MATERIAL GENERADO

Uso de balances de nutrientes como herramienta de buenas prácticas ganaderas en sistemas productivos del sur de Chile.

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resulting in reduced N offtake. The inhibition could be due to less mineralization of soil N, physical impedance of root development and reduced N uptake, the latter resulting from changed physical and chemical conditions such as anaerobism and reduced available water. In the current experiment Ca and Mg concentrations were depressed by wheel tracking but P and K concentrations were less affected, probably because NPK fertilizer was applied for each cut. The offtakes of N and minerals were generally lower in the wheeled treatments, being a reflection in most cases of lower concentrations allied to lower DM yields.

In terms of herbage yield and quality and the soil parameters measured, there was no difference between sward types in their response to wheel tracking treatments, yet tetraploid cv. Condesa had 35% fewer tillers on average than diploid cv. Contender. However, the larger size of individual tillers, typical of tetraploids, may have conferred upon the Condesa swards the same overall vegetative cover and cushioning capacity against wheel tracking as Contender swards.

Conclusions

Wheel traffic reduced herbage DM yield and adversely affected N and mineral composition mainly owing to the indirect effects of soil compaction on the rooting environment, although some direct plant injury was also evident. The wheel tracking simulation of silage operations resulted in a mean reduction of 8% in DM yield, which was at the lower end of published yield reductions as the tractor was relatively light in relation to most machinery loads in other work. Frequent or delayed wheel passes exacerbated the negative effects compared with infrequent or immediate wheel passes. The two sward types responded similarly to wheel tracking treatments. It is concluded that in practice silage operations in the field should be carried out with the lightest equipment and fewest traffic activities consistent with an efficient system and without undue delay between operations at individual harvests.

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Opportunities for reducing the environmental impact of dairy farming managements: a systems approach

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Abstract

Dairy farming systems are important sources for the emission of a number of materials that include various forms of nitrogen (NO_3^-, N_2O and NH_3) with potential environmental impact. The present paper is a systems synthesis study and assesses the likely impact of changes in management on N flows and losses. These include tactical fertilizer adjustment, slurry injection, maize silage production and the use of white clover as an alternative to fertilizer N. Implications for greenhouse gases (N_2O and CH_4) and support energy have also been considered. Substantial reductions in inputs and total and proportional losses by all the options considered were predicted by this study. Thus, using a tactical approach to fertilizer application and injecting slurry or using 50% maize silage reduced overall N losses from 160 (under conventional management) to 86 and 109 kg N ha^{-1} respectively. Combining both possibilities reduced losses further to 69 kg ha^{-1}. Although use of white clover, especially at low contents in the sward, was the most effective regime to reduce losses, this was at some cost to production so that losses per livestock unit (LU) did not always differ from those under other managements. Changing the N management had consequences for greenhouse gas emission with an estimated maximum 70% reduction in N_2O release. The effects on CH_4 emissions were relatively small. Substantial reductions in support energy costs were also obtained: these arose mainly from the reduction in fertilizer N use, which represented 66% of the total support energy in the original system.

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Introduction

Dairy farming is a major agricultural enterprise in the UK which has adopted new techniques rapidly to improve productivity. Intensive grass-based dairy farming managements are, in the main, dependent upon large inputs of nitrogen (N)-based fertilizer to stimulate sufficient dry matter production to sustain milk production at economically attractive levels. Inputs of fertilizer N have generally been determined by economic optima as established in fertilizer response trials without necessarily having regard to efficiency of utilization. It has become increasingly obvious that continued high inputs of N can lead to imbalances that result in the transmission of excesses from the farm to waters and the atmosphere, with potential adverse environmental impact (Jarvis *et al.*, 1995). As well as environmental concerns, there are also current economic and sociopolitical pressures to encourage more extensive farming systems. Because of these issues (Wilkins, 1993), current managements are being assessed and options for the future considered.

It is clear that dairy farms in temperate regions can provide sources for a number of important solutes and gases. Much leaching, denitrification and ammonia (NH_3) volatilization can take place from intensively managed grassland soils and the other components of animal production, i.e. from stored and applied manures and other farm wastes or directly from the animal houses (Jarvis *et al.*, 1995). Grazed swards can lose much NO_3^- through leaching (Scholefield *et al.*, 1993) and current EC legislation on quality of potable and other waters requires actions to reduce NO_3^- concentration in drainage waters. Animal production systems are the primary sources for the increases in NH_3 in the atmosphere (Jarvis and Pain, 1990) that have occurred over recent years with consequences for atmospheric chemistry (INDITE, 1994) and the enrichment of natural/semi-natural ecosystems with N (Pearson and Stewart, 1993). Recent calculations indicate that grassland-based agriculture in the UK is also one of the major sources of emission of nitrous oxide (N_2O) (INDITE, 1994), with a significant contribution therefore to global warming effects (IPCC, 1992). Ruminant digestive systems are one of the major global sources of methane (CH_4) generation, which, as another important greenhouse gas, is also causing concern (Crutzen, 1991).

There is much fragmentary information about N flows, transformations and losses that could be used to provide the basis for greater efficiency of utilization within particular components of dairying systems. Substantial reductions in losses could be made immediately from alternative management of farm manures (Rees *et al.*, 1992) and leaching losses could be reduced by using new fertilizer managements (Titchen and Scholefield, 1992). However, for long-term success and sustainability it is essential that whole systems are considered because changes introduced to remedy one loss process may exacerbate other problems.

There have been recent descriptions and assessments of N flows and losses (Jarvis, 1993) and of CH_4 and N_2O emissions (Jarvis and Pain, 1994) from a UK dairy farm that has characteristics and management that are typical of many intensive systems. Another recent study has described N flows and losses in Dutch dairy farms (Aarts *et al.*, 1992). In the present paper we examine the implications of using alternative management options for the dairy farm described previously (Jarvis, 1993) for N and trace gas losses. Manipulation of N inputs provides the basis for each of the options discussed because of its pivotal role in (a) determining production responses and (b) environmental issues. To do this we use the structure of the original case study farm and apply findings from recent experimental data, calculations based on model descriptions and current best estimates in relation to a number of management technologies.

The case study dairy farming system

The rationale behind, and full characteristics of, the case study farm are described elsewhere (Jarvis, 1993) and key features are shown in Table 1. Our study is based on a hypothetical, but typical, dairy farm in SW England, which relies on 250 kg N fertilizer ha^{-1} to produce sufficient grazing and ensiled grass to maintain 102 milking cows and a total of 164 livestock units. The assumptions for calculating the flows of N and of N_2O and CH_4 emissions are as described by Jarvis (1993) and Jarvis and Pain (1994). We consider the effects of modifying the management in various ways with the objective of improving N management by (a) making more efficient use of fertilizer N and that recycled in slurry, (b) depending on inputs of biologically fixed N_2 from white clover in mixed swards, and (c) partly substituting maize silage for conserved grass. Five alternative managements are examined for which recent research findings have allowed greater understanding and for which appropriate technology has become available. It is assumed throughout that no silage effluent is produced and that a wilted silage (> 28% dry matter) is used. Forage conserved in this way gives similar levels of milk production ha^{-1} to those obtained with well-preserved, wetter silage (Zimmer and Wilkins, 1984). Fertilizers are assumed to be applied evenly across all fertilized areas. As well as losses of N to waters and the atmosphere, emissions of the greenhouse gases N_2O and CH_4 are also considered, and although we do not address changes in the total C-balance of the farm, effects on the support energy requirements are discussed.

Table 1. Characteristics of dairy farm case study (from Jarvis, 1993)

Soil type/grassland	25 ha clay loam: poor drainage: long-term swards 51 ha loam: moderate drainage: reseeded swards
Animals	102 cows yielding 5554 l per cow 110 other cattle (followers and beef calves) Total livestock units 165
Purchased feeds	129·4 t DM (concentrates 1390 kg per cow − 69 kg per cow other supplements)
Farm produced feeds	393·5 t silage DM (26·5% DM) 574·0 t grazed grass DM
Purchased bedding	45·6 t DM
Wastes	1693 m^3 slurry 670 m^3 dirty water
Annual N inputs	
Fertilizer	19·00 t
Atmosphere	1·90 t
Fixation	0·76 t
Feeds and bedding	3·93 t
Total	25·59 t

DM, dry matter.

Option 1: tactical fertilizer N use and slurry injection

Current advice for N fertilizer use on grassland in the UK is based on economic optima responses with only minimal regard to efficiency of use of the applied N and relatively crude adjustments made in relation to supplies of recycled N from either mineralization of soil organic matter or excretal returns (MAFF, 1994). Model calculations (Scholefield *et al.*, 1991) showed that in the case study farm nearly 12 N (157 kg N ha^{-1}) would be mineralized. Recent *in situ* measurements of net mineralization in soil similar to that of the model farm have indicated that even greater rates of mineralization may occur, i.e. > 300 kg ha^{-1} (Gill *et al.*, 1995). As well as this source, the estimated amounts of recycled N deposited in excreta during grazing in the conventional management was 10·5 t (138 kg ha^{-1}), much of which would have been in an available form: a further 8·5 t was available in slurry and dirty water (equivalent to 111 kg ha^{-1}) (Jarvis, 1993). Current recommendations (MAFF, 1994) for N fertilizer discriminate between cut or grazed swards but the difference in the amounts to be applied (40 kg ha^{-1}) is small relative to excretal returns. Thus, there is also only a simple separation into soils with low, medium or high background N status either with (Thomas *et al.*, 1991), or without (MAFF, 1994), some recognition of site characteristics. There is, therefore, opportunity to improve fertilizer N recommendations to take much better account of background soil/site/management differences and supplies of recycled N.

The integrated effect of all inputs to the soil mineral N pool is to produce an erratic level of available N in the soil profile. A recent approach (Titchen and Scholefield, 1992) adjusts fertilizer N inputs tactically in relation to the current soil mineral N contents, the estimated grass crop requirements and predicted effects of mineralization and denitrification. Using this approach on commercial dairy farms annual fertilizer requirements were reduced from, on average, 300 to 211 kg N ha^{-1} compared with the farmer's routine methodology with no effect on dry matter production (Titchen and Scholefield, 1993). A similar effect on the case study farm would reduce overall annual fertilizer N application from 250 to 175 kg N ha^{-1}.

Additionally, there are also opportunities to use N in slurry more efficiently. The case study farm was estimated to hold the equivalent of 120 kg N ha^{-1} in store as excreta or dirty water, of which 45% (54 kg ha^{-1}) was as NH_4^+ with potential, therefore, to be lost. Calculations indicated that 22 kg N ha^{-1} would be lost as NH_3 volatilization during surface applica-

Table 2. Inputs and production characteristics of management options for dairy farming systems (derivation of value as described by Jarvis, 1993, with other assumptions as defined in the text)

Option	N inputs (kg ha^{-1})					Stocking rate (LU ha^{-1})	Silage production (t DM)	
	Fertilizer	Atmosphere	Fixation	Concentrates and bedding	Total		Grass (/clover)	Maize
Case study	250	25	10	52	337	2·17	393·5	–
1†	155	25	10	52	242	2·17	393·5	–
2a‡	–	25	144	41	210	1·74	314·8	–
b§	–	25	72	41	138	1·74	314·8	–
3††	185	25	10	52	272	2·17	196·7	196·7
4‡‡	115	25	10	52	202	2·17	196·7	196·7
5§§	–	25	107	41	173	1·74	157·3	157·2

†Tactical N − injected slurry. ‡High clover swards. §Low clover swards. ††50% maize silage. ‡‡Option 1 − Option 3. §§Option 2a − Option 3.
DM, dry matter.

Table 3. [illegible] losses over whole farm

Option	Leaching		Denitrification		NH_3 volatilization			Total loss	Loss per LU
	Grazing cutting	Slurry‡ application	Grazing cutting	Slurry‡ application	Grazing cutting	Slurry‡ application	Store/ House		
Case study	44·2	11·8	44·5	10·8	12·3	22·4	13·8	160	74
1	20·6	11·8	20·7	10·8	5·8	2·2	13·8	86	39
2a	18·7	9·4	18·7	8·6	4·7	17·9	11·1	89	51
b	6·7	9·4	6·7	8·6	1·6	17·9	11·1	62	36
3	32·3	2·9	34·5	9·2	7·6	11·1	11·3	109	50
4	15·9	2·0	16·1	9·2	3·8	9·5	11·3	69	32
5	13·6	2·3	14·3	8·4	2·0	6·8	7·7	55	32

†Losses calculated according to the model of Scholefield *et al.* (1991) and/or as described by Jarvis (1993).
‡Includes dirty water

tion by a conventional vacuum tanker spreader. Studies have shown that injection of slurry reduces NH_3 volatilization substantially (Phillips *et al.*, 1990). Option 1 assumes that the slurry is injected with a shallow injector (to 60 mm), and that NH_3 volatilization is reduced to 10% of that from spreading (B. F. Pain, personal communication). This would require careful timing as other studies (van der Weerden *et al.*, 1994) have shown that under some environmental conditions, losses from shallowly injected slurry can be at least as great as from conventional spreading. We assume that the method is effective and provides another 20 kg of N ha^{-1} in the soil that is available for uptake and to substitute for N fertilizer (Table 2). Losses through leaching and denitrification are assumed not to have been affected by injection. Although this may not be the case (van der Weerden *et al.*,1994), there are insufficient data to be able to predict effects with any confidence. Furthermore, run-off would also be reduced and if injection took place at the appropriate time, any leaching would also be reduced: the assumption for the present purposes is that these effects would balance.

The tactical N approach would take account of the immediate effects of conserving available N in this way, and over the longer term, of any enhanced build up of potentially mineralizable N either through the reduction in the loss of mobile organic materials transported in run-off or eventual release from more stable forms of organic matter. However, if it is assumed that the present impact is restricted to reducing NH_3 volatilization from 22 to 2 kg ha^{-1} (Table 3), the fertilizer requirement is 155 kg N ha^{-1}, i.e. an overall reduction in fertilizer N of 38%, and a total annual input to the farm therefore, of 242 kg N ha^{-1} compared with 337 kg N ha^{-1} (Table 2). This reduction would be achieved at no cost to dry matter production, and stocking rates and animal production could be maintained.

Losses of N for this management system can then be calculated as before, i.e. from the model of Scholefield *et al.*, (1991) or as described previously (Jarvis, 1993).

Option 2: replacement of fertilizer N by biologically fixed N_2 by white clover

Losses of N from mixed grass/white clover swards have been substantially lower than those from highly fertilized ones, especially where these were grazed (Ryden *et al.*, 1984). This finding encouraged the perception that biologically fixed N_2 had less environmental impact than that derived from mineral N fertilizers. More recently, studies have shown that N losses are related to the N inputs into the system, regardless of their origin, and consequently depend on the production and animal carrying capacity of the enterprise. Thus, when unfertilized (N) grass/clover pastures were compared with N-fertilized grass swards there was little difference in the quantities of NO_3^- N leached (Cuttle *et al.*, 1992), and determined by stocking rate irrespective of the origin of the N supplied (Cuttle and Jarvis, 1995), with N loss ha^{-1} increasing with increasing animal numbers.

Although few direct comparisons have been made, one of the consequences of reliance on white clover would be a reduction in the stocking density and overall production from the system. For the

[illegible] on a white clover-based sward are 30% of those achieved in an adequately fertilized one (see Scholefield and Tyson, 1992) and that production per animal remains the same. This would require 20% less herbage dry matter. The model of Scholefield *et al.* (1991) can be used to estimate the N supply required to produce this quantity of dry matter (assuming that there would be no difference in requirements with fertilizer or fixed N_2) and the associated losses then estimated as before. The annual input from white clover required for this level of production is 144 kg N ha^{-1} over the whole farm and this is the basis of Option 2a (Table 2). Proportionate decreases in farm slurry production (and its impact) would also result and losses are calculated on that basis. Livestock carrying capacity decreases from 2·17 livestock units (LU) ha^{-1} in the case study to 1·74 LU ha^{-1} (Table 2).

In fact, the requirements for the white clover to make inputs of N of this order may be overestimated. It has been shown that under carefully controlled and sustained management guidelines, 80% of the sheep carrying capacity on grass swards with 420 kg N on mixed swards can be achieved with very low (< 5%) white clover contents (Parsons *et al.*, 1991). Parsons *et al.* (1991) estimated that only 24 kg N ha^{-1} were fixed by the clover under this regime and there were concomitant low N losses compared with the highly fertilized sward. Although it may not be possible to transfer directly these findings to grazing cattle, it does indicate a potential to reduce the requirement for the amounts of N_2 fixed while maintaining dry matter production. As a further example therefore (i.e. Option 2b), the effects of reducing fixation rate by 50% to 72 kg N ha^{-1} on N losses are also shown (Table 3). The stocking rate was presumed to have remained at 1·74 LU ha^{-1}.

Option 3: use of forage maize

Most of the inefficiencies in N utilization within animal production systems relate to poor utilization of dietary N by ruminants (Beever and Reynolds, 1994). Under average conditions, the efficiency of utilization by dairy cows is 16–23% (van Vuuren and Meijs, 1987). For the case study farm (Table 1), the estimate for the overall conversion of N into milk by cows and protein into growth of the young animals was assumed to be 20%.

Dietary N utilization is more efficient with mixtures of maize and grass silage than with grass silage alone (van Vuuren and Meijs, 1987: [illegible] is available and the climate is suitable for maize production, substitution of maize silage for a proportion of the ensiled grass may be an effective option. Milk production per cow was assumed to be unaffected by this substitution. This is probably a conservative assumption as experiments have shown that milk yield is increased by incorporation of maize silage in the diet (Phipps, 1990). Stocking density is assumed to remain at 2·17 LU ha^{-1}. At a harvestable dry matter yield of 10 t ha^{-1}, an area of 19·7 ha is required to grow maize to substitute for half of the ensiled grass in the case study. This can be grown without use of fertilizer N (Phipps and Pain, 1978) and, therefore, has the immediate impact of reducing the overall fertilizer requirement for the farm from 19 t to 14·1 t (i.e. to, on average, 185 kg N ha^{-1}, for the whole farm, Table 2).

In Option 3, maize is grown on the better drained land currently occupied by younger swards. As already indicated, the N requirement of the maize crop is provided from the stored manures and would produce forage with 1·5% N in the dry matter. This means that in total the dietary intake of N in conserved forage and concentrates (which remain as before) in this option would be 11·2 t compared with 12·7 t with ensiled grass in the case study farm (Table 4). Assuming that, because production is maintained, N in milk and animal tissues remains the same (i.e. 5·1 t), total amounts of N excreted would decrease from 20·1 t to 19·1 t and from 10· t to 8·7 t in winter while the animals were indoors. Thus, as well as reducing N inputs, use of maize produces a slight increase in utilization efficiency of dietary N from 20% to 21%. At the same proportional rate of loss by NH_3 from the houses and stores as in the case study (i.e. 10·3% of the excreted N), some 7·8 t N would be available for application to land compared with 9·1 t.

Cultivating an established sward to provide a tilth suitable for maize may cause release of N through an enhanced rate of mineralization. Surges in the soil mineral N content below disturbed grass swards indicate that this could be substantial (Lloyd, 1992), but this could be minimized by growing maize continuously on the same area of land. There are few data as yet on which to base estimates of leaching and other losses but recent Dutch studies (Schröder *et al.*, 1993) on a sandy soil have shown that NO_3^- leaching can be substantial. The present farm has much less freely draining soils and so leaching losses from the maize area are assumed to be less than on the Dutch sandy soil and equal to 25% of that under grass in the conventional management. To achieve this requires that the slurry N is utilized efficiently and ideally applied either immediately before or during the cultivation phase with rapid incorporation, or during the rapid growth stage of the maize (by band spreading, for example). If this took place during spring, then leaching and run-off resulting from slurry application to the maize is minimized. NH_3 volatilization would be reduced because of rapid incorporation, but has been assumed to be 10% of the total N added (compared with 18·6% when slurry is surface spread) and denitrification to be equivalent to 20% of the NH_4^+−N as before. Again it is probable that inaccuracies in assumptions for any one component of the losses will be balanced by differential effects in others.

The assumed 10 t of harvestable maize dry matter ha^{-1} will represent only a proportion of the total dry matter produced, which is assumed to be 12 t ha^{-1}. At 1·5% N in the dry matter, this overall yield would require 3·55 t N. If older swards were used there would be a significant supply through mineralization; this would be reduced with younger swards but if maize cropping were repeated on the same area, repeated applications of slurry would build up potential for release. Forty-five per cent of the slurry N is in NH_4^+ form and available for plant uptake: some of the remainder will be in an easily mineralized form and become available during the growing season. This is assumed to be 50–60% of the non-NH_4^+ N. With this composition and behaviour, and taking into account losses as defined above, a total of 7·2 t N is needed to be supplied from slurry to produce the required amounts of harvestable maize silage, i.e. most of that held in the store (Table 4). Utilization of N in this way therefore provides a sink for most of the N in slurry and leaves a residual 0·7 t N to be applied to the grassland. Denitrification and NH_3 losses from this component were calculated as before and leaching/surface run-off losses have been assumed to be zero because, on average, lower application rates would be used and/or more appropriate environmental conditions can be chosen for the smaller volume to be dispersed.

Options 4 and 5: management combinations

The options so far considered have each been based upon a single major change to existing management procedures. It is obviously possible to integrate a number of options and we therefore also consider the effects of combining maize silage production with tactical fertilizer application (Option 4) or with grass/clover (Option 5). In this latter case, mixed swards with the higher white clover contents (Option 2a) were assumed. In both options the assumptions were as before and the residual slurry was injected into the respective grass or grass/clover swards. In Option 4, maize production reduced the area requiring N fertilizer and lowered the average overall requirement to 115 kg ha^{-1} (Table 2) with concomitant reduction in losses, which were calculated as described before, as were those associated with the slurry applied to maize (Option 3) or injected into the sward (Option 1). Livestock production was maintained at 2·17 LU ha^{-1}. For Option 5 (grass-clover with maize silage), the assumptions made about the grass-clover area and the production of maize to provide 50% of the total silage were as for Options 2a and 3 respectively. Because maize allows an increase in dry matter production per unit area compared with the mixed sward, the area required to produce half the silage needed for the same stocking rate as Option 2a (i.e. 1·74) is less at 15·7 ha. An alternative would be to

Table 4. Amounts of N in components of housed animal systems under different management regimes (t per farming system per year)

Option	N in conserved feeds: Grass (/clover) silage	N in conserved feeds: Maize silage	N in conserved feeds: Concentrates	N excreted	N stored	N in slurry applied to: Grass (/clover) area	N in slurry applied to: Maize area
Case study	3·93	–	3·75	10·19	9·14	9·14	–
1	3·93	–	3·75	10·19	9·14	9·14	–
2a	7·14	–	3·00	8·15	7·31	7·31	–
b	7·14	–	3·00	8·15	7·31	7·31	–
3	4·46	2·95	3·75	8·68	7·79	0·75	7·04
4	4·46	2·95	3·75	8·68	7·79	0·75	7·04
5	3·57	2·36	3·00	7·18	6·44	0·82	5·62

maintain the maize area at 19·7 ha and increase the stocking rate slightly, accepting that this would increase the emission of pollutants.

Effects of managements on nitrogen balances and losses

It was clear from the previous analysis (Jarvis, 1993) that the calculated emissions of N from the dairy farm to atmosphere and drainage were substantial and equivalent to 160 kg N ha^{-1} $year^{-1}$, or 74 kg N per LU (Table 3). This represented 47% of the total annual input but is possibly a substantial underestimate because a further 32% of the annual inputs were not accounted for. Although this may represent inadequate estimation of some of the losses, especially of denitrification (Jarvis, 1993), a substantial proportion of the unaccounted for N may be immobilized in soil organic matter. Studies on similar soil type with similarly aged swards in SW England have shown that 64 kg N ha^{-1} per year were immobilized (Tyson, 1993). This rate occurred with long-term swards, but would not remain constant and N accumulation would decrease with time. However, it can be estimated that at the current status of the present swards 4·9 t N per year could be immobilized.

All the proposed options had effects in reducing N losses by at least 32% (Option 3, maize silage) and at best by 66% (Option 5, grass/clover swards plus maize) compared with the case study farm (Table 5). The relative importance of individual components of the loss changed with each option. In the case study, the ratios of leaching to denitrification and NH_3 losses were approximately 1:1:0.9. Changing to mixed grass/clover swards (Option 2) and tactical fertilizer and maize (Option 4), although reducing all the losses, increased the relative importance of NH_3 volatilization so that the equivalent ratios were 1:1:1.2 and 1:1.3:1.3 respectively. Slurry injection or maize production on their own reduced the amounts, and the relative importance of NH_3 loss. None of the proposed options made specific attempts to reduce NH_3 emission from houses and stores, which was always 33% or greater (on average, 49%) of the total NH_3 loss. New approaches to reducing NH_3 fluxes from animal houses are being investigated (Ketelaars and Rap, 1994). Although the on-farm technology for this is not yet developed, this may become more important as the consequences of NH_3 emissions from livestock farming for environmental quality become more precisely determined (Hutchings *et al.*, 1995). In the meantime, manipulation of management, either to reduce the quantities of excretal N from housed animals or to make better use of that applied to land, provides the best opportunity to reduce NH_3 output.

Leaching losses were also reduced by all the options with the lowest quantities lost from the grass/clover and maize silage combination (Option 5). The case study farm is based on a generally heavy soil, which would lead to relatively low leaching losses. Nevertheless, the losses in all of the options, even when marked reductions have been

Table 5. Nitrogen inputs, output and losses from dairy farming systems (t per year)

	Case study	Option 1	Option 2a	Option 2b	Option 3	Option 4	Option 5
Input							
Fertilizer	19·00	11·78	–	–	14·06	3·74	–
Atmosphere	1·90	1·90	1·90	1·90	1·90	1·90	1·90
Fixation	0·76	0·76	10·94	5·47	0·56	0·56	8·13
Concentrates and bedding	3·93	3·93	3·14	3·14	3·93	3·93	3·14
Farm total	25·59	18·37	15·98	10·52	20·45	[illegible]	13·18
Outputs and losses							
Outputs							
Milk and protein	5·10	5·10	4·08	4·08	5·10	5·10	4·08
Losses							
Leaching	4·28	2·46	2·13	1·22	2·67	1·43	1·21
Denitrification	4·20	2·39	2·07	1·16	3·32	1·92	1·72
NH_3 volatilization	3·69	1·66	2·56	2·32	2·28	1·87	1·25
Farm total	12·17	6·52	6·00	4·70	8·28	5·22	4·19
Unaccounted for	8·32	6·75	5·91	1·73	7·08	4·81	4·91

achieved, were still significant. Results from a similar soil type in SW England (Scholefield *et al.*, 1993), showed that a loss of only 31 kg ha^{-1} per year resulted in initial NO_3^--N concentrations greater than 11·3 mg l^{-1} in drainage, i.e. greater than the EC limit. Assuming similar effects in our study, only options 2, 4 and 5 would have produced initial leachate concentrations lower than the EC limit. In general, the estimated denitrification and leaching losses were almost equal although, as indicated previously, it is probable that the denitrification estimates may have been too low. Basing the farming system on other soil types with different drainage characteristics would influence the balance of loss from these two processes considerably, i.e. more leaching from coarser textured, or more denitrification from heavier soils.

The impact of a change to a white clover-based N economy was in all cases considerable, with reductions per unit area in all the individual components of loss. It is also informative to examine the losses in relation to animal production. Strong positive relationships between sheep stocking rate and NO_3^- leaching have been established in the past (Cuttle *et al.*, 1992). With the present options it can be seen clearly that whereas low losses per unit area could be achieved with a 'normal' clover-based sward (Option 2), production was reduced and losses per LU were greater than, or equivalent to, those based on improved fertilizer management. Nevertheless, it was clear that total losses were low, especially where white clover contents were low (Option 2b) when they were of a similar magnitude to those recorded by Parsons *et al.* (1991) with sheep-grazed pastures. To achieve and sustain such a sward with a low clover composition under cattle farm managements may not be practically achieved or easily sustained, but would apparently provide a very efficient utilization of the N cycling within the system. For a system such as Option 2b to function properly requires that N is recycled rapidly and effectively (Sheehy, 1989). Thus, Parsons *et al.* (1991) estimated that 238 kg of N were recycled by excreta in the pasture system with an annual input of 24 kg of fixed N. Where inputs of N from clover have been increased, for example in a grazed white clover monoculture, NO_3^- leaching (Macduff *et al.*, 1989) and NH_3 volatilization (Jarvis *et al.*, 1991) were increased to rates that were similar to those from highly fertilized grass swards.

The annual input of N, from all sources, to an intensive animal production system is large, i.e. over 25·5 t in the case study farm, of which only 20 per cent was converted into animal products, 47% was estimated as being lost and 32% was not accounted for. Inputs were reduced in all the options considered, to 10·5 t in Option 2b. In this latter case, the apparent efficiency of conversion into usable product was increased to 39%, while losses remained proportionately the same, and the unaccounted for N was reduced to 16·4% of the total. Assuming that immobilization of N in plant residues in the soil accounted for 4·86 t (Tyson, 1993) in all the options, this would remove all of the excess N in Options 2b, 4 and 5 and at least 59% of that in Options 1, 2a and 3. Although there are opportunities for losses, as yet unquantified, in soluble organic forms in drainage or silage effluents for example, the remainder of the excess N is most likely to be accounted for by denitrification. Denitrification assessments are currently likely to underestimate losses through shortcomings in techniques (Smith and Arah, 1990) and an inability to make measurements in all components of the system. In some of the options, particularly those in which maize was involved, there may have been greater opportunity over time for proportionately more N to be incorporated in the soil as organic matter from dung and plant residues. Although this, in turn, would reduce the immediate losses through denitrification, there is the possibility that enhanced mineralization would occur because of the cultural requirements of the maize crop and this would promote further loss by both denitrification and leaching.

Effects on greenhouse gases, N_2O and CH_4

This paper has centred thus far on effects on N cycling and losses *per se* and the modified managements have been devised with N efficiency in mind. The original case study farm has also been examined (Jarvis and Pain, 1994) in terms of its impact on emission of N_2O and CH_4 and it is of value to look at the effect that the options have had on these two gases. N_2O losses were greatest in the case study system and reduced to varying degrees in the alternative managements by over 70% in Option 2b (low clover) but by at least 43% in the remainder (Table 6). The assumption throughout has been that the products of denitrification (N_2 and N_2O) remain in a fixed proportion (i.e. 3:1) in all situations; this is unlikely to be the case. Changing amounts and forms of N inputs are also likely to have had effects on the proportions of N_2O released during

Table 6. Nitrous oxide emissions from dairy farming systems (values derived as described by Jarvis and Pain, 1994)

Option	kg N ha^{-1}		Farm total		
	Fertilized/grazed swards	Wastes return	t N	kg N LU^{-1}	kg CO_2* equivalent
Case study	11·1	2·7	1·05	6·36	7493
1	5·2	2·7	0·60	3·64	4286
2a	4·7	2·1	0·52	3·95	3721
2b	1·3	2·1	0·30	2·25	2115
3	8·6	2·3	0·83	5·02	5914
4	4·0	2·3	0·48	2·40	3421
5	3·6	2·1	0·43	3·27	3079

*Assuming a radiative effect per N_2O molecule of 200 × that of CO_2.

denitrification. Recent studies have shown that relatively less N_2O is produced when NO_3^- substrate levels are low (Scholefield *et al.*, 1994), so that the N_2O losses from the lower input systems may have been lower than indicated.

Methane release was little affected by the present management changes. The case study farm was estimated to emit nearly 12 t CH_4-C during the year (Jarvis and Pain, 1994), in the main from the ruminants themselves. There may have been some effects of the dietary changes imposed by the options involving grass/clover and the incorporation of maize silage (Jarvis and Moss, 1994) but the calculated effects of these would be relatively small. The major differences were between white clover-based and the other systems, because of lower livestock carrying capacity of the white clover-based farms. Thus, the total emission (but not that per LU) would be reduced, on a proportional basis to 9·6 t CH_4-C in Options 2a, 2b and 5, i.e. from 30·0 t to 24·0 t CO_2 equivalents (assuming a radiative effect per CH_4 molecule of thirty times that of CO_2) for the whole farm. Methane was always 80% or more of N_2O-CH_4 emission (in CO_2 equivalents). Changing the N management of the farm had a substantial effect in changing the total CH_4 contribution from 80% of the effect in the case study farm to 92% in Option 2a.

Effects on support energy

Support energy use is of importance not only because it represents a major cost to the farmer, but also because of its relevance to the overall sustainability and environmental impact of the system and to emissions at various points 'up-stream' from the farm. Support energy costs for forage production and conservation, based on White *et al.* (1983) varied between 433 and 1923 GJ (Table 7). All of the system modifications reduced the support energy required compared with the case study with the requirement per LU being more than halved in Options 2a, 2b and 5. All the reductions arose, in the main, from the reduced fertilizer N use, which represented 66% of the total support energy in the case study farm.

Table 7. Support energy required to produce herbage and conserved forage in dairy farming systems

Option	GJ ha^{-1} (whole-farm basis)			Total		
	Herbage production†	Herbage ensiling‡	Maize product./ensiling§	GJ ha^{-1}	GJ LU^{-1}	Farm GJ
Case study	20·7	4·6	–	25·3	11·7	1923
1	16·8	4·6	–	21·4	9·9	1626
2a	2·1	3·6	–	5·7	3·3	433
2b	2·1	3·6	–	5·7	3·3	433
3	15·5	2·3	3·9	21·7	10·0	1649
4	12·4	2·3	3·9	18·6	8·6	1414
5	1·7	1·8	3·2	6·7	3·9	509

†Based on 73 MJ kg^{-1} N, 7 MJ kg^{-1} P and 7 MJ kg^{-1} K (and an assumed average application rate of 78 kg P ha^{-1}, and 112 kg K ha^{-1}) and with N rates according to particular management option. Allowance of 1·2 GJ ha^{-1} for fuel and machinery and with extra 3 GJ ha^{-1} for slurry injection in 1 and 4 (based on White *et al.*, 1983). ‡For areas devoted to grass silage, inputs of 7·2 GJ ha^{-1} for fuel and machinery and 3·9 GJ ha^{-1} for storage; figures reduced by 20% for grass/clover to allow for lower yields (based on White *et al.*, 1983). §A total of 15·0 GJ ha^{-1} for maize production and ensiling without fertilizers (Wilkinson, 1981).

Conclusion

The possibilities for reduction in environmental impact of dairy farms are evident and the options considered here all have apparent potential for reduction in input and losses. As well as immediate effect on N balances, there were benefits for each option in reducing N_2O emissions and support energy inputs. The implications of each change for N are complex because of the nature of the grassland N cycle and have consequences for other phases of the system. Although it was clear that N derived from white clover was the best option as far as total losses were concerned, a lower stocking rate resulted in similar rates of loss per unit of production to those systems based on improved fertilizer management. Introduction of maize had important effects as demonstrated elsewhere (Weissbach and Ernst, 1994). However, it should be noted that there are still many poorly defined factors influencing N transfer where maize is concerned, especially long-term effects on mineralization rates and effective utilization of mineral N supplied from slurry. The availability of current models and information for component parts of the system is a necessary prerequisite for a systems synthesis approach. In order to provide confidence in the estimates, some sensitivity analysis is required but until data from experimental systems are available, this will be difficult to achieve. In the meantime, the present estimates provide a basis for detailed consideration of effects of these and further changes on N budgets. However, there will be practical difficulties associated with a greater reliance on fixed or excretal N because of uncertainties in rates of N supply from these sources. There will also be significant economic implications of the various management options described here. The effects of options such as these on economic returns from the dairy farm system are being examined in a further study.

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environmental effects of various systems. Environmental balances for different materials and systems are being evaluated.

Finally, the general concern of the public about the effects of soilless culture on environment, consumers' health and traditional agricultural production, should be anticipated by information on energy and nutrient input/output sheets, quality of products and environmental risks. Studies by Gysi & Reist (1990) reveal that, with respect to the above-mentioned criteria, intensive soilless production can to be considered more favourable than soil-based intensive production. If consumers are correctly informed, it may stimulate their appreciation of integrated horticultural products, and enable the widespread introduction of these environmentally safer production systems.

Conclusions

In order to establish a sustainable, safe and competitive glasshouse horticulture, production systems and management have to be improved. Clean production systems are needed not only because of governmental aims, but also because of changing consumer attitudes towards the product quality in relation to the production process. Only advanced integrated production systems with minimim inpunts and emission can fulfil the strict environmental aims. However, also the reluctance of consumers must be overcome to allow the horticultural industry a successful conversion from traditional soil-based systems to integrated (soilless) systems.

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Dairy farming systems based on efficient nutrient management

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Abstract

In Dutch dairy farming, dramatic nutrient losses occur, causing serious environmental problems and representing an economic and energy waste. So farming systems have to be developed based on efficient nutrient management. A dairy farm is characterized as a system with soils and crops, forage, cattle and manure as main components. Simple models of nutrient flows in and between components of the farming system were used to design a prototype system for a new experimental farm on sandy soil, that has to meet strict environmental demands. Experimental results of this farm will be used to improve the models and the models will be used again to optimize the prototype system. Initial results of modelling suggest that nutrient losses can be reduced considerably by more accurate management and introduction of rather cheap and simple measures. However, more radical and expensive modifications of the farming system are necessary to meet future standards of the Dutch government for maximum allowable emissions.

Keywords:
dairy farming, forage production, milk production, environmental demands, nutrient flow, nutrient utilization, nitrogen, phosphorus

Introduction

Dairy farming, occupying about 65% of the cultivated area and providing the main income to some 35% of its farmers, is the most important sector of Dutch agriculture. In the past, milk production systems were characterized by careful use of animal manure, limited milk production per hectare and integration of arable and dairy farming (Damen, 1978). Over the last decades, however, these systems were strongly intensified by increased inputs of anorganic fertilizers and purchased feeds. Introduction of the milk quota system in 1984 has resulted in a decrease of about 15% in milk production. At present, average annual milk production on dairy farms is about 11 500 kg ha^{-1}, but considerable differences exist between regions and individual farms, with systems on sandy soils being more intensive than on clay or peat soils.

Intensification has also led to a serious imbalance between inputs of nutrients in

Table 1. Average annual nitrogen, phosphorus and potassium balances of specialized dairy farms on sandy, peat and clay soils in the Netherlands, 1983-1986 (Aarts et al., 1988).

	Sand			Clay			Peat		
	N	P	K	N	P	K	N	P	K
Input (kg ha^{-1})									
- fertilizers	331	15	30	340	19	6	295	12	17
- concentrates	137	25	74	122	21	65	127	23	68
- purchased roughage	44	7	34	28	4	18	33	5	20
- atmospheric deposition	48	1	4	39	1	4	42	1	4
- miscellaneous	8	0	4	9	1	3	37	3	3
total	568	48	146	538	46	96	534	44	112
Output (kg ha^{-1})									
- milk	67	12	19	60	11	17	61	11	17
- sold livestock	14	4	1	11	3	1	11	3	1
- miscellaneous	1	0	0	1	0	0	0	0	1
total	82	16	20	72	14	18	72	14	19
Input - Output (kg ha^{-1})	486	32	126	466	32	78	462	30	93
Output/Input (%)	15	33	14	13	30	19	14	32	17

purchased fertilizers, concentrates, roughage and atmospheric deposition and outputs in milk and meat (Table 1). On Dutch dairy farms, output represents on average about 14% of the input for nitrogen (N), 32% for phosphorus (P) and 17% for potassium (K). The average annual surplus of 32 kg P ha^{-1} mainly accumulates in the soil, but continued accumulation will lead to saturation and leaching. The N surplus (about 470 kg ha^{-1}) contributes to environmental pollution by ammonia volatilization, runoff, leaching and denitrification. Dairy farming appears to be the major source of ammonia volatilization and associated acidification (Heij et al., 1991; Van Breemen et al., 1982). Denitrification would seem harmless to the environment, but in addition to N_2, N_2O is produced, a greenhouse gas that also affects the ozone layer (Bach, 1989), while denitrification in the subsoil may result in groundwater pollution by sulphates and heavy metals. Accumulation of N in soil organic matter is difficult to quantify, because of the large quantities present, but available evidence suggests that it accounts for a negligible part of the N surplus (Janssen, 1984).

Recently, the Dutch government has presented target values for emissions of N into the air and into groundwater and surface water, and for additions of P to the soil in the year 2000. For ammonia volatilization it is set at a reduction of 70% compared to the level in 1980. The nitrate concentration in the water at a depth of 2 m below groundwater level should be below 11.3 mg N l^{-1}. Total P application in anorganic fertilizer and manure should not exceed output in crop products. On P-saturated soils input will be even more restricted, to lower the P content of the soil. Therefore, societal demands on dairy farming systems with respect to N losses and P inputs are to become more stringent, requiring more efficient nutrient utilization. In addition, the health and well-being of men and animals, nature and landscape conservation and

the use of energy should be taken into account. Nevertheless, costs of milk production, including reasonable income, should not exceed financial returns.

The ultimate goal of farming systems research is to offer guidelines to farmers to design and develop their farms, taking into account their specific circumstances, such as soil type or farm size, and both short- and long-term objectives. This paper describes a research approach to generate the knowledge required for improved nutrient utilization in an economically optimum way. Integration of dairy farming and nature conservation is discussed elsewhere (Hermans & Vereijken, 1992).

Nutrient flows in dairy farming

Characteristic for dairy farming systems is the combination of plant and animal production. By exchanging manure and forage between the plant and animal components, nutrients cycle through the system, but nutrient losses also occur. In Figure 1,

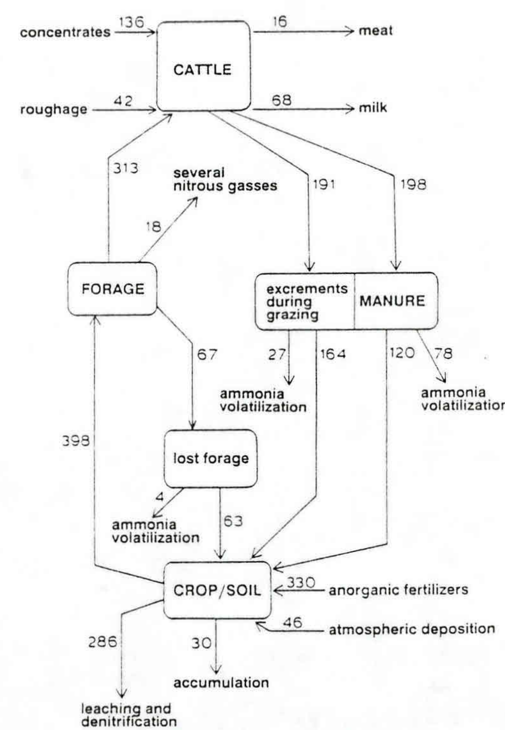

Fig. 1. Main N flows in an average dairy farming system on sandy soil, 1983-1986. Numerical values are kg N ha^{-1} yr^{-1}.

the main N flows on an 'average' farm on sandy soil are quantified. Only a minor part of the N consumed by cattle is converted into milk and meat: about 80% is excreted in urine and faeces. A quarter of this is lost through ammonia volatilization, especially from the excrements produced indoors. The excessive input of N fertilizer and the irregular distribution of excrements during grazing result in 40% loss of N input from the soil, by leaching of nitrate and denitrification (Van der Meer & Van Uum-Van Lohuyzen, 1986). A small part of the N taken up by the crop is lost by volatilization of ammonia and other nitrous gasses during growth and forage conservation. Moreover, some ammonia volatilizes from crop residues, left after grazing and harvesting.

Efficient nutrient management; constraints and perspectives

Single nutrient flows can be influenced by changing management. However, intervening in one step of the cycle may affect nutrient flows elsewhere, i.e. covering a slurry storage reduces direct ammonia emissions, but most of that ammonia will volatilise soon after slurry application, unless a low-emission technique is applied. Injection of slurry into the soil may reduce ammonia emissions considerably, but will lead to increased leaching of nitrate if the input of anorganic N fertilizers is not reduced. Therefore, in a strategy aiming at minimum losses, all the components of the system should be taken into account. Efficient nutrient management implies efficient utilization of nutrients in all stages of the cycle. Hence, conversion of nutrients from manure into forage and from feed into milk and meat should be maximized. Quantifying nutrient balances of the main components of the system can be useful to identify major losses and to find potential limiting or preventive measures.

In research ample attention has been paid to single measures to reduce losses, in the past mainly to economize on fertilizer and feed use, recently mainly to reduce environmental contamination. A considerable amount of information is, therefore, available on the effect of various measures on nutrient losses (e.g. Aarts et al., 1988; Korevaar & Den Boer, 1990). Unfortunately, the effects of most measures have not been tested in combination with other measures. Some measures, like avoiding application of slurry in autumn, or slurry injection, are almost always very effective. The costs and benefits of other measures, both in environmental and economic terms, may depend strongly on specific farm conditions. For instance, growing silage maize to balance the high N content in the summer diet of cows is difficult on peat soils. On these soils, however, a comparable result can be achieved by reducing the N fertilization level of the grass. Growing fodder-beets to partially substitute concentrates is most attractive on farms with a relatively low milk production level per hectare and consequently a surplus roughage. The costs of reducing N losses by restricted grazing strongly depend on farm infrastructure. Therefore, in principle for each group of farms with the same relevant characteristics, or even for each farm, a specific set of consistent measures to meet environmental and economic goals should be developed. Nevertheless, some aspects of optimizing nutrient utilization in the main farm components can be formulated in general terms.

Cattle

At maximum, 43% of the N ingested by lactating cows can be converted into milk and liveweight gain (Van Vuuren & Meijs, 1987), whereas the actual utilization is only 15-25%. The quantity of faeces N of grazing cows is rather constant (Kemp et al., 1979; Van Vuuren & Meijs, 1987). Hence the N surplus in the diet is excreted mainly in urine.

The quality of proteins in the diet also influences the utilization of N. Part of the feed protein is degraded in the rumen. The amino-acids and ammonia produced in the process will be used for microbial growth if enough energy is available, or the surplus will be excreted in urine. As heavily fertilized grass contains a substantial surplus of rumen degradable protein, reducing the protein content by restricted fertilizer application is an important option to improve utilization during grazing (Van Vuuren & Meijs, 1987). Including silage maize, with its low protein content, in the diet has a similar effect (Valk et al., 1990). Where that is not feasible, low protein/high energy concentrates are an alternative.

Utilization of dietary N can also be improved by higher milk production per lactation, because feed requirements per kilogram milk are lower. A higher lifetime milk production means lower replacement rates, requiring less calves to be reared, a rather inefficient form of protein utilization.

Animal manure

To restrict ammonia losses, transport time of excrements produced indoors to closed storage should be minimized by a slightly sloping floor with a central urine-drain and the use of a dung scraper. Preliminary research results suggest a reduction in ammonia volatilization of 50-60% compared to a conventional stable. Storage capacity for slurry, covering at least eight months of slurry production at the farm, should be available to avoid the necessity of slurry application during autumn and winter.

Under the common method of surface spreading of slurry, 25-35% of its N is lost through ammonia volatilization (Van der Meer et al., 1987). Injection of slurry into the soil will prevent ammonia losses almost completely. Also diluting slurry before spreading, or sprinkling during or shortly after application, may reduce losses considerably.

During grazing, faeces and urine are excreted in patches, leading to very high N concentrations locally: in urine patches about 500 and in dung patches about 2 000 kg ha^{-1} (Lantinga et al., 1987). As N in the dung is mainly in stable organic form, only some 13% is lost as ammonia. Nitrogen in urine mainly consists of urea, which easily dissociates in ammonia and carbon dioxide. However, most urine penetrates into the soil, restricting ammonia losses to about 13% (Van der Molen et al., 1989; Vertregt & Rutgers, 1988), while the remainder is transformed into nitrate. Such high amounts of mineral N cannot be taken up by a crop, so a considerable proportion may leach. Restricting grazing to daytime reduces the number of urine and dung patches. If maize is fed indoors, N content in urine will be reduced concurrently, so that, at a given level of fertilizer input, nitrate leaching will be lower (Korevaar &

Den Boer, 1990). A zero-grazing system would reduce nitrate leaching even more, but is detrimental to animal welfare and also labour-intensive.

Soil and crop

Increasing forage production by higher fertilizer application reduces the need for purchased feed, but increases the risk of nitrate leaching. The current Dutch fertilizer recommendation is based only on economic cost-benefit analyses, which at current prices results in a break-even point of about 7-8 kg DM kg^{-1} applied. For grassland on clay and sandy soils, the recommendation is 400 kg N ha^{-1} yr^{-1}, including effective N in applied manure (Prins et al., 1988); for silage maize it is 150 kg N ha^{-1} yr^{-1}. On many farms, N inputs are higher, partly because effective N in animal manure is underestimated. As a consequence, substantial quantities of N will leach to the groundwater.

Sub-optimal growing conditions are another cause of low nutrient utilization. Quite often the expected yield, on which fertilizer application was based, is not realised, because of management errors or water stress. Hence, improving growing conditions can reduce nutrient emissions. At a given level of N input, nitrate leaching on grassland is much higher under grazing than under mowing. Hence, fertilizer application should be based on the prevailing growing conditions and the mode of exploitation of the grass.

Maize growth stops rather early in autumn, compared with grass or fodder-beets. Moreover, nutrient uptake at the end of its growing season is rather inefficient. As a result, substantial amounts of mineral N may accumulate in the soil towards the end of the year. To reduce the risk of nitrate leaching in autumn and winter, a catch crop can be grown immediately after harvest (Scott et al., 1987).

If the land is suitable for both grassland and arable cropping, the ratio of the areas of the different crops may be optimized, taking into account both the desired quantities and qualities of forages, and the possibilities to apply animal manure. For example: on soils with limited moisture supply, feed energy production per unit area of maize is higher than for grass, so from that point of view, expanding the area of maize is attractive. However, nutrient uptake is lower, so on maize land less slurry is permitted. Therefore, expanding the maize area to increase feed energy production may lead to slurry surplus and increased demand for purchased proteins, which puts a limit on that expansion.

In current practice, a large proportion of the manure of the dairy farm is used on permanent maize land. Besides, part of the slurry of pig farms is used on these fields. As a result P accumulates in the soil, whereas on fields permanently cropped with grass, purchased P fertilizers have to be applied to maintain their P status. Therefore, it is essential that the manure is distributed according to crop nutrient requirements and P status of the soils.

Forage

The proportion of farm-produced forage that is utilized by cattle should be kept as high as possible to restrict nutrient losses from crop residues and the need for purchased feeds that cause extra inputs of nutrients. In practice, losses up to 20% of the potentially 'harvestable' grass production, and about 10% of maize and fodder-beets, are common. Losses can be reduced by restricting the daily grazing time and the total grazing period per field. Better timing and methods of harvesting also will reduce losses in grass and maize considerably and improve forage quality. Generally these measures will be economically viable, though they put high demands on the motivation and skills of the farmer.

Research approach

Whole-farm system research on interactions and overall effects of integrated measures is needed for a substantial improvement in nutrient utilization. However, that presents two major methodological problems. First replicates are hardly possible, because of specific farm conditions, such as soil type and skill of the farmer. Second, these specific conditions hamper extrapolation of results to other farms. Moreover, system research is expensive because a complete farm is needed and extensive observations have to be made. An attractive solution to these problems is to combine system modelling and system prototyping. Modelling is used to generate and evaluate prototype systems (Van Keulen & Wolf, 1986; Spedding, 1990; Spedding, 1988). Prototyping is needed to validate and improve the models. Therefore an important part of the experimental work aims at quantifying flows of nutrients, especially N. That includes measurement of animal protein intake, N content of manure and ammonia volatilization, both indoors and outdoors. Also leaching, mineralization and crop uptake of N have to be measured. If models have proven to be valid, they can be used as a tool for farming systems design and development in general, thus serving as a means of 'translating' research results to other conditions.

System modelling

Models represent simplified descriptions of reality. Accurate and reliable predictions require simple models, calibrated on rather extensive data collected in the environment where the models have to be used. Comprehensive models are not suitable for predictive purposes. These models contain many functions and parameters, each with its own uncertainty, accumulating in the simulated final result. Their usefulness is more in deriving the concepts for simple, causal relationships, needed in the simple models (Spitters, 1990). Therefore, simple models have been used to describe and integrate the main components of the dairy farming system. They were used to calculate the effects of N and moisture supply on dry matter yield and N content of crops, the influence of method of manure application on ammonia volatilization, and the effects of diet composition on milk production and excretion of nutrients in urine and faeces. Some of the relationships between management practices and nutrient flows have been reliably quantified, such as the effect of slurry injection on ammonia volatilization, others are more speculative, like the influence of grazing systems on yield losses, which appear to be strongly affected by the management skill of the farmer. Examining the range of responses to this type of measures allows analysis of the effect of farmer's skill on farming results.

System prototyping

The most promising prototype, according to the initial results of modelling, will be examined in practice by a joint effort of the Centre for Agriculture and Environment, the Centre for Agrobiological Research and the Research Station for Cattle, Sheep and Horse Husbandry. Recently, the Experimental Farm for Dairy Husbandry and Environment ('De Marke') came available for this purpose. The farm is located in the east of the Netherlands on a well-drained sandy soil. It comprises 55 ha, twice the size of an 'average' farm, but necessary to produce feed for the number of cows needed for a valid experiment. Some of the environmental restrictions imposed on the farm have been quantified explicitly in government regulations, others have been derived from more generally formulated targets and converted into quantitative criteria at farm level. A reduction in ammonia volatilization of at least 70%, compared to the average situation in 1980, implies an upper limit of 40 kg N ha^{-1} yr^{-1}. For a well-drained sandy soil and an annual precipitation surplus of 300 mm, nitrate leaching should not exceed 34 kg N ha^{-1} yr^{-1}, to comply with the standard of 11.3 mg N l^{-1}. Total annual N losses, including denitrification and runoff, should not exceed 128 kg ha^{-1}. Accumulation of P should be avoided, if the P status of the soil is within the agronomically satisfactory range. If it exceeds that, the P reserves should be reduced. For the experimental farm as a whole this implies an annual permitted surplus of 0.4 kg N ha^{-1}. Moreover, all manure produced should be applied on the farm. In addition, K leaching should be minimized, pesticide and energy use limited, well-being of man and animal taken into account, and financial output maximized.

Initial results of modelling

System design

With the 'current' system (I) as a reference, an 'improved' system (II) and a 'further improved' system (III) have been designed, based on constraints and perspectives of efficient nutrient management as described above. The latter aims at meeting the environmental standards of 'De Marke'. System I can be characterized as a dairy farm on well-drained sandy soil, comparable to the 'average' dairy farm in the mid-eighties (Table 2). The risk of nitrate leaching is high on this soil, but grass and other fodder crops can be grown and slurry injection is possible. Measures to reduce nutrient losses have been selected on the basis of cost-benefit analyses. System II is characterized by more accurate management, including concentrate supply according to the individual needs of the animals and more attention to grazing, harvesting and conservation losses. The measures introduced are relatively simple and cheap, being already practiced by innovative farmers. There is one exception, requiring substantial investments: the slurry storage has to be expanded and covered. System III is characterized by additional investments and measures to further improve the efficiency of nutrient management (Tables 2 and 3).

Table 2. Characteristics of a 'current' Dutch dairy farm on sandy soil, 1983-1986, and of the same farm after (further) improvement (data on annual basis).

	'Current' farm (system I)	'Improved' farm (system II)	'Further improved' farm (system III)
milk production (kg ha^{-1})	13 195	13 195	13 195
number of cows ha^{-1}	2.3	1.9	1.5
grass (ha)	22	16	16
maize (ha)	3	9	6
fodder-beets (ha)	0	0	3
purchased fertilizer N (kg ha^{-1})	330	159	91
purchased fertilizer P (kg ha^{-1})	15	2	3
purchased concentrates (kg ha^{-1})	4 995	2 348	1 778
purchased maize silage (kg DM ha^{-1})	2 135	0	0
part of feed energy produced on the farm (%)	57	83	84

Table 3. Measures to reduce nutrient losses on a 'current' Dutch dairy farm on sandy soil.

System components/measures	'Improved' farm (system II)	'Further improved' farm (system III)
Cattle component		
– milk production	• 7 000 kg cow^{-1} yr^{-1}	• 8 500 kg cow^{-1} yr^{-1}
– young stock	• 77% of dairy cows	• 57% of dairy cows
– grazing regime	• only by day	• only by day • restricted grazing season • rotational grazing from 4 to 2 day
– diet grazing season	• additional maize silage	• additional maize silage
– diet winter season	–	• fodder-beets and corn-cob mix to replace concentrates
Manure component		
– slurry storage	• expanded and covered	• expanded and covered
– method slurry application	• injection	• injection
– time slurry application	• spring and summer	• spring and summer
– stables	–	• improved floor • dung scraper
Soil/crop and forage components		
– fertilizer for grass	• 310 kg N ha^{-1} yr^{-1}	• 235 kg N ha^{-1} yr^{-1}
– fertilizer for maize	• 145 kg N ha^{-1} yr^{-1}	• 130 kg N ha^{-1} yr^{-1}
– catch crop	• after silage maize	• after silage maize
– land use	• more maize	• more maize and fodder-beets

In systems II and III, slurry is only applied in spring and summer by injection, soon to be compulsory in the Netherlands. The costs of this measure, in terms of prevented volatilization losses, are about NLG 3.5 kg^{-1} N, twice the costs of anorganic-fertilizer N. For economic reasons the stable has not been adapted in system II. In system III such adaptations have been assumed to minimize N losses, at a cost of about NLG 23 kg^{-1} N saved. To reduce N intake, and consequently the amount of N in excrements, grazing has been changed from continuous grazing to daytime grazing only, combined with feeding of maize silage, as is common practice already at about 50% of the dairy farms on sandy soils. In system III the grazing season ends on 1 October, one month earlier than in both other systems. As grazing is restricted, more maize silage is needed in systems II and III, so grassland is partly replaced by arable land. In system III, fodder-beets and maize (corn-cob mix) partly replace purchased concentrates. In system II, grass is fertilized annually with 310 kg N ha^{-1}, including effective N in animal manure, and maize with 145 kg N ha^{-1}. In system III, where annual leaching has to be restricted to 34 kg N ha^{-1}, annual N input on grassland is reduced to 235 kg ha^{-1}. Maize is fertilized at 130 N kg ha^{-1}, fodder-beets at 170 N kg ha^{-1}. In both systems, maize is followed by a catch crop to reduce nitrate leaching. Fertilization rates of crops following a catch crop or grassland, are reduced in accordance with the expected increased N mineralization. In system III, utilization of harvestable forage is improved by a two-day rotational grazing scheme. To reduce feed requirements, annual milk production per cow has been increased from 5 700 kg to 7 000 kg (system II) or 8 500 kg (system III). In 1989, average milk production per cow was already 6 700 kg, so this improvement has been realised partially already on most farms. Increasing milk production to the level of system III cannot be realised at once, but it can be achieved by continuous genetic improvement of the cattle. In system III, the number of calves is reduced to that needed for replacement of milking cows, so meat production is minimized.

System evaluation

As shown in Table 4, conversion of N intake to milk and meat will increase from 17% to 23% (system II) or 26% (system III). Higher efficiencies could be attained by further adaptations, such as zero-grazing, lower fertilizer rates on grass, higher milk production levels or off-farm rearing of calves. However, most of these adaptations will increase costs of milk production considerably.

As a result of improved manure management in the improved systems, a higher proportion of N from the excrements reaches the soil component (82% and 90%, compared to 73% in system I). This results, in combination with the lower N contents of excrements, in a sharp decline in ammonia losses from manure: from 101 kg N ha^{-1} to 50 kg N ha^{-1} and 21 kg N ha^{-1}, respectively. Similarly, reduced inputs and improved utilization of nutrients lead to a strong reduction in nutrient surpluses from the soil/crop component: from 325 kg N ha^{-1} to 113 kg N ha^{-1} and 77 kg N ha^{-1}, respectively; and from 29 kg P ha^{-1} to 0 kg P ha^{-1} for both systems. Lower fertilization rates on grass, and replacement of grass by fodder crops lead to lower average protein production per hectare but total dry matter production on the farm is hardly

Table 4. The model-calculated annual nutrient balances (kg ha^{-1} yr^{-1}) of a 'current' Dutch dairy farm on sandy soil, 1983-1986, and of the same farm after (further) improvement.

	'Current' farm (system I)		'Improved' farm (system II)		'Further improved' farm (system III)	
	N	P	N	P	N	P
Cattle component						
input:						
– purchased feed	178	31	63	12	58	10
– forage	313	42	293	42	234	35
output:						
– milk	68	12	68	12	68	12
– meat	16	5	12	4	8	2
output/input	17%	23%	23%	30%	26%	31%
Manure component						
input:						
– excrements during grazing	191	25	88	11	62	8
– excrements indoors	198	30	185	28	151	23
output:						
– excrements reaching soil	164	25	76	11	56	8
– manure applied	120	30	147	28	136	23
input - output	105	0	50	0	21	0
output/input	73%	100%	82%	100%	90%	100%
Soil/crop component						
input:						
– excrements reaching soil	164	25	76	11	56	9
– manure applied	120	30	147	28	136	23
– artificial fertilizer	330	15	159	2	91	3
– atmospheric deposition	46	1	45	1	45	1
– returning forage losses	63	6	24	3	19	3
output:						
– harvestable crop	398	48	338	45	270	38
input - output	325	29	113	0	77	0
output/input	55%	62%	75%	100%	77%	100%
Forage component						
input:						
– harvestable crop	398	48	338	45	270	38
output:						
– consumed forage	313	42	293	42	234	35
input - output	85	6	45	3	36	3
output/input	79%	87%	87%	93%	87%	93%
Whole farm						
input:						
– artificial fertilizer	330	15	159	2	91	3
– purchased concentrates	136	25	63	12	58	10
– purchased roughage	42	6	0	0	0	0
– deposition	46	1	45	1	45	1
– miscellaneous	7	0	4	0	4	0
output:						
– milk	68	12	68	12	68	12
– meat	16	5	12	4	8	2
input - output	477	31	191	0	122	0
output/input	15%	35%	30%	100%	39%	100%
volatilization: from manure	105		50		21	
nitrogen accumulation in soil	30		0		0	
leaching/denitrification	286		117		82	
other losses	56		24		19	

affected, because of the higher yields of fodder crops on soils with a limited water-supplying capacity. The adapted grazing system and reduced harvest losses lead to increased utilization of forage nutrients. At farm level, the annual surplus of nutrients already decreases sharply in system II: from 477 kg N ha^{-1} to 191 kg N ha^{-1}; and from

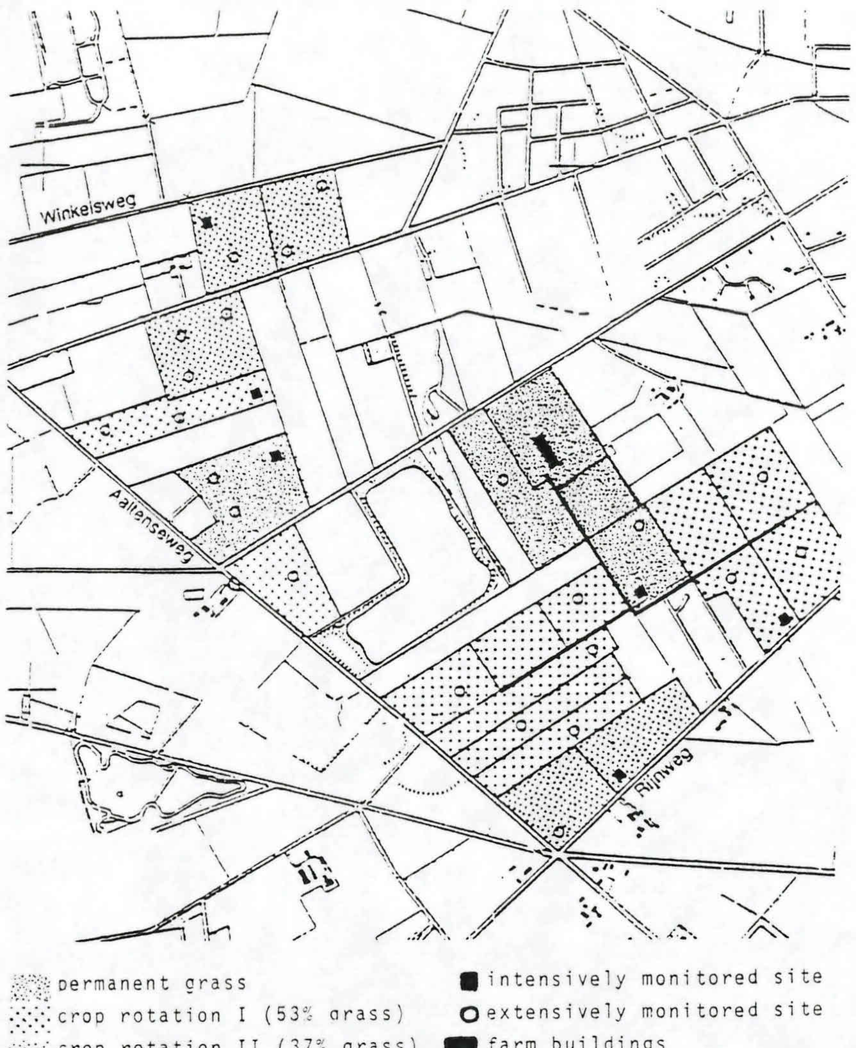

Fig. 2. Plan of 'De Marke'.

31 kg P ha^{-1} to 0 kg P ha^{-1}. In system III, N losses even decline to 122 kg ha^{-1}, just below the 128 kg ha^{-1} aimed for. Most of the N surplus is lost by leaching and denitrification. In system I, annual leaching is approximately 135 kg N ha^{-1}, and in the improved systems 44 kg N ha^{-1} and 28 kg N ha^{-1}, respectively.

Layout of the prototype system

Based on the initial results of model calculations, system III was selected as the prototype system to be examined on the experimental farm 'De Marke'. The farm area was divided into permanent grassland and two crop rotations, varying in the ratio of grass to arable crops (Figure 2). Most grassland is situated near the farm buildings, favourable from a farm management point of view, as the cows are milked indoors. Nevertheless, arable crops can be alternated with grass, for example to reduce soil-borne diseases. To recover nitrogen mineralized after ploughing the grassland, fodder-beets are grown first, because of their higher N uptake capacity compared to silage maize. Only permanent grassland and grass in rotation I will be irrigated if urgently needed for grazing.

Yields of arable crops are assessed per field. Grass yield is estimated by measuring silage yield and calculating intake of the grazing herd. More detailed information is collected by monitoring the soil-crop system at 28 permanent sites, six of which are monitored intensively. On permanent grassland and on each of the rotations two 'intensive' sites are situated, one on a field with a relatively shallow groundwater level, influencing soil water supply to the crop, and one on a field with a deep groundwater level. On the 22 less intensively monitored sites, soil mineral N is measured (in spring and autumn), crop phenological development is estimated (weekly), crop yield and nutrient contents are quantified (at harvest) and soil fertility is analysed (every three years). At the intensively monitored sites information is collected more frequently and additional observations include: soil-water content, soil-water tension, groundwater depth, nitrate concentration in drain water and soil temperature. Meteorological data, including rainfall, radiation, relative humidity, wind speed and temperature are recorded near the farm buildings.

Plot experiments, such as varying N fertilization levels, are only acceptable insofar their influence on performance of the prototype system is acceptable. Within that restriction, detailed experiments can provide important information about, for example, N leaching as a function of the fertilization level.

Perspectives

The calculated results for the 'improved' system (II) illustrate that nutrient losses may be reduced considerably by more accurate management and the introduction of an integrated set of cheap and profitable measures into existing farming systems. As the improved system does not fully meet the environmental goals with respect to N, further steps are necessary. However, the performance of system III suggests that environmental standards can be attained, even at a relatively high milk production level per hectare.

From agricultural practice, results have been reported from farms reducing the N surplus within a couple of years by 25-50% (Korevaar, 1992), mainly by better utilization of roughage, hence reduced concentrate input and reduced fertilizer input, partly compensated by better utilization of animal manure.

First steps in farm development towards integrated farming systems seem at least cost-neutral and may even be profitable because of lower inputs. Subsequent steps need more skill and are more expensive. However, preliminary calculations on the financial results of system III suggest an acceptable income, provided milk prices remain stable. Therefore, most dairy farms characterized by average or below-average milk production per hectare have rather favourable prospects. However, farms that are very dependent on purchased feed either have to reduce milk production per hectare or export animal manure. Under present Dutch conditions of substantial manure surpluses, reducing milk production seems the least expensive.

The experimental farm 'De Marke' aims at development of advanced prototypes of integrated farming systems. Subsequently these prototypes will be introduced on commercial farms for testing and further improvement. Introduction of the systems on a large scale requires the support of extension and education, and a stimulating environmental policy. Registration of inputs and outputs of nutrients at farm level was started in the Netherlands in 1990 on a voluntary basis, as a first step. In addition an adequate system of levies and premiums should be introduced to accelerate transition from the current farming systems to integrated systems with due attention to different goals.

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Environmentally Sensitive Areas in the UK and their Grassland Resource

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ABSTRACT

There are 43 Environmentally Sensitive Areas (ESAs) in the United Kingdom: 22 in England, 6 in Wales, 10 in Scotland and 5 in Northern Ireland. Farmers within these designated areas are able to enter into voluntary, 10-year management agreements for which they receive annual payments for farming in a way which seeks to maintain and enhance the landscape, wildlife and historic or archaeological interest of the area. The combined area of the ESAs in the UK is 3,356,000 ha, covering around 15% of the agricultural land area. ESAs encompass the range of habitats which are influenced by agricultural operations, although grassland has been particularly targeted, there being 1,058,733 ha in total within the ESAs, 353,999 ha (33%) of which was under agreement at the end of 1996. A range of grassland types occurs in ESAs, including a significant proportion of the hay meadow, chalk downland and wet grassland communities which are important for their nature conservation interest. This paper provides an overview of the ESA Scheme, with particular reference to the grassland resource, and summarises the associated monitoring and research and development programmes which have been established in support of the scheme.

THE ENVIRONMENTALLY SENSITIVE AREA SCHEME

Background

The Environmentally Sensitive Area (ESA) Scheme was introduced in the UK in 1987 following the passage of the enabling legislation (Section 18 of the Agriculture Act 1986). The Broads Grazing Marshes Conservation Scheme had previously been run as a pilot in 1985-6 (MAFF, 1989) and similar voluntary management agreements were operated by the Exmoor National Park in the early 1980s. The aim of the ESA Scheme is to maintain the landscape, wildlife and historic value of defined areas by encouraging beneficial agricultural practices. The scheme is voluntary and farmers and other land managers with responsibility for agricultural land within the designated areas are able to enter into 10-year management agreements and to receive annual payments in return for adhering to specified management prescriptions.

Since 1987 the ESA Scheme has been expanded and there are now 43 in the UK covering some 15% of the agricultural land area. Of these, 22 are in England, 6 are in Wales, 10 in Scotland and 5 in Northern Ireland. A list of the ESAs in the UK is given in Table 1, with details for each of the date of designation, the area which is eligible to be entered into the scheme, the approximate area of grassland (all types) and the area of

Table 1. Environmentally Sensitive Areas in the UK (position as at end 1996)

ESA	Date of designation	Eligible area (ha)	Permanent grassland (ha)	Grassland un agreement (h
England				
Avon Valley	1992	3,800	3,300	968
Blackdown Hills	1993	32,014	29,389	3,900
Breckland	1988	51,600	3,500	2,639
Broads	1987	24,000	17,334	15,352
Clun	1988	18,900	15,000	4,201
Cotswold Hills	1993	66,100	28,000	14,087
Dartmoor	1993	84,514	34,600	10,171
Essex Coast	1993	23,000	12,700	2,486
Exmoor	1992	68,362	35,440	24,008
Lake District	1992	219,300	162,800	87,865
North Kent Marshes	1992	11,600	7,300	3,400
North Peak	1988	50,500	3,900	2,876
Pennine Dales	1987	39,100	39,132	25,471
Shropshire Hills	1993	34,900	34,400	7,760
Somerset Levels & Moors	1987	25,900	23,722	15,032
South Downs	1987	51,700	14,865	5,584
South Wessex Downs	1992	38,737	10,771	4,784
South West Peak	1992	27,000	23,500	14,031
Suffolk River Valleys	1988	32,600	13,100	8,917
Test Valley	1988	3,300	2,400	1,178
Upper Thames Tributaries	1993	23,194	10,686	4,081
West Penwith	1987	6,914	3,752	3,392
	Total:	937,035	529,591	262,183
Wales				
Anglesey	1993	60,000	45,000	3,346
Cambrian Mountains	1987	119,600	83,720	24,078
Clwydian Range	1994	26,000	19,500	937
Lleyn Peninsula	1988	39,700	27,790	2,205
Preseli	1994	104,000	78,000	1,325
Radnor	1993	80,000	60,000	7,812
	Total:	429,300	314,010	39,703
Scotland				
Argyll Islands	1994	177,400	14,175	1,723
Breadalbane	1987	140,000	10,926	948
Cairngorms Straths	1993	148,700	15,686	297
Central Borders	1988	28,800	11,339	89
Central Southern Uplands	1993	220,300	36,732	0
Loch Lomond	1987	36,900	4,017	130
Machair of the Uists, etc.	1988	12,000	4,025	641
Shetland Islands	1994	144,900	15,477	910
Stewartry	1988	48,500	22,220	743
Western Southern Uplands	1993	131,000	29,422	34
	Total:	1,088,500	164,019	5,515
Northern Ireland				
Antrim Coast	1993	21,840	12,977	12,495
Fermanagh	1993	28,280	17,953	15,779
Mourn Mountains	1993	14,559	3,029	2,692
Slieve Gullion	1994	2,462	1,119	882
Sperrins	1994	46,928	16,035	14,750
	Total:	114,069	51,113	46,598

grassland that had come into agreement by the end of 1996.

The criteria for ESA designation used in the latest round of English designations are that the area must be of national environmental significance and represent a discrete and coherent unit of environmental interest. In addition, its conservation interest must be dependent upon the adoption, maintenance or extension of particular farming practices which have either changed or are likely to change, or which could, if modified, result in a significant improvement in that interest (Harrison, *in press*).

Environmental Aim and Objectives

The overall environmental aim of the ESA Scheme is to maintain the landscape, wildlife and historic value of the designated areas by encouraging beneficial agricultural practices. This is achieved through specific objectives which, although broadly similar, are tailored to each ESA and focus on the priorities of the area concerned. The ESA Scheme is designed and run so as to integrate the landscape, wildlife and historic interests of each area and the specific objectives have been drawn up with this in mind. Details of the objectives for each ESA are given in MAFF (1994) but they typically make reference to:

- maintaining and enhancing the wildlife conservation value of each of the habitats for which the area is particularly noted;
- maintaining and enhancing landscape quality; and
- maintaining and enhancing the archaeological and historic resource.

Performance Indicators

'Performance indicators' have been defined for each objective in each ESA and specify the targets which should be achieved during the five year period following the launch of the ESA (or re-launch after each quinquennial review). They provide a means of measuring the success of the scheme and a framework for its management and evaluation through an associated monitoring programme. Performance indicators cover uptake and environmental impact and include a combination of:

- overall uptake targets (usually in the form of a percentage of a type of eligible land that should be under agreement);
- targets that relate only to agreement land (*e.g.* the percentage of a certain type of feature which should be renovated); and
- environmental impact indicators which relate to the desired result of the imposition of ESA management agreements on various types of land (*e.g.* the maintenance or enhancement of botanical diversity or bird populations).

Full details of the performance indicators for each ESA are presented in MAFF (1994). These, and the associated objectives, are being reassessed as part of the current round of quinquennial reviews.

Management Tiers and Prescriptions

Under the current rules of the ESA Scheme, farmers and land managers with land in the designated areas are able to enter into ten-year management agreements (although there is an option for termination after 5 years) for which they receive an annual payment.

Each ESA has one or more tiers of entry in which the agricultural practices, or prescriptions, which have to be followed are specified. The nature of each tier depends upon the particular circumstances of each ESA, but may include, for example:

- the protection and management of existing habitats and features;
- the traditional management of hay meadows;
- the maintenance of high water levels in wetland habitats; and
- the reversion of arable land to grassland.

Although the precise details of the management prescriptions within each tier vary, they usually place a restriction on the use of certain cultivation techniques, stocking rates, the use of fertilisers, fungicides and insecticides and the introduction of new drainage infrastructure. They also usually include a requirement to maintain watercourses and boundary features. Some tiers impose more stringent management regimes than others and may, for example, require specified water levels to be maintained or place restrictions on the level and timing of grazing and/or the application of fertilisers. The payment rates which apply to each tier reflect the income foregone and, if appropriate, the need to provide an incentive.

THE GRASSLAND RESOURCE IN ESAs

The habitats specifically targeted by each of the ESAs in England and Wales are summarised in Table 2. Grassland is of particular importance in the context of the scheme and is the only habitat targeted by all ESAs. The total amount of grassland (all types) in each ESA is shown in Table 1; the overall extent of the resource in the ESAs in the UK being approximately 1,058,733ha, some 353,999ha of which (33%) has come into agreement.

A wide range of grassland types is represented in ESAs and many are particularly important for nature conservation. The significance of the lowland grassland resource in England has been assessed for each of English Nature's Natural Areas (Jefferson, 1996). This involved scoring each Natural Area on a five point scale from 'outstanding' to 'negligible', based on the estimated extent of lowland semi-natural grassland. Since the boundaries of most ESAs correspond quite closely with Natural Areas, a subjective assessment of the relative importance of the grassland resource in each ESA can be obtained. Table 3 shows the assessments for Natural Areas which include or coincide with ESAs. This indicates that the grasslands in seven ESAs are classed as 'outstanding', with the remainder being at least of 'some' significance. In England as a whole the grassland resource is classified as 'outstanding' in 12 ESAs.

There are a large number of neutral, calcareous and acid grassland and mire plant communities of high botanical interest represented in English and Welsh ESAs. The presence of significant National Vegetation Classification (NVC) communities (Rodwell, 1991 *et seq.*) is summarised in Tables 4 and 5. These include important concentrations of the English resource of wet grassland associated with grazing marshes and river floodplains; calcareous grassland, in particular chalk downland; and hay meadows. For example, there is an estimated 53,000ha of wet grassland in English ESAs (Glaves, *in press*). This represents around 27% of the total English resource, estimated as 200,000ha

(Dargie, 1993; DoE, 1995). The South Downs and South Wessex Downs ESAs include around 9,200ha of calcareous grassland on chalk (ADAS, 1996; MAFF, *in prep*. j) which represents around 29% of the English resource of 32,000ha (Jefferson & Robertson, 1996). In addition, important concentrations of calcareous grassland on limestone occur in the Cotswold Hills, Lake District and Pennine Dales ESAs.

Table 2. Habitats targeted in English and Welsh ESAs.

ESA	Grass -land	Wet grass -land	Wet-land	Arable (rever -sion)	Arable field m'gins	Low-land heath	Moor -land	Water -cour ses	Stone walls	Hedge -rows	Wood -land
England											
Avon Valley	✓	✓		✓							
Blackdown Hills	✓					✓				✓	
Breckland	✓			✓	✓	✓					
Broads	✓	✓		✓	✓			✓			
Clun	✓			✓	✓					✓	
Cotswold Hills	✓			✓					✓	✓	
Dartmoor	✓					✓	✓		✓	✓	
Essex Coast	✓	✓		✓							
Exmoor	✓					✓	✓			✓	
Lake District	✓		✓				✓		✓		
North Kent Marshes	✓	✓		✓							
North Peak	✓						✓		✓		
Pennine Dales	✓								✓		✓
Shropshire Hills	✓						✓			✓	
Somerset Levels & Moors	✓	✓						✓			
South Downs	✓			✓	✓						
South Wessex Downs	✓			✓	✓						
South West Peak	✓						✓		✓		
Suffolk River Valleys	✓	✓		✓							
Test Valley	✓			✓							
Upper Thames Tributaries	✓	✓		✓						✓	
West Penwith	✓					✓			✓		
Wales											
Anglesey	✓		✓	✓	✓	(✓)			✓	✓	✓
Cambrian Mountains	✓		✓	✓			✓		✓	✓	✓
Clwydian Range	✓		✓	✓	✓		✓		✓	✓	✓
Lleyn Peninsula	✓		✓	✓	✓	(✓)			✓	✓	✓
Preseli	✓		✓	✓	✓	(✓)			✓	✓	✓
Radnor	✓		✓	✓	✓		✓		✓	✓	✓

(✓) includes mosaics of maritime heath, maritime grassland and dune communities

Table 3. Lowland grassland significance assessments for English Nature Natural Areas which include English ESAs

ESA	Principal Natural Area	Significance assessment[1]
Blackdown Hills	Blackdowns	Some
Breckland	Breckland	Outstanding
Broads	Broadland	Significant
Clun	Central Marches	Some
Cotswold Hills	Greater Cotswolds	Outstanding
Dartmoor	Dartmoor	Considerable
Essex Coast	Thames Marshes	Some
Exmoor	Exmoor and the Quantocks	Some
Lake District	Cumbrian Fells and Dales	Outstanding
North Kent Marshes	Thames Marshes	Some
North Peak	Dark Peak	Some
Pennine Dales	North Pennines/Yorkshire Dales	Outstanding
Shropshire Hills	Shropshire Hills	Notable
Somerset Levels & Moors	Somerset Levels and Moors	Outstanding
South Downs	South Downs	Considerable
South Wessex Downs	South Wessex Downs	Outstanding
South West Peak	South West Peak	Some
Suffolk River Valleys	Suffolk Coast and Heaths	Notable
Upper Thames Tributaries	Oxford Clay Vales	Considerable
West Penwith	Cornish Killas and Granite	Some

Note: The small Avon Valley and Test Valley ESAs form or cross the boundaries of a number of Natural Areas and are thus not included in the table.
[1] after Jefferson (1996).

MONITORING AND EVALUATION

An extensive programme of environmental monitoring has been established in order to assess the degree to which the performance indicators are being met and thus the achievement of the objectives of the scheme. An evaluation of the monitoring results is undertaken to inform the review of each ESA which takes place every five years. An analysis of the results of the monitoring exercise for the initial tranche of five ESAs which were launched in England in 1987 was completed in 1996 and is presented in MAFF (1996a–e). Grassland monitoring has also taken place in second and third tranche ESAs (MAFF, *in prep.* a–k). Critchley (*in press*) provides an overview of the botanical monitoring methods which have been used in ESA monitoring.

Although it is only possible within the scope of this paper to provide a broad overview of the results from the monitoring programme, the overall indication is that the wildlife conservation interest of the areas concerned is being maintained. This compares favourably with the situation prior to designation where the interest of many of the areas was deteriorating. The wildlife conservation value of hay meadows in the Pennine Dales, for example, appears to have benefitted from the ESA Scheme; the species diversity of unimproved meadows having been maintained, whereas a general decline was noted during the period from 1978 to 1990 (DoE, 1993). Another indication that the scheme is bringing positive benefits comes from the Somerset Levels and Moors where there is evidence of an increase in bird numbers. Other examples are described in the detailed monitoring reports (MAFF, 1996a–e; 1997a–k).

Table 4. Neutral grassland and fen-meadow or rush-pasture National Vegetation Classification communities (excluding moorland) of high botanical interest in ESAs in England and Wales (community types are described in Appendix 1)

ESA	Neutral grasslands							Fen-meadows and rush-pastures					
Community:	MG 2	MG 3	MG 4	MG 5	MG 8	MG 11	MG 13	M 22	M 23	M 24	M 25	M 26	M 27
England													
Avon Valley					*		*		*	*	*		
Blackdown Hills				*				*	**	**	**		*
Breckland				**									**
Broads					*	*	*	**	**	**			*
Clun				*					*		*		*
Cotswold Hills				**									
Dartmoor				*					**	**	**		*
Essex Coast						*							
Exmoor				*		*			**	**	**		*
Lake District		**		**	*				*		*	*	
North Kent Marshes						*							
North Peak				*					*		*		
Pennine Dales	**	**		**	**				*		*	*	
Shropshire Hills				**									
Somerset Levels & Moors			?	*	**	**	**	**	**	**	**		**
South Downs				**		*							
South Wessex Downs				**	*								
South West Peak				*					*		*		
Suffolk River Valleys				*		*	*	**	**		**		
Test Valley					*			*	*	*	*		*
Upper Thames Tributaries			**	*			**	*					
West Penwith				*					*		*		
Wales													
Anglesey				*				*	*		*		
Cambrian Mountains		?		*					*				
Clwydian Range					*								
Lleyn Peninsula					*			*	*		*		
Preseli				*		*			*		*		
Radnor		?		*	*			*	*		*		

Notes: Where the resource is important in terms of quality and/or extent this is indicated as **

Other NVC mestorophic grassland communities of lower botanical interest, in particular MG1, MG6, MG7, MG9 and MG10 are also widespread in ESAs

? indicates that similar communities have been described, but these are not typical or have yet to be confirmed

The monitoring programme has also shown that the scheme has been successful in maintaining the landscape quality of the five first tranche ESAs. Indeed, there has been a significant enhancement of the landscape quality of two of these ESAs where the reversion of arable land to grassland has led to major changes in the landscape. The objective of maintaining the archaeological and historical resource within ESAs seems largely to have been met, principally through affording increased protection to such features by encouraging the reversion of arable land to grassland (Harrison, *in press*).

Table 5. Calcareous, acidic and miscellaneous National Vegetation Classification grassland communities (excluding moorland) of high botanical interest in ESAs in England and Wales (community types are described in Appendix 1)

ESA	Calcareous grasslands										Acid grasslands				Misc.	
Community:	CG 1	CG 2	CG 3	CG 4	CG 5	CG 6	CG 7	CG 9	CG 10	OV 37	U 1	U 2	U 3	U 4	SD[1]	MC[2]
England																
Blackdown Hills		*		*							*	*	*	*		
Breckland		**	**			**	**				**	**			**	
Broads											*					
Cotswold Hills		**	**	**	**											
Dartmoor												*	*	**		
Exmoor												*	*	**		**
Lake District		**						**	**					*		
North Peak														*		
Pennine Dales		**						**	*	**		*		*		
South Downs	**	**	**	**	**						*			*		**
South Wessex Downs		**	**	**	**	**	**									
South West Peak														*		
Suffolk River Valleys											**					
West Penwith				*									*	*		**
Wales																
Anglesey	*	**												*	**	**
Cambrian Mountains											*			*		
Clwydian Range	**	**				**			**					*		*
Lleyn Peninsula											*			*		*
Preseli											*			*		*
Radnor														*		

Notes: Where the resource is important in terms of quality and/or extent this is indicated as **
Other NVC acidic grassland communities of lower botanical interest, in particular U20 (and other bracken communities) are also widespread in ESAs
[1] various sand dune communities
[2] various maritime grassland communities

RESEARCH AND DEVELOPMENT

An extensive programme of research and development has been and continues to be carried out in connection with MAFF's agri-environment programme. A summary of the research projects relevant to ESAs which had been completed or were ongoing in 1995 is presented in MAFF (1995). Many other projects have been initiated since then and MAFF's current funding commitment amounts to around £800,000 year. A number of the research projects funded by MAFF are the subject of papers at this conference.

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MAFF (*In prep.* c) *Environmental Monitoring in the North Peak ESA 1988–1996.*
MAFF (*In prep.* d) *Environmental Monitoring in the Suffolk River Valleys ESA 1988–1996.*
MAFF (*In prep.* e) *Environmental Monitoring in the Test Valley ESA 1988–1996.*
MAFF (*In prep.* f) *Environmental monitoring in the Avon Valley ESA 1993–1996.*
MAFF (*In prep.* g) *Environmental monitoring in the Exmoor ESA 1993–1996.*

MAFF (*In prep.* h) *Environmental monitoring in the Lake District ESA 1993–1996.*
MAFF (*In prep.* i) *Environmental monitoring in the North Kent Marshes ESA 1993–1996.*
MAFF (*In prep.* j) *Environmental monitoring in the South Wessex Downs ESA 1993–1996.*
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APPENDIX 1

National Vegetation Classification grassland communities of high botanical interest in ESAs in England and Wales.

MG2	Meadowsweet–false oat-grass 'northern tall-herb grassland'
MG3	Sweet vernal-grass–wood crane's-bill 'northern hay-meadow'
MG4	Meadow foxtail–great burnet 'flood meadow'
MG5	Crested dog's-tail–common knapweed 'lowland hay-meadow/pasture'
MG8	Crested dog's-tail–marsh-marigold 'flood pasture'
MG11	Red fescue–creeping bent–silverweed 'inundation grassland'
MG13	Creeping bent–marsh foxtail 'inundation grassland'
CG1	Sheep's fescue–carline thistle 'warm temperate limestone grassland'
CG2	Sheep's fescue–meadow oat-grass 'lowland species-rich calcareous grassland'
CG3	Upright brome 'lowland calcareous grassland'
CG4	Tor-grass 'lowland calcareous grassland'
CG5	Upright brome–tor-grass 'lowland calcareous grassland'
CG6	Downy oat-grass 'lowland limestone grassland'
CG7	Sheep's fescue–mouse-ear hawkweed–wild thyme 'lowland calcareous grassland'
CG9	Blue moor-grass–small scabious 'northern sub-montane/montane carboniferous limestone grassland'
CG10	Sheep's fescue–common bent–wild thyme 'northern sub-montane calcareous grassland'
OV37	Sheep's fescue–spring sandwort 'metalliferous (calaminarian) grassland'
U1	Sheep's fescue–common bent–sheep's sorrel 'lowland acid grassland'
U2	Wavy hair-grass 'acid grassland'
U3	Bristle bent 'south-western acid grassland'
U4	Sheep's fescue–common bent–heath bedstraw 'sub-montane acid grassland'
M22	Blunt-flowered rush–marsh thistle 'lowland fen-meadow'
M23	Soft/sharp-flowered rush–marsh bedstraw 'western lowland rush-pasture'
M24	Purple moor-grass–meadow thistle 'lowland fen-meadow'
M25	Purple moor-grass–tormentil 'western lowland mire'
M26	Purple moor-grass–marsh hawk's-beard 'northern sub-montane mire'
M27	Meadowsweet–wild angelica 'lowland mire'
SD	Various sand dune communities
MC	Various maritime grassland communities


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Nitrogen cycling and losses from dairy farms

S. C. Jarvis

Abstract. The concern over leakage of nitrate into waters and loss of other forms of N to the environment demands an appraisal of N flows within complete systems. The grassland N cycle is complex, with interactive controls over fluxes and transformations, and has the potential for considerable losses. Although there are data from experimental systems, a total comprehension of flows is not yet possible.

Intensive dairy farming has a number of opportunities for leaks. A 'model' system in SW England has an annual input of 25.6 tonnes of N: of this only 20% is transferred into protein or milk, a further 46% is lost to the wider environment, 34% is as yet unaccounted for and much is recycled. Recent research has provided new techniques to decrease losses. To meet the joint requirements of production and environmental concerns we need to consider N flows and supplies on an integrated, whole farm basis, and to take better account of mineral N in the soil profile in relation to current crop demand, local climate and past sward management.

INTRODUCTION

PERCEPTIONS of the way in which nitrogen (N) is utilized and of the amounts that are recycled within grassland farming systems have changed enormously in recent years. All nutrient cycles are complex, and that of grassland N is particularly so because it involves not only soils and plants (and their constituent components) but also the ruminant animal. Unlike arable cropping, grassland agriculture also involves perennial crops, which may include legume species capable of fixing atmospheric N_2, and is targeted at the efficient husbandry of both plants and animals.

Nutrients such as N cycle between many of the various components of grassland systems at rates which are as yet poorly defined, especially where they are influenced by soil characteristics and conditions. The controls over the fluxes and transformations of grassland soil N are complex and highly interactive; mathematical (Thornley & Verberne, 1990; Scholefield *et al.*, 1991) and conceptual (t'Mannetje & Jarvis, 1990) models describe flows of N within grassland systems, but there are many missing details. However, there is a growing awareness that within animal production systems there is a very inefficient transfer of N into the final products (meat, milk or wool). Consequently, there is potential for harmful leakages of excess, mobile N into the wider environment, either as nitrate into aquifers or as gases into the atmosphere. Leakage of other nutrients can also occur but it is particularly important with N, which can change into forms that readily 'escape' from agricultural control.

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Nitrogen fertilizers have been an important component of grassland production in recent years. Responses to very high levels of addition, whether gauged by plant or animal production, have been well documented (Baker, 1986), and as a result large amounts of N are used in intensive lowland grass production. Recently there has been a levelling off in application rates in the UK, but over 11% of intensively managed systems receive more than 300 kg N/ha (Dampney, 1993). For intensive dairying, the recommended maximum rates are 300–380 kg N/ha for grazing and 340–420 kg N/ha for cut swards (ADAS, 1990). Recommendations for grassland on non-peat soils in The Netherlands also range up to 400 kg N/ha (Oenema *et al.*, 1992). Recent studies have shown that where inputs are high there are often opportunities for the equivalent of much of the added N to be lost by leaching (Macduff *et al.*, 1989; Tyson, 1992), denitrification (Jarvis *et al.*, 1991) or ammonia volatilization (Jarvis, 1991) from various components of the system. Such losses are no longer acceptable and current legislative, economic and public pressures demand that N use is optimized and that measures are defined to allow both environmental and economic goals to be attained. This will require a full appreciation of N movements in grassland systems. This paper provides a description of N flows within a dairy farm using recent experimental data, model calculations and best estimates.

NITROGEN CYCLING WITHIN A DAIRY FARM

In order to sustain the production requirements of intensive dairy systems, considerable flows of N into, within and from the farm are required. This paper examines these in a 'model' enterprise which encompasses the main features of a hypothetical, but typical, dairy farm in SW England. The characteristics of the farm (Table 1) are taken directly or calculated from results achieved by grassland farmers participating in the ADAS 'Milk Cheque' consultancy scheme (Whipps, 1991). The scheme covers 1447 herds of all breeds; because of their participation these are likely to represent the better-managed enterprises in the UK. Discussion with ADAS consultants provided further information for Table 1.

The model dairy farm is a holding of 76 ha and has soils ranging from moderately well-drained loams to poorly drained clay loams. The milking herd consists of 102 cows (Whipps, 1991), with 70 replacement heifers and 40 young animals for beef production (ADAS, pers. comm.), equivalent to a total of 165 livestock units (LUs). The milk production of 5554 litres per cow is sustained by silage production (at 9 t fresh weight per LU), total bought-in feeds equivalent to 1459 kg per cow and by grazed grass during a grazing period of 185 days. Other materials brought into the farm which ultimately contribute to the mobile N pool include bedding and fertilizers.

Annual inputs to the farm

Production responses in dairy systems have been very much stimulated by N fertilizer, supplied not because there are shortfalls of N as far as the animal is concerned, but to promote sufficient carbon fixation and plant growth (dry matter) for high rates of animal production. For the farms within the Milk Cheque scheme the estimated average fertilizer N input was 250 kg/ha (ADAS, pers. comm.); this is a lower rate of application than currently recommended (ADAS, 1990), but is substantially more than the average rate for all lowland grass systems. It is assumed in the present calculations that fertilizer is applied uniformly across grazed and ensiled areas to give a farm total of 19 t N (Table 2). Inputs from the atmosphere are probably less than the 35–40 kg/ha quoted for SE England (Goulding, 1990), but are still significant because of the high density of livestock and the likely patterns of ammonia volatilization and deposition; they are assumed to be 25 kg/ha.

Although clover-based pastures do not feature in this farm management system, some white clover may exist within the sward, especially in the older pastures. At an average overall clover content of 1–5%, Cowling (1982) estimated that approximately 10 kg N/ha would be symbiotically fixed. This additional N, plus that from atmospheric deposition, represents a further input to the farm of 2.66 t. Even if the farm relied much more on fixation of atmospheric N, the cycling, utilization and loss of N would not differ significantly from a farm based entirely on fertilizer N, provided total inputs of N did not change (Jarvis, 1992). The other major sources of imported N are concentrates and other feeds, and the straw provided as bedding for the cow cubicles and for the other animals: together these total 3.93 t N. In total, the annual input of N to the model dairy farm is 25.59 t N (Table 2).

Recycled N

The annual flows of N through the different components of the dairy farm are considerable. As well as the direct supply of N from new inputs (fertilizer, feed, etc.), plant available mineral N is also supplied from mineralization of soil organic matter. In the model farm there are 25 ha of old (10–25 yr) pasture and 51 ha of recently (4–6 yr) reseeded swards. If we assume that the organic N content and bulk density of the soils are similar to those quoted for experimental pastures in SW England by Tyson (1992), the top 10 cm would contain 5015 and 4300 kg N/ha

Table 1. Characteristics of a model dairy farm enterprise in SW England

Land area†:	76 ha			
Soil type:	loam→clay loam: moderate→poorly drained			
			Numbers	*LU*
Stocking:	milking cows†		102	102
	replacement heifers‡	(yr 1)	35	26
		(yr 2)	35	14
	beef calves‡	(yr 1)	20	15
		(yr 2)	20	8
		Total	212	165
Milk production†:	5554 l/cow Total 566,508 l			
				tonnes DM
Silage production:	9 tonnes FW/LU‡ (26.5% DM†)			393.5
Bought-in feeds†:				
concentrates	1390 kg/cow (@ 87% DM)			123.3
others	69 kg/cow§ (@ 87% DM)			6.1
Bought-in bedding¶:				
chopped straw for cow cubicles (@ 120 kg/LU/180 days)				12.2
loose housing for other animals (@ 530 kg/LU/180 days)				33.4
Volume of wastes¶:				m^3
slurry/farm wastes (@ 57 l/LU/180 days)				1693
dirty water (@ 18 l/cow/day)				670

†Taken or calculated from Milk Cheque (Whipp, 1991).
‡ADAS, personal communication.
§Calculated from Whipp (1991) and assumes same prices as concentrates.
¶MAFF (1991).

Table 2. Nitrogen inputs to a model dairy farm

Source	Inputs (kg/ha)	Farm total (t)
Atmosphere†	25	1.90
Fertilizers‡	250	19.00
Fixation§	10	0.76
Purchased feeds¶:		
concentrates (@ 2.88% N)	—	3.55
other feeds (@ 3.20% N)	—	0.20
bought-in straw (@ 0.4% N)	—	0.18
Total		25.59

†Assumed figure for SW England.
‡ADAS (S. Peel), personal communication.
§Cowling (1982): assumes an average clover content of 1–5%.
¶Whipp (1991).

respectively, giving a total of 345 t organic N in this layer of the soil over the whole farm. The flows of N from and into this large pool of soil N are difficult to predict, but it is likely that the net release through mineralization is substantial. A similar poorly drained soil in SW England (Blantern, 1991) with a previously well-fertilized grazed sward mineralized 141 kg N/ha/yr; draining the soil and increasing aeration increased the rate to 279 kg N/ha/yr. On a well-drained soil in SE England Hatch *et al.* (1991) estimated an annual net mineralization rate of 310 kg N/ha for a fertilized grass sward. The 'N-CYCLE' model of Scholefield *et al.* (1991) predicts that 173 and 149 kg N/ha is mineralized from soils of the old and reseeded swards of the model farm, respectively. From these estimates a total of nearly 12 t N is released by mineralization of soil organic matter (Table 3).

Table 3. Amounts of N cycling through components of a model dairy farm

	Per unit area (kg/ha)		Total (t)
Mineralized from soil†			
25 ha old sward	173		4.34
51 ha reseeded	149		7.62
		Subtotal	11.96
Contents in harvested herbage			
in ensiled grass (@ 2.27%)‡			8.93
in grazed grass§			13.03
Excreted N¶			
at pasture			10.47
during housing			10.19
Stored in wastes			
dirty water (@ 0.1% N)‖			0.67
farm wastes (@ 0.5% N)‖			8.47
		Subtotal	9.14

†Calculated from Scholefield *et al.* (1991).
‡Whipp (1991).
§Calculated from a daily N intake of 0.427 kg/LU (as derived from winter period) for 185 day grazing season.
¶See Table 4 and text for derivation.
‖Pain, personal communication.

Under most circumstances the apparent utilization of applied N by grass swards is very efficient. Grass roots are effective absorbers of N and where grass is cut and removed, apparent recovery in the herbage is high. Van der Meer & Van Uum-Van Lohuyzen (1986) showed that apparent recovery from fertilizer increased from 60% or less before 1975 to up to 90% in the 1980s. On the model farm the total N removed in the harvested silage is nearly 9 t, most of which is consumed by animals. If it is assumed that the total consumption of N per LU is the same during the summer as that calculated for the winter from ensiled grass plus concentrates, i.e. 0.427 kg N/LU/d, an N intake of over 13 t can be calculated for the 185 d grazing period. The total annual N consumption by the animals (21.9 t) represents an apparent efficiency of utilization of all annual inputs by harvested plants of over 85%. Efficiency of uptake into the sward would be even greater because this calculation takes no account of the N returned directly back to the soil during harvesting and through normal senescence of plant tissues, or of the losses in effluents from silage.

The problem of N losses from animal production systems usually arises because of the inefficiency of the ruminant at converting ingested N into milk or protein and liveweight gain. A maximum of 43% of the N ingested by dairy cows can be transferred in these forms (Van Vuuren & Meijs, 1987). Often the conversion rate is much less (Table 5). The excess N is excreted in dung and urine and, on a whole farm basis (Table 3), nearly 21 t N are excreted during the year; Table 4 shows the basis for this estimate. Excreted N is returned directly to the pasture during grazing or accumulated into farm wastes. Depending on whether it is in dung or urine, it can be partitioned into relatively immobile forms as soil organic matter, or into the mobile mineral N pool from which it will be taken up by the grass crop or become vulnerable to loss. In practice, much of the mobile N returned to the sward exceeds immediate local plant demands and can therefore be lost. Ryden *et al.* (1984) demonstrated the effect of excreta by comparing soil nitrate contents under cut and grazed swards.

Nitrogen excreted during housing is stored as farm wastes or collected in dirty water (Table 3), the volumes of which have been calculated using information from MAFF (1991) (Table 1). Typical N contents for these components of the dairy farm are 0.5 and 0.18%, respectively (Pain, pers. comm.). The difference between the total amount of N excreted during housing (Table 4) and that found in the stored wastes represents a loss from the system.

Many of the components cycling through the various broad N pools within the farm need further quantification. It would also be of value to define retention times within each pool in order to manipulate rates of recycling and thus maximize the utilization of N.

OUTPUTS AND LOSSES OF NITROGEN

Outputs

Much of the N ingested by farm animals is excreted. The annual offtake of 2940 kg N in milk from the dairy farm,

Table 4. Annual excretion of N from animals in a model dairy farm

Component	Total per unit area (kg/ha)	Total (t†) Dung	Urine
Grazing			
cows	87	2.24‡	4.35
others	51	1.59§	2.29
Housing			
cows	—	2.18‡	4.23
others	—	1.55§	2.23

†Data for cows calculated from difference between consumed N (grass + silage + feeds) and that removed in milk (Table 5) (see text): it is assumed that daily intake of N during summer is the same as in winter. Data for other, young animals is calculated on the basis that (1) intake per LU is as for mature animals, and (2) 78% of consumed N is excreted (Jarvis *et al.*, 1989).
‡Assumes 66% of excreted N is as urine (Van Vuuren & Meijs, 1987).
§Assumes 59% of excreted N is as urine (Jarvis *et al.*, 1989).

plus that assimilated into the protein of the younger animals, represents 27% of the fertilizer N input and only 20% of the total annual input. There are also other minor transfers of N into maintenance of the lactating animals which are not included in the present estimations.

Losses of N from grassland

The current major environmental concern is leakage of nitrate from well-drained soils into waters to be used for drinking. Other pathways of loss also have to be considered as potential causes of pollution in their own right and because they may influence the amount of nitrate leached. Volatilization and denitrification both result in major losses of N through emission of gases (ammonia and nitrous oxide, respectively) which affect atmospheric quality. Gaseous emissions and movement of nitrate into surface waters will be increasingly scrutinized.

Full definition of the factors controlling leaching from grassland soils is far from precise, though there is some information for a few UK sites and soil types. For example, in Northern Ireland relatively new, grazed swards, to which 200–300 kg fertilizer N/ha/yr was applied, lost on average 33 kg N/ha (Watson *et al.*, 1992), but losses from well-fertilized long-term swards in Institute of Grassland and Environmental Research experimental systems on well-drained soils were greater—up to 200 kg N/ha from swards where 420 kg fertilizer N/ha was applied (Macduff *et al.*, 1989). Leaching losses from an experimental system on a very poorly drained soil in SW England receiving 200 kg N/ha/yr have ranged over an 8-year period from 3 to 34 (mean 17) kg/ha/yr (Tyson, 1992). Leaching losses at particular sites increase with increasing fertilizer addition, and escalate above a critical range of fertilizer input (Barraclough *et al.*, 1992; Watson *et al.*, 1992).

Computer predictions using the N-CYCLE model (Scholefield *et al.*, 1991) suggest that, averaged across the two soil types on the model farm, 48 and 43 kg N/ha/yr would be leached from the old and reseeded swards, respectively. The model's prediction of the small difference between the older and newer swards is surprising because losses were almost three times greater from old than from reseeded swards on undrained soils at North Wyke (Tyson, 1992); larger fertilizer inputs at North Wyke may have been responsible for the difference. The computed values for leaching losses from the grazed swards in the model farm indicate a loss of nearly 18% of the applied fertilizer N (Table 5). There may also be significant transfer of N in organic forms which are not yet accounted for. Leaching losses occurring after slurry application are more difficult to compute because of a lack of data. Losses ranging from 55 to 90 kg N/ha/yr have been found for slurry applied to grassland in autumn (Smith & Chambers, 1993). An Irish study indicated that whereas 15% of the N applied in slurry was lost by leaching from a well-drained loam, only 2% was leached from an impermeable gley soil (Jarvis *et al.*, 1987). Overall losses from less permeable soils may be increased by surface runoff. Total losses by leaching and runoff from farm waste application on

Table 5. Outputs and losses of N from a model dairy farm

Form	kg/ha		t
Utilized			
removed in milk (@ 0.519%)†	—		2.94
assimilated into growth (protein)‡	—		2.16
		Subtotal	5.10
Lost at grazing			
leaching§: old sward	48		1.21
reseeded	43		2.17
denitrification§: old sward	48		1.21
reseeded	42		2.17
volatilization¶	10		0.74
Lost from wastes			
leaching‖			0.90
denitrification††			0.82
volatilization: animal house‡‡ + store			1.05
spreading§§			1.70
		Subtotal	11.97

†Whipps (1991).
‡Calculated from Jarvis *et al.* (1989). Assumes difference between intake and excretion = assimilation.
§Calculated from Scholefield *et al.* (1991).
¶Calculated from Jarvis (1991).
‖Assumes leaching = 5% of added N and runoff kg/ha = 0.0016 × applied N + 5.5 (Khaleel *et al.*, 1978).
††Calculated as 20% of NH_4^+ applied in total farm wastes (NH_4^+-N = 45% of total N) (Pain *et al.*, 1989).
‡‡Calculated from difference between winter excretion of N and that in stored wastes.
§§Calculated as 18.6% of total N applied (Jarvis & Pain, 1990).

the model dairy farm have been calculated as 897 kg N/yr (i.e. 11.8 kg N/ha/yr); this assumes a 5% leaching loss which is greater than from the impermeable gley soil but less than from the well-drained loam described by Jarvis *et al.* (1987), and uses the relationship of Khaleel *et al.* (1978) to calculate runoff.

Denitrification, the biological reduction of excess nitrate in soils by micro-organisms to N_2 and N_2O under anaerobic conditions, is another important loss process. Despite much data on denitrification losses in agricultural soils, rates are still uncertain because of problems associated with spatial and temporal variability, which make sampling and measurement under field conditions very difficult (Smith & Arah, 1990). The problem is particularly important in grassland soils which are inherently very heterogeneous. Scholefield *et al.* (1990) demonstrated the variation associated with one commonly used field technique, and concluded that a greater frequency of sampling rather than greater replication on each occasion would improve the accuracy of the estimates. Furthermore, most estimates have been based on measurements in the upper 10 cm of the soil profile and take no account of the significant denitrification that occurs below this depth (Jarvis *et al.*, 1991). The calculations for the fertilized, grazed swards on the model farm are based on estimates from the model of Scholefield *et al.* (1991), which indicate that on average 45 kg N/ha/yr is denitrified; this rate agrees with average losses at comparable levels of input on a poorly drained soil in SW England (Tyson, 1992), but is more than the annual average of 25 kg/ha measured on a similar soil type in SE England (Barraclough *et al.*, 1992). Application of slurry also stimulates denitrification losses by providing a mobile source of N, an anaerobic environment and an available carbon supply. Several studies indicate that these losses are substantial (Pain *et al.*, 1989). Total denitrification losses from the farm resulting from fertilizer inputs and excretal returns are 4.20 t N/yr (Table 5), equivalent to 55 kg N/ha/yr, though this is probably an under-estimate.

Another major loss process, ammonia volatilization, accounts for 3.49 t N (an average of 46 kg N/ha) released from the system. There is good information on losses from grazed swards and relationships have been established which can be used to provide realistic estimates (Jarvis, 1991). Much information also exists on losses of ammonia after spreading slurry and farm wastes onto grassland, and the figure quoted in Table 5 is based on average losses from UK studies (Jarvis & Pain, 1990). Much less information is available for ammonia losses directly from animal houses and waste stores. It is reassuring that the estimate of loss calculated by the difference between the N excreted during housing (Table 4) and that contained within the stored slurry (Table 3), i.e. 1.05 t NH_3-N, is similar to that calculated from Dutch information for losses per animal place using rates of 8.8 kg per cow, 3.9 kg per heifer/steer and 1.5 kg per calf (Klarenbeek & Bruins, 1988), i.e. 1.20 t NH_3-N.

The total calculated losses of N from the model dairy farm are therefore equivalent to 12.55 t, i.e. over 46% of the annual N input. A further 20% of the total input is assimilated into milk and protein in the young animals, leaving 34% (8.54 t or 112 kg/ha) as yet unaccounted for.

INTERACTIONS WITHIN THE N CYCLE

Improved methods in the future will allow better estimates of losses (especially those from denitrification) and resolve much of the current imbalance between inputs and outputs. Some of this imbalance probably results from the return to the soil of N immobilized as organic components of plant shoots and roots. Much of the model farm has relatively young swards and some N is likely to be sequestered in the soil in this way. Similar young swards in SW England, reseeded some 10 years ago and receiving 400 kg N/ha, had an average annual increase in soil organic matter N of 64 kg/ha (Tyson, 1992). Because of the smaller N input to the soils on the model farm, rates of immobilization are likely to be less than this but perhaps not substantially so.

The extent of mineralization of N in grassland soils is poorly defined but depends particularly upon sward age and the soil aeration status. This component of the N cycle plays an important role in the overall fluxes of N, and changes in grassland management can influence utilization and losses of N through effects on net mineralization. For example, drainage, by increasing aeration, can decrease denitrification potential and, most importantly, enhance mineralization at the same time. Both these effects may increase nitrate leaching. Cultivation and reseeding act similarly. Increased aeration after cultivation stimulates mineralization; this is followed by a period of accumulation and immobilization of N into the soil organic pool and a slower rate of release of mineral N. After a period, which varies according to site, soil, climate and management, the soil system achieves an equilibrium with respect to organic matter accumulation, and the differences between, for example, a cultivated, reseeded system and a long-term pasture, narrow. The net effects of some of these changes are illustrated in Figure 1; they demonstrate the complexity of the cycle and the difficulty of estimating N flows in grassland for all soil types and all management histories.

Despite the difficulties in quantifying this part of the N cycle, much of the unaccounted inputs to the model dairy farm can be allocated to immobilization. If it is assumed that 60 kg N/ha is immobilized in plant organic materials returned to the soil, this represents a total of 4.56 t N. It seems likely that much of the remaining 3.98 t (52 kg/ha) may be denitrified, but there may also be significant loss of soluble organic N in surface runoff and leachates.

FUTURE PROGRESS: IMPROVED FERTILIZER MANAGEMENT

Many of the problems in minimizing leaching and other losses are related to inadequate knowledge of the amounts, forms and patterns of change in mineral N ($NO_3^- + NH_4^+$) in the soil profile throughout a production cycle. Under most typical grazing managements, there are considerable seasonal fluctuations in the amounts and forms of mineral N and a marked accumulation in the soil towards the end of the growing season. Any major increase over and above that immediately required by the grass crop is potentially available for loss; that present at the end of grazing is particularly vulnerable because there is then little opportunity for plant uptake. One approach taken at North Wyke (Titchen *et al.*, 1989) has been to decrease this accumulation by shortening the grazing period, thereby minimizing excretal returns, and allowing the sward to act as a sink

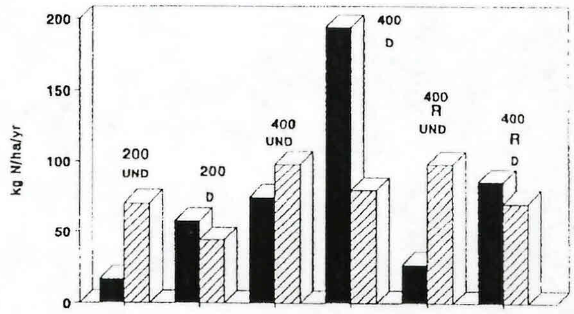

Fig. 1. Leaching and denitrification losses from grazed permanent or reseeded (R) swards on a poorly drained soil in SW England (from Tyson, 1992). Treatments were drained (D) or undrained (UND) soils, receiving 200 or 400 kg fertilizer N per year. ■, leaching; ▨, denitrification.

for excess N which is then removed in conserved forage. However, the decrease in loss potential at this stage may have a 'knock-on' effect with increased N losses resulting from the greater quantity of N in farm wastes.

Another recent approach (Titchen & Scholefield, 1992) has been to develop a rapid field test for soil mineral N to provide guidance for fertilizer additions, and thus achieve a smooth profile of mineral N levels throughout the grass growing period. When this 'tactical' method was compared with a conventional application of 210 kg N/ha/yr in regular doses at regular intervals, the decrease in the potentially leachable mineral N was over 30% and animal production was increased by 16%. Further refinement of 'tactical' fertilizer applications and greater knowledge and better integration of all N sources (from fertilizer, soil organic matter, farm waste and white clover) and their utilization will be necessary to decrease losses. This must be done on an integrated, whole farm basis with due regard to economic and environmental demands and constraints, soil type and conditions, local climate and past sward management.

As demonstrated in the model dairy farm example, the flows of N within a complete contemporary system are considerable and there are opportunities for much loss. There are also many opportunities within the cycle to enhance the efficiency of utilization at particular stages: improvement of capture of fertilizer N by forages, decrease in immediate losses and better use of slurry N (Jarvis & Pain, 1990), and increased efficiency of dietary N by ruminants are all feasible and practical options. However, it should be remembered that unless inputs are changed, decreasing flows and losses in one particular component of the overall farm cycle are likely to promote losses elsewhere. Further, enhanced capture of N into the products of grasslands ultimately results in release of the excess N (most likely through denitrification and a release of N_2O to the wider environment) at the human end of the food chain.

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Utilizing the nitrogen content of organic manures on farms—problems and practical solutions//

K. A. Smith[1] & B. J. Chambers[2]

Abstract. Organic manures contain valuable quantities of nitrogen, phosphate and potash, but many farmers regard them as 'waste materials' rather than as sources of plant nutrients. Utilization of the plant-available nitrogen content is poor at present because of manure management practices which lead to leaching and atmospheric losses. Experiments studying the effect of timing suggest that, in order to decrease nitrate leaching, applications of manures which contain much available nitrogen should not be made during the period September to December on freely draining grassland and arable soils. Spring top dressings of dilute pig or cattle slurries and poultry manures to growing cereal crops are generally more efficient than autumn applications, particularly on freely draining soils. Legislation requiring manures to be applied in an environmentally acceptable manner and the economic need for farmers to realize the nutrient value of organic manures are likely to change the farming industry's perception of manures as 'waste materials'.

INTRODUCTION

HISTORICALLY, organic manures have been important for maintaining soil fertility. However, in recent decades a ready supply of cheap inorganic fertilizers has meant that these have dominated farm fertilizer policies. Concurrently, increased intensification of livestock production has encouraged farmers to regard application of organic manures as a 'waste disposal' process. Consequently large quantities have been applied to limited land areas in the autumn/winter period without considering their manurial value and the potential pollution hazards.

Estimates of livestock waste production on UK farms based on June 1990 census figures (MAFF, 1990) suggest a total annual output of around 191 million tonnes, of which approximately 78 million tonnes is slurry or manure collected from buildings and yards and requiring handling and storage. The plant-available nitrogen content of these manures has a potential annual value of £56 million, based on a fertilizer price of 30 pence/kg N. However, its utilization is poor because management practices lead to nitrate leaching, ammonia volatilization and denitrification losses.

Recent statistics from Chalmers *et al.* (1992) suggest that farmer perception of the fertilizer value of organic manures is poor. The figures in Table 1 represent the difference between inorganic fertilizer rates applied to crops receiving no organic manures and those receiving organic manures. They show that farmers make small, inconsistent decreases in inorganic fertilizer following applications of organic manures. Even a modest improvement in allowances for nutrients supplied from organic manures could result in major savings in fertilizer costs without loss of yield and with less environmental pollution.

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Nutrient management from a farming systems perspective

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Abstract

Current intensive dairy farm managements may be "leaking" large quantities of nutrients (N and P) and in so doing threatening the wider environment. Such losses can be reduced considerably by an increased utilisation of nutrients in all components of the farm and by understanding their cycling within and between these components. A systems approach provides one important means of developing the understanding and integration of the complex interactions that occur with nutrient flows and losses in agriculture. Systems analysis indicates that inputs of N in fertilizers and feeds could be halved for intensive dairy systems in the UK and the NL. Important factors include reduced feed intake per unit of milk produced, as the result of improved milk yield per cow, an improved utilisation of nutrients in manures and slurries produced on the farm and a changing balance between the area of grassland and that for the production of other fodder crops where this is possible.

Keywords: dairy farms, environmental impact, indicators, losses, fertilizers, nitrogen, phosphorus.

Introduction

Over recent years, there have been increasing pressures on those who manage agricultural land to reconsider their nutrient use policies. As well as those operational drivers which farmers use to manipulate their nutrients for optimal economic returns, environmental issues have recently had substantial impact in influencing the way that we consider nutrients in grasslands and in dairy farming systems in particular. Throughout Europe, policies have been put into place to meet the requirements of the Nitrate Directive, accepted by member states of the EU in 1991. Other policies relating to diffuse sources of phosphorus (P), ammonia (NH_3) and nitrous oxide (N_2O) are currently under various stages of consideration. For all of these concerns, dairy farming is an important emission source (Jarvis, 1999). Fertilizers have become cheap compared with the costs of land, labour and quotas and because of this and because they provide flexibility for the farmer, fertilizing above recommended optimum levels has often been the case. In the Netherlands, for example, grasslands receive 50 kg N ha^{-1} more than recommended optima (Reijneveld *et al.*, 2000).

The research effort that has taken place over recent years has modified, in many respects (and especially for N), the approaches that have been taken to identify problems, to define opportunities to reduce these and to maintain sustainable systems for meat and milk production. One important step was the realisation that, whilst a knowledge of effects at soil and field scales was essential for defining economic responses, only by understanding the relationships between these and other components of the whole system could effective optimisation for both production and environment targets be achieved. To do this properly demands a knowledge not only of inputs and outputs to and from the soil (a surface balance), but also of inputs and outputs to and from the farm (a "farm gate" balance) and of all losses, internal flows and transfers (to provide a systems balance and understanding). Surface balances have been important tools for fertilizer recommendations and monitoring soil fertility but do not provide

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direct information on losses, nor of the fate of the nutrients partitioned in forages to be consumed by animals. Farm gate balances provide a simple means of determining nutrient surpluses within a system but not of their fate. Farm gate balances do, however, provide a means of indicating efficiency/inefficiency (of outputs in relation to inputs) within the system and the surplus (inputs minus outputs) is indicative of the potential for loss. Figure 1 shows, perhaps not surprisingly, that for N, there is a strong relationship between system surpluses and losses (leaching + denitrification + volatilization). The fact that there is not a 1:1 ratio, indicates that either not all the losses are being accounted for and/or that the system is not in equilibrium and is accumulating N (in soil organic matter). As the sward age increases, it can be expected that the differential between surplus and loss will grow smaller.

Of most value, and without which the relationship in Figure 1 could not be demonstrated, but more difficult to achieve because of the greater requirement for data, is the systems balance. This allows features of a nutrient's behaviour within, as well as movement to and from, the system to be examined. Most usually this has been used as a research tool either as small-scale experimental units (Peel *et al.*, 1997; Ledgard *et al.*, 1999) or complete whole farm systems (Aarts *et al.*, 1999). Modelling and systems analysis have also been used to demonstrate impacts, trends and opportunities for changes in nutrient management (Jarvis *et al.*, 1996). These approaches to managing nutrients have demonstrated new methods for developing management structures to satisfy the dual demands of production and reduced environmental impact. The advantages of understanding nutrient behaviour and management at a farm scale are therefore as follows: (i) it provides an appreciation of all sources and fates of nutrients, (ii) it requires a knowledge of transfers and recycling which is essential to maximise the efficiency of utilisation of the internal nutrient resources and to identify 'leaky' components of the system and (iii) because it is integrative, it also allows an understanding of the 'knock-on' effects that any change in nutrient management may have within other parts of the farm. For N, which has much potential for loss, this is especially important.

Dairy farms are particularly complex in terms of their management structure and the way in which nutrients are utilised and recycled. Many farms have relied on large inputs of nutrients in fertilizers, especially of N, but also of phosphorus (P) and potassium (K) to provide the dry matter production and the flexibility in management required to meet economic targets. There have also been increased inputs of concentrates because of their low costs and the need to improve quality of the diet of high yielding cows; these also contribute large quantities of nutrients to the system. In the present paper we concentrate on nutrient management within dairy farming systems with particular reference (because of the current environmental issues) to N and P. An improved appreciation of the potential to improve nutrient management will be to the benefit of farmers and the public at large alike. There are many other issues within the EU which will also interact with nutrient management and demand cross compliance, for example reduction of subsidies on milk and payment for the quality of the environment.

The dairy farm as a system

There are many opportunities for losses of nutrients from dairy farms. All farms operate with a surplus of nutrients, i.e. an excess of inputs over those removed in products. Surpluses generally increase (Jarvis, 1999) in a more or less linear way (Fig. 1) with increasing inputs to dairy farms. Their multi-component, multi-phasic nature means that nutrients are transferred from one component of the system to another (Fig. 2). Each transfer provides an opportunity for inefficiency and loss, especially in the case of N with which there is often an associated change in physical and chemical forms. Table 1 illustrates the efficiencies in the various parts of the system.

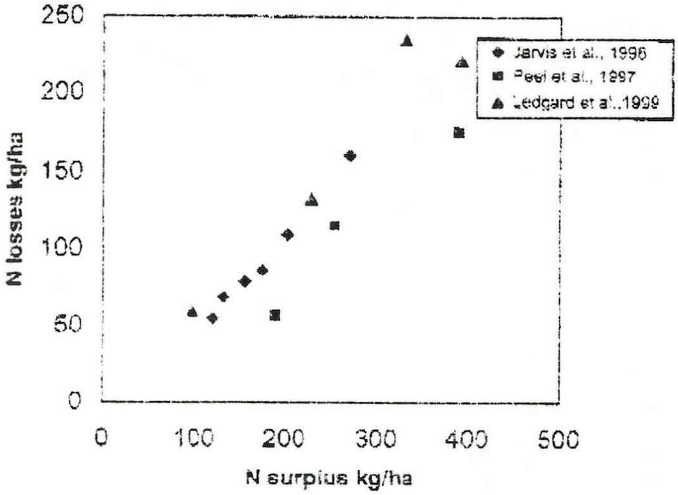

Figure 1. Relationship between nitrogen surplus and predicted losses from experimental dairy farm systems.

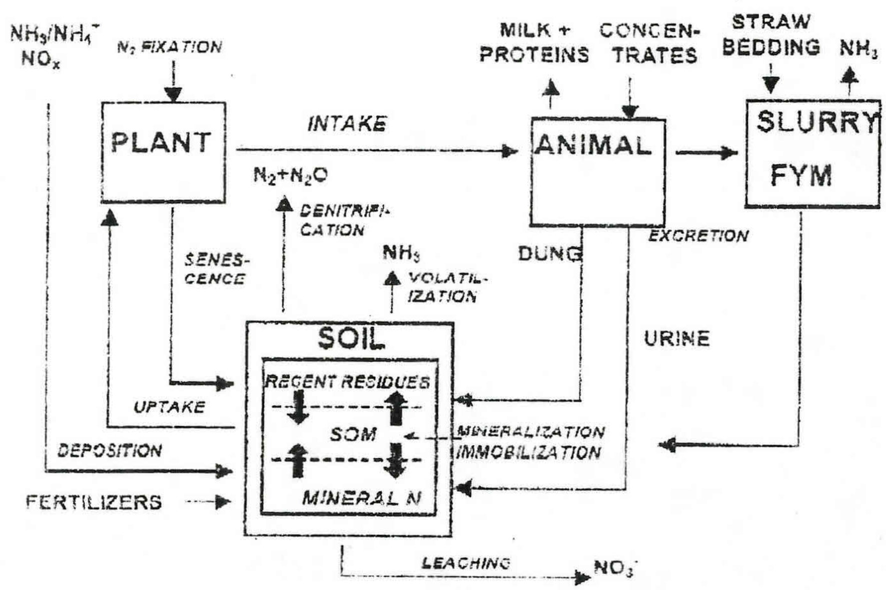

Figure 2. Simplified nitrogen cycle in dairy farming systems.

Table 1. Efficiencies (%) of component parts of specialised intensive dairy farming systems in the Netherlands on sandy soil: (a) technically attainable and (b) realised in practice by skilled farmers (Aarts *et al.*, 1999; Aarts *et al.*, 2000).

Component	Technically attainable		Average realised in practice	
	N	P	N	P
Soil: transfer from soil to harvestable crop	77	100	53	60
Crop: transfer from harvested crop to feed intake	86	92	71	75
Animal: conversion from feed to milk+meat	25	32	18	22
Slurry/dung+urine: transfer from excreta to soil	93	100	80	100
Whole farm: from inputs to outputs	36	100	16	27

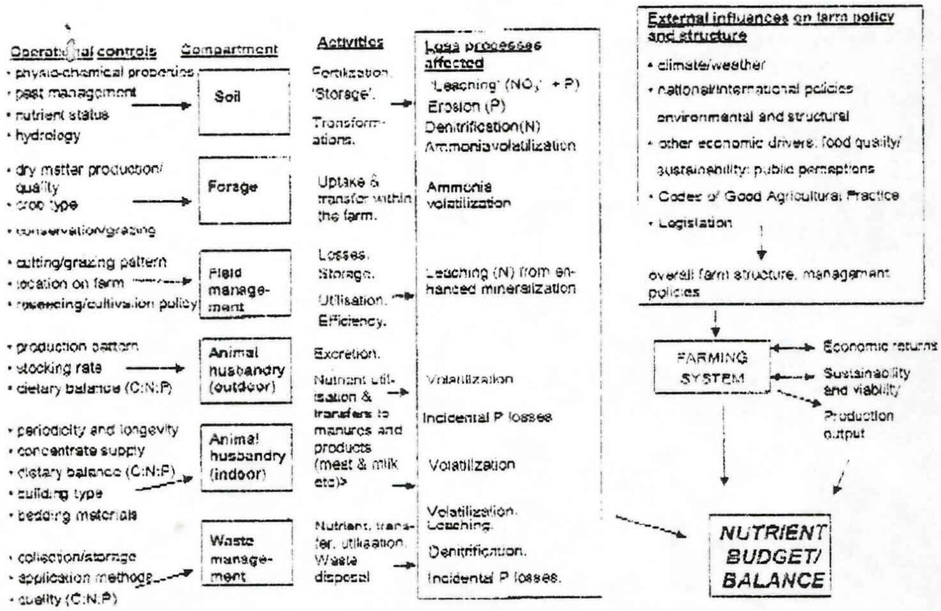

Figure 3. Controls over flows and losses of nitrogen and phosphorus in dairy farms (modified from Jarvis, 1997).

Much of the opportunity for loss is connected with the ruminant. Whereas grass and other forages are usually highly efficient removers of available nutrients from the soil into their biomass, the ruminant is not and excretes large proportions of ingested N and P (Table 1). The needs of the ruminant are also responsible for the often large imports of nutrients into the farming system in concentrates and other additional feeds. These are often neglected when the nutrient status of the system is considered but often comprise a major proportion of the nutrient import to the farm and therefore to the total balance/surplus in the system.

In order to provide truly effective improvements, it is important that all parts of the system are considered in an integrative way. There are a number of reasons for this. Firstly, as indicated above, it is important that all inputs and outputs to the system are considered. Each component and phase of management interacts with others and manipulation of one may have consequent effects in other parts of the farm (Figs. 2 and 3). The inefficient stages and therefore the "weak" phases which offer opportunity for loss and improvement are identified in Figure 2. In general, N has many opportunities for loss because of the changes in valence and form which occur especially after passage through the ruminant gut. Modelled (Jarvis, 1993; Jarvis *et al.*, 1996; Aarts *et al.*, 1992) studies indicate that at least 50% of the N inputs to dairying are lost to the wider environment by leaching, volatilization or denitrification. As is demonstrated in Figure 3, as well as the on-farm, direct activities which are directed at nutrient use, transfers losses and balances, off-farm factors (economics, policies, weather/climate) also determine the nutrient status and balance of the system.

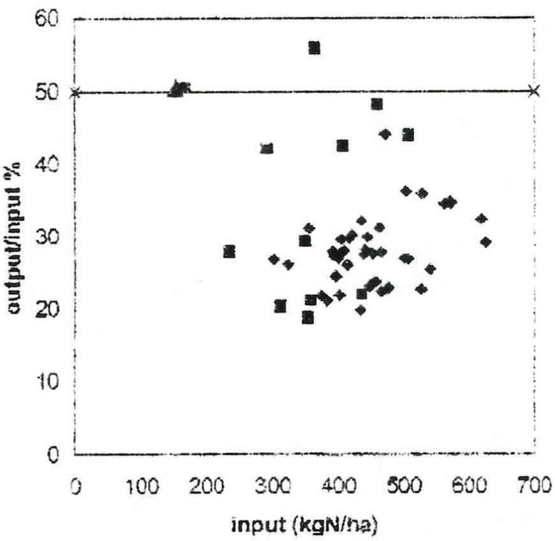

Figure 4. Relationship between nitrogen inputs and efficiency of nitrogen use in Dutch dairy farms (data from Reijneveld *et al.*, 2000; ◆ typical Dutch farms; ■ " cows and opportunities" scheme; ▲ de Marke; -✱- de Marke line).

Figure 4 demonstrates the efficiency of N use in practice in 1997 (Reijneveld *et al.*, 2000). Inputs to typical Dutch commercial farms are shown (for fertilizers and feed only); these represent the complete range of soil type and intensity of milk production in the Netherlands. The data from "Cows and Opportunities" scheme are for 12 selected commercial farms just before the start of a new project in which farmers aim to improve nutrient managements following the experiences of De Marke. The wide range in the data illustrates big differences between commercial farms and therefore opportunities and potential to make improvements. For P, the proportions that are lost are much less than for N because the generally conservative nature of P means that it will accumulate within the farm. A recent analysis has indicated that only *c.* 2% of the annual P input is lost from a dairy farm (Haygarth *et al.*, 1998). Although this represents only a small proportion, it has some significance in terms of environmental impact and eutrophication of surface waters (and this impact can increase substantially when the fixation capacity of the soil is exceeded).

The vulnerable components of dairy farm management

Losses to water

(i) Nitrogen

Although there may be environmentally significant transfers of other forms of N (*viz.* NO_2^- and NH_4^+), NO_3^- is the main form that is lost to waters. This arises because of accumulations in the soil profile immediately prior to periods when plant uptake has diminished and there is excess rainfall. There may be some direct loss from fertilizers and as the result of manure applications, but these are small and most NO_3^- leaching from grassland can be related to mineral N derived from the excreta of grazing animals. Rates of leaching from grazed areas are controlled by the inputs to, and management of, the sward and the extent of soil moisture deficits during the preceding growth period (Scholefield *et al.*, 1993). Options for reducing NO_3^- leaching by reducing, for example, the grazing period, have been suggested; current trends to extend grazing seasons may exacerbate the problem. As well as that derived from excreta, NO_3^- is also generated from the mineralization and nitrification processes in the soil acting upon native and added (in manures) organic materials. Stimulating mineralization by, for example, cultivating long-term pastures has an immediate impact on the generation of excess NO_3^- which may be lost if cultivations are made at inappropriate times of the year.

(ii) Phosphorus

Under most circumstances only small amounts of P (in various inorganic and organic forms) are leached in drainage unless the sorption capacity of the soil has been exceeded. Nevertheless, the amounts that are lost are of importance in acting as diffuse supplies of P with environmental impact. Accentuated losses of P occur when there is a combination of poorly timed applications of fertilizers or manures/slurries to land when the soil are above field capacity (= "incidental" losses: Haygarth and Jarvis, 1999).

Losses to air

These relate entirely to N and involve denitrification and NH_3 volatilization processes. Denitrification, the anaerobic reduction of NO_3^- to N_2 and N_2O, is a soil-based process occurring in pastures when they are wet and there is an excess of NO_3^-. There is therefore opportunity for direct losses when mineral (especially NO_3^--based) fertilizers are applied, when excreta are deposited in the field and when slurries/manures are applied. Some smaller denitrification losses may also occur from stored manures. Estimates indicate that, in total, for a dairy farm based on a poorly drained soil, the equivalent of at least 22% of the annual fertilizer input is lost through this process (Jarvis, 1993). Improved management of fertilizers and alternative strategies for manures all affect denitrification and the losses that may result.

Ammonia volatilization occurs wherever free NH_3 is generated, which in the context of a dairy farm is largely through the degradation of urea. Ammonium based fertilizers provide small emission sources, but unless urea fertilizer is being used, the major (and substantial) source is the urea excreted by the cattle. Volatilization can therefore occur throughout the farming system – during grazing, in the collection yards and houses, and from stored and applied manures and slurries. There are opportunities to reduce losses of NH_3 at each stage of production and these have been extensively documented. This is an area where a complete systems understanding is particularly important because of the "downstream" effects that may occur when a specific action is taken to mitigate a particular loss pathway.

Managing nutrients on a farm scale

Strategies

There are many options to reduce nutrient inputs, surpluses and losses within dairy farm systems; some of these are summarised in Table 2. In many instances care needs to be taken that measures taken to reduce one loss do not conflict with measures that may be taken to reduce another. For example, reducing or increasing the grazing period as measures to reduce leaching and overall farm NH_3 emissions, respectively, may incidentally raise NH_3 and lower leaching, respectively (Table 2).

Table 2. Measures to improve the nutrient flows of a dairy farming system.

Component of system	Measure/activity	Effect intended
Soil	Reduce fertilizer inputs	Enhance utilisation of system nutrients; decreased inputs purchased fertilizers
	Reduce fertilization period	Increase utilisation of fertilizers
	Increase maize area	Less fertilizer needed
	Catch crop after maize	Reduce NO_3^- leaching after harvest
	Regular soil testing	Improve fertilizer application
	Improve fertilizer application accuracy: avoid vulnerable areas	More effective supply; reduce direct transfer of N and P into waters
Crop	Promote rotational grazing	Reduce grazing losses
	Use best management practice for reseeding 'permanent' grassland	Reduce nitrate leaching from increased mineralization
	Use "nutrient efficient" genotypes	Enhance capture of nutrients in crop
Animal	Include maize or other low C:N products in diet	Reduce N content -> reduced N excretion
	Increase milk production/cow	Reduce feed needs/kg milk quota: less excreta per unit of product
	Reduced replacement rate of cows	Reduce feed needs/kg milk quota
	Rear young stock outside farm	Reduce feed needs/kg milk quota
	Reduce P contents of supplements	Reduce P excretion and farm surplus
	Reduced grazing period	More efficient use of grass crop
	Increase grazing period	Reduce overall farm NH_3 emissions
Dung+urine	Limiting daily grazing time	Reduce urine patches, increasing quantity slurry and reduce mineral fertilizers. Reduce grazing losses
	Restrict length of grazing season	Reduce urine patches, esp. in autumn, increasing slurry quantity. Reduce grazing losses
Slurry/manure	Fast removal to storage (e.g. scrapers)	Reduce NH_3 volatilization
	Covering store	" " "
	Incorporation , injection, band spreading	" " "

Strategies for the soil-component aim to make more effective use of fertilizers and include increasing those crops with low requirements compared with grassland, such as maize. Utilisation of fertilizers can also be improved by taking better account of soil supplies and better timing: to reduce inefficiency, fertilization should start later in spring and end earlier in autumn. The effect of careful fertilizer policies for grassland is clearly demonstrated in Fig. 5. Farmgate surpluses for Dutch commercial farms are strongly related to the difference between actual fertilizer application rates and those which are recommended. On average for the farms in Figure 5, surpluses could be reduced to c. 268 kg N ha^{-1}, if recommendations had been followed.

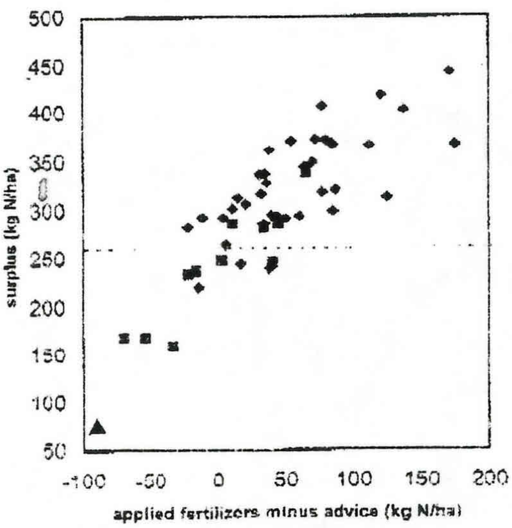

Figure 5. Efficiency (output/input) as affected by the difference between actual fertilization and recommended (financial optimum) fertilization (data from Reijneveld *et al.*, 2000◆ typical Dutch farms; ■ " cows and opportunities" scheme; ▲ de Marke; -✱- de Marke line).

The main strategy for the crop component can be related to a changed balance between grazing and cutting for conservation. On the one hand, there is potential to improve utilisation by reducing grazing and therefore harvest losses; as a result less feed has to be purchased (less inputs) and the risk of losses from crop residues is reduced. Stopping grazing earlier in the season also reduces leaching losses and the incidental losses of P. On the other hand, grazing the animals for longer may reduce the overall NH_3 losses from the system because that resulting from the housed phase of production in most countries is substantially greater than when animals are grazing.

For the animal component, the strategy is to reduce imported feeds, for example by reducing the number of animals whilst maintaining milk production by using cows with a high genetic potential and/or improved feeding regimes to increase yield per cow. Reduced N and P intake will reduce total excretion on the farm. This automatically reduces the opportunity for losses. For better management of the urine+dung component, as noted above, avoidance of generation of urine- and dung patches as much as possible, especially those in late summer or autumn. Restricted grazing has some impact because nutrient concentrations in these patches are much higher than a crop can utilise and much of the N can be lost. The potential to use nutrients in dung and urine collected indoors is much greater and reduces the need for mineral fertilizers but only if the slurry or manure is used effectively. The value of slurry as a fertilizer can be improved particularly by reducing NH_3 volatilization and there are a number of measures by which this can be achieved.

Case studies.

Many of the options identified in Table 2 have been incorporated into the N managements of experimental systems: some case study examples of these are summarised in Table 3. These also illustrate three different systems approaches taken to investigate the impact of various management strategies to improve N use efficiency and reduce losses. Full details are given in the relevant publications but briefly the systems are as follows.

A. Farm scale comparisons between a commercial enterprise and a prototype system (De Marke) which includes a reduced stocking rate, lower N fertilizer inputs, crop rotations and catch cropping after increased maize production (Aarts *et al.*, 1999)

B: Modelling/systems synthesis of the potential improvements to a typical UK dairy farm with modelled prediction of effects on transfer and losses. Management changes include using a tactical fertilizer method and injecting slurry, using mixed grass/clover swards and no fertilizer N, and growing maize instead of producing grass silage (Jarvis *et al.*, 1996)

C: Experimental farmlet scale experiments to compare a high output, commercial production system in the UK with (a) a reduced loss high output system based on improved fertilizer and slurry N utilisation and maize production and (b) animal loss and reduced intensity system with further reductions in N inputs and injected slurry (Peel *et al.*, 1998).

D. Experimental New Zealand farmlets to compare a grass/clover management with fertilizer at 200 or 400 kg ha^{-1} and high or low stocking rates (Ledgard *et al.*, 1999).

The data in Table 3 can be used to illustrate a number of points i.e. that (i) there are effective options to reduce fertilizer N inputs substantially and to better utilise N in manures and slurries with consequent effects on N surpluses and losses; (ii) as demonstrated in Figure 1, predicted and measured losses are highly dependent upon the system surplus; (iii) there is a good deal of similarity in the effects demonstrated between the systems that have been investigated, especially in the UK and NL; (iv) consideration of nutrients from dietary sources is important: only by understanding all the system transfers is the full impact of this appreciated; and (v) it is important that when comparisons are made all impacts are considered so that true effects can be quantified: it will also be important to take account of the economic implications of the changes proposed.

Table 3 includes some key indicators of the changes that can be achieved which link production and environmental effects and illustrate, within these experimental systems, the substantial effects that can be attained to the benefit of both targets. The indicators can be considered as diagnostics of efficiency of production, i.e. capture into product and litres of milk produced per unit of N input into the system, and as environmental diagnostics, i.e. farm surplus (an indicator of potential loss) and kg N lost per unit of milk produced (to provide a quality assurance guide). The latter, for example, can be used to demonstrate the effects of having high yielding cows (B) or high stocking rates (D) in terms of N emissions associated with each litre of milk produced. The final indicator, kg N lost per kg N surplus, could be considered an indicator of the extent to which the system is away from a steady state and shows distinct differences between the four basic farming systems experiments. In many cases the management changes have halved the values of the indicators. This is important because a

Table 3. Nitrogen use, loss and efficiency indices for "experimental" dairy farming systems.

	A: NL farm scale		B: UK "modelled" farms				C: UK experimental farmlets			D: NZ experimental farmlets			
	1	2	1	2	3	4	1	2	3	1	2	3	4
N inputs (kg ha^{-1})													
fertilizers	242	69	250	155	-	185	321	189	135	0	215	413	411
fixation		9	10	10	144	10	-	-	-	174	117	40	37
total (including feeds, deposition etc)	486	226	337	242	210	272	457	322	247	235	339	489	494
N capture (kg ha^{-1})													
milk	64	64	67	67	54	67	64	63	55	74	89	92	113
meat	14	10					4	5	3	6	6	6	6
% input	16	32	20	28	26	25	15	21	24	34	28	20	24
N surplus (kg ha^{-1})	407	153	270	175	156	202	389	254	189	97	228	332	392
N losses (kg ha^{-1})	407	113	160	86	79	109	175	115	57	60	132	235	221
Other Indicators													
Production: litres milk/kg N input	24.5	52.6	22.1	30.5	28.4	27.4	27.2	37.4	41.6	*47.0	*32.6	*22.6	*29.8
Losses: kg N per 1000 l milk	34.2	9.5	21.5	11.5	13.2	14.6	14.0	9.5	5.5	*5.4	*11.9	*21.3	*15.0
kg N per kg N surplus	1.00	0.73	0.59	0.49	0.50	0.54	0.45	0.45	0.30	0.62	0.59	0.71	0.56

A. Aarts *et al.*, 1999
1. Commercial dairy farm
2. De Marke experimental system

B. Jarvis *et al.*, 1996
1. 'Typical' management
2. Injected slurry and tactical fertilizer
3. Mixed clover swards, no fert. N
4. Maize silage

C. Peel *et al.*, 1997
1. Commercial practice, high outputs
2. Reduced loss, high outputs
3. Minimal loss reduced intensity

D. Ledgard *et al.*, 1999
1. Clover N only
2. " + 200 kg N
3. " + 400 kg N low stocking
4. " + 400 kg N high stocking

* assumes 3348 l/cow (Jarvis and Ledgard 2000

recent survey of N in UK diary farms showed that there were many-fold ranges in a number of these indicators (Jarvis, 1999), demonstrating that there is much opportunity to make improvement in overall N efficiency.

Future progress

There is much opportunity to improve nutrient use within dairy farms and many new, relevant technologies which could be employed. For maximum effect, it is important that the adoption of new practices are seen within the context of the whole farming system so that both advantages and disadvantages can be observed. Understanding the nutrient budget of farming system should enable farmers to better appreciate all the nutrient sources and fluxes and enhance efficiency and profitability. The development of indices and indicators will enable changes to be observed and comparisons to be made, but these will need to be compared with appropriate standards and yardsticks.

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Nutrient cycling and losses based on a mass-balance model in grazed pastures receiving long-term superphosphate applications in New Zealand.

1. Phosphorus

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SUMMARY

Phosphorus (P) cycling and losses in irrigated, sheep-grazed pastures receiving superphosphate (SP) applications for 35 years at annual rates of 0, 188 and 376 kg/ha were studied using a mass-balance approach which accounted both for P inputs to and outputs from the soil–plant–animal system. Total recoveries of applied P in the soil–plant–animal systems in the 188 and 376 kg SP/ha treatments were 94 and 83% respectively. Approximately 52–53% of the applied P was recovered in the soil within the major plant rooting zone (0–300 mm soil depth). These data suggest that P leaching losses from SP fertilizer, plant litter, root residue and sheep faeces were unlikely to occur beyond the major plant rooting zone. However, the transfer of excretal P to stock camps and the transport of P from SP fertilizer, plant litter and sheep faeces via the irrigation water along the border from the top to the bottom of the irrigated border strip accounted for less than 6% of the applied P. Superphosphate applications resulted in the accumulation of both soil inorganic and organic P fractions to a depth of 225 mm. The accumulation of soil inorganic P was most pronounced when SP was applied annually at the rate of 376 kg/ha, which was in excess of pasture P requirements.

INTRODUCTION

Superphosphate (SP) has traditionally played an important role in the maintenance of phosphorus (P) and sulphur (S) in ryegrass–white clover pastures in New Zealand (During 1984; Nguyen *et al.* 1989) and Australia (Blair 1983; Lewis *et al.* 1987*a*, *b*). Efficient use of the applied SP in grazed pastures depends on the extent of excretal transfer loss to stock camps, leaching losses, surface run-off, mineralization of soil organic P and organic S, and the immobilization of the applied P and S (Blair *et al.* 1977; Barrow 1980; Cornforth & Sinclair 1984; Sinclair & Saunders 1984).

In recent years, a downturn in the agricultural economy in Australasia, together with an increasing awareness of the need to sustain agriculture with non-renewable fertilizer resources, has focused greater attention on the importance of (i) identifying the major processes influencing the efficient use of applied P and S, (ii) estimating P and S losses from grazed pastures and (iii) assessing the maintenance P and S requirements in grazed pastures (Cornforth & Sinclair 1984; Sinclair & Saunders 1984; Till *et al.* 1987).

In response to these issues, various New Zealand researchers have constructed mass-balance models of P and S for sheep farms (Karlovsky 1982; Saggar *et al.* 1990*a*, *b*) and dairy farms (Middleton & Smith 1978; Parfitt 1980). A similar approach has been taken by Lewis *et al.* (1987*a*, *b*) for pastures in the sandy soils of south-eastern South Australia. In fact, the P and S models developed by the New Zealand Ministry of Agriculture and Fisheries (MAF) for estimating pasture maintenance P and S requirements are based on the mass-balance modelling approach (Cornforth & Sinclair 1984; Sinclair & Saunders 1984). This approach accounts for P and S inputs and outputs from fertilizers, irrigation, rainfall, pasture uptake, excretal returns and removal of nutrients in animal products. By constructing mass-balance models, Lewis *et al.* (1987*a*, *b*) and Saggar *et al.* (1990*a*, *b*) showed that P was recycled more efficiently in grazed pastures than S from SP. These workers

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attributed the more efficient recycling of applied P to the greater adsorption of P by soil colloids in the form of $H_2PO_4^-$ and HPO_4^{2-} than of S in the form of SO_4^{2-} ions.

Since the extent of applied P and S remaining in soils from SP applications to grazed pastures depends on factors such as pasture management practices, soil phosphate and SO_4^{2-} retention capacities, annual rainfall distributions and topography (Gillingham 1980; Till 1981; Cornforth & Sinclair 1984; Sinclair & Saunders 1984), information obtained from hill-country sheep farms or dairy farms (Middleton & Smith 1978; Parfitt 1980; Saggar *et al.* 1990*a*, *b*) may not be applicable to sheep-grazed pastures in flat land, which comprises the major part of Australasian agriculture. The long-term grazing experiment conducted at Winchmore Irrigation Research Station in New Zealand since 1952 (Nguyen *et al.* 1989; Condron & Goh 1990; Nguyen & Goh 1990) offers a unique opportunity for constructing mass-balance models of P and S to assess the effects of grazing animals and long-term SP application on the recovery of applied P and S in irrigated sheep-grazed pastures.

This paper reports on the results of using the mass-balance approach in identifying the major processes that affect P cycling and losses in irrigated sheep-grazed pastures, while a similar approach for S cycling and losses is presented in another paper (Nguyen & Goh 1992).

MATERIALS AND METHODS

Experimental sites and treatments

The experimental site was on a shallow (300–450 mm) Lismore stony silt loam (Udic Ustochrept), derived from moderately weathered greywacke loess over gravels (Fieldes 1968). The soil was well drained, with low phosphate (< 30%) and sulphate (< 15%) retention capacities (Nguyen 1990).

The experiment was initiated in 1952 on a 2-year-old pasture which had been sown with perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.) (Nguyen *et al.* 1989). Three fertilizer treatments were used, which consisted of annual applications of SP (9·3% S; 11% P) in July of each year since 1952 at rates of 0 (control), 188 and 376 kg/ha, to provide annual inputs of P of 0, 17.5 and 35 kg P/ha and S of 0, 21 and 42 kg S/ha. These treatments were arranged in separately fenced border strips of 0·07 ha (64 m × 11 m) per border (Fig. 1) in a randomized complete block design with four replicates per treatment. All treatments received lime at the rate of 4 t/ha in 1972 to maintain soil pH within the optimum range of 5·8–6·0 for pasture growth (Edmeades *et al.* 1985).

Each treatment was grazed by a separate flock of dry ewes to ensure that there was no excretal transfer of P and S between different treatments. The stocking rate (SR) in each treatment was adjusted to give adequate control of pasture growth and an annual average pasture utilization (PU) of 80% (Rickard & McBride 1987; Nguyen *et al.* 1989). The average annual SRs in the 0, 188 and 376 kg SP/ha per year treatments were 8, 17 and 20 ewes/ha, respectively. Each dry ewe (50–60 kg liveweight) was replaced every 4–5 years and produced 4·3–4·4 kg of wool annually (Nguyen 1990).

The site receives an average annual rainfall of 600–750 mm (Taylor 1981). Mean monthly soil temperature at a depth of 50 mm was 3 °C in July (winter) and 16 °C in January (summer). Soil temperature ranges in winter, spring, summer and autumn were 3·4–6·1, 9·6–14·4, 15·2–20·6 and 10·2–13·8 °C respectively (Rickard & McBride 1986). Irrigation water was applied using the border-strip system (Taylor 1981) during the December–April period (summer to early autumn), whenever the gravimetric soil moisture content (w/w) in the top 100 mm depth was *c.* 15%, with an average of five irrigations per year and 70–100 mm per irrigation (Nguyen *et al.* 1989).

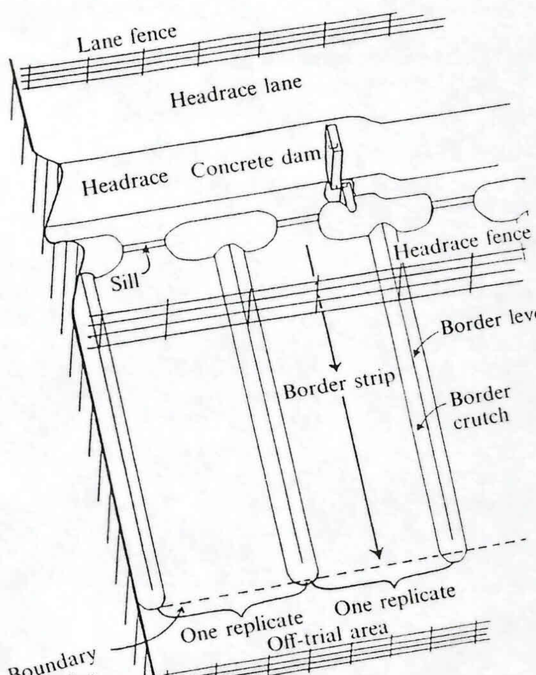

Fig. 1. Schematic layout of a border under border-strip irrigation (after Taylor 1981) and a replicate of each superphosphate treatment.

Table 1. *Parameters used in the phosphorus transfer model for determining phosphorus transfer losses from non-camp to camp areas (adapted from Saggar* et al. *1988, 1990*a, b)

Symbol	Definition
	Herbage P uptake from camp (c) and non-camp (nc) areas
HPc	Herbage P uptake (kg/ha per year) from camp areas.
HDMc	Herbage dry matter (DM) (kg/ha per year) from camp areas.
[HP]c	Herbage P concentration (%) in camp areas.
HPcf	Herbage P uptake (kg/ha per year) from camp areas after correcting for the proportion (Sc%) of these areas on 1 ha basis.
HPnc	Herbage P uptake (kg/ha per year) from non-camp areas.
HDMnc	Herbage DM (kg/ha per year) from non-camp areas.
[HP]nc	Herbage P concentration (%) in non-camp areas.
HPncf	Herbage P uptake (kg/ha per year) from non-camp areas after correcting for the proportion (Snc%) of these areas on 1 ha basis.
	Animal P uptake from camp (c) and non-camp (nc) areas
APcf	Animal P uptake (kg/ha per year) from camp areas.
APncf	Animal P uptake (kg/ha per year) from non-camp areas.
ΣAP	Sum of animal P uptake (kg/ha per year) from both camp and non-camp areas.
	Faecal P returns to camp (c) and non-camp (nc) areas
ΣFaP	Faecal P return (kg/ha per year) to camp and non-camp areas.
FaPc	Faecal P return (kg/ha per year) to camp areas after taking into account the proportion (Dc%) of faecal DM return to camp areas.
FaPnc	Faecal P return (kg/ha per year) to non-camp areas after taking into account the proportion (Dnc%) of faecal DM return to non-camp areas.
Pgc, Plnc	Amounts (kg/ha per year) of P transfer gains (Pgc) and losses (Plnc) from camp and non-camp areas respectively.
Pgc%, Plnc%	Proportion of excretal P gains (Pgc%) and losses (Plnc%) from camp and non-camp areas respectively.

The annual P inputs to the experimental site from irrigation water and rainfall were negligible (Quin & Woods 1978), while the amount of annual P released from soil weathering was estimated to be 4 kg P/ha (Quin & Rickard 1981).

The area of each border which was irrigated was termed the border strip (Fig. 1). Water was admitted to each border strip through the headrace and sills (Taylor 1981) and the border strips were separated by mounds (border levees) of soil (350 mm wide, 300 mm high) running the length of the strips. The area that was within *c.* 1000 mm of the base of the levee was referred to as the border crutch. Sheep were found to camp and deposit substantial amounts of P, S and K on both border crutches and border levees (Close & Woods 1986; Nguyen & Goh 1988; Nguyen 1990), and these were defined as camp areas.

Development of mass-balance phosphorus models

The development of the mass-balance P model involved the integration of the following components: annual P inputs from SP fertilizer applications, rock weathering, irrigation and rainfall; annual herbage P uptake and animal P intake; the return of root and plant P residues to soils for pasture plant uptake; excretal P transfer losses to stock camps and changes in soil P after 35 years of SP applications.

Data on some of these components, such as herbage nutrient uptakes and changes in soil P and S in the topsoil (0–75 mm) over a 35-year period have been reported (Nguyen *et al.* 1989). However, additional information is required on excretal transfer losses, root P uptake, inorganic and organic P contents in pasture herbage, pasture roots and sheep faeces and changes in soil P in camp areas and in the lower soil depths (75–300 mm) which are within the major rooting zone (0–300 mm) of border strip, irrigated, grazed pastures (Nguyen 1990). Detailed methods for obtaining values of these components are presented below.

In the mass-balance model, it was assumed that the inorganic P (P_i) fractions in plant litter, root residues and sheep faeces were returned directly to the soil inorganic P (soil P_i) pool while their organic P (P_o) fraction contributed to soil organic P (soil P_o) before being decomposed and mineralized to phosphate (Rowarth *et al.* 1985). Total P was designated P_t.

Transfer losses of excretal phosphorus

Excretal P transfer losses from the main grazing area (border strips) to stock camps (border crutches)

Table 2. *Equations used in the phosphorus transfer model for calculating phosphorus transfer gains and losses in camp and non-camp areas respectively (adapted from Saggar* et al. *1988, 1990*a, b)

Symbol	Equation
HPc	HDMc × [HP]c
HPcf	HPc × % proportion of camp areas (Sc%) on 1 ha basis*
HPnc	HDMnc × [HP]nc
HPncf	HPnc × % proportion of non-camp areas (Snc%) on 1 ha basis*
APcf	HPcf × % pasture utilization (PU%)†
APncf	HPncf × % PU
ΣAP	APcf + APncf
ΣFaP	ΣAP × 0·9‡
FaPc	ΣFaP × % proportion of faecal DM (Dc%) returned to camp areas.
FaPnc	ΣFaP − FaPc
Pgc	FaPc − APcf
Pgc%	(Pgc ÷ APcf) × 100
Plnc	APncf − FaPnc
Plnc%	(Plnc ÷ APncf) × 100

* Proportions of camp and non-camp areas of 18% and 82% respectively.
† Pasture utilization of 80%.
‡ A constant representing 90% of animal P intake returned as excretal P and 100% of this excretal P present in faeces (Barrow & Lambourne 1962; During 1984).

were estimated by using the P transfer model proposed by Saggar *et al.* (1988, 1990*a*, *b*) assuming that 90% of the P consumed by grazing animals was returned in excreta and almost all (> 99%) of the excretal P was present in faeces (Burrow & Lambourne 1962). The parameters and equations used in the P transfer model are presented in Tables 1 and 2.

Excretal P transfer losses to stock camps were calculated from the difference between the estimated amounts of faecal P returns and pasture herbage P uptake in camp areas.

Determination of pasture yield and pasture herbage phosphorus content in camp and non-camp areas

Annual pasture dry matter (DM) production from the non-camp area of the three SP treatments over a 35-year period was determined by harvesting pasture herbage from the border strip of each replicate of the three treatments at 1–1·5 month intervals, using two 4 m^2 enclosure cages (Nguyen *et al.* 1989). In addition, DM production from the border crutches (Fig. 1) of each replicate was determined during the 1983/84 growing season by using one smaller (0·75 × 1 m) enclosure cage for each replicate. Herbage in each cage on the border crutches was cut on the same day as the pasture herbage from cages on the border strips. Harvested herbage was oven-dried at 60 °C for 48 h before DM determination. Subsamples of oven-dried herbage were finely ground (< 1 mm) before P analysis (Quin & Woods 1976). Uptake of P in pasture herbage (kg/ha) was calculated as the product of mean annual DM yield (kg/ha) and pasture herbage P content (%). Data on DM production and pasture P uptake in non-camp areas over a 35-year period have been published (Nguyen *et al.* 1989).

Finely ground (< 0·15 mm) samples of herbage, harvested from the border strips of all treatments at the end of the 1984/85 growing season (May 1985) were also analysed for P_t and P_i contents (Walker & Adams 1958).

Faecal sampling for assessing faecal distribution

Amounts of faecal DM return to camp and non-camp areas of replicates 1, 2 and 3 of each treatment during the 1983/84 period were assessed during spring (September 1983), summer (January 1984), autumn (March 1984) and early winter (April/May 1984). Prior to sheep entering each of these replicates, ten random circles (1000 mm diameter) from each of the camp and non-camp areas were selected. The centre of each circle was marked with a wooden peg, and visible faeces that had been deposited from previous grazings within the area of the circle were removed using a hand-held leaf rake. Subsequently, after these replicates had been grazed by sheep, faeces from within each circle were collected, oven-dried at 105 °C for 48 h and then weighed for DM determination.

Faecal sampling from determining phosphorus content

Faecal samples collected for DM distribution were not suitable for analysing P content because of possible soil contamination by animal trampling and/or P leaching losses from faecal materials during the period between deposition and sampling (Rowarth *et al.* 1985). Thus only faecal materials freshly deposited on the sampling date in May 1984 were collected from each replicate of all treatments. These faecal materials were dried at 60 °C for 48 h and ground (< 0·15 mm) before analysis for P_t and P_i (Walker & Adams 1958).

Determination of root dry matter and root phosphorus content

Composite soil samples of four cores (100 mm diameter) were collected to a depth of 0–600 mm at intervals of 0–75, 75–150, 150–300, 300–450 and 450–600 mm from the border strip and border crutch of each replicate of the three treatments and on four occasions in spring (October 1985), summer (February 1986), autumn (April 1986) and winter (June 1986). These samples were passed through a series of sieves with mesh sizes of 2, 1, 0·5, 0·25 and 0·125 mm. Root materials from each composite soil sample retained by these sieves were combined, washed free of soil with distilled water and oven-dried at 60 °C for 48 h before weighing for DM determination. They were then ground (< 0·15 mm) for P_t and P_i determination.

Soil sampling for chemical analyses

Composite soil samples of ten cores (25 mm diameter) from each depth (0–75, 75–150, 150–225 and 225–300 mm) of the border strip and border crutch of each treatment were collected in July 1987 before SP application. Subsamples were air-dried and finely ground (< 0·15 mm) before analysis for P_t and P_i.

Soil P_o was calculated as the difference between soil P_t and P_i (Walker & Adams 1958). Soil samples which had been collected from the same four soil depths in 1952, at the beginning of this long-term grazing experiment, were also retrieved from storage (at room temperature) to determine P_t and P_i contents.

Determination of soil bulk density

Four replicates of soil samples from each border crutch and border strip of one randomly selected replicate of each treatment were collected in July 1987 at 75 mm intervals to a depth of 300 mm using a percussion auger (43 mm diameter). The bulk density of each soil depth in border strips or border crutches was determined as the ratio of oven-dried (105 °C for 72 h) weight (g) of soil over the total volume (cm^3) of soil collected in the auger. The measured soil P concentrations (μg/g) in both border strips and border crutches were corrected for soil bulk density (g/cm^3) to calculate the amounts (kg/ha) of soil P in these areas.

Determination of the proportion of applied P recovered in the soil–plant–animal systems

The proportion (%) of applied P recovered in soils after 35 years of SP applications was determined as the proportion of the difference in soil P_t in the major rooting zone (0–300 mm) between the SP and control treatments over the cumulative amount of P applied over a 35-year period.

The proportion of applied P recovered in the soil–plant–animal systems in SP-treated pastures was determined using the following equation:

$$P\ \text{recovery} = [(\Delta_2 + A_2 + P_2 + R_2) - (\Delta_1 + A_1 + P_1 + R_1)] \div [(WP + Pf) - WP] \times 100 \quad (1)$$

or

$$P\ \text{recovery} = [(\Delta_2 + A_2 + P_2 + R_2 - \Delta_1 - A_1 - P_1 - R_1) \div Pf] \times 100 \quad (2)$$

where P recovery was the recovery (%) of applied P in soil–plant–animal systems in SP-treated pastures; Δ_2, A_2, P_2 and R_2 represented various P pools in SP treatments: Δ_2 was an annual change (kg P/ha) in total soil P, while A_2, P_2 and R_2 were annual P removal in animal products and animal P returned to soil in plant litter and root residues respectively; Δ_1, A_1, P_1 and R_1 represented the above parameters in the same order for the control treatment; WP was the annual P release from soil weathering and Pf was the amount of annual P input from SP fertilizer application.

Statistical analyses

Analysis of variance of soil chemical data, faecal P concentration and annual herbage P uptake was done using the Statistical Analysis System (1987) in accordance with the randomized complete block design of the experiment. Root P data and total root DM production in the 0–600 mm depth were analysed using GENSTAT (Genstat 5 Committee 1987), with sampling dates as split plots and SP treatments as main plots. Sampling areas and sampling dates were treated as split plots in statistical analyses of data from faecal DM deposition, pasture DM production and pasture P status in camp and non-camp areas. Sampling dates and soil depths were treated as split plots in the analysis of variance for the distribution of root DM with depth over a 1-year period. $P < 0.05$ was the minimum acceptable level of significance.

RESULTS AND DISCUSSION

Animal phosphorus intake

Based on the assumption that 80% of herbage P uptake was consumed by grazing animals with an average PU of 80% (Nguyen 1990), annual amounts of animal P intake from pastures with 0, 188 and 376 kg SP/ha per year treatments were estimated to be 6·5, 23·3 and 35·7 kg P/ha respectively. Annual amounts of herbage P uptake in 0, 188 and 376 kg SP/ha treatments were reported to be 8·1, 29·1 and 44·6 kg P/ha respectively (Nguyen *et al.* 1989).

Annual returns of plant litter phosphorus

Since 80% of herbage P uptake was considered to be utilized by grazing animals, the remaining 20%, estimated as 1·6, 5·8 and 8·9 kg P/ha per year, was returned as plant litter P to soils in grazed pastures with 0, 188 and 376 kg SP/ha applied per year respectively.

Approximately 44–62% of herbage P content was

Table 3. *Total phosphorus (P_t), inorganic phosphorus (P_i) and phosphorus uptake (obtained from Nguyen* et al. *1989) in pasture herbage, annual animal phosphorus intake and annual herbage phosphorus residue in pasture receiving three different superphosphate (SP) inputs*

SP input (kg/ha per year)	Herbage P: P_t (% DM)	Herbage P: P_i (% DM)	Herbage P: P_i (% P_t)	Herbage P: P uptake (kg/ha per year)	Animal P intake (kg/ha per year)	Herbage P residue: P_t (kg/ha per year)	Herbage P residue: P_i (kg/ha per year)
0	0·22	0·09	44	8·1	6·5	1·6	0·7
188	0·32	0·17	53	29·1	23·3	5·8	3·1
376	0·42	0·26	62	44·6	35·7	8·9	5·5
S.E. (6 D.F.)	0·006	0·003	—*	1·24	—	—	—

* — not applicable.

Table 4. *Amounts of faecal dry matter (kg DM/ha) deposited in camp and non-camp areas sampled at different times in grazed pastures receiving three different superphosphate (SP) inputs*

Sampling date	Camp SP input (kg/ha per year) 0	Camp 188	Camp 376	Non-camp SP input (kg/ha per year) 0	Non-camp 188	Non-camp 376
September (spring) 1983	267 (48)*	494 (89)	533 (96)	88 (72)	238 (195)	268 (220)
January (summer) 1984	644 (116)	1044 (188)	1317 (237)	307 (252)	839 (688)	891 (731)
March (autumn) 1984	178 (32)	261 (47)	261 (47)	83 (68)	174 (143)	199 (163)
April/May (early winter) 1984	417 (75)	655 (118)	683 (123)	152 (125)	437 (358)	488 (400)
	D.F.	S.E.				
Area	6	15·9				
Treatment	4	22·9				
Sampling date	522	25·3				

* Values in parentheses indicate amount of faecal DM after correction for the proportions of camp (18%) and non-camp (82%) areas in grazed pastures.

Table 5. *Proportions (%) of faecal dry matter returned to camp and non-camp areas sampled at different times in grazed pastures with three different superphosphate (SP) inputs*

Sampling date	Camp SP input (kg/ha per year) 0	Camp 188	Camp 376	Non-camp SP input (kg/ha per year) 0	Non-camp 188	Non-camp 376
September (spring) 1983	40·0	31·3	30·4	60·0	68·7	69·6
January (summer) 1984	31·5	21·5	24·5	68·5	78·5	75·5
March (autumn) 1984	32·0	24·7	22·4	68·0	75·3	77·6
April–May (early winter) 1984	37·5	24·8	23·5	62·5	75·2	76·5
Mean ± S.E.	35·3 ± 2·09	25·6 ± 2·06	25·2 ± 1·79	64·8 ± 2·09	74·4 ± 2·06	74·8 ± 1·79

found to be in the P_i form (Table 3), in agreement with the results of other workers (Bromfield & Jones 1972; Playne 1976). Thus *c.* 0·7–5·5 kg P/ha per year from plant litter would be considered to enter the soil P_i pool.

Annual phosphorus removal in animal products

Amounts of annual P removal in animal products from ewe-grazed pastures which had been treated with 0, 188 and 376 kg SP/ha per year were 0·6, 1·2 and 1·4 kg P/ha respectively. These estimates were based on the assumption that each ewe causes the total annual removal of 0·074 kg P, which is the sum of the P removal in 4·4 kg wool ($6{\cdot}6 \times 10^{-4}$ kg P) with a P content of 0·014–0·016% (Blair 1983; Grace 1983) and annual P removal in liveweight (0·073 kg P) resulting from a replacement of a 55 kg ewe at 0·6% P (Grace 1983) every 4·5 years.

Phosphorus removal in animal products was considered to represent a relatively small P output from the soil–plant–animal system in the different SP treatments, since it accounted for < 10% of the P intake by grazing animals (Table 3).

Excretal phosphorus output and phosphorus lost by excretal transfer

Since < 1·5 kg P/ha per year of the animal P intake was retained in wool and animal tissues, *c.* 5·9, 22·1 and 34·3 kg P/ha per year ingested by grazing animals was returned as excreta to pastures in the 0, 188 and 376 kg SP/ha per year treatments respectively. This excreta was subject to transfer losses, since a significantly higher amount of faecal DM was returned to camp than to non-camp areas (Table 4).

Since camp areas accounted for 18% of the total grazing area in each replicate, the proportions of faecal DM returned to camp areas of the 0, 188 and 376 kg SP/ha treatments were estimated to be 35, 26 and 25% respectively (Table 5). The higher excretal transfer observed in the control treatment, particularly during the spring and late autumn-early winter periods (Table 5), could be due to the low SR (8 ewes/ha) in this treatment, as other workers (Taylor 1980; Morton & Baird 1990) have shown that with a low SR, animals tend to spend more time in camp areas. This effect is likely to be more pronounced when pasture growth is active in the spring and autumn periods.

Pasture DM production and pasture P uptake on a kg P/ha basis were found to be significantly higher in camp than in non-camp areas (Table 6). This could be due to higher excreta deposition and hence higher returns of excretal nutrients such as nitrogen (N), potassium (K), P and S to the former areas (Nguyen & Goh 1988; Nguyen 1990).

Sheep faeces deposited in pastures with annual SP applications of 0, 188 and 376 kg/ha were found to contain between 0·52 and 1·07% P, and *c.* 56–82% of faecal P was present in the inorganic fraction (Table 7). The observed increase in faecal P_i concentration and faecal P_i fraction with an increase in the rate of SP application (Table 7) probably reflects the increase in herbage P uptake and hence animal P intake (Table 3). This is supported by results obtained from other studies, which showed that faecal P_i concentration (Floate & Torrance 1970; Rowarth *et al.* 1988) and the faecal P_i fraction (Bromfield 1961; Barrow & Lambourne 1962) increased with increasing herbage P concentration.

The transfer of excretal P to stock camps could account for the annual P gains in camp areas, where the annual estimated amounts of faecal P returns (FaPc; Table 8) were higher than the annual amounts of animal P intake (APcf; Table 8) from these areas. These P gains in camp areas were found to be at the expense of P losses in non-camp areas, where the amounts of faecal P returns (FaPnc; Table 8) were lower than those ingested by grazing animals from these areas (APncf; Table 8). The proportions of excretal P that were subjected to transfer losses from non-camp to stock camp areas in pastures with annual SP applications of 0, 188 and 376 kg/ha were estimated to be 22·4, 13·9 and 13·7% respectively (Plnc %: Tables 1, 2 and 8). Thus the estimated amounts of excretal P transfer to stock camps in the 0, 188 and 376 kg SP/ha treatments were 1·3, 3·1 and 4·7 kg P/ha per year respectively. These transfer losses in the 188 and 376 kg SP/ha treatments were similar to those (12·5%) quoted in the New Zealand MAF P and S models (Cornforth & Sinclair 1984; Sinclair & Saunders 1984) for pastures under intensive rotational grazing systems where the ratio of SR:potential carrying capacity is > 0·75 (Metherell & Morrison 1984). However, the estimated transfer loss in the control pastures receiving no SP applications was substantially higher, probably because of the low SR (8 ewes/ha) in these pastures.

In the development of the P transfer model for estimating excretal P transfer to stock camps (Table 8), border levees being on top of a mound of soil (Fig. 1) and not in direct contact with irrigation water were not considered to be the major source or sink of P in nutrient cycling in grazed pastures because of their lack of adequate soil moisture for pasture growth (Nguyen 1990). In addition, very limited camping occurred on border levees because of the fence running along the top of these levees.

The above assumption is justified, since Nguyen (1990) in another study on border strip, irrigated, grazed pastures with a similar soil type observed that P transfer losses to stock camps with the inclusion of border levees as a source or sink of P in grazed pastures were similar to those where border levees were not considered to be involved in P cycling in grazed pastures.

In addition to the excretal P transfer to stock camps, some excretal P was also lost from the grazing trial to either the off-trial area (Fig. 1) whenever grazing animals were shifted from one replicate to another, or to the shearing shed for crutching, shearing or drenching (i.e. off-farm losses). Since the period of

Table 6. *Dry matter production (kg DM/ha), herbage phosphorus concentration (% P in DM) and phosphorus uptake (kg P/ha) from camp and non-camp areas sampled at different times in grazed pastures with three different superphosphate (SP) inputs*

Sampling time (month)	Camp DM (kg/ha)	Camp P concentration (%)	Camp P uptake (kg/ha)	Camp P uptake (kg on area basis)*	Non-camp DM (kg/ha)	Non-camp P concentration (%)	Non-camp P uptake (kg/ha)	Non-camp P uptake (kg on area basis)*
			0 kg/ha SP input					
September	146	0·21	0·31	0·06	73	0·19	0·14	0·11
October	600	0·25	1·50	0·27	405	0·23	0·93	0·76
November	780	0·24	1·87	0·34	520	0·26	1·35	1·11
December	974	0·27	2·63	0·47	695	0·24	1·67	1·37
January	1015	0·24	2·44	0·44	620	0·21	1·30	1·07
March	765	0·28	2·14	0·39	450	0·27	1·21	0·99
April	627	0·20	1·25	0·23	375	0·22	0·83	0·68
May	180	0·21	0·38	0·07	114	0·16	0·18	0·15
Total	5087	—	12·5	2·3	3252	—	7·61	6·2
			188 kg/ha SP input					
September	595	0·37	2·20	0·40	384	0·35	1·34	1·10
October	2356	0·45	10·60	1·91	1885	0·39	7·35	6·03
November	1733	0·38	6·59	1·19	1284	0·40	5·14	4·21
December	2083	0·37	7·71	1·39	1488	0·35	5·21	4·27
January	3607	0·35	12·62	2·27	2060	0·33	6·80	5·57
March	1473	0·35	5·16	0·93	1133	0·33	3·74	3·07
April	1110	0·29	3·22	0·58	926	0·31	2·87	2·35
May	139	0·30	0·42	0·08	110	0·31	0·34	0·28
Total	13096	—	48·5	8·8	9270	—	32·8	26·9
			376 kg/ha SP input					
September	683	0·50	3·42	0·61	488	0·45	2·20	1·81
October	3145	0·48	15·10	2·72	2030	0·49	9·95	8·16
November	1750	0·53	9·28	1·67	1400	0·50	7·00	5·74
December	2180	0·47	10·25	1·84	1615	0·48	7·75	6·36
January	4140	0·38	15·73	2·83	2325	0·41	9·53	7·82
March	1955	0·45	8·80	1·58	1448	0·46	6·66	5·46
April	1465	0·47	6·88	1·24	1045	0·43	4·49	3·68
May	225	0·43	0·97	0·18	180	0·39	0·70	0·58
Total	15543	—	70·4	12·7	10530	—	48·3	39·6

	DM D.F.	DM S.E.	P concentration D.F.	P concentration S.E.	P uptake (kg/ha) D.F.	P uptake (kg/ha) S.E.
Area (camp *v* non-camp)	9	46·9	9	0·003	9	0·125
SP inputs	6	72·1	6	0·007	6	0·130
Sampling month	126	126·1	126	0·011	126	0·198

* Based on an area basis of 18% (camp) and 82% (non-camp) of total grazing area.

time in which the grazing animals were involved in these activities accounted for *c.* 10 days per year, potential amounts of off-trial and off-farm P losses were estimated to be 2·7% of the total excreta P output per year (Table 8) or 0·15, 0·61 and 0·94 kg P/ha per year for pastures with annual SP applications of 0, 188 and 376 kg/ha respectively.

Phosphorus uptake in plant roots and phosphorus returns to soils from root residues

Most of the pasture root DM production to a depth of 600 mm was in the top 300 mm (Table 9). Over 95% of root DM production was in the topsoil (0–75 mm). This suggests that soil sampling to a depth of 300 mm for P and S analyses was adequate when assessing soil P status for predicting pasture P

[T]able 7. *Faecal total (P_t) and inorganic phosphorus (P_i) concentrations and the proportion of faecal phosphorus [a]s inorganic fraction in sheep faeces collected from pastures with three different superphosphate (SP) inputs*

SP input (kg/ha per year)	Faecal P_t (% P in DM)	Faecal P_i Concentration (% P in DM)	Faecal P_i Proportion (% faecal P_t)
0	0·52	0·29	56
188	0·80	0·60	75
376	1·07	0·88	82
S.E. (6 D.F.)	0·003	0·003	—

[T]able 8. *Amounts (kg P/ha) of herbage phosphorus utilized by grazing sheep and returned as faeces to camp and [n]on-camp areas sampled at different times in grazed pastures with three different superphosphate (SP) inputs*

Sampling time (month)	Animal P intake* Camp (APcf)	Animal P intake* Non-camp (APncf)	Total ΣAP	Faecal P* (ΣFaP)	Proportion (%) of faecal DM in camp areas (Dc)†	Faecal P to* Camp (FaPc)	Faecal P to* Non-camp FaPnc)
			0 kg/ha SP input				
September	0·05	0·09	0·14	0·13	40·0	0·05	0·08
October	0·22	0·61	0·83	0·75	40·0	0·30	0·45
November	0·27	0·89	1·16	1·04	40·0	0·42	0·62
December	0·38	1·10	1·48	1·33	40·0	0·53	0·80
January	0·35	0·86	1·21	1·09	31·5	0·34	0·75
March	0·31	0·79	1·10	0·99	32·0	0·32	0·67
April	0·18	0·54	0·72	0·65	37·5	0·24	0·41
May	0·06	0·12	0·18	0·16	37·5	0·06	0·10
Total	1·82	5·00	6·82	6·14	—	2·26	3·88
			188 kg/ha SP input				
September	0·32	0·88	1·20	1·08	31·3	0·34	0·74
October	1·53	4·82	6·35	5·72	31·3	1·79	3·93
November	0·95	3·37	4·32	3·89	31·3	1·22	2·67
December	1·11	3·42	4·53	4·08	31·3	1·28	2·80
January	1·82	4·46	6·28	5·65	21·5	1·22	4·43
March	0·74	2·46	3·20	2·88	24·7	0·71	2·17
April	0·46	1·88	2·34	2·11	24·8	0·52	1·59
May	0·06	0·22	0·28	0·25	24·8	0·06	0·19
Total	6·99	21·51	28·50	25·66	—	7·14	18·52
			376 kg/ha SP input				
September	0·49	1·45	1·94	1·74	30·4	0·53	1·21
October	2·18	6·53	8·71	7·84	30·4	2·38	5·46
November	1·34	4·59	5·93	5·34	30·4	1·62	3·72
December	1·47	5·09	6·56	5·90	30·4	1·79	4·11
January	2·26	6·26	8·52	7·66	24·5	1·88	5·78
March	1·26	4·37	5·63	5·07	22·4	1·13	3·94
April	0·99	2·94	3·93	3·54	23·5	0·83	2·71
May	0·14	0·46	0·60	0·54	23·5	0·13	0·41
Total	10·13	31·69	41·82	37·63	—	10·29	27·34

[*C]alculated using parameters and equations from Tables 1 and 2. See Table 1 for abbreviations.
[†O]btained from Table 5.

[req]uirement. Root DM production was higher in [spr]ing and autumn than in summer or winter (Table [...]). This seasonal pattern was similar to that shown by pasture herbage (Nguyen *et al.* 1989). Lower root DM production in the SP-treated pastures compared with the control (Table 9) was attributed to the higher

Table 9. *Root dry matter production (kg DM/ha) and its proportion (%) in soil samples from five different depths collected over four seasons (1985/86) from three superphosphate (SP) treatments*

Soil depth (mm)	SP input (kg/ha per year)											
	Spring (October 1985)			Summer (February 1986)			Autumn (April 1986)			Winter (June 1986)		
	0	188	376	0	188	376	0	188	376	0	188	376
0–75	7296 (83)*	7725 (85)	6579 (85)	8190 (82)	6110 (84)	5314 (83)	7816 (85)	8210 (85)	5635 (86)	7493 (85)	6890 (84)	5973 (85)
75–150	972 (11)	846 (9)	648 (8)	1040 (10)	600 (8)	645 (10)	784 (8·5)	835 (9)	448 (7)	781 (9)	825 (10)	628 (9)
150–300	294 (3)	258 (3)	310 (4)	425 (4)	336 (5)	289 (45)	385 (4)	390 (4)	276 (4)	294 (3)	289 (3·5)	244 (3)
300–450	170 (2)	178 (2)	133 (2)	212 (2)	184 (3)	138 (2)	125 (1)	164 (1·5)	136 (2)	171 (2)	163 (2)	115 (1·5)
450–600	68 (1)	66 (1)	50 (1)	65 (1)	57 (1)	39 (0·5)	51 (0·5)	60 (0·5)	51 (1)	45 (0·5)	54 (0·5)	45 (0·5)
Total	8800	9073	7720	9932	7287	6425	9161	9659	6546	8784	8221	7005

	Root DM distribution with depth		Total root DM production	
	D.F.	S.E.	D.F.	S.E.
Sampling date	171	31·2	27	148·3
SP treatments	6	15·7	6	43·3
Depth	171	34·8	—	—

* Proportion (%) of root DM production (0–600 mm depth) present at each depth.

Table 10. *Effects of three different superphosphate (SP) treatments and four sampling seasons (1985/86) on root phosphorus concentration and root phosphorus uptake*

	SP input (kg/ha per year)					
	Root P concentration (% P in DM)			Root P uptake (kg P/ha)		
Sampling date	0	188	376	0	188	376
Spring (October 1985)	0·13	0·20	0·24	11·4	18·1	18·5
Summer (February 1986)	0·12	0·19	0·21	11·9	13·8	13·5
Autumn (April 1986)	0·12	0·15	0·18	11·0	14·5	11·8
Winter (June 1986)	0·12	0·20	0·24	10·5	16·4	16·8
Mean	0·12	0·19	0·22	11·2	15·7	15·2

	D.F.	S.E.	D.F.	S.E.
Sampling date	27	0·012	27	0·19
Treatment	6	0·009	6	0·12
Treatment × sampling date	27	0·014	27	0·30

Table 11. *Amounts (μg P/g) of soil total phosphorus at four different soil depths of the main grazing area (border strips) prior to (1952) and 35 years after (1987) receiving three different annual superphosphate (SP) inputs, including soil bulk density and values of soil total phosphorus (kg/ha) after correction for bulk density*

Soil sampling depth (mm)	Soil bulk density (g/cm³)	1952	1987 SP input (kg/ha per year) 0	188	376	S.E. (6 D.F.)
		Uncorrected for soil bulk density				
0–75		710 ± 13*	750	930	1150	10·1
75–150		670 ± 10	715	885	1010	12·1
150–225		600 ± 16	625	680	740	11·5
225–300		505 ± 11	500	500	510	8·7
		Corrected for soil bulk density				
0–75	1·01 (3·1)†	538	568	704	871	9·2
75–150	1·11 (2·5)	558	595	737	841	10·1
150–225	1·23 (1·5)	554	577	627	683	8·7
225–300	1·24 (2·2)	470	465	465	474	8·7
Soil P_t (0–300 mm)		2120	2205	2533	2869	21·7
Changes in P_t (0–300 mm)						
Over a 35-year period		—	85	413	749	—
Annually		—	2·4	11·8	21·4	—

* Mean ± S.E.
† Coefficient of variation in parentheses.

DM production of the plant tops at the expense of the root DM production in the former.

Phosphorus content in pasture roots was found to reflect the previous SP top-dressing history (Table 10) and was higher in SP-treated pastures that in the control. A similar pattern was observed for herbage P content (Table 3).

The mean annual P uptakes by pasture roots in the 0, 188 and 376 kg/ha treatments were 11·2, 15·7 and 15·2 kg P/ha respectively (Table 10). Published data suggests that there is a 50% turnover of root biomass each year in frequently grazed pastures (Middleton & Smith 1978; Parfitt 1980). Thus the annual root P residues in the 0, 188 and 376 kg SP/ha treatments were estimated to be 5·6, 7·9 and 7·6 kg P/ha. Since over 98% of root P in these SP treatments was present as P_o (data not presented), almost all root P residues were considered to be assimilated into the soil P_o pool.

Changes in soil phosphorus over long-term pasture development and superphosphate applications

Significant amounts of soil P_t accumulated to a depth of 0–225 mm in all treatments after 35 years of pasture development with or without annual SP applications (Table 11). Changes in soil P_i and P_o with time in the topsoil (0–75 mm) over a 35-year period have also been reported (Condron & Goh 1989; Nguyen *et al.* 1989). Except for the 7-year period following lime application in 1972, the accumulation of soil P_t in all treatments was linear (Quin & Rickard 1981; Nguyen *et al.* 1989). Annual rates of P accumulation in the major plant rooting zone (0–300 mm) in the control, 188 and 376 kg SP/ha treatments measured by the increase in soil P_t (kg P/ha) after 35 years of the grazing experiment, relative to that in the soil at the beginning (1952) of this experiment, were estimated to be 2·4, 11·8 and 21·4 kg P/ha per year respectively (Table 11). Soil bulk density at each sampling depth of the main grazing area (border strips) was found to be unaffected by different SP treatments (data not presented) and unchanged over 35 years (D. S. Rickard & B. F. Quin, unpublished). Thus only data of the mean of soil bulk density for these SP treatments (Table 11) was used in the conversion of soil P_t from μg P/g to kg P/ha basis.

The accumulated soil P_t in the main grazing area (non-camp soil) of the control treatment was attributed to an increase in the soil P_o fraction (Table 12), while SP applications enhanced the accumulation of both soil P_o and P_i fractions and increased the ratio of soil P_i : soil P_o in the 0–225 mm depth (Table 12). The higher soil P_i : P_o in SP-treated pastures was attributed to a higher accumulation of soil P_i relative to soil P_o in these pastures (Table 12).

The higher accumulation of both soil P_o and P_i fractions even to the depth of 150–225 mm (Table 12) in SP treatments relative to that in the control, suggests some leaching of either or both P fractions from soil P_o and P_i pools in the topsoil (0–75 mm), applied P fertilizers, and faecal P (Bromfield & Jones 1970; Campbell & Racz 1975; Schoenau & Bettany 1987). This could also be due to a number of other factors such as the preferential flow of dissolved particulate soil organic matter or faecal materials down soil macropores by irrigation water (Kanchanasut *et al.* 1978; Kanchanasut & Scotter 1982; Blair 1983), the distribution and decomposition of plant root P residues (Clark *et al.* 1980) and the incorporation and transportation of faecal P, plant litter P and soil P from the 0–75 mm depth to the lower soil depth (150–225 mm) by earthworms and other soil fauna (Syers & Springett 1984). These suggestions were further supported by the finding that, compared with the control treatment, higher DM production in SP-treated pastures (Nguyen *et al.* 1989) allowed higher SR to be carried and hence there was higher return of animal excreta and plant litter (Tables 3, 4 and 8). Consequently the amounts of plant and faecal P residues that were likely to be exposed to earthworm activity, leaching or preferential flow from soil surface to a depth of 225 mm would be higher in SP-treated pastures than in the control treatment. This P movement is unlikely to occur beyond a depth of 225 mm since there was no significant difference in soil P_t, soil P_o or soil P_i at 225–300 mm between different treatments.

Accumulation of soil phosphorus in camp areas

Amounts (μg P/g) of soil P_t, soil P_i and soil P_o within 0–225 mm soil depth of the camp areas were significantly higher in SP-treated pastures than in the control (Table 12), probably reflecting the higher faecal P return in the SP treatments (Table 8). The presence of higher soil P_i at the 0–225 mm depth in the camp area of the 376 kg SP/ha treatment, relative to that in the 188 kg SP/ha treatment (Table 12) was in agreement with the result obtained from the non-camp soil (Table 12). This showed that an increase in annual SP applications from 188 to 376 kg SP/ha enhanced the accumulation of soil P_i in both camp and non-camp areas. This was attributed to the oversupply of P input from the 376 kg SP/ha treatment for pasture maintenance P requirement (Nguyen *et al.* 1989), leading to an accumulation of P_i in labile and Ca-bound forms (e.g. $NaHCO_3$-extractable P and HCl-extractable P) (Condron & Goh 1989). The mean of soil bulk density for each sampling depth in camp areas (border crutches) of the three SP treatments (Table 13) was used in the conversion of soil P_t from μg P/g to kg P/ha, since it was found that these treatments did not significantly affect soil bulk density in camp areas (data not presented).

Amounts of soil P_t, soil P_i and soil P_o at the depth of 225–300 mm in the camp areas were found to be unaffected by different SP inputs (Tables 12 and 13). In addition, these amounts were similar to those in non-camp areas at the same soil depth (Table 11). The results suggest that leaching of both P_i and P_o beyond 225 mm depth in this low phosphate-retaining soil is unlikely to occur not only in the non-camp but also in camp areas, although the latter received higher faecal P returns (Table 8) and showed higher soil P_i and P_o in the top 225 mm soil depth than that of non-camp areas (Table 12).

By comparing the amounts (kg P/ha) of soil P_t within the major rooting zone (0–300 mm) in camp and non-camp areas, the estimated amounts of P gains in camp areas of the control, 188 and 376 kg SP/ha treatments due to excretal P transfer (Table 8)

Table 12. *Amounts (μg P/g) of soil inorganic (P_i) and organic (P_o)* phosphorus and their ratios at four different soil depths of the main grazing area (border strips) prior to (1952) and after 35 years (1987) and in camp area (border crutches) in pastures after 35 years (1987) of three different annual superphosphate (SP) inputs*

Soil sampling depth (mm)	1952			1987: SP input (kg ha per year): 0				188				376				S.E. (6 D.F.)		
	P_i	P_o	P_i:P_o	P_t	P_i	P_o	P_i:P_o	P_t	P_i	P_o	P_i:P_o	P_t	P_i	P_o	P_i:P_o	P_t	P_i	P_o
Main grazing area																		
0–75	260	450	0·58	—†	220	530	0·42	—	325	605	0·54	—	540	610	0·89	—	8·7	13·0
75–150	210	460	0·46	—	195	520	0·38	—	275	610	0·45	—	365	645	0·57	—	7·2	11·6
150–225	195	430	0·45	—	185	440	0·42	—	220	460	0·48	—	280	460	0·61	—	6·6	10·4
225–300	195	305	0·64	—	190	310	0·61	—	205	295	0·69	—	210	330	0·70	—	4·3	6·6
Camp area																		
0–75				920	300	620	0·48	1400	675	725	0·93	1670	950	720	1·32	36·1	26·0	27·5
75–150				820	280	540	0·52	1150	490	660	0·74	1375	720	655	1·10	27·5	23·1	24·6
150–225				710	250	460	0·54	825	310	515	0·60	895	350	545	0·64	18·8	17·6	17·3
225–300				515	195	320	0·61	520	195	325	0·60	525	200	325	0·62	14·4	14·4	11·6

* P_o estimated as the difference between P_t and P_i.

† — not applicable as P_t values for main grazing area were presented in Table 11.

Table 13. *Amounts of soil total phosphorus (kg P/ha) corrected for soil bulk density at four different soil depths in camp areas of grazed pastures after 35 years with three different superphosphate (SP) inputs*

Soil sampling depth (mm)	Soil bulk density (g/cm^3)	SP input (kg/ha per year)			S.E. (6 D.F.)
		0	188	376	
0–75	1·01 (2·7)*	697	1061	1265	20·2
75–150	1·01 (2·4)	621	871	1042	18·8
150–225	1·15 (1·9)	612	712	772	14·4
225–300	1·19 (1·5)	460	464	469	10·1
Total (0–300)	—	2390	3108	3548	24·6

* Coefficient of variation (%) in parentheses.

Table 14. *Amounts of soil total phosphorus (P_t) at 0–300 mm depth in camp and non-camp areas of three different superphosphate (SP) treatments before and after correction for the proportion of these areas in each treatment and estimated transfer gains of phosphorus in camp soils*

SP input (kg/ha per year)	Soil P_t: kg P/ha basis*		Soil P_t: kg P on area basis†			P transfer‡ (kg/ha per year)
	Camp P_{t1}	Non-camp P_{t2}	Camp P_{t11}	Non-camp P_{t12}	Total $(P_{t11}+P_{t12})$	
0	2390	2205	430	1808	2238	0·9
188	3108	2533	559	2077	2636	2·9
376	3548	2869	639	2353	2992	3·5

* Obtained from Tables 11 and 13.
† Based on camp and non-camp areas occupying 18 and 82% of the total grazing area respectively.
‡ Estimated as the proportion of the difference between $(P_{t11}+P_{t12})$ and P_{t2} over a period of 35 years.

were 0·9, 2·9 and 3·5 kg P/ha per year (Table 14) respectively. These estimates based on soil P_t were similar to those (1·3, 3·1 and 4·7 kg P/ha per year) estimated earlier using the P transfer model.

Recovery of applied phosphorus in soil–plant–animal systems

The proportion of applied P recovered in the major soil rooting zone (0–300 mm depth) was found to be unaffected by the rate of SP application and was estimated to range from 53·6 to 54·2% (Table 15). By including the annual amounts of P cycling in both plant and animal P pools, the proportions of applied P recovered in the soil–plant–animal system in the 188 and 376 kg SP/ha treatments were estimated to be 94·3 and 83·1% respectively (Table 15). These results suggested that 5·7–16·9% of the applied P was not recovered. Approximately 11·7–12·6% of the applied P was estimated to be transferred to stock camps or lost to off-trial and off-farm sites (Table 15). Thus the remaining applied P that was not accounted for in the soil–plant–animal system (i.e. 0–5·2% of applied P; Table 15) was postulated to be lost by lateral P transfer via the floating of dry faeces in irrigation water as it moved along the border strip from the headrace to the bottom of the border strip (Fig. 1). Close & Woods (1986) recorded a significant increase in P concentration in irrigation water as it moved from the top to the bottom of the border strip.

Practical implications

As shown in this study, long-term SP applications to irrigated, sheep-grazed pastures resulted in the accumulation of both soil P_i and P_o fractions to a depth of 225 mm. This accumulation was mainly from plant litter, sheep excreta, microbial P immobilization and adsorption of fertilizer P on soil colloids. The accumulated soil P may play a significant role in sustaining both pasture and animal production, especially after the withholding of SP applications. The degree of this P sustainability needs to be examined in the light of the increasing economic pressure on Australasian farmers to reduce fertilizer inputs.

Table 15. *Phosphorus pools and phosphorus inputs to and outputs from the soil–plant–animal system in grazed pastures with three different superphosphate treatments*

P pools, inputs and P outputs	Amounts of P (kg/ha) SP input (kg/ha per year) 0	188	376
Soil weathering	4	4	4
SP fertilizer			
per year basis	0	17·5	35
over 35 years	0	612·5	1225
Soil P pool	2205	2533	2869
Δ soil P due to pasture development and SP applications			
over 35 years	85	413	749
per year basis	2·4	11·8	21·4
Δ soil P over 35 years due to SP applications	—	328	664
Recovery (%) of applied P in soil	—	53·6	54·2
Herbage P pool	8·1	29·1	44·6
Animal P removal	0·6	1·2	1·4
Excretal P output	5·9	22·1	34·3
Excretal transfer to			
stock camps	1·3	3·1	4·7
off-farm and off-trial sites	0·2	0·6	0·9
total transfer losses	1·5	3·7 (12·6)*	5·6 (11·7)*
Plant P residue	1·6	5·8	8·9
Root P residue	5·6	7·9	7·6
P in soil–plant–animal pools	10·2	26·7	39·3
Recovery (%) of applied P in soil–plant–animal pools	—	94·3	83·1

* Excretal P transfer expressed as a percentage of applied P after taking into account the amount of excretal P transfer in the control treatment.

The accumulated soil P_o and P_i to a depth of 225 mm suggests that both P_i and P_o from SP fertilizer, plant residues, sheep faeces and soil P may be leached or transported by earthworms from the topsoil (0–75 mm depth) to lower soil depths (75–225 mm). However, this P movement was unlikely to occur beyond the major rooting zone (0–300 mm depth), and hence most of the applied P even at the highest rate of 376 kg SP/ha, which exceeded pasture plant P requirements, was accounted for in the soil–plant–animal system. Phosphorus losses in surface run-off were unlikely to be the major concern on these irrigated, sheep-grazed pastures since these losses accounted for < 6% of the applied P.

Grazing animals played a significant role in the recycling of P from pasture herbage to the soil via the return of sheep faeces. Sheep faeces consisted of both P_i and P_o fractions, and the proportion of faecal P as P_i increased with increasing SP inputs. The return of faecal P to the soil may play a significant role in providing P for pasture plants. Future studies on the decomposition of sheep faeces and the plant availability of P in sheep faeces of irrigated, grazed pastures are warranted.

Although grazing animals might play a major role in P recycling in grazed pastures, they were also responsible for a considerable amount (14–22% of excretal P) of P transfer as excreta from the main grazing areas to stock camps. This transfer may be reduced by management practices such as rotational grazing at high SRs. In addition, the efficiency of the utilization of P fertilizers in irrigated, sheep-grazed pastures can be improved when P fertilizers are applied only to the main grazing areas (border strips) and not to the entire grazing area which includes the animal camp (border crutches) area.

CONCLUSIONS

This study shows that long-term SP applications resulted in the accumulation of both soil P_i and P_o fractions. The accumulated soil P provided significant amounts of P for pasture requirements. Leaching losses of P from SP fertilizer, plant litter, root residues and sheep faeces were unlikely to occur beyond the major soil rooting zone (0–300 mm) in this low phosphate-retentive soil. Most of the applied P was accounted for in the border strip of the irrigated, sheep-grazed pastoral system. To improve the efficient use of applied P, SP should be applied to border strips (non-camp) and not to the entire grazing area. The excretal P transfer loss may be reduced by management practices which involve rotational grazing at high SRs.

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REVIEW ARTICLE

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Soluble organic nitrogen in agricultural soils

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Abstract The existence of soluble organic forms of N in rain and drainage waters has been known for many years, but these have not been generally regarded as significant pools of N in agricultural soils. We review the size and function of both soluble organic N extracted from soils (SON) and dissolved organic N present in soil solution and drainage waters (DON) in arable agricultural soils. SON is of the same order of magnitude as mineral N and of equal size in many cases; 20–30 kg SON-N ha^{-1} is present in a wide range of arable agricultural soils from England. Its dynamics are affected by mineralisation, immobilisation, leaching and plant uptake in the same way as those of mineral N, but its pool size is more constant than that of mineral N. DON can be sampled from soil solution using suction cups and collected in drainage waters. Significant amounts of DON are leached, but this comprises only about one-tenth of the SON extracted from the same soil. Leached DON may take with it nutrients, chelated or complexed metals and pesticides. SON/DON is clearly an important pool in N transformations and plant uptake, but there are still many gaps in our understanding.

Key words Dissolved organic nitrogen · Soluble organic nitrogen · Nitrogen transformations · Nitrogen loss · Leaching

Dedicated to Prof. K. Vlassak on the occasion of his 65th birthday

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Introduction

The forms of N present in the soil and lost in drainage have been the subject of research for many years. Lawes and Gilbert (1881) made gravimetric analyses of the organic and inorganic contents of lysimeter drainage waters after laboriously distilling >100-l samples. They reported that the amount of dissolved organic N (DON; N dissolved in soil solution and drainage water collected from agricultural land) leached was no more than 2 kg N ha^{-1} and mostly <1 kg N ha^{-1} out of a total leached of 50 kg N ha^{-1}, and that it was highly nitrogenous, with a mean C:N ratio of 2.6:1. They said that: "Respecting the nature of these nitrogenous organic bodies, and the part they possibly play in plant nutrition, very little is at present known." Russell and Richards (1919) reported analyses of N in rainwater. They calculated that rainwater deposited ca. 1.5 kg organic-N ha^{-1}, one-quarter of the total N deposited at the beginning of this century. Thus it was known almost 100 years ago that rain and drainage waters contained DON. What progress have we made? Fortunately labour-intensive distillation and gravimetric analysis is a thing of the past and we are beginning to understand the composition and dynamics of DON. This paper reviews that progress.

The plough layer of arable soils may contain >3000 kg N ha^{-1} (Stevenson 1982; Streeter and Barta 1988), but most of this is composed of a continuum of complex organic forms, which can be divided conceptually into a number of pools (Paul and Juma 1981). These pools may include organic N which is virtually inert (Hsieh 1992) as well as N present in the living bodies of the soil microbial biomass (SMB) (Jenkinson and Powlson 1976). Mineral N comprises only a small part of the total N in the soil (Harmsen and Kolenbrander 1965; Bremner 1965), usually about 1% in arable soils (Jarvis et al. 1996), except after fertiliser application. But mineral N cycles rapidly. It is supplied by the mineralisation of soil organic matter, as well as from fertiliser, manure and atmospheric deposition, and depleted by plant uptake and immobilisation by microorganisms, by denitrification and by leaching. Although microbial N comprises only 3–5% of total N in soil, mineralisation, immobilisation and denitrification are all microbially mediated processes.

The importance of NH_4^+-N and NO_3^--N in crop nutrition, and the environmental and possible health impacts of NO_3^- in groundwater has focused attention on the study of mineral N (Archer et al. 1992). However, a pool of soluble forms of organic N of equal size to mineral N (about 0.3–1% of total organic N in arable soils; Mengel 1985) also exists in soil. Only recently has interest in soluble organic N (SON; N extracted from soil by water, KCl, etc.) and DON increased. We make the distinction between SON and DON because, as we shall show later, the two pools are neither the same size nor of the same composition. Soluble pools of organic N are composed, at least partially, of easily mineralisable N, and so have a major impact on the usually very small but rapidly cycling N pools such as NH_4^+-N (Mengel et al. 1999). SON is therefore likely to be an important pool in N transformation pathways and plant uptake (Németh et al. 1988). The flux of N through the microbial biomass is large compared to its size and the size of the mineral N and SON pools at any given time. Determination of the relative importance of mineral N and SON in specific N transformation pathways and crop nutrition is essential for a proper understanding and prediction of the production, transformation and fate of N in agricultural systems.

In forest systems increased levels of atmospheric deposition have increased the N content of soil and brought many forest soils close to N saturation (Aber et al. 1989; Aber 1992; Currie et al. 1996; Koopmans et al. 1997; Goulding et al. 1998). Quite large pools of DON have been measured in leachates from forest floors (Yavitt and Fahey 1986; Stevens and Wannop 1987; Qualls et al. 1991; Currie et al. 1996). Qualls et al. (1991) found that 94% of the dissolved N leaching through a deciduous forest soil was present in organic form. Yu et al. (1994) also found that SON was the dominant form of N in a coniferous forest soil. SON has been identified as a key pool in soil-plant N cycling in forest systems (Qualls and Haines 1991), arctic tundra (Atkin 1996) and subtropical wet heathland (Schmidt and Stewart 1997) and DON represents a major input of N to surface water in forested watersheds (Wissmar 1991; Hedin et al. 1995), suggesting that the leaching of organic N could be a major pathway for N loss from at least some soils.

Compared to semi-natural systems, little is known about the form and function of SON/DON and the role that it plays in soil N cycling in agricultural soils. Here we review the progress that has been made since Lawes and Gilbert (1881) first analysed rain and drainage waters. We review methods of extraction, pool sizes and the role of SON and DON in N transformations and losses.

Measuring SON and DON

Extraction of SON from soil

Soluble forms of N can be extracted from soils by shaking with water. However, such extractions cause the dispersion of clays and it can be difficult to obtain clean solutions for analysis (Young and Aldag 1982). A range of salt solutions have been used for extracting N from soils, most commonly solutions of $CaCl_2$, KCl and K_2SO_4. Salt extracts may disturb adsorption equilibria on soil surfaces and release organic N, which was not originally dissolved (Fig. 1). The SON pool in soils cannot be measured directly by extraction, but instead is determined by subtracting the mineral N concentration from the total soluble N (TSN) concentration. Kjeldahl digestion was first used in this way to determine TSN in seawater; it has also been used for soil solutions (Beauchamp et al. 1986). This method is based on the reduction of N to NH_4^+-N in an acid solution, and has been described in detail by Bremner and Mulvaney (1982). However, the method is slow and cumbersome, and large N contents in control samples can decrease the accuracy and sensitivity of the procedure (Smart et al. 1981).

The development of simple, rapid and automated methods by which TSN can be routinely analysed has encouraged more measurements of SON to be made in recent years. Persulphate ($K_2S_2O_8$) oxidation was originally used for the analysis of seawater (D'Elia 1977; Koroleff 1983) but has been modified to determine TSN in fresh water (Solórzano and Sharp 1980) and soil extracts (Ross 1992; Cabrera and Beare 1993; Sparling et al. 1996). $K_2S_2O_8$ oxidation is based on the principle that, in the presence of a strong oxidising agent, both NH_4^+-N and SON are converted to NO_3^--N. Complete oxidation is achieved by autoclaving the soil extract or by ultra-violet digestion (Cabrera and Beare 1993; Williams et al. 1995; Sparling et al. 1996). Both approaches are suitable for the processing of large batches of samples (100 per day) and require only simple laboratory equipment. This technique has become popular in recent years for determining SMB-N from the difference in TSN between chloroform-fumigated and non-fumigated soils (Ross 1992; Sparling and Zhu 1993; Murphy

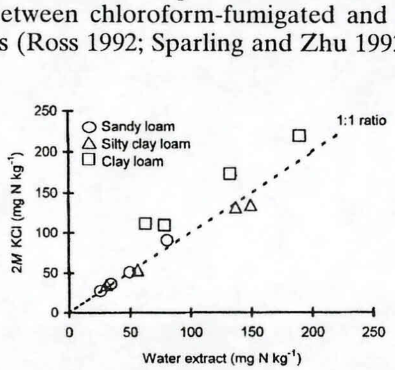

Fig. 1 Effect of extractant on the size of the soluble organic N pool in soils of differing texture

et al. 1998a). By also measuring the mineral N content in non-fumigated soil, SON can be estimated by difference (Jensen et al. 1997; McNeill et al. 1998). Alternatively, KCl extracts are routinely used to measure mineral N in soil to support fertiliser recommendations (Shepherd et al. 1996; Wilson et al. 1996). It is possible to use samples of the same soil extract for determination of TSN, mineral N and thus SON (Bhogal et al. 1999).

Smart et al. (1981) reported that $K_2S_2O_8$ oxidation was more precise than Kjeldahl digestion for measuring TSN in samples collected from a range of aquatic habitats. Studies employing water and 0.5 M K_2SO_4 soil extracts, comparing Kjeldahl digestion and $K_2S_2O_8$ oxidation, have found no significant difference in the amount of TSN determined (Cabrera and Beare 1993; Yu et al. 1994; Sparling et al. 1996). However, in 1 M KCl soil extracts, TSN was overestimated at low concentrations and underestimated at higher concentrations by $K_2S_2O_8$ oxidation compared to Kjeldahl digestion (Cabrera and Beare 1993). More recently, Merriam et al. (1996) proposed a high-temperature catalytic oxidation technique to measure TSN and found that results compared well to those of $K_2S_2O_8$ oxidation.

Our measurements of arable and cultivated ley crops growing on a sandy loam soil suggest that the extractant has only a slight influence on the size of both the mineral N and SON pools in this soil type. Extractions made with routine procedures for mineral N (2 M KCl) or SMB-N (0.5 M K_2SO_4) all show the same quantitative differences in the effects of management on N pool size and distribution; all are suitable for determining SON. However, other measurements of TSN on three contrasting soil types, sandy loam, silty clay loam and clay loam (Fig. 1) suggest that water and KCl extract comparable amounts of SON from non-clay soils, but that clay soils contain TSN only extractable with KCl. This is likely to be NH_4^+-N or organic N adsorbed on cation exchange surfaces on the clay.

Measuring SON by electroultrafiltration

Electroultrafiltration (EUF) applies an electric field to soil suspensions to separate fractions of soluble N by forced diffusion through membrane filters (Németh 1979, 1985). The method removes both mineral (EUF-NO_3) and organic (EUF-N_{org}) N, and is therefore considered by its advocates to remove all forms of N that are available for plant uptake, loss or microbial transformations over the short term. The advantage of EUF is that the rate of nutrient release can be determined, whereas soil extraction determines only pool sizes (Németh 1985). However, EUF is labour intensive, costly, and produces variable results compared to soil extraction (Houba et al. 1986). Houba et al. (1986) found that amounts of N extracted by EUF and 0.01 M $CaCl_2$ were highly correlated, and concluded that the two techniques were interchangeable. Feng et al. (1990) found that the EUF-N_{org} fraction was larger than the pool of N extracted with $CaCl_2$ 1 week after the incorporation of ground and dried rape tops into soil. Recent results from Mengel et al. (1999) indicated that EUF removed a larger N_{org} fraction than $CaCl_2$ in 13 of 20 soils. Thus EUF appears to measure SON plus a part of some other soil N pool.

DON in soil solution

Dissolved mineral and organic N in soil solution is found in mobile water flowing through pores, cracks and channels, and in immobile water within pores in soil peds. Sampling techniques must be selected to extract the pool of interest. Centrifugation of soil at natural soil moisture contents has been shown to be a suitable method for obtaining solutions containing dissolved organic C similar in composition to the soil solution in macro-, meso- and micro-pores (Raber et al. 1998). However, most measurements are made to estimate leaching losses, and these can be made in a number of ways, summarised below (Addiscott et al. 1991; Goulding and Webster 1992; Titus and Mahendrappa 1996). All of the methods have limitations, especially in structured soils with heterogeneous preferential flow through soil macropores.

Suction cups (also called porous cups, porous probes and, by some, lysimeters) are made of a hydrophilic material containing small pores, typically kaolin clay. When they are buried in the soil and connected to the surface by one or more narrow tubes, soil solution can be collected by creating a vacuum in the cup (Grossmann and Udluft 1991). They have been very widely used to measure NO_3^--N leaching, but only work well in homogeneous (e.g. sandy) soils where water flow is not dominated by cracking and preferential flow. In such soils, solute movement as measured by suction cups and monolith lysimeters is in close agreement (Webster et al. 1993). They do not work well on poorly drained, heavy clay soils because they have been found to preferentially sample immobile water from soil peds (Goulding and Webster 1992; Hatch et al. 1997), whereas macropores are the dominant drainage route, especially after heavy rain (Grossmann and Udluft 1991) or irrigation. However, suction cups may still be useful for determining the composition of the pool of water that plant roots utilise.

It is difficult to determine the volume of soil from which suction cups remove solution; the reproducibility of both the volume of water removed and the concentration of solutes in it is often unsatisfactory (Addiscott et al. 1991). Suction cups with ceramic heads are normally used to obtain soil solution for the analysis of NO_3^--N (Webster et al. 1993; Poss et al. 1995; Hatch et al. 1997) but their use is questionable for measuring a strongly adsorbed ion such as PO_4^{3-} (Hansen and Harris 1975). Because DON contains molecules that adsorb and absorb strongly, ceramic suction cups are unlikely to be suitable for measuring DON. Quartz/Teflon samplers are inert (Zimmermann et al. 1978) and are now widely used, especially in P leaching studies (Bottcher et al. 1984). However, they are approximately 10 times more expensive than ceramic cups.

DON in drainage water

Nutrient and pollutant losses have been extensively studied by collecting drainage water from field drains (Lawes et al. 1882; Tyson et al. 1997), and lysimeters (Smolander et al. 1995; Titus and Mahendrappa 1996). However, field drains may only partially intercept soil drainage, the rest percolating to the aquifer when subsoils are permeable. Drains also require constant (preferably automatic) sampling. Very detailed results are available from experimental systems of fully automated, hydrologically isolated plots such as the Brimstone experiment (Cannell et al. 1984; Vinten and Redman 1990). Such experiments enable all drainage to be collected and analysed, giving a quantitative water and solute budget from plots on which agricultural operations can be carried out as normal, but at great cost (Cannell et al. 1984). Lysimeters are in the middle price range at approximately one-tenth of the cost of a field drainage system but 10 times the cost of porous cups. Although subject to soil shrinkage problems, they are the most reliable and economic tools with which to measure total water and solute loss from a block of soil large enough to minimise spatial variation and allow management practices almost as normal.

Size of the SON and DON pools

Smith (1987) measured SON pools as large as, or larger than, mineral N pools in air-dried agricultural soils. He suggested that this reflected the presence of freshly decomposed plant material and/or organic matter components disrupted during sample preparation. Recent studies with fresh soil have confirmed that as much SON as mineral N can exist in soil under agricultural cropping systems (Jensen et al. 1997; McNeill et al. 1998; Bhogal et al. 2000). Jensen et al. (1997) showed that 0.5 M K_2SO_4-extractable SON varied seasonally between 8–20 kg SON-N ha^{-1} in a coarse sand and 15–30 kg SON-N ha^{-1} in a sandy loam (0–15 cm); the minimum occurred during winter and the maximum in late summer. McNeill et al. (1998) showed that SON comprised 55–66% of the TSN under wheat (18 kg SON-N ha^{-1}) and pasture (28 kg SON-N ha^{-1}) on a loamy sand (0–10 cm) of low organic matter content. Assuming a bulk density of 1.5 g cm^{-3} for the 0 to 30-cm layer, Mengel et al. (1999) removed a $CaCl_2$-extractable SON pool (fraction 1, 20 °C) of approximately 30–45 kg N ha^{-1} from 17 arable soils. Our data (Table 1) show, for a wide range of soil types (0 to 30-cm layer), a KCl-extractable SON-N content of generally 20–30 kg SON-N ha^{-1} and a very constant ratio between mineral N and SON in arable soils, with SON comprising about 40–50% TSN. Both data sets were collected in spring, although our measurements were made on field-moist soil while those of Mengel et al. (1999) were conducted on air-dried sieved (<1 mm) soil.

On the sandy loam soil of the long-term Woburn ley-arable experiment (described by Johnston 1973) we measured SON contents in the 0 to 25-cm layer ranging from 7 kg SON-N ha^{-1} under continuous arable cropping to 18 kg SON-N ha^{-1} after 8 years of grass ley. These accounted for between 33% and 60% of the TSN. Our measurements of SON in the 0 to 25-cm layer of a range of soils on an organic farm showed that it accounted for 80% of TSN and ranged from 24 to 46 kg SON-N ha^{-1}, increasing with the number of previous years under grass/clover ley. Figure 2 shows mineral N and SON pool sizes in the plough layer (0–23 cm) under continuous arable (under wheat dur-

Table 1 Mineral and soluble organic N pools (kg N ha^{-1}) extracted in 2 M KCl for 12 soils (0–30 cm) under arable cropping in spring 1999. Values in parentheses are SEMs (n=3). *BD* Bulk density, *SON* soil organic N

Soil	Soil subgroup (SSLRC)	USDA great group	Texture	Previous crop	BD (g cm^{-3})	Mineral N	SON
1	Typical brown earth	Dystrochrept	Sandy loam	Winter wheat	1.01	26 (6)	29 (8)
2	Typical argillic brown earth	Hapludalf	Silty loam	Winter wheat	1.44	35 (3)	33 (3)
3	Stagnogleyic argillic brown earth	Hapludalf	Sandy clay loam	Winter oilseed rape	1.42	37 (3)	22 (4)
4	Stagnogleyic argillic brown earth	Hapludalf	Sandy clay loam	Winter wheat	1.27	81 (8)	30 (1)
5	Gleyic argillic brown earth	Hapludalf	Silty clay loam	Celery	1.31	26 (1)	34 (1)
6	Typical humic-alluvial gley soils	Haplaquoll	Clay loam	Winter wheat	1.41	32 (2)	28 (3)
7	Brown rendzina	Eutrochrept	Clay loam	Field peas	1.31	21 (3)	33 (1)
8	Brown rendzina	Eutrochrept	Clay loam	Winter wheat	1.13	25 (1)	24 (2)
9	Pelo-stagnogley soil	Haplaquept	Clay	Winter wheat	0.97	33 (6)	24 (3)
10	Typical calcareous pelosol	Haplaquept	Clay	Winter wheat	1.26	33 (2)	27 (2)
11	Typical calcareous pelosol	Haplaquept	Clay	Winter wheat	1.36	27 (2)	33 (3)
12	Typical calcareous pelosol	Haplaquept	Clay	Winter barley	1.08	27 (2)	23 (3)

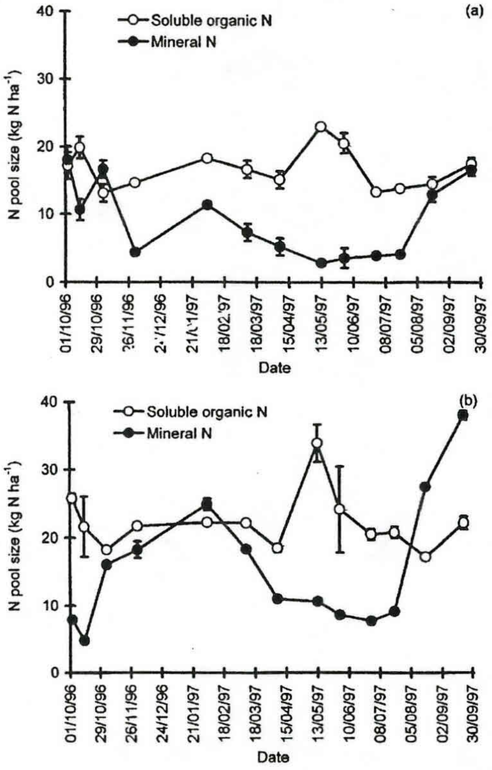

Fig. 2 The dynamics through a year (1 October 1996–30 September 1997) of the mineral N and soluble organic N (*SON*) pools (extracted in 2 M KCl) in the surface soil (0–23 cm) under a continuous arable plot (**a**) and a short-term ley ploughed up immediately after the first sampling and planted with wheat (**b**). *Capped bars* are SEM

ing year of sampling; Fig. 2a) and an 8-year ley ploughed up immediately after the first sampling and sown to wheat (Fig. 2b) in the Woburn ley-arable experiment. More SON and mineral N were found under the ploughed up grass, as was expected. Even larger SON pool sizes (70–500 kg SON-N ha^{-1}) have been measured in the soil profile (0–90 cm) after ploughing and reseeding long-term (>50 years) grasslands (Bhogal et al. 1999). Clearly SON is a significant pool within agricultural soils.

Composition of SON

Dissolved organic matter leaching from forest floor litter is dominantly composed of decay-resistant organic acids formed by partial decomposition of plant, microbial and animal tissues (Currie et al. 1996). It is often fractionated by polarity and charge using resins and/or the optical absorbance of solutions determined at 340 nm (Raber et al. 1998; Scott et al. 1998; Tipping et al. 1999). In organic-rich layers, such as leaf litter or peat, hydrophobic and hydrophilic acids occur in equal concentrations. However, hydrophobic compounds are retained by adsorption and aggregation in mineral horizons, and the relative concentration of hydrophilic acids increases in drainage from upland soils as the thickness of the mineral horizons increases (Tipping et al. 1999). Ligand exchange between the dissolved organic matter and carboxyl (Kaiser et al. 1997) or hydroxyl groups (Shen 1999) on the surface of soil minerals is thought to be an important mechanism for sorption, although a number of other mechanisms have also been proposed (see Chiou et al. 1983). Adsorption isotherms for interactions between dissolved organic C and arable agricultural soils can be represented by linear isotherms, where the different adsorption capacities of different soils are related to the clay content (Riffaldi et al. 1998). Shen (1999) found the sorption capacity of the soil to be influenced by pH (maximal at pH 4–5 and decreasing with pH>5), clay content (through its relationship with the amount of mineral surface), the ionic strength of the soil solution and the ion species present in soil solution (e.g. Ca^{2+} binding to mineral sites or complexing with dissolved organic matter). Thus management practices in agricultural systems, such as manure application and liming, will change the sorption capacity of soil and the resulting composition of the dissolved organic matter in solution.

Dissolved organic matter from arable agricultural soils has a higher proportion of hydrophobic compounds when compared with extracts from grassland and forest soils (Raber et al. 1998). Total amounts of SON are usually higher under grassland and forest and the proportion of amino compounds is greater than in arable soils (Németh et al. 1988; Mengel et al. 1999). In arable soils, 23–55% of SON is hydrolysable. Free amino acids only make up 3% of DON, amino sugars and heterocyclic-N bases, on average, 15%; the remainder of the hydrolysable fraction is present in amino compounds.

The availability of suitable analytical techniques has restricted progress in determining the composition of SON/DON. However, new developments in techniques such as fluorescence spectroscopy (Senesi et al. 1991; Tam and Sposito 1993; Erich and Trusty 1997), Fourier-transform infrared spectroscopy (Candler et al. 1988; Gressel et al. 1995a; Kaiser et al. 1997), ^{13}C-nuclear magnetic resonance spectra (Candler et al. 1988; Novak et al. 1992; Kaiser et al. 1997) and ultraviolet-visible spectra (Candler and Van Cleve 1982; Cronan et al. 1992) are now being employed to characterise dissolved organic matter into functional or structural groups. Most progress has been made from the analysis of humic substances (Gressel et al. 1995a) but research into dissolved organic matter (Gressel et al. 1995b) suggests that progress will be rapid. We need to further characterise specific components of SON rather than considering it as a single pool. Only then will we quantify the size of SON fractions that are actually involved in N cycling and loss processes.

Role of SON in N transformations

Mineralisation

The SON pool and its transformations have been largely overlooked in research into mineralisation, which has usually concentrated on changes in the size of the soil mineral N pool. However, the form of N determines which micro-organisms and plants can utilise it, and so the distinction between the mineralisation of organic matter through to NH_4^+-N or only to SON is important.

Appel and Mengel (1993) have suggested that $CaCl_2$-extractable SON is a reliable indicator of the pool of organic N available for mineralisation in sandy soils because the size of this pool correlates with net N mineralisation (Appel and Mengel 1992, 1993; Groot and Houba 1995). However, Appel and Xu (1995) attempted to relate the mineralisation of EUF-N_{org} to the appearance of mineral N by using ^{15}N-labelled rape residue as a tracer. They found that, while EUF was able to selectively extract organic N derived from the rape, the decline in the size of the EUF-N_{org} fraction was not sufficient to account for all of the production of ^{15}N-labelled mineral N. The authors also found that a large (but unknown) fraction of the EUF-N_{org} derived from the rape was not easily mineralisable.

Murphy et al. (1998b) found that changes in the size of the KCl-extractable SON pool in a loamy sand under pasture and wheat paralleled gross N mineralisation rates. We have recently found that the production of KCl-extractable SON during aerobic incubation is significantly correlated to gross ($r=0.73$, $P<0.001$) and net N mineralisation rates ($r=0.68$, $P<0.01$) in agricultural soils ranging in texture from sandy loam to clay loam. However, no direct link between SON turnover and gross N mineralisation rates has yet been established.

Smith (1987) showed that, during the long-term aerobic incubation of soil with periodic leaching [using the method of Stanford and Smith (1972)] DON was produced between leaching episodes, although the majority of leached N was removed as NO_3^--N. Smith (1987) determined the mineralisation potential of the leachates in the absence of soil and concluded that the leached DON was not "exceptionally susceptible" to mineralisation. This may imply that DON is stable in soil and, after leaching, in streams and rivers. However, since the mineralisation potential was determined *outside* of the soil, the findings of Smith (1987) cannot be used to indicate an in situ rate of turnover for this pool.

Since the breakdown of soil organic matter can result in the production of organic compounds that are soluble but recalcitrant to further microbial decomposition (Smolander et al. 1995), it is likely that only a fraction of the SON pool will be mineralised. Better relationships may exist between N mineralisation and the labile fractions of SON. DeLuca and Keeney (1994) measured a decline in the size of the soluble amino-N pool in soil under tallgrass prairie that coincided with an increase in SMB and N mineralisation. Kielland (1995) has suggested that the rapid turnover of amino acids in arctic tundra soil results in high rates of gross N mineralisation. Recently, Mengel et al. (1999) examined the relationship between net N mineralisation and fractions of SON (amino-N, amino sugars, heterocyclic-N of nucleic acids). They found a significant correlation between amino-N and net N mineralisation across 17 arable, 1 forest and 2 grassland soils. However, this relationship was not significant when the arable soils were examined independently: the range of amino acid contents of the arable soils was narrow compared to the variation in net N mineralisation rates, suggesting that N pool *size* does not necessarily reflect N process *rates*. Jones (1999) showed that the degradation of amino acids is extremely rapid with half-lives, for a range of amino acids, being only 1–12 h in a range of soils.

One difficulty with interpreting net N mineralisation rates is the confounding problem of immobilisation (Barraclough 1991; Davidson et al. 1991; Mary and Recous 1994; Murphy et al. 1998b; Recous et al. 1999). We have found that gross N mineralisation rates (i.e. ammonification) vary less than net N mineralisation across arable soils (unpublished data), highlighting the importance of immobilisation in the cycling of N in agricultural soils (Gaunt et al. 1998). Similarly, Barraclough et al. (1998) found that gross N mineralisation rates, associated with the decomposition of plant residues, were more constant than net N mineralisation rates. This suggests that, if the amino-N and amino sugar fractions are the direct sources from which inorganic N is produced (Mengel et al. 1999), then their pool size may be better related to total soil N supply (i.e. gross N mineralisation) than the net product of mineralisation-immobilisation turnover. However, fluxes in and out of the amino-N and amino sugar pools may still mask the relationship between N fraction and N transformation.

Immobilisation

NH_4^+ is the dominant form of N assimilated and immobilised by soil micro-organisms (Jansson and Person 1982; Recous et al. 1988; Shen et al. 1989), although NO_3^--N can also be assimilated when C is available (Azam et al. 1986; Recous et al. 1988). Mineralisation-immobilisation theory (MIT) assumes that all N uptake is from the mineral pool. However, micro-organisms can also utilise low molecular weight SON compounds (Molina et al. 1983; Barak et al. 1990; Barraclough 1997), suggesting that classic MIT theory may be incorrect. Hart et al. (1994) found that the C:N ratio of the substrate utilised by heterotrophic micro-organisms in a forest soil was similar to that of the K_2SO_4-extractable organic pool in that soil. They also showed that the K_2SO_4-extractable SON pool declined when the chloroform-labile N pool (i.e. SMB-N) increased, and suggested that extractable SON is a major source of N for micro-organisms in forest soil (Hart and Firestone 1991; Hart et al. 1994). In a soil under winter wheat, only 44% of added leucine-N and 82% of glycine-N was mineralised through to NH_4^+-N, while all of the amino-N disappeared from the soil, presumably due to direct assimilation by the SMB (Barraclough 1997). Thus conventional views of N transformation within soil may be an over-simplification.

Plant residues contain significant amounts of water-soluble C and N (Mengel and Kirkby 1978), and water-soluble fractions of straw have been shown to be relatively rich in N (Jensen et al. 1997). N from plant residues is rapidly assimilated by the SMB, and Jensen et al. (1997) found no significant increase in the amount of SON in soil after amendment with 4 and 8 t ha^{-1} oilseed rape straw compared to unamended soil. By contrast, both SON and DON were larger in the FYM plot of the Broadbalk wheat experiment than in the other plots receiving fertiliser (Table 2). Clearly the quality of the organic material has a major impact on SON/DON. This is further supported by studies of nutrient dynamics on tropical soil amended with green manures, which showed the concentration of SON to increase substantially at the same time as immobilisation of mineral N (Mulongoy and Gasser 1993).

Figure 3 shows mineral N and SON pools in soil into which three contrasting crop residues had been incorporated. Both SON and mineral N pools were smaller in the soil to which maize residues had been added than in the control soil. Maize residues have a C:N ratio of 108:1, and it is probable that both mineral N and SON were immobilised in this treatment. By contrast, forage rape and rye have C:N residues of 11:1 and 14:1, and both the mineral N and SON contents of the soil to which these residues were applied increased greatly. During the residue incubation, rain fell after day 50. The SON content of the soil in the residue treatments returned to a value similar to that of the control after rainfall, perhaps because the organic N released from the residues was leached below the layer from which samples were taken. If this was the case, it indicates a difference in leachability and thus chemical composition between SON derived from newly incorporated residue and older SON.

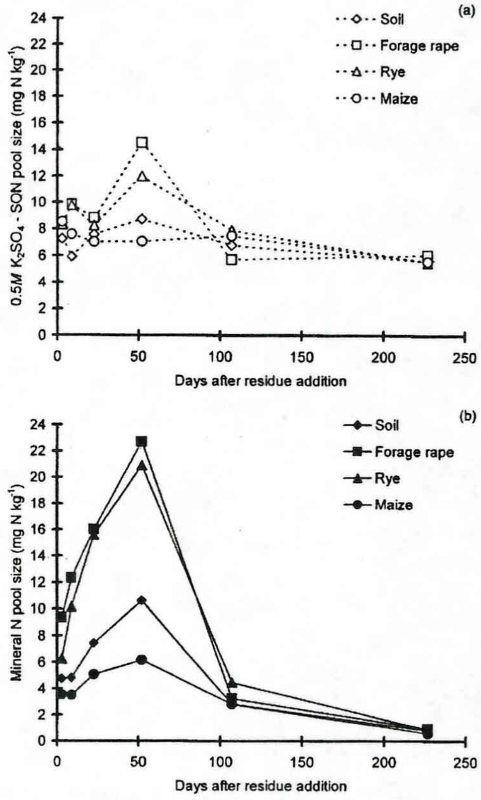

Fig. 3 The dynamics of the SON pool (**a**) and the mineral N pool (**b**) in soils into which forage rape, rye and maize residues had been incorporated

Leaching

Concentrations of DON in forest floor leachates exceed concentrations of mineral N (Yu et al. 1994), and DON has been found to be a dominant source of N leached into lakes in forested watersheds (Wissmar 1991). Stevens and Wannop (1987) studied the composition of TSN in leachate from lysimeters under the organic horizon, and from ceramic cup samples in the soil mineral layers, after the clearfelling of Sitka spruce in north

Table 2 Mineral N, soluble organic N SON or dissolved organic N (*DON*) contents (kg N ha^{-1}) in the soil profile (0–75 cm) and drainage solution collected from tile drains (September–November 1998) located at a soil depth of 65 cm under plots of the Broadbalk wheat experiment. Values in parentheses are SEMs ($n=4$) *N0* No N applied, *N144* 144 kg $(NH_4)_2NO_3$-N ha^{-1} $year^{-1}$, *N288* 288 kg $(NH_4)_2NO_3$-N ha^{-1} $year^{-1}$, *FYM* farmyard manure with a N content equivalent to ca. 240 kg N ha^{-1} $year^{-1}$

Method	Measurement	N0	N144	N288	FYM
Soil extraction[a]	Mineral N	19.3 (1.3)	35.9 (1.3)	49.3 (2.2)	82.9 (3.0)
	SON	46.9 (8.9)	23.7 (8.7)	55.3 (11.7)	60.5 (11.9)
Total drainage[b]	Mineral N	9.9	6.3	29.0	52.0
	DON	1.2	1.1	2.5	7.0

[a] Soil profiles (0–75 cm) sampled on 7 December 1998 and extracted using 2 M KCl
[b] Assumes that 200 mm of total drainage (tile drains plus leaching through to chalk) occurs over the total leaching period (September–March). Values calculated using an average concentration of mineral N and DON concentration from the tile drains over the period September–November 1998

Wales. They found that, within the organic horizons, DON was >90% of TSN, but that deeper in the soil profile NO_3^--N predominated, suggesting the transformation of DON to NO_3^--N during percolation.

The current perception is that, in agricultural soils in temperate climates, NO_3^--N is the main source of N in drainage water. It now seems that DON may also be a vehicle for significant loss of N from the soil. Table 2 shows some data recently collected from the Broadbalk continuous wheat experiment at Rothamsted [described by Dyke et al. (1983)]. This 156-year-old experiment compares the effects of different amounts and types of fertiliser and manures on wheat grown continuously and in rotation. Each plot is drained and samples collected from the drains and suction cups. DON in drain waters is compared with SON measured in KCl soil extracts. The data show that the SON pool in the soil is much larger than the amount of N leached as DON; the latter being equivalent to only 2–10% of the SON pool. However, approximately 10% of the N leached from drains is still lost in an organic form. Thus the amount of N that is likely to be leached in organic form is significant but much less than the amount that remains within the soil.

It is apparent that more total N and DON is leached from plots receiving farmyard manure (FYM) compared to inorganic N. Loss of DON increased in the order nil-N plot<N-fertiliser plots<FYM (Table 2). The increase in DON under FYM is related to the greater amount of N in the soil (N0=0.104% N, N144=0.127% N, N288=0.129% N, FYM=0.297% N) but may also be due to the reduced sorption of DON onto soil in the FYM plot. Soil organic matter blocks active sites on soil minerals, reducing the sorption of dissolved organic matter.

Very large amounts of DON (up to 20% of total N lost) have been found in drainage waters leaving grassland lysimeters in Devon, UK (Hawkins et al. 1997). Certainly the potential exists for large leaching losses of DON: substantial amounts of SON have been found at depth (0–90 cm) when extracted from soil under wheat and especially permanent pasture (Fig. 4; Bhogal et al. 1999). To date it is unclear whether SON is mineralised before leaving the surface soil, thus leaching as NO_3^--N, or whether it is retained in the soil. It is also unclear whether DON leaving soils can be transformed to NO_3^--N in surface- or groundwaters. If SON is being mineralised during percolation through soil then it is included in measured losses (as NH_4^+-N, always small, and NO_3^--N) and accounted for in the N balance. However, SON sorbed in lower soil horizons may not be accounted for in measurements. Losses of N from soil may therefore be greater than previously considered from NO_3^- leaching studies, and leached DON may partially explain the imbalance that is sometimes calculated in experiments tracing the fate and recovery of ^{15}N-labelled fertiliser (Glendining et al. 1997). This is supported by findings of Németh (1985), who found that fertiliser N was recovered in both EUF-NO_3 and EUF-N_{org} fractions, and

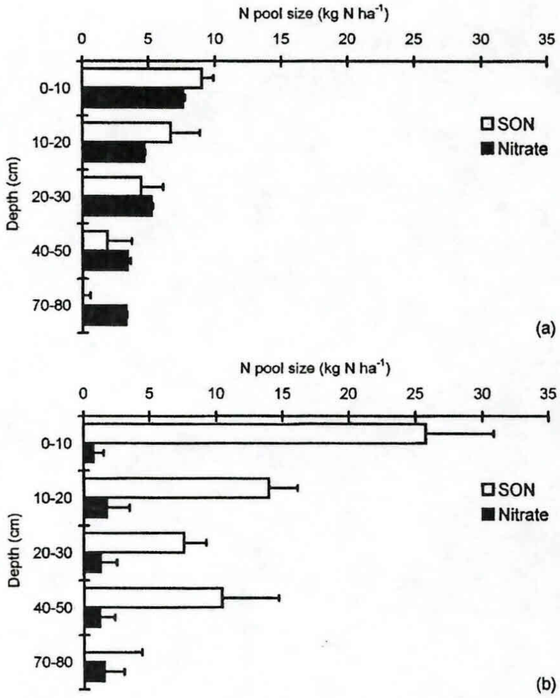

Fig. 4 Distribution of 2 M KCl-extractable SON and NO_3^--N within the soil profile under continuous arable cropping (**a**) and permanent unimproved grassland (**b**) on a sandy loam soil. *Capped bars* are the SEM

concluded that it is necessary to measure the easily mineralisable EUF-N_{org} fraction in fertiliser-recovery studies. However, Appel and Mengel (1992) did not measure an increase in the size of the EUF-N_{org} fraction following fertiliser addition.

Plant uptake

The release of N through the mineralisation of SOM and plant residues has long been identified as an important source for plant uptake (Scarsbrook 1965; Paul and Voroney 1980). Plants are usually thought to take up NH_4^+-N and NO_3^--N, with NO_3^--N being favoured when it is available in abundance (Streeter and Barta 1988). Plants are able to take up urea directly in the absence of hydrolysis (Harper 1984), but more slowly than mineral N. Bollard (1959) demonstrated that many plants are able to use SON compounds as their sole N source. In some natural environments (e.g. temperate and arctic heathlands) mineralisation rates are too slow to supply all of the N required by plants as NO_3^--N or NH_4^+-N (Chapin et al. 1988; Nadelhoffer et al. 1991; Jonasson et al. 1999; Schmidt et al. 1999), and soluble forms of organic N are a major source of plant-available N (Chapin et al. 1993; Kielland 1994). Schmidt and Stewart (1997) found that, over 1 year of measurements, NH_4^+-N and amino acids were the dominant forms of N under subtropical heathland. It is well established that some plants are capable of utilising SON either directly (Chapin et al. 1993; Kielland 1994) or in association with mycorrhiza and ectomycorrhizae (Atkin 1996; Kielland 1994; Michelsen et al. 1996). Mycorrhizal roots may transport amino acids from the bulk soil via the extracellular mycelium, thus preventing their rapid degradation in the rhizosphere. This may be the main route for the direct uptake of SON by plants (Jones 1999). However, there is still little information on the contribution of SON to the total intake of N by mycorrhizal plants.

Within agricultural soils, the availability of organic N compounds to plant roots is usually assumed to be minimal (Harper 1984). It is not clear whether agricultural crops benefit from SON either by direct uptake or via microbial immobilisation of organic molecules and then re-mineralisation of NH_4^+-N. Appel and Mengel (1990, 1992) found that the size of the organic N pool extracted by EUF or $CaCl_2$ did not decline in sandy soils during the growth of rape plants, although this lack of change in the size of the SON pool does not indicate that the fraction is not turning over and supplying N to plants. Jones and Darrah (1994) demonstrated the presence of amino-acid-specific transporters in the cell membranes of plant roots capable of operating at low solution concentrations, and established that maize roots possess a highly efficient mechanism for scavenging SON. Further studies using ^{15}N as a tracer are required to determine what proportion of the extractable SON is used by plants.

Mercik and Németh (1985) showed that long-term (since 1923) annual applications of inorganic N (90 kg N ha^{-1}) increased both the EUF-NO_3 and EUF-N_{org} fractions compared to soils given no N. Mercik and Németh (1985) also found that leaching of EUF-NO_3 was greater under potato monoculture compared to rye monoculture. On the Broadbalk experiment at Rothamsted, less SON was present where wheat followed potatoes compared to continuous wheat. At present we are unsure whether this was related to: (1) a difference in the quality and quantity of residue inputs to soil under potatoes compared to wheat, or (2) a difference in the utilisation of SON between plant species.

Seasonal dynamics of SON

Seasonal variations in dissolved organic matter fluxes have been measured in the seepage water from forest soil and drainage from upland peat soils, with maximum fluxes occurring during summer (Tegen and Dörr 1996; Scott et al. 1998). The Woburn dynamic study (Fig. 2) was an attempt to measure the dynamics of all the major pools and transfer processes of N through an agricultural year. Figure 2 shows the dynamics of the mineral N and SON pools. Clearly the SON responds to periods of plant uptake and mineralisation/immobilisation, but it is not as dynamic as the mineral N pool. Under continuous arable cultivation, the size of the SON pool was relatively constant at ca. 15–20 kg SON-N ha^{-1} (0–23 cm), decreasing with leaching in early winter and increasing markedly during the period of rapid root growth in spring. Under ploughed-out grass, the SON pool size was larger at ca. 20–25 kg SON-N ha^{-1} (0–23 cm), and the changes were larger than under continuous arable, but occurred at the same times of the year and in the same way. The dynamics of the SON and mineral N pool in the plough layer (0–23 cm) were reflected in subsoil down to 90 cm. The period of rapid plant growth (April to June) both above and below-ground (root growth/turnover and exudates) appeared to play an important role in SON dynamics. We hypothesise that in this soil there is a relatively constant pool of SON (related to soil organic matter content and soil texture) and a more dynamic pool of SON, which reflects current plant dynamics. Presumably these two fractions of SON vary greatly in their chemical composition, stability and transformation rates.

Transport of pollutants with DON

Early work appreciated that dissolved organic matter leached from soil could carry with it nutrient cations such as Ca^{2+} and Mg^{2+} and pollutants such as toxic metals (low affinity for DON – Cd, Zn; high affinity for DON – Cu, Cr, Hg) and pesticides (Chiou et al. 1986; Berggren et al. 1990; Wissmar 1991; Liu and Gary 1993). It is the concentration of a heavy metal in soil solution that is of importance in determining its environmental impact, not the total metal content (Temminghoff et al. 1998). This makes the study of dissolved organic matter/heavy metal relations even more important. Many of the functional groups of dissolved organic matter are acidic and deprotonated, resulting in anionic charged matter which facilitates its solubility and ability to complex with metals (Shen 1999). The availability and mobility of heavy metals in soils is dependent on their chemical speciation and relative distribution in soil solution. Heavy metals such as Cu that complex with dissolved organic matter are potentially highly mobile (Temminghoff et al. 1997; Kalbitz and Wennrich 1998; Romkens and Dolfing 1998). Reddy et al. (1995) showed that Cu was predominately complexed with dissolved organic matter at neutral pH, while under acidic conditions free ionic forms dominated. Animal manure is a source of Cu in agricultural systems (Romkens and Dolfing 1998) and we have measured a higher concentration of Cu in the drainage solution from the Broadbalk wheat experiment under FYM compared to mineral N plots.

The use of SON in fertiliser recommendations

To predict the availability of N to crops from soil it is necessary to determine both mineral N and N which is easily mineralisable and likely to be released during crop growth. Many methods and models have been developed for measuring or predicting N released by mineralisation. These were reviewed by Jarvis et al. (1996) and Shepherd et al. (1996). Very little research has concentrated on SON. Smith et al. (1980) recognised the need to measure what they described as "organic N leached" when using extractants to estimate mineralisable N. Németh (1985) and Recke and Németh (1985) also concluded that it is necessary to measure both the EUF-NO_3 and EUF-N_{org} fractions of soil N when predicting N fertiliser requirements. The EUF method has been used to predict fertiliser requirements for sugar beet (Wiklicky 1982; Recke and Németh 1985; Sheehan 1985), grapes (Eifert et al. 1982) and other agricultural crops (Appel and Mengel 1990; Saint Fort et al. 1990; Linden et al. 1993). Rex et al. (1985) found a highly significant correlation between the grain yield of winter wheat and EUF-N (EUF-NO_3 plus EUF-N_{org}) content of the soil.

We found no strong relationship between the size of the KCl-extractable SON pool and the amount of potentially mineralisable N (PMN; determined by anaerobic incubation) in soil collected from a range of soil types and land uses (Fig. 5). Both the mineral N and SON pool size increased after anaerobic incubation, with the ratio between mineral N and SON being identical at the beginning and at the end of the incubation. The absence of any change in this ratio did not indicate that SON was not the source of mineral N production, since input and output fluxes through this pool may have been in balance during the incubation. As an index, PMN would rank soils in the same order whether SON was included as a component of the total pool or not. However, if the size of the PMN pool released during incubation is thought to reflect the amount of the N that will become available in the short-term to plant uptake, then inclusion of SON within the PMN pool may be warranted. Measurement of both mineral N plus SON (i.e. TSN) may improve subsequent fertiliser recommendations but, with the exception of EUF recommendations, few attempts have been made to incorporate measurements of SON into decision support systems. This may be because the processes and conditions which promote and consume SON are not, as yet, well defined.

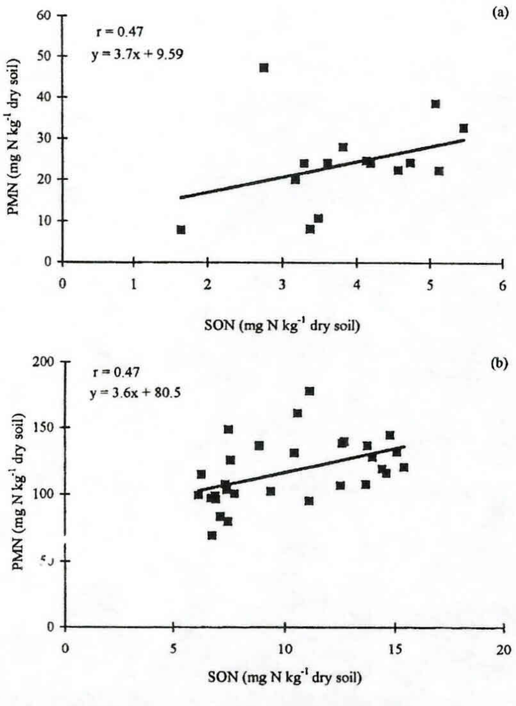

Fig. 5 Correlation between the amount of SON extracted in 2 M KCl (*x axis*) and potentially mineralisable N (*PMN*; *y axis*) in soil collected from the Woburn ley-arable experiments (**a**) and ley-arable plots on Duchy home farm (**b**)

Conclusions

Concerns over the environmental and health impacts of NO_3^--N leached from agricultural soils, as well as the importance of NH_4^+-N and NO_3^--N in crop nutrition, have focused attention on the study of mineral N in agricultural soils. In contrast, SON in soil, its transformations, and losses by leaching of DON have received relatively little attention. Recent research suggests that the SON pool is as large as the mineral N pool, with at least a proportion of it leaching either untransformed as DON or transformed into NO_3^--N. DON carries with it nutrients and pollutants into surface- and groundwaters. It is very likely that SON plays an important role in mineralisation/immobilisation. However, the absence of a relationship between SON and other indices of N availability suggest that merely measuring SON will not indicate the size of the mineralisable N pool. Information on the role of SON in N transformations is sparse and often contradictory. Until the many questions surrounding SON are answered we will be limited in our ability to incorporate it into N-cycle models and decision support systems. We need to quantify the size and composition of SON pools in different systems, identify those fractions of SON that are involved in N mineralisation, microbial assimilation and plant uptake, and determine the ultimate fate of SON. The recent development of simple methods to measure SON and improved techniques to characterise SON will stimulate research and enable workers to begin to answer these questions.

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Natural and artificial sources of nitrogen and phosphate pollution of waters in the Netherlands surface

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NATURAL AND ARTIFICIAL SOURCES OF NITROGEN AND PHOSPHATE POLLUTION OF SURFACE WATERS IN THE NETHERLANDS

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SUMMARY

Nitrogen and phosphate pollution of surface waters by various potential sources is discussed in general. Attention is paid to the more or less natural sources: precipitation, leaching of natural soil constituents, seepage of saline groundwater and to the artificial sources: domestic waste water, inlet water from storage canals, leaching of fertilizers, surface runoff and discharge of dune-water.

For three polders and three brook catchment areas the total nitrogen and phosphate load has been calculated. Polders and catchment areas differ with respect to the geo-hydrological situation, soil type, agricultural soil use and population density. The contribution by the more or less natural sources of pollution is much higher in the polders than in the brook catchments because of the influence of seepage and peat soil in the former. For polder waters the role of agricultural sources is not important neither for phosphate nor nitrogen.

In the sandy soil of the brook catchment areas on the other hand, the nitrogen pollution originates for the largest part from scattered agricultural sources. The contribution to the phosphate load by domestic sewage depends on the population density and the treatment method of waste water, but generally speaking it is an important factor. The use of P-fertilizers in agriculture up to now does not lead to an increase in the leaching of phosphate to the groundwater. Via surface runoff from badly drained grassland, P-fertilizers can be transported to surface waters. The quantification of this source needs further investigation.

For the explanation of the water quality information must be available about geology, hydrology, soil type and human activities.

1. INTRODUCTION

The fight against pollution of surface waters is heavily focussed on the removal of phosphate and nitrogen compounds because of their adverse effect on water quality. Each addition above the natural load causes undesirable changes in the biological equilibria of the receiving waters. Small or large fluctuations in the oxygen content of the surface waters and even anaerobic situations can occur as a result of increased algae growth. Pollution of surface waters with ammonium or organic nitrogen compounds can lead to a considerable consumption of oxygen when oxydation to nitrate occurs. The suitability of surface waters for some functions will be lowered by the presence of certain compounds. For example when surface waters are used for the production of drinking water, the nitrogen concentration should be lower than 11 $g.m^{-3}$ N (Ministerie van Verkeer en Waterstaat, 1975).

In many surface waters in The Netherlands high concentrations of nitrogen and phosphorus are periodically or constinously recorded (table 1). The question arises which

Table 1 Short and long term water quality aims for nitrogen and phosphate in surface waters in The Netherlands (Ministerie van Verkeer en Waterstaat, 1975) and the recorded concentrations in two areas

Parameter		Water quality aims		Recorded	
		short term	long term	Barneveldse Beek	Polder near Oranjeplaat
Kjeldahl-N	($g.m^{-3}$ N)	3.0	1.0	3.4	7.8
NH_4^+	($g.m^{-3}$ N)	2.0	<0.5	2.1	6.0
NO_3^-	($g.m^{-3}$ N)	4.0	2.0	5.6	0
Total-P	($g.m^{-3}$ P)	0.3	0.05	1.2	1.4

pollution source is to blame for this and what can be done to minimize pollution. A necessary and useful tool to answer this question is the nitrogen and phosphorus balance for surface waters. The balance for The Netherlands as a whole does not give adequate information to be used as basis for measures to be taken in a specific area, since even in dry summers more than 80 per cent of the phosphate imported by the river Rhine is transported directly to the sea. Moreover, large differences in nitrogen and phosphorus load between areas can be expected to occur due to differences in hydrological situation, soil type and soil use. In this article information will be given about the contribution of nitrogen and phosphate by some potential sources for the situation in The Netherlands. With this knowledge an explanation will be given for the nitrogen and phosphorus load as present in three polders and three catchment areas.

2. POLLUTION SOURCES OF NITROGEN AND PHOSPHATE

2.1. *Precipitation*

Research carried out about the chemical composition of precipitation in general gives information on the total load by dry as well as wet deposition. It therefore is not correct to ascribe this pollution to rain alone.

Table 2 Nitrogen ($g.m^{-3}$ N) and phosphate contents ($g.m^{-3}$ P) of precipitation

Measuring period	Number of stations	Nitrogen		Phosphate		Literature
		mineral*	total**	ortho	total	
'32–'37	1	0.6	–	–	–	Leeflang (1938)
'73–'75	1	2.4	3.2	0.03	0.08	Steenvoorden and Oosterom (1975)
'73–'74	14	2.0	–	–	0.15	Henkens (1976)
'78	12	2.4	–	0.01	–	KNMI/RIV (1978)

*NH_4^+ and NO_3^-; **Kjeldahl-N and NO_3^-; – not analyzed.

The chemical composition of the precipitation (table 2) has changed considerably as a result of increasing agricultural and industrial activities and traffic intensity. In the period 1932 through 1937 an average concentration of 0.6 g.m^{-3} N has been measured. Now a total-nitrogen concentration of 3.0 g.m^{-3} N seems to be an acceptable value. For total-phosphate the average concentration is 0.10^{-3} P. The yearly amount of rain in The Netherlands is roughly 750 mm. If all the precipitation would fall on open water in a certain area this would give a load of 22.5 kg N and 0.75 kg P per hectare.

2.2. *Groundwater*

In this paragraph attention will be paid to the natural contribution by groundwater to the load of surface waters. In paragraph 2.3. the influence of agricultural activities on the concentrations of nitrogen and phosphorus compounds will be discussed.

The groundwater not influenced by man's activities already contains a certain amount of nitrogen and phosphorus compounds. Discharge of a precipitation surplus via the groundwater system on open water always causes a natural load. The groundwater discharge not only depends on the local precipitation surplus, but also on the regional geo-hydrological situation. The discharge can be higher because of seepage and can be smaller as a result of infiltration.

Information about the natural chemical composition of the groundwater has been gained from analyses in the upper meter of the groundwater under nature areas. The concentrations found are to a very large extent determined by the soil composition. Especially the organic matter content is of influence with regard to the concentrations of nitrogen and phosphate (table 3). The dispersion of phosphate concentrations for one type of soil was rather wide. For sandy soils total-phosphate concentrations of 0.15 g.m^{-3} P as well as 0.01 g.m^{-3} P have been found. Therefore the calculation of the natural mineral load of groundwater in a certain area should be based on analyses of groundwater from the investigated area. The very high concentrations found under nature areas on a marine clay soil are influenced by the seepage of saline groundwater (table 4).

Table 3 Average nitrogen and phosphate concentrations in the upper meter of the groundwater under nature areas for different soil types (Steenvoorden and Oosterom, 1973; Bots et al., 1978)

Soil type	Number of nature areas	Nitrogen (g.m^{-3} N)			Phosphate (g.m^{-3} P)	
		Kjeldahl-N	NO_3^-	total	ortho	total
Sand	3	0.9	0.3	1.2	0.02	0.05
River clay	1	0.5	0.4	0.9	0.01	0.11
Cut-over high moor peat	4	4.8	0.3	5.1	0.01	0.09
High moor peat	2	5.8	0.6	6.4	<0.01	0.15
Mesotrophic low moor peat	4	5.1	0.5	5.6	0.04	0.28
Marine clay	5	11.1	0.3	11.4	2.6	3.2

Table 4 Nitrogen and phosphate concentrations in the deep groundwater (10 m – 100 m) of some areas in The Netherlands. The number of analyses placed between brackets

Area	Nitrogen ($g.m^{-3}$ N)	Phosphate ($g.m^{-3}$ P)	Literature
Groningen, Friesland*	6.6 (240)	0.9 (156)	Bots et al. (1978)
North-Holland**	18 (88)	2.9 (88)	Toussaint and Boogaard (1978)
Mid-western Netherlands*	8.0 (600)	1.0 (430)	Steenvoorden (1976)
Zeeland (South-west Netherlands)	11.8 (52)	2.3 (49)	ICW

*NH_4^+ and ortho-P; **total-N and total-P

In marine sediments the groundwater generally can be characterised not only by high chloride concentrations but also by high nitrogen and phosphate concentrations. The contents can reach values of 5 $g.m^{-3}$ P and 50 $g.m^{-3}$ N or even more. This eutrophic groundwater can be found in a small strip along the coast of the provinces Groningen and Friesland in the northern Netherlands and in the largest part of the provinces North-Holland, South-Holland (mid-western Netherlands) and Zeeland in the western part of the country. For the provinces Groningen and Friesland only groundwater samples have been used with a Cl^- concentration of 200 $g.m^{-3}$ or higher and the analyses in North-Holland have been performed on water from gas wells and from wells for cooling water. The high concentrations not only are found at depths of more than 10 meters below sea level, but also in the upper meter of the groundwater (table 3). The absence of nitrate and the high concentrations of ammonium in the groundwater have been caused by the long residence time in the subsoil and the anaerobic conditions in the eutrophic marine sediments.

The saline and eutrophic groundwater can reach the surface waters by several ways. A more or less natural way is seepage. Seepage of loaded water takes place when the pressure in loaded deep groundwater is higher than in the shallow groundwater and when between shallow and deep groundwater a soil layer with a low hydrological permeability is missing. Nutrient rich groundwater can reach the surface water also by an artificial way when an impermeable layer is perforated to win gas or cooling water. In the province North-Holland this has been done on a large scale and the average yearly load per square meter of water in the polder Schermerboezem, having an area of 80,000 ha, is roughly 16 g N and 1.7 g P (Toussaint and Boogaard, 1978).

2.3. *Leaching of fertilizers*

The application of phosphate fertilizers on grassland and arable land has not yet had a measurable effect on the transport of phosphate to the groundwater, when the dose is based on soil fertility and crop production. The phosphate concentrations in shallow

Table 5 Total phosphate concentrations in the drainage water of arable land (Henkens, 1971) and in the shallow groundwater under grassland (Steenvoorden and Oosterom, 1977) for different soil types

Arable land	$g.m^{-3}$ P	Grassland	$g.m^{-3}$ P
Sand	0.02	Sand	0.04
River clay	0.04	River clay	0.05
Old cut-over high moor peat	0.02	Low moor peat	0.11
Newly cut-over high moor peat	0.73		

groundwater under grassland and in drainage water from arable land (table 5) are the same as the concentrations in the shallow groundwater under nature areas on a comparable soil type (table 3). An exception must be made for newly cut-over peat soils, where higher phospate concentrations are measured because of the high mobility of the soil organic matter. The application of large amounts of cattle manure, up to 300 $m^3.ha^{-1}.year^{-1}$ with a dry matter content of 9%, during five years on a sandy soil did not have a measurable effect. One can expect, however, that continuously giving a high dosis of manure will, because of the mobility in the soil of organic phosphates, in the long term result in an increased phosphate leaching (Gerritse, 1977).

The consequences of fertilizing for the nitrogen pollution of the groundwater are dependent on:

- soil use (grassland, arable land);
- soil type;
- the level and type of fertilizing (fertilizers, manure);
- the hydrological situation.

The leaching of nitrogen occurs for nearly 100% in the form of NO_3^-. At the same fertilization level the nitrate concentration in the shallow groundwater is higher for arable land than for grassland (fig. 1). This is caused by mineralization of the organic matter of the remainder of crop and roots on arable land and moreover, by the absence in early spring of a growing crop that can take up the mineralized nitrate.

With respect to the results given in fig. 1 one should take into account that the arable land only received manure which causes a much higher nitrogen leaching than fertilizer gifts. The research on grassland and arable land has been performed in different years so that the total amount of leachate is not the same. The calculated groundwater feed was roughly 100 mm for the grassland plots and 300 mm for the ones on arable land. The leaching of nitrogen at the different manure doses on arable land is roughly 30% of the applied amount of nitrogen. On grassland the leaching was 5%.

Soil type plays an important role in leaching. At a comparable fertilization level the nitrate concentrations in the shallow groundwater under grassland are much higher for sandy soils than for soils with some organic matter in the profile and for soils with

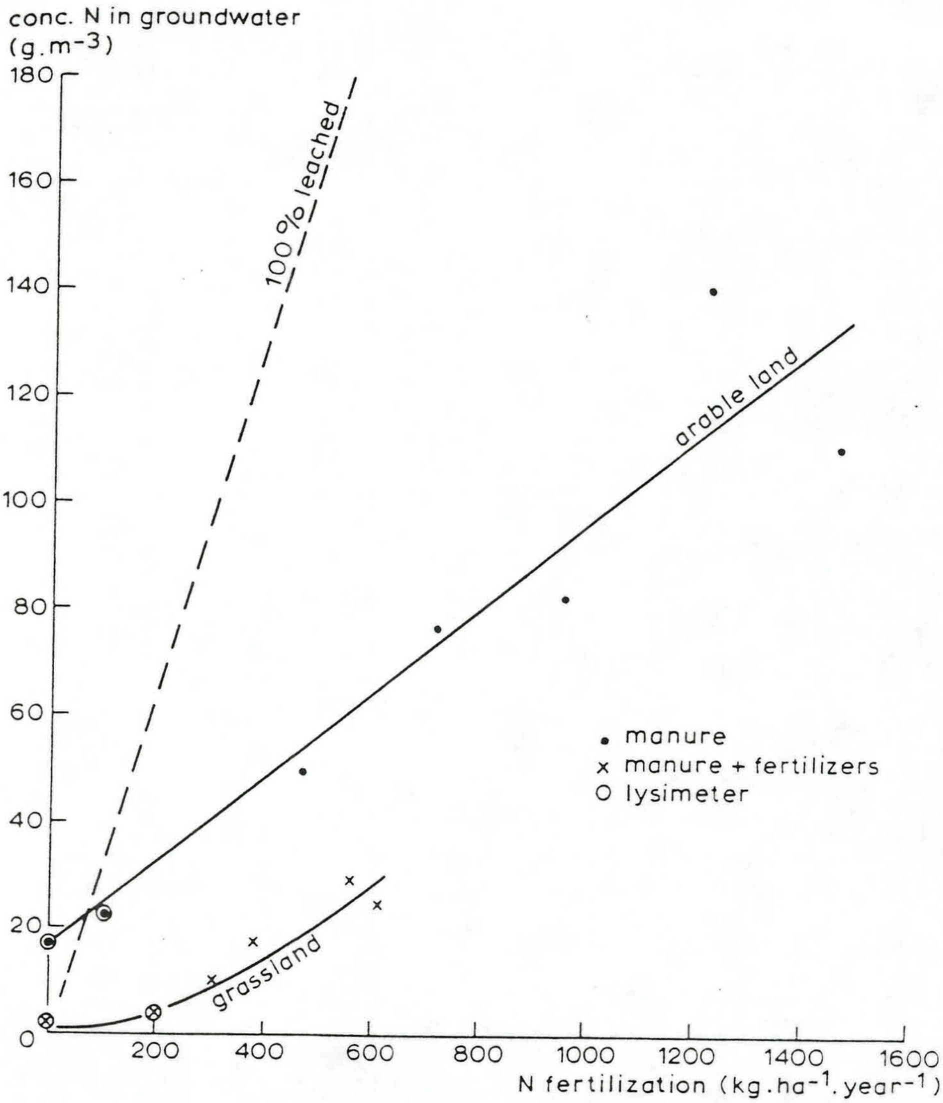

Fig. 1 Nitrate concentration in the upper meter of the groundwater as influenced by nitrogen fertilizing on sandy arable land and sandy grassland in field studies and lysimeter experiments. The line, '100% leached' has been calculated for a net groundwater feed of 300 $mm.year^{-1}$ (after Steenvoorden, 1978b; Kolenbrander, 1978)

a finer texture, like clay soils (table 6). The same effect has been noted for arable land by Kolenbrander (1971).

The drainage situation has a large influence on the leaching of nitrogen. For average climatic conditions in The Netherlands the leaching of sandy soils in a good drainage

Table 6 Influence of soil type on the nitrate concentration in the upper meter of the groundwater under grassland with a comparable fertilization level (Steenvoorden and Oosterom, 1977)

Soil type	Fertilization (N kg.ha^{-1}.year^{-1})			NO_3
	fertilizer	manure	total	N g.m^{-3}
Sand	350	205	555	29
Loamy and peaty sand	335	165	500	1
Clayey sand	435	170	605	5
Heavy clay	360	130	490	0

situation amounts to a loss of 50% of the mineral nitrogen which is present at the end of the growing season. For a moderate drainage situation the percentage is 80 because of an accelerated transport to open water via surface runoff and interflow (Rijtema, 1977). Before the groundwater reaches surface waters the nitrate content can decrease as a result of denitrification in the groundwater during transport and mixing with less polluted groundwater. Type of soil, residence time and geo-hydrological situation are important factors with regard to these processes.

Transport of precipitation surplusses can occur not only in a downward direction but also horizontally through or over the top soil. This will happen when in a certain period the quantity of precipitation exceeds the sum of evapotranspiration, groundwater feed and storage in and on the soil. The phosphate concentration in this surface runoff water depends on soil fertility and soil type (Kolenbrander, 1977; Sharpley et al., 1977). The more the phosphate dosage exceeds the uptake by plants, the higher the storage in the soil and the higher the phosphate concentration in the soil solution (fig. 2). At the same phosphate dosage the potential contribution on a sandy soil is higher than on a clay soil. For the climatic situation in The Netherlands one can expect that surface runoff will be a rare phenomenon and will be restricted to grassland which in general has a higher groundwater level and the soil therefore a lower storage capacity than arable land.

The nitrogen concentrations in the surface runoff are dependent on the mineral nitrogen stock in the percolated soil layer and on the amount of runoff water. The nitrogen concentration, in contrast to that of phosphate, will be in the same range as the concentration in the shallow groundwater. At higher precipitation surplusses the concentration will decrease because of the limited quantity of mineral nitrogen in the soil (Kolenbrander and Van Dijk, 1972).

2.4. *Domestic waste water*

The quantity of nutrients yearly produced as domestic sewage by the average individual include 5 kg nitrogen (Kolenbrander, 1971) and 1.5 kg phosphorus (Koot, 1970). The part reaching open water heavily depends on the way of handling the waste water. Most of the waste water nowadays is treated in biological purification plants.

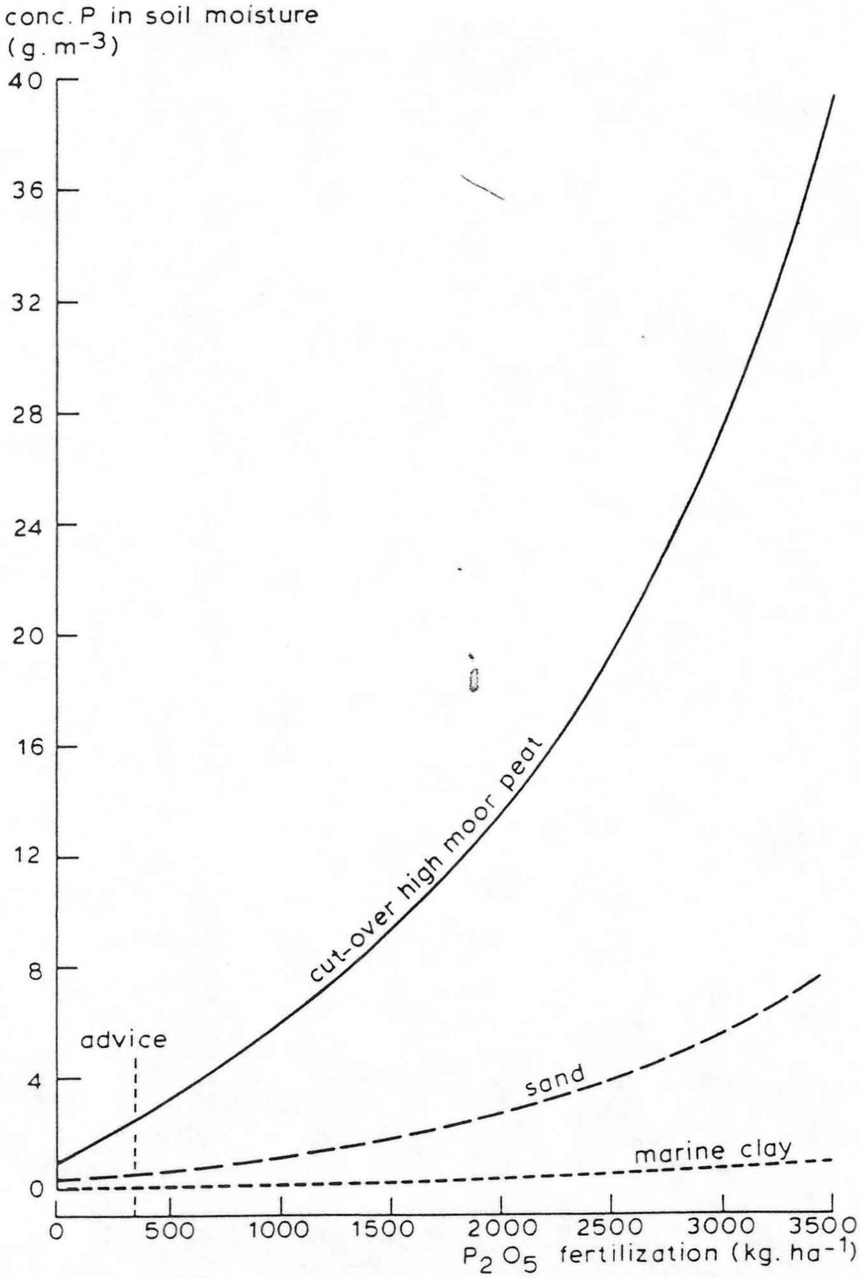

Fig. 2 Relation between total-P concentration in the soil moisture in the upper 20 cm of the unsaturated zone and the cumulative quantity of P_2O_5 added in 5 to 7 years in the form of fertilizers or organic manure (after Kolenbrander, 1977). The broken vertical line indicates the cumulative fertilization advice

An impression of the contribution to the nitrogen and phosphorus load by effluents of biological treatment plants can be given from the results of a research carried out in the area of the Schermerboezem (Hoogheemraadschap, 1976) and the Barneveldse Beek (Beunders, 1978). The concentrations found are slightly influenced by industrial discharge, but in the area of the Barneveldse Beek this applies only to one plant. The average nitrogen contribution per inhabitant equivalent is quite comparable for the two areas (table 7).

For the scattered inhabited sites sometimes a septic tank is used for the treatment, or the waste water is directly discharged on open water. The effect of septic tanks on the reduction of the nitrogen and phosphorus concentrations in effluents is largely unknown.

2.5. *Fresh water inlet*

The inlet of fresh water has several reasons. Rather general motives are the water supply of agriculture and horticulture and the water management for shipping purposes. To combat saline seepage and penetration of saline water near sluices sometimes large quantities of water are used to flush the canals (table 8). Large differences in the quantity

Table 7 Average discharge of nitrogen and phosphate via effluents of water purification plants in the area of the Schermerboezem and the Barneveldse Beek

Item	Dimension	Schermerboezem	Barneveldse Beek
Total-N	N $g.m^{-3}$	35	27
Total-P	P $g.m^{-3}$	–	13.5
Water discharge	$1.inhabitant^{-1}.day^{-1}$	203	200
N-discharge	N $kg.inhab.^{-1}.year^{-1}$	2.6	2.0
P-discharge	P $kg.inhab.^{-1}.year^{-1}$	–	1.0
Number of plants		9	6

Table 8 Quantity of inlet water (mm) to flush saline water ways and used for other purposes in the dry summer of 1976 for different soil types and averaged for a number of water supply areas (Van Boheemen, 1977)

Soil type	Number of supply areas	Flushing	Other purposes
Low moor peat	2	0	123
Marine clay	3	35	36
Low moor peat and marine clay	4	106	120
River clay	3	0	63
Sand	6	0	64

of input water within the same soil type can occur between the water supply areas, this depends on the differences in seepage of saline water, availability of water, etc.

The fresh inlet water in The Netherlands mainly originates from the river Rhine and it has very high nitrogen and phosphate concentrations. The average total-nitrogen and total-phosphate concentrations are roughly 7 $g.m^{-3}N$ and 1 $g.m^{-3}P$ (RWS/RIV/RID, 1976 and 1977). During the transport through channels towards the area where it is needed, important changes in the chemical composition may occur, for example by polluted water discharges. Analyses in the inlet water may be necessary therefore to calculate the total load of surface waters.

2.6. *Discharge of agricultural waste water*

Since the law of 1970 on combating pollution of surface waters is in action, many Water Authorities have played an active role in repelling discharges of agricultural waste waters. It can be presumed that possible illegal discharges will be a scarce phenomenon and thus will give only a small contribution to the mineral load of surface waters. The high concentrations in spread liquid manure, however, can have an important temporary influence on local water quality. These concentrations can be in the order of some thousands $g.m^{-3}$ for nitrogen and some hundreds $g.m^{-3}$ for total-P (Kolenbrander and De La Lande Cremer, 1967). The exact contribution to the total load of surface waters will always remain rather unsure and only a rough estimation.

3. DISCHARGE OF NITROGEN AND PHOSPHATE

The nitrogen and phosphorus compounds added to open water by pollution sources partly will be discharged in either a dissolved form or an undissolved but floating one. Partly the compounds will be involved in physical, chemical and biochemical processes by which products can escape to the atmosphere or can be stored in bottom sediments or in living organisms like reed, fish, etc.

Nitrogen can be lost to the atmosphere by denitrification and NH_3-evaporation. The importance of denitrification mainly depends on the availability of a biochemical oxidizable substrate and on water temperature. Especially in waterways receiving effluents of purification plants favourable denitrification conditions will exist. In such a situation denitrification rates have been measured of 35 $g.m^{-2}.year^{-1}N$ at 4°C and of 330 $g.m^{-2}.year^{-1}N$ at 21°C (Tiren et al., 1976; Van Kessel, 1976).

Little is known about the quantitative importance of NH_3-evaporation. In watery solutions ammonium is dissociated as follows: $NH_4^+ \rightleftarrows NH_3 + H^+$. When the pH-value of the temperature rises, the equilibrium shifts to the right. At a pH of 8 and a temperature of 0°C still all NH_4^+ is undissociated. At a pH-value of 9 and a temperature of 20°C already 30% of the NH_4^+ is in the dissociated form. In surface waters with high ammonium concentrations, for example in eutrophic polders, important quantities

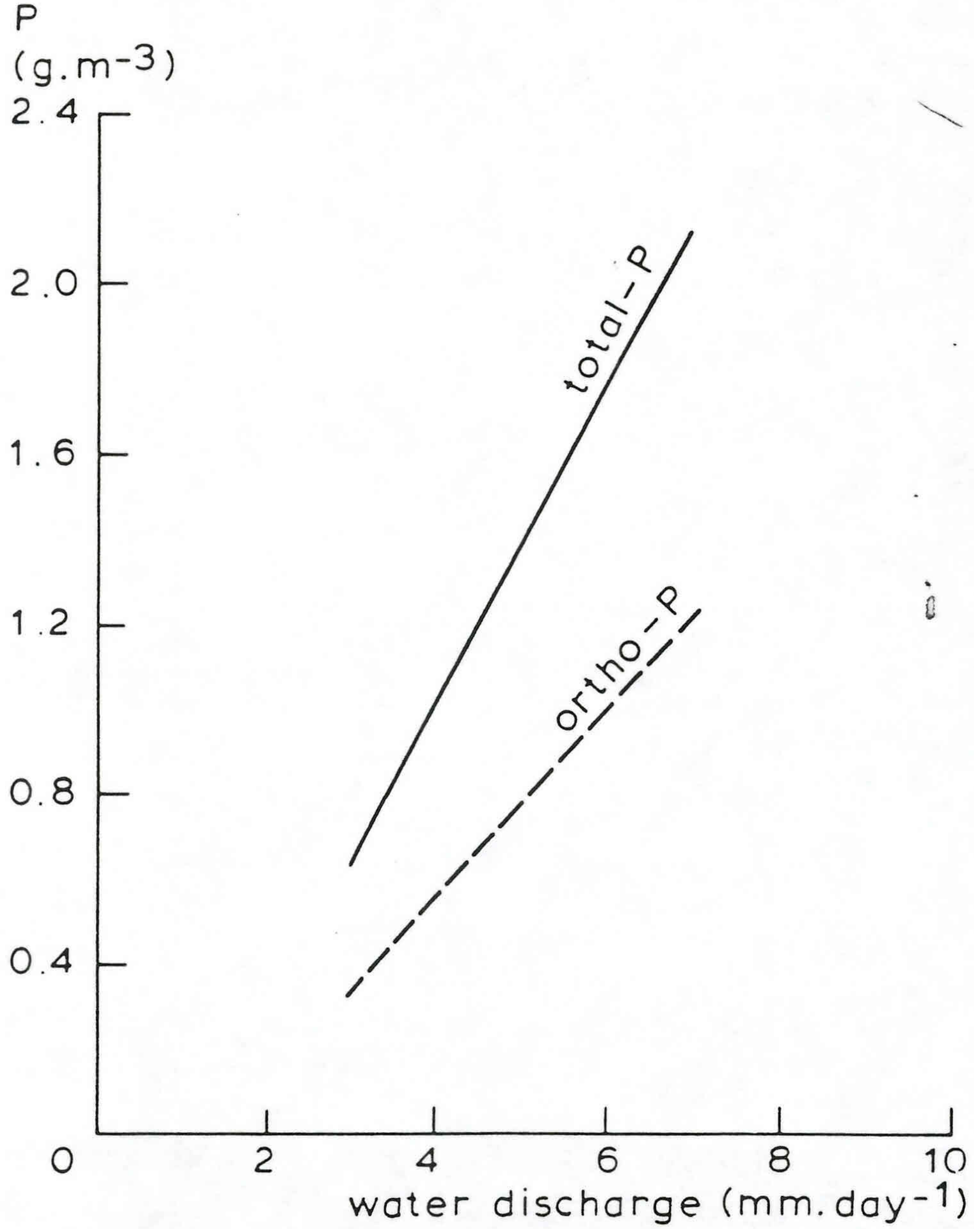

Fig. 3 Relation between the water discharge of the Barneveldse Beek and the concentration of orthophosphate and total-phosphate

of nitrogen may be lost via this process during algal blooms which create favourable conditions by a rise in pH.

Accumulation of nitrogen compounds and phosphate in the bottom sediment can proceed by precipitation of phosphate, adsorption of phosphate and ammonium and by settling of floating particles and dead water organisms and plants.

The disappearance of soluble phosphates from surface waters has been observed in a polder with saline seepage (Steenvoorden and Pankow, 1976) and in the Barneveldse Beek after the discharge of effluent by a purification plant in a dry summer period (Beunders, 1978).

All processes by which accumulation can occur in principle are reversible. When conditions change, nitrogen and partially phosphate can be released from the sediment to the surrounding water. For phosphate this reversibility is proved by the concentration increase during summertime in the lake Veluwe Meer. The only source in that period is the bottom sediment (Hosper, 1978). For the release of nitrogen and phosphate by sediments in some lakes, Vollenweider (1968) mentions values of 0.01 $g.m^{-2}.day^{-1}$ P and 1.2 $g.m^{-2}.day^{-1}$N. The rate of exchange will depend among other things on the concentrations in the water, the oxygen conditions at the water – sediments interface, and the flow velocity of the water.

During a peak flow of nearly five days a close correlation existed between the water discharge and the concentrations of ortho- and total-phosphate (fig. 3). The concentration of dissolved P in the surface runoff from agricultural land depends on the fertility of the soil (Sharpley et al., 1976), which is nearly constant during a peak flow period of some days, so if surface runoff from agricultural land would have been the P-source the concentration in the water in the Barneveldse Beek would have been constant. It is very probable that in dry periods some phosphate is stored in the sediment, which is resuspended in periods with peak flows.

4. BALANCES FOR NITROGEN AND PHOSPHATE IN SURFACE WATERS

4.1. *Nitrogen and phosphate balances in polders*

Nitrogen and phosphate balances are given for three polders, all situated in the western part of The Netherlands. Industrial activities are not present and population density is low. Two of them, Frederikspolder and Veenderpolder, are under grass and consist of a eutrophic peat soil and a peaty clay soil respectively. The polder near Oranjeplaat consists of an arable clay soil. Other differences are the seepage intensity, the ratio between land and open water and the presence of gas wells. The last aspect is important for the Veenderpolder alone (table 9) (Steenvoorden, 1977).

The diffuse potential agricultural sources of pollution are leaching of fertilizers via the groundwater and disposal of agricultural waste water during the stabling period. In the groundwater nitrate could not be analysed, which is not such a big surprise as the level of fertilization is much lower than the level mentioned in table 6. All the nitrogen in the groundwater is in the form of ammonium and organic nitrogen and originates from

Table 9 Some characteristics of the Frederikspolder, the Veenderpolder and the polder near Oranjeplaat

Characteristic	Dimension	Frederikspolder	Veenderpolder	Polder near Oranjeplaat
Area	ha	68	156	40
Land use:				
grassland	%	85	96	–
arable land	%	–	–	99
water	%	15	4	1
Population density	inhabitants.ha^{-1}	0.20	0.25	0.10
Animal density agric. soil:				
cattle*	cu.ha^{-1}	2.1	2.4	–
total stock*	cu.ha^{-1}	2.3	2.7	–
Seepage	mm.day^{-1}	–0.1	0.1	1.0

*1 cu (cattle-unit) is the added value equivalent with that of 1 milk cow

Table 10 Load and discharge of nitrogen and phosphate in surface waters in three polders

Polder	Nitrogen (N kg.ha^{-1}.year^{-1})		Phosphate (P kg.ha^{-1}.year^{-1})	
	load	discharge	load	discharge
Frederikspolder	22	18	6.0	4.3
Veenderpolder	46	21	12.3	3.8
Polder near Oranjeplaat	110	52	19.3	9.1

natural soil resources. The same holds for phosphate (see chapter 2.3). In grassland polders a part of the dung-water produced in the stabling period can reach the ditches when storage facilities are lacking. From a census of storage facilities at farms in the north-western part of The Netherlands it appeared that roughly 8% of the farms had no facilities at all (Hoogheemraadschap, 1976). For the two grassland polders the discharge of agricultural waste water has been calculated to be 8% of the dung-water production in the stable period. The contribution by other sources could be calculated from hydrological and water quality data collected in the field.

The nitrogen and phosphorus load of the surface waters in the polders varies largely (table 10), which to a large extent is caused by differences in seepage intensity and the contribution by gas wells (table 11). The discharge of nitrogen and phosphate via the pumping stations controlling the polder water management is much lower than the total input, so that loss of nitrogen to the atmosphere and storage of nitrogen and phosphorus in the sediment will have played an important role. In the Frederikspolder and Veenderpolder the conditions for NH_3-volatilization have been favourable as in summertime pH-values between 8 and 9 frequently have been measured.

The N/P ratio of the total load lies between 3.5 and 5.5, which makes it very probable that in these polders nitrogen is the limiting factor for algal growth.

Sources which cause an inevitable pollution of surface waters and which can not be forced back by water authorities may be named more or less "natural" pollution sources. This background pollution is created by: precipitation which directly falls upon open water, leaching of natural soil components and seepage of saline water. The contribution by these sources is more than 25% of the total load for the three investigated polders and can even reach 99% (table 11). The artificial pollution is mainly caused by the inlet water from storage canals and saline water from gas wells. Discharge of dung-water from stables account for a relative small part of the nitrogen and phosphorus load in grassland polders. The significance of the last mentioned pollution source will increase in polders with a lower natural load. Especially in polders without seepage of saline water and in soil consisting of mesotrophic or oligotrophic peat this source will be of great concern. Even when all artificial sources of pollution would be eliminated it will be impossible to meet the short and long-term water quality aims (table 1) for a great number of polders in The Netherlands.

4.2. *Nitrogen and phosphate balances in catchment areas*

The nitrogen and phosphate balances have been calculated for the brooks Barneveldse Beek (Beunders, 1978; Steenvoorden, 1978a), Hupselse Beek (Kolenbrander and Van Dijk, 1972; Study Group Hupselse Beek, 1973) and Raalterwetering (Steenvoorden and

Table 11 Contribution (in %) by sources of pollution to the nitrogen and phosphate load of surface waters in three polders

Sources of pollution	Frederikspolder		Veenderpolder		Polder near Oranjeplaat	
	N	P	N	P	N	P
Natural sources:						
precipitation on open water	17	2	2	1	<1	<1
leaching of natural soil constituents	8	60	8	13	6	3
saline seepage	–	–	24	23	93	96
sub-total	25	62	34	37	99	99
Artificial sources:						
domestic waste water	4	4	2	4	<1	1
gas wells	–	–	32	29	–	–
leaching of fertilizers	0	0	0	0	1	0
discharged stable water	14	2	7	1	–	–
inlet water	57	32	25	29	–	–
sub-total	75	38	66	63	1	1
Total	100	100	100	100	100	100

Oosterom, 1973), all situated in the eastern sandy part of The Netherlands.

The soil in an important part of the catchment area of the Barneveldse Beek consists of fine loamy sandy soils. In combination with a bad drainage situation this can lead to surface runoff in wet periods. At a depth of 15 to 20 meters a clay layer with a low permeability can be found. This causes a shallow groundwater flow pattern. Roughly 75 per cent of the population lives in dwellings connected to sewage purification plants of which the effluents are discharged into the surface waters inside the catchment area.

In the area of the Hupselse Beek the top soil consists of medium coarse sand which contains some gravel. The thickness of the layer varies from nearly 0 to 10 meters. The underlying sediment is a clay layer with a very low permeability. The flow in the brook reacts very quickly on precipitation.

The top 40 cm of the sandy soil in the catchment area of Raalterwetering is slightly loamy. At greater depths the soil consists of a well-permeable coarse sandy soil. The groundwater flow pattern is rather deep as compared with the two previous catchment areas and the response of the flow on precipitation is slow.

The discharge of the three areas in the years of investigation was 300, 145 and 150 $mm.year^{-1}$ for respectively Barneveldse Beek, Hupselse Beek and Raalterwetering. Information about soil use, cattle intensity and population density can be found in table 12.

In sandy soil areas the cattle intensity normally lies on a much higher level than in other regions. The consequence might be that in periods that the manure can not be spread over the land, part of the dung-water is illegally dumped into the surface waters. Because of this uncertainty the contribution by diffuse agricultural sources will be calculated from the difference between the total nitrogen and phosphorus discharge from the catchment area via the water and the input by the known sources of pollution. By using this method one neglects the processes in the waterways by which nitrogen and

Table 12 Some characteristics of the catchment areas of the Barneveldse Beek, the Hupselse Beek and the Raalterwetering

Characteristic	Dimension	Barneveldse Beek	Hupselse Beek	Raalter-wetering
Area	ha	15,400	650	1525
Land use:				
grassland	%	59	64	94
arable land	%	6	16	5
water	%	1	1	1
other	%	34	19	–
Population density	$inhab.ha^{-1}$	2.8	0.4	0.3
Animal density agric. soil:				
cattle*	$cu.ha^{-1}$	2.7	2.0	2.2
total stock*	$cu.ha^{-1}$	12.4	4.1	4.5

*1 cu (cattle-unit) is the added value equivalent with that of 1 milk cow

Table 13 Annual load and discharge of nitrogen (N $kg.ha^{-1}.year^{-1}$) and phosphate (P $kg.ha^{-1}.year^{-1}$) for the surface waters of the three sandy catchment areas: Barneveldse Beek (BB), Hupselse Beek (HB) and Raalterwetering (RW)

Sources of pollution	Nitrogen			Phosphate		
	BB	HB	RW	BB	HB	RW
Natural sources:						
precipitation on open water	0.2	0.2	0.2	0.01	<0.01	<0.01
leaching of natural soil constituents	3.0	1.5	1.5	0.3	0.07	0.15
sub-total	3.2	1.7	1.7	0.31	0.07	0.15
Artificial sources:						
domestic waste water and industry	9.5	0.5	0.3	3.6	0.15	0.10
leaching of fertilizers	} 13.0	} 23.5	} 0.9	0	0	0
surface runoff and agric. discharges				−0.2	0.03	−0.01
sub-total	22.5	24.0	1.2	3.4	0.18	0.09
Total discharge	25.7	25.7	2.9	3.7	0.25	0.24

phosphorus can disappear from the water. As a first approximation it is allowable, however.

The natural leaching of P from the soil via the groundwater has been calculated from the total water discharge and the average concentration in the groundwater of the catchment areas. These concentrations are 0.10 $g.m^{-3}$ P for the Barneveldse Beek and the Raalterwetering areas and 0.05 $g.m^{-3}$ P for the Hupselse Beek area. For nitrogen the natural content is roughly 1.0 $g.m^{-3}$ N for all three areas. About pollution by domestic sewage of the scattered population very little information is available. For the calculation of this contribution some approximations have been made. The first is that in the summer half-year no domestic sewage reached the open water because of infiltration into the subsoil. For the winter half-year the nitrogen in the domestic sewage was supposed to reach open water for 100% and the phosphate for 50%.

The load with N and P by natural sources of pollution in the sandy soil catchment areas has a much lower level than in polders (table 13, compare table 10 and 11). As the N/P-ratio of the natural load varies between 11 and 24 for the three investigated brooks phosphate seems to be the limiting element for algal growth. An important part of the artificial nitrogen pollution originates from agricultural sources. For the Barneveldse Beek, the Hupselse Beek and the Raalterwetering the share is respectively 51%, 91% and 31%. The much higher contribution by agriculture for the Hupsele Beek as compared with the Raalterwetering can be explained by the higher percentage of arable land and the speedy and shallow groundwater flow pattern. Both factors have a negative effect on the leaching of nitrate. With a higher residence time of groundwater in the subsoil denitrification and immobilization can reduce the nitrate concentration before the groundwater reaches open water.

Despite the very high cattle intensity in the Barneveldse Beek area, the share of agricultural sources in the N-load is much smaller than for the Hupselse Beek with the same total load. Factors which can explain the lower share in the Barneveldse Beek area are: the higher percentage of nature areas, the lower percentage of arable land and the heavier structure of the top soil (see fig. 1 and table 6).

The contribution by agriculture to the N-load in sandy soil areas is underestimated by the method used, because the N-losses to the atmosphere by denitrification of bottom sediments have been neglected. An estimation for the Barneveldse Beek area, where the disposal of sewage-plant effluents give a fair supply of organic material, leads to a yearly loss of roughly 5.5 $kg.ha^{-1}$ N on average for the whole area. Introducing this correction, the contribution by agricultural sources increases from 13.0 to 18.5 $kg.ha^{-1}$ N and the share in the total load from 51 to 61%. The influence of denitrification by bottom deposits in the other two brooks will be much smaller because there is no disposal of sewage-plant effluents. Moreover, in the Hupselse Beek sometimes high peak flows occur which remove the bottom deposits.

The phosphate load of the Barneveldse Beek is highly influenced by the disposal of sewage-plant effluents and domestic sewage of the scattered population. The share of this source in the total P-load of Barneveldse Beek, Hupselse Beek and Raalterwetering is 95, 60 and 40% respectively. The share of domestic sewage is overestimated by using this calculation method because the probable increase of P in the bottom deposits has been neglected. In eleven sediment cores taken over a distance of some kilometers downstream from a heavy disposal of sewage-plant effluents, the total P-storage has been analysed. Per square meter to a depth of 30 cm an amount of 120 g P is present (Hoekstra, 1979). So the contribution of agricultural sources is underestimated in this area. This is confirmed by some analyses in the runoff from grassland in the Barneveldse Beek area. The concentration of total-P ranged between 0.02 at low flows and 0.9 $g.m^{-3}$ at peak flows (Vasak, 1978; Steenvoorden, 1979). The negative values for agricultural P-sources in table 13 must also partly be explained by the errors made in the measurements or estimations of other sources. A more precise calculation of the P-pollution by agriculture can only be achieved when more information is available about changes in storage in bottom deposits. But also the contribution by domestic sewage from the scattered population is an uncertain item in the P-balance.

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Organic N losses from a poorly drained grassland soil

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Summary

The environmental impact of large inputs of fertilizer N applied to intensive agricultural systems has been well documented. However, in order to determine the full impact of N loading to surface waters, the relative contributions of both dissolved inorganic and organic fractions must be considered. Few studies have so far addressed either the losses of the organic N fraction or the hydrological pathways through which losses occur.

Introduction

Intensively managed grassland systems have long been recognised as potential diffuse sources of pollution to freshwater bodies. Garwood and Ryden (1986) showed that substantial losses of nitrogen (N) could occur from grassland receiving inorganic fertilizer inputs exceeding 250 kg N ha^{-1}. Additionally, a large proportion (50-80%) of the N ingested by grazing cattle is returned to the soil in their excreta and in particular urine (Bristow, 1992)

As yet, few attempts have been made to measure mobile organic N and this may be attributable to the following reasons. Inorganic N was considered to be the major problem and most leaching studies concentrated on this area. Total nitrogen analysis involving kjeldahl digestion is slow, expensive, and involves chemicals that are hazardous to health. Sample collection using porous ceramic cups has problems associated with the potential adsorption of organic compounds onto the ceramic.

Data from a catchment study by Johnes & Burt (1991) however show that losses of organic N may be 40 % of the total N lost. This suggests that there may be a serious shortfall in previous estimates of bio-available N in downstream waters. This paper reports the preliminary results from a three year study of the extent of organic N losses in surface runoff and drainage waters from grazed grassland in SW England, under different fertilizer managements.

Methods

Hydrologically isolated plots (1ha) were set up on grassland with a poorly drained clay loam soil. The plots were grazed continuously by beef cattle, were either drained or undrained and were managed under different fertilizer regimes (0, 200, 300 kg mineral N ha^{-1} yr^{-1}). Water movement pathways (drainage to 85cm, or runoff and surface lateral flow to 30cm), were separately channelled to V-notch weirs. More details of the grassland management and drainage treatments are given in Tyson *et al.* (1992) and Armstrong & Garwood (1991).

Water samples were collected from the outflows of field drains and surface weirs once per day throughout the winter drainage period (Oct-Mar) and more intensively during storm events. The samples were separately analysed for soluble inorganic N and total UV digestible N both before and after filtering (<0.2 microns). The total N fraction was determined using an automated continuous flow method based on conversion of organic N to nitrate by digestion with di-potassium peroxodisulphate at 70°C, catalysed by UV. The nitrate was colorimetrically determined as nitrite by the Griess reaction following reduction by a cadmium/copper reductor. The organic N fraction of the sample was determined by difference.

Results

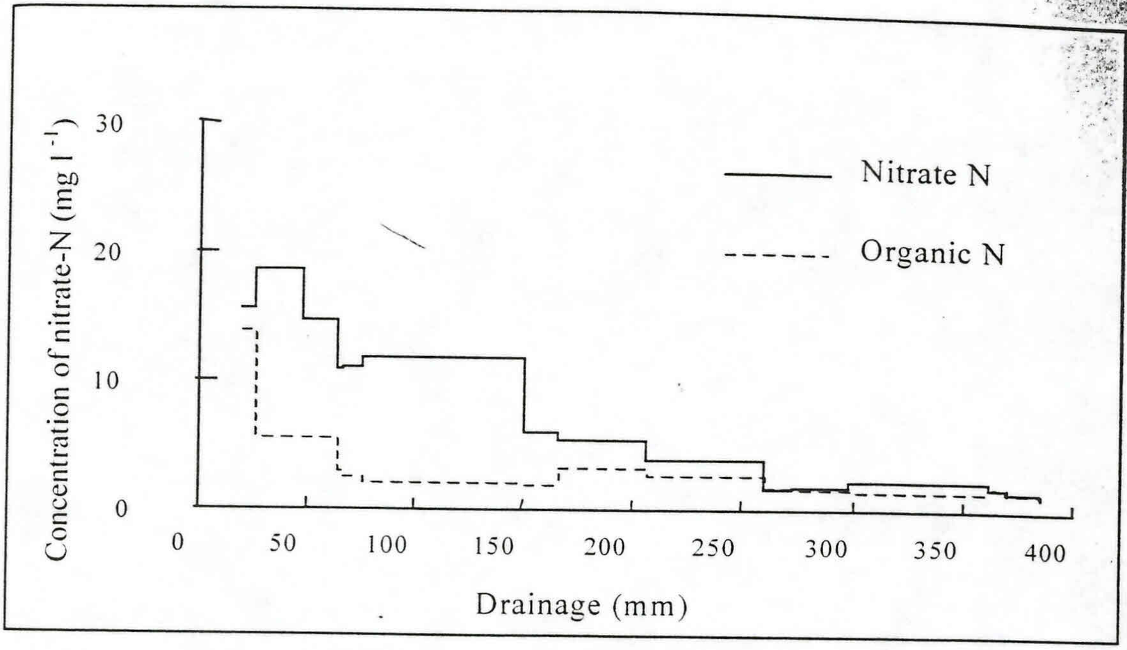
Preliminary results show that losses of total N from pasture receiving 300 kg N ha^{-1} yr^{-1} were higher from drained swards compared with undrained swards (88.8 kg N ha^{-1} and 31.7 kg N ha^{-1}, respectively). The largest organic N concentration (40 mg N l^{-1}) was in drainage to 85cm from drained pasture receiving 300 kg N ha^{-1} yr^{-1}. Overall, the proportion of organic N was generally larger (38% compared with 19%) in runoff and surface lateral flow from undrained grassland receiving the same mineral N input. The largest organic N losses (18.6 kg organic N ha^{-1} yr^{-1}) were from undrained grassland receiving 200 kg N ha^{-1} yr^{-1} and represented 22% of the total N lost.

The largest proportion of N lost in the organic form (60%) was in runoff and surface lateral flow from undrained grassland with ryegrass/white clover swards.

There were insignificant differences in organic N concentrations between unfiltered and filtered samples.

Fig 1. Shows a typical chemograph (elution profile) of inorganic and organic N during a winter drainage period. It shows that although the overall organic N contribution to total N is smaller than that of inorganic N, initial organic N concentrations may be relatively large.

Fig. 1 Concentration profile of inorganic and organic N in surface lateral runoff to 30 cm from grazed grassland receiving 300 kg N ha^{-1} yr^{-1}.



Conclusions

Undrained grassland may yield a larger proportion of total N losses as organic N than drained grassland. A significant proportion of total N losses from grazed grassland may be as organic N and may not have been previously accounted for in existing estimations of N balances and models of the N cycle in pasture systems.

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Leaching of dissolved organic N from grass–white clover pasture in SW England.

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Abstract

Although it has long been suspected that some of the nitrogen (N) lost in waters draining from grassland systems is in soluble organic forms, few studies have attempted to determine the extent of such loss. This paper gives the details of a two year study on losses of organic N in surface runoff and drainage waters from poorly draining grass and grass-white clover grazed lysimeters. The results show that as much as 66% of the total N lost from grass-white clover swards is in an organic form. Organic N represents a significant proportion of total N losses from grazed grassland which has not previously been accounted for in existing estimations of N balances and models of the N cycle in pasture systems.

Keywords: Organic N, N Cycle, total N, grass-white clover, grassland.

Introduction

The use of legumes, and in particular white clover, is fundamental for providing a nitrogen input to organic grassland systems. Grass-clover swards are considered less N leaky in comparison to grass swards receiving mineral N fertilizer. Few studies have so far addressed either the losses of dissolved organic N or the hydrological pathways through which losses occur from pasture. Consequently, models of the N cycle in pasture systems have tended to ignore the contribution of N leaching from organic matter. Data from a catchment study by Johnes & Burt (1991) show that losses of organic N may be 40 % of the total N lost, and suggest that there may be a serious shortfall in previous estimates of N leaching losses. This paper reports the results of the extent of organic N losses in surface runoff and drainage waters from grazed grass-white clover and grass swards in SW England, under different hydrology regimes over two winter drainage periods between 1995-97.

Method

Hydrologically isolated pasture lysimeters (1 ha) were set up on poorly draining clay loam soil in SW England. The lysimeters were continuously grazed by beef cattle, typically from March to October, and received either an application of 200 or 0 kg N $ha^{-1} yr^{-1}$ (grass-white clover). The average percentage clover in the grass-white clover swards was 13 % in 1995 and 9% in 1996 and N fixation levels were estimated to be ha^{-1} yr^{-1} and 34 kg N ha^{-1} yr^{-1} respectively. The lysimeters receiving mineral lizer were drained and the grass-white clover lysimeters were either drained or nned. Each fertilizer treatment was replicated. Water movement pathways inage to 85cm, runoff and surface lateral flow to 30 cm), were separately channelled through V-notch weirs. More details of the grassland management and drainage treatments are given in Tyson *et al.* (1992) and Armstrong & Garwood (1991).

Water samples were collected from the outflows of field drains and surface weirs once per day throughout the winter drainage period (October-March). The samples were separately analysed for soluble inorganic N and total UV digestible N after filtering (<0.2 microns). The dissolved organic N fraction of the sample was determined by difference. The drainage volumes for the two winters of the study were 383 and 364 mm respectively.

Results

Results from both winter drainage periods show that total N losses were larger from drained grass swards receiving 200 kg ha^{-1} yr^{-1} compared with those from drained grass-white clover. The largest proportion of organic N lost was in runoff and surface lateral flow from grass-white clover grassland, which contained up to 66% of total N lost in an organic form (Table 1.) Total N losses (19.8 kg ha^{-1} yr^{-1}, 8.28 kg ha^{-1} yr^{-1}) from the drained grass-white clover swards in 1995-96 and 1996-97 amounted to 30.6% and 24.35 % of the N fixed respectively. Organic N concentrations in surface runoff from grass-white clover exceeded those of inorganic N at the start of drainage and periodically throughout the winter (Fig 1.).

Conclusions

Results suggest there is an increase in N leached in an organic form where clover is present in the sward. Although organic N exports from grass-white clover swards are smaller than those from swards receiving fertilizer, the proportion lost effectively doubles the total N export. A significant proportion of total N losses from grazed grassland may be as organic N and may not have been previously accounted for in existing estimations of N balances and models of the N cycle in pasture systems.

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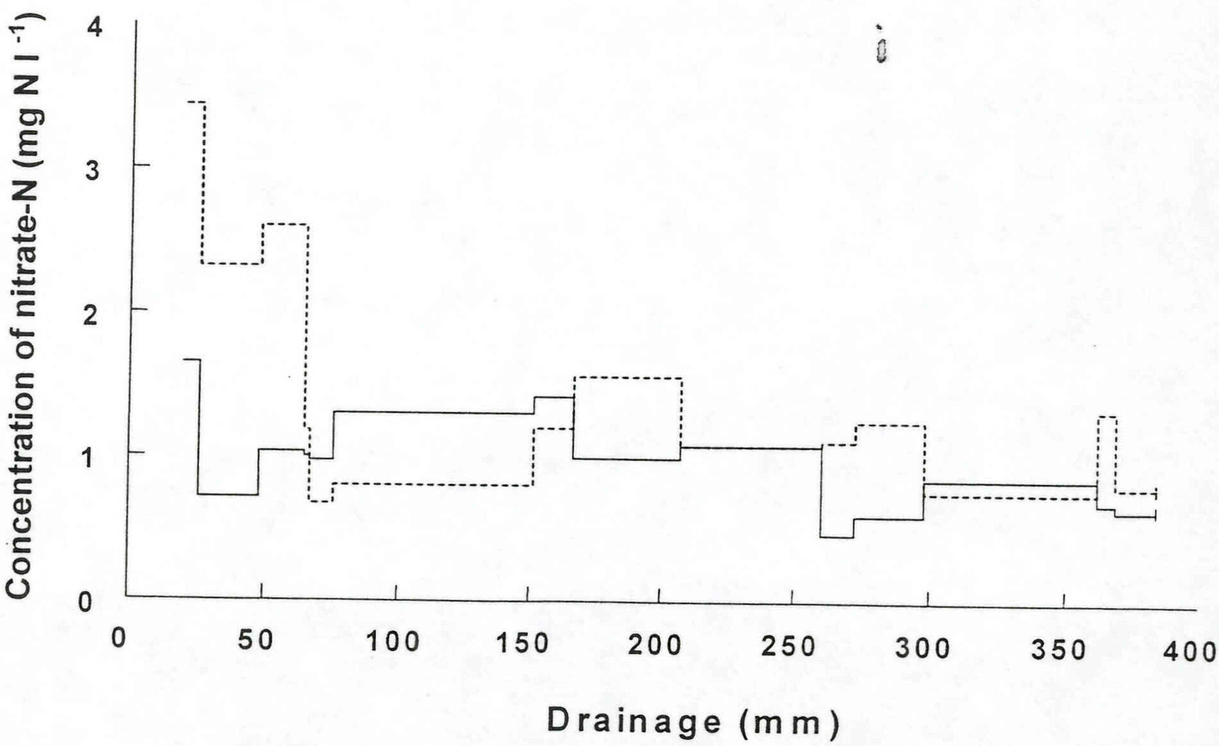
Table 1. Total N and percentage organic N leaching from grass-white clover and grass receiving 200 kg N ha^{-1} yr^{-1}.

Treatment	1995-96 Total N (kg N ha^{-1})	1995-96 Percentage organic N	1996-97 Total N (kg N ha^{-1})	1996-97 Percentage organic N
Grass-clover undrained	8.94	60	5.35	66
Grass-clover drained	19.84	25	8.28	22
200 kg N ha^{-1} yr^{-1} drained	84.66	22	44.9	13

Figure 1. Patterns of inorganic N (solid line) and organic N (broken line) inrunoff from an undrained grass–white clover sward.



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ORIGINAL PAPER

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Mineralization of nitrogen in permanent pastures amended with fertilizer or dung

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Abstract Nitrogen mineralization was measured in three permanent pastures – either fertilized or unfertilized grass, or a mixed grass-clover sward – which were further amended with either fertilizer or cattle dung over a summer growing season. Measurements were made at 4-weekly intervals from June to October. Rates of net mineralization were similar in each of the background treatments (overall mean 0.99 ± 0.091 kg N ha^{-1} day^{-1}) and did not change markedly during the experiment. From the second sampling (July) onwards, rates of mineralization in all the dung treatments were higher than in the control by a factor of up to 2. In the fertilizer-amended treatments, rates were also consistently (but not significantly) higher than in the control. However, the relatively small effect of fertilizer detected at each sampling had a significant cumulative effect by the end of the experiment. There was no interaction between the background and current treatments. Potential mineralization, measured by anaerobic incubation, increased in all the treatments over the period of the experiment, showing an accumulation of readily mineralizable residues. Total N mineralized and the N accumulated during the experiment were calculated and compared. This approach suggests that potential measurements could provide a good estimate of changes in soil N supply that would not be otherwise detectable in changes in soil total N in the short-term.

Keywords Soil fertility · Net mineralization · Potential mineralization · Fertilizer · Dung

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Introduction

Soils vary widely in their physical, chemical and biological properties and in their ability to support plant growth. The time taken for a soil to recover to the equivalent structure of a long-term pasture, following long-term arable cultivation, can be more than 50 years (Low 1955). Along with the physical changes, increasing ranges in complex chemical and biological processes will also become established. These changes in soil characteristics may be accompanied by a gradual accumulation of soil organic matter (SOM) and will combine to determine the overall soil fertility (Clement and Williams 1964; Tyson et al. 1990).

Soil fertility may also be thought of in terms of the relationship between the nutrient requirements of a crop and the ability of the soil to satisfy these requirements. Since the productivity of soil is often driven by the supply of N, an estimation of the release of N from SOM may be regarded as a prime indicator of soil fertility. The capacity of soils under grassland management to accumulate organic N can be large, e.g. equivalent to 160 kg N ha^{-1} year^{-1} (Clement and Williams 1967). However, at any one time, only a small proportion (generally $<2\%$ of total soil N) is present in inorganic forms that will be readily available to plants (Bremner 1965). The proportion of the total soil organic N that is released annually varies widely from $<2\%$ to more than 10%, depending on soil type and conditions (Bartholomew and Kirkham 1960). Thus, the accumulation of organic N in the longer term may not be related directly to the ability of the soil to sustain the rate of N released. However, the net amount of N mineralized (i.e. the difference between gross mineralization and gross immobilization) will indicate the ability of the soil to satisfy the immediate requirements of the plant.

Changes in soil organic N may be relatively rapid in the first years following cultivation and the establishment of a new sward, but the trend becomes asymptotic after some 10–15 years as grassland approaches an equilibrium (Tyson et al. 1990). During the last 15 years of a 30-year study on grass/clover swards, Tyson et al. (1990) found an increase of only about 0.001% in soil total N per annum to 15 cm depth (equivalent to 25 kg N ha^{-1} $year^{-1}$). Even though this rate of change represents an annual increase of about 0.5% of the total N present, much of which may be readily decomposable, such changes are still difficult to detect in the short term, without a large-scale soil sampling programme and a high degree of analytical resolution. The process of N accumulation may be accelerated by the application of fertilizer N and also under grazed as opposed to cut swards, since much of the organic N will be returned in excreted waste (Whitehead et al. 1990).

The recycling of organic materials from both animal excreta and plant residues are important sources of N in grassland production, but their contribution to plant growth is difficult to quantify. Potential mineralization measures the component of SOM that is likely to be mineralized (Hassink 1992) and can therefore be considered to provide a useful measure of fertility. Previous measurements using field incubation techniques (Hatch et al. 1991; Gill et al. 1995) have shown that in long-term grassland the annual release of mineral N from SOM is considerable. There is a need to improve the predictive capability for mineralization so that external N inputs can be optimized (Jarvis et al. 1996). Here we report on both net and potential rates of mineralization in grass swards. Measurements were made over several months, from the time when fresh dung was deposited until the pats had begun to fragment and these were compared with the effects of periodic applications of fertilizer N.

Materials and methods

The study was conducted between June and October 1997 on an established experimental site (Rowden Drainage Experiment) on the farm of IGER, North Wyke Research Station, Devon in S.W. England. Further details can be found in Hatch et al. (2000) but briefly, long-term grazed swards were divided into 1-ha treatment paddocks which had been maintained under the same management systems since 1982. Three background systems were used in the present experiment: (1) a fertilized perennial ryegrass sward (predominantly *Lolium perenne* L.) receiving 200 kg N ha^{-1} $year^{-1}$ (GF); (2) a grass-clover sward containing about 20% clover (GC); (3) an unfertilized grass sward (containing mixed grass species dominated by *Agrostis* L. spp., *Anthoxanthum odoratum* L. and *Holcus lanatus* L.) receiving no fertilizer N (G0). All three treatments had been grazed by beef cattle and received 25 kg P and 50 kg K ha^{-1} in April; fertilizer N was also applied at that time to the GF sward at a rate of 40 kg N ha^{-1}.

Four replicate field plots, approximately 30 m^2, were fenced off within each of the background treatments to exclude cattle during the 1997 grazing season. Within these field plots, three individual sub-plots (each of 6 m^2) were established to provide three current treatments, namely:

1. Control: unamended background treatment i.e. no additional fertilizer added (C).
2. Fertilizer: N-fertilizer applied in three or four applications, totaling 200 kg N ha^{-1} as NH_4NO_3 (F).
3. Dung: artificial dung pats providing the equivalent of 660 kg (total) N ha^{-1} within the dung pat area (D).

There were, therefore, nine current treatments: GFC, GFF, GFD, GCC, GCF, GCD, G0C, G0F and G0D, each separately replicated by four field plots within the background treatments.

After the swards had been cut to a uniform height (4 cm) on 30 May, fertilizer was added to the three current fertilized (F) treatments at a rate of 60 kg N ha^{-1} for the GFF (which had already received 40 kg N ha^{-1}), and 100 kg N ha^{-1} for the GCF and G0F. Fresh dung from dairy cattle was applied as pats to the dung treatment plots (GFD, GCD and G0D) on 25 June. Aliquots of dung (4 kg) were formed into circular pats (0.125 m^2) on top of 500-mm squares of open plastic mesh (10 mm × 10 mm) that had been positioned on the sward. In this way, dung material could pass into the soil whilst the residual material, supported on the plastic mesh, could be lifted periodically to allow soil cores to be taken from beneath them. The fresh dung comprised 7.9% dry matter, 34.5% C and 2.6% N (dry weight basis) with a C:N ratio of 13.1. The swards were cut again on 21 July, after which the GFF, GCF and G0F plots received 60 kg N ha^{-1} and cut once more on 1 September, followed by a further 40 kg N ha^{-1} to each of the fertilized plots.

Measurements were made at approximately 4-week intervals from 16 June to 13 October (17 weeks in total). Rates of net mineralization were determined using the technique described by Hatch et al. (1990). Four (37 mm diameter) pairs of soil cores were taken to a depth of 70 mm from each treatment sub-plot: one core from each pair was placed in a 1-l Kilner jar and the remaining four cores were bulked and analysed for NH_4^+-N. The Kilner jars (four per treatment) were sealed and acetylene (C_2H_2) was added as a nitrification inhibitor to produce a 2% concentration (v/v) in the headspace. The jars were then incubated in holes in the ground, adjacent to the experimental area, for 7 days before analysis.

On the day of sampling, the bulked soil from each sub-plot was crumbled, mixed by hand and the above-ground plant material (shoots and stubble) plus undecomposed root material and stones were removed. Moist soil samples (50 g) were shaken with 2 *M* KCl (250 ml) on an orbital shaker for 2 h. The suspension was then filtered through Whatman No. 1 paper and the filtrate was collected after the first 5 cm^3 had been discarded. The concentrations of NH_4^+ and NO_3^- in the extracts were determined by automated segmented-flow colorimetry (Searle 1984; Kempers and Luft 1988). Net mineralization was calculated from the difference in NH_4^+-N contents of the soil cores at the start and end of the incubation period.

Potential mineralization rates were determined by an anaerobic incubation method (Lober and Reeder 1993) based on a modification of an earlier technique (Waring and Bremner 1964). Soil was air-dried at 30 °C and ground to pass a 2-mm sieve; 5 g was placed in 60 ml polypropylene syringes fitted with Luer-Lok taps. Deionized water (13 ml) was added and air was eliminated by gentle shaking and advancing the plunger, until the soil slurry reached the Luer tip. The syringes were clamped needle-end down to a stand and placed in an incubator at 37 °C. After 7 days of incubation, 37 ml of 2.7 *M* KCl was added to each syringe resulting in a 2 *M* KCl to soil extract ratio of 10:1. The plungers were pulled down to introduce headspace and the syringes were shaken vigorously for 1 h. The soil solutions were filtered and analysed for NH_4^+ and NO_3^- as previously described. Mineralizable N was calculated as the difference between post- and pre-incubation NH_4^+-N values.

An aliquot of each soil sample was dried at 105 °C overnight for gravimetric soil moisture determination. Sub-samples of air dried soil were ground to pass a 2 mm sieve and analysed for total C and total N, using a Carlo Erba NA 1500 analyser (Erba Science UK Ltd). Bulk density was calculated from the dry weight of soil in the cores divided by their volume.

A randomized design was adopted for replicates within each of the background treatments. Data were examined by analysis of variance (ANOVA), grouped regression analysis and Student's *t*-test (using Genstat 5 software) and an antedependence analysis was employed for the repeated measurements of net mineralization on the background treatments (Kenward 1987).

Results

Table 1 shows some of the soil characteristics of the background treatments measured at the beginning of the experiment. Total soil C and N contents increased in the order G0<GC<GF ($P<0.05$), but there were no differences in soil C:N ratio between the treatments. Inorganic N ($NH_4^+ + NO_3^-$) content was similar in each of the background treatments (5.9±1.16, 5.5±0.75 and 5.2±0.96 mg kg^{-1}, respectively) but the NH_4^+: NO_3^- ratio was lower ($P<0.05$) in the GC and GF soils (Table 1). Three weeks after the dung pats had been applied (14 July), there was an increase in soil C content of 18% and 30% in G0D and GCD, respectively ($P<0.05$). However, although inorganic N increased, no significant change in total soil N content was detected at this time and there was a corresponding increase in soil C:N in G0D and GCD with ratios of 9.3 and 10.2:1, respectively ($P<0.05$). The increase in soil C content with dung was short-lived; subsequent measurements showed no differences between treatments in soil C, or in soil C:N. Soil bulk density (BD) was similar in each of the background treatments and, as reported earlier (Hatch et al. 2000), the effect of dung was to lower the BD in G0D and GFD. A similar effect was found for GCD ($P<0.05$) in this study. No effect of fertilizer N (G0F, GCF and GFF treatments) was found in terms of soil total C and total N or BD. The dung pats (in G0D, GCD and GFD) became progressively depleted in both C and N and by September had begun to fragment and were penetrated by new grass shoots. By the end of the experiment, the total C and N contents of the remnants of the dung pats had fallen by 52% and 42%, respectively, with a change in C:N ratio from 13:1 to 11:1 (during the experiment). Soil water content was similar in all the treatments and is shown as a mean (of all the treatments) together with soil temperature, on each sampling occasion (Fig. 1).

Higher ($P<0.05$) soil inorganic N ($NH_4^+ + NO_3^-$) contents (Fig. 2) were measured in the N treatments (G0F, GCF and GFF) in the weeks following two of the fertilizer applications (30 May and 1 September). Where the interval was more than 3 weeks between fertilizer N application (21 July) and sampling (19 August), levels of inorganic N were no longer significantly different, except in the long-term fertilized (GFF) treat-

Table 1 Soil characteristics of three grass swards, unfertilized (*G0*), grass/clover (*GC*) and fertilized (*GF*). Values are means ($n=4\pm$SEM) and relate to dry wt. soil

Background treatment	G0	GC	GF
% C	4.39 (0.158)	5.17 (0.236)	5.98 (0.332)
% N	0.54 (0.016)	0.63 (0.024)	0.73 (0.037)
C:N	8.1 (0.25)	8.2 (0.14)	8.1 (0.13)
pH (water)	5.3	5.5	5.6
NH_4^+:NO_3^-	7.1 (1.53)	3.8 (0.34)	3.2 (0.91)

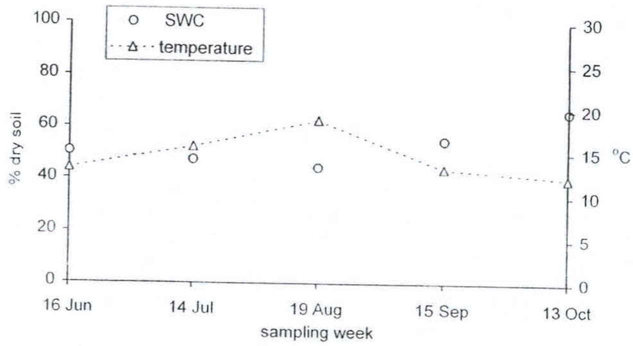

Fig. 1 Soil water content (○) and temperature at 7 cm (△) in a silty clay loam under permanent pasture measured from June to October±SEM (*vertical bars*)

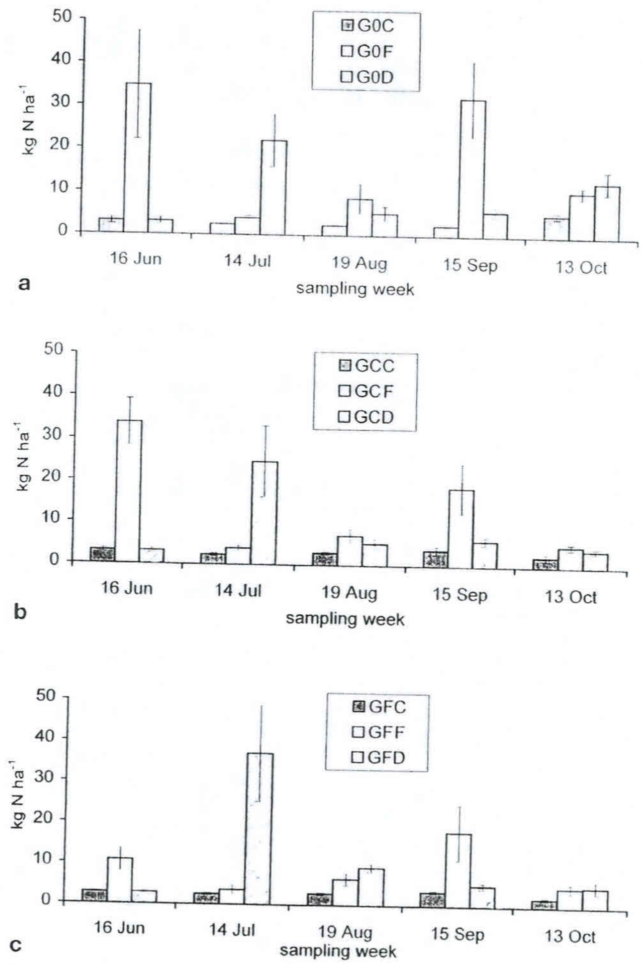

Fig. 2a–c Inorganic N ($NH_4^+ + NO_3^-$) in soil from background treatments: **a** unfertilized (*G0*), **b** grass/clover (*GC*) and **c** fertilized (*GF*) swards, shown as histograms which received either zero N (*C, shaded*), fertilizer N on 30 May, 21 July and 1 September (*F, open*), or dung applied 25 June (*D, part shaded*) ±SEM (*vertical bars*)

ment ($P<0.05$). On 14 July, higher levels of soil inorganic N ($P<0.05$) were found under all the dung treatments (G0D, GCD and GFD). By the next sampling date (19 August), levels had fallen and were similar to the controls. However GFD, which had the highest value on 14 July, remained significantly higher than GFC ($P<0.05$). Over the period of the experiment, NH_4^+:NO_3^- ratios in the controls were, on average, 8.1:1, 4.7:1 and 3.9:1 for the G0C, GCC and GFC soils, respectively, but the distribution between the two forms of N became more even with the addition of either fertilizer or dung. Thus at the end of the experiment, NH_4^+:NO_3^- ratios were 1.5:1, 1.4:1 and 1.6:1 for the G0F, GCF and GFF soils and 5.1:1, 4.5:1 and 3.9:1 for the G0D, GCD and GFD soils, respectively.

Rates of net mineralization were similar on each sampling occasion in the three control treatments (G0C, GCC and GFC) throughout the experiment (overall mean 0.99 ± 0.091 kg N ha^{-1} d^{-1}); each followed the same seasonal pattern with a small increase in September followed by a decrease in October (Fig. 3). From 14 July onwards, net mineralization was higher ($P<0.05$) in the dung treatments (G0D, GCD and GFD) than the controls, with the highest rates in each case recorded in September. After this, control and dung values tended to converge. The fertilized treatments (G0F, GCF and GFF) recorded consistently higher rates of net mineralization than the controls on each sampling occasion, but the differences were not significant, with the exception of GFF on 13 October ($P<0.05$). However, antedependence analysis showed that starting from the August sampling, a cumulative difference in all background treatments was established by the amendments of N, as well as dung ($P<0.001$), which persisted through to the final sampling in October. Mean rates obtained from fitted regressions were used to compare rates of mineralization, and estimates of the total amounts of N mineralized in each treatment were obtained for the experimental period of 119 days (Table 2). Within each of the background treatments

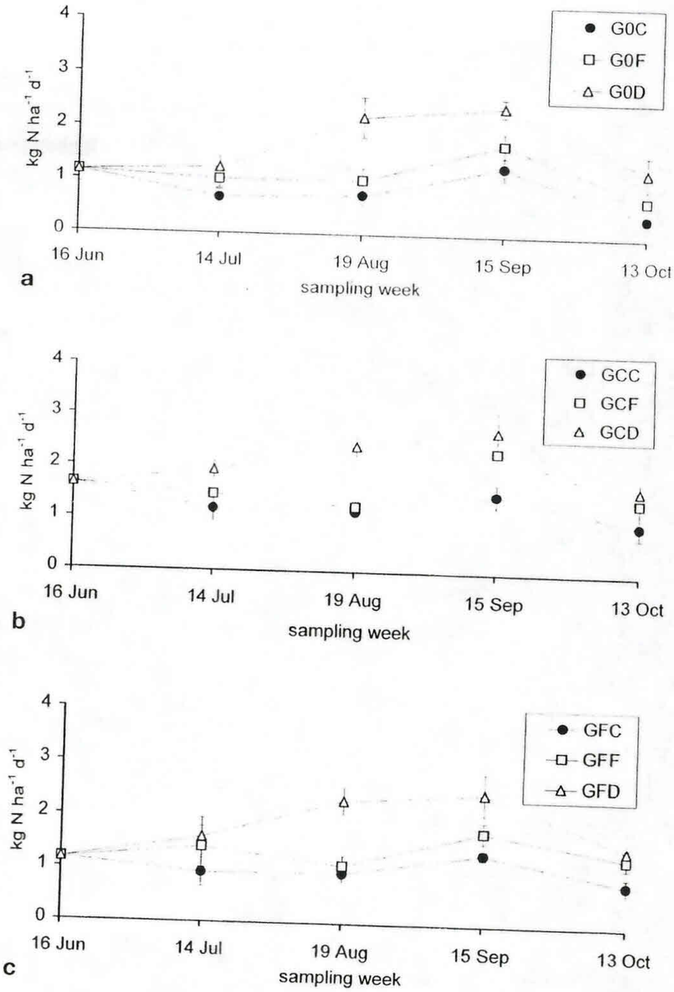

Fig. 3a–c Net mineralization in soil from background treatments: **a** unfertilized (*G0*), **b** grass/clover (*GC*) and **c** fertilized (*GF*) swards which received either zero N (*C*, ●), fertilizer N on 30 May, 21 July and 1 September (*F*, □), or dung applied 25 June (*D*, △) ±SEM (*vertical bars*)

Table 2 Bulk density (measured in September 1997) and mean rate of increase in potential mineralization ($n=4\pm$SEM) over an experimental period of 119 days from which the overall increase in potential over the same period is derived for comparison with net mineralization.

Treatment	Bulk density	Potential mineralization		Net mineralization (kg ha^{-1})[b]
		Rate of increase (µg g^{-1} day^{-1})[a]	Overall increase (kg ha^{-1})	
G0C	0.85	0.42 (0.049)	30	92
G0F	0.89	0.53 (0.061)	39	134
G0D	0.71	0.71 (0.104)	42	212
GCC	0.77	0.83 (0.140)	53	142
GCF	0.84	0.85 (0.273)	60	191
GCD	0.67	1.43 (0.308)	79	259
GFC	0.78	0.52 (0.153)	33	120
GFF	0.81	0.60 (0.110)	40	165
GFD	0.62	1.03 (0.286)	53	233

[a] Obtained from the slopes of the regressions using the data shown in Fig. 4
[b] Obtained from the mean of the rates over the experimental period (Fig. 3)

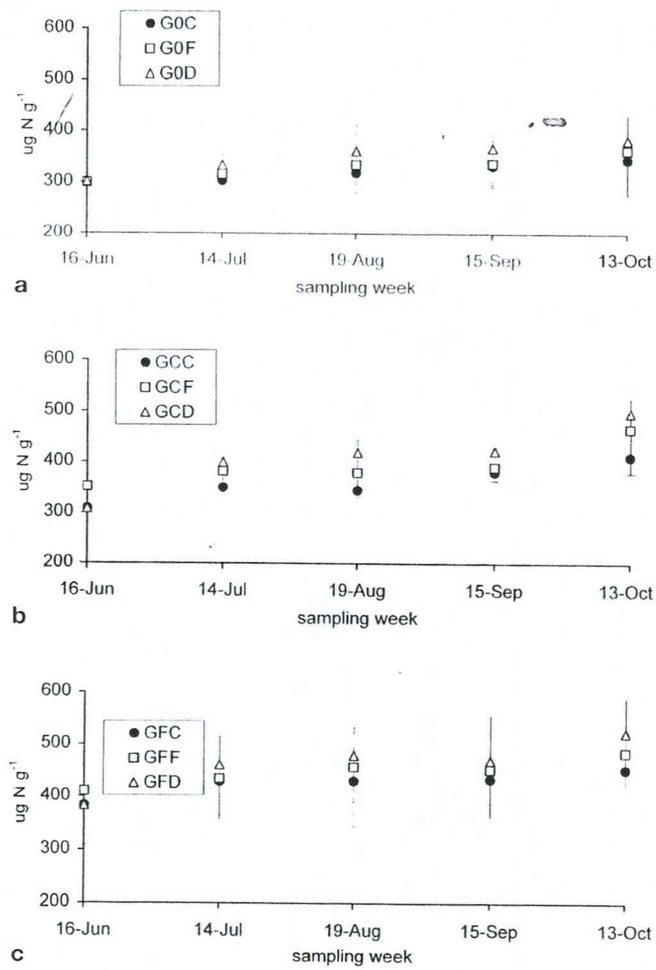

Fig. 4a–c Potential mineralization in soil from background treatments: **a** unfertilized (*G0*), **b** grass/clover (*GC*) and **c** fertilized (*GF*) swards which received either zero N (*C*, ●), fertilizer N on 30 May, 21 July and 1 September (*F*, □), or dung applied 25 June (*D*, △) ±SEM (*vertical bars*)

(G0, GC and GF), the amended sub-treatments were in the order C<F<D ($P<0.001$). There was no interaction between the background treatments and the amended treatments.

At the start of the experiment, potential mineralization in the background treatments (Fig. 4) was in the order 301, 308 and 383 μg N g^{-1} (dry soil) for G0, GC and GF, respectively, but the values were not significantly different. On any particular sampling date, no effect of fertilizer or dung could be detected, but there was a consistent trend for an increase in potential mineralization in all treatments during the experiment. In the control treatments, the increases were between 14 and 33%, the largest being in GCC, from 308 to 410 μg N g^{-1} (dry soil). Using a grouped regression analysis, it was established that the trend for increasing potential mineralization with time was significant ($P<0.001$) in all treatments (Fig. 4). Fitted regressions showed that although the rates of increase (i.e. slopes) were similar in all treatments, the dung treatments had consistently higher values ($P<0.05$) than the other two treatments (i.e. significantly different intercepts). In Table 2, overall increases in potential mineralization for the period of the experiment (calculated from the slopes of the regressions) are compared with the net N mineralized (measured by jar incubation). There was an increased potential in the controls (G0C and GFC) of about 30 kg N ha^{-1}, but a nearly two-fold increase in the control of the clover-based sward (GCC) and in all treatments receiving either fertilizer or dung. Net mineralization values were some 3–5 times greater than potential measurements.

Discussion

Soil characteristics of the background treatments were related to their long-term management in that soil C and N contents increased with the scale of previous N inputs, i.e. unfertilized<clover<fertilized. The greater amounts of N mineralized in the GCC and GFC treatments probably reflected the return of higher quality plant residues (with a lower C:N ratio) than those in the G0C treatment (Ledgard et al. 1998). There were also high levels of inorganic N in the soils following the application of fertilizer. Soil sampling was timed to be at least 2 weeks after fertilizer was applied and there was no indication that high levels of inorganic N interfered with the measurements of mineralization. Inorganic N was also higher in the soil under dung and this persisted for the first few weeks, following placement of the dung pats. Dickinson and Craig (1990) and Lovell and Jarvis (1996) also found that an increase in inorganic N under dung pats lasted for only a few weeks after deposition. Despite large inputs of N, in the present study the relative stability in the soil inorganic N pools would suggest rapid immobilization by plants and/or soil microbiota, as found by Jackson et al. (1989).

Higher rates of net mineralization were measured in all the dung treatments from July to September, after which rates began to converge. This pattern is consistent with the breakdown of a finite amount of readily available labile C substrate (i.e. from dung) and resembled the same pattern of increased microbial respiration when dung was added to soil, as reported by Lovell and Jarvis (1996). In a grazed pasture, the effect of dung will be localized, e.g. over a grazing season (180 days), and assuming average stocking rates of cattle and no overlapping, dung would cover about 260 m^2 in a 1-ha pasture. This represents less than 3% coverage of the 1-ha grazed area and so the considerable increase in net mineralization that was found in the dung treatments would translate into only a small change in the actual amount of N mineralized. Although the increases in rates of net mineralization were lower in the N treatments than with dung, the effect of N applies to the whole area of sward and would, therefore, result in larger total amounts of N mineralized per unit area. For example, Gill et al. (1995) found that net annual mineralization was 317 kg N ha^{-1} in a fertilized and only 135 kg N ha^{-1} in an unfertilized grass sward.

Whereas dung had a major impact on mineralization rates by increasing soil C directly, the effect of fertilizer N was less pronounced since it increased C only indirectly by improving plant growth and increasing the return of plant residues, as was shown in arable (Glendining et al. 1992) and pasture (Gill et al. 1995) soils. However, over the course of this experiment a statistically significant difference was found in the overall increase in rates of net mineralization in N treatments compared with the controls. Therefore, small, but consistent changes in the short term that are difficult to detect because of soil heterogeneity will contribute to the establishment of longer-term changes that develop in 'improved' grassland systems. This effect has also been described recently (Hatch et al. 2000) for two of these same background treatments (G0 and GF).

Potential mineralization is usually measured only as an index of the general nutrient supplying ability of a soil. For example, there is a range of soil tests that are intended to indicate the amount of N that may become available for crop growth by mineralization of SOM during the growing season (Gianello and Bremner 1986). However, it is unlikely that any one particular test will be able to predict accurately the release of soil N, since the continual return of plant litter and recycling of N cannot be accounted for adequately. A standard test was not identified in the review of methods undertaken by Meisenger (1984). The situation is further complicated by changes in the composition and activity of the soil microbiota in response to the composition of SOM residues, or during seasonal variations. There is unlikely to be a simple relationship between SOM accumulation and subsequent net release because of the many physical, chemical and climatic/seasonal conditions that will affect soil microbiota (Nannipieri et al. 1990). Only moderate or poor correlations were found with different chemical indexes of soil N availability (Hong et al. 1990), but more promising results have been obtained with soil incubations (both aerobic and anaerobic) when compared with plant uptake (Fyles et al. 1990). Soil incubation has also been used to measure potential mineralization in whole soil and light fractions of SOM, which can be used to assess the labile component of SOM that is biologically mediated (Barrios et al. 1996).

In the present study, potential mineralization (anaerobic incubation) was measured regularly and in parallel with net mineralization. It was possible, therefore, to examine changes over an extended period and to identify a clear increase in potential mineralization in all treatments, representing an accumulation of readily mineralizable (i.e. labile) residues. This trend occurred concurrently with the net release of NH_4^+ from SOM, estimated by measurements of net mineralization. With no additional 'inputs' (see G0C and GFC), potential mineralization might be expected to remain unchanged, or even to decline over a season, but returns of organic N from plant shoots and roots contribute to the net accumulation of SOM, particularly in undisturbed grassland soils (Jarvis et al. 1996).

From previous records (K. Tyson, personal communication) covering the 10-year period (1987–1996) prior to this experiment, the total soil N had increased, on average, by 41, 44 and 83 kg N ha^{-1} $year^{-1}$ in the G0, GC and GF swards, respectively. It is reasonable to assume that these amounts are probably somewhat higher than when our study was made (1997), since the rate of accumulation lessens in a soil as it approaches an equilibrium (Tyson et al. 1990). Therefore, the rates derived from soil analysis over the long term appear to be very similar to those calculated from the increase in potential mineralization values that were measured in the present study (Table 2). We therefore suggest that the periodic assessment of potential mineralization could be used as an indicator of the accumulation of readily mineralizable residues which then contribute to the release of inorganic N over the succeeding growing season(s).

The contrasting result of net mineralization, whilst at the same time registering an increase in labile residues (i.e. potential mineralization) is an enigma which is frequently encountered in the N cycle of soils under semi- or permanent grassland. Thus, it is quite possible to have a positive net mineralization combined with a positive accumulation of organic N. This can be explained if the total inputs of N from fertilizer, atmospheric deposition, in precipitation and N fixation, exceed the total outputs of N in animal products and losses through denitrification, ammonia volatilization and leaching (Jarvis et al. 1995; Whitehead 1995). Moreover, measurements of net mineralization, based on fluxes through the inorganic N pool, will include N that has been recycled perhaps several times through the soil system (Hatch et al., in press). Consequently, net values may not necessarily relate directly to the overall change in organic N when expressed on an annual basis.

Whilst the actual mineralization rate of new labile residues over succeeding seasons still needs to be established (depending on local climatic conditions etc.), the similarities between increases in soil organic N and potential mineralization warrant further investigation. Regular assessments using potential measurements could provide a more accurate estimate of changes in soil N supply (and perhaps other aspects of soil fertility) than would be identifiable from soil total N analyses in the short term, or from other methods which rely on chemical extraction.

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Relationships between soil thermal units, nitrogen mineralization and dry matter production in pastures

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Abstract. Nitrogen (N) is of environmental concern if it leaches or is released as nitrous oxide (N_2O). In order to utilize N efficiently in grazed pasture systems, the fluxes of N from various sources need to be quantified. One flux is N mineralization from organic sources. Previous work has examined incubation and chemical extraction of soils as methods to determine N mineralization potential. This paper re-examines new and previously published data on net mineralization, with the aim of examining the relationships between soil thermal units, net N mineralization (measured using acetylene incubations) and dry matter production in pastures. Net N mineralization is expressed as N turnover (net N mineralization as a % of total soil N). Relationships are developed between soil thermal units, dry matter production, and N turnover. These relationships have potential in advising farmers on potential N mineralization from soil organic matter. A second use of such relationships is the modelling of N transformations in pasture systems. Further work should explore the effect of soil moisture on such relationships and examine the relationship between soil thermal units and uptake of N by pasture.

Keywords: Nitrogen, mineralization, models, prediction, soil temperature, pastures

INTRODUCTION

Soil inorganic-N may be derived from fertilizer, the mineralization of organic matter and deposition on to the pasture from atmospheric and animal sources. Grassland soils are often high in organic matter and the potential supply of inorganic-N released by mineralization can be correspondingly large (Jarvis *et al.*, 1996). Inorganic-N may undergo further transformations such as denitrification, nitrate leaching, plant uptake and immobilization. In the interests of efficient farming practices, for both environmental and economic goals, it is desirable for inorganic-N from all sources to be utilized efficiently by pasture plants. N is of environmental concern if it leaches as nitrate (NO_3^-), (Jarvis, 1992), or if N_2O is produced since it is involved in global warming and ozone depletion (Crutzen, 1981; Robertson, 1993): both emissions may be substantial in grassland especially when grazed (Jarvis *et al.*, 1995). To achieve efficient N utilization and avoid excess N in the soil, the rate of soil inorganic-N production from soil organic matter must be known.

Methods to determine the potential mineralization indices of soils include laboratory and field incubations, chemical extractions, measurement of N mineralization in the field, and ^{15}N labelled fertilizer techniques (Jarvis *et al.*, 1996). Another method to predict N mineralization examined the density fractions of soil macro-organic matter (Hassink, 1995a). While good agreement can be achieved between different laboratory procedures (Gianello & Bremner, 1986; Groot & Houba, 1995), few if any tests relate the inorganic-N measured, during incubation or extraction, to either the release under field conditions or to actual N uptake and dry matter (DM) production of pasture. At best these laboratory tests provide measurements of the potential to release N measured under specified conditions (Jarvis *et al.*, 1996).

Factors which influence the rate of soil N mineralization include soil moisture, soil pH and soil temperature (e.g. Stanford *et al.*, 1973). Soil thermal units (STU) have been used to predict N mineralization in sludges (Honeycutt *et al.*, 1988), manures and crop residues added to soils under field conditions (Honeycutt & Potaro, 1990). The approach has been further developed, to estimate the availability of N to plants from soil organic matter and other organic N sources, and to improve N use efficiency in cropping systems (Honeycutt *et al.*, 1994).

The method has been used over a limited range of conditions and has not been applied to grazed pasture systems. There are few comprehensive datasets, containing field data on N mineralization rates and soil temperature, which allow relationships between N mineralization and STU to be examined. Gill *et al.* (1995) and Hatch *et al.* (1990, 1991) measured net mineralization in pastures using soil core incubation with acetylene inhibition of nitrification. In these studies variation in soil temperature accounted for a statistically significant proportion of the variation in net mineralization rates recorded. The objective of this study was to re-examine these datasets, and some new New Zealand data, to try and determine relationships between net N mineralization, dry matter production and soil temperature in pasture systems.

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MATERIALS AND METHODS

At 3 sites in the Waikato region of New Zealand net N mineralization was measured under ryegrass/white clover (*Lolium perenne* L.–*Trifolium repens* L.) pastures. These pastures were on a free draining silt loam soil and had received split dressings of urea totalling 0, 200, or 400 kg N/ha per year. Net N mineralization was measured, at each site, using the method of Hatch *et al.* (1990). In brief, 12 soil cores (7.5 cm × 2.5 cm) were incubated in a glass jar with acetylene (2% v/v), in the field, for a 14 day period (Hatch *et al.*, 1990), replicated thrice. Jars were placed in holes in the ground adjacent to experimental areas. After 14 days the cores were sieved and extracted with 2M KCl and the soil water content determined. The amount of inorganic-N in the soil cores after incubation, less that initially present, was taken as the amount mineral- ized. Methods used to determine soil inorganic-N and soil total N are described fully in Hatch *et al.* (1990). Previously published net mineralization data measured in the UK (Gill *et al.*, 1995; Hatch *et al.* 1991, 1990) were examined as described below.

DM production data (coinciding with the 14 day mineralization measurements) were also available for part, or the whole, of these experiments. In total 13 datasets were available, varying with time of sampling, soil type, pasture composition, N fertilizer history and sampling depth (Table 1). Soil temperatures at 10 cm depth were either recorded during the experiments or obtained from meteorological station records at the study sites. Total soil organic-N was calculated for the sampling depth used, from the total N concentration in the soil and the soil bulk density. Net N mineralization is expressed as a percentage of the total soil organic-N pool and is referred to as N turnover:

$$\%\text{N turnover} = \frac{\text{Net N mineralization}}{\text{Total soil organic N}} \times \frac{100}{1}$$

Soil thermal units (STU) over the whole of the measurement periods were calculated by summing the average daily soil temperature above 0 °C (Honeycutt *et al.*, 1988). Relationships between STU, DM production and N turnover were determined from both the data obtained in New Zealand and the UK (Table 1).

RESULTS AND DISCUSSION

A summary of the total STU, DM production, N turnover and average N content of DM is presented in Table 2. Net N mineralization rate fluctuated considerably with time (Fig. 1a). Converting net N mineralization to N turnover and plotting cumulative N turnover versus STU produced a close relationship between the two variables (Fig. 1b). It is clear that in this example from a long-term perennial ryegrass sward, that with some initial characterization of the soil (bulk density and total soil N), STU could be an effective predictor of N turnover.

Temperature is also a controlling factor for plant growth (Langer, 1973; Downs & Hellmers, 1975). Plotting the measured DM, as cumulative DM production, versus STU also resulted in a close relationship (Fig. 1c). As well as cumulative N turnover and cumulative DM production being expressed with respect to STU a correlation also exists when cumulative N turnover is plotted versus cumulative DM production (Fig. 2). For each of the 13 datasets considered, over a range of management conditions, relationships could be demonstrated for cumulative N turnover versus STU; cumulative dry matter production versus STU and cumulative N turnover versus cumulative DM production (Table 3). While a strong correlation occurs between cumulative DM production and N turnover this does not necessarily mean there is a causal link. However, if other factors are at an optimum (e.g. soil moisture, nutrient supply, temperature) then N availability could be a major factor in determining DM production.

Pooling all the datasets and plotting cumulative N turnover versus STU produced no clear overall relationship, suggesting that N turnover was site or soil specific. Plotting pooled data of cumulative DM production versus cumulative N turnover clearly showed three distinct subsets (Fig. 3). DM yield per unit of N turnover was lowest in the grouped New Zealand data (grass/clover pasture) and highest in that for the all grass only treatments (sets 1–7, 9, 10), with the exception of dataset 6, a grass/clover site. Intermediate between these two groupings was a third subset of data (8) from a grass/clover site. These groupings suggest that the correlation between DM yield and N turnover may be influenced by pasture composition, possibly due to the plant material returned to the pasture decomposing and mineralizing at different rates. Alternatively differences could be related to the

Table 1. Summary of the 13 datasets used to describe the relationships between soil thermal units, dry matter production and N turnover.

Reference	Dataset	Date of start of experiment	Number of days measured	Total soil N (kg N ha)	Pasture type†	N applied‡ (kg N ha per year)	Soil sample depth (cm)	Soil texture	Prior management history§
Gill *et al.*, 1995	1	24 Apr 1992	217	5940	pr	200	10	clay	200, rg
	2	24 Apr 1992	217	4960	pr	200	10	clay	200, rg
	3	24 Apr 1992	217	5120	pr	0	10	clay	0, rg
Hatch *et al.*, 1990	4	10 Mar 1986	120	4620	pr	0	15	loam	420, c
	5	2 Mar 1987	85	4620	pr	0	15	loam	420, rg
	6	2 Mar 1987	85	4620	pr wc	0	15	loam	0, rg
	7	4 Mar 1987	155	3150	pr	0	15	clay	0, lg
Hatch *et al.*, 1991	8	10 Mar 1988	210	4620	pr wc	0	15	loam	0, rg
	9	10 Mar 1988	210	6090	pr	0	15	loam	420, rg
	10	10 Mar 1988	210	6090	pr	420	15	loam	420, rg
New Zealand data (unpublished)	11	1 Apr 1996	152	5850	pr wc	0	7.5	silt loam	0, rg
	12	1 Apr 1996	152	4200	pr wc	0	7.5	silt loam	200, rg
	13	1 Apr 1996	152	3300	pr wc	0	7.5	silt loam	400, rg

† pr = perennial rye grass (*Lolium perenne*); wc = white clover (*Trifolium repens*). ‡ N applied (as NH_4NO_3) during measurement period. § N applied (kg ha per yr) prior to acetylene incubation measurements followed by defoliation treatment where rg = rotational grazing, lg = lax grazing and c = cutting.

Table 2. Summary of STU, DM production, N turnover and % N in herbage.

Dataset	Total STU (°C)	Total DM (kg/ha)	Total N turnover (%)	Mean herbage N (%)
1	2528	8130	4.63	3.39
2	2528	9460	463	2.82
3	2345	4600	1.68	2.31
4	1241	3102	1.86	2.13
5	660	5873	1.96	3.20
6	660	5743	1.51	2.90
7	1331	4642	1.23	1.72
8	2677	8514	9.52	2.16
9	2677	13607	5.42	2.23
10	2677	9323	5.54	2.83
11	1503	2557	7.75	4.34
12	1503	2555	991	4.09
13	1503	1542	7.33	3.70

different sampling times, soil sampling depths and rainfall events (see below).

Once N is mineralized in the soil only a proportion of it may actually be taken up by the pasture plants. Honeycutt *et al.* (1994) showed that in a cropping system, N recovery from soil organic matter and crop residue was a dynamic process reflecting changing nutrient availability and plant demand.

Using STU Honeycutt *et al.* (1994) determined the N uptake efficiency from historical growth data of crops and compared it to predicted net N mineralization. Nitrogen uptake requirements in excess of N mineralization could be met by fertilizer N. Nitrogen use efficiency in mixed pastures is more complex with multiple N sources comprising N fixation by clover, mineralization of N from soil organic matter as well as fertilizer N and grazing animal returns. This may be further complicated by the possibility of mixed pasture species. Hatch *et al.* (1991) and Gill *et al.* (1995) found the total uptake of mineralized N to average 74 and 144% respectively over the experimental periods. In Hatch *et al.* (1990) and at the New Zealand sites total uptake of N by the pastures averaged 91 and 30% respectively. The apparently low efficiency of use of the mineralized N at the New Zealand sites, measured during an atypically wet autumn/winter period, may be due to the season in which data collection commenced. Measured losses of N as leaching and denitrification were significant (Ledgard *et al.*, 1996) resulting in a reduced amount of N available for plant uptake. In contrast the UK data collections commenced in spring and coincided with rapid plant growth and N uptake and low potential for leaching. For example, total rainfall (over 85 to 155 days) during the collection of the Hatch (1990) datasets varied between sites from 148 to 329 mm whereas at the New Zealand sites total rainfall was 820 mm for the 152 day study period. It is likely that N turnover was over-estimated since short-term isolation of the soil from natural wetting/drying cycles during acetylene incubation can eliminate the potential for leaching and reduce denitrification loss, and soil coring can potentially increase the oxygen status and amount of substrate available and thus mineralization occurring in the soil. Potentially, over-estimation of N turnover could have occurred at all sites due to reasons mentioned above. However, in an extremely wet season, with evenly distributed rainfall such over estimation may be accentuated using the acetylene incubation technique. The moisture content of the soil at any given time can affect N mineralization. Doel *et al.* (1990) found that soil thermal units appeared to adequately predict net N mineralization from organic residues at soil water potentials of −0.03 to −0.01 MPa but were not valid for prolonged dry conditions.

The relationships described above could have potential roles in advising farmers on the expected contribution of N mineralization to their pasture and in modelling N dynamics in pasture. In an advisory situation the prediction of N turnover would be limited by the time consuming acetylene technique. To avoid this lengthy and time consuming process of characterizing N turnover rate there needs to be a laboratory incubation method of soil calibration linked to the field thermal data, readily obtainable from meteorological records. Honeycutt *et al.* (1988) successfully used laboratory incubations to predict net N mineralization in the field from paper mill sludge, indicating a potential for linking laboratory and field studies. Such an approach should be examined for characterizing N turnover from soil organic matter in the field. N turnover following application of sludges and slurries to grazed pastures could also be examined using similar proce-

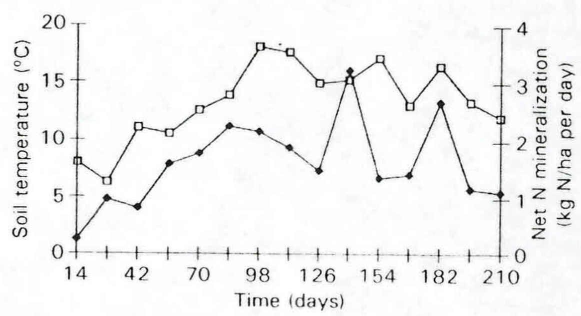

Fig. 1a. An example of the fluctuating rate of net mineralization (◆) and soil temperature (□) over time (data set 10, from Hatch *et al.*, 1991).

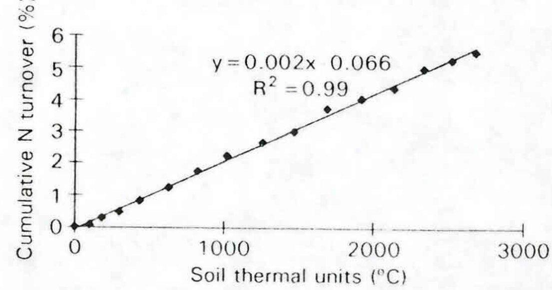

Fig. 1b. Cumulative N turnover plotted against STU (data set 10, Hatch *et al.*, 1991).

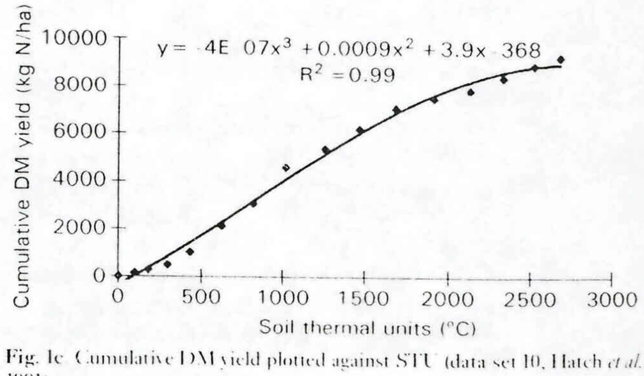

Fig. 1c. Cumulative DM yield plotted against STU (data set 10, Hatch *et al.*, 1991).

Table 3. Summary of the coefficients for the correlations between cumulative N turnover, cumulative DM and STU

Dataset	Cumulative N turnover vs. STU $y = ax + b$			Cumulative DM vs. STU $y = ax^3 + bx^2 + cx + d$					Cumulative N turnover vs. cumulative DM $y = ax^3 + bx^2 + cx + d$				
	a	b	R^2	a	b	c	d	R^2	a	b	c	d	R^2
1	1.7E-3	0.54	0.97	1E-7	−1E-3	5.6	160	0.99	73	495	1056	28	0.99
2	1.6E-3	0.33	0.98	2E-7	−2E-4	7.2	19	0.99	184	1005	1259	83	0.99
3	0.7E-3	0.09	0.98	−1E-7	3E-5	2.6	136	0.99	1319	2594	2047	67	0.97
4	1.6E-3	0.22	0.93	1E-6	8E-3	1.5	42	0.99	440	−8356	4301	−101	0.95
5	2.8E-3	0.26	0.92	−1E-5	2E-2	2.3	14	0.99	803	−184	334	31	0.99
6	2.3E-3	0.13	0.93	−2E-5	3E-3	0.4	83	0.99	2243	−1581	1141	66	0.98
7	0.8E-3	0.27	0.86	1E-6	1E-3	4.4	21	0.98	−1761	6444	−1433	207	0.94
8	3.7E-3	−0.24	0.99	−9E-8	4E-5	3.8	337	0.99	−1	−19	−1150	−183	0.99
9	2.1E-3	0.16	0.95	−6E-7	2E-4	4.9	451	0.99	108	945	405	218	0.98
10	2.1E-3	0.06	0.99	−4E-7	9E-4	3.9	368	0.99	40	212	1719	−160	0.99
11	5.1E-3	0.15	0.99	5E-7	−2E-4	2.7	5	0.99	3	−45	471	−2	0.99
12	6.6E-3	−0.41	0.99	2E-7	−7E-4	2.3	22	0.98	2	−48	505	−26	0.99
13	4.9E-3	0.04	0.99	2E-7	−7E-4	1.6	1	0.99	1	−21	302	−1	0.99

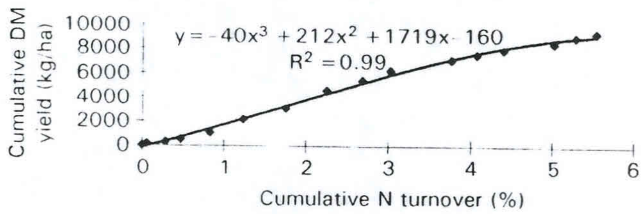

Fig. 2. Cumulative DM plotted against cumulative N turnover (data set 10, Hatch *et al.*, 1991).

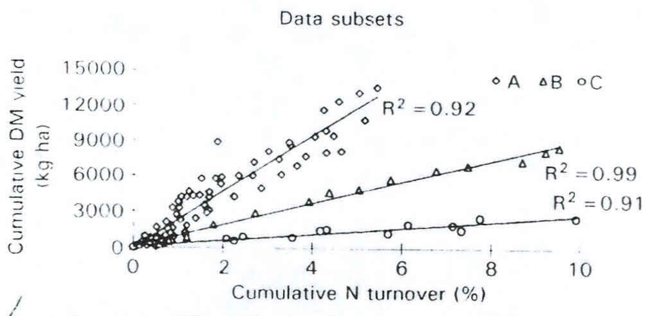

Fig. 3. Cumulative DM yields plotted against cumulative N turnover for pooled data subsets A, B and C (A = datasets 1–7, 9 and 10; B = dataset 8; C = datasets 11–13).

dures. Incubation of a pasture soil sample at several moisture contents with successive subsampling for net mineralization, might directly link N turnover in the laboratory to soil temperatures and moisture contents in the field. This information would enable farmers to estimate the supply of mineral N provided they had access to data on STU and soil moisture. Hassink (1995b) demonstrated that increases in mineralized N can reduce fertilizer N requirements, after examining data from 21 mown field sites it was found that increases in non-fertilizer N supply (NFNS) resulted in decreases in the optimum fertilizer application rates required. For a 100 kg N/ha increase in NFNS, fertilizer N could be reduced by 80 kg N/ha at a marginal N efficiency of 7.5 kg DM per kg N applied. NFNS was the difference in N uptake between fertilized and unfertilized plots and would have under-estimated total N mineralization since leaching or denitrification of N would not have been accounted for.

The relationships between STU, N turnover and DM production may also have the potential for use in modelling N mineralization and promoting the efficient use of N. Campbell *et al.* (1995) suggested coupling the potentially mineralizable N concept with deterministic models (such as CERES and LEACHM) and to quick routine laboratory methods (such as hot 2M KCl extraction), thereby enabling prediction of N mineralization. Also, the NCYCLE model (Scholefield *et al.*, 1991) includes a mineralization sub-model which predicts N mineralization in pastures on an annual cycle. Current developments of NCYCLE (Scholefield *et al.*, 1996) include shorter time steps (e.g. monthly). Once the mineralization potential has been estimated for a soil at a given moisture content, from predicted soil thermal units, other leaching and/or denitrification sub-models could also be included to examine the fate of N mineralized.

CONCLUSIONS

This analysis indicates that it may be possible to predict DM production and N turnover for a specific site or soil using simple procedures. The studies defined here indicate that the key factors required are soil temperature, total soil N and bulk density at the beginning of the period in question and pasture composition. The relationship of STU with DM yield and N turnover provides a useful correlation that could possibly be used in the modelling of N use in pasture systems. Future work should first examine linking a quick laboratory based incubation technique to the acetylene technique so the relationship between STU and N turnover can be used to characterize a farmer's soil N supply from organic matter and thus be of use in predicting N mineralization. Secondly, the relationships should be further explored for modelling N dynamics by examining different threshold temperatures, based on limits of microbial activity or pasture species, and possible relationships between N uptake and STU.

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Letter to the editor

Deep soil compaction

Dear Sir,

Sullivan & Mongomery (1998) consider two hypotheses to explain deep subsoil compaction: clay translocation and traffic loading. There seems to be at least one more possible cause of this phenomenon.

The soils concerned have clay contents of 50–60%. Some soils with clay contents as high as this shrink during the dry season and form cracks reaching to the surface; sometimes these cracks are wide enough to slip a hand into. At the same time, some at least of these soils break up at the surface to form small fragments which may be some 2–3 mm across. These fragments could easily fall down into the cracks; and the tendency to do so might be higher under arable than under pasture. If crumbs did fall into cracks in this way, they would impede swelling during the wet season, leading to the development of high horizontal stresses, which might in turn result in compaction (and shear). In particular, the high bulk densities at 0.5–1.0 mm depth under cotton at Pilliga (Fig. 1b in the original report) would be consistent with the hypothesis that fragments of surface soil become lodged in the cracks at this depth.

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Leaching of dissolved organic N from grass-white clover pasture in SW England

Jane Hawkins and David Scholefield

Background

In the mid 1980's concern arose that water draining from intensively managed grassland could exceed the EC imposed levels of 11.3 mg l^{-1} NO_3-N for drinking water.

Arable enterprises - thought to be the major concern.

Comparisons were made with cut and not grazed grassland.

Large proportion (50-80%) of the N ingested by the animal is returned to the soil in their excreta and in particular urine

Background

Waters from intensively grazed grassland soils found to contain nitrates in excess of EC limit

Studies focus mainly on inorganic Nitrogen (N)

Catchment study by Johnes and Burt - losses of organic N may be up to 40% of total N

Models of the N cycle in pasture systems have tended to ignore the contribution of N leaching from organic matter.

1 ha lysimeters
Hydrologically separated with interceptor drains
Half drained down to 85 cm. The rest undrained but surface lateral runoff to 30cm.

4 inorganic N fertilizer treatments:-

Conventional N ~ 280 kg N- regular doses
Tactical N ~ 200 kg strategic dose based on mineral N in soil
Grass/clover ~ no mineral N
Zero N No N input

Slurry inputs kg N ha^{-1} yr^{-1}:-

Conventional N - 40-50kg
Tactical N - 90kg
Grass/clover - 80 kg only on undrained
Zero N - no slurry

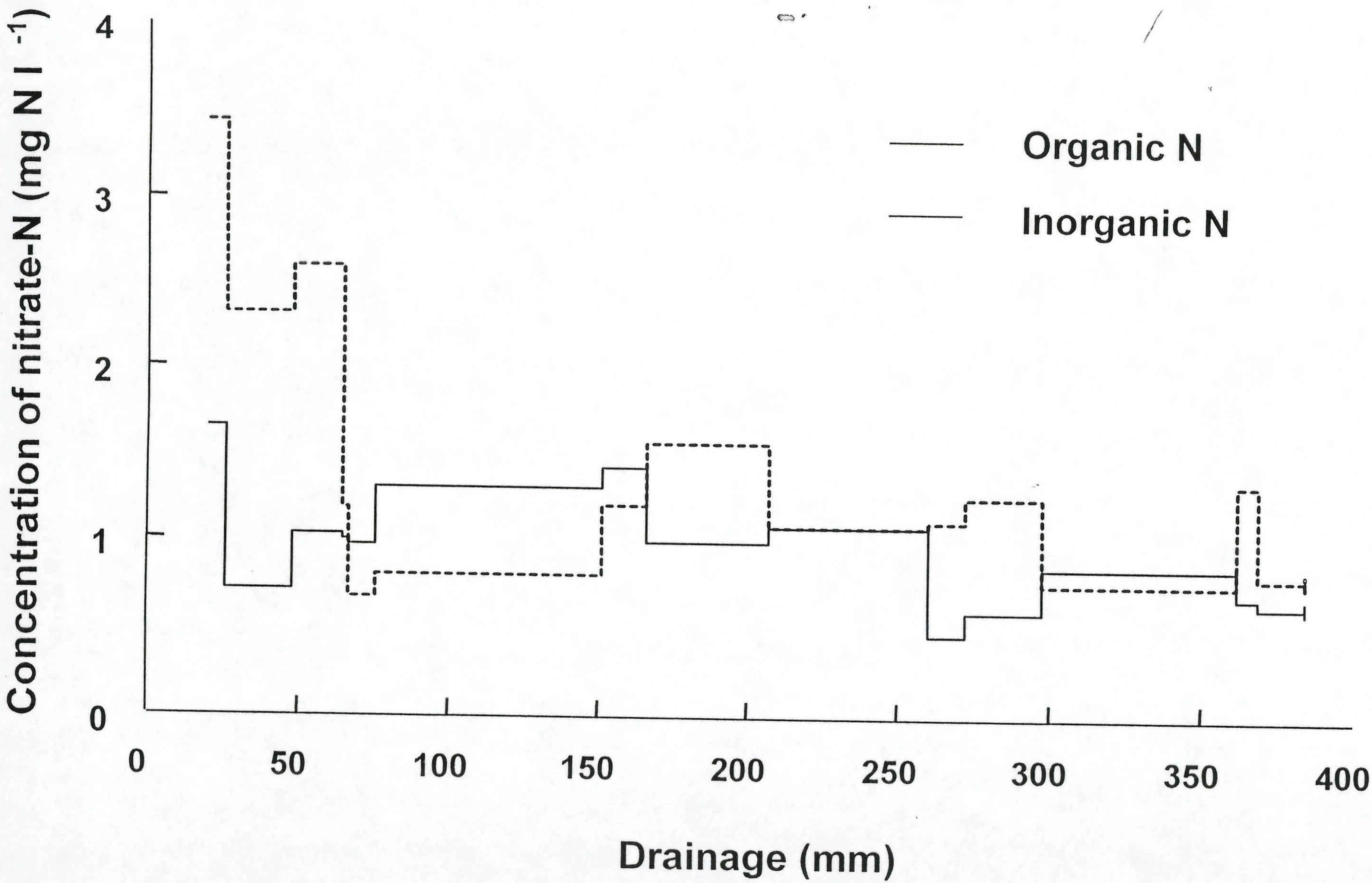
Clover content of swards

	% clover	Estimated N fixed ($kg\ N\ ha^{-1}\ yr^{-1}$)
1995	13	62
1996	9	34

Total N and percentage organic N leaching from grass-white clover and grass receiving 200 kg N ha^{-1} yr^{-1}.

Treatment	1995-96		1996-97	
	Total N (kg N ha -1)	% organic N	Total N (kg N ha -1)	% organic N
Grass-clover undrained	8.94	60	5.35	66
Grass-clover drained	19.84	25	8.28	22
200 kg N drained	84.66	22	44.9	13

Patterns of inorganic N and organic N in runoff from an undrained grass–white clover sward.



Organic forms of soil nitrogen

	% of soil N
Amino acid N	30-45
Amino sugar N	5-10
Ammonia-N	20-35
Acid insoluble-N	20-35
Unknown fraction	10-20

Conclusions

Larger total N losses from swards receiving inorganic fertilizer N

Increase in proportion of organic N where clover is present

Estimations of N losses from grazed grassland systems should take organic N losses into account

SYSTEM 2

FARM STATUS

Area	:	19 ha
Stocking rate	:	1.9 LU/ha = 36 cows
Milk	:	6000 l/cow = 216000 l
Silage	:	2.53 t DM/cow = 91.8 t (DM)
Concentrates	:	1400 kg/cow = 42.8 t (DM)
Volume of wastes: housed winter	:	57/LU/d (200 d) = 410.4 m^3
Volume of wastes: summer	:	10 l/LU/d (165 d) = 59.4 m^3
Volume of dirty water	:	18 l/LU/d (200 d) = 129.6 m^3

(Volumes from Code of Good Agricultural Practice)

N INPUTS

	kg/ha	total t
Atmosphere	30.0	0.570
Concentrates @ 3.00% N (SP)	67.3	1.285
Fertilizers (from systems description Appendix 1C and 20 kg/ha to maize)	175.4	3.325
	-------	-------
Total for system	272.6	5.180

N IN CROP

	kg/ha	total t
Harvested for silage: 91.1 t (DM)		
- grass 39.8 t (DM) @ 2.55% N (SP)	53.4	1.015
- maize 51.3 t (DM) @ 1.4% N (SP)	37.8	0.718
Present in grazed grass		
93.0 t (DM) @ 2.7% N (SP)	132.1	2.511
(grazed grass yield estimated from *Milk from Grass* at appropriate N fertilizer input)	--------	--------
Total for system	223.4	4.244

N IN MILK

	kg/ha	total t
21600 l @ 0.% (SP)	58.5	1.112
(uses 1.03 x l to kg conversion)		
Assume some maintenance/conversion as system 1)	23.8	0.453
	-------	-------
Total for system	82.8	1.567

EXCRETED N DURING HOUSING

	kg/ha	total t
Total N intake by cattle (grazed + ensiled grass+maize+concentrates)	291.0	5.529
Excreted N calculated as difference between intake and utilised.		
(i) Assume 67% of N utilised during housing (SP) (adjusted to 200 d) [(5.529 - 1.567) x 0.67]	139.5	2.655
(ii) Assume 4 hrs spent at milking during grazing [(3.962 - 2.650) x 0.17]	11.5	0.219
	--------	-------
Total N excreted in house for system	150.9	2.874

N STORED IN WASTES

	kg/ha	total t
469.8 m^3 slurry @ 0.5% N	123.6	2.349
129.6 m^3 dirty water @ 0.1% N	6.8	0.130
	--------	--------
Total for system	130.5	2.479

N LOSSES FROM SYSTEM

(i) from cutting/grazing etc.

Leaching and denitrification: estimated (except where indicated) from NCYCLE for loam soil: good drainage: long term grassland: 2-10 year swards.

Leaching

		kg/ha	total t
(a) 2 cut silage 3 ha @ 245 kg/ha		30.6	0.092
(b) 1 cut silage 4.5 ha @ 245 kg/ha	30.6}		
	64.5}	47.5	0.214
(c) grazing 5.5 ha @ 250 kg/ha		66.2	0.364
(d) maize 6 ha @ 20 kg/ha		30.0*	0.180
		-------	-------
	Total for system	44.7	0.850

(*uses Bridgets best estimate)

Denitrification

(a)		10.2	0.031
(b)	10.2}		
	21.5}	15.9	0.071
(c)		22.2	0.122
(d)		1.3*	0.078
		-------	-------
	Total for system	12.2	0.232

(*NCYCLE value for cut grass)

NH_3 volatilization: calculated as before
(i.e. NH_3^- N volatilized = 0.000347 x kg N/ha $^{1.857}$)

(b) 5.5 ha grazed at 250 kg/ha	9.9	0.054
(c) 4.5 ha grazed at 245 kg/ha	9.5	0.043
	-------	-------
Total for system	5.1	0.097

(ii) from wastes etc. (all calculated as per Jarvis, 1993)

(a) leaching (no run-off assumed)
Assume 57.5% of slurry N applied to maize (see Append 1B)
i.e. 1.413 tN applied to maize and 1.066 tN to grass.
loss from maize:- taken into account in calculation above.

loss from grass @ 5% of total N in wastes	2.8	0.053

(b) denitrification: (assume 20% of applied NH_4^+ and that NH_4^+ N = 45% of total N: same rate of loss from maize as for grass

for grass	11.7	0.223

(c) volatilization - spreading

- from grass (assume 18.6% of total N applied)	10.4	0.198
- from maize (assume 10% of total N applied)	7.4	0.141
- from shed + store (calculated as difference between N excreted during housing and stored N)	20.8	0.395
	-------	-------
	38.6	0.734

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Edited by

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