

A lysimeter study of the fate of ^{15}N -labelled nitrogen in cow urine with or without farm dairy effluent in a grazed dairy pasture soil under flood irrigation

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Abstract Nitrate (NO_3^-) leaching from cow urine patches is considered to be a major contributor to the overall NO_3^- leaching loss in grazed dairy pastures. Farm dairy effluent (DE) is usually applied to grazed pastures to recycle nutrients. The objective of this study was to determine the fate of cow urine nitrogen (N), labelled with ^{15}N , applied to soil monolith lysimeters with or without DE application. The soil was a Templeton fine sandy loam (Haplustepts), and the pasture was perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*). Cow urine was applied in the autumn at $1000 \text{ kg N ha}^{-1}$ either alone or with DE ($400 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ split into four applications). The lysimeters were flood irrigated. One year after application, 6.4–9.1% of the ^{15}N -labelled urine N was lost by leaching, 29.3–38.8% was removed in the cut pasture, 45.7–47.5% remained in the soil and plant roots, and <2% was lost by volatilisation. The application of DE plus urine significantly increased pasture yield and pasture N offtake above

those in the Urine alone treatment, but reduced the percentage recovery of the ^{15}N -labelled urine N in the pasture top compared with the Urine alone treatment. The amount of N unaccounted for was slightly higher in the Urine + DE (13.9%) treatment than in the Urine alone treatment (5.3%), probably due to increased denitrification as a result of organic carbon inputs from the DE. The urine N application significantly suppressed the clover component of the pasture, and this has implications for N budgeting on grazed dairy pastures. The amount of ^{15}N lost by leaching from the Urine alone treatment confirms that this is a major contributor to the overall effect of dairying on drainage water quality.

Keywords nitrate leaching; pasture; dairy; urine; denitrification; volatilisation

INTRODUCTION

Nitrate (NO_3^-) leaching from agricultural land and the subsequent contamination of water resources is a major environmental issue that could undermine the long-term sustainability of intensive agricultural production systems (Spalding & Exner 1993; Jarvis et al. 1995; Addiscott 1996; Cameron et al. 1997; Di & Cameron 2002a). High concentrations of NO_3^- in drinking water are deemed to be detrimental to human health; world and national health organisations have established drinking water guidelines limiting NO_3^- -N concentration to a maximum of 10–11.3 mg NO_3^- -N l^{-1} . Nitrate in surface water can cause eutrophication resulting in algal blooms (Howarth 1988). To protect water quality, national and regional authorities and industry groups have developed recommendations to limit N inputs into farming systems, established nitrate-vulnerable zones (e.g., EC 1991), and produced codes for good agricultural practices (e.g., MAFF 1991) or environmental assurance programmes (DEC 2001). However, the leaching

of NO_3^- -N is a complex issue and is affected by a number of soil, environmental, and management factors (Di & Cameron 2002a). Its mitigation requires a sound understanding of the processes and factors that affect the fate of nitrogen (N) applied to the soil-plant system.

In intensively grazed dairy pasture systems, although the amount of NO_3^- -N leached is affected by inputs of N fertilisers and organic waste effluents (Scholefield et al. 1993; Di et al. 1998a,b, 1999; Ledgard et al. 1999a; Silva et al. 1999; Monaghan et al. 2000), the largest contribution comes from the N returned from the animal urine (Ball et al. 1979; Ryden et al. 1984; Field et al. 1985; Silva et al. 1999; Di & Cameron 2000, 2002b). In a grazed pasture, between 60–90% of the N ingested by the grazing animal is returned to the pasture in the urine and dung and more than 70% of the N returned is in the urine (Haynes & Williams 1993; Jarvis et al. 1995). Therefore, a sound understanding of the fate of urine N deposited to grazed pastures is essential for understanding nitrogen dynamics on dairy farms, developing N budgets, and developing better management systems to mitigate NO_3^- -N leaching.

On most New Zealand dairy farms, farm dairy effluent (DE, comprising urine, dung, and washing water) is applied back to the pastures to recycle the nutrients. This effluent is not only rich in N, but also contains a significant amount of organic carbon (C). When applied to the soil, the organic C may affect the transformations and fate of N in the soil (Di et al. 1998a,b, 1999; Zaman et al. 1999; Di & Cameron 2002a). However, the effect of DE on the fate of urine N is largely unknown.

Dairy farming in New Zealand has been expanding rapidly, with the dairy cattle population increasing from c. 3 million in 1982 to c. 4 million at present. Much of the expansion has been occurring in the South Island, e.g., in Canterbury and Southland. On the plains of Canterbury, a number of new dairy farms have been, or are being, established on sandy or stony free-draining soils and all the dairy farms are either flood or spray irrigated. There is a general concern that NO_3^- -N leaching may be significantly increased when land use is changed from dryland sheep farming to irrigated dairy farming (Ecan 2002). There is a need to better understand the fate of urine N from the grazing animal to develop best management practices to reduce NO_3^- -N leaching in grazed pastures.

The objective of this study was to determine the fate of N in cow urine labelled with ^{15}N applied to

soil monolith lysimeters with or without dairy effluent under flood irrigation.

MATERIALS AND METHODS

Soil and pasture

The soil used was a Templeton fine sandy loam (Immature Pallic soil, Hewitt 1998; Haplustepts, Soil Survey Staff 1998) from a Lincoln University farm located c. 20 km south of Christchurch on the Canterbury plains (Table 1). The pasture was a mixture of perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*).

Lysimeter collection and installation

Undisturbed soil monolith lysimeters, 50 cm diam. and 70 cm deep, were collected following well-established protocols and procedures (Cameron et al. 1992). In brief, this involved placing a metal cylinder casing on the soil surface, digging around the casing, making sure to minimise disturbance to the soil structure inside, and gradually pushing the casing down by small increments. Once the casing had reached the desired depth (70 cm), the soil monolith was then cut off from the subsoil with a cutting plate and lifted out of the collection site. The lysimeters were transported to a lysimeter facility at Lincoln University, using a specially designed trailer with air-bag suspension to minimise disturbance to the soil monolith. The gap between the soil monolith and the metal casing was sealed using petroleum jelly to stop edge-flow effects (Cameron et al. 1992). The lysimeters were then installed in the field lysimeter facility with the surface of the lysimeters at the same level as that of the surrounding soil surface to maintain normal plant growing conditions.

Lysimeter treatments and maintenance

There were three treatments: Control (no N applied); urine applied at 1000 kg N ha⁻¹ yr⁻¹

Table 1 Properties of the Templeton soil used.

Depth (cm)	pH	Organic C (g kg ⁻¹)	Total N (g kg ⁻¹)	Bulk density (t m ⁻³)	Total porosity (v/v%)
0–20	5.3	24.6	2.0	1.32	49
20–40	5.4	8.0	0.7	1.49	44
40–60	5.8	2.3	0.3	1.52	43
60–80	6.3	2.0	0.3	1.44	46

(equivalent to the N loading rate under a cow urine patch); and urine (1000) plus farm dairy effluent (DE) at 400 kg N ha⁻¹ yr⁻¹. Each treatment had four replicates. The treatments were allocated to the lysimeters in a randomised design. The combination of urine with DE represents a situation where the pasture receives farm dairy effluent and is subsequently grazed by dairy cows (i.e., with urine inputs).

The urine was applied in one application in May 1997. Fresh urine was collected early in the morning during the milking of Friesian cows, and was analysed immediately. Before the urine was applied to the lysimeters, it was labelled with ¹⁵N to determine the fate of urine N and calculate a mass balance. The ¹⁵N label consisted of highly enriched (>99% atom% ¹⁵N) urea N and glycine N in a 9:1 ratio to give a final enrichment of 5% (atom%) ¹⁵N after mixing with the urine. This was done in recognition that urine N consists of urea N and glycine N in a ratio of c. 9:1 (Bathurst 1952). Past studies have shown that uniform labelling of urine N is not easy to achieve, even when feeding the animal with ¹⁵N-labelled food, due to differences in digestibility of dietary protein (Orskov 1992). Therefore, other researchers have used labelled synthetic urine (Fraser et al. 1994). The approach adopted in this study provided a good compromise, where real cow urine was used with a reasonably uniform labelling of ¹⁵N for most of the urine N. After labelling, the mixed urine was applied uniformly onto the lysimeters. The same volume of water was applied to the other lysimeters that did not receive urine to maintain the same moisture input to all lysimeters.

DE was applied in four split applications at about quarterly intervals. DE was collected from a nearby dairy farm in the morning after milking. It essentially comprised a mixture of cow urine and dung, deposited during the milking session, and washing water. Analysis of the DE showed that a large proportion of the N was in organic forms originating from the dung (Table 2). The main form of inorganic N was NH₄⁺-N, with only traces of NO₃⁻ and NO₂⁻-N. The DE also contained organic C. The concentrations of N and C in DE may vary considerably depending on the number of cows milked and amount of water used for washing. After the DE was analysed for total N, the volume of DE to give the desired amount of total N was then applied evenly to the lysimeters. The same volume of water was applied to other lysimeters which did not receive DE.

A phosphate (P) fertiliser, single super-phosphate, was applied to all the lysimeters at the rate of 40 kg P ha⁻¹ yr⁻¹. This was done 1 week before the application of urine and DE.

From November to April (late spring–mid autumn), flood irrigation at 100 mm was applied to all the lysimeters at about monthly intervals with a total of six applications. The amount of water applied represented the average amount of water applied on commercial farms, but this, in reality, might vary between farms. Irrigation water was applied by pouring carefully onto the surface of the lysimeters. From May to October (late autumn–mid spring), simulated rainfall was applied at the end of each month (if necessary), to supplement the natural rainfall received and to equal the 75th percentile of local rainfall records for the same period of the year. This was done to create a so-called “worst case scenario” in terms of rainfall inputs.

The herbage was cut periodically to simulate typical grazing practice. All the harvested herbage was removed and dry matter yield recorded. Subsamples of the harvested pasture were dissected by hand to separate grass and clover components.

Leachate and herbage analysis

The experiment started in May 1997 and was finished in April 1998. Leachate from the lysimeters was collected as required (i.e., when drainage was above 200 ml), or weekly, and analysed for nitrate, nitrite, and ammonium by flow injection analysis (Tecator Inc., Sweden), or ion exchange chromatography (IEC) (Waters Inc., USA). Herbage nitrogen content was analysed on a LECO CNS-2000 analyser. ¹⁵N in the leachate was recovered by the diffusion method described by Brookes et al. (1989) and was analysed on an isotope ratio mass spectrometer (Europa Scientific, UK). A subsample of the herbage was ground and its ¹⁵N content was determined using the isotope ratio mass spectrometer.

Table 2 Chemical properties of the farm dairy effluent used. Only traces of NO₃⁻-N were detected.

Time of application	pH	Total N (mg l ⁻¹)	NH ₄ ⁺	Organic C (mg l ⁻¹)
			concentration (mg l ⁻¹)	
May 1997	7.9	423	115	5300
Aug 1997	7.1	150	27	725
Nov 1997	8.8	138	37	730
Feb 1998	7.3	240	39	715

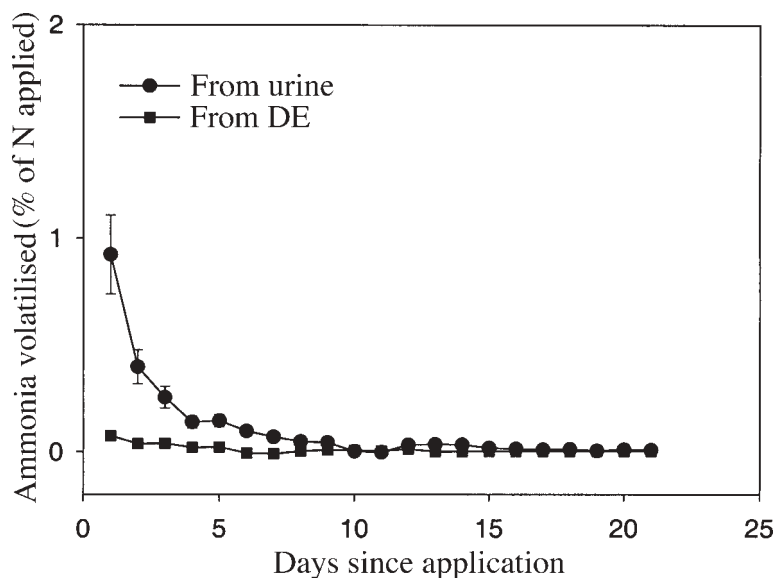


Fig. 1 Ammonia loss by volatilisation from cow urine and DE (\pm SE).

Ammonia volatilisation

The amount of N lost by volatilisation from the Urine only, DE, and Control was measured using a chamber method as described by Black et al. (1985) immediately after the application of the N sources. Measurements were made from four replicates of each treatment and were continued for a 3-week period.

Destructive sampling of the lysimeters

At the end of the experiment (April 1998), the soil monolith was pushed out of each lysimeter casing and the soil was subsampled at different depths to determine the amount of ^{15}N retained in the soil and in plant roots. Plant roots were recovered from the soil by washing under running water on a sieve. The roots were dried in an oven at 80°C for 2 days, then ground before being analysed on the mass spectrometer. The soil samples were also ground and analysed for ^{15}N .

Statistical analysis

Analysis of variance was performed using Minitab (Version 13, Minitab Inc.). Standard errors were calculated for the mean values that are presented.

RESULTS

Water balance

Total water input was 1162 mm, comprising 500 mm of natural rainfall, 62 mm of supplementary

rainfall, and 600 mm of flood irrigation. Cumulative drainage reached c. 240 mm, and drainage occurred in both summer and winter. About 55% of the drainage occurred during the winter half of the year.

Ammonia volatilisation from cow urine

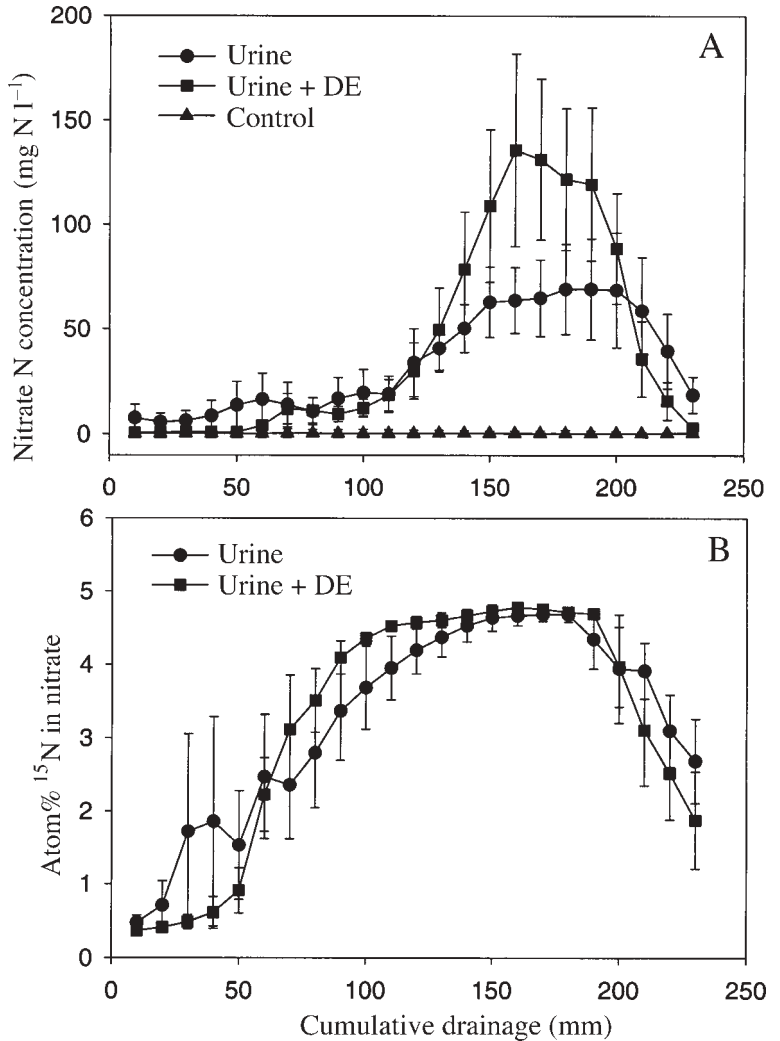
Nitrogen loss by ammonia volatilisation from the cow urine was <2% of the N applied (Fig. 1). More than 50% of this loss occurred during the first 2 days after urine application. Daily losses reached background levels 18 days after urine application. Ammonia volatilisation loss from DE was very low and was <1% of the N applied. Losses from the Control were close to zero and are not reported.

Nitrate and ^{15}N leached

There was a trend of greater peak NO_3^- -N concentration in the Urine + DE treatment than in the Urine alone treatment (Fig. 2A), although this was not always significant. The concentration in the Control was low. The ^{15}N enrichment in the NO_3^- -N in the leachate increased with increasing drainage volume, peaking near 5% (atom%) which was the ^{15}N enrichment of the labelled urine N (Fig. 2B).

The amount of NO_3^- -N leached over the year was 1.5 kg N ha^{-1} in the Control, but this increased to 77 kg ha^{-1} in the Urine alone treatment and 90 kg ha^{-1} in the Urine + DE treatment (Table 3). However, the difference between the Urine alone

Fig. 2 A, Nitrate N concentrations in the leachate (\pm SE); and B, 15 N enrichment in the leachate NO_3^- -N (\pm SE).



and Urine + DE treatments was not statistically significant.

Pasture N offtake and yield

In the first harvest of the pasture following urine application, >90% of the N in the grass component was derived from the ¹⁵N-labelled urine N (Fig. 3A). The percentage of N derived from the urine then decreased with the increasing number of harvests, particularly after the 5th harvest in January. In the clover component, <30% of the N in the first harvest (Dec 1997) was derived from the urine N applied (Fig. 3B), and this also decreased with time in the Urine alone treatment. The amount of N in the clover component of the

Urine + DE treatment was very small (Fig. 3B). The percentage of N derived from the urine N was lower in the Urine + DE treatment than in the Urine alone treatment in several of the harvests (Fig. 3).

Table 3 Annual total NO_3^- -N leached, herbage dry matter, and N offtake, determined on the lysimeters. Figures followed by the same letters within a column are not statistically different ($P < 0.05$).

Treatments	NO_3^- -N leached (kg N ha ⁻¹)	Dry matter yield (t ha ⁻¹)	N offtake (kg N ha ⁻¹)
Control	1.5a	9.2a	282.5a
Urine	77.0b	17.1b	489.4b
Urine + DE	90.0b	21.2c	605.2c

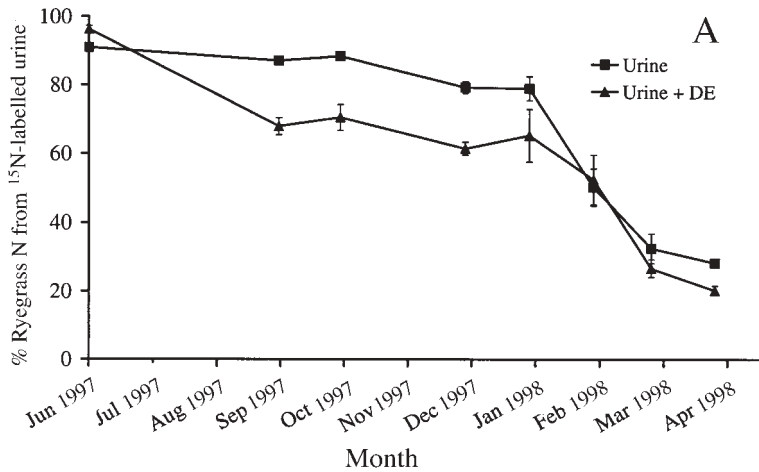
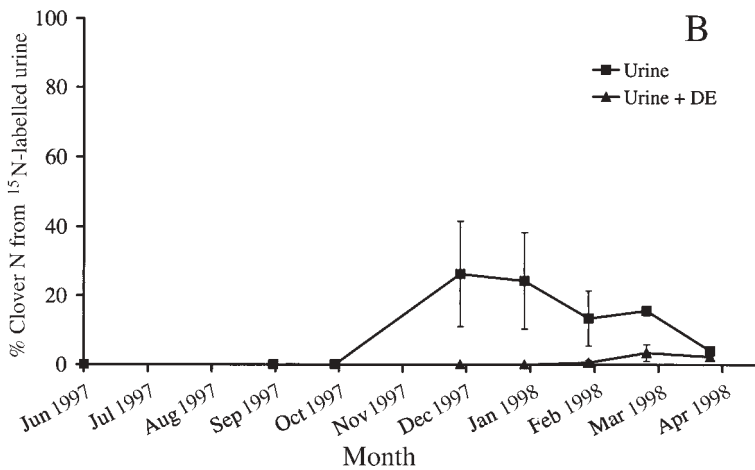


Fig. 3 A, Percentage of N in grass; or B, clover derived from the ¹⁵N-labelled urine N (\pm SE).



The pasture yield increased from 9.2 t ha⁻¹ in the Control to 17.1 t ha⁻¹ in the Urine alone treatment (Table 3). The application of DE in addition to urine further increased dry matter yield to 21.2 t ha⁻¹. Pasture N offtake increased from 283 kg N ha⁻¹ in the Control to 489 kg N ha⁻¹ in the Urine alone treatment, and to 605 kg N ha⁻¹ in the Urine + DE treatment (Table 3).

Impact of urine on grass/clover composition

In the Control, the percentage of white clover in the pasture increased from June 1997 to February 1998, reaching >60%, before declining again towards April (Fig. 4). However, in the Urine alone and Urine + DE treatments, the percentages of clover were significantly lower than that in the Control, and were below 10% for much of the

experimental period (Fig. 4). They only increased towards the end of the experimental period in March and April.

¹⁵N recovered from soil and roots

A large proportion of the ¹⁵N recovered from below ground was found to be in the top 5-cm soil layer (Fig. 5). Between 50 and 60% of the ¹⁵N recovered from the roots was from the top 5-cm depth, and between 30 and 40% of the ¹⁵N recovered from the soil was also found in the top 5-cm soil layer. The amounts of ¹⁵N recovered from both roots and soil decreased rapidly with soil depth.

¹⁵N balance

Of the ¹⁵N applied in the urine, 6.4–9.1% was recovered in the leachate, 29.3–38.8% was

Fig. 4 Percentage of white clover in the pasture. Vertical bars indicate LSD (0.05).

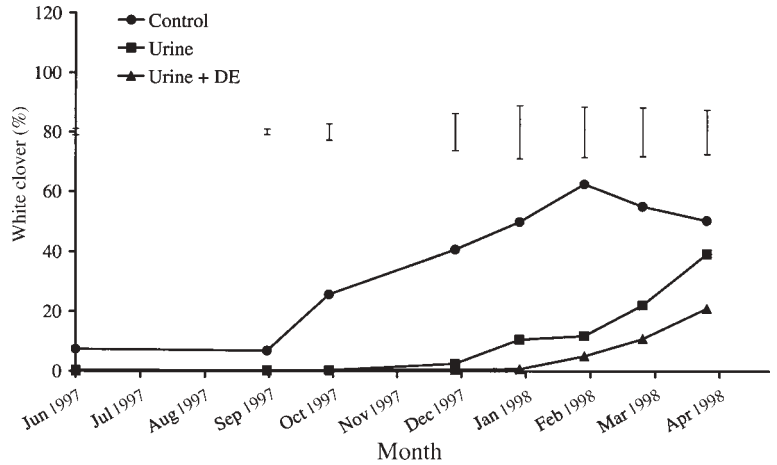
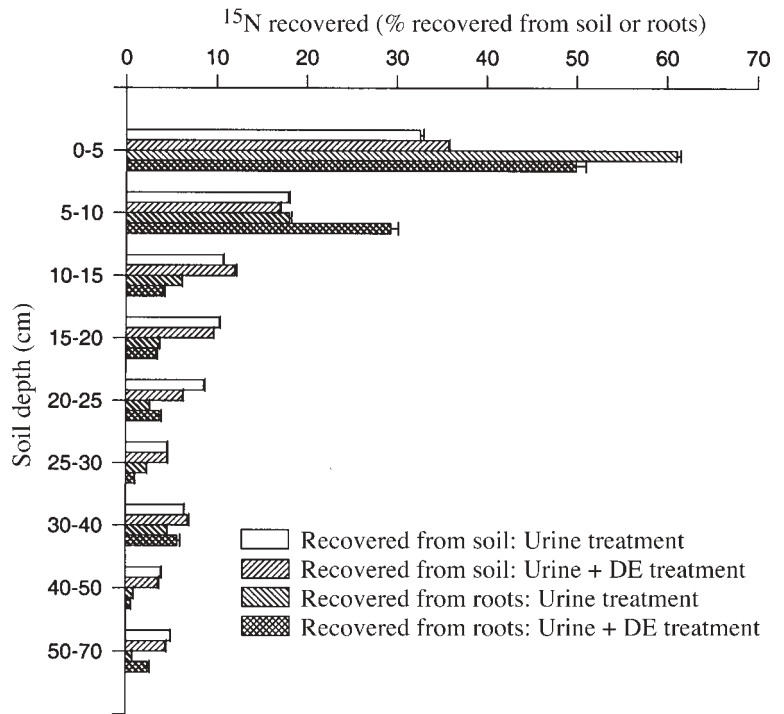


Fig. 5 ¹⁵N recovered from different depths of the soil as a percentage of total ¹⁵N recovered from roots or soil (\pm SE).



recovered in the pasture tops, 25.4–28.3% remained in the soil, and 19.2–20.3% was retained in the plant roots (Table 4). The recovery of ¹⁵N in the harvested pasture was significantly higher ($P < 0.01$) in the Urine alone treatment than in the Urine + DE treatment. The amount of ¹⁵N unaccounted for was higher in the Urine + DE treatment (13.9%) than in the Urine alone treatment (5.3%), although this was not statistically significant.

DISCUSSION

Previous studies have shown that NO_3^- -N leaching from cow urine patches in grazed dairy pastures is a major contributor to high NO_3^- -N leaching losses in free-draining dairy pasture soils (Silva et al. 1999; Di & Cameron 2002b). Results from this study show that between 6.4 and 9.1% of the ¹⁵N-labelled urine N may be leached. Although these percentage

values do not appear large, they are equivalent to reasonably large total annual amounts of NO_3^- -N, mainly because of the high nitrogen loading rates under the urine patch ($1000 \text{ kg N ha}^{-1}$). The actual amounts of NO_3^- -N leached varied from $77\text{--}90 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, and these values were similar to those values that would have been obtained through a calculation of the percentage ^{15}N leaching losses ($64\text{--}90 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). The application of DE at $400 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ did not result in a significant increase in NO_3^- -N leaching loss. This was probably partly related to the fact that a large proportion of the N in the DE applied was in organic forms, from the cow dung, and therefore it would take some time for this N to be released by mineralisation. The slowly released N had a greater chance of being taken up by the pastures present than N applied at large quantities in a single application (e.g., in the urine).

The NO_3^- -N leaching losses under the cow urine patches from this study were lower than those found on a stony Lismore soil where NO_3^- -N leaching losses of $300\text{--}600 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ were recorded under urine patches (Di & Cameron 2002b). The significantly greater leaching losses from the Lismore soil compared with the Templeton soil were mainly attributed to the shallow and free-draining nature of the Lismore soil which had only c. 30 cm of fine top soil below which the stone content increased significantly. In addition, the drainage volume from the Lismore soil study was also higher ($610\text{--}880 \text{ mm}$) compared with this study (240 mm). The percentage recoveries of ^{15}N in the leachate in this study were similar to the 8% recovery reported for the same Templeton soil type by Fraser et al. (1994), despite the lower

urine application rate (500 kg N ha^{-1}) used in their study.

It must be noted that the leaching losses recorded here are those directly under the urine patch in a grazed paddock. However, the whole paddock is not covered by urine patches in any one year. For instance, if the stocking rate is 3 dairy cows ha^{-1} , the area covered by urine patches is c. 25% of the paddock per year. Therefore annual NO_3^- -N leaching losses from a grazed paddock should be calculated on the basis of the areas that are covered by urine and non-urine patch areas (Di & Cameron 2000). For instance, if the NO_3^- -N leaching loss is $77 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ from the urine patch areas and $1.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ from the non-urine patch areas (Control, assuming no other N was applied) (Table 3), then the average NO_3^- -N leaching loss from the grazed paddock is estimated to be $20.4 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. Where urea or dairy effluent is applied to the whole grazed paddock, the calculated leaching losses are expected to be slightly greater than this due to slightly greater leaching losses from the urea or DE treatments than from the Control (Di & Cameron 2002b). In addition, urine deposited at different times may overlap and the effect of this on NO_3^- leaching is not well documented.

The significant increases in pasture yield and N offtake as a result of urine application (Table 3) were in agreement with results from the previous study using the Lismore soil (Di & Cameron 2002b). The application of DE further increased total pasture yield and N offtake above those in the Urine alone treatment (Table 3). The increased N uptake was probably derived from the additional N that was applied from DE. However, as DE also contains other nutrients (e.g., phosphate and sulphate), these nutrients may also have contributed to the higher dry matter yield (Di & Cameron 2002b). The increased supply of N from the DE was probably the main reason for the lower recovery of ^{15}N -labelled urine N in the pasture tops in the Urine + DE treatment compared with that in the Urine alone treatment (Table 4).

The significant suppression of clover growth by the urine N (Fig. 4) has important implications in N budgeting in grazed dairy pastures (Ledgard et al. 1999a,b), as this will significantly reduce the amount of N fixed under the cow urine. It is interesting to note that the percentages of N in the clover component derived from the ^{15}N -labelled urine N were very low despite the high N application rate (Fig. 3), indicating that most of the N in the clover was still fixed from the atmospheric N_2 .

Table 4 Recoveries of ^{15}N in the leachate, pasture top, soil, and roots. NS, not significant.

Components	% ^{15}N recovery		Level of significance
	Urine (\pm SE)	Urine + DE (\pm SE)	
Volatilisation	2.0 (0.2) ¹	2.0 (0.2) ¹	NS
Leachate	6.4 (1.7)	9.1 (2.5)	NS
Pasture top	38.8 (1.8)	29.3 (0.4)	$P < 0.01$
Pasture roots	19.2 (1.5)	20.3 (2.0)	NS
Soil	28.3 (0.9)	25.4 (1.0)	NS
Total	94.7 (2.5)	86.1 (3.5)	NS
Unaccounted for	5.3 (2.5)	13.9 (3.5)	NS

¹Estimates based on volatilisation measurements (^{15}N was not measured in the volatilisation experiment).

This was in clear contrast to the N in the grass component, which was mostly derived from the ^{15}N -labelled urine N (Fig. 3).

The data in Fig. 5 and in Table 4 clearly show that significant amounts of the urine N remained in the top part of the soil profile 1 year after the urine application. As most of this N is either in plant roots or in soil organic matter, it will only become available after being released by decomposition or mineralisation in subsequent years.

The ^{15}N that was unaccounted for was presumably lost mostly by denitrification. The slightly higher amount that was unaccounted for in the Urine + DE treatment compared with the Urine alone treatment (Table 4) was probably due to increased denitrification stimulated by the addition of organic carbon in the DE. Denitrification is known to be enhanced by higher contents of organic substrates (de Klein et al. 2001). The amount of N that may be lost by denitrification can vary significantly depending on soil and environmental conditions (Barton et al. 1999).

It should be noted that the urine was applied in the autumn in this study. The N applied, once converted to NO_3^- -N, was thus vulnerable to leaching during the winter season. Therefore, the results probably represent a worst case leaching scenario. The amount of N leached from cow urine was found to be lower for urine applied in the spring than that applied in the autumn (Di & Cameron 2002b) and it is likely that other dissipation pathways, e.g. volatilisation, uptake by pasture, and immobilisation will also be affected by the timing of application (Carran et al. 1982; Sherlock & Goh 1984; Di & Cameron 2002b). For example, when cow urine N was applied in mid winter (July) to the same Templeton soil type at $500 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (Fraser et al. 1994), only 20.4% of the N was retained below ground in the soil and plant roots compared with 45.7–47.5% of ^{15}N retained below ground in this study. In addition, the amount of ^{15}N unaccounted for by Fraser et al. (1994) was 27.9%, and this is considerably higher than the 5.3–13.9% unaccounted for in this study. The warmer soil temperatures in autumn in this study compared with those in mid winter (Fraser et al. 1994) were probably responsible for the greater immobilisation in the present study. A lower soil redox potential following the mid winter application (Fraser et al. 1994) might have been responsible for the greater amount of denitrification (as indicated by the amount of N unaccounted for) compared with the present study.

ACKNOWLEDGMENTS

The authors thank the New Zealand Dairy Industry and the Foundation for Research, Science and Technology for funding, and Trevor Hendry, Steve Moore, and Neil Smith for technical support.

REFERENCES

- Addiscott, T. M. 1996: Fertilizers and nitrate leaching. *In*: Hester, R. E.; Harrison, R. M. ed. *Agricultural chemicals and the environment. Issues in Environmental Science and Technology* 5: 1–26.
- Ball, P. R.; Keeney, D. R.; Theobald, P. W.; Nes, P. 1979: Nitrogen balance in urine-affected areas of a New Zealand pasture. *Agronomy Journal* 71: 309–314.
- Barton, L.; McLay, C. D.; Schipper, L. A.; Smith, C. T. 1999: Annual denitrification rates in agricultural and forest soils: a review. *Australian Journal of Soil Research* 37: 1073–1093.
- Bathurst, N. O. 1952: The amino acids of sheep and cow urine. *Journal of Agricultural Science* 42: 476–478.
- Black, A. S.; Sherlock, R. R.; Smith, N. P.; Cameron, K. C.; Goh, K. M. 1985: Effects of forms of nitrogen, season, and urea application rate on ammonia volatilisation from pastures. *New Zealand Journal of Agricultural Research* 28: 469–474.
- Brookes, P. D.; Stark, J. M.; Minter, B. B.; Preston, T. 1989: Diffusion method to prepare soil extracts for automated nitrogen-15 analysis. *Soil Science Society of America Journal* 53: 1707–1711.
- Cameron, K. C.; Smith, N. P.; McLay, C. D. A.; Fraser, P. M.; McPherson, R. J.; Harrison, D. F.; Harbottle, P. 1992: Lysimeters without edge-flow: an improved design and sampling procedure. *Soil Science Society of America Journal* 56: 1625–1628.
- Cameron, K. C.; Di, H. J.; McLaren, R. G. 1997: Is soil an appropriate dumping ground for our wastes? *Australian Journal of Soil Research* 35: 995–1035.
- Carran, R. A.; Ball, P. R.; Theobald, P. W.; Collins, M. E. G. 1982: Soil nitrogen balances in urine-affected areas of pasture under two moisture regimes in Southland. *New Zealand Journal of Experimental Agriculture* 10: 377–381.
- DEC 2001: Market focused. An environmental management system for New Zealand dairy farmers. The Dairying and Environment Committee (DEC), New Zealand Dairy Board, Wellington, New Zealand.
- de Klein, C. A. M.; Sherlock, R. R.; Cameron, K. C.; van der Weerden, T. J. 2001: Nitrous oxide emissions from agricultural soils in New Zealand—a review of current knowledge and directions for future research. *Journal of the Royal Society of New Zealand* 31: 543–574.

- Di, H. J.; Cameron, K. C. 2000: Calculating nitrogen leaching losses and critical nitrogen application rates in dairy pasture systems using a semi-empirical model. *New Zealand Journal of Agricultural Research* 43: 139–147.
- Di, H. J.; Cameron, K. C. 2002a: Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. *Nutrient Cycling in Agroecosystems*. In press.
- Di, H. J.; Cameron, K. C. 2002b: Nitrate leaching and pasture production from different nitrogen sources on a shallow stony soil under flood irrigated dairy pasture. *Australian Journal of Soil Research* 40: 317–334.
- Di, H. J.; Cameron, K. C.; Moore, S.; Smith, N. P. 1998a: Nitrate leaching from dairy shed effluent and ammonium fertiliser applied to a free-draining soil under spray or flood irrigation. *New Zealand Journal of Agricultural Research* 41: 263–270.
- Di, H. J.; Cameron, K. C.; Moore, S.; Smith, N. P. 1998b: Nitrate leaching and pasture yields following the application of dairy shed effluent or ammonium fertilizer under spray or flood irrigation: results of a lysimeter study. *Soil Use and Management* 14: 209–214.
- Di, H. J.; Cameron, K. C.; Moore, S.; Smith, N. P. 1999: Contributions to nitrogen leaching and pasture uptake by autumn-applied dairy effluent and ammonium fertilizer labelled with ¹⁵N isotope. *Plant and Soil* 210: 189–198.
- EC 1991: Council directive of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources. 91/676/ECC, Legislation 1375/1–1375/8, European Community.
- Ecan 2002: Environmental guidelines for dairy farming in Canterbury. Environment Canterbury (Ecan) Report R01/35. Christchurch.
- Field, T. R. O.; Theobald, P. W.; Ball, P. R.; Clothier, B. E. 1985: Leaching losses of nitrate from cattle urine applied to a lysimeter. *Proceedings of the Agronomy Society of New Zealand* 15: 137–141.
- Fraser, P. M.; Cameron, K. C.; Sherlock, R. R. 1994: Lysimeter study of the fate of nitrogen in animal urine returned to irrigated pasture. *European Journal of Soil Science* 45: 439–447.
- Haynes, R. J.; Williams, P. H. 1993: Nutrient cycling and soil fertility in the grazed pasture ecosystem. *Advances in Agronomy* 49: 119–199.
- Hewitt, A. E. 1998: New Zealand soil classification. 2nd ed. Lincoln, Canterbury. Manaaki Whenua–Landcare Research New Zealand Ltd Press.
- Howarth, R. W. 1988: Nutrient limitation of net primary production in marine ecosystems. *Annual Review of Ecological Systems* 19: 898–1110.
- Jarvis, S. C.; Scholefield, D.; Pain, B. 1995: Nitrogen cycling in grazing systems. In: Bacon, P. E. ed Nitrogen fertilization in the environment, New York, Marcel Dekker. Pp. 381–419.
- Ledgard, S. F.; Penno, J. W.; Sprosen, M. S. 1999a: Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. *Journal of Agricultural Science, Cambridge* 132: 215–225.
- Ledgard, S. F.; Williams, P. H.; Broom, F. D.; Thorrold, B. S.; Wheeler, D. M. 1999b: OVERSEER™—a nutrient budgeting model for pastoral farming, wheat, potatoes, apples and kiwifruit. In: Currie, L. D.; Hedley, M. J.; Horne, D. J.; Loganathan, P. ed. Proceedings of the workshop: best management practices for production. Massey University, Palmerston North, New Zealand. Pp. 143–152.
- MAFF 1991: Code for good agricultural practice for the protection of water. Ministry of Agriculture, Fisheries and Food. London, MAFF Publications.
- Monaghan, R. M.; Paton, R. J.; Smith, L. C.; Binet, C. 2000: Nutrient losses in drainage and surface runoff from a cattle grazed pasture in Southland. *Proceedings of the New Zealand Grassland Association* 62: 99–104.
- Orskov, E. R. 1992: Protein nutrition in ruminants. London, Academic Press.
- Ryden, J. C.; Ball, P. R.; Garwood, E. A. 1984: Nitrate leaching from grassland. *Nature* 311: 50–53.
- Scholefield, D.; Tyson, K. C.; Garwood, E. A.; Armstrong, A. C.; Hawkins, J.; Stone, A. C. 1993: Nitrate leaching from grazed grassland lysimeters: effects of fertilizer input, field drainage, age of sward, and patterns of weather. *Journal of Soil Science* 44: 601–613.
- Silva, R. G.; Cameron, K. C.; Di, H. J.; Hendry, T. 1999: A lysimeter study of the impact of cow urine, dairy shed effluent and nitrogen fertilizer on drainage water quality. *Australian Journal of Soil Research* 37: 357–369.
- Sherlock, R. R.; Goh, K. M. 1984: Dynamics of ammonia volatilisation from simulated urine patches and aqueous urea applied to pasture. I. Field experiments. *Fertilizer Research* 5: 181–195.
- Soil Survey Staff 1998: Keys to soil taxonomy. 8th ed. United States Department of Agriculture, Washington DC.
- Spalding, R. F.; Exner, M. E. 1993: Occurrence of nitrate in groundwater—a review. *Journal of Environmental Quality* 22: 392–402.
- Zaman, M.; Di, H. J.; Cameron, K. C. 1999: A field study of gross rates of N mineralisation and nitrification and their relationships to microbial biomass and enzyme activities in soils treated with dairy effluent and ammonium fertilisers. *Soil Use and Management* 15: 188–194.