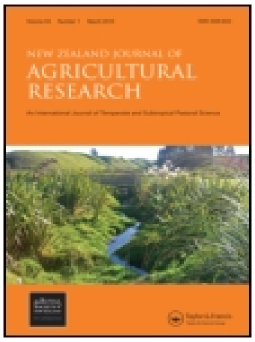

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Calculating nitrogen leaching losses and critical nitrogen application rates in dairy pasture systems using a semi-empirical model

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Abstract A simple, semi-empirical model for estimating nitrogen (N) leaching losses and critical N application rates in dairy pasture systems is described. The model uses the annual rates of major N flux processes in the soil-plant system to determine the potentially leachable N pool (mineral N and mineralisable N), and estimates the N leaching loss based on measured relationships between the N leaching loss and the potentially leachable N in the soil. The N flux processes considered in the model include fertiliser or effluent N applications, biological N fixation, soil N mineralisation and immobilisation, plant N uptake, animal N return at the urine patches, ammonium volatilisation, and denitrification. The impact of drainage on N leaching is taken into account by normalising the N leaching loss to a per 100 mm drainage basis. A quadratic equation is used to describe the relationship between the N leaching loss and potentially leachable N. Tests of the model predictions against other experimental data showed reasonable agreements between the estimated N leaching losses with those measured. The modelled critical N application rates which would cause the annual average N concentration in the drainage water to reach the drinking water standard (11.3 mg N l^{-1}) are: 390–392 kg N ha^{-1} for cut and carry, and 162–192 kg N ha^{-1} for grazed pastures if urea is used; and 588–600 kg N ha^{-1} for cut and carry, and 248–301 kg N ha^{-1} for grazed pastures if dairy shed effluent is used.

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Keywords nitrate; cycling; urine; urea; effluent; water quality; modelling

INTRODUCTION

Nitrate leaching and the contamination of ground and surface waters are a major concern in many countries. Nitrate leaching from intensive agricultural systems, e.g., dairy farming systems where N fertilisers and waste effluents are applied, is considered a major contributor to increased nitrate concentrations in ground water (Jarvis 1993; Spalding & Exner 1993; Addiscott 1996; Cameron et al. 1997). High concentrations of nitrate in drinking water are deemed to be harmful to human health and the drinking water standard established by the New Zealand Department of Health thus limits the nitrate concentration in drinking water to 11.3 mg NO_3-N l^{-1}. Regulatory authorities in many countries have established, or are in the process of establishing, guidelines regulating the application rates of N fertilisers or waste effluents on land (e.g., European Community 1991).

However, nitrate leaching is a complex process and is affected by a number of soil, environmental, and management conditions. Numerous studies have been conducted to determine nitrate leaching as affected by the application of N fertilisers, organic wastes, and other soil, environmental, and management factors (e.g., Clothier & Sauer 1988; Jarvis 1993; Scholefield et al. 1993; Cameron et al. 1996, 1999; Ledgard et al. 1996; Di et al. 1998a, 1998b). Computer simulation models have also been developed to improve our understanding of N leaching processes and provide management guidance (e.g., Berstrom & Jarvis 1991; Scholefield et al. 1991; Hutson & Wagenet 1992). Some of the models that have been developed are mechanistic in their representation of the solute transport and N

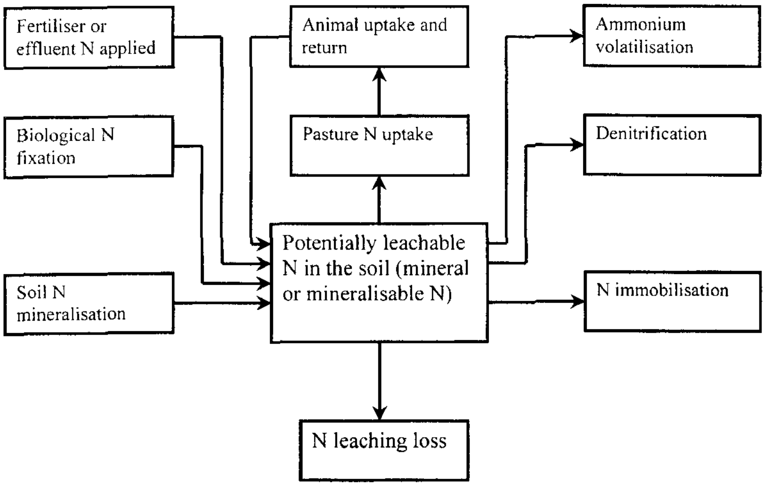

Fig. 1 The NLE conceptual model showing the N input and output processes to and from the potentially leachable N pool.

transformation processes in the soil (e.g., LEACHN, Hutson & Wagenet 1992), whilst others are more empirical and are based on N mass balance data collected from local farms (e.g., Scholefield et al. 1991), or on the probability density function of solute travel times (e.g., White & Magesan 1991). A limitation with many models is their inadequate account for the impact of N excreted by grazing animals on N leaching, either because the models were developed for cropping conditions or because of a lack of quantitative understanding. Recent studies have shown that in grazed dairy pasture systems, a large amount of the N consumed by the animal is returned to the pasture in the form of urine and dung to patches of the paddock. The N loading rate under each urine patch, for example, is equivalent to about 1000 kg N ha^{-1}, and this high rate of N can have a dramatic impact on the overall N leaching loss from the paddock (Ryden et al. 1984; Stout et al. 1997; Silva et al. 1999). A model for estimating N leaching losses in grazed pasture systems should thus take this impact into account.

Di et al. (1998a, 1998b) and Silva et al. (1999) reported N leaching losses from a series of lysimeter studies following the application of N fertilisers, dairy effluents, and cow urine. A simple, semi-empirical model has been developed based on these studies to estimate N leaching losses in dairy pasture systems for management purposes. This paper describes the model, tests it against other experimental data, and reports critical N application rates estimated using the model in relation to the drinking water standard.

DESCRIPTION OF THE MODEL

Model concept

The nitrogen leaching estimation (NLE) model is based on the concept that the annual N leaching loss is related to the annual average amount of potentially leachable N (mineral N and mineralisable N) in the soil, and the annual drainage through the soil. The potentially leachable N is determined by the sum of annual flux rates of the major N cycling processes (Fig. 1). Therefore, all the processes that add mineral or mineralisable N to the soil will increase the potentially leachable N pool and thus N leaching loss, whereas those processes that remove N from the potentially leachable N pool will reduce N leaching loss.

An annual flux rate was estimated for each of the processes in Fig. 1, and the potentially leachable N was then calculated as:

$$N_{PL} = N_F + N_B + N_M + N_A - N_P - N_V - N_D \quad (1)$$

where N_{PL} is potentially leachable N (kg N ha^{-1}); N_F is annual fertiliser or effluent N application rate; N_B is annual biological N fixation; N_M is annual net N mineralisation; N_A is annual animal N returns (mainly urine) to the pastures; N_P is annual pasture N uptake; N_V is volatilisation losses after the application of N fertilisers or effluents; and N_D is denitrification loss.

The effect on N leaching by drainage is taken into account by normalising the annual N leaching loss to a per 100 mm drainage basis, assuming that annual N leaching losses increase with drainage volume. The normalised N leaching loss is then expressed as a function of the potentially leachable N pool:

$$N_L = f(N_{PL}) \quad (2)$$

where N_L is the annual N leaching loss.

Although some other variables which may also affect N leaching are not explicitly considered in the model, they are indirectly accounted for by the variables that are considered in the model. For example, the timing of fertiliser application with regard to plant demand, the number of annual split applications, the amount of N applied at each application, and the climate may all affect N leaching loss (McLay et al. 1991; Di et al. 1999; Cameron et al. 1999). These effects are indirectly reflected in the pasture N uptake data, or in the N input and output processes as shown in Fig. 1. For example, N applied at the wrong time or in excessively large amounts will result in lower efficiency in pasture uptake and a greater amount of potentially leachable N. Climatic conditions which result in lower N recoveries in the harvested pasture may also result in greater amounts of potentially leachable N in the soil and thus greater leaching losses. Animal dung returns are not considered as a separate input variable but are included as part of the soil N mineralisation input to the potentially leachable N pool. A large proportion of the N (>75%) in the dung material is in organic forms which are only slowly decomposable (Scholefield et al. 1991; Haynes & Williams 1993). The N will thus be released over the following years as part of the soil N mineralisation process. Atmospheric deposition of N is negligible in New Zealand and is thus ignored in the model. However, it can be significant in some parts of the world and should then be included as one of the N input processes to the potentially leachable N pool in Fig. 1.

Experiments from which the input variables were derived

Di et al. (1998a, 1998b) and Silva et al. (1999) reported mineral N (predominantly NO_3^-) leaching losses measured from large, undisturbed soil lysimeters (50–80 cm diameter and 70–120 cm depth). The soil used for the lysimeter studies was a Templeton fine sandy loam (Kear et al. 1967) (Immature Pallic soil, Hewitt 1998; Haplustepts, Soil Survey Staff 1998). N leaching losses were measured after the application of a range of N sources, including urea, ammonium chloride, dairy shed effluent (DSE), and cow urine. Annual application rates of N were 0, 200, 400, and 1000 kg N ha^{-1}, and these rates were applied in one application or split into 2 or 4 annual applications. The lysimeters received different irrigation treatments, e.g., spray (50 mm mo^{-1}) or flood (100 mm mo^{-1}) during the summer months.

The input variables for the model shown in Fig. 1 were determined on the lysimeters. Biological N fixation was determined by using a ^{15}N dilution method (Ledgard et al. 1990). N loss by denitrification was determined to be the amount that was unaccounted for from the ^{15}N mass-balance data collected at the end of the experiments. Nitrogen mineralisation and immobilisation were determined in separate experiments in the laboratory and field using the same soil and N sources as those on the lysimeters, using ^{15}N dilution methods (Zaman et al. 1999a, 1999b). The net annual N mineralisation rate was then estimated.

Relationship between potentially leachable N and N leaching losses

The annual flux rates determined or estimated from the above studies were fed into Equation 1 to calculate the annual amount of potentially leachable N. Where mineral N fertilisers such as urea or ammonium chloride were applied, the N was assumed to become part of the potentially leachable N immediately after application and subject to the subsequent N transformation processes. Where dairy shed effluent was applied, 65% of the N was assumed to have become part of the potentially leachable N pool within the year of application (Di et al. 1998b; Zaman et al. 1999a, 1999b). As most of the N in urine is urea, this became part of the potentially leachable N.

Annual N leaching losses (N_L) measured from the lysimeters were normalised to a per 100 mm drainage basis by dividing the annual drainage volumes (mm) into the annual N leaching losses. The normalised N leaching loss was then related to the potentially leachable N by the weighted least squares method (Snedecor & Cochran 1980). Linear, power function, and quadratic equations were assessed, and the quadratic function (3) was found to best describe the relationship:

$$N_L = a \cdot (N_{PL})^2 + b \cdot N_{PL} \quad (3)$$

where a and b are constants.

Since data on N losses by volatilisation and denitrification are not always available, two equations were produced, one considering

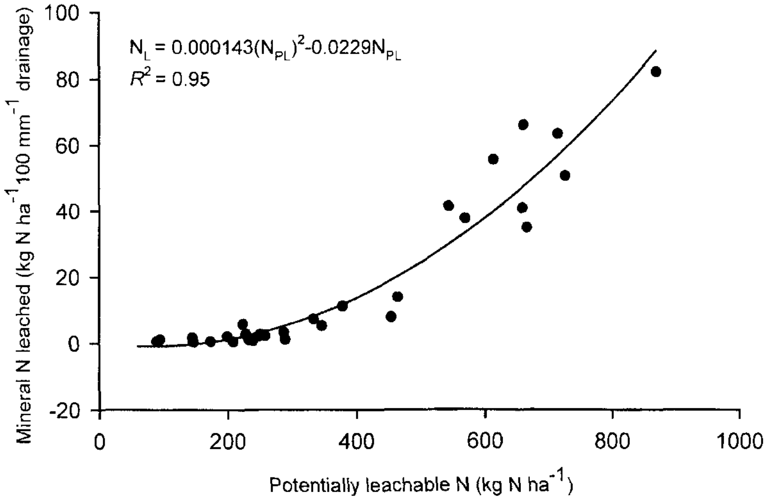

Fig. 2 Relationship of annual N leaching losses (N_L) with the potentially leachable N (N_{PL}), where volatilisation and denitrification were considered.

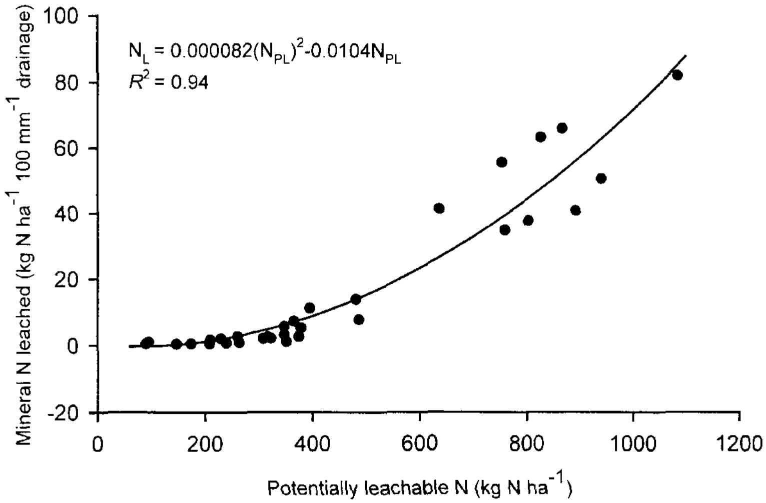

Fig. 3 Relationship of annual N leaching losses (N_L) with the potentially leachable N (N_{PL}), where volatilisation and denitrification were not considered.

volatilisation and denitrification as separate input variables and the other without considering the two variables as separate input variables. This was to assess the impact of these two variables on the model performance in estimating N leaching losses. When these two factors were not considered as input variables, they were essentially assumed to remain constant and were only implicitly accounted for by their impacts on other N processes and leaching losses as determined on the lysimeters.

The relationships between the potentially leachable N and the N leaching loss are shown in Fig. 2 (where volatilisation and denitrification were considered as input variables) and Fig. 3 (volatilisation and denitrification were not considered).

The relationships were described by the following equations.

Model 1, where volatilisation and denitrification were included as input variables:

$$N_L = 0.000143(N_{PL})^2 - 0.229N_{PL}$$
$$R^2 = 0.95\ (P < 0.001) \quad (4)$$

Model 2, where volatilisation and denitrification were not considered:

$$N_L = 0.000082(N_{PL})^2 - 0.0104N_{PL}$$
$$R^2 = 0.94\ (P < 0.001) \quad (5)$$

The good relationships between the N leaching loss and the potentially leachable N (Fig. 2, 3) demonstrate the ability of the model to account for the impact of the wide range of factors considered in the lysimeter experiments (including N sources, application rates, number of applications, time of applications, and irrigation conditions) (Di et al. 1998a, 1998b; Silva et al. 1999) on the N leaching loss. Equation 2 seemed to describe the relationship between N leaching loss and potentially leachable N equally as well as Equation 1 despite the fact that volatilisation and denitrification were ignored in Equation 2. This may be because the effect on N leaching by volatilisation and denitrification was overshadowed by effects of the other N flux processes. When the potentially leachable N was less than 200 kg N ha^{-1}, the N leaching loss was low (Fig. 2, 3). Some of this N was retained by the soil, e.g., in ammonium form, while a proportion of the N (e.g., some of the N from biological fixation) was not immediately available for leaching.

The application of these models to calculate the N leaching loss for cut and carry pastures involves firstly calculating the potentially leachable N using Equation 1, and then calculating the N leaching loss using Equation 4 or 5, depending on the availability of data for volatilisation and denitrification. In grazed pastures, however, cow urine is returned to the paddock by the grazing animal, creating urine patches with N loading rates equivalent to 1000 kg N ha^{-1} under each urine patch. The impact of the urine N on N leaching therefore needs to be considered in the model (Ruz-Jerez et al. 1995; Silva et al. 1999). N leaching losses from the urine and non-urine patch areas need to be calculated separately first using Equation 4 or 5, and a weighted average of N leaching losses from the paddock can then be calculated on the basis of the average paddock areas that are occupied by urine and non-urine patches. Under typical grazing conditions in New Zealand, about 25% of the paddock area (varying depending on stocking rate) is covered with urine per year (Haynes & Williams 1993; Silva et al. 1999). The leaching loss from the paddock can thus be calculated from the following equation:

$$N_L = N_{L1} \times 0.25 + N_{L2} \times 0.75 \qquad (6)$$

where N_{L1} and N_{L2} are the N leaching losses at the urine patch and non-urine patch areas, respectively. Substituting Equation 4 or 5 into Equation 6 gives the equations for calculating N leaching loss from grazed paddocks.

Model 1: where volatilisation and denitrification were considered:

$$N_L = [-0.0229N_{PL1} + 0.000143(N_{PL1})^2] \times 0.25 + [-0.0229N_{PL2} + 0.000143(N_{PL2})^2] \times 0.75 \qquad (7)$$

Model 2: where volatilisation and denitrification were not considered:

$$N_L = [-0.0104N_{PL1} + 0.000082(N_{PL1})^2] \times 0.25 + [-0.0104N_{PL2} + 0.000082(N_{PL2})^2] \times 0.75 \qquad (8)$$

where N_{PL1} and N_{PL2} are the potentially leachable N pools at the urine patch and non-urine patch areas, respectively. These two values need to be estimated for urine and non-urine patch areas separately using Equation 1.

TEST OF THE MODEL AGAINST EXPERIMENTAL DATA

To assess the applicability of the NLE model to wider conditions, the model was tested against N leaching data measured after the application of other N sources or from other soils reported in the literature (Table 1). The data selected for this test were from studies where the variables considered in Equation 1 were reliably determined or could be estimated based on the information given in the literature. The studies selected included the application of urea (Ledgard et al. 1996), dairy pond sludge (Cameron et al. 1996), and pig slurry (Cameron et al. 1995; Carey et al. 1997). The soils used were a Horotiu silt loam (Orthic Allophanic soil, Hewitt 1998; Udivitrands, Soil Survey Staff 1998), Templeton soil (Immature Pallic soil; Haplustepts), and a Lismore very stony silt loam (Immature Orthic Brown soil; Haplustepts) which is a shallow stony soil with <30 cm soil above continuous gravels.

The N leaching losses estimated by the model were in the same order of magnitude as those measured (Table 1), despite the different N sources applied and the contrasting soils used. The largest discrepancy between the calculated and the measured N leaching losses occurred where pig slurry was applied at 200 and 400 kg N ha^{-1}, from the study by Carey et al. (1997). This was probably because of a slightly lower N mineralisation rate used for the model which was estimated on the basis of that in the control treatment (no pig slurry applied). The application of organic wastes, e.g., pig slurry, was likely to have increased the N mineralisation rate above that in the control because of stimulated microbial activities (Kandeler et al. 1994; Zaman et al. 1999a). Model 2 seemed to perform equally well compared with Model 1 against measured data. This illustrates that, in modelling, more input variables do not necessarily lead to improved performance of the model, particularly when the necessary data are not readily available or can not be reliably estimated.

CALCULATION OF CRITICAL N APPLICATION RATES

The model can also be used to estimate the amount of fertiliser or effluent N applied that would cause the annual average N concentrations in the drainage water to reach the drinking water standard (11.3 mg N l^{-1}). This can be achieved by solving Equation 4 or 5 (for cut and carry conditions) and Equation 7 or 8 (for grazed conditions) for N_{PL} when the N leaching loss gives 11.3 mg N l^{-1} in the drainage water. Calculations were carried out for conditions similar to those reported by Di et al. (1998a, 1998b) and Silva et al. (1999) (Table 2). The critical application rates calculated in Table 2 should be applicable to annual drainage rates of about 150–600 mm, which were the conditions under which the experimental data were generated for developing the model.

The N application rates that would cause the N concentration in the drainage water to reach the drinking water standard were found to be substantially lower under grazing than under cut and carry systems (Table 2). This was because of the high potential for N leaching from urine patches under grazed conditions. The high rate of N input under each urine patch is significantly above the amount of N that can be taken up by the pasture; the surplus N remaining in the soil is then subject to leaching during the wet season. The two models produced similar results for the cut and carry conditions, but produced slightly different results for the grazed conditions. The latter was caused by interactive effects between the urine N and urea or DSE N on volatilisation or denitrification. For instance, when DSE was applied to the paddock (to both the urine and the non-urine patch areas), this was likely to increase denitrification of the urine N because of the organic carbon provided by the DSE (Di et al. 1998a). This effect was shown in Model 1 where denitrification was taken into account, but not in Model 2 where denitrification was not considered as an input variable.

Current recommendations by regional councils in New Zealand usually limit the annual application rate of N to about 150–200 kg N ha^{-1} without regard to the forms of N applied or management conditions. The code for good agricultural practice for the protection of water in the UK recommends that the total N in the livestock wastes and other organic wastes applied should not exceed 250 kg ha^{-1} per annum (MAFF 1991). Judged by the model outputs (Table 2), such guidelines are appropriate for grazed pastures, but are too low for cut and carry conditions.

The above calculations were based on annual average N concentrations in the drainage water. However, the concentration of N in the drainage

Table 1 Comparison of N leaching losses estimated using the model with those measured. Denitrification and volatilisation N losses were considered in Model 1, but not in Model 2. Dairy pond sludge from the anaerobic treatment pond.

Total N applied (kg N ha^{-1})	Forms of N applied	Soil type	N leaching losses (kg N ha^{-1} 100 mm^{-1}) Measured	Estimated Model 1	Estimated Model 2	References
0	–	Horotiu silt loam	6.3	6.1	6.3	Ledgard et al. (1996)
225	Urea	Horotiu silt loam	8.7	8.0	10.0	Ledgard et al. (1996)
360	Urea	Horotiu silt loam	18.0	11.8	15.5	Ledgard et al. (1996)
291–319	Dairy pond sludge	Templeton fine sandy loam	1.4–2.9	0–3.4	1.5–4.3	Cameron et al. (1996)
200	Pig slurry	Lismore very stony silt loam	5.8	2.5	5.6	Cameron et al. (1995)
0	–	Templeton fine sandy loam	0.3–4.7	0–4.6	0.2–3.4	Carey et al. (1997)
200	Pig slurry	Templeton fine sandy loam	0.4–14	0–2.4	1.6–6.2	Carey et al. (1997)
400	Pig slurry	Templeton fine sandy loam	11.3	0.2	6.4	Carey et al. (1997)

water usually varies with time depending on the time and amount of N applied (Ledgard et al. 1996; Di et al. 1998b, 1999; Silva et al. 1999). Therefore, the concentration of N in the drainage water may exceed the drinking water standard at certain times of the year even when N is applied at an amount less than that calculated in Table 2. If the regulatory requirement is that the N concentration should not exceed 11.3 mg N l^{-1} at any time, then a safety factor should be introduced, which would further reduce the critical N application rate.

DISCUSSION

Although the model was developed on a semi-empirical basis, because the major N input and output processes which affect the fate of N in the soil-plant system were accounted for in the model, it may have the potential to be applied to a wider environment. However, the model has only been tested against a limited number of studies, and more tests against a wider environment are required. Data used for developing the model were mainly from experiments of relatively short-duration (<3 yrs) on free-draining soils. On heavier soils, the component of N losses by denitrification, for instance, might become more important. More data will continue to be collected from our lysimeter studies over a longer term and from a wider range of soils. In the experiment of Silva et al. (1999), from which some of the data were derived for developing the model, the urine was applied in the autumn to simulate a worst case scenario of urine impact on nitrate leaching. The critical N application rates calculated in Table 2 may thus represent the lower end or "safer side" of the scale. In our current research programme, cow urine is applied at different times of the year and the nitrate leaching losses from these experiments will be compared with those reported by Silva et al. (1999). The new data will be incorporated into the model to further refine it. Net N mineralisation rate is an important but difficult input variable to determine or estimate; more studies are being conducted to determine the N mineralisation rates in a range of soils collected from different parts of New Zealand, with the aim of developing a submodel for estimating N mineralisation rates.

One of the features of the NLE model is its conceptual simplicity and the small number of parameters required. The model only requires 8–10 input parameters, depending on whether

Table 2 Input variables and estimated critical annual application rates of N in the forms of urea or dairy shed effluent (DSE) which would result in annual average mineral N concentrations in the drainage water reaching the drinking water guideline of 11.3 mg N l^{-1}. Volatilisation and denitrification N losses were considered in Model 1, but not in Model 2. The area covered by urine patches per year was assumed to be 25% of the paddock area. It was assumed that 65% of the N in DSE was mineralisable in one year. NA, not applicable.

	Cut and carry		Grazed pastures			
	Model 1	Model 2	Model 1		Model 2	
			Urine patch area	Non-urine patch area	Urine patch area	Non-urine patch area
Input variables (kg N ha^{-1})						
N fixed	150	150	10	150	10	150
Soil N mineralised	200	200	200	200	200	200
Urine N input	0	0	1000	0	1000	0
Pasture N uptake	300	300	600	300	600	300
Volatilisation losses						
-Urea applied	50	NA	60	50	NA	NA
-DSE applied	0.4	NA	20	0.4	NA	NA
Denitrification losses						
-Urea applied	20	NA	100	20	NA	NA
-DSE applied	60	NA	150	60	NA	NA
Critical application rate to reach 11.3 mg N l^{-1}			**Combined paddock**		**Combined paddock**	
-Urea	392	390	192		162	
-DSE	588	600	301		248	

volatilisation and denitrification are taken into account, which are considerably fewer than those required by complex mechanistic models. Mechanistic models such as LEACHN will have an important role to play for research purposes in improving our understanding of N leaching processes, whilst more empirically based, mass-balance models are probably more suitable for providing management guidance.

Results from the NLE model show that N returned in urine patches has a large impact on N leaching from a grazed paddock and needs to be taken into consideration when modelling N leaching for grazed pastures. However, there is some uncertainty as to the area covered by urine patches per year as affected by stocking rates. If 35% of the paddock area is assumed to be covered by urine patches instead of the 25% used in Table 2, then the critical N application rates are 83–123 kg N ha^{-1} for urea and 128–202 for DSE. These rates are significantly lower than those calculated in Table 2. In addition, it is assumed in the NLE model that each urine deposition occupies a discrete area in the paddock without overlapping, an assumption which may not always hold. Further studies are necessary to ascertain the area and spatial pattern of urine patches in grazed paddocks.

The NLE model has also shown that the critical N application rates differ significantly between cut and carry and grazed conditions, mainly because of the effect on N leaching by urine returns. These differences should be recognised in regulations governing the application of different N sources on land.

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