# Environmental impacts of grazed pastures

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Abstract
Large nitrogen (N) surplus and return of excreta-N in localised patches at high N rates in intensively grazed pasture systems markedly increases the risk of N losses to waterways and the atmosphere. Here are described the main routes of N input to grazed pastures, losses via N leaching, methane ($CH_4$) and nitrous oxide ($N_2O$) emissions. Furthermore farm N budgets and N use efficiency in relation to management strategies that can be applied to reduce N losses are discussed. Nitrate leaching increases exponentially with increased inputs and is closely related to urine patches, which also influence the leaching of dissolved organic N. High $N_2O$ emission rates in grazed pastures are related to fertiliser-N or N in excreta combined with compaction by animal treading. Grazing may considerably reduce $CH_3$ emissions compared to indoor housing of cows. Pastures are occasionally cultivated due to sward deterioration followed by a rapid and extended period of N mineralization, contributing to an increased potential for losses. Good management of the pasture (e.g. reduced fertiliser input and reduced length of grazing) and of the mixed crop rotation during both the grassland and the arable phase (e.g. delayed ploughing time and a catch crop strategy) can considerably reduce the negative environmental impact of grazing. It is important to consider the whole farm system when evaluating environmental impact. In particular for green house gasses since the pasture may serve as a source of $N_2O$ and indirectly of $CH_3$, but also as a sink of $CO_2$ influenced by management practices on the farm.

Keywords: Clover/grass pastures, grazing, nitrogen losses, pasture management

## Introduction

Grazing animals have profound effects on 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 quantity of N (>70%) is recycled through the direct deposition of animal excreta. This low N utilization by grazing animals reflects the relatively high concentrations of N required for metabolic functions and optimum growth of plants 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 N fertiliser 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 N $ha^{-1}$ $yr^{-1}$ 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 4% N.

Grazing cattle return N in urine patches at rates of up to about 1000 kg N $ha^{-1}$, which is far in excess of plant requirements (Haynes and Williams, 1993). Urine N is in highly mineralisable forms compared to dung N, and within 3-5 days is rapidly converted to plant-available N in soil. This can result in 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).

The large N surplus and return of excreta-N in localised patches at high N rates in intensively grazed pasture systems markedly increases the risk of N losses to waterways and the atmosphere. 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 ammonia ($NH_3$) volatilisation, nitrification and denitrification (Bolan *et al.*, 2004). 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).

In grazed pastures, animal treading damage during grazing under wet soil conditions limits pasture growth and reduces soil infiltration rates (Drewry *et al.*, 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).

Pastures are permanent or occasionally cultivated due to sward deterioration. As a consequence of the substantial N surplus and the soil N build-up, the cultivation of grazed grassland is followed by a rapid and extended period of N mineralization, which may also contribute significantly to an increased potential for losses.

The objectives of the present review are to describe the main routes of N input to farming systems with grazed pastures, to quantify environmentally harmful losses from these systems via $NO_3$ leaching, atmospheric $CH_4$ and $N_2O$ losses, and to discuss farm N budgets and N use efficiency in relation to management strategies that can be applied to reduce N losses.

## Nitrogen leaching

### *Permanent pastures*

Research on grazed systems indicates that $NO_3$ leaching increases exponentially with increased N input (Figure 1). This is mostly associated with an increase in dry matter (DM) 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 urine N compared to fertiliser N in contributing to $NO_3$ leaching (because of the much larger specific rate of N application in urine); urine typically contributes 70-90% of total N lost through leaching (Monaghan *et al.*, 2007).

Many permanent pastures include legumes, particularly white clover in temperate pastures, to supply N from clover $N_2$ fixation for long term production (Ledgard, 2001). The N concentration of pasture exceeds that required by grazing animals, and furthermore 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 urine-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. In areas where N inputs from excreta occur there will be low associated input from $N_2$ fixation.

Fertiliser N is generally used efficiently by pastures for plant growth but it enhances pasture N uptake and grass N concentrations, thereby increasing both N excretion in urine and the risk of environmental loss. Estimates of N leached from managed pastures vary widely, ranging from about 5 to 200 kg N $ha^{-1}$ $yr^{-1}$, due to many factors including differences in N input, N output in excreta, soil drainage and animal type (e.g. Monaghan *et al.*, 2007).

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

One of the options to mitigate $NO_3$ losses is to reduce the length of the grazing season. A $NO_3$ leaching model of Vellinga *et al.* (2001) was used to exemplify the effect of grazing season length on $NO_3$ losses (Figure 2). Under full grazing, i.e. 20 hours a day from 15 April to 15 October, the $NO_3$ concentration is 69 mg/l. In this experiment, the best strategy to reduce $NO_3$ losses was to shorten the end of the grazing season in the autumn. For example, $NO_3$ concentration was reduced to 59 mg/l by ending grazing on the 15 September, and the Nitrate Directive target was achieved by ending grazing around 15 August. However, some farmers especially in mainland Europe choose to start grazing their herds later in the spring/summer and do not turn out cows until after the whole grassland area has been cut for silage. Farmers follow this strategy to ensure that high quality silage can be harvested. From an environmental viewpoint this strategy is less efficient than shortening the end of the grazing season as delaying turnout from 15 February to 15 March only reduces $NO_3$ concentration to 66 mg/l.

*Pastures in crop rotation*

Dairy production systems in parts of Europe are 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). Generally, the ploughing-out phase carries the highest risk of $NO_3$ leaching as N accumulated in the soil during the ley phase is released upon cultivation. However, in every phase there are options to reduce $NO_3$ leaching.

Table 1 shows N budgets for temporary pastures as influenced by different sward types and uses, and that the N surplus is considerable for pasture grazed by dairy cows with high feed-N (Eriksen, 2001). However, in the early pasture phase $NO_3$ leaching losses are usually low as much N can be accumulated in the sward, but over time the N loss depends on the equilibrium between inputs and the soil organic N pool. This equilibrium is not reached within the first years of the clover pasture (Johnston *et al.*, 1994) and also takes longer to be reached in grass-clover compared to grass-only swards due to the self-regulatory nature of legumes. Even though $NO_3$ leaching losses in young swards are much less than indicated by the surplus of the N budgets, losses occur depending on management of the defoliation system and N input. In terms of leaching, cutting-only systems are the most advantageous

(Wachendorf *et al.*, 2004) but a management system that combines cutting and grazing is preferable to a pure grazing system. The dual advantage is in less recycling of animal excreta and a lower N surplus because of herbage removal, both leading to less $NO_3$ leaching. In a Danish experiment, grazing with spring application of cattle slurry showed $NO_3$ losses dramatically higher than other managements (Figure 3, Eriksen *et al.*, unpublished) but excluding the use of slurry under grazing management, or alternatively using combinations of cutting and grazing, reduced $NO_3$ leaching considerable to an average level of approximately 30 kg $NO_3$-N $ha^{-1}$. This was close to, and not statistically different from the leaching under cutting.

The release of N following grassland cultivation is often substantial in the first year, with N fertilizer replacement values often exceeding 100 kg N $ha^{-1}$ (Eriksen *et al.*, 2008), but with relatively little effect of both grassland management and age, even in situations with huge differences in N input during the grass phase (Eriksen, 2001; Hansen *et al.*, 2005). It has been demonstrated that mineralisation of N following grassland cultivation is a two-stage process with a rapid mineralisation over the first 160-230 days followed by a second phase with mineralisation rates 2-7 times lower than in the first phase (Vertes *et al.*, 2007). However, it is recognised that grass-rich crop rotations are more beneficial to soil fertility than arable rotations and although differences in the initial residual value between grasslands of different age or management often seem small, the accumulated effect can be considerable over the years. It is well-known that cultivation of permanent pasture has an impact on soil C and N dynamics for decades (Springob *et al.*, 2001), but it has also been shown that even grasslands of 2 to 8 years of age may affect the availability of soil N for some time (Eriksen *et al.*, 2008).

Delayed ploughing in late winter or spring can reduce $NO_3$ leaching especially on sandy soils and where rainfall occurs early in the autumn or winter (Djurhuus and Olsen, 1997). However, this must be set against possible lower yields of spring versus winter crops. It has also been shown that rotary cultivation of grass swards prior to ploughing can result in faster N availability and better synchrony between N mineralization and plant uptake (Eriksen and Jensen, 2001).

The release of large quantities of N from grass-clover residues means that fertiliser-N use on subsequent cereal crops can be reduced or even eliminated in the first following crop. Catch crops are useful during winters in the arable phase of the crop rotation to reduce $NO_3$ leaching, by removing soil mineral N from the soil profile before winter drainage occurs. An example is given in Figure 4, where two grass-clover swards were ploughed on coarse sandy soil in Denmark. Perennial ryegrass as a catch crop reduced $NO_3$ leaching by 66-88% compared to bare soil, and in the treatment with barley harvested green and followed by Italian ryegrass it was reduced by more than 90% to less than 10 kg $NO_3$-N $ha^{-1}$ $yr^{-1}$.

A key objective in designing grass-arable crop rotations is to optimise the grass phase, i.e. the number of grassland cultivations in relation to the length of the grass phase. For the individual farmer this depends on the requirement for feed and access to grazing. The common motivation for grassland cultivation is yield loss due to sward deterioration caused by e.g. compaction from wheel traffic and invasion of less productive natural grasses (Hoving and Boer, 2004), but also the maintenance or increase of soil fertility and nutrient utilisation play a role.

*Leaching of dissolved organic N*

Leaching of N from cultivated soils occurs mainly in $NO_3$ form, but leaching of dissolved organic N (DON) has recently received increasing attention. Urine patches by cows have been indicated to influence leaching of DON as a consequence of the increase in soil solution pH following urea hydrolysis leading to increased solubility of soil organic matter

(Wachendorf *et al.*, 2005; Van Kessel *et al.*, 2009). Rasmussen *et al.* (2008) reported a total N leaching of 10 kg N $ha^{-1}$ from cut grass-clover with DON accounting for 15% of the total loss, and Wachendorf *et al.* (2005) reported a total N leaching of 30 kg N $ha^{-1}$ from a grazed pasture with DON accounting for one third of total N. Nitrogen leaching in both experiments was estimated on the basis of soil solution sampled with suction cups/plates. Other studies reporting much higher DON proportion of the total dissolvable N are most often based on soil extractions (e.g. Jones and Willet, 2006) mobilizing disproportional high amounts of DON. The trend often found in studies of the contribution of DON to total N leaching is that the amount of DON lost is relatively constant, whereas changes in total N losses occur due to changes in the leaching of $NO_3$ (Rasmussen *et al.,* 2008; Vinther *et al.*, 2006).

Nitrous oxide and methane emissions

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

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 $N_2O$-N $ha^{-1}$ $yr^{-1}$ (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, losses of up to 29 kg $N_2O$-N $ha^{-1}$ $yr^{-1}$ have been measured in grass pastures in Ireland that received N fertiliser application at a rate of 390 kg N $ha^{-1}$ $yr^{-1}$ (Hyde *et al.*, 2006). Ryden (1983) reported losses of 1.3 kg N $ha^{-1}$ $yr^{-1}$ for grass pasture that did not receive N fertiliser inputs and of 11 kg N $ha^{-1}$ $yr^{-1}$ after N fertilisation at a rate of 250 kg N $ha^{-1}$ $yr^{-1}$. 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 200 kg N $ha^{-1}yr^{-1}$ (as either ammonium nitrate or urea), at 6 kg N $ha^{-1}$ $yr^{-1}$ without N fertiliser and 15 and 13 kg N $ha^{-1}$ $yr^{-1}$, respectively, for the two N fertilisers. Similar denitrification losses have been reported by other workers on clover/grass pastures in New Zealand. For example, Ruz-Jerez *et al.* (1994) reported annual losses of 3.4 kg N $ha^{-1}$ from grass/clover swards compared to 19.3 kg N $ha^{-1}$ after application of 400 kg $ha^{-1}$ $yr^{-1}$ of N fertiliser and Ledgard *et al.* (1999) reported total denitrification losses of 3-7 kg N $ha^{-1}$ $yr^{-1}$ without added N compared to 10-25 kg N $ha^{-1}yr^{-1}$ after application of 200 kg N $ha^{-1}yr^{-1}$ as N fertiliser. Much higher denitrification losses were measured from N-fertilised pasture in Northern Ireland at up to 154 kg N $ha^{-1}$ $yr^{-1}$ after application of 500 kg N $ha^{-1}yr^{-1}$ as N fertiliser (Watson *et al.*, 1992).

Methane emission from animal excreta occurs under strictly anaerobic conditions in slurry stores and is expected to be neglectable during grazing. Thus, by lowering the quantity of slurry stored especially during summer when emissions are highest (Sommer *et al.*, 2009) grazing systems may considerably reduce methane emissions compared to indoor housing of cows. Methane 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 DM of intake (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 urine-N with high risk of N loss.

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 the quantity 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, 2000b). 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. 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 N leaching 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 ha and per kg milk, mainly due to the much lower use of fertiliser-N.

Importance of managing whole systems

Intensively grazed pasture systems markedly increase the risk of N losses to waterways and the atmosphere. Traditionally, N losses are expected to increase with the age of the sward due to the likely loss of excess N when maximum accumulation has been reached. This picture of increasing N losses with sward age and comparatively huge losses following cultivation may well describe the situation on many farms, but it is not an inherent property of grassland and grass-arable rotations. Good pasture management (e.g. reduced fertiliser input and reduced length of grazing) and in the mixed crop rotation both during the grassland and the arable phase (e.g. delayed ploughing time and a catch crop strategy) can considerably reduce negative environmental impact of grazing.

Although there are environmental consequences of grazing, pollution swapping can occur with e.g. housing and cutting only systems due to transport and storage of feed – all of which require energy often from non-renewable sources. Therefore, when evaluating environmental impact it is necessary to examine the whole farming system. In particular for green house gasses since pasture may serve as a source of $N_2O$ and indirectly of $CH_3$, but also as a sink of $CO_2$ influenced by management practices on the farm. Furthermore, indirect emissions in other ecosystems from losses of N via $NO_3$ leaching and $NH_3$ volatilisation should be accounted for.

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Table 1. Annual N budget of six pasture management systems (kg N $ha^{-1}$ $yr^{-1}$). Data are mean of production years 1-3 (Eriksen, 2001).

| | Ryegrass-only | | | Clover/ryegrass | | |
|---|---|---|---|---|---|---|
| | cut | grazed low N[1] | grazed high N[1] | cut | grazed low N[1] | grazed high N[1] |
| N-Input | | | | | | |
| Fertilizer | 300 | 300 | 300 | 0 | 0 | 0 |
| $N_2$-fixation | 0 | 0 | 0 | 300 | 258 | 266 |
| Animal manure | 0 | 222 | 320 | 0 | 240 | 326 |
| N-Output | | | | | | |
| Herbage yield | 287 | 240 | 292 | 288 | 271 | 342 |
| Balance | 13 | 282 | 328 | 12 | 227 | 250 |

[1] Grazed low and high N refers to grazing by dairy cow with 140 and 310 g N $d^{-1}$ in supplements, respectively.

Table 2. N inputs and outputs from intensive dairy farm systems in NZ receiving N fertiliser at 0 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 input – N output in product) | | | |
| N use efficiency (product-N/input-N) | 46% | 23% | 22-36% |

Table 3. Characteristics of fertiliser-N and clover-N dairy farm systems in the study undertaken by Schils *et al.* (2000 a,b).

| | Fertiliser-N | Clover-N |
|---|---|---|
| Cows (number) | 59 | 59 |
| Area (ha) | 34 | 41 |
| Milk production (kg FPCM[a] $ha^{-1}$) | 13884 | 12053 |
| Fertiliser N (kg N $ha^{-1}yr^{-1}$) | 208 | 17 |
| Manure effective N (kg N $ha^{-1}yr^{-1}$) | 67 | 52 |
| Clover fixed N (kg N $ha^{-1}yr^{-1}$) | 0 | 176 |
| Grazing system (hours $day^{-1}$) | 24 | 24 |
| | | |
| N input (kg N $ha^{-1}yr^{-1}$) | 333 | 279 |
| N output (kg N $ha^{-1}yr^{-1}$) | 80 | 69 |
| N surplus (kg N $ha^{-1}yr^{-1}$) | 253 | 212 |
| Nitrate leaching (kg N $ha^{-1}yr^{-1}$) | 20 | 22 |
| Nitrous oxide (kg N $ha^{-1}yr^{-1}$) | 9.4 | 6.6 |
| GHG total (kg $CO_2$-equiv $ha^{-1}$) | 16065 | 12198 |
| Nitrate leaching (kg per 1000 kg milk) | 1.4 | 1.8 |
| Nitrous oxide (kg per 1000 kg milk) | 0.7 | 0.5 |
| GHG total (kg $CO_2$-equiv per kg milk) | 1.2 | 1.0 |

a) Fat and protein corrected milk

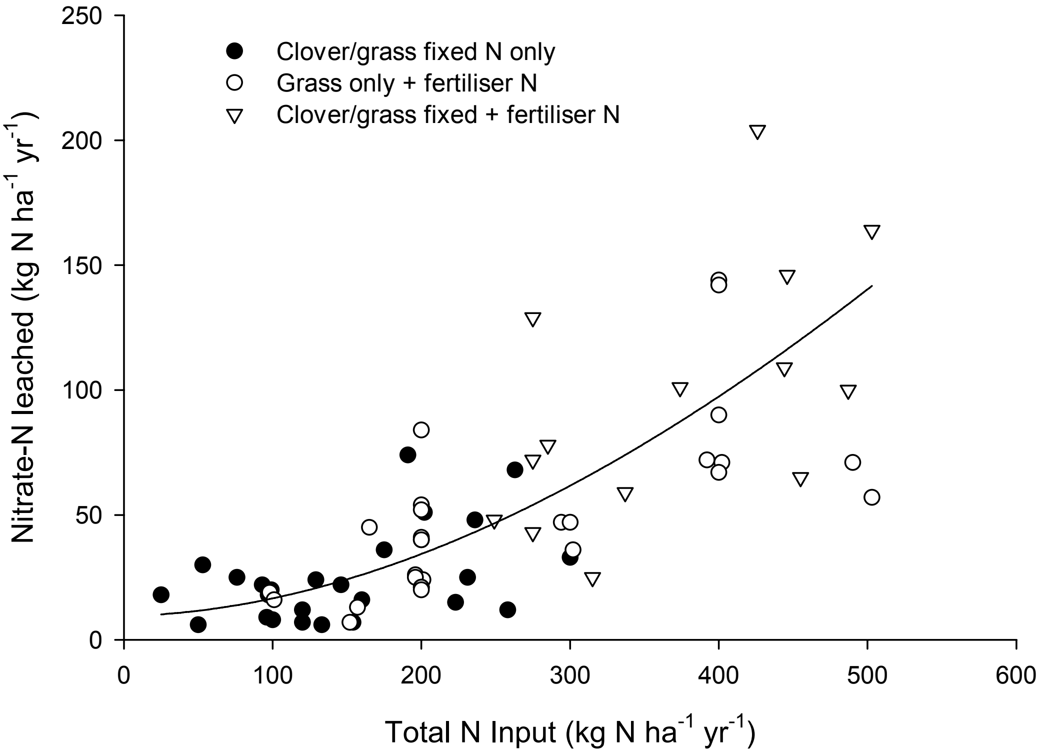


Figure 1. Nitrate leaching from grazed pasture systems as affected by total N input. Data are a summary of studies from NZ, France, UK and Denmark. The line of best fit is an exponential function obtained by fitting the data on the log scale. Updated from Ledgard (2001).

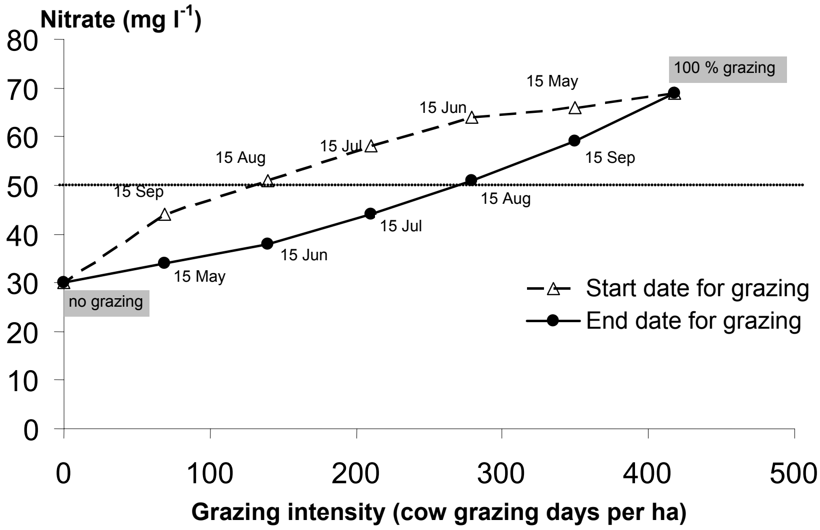


Figure 2. The effect of the length of the grazing season on $NO_3$ leaching. The upper line represents the starting dates of the grazing season, while the lower line represents the end dates of the grazing season. The modeled dairy farm has 80 cows, with a milk production of 8000 kg $cow^{-1}$, on 35 ha of grassland on sandy soil, fertilised with 250 kg N $ha^{-1}$.

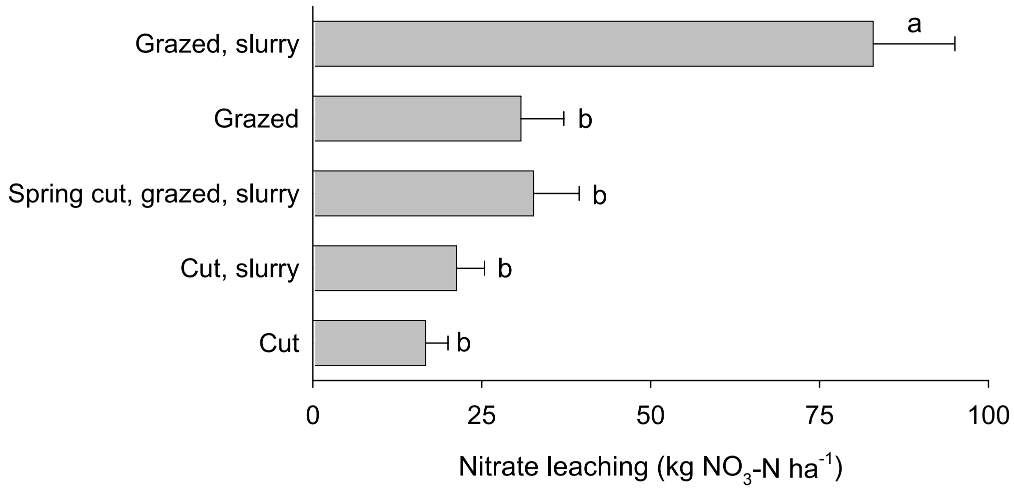


Figure 3. Nitrate leaching from grass-clover pastures with different management. In grazed grass-clover slurry was injected in the spring (100 kg total-N $ha^{-1}$ cattle slurry), in cut-only grass-clover an additional injection of slurry was made following a spring harvest of herbage. Bars with the same letter indicate no significant difference ($P$>0.05)

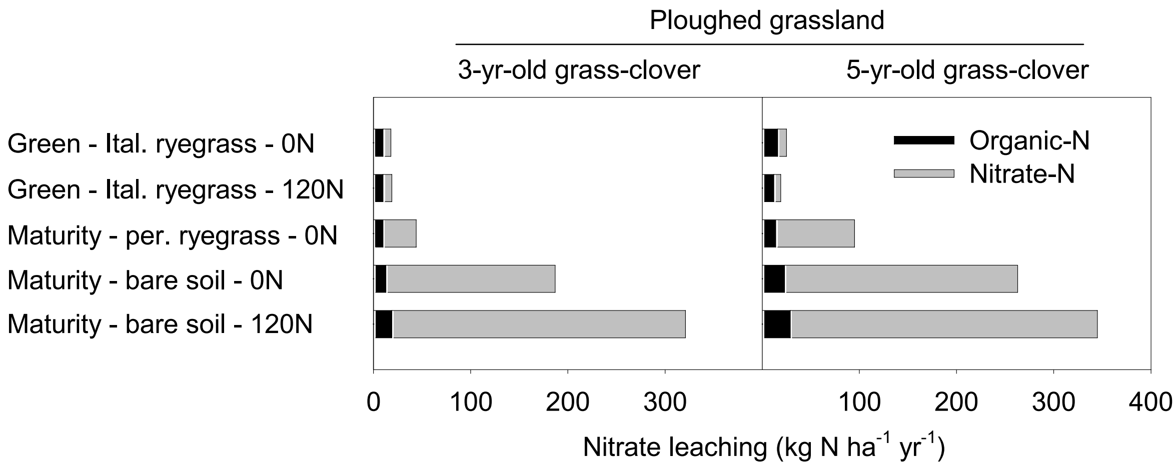


Figure 4. Leaching of $NO_3$ and dissolved organic nitrogen after spring cultivation of 3 or 5 year old grassland followed by 1) barley harvested green with Italian ryegrass undersown, 2) barley harvested at maturity with perennial ryegrass undersown, and 3) barley harvested at maturity without catch crop. From Hansen *et al.* (2007).