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Environmental impacts of grazed clover/grass pastures

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Grazed clover/grass pastures are important for animal production systems and the clover component is critical for its contribution to N inputs *via* biological fixation of atmospheric N_2. The resource efficiency and environmental emissions for clover/grass pastures can differ from that of N-fertilised grass-only pastures. Fixation of N_2 by clover uses photosynthetically-fixed carbon, whereas fertiliser N production consumes fossil fuels and has net greenhouse gas (GHG) emissions. Clover has a higher phosphorus (P) requirement than grass and where extra P fertiliser is used for clover/grass pastures the risk of P loss to waterways is greater than for grass-only pastures. Nitrogen leaching from grazed pasture increases exponentially with increased N inputs and urinary-N contributes 70 to 90% of total N leaching. However, the few studies comparing clover/grass and N-fertilised grass-only pastures at similar total N inputs indicated similar N leaching losses. Nitrous oxide emissions from grazed pastures due to N-cycling of excreta are similar for clover/grass and N-fertilised grass-only pastures at similar total N inputs. However, grass-only pasture requires the application of N fertiliser, which will result in additional specific losses that don't occur from clover-fixed N. Thus, total N_2O emissions are generally higher for N-fertilised grass pastures than for clover/grass pastures. A summary of various whole-system and life cycle assessment analyses for dairy farms from various countries indicated that at similar total N inputs, clover/grass pasture systems can be more efficient than N-fertilised grass systems per kilogram of milk produced from an energy use and GHG perspective whereas results for nutrient losses to waterways were mixed and appear to be similar for both pasture types. In practice, other management practices on farm, such as crop integration, supplementary feeding strategy and winter management, can have a larger overall effect on environmental emissions than whether the N input is derived from fertiliser N or from N_2 fixation.

Keywords: grazed clover/grass pastures; life cycle assessment; nitrogen losses; whole-system

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Introduction

In many temperate regions of the world, legume-based pastures are important for milk, meat and fibre production. In these pasture systems, white clover (*Trifolium repens L.*) is one of the key perennial legumes (Abberton and Marshall, 2005). Perennial legumes are used because of their valuable contribution to production, feed quality and N inputs *via* biological fixation of atmospheric N_2 (e.g., Ledgard, 2001; Woodfield and Clark, 2009). The amount of biological N_2 fixation by clover in temperate pastures ranges from about 10 to 300 kg N ha^{-1} $year^{-1}$ and depends on a number of factors including legume species, soil and climatic conditions, nutrient supply and grazing management (e.g., Ledgard, 2001; Hansen and Vinther, 2001). This fixed N becomes available slowly over time to the grass in pastures after it is released into soil *via* exudates from living legume roots, by mineralisation of senesced legume tissues and in excreta after consumption by grazing animals (Ledgard, 1991). For perennial white clover/ryegrass pastures receiving no N fertiliser, total pasture production is generally correlated with amounts of N fixed by white clover (Hansen and Vinther, 2001; Carlsson and Huss-Danell, 2003).

Use of N fertiliser on clover/grass pastures has generally increased during the last few decades, particularly in more intensively managed pastures under dairy farming in southern hemisphere countries. N-fertilised ryegrass-only pastures are common in Europe. Generally, growth of perennial ryegrass pasture is limited by N and dry matter (DM) production increases as applied N is increased up to a maximum at N input of about 500 kg ha^{-1} $year^{-1}$ (e.g., Vellinga and Andre, 1999; Wilkins and Jones, 2000). Perennial white clover/ryegrass pastures produce a lower DM yield compared to pure ryegrass pastures under high fertiliser N inputs. However, in a recent review on white clover, Andrews *et al.* (2007) concluded that with appropriate management and utilisation of recommended cultivars, pasture production from a perennial ryegrass/white clover pasture is likely to be similar to that from a perennial ryegrass pasture receiving N at 200 kg ha^{-1} $year^{-1}$ and around 70% of that obtained from perennial ryegrass with N input at 350 to 400 kg ha^{-1} $year^{-1}$. Under low fertiliser N input conditions, the ability of clover to contribute N through N_2 fixation and N transfer becomes important (Høgh-Jensen and Schjoerring, 1997).

This review concerns the general environmental impacts of grazed clover/grass pastures with emphasis on N and greenhouse gases. The theoretical implications of using N_2 fixation by clover and/or fertiliser N inputs are considered and research on N leaching and N_2O emissions from clover or fertiliser-N based pastures is reviewed. This is then expanded to cover farm and whole-system life-cycle based environmental analyses of clover/grass and N-fertilised pastures.

Grazed pastures and general environmental implications

Grazing animals have profound effects on legume-based pastoral systems including nutrient removal by grazing and redistribution through excreta. Generally, 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. This low N utilization reflects the relatively high concentrations of N in pasture plants required for metabolic functions and optimum growth compared to that needed by the grazing ruminant for amino acid and protein synthesis (Haynes and Williams, 1993). Increasing the N concentration in grass, such as by increasing the rate of fertiliser N application, can result in a substantial N surplus (i.e. N inputs – N outputs in products). For example, N surpluses of 150 to 250 kg ha^{-1} $year^{-1}$ occur in highly productive dairy farm systems in the Netherlands and northern Germany (Rotz *et al.*, 2005). Increasing the N concentration of the diet generally increases the excretion of urinary N in both absolute terms and as a percentage of the total N excreted. There is an exponential relationship between N intake and N excretion in urine and Scholefield *et al.* (1991) predicted that about 80% of N intake is excreted in urine with a dietary N concentration of 40 g/kg. Grazing cattle return N in urine patches at rates of up to about 1000 kg ha^{-1}, which is far in excess of plant requirements (Haynes and Williams, 1993). Urinary N is in highly mineralisable forms compared to faecal N, and within 3 to 5 days is rapidly converted to plant-available N in soil. This can result in concentrations of inorganic soil N under urine patches up to 10 times greater than under dung patches, and more than 30 times greater than areas unaffected by excreta (Afzal and Adams, 1992). This has a marked effect on spatial and temporal variability in clover/grass dynamics. High soil N from urine and animal rejection of herbage on or near dung patches can adversely affect legume growth and N_2 fixation by altering the legume-grass competitive interaction and depressing N_2 fixation (Vinther, 1998). Decline in clover contribution in clover/grass pastures with increased fertiliser N application and soil mineral N has also been well documented (e.g., Ledgard *et al.*, 2001; Schils and Snijders, 2004). In soils where the concentration of inorganic N is high, legumes utilise soil N and fix less N_2. A review of developments in germplasm improvement in white clover reveals that progress has been made in the breeding of white clover varieties less affected by applied N (Abberton and Marshall, 2005). Furthermore, there is ample evidence that strategic N application in spring increases total dry matter yield, with only limited adverse effects on clover content (Frame, 1987; Eckard and Franks, 1998; Schils, 1997).

The large N surplus and return of N in localised patches of excreta at high N input rates in intensively grazed pasture systems markedly increases the risk of N loss to waterways and the atmosphere. The transformations and losses of N in managed grazed pastures have been previously reviewed (e.g., Haynes and Williams, 1993; Bolan *et al.*, 2004). The excretal returns, particularly urinary N, from grazed animals are typically the major sources of N lost from grazed pastures. The primary transformations leading to N losses are NH_3 volatilisation, nitrification and denitrification. Leaching losses of NO_3^- to waterways and emissions of NH_3 and N_2O to the atmosphere from grazed pastures have significant environmental implications (Oenema *et al.*, 1997; Di and Cameron, 2002).

Methane (a greenhouse gas) emissions from enteric digestion of pasture by grazing ruminants are large (e.g., Clark, 2001). The main determinant of methane emissions is the amount of feed consumed by the ruminant animal and is likely to be similar for perennial ryegrass and white clover/ryegrass pastures. However, other temperate forage legumes such as *Lotus corniculatus* and *Hedysarum* that contain compounds such as condensed tannins can lead to a reduction in methane emissions on a per unit dry matter intake basis (Woodward *et al.*, 2001; Ramirez-Restrepo and Barry, 2005). They also have the potential to reduce N leaching and N_2O emissions by increasing the relative N excretion in faeces compared to urine

(Carulla *et al.*, 2005), thereby reducing the amount of urinary N, with high risk of N loss.

In grazed pastures, animal treading damage during grazing under wet soil conditions limits pasture growth and reduces soil infiltration rates (Drewry, Cameron and Buchan, 2008). Animal treading of pasture can also increase soil bulk density and consequently cause an increase in mechanical impedance to root penetration and a reduction in aeration and/or an increase in waterlogging of soil. This will have a negative effect on legume growth, productivity, and N_2 fixation in pasture (Menneer *et al.*, 2004). In addition, the treading damage also increases the risk of run-off losses of other nutrients, such as P, from grazed pastures (Monaghan *et al.*, 2005).

Clover N_2 fixation versus N fertiliser

N_2 fixation process

Biological N_2 fixation places a carbon burden on a host plant (Schulze, 2004), so that it is associated with a higher metabolic cost than N acquisition by other routes such as nitrate or ammonium uptake from soil (Pate and Layzell, 1990). For example, although the carbon cost of N_2 fixation is variable across legume species, it has been estimated at an average of 6 g C per 1 g N fixed (Vance and Heichel, 1991). However, it has recently been proposed that although rhizobial and arbuscular mycorrhizal symbiosis may consume 4 to 16% of the carbon recently fixed by photosynthesis these symbioses actually stimulate photosynthesis allowing legumes to take advantage of the nutrients supplied by the microsymbionts without compromising the total amount of photosynthates available for plant growth (Kaschuk *et al.*, 2009). The process of N_2 fixation using photosynthetically-derived C is "greenhouse gas neutral". In contrast, manufacturing of N fertilisers consumes fossil fuel, resulting in a net increase in GHG emissions.

Product feedback regulation is widespread in biochemical reaction chains, including the N_2 fixation process. It is well known that nitrate inhibits nitrogenase activity, reducing N_2 fixation itself in legume nodules and can also reduce the mass of N_2-fixing nodules (e.g., Lucinski, Polcyn and Ratajczak, 2002). Many field studies have shown that increasing soil N status, including following N fertiliser or from excreta, can reduce clover growth in mixed clover/grass pastures and further reduce N_2 fixation due to clover substituting N uptake for N_2 fixation. For example, Ledgard *et al.* (2001) showed a 30 to 70% decrease in total N_2 fixation, depending on the time of application and the grazing management. In grazed clover/grass pastures, this N feedback mechanism means an increase in overall N efficiency and reduced potential for loss. In areas where N inputs from excreta occur there will be low associated input from N_2 fixation whereas fertiliser N is applied uniformly and N loss risk is increased where it coincides with patches of excreta.

N fertiliser

Production of N fertiliser has a high fossil-fuel energy requirement, particularly associated with the ammonia production stage which commonly uses natural gas, and in total can equate to approximately 60 MJ/kg N (e.g., Jenssen and Kongshaug, 2003). Greenhouse gases are released in the production and use of synthetic N fertiliser. For urea production, transport, spreading and CO_2 emission after application, the total GHG emissions (excluding those associated with N_2O from soil) can equate to approximately 3.5 to 4.0 kg CO_2-equivalent per kilogram of urea-N, of which less than 10% is from transport and spreading (e.g., Williams, Audsley and Sandars, 2006; Ledgard, unpublished).

Environmental implications

The N concentration in pasture exceeds that required by grazing animals and white clover has higher digestible protein (N) and lower soluble carbohydrate concentrations than perennial ryegrass (e.g., Vinther and Jensen, 2000; Wilkins and Jones, 2000). This can result in poor utilisation of clover-protein, increased urinary N output and consequently greater risk of environmental N pollution (Weller and Jones, 2002). For example, Wilkins and Jones (2000) measured a greater proportion of N intake by cattle partitioned to urinary-N output with a white clover diet than with a ryegrass-based diet. However, the clover-N feedback mechanism, whereby N_2 fixation decreases with high N inputs, acts to enhance N efficiency. While some studies with clover/grass pastures have shown lower N losses than from N-fertilised pastures with higher total N inputs, there have been few studies where clover-N and fertiliser-N have been compared in grazed systems with similar total N inputs. Two studies that attempted this comparison under sheep grazing (Cuttle *et al.*, 1992) or dairy cow grazing (Sprosen, Ledgard and Thom, 1997) showed similar N leaching from clover/grass and N-fertilised grass-only systems, although large temporal variability affected the precision of these comparisons. However, white clover/grass pastures are more N-efficient than white clover-only pasture and Loiseau *et al*. (2001) measured high N leaching from the latter, which could be attributed to high clover N concentration, greater N excretion and lack of associated grasses capable of rapid uptake of recycled N.

In mixed clover/grass pasture, relatively high clover levels in autumn could increase the risk of subsequent N loss, and there may be limited opportunity to reduce the effects of this except by supplementary feeding of low-N feeds such as maize silage. In contrast, fertiliser N use can be timed for optimum N efficiency with autumn application avoided and spring and summer applications targeted instead. Nevertheless, the pulse of N in soil following fertiliser N application at any time result in greater risk from ammonia loss and from leaching following high rainfall events, than from the steady release from mineralisation of N from clover residues in soil.

Phosphorus requirements for good clover performance are relatively high (Caradus, 1994; Sinclair *et al.*, 1996). Additionally, white clover is less competitive than perennial ryegrass in mixed clover/grass pastures, which is generally attributed to their coarser less-branched root system with low absorptive surface area. In the Netherlands, fertiliser P recommendations for clover/grass pastures are lower than for N-fertilised grass pastures at the same soil P status, due in part to differences in overall productivity but also to research indicating greater P responses in N-fertilised grass pastures (Schils and Snijders, 2004). The latter was attributed to a lack of observed response by clover to P associated with reduced competitiveness of clover. In contrast, in New Zealand the optimum soil P test has been set based on optimising total production and clover content of pasture, and application of relatively high rates of P fertiliser is a common practice to maintain white clover levels in pastures (e.g., Kemp, Condron and Matthew, 1999). However, the potential for P loss to waterways increases with increased soil P concentration (Monaghan *et al.*, 2007) and therefore can be greater for clover/grass pastures.

Research on N leaching and N_2O loss

N leaching from grazed pastures

Research on grazed systems indicates that NO_3^- leaching increases exponentially with increased N input from fertilisers and/or N_2 fixation by clover (Figure 1). This is mostly associated with an increase in dry matter production, N uptake and recycling in animal excreta resulting in a corresponding increase in leaching losses from urine patches (Ledgard, 2001). Various studies have also shown the much greater importance of urinary N compared to fertiliser N in contributing to NO_3^- leaching (because of the much larger specific rate of N application in urine), and urine typically contributes 70 to 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 grass N concentration, thereby increasing both N excretion in urine and the risk of environmental loss. For example, Ledgard, Penno and Sprosen (1999; unpublished data) measured N leaching losses (averaged over 5 years) from clover/grass pasture systems in grazed dairy farmlets receiving N inputs of 0, 207 or 410 kg ha^{-1} year^{-1} at 30, 63 and 130 kg ha^{-1} year^{-1}, respectively. Losses of NO_3^- through leaching are much higher during winter as a result of high soil moisture status, rainfall and low evapotranspiration. Winter leaching of N can be further exacerbated by dry summer/autumn conditions and an

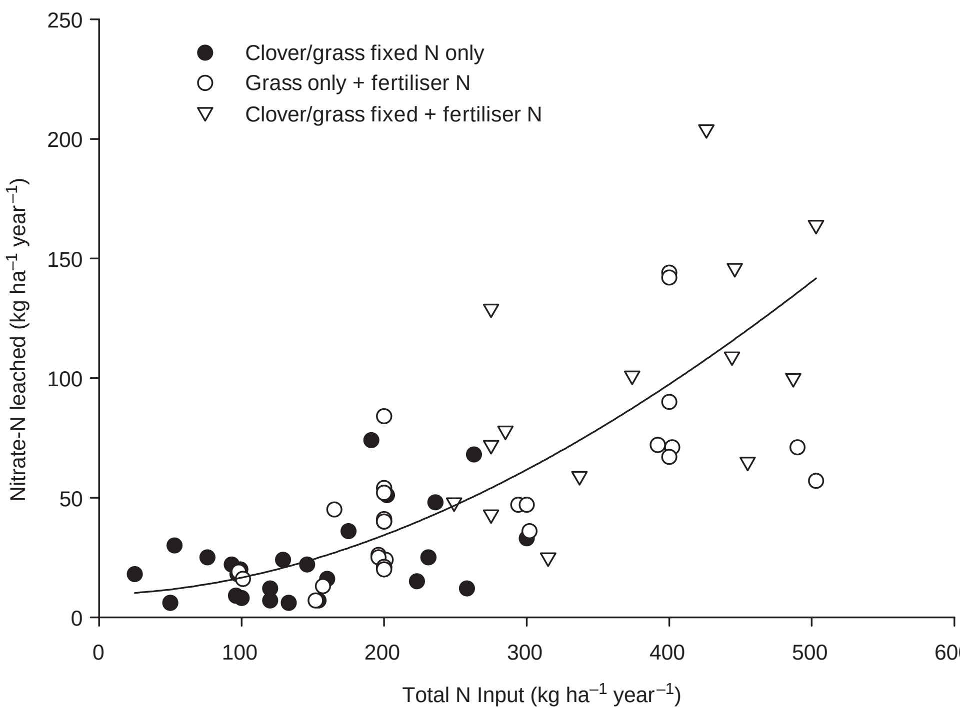

Figure 1. Nitrate-N leaching from grazed pasture systems as affected by total N input from fertilisers and/or N_2 fixation by clover. Data are a summary of studies from NZ, France, UK and Denmark. The line of best fit is an exponential function obtained by fitting the data on the log scale. Updated from Ledgard (2001).

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 about 5 to 200 kg ha^{-1} year^{-1} and this is due to many factors including differences in N input, N output in excreta, soil drainage and animal type (e.g., Monaghan *et al.*, 2007). Leaching of N in forms other than NO_3^- is generally low. However, ammonium leaching can occur in some soils and may be enhanced where mitigation practices target reduced nitrification. Research also indicates that in some situations, dissolved organic N can be a significant source of N leached (Jones *et al.*, 2004).

Eriksen, Vinther and Søegaard (2004) observed higher leaching losses of N from grazed N-fertilised ryegrass pasture (on average 47 kg ha^{-1} year^{-1}) than from grazed non-N-fertilised clover/ryegrass pasture (on average 24 kg ha^{-1} year^{-1}). Over time the losses from the clover/ryegrass pasture decreased due to a reduction in N_2 fixation together with a reduction in dry matter production that in turn led to a lower grazing intensity and lower rate of recycling of animal excreta. The summary of research findings on N leaching from grazed pastures (Figure 1) shows overlap of N leaching values estimated from pastures with or without clover at similar N inputs. However, in long-term pastures, N inputs from N_2 fixation are usually less than about 200 kg ha^{-1} year^{-1} thereby limiting maximum N leaching from non-N-fertilised clover/grass pastures, whereas fertiliser N may be used at much higher annual rates of application, with potential for high N losses.

N leaching from clover/grass pastures in crop rotations

Dairy production systems in Europe are, to a large extent, based on ley-arable rotations (Vertés *et al.*, 2007) that are characterised by three phases: pasture, ploughing out, and subsequent arable cropping (Watson *et al.*, 2005). An example from Denmark is shown in Figure 2 with two of six years in grazed pasture associated with the lowest N leaching. Generally, the ploughing-out phase after the pasture ley phase carries the highest risk of NO_3^- leaching (for several years) as N accumulated in the soil during the pasture phase is released upon cultivation. However, in every phase there are options to reduce NO_3^- leaching.

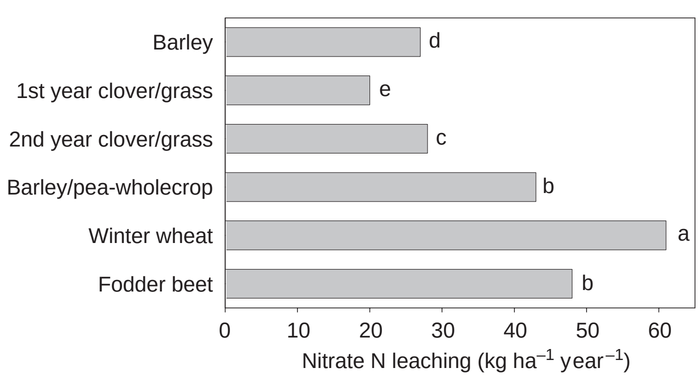

Figure 2. Nitrate-N leaching from a ley/arable crop rotation. Mean of 4 years. Values with the same letter are not significantly different. From Eriksen, Askegaard and Kristensen (1999).

The pasture phase: The potential for N leaching from the pasture phase depends on the sward type and use, particularly whether it is cut or grazed (e.g., Table 1). Wachendorf *et al.* (2004) also showed much lower N leaching from cut pasture than from grazed pasture. High N surpluses can occur from pastures grazed by dairy cows and supplemented with high feed-N (Table 1), although N leaching losses from young pasture swards are much less than indicated from the N budget surplus (Eriksen *et al*., 2004). In the early pasture phase, NO_3^- leaching losses are usually low as much N can be accumulated in the sward and in soil organic N, but over time the N loss depends on the equilibrium between inputs and changes in the soil organic N pool. This equilibrium is not reached within the first years of the clover pasture (Johnston *et al.*, 1994) and it also takes longer to be achieved in clover/grass pastures than in grass-only pastures due to the self-regulatory nature of legumes (Eriksen *et al.*, 2004).

Pasture cultivation: The ploughing of grazed pasture is followed by a rapid and extended period of N mineralisation. The release of accumulated soil N following pasture cultivation is substantial in the first year, with fertiliser N replacement values often exceeding 100 kg N ha^{-1} (Eriksen, Askegaard and Søegaard, 2008), but with relatively little effect of pasture management and age, even in situations with huge differences in N input during the pasture phase (Eriksen, 2001; Hansen, Eriksen and Jensen, 2005). It has been demonstrated that mineralisation of N following pasture cultivation is a two-stage process with a rapid mineralisation over the first 160 to 230 days followed by a second phase with mineralisation rates 2 to 7 times lower than in the first phase (Vertés *et al.*, 2007). However, it is recognised that pasture-rich crop rotations are more beneficial to soil fertility than arable rotations and although differences in the initial residual value between pastures of different age or management often seem small, the accumulated effect can be considerable and impacts on soil C and N dynamics for decades (Springob *et al.*, 2001). Delayed ploughing or reduced cultivation methods can reduce the amount of N mineralised prior to winter leaching (e.g., Djurhuus and Olsen, 1997).

The release of large quantities of N from the clover/grass residues means that fertiliser-N use on subsequent cereals can be reduced or even eliminated in the first

Table 1. Annual N budget (kg ha^{-1} year^{-1}) of six pasture management systems[1]

	Ryegrass-only			Clover/ryegrass		
	Cut	Grazed low[2] N	Grazed high[2] N	Cut	Grazed low[2] N	Grazed high[2] N
Input						
Fertilizer	300	300	300	0	0	0
N_2 fixation[3]	0	0	0	300	258	266
Animal manure	0	222	320	0	240	326
Output						
Herbage yield	287	240	292	288	271	342
Balance	13	282	328	12	227	250

[1] Data are mean of production years 1 to 3 (Eriksen, 2001).
[2] Grazed low and high N refer to grazing by dairy cows with 140 and 310 g N per day, respectively, from supplements.
[3] Rates of N_2 fixation are estimated.

following crop. Catch crops are useful during winters in the arable phase of the crop rotation to reduce N leaching, by removing soil mineral N from the soil profile before winter drainage starts. An example is given in Figure 3, where two grass-clover swards were ploughed on coarse sandy soil in Denmark. Perennial ryegrass as a catch crop reduced NO_3^- leaching by 66 to 88% compared to bare soil and in the treatment with barley harvested green and followed by Italian ryegrass NO_3^--N leached was reduced by more than 90% to less than 10 kg ha^{-1} year^{-1}.

Organising pasture-arable crop rotations: A key objective in designing pasture-arable crop rotations is to optimise the period of the pasture phase. For the individual farmer this depends on the requirement for feed and access to grazing. The common motivation for cultivation of pasture is yield loss due to sward deterioration caused by factors such as pest damage, compaction from wheel traffic and invasion of less productive natural grasses (Hoving and de Boer, 2004). Leaching losses of N generally increases with increased age of the pasture sward and with maximum soil organic N accumulation. However, the huge residual N effect from pastures may, to a large extent, be utilised by crops instead of being leached if an efficient strategy is used.

N_2O emission

Reported N_2O emission rates from soils under clover/grass pasture grazed by dairy cows in New Zealand and Australia range from 6 to 11 kg of N_2O-N per hectare per year (Dalal *et al.*, 2003; Luo *et al.*, 2008a). At comparable levels of production it is likely that the N_2O emissions resulting from N-cycling of animal excreta will be similar for both clover/grass and grass pasture. However, because grass pasture requires inputs of N fertiliser, this type of pasture will have additional fertiliser-specific losses. For example, N_2O-N losses of up to 29 kg ha^{-1} year^{-1} have been measured in grass pastures in Ireland that received N application of 390 kg ha^{-1} year^{-1} (Hyde *et al.*, 2006). Ryden (1983) reported N losses of 1.3 kg ha^{-1} year^{-1} by denitrification in grass pasture that did not receive fertiliser N inputs and of 11 kg ha^{-1} year^{-1} after N fertilisation at a rate of

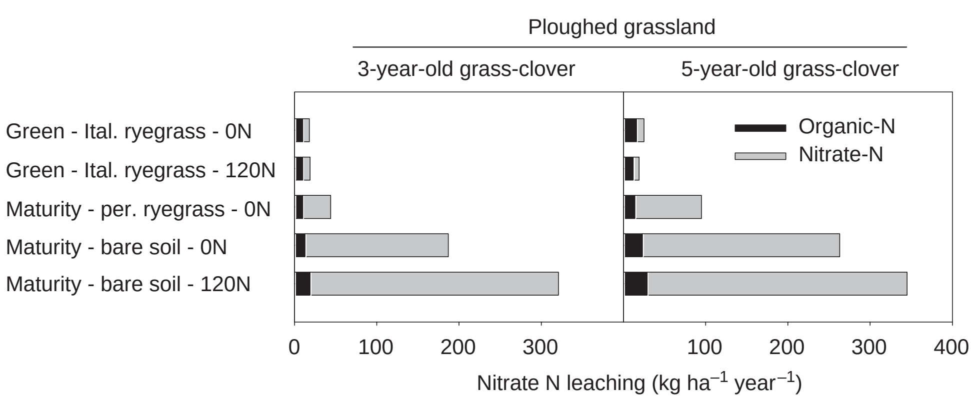

Figure 3. Leaching of nitrate N and dissolved organic N after spring cultivation of grassland followed by 1) barley harvested green with Italian ryegrass undersown, 2) barley harvested at maturity with perennial ryegrass undersown, and 3) barley harvested at maturity without catch crop. From Hansen, Eriksen and Vinther (2007).

250 kg ha^{-1} year^{-1}. In an Australian study (Eckard *et al.*, 2003), N losses from total denitrification were significantly less from unfertilised clover/ryegrass pasture than from the same pasture receiving N input of 200 kg ha^{-1} year^{-1} (as either ammonium nitrate or urea), at 6 kg ha^{-1} year^{-1} without N fertiliser and 15 and 13 kg ha^{-1} year^{-1}, respectively, for the two N fertiliser treatments. Similar denitrification losses have been reported by other workers on clover/grass pastures in New Zealand. For example, Ruz-Jerez, White and Ball (1994) reported annual N loss of 3.4 kg ha^{-1} compared to 19.3 kg ha^{-1} after application of 400 kg ha^{-1} year^{-1} of fertiliser N and Ledgard *et al*. (1999) reported total N loss through denitrification of 3 to 7 kg ha^{-1} year^{-1} without added N compared with 10 to 25 kg ha^{-1} year^{-1} after N application of 200 kg ha^{-1} year^{-1} as fertiliser N. Much higher denitrification losses were measured from N-fertilised pasture in Northern Ireland at up to 154 kg ha^{-1} year^{-1} after N input of 500 kg ha^{-1} year^{-1} as N fertiliser (Watson *et al.*, 1992).

Eckard *et al*. (2003) noted that the denitrification losses were highest in winter when the soil moisture was highest. Very high N_2O emission rates have been observed in grazed pastures (e.g., de Klein, Smith and Monaghan, 2006a; Luo, Ledgard and Lindsey, 2008b; Luo *et al.*, 2008c) when wet soils become compacted by animal treading. Treading causes anaerobic conditions and animal excreta provide abundant N and C. Thus, high N_2O emission rates can occur on wet soils soon after N fertilisation or grazing.

Rochette and Janzen (2005) reported average N_2O-N emissions from annual crop legumes of 1.0 kg ha^{-1}, 1.8 kg ha^{-1} for pure forage crops and 0.4 kg ha^{-1} for legume/grass mixes, and noted these values were much lower than those predicted using 1996 IPCC methodology. As a consequence, these authors recommended that biological N_2 fixation as a process be removed from the IPCC N_2O inventory methodology and N_2O emissions induced by the growth of legumes be estimated only as a function of crop residue decomposition. In a recent review and update of the IPCC guidelines (de Klein *et al.*, 2006b), no indirect N_2O emissions have been attributed to biological N_2 fixation. Nevertheless, the effects of fixed N are accounted for in grazed clover/grass pastures *via* N_2O emissions from excreta (derived from consumed clover) and from increased grass growth (which is consumed and excreted) from mineralised clover N residues.

There have been few attempts to determine N_2O emission factors for clover/grass pastures compared to N-fertilised grass-only pastures. Corré and Kasper (2002) reported an experiment carried out in The Netherlands (with only one repetition), in which N_2O emission from clover/grass plots was compared with that from grass-only plots (from the end of the growing season until the next spring). They determined that the emission factor for clover-N was 0.2%, compared to 1.3% for fertiliser-N. However, due to lack of repetition this calculation has a high level of uncertainty.

There is limited research data on the ratios of N_2:N_2O emitted from pasture, including clover/grass. Research is particularly needed to determine strategies that will increase the ratio of N_2:N_2O emitted. Such research will allow identification of practical options to increase losses of benign N_2 gas relative to the potent greenhouse gas N_2O.

Farm N budgets and N use efficiency

The previous two sections referred to measurements from plots or paddocks with pasture but there is also a need to consider the whole farm and account for effects of nutrients in external inputs such as brought-in feed and within-farm transfers such as from farm dairy effluent or stored slurry. The magnitude of N input to grazed farm 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 2). A summary of dairy farm systems across western Europe (Bossuet *et al.*, 2006) 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 than in the New Zealand study.

A three-year comparison between fertiliser-N and clover-N dairy systems using ryegrass-based pasture was carried out by Schils *et al.* (2000a,b). The farm systems had the same number of cows but more land was used in the clover system due to lower pasture production (Table 3). An intensive monitoring programme was carried out, with measurements including forage production and quality, feed intake and milk production. Biological N_2 fixation was calculated from clover contents and clover/grass yields. Nitrate concentration in drain water was measured on a weekly basis, while GHG emissions were calculated using IPCC emission factors (Schils *et al.*, 2005). The N surplus per ha was higher for the N-fertilised system, but this was mainly related to the higher milk production per ha. The N surplus per kg milk was similar for both systems. The average NO_3^--N concentration in drain water was 26 mg/L for the N-fertilised system and 28 mg/L for the clover system. However, peak concentrations in clover paddocks were as high as 100 mg/L, whereas peak concentrations on N-fertilised

Table 2. N inputs and outputs from intensive dairy farm systems in NZ receiving N fertiliser at nil or 410 kg ha^{-1} year^{-1} and data from a range of farm systems in western Europe

	New Zealand[1]		EU farms[2] (range)
	0N	410N	
N Inputs (kg ha^{-1} year^{-1})			
N_2 fixation + atmospheric deposition	170 (90–220)[3]	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 ha^{-1} year^{-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 ha^{-1} year^{-1})	−11 (−74 to +47)	7 (−11 to +24)	
Farm N surplus (kg ha^{-1} year^{-1})	92	387	70–463
N use efficiency (product-N/input-N)	0.46	0.23	0.22 to 0.36

[1] Ledgard *et al.*, 1999 and unpublished data.
[2] Bossuet *et al.*, 2006.
[3] The range of N flows measured over 5 years.

Table 3. Characteristics of fertiliser-N and clover-N dairy farm systems[1]

	Fertiliser-N	Clover-N
Cows (number)	59	59
Area (ha)	34	41
Milk production[2] (kg/ha)	13884	12053
Fertiliser N (kg ha^{-1} year^{-1})	208	17
Manure effective N (kg ha^{-1} year^{-1})	67	52
Clover fixed N (kg ha^{-1} year^{-1})	0	176
Grazing system (h/day)	24	24
N input (kg ha^{-1} year^{-1})	333	279
N output (kg ha^{-1} year^{-1})	80	69
N surplus (kg ha^{-1} year^{-1})	253	212
Nitrate-N leaching (kg ha^{-1} year^{-1})	20	22
Nitrous oxide N (kg ha^{-1} year^{-1})	9.4	6.6
GHG[3] total (kg/ha)	16065	12198
Nitrate-N leaching (kg/t milk)	1.4	1.8
Nitrous oxide N (kg/t milk)	0.7	0.5
GHG[3] total (kg/kg milk)	1.2	1.0

[1] Schils *et al*. (2000a,b).
[2] Fat and protein corrected milk.
[3] Greenhouse gas emissions as CO_2-equivalent.

paddocks were never higher than 60 mg/L. While there was no significant difference in the calculated N leaching between the systems, when expressed as N leached per kg milk there was a trend for it to be 25% higher from the clover system than from the N-fertilised system. Calculated N_2O emissions (direct and indirect) were lower on the clover system, both per hectare and per kilogram of milk, mainly due to the much lower use of fertiliser-N.

Whole-system life-cycle-based environmental analysis

Ideally, overall efficiency of a farm system and environmental emissions should account for the life-cycle of the whole system and its associated inputs and emissions. The use of life cycle assessment (LCA) methodology in agriculture has increased during the past 5 years. While LCA has typically been used to evaluate products throughout their full life cycle including all transportation, consumption and waste stages (e.g., Guinée *et al*., 2002), it has also been used to examine the "cradle-to-farm-gate" stage of the life cycle. Thus, it accounts for extraction of raw materials and the production, delivery and use of resources on farm such as fertilisers, lime, fuel and electricity as well as all on-farm emissions (e.g., Cederberg and Mattson, 2000). These important sources of environmental emissions are otherwise excluded in most studies and this can result in underestimation of total emissions, particularly for farm systems with relatively high inputs of brought-in feed and fertiliser.

LCA has been applied to a study by Ledgard *et al*. (1999) of two New Zealand dairy farmlet systems with clover/ryegrass pasture receiving N inputs of 0 or 207 kg ha^{-1} year^{-1} (Table 4; Figure 4). This is compared with an associated hypothetical system based on ryegrass-only pasture receiving N at 160 kg ha^{-1} year^{-1} which was assumed to achieve the same productivity as the nil-N clover/ryegrass pasture. The rate of 160 kg ha^{-1} year^{-1} was based on the average biological N_2 fixation rate measured in the latter pasture over 5 years (Ledgard, unpublished). Total energy use in the 207N and 160N systems was 109 and 109% higher per 1 kg of milk than the 0N system, while fossil fuel use was 155 and 149% higher, respectively. Total GHG emissions per 1 kg milk were 11 and 12% higher for the 207N and 160N systems and were dominated by methane and N_2O. Corresponding increases for eutrophication potential per 1 kg of milk (using the method of Guinée *et al*., 2002) were 84 and 33% higher than the 0N system, due

Table 4. Inputs, productivity and environmental efficiency[1] of dairy farmlets with white clover/ryegrass pastures receiving N at 0 or 207 kg ha^{-1} year^{-1} compared to a hypothetical farmlet with ryegrass-only and receiving N at 160 kg ha^{-1} year^{-1}

	0N	207N	160N grass-only
Cows/ha	3.3	3.3	3.3[2]
Milk production (kg ha^{-1} year^{-1})	13210	15460	13210[2]
Fertiliser N (kg ha^{-1} year^{-1})	0	207	160
N_2 fixation (kg ha^{-1} year^{-1})	160	100	0
Dry matter intake (kg cow^{-1} year^{-1})	4520	5296	4520[2]
N leaching (kg ha^{-1} year^{-1})	30	63	30[2]
Energy use[3] (MJ ha^{-1} year^{-1})	9069	22438	18926
Fossil fuel use[3] (MJ ha^{-1} year^{-1})	6597	19966	16453
Greenhouse gasses[3] (kg ha^{-1} year^{-1})	13232	17525	14892

[1] Ledgard *et al.*, 1999 and unpublished data.
[2] Assumed to be the same as for the 0N farmlet.
[3] Energy use and greenhouse gas emissions were estimated using a whole-system LCA method.

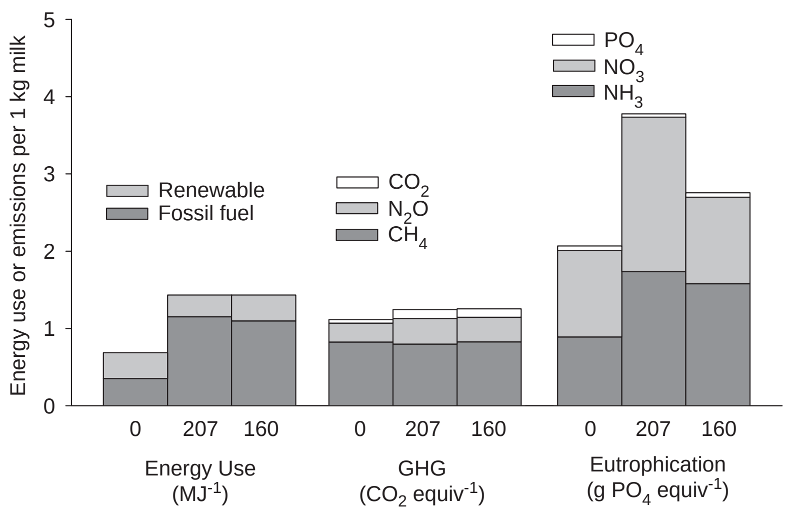

Figure 4. Energy use, GHG (greenhouse gasses as CO_2 equivalents) emissions and eutrophication (as PO_4 equivalents) potential (based on life cycle assessment) from dairy farmlet systems (Ledgard et al., 1999; Table 4) with clover/grass pasture receiving fertiliser-N at 0 or 207 kg ha^{-1} year^{-1} or grass-only receiving fertiliser-N at 160 kg ha^{-1} year^{-1}.

to increased N leaching and/or ammonia emissions.

In the Netherlands, Thomassen *et al.* (2008) used LCA to compare results from a selection of organic and conventional dairy farms and found similar GHG efficiency for both but lower total energy use and eutrophication potential per kg milk from the organic farms. In contrast, similar comparative studies in Sweden (Cederberg and Mattson, 2000; Cederberg and Flysjo, 2004) showed lower energy use and GHG emissions per 1 kg of milk from organic farms but higher eutrophication potential per 1 kg of milk. An important factor confounding environmental emissions estimated for the organic farms in the European studies was the use of concentrates, roughage and crops which differed from that on conventional farms, whereas the New Zealand study was solely based on long-term perennial white clover/ryegrass pasture. Another New Zealand grazing system study with white clover/ryegrass pasture as the sole feed source on farmlets receiving N at 0 or 139 kg ha^{-1} year^{-1} also showed lower energy use (by 51%), GHG emissions (by 15%) and eutrophication potential (by 36%) per 1 kg of milk produced from the system based on clover N_2 fixation as the only N input (Basset-Mens, Ledgard and Boyes, 2009).

Overall, these farm system and LCA studies indicate that at similar N inputs, clover/grass pasture systems can be more efficient than N-fertilised grass systems from an energy use and GHG perspective whereas results for nutrient losses to waterways were mixed and appear to be similar for both systems. In practice, other management practices on farm, such as crop integration, supplementary feeding strategy and winter management options (e.g., Eriksen, Askegaard and Kristensen, 1999; de Klein and Ledgard, 2005; Monaghan *et al.*, 2007) can have a larger overall effect on environmental emissions than whether the N input is derived from fertiliser N or from N_2 fixation.

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