Environmental impacts of nitrogen in pastoral agriculture

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Abstract

The principal environmental impacts of nitrogen in pastoral agriculture are identified as: ammonia volatilisation, nitrous oxide emission, reduction of methane oxidation, and contamination of waters with organic nitrogen or inorganic nitrogen (nitrate, nitrite and ammonium). Each of these impacts is analysed in terms of its sensitivity to the form in which N enters the farming system, symbiotically or as fertiliser. Indirect effects that flow through from any changes in productivity are also examined. With the exception of organic N pollution of waters, all the impacts are shown to be directly affected by fertiliser N.

Keywords: ammonia, leaching, methane, nitrate, nitrous oxide

Introduction

Nitrogen has a large effect on productivity of many ecosystems. It is also an element that exists in nature in a wide range of forms which vary in their toxicity and their ability to be transported across system boundaries. This paper considers the significance of N entering pastoral farming systems for production purposes, in terms of its escape from farming into other environments, and the impact which it has on them. A central part of this discussion relates to the way N enters the system, whether by symbiotic fixation, or as fertiliser. Direct effects of mode of entry are separated from the indirect effects that flow through from production changes.

Grassland was , for a long while, considered to be an efficient system which recovered most of the fertiliser N applied . This view was based on measurements of cut swards where 80% of applied N may be removed, and which contrast strongly with grazed swards where export of added N is typically less than 15–20 % (Ball 1979; Ball & Ryden 1984; Ryden et al. 1984). For 50 years it had been known that animal excreta, especially urine, had profound effects on the distribution of available N in the field (Thompson & Coup 1940), plant composition and productivity (Sears & Evans 1953) and some loss processes (Doak 1952). It was not until the 1970–80 however, that the larger view emerged, and animals and the amount of N being turned over in systems, were related to loss processes and N impacts on the environment. The 1982 publication ‘Nitrogen balances in New Zealand ecosystems’ (Gandar 1982) gives a comprehensive account of a range of grassland farming systems in terms of N inputs, turnover and environmental impacts and clearly identifies the importance of grazing animals in these processes

As a starting point for this discussion the forms of N that must be considered need to be identified. Much of the public concern is focused on nitrate-N (NO_3-N) in water supplies, because of its perceived toxicity to humans and strong association with agricultural activities in Europe and North America in particular. A much broader range of issues needs to be considered however.

Nitrogen exists in soil and excreta in organic forms, which can be moved across boundaries in the mass flow of water, as in surface runoff, or flow in drains which receive direct inputs of, for instance, manure. Some movement of soluble organic-N may also occur but this is likely to be a small proportion of the total. Nitrogen in organic forms is the substrate that is mineralised to form ammonium ions (NH_4^+) which, in turn, can be nitrified to nitrite ions (NO_2^-) and nitrate ions (NO_3^-). Thus organic-N can have a flow through effect on all the impacts associated with those forms as well as the direct effects of organic matter itself on water quality.

Nitrogen can move across ecosystem boundaries to the atmosphere as ammonia (NH_3), as nitrous oxide (N_2O) and as molecular nitrogen (N_2). Both NH_3 and N_2O have powerful impacts within the atmosphere, N_2 as the dominant component of the air is thus impact free. The impact of NH_3 is through the complex interactions of acid rain formation and nitrogen deposition in rainfall whereas N_2O has its impact in the stratosphere as a persistent greenhouse gas, and in the troposphere as an ozone destroying agent. A further atmospheric impact of nitrogen cycling, which is far less obvious, is on methane (CH_4) emission from agricultural systems. This impact arises through the activities of CH_4 oxidising bacteria in the soil which strip CH_4 from the air. The rate at which this proceeds is strongly affected by N cycle processes and fertiliser input in particular.

Acidification of soils is a process which impacts on the production base and is driven, at least in part, by N cycle processes. While this impact is an on site effect it will be considered here because it can be cumulative or discontinuous and thus affect people other than the current land user. In the same way changes to the quantity or quality of soil organic matter can be considered as environmental impacts as well as on site changes to the productive base.

Each of these impacts can be examined by asking two questions:

1. Is there any direct effect of N source on the impact?
2. Are there indirect effects from production or associated managements?

Organic Nitrogen in water

This is perhaps the simplest issue as the N form is itself a product of the system. As N inputs and production rise the amount excreta N rises and hence the quantity at risk of loss rises, and will be greatest where animals are yarded regularly. Soil erosion provides another source which may be sensitive to stock densities.

Nitrate in water

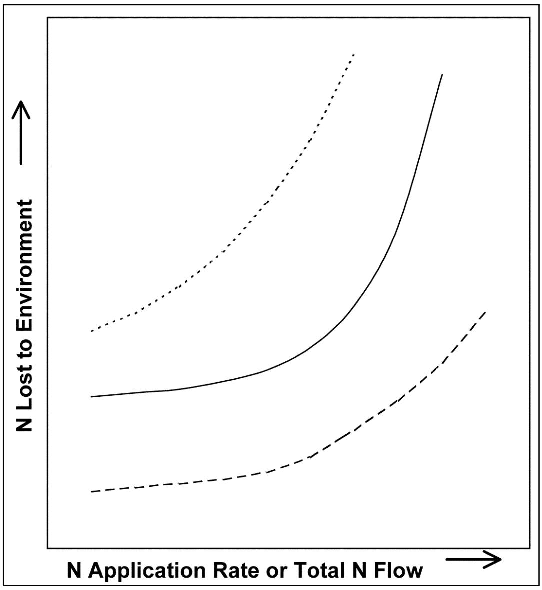
This is a far more difficult and contentious issue as it impinges on human health and has a very high profile as a developed world problem. For NO_3^- leaching to be a problem there must be prior accumulation of, or concurrent formation of, NO_3^- in sufficient quantity to enrich percolating water. Where the source of N is symbiotic fixation there is no direct effect on these processes and hence no coupled input-output relationship. Fertiliser N use does, however, offer that possibility. Nitrate may be added in fertiliser or accumulate as NH_4^+ yielding products are transformed and leaching can occur as a direct consequence. Examples may be helpful to test the scale of this direct effect in relation to the whole system effect.

Rotationally grazed sheep example (Ruz Jerez 1991)

A system receiving 400 kg N ha^{-1} applied as Urea in 8 dressings (400N), and another system wholly dependent on symbiotic fixation (GC) were compared. The 400N system maintained higher NO_3^- -N levels in the soil, about 50 mg N per kg soil, than did the GC system, about 5 mg N per kg, and therefore presented increased opportunities for leaching. Yield increases also meant more grazing days (42%) and N circulating (65%) in the 400N system. Ruz Jerez estimated NO_3^- -N leaching as 41 kg N ha^{-1} and 6 kg N ha^{-1} in the 400N and GC treatments respectively. The proportion leached from urine patches was calculated as about 25% and 50% respectively indicating a substantial direct fertiliser effect as well as an effect through production. In terms of N leached per grazing-day the GC system was more efficient at 0.6 g N compared with 3.1 g N leached per grazing-day.

A study of this sort, with a single N rate, does not ask whether equal production was attainable with less N fertiliser and with a lower environmental cost. It is unlikely, however, that the N input-yield relationship is optimal where available N levels are elevated for much of the year. Strongly curvilinear relationships between N input rate and N escaping the system are common (Figure 1). Excessive N rates will tend to lead to high direct impacts on the environment and the task of defining the point where excess starts remains a major problem. Soil type exerts a major effect and similar applications can result in quite different risks of loss on different soils.

Figure 1: The generalised relationship between N fertiliser input, or N turnover within a system, and the loss of N from the system to the environment. The upper and lower dashed lines indicate the range of relationships that may occur.



Similar arguments apply to the other ionic forms of N which may enter water. The significance of NH^+_4 and NO_2^- runoff and leaching is far less extensively studied than is NO_3^- and soil type and environmental factors are likely to affect these processes to a great extent. Both forms of N have higher toxicity to a wide range of organisms than does NO_3^- and are associated with transformation of urea (Burns et al. 1995).

Dairy f*arming* e*xample (Ledgard et al. 1996)*

A long-term N grazing trial is currently in progress at the Dairy Research Corporation (DRC) in the Waikato, to examine the effects of increased N rates on dairy production. This has also provided an opportunity to study potential environmental impacts of fertiliser N (200 or 400 kg N ha^{-1} yr^{-1}) versus symbiotically fixed N.

In year 1 of the trial gaseous losses of N (NH_3 and N_2O) increased with increasing fertiliser application.

Denitrification losses were 6, 11 and 15 kg N ha^{-1} yr^{-1} for the control, 200N and 400N treatments respectively while the ammonia losses were 15, 45 and 63 kg N ha^{-1} yr^{-1} respectively. However, the major loss pathway of N from the grazed pastures was leaching.

The net environmental impacts of fertiliser N use in pasture may take time to emerge due to grazed pastures having to reach new equilibria and the need for long-term monitoring to allow for seasonal variations. For example under the 400N treatment leaching losses as nitrate in years 1 and 2 were 15 and 204 kg N ha^{-1} yr^{-1} while under the unfertilised pasture leaching losses in years 1 and 2 were 12 and 74 kg N ha^{-1} yr^{-1} respectively. Losses were low in the first year under the 400N treatment due to the system becoming established and it was a relatively dry winter. In the second year it was an extremely wet winter which may have exacerbated leaching losses. Hence the need for long-term monitoring to study environmental impacts.

Increased leaching losses of nitrate may be a result of increased cycling and subsequent loss of excreta N or direct leaching of fertiliser N. In this grazing trial, to date, increased leaching losses appear to be due to increased cycling and excretal return. Associated ungrazed lysimeters also receiving 400 kg N ha^{-1} yr^{-1} showed direct leaching loss of applied fertiliser to be negligible. This contrasts with the results of Ruz-Jerez discussed earlier.

Nitrous Oxide emission

The scientific literature records many examples of sharp peaks of N_2O emission following the application of N fertiliser. Reasons for this "fertiliser effect" are not known but it does appear to be general (Jarvis et al. 1991) and a direct impact associated with N fertiliser needs to be recognised. Daniel et al. (1980) proposed that some *Rhizobium* species could denitrify and thus produce N_2O while living in the soil. This was initially seen as a potential environmental hazard associated with biological N fixation, but demonstration of it's importance is still lacking.

Indirect effects driven by production are harder to characterise. Small plot experimentation suggests that urine N has a major effect on emission, similar to the fertiliser effect (Müller 1995). Paddock scale measurement in a range of farming systems, however, provides no evidence that N_2O emissions vary with the amount of N cycling through the system (Carran et al. 1995) although soil structural damage may be important and related to production through stock density at grazing.

Ammonia volatilisation

This issue is well researched; application of urea can result in a direct loss of NH_3 from the fertiliser at rates which vary with soil (Selvarajah et al. 1989) and weather conditions and application rate. Similarly urea from urine suffers the same sort of loss and strong rate dependence has been shown. Jarvis et al. (1989) compared a grass-clover (GC) system with systems receiving 210 and 420 kg N ha^{-1} $year^{-1}$, and showed that during a 7-day grazing 23, 27 and 64 kg N of N were excreted respectively with the difference being dominated by urinary-N. Small amounts of NH_3 were volatilised from the GC and 210 N systems (7 and 10 kg N ha^{-1} $year^{-1}$) while substantially more volatilised from the 420 N system (25 kg N ha^{-1} $year^{-1}$). Application rate of N fertiliser thus emerges again as a key determinant of impact.

Methane oxidation

Methane (CH_4)is oxidised in to CO_2 soils by the same groups of micro-organisms that oxidise NH_4^+ to NO_3^-, in this way a bad greenhouse gas (CH_4) is converted to a les harmful one (CO_2). Active CH_4 oxidation is a positive environmental attribute for a farming system to have. There is clear evidence that use of NH_4^+ yielding fertilisers suppresses methane oxidation (Mosier et al. 1991). Increasing the amount of N cycling, by whatever means will similarly affect the process.

Soil acidification

Both N fixation and use of NH_4^+ yielding fertiliser are acidifying processes. Evaluating the actual short- and long-term impacts of these in systems is complex and beyond the scope of this paper. The generalisation can be made, however, that where any strategy that results in extra N entering the system is used, extra care in monitoring soil acidity is warranted.

Discussion

The environmental impacts of N in pastoral agriculture are influenced by source of N. Fertiliser N use carries with it a risk of direct contamination of waters and emission of NH_3 and N_2O to the atmosphere. The size of this risk is, however, affected by the quantities of N fertiliser applied and the relationship is non-linear. Across a range of rates there is little direct effect but beyond a threshold value, both direct and indirect effects increase. Figure 1 shows this as a generalised set of relationships, Soil type, season, management and possi-bly some unidentified factors all appear to modify the relationship between fertiliser rate and loss to a high degree and make it unpredictable. An individual site may therefore, show curves of different characteristics from year to year.

Direct impacts from symbiotic N fixation are less obvious. The crucial question is however, whether fertiliser N and symbiotic systems differ in their impact at similar levels of production or N turnover within the system. Good examples of such comparisons are hard to find even if set up experimentally since N inputs are timed in fertiliser systems and are not a function of a continuous system such as fixation of N. Increased N input through symbiotic fixation may, for instance, occur mainly in summer and have associated large losses from NH_3 volatilisation. If fertiliser N is used to spread production in autumn or spring, impacts on N_2O emission or NO_{3-} leaching may be expected. Ultimately comparison of the environmental impacts will need to be made on a farming system basis rather than a strict N source or quantity basis. Total N turnover through animals or the soil mineral N pool may provide the most satisfactory basis for comparison. However, from an environmental loading point of view the severity of the impacts alone may be the only issue to be considered and associated increased productivity or economic activity not deemed an issue. It must be concluded that there is no absolute basis for comparison that will suit all interested sectors of the community.

In this paper we have not considered some of the wider environmental impacts of nitrogen in pastoral farming. Energy issues have been dealt with by Walker (1996) and while some updating of costs may be necessary the underlying analysis remains valid. The offshore impacts of the phosphate mining that has supported legume based agriculture in this country, is noted here but not analysed.

Symbiotic N fixation is less likely to have a direct influence on environmental impacts but similar indirect effects may occur due to subsequent cycling of N through the grazing animal and resulting excretal returns. However, these effects may be of a lesser magnitude than those systems receiving high rates of fertliser N (e.g. 400 kg N ha^{-1} yr^{-1}). Strict comparisons between forms of N input are inherently difficult and issues surrounding this are discussed.

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