# NUTRIENT CYCLING WITHIN TEMPERATE AGRICULTURAL GRASSLANDS

D. Scholefield[1] and O. Oenema[2]

[1] Institute of Grassland and Environmental Research, North Wyke, Okehampton, Devon, EX20 2SB, UK.

[2] DLO-Research Institute for Agrobiology and Soil Fertility (AB-DLO), P.O. 129, NL-9750 AC Haren, The Netherlands

## ABSTRACT

Productive grasslands are no longer regarded as environmentally benign, especially where high stocking rates are employed and the storage and disposal of large amounts of farm wastes are necessary. The nutrient cycles within the soil/plant/animal system are both complex and inefficient, particularly for nitrogen, with poor assimilation by both the plant and animal components under intensive managements. While all transfers are important to the overall efficiency of nutrient use within the system, it is ultimately plant uptake from the soil that determines both economic output and environmental impact. This paper focuses on the soil as a medium for manipulation and management to effect improvements in the efficiency of nutrient use. First we present some quantitative information about the various inputs to, pools and losses from different kinds of system, with emphasis on the nitrogen cycle. Secondly, a concept of the efficiency of nutrient use is developed to provide a simple quantitative framework with which to evaluate different systems and strategies that might improve efficiency. Some strategies currently being evaluated are described and are identified as being of two types: those which merely reduce inputs, and those which improve the inherent efficiency of one or more transfers within the system. If production targets are to be reconciled with environmental constraints for intensively managed systems, type 2 strategies will need to be developed. This will involve much basic science, the development of new kinds of mathematical model and very effective methods of technology transfer.

## KEYWORDS

Waste management, nutrient cycles, environment, resource inputs, soil, models

## INTRODUCTION

The cycles of the nutrient elements within a grazed grassland system are extremely complex and, although they interact one with another to varying degrees, we have tended to study each in isolation. Currently, our greatest knowledge is of the nitrogen (N) cycle, because herbage production in most agricultural grassland systems is N limited (Wedin, 1996), because of the tremendous importance of the N-fixing legume in agriculture generally and because of growing concerns about the environmental impact of N losses (e.g. the EC and WHO Nitrate Directives). During the past decade there has been considerable progress in the quantification of the N pools and fluxes, due mainly to the development of suitable field techniques for the measurement of N losses and of net N mineralization (Jarvis et al., 1996a). These new data have been used in the construction, calibration and testing of mathematical models of the pasture N cycle (Parton et al., 1987; Thornley and Verberne, 1989; Scholefield et al., 1991; van der Ven, 1996). These are now being applied to identify novel management strategies for achieving reduced N losses while maintaining economic levels of production.n (Kriel and Oenema, 1990; Oenema et al., 1992; Cuttle and Scholefield, 1995; Jarvis et al., 1996b).

Progress towards greater understanding and quantification of the cycles of other macro-nutrients has been less rapid compared to that with N. With phosphorus (P), for example, while there is a general concern about its contribution to the eutrophication of surface waters (Vollenweider, 1968) there is yet a rather poor appreciation of mechanisms controlling either plant availability (Harrison, 1985) or transport within the soil (Haygarth and Jarvis, 1996). Nevertheless, simple models of the P cycle in grazing systems have been developed (Haynes and Williams, 1993a) as aids to management and further study. Research into the cycling of potassium (K) and sulphur (S) has lacked the impetus of environmental concern that stimulated research into N and P cycling. There is a growing awareness however of soil K and S deficiencies due to reduced inputs from fertilizer and the atmosphere, and to large losses by leaching and in runoff, and that these deficiencies could be responsible for sub-optimal efficiency of N use.

Because of its complexity, the study of nutrient cycling has tended to be fragmented, with scientists focusing on their own areas of expertise. Now, with the development of mathematical models it should be possible, in principle, to obtain a detailed understanding of how the processes operating in each nutrient pool affect the outputs from the whole system, and thus how these processes might be manipulated for desired goals. Unfortunately however, the modellers in turn have also tended to focus on discrete parts of the system, such that there exist few models, except very simplified ones, that have been developed equally well for all parts of the system. Despite these constraints, there is sufficient knowledge of the N cycle at least, to enable the trade-offs between production and environmental impact to be quantified at different levels of input to a grassland system. Thus, a question of major importance that this knowledge should be directed towards is under which conditions, if any, can economic levels of production be reconciled with acceptably small environmental impacts? To answer this question demands definition and quantification of expressions of the efficiency of nutrient use, which in turn can be used as measures of the sustainability of the system.

While there have been recent comprehensive reviews of nutrient and N cycling in grazed grassland (Haynes and Williams, 1993a; Whitehead, 1995; Russelle, 1996), it is our purpose, in this paper, to pursue the following somewhat narrower objectives:

(i) identification and quantification of those pools and fluxes critical to the efficiency of nutrient use in the field;

(ii) reporting technically feasible management strategies that might be applied to increase efficiency of nutrient use, and

(iii) discussion of future research needs.

The arguments are presented from a soil scientist's perspective, with emphasis on the N cycle; although where possible, the relevance of these arguments to P, K and S cycling are pointed out.

## NUTRIENT INPUTS, POOLS AND FLUXES

The efficiency of nutrient cycling and transformations within the soil-plant-animal system and the balance between nutrient inputs (fertilizer, feed) and outputs (products, losses) are major factors determining the productivity and long-term sustainability of the system. Nutrients cycle between the major pools of soil, forage, animal, excrement/slurry and then back to soil again. The system loses nutrients as useful products (meat, milk), and also by ammonia volatilization, denitrification and leaching. These losses have to be compensated for by nutrient inputs of fertilizer, atmospheric deposition, N-fixation and purchased feed. The input:output ratios of the various pools vary widely, especially for N. However, the whole coupled system has typically a rather small output:input ratio: only a minor fraction of the total input is retained in useful products. Furthermore, this ratio tends to decrease with total input to the system, because the output:input ratios for all the pools tend to decrease. The soil is by far the largest pool and resource of nutrients and functions as a major buffer for the whole system. Much of the soil nutrient reserve is in organic combination and is made available for plant uptake only slowly through microbial degradation.

Quantitative information about the nutrients in the different pools and about the fluxes between and the losses from the pools is the key to understanding how the whole system operates and to the designing and implementation of novel systems with improved efficiency of nutrient use. The following sections describe current quantitative conceptual information with emphasis on the soil.

**Nutrient inputs**. Although nutrients can be taken up through the leaves of plants, for example, S can be readily taken up from the atmosphere as $SO_2$ (Cowling et al., 1973), the main pool of plant available nutrients is the aqueous phase of the soil. The rate of supply of nutrients into this pool and the plant's effectiveness in competing with other soil processes for them determine plant uptake and growth, in relation to the plant's genetic potential and the ambient conditions of light, temperature and soil water content. The other (competing) soil processes are those resulting in loss, such as volatilization and leaching, and those resulting in immobilization, either by microbial activity or by chemical and physical adsorption to soil colloids.

In order to maintain economic yields of herbage the supplies of the nutrients N, P, K and S to the soil solution are enhanced through top-dressings of mineral fertilizers. Fertilizer use for all crops throughout the world peaked in the late 1980's but has declined only at a slow rate since then. Fertilizer use for grassland agriculture in most temperate regions has followed this general trend. For example, estimates for application of N + $P_2O_5$ + $K_2O$ to forage crops and pastures in the USA were 3.6 million tons in 1970 (Beaton and Berger, 1974) and 2.8 million tons in 1992 (Griffith and Murphy, 1996). The average amounts N, $P_2O_5$ and $K_2O$ applied per hectare in 1992 ranged from 144, 64, 87 kg respectively, for corn silage to 34, 17, 17 kg, respectively for rangelands. In the UK and western Europe generally the average amounts applied have been considerably greater, particularly with intensive dairy farming. The current recommendations in the UK for intensive livestock farming are centred on 340, 60, and 60 kg $ha^{-1}$ under grazing management, and 380, 90, and 320 kg $ha^{-1}$ under cutting of N, $P_2O_5$ and $K_2O$, respectively (MAFF, 1994). In order to avoid over-fertilization some allowance is made in these recommendations for differences in site fertility, estimated from knowledge of previous managements, and for nutrients applied in manures and slurries. Nevertheless these amounts are aimed at achieving the 'economic optimum' yield for the site, (see Deenen and Lantinga, 1993) with little regard until very recently to the likely extent of nutrient losses and their environmental impacts.

Where grassland production is based predominantly on legumes, as in New Zealand for example, the main fertilizer requirement is for maintenance of soil P, K and S status. Amounts of N fixed by grass-legume swards comprising 25-50% legume can be as great as 285 kg $ha^{-1}$ (Boller and Nosberger, 1987), but annual applications of P ($P_2O_5$) and S ($SO_3$) of 80 and 40 kg $ha^{-1}$, respectively, are required to maintain economic outputs from these systems (Haynes and Williams, 1993b; Nguyen and Goh, 1993; Sinclair and Rodriguez Julia, 1993). In trials aimed at evaluating the use of foliar P anf K concentrations for diagnosing optimal fertilizer applications to grass/white clover swards in northern Spain, it was found that clover yields were particularly limited by K at higher levels of P input and that clover K/P ratios of 5.5-6.0 were associated with the highest yields (Rodriguez et al., 1992).

Fertilizer form could potentially have important effects on subsequent nutrient cycling. There is some evidence that ammonium is taken up more readily than nitrate when the soil temperature is below 9$^{o}$C (Clarkson et al., 1986; Stevens, 1988). Certainly, ammonium should be more resistant to loss by leaching because of its strong affinity for negatively charged surfaces. In an experiment where ammonium nitrate, ammonium phosphate and urea were applied in early spring, when prone to loss in runoff, there was only a small advantage indicated with the ammonium-rich forms in terms of yield of first cut silage (Scholefield and Stone, 1995). While urea is now more widely used than ammonium nitrate for forage production, possibly because it is cheaper per unit of N, there is greater potential for N loss as ammonia gas from urea applied under warm dry conditions. (Griffith and Murphy, 1993).

Triple superphosphate is now the most widely used form of mineral P fertilizer. It provides readily available P but is deficient in S compared to single superphosphate, and therefore S deficiencies that have developed as a consequence of reduced industrial emissions can be exacerbated. Ground rock phosphate is the only permissible form of mineral P fertilizer in organic farming. It is cheaper and richer in P than triple superphosphate, but is generally less effective (Bolland and Gilkes, 1995) except when finely ground and partially acidulated (Rajan et al., 1993). Muriate of potash (KCl) is still the most popular K source for forage production, although potassium sulphate is used as an effective combined K and S source, particularly for legumes. Sulphur is applied as ammonium or potassium sulphate or, in New Zealand particularly, as elemental S. This latter S source is better applied in the autumn to allow time for effective oxidation (Ledgard et al., 1993)

**Plant uptake**. Plant response to increasing supply of applied fertilizer N follows a characteristic relationship, which is usually linear from zero to between 250 and 700 kg ha $^{-1}$ in north western Europe (t'Mannetje and Jarvis, 1990). The yield response per increment of N then gradually diminishes to zero, at which point N is no longer a limitation to yield (Fig. 1). Applications at levels above this point may result in continued uptake but reductions in yield. Further response to N may be obtained if deficiencies of other nutrients or soil water are rectified. Thus, the most efficient fertilizer strategies are those which optimise and balance all nutrients such that plant growth is limited only by genetic potential and ambient conditions. However, fertilizer levels have been dictated by the desire to obtain the economic optimum yield, which has been defined as the level of fertilizer N above which the yield return per unit of added N per ha falls below a certain value, for example 10 kg $ha^{-1}$ (Morrison et al., 1980; Deenen and Lantinga, 1993). The yield response to increasing fertilizer is displaced under grazing, relative to cutting, because of the considerable amounts of nutrients returned in dung and urine.

The pattern of nutrient uptake and herbage production through the growing season is determined by the physiology of the plant, its frequency of defoliation and the ambient conditions of temperature, light and soil water content. For the most efficient use of nutrients from fertilizer the pattern of application should be the same as the pattern of relative potential for uptake and growth. However, this may not be the most efficient pattern of herbage production for the farmer, who may need to use fertilizer at sub-optimal efficiency in drier months in order provide sufficient herbage for grazing dairy cows, for example. Fig. 2 shows the monthly pattern of fertilizer N application calculated for the most efficient use of 240 Kg ha$^{-1}$ applied annually to cut grass grown under average weather conditions in the south west of the UK. The distribution was calculated from analysis of N-yield data from a multi-site experiment (Morrison et al., 1980), in which cut swards received equal amounts of fertilizer N per cut.

N-yield distributions of a continuously grazed sward at the same location given equal applications of fertilizer N are displaced to later in the season, with maximum yields harvested in late June-early July (Tyson et al., 1992). The extent to which this displacement is due to the change in either the physiology of the plant or to the delayed supply of nutrients from excretal returns has not been ascertained.

The yields of herbage grown at the economic optimum levels of fertilizer input, or with optimal grass/legume combinations vary enormously, depending on soil properties, site conditions of light intensity, temperature and rainfall, plant species, sward management and patterns of weather. For example, dry matter yields of timothy may be 3.5 t ha$^{-1}$ in northern Europe (Nissinen, 1992), whereas in the US 10 t ha$^{-1}$ would be more commonly achieved (Griffith and Murphy, 1996). Deenen and Lantinga (1993), working in the Netherlands reported yields of 10.094 and 10.950 t ha$^{-1}$ under cutting and grazing respectively, for swards applied with 244 kg N ha$^{-1}$ and 14.756 and 14.601 t ha$^{-1}$ under cutting and grazing respectively for swards applied with 540 kg N ha$^{-1}$. They attributed the small difference between cutting and grazing at the lower fertilizer rate to the effects of excretal returns and better regrowth after grazing in late season. The N content of herbage can vary between about 1.5% and 4.5% in relation to annual N input, but also over the growing season, depending on N supply and the physiological state of the sward (e.g. Tallowin et al., 1990).

The N content of the legume component in mixed grass/legume swards is less variable and tends to be close to 4%, whereas the grass component is relatively N-deficient and is often less than 2%. This has led to the view that systems based on a grass/legume sward should use N more efficiently (Parsons et al., 1991; West and Mallarino, 1996). While this has been difficult to demonstrate experimentally (Scholefield and Tyson, 1992), a model developed by Hutchings and Kristensen (1995) predicts slightly smaller nitrate leaching losses under grazing from below a grass/white clover sward than from a N-fertilized grass sward giving the same level of production.

The P, K and S contents of both grasses and legume species in temperate grasslands are normally within the ranges 0.21-0.42%, 2.0-3.1% and 0.27-0.40%, respectively (Tyson et al., 1992; Ledgard et al., 1993; Martz et al., 1996). Typical nutrient offtake values in a harvested annual dry matter yield of 10 t ha$^{-1}$ would be 400, 35, 250 and 35 kg ha$^{-1}$ of N, P, K and S, respectively. It is important to realise that these values are not the total amounts of each nutrient taken up from the soil: to calculate uptake, allowance must be made for unharvested nutrients in roots and stubble and for the amounts continually being returned to the soil in senescent plant material. Both of these are difficult to quantify. Data for the partitioning of soil mineral N between roots and shoots (Ball and Field, 1987; Hansson and Pettersson, 1989)indicate that about 62% of the total N in the plant is harvested under cutting or good grazing management. There is little information on the partitioning of other nutrients, in plants growing in the field.

The proportion of the above-ground plant tissue that is harvested as opposed to that which senesces depends on the frequency of defoliation and the leaf area index maintained for photosynthesis (Fig. 3; Parsons et al., 1983). Under lenient continuous grazing about one third as much plant carbon was eaten as senesced, whereas under hard grazing the two fractions were about equal. A study reported by Tallowin and Brookman (1996), demonstrated however that because of considerable translocation of N within leaf tissue, the average C:N ratio for newly emerging leaves of 4 different grass species receiving 400 kg N ha$^{-1}$ was about 11:1, whereas that for the senescent leaves was close to 23:1. This has two consequences: the nutrient losses via continuous senescence are reduced, and the mineralization of N from the senecenced leaves is retarded.

**Animal intake and excreta**. Whether eating conserved herbage or grazing at pasture, ruminants are rather inefficient at incorporating the nutrients in herbage into the animal products, milk and meat. For example, the proportion of the N in diet that is retained by grazing beef animals is found to vary between about 10 and 20% according to the concentration of N in the diet (Fig. 4). At low concentrations of dietary N, the N in excretal returns is equally divided between faeces and urine, but with increasing dietary N concentration to 4%, the proportion in urine increases to about 80% of the total N returned (Barrow and Lambourne,1962) and thus the amount returned in faeces remains roughly constant (Lantinga et al., 1987). Reporting earlier studies, Haynes and Williams (1993a) showed that with lactating dairy cows 25, 35 and 12% of the ingested N, P, and K respectively, is incorporated into animal product. While most of the returned P is in faeces, most of the returned K is in urine, with concentrations of both increasing with increasing concentration in the diet. The excretal returns during grazing are spatially and temporally very heterogeneous, making dramatic localized impacts on soil microbial and faunal populations and on the growing plant. An extreme effect is the disproportionate redistribution of nutrients contained in herbage grazed on hillslopes to the flatter campsite areas (Rowarth and Gillingham, 1990). It is argued generally that nutrients returned in excreta are more susceptible to loss from the system because of their relatively high concentrations.

The excreta collected from housed animals (slurry and manure) are also an important source of plant nutrients. The large variability in nutrient composition due to type of animal, time and manner of storage, and water content, coupled with variation in method of application, make efficient use of these nutrients a difficult task. Because of the considerable loss of N that occurs by volatilization during storage and on application to land, slurry is relatively rich in K and N, whereas farmyard manure is relatively rich in P. The development of techniques to enable the balancing of the N, P and K inputs to pastures from both fertilizer and farm waste sources should provide considerable improvement in the efficiency of nutrient use on livestock farms.

**Mineralization.** The microbially mediated process of mineralization of organic compounds releases major supplies of nutrients into soil solution. Most of our knowledge concerns the mineralization of N from the various organic fractions to produce ammonium. This, in turn, is used in the production of free nitrate through the process of nitrification. Both ammonium and nitrate can be re-assimilated by the microbial biomass, in the process of immobilization. The two processes, mineralization and immobilization thus operate concurrently, and it is net mineralization (or net immobilization) that determines the supply of mineral N. Many different chemical, biological and physical factors control net mineralization and it is difficult to measure in the field.

The chemical constitution of the organic substrate has a major effect on the rate of microbial degradation: fresh residues containing much α-amino N are rapidly decomposed, whereas old, humified material, with N mainly in heterocyclic combination is relatively recalcitrant. Although a continuum exists in soil organic matter it has been convenient for modelling purposes to identify only a few discrete classes with different degrees of recalcitrance (Parton et al., 1987). The C:N ratio of the substrate is considered a useful indicator of net mineralization, wide ratios leading to net immobilization. There is some disagreement as to the C:N value that marks the mineralization/immobilization balance in any given system. A C:N ratio of >25 has been stated as the litter quality at which decomposers change from effecting net mineralization to net immobilization (Aber and Melillo, 1991), whereas a C:N ratio of 15 was thought to be the critical value for manures (Castellanos and Pratt, 1981). The texture of the soil has a large influence on the turnover of soil organic matter. As organic matter ages and becomes increasingly recalcitrant, it becomes increasingly bound up with the finer (clay size) soil fractions, affording further protection from decomposition (Hassink, 1994), and offering one explanation for the greater accumulations of organic matter in clayey compared with sandy soils. Another possibility for the latter is the greater degree of aerobicity in sandy soils to enable more rapid oxidative respiration of carbon substrates.

Mineralization is strongly influenced by both temperature and soil water content, but these factors tend to counteract each other in the field as the season progresses, such that in a UK permanent pasture, 21-38% of annual net mineralization occurred during the period November-February. (Gill et al., 1995). Periods of freeze/thaw and wet/dry tend to enhance mineralization through disruption of aggregates and release of substrate (Jarvis et al., 1996a). Desiccation and heating either under natural or laboratory conditions has been found to cause large transient increases in net mineralization on subsequent rewetting (Birch, 1958; Farooqi et al., 1983). This indicates that freshly regenerated populations of soil microorganisms are much more effective than old populations 'in equilibrium' with soil conditions.

Soil macro-fauna such as protozoa and earthworms have generally been found to have a stimulatory effect on net mineralization. (Anderson, 1988). Earthworm-worked soils have enhanced respiratory activity, but smaller microbial populations (Ruz- Jerez et al., 1988), with the effects persisting for months after the removal of earthworms (Scholefield, 1990; unpublished results).

There are direct and indirect methods for estimation of net mineralization of N in intact field soils. Direct methods developed recently include the incubation of intact cores under field conditions with measurement of the changes in nitrate and ammonium over a specified period (e.g. Macduff and White, 1985). Refinements to this general method include incubation in sealed containers with acetylene to inhibit nitrification (Hatch et al., 1991; Gill et al., 1995), the fitting of sheaths to each core to restrict aerobicity and soil atmosphere replacement with 20% oxygen in helium in order to measure denitrification concurrently (Blantern, 1991). Although these methods represent a considerable improvement over the earlier laboratory methods involving sieved, dried soil, it is felt generally that accurate estimation of net mineralization in the field will require yet more technique development (Jarvis et al., 1996a). Nevertheless, where long runs of core incubations have been conducted some important effects have been revealed: annual net mineralization of N in temperate pastures can range between 65-415 kg ha$^{-1}$; it increases with increasing fertilizer input, although rates under grass/white clover swards were also high; the fertilization effect was measurable in years after fertilizer had been withdrawn; and, there was greater mineralization in drained than in undrained clay soil on adjacent plots receiving the same agronomic treatments (Jarvis et al., 1996) These results indicate that mineralization results in a major input of N to intensively managed pastures, that it is strongly influenced by the C:N ratio of the current organic residues, and that soil aerobicity as distinct from soil texture is an important control.

While the mineralization of organic S compounds might be subject to similar controls and influences as those determining mineral N supply, the supply of P through mineralization will be complicated by the many fixation and transformation mechanisms that limit P mobility in soils. Therefore the C:P ratio will be less useful than the C:N ratio for predicting the balance between mineralization and immobilisation of P and N, respectively. Another difference with P is that in organic combination it is mostly already in a fully or partially oxidised state.

**Loss processes**. As stated earlier it is the processes of nutrient fixation and loss that compete with the growing plant for the nutrients supplied to the soil from fertilizer, net mineralization, atmospheric deposition, N-fixation, the urinary returns of grazing animals and slurries and manures from housed animals. The major loss processes are leaching and volatilization, although important losses of some nutrients (especially P) occur through erosion of particulates. Some loss of each macro-nutrient occurs through leaching. Leaching of N has potentially both serious agronomic and environmental significance; leaching of P has only environmental significance, while that of K and S is recognised as only having economic significance at present. Only N suffers serious loss through volatilization, but with both economic and environmental significance. Both P and K and to some extent ammonium-N can become unavailable to varying degrees through physical and chemical fixation processes, involving soil mineral and organic colloids.

The volatilization of ammonia takes place mainly from urine patches, dung pats, urea fertilizer, slurry and manure applied to the land surface, animals themselves, cattle pens, waste storage vessels and in small amounts from living and senescent plant tissues (Whitehead and Lockyer, 1989; Sutton et al., 1993; Whitehead, 1995). Losses of ammonia-N from urine patches in the field measured using wind tunnels (Lockyer and Whitehead, 1990; Sherlock and Goh, 1984) suggested that about 15% of urinary N is volatilized during the few days after deposition. More recent measurements using micro-meteorological techniques indicate losses to be smaller, at 11% of urinary N. Losses increase with increasing temperature, soil pH, N content of the urine (N inputs) and decreasing soil water content, although it has not always been possible to explain the losses measured under grazing by the environmental variables (Hatch et al., 1990). Annual losses from grazed swards in the UK were reported to range from 1 kg N ha$^{-1}$ for sheep grazing grass/white clover to 41 kg N ha$^{-1}$, for dairy cows grazing grass receiving 550 kg N ha$^{-1}$ (Jarvis et al., 1995). Losses at grazing from dairy 'farmlets' in New Zealand were 15, 45 and 63 kg N ha$^{-1}$ for fertilizer inputs of 0, 225 and 360 kg N ha$^{-1}$, respectively (Ledgard et al., 1996). Losses from slurry applied to the land surface can be 31-84% of the ammonium N content of the slurry (29.3-94.5 kg N ha$^{-1}$), if no methods to reduce such loss are employed (Pain et al., 1990). Acidification prior to application reduced the loss to 14-57% of the ammonium N. Much of the ammoniacal N volatilized from urine and slurry can be re-deposited on terrestrial vegetation close to the site of volatilization. Whitehead (1995) reported annual deposition of ammoniacal N to moorlands in the UK and forested areas in The Netherlands of 16.5 and 115 kg ha$^{-1}$, respectively.

Denitrification in soil is a major cause of N loss from grassland systems and the major source of atmospheric nitrous oxide. It occurs under warm anoxic conditions that allow carbon to be oxidised while nitrate acts as the terminal electron acceptor for the denitrifying organisms. The two main products are nitrous oxide and nitrogen gas. The potential for N loss through denitrification is very high: loss equivalent to 30 kg N $ha^{-1}$ $d^{-1}$ has been measured under optimal conditions in the laboratory (Bijay-Singh et al., 1989). In the field however, peak rates >2 kg N $ha^{-1}$ $d^{-1}$ are uncommon, as measured with existing techniques. Although the general effects of the major controls on denitrification are well known from laboratory studies based on homogenized soils (e. g. Nommik, 1956), attempts to account for field variability have met with only limited success (Jarvis et al., 1991; Colbourn, 1992). Obtaining accurate field measurements is beset with difficulties, which include the need to accommodate large spatial and temporal variability, and the inherent shortcomings of available techniques.

The acetylene inhibition technique has been most used to obtain long term estimates of denitrification in the field. Annual N losses from pasture measured using this technique were determined by N input and site conditions: with sheep grazed pastures on a sandy loam soil in the UK, N losses of 6.1 and 51.2 kg N $ha^{-1}$ from grass/white clover or grass receiving 420 kg N $ha^{-1}$ fertilizer input, respectively, were reported (Parsons et al., 1991); N losses from a beef grazed grass sward receiving 400 kg N $ha^{-1}$ fertilizer, on a poorly drained clay loam soil in a high rainfall area averaged 113 kg $ha^{-1}$ over 4 years (Scholefield et al., 1988). In contrast, N loss through denitrification from a dairy pasture in New Zealand receiving a 400 and 74 kg N $ha^{-1}$ input from fertilizer and N-fixation respectively, was only 15 kg $ha^{-1}$ in the first year (Ledgard et al., 1996).

Urine patches are 'hot spots' for denitrification activity. Ruz-Jerez (1995), working with grass/clover pastures in New Zealand, noted that denitrification losses were generally low but that urine patches, which constituted only 10-15% of the area contributed more than the rest of the pasture to N loss. On application of artificial urine to a intensively managed beef grazed sward on a sandy soil in the Netherlands, rates of N loss of 9 and 6 kg $ha^{-1}$ $d^{-1}$ for denitrification and nitrous oxide emission, respectively, were obtained (de Klein and Logtestijn, 1994). Nitrogen losses through denitrification from surface applied slurry can also be large: 32.9 kg $ha^{-1}$ were lost from an autumn application of 80 $m^3 ha^{-1}$ to a sandy soil (Pain et al., 1990). Attempts to reduce ammonia loss from applied slurry by the use of shallow injection techniques has had the effect of increasing denitrification (Thompson et al.,1987), a result which was supported by a recent laboratory study to assess the impact of injection on the nitrous oxide:nitrogen ratio, (Scholefield et al., 1997, Poster in ICG).

The leaching of N and P has received much attention over the last 10 years in relation to their perceived deleterious impacts on the aqueous environment. Grassland agriculture had been viewed generally as conservative rather than leaky, until it was demonstrated that nitrate leaching losses under intensively grazed swards were comparable to losses obtained with arable cropping (Ryden et al., 1984; Garwood and Ryden, 1986; Haigh and White, 1986). More recent experiments using grazed grassland lysimeters installed in a clay loam soil have shown that for the same N input (400 kg $ha^{-1}$) annual N leaching can vary dramatically according to age of sward, soil conditions, drainage status (percolation or runoff) and patterns of weather. Thus on an old sward with under-drainage, or with no drainage (runoff and surface lateral flow) and recent reseeding, average annual nitrate leaching (4 years) was 186 or 24 kg N $ha^{-1}$, respectively (Scholefield et al., 1988) However, with any given treatment, there was twice as much leaching after a hot, dry summer as after a cool, wet one (Scholefield et al., 1993). These large loads of leached nitrate were associated with peak N concentrations far in excess of the public health limits of 10 and 11.3 mg $l^{-1}$ nitrate-N for drinking water in the USA and Europe, respectively. For example, peak concentrations of nitrate-N in water draining from the 1 ha lysimeters ranged from 55 mg $l^{-1}$ to 12 mg $l^{-1}$, for the under-drained old sward receiving 400 kg N $ha^{-1}$ and the undrained sward receiving 200 kg N $ha^{-1}$, respectively. Similar loads and peak concentrations of nitrate-N were leached from grazed, fertilized pastures in Germany (Spatz, 1992).

The leaching of N has been shown to be generally smaller from below grass/clover pastures than from intensively managed grass with high inputs of fertilizer N (Ryden et al., 1984; Ruz-Jerez et al., 1995), but where leaching was compared at approximately equal levels of production (Cuttle, 1992; Scholefield and Tyson, 1992), there were no consistent differences. Indeed, leaching under legume-rich swards can be as high as that with large inputs of fertilizer (Baber and Wilson, 1972; Ball, 1982; Macduff et al., 1990). Furthermore, recent measurements of the concentration of organic N compounds in the drainage water from clay soils under pasture show that there is substantial leaching of organic N from pastures that may subsequently add to the mineral N burden of rivers and estuaries; and, a greater proportion of N leaching from grass/clover systems as compared to fertilized grass is in organic combination (Fig. 5).

While earlier studies made around the world have indicated that losses of P from pasture were negligible from an economic perspective (Kilmer et al., 1974; Costin, 1980; Jordan and Smith, 1985), P leaching from intensively managed grassland is now recognised as a potential cause of the eutrophication of surface waters. The amounts lost from grazed grassland during the winter drainage period are small however, with annual losses of molybdate reactive P of 0.4 and 0.2 kg $ha^{-1}$ in surface lateral flow and field drains respectively, being measured recently in the UK (Hawkins and Scholefield, 1996). Peak P concentrations as high as 700 µg $l^{1}$ were measured in surface drainage from grass/white clover swards receiving 50 kg P $ha^{-1}$ as triple superphosphate. Haygarth and Jarvis (1997) measured total P and molybdate reactive P during storm events and found that although total P was correlated to discharge, molybdate reactive P was not. Storm events resulting in total P concentrations as high as 1700 µg $l^{1}$ were reported, with one intense storm responsible for loss of 0.5 kg P $ha^{-1}$. Haygarth and Jarvis (1996) also reported that the proportion of P lost in the organic form was greater in water percolating to field drains than in surface lateral flow, which is contrary to the effect of drainage hydrology on the distribution of N forms between organic and inorganic reported earlier.

Losses of P in early spring runoff can be relatively large: Uhlen (1989) reported P losses of 1.9 kg $ha^{-1}$ in snowmelt running down a Norwegian hillslope, while >0.5 kg P $ha^{-1}$ was lost after application of fertilizer for first cut silage in the UK (Scholefield and Stone, 1995). Losses of P in runoff from farm wastes applied under wet conditions can be extremely high: Edwards and Daniel (1993) reported a peak P concentration of 76 mg $l^{-1}$ after an application of poultry manure at 20 Mg $ha^{-1}$.

Losses of K by leaching are small relative to amounts applied and the levels extractable from the soil. Measurements made during winter in water draining from monolith lysimeters of a sandy loam under fertilized grass cumulated in losses of 3.6 kg $ha^{-1}$ (Garwood and Tyson, 1973). This value is similar to those measured in runoff from grazed grassland (Uhlen, 1989; Scholefield and Stone, 1995), but much smaller than that measured in runoff from a pasture applied with various farm wastes (Misselbrook et al., 1995). In this study, 10.1, 16.0 and 29.1 kg K $ha^{-1}$ were lost during winter after application of dairy washings, cattle slurry and farmyard manure, respectively, in October.

Losses of sulphate S can be relatively large and tend to be correlated with N leaching. Garwood and Tyson (1973) reported losses of 19.4 and 29.1 kg S $ha^{-1}$ from grass lysimeters applied with 250 and 500 kg N $ha^{-1}$, respectively. The range of S loss through leaching in New Zealand was quoted by Nguyen and Goh (1993) as 11-43 kg $ha^{-1}$.

Because of the increased nitrate leaching caused by the presence of the grazing animal, much research activity has focused on the urine patch as a direct source of loss. Although it might be supposed that urine voided under dry conditions may be transported to depth by macropore flow, the great spatial variability that is often observed in nitrate concentration from water extracted by ceramic cup samplers may not be a result of urinary deposition. Uniform application of nitrate solution to the soil surface may give rise to the same degree of variability when sampled, due to preferential flow (Holden et al., 1995). Moreover, the effect of urine application to pastures is rather short-lived (Sherwood and Fanning, 1989; Scholefield and Titchen, 1995). When artificial urine was applied to grass plots at various times of the year, only patches that were applied in September or later contained significantly greater quantities of mineral N in November than the control plots (Cuttle and Bourne, 1993).

**EFFICIENCY OF NUTRIENT USE IN GRAZED PASTURES**

The processes of plant uptake from the soil, and herbage intake and assimilation by the animal have the greatest influences on the efficiency of nutrient cycling within the grassland system. Ultimately it is the efficiency of plant uptake that determines the efficiency of the whole system because the processes of nutrient loss, involving volatilization and leaching, occur mainly from within the soil. The efficiency of nutrient use can be defined as the ratio of the amount of nutrient in animal product to the amount supplied as inputs. However, efficiency expressed in this way conveys only limited information about the sustainability and environmental impact of a system, as the types of inputs are not specified and losses are ignored. Several indices of efficiency of N use were calculated by Scholefield and Smith (1996) for a grass ley on a loam soil in southern England. Apart from 'Product N'/'Total N Input', these included 'Product N' + 'Total N losses'/'Total N Input', 'Product N'/'Fertilizer N', and 'Product N'/'Total N losses'. Although these indices are not independent of each other, they can be used to view efficiency of nutrient use from different standpoints. Thus the first two indices are measures of relative sustainability: values greater than one indicate that the system is undergoing net nutrient loss and cannot be sustained over the longer term. 'Product N'/'Fertilizer N' is an economic index, whereas 'Product N'/'Total N losses' is a measure of relative environmental impact. Table 1 shows these indices of efficiency calculated using a derivative of the NCYCLE model (Scholefield et al., 1991) for average N flows through a dairy pasture system at two levels of management intensity, either with N-fixation from white clover as the main N input, or with 350 kg N $ha^{-1}$ as mineral fertilizer.

The table shows that, while both systems are rather inefficient, they are at least sustainable in the sense that N is not being depleted. The fact that grassland systems accumulate organic N is of course the basis of ley-arable farming. Index 3 shows that, with the high N input system, the equivalent of only one quarter of the N supplied as fertilizer is incorporated into the milk product. Index 3 is clearly inappropriate for assessing legume-based systems as the amount of "bag" fertilizer is zero, which results in the infinitely efficient use of it. Index 4 shows that the legume-based system is relatively much more environmentally benign than the high input system. However, it must be borne in mind that this may be only a consequence of the smaller N inputs to the legume-based system, rather than to any inherent advantage to efficiency of N use with N-fixation. Model calculations show that index 4 for a fertilizer-based system was the same as that for a legume-based system at equivalent levels of N exported in product.

Index 4 is perhaps the most informative index of efficiency of N use if it is linked to the total N input to the system. That is, the best statement of the efficiency of N use is the proportion of the total soil mineral N flux that is partitioned to the plant, as opposed to the processes of loss. This statement is embodied in the model NCYCLE (Scholefield et al., 1991), which simulates the annual N pools, transfers and losses under grazing in the UK. The relationship in Fig. 6 was derived from inputs and measurements of annual fluxes and pools within 10 grazing systems, each over a 4 year period. It can be seen that as the flux of soil mineral N increases, the proportion of it that is partitioned to the plant decreases; that is, the efficiency of the system decreases. Novel systems that are relatively more efficient could be identified since they would lie above the line (e. g. at position A), whereas systems with poorer efficiency would lie below it (e. g. at position B). This relationship would also be useful to reveal any gross differences in the efficiency of the different grazing managements. For example, rotational versus continuous grazing, and sheep versus dairy cattle. Moreover, if the urine patch *per se* had a major effect on the efficiency of N use, any point derived for a cutting-only system would lie above the line in Fig. 6.

Watson et al., (1992) demonstrated the efficiency of N use in grazed grassland by a plot of total N loss against N consumed by the animals. The plot was linear such that for every 1.0 kg of N consumed in herbage in the range 100-450 kg $ha^{-1}$, 0.5 kg N was lost to the environment. Their main conclusion was that the N cycle is inherently "leaky" and that production is always accompanied by loss. This is at odds with the concept of nutrient requirement or 'demand' that has tended to direct fertilization strategies in both grassland and arable agriculture. This concept is that the plant has a nutrient demand to satisfy its growth potential in a given environment, and only if supply exceeds this value will appreciable losses take place. Although there is some evidence for the existence of a 'breakpoint' in the N input/N loss relationship (Barraclough et al., 1992), its existence is more likely to be due to a discontinuity in N supply mediated through mineralization (as explained by Wedin, 1996) than one in N input defined simply by demand. Titchen and Scholefield (1992) demonstrated that, by applying fertilizer tactically according to the mineral N supply from other sources, it was possible to partition more of the applied N to the product and so reduce the residue of potentially leachable N left in the soil in the autumn.

It is the extent of net mineralization relative to the uptake potential of the plant that is critical to the efficiency of N use in pasture systems. Wedin (1996) argued that it is not valid to consider "availability" of N as a soil property in isolation from the chemical characteristics of the plant debris that contributed to a major part of that available N. In an experiment in which 5 perennial grasses (three $C_3$ and two $C_4$ species) were grown in monoculture for 3 years, it was found that the C:N ratio of the plant debris exerted strong control over the net mineralization rate in the soil (Wedin and Tilman, 1990).

Other factors that can change net mineralization and thus the efficiency of N use include soil physical conditions, the age of the sward since cultivation and the seasonal pattern of weather. There may be scope for introducing managements that enable the tactical manipulation of at least some of these (e.g. Scholefield et al., 1988), to optimise efficiency of N use.

There are thus two types of approach to improve the efficiency of N use in grassland systems: one is to optimise N use within our existing systems, that is to displace the system along the relationship in Fig. 6 to the left; the second is to displace the system vertically away from this relationship to make it inherently more efficient. The second type will probably involve increasing the uptake efficiency of the plant and/or the assimilation efficiency of the animal. Most of the strategies being considered currently may be of the first type.

## STRATEGIES FOR IMPROVING EFFICIENCY

**Tactical fertilizer applications.** In this approach the mineral N in the soil is not allowed to accumulate during the growing season, but is limited to a predetermined level or 'profile'. This mineral N profile is maintained by adding fertilizer tactically, in amounts determined by the use of a rapid soil test (Scholefield and Titchen, 1995). Profiles were maintained at an arbitrary level (<45 kg N $ha^{-1}$) for early field experiments in which 20-30% improvement in the efficiency of N use was obtained (Titchen and Scholefield, 1992). In these experiments the profile of soil mineral N could not be linked to a given level of production, or any environmental goal.

A similar approach has olso been developed and is now being evaluated in The Netherlands (Oenema et al.,1992; Vellinga et al., 1996), which is described as a System for Adjusted Nitrogen Supply (SANS). This uses model calculations to obtain N yield, N mineralization, N recovery in herbage and uses laboratory tests to obtain values for available soil N. Superior efficiency of N use was obtained using SANS compared with using current N fertilizer recommendations on both sandy and clay soils.

The UK approach is currently being further developed with the use of a model (NCYCLE; Scholefield et al., 1991) to link the mineral N profile to target yields of herbage and levels of N loss (Scholefield et al., 1996). An example of a model-derived mineral N profile is given in Fig. 7.

Because of the highly heterogeneous returns of N and K in urine and P in dung to grazed swards, particularly to animal 'camping' areas, the precision application of fertilizers to the areas that are relatively nutrient poor is being considered in The Netherlands and elsewhere (van der Putten, pers. communication).

**Farm nutrient balances.** Improving the efficiency of nutrient use is the key to decreasing losses of N and P to environmentally acceptable levels. This holds especially for intensively managed temperate pastoral systems. Successful implementation of new techniques to improve efficiency (e.g. De Haan and Ogink, 1994) requires a thorough understanding of all of the nutrient transformation and loss processes, as previously discussed.

The need for reducing nutrient losses and improving the efficiency of nutrient use is very urgent for intensively managed dairy farming systems in many parts of the world, particularly The Netherlands (Aarts et al., 1992; van der Meer and van der Putten, 1995; Bussink and Oenema, 1997). In order to comply with government legislation dealing with slurry applications, a nutrient balance accounting system is to be implemented by 1998. Under the legislation, farmers are required to reduce nutrient losses by a step-wise reduction in the annual surpluses of N and P between 1998 and 2008.

All inputs of nutrients via fertilizers, purchased concentrates, roughage, manure and cattle, and all outputs via milk meat, wool manure, and cattle must be recorded by all farmers. When the surplus of N or P is larger than that permissible (Table 2) the farmer must pay a levy for each kg in excess.

Clearly this is a complex system and difficult to implement nation-wide. It emphasizes the resolve needed to solve the huge emissions problem that have originated from the large-scale intensification of dairy and pig farming during the last 30 years.

The nutrient budget accounting system can be considered as a 'top down' approach that forces farmers to implement the most effective techniques to suit their individual circumstances for improving efficiency of nutrient use. To select and implement these techniques requires integrated modelling and experimentation at the farm scale and an effective system of technology transfer and advice.

Two recent modelling studies (van der Meer and van der Putten, 1995; Jarvis et al., 1996b) have specified the nutrient budgets of complete dairy farms for several scenarios incorporating nutrient efficient strategies. The two studies were conducted from different standpoints: in the Dutch study, scenarios were determined according to target outputs for the stepwise reduction in N and P surpluses required during the next 10 years to meet environmental legislation (Table 2); in the UK study the objective was more to explore the potential of different strategies in feasible combinations, for reducing N emissions while maintaining production.

The strategies examined in the Dutch study were reduction of fertilizer, injection of slurry, restricted grazing and modification to livestock buildings to reduce volatilization of ammonia. It was calculated that surpluses of N and P could be reduced from 427 and 68 kg $ha^{-1}$ to 123 and -0.3 kg $ha^{-1}$ respectively, by implementation of the strategies. In the UK study, the strategies examined were reduction in fertilizer input through tactical application, injection of slurry, use of legume-derived N rather than mineral fertilizer and the use of forage maize (a $C_4$ plant with lower N requirement than ryegrass). It was calculated that total N losses per ha and per livestock unit could be reduced from 160 and 74 to 55 and 32, respectively, by the best combination of strategies. However, the full implications of the strategies for the farmer have yet to be evaluated.

**Improved management of animal wastes.** When properly collected and stored, animal wastes contain nearly all the nutritive elements in the forage and concentrates ingested by the animal that have not been utilized for the production of milk, meat and wool. This fact has been known for a long time, but the need and potential for effective utilization of these nutrients in intensively managed grassland systems has been appreciated for only about 20 years. When wastes are properly managed, nearly all N, P and K contained in them can be utilized.

Proper management of farm wastes starts with effective collection and storage of wastes from livestock in non-leaky, covered structures. Also the rinse water and runoff from the barn must be collected and stored. Covering the structures minimizes the escape of the highly volatile ammonia and sulphide. Various kinds of cover have been evaluated and it has been demonstrated that ammonia loss can be reduced by an average of 70%, depending on temperature and wind speed. Acidification of slurry has also proved to be effective at reducing ammonia loss during storage, but serious side effects have hindered practical applications. While appropriate systems for slurry storage are currently available, little progress has been made to reduce volatilization from the animal houses. The urease activity on the floors of cubicle houses is extremely high so that urea hydrolysis is rapid. Rinsing the floors with acid, water and formalin has had some success in reducing N losses but would be rather costly to implement.

In The Netherlands a great effort has been made to process wastes to produce an attractive and easy-to-use fertilizer. A number of pilot industrial processing units have been in operation over the last 10 years, but nearly all have failed due to the high costs associated with transport and drying of the wastes. The processing operations include separation of solids from liquid phases, anaerobic fermentation to convert easily metabolizable organically bound N and to collect the generated methane, acidification to prevent ammonia loss and drying to reduce bulk and facilitate better handling.

Successful implementation of low emission techniques for the application of animal wastes to grassland have greatly contributed to increased efficiency of nutrient use in intensively managed grasslands in western Europe during the last 10 years. Homogenization is a prerequisite for good utilization because stored slurry becomes segregated, with P-rich fractions sinking to the bottom layers. However, if homogenization cannot be done, the upper layers which are relatively impoverished in P should be applied where soil tests reveal particularly high P levels. Acidification and injection have proved effective in reducing ammonia emission as reported earlier, but where one route of loss has been suppressed there is increased potential for another route (i.e. denitrification or leaching) to be enhanced (Jarvis et al., 1987; Pain et al., 1990).

**Use of low input, legume-based systems.** There is yet much interest in evaluating and developing systems of grassland production based on legumes grown alone or in combination with grasses. While several field experiments have demonstrated that legume-based systems are more efficient in N use (Parsons et al., 1991; Ruz-Jerez et al., 1995), it has been difficult to compare efficiencies within the two systems at the same level of production. There are several pieces of evidence that indicate that grass/legume combinations should be more efficient at utilizing N than grass alone, applied with mineral fertilizer. One is simply that because no fertilizer is applied, the level of soil mineral under grass/clover swards should, on average, be smaller. A second is that the soil under a grass/clover sward has been found to contain a greater proportion of N in the ammonium form (Jarvis and Barraclough, 1991; Scholefield and Titchen, 1995). A third is that the grass component in grass/legume swards is maintained in an N-deficient state so that mineral N levels should be kept low (Parsons et al., 1991).

Acceptance of the use of grass/legume combinations for improved efficiency of N use will be gained only by obtaining incontrovertible evidence from experiments. The use of models (e.g. Hutchings and Kristensen, 1995) will help to resolve this issue only if the models are based on the appropriate mechanisms and are parametized realistically.

## FUTURE RESEARCH TRENDS AND NEEDS

Recent research has produced much quantitative information about nutrient pools and fluxes within temperate, agricultural grasslands. Those fluxes critical to the efficiency of the whole system have been identified and mostly quantified. This information has enabled concepts of the efficiency of nutrient use (N, in particular) to be formalized to provide a framework for evaluation of novel systems. Field experimentation and simulation modelling have each been successful in the development and evaluation of novel strategies for implementation at the field and farm scale. These strategies can be of two types and it is important to distinguish between them: the first type decreases nutrient inputs only, whereas the second improves the inherent efficiency of one or more of the nutrient transfers within the system. Most modelling and whole-farm budgeting studies reveal that, with the most intensive managements currently operating, it is improbable that economic levels of production can be reconciled with acceptable impacts on the environment. New, more efficient strategies of the second type will be needed, which will involve considerable activity in the basic animal, plant and soil sciences. Moreover, there is a need to provide effective means for demonstration and implementation of the new strategies by the provision of large scale, farmer-friendly models and decision support systems as well as the use of integrated, field-scale, farmlet experiments. Highly integrated studies driven by top down objectives, (i.e. from the Industry via the animal) are essential in order to optimize nutrient assimilation by the animal, provide forages of a quality suitable for their optimal assimilation and to enable the management of inputs to the soil and sward that provide economic levels of forage at acceptable environmental cost.

Specifically, we need to know more about the process of mineralization: methods for better quantitative measurement and prediction of net N mineralization must be developed that enable the effects of the physical location within the soil and of the quality of the carbon substrates to be accurately evaluated. Also, we need the basic knowledge to allow the manipulation of the denitrification process through farm managements, for minimal loss and especially for minimal nitrous oxide flux. It will be important for progress with both mineralization and denitrification that we acquire an understanding of how antecedent conditions in the soil affect the concentrations and activities of the different enzymes operating. There is evidence now accumulating that substantial amounts of N and P are transported within and from soils in organic combination; the implications of this for the efficiency of nutrient transfers and the overall biological activity within the soil as well as impacts on 'downstream' processes need elucidation. The coupling of nutrient transfers to carbon dynamics and energy dissipation is implicit rather than explicit in many models. Re-examination of indices of efficiency for different management strategies with energy flows taken into account may change the ranking of those strategies. The potential benefits of rotational managements including mixed farming at a range of intensities needs re-examination, for it offers much opportunity for manipulation of net mineralization through changes in soil physical conditions and the chemical constitution of plant residues.

There are now several models of nutrient cycling in grassland that are proving useful for extrapolation to unknown scenarios and in making farm management decisions. These include the Hurley Pasture Model (Thornley and Verberne, 1989), CENTURY (Parton *et* al., 1987), NCYCLE (Scholefield et al., 1991) and FARM-MIN (van der Meer and van der Putten, 1995). While each of these existing models has different strengths and weaknesses, as mentioned earlier, there is now the need for further modelling developments that may or may not require the construction of new models. One is the need to provide usable models that operate at the catchment and landscape scales, and another is to integrate models that operate within the scientific and technical spheres with farm economic models, to provide user-friendly decision support systems for farmers and advisors. Because of the great influence of changing patterns of weather on nutrient fluxes, the latter will probably need the provision of risk assessment-based or stochastic outputs, rather than the deterministic outputs common to most currently used models. The linkage of the cycles of all nutrients through carbon and energy flux is a distant yet worthwhile modelling objective.

## REFERENCES


**Aarts, H. F. M., E. E. Biewinga and H. van Keulen.** 1992. Dairy farming systems based on efficient nutrient management. Neth. J. Agric. Sci. **40**: 285-290.

**Aber, J. D. and J. M. Melillo**. 1991. Terrestrial Ecosystems. Saunders College Publishing, Philadelphia, PA.

**Anderson, J. M**. 1988. The role of soil fauna in agricultural systems. Pages 89-112 *in* J. R. Wilson, ed. Advances in Nitrogen Cycling in Agricultural Ecosystems. CAB International, Wallingford, UK.

**Baber, H. L. and A. T. Wilson**. 1972. Nitrate pollution of groundwater in the Waikato Region. J. New Zealand Inst. Chem. **36**: 179-183.

**Ball, P. R.** 1982. Nitrogen balances in intensively managed pasture systems. Pages 47-66 *in* Nitrogen Balances in New Zealand Ecosystems. DSIR, Palmeston North.

**Ball, P. R. and T. R. O. Field**. 1987. Nitrogen cycling in intensively managed grasslands: a New Zealand viewpoint. Pages 91-112 *in* P. E. Bacon, J. Evans, R. R. Storrier and A. C. Taylor, eds. Nitrogen Cycling in Temperate Agricultural Systems. Australian Society of Soil Science, Riverina.

**Barraclough, D., S. C. Jarvis, G. P. Davies and J. Williams**. 1992. The relation between fertilizer nitrogen applications and nitrate leaching from grazed grassland. Soil Use and Management **8**: 51-56.

**Barrow, N. J. and L. J. Lambourne.** 1962. Partition of excreted nitrogen, sulphur and phophorus between the faeces and urine of sheep being fed pasture. Austr. J. Agric. Res. **13**: 461-471.

**Beaton, J. D. and J. Berger**. 1974. Present and potential use of fertilizer for forage production in temperate zones. Pages 17-37 *in* D. A. Mays, ed. Forage Fertilization. ASA, SSSA, Csssa. Madison, WI.

**Bijay-Singh, J. C. Ryden and D. C. Whitehead**. 1989. Denitrification potential and actual rates of denitrification in soils under long-term grassland and arable cropping. Soil Biol. Biochem. **21**: 897-901.

**Birch, H. F.** 1958. The effect of soil drying on humus decomposition and nitrogen availability. Plant and Soil **10**: 9-31.

**Blantern, P. J.** 1991. Factors affecting nitrogen transformations in grazed grassland soils with specific reference to the effects of artificial land drainage and N-fertilization. Unpublished PhD Thesis, University of Exeter, UK.

**Bolland, M. D. A. and R. J. Gilkes**. 1995. Long-term value of North Carolina and Queensland rock phosphates compared with triple superphosphate. Fert. Res. **41**: 151-158.

**Boller, B. C. and Nosberger**, J. 1989. Symbiotically fixed nitrogen from field-grown white and red clover mixed with ryegrasses at low levels of $^{15}$N fertilization. Plant and Soil **104**: 219-226.

**Bussink, D. W. and O. Oenema**. 1997. Ammonia volatilization from dairy farming systems in temperate areas; a review. Nutrient Cycling in Agroecosysems (in press).

**Castellanos, J. Z. and P. F. Pratt**. 1981. Mineralization of manure nitrogen-correlation with laboratory indices. Soil Sci. Soc Amer. J. **57**: 63-66.

**Clarkson, D. T., M. J. Hopper and L. H. P. Jones**. 1986. The effect of root temperature on the uptake of nitrogen and the relative size of the root system in *Lolium perenne*. 1. Solutions containing both $NH_4^+$ and $NO_3^-$. Plant, Cell and Environ. **9**: 535-545.

**Colbourn, P**. 1992. Denitrification and $N_2O$ production in pasture soils. The influence of nitrogen supply and moisture. Agric. Ecosys. and Environ. **39**: 267-278.

**Costin, A. B**. 1980. Runoff and soil nutrient losses from an improved pasture at Ginninderra, Southern Tablelends, New South Wales. Aust. J. Agric. Res. **31**: 533-546.

**Cowling, D. W., L. H. P. Jones and D. R. Lockye**r. 1973. Increased yield through correction of sulphur deficiency in ryegrass exposed to sulphur dioxide. Nature **243**: 479-480.

**Cuttle, S. P. and P. C. Bourne**. 1993. Uptake and leaching of nitrogen from artificial urine applied to grassland on different dates during the growing season. Plant and Soil **150**: 77-86.

**Cuttle, S. P.** 1992. Prospects for reduced nitrate leaching by the use of clover in grassland systems. Pages 239-243 *in* J. R. Archer, K. W. T. Goulding, S. C. Jarvis, C. M. Knott, E. Lord, S. E. Ogilvy, J. Orson, K. A. Smith and B. Wilson, eds. Nitrate and Farming Systems. Aspects of Applied Biology 30. Ass. Appl. Biol. 30. H. R. I. Wellesbourne, UK.

**Deenen, P. J. A. G. and E. A. Lantinga.** 1993. Herbage and animal production responses to fertiliuzer nitrogen in perennial ryegrass swards. 1. Continuous grazing and cutting. Neth. J. Agric. Sci. **41**: 179-203.

**De Haan, M. H. A. and N. W. M. Ogink.** 1994. Towards a balance between livestock farming and environment; 10 years FOMA research. Onderzoek inzake de mest-en ammoniakproblematiek in de veehouderij 19 (Rundvee). DLO, Wageningen, 164 pp.

**Edwards, D. R. and T. C. Daniel**. 1993. Effects of poultry litter application rate and rainfall intensity on quality of of runoff from fescuegrass plots. J. Environ. Qual. **22**: 361-365.

**Farooqi, M. A. R. M. Hanif, and C. J. deMooy**. 1983. Nitrogen mineralization potential and urea hydrolysis under aerobic and anaerobic conditions. Commun. in Soil Sci. and Plant anal. **14**: 29-47.

**Garwood, E. A. and K. C. Tyson**. 1973. Losses of nitrogen and other plant nutrients in drainage from soil under grass. J. Agric. Sci. Camb. **80**: 303-312.

**Garwood, E. A. and J. C. Ryden.** 1986. Nitrate loss through leaching and surface run-off from grassland: effects of water supply, soil type and management. Pages 99-113 *in* H. G. van der Meer, and G. C. Ennik, eds. Nitrogen Fluxes in Intensive Grassland Systems. Martinus Nijhoff, Dordrecht, The Netherlands.

**Gill, K., S. C. Jarvis and D. J. Hatch**. 1995. Mineralization of nitrogen in long-term pasture soil: effects of management. Plant and Soil **173**: 153-162.

**Griffith, W. K. and L. S. Murphy**. Macronutrients in forage production. Pages 13-25 *in* R. E. Joost and C. A. Roberts, eds. Nutrient Cycling in Forage Systems. Potash and Phosphate Institute. Norcross, GA.

**Haigh, R. A., and R. E. White**. 1989. Nitrate leaching froma small uderdrained grassland catchment. Soil Use and Management **2**: 65-70.

**Hansson, A. C. and R. Pettersson**. 1989. Uptake and above- and below-ground allocation of soil mineral-N and fertilizer-$^{15}$N in a perennial grass ley (*Festucal pratensis*). J. Appl. Ecol. **26**: 259-271.

**Hassink, J**. 1994. Effects of soil texture and grassland management on soil organic C and N and rates of C and N mineralization. Soil Biol. Biochem. **26**: 1221-1231.

**Hatch, D. J., S. C. Jarvis and G. J. Dollard.** 1990. Measurements of ammonia emission from grazed grassland. Environ. Poll. **65**: 333-346.

**Hatch, D. J., S. C. Jarvis and S. E. Reynolds.** 1991. An assessment of the contribution of net mineralization to N cycling in grass swards using a field incubation method. Plant and Soil **138**: 23-32.

**Hawkins, J. M. B. and D. Scholefield.** 1996. Molybdate-reactive phosphorus losses in surface and drainage waters from permanent grassland. J. Environ. Qual. **25**: 727-732.

**Harrison, A. F**. 1985. Effects of environment and management on phosphorus cycling in terrestrial ecosystems. J. Environ. Management **20**: 163-179.

**Haygarth, P. N. and S. C. Jarvis.** 1996. Pathways and forms of phosphorus losses from grazed grassland hillslopes. Pages 283-294 *in* M. G. Anderson and S. M. Brooks, eds. Advances in Hillslope Processes. Wiley, London.

**Haygarth, P. M. and S. C. Jarvis.** 1997. Soil derived phosphorus in surface runoff from grazed grassland lysimeters. Wat. Res. **31**: 140-148.

**Haynes, R. J. and P. H. Williams**. 1993a. Nutient cycling and soil fertility in the grazed pasture ecosystem. Adv. in Agron. **49**: 119-199.

**Haynes, R. J. and P. H. Williams**. 1993b. An overview of pasture response, nutrient turnover and nutrient accumulation on the grazed, long-term super phosphate trial ar Winchmore, New Zealand. Proc. 17th Int. Grass. Congr. New Zealand. pp. 1430-1433.

**Holden, N. M., D. Scholefield, A. G. Williams and J. F. Dowd**. 1995. The simultaneous use of three common hydrological tracers for fine spatial and temporal resolution investigation of preferential flow in a large soil block. Pages 77-86 *in* Tracer Technologies for Hydrological Systems. IAHS Publication No. 229.

**Hutchings, N. J. and I. S. Kristensen**. 1995. Modelling mineral nitrogen accumulation in grazed pasture: will more nitrogen leach from fertilized grass and unfertilized grass/clover. Grass and Forage Sci. **50**: 300-313.

**Jarvis, S. C., M. Sherwood and J. A. M. Steenvoorden**. 1987. Nitrogen losses from animal manures: from grazed pastures and from applied slurry. Pages 195-212 *in* H. G. van der Meer, R. J. Unwin, T. A. van Dijk and G. C. Ennik, eds. Animal Manure on Grassland and Forage Crops. Fertilizer or Waste? Developments in Plant and Soil Sciences, Vol. 30. Martinus Nijhoff, Dordrecht, The Netherlands.

**Jarvis, S. C. and D. Barraclough.** 1991. Variation in mineral nitrogen under grazed grassland swards. Plnat and Soil **138**: 177-188.

**Jarvis, S. C., D. Barraclough, J. Williams and A. J. Rook**. 1991. Patterns of denitrification loss from grazed grassland: effects of N fertilizer inputs at different sites. Plant and Soil **131**: 77-88.

**Jarvis, S. C., D. Scholefield and B. F. Pain**. 1995. Nitrogen cycling in grazing systems. Pages 381-419 *in* P. E. Bacon ed. Nitrogen Fertilization and the Environment. Marcel Dekker, New York.

**Jarvis, S. C., E. A. Stockdale, M. A. Shepherd and D. S. Powlson.** 1996a. Nitrogen mineralization in temperate agricultural soils: processes and measurement. Adv. in Agron. **57**: 187-235.

**Jarvis, S. C., R. J. Wilkins and B. F. Pain**. 1996b. Oppotunities for reducing the environmental impact of dairy farming managements: a systems approach. Grass and Forage Sci. **51**: 21-31.

**Jordan, C. and R. V. Smith**. 1985. Factors affecting leaching of nutrients from an intensively managed grassland in Count Antrim, Northern Ireland. J. Environ. Management **20**: 1-15.

**Kilmer, V. J., J. W. Gilliam, J. F. Lutz, R. T. Joyce and C. D. Eklund.** 1974. Nutrient losses from ferilized grassland watersheds in western North Carolina. J. Environ. Qual. **3**: 214-219.

**de Klein, C. A. M. and R. S. P. van Logtestijn**. 1994. Denitrification and $N_2O$ emissions from urine-affected grassland soil. Plant and Soil **163**: 235-242.

**Kriel, W and O. Oenema** 1990. Grassland ferilization and environment. Proc. 13th General Meeting of the European Grassl. Fed. pp.132-143.

**Lantinga, E. A., J. A. Keuning, J. Groenwold and P. A. G. Deenen**. 1987. Distribution of excreted nitrogen by grazing cattle and its effects on sward quality, herbage production and utilization. Pages 103-117 *in* H. van der Meer, R. J. Unwin, T. A. van Dijk and G. C. Ennik, eds. Animal Manures in Grassland and Fodder Crops: Fertilizer or Waste. Martinus Nijhoff, Dordrecht, The Netherlands.

**Ledgard, S. F., D. A. Clark, M. S. Prosen, G. J. Brier and E. K. K. Nemaia.** 1996. Nitrogen losses from grazed dairy pasture, as affected by nitrogen fertilizer application. Proc. New Zealand Grassland Assoc. **57**: 21-25.

**Ledgard, S. F., B. S. Thorrold, M. S. Sprosen and M. F. Hawke**. 1993. Effectiveness of sulphate and elemental sulphur fertilizers applied to pastures in autumn or spring. Proc. 17th Int. Grass. Cong. New Zealand. pp.1464-1465.

**Lockyer, D. R. and D. C. Whitehead**. 1990. Volatilization of ammonia from cattle urine. Soil Biol. Biochem. **22**: 1137- 1142.

**Macduff, J. H. and R. E. White**. 1985. Net mineralization and nitrification rates in a clay soil measured and predicted in permanent grassland from soil temperature and moisture content. Plant and Soil **86**: 151-172.

**Macduff, J. H., J. H. A. M. Steenvoorden, D. Scholefield and S. P. Cuttle.** 1990. Nitrate leaching losses from grazed grassland. Proc. 13th Meeting of the European Grassland Fed. Lahti, Finland. pp. 18-24.

**MAFF,** 1994. Recommendations for grassland. Pages 91-109 *in* Fertilizer Recommendations for Agricultural and Horticultural Crops (Reference Book 209), HMSO, London.

**t'Mannetje, L. and S. C. Jarvis.** 1990. Nitrogen flows and losses in grasslands. Proc. 13th Meeting of the European Grassland Fed. Banska Bystica, Czechoslovakia. pp.114-131.

**Martz, F. A., M. F. Weiss and J. R. Gerrish**. 1996. Macromineral availability and utilization by grazing livestock. Pages 177-191 *in* R. E. Joost and C. A. Roberts. Nutrient Cycling in Forage Systems. Potash and Phosphate Institute. Norcross, GA.

**Misselbrook, T. H., B. F. Pain, A. C. Stone and D. Scholefield**. 1995. Nutrient run-off following application of livestock wastes to grassland. Environ. Poll. **88**: 51-56.

**Morrison, J., M. V. Jackson and P. E. Sparrow.** 1980. The response of perennial ryegrass to fertilizer nitrogen in relation to climate and soil. G. R. I. Technial Report No. 27. G. R. I., Hurly, UK.

**Nguyen, M. L. and K. M. Goh**. 1993. Factors affecting plant sulphur requirements in New Zealand pastoral and arable systems. Proc. 17th Int. Grass. Congr. New Zealand. pp. 1433-1436.

**Nissinen, G.** 1992. The suitability of green forage plants for grazing in Northern Finland. Proc. 14th Meeting of the European Grassland Fed. Lahti, Finland. pp. 329-332.

**Nommik,** H. 1956. Investigations on denitrification in soil. Acta Agric. Scand. **6**:195-228.

**Oenema, O., F. A. Woperis and G. H. Ruitenberg**. 1992. Developing new rscommendations for nireogen fertilization of intensively managed grasslands in The Netherlands. Pages 249-253 *in* J. R. Archer, K. W. T. Goulding, S. C. Jarvis, C. M. Knott, E. Lord, S. E. Ogilvy, J. Orson, K. A. Smith and B. Wilson. Aspects of Applied Biology 30.

**Pain, B. F., R. B. Thompson, Y. J. Rees and J. H. Skinner**. 1990. Reducing gaseous looses of nitrogen from cattle slurry applied to grassland by the use of additives. J. Sci. Food and Agriculture **50**: 141-153.

**Parsons, A. J., E. L. Leafe, B. Collett, P. D. Penning and J. Lewis.** 1983. The physiology of grass production under grazing. II. Photosynthesis, crop growth and animal intake of continuously-grazed swards. J. Appl. Ecol. **20**: 127-139.

**Parsons, A. J., R. J. Orr, P. D. Penning, and D. R. Lockyer**. 1991. Uptake, cycling and fate of nitrogen in grass-clover swards continuously grazed by sheep. J. Agric Sci. Camb. **116**: 47-61.

**Parton, W. J., D. S. Schimel, C. V. Cole and D. S, Ojima**. 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. Soil Sci. Soc. Amer J. **51**: 1173-1179.

**Rajan, S. S. S., S. F. Ledgard and B. S. Thooold**. 1993. Field evaluation of partially acidulated phosphate rocks as fertilizers for permanent pastures. Proc. 17th Int. Grass. Cong.New Zealand. pp. 1441-1442.

**Rodriguez, M., G. Besga and M. Oyanarte.** 1992. The responses of perennial ryegrass-white clover to P and K fertilization in terms of yield and chemical composition. Proc. 14th Gen. Meeting of the European Grasl. Cong. Lahti, Finland.

**Rowarth, J. S. and A. G. Gillingham**. 1990. Effects of withholding fertilizer on pasture production and phosphate cycling in hill country. Proc. N.Z. Grasl. Ass. **51**: 17-20.

**Russelle, M. P**. 1996. Nitrogen cycling in pasture systems. Page 125-157 *in* E. J. Joost and C. A. Roberts, eds. Nutrient Cycling in Forage Systems. Potash and Phosphate Institute. Norcross, GA.

**Ruz-Jerez, B. E., P. R. Ball and R. W. Tillman**. 1988. The role of earthworms in nitrogen release from herbage residues. Pages 355-371 *in* D. S. Jenjinson and K. A. Smith, eds. Nitrogen Efficiency in Agricultural Soils, Elsevier, New York.

**Ruz-Jerez, B. E., R. E. White and P. R. Ball**. 1995. A comparison of nitrate leaching under clover-based pastures and nitrogen-fertilized grass grazed by sheep. J. Agric. Sci. Camg. **125**: 361-369.

**Ryden, J. C., Ball, P. R. and Garwood, E. A.** 1984. Nitrate leaching from grassland. Nature. London **311**: 50-53.

**Scholefield, D., E. A. Garwood and N. M. Titchen.** 1988. The potential of management practices for reducing losses of nitrogen from grazed pastures. Pages 220-229 *in* D. S. Jenkinson and K. A. Smith, eds. Nitrogen Efficiency in Agricultural Soils. Elsevier, New York.

**Scholefield, D, D. R. Lockyer, D. C. Whitehead and K. C. Tyson**. 1991. A model to predict transformations and losses of nitrogen in UK pastures grazed by beef cattle. Plant and Soil **132**: 165-177.

**Scholefield, D. and K. C. Tyson**. 1992. Comparing the levels of nitrate leaching from grass/clover and N-fertilized grass swards grazed by beef cattle. Proceedings of the 14th General Meeting of the European Grassland Federation, Lahti, Finland. pp. 530-533.

**Scholefield, D., K. C. Tyson, E. A. Garwood, A. C. Armstrong, J. Hawkins and A. C. Stone**. 1993. Nitrate leaching from grazed grassland lysimeters: effects of fertilizer input, field drainage, age of sward and patterns of weather. J. Soil Sci. **44**: 601-613.

**Scholefield, D. and A. C. Stone**. 1995. Nutrient losses in runoff water following application of different fertilizers to grassland cut for silage. Agric. Ecosys. and Environ. **55**: 181-191.

**Scholefield, D. and N. M. Titchen**. 1995. Development of a rapid field test for soil mineral nitrogen and its application to grazed grassland. Soil Use and Management **11**: 33-43.

**Scholefield, D., D. R. Lockyer and S. F. Ledgard**. 1996. The simulation of N fixation by white clover within a model describing N flows in grazed grassland. Abstracts of Communications, Proceedings of the twenty seventh Meeting of the Agricultural Research Modeller's Group. J. Agric Sci. Camb. **126**: 117-122.

**Scholefield, D. and S. U. Smith**. 1996. Nitrogen flows in ley-arable systems. Pages 96-105 *in* D. Younie, ed. Legumes in Sustainable Farming Systems. BGS Occaisional Symposium No. 30. Aberdeen, Scotland.

**Sherlock, R. R. and K. M. Goh**. 1984. Dynamics of ammonia volatilization from simulated urine patches and aqueous urea applied to pastures. Fert. Res. **5**: 181-195.

**Sherwood, M. and A. Fanning**. 1989. Leaching of nitrate from simulated urine patches. Pages 32-44 *in* J. C. Germon, ed. Management Systems to Reduce Impacts of Nitrates. Elsevier, London.

**Sinclair, A. G. and M. Rodriguez Julia**. 1993. Defining economically optimal phosphorus fertilizer rates for permanent grass-clover pastures in Spain and New Zealand. Proc. 17th Int. Grass. Cong. New Zealand. pp. 1424-1425.

**Spatz, G**. 1992. Nitrate leaching under pastures. Proceedings of the 14th Meeting of the European Grassland Federation. Lahti, Finland. pp. 208-212.

**Stevens, R. J.** 1988. Some factors affecting the efficiency of fertilizer nitrogen for grass production in spring. Pages 177-191 *in* D. S. Jenkinson and K. A, Smith, eds. Nitrogen Efficiency in Agricultural Soils. Elsevier, London.

**Sutton, M. A., C. E. R. Pitcairn and D. Fowler**. 1993. The exchange of ammonia between the atmosphere and plant communities. Adv. in Ecol. Res. **24**: 301-393.

**Tallowin, J. R. B. and S. K. E. Brookman.** 1996. The impact of differences in nitrogen content, nitrogen utilization and loss from laminae on competition between four grass species in an old pasture. J. Agric. Sci. Camb. **126**: 25-35.

T**allowin, J. R. B., F. W. Kirkham, S. K. E. Brookman and M. Patefield.** 1990. Response of an old pasture to applied nitrogen under steady-state continuous grazing. J. Agric. Sci. Camb. **115**: 179-194.

**Titchen, N. M. and Scholefiel**d. 1992. The potential of a rapid test for soil mineral nitrogen to determine tactical applications of fertilizer nitrogen to grassland. Pages 223-230 *in* J. R. Archer, K. W. T. Goulding, S. C. Jarvis, C. M. Knott, E. Lord, S. E. Ogilvy, J. Orson, K. A. Smith and B. Wilson, eds. Nitrate and Farming Systems. Aspects of Applied Biology 30. Assoc. of Appl. Biol., H. R. I., Wellesbourne, UK.

**Thompson, R. B., J. C. Ryden and D. R, Lockyer.** 1987. Fate of nitrogen in cattle slurry following surface application or injection. J. Soil Sci. **38**: 689-700.

**Thornley, J. H. M. and E. I. J. Verberne**. 1989. A model of nitrogen flows in grassland. Plant, Cell and Environ. **12**: 863-886.

**Tyson, K. C., E. A. Garwood, A. C. Armstrong and D. Scholefield**. 1992. Effects of field drainage on the growth of herbage and the liveweight gain of grazing beef cattle. Grass and Forage Sci. **47**: 290-301.

**Uhlen, G.** 1989. Surface runoff losses of phosphorus and other nutrient elements from fertilized grassland. Norwegian J. Agric. Sci. **3**: 47-55.

**Van der Meer, H. G. and H. J. van der Putten**. 1995. Reduction of nutrient emissions from ruminant livestock farms. Pages 118-135 *in* G. Pollott, ed. Grassland into the 21st Century: Challenges and Opportunities. British Grassland Society Occaisional Symposium No. 29. BGS, UK.

**Van de Ven, G. W. J.** 1996. A mathematical approach to comparing environmental and economical goals in dairy farming on sandy soils in The Netherlands. Unpublished PhD Thesis. Wageningen Agricultural University. 239 pp.

**Vellinga, Th. V., A. P. Wouters and R. G. M. Hofstede**. 1996. System of adjusted nitrogen supply (SANS) for fertilization of grassland. Pages 381-385 *in* O. van Cleemput, ed. Progress in Nitrogen Cycling Studies. Kluwer Academic Press, The Netherlands.

**Vollenweider, R. A.** 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters with particular reference to nitrogen and phosphorus as factors of eutrophication. GP OE/515, Organization for Economic Co-operation and Development, Paris.

W**atson, C. J., C. Jordan, P. J. Taggart, A. S. Laidlaw, M. K. Garrett and R. W. J. Steen**. 1992. The leaky N-cycle in grazed grassland. Pages 215-222 *in* J. R. Archer, K. W. T. Goulding, S. C. Jarvis, C. M. Knott, E. Lord, S. E. Ogilvy, J. Orson, K. A. Smith and B. Wilson, eds. Nitrate and Farming Systems. Aspects of Applied Biology 30. Association of Applied Biologists, H. R. I. Wellesbourne. UK.

**Wedin, D. A**. 1996. Nutrient cycling in grasslands: an ecologist's perpective. Pages 29-42 *in* R. E. Joost and C. A. Roberts, eds. Nutrient Cycling in Forage Systems. Potash and Phosphate Institute. Norcross, GA.

**Wedin, D. A. and D. Tilman**. 1990. Species effects on nitrogen cycling: a test with perennial grasses. Oecologia **84**: 433-441.

**West, C. P. and P. Mallarino.** 1996. Nitrogen transfers from legumes to grasses. Pages 167-173 *in* R. E. Joost and C. A. Roberts, eds. Nutrient Cycling in Forage Systems. Potash and Phosphate Institute. Norcross,GA.

**Whitehead, D. C**. 1995. Grassland Nitrogen. CAB International, Wallingford, UK. PP. 152-176.

**Whitehead, D. C. and D. R. Lockyer**. 1989. Decomposing grass herbage as a source of ammonia in the atmosphere. Atmospheric Environment **23**: 1867-1869.

**Figure 1**
Typical herbage dry matter yield (full line) and N yield (dashed line) response to the application of mineral fertilizer to cut grassland.

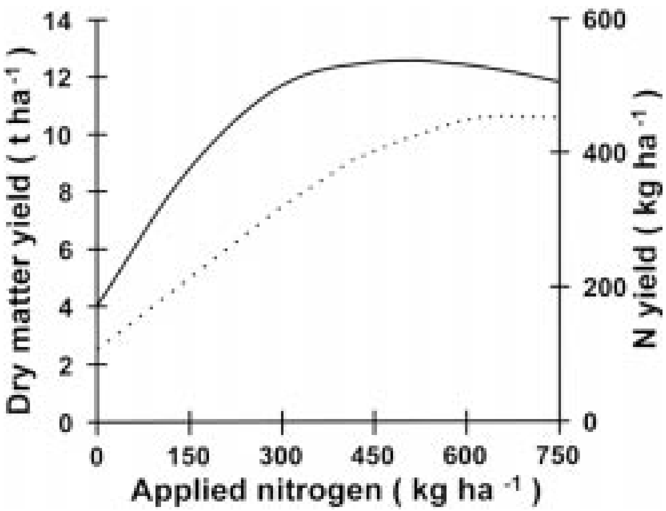


**Figure 2**
Model optimised pattern of fertilizer application to obtain equal efficiency of N use during the year (shaded bars), compared to a typical pattern of fertilizer currently recommended in the UK (clear bars).

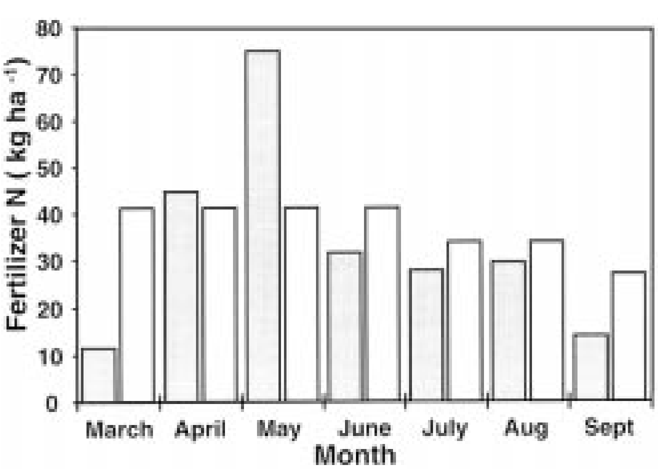


**Figure 3**
The relationship between uptake and loss of matter during continuous grazing at a range of leaf area index (re-drawn from Parsons *et al*., 1983), developed from measurements of gross photosynthesis and leaf area index and from the assumption that one third of the standing live weight dies every 11 days.

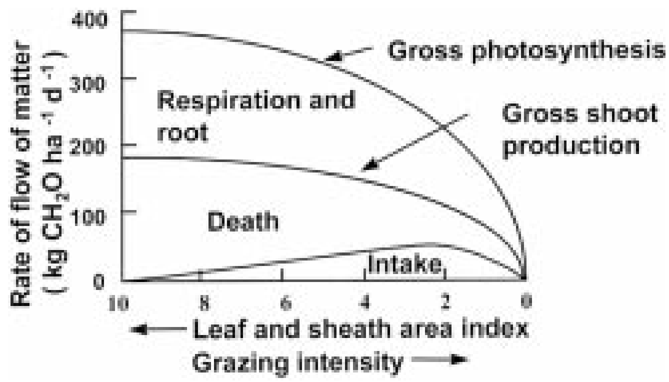


**Figure 4**
Relationship between the concentration of N in the herbage consumed and the proportion of intake N retained by the animal. (Re-drawn from Scholefield et al., 1991).

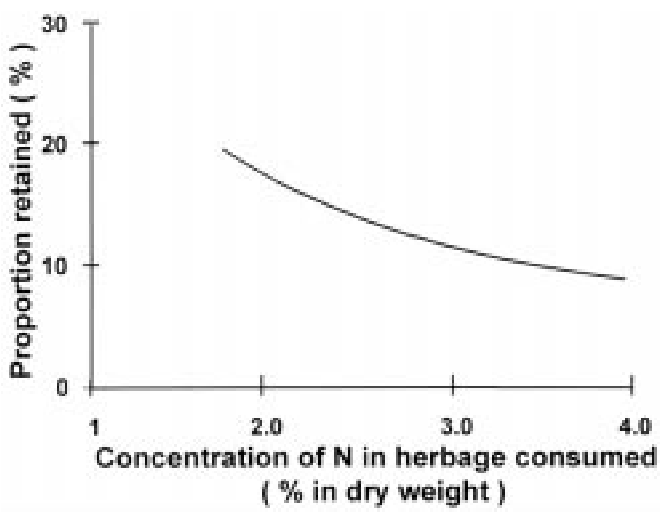

**Figure 5**
Charts showing the proportion of N leached to surface water in organic combination (shaded area) during the 1995-96 winter from 'farmlet' lysimeters on a clay loam soil in sw England. Note the contrast between grass receiving 280 kg N $ha^{-1}$ fertilizer (left chart) and grass/white clover receiving zero fertilizer (right chart).

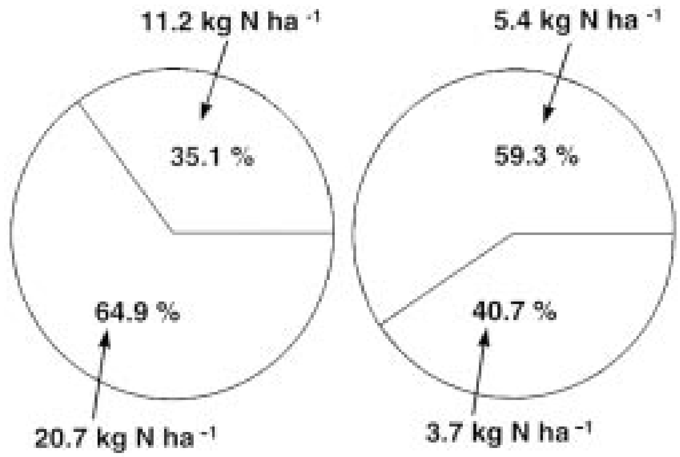


**Figure 6**
Relationship between the annual flux of soil mineral N and the proportion of it partitioned to plant uptake. (Re-drawn from Scholefield *et al*., 1991).

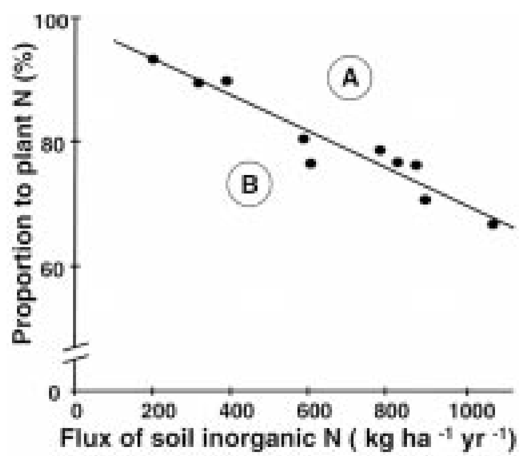


**Figure 7**
Mineral N 'profile' for optimizing monthly additions of fertilizer N to a two year old cut sward on a sandy loam with a history of ley-arable management in sw England, for the production of 9.7 t $ha^{-1}$ dry matter from 340 kg N $ha^{-1}$ fertilizer.

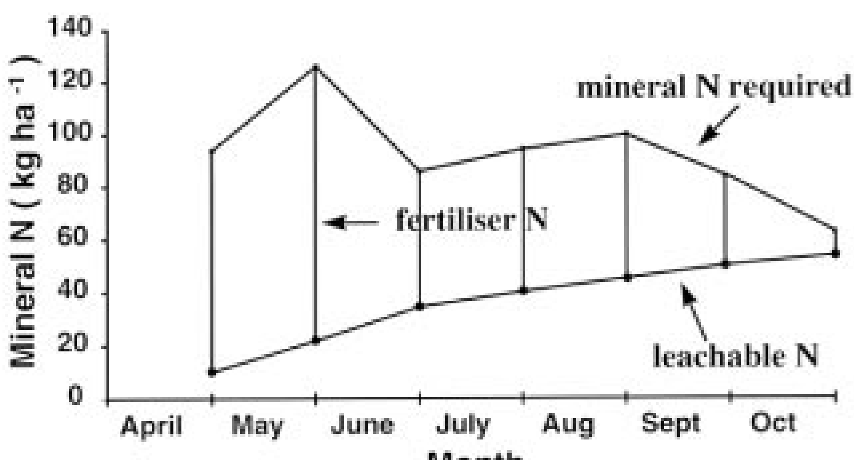

**Table 1**
Efficiency indices for N use within a grazed field of a dairy pasture system, with N input either from N fixation or from mineral fertilizer.

| Efficiency index | Low input system (grass/ white clover) | High input system (mineral fertilizer) |
|---|---|---|
| 1. Product N/Total N input | 0.33 | 0.22 |
| 2. Product N + Total N Loss/Total N Input | 0.47 | 0.65 |
| 3. Product N/Fertilizer N | × | 0.26 |
| 4. Product N/Total N losses | 2.40 | 0.51 |

**Table 2**
Step-wise decrease of permissible surpluses of N and P (kg $ha^{-1}$) required by livestock farmers in The Netherlands.

| Year | N | P |
|---|---|---|
| 1998 | 300 | 17 |
| 2000 | 275 | 15 |
| 2002 | 250 | 13 |
| 2005 | 200 | 11 |
| 2008 | 180 | 9 |