Understanding the soil nitrogen cycle — Source link

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Abstract. A quantitative knowledge of nitrogen cycle processes is required to design strategies for decreasing leakage of N from agriculture to the wider environment. However, it is remarkably difficult to make reliable measurements of many of the key processes under realistic field conditions. In impermeable soils hydrologically separated plots provide an invaluable method of measuring leaching and runoff. Estimates of nitrate leaching using porous ceramic cups agree well with lysimeter measurements on sandy soil but are suspect on more structured soils. Estimates of N_2O flux from soil are subject to great spatial heterogeneity; developing long path-length measuring techniques may overcome this problem.

^{15}N labelling is valuable for assessing fertilizer N loss, forms of N left in soil and the fate of N from crop residues. The combination of experimental and modelling approaches can provide insights that are otherwise unattainable, including a basis for more precise advice on N fertilization.

Mineralization of soil organic matter and crop or animal residues provides much of the nitrate leached during winter under the climatic conditions of north-west Europe, because mineralization is poorly synchronized with crop N uptake. Maintenance of crop cover during winter can greatly decrease leaching but the long-term effects on the N cycle of winter cover crops or incorporating cereal straw are not yet clear.

INTRODUCTION

Many of the nitrogen cycle processes were discovered more than a century ago but unexpected aspects continue to be found. Past work was usually directed towards increasing the efficiency of use of nitrogen fertilizer in order to increase agricultural production. Current work is largely stimulated by concerns over the environmental effects of leakages of nitrogen from the soil, especially nitrate leaching and the production of nitrous oxide, but the two aims are by no means incompatible. Even where the processes and principles are clear (or are thought to be) it is now necessary to obtain reliable quantitative measurements of the key processes, under realistic field conditions, for a range of agricultural situations. This information is vital as a basis for rational judgments on ways of limiting the leakage of N from agriculture into the wider environment.

This article draws attention to the main N cycle pools and processes that are relevant to considerations of the environmental impact of agriculture, with particular reference to arable systems in the maritime climate of north-west Europe. For a more detailed consideration of the global N cycle see Jenkinson (1990a) and references therein.

CENTRAL ROLE OF SOIL ORGANIC MATTER

Table 1 shows current estimates of the quantities of nitrogen, in different forms, on a global scale. Soil organic nitrogen is the dominant pool in terrestrial ecosystems, being an order or magnitude greater than the nitrogen in plant biomass. Figure 1 shows this diagrammatically for an arable field; the figures are based on data from experimental fields at Rothamsted but are broadly representative of many arable fields throughout north-west Europe. An obvious feature is the very large size of the soil organic nitrogen pool: the surface layer of an arable soil typically contains 2000–6000 kg N/ha in organic matter. The amount reflects the past history of the soil and its mineral composition, being greatest in soils that have had long periods under grass or forest and contain much clay which can stabilize organic matter. Some fractions of soil organic matter are very stable,

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Table 1. Estimates of the active pools in the global nitrogen cycle (from Jenkinson, 1990a)

	N (10^6 tonnes)
Air	
N_2	3,900,000,000
N_2O	1400
Land	
Plants	15,000
Animals	200
of which people	10
Soil organic matter	150,000
of which microbial biomass	6000
Sea	
Plants	300
Animals	200
In solution or suspension	1,200,000
of which NO_3-N	570,000
of which NH_4-N	7000
Dissolved N_2	22,000,000

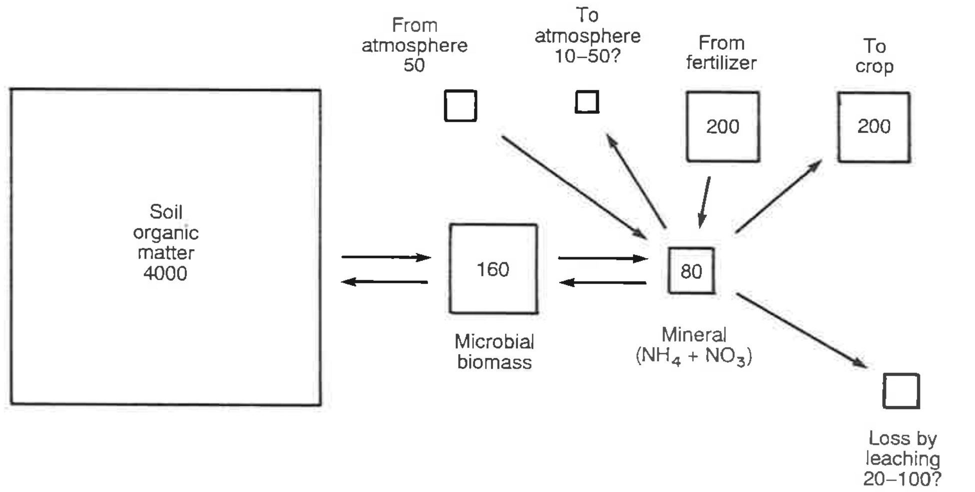

Fig. 1. Nitrogen pathways in soil. The quantity of nitrogen in each pool (kg/ha) or undergoing each process (kg/ha/yr), is proportional to the size of the square.

having half-lives measured in hundreds or even thousands of years. Other fractions, such as remnants of fresh plant or animal material, are easily broken down by micro-organisms to yield carbon dioxide and inorganic nitrogen over periods of days or weeks; Jenkinson (1990b) described mathematical models of soil organic matter turnover that reflect this diversity of activities. The older more stable fractions cannot be ignored because even the slow decomposition of a very large pool can produce a significant quantity of inorganic N. Indeed there is evidence that the long-continued decomposition of organic matter that follows the conversion of old grassland to arable production is a significant source of the nitrate currently entering certain aquifers (Whitmore *et al.*, 1992). In addition to the slow component of the biological process of decomposition, the physical movement of water and nitrate through certain rocks, particularly unfissured chalk, is also slow. Thus, the full consequences of a major change in land use on the nitrate content of natural waters may only become apparent many years later.

MINERALIZATION AND IMMOBILIZATION

Even when no net change in total soil organic N is occurring, N is both entering the organic pool (the process of immobilization) and leaving it as inorganic forms (mineralization), both being mediated by the soil microbial population. Figure 2 shows, diagrammatically, the time-course of mineralization during a year in a typical agricultural soil under the climatic conditions of north-west Europe. What is shown is the *net* rate (in kg N/ha/day) of mineralization, i.e. the excess of mineralization over immobilization. During winter, mineralization is limited by low temperature but as soil warms in spring the rate increases rapidly, roughly coinciding with increased crop growth—for comparison Figure 2 includes typical rates of N uptake by winter wheat and grass. N mineralized during spring has a reasonable chance of being absorbed by the crop, although some losses will occur as with fertilizer applied at this time. Mineralization usually continues long after uptake by an arable crop has ceased, causing a considerable accumulation of nitrate during the late summer, autumn and early winter; values of 30–100 kg N/ha as nitrate to a depth of 1 m are common in arable fields. Only a small fraction of this is absorbed by an autumn-sown crop and the remainder is exposed to leaching throughout the late autumn, winter and early spring period when soil is generally saturated. In soils under grass, the late flush of growth removes some residual nitrate, thus decreasing winter leaching, but if the grass is grazed nitrate formed from mineralization and nitrification of N in animal excreta adds considerably to the quantity present and can lead to large leaching losses (Jarvis, 1993).

The addition of various organic materials to soil can profoundly influence the balance between immobilization

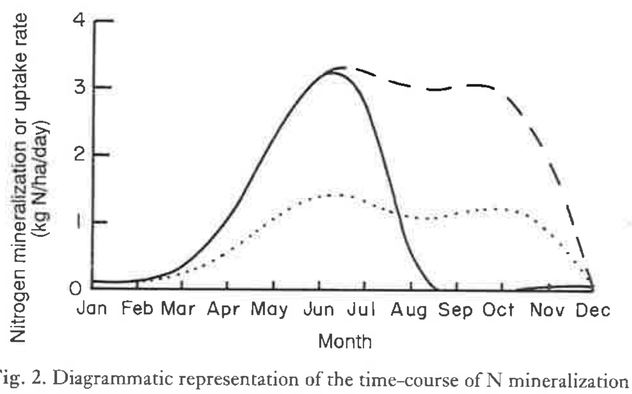

Fig. 2. Diagrammatic representation of the time-course of N mineralization (······) and uptake by winter wheat (———) and grass (- - - - - -) showing rates in kg N/ha/day.

and mineralization in the short term. Materials rich in N (i.e. having a narrow C to N ratio) favour mineralization; these include residues of legumes, animal slurry and organic fertilizers based on dried blood or other proteins. Those with a low N content (i.e. a wide C to N ratio such as cereal straw) favour immobilization, because such materials contain too little N, at least in readily decomposable forms, to satisfy the requirements of the microbial population responsible for their decomposition. This requirement can be met by absorbing inorganic N from the surrounding soil and, in the short term, this can decrease the quantity of nitrate at risk to leaching (e.g. Powlson *et al.*, 1985; Bertilsson, 1988). However, recent studies on N immobilization following incorporation of wheat straw have cast some doubt on the mechanism by which this occurs (Ocio *et al.*, 1991). The combined use of ^{15}N labelling and an appropriate mathematical model can provide a method for estimating gross rates of mineralization and immobilization (Barraclough, 1991).

THE PROCESSES CAUSING LOSS OF NITROGEN

There are two main processes of nitrate loss from soil—leaching (and/or surface runoff in some situations) and denitrification. Under the climatic conditions of north-west Europe, leaching is mainly a winter phenomenon starting when soils reach field capacity in autumn and ending as they begin to dry in spring. In colder regions or in those with a more continental climate, where soils are frozen for much of the winter, leaching or surface runoff tends to occur as a flush in spring as soil and snow thaw.

Throughout the world there is political and legislative pressure to decrease leaching and runoff of nitrate from agricultural land in order to meet maximum permitted concentrations of nitrate in drinking water, such as 50 mg/l in the European Community or 45 mg/l in the USA, equivalent to 11.3 or 10 mg N/l, respectively. Whether or not these limits are justified on medical grounds has been hotly debated and is discussed by Addiscott *et al.* (1991). An environmental reason for limiting nitrate leakage is to minimize eutrophication of surface waters. Although eutrophication of fresh water is usually limited by phosphate, nitrogen can sometimes be the limiting factor in estuaries and in seas subject to limited mixing by currents, such as the Baltic (Schrøder, 1990).

Denitrification, the other main loss process affecting nitrate, is the conversion of nitrate to a mixture of nitrogen (N_2) and nitrous oxide (N_2O) which is evolved to the atmosphere. This can occur in soil that is wet (but by no means waterlogged) and sufficiently warm for microbial activity (Fillery, 1983; Sahrawat & Keeney, 1986). Denitrification has long been recognized as loss of a valuable plant nutrient but, more recently, environmental problems associated with nitrous oxide have been recognized. Nitrous oxide is a greenhouse gas, each molecule being 150 times more effective at causing warming than carbon dioxide. It is thought that this gas contributes 5–10% of current global warming and it is also involved in reactions that lead to ozone depletion in the stratosphere (Warneck, 1988). Much more information is needed on the soil conditions that lead to production of nitrous oxide, as opposed to nitrogen, during denitrification and also on the significance of nitrous oxide production during nitrification (Bremner & Blackmer, 1978). It is important to ensure that future changes in agricultural practice, including those designed to decrease nitrate leaching, do not lead to an increase in nitrous oxide production because they increase the residence time of nitrate in soil. Nitric oxide (NO) and nitrogen dioxide (NO_2) also influence atmospheric chemistry and can be evolved from and absorbed by soils and crops. As yet information on the factors involved is rather limited (Jenkinson, 1990a).

Ammonia volatilization represents a major loss of nitrogen from agricultural systems involving animals (Jarvis & Pain, 1990). It has also been found that some ammonia can be evolved from the foliage of arable plants. This may be serious in crops that are heavily over-fertilized with nitrogen, or are badly diseased (Goulding *et al.*, 1993), but in normal crops the quantities involved are probably less than 10 kg N/ha, although there is some evidence to the contrary (Schorring *et al.*, 1989). Ammonia can also be lost if urea fertilizer is applied to the surface of soils under certain conditions, for example dry calcareous soils (Fenn & Hossner, 1985). Ammonia volatilization can often be a major cause of loss where urea fertilizer is used in paddy rice systems, but it seems to be much less common in temperate agriculture.

THE FATE OF N FROM INORGANIC FERTILIZER IN TEMPERATE ARABLE AGRICULTURE

Losses during the growing season

Field experiments in which different rates of N fertilizer are applied to crops can yield valuable information on the fate of fertilizer N. For example, the proportion taken up by the crop can be calculated, and inorganic residues left in soil can be measured. The use of the heavy isotope of nitrogen, ^{15}N, permits two types of experiment to be conducted. First, an additional ^{15}N balance can be constructed in order to calculate the quantity of fertilizer nitrogen lost from the crop–soil system, and second, the forms of N left in soil or crop residues, and their subsequent fate, can be traced. Figure 3 shows an example of ^{15}N balances in field experiments with winter wheat and shows how the distribution of fertilizer N can vary greatly from year to year. In a series of experiments in which ^{15}N-labelled fertilizer was applied, in spring, to winter wheat on three different soil types, recovery of fertilizer N in the above-ground parts of the crop ranged from 46 to 87% with a mean of 68%. The proportion retained in soil was remarkably constant, averaging 18% where N was applied as $^{15}NH_4{}^{15}NO_3$, but less (7–14%)

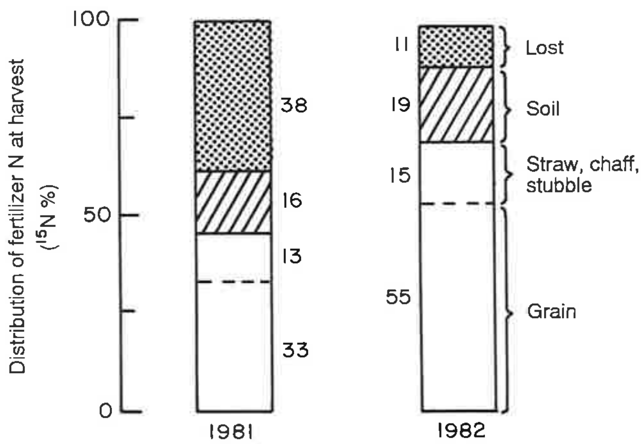

Fig. 3. Distribution of ^{15}N at the time of harvesting winter wheat at Saxmundham, Suffolk (Beccles series soil) in separate experiments in 1981 and 1982. Labelled fertilizer was applied as $^{15}NH_4{}^{15}NO_3$ in mid-April.

where it was applied entirely as nitrate. The proportion not recovered in either crop or soil varied between years from 2% to over 35% (Powlson *et al.*, 1992). There was a linear relationship between loss of fertilizer N and rainfall in the 3-week period following application; each additional 10 mm of rain increased the loss by 2.6% (Fig. 4). On the basis of this relationship it was suggested that the second part of a divided dressing could be adjusted to take account of whether or not significant losses would have affected the first application.

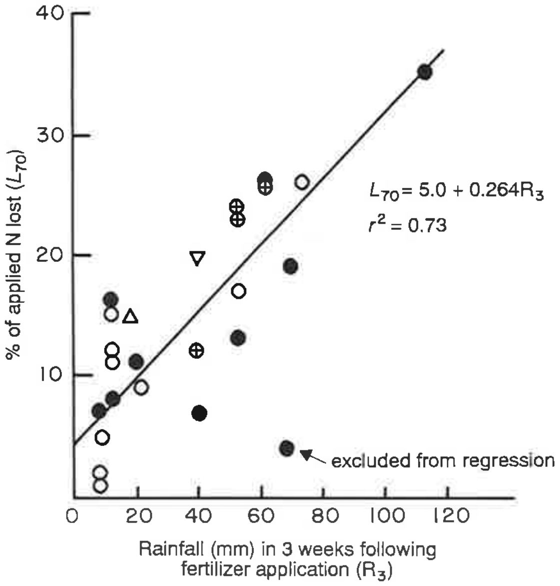

Fig. 4. Relationship between rainfall after application of ^{15}N-labelled fertilizer to winter wheat (R_3) and percentage loss (L_{70}) of fertilizer N from the crop–soil system. Loss defined as labelled N not recovered in crop or soil to a depth of 70 cm at the time of harvest. Different symbols indicate rates of fertilizer N known to be above (⊕) or below (○) that giving maximum grain yield of winter wheat. All other results indicated by ●. Results included for experiments with forage maize (▽) and winter oilseed rape (△) but not used in calculating regression (Powlson *et al.*, 1992).

Rainfall could favour loss by increasing leaching, denitrification, or both. A major limitation of ^{15}N balance experiments is that they give no direct information on the cause of loss unless other measurements of specific processes are conducted concurrently. Addiscott & Powlson (1992) proposed a method for distinguishing between the two loss processes based on the use of a mathematical model for leaching (Addiscott & Whitmore, 1987). In three of the 13 experiments leaching seemed to have been the major cause of loss, but in the others there was too little movement of water down the soil profile after spring fertilizer application to cause substantial leaching (Fig. 5). In all but two experiments there was good circumstantial evidence that the non-leaching loss was caused by denitrification. These studies, together with direct measurements of leaching (Dowdell & Webster, 1984; Goss *et al.*, 1988) indicate that leaching of fertilizer N shortly after its application to crops in spring is not a major cause of loss, at least in the climatic conditions of southern and eastern England. In wetter regions, where the main leaching period extends further into the spring, the situation could well be different.

N applied to spring crops at the time of sowing remains in the soil for several weeks before uptake begins. Nitrate is then at greater risk of loss than that from equivalent applications to autumn-sown crops that are already established. Nitrate from very early applications, such as those sometimes given to oilseed rape in February, is present in soil, while there is still serious risk of leaching even under the relatively dry conditions of south-east England.

Fertilizer residues remaining in soil

The use of ^{15}N allows the forms of residual fertilizer-derived N in soil to be identified and their subsequent fate studied. In some recent experiments with winter wheat (MacDonald *et al.*, 1989) at least 80–90% of the fertilizer-derived N in soil at harvest was in organic forms—roots, stubble, root exudates, microbial cells, metabolites and humus. For crops

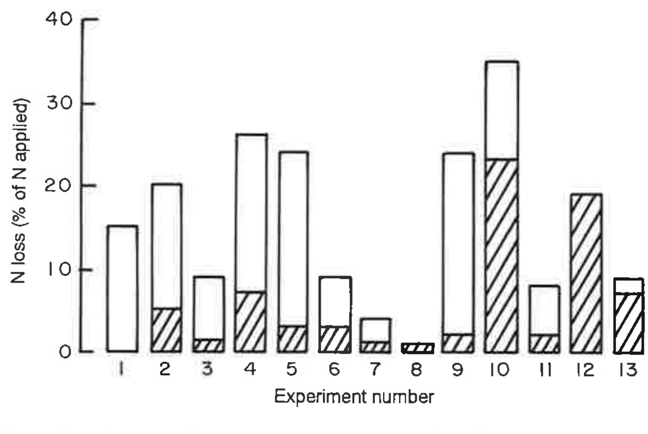

Fig. 5. Partitioning of N loss between nitrate leaching (▨) and gaseous loss (□). Calculated for a fertilizer application of 150 kg N/ha applied in spring to winter wheat in 13 separate experiments (Addiscott & Powlson, 1992).

given fertilizer N at currently recommended rates, it was rare for more than 5 kg/ha of fertilizer-derived N to be left in soil as nitrate unless severe drought or disease greatly decreased crop growth. Usually more than 90% of the inorganic N in soil at this time was unlabelled, indicating that it was derived from sources other than the current year's fertilizer, mainly mineralization of soil organic matter. Not surprisingly the amount tended to be greatest in soils of large organic matter content, or those which had recently been ploughed out of a ley or had received heavy dressings of organic manures. The reason for the predominance of soil-derived nitrate in the profile after harvest is the lack of synchronization between mineralization and crop uptake, as discussed above and illustrated in Figure 2. Table 2 shows some results from a series of experiments with crops other than cereals. These show a different pattern. With potatoes more inorganic N derived from fertilizer remained in soil after harvest, and the total quantity of inorganic N left in soil was also greater than with winter wheat. The situation with oilseed rape was intermediate.

The quantity of fertilizer N applied to a crop can affect the amount of nitrate present in soil and at risk of leaching during the following autumn and winter. However, the relationship between rate of N application and residual nitrate is not a simple linear one. Glendining *et al.* (1992), studying plots of the Broadbalk Wheat Experiment, found

Table 2. Labelled and unlabelled N in crop and soil after growing winter wheat, winter oilseed rape and potatoes on a flinty silty clay loam soil (MacDonald *et al.*, 1990)

	N content (kg N/ha)		Percentage of labelled N applied
	^{15}N labelled	Unlabelled	
	Winter wheat		
In above-ground crop	124	80	55
In soil (0–100 cm)			
Organic	41	36†	18
Inorganic	6	49	3
Total ^{15}N accounted for	176	—	76
^{15}N lost	48	—	24
	Oilseed rape		
In above-ground crop	114	70	48
In leaf litter	8	7	3
In soil (0–100 cm)			
Organic	51	36†	22
Inorganic	10	62	4
Total ^{15}N accounted for	183	—	77
^{15}N lost	54	—	23
	Potatoes		
In tops	31	31	14
In tubers	87	60	39
In soil (0–100 cm)			
Organic	33	33†	15
Inorganic	16	95	7
Total ^{15}N accounted for	167	—	75
^{15}N lost	56	—	25

†Unlabelled N associated with the labelled N residue in soil was estimated by assuming that the ratio labelled:unlabelled N was the same as in stubble for wheat and rape, or in tops for potatoes.

that soil nitrate content measured at harvest or later was approximately constant for N applications between nil and that above which there was little further increase in crop yield: at higher rates residual nitrate increased sharply. Chaney (1990) observed a similar pattern in experiments with winter wheat although, in contrast to the Broadbalk results, the increase in residual nitrate did not occur until well after the estimated optimum economic rate of N.

Long-continued applications of N fertilizer over many years can have a significant impact on the soil N cycle. Crops grown with larger rates of N leave larger quantities of N in organic residues (e.g. straw, stubble, roots), which may increase the amount of soil organic N and the amount of nitrate formed from mineralization, some of which may be leached. This may account for recent observations of a more linear relationship between N fertilizer rate and nitrate leaching in the Broadbalk Experiment (K.W.T. Goulding & C.P. Webster, unpublished data).

Post-harvest mineralization of crop residues has a major effect on the quantity of nitrate at risk of leaching. The extent of mineralization depends on the composition of these residues and the method of incorporation or disposal. The use of ^{15}N permits these transformations to be studied and quantified, even in the presence of a very large background of soil organic N. In the experiments of Macdonald *et al.* (1990) the rate of release tended to be least following winter wheat with straw removed; 29% of the labelled residue was mineralized and either lost from the soil–plant system or absorbed by the following crop. The corresponding values were 46% for potatoes (residues incorporated), 36% for oilseed rape (residues removed) and 39% for sugarbeet (tops incorporated).

Following the fate of labelled N in crop residues and that which has recently entered the soil facilitates the development of mathematical models of soil N turnover (Jenkinson & Parry, 1989). These can form the basis of practical models designed to improve predictions of the fertilizer N requirements of specific crops or to assess the nitrate leaching risk associated with different cropping systems (Bradbury *et al.*, 1990; Whitmore *et al.*, 1991).

INPUTS OF N FROM THE ATMOSPHERE

It has been known since the beginning of this century that rain contains nitrogen in the forms of ammonium and nitrate. More recently it has been established that there are substantial inputs of N to the crop–soil system from dry deposition of gases such as ammonia and oxides of nitrogen. Goulding (1990) estimated that 50–60 kg N/ha/yr enters cereal growing systems at Rothamsted. These inputs are partly offset by gaseous losses giving a net annual input of about 40 kg N/ha. The fate of this input has been calculated using mathematical models for the nitrogen cycle of the Broadbalk Experiment (Whitmore & Goulding, 1992). These show that 51% of the atmospheric input is taken up by the wheat crop on the plot receiving 192 kg N/ha as fertilizer, but 29% is leached. This represents 29% of the estimated total leaching from the system—a large proportion considering atmospheric inputs are only 17% of total N inputs.

DIFFICULTIES OF MEASUREMENT

Almost all of the nitrogen cycle processes are difficult to measure accurately in field conditions. This hampers research and means that conclusions are often based on deductions from indirect observations. Even the use of ^{15}N as discussed above is not without its difficulties and results must be interpreted with care.

Nitrate leaching

The measurement of nitrate leaching is surprisingly difficult. The collection and analysis of water from drains in hydrologically separated plots, as in the Brimstone Experiment, provides invaluable information on leaching in an impermeable clay soil, but the technique cannot be applied to freely drained soils. During the first 10 years of the Brimstone Experiment, when autumn-sown crops were grown, over-winter leaching losses were usually in the range 20–50 kg N/ha. Winter losses averaged 24% less from land that had been direct drilled instead of ploughed (Goss *et al.*, 1988). Only in one year did leaching in spring, after N fertilizer was applied, exceed 10 kg N/ha. More recently nitrate leaching has been measured in a number of different crop rotations; Table 3 shows some results for one year taken from Catt *et al.* (1992). Leaching was greatest when soil was bare during the winter (the Fallow/Spring wheat treatment), and any crop cover during winter decreased leaching to some extent. However, nitrate concentrations in many of the drainflow samples collected over the last 13 years have exceeded 50 mg nitrate/l, even in treatments involving good agricultural practices to minimize leaching.

The measurement of nitrate concentrations and water flow in streams and rivers draining a well-defined catchment offers a method for assessing nitrate leaching under realistic conditions. An example was given by Burt & Arkell (1987). As with hydrologically isolated plots, the method is only applicable to impermeable soils in which almost all of the water leaving the catchment does so via the monitored rivers and streams with little vertical leaching.

Table 3. Nitrate N losses (kg/ha) in winter drainflow at the Brimstone Experiment in 1988–89 (from Catt *et al.*, 1992)

Crop	Mean loss
Grass	9.8
Fallow/Spring wheat	25.0
Mustard/Spring wheat	10.6
Winter oats (straw burnt, ploughed)†	10.2
Winter oats (straw incorporated, ploughed)†	6.3
Winter oats (straw burnt, shallow tined)†	6.1

†Straw disposal and cultivation methods applied prior to establishment of winter oats in autumn 1988.

On freely drained soils, where isolated drained plots cannot be used, lysimeters are a well-established method of measuring nitrate leaching directly. They were used by Dowdell & Webster (1984) to monitor leaching losses from a shallow chalky soil growing spring barley: annual leaching losses were between 65 and 83 kg N/ha and, again, occurred mainly in winter.

An alternative technique for estimating nitrate leaching is to measure nitrate in samples of soil water and combine this with a measure or estimate of water flow. Porous ceramic probes are now being used widely to obtain samples of soil water although their use has preceded understanding of their mode of action. Addiscott (1990) drew attention to several difficulties with porous cups, not least of which is extracting any water at all from some heavy textured soils.

On a sandy soil, Goulding & Webster (1992) found that estimates of nitrate leaching obtained from porous probes and lysimeters were in good agreement provided differences in water flow between lysimeters and the open field were taken into account. There was a delay in drainage from the lysimeters because of the absence of matric suction. An initial comparison of nitrate concentrations in water collected in porous cups and from drains in a more structured soil showed very poor agreement.

An indirect approach to assessing nitrate leaching is sequential sampling of the soil profile to measure inorganic N combined with the use of mathematical models for mineralization and leaching (Powlson *et al.*, 1989). If agreement between measured and simulated results is good it is assumed that the modelled value for the quantity of nitrate leached below the sampling depth is a reasonable estimate of actual leaching. This approach is not without pitfalls as it is possible that the model could match the data for erroneous reasons, for example if *both* mineralization and leaching were over- or under-estimated. Despite this and other limitations, it does offer a means of estimating leaching losses using readily available data if direct measurements cannot be made.

Gaseous fluxes

Measuring gaseous emissions from soil, under field conditions, presents formidable difficulties. The measurement of an N_2O flux from soil is relatively straightforward using a cover box and gas chromatographic measurement, but measuring total denitrification loss, N_2O plus N_2, against an atmospheric background of 78% N_2 is not. Because the ratio of N_2O to N_2 can vary, it is not possible to estimate total N loss from N_2O measurements alone, except in acid soils where almost 100% of the total denitrification loss is as N_2O. The discovery (Yoshinari & Knowles, 1976) that acetylene blocks the reduction of N_2O to N_2 was a considerable breakthrough as the total products of denitrification can be measured as N_2O. This has formed the basis of a field technique (Ryden *et al.*, 1979) used successfully on medium- and light-textured soils but thought to have limitations in soils of high clay content in which gaseous diffusion can be very slow. A variant of the technique is to treat intact soil cores with acetylene in a bottle that is incubated at field temperature (Ryden *et al.*, 1987). This is undoubtedly a valuable technique for com-parative studies but the validity of the absolute denitrification rates obtained is uncertain (Webster & Goulding, 1989).

A serious problem in measuring denitrification, or N_2O flux from soil, is spatial heterogeneity. Parkin (1987) demonstrated that denitrification occurred at 'hot spots' in soil and replicate measurements commonly vary by an order of magnitude. An approach now being developed is to use long path-length techniques for measuring N_2O with tunable diode lasers (Measures, 1989; Jarvis, 1990). Techniques for measuring ammonia fluxes from soils or crops, using micro-meteorological methods or open-ended chambers, are well developed and widely used (e.g. Jarvis & Pain, 1990).

STRATEGIES TO DECREASE NITRATE LEACHING

The underlying principles to be followed in attempts to decrease nitrate leaching are now becoming much clearer but translating these into practical strategies presents considerable problems. In addition to balancing the agricultural and environmental considerations, which can often conflict, the effects of a given strategy on nitrate leaching and gaseous emissions must also be balanced. The long-term implications of any strategy and the overall effect at the farming system level must also be considered. Areas needing consideration include the rational use of inorganic nitrogen fertilizer, the balance between different N cycle processes in autumn, and the impact of crop cover during winter. The management of organic sources of N, especially animal manures, is also of great importance and is discussed in detail by Jarvis (1993) and Smith & Chambers (1993).

Inorganic N fertilizer

Although excessive applications of inorganic N fertilizers are not the major cause of the nitrate leaching problems now being encountered, this does not imply that there is no relationship at all between fertilizer use and leaching. A 'breakthrough' effect, with residual nitrate increasing sharply above a certain optimum rate, has been observed for grassland (Barraclough *et al.*, 1992) and winter wheat (Chaney, 1990; Glendining *et al.*, 1992). It is therefore important to improve current systems of providing advice to farmers on a field-specific basis, but this raises several difficult problems. First, it is not simple to predict opitmum N rates. Much current research is directed towards improving the accuracy and precision of fertilizer recom-mendations based on a better quantitative understanding of the nitrogen cycle in soil, especially the supply of N from the mineralization of organic matter (e.g. Jenkinson & Parry, 1989; Whitmore *et al.*, 1991). Second, even with the best possible prediction made in spring, prior to fertilization application, factors later in the season (e.g. drought, disease) may decrease crop growth and uptake of N and lead to larger nitrate residues than expected. It could therefore be argued that it would be prudent, at least in sensitive areas, to apply less N fertilizer to wheat than currently recommended in order to decrease the frequency with which this occurs. Clearly this has economic implications which are recognized in the UK Nitrate Sensitive Areas Scheme. Thirdly, the long- continued use of high rates of N fertilizer undoubtedly leads to an accumulation of potentially mineralizable organic N in soil. This can lead to more nitrate being available for both crop uptake and loss and is currently being studied in the Broadbalk Experiment at Rothamsted (Glendining *et al.*, 1992; Goulding & Webster, 1992) and by ADAS at Ropsley in Lincolnshire (Sylvester-Bradley & Chambers, 1992). With some non-cereal crops it appears that fertilizer makes a more direct contribution to residual nitrate, and hence leaching, than with cereals. This is especially so for some vegetable crops which give an economic response to large dressings of N fertilizer but use it very inefficiently (Rahn, 1992). It may be that the area under crops known to leave larger residues of nitrate, or readily decomposable organic N, will need to be limited in the most sensitive catchment areas. Computer models of N cycle processes that integrate from the field to the catchment level should be of great value in planning any such restrictions.

Crop cover and N cycle processes in autumn and winter

The presence of a growing crop during the autumn and winter period will generally decrease leaching compared with that from bare soil (see, for example, Table 3). Sowing a crop in autumn, rather than in spring, is often the simplest practical method of minimizing leaching but it is not a panacea. To be effective, an autumn crop must be sown early but this is not always possible. Indeed, early sowing can conflict with other aspects of good husbandry such as the control of weeds, with minimal use of herbicides, or the carryover of pests or diseases.

Winter cover crops can be effective in decreasing nitrate leaching where soil would otherwise be bare prior to sowing a spring-sown crop. These will almost certainly play a valuable role in strategies to decrease leaching but various practical difficulties still remain to be solved if they are to be used reliably in a wide range of soil types and agronomic situations (Christian *et al.*, 1992). It is particularly important to discover more about the time-course of N release following the incorporation of cover crops in spring. A proportion of the N released will not be well synchronized with uptake by the subsequent crop and will be subject to leaching; evidence of this has been noted in the Brimstone Experiment (Catt *et al.*, 1992).

The various possible combinations of method and timing of cultivation and method of disposal of crop residues need to be carefully assessed because they have a major influence on N mineralization. There is evidence that delaying cultivation can decrease mineralization (Stokes *et al.*, 1992), but this is incompatible with early sowing of the next crop.

Incorporation of low-N crop residues, such as cereal straw, causes some immobilization of N into the soil microbial biomass leaving slightly less at risk of leaching. However, in the long term, this additional organic N in soil leads to an increase in the basal mineralization rate of the soil (Powlson *et al.*, 1987). Various agronomic packages suitable for a range of different crop rotations, soil types and local weather conditions need to be designed. They must be based on a sound knowledge of the N cycle, take full account of any long-term implications and also be practical.

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Minimizing nitrate losses from arable soils

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Abstract. Recent experiments on soils overlying sand, chalk and limestone aquifers have shown that nitrate leaching losses can be decreased by modifying crop husbandry. Green cover during winter, if established early enough, can reduce nitrate loss. Cultivations can be timed to minimize leaching, and the advantages of irrigation (increased nitrogen offtake and smaller post-harvest soil mineral nitrogen residues) outweigh the potential disadvantage of increased leaching risk during the growing season. It is important not to over-fertilize crops. Using these techniques within farm rotations has decreased nitrate losses in small plot experiments. The next step is to measure the effects on commercial farms where the scale of operation might preclude the high level of husbandry that is required for successful nitrogen management.

INTRODUCTION

Legislation by the European Community (EC) and the UK Government increasingly aims to minimize the amount of nitrate leached from agricultural soils to aquifers. Almost certainly this will require growers to modify farming practices. The most extreme solution would be to convert productive arable land into low input grassland. Before this is advocated on a large scale it is necessary to ascertain whether less disruptive changes would decrease nitrate leaching to an acceptable level. Many experiments have investigated a single aspect of nitrogen management. Although this is a valuable approach, there is a need to incorporate these principles into whole rotations to measure the success of simple modifications for decreasing nitrate loss. The development of integrated practices to decrease nitrate leaching is therefore a high priority for agricultural research (Anon., 1992). Early results from experiments on three of the most important aquifer types (sandstone, limestone and chalk) have shown how farming practices can affect nitrate loss (Davies & Rochford, 1992; Shepherd *et al.*, 1992). These experiments also provide information on the commercial viability of the practices. We review results from these and other experiments and discuss the practicalities of nitrate management within arable rotations.

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MEASUREMENT OF NITRATE LEACHING

Measuring the true amount of nitrate leached from an area of land is difficult. Before the effects of changed farming practices on nitrate leaching could be assessed, inexpensive and reliable methods of measurement had to be developed. Work at both ADAS Gleadthorpe and the AFRC Institute of Arable Crops Research, Rothamsted, has shown that porous ceramic cups, buried at depth to sample soil water, are satisfactory for measuring nitrate leaching on light soils (Webster *et al.*, 1993). This was an important step forward, as previously lysimeters were required, which are expensive to install, limited in application and subject to artificial conditions (Wagner, 1962). Porous cups are now used in many field experiments (e.g. Jarvis *et al.*, 1987; Cuttle, 1992) and within the Nitrate Sensitive Areas (NSAs) of the Ministry of Agriculture's Pilot Nitrate Scheme (Archer, 1992).

Soil sampling for mineral nitrogen can also help to indicate leaching potential if measured in autumn, although this does not provide a direct measure of leached nitrogen.