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# Future trends in nitrogen research

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## Abstract

N research effort has undergone major changes over recent decades with changing emphasis because of environmental problems and issues. This driving force, coupled with a universal desire to improve N-use efficiency, appreciation of the importance of maintaining soil resource quality and a need to provide integrated landscape managements, will continue to prompt new research areas and issues for study. Already, much information has been provided and new approaches and needs defined. It will be essential in future research to take full note of the many interactions that occur and to provide a mechanistic basis so that scaling of effects can be undertaken with the appropriate simplification without being superficial. Examples of interactions, as well as fundamental gaps in the basic processes are discussed and needs for future research identified.

## Introduction

For the efficient functioning of all ecosystems, whether under agricultural management or in natural/semi-natural environments, the importance of nitrogen (N) cycling is profound. This has been appreciated for decades and researchers have therefore investigated many of the components of the cycle. However, despite the longevity of interest and the intensity of research activity, and despite a good appreciation of the basic processes and structure of the cycle, many components are still poorly understood and/or quantified. Further, as demands for new information have arisen, so it has become apparent that results from all situations do not necessarily fit well with existing general concepts and model descriptions.

In large part, the stimulus to research various components of the cycle afresh has come about through environmental pressures. Whilst it is true that many research projects in many parts of the world are still driven simply by the need to achieve crop yields that are closer to the potential for a given site, many others have arisen because of problems of pollution and the needs of policy makers to make attempts to predict pollution effects and/or meet legislative demands. The overall result of this has been, so far as the N-cycle is concerned, (i) a refocusing of effort to meet specific demands, often to provide quantification (e.g. how much $NO_3^-$ leaches from particular managements), (ii) development of novel approaches to tackle hitherto neglected areas (e.g. determination of gaseous N fluxes), and (iii) a much wider appreciation of the complexity and interactions that occur and the need to develop models of varying degrees of sophistication to aid prediction.

At least in the foreseeable future it seems likely that problems of excess, mobile N leaking in one form or another to the wider environment will continue to dictate research demands. Nitrate in waters remains, because of current legislation, a major issue, but volatilization of ammonia ($NH_3$) and the release of nitrous oxide ($N_2O$) and the other oxides of N (NO, $NO_x$) into the atmosphere and waters are all of concern and interest and will require greater understanding. This must also be seen in the context of other driving forces for research on N:

(1) The N cycle is scientifically challenging: its complexity and interactions attract research initiatives, and the availability and applicability of new tools and approaches provide stimuli to researchers.

(2) There is still a need to provide crop (plant) response information. Many areas require increased agricultural production and N, in many circumstances, is a major limiting factor. In all systems, there is a

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demand for increased efficiency of utilization and this requires new data for many existing managements.

(3) Novel cropping regimes, crops and management practices that become available have consequences for N flows and an awareness of these is essential. Organic farming, use of land for 'set-aside' and other new managements, put particular demands on nutrient flows which are not yet understood and will require new knowledge.

(4) Increasingly, there will be demands to integrate agriculture more closely with other components of the countryside. Nutrient flows, and that of N in particular, are important features of this integration and will require control and manipulation and, therefore, understanding.

(5) There is growing appreciation of soil as an important resource and of the need to 'protect' that resource. Soil quality is not easily defined ( it depends on 'quality for what?') but N status is an essential component and will require characterization and definition in relation to the demands of the particular system.

The present paper is not an attempt to review the overall current status of N research and pinpoint detailed gaps in our knowledge: this would have been difficult because of the breadth and depth of the subject. Instead, it is rather to identify and illustrate, using recent information based in large part on grassland N research from the Institute of Grassland and Environmental Research, general areas and approaches which will be required to meet the demands of the issues already noted. It will concentrate particularly on soil N processes and losses and their impact over a range of spatial and temporal scales.

## Soil transformations: current status and research needs

An emphasis on environmental issues with respect to N has brought about changes in research activities over the last 10 years or so. In the first instance there was a need for quantification of losses under various defined circumstances so that some preliminary assessments of impact could be made. Much information of this nature has accumulated, especially with regard to $NO_3^-$ leaching and there is now a substantial literature on the rates of loss of N by leaching and by other means from various managements. Although the emphasis may change between particular processes, there will still be a requirement to provide direct quantification of losses of this nature.

However, numerous measurements have demonstrated the complexity of the various systems, and that, in order to make confident model predictions for other circumstances, a much better understanding of the transformation processes is required. Future research will not only have to probe the processes further but also recognize the very high degree of interaction that occurs and the need to integrate information so that effective, reliable extrapolation can be made over a range of spatial and temporal scales. In the following, by no means exhaustive, examples some of the problems and issues are discussed.

### *Mineralization/immobilization*

The balance between these two microbially based processes is central to the flows and availability of mobile forms of N in the soil. Much research effort has been undertaken to determine rates in the field and under controlled conditions and much information already exists. This has been reviewed extensively recently (Powlson et al., 1994). However, it is not yet possible to provide adequate reliability for prediction which is of immediate relevance to, and needed for, decisions on fertilizer requirements and recommendations, $NO_3^-$ leaching, recycling of N from crop residues and animal manures, maintenance of soil organic matter contents, emissions of trace greenhouse gases and the future management of changing agricultural practices.

Much is known on a qualitative basis about many of the soil factors which influence mineralization but the extent of data is limited and does not necessarily allow a practical interpretation of impact on N cycling in other than a few restricted circumstances. There is little doubt that the practical effects of mineralization can be substantial. Recent studies have attempted, using a field based incubation technique with regular, frequent sampling, to produce an annual net mineralization rate for a range of different grassland soils (Table 1). The results showed a wide range of maximum daily rates from 1.01 to 3.19 kg N $ha^{-1}$ and net annual rates ranged up to 370 kg N $ha^{-1}$ in an intensively managed grazed pasture. Even where there had been no fertilizer N applied for many years 135 kg N $ha^{-1}$ were released. Addition of fertilizer to a previously unfertilized sward significantly increased the net release of N, but withholding N had no effect on mineralization during the year of measurements. Changes in temperature accounted for 35% of the variability in daily rates but there was little significant effect of soil moisture. It is of immediate relevance to the problems of pollution to note that, on average, 27% of the annual release occurred during the period November - January, i.e. when any $NO_3^-$ in the system would have been particularly vulnerable to losses in winter drainage.

The amounts of net release were large and must have considerable impact on the overall efficiency of N in these grassland soils. Preliminary studies of the same soils have indicated significant differences between the distribution of organic materials between different physical fractions of the soils and between the different treatments and differential mineralization/immobilization rates of those materials when measured under controlled conditions. Soil texture apparently exerts control over mineralization by influencing aerobicity, affecting the physical distribution of organic materials and conferring some degree of "protection" through an association of organic matter with clay particles (Hassink et al., 1993). However, there is not yet enough appropriate information to allow the development of effective 'broad-brush' predictive models. This will require better definition of soil organic matter, its location over a fine spatial scale and its interaction with soil biology and environmental status. An important component of this will be to derive/link relationships between soil texture, structure and pore size and to determine whether protected soil organic matter is of practical relevance. We will also need to define the quality of soil organic materials and added residues in terms of N (and other nutrient) supplying potential rather than by chemical composition.

Another important aim in studies of mineralization, and indeed of all soil N cycling processes, will be to consider the architecture of the soil (Dexter, 1988) and the location of the internal N cycle and its relationship to soil organisms. A knowledge of the diffusional constraints to $NH_4^+$ and $NO_3^-$ movement (i.e. between different pools) will also be required. This would help to provide the mechanistic basis for understanding the controls over mineralization at a fundamental level: the concepts of microsites and different 'pools' of potentially mineralizable N and of mineral N are implicit in this. With this as a basis it should be possible to integrate information sequentially into a series of larger scales.

*Soil microbial biomass*

Soil microbial biomass has a profound effect on the net release of mineral N but we cannot yet assess the possibility of being able to manipulate N flows though the complex food webs that this forms part of in soils to the advantage of agricultural and other systems. An understanding of this is essential to aid the development of all management systems. Microbial biomass is important as an agent of change, decomposition and release of N from fresh organic residues and native soil organic matter into more labile forms, as a major sink for 'active' soil N and as a potential source of labile N. One description has been as "the eye of the needle though which all organic matter must pass" (Jenkinson, 1990).

In grassland soils the amounts of soil microbial biomass present were substantial, but were not apparently influenced to a very marked degree by background management (Bristow and Jarvis, 1991) . In other studies summarized in Table 2, there were few differences in trends of biomass N (or C) with time, drainage, withdrawal of fertilizer N from a previously fertilized soil, or addition of fertilizer N to a previously unfertilized soil. There was, however, substantially more biomass in the previously unfertilized than fertilized treatments. In contrast, measurements of ATP, enzyme activity and respiration showed significant effects of both short and long term treatments. It is also apparent that the patterns and effects demonstrated with soil microbial biomass (Table 2) do not conform with those found in the same soils for net mineralization rates (Table 1). There is therefore a need, firstly, to understand the differences in microbial activities which occur in response to short term management changes and to identify and quantify differences in microbial community structure. Secondly, if the aim of making progress in increasing the efficiency of N utilization from all sources is to be realized, the integration of this information with better understanding of the soil organic matter and added residues is essential and will require research at a fundamental level.

*Nitrification*

As far as the fate of any N released through interaction of mineralization/immobilization processes is concerned, nitrification is an important step. The transformation from a relatively immobile ($NH_4^+$) to a highly mobile ($NO_3^-$) stage via $NO_2^-$ provides opportunities for escape of a number of materials ($NO_3^-$, $NO_2^-$, $NO_x$, $N_2O$, $N_2$) with the obvious consequences. Despite its importance as a rate limiting process and a reasonable knowledge of the ecology and environmental demands of the organisms involved, it is a poorly defined pro-

*Table 1*. Mineralization rates in soils under long-term pastures (from Gill et al., 1995)

| N treatment (kg N $ha^{-1}$) Past | Present | Drainage | Total net annual mineralization (kg N $ha^{-1}$) | Annual turnover (% total soil N) |
|---|---|---|---|---|
| +200 | +200 | Drained | 376 | 6.5 |
| +200 | +200 | Undrained | 317 | 5.7 |
| +200 | 0 | " | 292 | 5.5 |
| 0 | 0 | " | 135 | 2.5 |
| 0 | +200 | " | 270 | 5.3 |

*Table 2*. Effects of past and present sward mangement on soil microbial biomass contents and activities (from Lovell et al., 1995)

| N treatment (kg N $ha^{-1}$) Past | Present | Drainage | Biomass contents (mg $kg^{-1}$ dry soil) C | N | Dehydrogenase activity (n mol INTF $g^{-1}2\,h^{-1}$) | ATP content ($\mu g\ g^{-1}$) | Basal respiration ($\mu g$ $CO_2$-C ($g^{-1}\,h^{-1}$)) |
|---|---|---|---|---|---|---|---|
| +200 | +200 | Drained | 1056 ($\pm$ 34.9) | 159 ($\pm$ 3.0) | 889 a | 3.1 a | 0.9 a |
| +200 | +200 | Undrained | 1055 ($\pm$ 28.9) | 169 ($\pm$ 4.0) | 1343 b | 4.7 b | 1.8 b |
| +200 | 0 | " | 1009 ($\pm$ 21.6) | 161 ($\pm$ 4.7) | 1204 b | 3.4 a | 1.8 bc |
| 0 | 0 | " | 1616 ($\pm$ 53.3) | 255 ($\pm$ 7.7) | 1362 b | 5.7 b | 2.6 d |
| 0 | +200 | " | 1716 ($\pm$ 49.9) | 275 ($\pm$ 4.7) | 1124 ab | 7.3 d | 2.1 be |

Significant differences between activities ($p$<0.05) shown by values without same letter.

cess in many soils. In all tillage systems it is usually assumed that nitrification is not a limiting process and that nitrification rates exceed the rate of net mineralization and do not impose limitation to further transformation. For many grassland (Jarvis and Barraclough, 1991) and natural systems, however, there may be significant quantities of $NH_4^+$ accumulated in the soil profile (Table 3). Nitrification interacts strongly with local ambient soil conditions and there is a high degree of spatial compartmentalization of $NH_4^+$ production and consumption sites. These, and the diffusional constraints between microsites in the soil, exert controls over nitrification rates. Latent potentials can apparently develop which are dependent upon background management and which can be displayed over the relatively short (Jarvis and Barraclough, 1991) or longer term (Willison and Anderson, 1991). There are also important competition effects with plant uptake, for example (Barraclough et al., 1983). Interactions with denitrification (see later) may also be of some importance with respect to removal of $NO_3^-$ and the production of trace gases.

Nitrification is central to the flows, transfers, losses or utilization of N. It is also a key interacting process which is linked to others by flows of substrates and spatial distribution in soil systems and a greater understanding will be important in maximizing, especially in grassland situations, N efficiency and reducing losses.

## Nitrogen loss processes

Concerns over emissions will continue, not only because of direct concerns over environmental issues, but also because there is a growing desire to make nutrient cycles within farming systems more efficient from economic and self-sufficiency stand points as well. Much information has been gathered over recent years on many aspects of the loss processes but there are still major problems in being able to predict losses in many situations and over a range of scales. Research will be required to continue to improve our understanding of the mechanisms involved, to have the basis to extrapolate information to other circumstances, to take account of various interactive forces and factors and to be able to produce predictive estimates of effects over a range of scales through from field to farm to catchment and region and to accurate national budgets where

*Table 3.* Effects of fertilizer inputs to grazed grassland soils on soil mineral N in autumn (to 40 cm). The ratio $NH_4^+$ : $NO_2^-$ is taken as an indicator of nitrification potential (data taken from Jarvis and Barraclough, 1991)

| Soil | Mineral N | Fertilizer rate (kg ha$^{-1}$) | | | |
|---|---|---|---|---|---|
| | ($\mu$g g$^{-1}$ soil) | 100 | 250 | 450 | 750 |
| Hurley | $NH_4^+ + NO_3^-$ | 24.1 | 22.7 | 64.3 | 207.3 |
| (loam) | $NH_4^+ : NO_3^-$ | 2.2 | 2.0 | 0.7 | 0.4 |
| Jealotts Hill | $NH_4^+ + NO_3^-$ | 30.3 | 27.5 | 48.7 | 310.3 |
| (fine loam) | $NH_4^+ : NO_3^-$ | 1.7 | 1.5 | 0.4 | 0.1 |
| Ravenscroft | $NH_4^+ + NO_3^-$ | 27.4 | 248.5 | 186.7 | 292.3 |
| (clay) | $NH_4^+ : NO_3^-$ | 0.4 | 0.1 | 0.3 | 0.1 |
| North Wyke | $NH_4^+ + NO_3^-$ | 89.6 | 55.1 | 76.5 | 239.8 |
| (clay) | $NH_4^+ : NO_3^-$ | 7.0 | 6.7 | 2.1 | 0.1 |
| Drayton | $NH_4^+ + NO_3^-$ | 2.6 | 1.8 | 3.1 | 60.7 |
| (clay/loam) | $NH_4^+ : NO_3^-$ | 0.4 | 1.6 | 0.3 | 0.2 |

these are required. In order to do this with confidence some important gaps need to be filled. Some examples follow for each of the three major loss processes.

### *Leaching*

Primarily because of legislative demands, much data on $NO_3^-$ leaching has been accrued over the last 5 years (see for example, Archer et al., 1992). Even accepting the problems associated with sampling to provide accurate assessment of solute transport in many soils, data are usually limited to quantification of particular agronomic/environmental combinations and do not necessarily provide the means of describing, on a mechanistic basis, leaching losses for many important soil types. Optimization of land management to meet the demands of legislation and to be able to continue viable production systems will require the use of simulation models which describe N losses over larger scales. Such models will be dependent on accurate sub-models which define the accumulation of potentially leachable soil $NO_3^-$ linked with others to describe $NO_3^-$ transport. Many excellent models of leaching exist (e.g. Addiscott and Whitmore, 1991; Barraclough, 1989), but the use of mechanistic models to simulate N transport in a number of soil types, notably those that have a defined structure, have met with only limited success. A new approach has been developed recently to generate 'real-time' data with which to test and validate and thereby modify and improve existing models. By the use of a range of tracers (Cl, $^{15}N$, deuterium), transport of $H_2O$ and solutes is being quantified at very fine spatial and temporal resolution in a large (3.4 × 5.4 × 1.2 m) undisturbed soil block (Scholefield and Holden, 1994). Initial results are already illustrating the degree of spatial variability in solute concentration fronts at depth in a well drained, structured soil which makes prediction very difficult. More such information will be required before effects at the larger scale can be predicted.

Previous studies have allowed a number of empirical relationships to be established and because these are of some considerable value, many of the existing data sets should be re-examined to develop further relationships. Thus long-term studies on grazed pasture (1 ha drained lysimeters) have provided clear relationships between recorded soil moisture deficients during summer and $NO_3^-$ leaching losses during the following winter (Scholefield et al., 1993b) with more leaching with increasing soil moisture deficit. Other relationships between the soil load of potentially leachable $NO_3^-$ and the peak concentration at which it leached were also established. These relationships were independent of drainage volume and only slightly influenced by drainage intensity. Other data from long term experiments have also been examined in the same way (Table 4) and demonstrate the possibility of employing such empirical approaches to account for preferential

*Table 4.* Relationships between peak $NO_3^-$ leachate concentration and soil load of $NO_3^-$ on different soils and under different managements (data from Scholefield and Holden, 1994)

peak (mg $dm^{-2}$) = a × $NO_3^-$ leached (kg N $ha^{-1}$) + b

| Soil type | Agricultural system | a | b | $r^2$ |
|---|---|---|---|---|
| Loamy sand | Grassland | 0.99 | 1.4 | 0.88 |
| " | Arable | 0.92 | 2.4 | 0.93 |
| Loam | Grassland | 0.62 | 3.5 | 0.99 |
| Clay loam (undrained) | Grassland | 0.58 | 3.9 | 0.96 |
| Clay (cracking) (mole drained) | Arable | 0.37 | 2.8 | 0.33 |
| Clay loam (mole drained) | Grassland | 0.28 | 5.6 | 0.97 |

*Table 5.* Potential denitrification rates to depth below a long-term grazed grass-clover sward. Data are calculated from laboratory measurements of samples incubated under anaerobic conditions in the presence of added $NO_3^-$ and with (NC) or without (N) added C (sucrose) and are calculated from Jarvis and Hatch (1994)

| Sample depth (m) | Potential denitrification (kg N $ha^{-1}$ $d^{-1}$) | |
|---|---|---|
| | N | NC |
| 0 - 0.5 | 14.3 | 21.9 |
| 0.5 - 1.0 | 6.9 | 8.8 |
| 1.0 - 2.0 | 15.3 | 30.7 |
| 2.0 - 4.0 | 4.7 | 67.2 |
| 4.0 - 6.0 | 3.9 | 35.3 |
| Total | 45.1 | 163.9 |

*Table 6.* Total denitrification and nitrous oxide emission rates (g $ha^{-1}$ $d^{-1}$) (determined over 4 hours using an enclosure method with or without acetylene, respectively) on a peat soil with added cattle slurry (from Jarvis et al., 1994)

| Days after spreading | Total denitrification | $N_2O$ |
|---|---|---|
| *Week 1* | | |
| 1 | 20 (7.5) | 7 (4.5) |
| 2 | 80 (34.1) | 30 (14.5) |
| 3 | 260 (63.1) | 54 (29.9) |
| 4 | 46 (15.9) | 24 (6.0) |
| 5 | 86 (51.0) | 47 (19.9) |
| *Week 2* | | |
| 1 | 66 (31.7) | 67 (5.4) |
| 2 | 15 (3.7) | 29 (5.7) |
| 3 | 31 (10.5) | 34 (14.9) |
| 4 | 167 (16.9) | 49 (2.6) |
| 5 | 86 (9.5) | 34 (11.5) |

flow in modelling $NO_3^-$ leaching which may be especially appropriate at the catchment and regional scales.

It will be essential to build into all such predictions an appreciation of the interactions that may occur. Again at an empirical level, it is clear that increased soil aerobicity (through, for example, drainage) can have a profound effect on $NO_3^-$ leaching not only though reduction in denitrification but also, it is suggested, an enhanced rate of net mineralization (Scholefield et al., 1993b). Quantification of these effects will also require good mechanistic understanding so that overall impact at the larger scale can be determined.

The fate of $NO_3^-$ moving away from agricultural systems also needs to be assessed. It is often assumed that $NO_3^-$ leaching from grassland systems is non-reactive and is transported to the saturated zone in confined aquifers without change. There is, however, evidence from a number of studies which suggests that there may well be denitrification effects which influence the final concentrations of $NO_3^-$ in ground water (Gillham, 1991). This denitrification may also be an important removal mechanism as $NO_3^-$ moves through the unsaturated zone as well (Lind and Eiland, 1989). Recent studies have shown that considerable potential for denitrification existed at least to 6–8 m below long term grassland and was increased substantially when labile carbon was added to the experimental system (Jarvis and Hatch, 1994). Calculations indicate (Table 5) that even if only a small proportion of this potential for removal was achieved in practice, this would have implications for the movement of $NO_3^-$ into aquifers. However, the consequences of this with regard to emissions of environmentally active gases (i.e. $N_2O$) must also be considered and a greater knowledge is again required to enable better management and policy decisions relating to potential pollution from agriculture.

### *Denitrification*

The problems associated with determining denitrification in soil are considerable and well known. The spatial and temporal variability impose considerable restrictions on our current ability to measure and determine its impact (Smith and Arrah, 1990). Despite the very large research effort that has, and is going on, much is still uncertain about the impact of denitrification with regard both to its influence on overall N use efficiency and to its contribution to $N_2O$ budgets.

Recent attempts to synthesize information to provide N budgets and flows at a farming system level indicated that at least 16.5% of an annual N input of 24.5 tonnes N to a 76 ha dairy farm would be lost through denitrification (Jarvis, 1993). It was suggested that, because of a current inability to estimate losses via this route a major proportion of the fraction of N still unaccounted for in this farming system could also be ascribed to denitrification losses. Research must continue to address this problem as a matter of urgency in order to increase efficiency and to reduce $N_2O$ emissions. It is perhaps worth noting, however, that the ultimate fate of N over time is to re-enter the global N cycle through denitrification and gaseous transmission to the atmosphere. If increasing agricultural efficiency results in greater capture of N in products, then denitrification will become more important at a later stage of the food chain after consumption by humans. In the meantime, there is much to be learnt about the process and its interaction with environmental conditions.

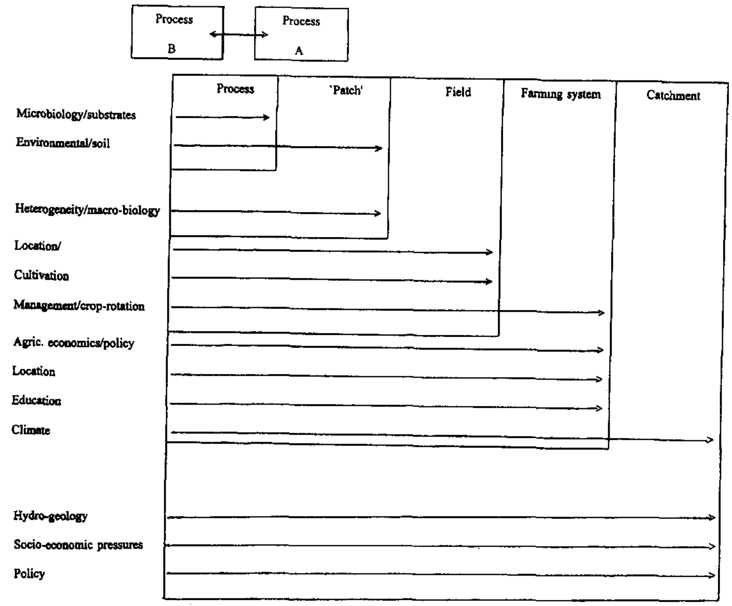
Studies with a poorly drained clay soil under controlled environment conditions and using a flow-through system have enabled definition of effects of $H_2O$, $O_2$ and $NO_3^-$ contents on the amounts and proportions of $N_2$ and $N_2O$ produced (Scholefield et al., 1993a). The next stage will be to use this information to develop models for field use. Other recent laboratory and field studies (Bisson et al., 1994) have further underlined the complexity of the controls and demonstrated the need to provide mechanistic explanation. In this latter case, the release of NO was used as a fingerprint to define nitrification. NO is short-lived gas, but its emission from the soil should provide an indicator of process activities. Although nitrification was the dominant source of NO, release was obvious on addition of either $NH_4^+$ or $NO_3^-$. The preliminary conclusions from this study were that nitrification was the primary source of NO but that there was some evidence that denitrification was also an important source under certain moisture conditions. This raises questions of (i) the extent of coupling between the two processes, (ii) the possibility of sharing joint metabolites if the processes occur concurrently, (iii) the relationships and interactions between the amounts of substrates ($NH_4^+$ and $NO_3^-$), and (iv) the reactions of the relevant microbiological populations and their enzyme systems and the release of the trace gases $N_2O$ and NO at various stages of either the oxidative or reductive transition. There may also be opportunities for losses of another short-lived intermediate, $NO_2^-$ , directly into waters in some circumstances.

With this degree of complexity it is not surprising that results of field measurements can often be difficult to interpret. Studies of $N_2O$ release and denitrification in a peat soil with added cattle slurry showed, over a range of time scales, a wide range in denitrification rates and N : $N_2O$ ratios as well as an expected high degree of spatial variability (Table 6). It is of interest to note that, over a range of treatments, the major proportion (0.635) of the daily variability in total denitrification could be accounted for by changes in soil moisture and $NH_4^+$ contents. Inclusion of soil temperature and $NO_3^-$ contents in the regression model did not improve the $R^2$ value. This further demonstrates, at a practical level, the importance of considering the coupling of nitrification and denitrification processes in making predictions of losses. This is in addition to providing a better means of determining denitrification under field conditions. Whilst there are new approaches and techniques available to determine net $N_2O$ fluxes and budgets (e.g. micro-meteorology methods, photoacoustic, infrared and laser instrumentation) these will not necessarily provide sufficient basic information to extrapolate information to other circumstances. The requirement will be for integration of controlled environment and field measurements to be coupled with appropriate model simulations.

### *$NH_3$ volatilization*

Of the three major loss processes, the chemical and physical controls over the transfer of $NH_3$ *in simple systems* are probably the best understood. However, despite this and although the sources are generally well known, emission rates and transfers of $NH_3$ within many agricultural and natural systems are not well defined. $NH_3$ volatilization is an important loss process especially from animal production systems and in a typical dairy farm approximately 13.6% of a total annual input of 25 tonnes of N are estimated to be lost (Jarvis, 1993). Measurement of $NH_3$ in the field has only been possible relatively recently so that data from many aspects of agricultural managements are restricted. There are a considerable number of data sets from some components of agricultural production (e.g. spread farm wastes, see Nielsen et al., 1991) a restricted number from others, (e.g. losses from grazed swards) and even less from others (e.g. from stored wastes and animal houses). The current need to provide national budgets for $NH_3$ because of its implications for atmospheric chemistry and critical loads for terrestrial and aquatic systems will require (i) much greater

*Table 7*. Interaction of effects of various factors and influences on the impact of N cycling processes over a range of scales




precision in estimates of emission factors from the major animal derived sources, especially excreta either deposited in fields and houses or stored, each under a wide range of conditions; (ii) a knowledge of the transfer coefficients for $NH_3$ from various sources over a range of scales, i.e. from urine patch to neighbouring sward, from fertilized, grazed or slurry-amended field to neighbouring areas, and from farming systems to adjacent natural systems; and (iii) some further definition of the role that crops have as sink or sources of $NH_3$. As remedial measures are introduced to reduce $NH_3$ emissions to the atmosphere (as they have been in the Netherlands), it is important that interactions with other processes and the consequences of these for the overall efficiency of the farming system are fully considered and quantified.

*Interactions, integration and scaling of effects*

Research approaches to N cycling have changed considerably over recent years and are opening up new insights to the transfer of N within various components and at various spatial levels within the whole complexity of the system. The demands of legislation and national policies have and, at least over the short term, will continue to raise issues which will require continued input into developing understanding and approaches to be able to do that. It is important on the one hand that basic processes and mechanisms are understood and, on the other, that these are translated into simulation models appropriate to the question being raised. It is important that models of N flows continue to be developed at various levels of complexity. There is a need to define models operating for each of the various sub-systems involved so that simplification can be made further up the hierarchy but without the final model being a superficial treatment of the system. Changes in our perception about the various interactions and controls over individual processes have been made apparent through detailed experimental work and will continue with the use of new tools. For example, stable isotope definition of changes using enriched and natural abundance techniques; definition of controls over microbial activities in relation to community structure and location as determined by soil architecture using enzyme measurements and genetically manipulated organisms; long path detection systems for trace gas measurement; and near real time measurement of solute and water fluxes etc. are now, or becoming available. These will ultimately lead to increased accuracy in the models used for the management of agricultural systems and in the prediction of the impact of these systems on the environment.

There is a need therefore to integrate information which must be based on sound principles over a range of scales i.e. from microsite to patch, from patch to

*Table 8.* Estimated effects of management changes on nitrogen losses from dairy farming: initial desk studies, A: conventional 'model' management (see Jarvis, 1993); B: using 'tactical' fertilizer approach based on regular analysis of soil mineral N and injected slurry; C: grass/white clover swards; D: substituting 50% of grass silage by maize silage to (i) utilize slurry N and (ii) improve conversion of dietary N into product; E: B + D.

| | A | B | C | D | E |
|---|---|---|---|---|---|
| N fertilizer use (kg $ha^{-1}$) | 250 | 200 | 0 | 185 | 148 |
| Stocking rate livestock (units $ha^{-1}$) | 2.17 | 2.17 | 1.74 | 2.17 | 2.17 |
| N losses (kg $ha^{-1}$) | 160 | 86 | 89 | 99 | 60 |

*Table 9.* Experimental management systems on 1 ha farmlets to determine the flows and losses of N from complete grassland production systems (from Pain et al., 1994)

| | |
|---|---|
| *Conventional management* | |
| N fertilizer: | Set times and amounts (280 kg $ha^{-1}$ $yr^{-1}$) |
| Slurry: | Surface applied in spring and post silage |
| Grazing: | May - October : beef steers |
| Silage: | 3 times per year |
| *Tactical nitrogen* | |
| N fertilizer: | 'Tactical' adjustment based on fortnightly analysis of soil mineral N - to reduce excess soil mineral N and leachate concentrations (to below 11.3 mg $L^{-1}$) |
| Slurry: | Injected - to reduce $NH_3$ volatilization - in spring and post-silage |
| Grazing: | May - August : beef steers |
| Silage: | 4 times per year - 1 post grazing to remove excess soil mineral N |
| Grass/white clover | |
| N fertilizer: | Nil |
| Slurry: | Injected in spring and autumn: + nitrification inhibitor to reduce denitrification/leaching |
| Grazing: | May - October: beef steers |
| Silage: | 3 times per year |

field, from field to farming system and from farming system to catchment (Table 7). Care must be taken to ensure that the models used to do this have the appropriate simplification and not superficiality. Throughout this discussion it has been clear that whilst a full understanding of the controls over a particular process is a prerequisite to be able to do this, full recognition must be made of interactive effects with other processes at the spatial/temporal level required. At the systems level the effects can be extremely important and require quantification.

Sufficient information and model development is available to be able to make estimates and calculations for the flows and transmission of N for existing farming systems (see Jarvis, 1993) and then to be able to manipulate managements to look at the impact on nitrogen losses. Information shown in Table 8 is a preliminary estimate of the impact that changing fertilizer and waste management on a dairy farm could have on leaching losses, taking into account the coincident effects that these changes have on mineralization, ammonia volatilization and denitrification for example. Exercises of this nature have value in that they provide an immediate perception of the overall impact and implications for a farming system, they identify and define gaps in knowledge and demonstrate the means of increasing N efficiency and reducing losses and environmental impact. Hand-in-hand with such desk studies, holistic system measurements should contribute to our understanding of N flows and transmission. As an example, a new series of experiments involving a series of grassland managements

(Table 9) has recently been initiated which is based on 1 ha farmlets and attempts to address the complexity of grassland production farm cycles within experimentally manageable areas. Each area is designed to contain elements of grazing, grass conservation and farm waste returns and measurements are made of herbage and animal production as well as intensive programmes to determine N losses and budgets. Whilst these farmlets and a range of managements (Table 9) provide a unique research opportunity for direct comparisons to be made in both environmental and production terms, it is critical that the controls over the processes that are deriving any changes in the flows of nutrients are fully appreciated. This will ultimately provide the means providing the model basis of extrapolating information to other systems or making prediction over longer time or large spatial scales.

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