clay to heavy clay, abundant in allophane and other amorphous materials. Water table depth fluctuates usually around 2 m. The unconfined aquifer is underlain by a clayey low-permeability layer.
Sweet corn (*Zea maize* L.) and Chinese cabbage (*Brassica pekinensis* Rupr.) were grown every year, and average chemical nitrogen applications in 1995–2002 were 168 and 191 kg ha^{-1}, respectively. Winter wheat (*Triticum aestivum.* L.) was grown in winter, without fertilizer application, to be incorporated into the soil as green manure in spring before maturity.

Volumetric water contents from the surface to a depth of 1 m and at a depth of 1 m were measured by Time Domain Reflectometry. Pressure potentials at depths of 90 and 110 cm were measured by tensiometers. Matrix flow across a depth of 1 m was calculated by applying Darcy's law. Unsaturated hydraulic conductivity of the undisturbed soil was determined by the steady state method. Preferential flow across a depth of 1 m was calculated by the water balance method (Hasegawa and Eguchi, 2002) assuming that evaporation, root water uptake and surface runoff were negligible during and after heavy rain events. Water table levels were measured almost every day at 61 monitoring wells installed in the field with a grid of about 5 m. Dynamic cone penetrometer measurements were conducted at 54 points with a grid of about 5 m for determining the spatial distribution of the low-permeability layer. NO_3^- and chloride (Cl^-) concentrations in the soil water, extracted by suction porous cups at depths of 1, 1.5 and 2 m, and those in the shallow aquifer were measured approximately twice a month.
Horizontal groundwater discharge flux was determined by numerical analysis based on the method of successive steady states (Bear, 1972) assuming that at every instant of time the phreatic surface has the shape of a steady phreatic surface and that the transient process may be regarded as a sequence of steady states. A two-dimensional, horizontal, steady-state unconfined groundwater flow equation, with net vertical groundwater recharge flux Δq_v (= vertical recharge flux – vertical discharge flux), taking into account the vertical profiles of porosity and saturated hydraulic conductivity, as well as the spatial distribution of the low-permeability layer, was applied to simulate the phreatic surface of the aquifer. By solving the equation, subject to the constant-head boundary conditions, using the measured field-boundary water levels, the horizontal discharge flux (= Δq_v) was obtained.
Mass balances of water, NO_3^- and Cl^- were determined in the subsoil, between a depth of 1 m and the upper boundary of the low-permeability layer (≈ 2.6 m on average), during the period from November 25, 2001 to October 11, 2002. NO_3^- and Cl^- fluxes were determined as the product of water flux and solute concentrations. The distribution coefficient of Cl^-, an example of a non-degradable monovalent inorganic anion, was calculated on the basis of the mass balance of Cl^- in the liquid phase, and was applied to estimate NO_3^- adsorption.

Results and discussion

Drained water from a depth of 1 m during the period from November 25, 2001 to October 11, 2002 was 222 mm, whereas horizontal groundwater discharge was only 28 mm. This indicates that groundwater discharge occurred mainly in the vertically downward direction (188 mm) through the low-permeability layer toward the underlying deeper aquifers. The average water table depth and the average storage water in the aquifer during this period were 2.01 m and 383 mm, respectively. Average residence time of water (= storage water / total discharge flux) in the shallow aquifer was calculated to be 1.55 yr.
The distribution coefficient of Cl^- was estimated to be 1.3×10^{-3} m^3 kg^{-1} based on the mass balance of Cl^- in the liquid phase; very close to the value obtained from the relationship between the total content of Cl^- in the soil and the concentration of Cl^- in the soil water. This indicates that the mass balance of solute was calculated sufficiently accurately.
The mass balance of NO_3^- (Figure 1) indicates that 4.6 kg ha^{-1} of NO_3-N was removed from the subsoil during this period. The average NO_3^- removal rate was 5.2 kg N ha^{-1} yr^{-1}. Ishizuka and Onodera (1997) showed evidence for denitrification in the shallow aquifer on Joso Upland by isotope enrichment of ^{15}N-NO_3^- within and just above the water table. They did not detect denitrifying microbes in the subsoil at depths of 0.5 and 1 m, whereas they detected 1.1×10^3 CFU (colony forming unit) of denitrifying microbes per g of dry soil at a depth of 1.5 m just above the water table. Yoshida and Oba (1982) measured oxidation-reduction potentials in the shallow aquifer on Joso Upland and showed seasonal variations from 40 to 360 (141 on average) mV, which is low enough for NO_3^- reduction. These results suggest that denitrification is a common process in the shallow aquifer on Joso Upland. Assuming that denitrification occurred only within the saturated zone, the degradation rate constant of first-order kinetics for NO_3^- was calculated to be 1.8×10^{-9} s^{-1}. This value is two orders of magnitude smaller than the value obtained in laboratory experiments using repacked soil columns (Ogawa *et al.*, 2000). This suggests that model predictions using denitrification rates derived from laboratory experiments using disturbed soil samples may overestimate NO_3^- removal from aquifers and underestimate the risk of NO_3^- pollution. Therefore, field monitoring-based denitrification rates should be used in prediction models for NO_3^- pollution in the groundwater. The half-life of NO_3^- in the aquifer was calculated to be 5.3 yr. This value was more than three times higher than the average residence time of water in the aquifer and approximately two times higher than that of NO_3^- if the effect of anion

adsorption on NO_3^- transport was taken into account. This indicates that the contribution of the denitrification process to mitigation of NO_3^- pollution in this aquifer is small and that significant quantities of NO_3^-, leached from the crop root zone, discharge through the low-permeability layer and are not denitrified. These results show that field-scale characterization of both hydrological and denitrification processes in soil and groundwater systems allow evaluation of the practical effect of denitrification on mitigating diffuse pollution by NO_3^-.

Figure 1. Mass balance of NO_3-N in the subsoil between a depth of 1 m and the upper boundary of the low-permeability layer (about 2.6 m depth on average) during the period from November 25, 2001 to October 11, 2002.

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Simulation of water and nitrogen dynamics in a fertigated potato crop

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Abstract

In potato production, correct timing of water and nitrogen (N) application can help minimize nitrate leaching while still enabling high yields. Simulation models are valuable tools for evaluating the effects of different management practices on crop yields and on the environment. The objective of this study was to adapt and parameterise an existing dynamic simulation model using field experiments with fertigated potatoes (*Solanum tuberosum*). The experiments were conducted during 2003 and 2004. The model was calibrated on data from the fully irrigated treatments from 2004 and validated on the treatments from 2003 and the remaining treatments from 2004. The tuber yields, water use and N uptake of the potato crop were simulated well in most plots. The calibrated model was applied on potatoes cultivated using standard N application and overhead irrigation. The simulations showed that the use of fertigation instead of a practice with application of fertilizers at planting and overhead irrigation did not affect the dry matter yields but the effect on the environment was positive.

Keywords: fertigation, modelling, nitrogen dynamics, potato

Background and objectives

Simulation models are valuable tools for evaluating the effects of different management practices on crop yields and on the environment. In potato production correct timing of water and N application is important for minimizing nitrate leaching and still obtaining high yields. Methods such as irrigation and fertilisation through drip-lines (fertigation) may save water and also decrease nutrient leaching. The objective of this study was to adapt and parameterise the Daisy model to simulate fertigation and reflect the differences between treatments in experiments with fertigated potatoes. The presented work is part of an ongoing EU funded project FertOrgaNic, whose main objective is to enhance the use of various organic fertilisers in farming systems and to develop new management strategies for improvements of the water and N-use efficiency and hence to reduce the environmental pollution (www.fertorganic.org).

Materials and methods

Daisy is a deterministic, one-dimensional, mechanistic model that simulates crop production and water and N balances in the root zone (Abrahamsen & Hansen, 2000, Hansen *et al.*, 1991). The model simulates C and N transformations in soils including turnover of organic C and N, immobilisation, nitrification, and denitrification processes. Drip irrigation is simulated as a source term at a certain depth interval in Richard's equation. The depth interval is user-defined and determined by the location of the drip line. Similarly, fertigation is simulated as a source term in the convection-dispersion equation.

The Daisy model was applied on data obtained from potatoes grown on coarse sandy soils in Denmark during 2003 and 2004. The field experiment included 5 treatments: T1) not irrigated, not fertilised, T2) application of pig slurry only, not irrigated T3) application of pig slurry only, irrigated, T4) application of pig slurry combined with mineral fertilisers in the irrigation water, and T5) the same as T4), but an alternative fertigation strategy with N in the irrigation water late in the growing season. About half of the crop demand for N was applied as pig slurry just before

planting (T2, T3, T4 and T5). Drip lines supplied additional amounts of mineral N (T4 and T5) and water during growth (T3, T4 and T5).

Figure 1. Placing of TDR-probes.

Measurements in the soil included soil-water content, soil water tension, mineral N and nitrate concentrations from suction cups. The plant measurements included LAI, dry-matter accumulation, and N-uptake. The model was calibrated on data from the fully irrigated treatments from 2004 and validated on the treatments from 2003 and the remaining treatments from 2004. The estimated hydraulic parameters of the soil were adjusted in order to minimize the difference between observed and simulated soil moisture contents. The same soil parameters were used in the model for both years although the experiments were located in different fields at the same farm. However, it was necessary to adjust the hydraulic conductivity of the B-horizon. The crop was calibrated on the basis of measurements of N-yields in T4 and T5 during 2004 and on the measurements of LAI and parts of the plants (leaf, stem, tuber).

Results and discussion

It is difficult to apply a one-dimensional model on crops cultivated in ridges and rows. A fictive zero level was used for the modelling of the soil surface defined as the mean of the level between ridges and the top of the ridges. Measurements of soil water content were made by several TDR-probes (Figure 1), and when comparing the simulations from 0-60 cm with TDR-measurements, the probes from fictive zero down to 60 cm was used. Generally, good agreements were found between measured and simulated soil water contents in 0-60 cm. However, in the beginning of the growing season during 2004, the simulated soil water contents generally were lower than the measured and likewise for T3 and T4 in 2003. TDR-measurements were also made horizontally in three depths (0, 23, 50 cm). The low simulated soil water content at the beginning of the growing season was most pronounced in 23 cm indicating that the model may overestimate root development at the beginning of the growing season, thereby increasing water consumption in deeper layers.

The simulations resulted in a relatively good agreement between observed and simulated dry-matter yields and N yields, although the stressed treatments (T1, T2, T3) often deviated more than T4 and T5. The differences between measurements and simulations were larger during 2004 than during 2003. However, the biomass development during the growing season and the partitioning of dry matter into plant parts were generally simulated well by the model for most plots (Figure 2). Also the N content in different plant parts was well simulated.

Figure 2. Measured and simulated dry matter of plant parts during 2003.

The model estimated larger N yields during 2004 than during 2003, as also observed. The distance between drip points was too large in the first year, indicating that the low yields this year resulted from local water stress. This was also reflected by the TDR probes. The tendency for larger N yields in T4 than in T5 during 2003 was also achieved in the simulations. During 2004 the observed N yields in T5 were larger than in T4 and this was similarly reproduced in the simulations.

Two model scenarios were followed for the same two years and compared to the 'fertigation practice'. Common for the two scenarios and the 'fertigation practice' was the initial application of slurry, the amount of fertiliser applied, and the amount of irrigation water. Scenario 1 included drip irrigation/one application of N fertiliser at planting, and scenario 2 included overhead irrigation/one application of fertilisers at planting.

Simulation results from the two scenarios were compared with simulation results from the practice with fertigation. Scenario 1 resulted both years in unchanged dry-matter yields, decreased N-yields (6-18 kg N/ha) and increased N leaching (6-9 kg N/ha, 1 May-31 December). Scenario 2 resulted during 2003 in increased dry matter yields (0.3 t DM/ha) and decreased N yields (4 kg N/ha) and during 2004 in decreased dry-matter yields (0.3 t DM/ha)) and decreased N yields (19 kg N/ha). In both years N leaching increased (7-15 kg N/ha) when compared with the practice with fertigation.

Conclusions

The tuber yields, water use and N uptake of the potato crop were simulated well in most plots. Generally simulations of T4 and T5 agreed better with measurements than the more stressed T1, T2 and T3. Although the agreement between simulations and measurements generally were acceptable, further improvement in the parameterisation of the potato may be possible, e.g. distribution of the roots. The simulations showed that the use of fertigation instead of 1) drip irrigation/application of fertilizers at planting and 2) overhead irrigation/application of fertilisers at planting did not affect the dry matter yields but the effect on the environment was positive.

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Improved simulation of net N mineralization for internet-based site-specific N fertilizer recommendations

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Abstract

To keep the surplus of agricultural N balances as low as possible, it is crucial to include the amount of N mineralized during crop growth into N fertilizer recommendations. Thus the main aim of this ongoing project is to improve the simulation of net N mineralization by providing adequate N mineralization parameters and functions that are adjusted to specific soil and crop conditions, in Lower Saxony, Germany.
In a first step the mineralization module is adjusted using long-term laboratory incubations from different treatments of agricultural field trials as well as other well-documented sites. Evaluations will be carried out by comparing simulation results based on the so-derived parameters and functions to field mineralization measurements. Other work in this joint project is done on adjusting a crop model and supplying daily site-specific soil water data. The joint model will be the basis for site-specific N fertilizer recommendations via the internet in a later phase of the project. First results show that the amount and the source of fertilizer N, the use of cover crops as well as the site itself influence N mineralization from laboratory incubations.

Keywords: fertilizer, internet-based recommendations, long-term incubation, mineralization, modelling, nitrogen

Background and objectives

Besides the N demand of the crops and the mineral N content of the soil, it is crucial to include the amount of N mineralized in the soil during crop growth into N fertilizer recommendations in order to further reduce the surplus of agricultural N balances. So far, for advisory purposes only very rough estimates of N release are available, but it is well known that N mineralization - especially of soils with high mineralization potential - needs to be more accurately considered. Various models for the simulation of net N mineralization exist. However, these models are usually not adjusted to specific soil and site characteristics, e.g. it was shown that the parameters and temperature functions taken from North German loess soils are by far not adequate for Northwest-German sandy arable soils (Heumann and Böttcher, 2004a; Heumann *et al.*, 2003). In addition, the complexity of available models makes them unsuitable for advisory purposes.
The main objective of this ongoing project is to provide adequate N mineralization parameters and functions that are adjusted to specific soil and crop conditions, with a focus on sandy arable soils as well as loess soils in Lower Saxony, Germany.

Material and methods

Samples were taken in spring 2005 from more than 50 treatments of 14 field trials carried out by the Agricultural Extension Service of Lower Saxony in 9 regions of Lower Saxony. Experimental factors were the amount (no N,

optimum N, optimum N minus 40%) and the source (mineral fertilizer, liquid manure) of fertilizer N as well as the use of cover crops vs. conventional management. The trials had mostly been running for 7 to 12 years.
The mineralization parameters of two organic N pools were derived by a long-term incubation-leaching technique (about 200 days) at optimum temperature and water conditions (Heumann *et al.*, 2003). A two-pool first-order kinetic equation was fitted to the cumulative net N mineralization curves. This equation was chosen in order to be able to differentiate between N mineralization from easily mineralizable, 'fresh' organic N and from slowly mineralizable organic N in 'old', more humified soil organic matter. In the near future, temperature and water functions of the two rate coefficients will be obtained by incubating characteristic soils at below optimum conditions. Evaluations will be done by comparing simulation results based on the so-derived functions and parameters to field mineralization measurements in undisturbed soil columns (compare Heumann and Böttcher, 2004b).
Other work in this joint project is done on adjusting a crop model and on supplying daily site-specific soil water data. The joint model will be tested on time series of mineral N contents in the field trials.

Results and discussion

Our first results concentrate on the effect of the experimental factors (amount and source of fertilizer N, use of cover crops) as well as the regional / site influence on N mineralization from the still running laboratory incubations. The various, yet provisional effects will be discussed on the poster.

Outlook

In a later phase of the project the joint model (water balance, N mineralization, N uptake and crop yield) will be used to develop a more simple, internet-based N fertilization software. Since this is meant to be used by farmers and advisors, it will rely to a certain extent on internal data bases (available at the Geological and Soil Survey of Lower Saxony) referring to the specific site. Starting in spring 2007 our simulation results for representative experimental sites will be available at www.isip.de, whereas monitoring data will already be presented in spring 2006.

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Fate of ^{15}N-labeled ammonium nitrogen applied to *Dactylis Glomerata* grassland

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Abstract

From a three year lysimeter study, applying ^{15}N-labeled ammonium nitrogen on orchardgrass (*Dactylis Glomerata* L.) grassland, most of the applied ^{15}N was taken up by the grass or remained at the surface soil layer in the form of organic N (plots receiving 250 and 500 kg N ha^{-1}). In conditions where 1000 kg N had been applied, ^{15}N quickly moved downward in the first year. The recoveries of ^{15}N in grass and soil were approximately 90% at 250 and 500 kg N ha^{-1}, and ^{15}N lost through leaching was less than 1% of the applied ^{15}N. The concentration of nitrate nitrogen in leachate never exceeded 10 mg N L^{-1} at 250 or 500 kg N ha^{-1}, while the concentration at 1000 kg N ha^{-1} increased markedly, amounting to 50.3 mg N L^{-1}. The non-vegetation plot of 1000 kg N ha^{-1} showed a higher nitrate concentration of leachate than that in the plot with vegetation. Based on these results, it was concluded that the low recovery of ^{15}N in leachate at 250 and 500 kg N ha^{-1} was due to grass uptake and the immobilization of ^{15}N in the upper soil layer, and the importance of having well-managed grass vegetation for regulating nitrate leaching was shown.

Keywords: ^{15}N, leaching, lysimeter, orchardgrass

Background and objectives

Potential nitrogen loading in Japanese agriculture is considerably high because of a high cattle density plus intensive management on the main island (Hojito *et al.*, 2003). Some of the grasslands are used as dumping sites for excess cattle manure, since most of the farms do not have a field area large enough to accommodate the manure that is produced. In such cases, quantification of nitrogen leaching is an important factor in reducing the environmental pollution, particularly in terms of nitrogen retention in the grassland vegetation and soil layer. Using ^{15}N as a tracer is an effective way to analyze the nitrogen flow among plant, soil and leaching water (Clough *et al.*, 1998, Barkle *et al.*, 2001, Kimura *et al.*, 1991, Fillery *et al.*, 2001, and Normand *et al.*, 1997). This paper reports the results of a lysimeter study focusing on the nitrogen flow in a grassland system under heavy nitrogen application conditions.

Materials and methods

During the autumn of the first year, ^{15}N-labeled ammonium sulfate was applied at a rate of 250, 500 and 1000 kg N ha^{-1} in three equal split applications to orchardgrass (*Dactylis Glomerata* L.) grassland in a lysimeter with a depth of 240 cm containing andosol. The ^{15}N concentration of the applied fertilizer was 10.4, 5.4, and 2.9 atom% at 250, 500 and 1000 kg N ha^{-1}, respectively. In the following spring, unlabeled nitrogen was applied at a rate of 62.5, 125 and 250 kg N ha^{-1} divided in three applications between April and July (Year 1). Similar amounts of unlabeled N were added during the second and third year, as shown in Table 1, divided in 6 applications in autumn and spring. A non-vegetation treatment was set on an unlabeled 1000 kg N ha^{-1} plot. The grass was harvested three times a year except in the non-vegetation plot. Leaching water was collected from the lysimeter and the collected grass, leachate and soil were analyzed for ^{15}N. For soil analysis, 2M-KCl solution was used to extract KCl extractable form of nitrogen and the residue soil sample was used for analyzing KCl non-extractable form of nitrogen.

Table 1. Total fertilizer applications during the lysimeter study.

Year	Application timing (3 split applications in autumn & spring)	Treatment			
		N250	N500	N1000	N1000 non-vegetation
		(kg N ha^{-1})			
Year 1 (2001)	Sep. Oct. Nov	250 (^{15}N)	500 (^{15}N)	1000 (^{15}N)	1000 (^{14}N)
	Apr. May. Jul	62.5	125	250	250
Year 2 (2002)	Sep. Oct. Nov	187.5	375	750	750
	Apr. May. Jul	62.5	125	250	250
Year 3 (2003)	Sep. Oct. Nov	187.5	375	750	750
	Apr. May. Jul	62.5	125	250	250
Former treatment*	1997～1999	No application	Slurry (60 Mg ha^{-1})	Slurry (80 Mg ha^{-1})	

* *Before starting the experiment, cattle slurry was applied to the lysimeter for three years, while no fertilizer was used in 2000. Grassland was established one year before the experiment.*

Results and discussion

Table 2 shows the vegetation's dry matter yield (orchardgrass + weeds). The grass yield of the 250 kg N ha^{-1} plot showed a normal growth rate while those of the 500 and 1000 kg N ha^{-1} plots showed growth reduction due to excessive nitrogen, particularly for 1000 kg N ha^{-1}.

Table 2. Dry matter yield of orchardgrass and weeds in the lysimeter study.

Duration	Treatment	Yield DM Mg ha^{-1}					
		Oct.	Nov.	May	Jul.	Sep.	Sum
Year 1	N250	1.18	1.07	6.60	2.20	2.81	13.87
	N500	1.31	1.49	9.89	3.31	4.30	20.30
	N1000	1.10	1.40	9.78	3.71	3.74	19.73
Year 2	N250	1.45		5.11	2.42	2.18	11.15
	N500	1.40		4.27	3.09	3.22	11.98
	N1000	0.92		1.52	1.93	6.19	10.55
Year 3	N250	0.99		5.99	2.52	2.48	11.97
	N500	1.16		5.92	2.65	5.34	15.07
	N1000	0.83		1.72	1.68	5.88	10.11

Distribution of NDF (Nitrogen derived from fertilizer) in the soil layers showed that the majority of NDF in the soil at 250 and 500 kg N ha^{-1} was retained in the upper soil layers (0-20cm), and over 89% of NDF in the soil was in a KCl non-extractable form (e.g. organic or fixed N). In the 1000 kg N ha^{-1} plot, only 20% of NDF was located in an upper

layer, of which 86% was KCl non-extractable, and a considerable amount of the KCl extractable NDF was found in the lower layers (20-240 cm) after the first year of the experiment. In the lower layer of this treatment, the amount of KCl extractable NDF decreased as each year passed, though the amount of non-extractable NDF was relatively stable annually.

The recovery for applied labeled N (NDF) in the harvested grass in the first 12 months (Year 1) was 51, 56 and 36% at 250, 500 and 1000 kg N ha^{-1}, respectively (Table 3). In the following two years, the recovery of NDF was only 3-5%. The recoveries of NDF in grass and soil were approximately 90% at 250 and 500 kg N ha^{-1}, and the NDF lost through leaching was quite small: 0.03% and 0.28% of applied NDF, respectively, during the three years of the experiment. While at 1000 kg N ha^{-1} plot, 22.5% of applied NDF was found in the leachate. The low proportion in the lower layer was consistent with the low level of leaching.

Table 3. Distribution of N derived from fertilizer among grass uptake, soil and leaching.

^{15}N input	(g Nm^{-2})	Duration	N250 (25)	N500 (50)	N1000 (100)
Grass uptake	%	Year 1	51.01	56.03	36.24
		Year 2	3.06	2.76	1.83
		Year 3	2.27	1.88	0.67
		Subtotal	56.33	60.67	38.74
Leaching	%	Year 1	0.02	0.05	1.45
		Year 2	0.01	0.12	10.69
		Year 3	0.00	0.11	10.40
		Subtotal	0.03	0.28	22.54
Soil accumulation	%	Year 3	33.32	30.59	33.42
Unknown (denitrification+α)	%	3 years	10.32	8.47	5.30

The concentration of nitrate nitrogen in the leachate never exceeded 10 mg N L^{-1} at 250 or 500 kg N ha^{-1} over the experimental period, while the concentration at 1000 kg N ha^{-1} increased markedly, amounting to 50.3 mg N L^{-1} at the end of the experiment. The non-vegetation plot under 1000 kg N ha^{-1} condition showed a higher nitrate concentration of leachate than that in the plot with vegetation during years 1 and 2. The proportion of NDF in the leachate was considerably high under conditions of 1000 kg N ha^{-1} while the 250 and 500 kg N ha^{-1} plots showed constantly low proportions. These results suggest an important role is being played by vegetation that is properly managed in terms of the amount of nitrogen application on nitrate leaching regulation through grassland soil layers.

Conclusions

Based on these results, it was concluded that the low recovery of NDF in leachate at 250 and 500 kg N ha^{-1} was due to uptake by the grass vegetation and immobilization in the upper soil layer, and that the importance of well-managed grass vegetation on regulation of nitrate leaching was shown.

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Evaluating nitrogen leaching from monolith lysimeters containing different soils using a transfer function approach

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Abstract

Nitrogen (N) leaching in different soils was evaluated with a stochastic transfer function approach. We cropped sweet corn (*Zea Mays* L.) in summer and Chinese cabbage (*Brassica rapa* L. var. *amplexicaulis*) or spinach (*Spinacia oleracea* L.) in autumn during 4 years on outdoor lysimeters of Andisols (CL), Ultisols (LiC) and Entisols (S) with application of ammonium N fertilizer. Bromide ion was also applied on each lysimeter to obtain a gamma probability density function of tracer travel time. Simulated total N (TN) in leachate obtained by the transfer function approach with pulse inputs of the excess N (applied N–N uptake) agreed well with the observations. In Andisols, N leaching was influenced by the N applications in the preceding year. In Ultisols, N unused by spinach was preferentially transported to the bottom of the lysimeter. In Entisols, TN concentrations increased just after N application and were the largest among the three soils. This study concluded that N leaching behaviours in different soil types can be well understood using the simple transfer function model.

Keywords: andisol, bromide, entisols, gamma probability function, ultisol

Background and objectives

For establishing optimal N management practices in fields, we have to determine the influence of soil type on N leaching. Some researchers have modelled solute infiltration processes under field conditions using stochastic transfer function approaches (Ren *et al.*, 2003; Maeda and Bergström, 2000), but few have used the models to evaluate N leaching from different soils over long periods.

The objective of this study was to analyze 4-year monitoring data on N leaching from outdoor monolith lysimeters containing different soils using a transfer function model, in which excess N for each crop was assumed to leach out of the lysimeters in the same manner as a tracer.

Material and methods

We investigated N leaching from undisturbed monolith lysimeters containing Andisols (n=3), Ultisols (n=3), or Entisols (n=2) to a depth of 1 m, with an inner diameter of 0.286 m. Sweet corn in summer and Chinese cabbage or spinach in autumn were cropped each year on the lysimeters during the period from May 2001 to March 2005. Ammonium N was applied to the plough layer at a rate of 24.6 g N m^{-2} per crop.

To serve as a tracer, Br^- was placed on top of each lysimeter at 20 g Br m^{-2} in the first fertilization. The Br^- concentration in each leachate was normalized by dividing it by the total amount of Br^- leached out of the lysimeter during the entire period. A breakthrough curve of these normalized Br^- concentrations was considered to be a gamma probability density function of tracer travel time (f, mm^{-1}) as a function of cumulative leachate (I, mm) as follows:

$$f[I] = \frac{1}{\beta^{\alpha}\Gamma(\alpha)} I^{\alpha-1} e^{-\frac{I}{\beta}} \tag{1}$$

where Γ is a gamma function, and α and β are fitting parameters for each soil, acquired by moment analysis (Toride and Leij, 1996). The value of $\alpha\beta$ is the mean cumulative leachate. Excess N for each crop was calculated by subtracting crop N uptake from applied N. A breakthrough curve of TN concentration was obtained for each crop

using the transfer function approach with a pulse input of the excess N. Lastly, breakthrough curves for all crops were overlaid during the experimental period.

Results and discussion

Leaching of Br^- accounted for 49%, 67%, and 81% of the input in Andisols, Ultisols, and Entisols, respectively. Tracer travel time was well described by a gamma probability function in each soil type (Figure 1). The mean cumulative amounts of leachate ($\alpha\beta$), at which half of the leached Br^- had passed through the bottom of the lysimeter, were 646 mm for Andisols, 477 mm for Ultisols, and 211 mm for Entisols (Figure 1).

Figure 1. Normalized Br^- concentration of leachate from monolith lysimeters containing Andisols, Ultisols and Entisols, and the curves fitted by a gamma probability function. Symbols and lines refer to measured concentrations and fitted curves, respectively. Between parentheses the parameters α and β are given for each soil type.

Leaching losses of N from lysimeters during the entire experimental period were 103 g N m^{-2} in Andisols (53% of the input), 113 g N m^{-2} in Ultisols (57%), and 151 g N m^{-2} in Entisols (77%), respectively. Simulated TN concentrations in leachate generally agreed with observations (Figure 2). Among the three soils, highest N leaching was found in Entisols and TN concentrations in leachate showed sharp responses to the preceding N inputs over the period, because Entisols had the shortest mean cumulative leachate (Figure 1). In Andisols N leaching was likely influenced by N management in the preceding year, because the mean cumulative leachate was similar to annual amounts of total leachate (471-1178 mm). For instance, TN concentration in leachate increased in summer of the third year because more excess N for crops in the second year leached out in this period, and TN increased again in the fourth year due to low N uptake by the preceding spinach (Table 1 and Figure 2). In Ultisols, the peak Br^- concentration appeared more than 100 mm before the mean cumulative leachate, suggesting preferential transport of N in this soil. Nitrogen leaching from Ultisols was larger in winter because more excess N resulted from cropping of spinach.

Conclusions

Different patterns of N leaching from three soils were well understood using the transfer function model in terms of tracer travel time and N balances in the field. This simple approach would be beneficial for evaluating optimal N management in fields.

Table 1. Uptake of N by crops during the experimental period.

Soil	2001 Sweet corn	Chinese cabbage	2002 Sweet corn	Spinach	2003 Sweet corn	Spinach	2004 Sweet corn	Spinach	Total
Andisols	10.3	27.1 [1)]	14.0	0.3	18.7	3.6	20.9 [1)]	0.2	95.1 (48%) [2)]
Ultisols	5.2	15.3	16.0	0.7	16.9	1.3	12.9	0.3	68.8 (35%)
Entisols	10.8	14.9	7.9	0.2	8.7	6.4	14.3	0.2	63.5 (32%)

[1)] *N Uptake exceeded the amount of N applied to the crop (24.6 kg N m^{-2}), so the excess N was set zero for the calculation.*

[2)] *Between parentheses total N uptake is expressed as percentage of total N input.*

Figure 2. Measured and simulated concentrations of total N in leachate from monolith lysimeters containing a) Andisols, b) Ultisols, and c) Entisols. Solid circles and lines refer to measured and simulated concentrations, respectively. Horizontal lines indicate crop growth periods.

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Nitrogen leaching under grassland from urea and slurry: its relation with pasture production. A lysimeter study

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Abstract

Two monolith lysimeters studies were conducted to evaluate nitrogen (N) leaching losses on an Andisol of Southern Chile. The first study (spring-summer 2000-2001) used *Lolium multiflorum* and increasing rates of N (0-300 kg N ha^{-1}; urea). The second study (winter-spring 2002) used *Lolium perenne*, urea (200 kg N ha^{-1}) and slurry (260-520 kg N ha^{-1}) plus a control treatment. Pasture dry matter (DM) yield and N losses by leaching as nitrate (NO_3-N) and ammonia (NH_4-N) were determined. In the first study, NO_3-N losses by leaching were lower than 1 kg ha^{-1}, with a maximum of 10 mg L^{-1} in the leachates. NH_4-N losses represented about the 10% of the NO_3-N leached below 60 cm and the highest pasture yield (16 ton DM ha^{-1}) was obtained with 300 kg N ha^{-1} (urea). In the second study, the highest NO_3-N leaching was 29 kg N ha^{-1}, with a maximum concentration in the leachates of 10.3 mg L^{-1}. The leaching of NH_4-N did not exceed the 3.6 kg ha^{-1} and the pasture production reached 6.6 ton DM ha^{-1} with 520 kg N ha^{-1} (slurry). N applications in winter should be avoided to minimize N leaching losses and this would increase the N use efficiency by pastures.

Keywords: ammonium, leaching, manure, nitrate, nitrogen, ryegrass, urea

Background and objectives

Organic manures arising from livestock production represent a potential source of major nutrients for crop production (Smith and Chambers, 1995), but if they are not carefully managed they represent a source of diffuse pollution (Lord, 1996). Nitrogen fertilizers can be lost by leaching, erosion and runoff, or by gaseous emissions. The relative importance of these processes depends on the agricultural system and the environment (Peoples *et al.*, 1995).

Nitrogen leaching from intensive agricultural systems, e.g. dairy farming systems where N fertilizers and waste effluents are applied, is considered a major contributor to increased nitrate concentrations in ground water (Jarvis, 1993) and represents a significant factor in soil fertility and pasture production. The aim of this study was to evaluate the effect of N-fertilizer applied as urea and cattle slurry on N leaching losses in two lysimeter systems.

Materials and methods

Two monolith lysimeter studies (0.16 x 0.60 m) were carried out on an Andisol of the Pemehue Series in Southern Chile (38º 54'S; 72º 18' W). The first study (spring-summer 2000-2001) used *Lolium multiflorum* cv. Tama pasture, and increasing rates of N application (0 and 300 kg N ha^{-1}; urea) applied on October 6, 2000. In the second study (winter-spring, 2002) a pasture of *Lolium perenne* cv. Aries was used. The treatments were urea (200 kg N ha^{-1}) and cattle slurry (260 to 520 kg N ha^{-1}) plus a control treatment. The chemical composition of the slurry is shown in Table 1. Both N sources were broadcast with three equal split applications (July 5, August 23 and October 18, 2002). For the first and the second study, the pasture was cut three times and dry matter (DM) yield was measured. N losses by leaching as nitrate (NO_3-N) and ammonia (NH_4-N) below 60 cm were determined.

Table 1. Chemical properties of the slurry applied to the lysimeters. Winter-spring, 2002.

DM (g kg^{-1})	N (g kg^{-1})	P (g kg^{-1})	Mg (g kg^{-1})	K (g kg^{-1})	Ca (g kg^{-1})	Na (g kg^{-1})
130	13.5	6.8	3.9	25.3	9.7	3.9

Results and discussion

Results of the first study (spring-summer) showed that NO_3-N losses by leaching were lower than 1 kg ha^{-1} (Figure 1a), with a maximum concentration of 10 mg L^{-1} in the leachates. The highest NO_3-N loss was observed when 200 kg N ha^{-1} was applied to the soil. On the other hand, NH_4-N losses represented about the 10% of the NO_3-N leached below 60 cm (Figure 1b) and the highest NH_4-N leaching was obtained with the application of 150 kg N ha^{-1} as urea. The amounts of NH_4-N leached was lower than those of NO_3-N because: (i) NH_4^+ can be adsorbed on the soil due to its high affinity by the negatively charged surfaces (Scholefield and Oenema, 1997) and (ii) NH_4^+ can be nitrified to NO_3^- in the soil. The application of urea at a rate of 300 kg N ha^{-1} minimized significantly both the NO_3-N and the NH_4-N leaching because of the greater N uptake and DM production of the pasture than the other treatments.

Figure 1. Leaching of (a) NO_3-N and (b) NH_4-N at increasing rates of N fertilizer applied as urea. Spring-summer 2000-2001.

In the second study, the highest NO_3-N leaching (29 kg N ha^{-1}) was observed with 520 kg N ha^{-1} applied as slurry (Figure 2a), and the maximum concentration in the leachates was 10.3 mg L^{-1}.

Figure 2. Leaching of (a) NO_3-N and (b) NH_4-N at increasing rates of N fertilizer applied as urea (200 N) and slurry (260 N and 520 N). Winter-spring 2002.

As a consequence of heavy rainfall during the winter, a considerable amount of $N\text{-}NO_3$ was leached in all the treatments and about 93% of $N\text{-}NO_3$ losses occurred between July and August. As expected, losses of NH_4-N were significantly lower than those of $N\text{-}NO_3$. The leaching of NH_4-N did not exceed the 3.6 kg ha^{-1} (Figure 2b) and the highest amounts in the leachates occurred in the treatments 200 N (urea) and 520 N (slurry). NH_4-N losses (86%) occurred between July and September and the leaching was decreasing gradually through the evaluation period.
For the spring-summer study it was observed that as the level of N supply was increased, DM yield was significantly raised. N was used efficiently, as indicated by the highest pasture production (16 ton DM ha^{-1}) when 300 kg N ha^{-1} was applied as urea (Figure 3a). This DM yield was about three fold higher than that of the control treatment. In the second study (winter-spring) pasture production reached 6.6 ton DM ha^{-1} with 520 kg N ha^{-1} applied as slurry (Figure 3b), which was about twice the yield of the control treatment (3.1 ton DM ha^{-1}). Furthermore, there were no significant differences among the treatments 200N (urea) and 260N (slurry) in DM yield, because it has been stated that DM produced with slurry could be similar to those obtained from inorganic N-fertilizers (Bittman, 2001). The differential yield response among the first and the second study, can be attributed to both the seasonal fluctuations in DM production and the differences in the ryegrass species evaluated.

Figure 3. Dry matter yield of ryegrass at increasing rates of N fertilizer applied as (a) urea and (b) urea (200 N) and slurry (260 N – 520 N).

Conclusions

In spring time, the application of N to pastures increased DM yield and the N losses by leaching were minimized. Nitrogen applications in winter should be avoided to minimize N leaching losses and this would increase the N use efficiency by pastures.

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Effect of quantity and source of N on ryegrass production and leaching on an Andisol

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Abstract

Nitrogen (N) is lost as volatile emissions to the atmosphere and through leaching and runoff to ground and surface waters. Leaching of N mainly occurs as NO_3^-, while leaching of NH_4^+ is a problem when applied in high quantities on coarse-textured soils. The effects of the N fertilizer source (urea and sodium nitrate) and rates on the yield and shoot N content of ryegrass, and the dynamics of the available mineral N in the soil profile under field conditions were evaluated in an Andisol of Southern Chile. The results showed that DM production increased as N supply increased from 0 kg N ha^{-1} to 300 kg N ha^{-1}. Both, urea and sodium nitrate induced an increase in the plant N content as the rate of N application was raised and a slightly higher N concentration was observed in the urea-treated plants than in sodium nitrate-treated plants. The highest rate of N application yielded the highest NO_3^- and NH_4^+ concentrations in the deepest soil layers evaluated, because the slow growth rate of the pasture generated N leaching down the soil profile, when the soil was treated with sodium nitrate and urea, respectively.

Keywords: ammonium, leaching, nitrate, nitrogen, ryegrass, sodium nitrate, urea

Background and objectives

Nitrogen is an essential element for plant nutrition being the most limiting factor of forage growth, under various circumstances. Intensive grazing systems utilizing *Lolium* pastures for dairy milk production are being increasingly adopted by dairy farmers in Southern Chile as a way of reducing operating costs. In Southern Chile, soil acidity limits grassland production (Mora *et al.*, 1999; 2002) and the acidification, which results in the leaching of nutrients, is a consequence of heavy rainfall in winter, the application of acidifying fertilizers (urea) and plant uptake (Mora and Demanet, 1999). The understanding of the effect of the N source on the plant uptake and N soil transformations will help to develop adequate strategies of N fertilization in acids soils.
The objectives of this research were: (i) to evaluate the effect of the N source at increasing rates of application on the yield and quality of ryegrass and (ii) to study the dynamics of the available mineral N in the soil profile under field conditions in an Andisol of Southern Chile.

Material and methods

A pasture of *Lolium perenne* cv. Nui, was sown in September 1999, on an Andisol of Temuco Series (39º S 73º W, 50 meters above sea level). Urea and sodium nitrate were applied as N source at rates of 150 and 300 kg N ha^{-1}. Both N sources were broadcasted with three equal split applications, at the 3-4 leaves stage (October 5, 1999) and right after the first and second harvests (December 9, 1999 and January 17, 2000, respectively) and control treatments (0 N) were included. The pasture was cut three times (December 6, January 13 and February 3). Dry matter (DM) production and shoots N contents were determined. An ANOVA was performed with the data, using a factorial model with completely randomized design and three replications. Soil samples were taken every fifteen days from October 15 at the 0-10; 10-20 and 20-40 cm depth to determine NH_4-N and NO_3-N at 30 °C on wet basis, for each treatment. These results were analyzed using the Fourier Series regression model.

Results and discussion

The total DM yield increased as N supply increased from 0 to 300 kg N ha^{-1} (Figure 1). However, the N source did not show any effect on the yield. The highest DM yields occurred early in the growing season (October) at the first cut date ($p<0.05$). Temperature and the suitability of fertilization and irrigation practices enhanced the mineralization of organic matter, which resulted in a high plant growth rate and decreased NO_3-N leaching losses. The delay of the last two N applications would explain the lower DM yields of the second and third cuts. This yield decrease could be associated with a temporal reduction in the NO_3-N uptake by defoliation after several cuttings during the growing period (Jarvis and Macduff, 1989) and a loss of regrowth capacity as the temperature increases in early summer.

Figure 1. Effect of N sources and rates on dry matter yield of a Lolium perenne pasture.

Both, urea and sodium nitrate induced an increase in plant N content as the rate of N application was raised ($p<0.05$). Total nitrogen uptake by leaves was slightly higher with N broadcasted as urea than as sodium nitrate, as pasture often exhibits higher NH_4-N uptake than NO_3-N, when both ions are evenly supplied to soil (Clarkson *et al.*, 1986).

Table 1. Shoot nitrogen concentration (g kg^{-1}) on each harvest of Lolium perenne pasture.

	Cut I	Cut II	Cut III
		g kg^{-1}	
Control	16.4b	24.1c	26.0c
150 urea	19.8ab	33.2b	38.2ab
300 urea	22.5a	38.1a	42.0a
150 S nitrate	17.3b	31.7b	35.2b
300 S nitrate	21.9a	32.7b	41.0a

Column means followed by the same letter are not significantly different at 0.05 probability level.

In late spring (November) the addition of N fertilizer raised soil NO_3-N content in the 0-10 cm of depth, and the highest concentration was obtained with 300 kg N ha^{-1} as sodium nitrate (Figure 2a). The decrease in soil NO_3-N in the 0-10 cm during the first half of November could be explained by plant uptake and N leaching produced by irrigation. In this period, the decrease in soil NO_3-N concentration in the 10-20 and 20-40 cm of depth (Figure 2a)

can be attributed to water flux downward soil profile. In early summer (December) plant uptake decreased due to a lower growth of ryegrass and the data clearly show that when sodium nitrate is applied, the higher NO_3^- availability induces a greater potential for N leaching, and therefore lower N fertilizer use efficiency.
Soil NH_4-N levels (0-10 cm) increased only in the treatment with the higher urea rate applied (Figure 2b). An increase in the NH_4-N concentration in deeper soil stratum (10-20 and 20-40 cm) was observed after each split of urea fertilizer. However, the high affinity of NH_4^+ with negatively charged surface diminished the NH_4-N losses. Temporal decrease or fluctuations in soil N-NH_4 might be explained by plant uptake and nitrification from urea input treatment. The immobilization by soil microorganisms could also explain the behavior of NH_4^+.

Figure 2. Variation of (b) NO_3-N and (b) NH_4-N concentration in the soil profile through the evaluation period at different rates of N application as urea and sodium nitrate.

Conclusions

Dry matter yield arose as the N supply increased from 0 to 300 kg N ha^{-1}. There was no effect of NH_4-N or NO_3-N sources on grass yield, and both N sources increased efficiently shoot N content. The highest rate of N application yielded the highest NO_3^- and NH_4^+ concentrations in the deepest soil layers, because the slow growth rate of the pasture leached N download the soil profile, when the soil was treated with sodium nitrate and urea, respectively.

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N mineralisation from grape residues

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Abstract

A long-term laboratory incubation experiment was performed with residues from hand-picked (complete clusters) and machine-harvested (destemmed) grapes applied to three different vineyard soils. 16% of the total N of the pomace made from hand-picked grapes was released on average from a slowly decomposing pool (half-life period t_{50} = 703 d), while 23% of the total N was estimated to be released from the same pool of destemmed grape pomace (t_{50} = 781 d). A rapidly decomposing pool could not be determined. Rapid decomposition was inhibited presumably because of high polyphenol contents in the pomace, causing a temperature-dependent delay. The net N mineralisation of complete grape cluster residues was retarded on average for 12.4 days (10°C), whereas destemmed grapes started to release N on average 5.5 days (10°C) later than the control soil. At higher temperatures the inhibitory effect was much smaller.

Key words: crop residues, grape, mineralization, nitrogen

Background and objectives

The waste products of winemaking are often deposed in the vineyard from which they have been removed earlier. At the same time, winegrowers are aware of the nutritional value of both the pressing residues (pomace) and the yeast and fine particles that sediment in the must after pressing (dregs). Pomace and dregs contain more than 90% of the N that was removed from the vineyard with the grapes. The utilisation of grape residues as a basic fertiliser is still very common in viticulture (Bertran *et al.*, 2004). For this reason the consideration of N mineralisation from grape residues is necessary when offering a fertiliser recommendation. This is particularly important, because pomace is often only applied to a limited area in the vineyard. If a simulation model for N dynamics in viticulture (Nendel and Kersebaum, 2004) is used for supporting fertiliser recommendations, N mineralisation parameters for grape pomace need to be provided to the user. To do this, a long-term *in-vitro* incubation experiment was carried out with two different kinds of white grape pomaces: residues from whole grape clusters that are manually cut from the shoots and residues from single grape berries that are shaken off the vine with a mechanical harvester.

Materials and methods

From the pressing residue of white Riesling grapes, harvested manually in October 2003, grape skins, seeds and stalks were separated (Table 1). 10 g soil aliquots (duplicates) of a control treatment without any added organic matter and a treatment containing grape pomace equivalent to 30 Mg ha^{-1} were prepared. The pomace equivalent was designed according to the ratio of stalks, grape skins and seeds found in the original sample, presuming two seeds a berry in average. In addition, another treatment was prepared containing only grape skins and seeds, representing the residues of mechanically harvested grapes. This set-up was prepared for three different vineyard soils.

The substrates were incubated in plastic syringe bodies of 60 ml volume for 420 days at 4°C, 20°C, 28°C, and 36°C, respectively, following the method of Stanford and Smith (1972).

Table 1. Properties of the grape residues.

Residues	C_{org} (mg kg^{-1})	N_{tot} (mg kg^{-1})	Mass fraction (kg kg^{-1})
Stalks	428.0	11.0	0.24
Seeds	523.5	15.0	0.24
Skins	448.0	18.1	0.52

A two-pool model with a temperature-dependent rate coefficient was fitted to the observed data.
Following the idea of Rahn and Lillywhite (2001), a delay factor h(T) was introduced to describe the temperature-dependent delay of mineralisation, as it was observed in the experiment (Equation 1).

$$N(t,T) = N_{fast} \cdot \left(1 - exp\left(-a \cdot exp\left(-\frac{b}{T+273}\right) \cdot \left(t - \frac{h}{T}\right)\right)\right) + N_{slow} \cdot \left(1 - exp\left(-c \cdot exp\left(-\frac{d}{T+273}\right) \cdot \left(t - \frac{h}{T}\right)\right)\right)$$

N(t,T)	mineralised N depending on time t and temperature T
N_{fast}, N_{slow}	parameters representing the size of rapidly and slowly decomposing N pools
a, b, c, d	Arrhenius parameters
h	delay factor

Results and discussion

52±19 mg N kg^{-1} could be released from a slowly decomposing pool of the residues of handpicked grapes and 87±30 mg N kg^{-1} from the residues of machine harvested grapes. A rapidly decomposing pool could not be determined. Rapid decomposition was inhibited, observed as a temperature dependent delay of N mineralisation (Figure 1). Mineralisation of complete grape cluster residues was delayed on average 5.5 days (10°C), destemmed grapes started to mineralise on average 12.4 days (10°C) later than the control soil (Table 2). At higher temperatures the inhibitory effect was much smaller. The different impact of the two kinds of added residues on the N mineralisation pattern leads to the assumption that high polyphenol contents in the grape residues rather than the C to N ratio of between 28 (destemmed grapes) and 30 (complete clusters) are responsible for the observed effects. Polyphenols are known to inhibit N mineralisation is soil (Schimel *et al.*, 1996). Grape stalks contain a considerably higher amount of polyphenols than the grape skins (Ribéreau-Gayon *et al.*, 1998).

Figure 1. Temperature dependency of the initial delay of net N mineralisation, exemplarily shown for pomace of complete grape clusters (po_1) incorporated in the Nierstein soil.

Table 2. Parameter estimates derived from fitting Equation 1 to the experimental data. The fit was biased by fixing the rapidly decomposing pool to zero. The rate coefficients k_{slow} are given for a temperature of 36°C.

Soil	Treatment	h (°C·d)	N_{slow} (mg kg^{-1})	k_{slow} (d^{-1})
Ruppertsberg	Control	0.0 ± 73.8	92.8 ± 2.9	7.47E-03
Ruppertsberg	st + sk + sd	75.9 ± 41.2	146.0 ± 3.4	6.45E-03
Ruppertsberg	sk + sd	41.9 ± 36.5	169.4 ± 3.6	6.39E-03
Nierstein	Control	0.0 ± 62.2	195.4 ± 6.0	7.35E-03
Nierstein	st + sk + sd	172.4 ± 28.8	266.2 ± 5.8	5.13E-03
Nierstein	sk + sd	34.4 ± 49.3	315.6 ± 14.6	3.77E-03
Bad Kreuznach	Control	0.0 ± 53.2	145.3 ± 4.3	6.70E-03
Bad Kreuznach	st + sk + sd	102.4 ± 33.1	178.2 ± 3.8	5.99E-03
Bad Kreuznach	sk + sd	74.9 ± 27.9	209.0 ± 3.6	5.58E-03

st = stalks, sk = berry skins, sd = grape seeds.

Conclusions

Although they have a similar C to N ratio, the residues of complete grape clusters and the residues of destemmed grapes show a distinct difference in the N mineralisation pattern. An observed lag phase was significantly longer in grape pomace containing stalks (typical for hand-picked grapes) compared to grape pomace containing only grape skins and seeds (typical for machine-harvested grapes). Grape stalks contain a much higher concentration of polyphenols that are held responsible for the observed initial inhibition of the N mineralisation process and also for the slower and less effective release of N. This once more underlines the assumption that N mineralisation is often governed by impact factors that can not be reduced to the plain availability of N in the decomposing material.

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Grassland management influence on soil quality assessed through several hydrolytic enzymes of the nitrogen cycle

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Abstract

Soil management can affect greatly soil quality. Soil enzymes are highly sensitive to disturbance in the environment, so they have been widely used to evaluate soil quality, despite the lack of accepted methodological standards. On the other hand, grassland management in most areas depends on the input of large amounts of cattle slurry amendments. We investigated the activities of three hydrolytic enzymes of the nitrogen cycle (BAA-protease, casein-protease, urease) in grassland soils to investigate whether fertilizer input could affect soil quality. Three different locations were chosen with different climatic parameters (Rodeiro, Boimorto, Trabada). In each location two soils were sampled, one unfertilized and another one fertilized. Data were collected monthly over 14 months. Differences due to soil management were found between each pair for urease content, although no important alteration in the nitrogen cycle has been found.

Keywords: grassland, long-term experiment, management, nitrogen cycle, soil enzymes, soil properties

Background and objectives

At the moment it has been confirmed that soil management must be as sustainable as possible, this means that it has to be able to increase the productivity, as well as avoiding an appreciable disturbance on soil and environment (Doran and Parkin, 1994). This idea involves the need to evaluate the impact of agricultural practices on soil quality and, although the term 'soil quality' comprises a lot of properties. Currently researchers tend to use biochemical properties to evaluate soil quality, because they show a rapid response to degrading factors (Gil-Sotres *et al.*, 2005). Biological attributes of soil quality include the many soil components and processes related to organic matter cycling, such as total organic carbon and nitrogen, microbial biomass, mineralizable carbon and nitrogen, enzyme activities and soil fauna and flora (Gregorich *et al.*, 1997).

Galicia (NW Spain) is an area with a high density of livestock, consequently a large amount of grasslands, whose productivity are maintained mainly through intense cattle slurry fertilization and occasionally through mineral fertilization, has been developed. In this work we search for alterations in the nitrogen cycle due to soil management. For this purpose some hydrolytic enzymes of the nitrogen cycle (urease, BAA-protease, casein-protease) were compared between grasslands with high-level fertilization and grassland with low-level fertilization with the aim of finding out management influence over several enzymes considered soil quality indicators (Doran and Parkin, 1994).

Material and methods

Three pairs of grasslands in different areas of Galicia (Rodeiro, Boimorto, Trabada), each pair consisting on a natural grassland (NF) and a grassland with a high level of fertilization (F), were investigated across a period of time of 14 months, starting the trial in September 2003 and finishing in November 2004.

Table 1. Location, site characteristic and climate parameters of the soil samples studied.

Sample	Longitude (W)	Latitude (N)	Altitude (m.a.s.l.)	Parent material	Mean annual temperature (ºC)
Rodeiro F	7º 58´ 17´´	42º 41´ 27´´	620	Schists	10.9
Rodeiro NF	7º 58´ 17´´	42º 41´ 27´´	620	Schists	10.9
Boimorto F	8º 07´ 28´´	43º 02´03´´	500	Schists	13.2
Boimorto NF	8º 08´ 10´´	43º 01´40´´	390	Schists	13.2
Trabada F	7º 10´ 42´´	43º 24´ 38´´	390	Slates	14.0
Trabada NF	7º 10´ 42´´	43º 24´ 38´´	390	Slates	14.0

The analyses were carried out monthly. In all of the cases the top 10 cm of soils were taken for analysis of total carbon, total nitrogen, pH and some of the hydrolytic enzymes of the nitrogen cycle: Casein-protease, BAA-protease and urease. Field-moist soil samples were sieved through a 4-mm screen and a portion of this was air-dried and sieved through a 2-mm screen for carbon, nitrogen and pH analysis.
Total carbon content was determined by wet oxidation (Guitián and Carballas, 1976), while nitrogen content was determined using a Kjeldahl oxidation method following Guitián and Carballas (1976). pH was determined by using a combination glass electrode (soil:water ratio = 1:2.5) (Guitián and Carballas, 1976). Particle size distribution was determined using a Robinson pipette with Calgon as dispersant (Guitián y Carballas, 1976). The activity of urease was determined as described by Nannipieri *et al.* (1980), using urea as substrate and incubating for 1.5 h at 37º and pH 8.0 and measuring the NH_4^+ released with an ammonium electrode. BAA-protease activity was determined using the same incubation conditions but with α-benzoyl.N-argininamide (BAA) as substrate (Nannipieri *et al.*, 1980). In both cases enzymatic activity is expressed in μmol NH_3 g^{-1} h^{-1}. Casein-protease activity (Nannipieri *et al.*, 1979) was determined with casein as substrate, incubating for 2 h at 50º C and pH 8.1 (Tris-HCl buffer 0.05 M) and determining the amino acids released by the Folin colorimetric method; enzymatic activity is expressed in □mol tyrosine g^{-1} h^{-1}. All analyses were carried out 15 days after soil sampling at latest. When not in use, the field-moist samples were stored at 4º C.

Results and discussion

Although fertilized grasslands show a lower level of total carbon and total nitrogen than natural grasslands, those also exhibit a lower C/N ratio. This reveals that fertilization leads to a relative increase in total nitrogen.

Table 2. General characteristics of the samples studied.

Sample	Carbon (g kg^{-1})	Nitrogen (g kg^{-1})	C/N ratio	pH (water)	Texture
Rodeiro F	7.75±0.81	0.595±0.042	13±2	5.46±0.21	Sandy-clay-loam
Rodeiro NF	10.00±0.76	0.783±0.042	13±2	5.08±0.16	Sandy-clay-loam
Trabada F	4.39±0.84	0.350±0.032	13±2	5.00±0.21	Loam
Trabada NF	7.73±0.99	0.571±0.055	14±1	5.13±0.16	Loam
Boimorto F	6.36±1.35	0.499±0.091	13±2	5.28±0.17	Sandy-loam
Boimorto NF	7.32±1.29	0.495±0.118	15±1	5.37±0.25	Loam

BAA-protease and casein-protease activity levels don't show differences in both groups of grasslands, being the soils in Trabada an exception, nevertheless, an important decrease was found in urease activity in fertilized grasslands. It must be pointed out that Trabada has a parent material different from the other two locations. These data suggest that the elevated content in inorganic nitrogen in cattle slurry inhibits soil urease activity. However, it must be pointed out that no important alterations took place in enzymes of the nitrogen cycle in the studied grasslands; these show a similar casein-protease/BAA-protease ratio to Galician soils developed over climax vegetation (Trasar-Cepeda *et al.*, 2000).

Table 3. Enzyme activities of the samples studied.

Sample	BAA-protease[1]	Casein-protease[2]	Urease[1]
Rodeiro F	38.86±12.55 a	1.41±0.25 a	76.34±32.42 a
Rodeiro NF	48.70±14.63 a	1.65±0.29 a	124.09±64.49 b
Trabada F	20.18± 5.98 a	1.03±0.27 a	23.58± 5.17 a
Trabada NF	36.42± 9.84 b	1.79±0.43 b	69.12±31.21 b
Boimorto F	20.28± 4.47 a	0.91±0.20 a	20.50±11.09 a
Boimorto NF	29.28± 8.26 a	1.43±0.29 b	63.82±26.83 b

[1] *$\mu mol\ NH_3\ g^{-1}\ h^{-1}$.*
[2] *μmol tyrosine $g^{-1}\ h^{-1}$.*
Same letters within a location and enzyme means that activities are not significantly different $P \leq 0.05$.

Conclusions

This work suggests that grassland management, in both cases, with a high input or with a low input of organic fertilizers, maintains soil biochemical quality. Further research is needed to assess if soils developed over schists have different biochemical behaviour than those developed over slates.

Acknowledgements

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Potential microbial N transfer

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Abstract

The potential N immobilization rate of a soil bacterial community was investigated in a laboratorial experiment. The amount of N immobilized by the microbial biomass in conditions which were not C limited depend on the growth rate of the microorganisms and their C- to N- ratio. Both factors were measured under optimal growth conditions with increasing amounts of NH_4NO_3 (0, 50 and 100 mg N l^{-1}). The microbial growth was four times higher in the 100 mg N l^{-1} fertilized treatment compared to the control without N and the C- to N- ratio varied between 6.1 and 8.6. If we take these fluctuations into account and calculate the potential N immobilization rate of the microbial community in a diluvial sandy soil, between 26 and 148 kg N ha^{-1} d^{-1} could be microbially immobilized. In which time course the microbial immobilized N will be remineralized or transferred into the soil organic carbon pool is not yet understood. Models describing N transfer rates in soils could probably be improved when these microbial immobilization rates would be included.

Keywords: C:N ratio, immobilization, microbial biomass, Microbial N transfer, nitrogen

Background and objectives

Microbially mediated N conversion rates can be extremely high under favourable conditions, e.g. after N fertilization or the application of easily decomposable organic material and particularly within the C rich rhizosphere soil. The importance of the soil microbial community composition and activity in gross N transfer processes is poorly understood yet. The temporarily immobilization of high N values within the soil microbial biomass in rhizosphere soil can probably explain the deviation of field N measurements from model based predictions (Magid *et al.*, 1997).

Material and methods

The potential N immobilization rate of a soil microbial community was investigated in a laboratorial experiment. The amount of N immobilized by the microbial biomass depend on the C-availability, the bacterial growth rate (measured as increase in protein content per day) and the C-to-N ratio of the microbial biomass. Microbial growth and C-to-N ratio were measured under optimal growth conditions in liquid culture (0.1 g K_2HPO_4, 0,1 g $CaCl_2 \cdot 6\ H_2O$, 0.3 g $MgSO_4 \cdot 7\ H_2O$, 0.1 g $FeCl_3 \cdot 6\ H_2O$ per 1 l destilled water) with increasing amounts of NH_4NO_3 (0, 50 and 100 mg N l^{-1}) and increasing glucose concentrations (0, 1500 and 3000 mg C l^{-1}). The microbial fraction was separated from a diluvial sand according to Ruppel and Augustin (1998) and two times washed in sterile 0.05 M NaCl solution by centrifugation (2450 x g at 5 °C for 30 min) to remove all adhering nutrients. 100 µl of the microbial fraction were inoculated into 40 ml liquid medium in three replicates, cultivated at a horizontal shaker (150 rotations per min) at 28 °C for 48 h. Sterile controls without microorganisms were added. After 48 h growth protein content was measured in a 1 ml aliquot according to Lowry *et al.* (1951) and the other microorganisms were harvested by centrifugation (2450 x g at 5 °C for 30 min) and dried at 60 °C. Then dry matter content was measured and total carbon and total nitrogen contents were determined using the elemental analyser CHN-O-Rapid (Heraeus).

Results and discussion

In C-rich conditions (3000 mg C l^{-1}) the bacterial growth increased four-fold after 100 mg N l^{-1} application compared to the unfertilized control (Figure 1A). Additionally the C-to-N ratio varied between 8.6 in the 0 N treatment and 6.1 in the 100 mg N l^{-1} treatment (Figure 1B).

Figure 1. Microbial dry matter production within 48h liquid culture (A) and C- to N- ratio within the microbial biomass (B) with increasing mineral N supply in C-rich conditions (3000 mg C l^{-1}).

If we take these fluctuations into account and calculate the potential N immobilization rate of the microbial community of a diluvial sandy soil (100 µg C_{mic} g^{-1} soil), between 26 and 148 kg N ha^{-1} d^{-1} could be microbially immobilized. This huge amount of N could be transferred within one day if C is not limited (3000 mg C l^{-1}) and the growth conditions for the bacterial community are set optimal at 28 °C and saturated oxygen concentrations.

Figure 2. Potential microbial N immobilization rate in C rich conditions depending on the N fertilization level and the C- to N- ratio within the soil microbial biomass.

These C enriched conditions occur within the root exudates which can amount from 30% to 70% of annual plant net C fixation values (Merbach *et al.* 1999). 40 to 90% of these C values were decomposed by root respiration activity or by associated microorganisms (Lynch and Whips 1990). Under such conditions the mineral nitrogen which is fertilized to young growing plants can be temporary completely immobilized into the soil microbial biomass within the first two to three days after fertilization. How fast these N values will be remineralized or be transferred into the soil organic carbon pool is not yet understood.

Conclusions

Since microbial gross N transfer rates in soils are still very difficult to measure, the knowledge about the potential activities of the soil microbial community to transfer mineral nitrogen into different N pools, such as soil organic matter, microbial biomass N, gaseous N or other mineral N compounds is very poor. Model experiments can improve our understanding of potential microbial activities.

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Nitrogen losses to surface water on a heavy clay soil in the Netherlands

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Abstract

Losses of nitrogen to surface water were measured from autumn 2002 to spring 2005 at a grassland plot on a heavy clay soil in the Netherlands. The water balance showed that ditches were the dominant pathway for water and nutrients in this system. Annual nitrogen losses ranged form 13 kg/ha/year in the dry winter of 2002/2003 to 19 kg/ha/yr in the other two winters. Incidental losses of nitrogen after manure application strongly influenced the annual nitrogen losses. In wet years the loss of nutrients in autumn by drains and ditches contributes substantially to the yearly losses.

Keywords: ditches, drain water, leaching, losses, nitrogen

Background and objectives

Diffuse pollution of groundwater and surface water with nitrogen is a serious problem in the Netherlands. As a result of the EU-Nitrates Directive, the pollution of groundwater with nitrogen has been extensively studied (Fraters *et al.*, 2004). Relatively little attention has been paid to the pollution of surface waters disregarding the fact that the relative contribution of agriculture to the pollution of the surface water with nitrogen has increased from 43% in 1985 to 57% in 2002. To obtain more information on the pollution of surface waters by dairy farms, a number of measurement projects has been set up at the end of the nineties on a sandy soil, a peat soils and a clay soil. This study focuses on the losses of nitrogen from a dairy farm on a clay soil.

Materials and methods

The losses of nitrogen to surface waters were measured from autumn 2002 to spring 2005 on a grassland plot at a dairy farm. The plot had a size of 3.2 ha and the grass was alternately mowed and grazed. The total N application by fertilizer and manure was 463 kg/ha/yr. The average N surplus was 107 kg N/ha/yr and declined from 220 kg/ha/yr in 2003 to 49 kg/ha/yr in 2004.

The plot was located on a heavy clay soil (57% < 2µm; 40% 2-50 µm) The site was level and the soil was drained by subsurface drains and drainage ditches. The subsurface drains were located at a depth of 80 cm below the surface. The ditches were shallow (50 cm depth) and located at intervals of 46 m. Both the subsurface drains and the ditches fed a small channel that surrounded the plot.

To construct a water and nitrogen balance for the plot the amount and composition of the discharge from the drains, ditches and channel were continuously monitored. Water samples were collected automatically depending on discharge. Bulked samples were analysed every week for NO_3, NH_4, Kjeldahl-N, PO_4 and total-P.

Results

Discharge of water and nutrients from the plots was generally limited to the winter period (October to April). The average precipitation surplus in winter over the three year measurement period was 290 mm. The average discharge of the channel (296 mm) was almost equal to the precipitation surplus in winter (Table 1). The discharge from the drains and ditches to the channel was slightly (10%) lower. This difference may be due to overland flow from the plot or from a small road adjacent to the channel. Most of the water (60%) was carried to the channel by

means of the drainage ditches (Table 1). The drains contributed to the drainage by 30%. Discharge to groundwater was negligible due to the extremely low conductivity of the heavy clay (sub)soil.
The relative contribution of drains and ditches to the discharge changed considerably during the season. In autumn, when cracks are present in the heavy clay soil, drains contributed significantly to the monthly discharge (30-50 mm/ month). During the winter, when cracks are closed due to swelling of the clay soil, the drainage reduces to values of 10-15 mm/month. In this period the ditches play a dominant role in the discharge. The importance of cracks for the functioning of the drains is also clearly shown when the results for 2003 and 2004 are compared. The summer of 2003 was dry and the discharge of the ditches remained relatively high until February leading to a total discharge of 90 mm. In the wet year 2004, the discharge by the drains was already strongly reduced in December and the total discharge was only 70 mm.

Table 1. Average water and nitrogen fluxes and concentrations in drains, drainage ditches and channel over the years 2002/2003, 2003/2004 and 2004/2005.

	Water fluxes (mm)	Flux (kg/ha)		Concentrations (mg/l)	
		Total N	Organic N	Total N	Organic N
Drains	85	5	4	5.6	4.2
Drainage ditches	183	12	9	6.3	4.6
Drains +ditches	268	17	12	6.1	4.5
Channel	296	13	10	4.1	3.1

The nitrogen discharge by the channel was on average 13 kg N/ha/yr (Table 1). Most of the nitrogen was in an organic form or present as colloids. Inorganic nitrogen consisted for approximately 50% of NO_3 and 50% of NH_4. De discharge of the drains and ditches was higher (17 kg N/ha/yr) than the discharge of the channel, due to a higher content of NO_3 (1.3 kg/ha/yr) and organic/colloidal N (2.8 kg N/ha/yr).

Figure 1. Discharge of total N by drains and ditches in the winter period (October-April) of the years 2002/2003, 2003/2004 and 2004/2005.

The discharge of nitrogen by drains and ditches varied considerably form year to year (Figure 1). The year 2002/2003 was rather dry (total discharge of 243 mm) and there was no discharge from February onwards. In 2003/2004 and 2004/2005 discharge of water and nutrients was considerably higher (approx. 280 mm, and 19 kg N/ha/yr). The higher nitrogen losses in 2003/2004 are partly caused by rainy weather (31 mm) in the week

following manure application in March. The rainfall led to a discharge of 19 mm of water and 9.4 kg N/ha by the ditches, accounting for 50% of the annual nitrogen loss in 2004. In 2004/2005 a similar incident took place. However, due to lower rainfall the nitrogen losses were only 3 kg N/ha/yr. Despite this the nitrogen losses in autumn 2004 were high compared to 2002, due to the higher water discharge in autumn.
During the three year period the N-surplus at the field decreased from 220 – 49 kg ha^{-1} yr^{-1}. This decline in N – surplus did not lead to a decline in N leaching during the observation period. Moreover the potential decline in N leaching upon a reduction in yearly N application rates will be limited as a substantial part of the N leaching is caused by incidental losses.

Figure 2. Total nitrogen concentrations in channels, ditches and drains during the monitoring period.

Average nitrogen concentrations in the channel were well above the Maximum Tolerable Risk (MTR- values) for stagnant surface waters in the Netherlands during the summer period (2.2 mg N/l). Nitrogen concentrations in the channel were generally quite close to the concentrations in the ditches. Concentrations in the drains are generally somewhat lower. Nitrogen concentrations in the discharge water were highest in autumn and spring and lowest during mid-winter (Figure 2). The high spring concentrations were closely linked to manure applications in spring. For example in spring 2004, concentrations of more than 40 mg/l were measured following manure application.

Conclusions

Discharge of water en nutrients on this heavy clay soil was limited to the winter period. The ditches played a dominant role in the drainage of water and the loss of nutrients from the plot. The contribution of the drains was strongly dependant on the presence of cracks in this heavy clay soil. The total annual loss of nitrogen is strongly influenced by incidental losses of nitrogen after manure application in spring. In wet years substantial losses of nutrients occur in autumn through both drains and channels.

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Evidence for widespread fungal denitrification in agricultural soils

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Abstract

Many fungi can denitrify, and as most lack N_2O-reductase, the major product of fungal denitrification is N_2O. The microbial biomass of temperate soils is often dominated by fungi, the fungal to bacterial ratio being affected by the land use and management practices. Fungal denitrification has previously been shown to be important in woodland soils and an intensively managed grassland soil. In this study the substrate-induced respiration inhibition method with cycloheximide as the fungal inhibitor was used in combination with the ^{15}N gas-flux method to determine the relative importance of fungi and bacteria to N_2O production in ten soils under different management histories (deciduous forest, coniferous forest, grazed grassland at three intensities, cut grassland at three intensities, barley, potatoes). The N_2O flux was depressed by cycloheximide in all soils, fungal processes being responsible for 51% of the N_2O emissions on average over all soils. All of the N_2O was derived from the reduction of NO_3^- and not from nitrification. Since fungi can perform aerobic respiration and denitrification simultaneously under specific conditions of O_2 stress, fungal denitrification could be widespread in the partially-aerated microsites of aggregated soils while bacterial denitrification proceeds in the anoxic zones.

Keywords: cycloheximide, denitrification, fungi, N_2O, respiration

Background and objectives

Denitrification has now been identified as a function of eukaryotes, including yeasts (Tsuruta *et al.*, 1998) and filamentous fungi (Shoun *et al.*, 1998) as well as prokaryotes. Shoun *et al.* (1998) identified a number of fungi with denitrifying ability, but behave differently to bacterial denitrifiers. Unlike bacterial denitrification were N_2 is the major end product, many fungi lack N_2O reductase so the major end product is N_2O (Shoun *et al.*, 1992).

By adding a bacterial inhibitor (streptomycin) and a fungal inhibitor (cycloheximide), Anderson and Domsch (1975) developed the substrate-induced respiration inhibition (SIRIN) method to measure the relative bacterial and fungal contributions to the respiring soil biomass. Castaldi & Smith, (1998) observed that low cycloheximide concentrations drastically reduced N_2O emissions in woodland soil with little effect on arable soil. Laverman *et al.*, (2000) also found high pH and presence of fungi to be factors for high denitrification rates in woodland soils. Laughlin and Stevens (2002) found that nearly 90% of the N_2O was being produced by fungi in a grassland soil. The microbial biomass of temperate soils is often dominated by fungi, the fungal to bacterial ratio being affected by the land use (Anderson and Domsch, 1975) and management practices (Frey *et al.*, 1999). Fungi may therefore make a significant contribution towards the global N_2O budget from soils.

In this study the role of fungi for N_2O production in ten soils with different management histories was determined using the SIRIN method and the ^{15}N gas-flux method (Mosier and Schimel, 1993). The SIRIN method using cycloheximide as the fungal inhibitor allowed the relative contributions of fungi and bacteria to N_2O production to be determined. Labelling of the NO_3^- pool in the ^{15}N gas-flux method enabled the origin of the N_2O to be determined.

Materials and methods

Soil samples were collected at a depth of 5-15 cm from ten sites in October 2004. The management history of the sites and properties of the soils are shown in Table 1.

Table 1. Management histories and soil properties at the ten sites in Northern Ireland.

Site	Management history	pH	Sand (%)	Silt (%)	Clay (%)	Total C (g kg^{-1})	Total N (g kg^{-1})
1	Forest, deciduous	4.2	46	38	16	175	8.8
2	Forestry, coniferous	4.7	42	32	26	75	5.2
3	Grazing, organic	5.8	23	30	45	57	5.9
4	Grazing, 150 kg N $ha^{-1}y^{-1}$	5.8	82	11	7	21	2.0
5	Grazing, 250 kg N $ha^{-1}y^{-1}$	6.2	53	25	21	33	3.2
6	Silage, 2-cut, organic	5.6	22	20	59	83	8.4
7	Silage, 1-cut, 150 kg N $ha^{-1}y^{-1}$	6.1	59	16	25	32	3.1
8	Silage, 3-cut, 300 kg N $ha^{-1}y^{-1}$	6.1	43	36	22	39	3.6
9	Arable, barley	5.7	79	12	9	30	2.5
10	Arable, potatoes	5.3	80	11	9	28	2.3

Air-dried, sieved (2 mm) soil was sub-sampled into 500 ml Kilner jars to supply 100 g soil on an oven-dry basis. Cycloheximide at 6 mg g^{-1} oven-dry soil was applied in 30 ml of aqueous solution. An equivalent volume of distilled water was applied to the control samples. Glucose and labelled $NH_4{}^{15}NO_3$ were added together to each jar in 5 ml of aqueous solution to supply glucose at 5 mg g^{-1} oven-dry soil and $NH_4{}^+$-N and $NO_3{}^-$-N at 7.14 µmol N g^{-1} oven-dry soil. The $NO_3{}^-$ was enriched to 40 atom % excess in ^{15}N. Additional water was added to each soil to attain field capacity. There were three replicates of each treatment arranged randomly in an incubator at 22°C. At 6 hours after the addition of glucose, the jars were sealed with gas-tight lids fitted with gas sampling ports. After 2 hours, two 12 ml samples of each headspace were transferred to evacuated septum-capped vials for analysis of CO_2 by gas chromatography and N_2O by isotope-ratio mass spectrometry. Further details of the laboratory procedures were as described by Laughlin and Stevens (2002).

Results and discussion

Previous studies have shown that the basal respiration rate is measured at 6 h during the lag phase before exponential microbial growth occurs in response to glucose. Basal respiration rates varied widely between soils from 0.21 µmol C $g^{-1}h^{-1}$ in the arable soil after potatoes (Soil 10) to 1.80 µmol C $g^{-1}h^{-1}$ in the soil managed organically for silage (Soil 6).

Table 2. Basal respiration rates and proportion of that respiration due to fungi in ten soils.

Soil	1	2	3	4	5	6	7	8	9	10
Basal respiration rate (µmol C $g^{-1}h^{-1}$)	0.84	0.36	1.23	0.27	0.56	1.80	0.68	0.43	0.35	0.21
Proportion of respiration due to fungi (%)	21	29	17	24	11	27	25	23	36	26

Over all soils, the proportion of fungi in the biomass calculated from the observed respiratory inhibitions was 24%, indicating that bacteria dominated these soils. In a review of studies in which the proportion of fungi and bacteria had been measured, Ruzicka *et al.* (2000) concluded that fungi often dominate temperate soils. In this study the same rate of moisture content, glucose and cycloheximide was used for all soils. Optimal soil moisture content, glucose concentration, and cycloheximide concentration should have been determined for each soil in preliminary experiments to comply with the selective inhibition criteria of Anderson and Domsch (1975). A previous study with Soil 8 found that cycloheximide at 15 mg g^{-1} was needed for proper fungal inhibition, so the rate used in this study (6 mg g^{-1}) may not have been too low, hence the proportion of the fungal biomass may have been underestimated. The N_2O flux varied widely between soils (Table 3) from 0.5 nmol N $g^{-1}h^{-1}$ in the coniferous forest soil (Soil 2) to 67.9 nmol N $g^{-1}h^{-1}$ in the organic-farmed grazed grassland soil (Soil 3).

Table 3. Rate of N_2O production and proportion of N_2O due to fungi in ten soils.

Soil	1	2	3	4	5	6	7	8	9	10
N_2O production (nmol N $g^{-1}h^{-1}$)	0.9	0.5	67.9	11.5	44.3	2.9	41.1	19.7	8.5	2.5
Proportion of N_2O due to fungi (%)	12	26	56	60	54	45	51	42	81	81

The effect of cycloheximide was highly variable and the proportion of the N_2O due to fungi ranged from 12% in Soil 1 to 81% in Soils 9 and 10, the two arable soils. Fungal processes were responsible for 51% of the N_2O emissions on average over all soils. The equations of Arah (1997) were used to calculate the fraction (d'_D) of the N_2O flux which was derived from the labelled NO_3^- pool and the ^{15}N mole fraction (a_D) of that pool. Values of d'_D were not significantly different from unity, so all of the N_2O was derived from the NO_3^- pool. The values for a_D agreed well with the calculated time-zero value for the enrichment of the labelled NO_3^- pool. All of the N_2O was being formed by reduction from NO_3^- and none from oxidation from NH_4^+.

Conclusions

Fungal denitrification may be an important mechanism for NO_3^- reduction in many agricultural soils. Fungi have a unique feature in being able to perform aerobic respiration and denitrification simultaneously due to their ability to use NO_3^- as an alternate electron acceptor to O_2 for respiration (Zhou *et al.*, 2001). Fungal denitrification is therefore likely to occur in the partially-aerated microsites of aggregated soils while bacterial denitrification proceeds in the anoxic zones.

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Long-term field experiment and modeling of nitrate leaching: a case study of irrigated maize in southern Bulgaria

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Abstract

The influence of the main natural and anthropogenic factors on nitrate leaching under irrigated maize grown on Fluvisols in Southern Bulgaria is characterized using long-term (1972-2004) experimental data and NLEAP model simulations (Shaffer *et al.*, 1991). The experimental design included four treatments with different nitrogen (N) and phosphorus fertilizer rates compared with non-treated plots. Sprinklers irrigation was scheduled to ensure a non-deficit water regime for the crop growth. Experimental data on N leaching were obtained by modified Ebermayer-Shilova type of lysimeters.

The tests of NLEAP model showed good agreement (R^2=56÷75%) between measured and simulated output parameters. The average NLEAP simulated drainage is between 70 and 87 mm for all treatments. The variation coefficients of the annual leachate volumes for the period 1972-2004 are high (50-60%). Increasing the optimum rates by 25% leads to near double increasing of the quantity of leached N below the root zone. The obtained information for the influence of natural and anthropogenic conditions on N leaching could be used as predictive tool for the risk analyses in vulnerable regions.

Keywords: fertiliser, irrigation, leaching, maize, nitrate, NLEAP model

Background and objectives

Information about nitrate nitrogen (N) leaching could be obtained using experimental research data and simulation models. Both approaches are often based on long-term data records accounting for the climate changes. The main parameters governing the N processes of irrigated maize in different agroecological regions of Bulgaria, have been monitored since 1972. The purpose of this study is to characterize the influence of the main natural and anthropogenic factors on nitrate-N leaching under irrigated maize grown on Fluvisols using long-term experimental data and NLEAP model simulations (Shaffer *et al.*, 1991).

Materials and methods

Field experiment with irrigated maize (*Zea mays*, L.) grown as monoculture was set up in 1972 at the field station of the Nikola Poushkarov Institute of Soil Science - village of Tsalapitsa (Lg 24°35' E, Lat 42°14' N, Alt 180 m) situated at Maritsa River watershed. Fluvisols spread over this area are characterised as sandy clay loam and main soil properties used as input parameters for the model are presented in Table 1.

Table 1. Main parameters for the upper (0-30 cm) and lower (30-90 cm) soil layers (OM – organic matter; CEC- cation exchange capacity; BD – bulk density; AWHC- available water holding capacity; WP – wilting point).

Depth, cm	Texture	pH in H_2O	OM	CEC,$cmol_{(c)}.kg^{-1}$	BD, $Mg.m^{-3}$	AWHC, v/v	WP, v/v
0-30	SL [a]	6.0	0.70	20.6	1.55	0.155	0.140
30-90	SCL [b]	6.4	0.49	23.7	1.49	0.150	0.160

[a] *SL – sandy loam.*
[b] *SCL – sandy clay loam.*

Table 2. Average and standard deviation values of some main climate characteristics (T – air temperature; P – precipitation; Eto- reference evapotranspiration).

T, °C	T, °C	ETo, mm		P, mm		Ir, mm
January	July	Oct-April	May-Sept	Oct-April	May-Sept	
0.7±2.0	23.7±1.2	245±26	566±59	265±61	232±93	265±102

Some of the main climate characteristics of the experimental site are presented in Table 2. Experimental layout consists of plots of 200 m^2 in six replications. Four variants of N and phosphorus fertilizers were applied. The rate of optimum treatment was calculated for full compensation (100%) of the N uptake by the crop production. Since 1975 an average N rate of 232 $kg.ha^{-1}$ (a. s.) for this treatment has been applied. The rates for the other treatments were 50%, 75% and 125% of the amounts of the optimum treatment. Of the fertilizer (ammonium nitrate) 2/3 was applied before sowing and 1/3 incorporated 40 days after planting. Sprinkler irrigation was scheduled to ensure a non-deficit water regime for the crop growth. Modified Ebermayer-Shilova type of lysimeters (Stoichev, 1997) to collect the leachate volumes were installed under non-fertilized, optimum and 50% reduced treatments at 1 m soil depth in three replications. N content in precipitation, irrigation, shallow groundwater (3 to 5 m depth) and residual N in the soil for all treatments were also monitored (Stoichev, 1997).
The sequential runs with the event-by-event time scale of the NLEAP were performed for the period 1972-2004. NLEAP model (version 1.13) computes water available for leaching after each precipitation and irrigation event using two-layer soil water capacity model. Soil C and N transformation in the model are confined to the upper layer (0 - 30.5 cm). The lower boundary of the second layer in this study was set to 91.5 cm, which is close to the depth of the lysimeters. In the case of non-deficit water regime, this should not lead to significant overestimation of water and hence N losses below the rooting depth. Nitrate-N leached is determined using an exponential relationship based on water and N available for leaching and soil porosity. Nitrate-N available for leaching is calculated from the N balance. The crop N uptake is simulated by a logistic growth curve adjusted for the crop yield. NLEAP model has two soil organic matter pools and one surface residue pool. Net N mineralization of the residue pool depends on its C:N ratio and decomposition rate.
The input experimental database includes: soil properties, climate data (daily precipitation, monthly air temperature and reference evapotranspiration (Allen *et al.*, 1998)), agro techniques, irrigation and fertilization management, N uptake, phenological phases and yields (Stoyanov and Donov, 1996).

Results and discussion

The tests of NLEAP model showed good agreement (R^2=56-75%) between measured and simulated output parameters (Stoichev *et al.*, 2001). The average NLEAP simulated drainage is between 70 and 87 mm for all treatments. The variation coefficients of the annual leachate volumes are high (50-60%) for all treatments. The

variations in leached nitrate-N increase with the rate of N fertilization (Table 3). The simulation of the long-term treatment with the highest rate of N fertilization, shows that a 25% increase of the N rate almost doubles the amount of leached NO_3-N below the root zone. The accumulation of residual soil N in this treatment is especially risky after dry year period (e.g. 1988). The dynamics of N budget components for vegetation and post-vegetation period through the all simulated years is illustrated with the optimum treatment case (Figure 1).

Figure 1. Dynamic of the main N budget components influencing the N leaching at fertilization rates calculated for full compensation of N uptake by maize (Nf – nitrogen (N) from fertilization, Np – N from precipitation; Nirr – N from irrigation, Nplt – N plant uptake, NL - NO_3-N leaching).

Table 3. Statistics (median±standard deviation) of the annual N budget components (measured sources and simulated sinks) and projected depth of leaching for the period 1975-2004 (Nf – nitrogen (N) from fertilization, Np – N from precipitation; Nirr – N from irrigation, Nplt – N plant uptake, NAL –NO_3-N available for leaching at the end of the year, NL - NO_3-N leaching, D – projected depth of leaching).

Variants of N rates	N_f (a.s.) kg.ha^{-1}	N_p kg.ha^{-1}	Nirr kg.ha^{-1}	N_{plt} kg.ha^{-1}	NAL kg.ha^{-1}	NL kg.ha^{-1}	D cm
non-fertilized	0	20±5	9±4	55±13	32±24	8±7	125±21
50%	116±16	20±5	9±4	180±35	31±30	14±13	122±16
75%	174±20	20±5	9±4	210±39	169±40	34±39	120±18
100%	232±24	20±5	9±4	256±48	227±64	50±52	120±17
125%	290±30	20±5	9±4	265±43	494±88	88±59	120±18

Conclusions

The long-term field data and model simulations allowed to quantify the risk for the environment from over N fertilization for the studied region. Increasing the N rate by 25% almost doubles the quantity of N leached below the root zone. The obtain information for the contribution of natural conditions and anthropogenic impact on N leaching could be used as a predictive tool for the risk analyses in vulnerable regions.

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Nitrate leaching losses from a recently developed intensive horticultural system in a previously disadvantaged region

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Abstract

The greenhouse-based vegetable production system of Almería, Spain is apparently associated with considerable NO_3^- contamination of underlying aquifers. Eighty percent of cropping occurs in soil and 20% in 'open' hydroponic systems. Nitrate leaching was measured from a pepper and tomato crop in 'open' hydroponic systems, and from two cropping sequences in a clay soil, one including a pepper crop, the other including a tomato crop. In the hydroponic systems, drainage occurred throughout the crop cycle; drainage represented 24-38% of applied nutrient solution, and had average NO_3^- concentrations of 995 and 944 mg NO_3^- L^{-1}, respectively, for the pepper and tomato crops. In the cropping sequences in soil, appreciable episodes of drainage and NO_3^- leaching (28-45 kg N ha^{-1}) occurred (a) following large irrigations given for soil management purposes (chemical disinfection, to stimulate manure decomposition), and (b) during the 4 week period following transplanting when irrigation exceeded ETc to ensure seedling survival. Nitrate concentrations in drainage from soil were generally 372-744 mg NO_3^- L^{-1}, and were >186 mg NO_3^- L^{-1} even when only water was applied. These data demonstrate that this agricultural system has large NO_3^- leaching losses, and that 'open' hydroponic systems make a disproportionate contribution to the overall loss. Because this industry has been largely responsible for recently transforming an economically deprived region into a wealthy one, it is suggested that it will be difficult to introduce management practices that appreciably reduce the on-going NO_3^- contamination of underlying aquifers.

Keywords: hydroponics, leaching, nitrate, pepper, tomato, vegetable

Background and objectives

Since the early 1980s, there has been rapid development of intensive vegetable production in greenhouses along the south-eastern coast of Spain. Climatic conditions and relative labour costs enabled earlier and cheaper production, much of which is exported to NW Europe. This greenhouse-based vegetable production system is mostly concentrated in the province of Almería, where there are 27,000 ha of highly concentrated, simple plastic greenhouses; additionally, there are several thousand hectares of greenhouses in neighbouring provinces. The recent rapid development of this industry has resulted in spectacular economic development, transforming a previously disadvantaged region into a wealthy one. This system attracts considerable international interest as a model for regions with similar climatic and initial socio-economic conditions. The rapid development of this system in Almería has coincided with a dramatic increase in NO_3^- contamination of aquifers. Appreciable areas of superficial aquifers have NO_3^- concentrations >300 mg NO_3^- L^{-1}, and these concentrations are rapidly increasing (Prof. Antonio Pulido-Bosch, Universidad de Almería, pers. com.). Approximately 80% of the area of greenhouse-based vegetable production in Almería occurs in soil, the rest in 'open' hydroponic systems. Fertigation and drip irrigation are used on all crops. N fertiliser and irrigation management are based on experience (Thompson *et. al.*, 2004). A series of studies were conducted to measure and characterise NO_3^- leaching loss from crops grown in 'open' hydroponic systems from cropping sequences in soil.

Materials and methods

Crops were grown in simple un-heated plastic greenhouses in the Cajamar 'Las Palmerillas' Research Station in Almería, Spain. All crops were grown with drip irrigation and continuous fertigation with a complete nutrient solution being applied in all irrigations. Irrigation and N management (nutrient solutions with 10-12 *mM* NO_3^- and 1.0-1.5 *mM* NH_4^+) followed local practice, unless otherwise indicated. All crops were transplanted as 6-8 week seedlings. Two crops were grown in 'open' hydroponic systems. Volumes of applied nutrient solution were measured daily, and daily samples were analysed for NO_3^- and NH_4^+ concentration. Drainage was collected daily in drainage trays, each with two 40 L bags of substrate. Sweet pepper (*Capsicum annum, L.*; cv. Vergarsa) was grown in perlite from 21 July 2004 to 6 January 2005. Drainage was collected in two replicate trays; each of the two 40 L perlite bags in each tray contained 6 plants and 5 drippers. Tomato (*Licopersicum esculetum* L.; cv. Boludo) was grown in rockwool from 6 March 2005 to 6 July 2005. Four drainage trays were used, the two 40 L bags in each tray contained 6 plants and 6 drippers. Two cropping sequences in a clay soil were followed. The soil was an 'enarenado' artificial soil typical of this horticultural system (Wittwer and Castilla, 1995), with a 30 cm layer of clay soil, imported from a quarry placed, over the natural silt loam soil, with a 10 cm layer of coarse river sand mulch placed over the imported clay layer. Drainage was collected daily from two free-draining re-packed lysimeters (4 m long x 2 m wide x 0.7 m deep). The first cropping sequence consisted of a 115 t/ha manure application (1,270 kg N ha^{-1}, which is common in this system), followed 2 weeks later with a 40 mm irrigation, a partial pepper crop (grown 25 July to 22 August 2003) and a complete pepper crop (grown 25 August 2003 to 2 January 2004). The second cropping sequence consisted of chemical soil disinfection with a total of 75 mm irrigation, a partial tomato crop and a complete tomato crop planted on (grown 27 September 2004 to 2 March 2005). In the tomato crop, only water was applied until 13 October 2004. In the soil-grown crops, irrigation was managed with tensiometers (-15 to -35 kPa) once the crops were established (by 3 weeks). In all cases drainage was collected daily, the volumes measured and the concentrations of NO_3^- and NH_4^+ determined.

Results and discussion

Of 950 kg N ha^{-1} (93% as NO_3^-) applied to the pepper crop in perlite, the NO_3^-–N leaching loss was 452 kg N ha^{-1}. Total drainage was 201 mm (from 526 mm of irrigation) with an average NO_3^- concentration of 995 mg NO_3^- L^{-1}. For the tomato crop in rockwool, from the total application of 555 kg N ha^{-1} (94% as NO_3^-) the total NO_3^- leaching loss was 162 kg N ha^{-1}. Total drainage was 76 mm (from 316 mm of irrigation) with an average NO_3^- concentration of 944 mg NO_3^- L^{-1}. From the pepper cropping sequence in soil, appreciable drainage and NO_3^- leaching occurred following the 40 mm irrigation after manure addition, and during first 4 weeks (crop establishment phase) of both the curtailed and the completed pepper crops. Nitrate leaching losses during these periods were 28, 43 and 43 kg N ha^{-1}, respectively. From the tomato cropping sequence in soil, major episodes of drainage and NO_3^- leaching occurred after disinfection (85 mm irrigation), following transplanting of the curtailed crop, and during establishment (first 4 weeks) of the completed crop; the respective losses were 45, 28, and 41 kg N ha^{-1}. During both cropping sequences in soil, the NO_3^- concentration in drainage was usually 372-744 mg NO_3^- L^{-1}, and was >186 mg NO_3^- L^{-1} even when only water was applied.

The hydroponic tomato crop was only grown for 4 months. Commonly hydroponic tomato is grown for 5-10 months; maintaining a similar drainage fraction and NO_3^- concentrations, to that reported here, for 5-10 months would result in appreciably larger NO_3^- leaching losses.

These results demonstrate considerable NO_3^- leaching losses from 'open' hydroponic systems Drainage represented 24-38% of total irrigation volume, and the average drainage NO_3^- concentration was almost 1,000 mg NO_3^- L^{-1}. Given that the applied nutrient solutions and drainage fractions are consistent with commercial practice, these results are likely to be generally representative of commercial farms. Considering that there are approximately 5,000 ha of crops grown in 'open' hydroponic systems in the province of Almería, within a very high geographical concentration of greenhouses, these hydroponic systems represent a substantial source of NO_3^- contamination of underlying aquifers. From the cropping sequences in soil, appreciable drainage occurred following the large soil management irrigations applied during non-cropping periods and during the first four weeks after transplanting. Appreciable drainage for several weeks following transplanting is a consequence of (a) large irrigations applied immediately prior to transplanting to ensure that the entire soil surface and profile are moist, and (b) the application of irrigations considerably in excess of ETc (crop evapotranspiration) to ensure the survival of seedlings, which are often planted

under very warm climatic conditions; these are both standard local practices (Thompson *et al.*, 2004). Once established, there was very little drainage from the completed crops in soil, in the present study, on account of the use of tensiometers for irrigation management during this period. Comparisons of applied irrigation volumes and calculated ETc on commercial farms, in this system, indicated that generally until crop establishment irrigation volumes were considerably in excess of ETc, and that once crops were established irrigation was broadly similar to ETc. Therefore the occurrence of NO_3^- leaching from soil grown crops, primarily after large soil management irrigations and during crop establishment, observed in the current experimental work is likely to be similar to that which occurs on commercial farms.

Conclusions

The data from the present studies suggest in the greenhouse-based vegetable production system in Almería, Spain, that while cropping in soil makes an appreciable contribution to NO_3^- contamination of underlying aquifers, the 'open' hydroponic cropping systems make a relative contribution that is likely to be much larger than the percentage of cropping area occupied by these systems. Much of the greenhouse area in Almería has been declared Nitrate Vulnerable Zones in accordance with the EU Nitrate Directive (Anon., 1991). However, the key role of this horticultural industry in promoting the very rapid economic development of this region creates large difficulties for implementing measures that effectively reduce the massive NO_3^- contamination of aquifers that is taking place. This observation has relevance to other previously disadvantaged regions where agricultural development is taking place.

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Land use influence on soil mineralizable nitrogen

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Abstract

Agricultural land use often involves application of large amounts of fertilizers, which may lead to serious environmental problems, such as acid rain, eutrophication and production of greenhouse gases, mainly associated with application of nitrogen. In the present study, the effect of type of land use on nitrogen mineralization was investigated in two areas in Galicia (NW Spain), to establish the potential for nitrate production and to assess the risk of nitrate leaching. The results of incubation experiments under controlled conditions showed that grasslands soils had a higher capacity for mineralization of nitrogen than soils under forest or arable crops. Furthermore, grassland soils also showed a greater nitrification potential, although the mineralization values obtained in the laboratory indicated that production of inorganic forms of nitrogen could not lead to environmental problems.

Keywords: eutrophication, Galicia, land use, mineralization, nitrogen

Background and objectives

Large amounts of fertilizers, most commonly nitrogenous and phosphate fertilizers are applied to agricultural land with the aim of meeting plant nutrient requirements. However, the extensive application of fertilizers may lead to serious environmental problems, such as eutrophication of nearby water bodies (due to losses of soil phosphorus and soil nitrogen) or may contribute to acid rain or to the greenhouse effect as a consequence of gaseous emissions of nitrogen. In other words, nitrogen fertilization is associated with serious environmental risks. This is particularly important in regions with large areas of grassland, where cattle and pig slurries are often used as fertilizers. In such areas, organic matter mineralization will be an essential source of nitrogen (along with the inorganic nitrogen provided by the slurry); a large part of the mineralized nitrogen will be taken up by the vegetation, whereas the rest will be leached or emitted to the atmosphere. In the present study, the extent to which land use modifies the nitrogen mineralization capacity of soils and the associated risk of environmental contamination were investigated in two areas in Galicia (NW Spain).

Materials and methods

The study was carried out in two areas in Galicia, Fervenza (43º00´N, 8º58´W) and Portodemouros (42º52´N, 8º08´W), with different climatic, geological and edaphic characteristics. Soils under different types of land use were studied in each area. In Fervenza, forest soils (Forest), grassland soils under a low intensity of fertilization (NF grassland) and grassland soils under intensive fertilization (F grassland) were studied; in Portodemouros, forest soils (Forest), grassland soils under medium intensity management, dominated by organic fertilization (Grassland) and arable soils with rotations of corn-potatoes-cereals (Arable) were studied. In all cases sampling depth was 0-10 cm. Mean values of some properties of each group of soils are shown in Table 1.
Net nitrogen mineralization was determined as the difference between the amount of nitrogen initially present in the soils and that present after incubation for 10 days under controlled conditions of temperature (25 °C) and humidity (field humidity maintained in closed jars). At both, the start and the end of the incubation period, inorganic N was extracted from the soil with 2 M KCl (1:50) and the different forms of inorganic nitrogen (total, ammonia) were

analysed according to Bremner's procedure (Bremner and Keeney, 1965). Given the absence of nitrite-N, nitrate-N was calculated as the difference between total inorganic N and ammonia-N.
pH, total C, total N and available inorganic P (Pi) were determined using methods described by Guitian and Carballas (1976).

Table 1. Mean (± s.d.) values of some properties of the soils studied (0-10 cm depth). For each area, numbers in the same column followed by the same letter are not significantly different (p<0.005).

	pH H_2O	pH KCl	Total C (%)	Total N (%)	C/N	Pi (mg/kg)
Fervenza						
Forest (61)[a]	4.99±0.31*a*	4.08±0.30*a*	9.13±2.09*a*	0.663±0.156*a*	14.1±2.9*a*	2.98± 1.04*a*
NF grassland (33)	5.13±0.36*a*	4.16±0.17*a*	6.84±2.19*b*	0.534±0.147*b*	12.9±2.3*a*	13.00±11.93*b*
F grassland (93)	5.46±0.33*b*	4.52±0.34*b*	6.83±1.07*b*	0.575±0.160*b*	12.0±2.3*a*	26.94±11.61*c*
Portodemouros						
Forest (65)	5.48±0.39*a*	4.28±0.37*a*	4.91±1.41*a*	0.31±0.104*ab*	16.7±3.5*a*	2.14± 0.57*a*
Grassland (126)	5.57±0.29*a*	4.36±0.28*a*	4.12±1.36*b*	0.354±0.132*a*	11.9±1.8*b*	14.28±10.06*b*
Arable (34)	5.31±0.30*a*	4.07±0.18*a*	3.39±0.92*c*	0.265±0.060*b*	12.9±2.6*c*	23.20±22.34*b*

[a] *Number of soil samples.*

Results

In Fervenza and Portodemouros, both, arable and grassland use have resulted in changes in soil properties. The most obvious changes are a slight increase in pH and a decrease in total C in the soil, which was particularly notable in the arable soils. Furthermore, soil cultivation led to a large increase in the levels of available P, generating improved conditions for microbial activity.
With respect to the initial amounts of inorganic nitrogen, the highest values (Table 2) were recorded in the intensively fertilized grassland soils. The differences were mainly due to higher nitrate contents.

Table 2. Mean (± s.d.) values of inorganic nitrogen forms initially and mineralized in the studied soil samples (0-10 cm depth). For each area, numbers in the same column followed by the same letter are not significantly different (p<0.005).

	initial N (mg/100g)			mineralized N (mg /100 g)		
	Tot inorg N	N-NH_4^+	N-NO_3^-	Tot inorg N	N-NH_4^+	N-NO_3^-
Fervenza						
Forest (61)[a]	1.79±051*a*	0.87±0.23*a*	0.92±0.39*a*	1.10±0.52*a*	0.51±0.25*a*	0.59±0.43*a*
NF grassland (33)	1.79±0.47*a*	0.64±0.18*a*	1.15±0.41*b*	1.50±0.56*a*	-0.06±0.29*b*	1.56±0.56*b*
F grassland (93)	3.58±1.40*b*	0.75±0.38*a*	2.82±1.45*c*	2.43±0.67*b*	-0.19±0.24*b*	2.62±0.51*c*
Portodemouros						
Forest (65)	1.27±0.41*a*	0.50±0.29*a*	0.77±0.17*a*	0.55±0.58*a*	0.39±0.40*a*	0.17±0.37*a*
Grassland (126)	2.26±2.05*b*	0.74±0.97*a*	1.52±1.11*b*	1.56±0.51*b*	0.10±0.32*b*	1.46±0.60*b*
Arable (34)	2.05±1.15*b*	0.31±0.03*b*	1.74±1.12*b*	0.97±0.44ab	0.07±0.04*b*	0.90±0.43*c*

[a] *number of soil samples.*

Net mineralization of nitrogen (Table 2) was highest in all grassland soils. However, the quantities mineralized were never very high, thus restricting possible environmental problems. Mineralized nitrogen was mainly in the form of nitrate in the cultivated soils, while in the forest soil ammonia-N also increased; in Fervenza, ammonium content in the cultivated soil decreased during incubation, suggesting that the rate of nitrification exceeded that of mineralization. It must be pointed out that the main difference between grassland and the other uses is the existence of a stronger nitrification process in grassland soils.

Discussion and Conclusions

The results obtained show that the grassland soils have a higher capacity for nitrification than the other soils. This potential can basically be attributed to the intensity of slurry application, providing large quantities of easily mineralizable nitrogen-rich organic substrates, as well as to higher soil pH, presenting a more favourable physicochemical environment for microbial action (Diaz-Fierros *et al.,* 1987).
However, it must be taken into account that although grassland use is associated with a higher nitrification potential than the other uses, and although mineralization values obtained in laboratory assays were not very high, there is an obvious need for careful management of grasslands to avoid nitrate leaching (which could create eutrophication) and conditions that lead to denitrification and the associated loss of nitrogen through gaseous emissions.

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A novel dual isotope enrichment method for distinguishing between N_2O sources in soils

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Abstract

We present a novel ^{18}O-^{15}N-enrichment approach for distinguishing between nitrification, nitrifier denitrification and denitrification as sources of N_2O. This new technique was based on a method with single- and double-^{15}N-labelled NH_4NO_3. A new treatment with ^{18}O-labelled H_2O was introduced, exploiting the fact that ammonia oxidisers use O_2 from soil air for the oxidation of NH_3 and thus N_2O from nitrification, but H_2O for the oxidation of the resulting hydroxylamine to nitrite. Thus, the O in N_2O from nitrifier denitrification can come either from soil O_2 or H_2O. It was assumed that a) there would be no preferential removal of ^{18}O or ^{16}O during nitrifier denitrification or denitrification, b) the ^{18}O signature of the applied ^{18}O-labelled water would remain constant, and c) any O-exchange between $H_2{}^{18}O$ and NO_3^- would be negligible under the chosen experimental conditions. These assumptions could be validated for the soil investigated. The results of the new method were compared with those of a conventional inhibition method using low concentrations of C_2H_2 and high concentrations of O_2. The inhibitor approach overestimated the contribution from nitrification at the expense of denitrification. We regard the new enrichment method as a more reliable method that allows distinction between more soil sources of N_2O than conventional methods.

Keywords: ^{15}N, ^{18}O, dual-isotope enrichment, N_2O, nitrification, nitrifier denitrification, soil

Background and objectives

Nitrous oxide (N_2O) can be produced by various soil sources, most importantly nitrification, nitrifier denitrification and denitrification (Figure 1). Current methodology to differentiate between these sources of N_2O is based on inhibitors (Webster and Hopkins, 1996) or stable isotopes (Baggs *et al.*, 2003, Tilsner *et al.*, 2003). Inhibitors often have side-effects and the inhibitors most commonly used for the distinction of N_2O sources, acetylene (C_2H_2) and oxygen (O_2), are not reliable in all soil conditions and for all microorganisms involved. Stable isotope techniques, both enrichment and natural abundance, have the disadvantage that they can so far only differentiate between nitrifier- and denitrifier-based N_2O production.

The objective of this study was to develop a novel dual-isotope enrichment technique to distinguish between N_2O production from nitrification, nitrifier denitrification, nitrification-coupled denitrification and fertilizer denitrification in soil.

Figure 1. *Sources of N and O in N_2O, after Wrage et al. Different font styles indicate different sources. The splitting off of water in the reduction reactions is only shown exemplarily. ONO-: nitrite; ONOO-: nitrate (drawn linearly for reasons of simplicity, note that each O is bound to the central N atom).*

Material and methods

Ammonia oxidisers use O_2 from soil air for the oxidation of NH_3, but H_2O for the oxidation of the resulting hydroxylamine to NO_2^-. Thus, N_2O from nitrification should reflect the ^{18}O signature of soil O_2, whereas the O in N_2O from nitrifier denitrification comes from soil O_2 and H_2O (Figure 2). Based on this, an incubation treatment was developed with addition of ^{18}O-labelled H_2O. Three further treatments were set up with addition of single- or double-^{15}N-labelled NH_4NO_3. It was assumed that a) there would be no preferential removal of ^{18}O or ^{16}O during nitrifier denitrification or denitrification, b) the ^{18}O signature of the applied ^{18}O-labelled water would remain constant over the experimental period, and c) any O-exchange between $H_2{}^{18}O$ and NO_3^- would be negligible under the chosen experimental conditions. These assumptions were tested. The new method was furthermore applied to a silt loam soil at 50% water-filled pore space following application of 400 mg N kg^{-1} dry soil. Results for the contribution of various processes were compared with those of a conventional method using small concentrations of C_2H_2 and large concentrations of O_2 as inhibitors (Robertson and Tiedje, 1987, Webster and Hopkins, 1996).

Figure 2. *N_2O production by different processes, measured at 6 and 24 hours with the new ^{18}O-^{15}N-enrichment method (Enr M) or the inhibition (C_2H_2 + O_2) method after Webster and Hopkins (Webster and Hopkins, 1996) (Inh M). The percentage of total N_2O production is given per process (Wrage et al., 2005).*

Results and discussion

The assumptions were validated for the soil investigated. The enrichment method and the inhibition method agreed on the importance of nitrifier denitrification (44 and 40% of N_2O production after 24 h for the enrichment and the inhibition method, respectively), but the inhibition method overestimated nitrification and underestimated denitrification in this soil. This was probably due to incomplete inhibition of nitrifier denitrification and denitrification by large concentrations of O_2, and a negative effect of C_2H_2 on denitrification. The enrichment method seems to be a reliable tool for the distinction of soil sources of N_2O.

Conclusions

The novel ^{18}O-^{15}N-enrichment method enables a separation between more sources of N_2O in soils than was possible before, and has provided the first direct evidence for the significance of nitrifier denitrification as an N_2O-producing process in soils. The method provides a more reliable and less disruptive alternative to the use of inhibitors for quantifying sources of N_2O in soil, and is associated with fewer conceptual uncertainties.

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