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Nitrogen Cycling in Intensively Grazed Pastures and Practices to Reduce Whole-Farm Nitrogen Losses

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Nitrogen cycling in intensively grazed pastures and practices to reduce whole-farm nitrogen losses

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Key points

1. A large proportion of the N (>70%) consumed by grazing animals is excreted and this excreta is the main source of N losses from grazed pastures by ammonia (NH_3) volatilisation, nitrous oxide (N_2O) emission and nitrate (NO_3^-) leaching.
2. Management strategies and practices that can reduce N losses in grazing systems include optimising N inputs, manipulating soil N cycling processes, selecting for plants and animals that maximise N utilisation and altering grazing and feeding management.
3. Using stand-off/feed pads or housing systems for removing grazing animals off pasture during greatest risk periods of N loss can reduce excreta deposition to soil at these times, thereby reducing N leaching and N_2O emissions. However, NH_3 losses as a result of N pollution swapping" need to be controlled.
4. Mitigation strategies and practices always need to be evaluated in a whole farm system context to ensure overall efficiency gains through decreasing N losses per unit of animal production and to achieve a tighter N cycle.

Key words: Nitrogen, grazed pasture, leaching, nitrous oxide, ammonia, nitrogen loss, mitigation strategy

Introduction

Grasslands cover between 20 and 40% of the earth's land surface (Reynolds et al. 2005). Some of the land occurs naturally in the semi-arid climate with no external inputs. These extensive low input systems may be legume-based, but production is often limited by N availability. In the more humid regions including Australia, New Zealand (NZ) and parts of North and South America and Europe, most pastoral land is managed. The managed pastures are generally more productive with higher per-hectare animal productivity as an important goal for the pastoral farmers. Input of resources including N fertiliser to the managed pastures can be substantial, resulting in a large N surplus (i.e. N inputs-N outputs in products). For example, N surpluses of 150 to 250 kg N ha^{-1} yr^{-1} occur in highly productive dairy farm systems in the Netherlands and northern Germany (Rotz et al. 2005).

In grazed pastures, the conversion of consumed N into product is low and a substantial amount of N (>70%) is recycled through the direct deposition of animal excreta. Such a low utilization of pasture N reflects a simple feature of the pasture-animal relationship; in most situations, pasture plants require significantly higher concentrations of N to grow at optimal rates than is needed by the grazing ruminant for amino acid and protein synthesis (Haynes and Williams 1993). The proportion of N in the urine increases with increasing N content of the diet. In most intensive high-producing pasture systems, where animal intake of N is high, more than half the N is excreted as urine.

The large N surplus and low N utilisation in intensively grazed pasture systems increases the risk of N losses to waterways and the atmosphere. There have been increasing concerns about the environmental impacts of the N losses, and accordingly, research has been focusing on developing strategies and practices to reduce the losses. This paper first outlines major N transformations and losses in managed grazed pastures and then presents a range of options that can be used to reduce N losses from intensively grazing systems.

Nitrogen transformation processes leading to N losses in grazed pastures

The transformations and losses of N in managed grazed pastures have been previously reviewed (e.g. Haynes and Williams 1993). The N in excreta following deposition undergoes microbial mineralisation before it is released as ammonium ion forms (NH_4^+) and NH_3. This mineralisation of N is much faster from urine than from dung. N can be lost to the atmosphere by NH_3 volatilisation, or converted to nitrate (NO_3^-) through nitrification process by nitrifying bacteria in soil. NO_3^- is then prone to leaching losses and denitrification. Denitrification is the conversion of NO_3^- to gaseous N products (N_2O and N_2). The primary transformations leading to N losses are ammonia volatilisation, nitrification and denitrification. The magnitude of N input to grazed systems is generally the main factor determining the N surplus and therefore the potential for N losses. For example, Ledgard et al. (1999) found that a three-fold increase in total N inputs to intensively-grazed dairy pastures in NZ resulted in a four-fold increase in N surplus, a four-to five-fold increase in gaseous and leaching losses, and a halving of the N use efficiency (Table 1). A summary of dairy farm systems across western Europe showed an even wider range in amount and form of N inputs, N outputs, and N surplus, with denitrification being generally higher overall and N leaching lower (Bossuet et al. 2006).

Table 1 *N inputs and outputs from intensive dairy farm systems in NZ receiving N fertiliser at nil or 410 kg N ha^{-1} yr^{-1} (Ledgard et al. 1999 and unpublished data). Bracketed values are range in N flows measured over 5 years. Data are compared with that from a range of farm systems in western Europe (Bossuet et al. 2006).*

	0 N (NZ)	410 N (NZ)	EU farms
N Inputs (kg N ha^{-1} yr^{-1}):			
Clover N_2 fixation + atm. dep.	170 (90-220)	50 (25-135)	6-133
Fertiliser N	0	410	0-262
Manure N (imported)	0	0	0-22
Purchased feed	0	41	6-489
N Outputs (kg N ha^{-1} yr^{-1}):			
Milk + meat	78 (68-83)	114 (90-135)	20-127
Transfer of excreta to lanes/sheds	53 (41-63)	77 (72-91)	
Denitrification	5 (3-7)	25 (13-34)	10-41
Ammonia volatilisation	15 (15-17)	68 (47-78)	18-81
Leaching	30 (12-74)	130 (109-147)	16-63
Immobilisation of fertiliser N		70 (60-84)	
N balance (kg N ha^{-1} yr^{-1}):	-11 (-74 to +47)	7 (-11 to +24)	
Farm N surplus (kg N ha^{-1} yr^{-1}):	92	387	70-463
N use efficiency (product-N/input-N)	46%	23%	22-36%

Ammonia volatilisation In grazed pastures, biological degradation of animal excreta and hydrolysis of fertilisers containing urea and ammonium ions leads to the continuous formation of NH_3 in the soil, which can volatilise to the atmosphere. Jarvis et al. (1989) found that NH_3 loss from urine patches increased under high N fertilisation because more N was excreted in urine. Less NH_3 is lost from grazing systems than from animal housing systems, where the combined loss from the animal houses, manure storage and field application can be large. Jarvis and Ledgard (2002) made a critical comparative analysis of NH_3 losses from two contrasting model dairy systems in the United Kingdom (UK) and NZ. The desk study has demonstrated distinct differences between the two farming systems in terms of total N input, N off-take, N surplus and per hectare NH_3 loss. These values were 1.7, 1.2, 1.8 and 2.4 times higher in UK than in NZ, respectively. The greater per hectare loss of NH_3 in the UK farm was attributed mainly to the higher fertiliser N input, and to the housing of animals and the subsequent spreading of the manure to the farm. However, when NH_3 loss was expressed in relation to the farm N surplus, there was little difference between the two farms with NH_3 loss being approximately 20% of the N surplus in each case.

Nitrogen leaching Review of research on grazed systems suggests that NO_3^- leaching increases exponentially with increased N inputs (Figure 1). Studies have also shown the greater importance of urine N compared to fertiliser N in contributing to NO_3^- leaching, and urine typically contributes 70%-90% of total N leaching loss (reviewed by Monaghan et al. 2007). Fertiliser N is generally used efficiently by pastures but it enhances pasture N uptake and pasture N concentrations, thereby exaggerating N excretion in urine and increasing risk of loss. NO_3^- leaching losses are much higher during winter as a result of high rainfall and low evapotranspiration. Winter leaching of N can be further exacerbated by dry summer/autumn conditions and an associated slowing down of plant growth, which results in a build-up of NO_3^- levels in soil by autumn (Scholefield et al. 1993). Estimates of N leached from managed pastures vary widely, ranging from 6 to 162 kg N ha^{-1} yr^{-1} and this is due to differences in N input, pasture N uptake, soil drainage and animal type (e.g. Stout et al. 2000). Leaching of Ca^{2+} and other base cations is associated with leaching of NO_3^-, which can potentially decrease soil pH (Haynes and Williams 1993). Leaching of N forms other than NO_3^- is generally low and not measured. However, ammonium leaching can occur on some soils and may be enhanced where

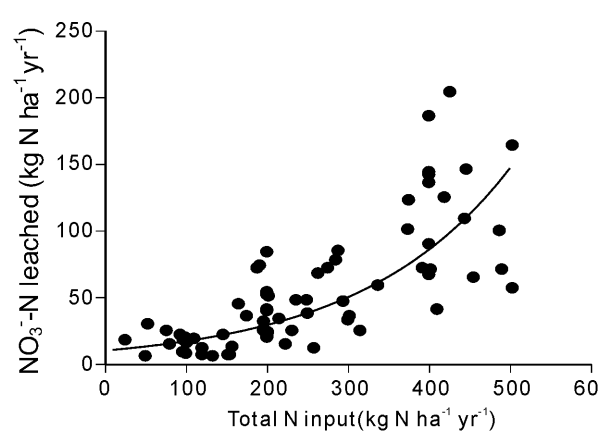

Figure 1 *Nitrate leaching from grazed pasture systems as affected by total N input. Data are a summary of studies in NZ, France and UK. The line of best fit is an exponential function obtained by fitting the data on the log scale.*

mitigation practices target reduced nitrification . Recent research also indicates that in some situations , dissolved organic N can be a significant source of N leached (Jones et al . 2004) .

Nitrous oxide emission High N_2O emission rates in grazed pastures have been observed (e .g . Hyde et al . 2006) and these high rates are associated with N and C from the deposition of animal excreta to the soil and anaerobic conditions as a consequence of soil compaction caused by animal treading . Wet soil conditions soon after N fertilisation or grazing resulted in high N_2O emissions from pastures . N_2O emissions from dairy pasture soils in New Zealand and Australia ranged from 6 to 11 kg N_2O-N ha^{-1} yr^{-1} (Dalal et al . 2003 ; Luo et al . 2008a) , where losses of up to 29 kg N_2O-N ha^{-1} yr^{-1} have been recorded from an Ireland grassland with N application rate of 390 kg N ha^{-1} yr^{-1} (Hyde et al . 2006) . There has been limited research on practices to increase the ratio of N_2 :N_2O emitted and more applied research is needed to identify options to increase loss of the benign N_2 relative to the potent greenhouse gas N_2O .

Management practices to reduce N losses

Strategies to reduce N losses from grazed pastures need to focus on reducing the N surplus in the system and on increasing N use efficiency through converting more N to animal products such as milk , meat and wool . As outlined in Figure 2 , there is a range of possible management strategies and practices that can be used to reduce N losses from grazed pastures . Some are in use , whilst others need further research and development before adoption . Larger reduction of N losses may be achievable through the use of multiple strategies . However , the individual effects of each strategy may not necessarily be cumulative .

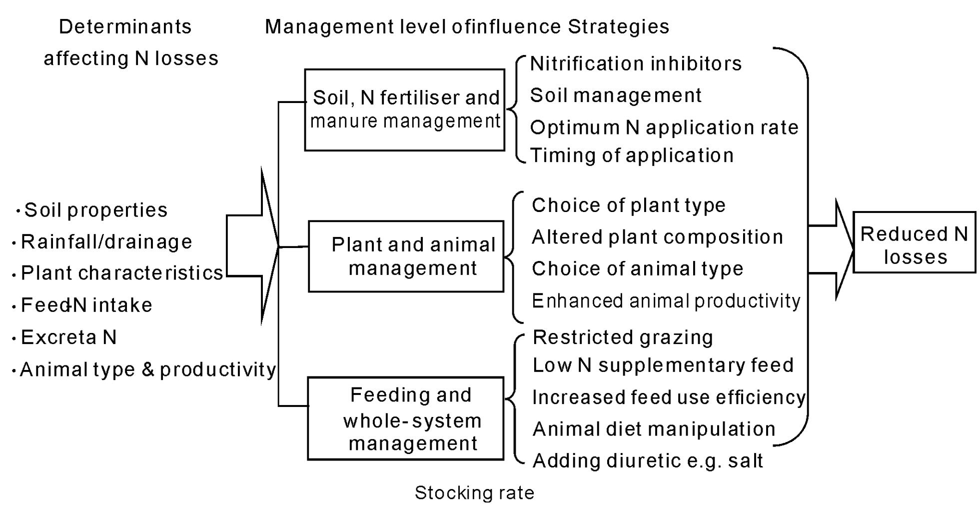

Figure 2 *On-farm management strategies for reducing N losses .*

Soil , N fertiliser and farm manure management

Soil management Soils differ in their risk of N losses . For example , poorer-draining clay-textured soils generally have higher denitrification and N_2O losses and lower N leaching . Reduction in N_2O losses could be achieved by altering soil conditions e .g . liming , improving drainage and avoiding soil compaction , although the general applicability of these methods is limited . However , this can result in the dichotomy of N pollution swapping , such as improved drainage reducing N_2O emissions but increasing N leaching . Farmers can also alter efficiency of N cycling in soil by strategic immobilisation of excess N prior to high N loss periods such as by carbon addition or by controlling N transformation processes in soil (e .g . inhibiting nitrification by use of inhibitors) .

N fertiliser or manure management Appropriate fertiliser or manure N management decisions should be made to optimise application rates and timing to ensure efficient use of the applied N . Limiting the amount of N fertiliser or manure applied under wet conditions in autumn and winter , when pasture growth is slow and soil is wet , can decrease direct leaching of fertiliser or manure N and N_2O emissions (van der Meer 2008) . Techniques , such as incorporation and injection of effluent and animal manure into the soil , are available that reduce the amount of NH_3 volatilisation during and after field application (Sommer et al . 2003) , but these techniques increase CO_2 emissions because more fuel is required (Hansen et al . 2003) . These NH_3 reduction techniques may also lead to N pollution swapping with associated increased N_2O emissions and leaching from the soil (Brink et al . 2001) .

N process inhibitors Technologies employing urease and nitrification inhibitors (NI) can be used as effective mitigation alternatives to control N losses from urine and N fertiliser by acting on the N processes of urea hydrolysis and nitrification respectively. For example, studies have shown that both NO_3^- leaching and N_2O emissions from urine patches can be potentially reduced by up to 70% with land application of NI onto pastures (reviewed by Monaghan et al. 2007). Alternatively, this reduction could be achieved by strategically targeting the urine patches. A detailed animal metabolism study has shown that NI can be delivered to animals (e.g. using a slow-release bolus) and excreted intact in the urine resulting in inhibition of nitrification of urine-N on deposition to soil (Ledgard et al. 2008). A recent field grazing study using this novel approach showed a 30% reduction in both NO_3^- leaching and N_2O emissions (Ledgard and Luo, unpublished). However, accumulation of NH_3 and NH_4^+ due to the use of NI may increase potential of NH_3 volatilisation and NH_4^+ leaching.

Plant and animal selection

Some strains of ryegrass have been shown to have increased rooting depth (Crush et al. 2007) which increases their ability to remove N from a greater depth of soil, thereby potentially reducing the risk of N losses. In plants with high tannin levels which are consumed by grazing animals, less N is excreted in urine relative to dung (e.g. Misselbrook et al. 2005). Studies by Merry et al. (2003) have shown that feeding beef cattle with grass silage containing elevated concentrations of water soluble carbohydrates increased the N use efficiency for microbial growth in the rumen from 46% to 68%. Similarly, Miller et al. (2001) found that dairy cows on a high sugar" variety of perennial ryegrass excreted 18% less N in total and 29% less urine N. Thus, manipulation of plant composition offers potential to reduce N excretion in the urine, thereby reducing the risk of subsequent N losses from this highly concentrated N source. Plants also have the potential to alter soil N cycling via the quality of their residues. We have measured a 10-fold difference in gross N immobilisation rate between non-N-fertilised grassland dominated by *Agrostis* and *Holcus* spp. compared to ryegrass with a regular N fertiliser history. Such differences in immobilisation potential may be important controllers of N losses from N sources such as animal urine.

Animal type influences the efficiency of N cycling. Our field research showed less NO_3^- leaching from sheep or deer than from beef cows for the same level of pasture N intake, associated with greater spread of urine-N and increased efficiency of N cycling (Hoogendoorn et al. unpublished data). Breeding and selection of grazing animals for increasing productivity is also an attractive option to reduce N losses. For example, increasing milk production efficiency in dairy cattle will partition more N to milk formation relative to maintenance and reduce the amount of N that ends up in excreta. Similarly, growing meat-producing animals to their finishing weight more quickly reduces the associated maintenance requirements, thereby reducing total intake and N excretion. This also reduces methane and N_2O emissions thereby reducing their greenhouse gas footprint.

Feeding and whole-system management

Low N feed supplement Pasture typically contain an excess of protein relative to animal requirements and supplements with low protein feed (e.g. maize silage) can increase efficiency of N utilisation (e.g. Kebreab et al. 2001). Potentially, diets can be managed to meet animal requirements such as by supplementing with low protein feed at high levels in non-lactating animals with lesser protein demand than for periods of high milk production. However, such strategies need to account for the whole system, as discussed later.

Diet additives or manipulation Animal supplementation studies have shown that salt addition to feed can also increase urine volume, decrease urine-N concentrations (Ledgard et al. unpublished data) and increase spread of urine, thereby possibly increasing N efficiency and decreasing N losses. Kool et al. (2006) showed that increasing the hippuric acid concentration in urine reduced N_2O emission by up to 50% in a laboratory study, and suggested that manipulating the diet of animals to increase the hippuric acid content of the urine could be potential N_2O mitigation strategy. Further research on this is required.

Nil or restricted grazing systems In temperate environments with winter grazing, practices involving the use of stand-off/feed pads or housing systems can reduce N_2O emissions and NO_3^- leaching. With this practice, animals are kept off grazing paddocks, so excreta deposition is reduced at a time when it leads to greatest N losses (e.g. late-autumn/winter). This practice provides opportunity for controlling N losses, as the animal excreta is collected and can be applied evenly to the pasture at targeted rates and optimum time when the risk for N losses is minimal (Luo et al. 2008b; van der Meer 2008). In these systems, collection and application of large quantities of manure become critical for N use efficiency, as there are many opportunities and places for N compounds to escape from animal manure management systems. Management techniques are increasingly important with these practices to avoid N pollution swapping (e.g. reducing N leaching from paddocks but increasing NH_3 loss from animal houses). However, studies (e.g. van der Meer 2008) suggest that N losses could be much higher for animals grazing on pastures than for housed animals with optimised effluent treatment using anaerobic lagoons. Anaerobic digestion of the animal manure during storage has an additional potential advantage of producing methane as biofuel.

Whole system efficiency Management of all factors involved in the N cycle to reduce N losses in animal grazing systems is complex, and requires a whole farm systems approach. If management practices are used to reduce N loss in one part of the system, the preserved N is prone to loss elsewhere if all parts are not equally well managed. The measure of the N efficiency needs to take account of N losses per unit of production as well as per unit of land. Management practices and technologies that increase the N efficiency within the soil/plant/animal system are likely to increase pasture and animal productivity , which in turn , is likely to increase methane emissions (e .g . increased stocking rates) . Preferably , increased N efficiency would be met by reducing N inputs and operating a tighter N cycle . Additionally , a whole-system approach needs to consider the whole food chain and to account for multiple environmental emissions and efficiency of use of resources such as energy , through the use of tools such as Life Cycle Assessment (LCA) . LCA use is relatively new to agriculture and studies have shown that the majority of environmental emissions are associated with the on-farm stages . A recent dairy farm system study used LCA to examine the effects of some intensification and N mitigation options (Basset-Mens et al . 2008 ; Figure 3) . Intensification using N fertiliser increased milk production per dairy-farm hectare by 23% and increased profitability , but led to a large reduction in energy efficiency and an increase in emissions of N sources and greenhouse gases per kg milk . In contrast , further intensification with maize silage (+78% milk production per dairy-farm hectare) resulted in little change in environmental efficiency (increased efficiency on the dairy farm but significant losses from the crop area) but increased energy use and decreased profitability . Integration of a winter stand-off or feed pad decreased N leaching , N_2O and total greenhouse gas emissions , but reduced profitability .

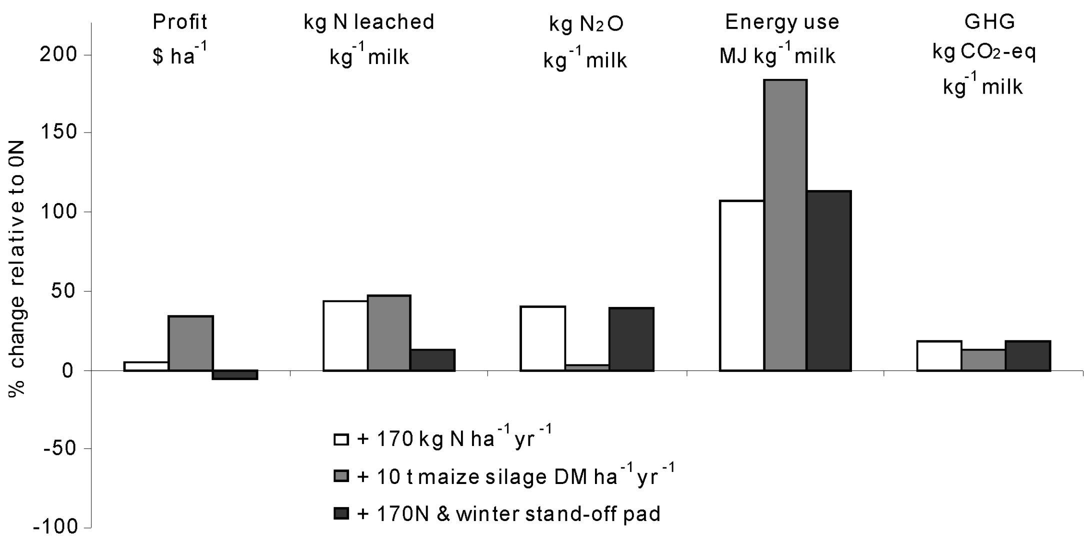

Figure 3 *Effects of intensification on profit ha^{-1} (bsaed on Economic Farm surplus) ,energy use efficiency and environmental efficiency indices estimated using Life Cycle Assessment for NZ dairy farmlet systems relative to a non-N fertiliser clover-based farmlet(from Basset-Mens et al .2008 Jensen et al .2005).*

The actual farmer practices and likelihood of adoption of environmental mitigation technologies are dependent on many factors . For example , a farmer group study using a Multiple Criteria Decision Making tool to examine a range of potential systems involving N mitigations ranked a system using a winter stand-off or feed pad poorly because of criteria such as capital requirements , risk , stress , extra skills and labour needs (Dooley et al . 2005) . These studies highlight that environmental efficiency can vary markedly with different farm systems and practices , but that achieving adoption of environmentally efficient practices by farmers requires consideration of economic and social factors as well as production and environmental factors .

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