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Impacts of management on leaching of nitrate from pastures

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Abstract

Clover-based grasslands as used in New Zealand, while considered "clean" relative to grasslands farmed intensively with fertiliser nitrogen (N), are nevertheless polluting. Recent measurements of nitrate-N down the soil profile below heavily stocked (22.5 ee/ha) pastures grazed by sheep, have shown that leaching losses under set stocking are 2-3 times those from rotational grazing (37 vs 16 kg N/ha/yr). As the input of N, its storage, and outgoings in animal products are similar in both systems, this indicates that considerably more urinary N is being lost to the aerial environment from rotationally grazed pastures, with nitrate leaching being the main avenue of loss under set stocking. Differences in sward structure are suggested as the major factor. Lower nitrate leaching from cocksfoot vs ryegrass-based pastures are thought to be associated with lower herbage quality factors reducing urinary N formation. While grazing management will not greatly influence total N losses to the environment, it can influence the avenues by which N enters the wider environment. Aquatically sensitive catchments may be better protected from leaching of nitrate by modification of the grazing management approaches.

Keywords pasture management, nitrate leaching, ammonia volatilisation, pasture structure, herbage quality, groundwater pollution

Introduction

There is general demand for better accountability of the environmental impacts of agricultural practices. Pollution affects the quality of life for people world wide, and water quality is becoming of major concern in many countries as population growth and improved living standards place greater demands on potable water. Nitrate pollution has been identified as one problem. Nitrate moves with the ground water into deeper aquifers, as a result of years of intensive farming, especially where fertiliser nitrogen (N) has been used. Clover-based pastures, as used in New Zealand for animal production, are seen as environmentally "clean", although they still have relatively large inherent N losses due to the grazing animal, and are therefore polluting (Field & Ball 1982), if to a lesser degree.

Pathways of N losses

Nitrogen losses from grazed pastures are driven primarily by the grazing animal. Most of the N ingested during grazing is excreted on to small areas in urine patches, at concentrations approaching the annual requirements of pasture (30-60 g N/m,), far in excess of the sward's capacity for uptake in the short term. The major part of this N is open to loss through two main pathways. First, hydrolysis of urea to ammonia occurs rapidly, and under appropriate environmental conditions (summer-autumn) some of the ammonia formed is readily volatilised into the atmosphere. There is evidence that some of this ammonia can be re-absorbed by herbage, thereby reducing losses (Denmead *et al.* 1976). Second, after nitrification of ammonuim to nitrate, any excess nitrate in the urine patch is open to leaching below the root uptake zone. Thereafter, nitrate moves into the ground water when drainage occurs (winter-). Both pathways operate in pastures, with the emphasis changing seasonally.

Can N losses by manipulated?

There appears to be little scope to change the total amount of N lost short of changing stocking rate (Field & Ball 1982), but the seasonality of the two main pathways of loss would suggest that manipulation of pasture management may be able to influence the balance between them.

To this end, the nitrate-N profile below a long term grazing management experiment was studied to see what influence differences in defoliation patterns may have had on the quantity of nitrate escaping in drainage beneath the pastures.

Experimental design and sampling

The experiment involved small, self-contained farmlets to compare 3 systems of grazing: (1) rotationally grazed (RG) all year with 10-12 grazings/year at rotations lengths varying from 24 days in spring to 60+ in winter; (2) set stocked (SS) all year; and (3) a combination (CC) of rotational grazing, with a period of set stocking from lambing in late August to weaning in early January; and 2 pasture types: (1) ryegrass-based plus clover, and (2) ryegrass and cocksfoot base plus clover. The common stocking rate was 22.5 ee/ha. During the lo-year duration of this experiment (sown and established in 1978-79), many aspects of sward performance (production, structure and symbiotic N fixation) and changes in soil chemical properties (total N, C, P, organic matter, soil microbial biomass N and C, mineral N) were monitored.

In spring 1989 (26-27 October) the soil beneath each pasture was measured, using a Giddings hydraulic drill to extract 68 mm diameter cores to 2.1 m depth. Each core was divided into 300-mm sections and the top section divided again into two 150-mm sections, giving 8 sections in total. Each section was subsampled and deep frozen for later analysis for nitrate-N, which was extracted by shaking sub-samples for 1 hour in 2N KCl. Two cores were taken from each RG pasture, and one from each SS and CG pasture. There were 4 replications, making a total of 32 cores.

Management effects

Drainage down the soil profile, which leaches nitrate-N, occurs in specific events whenever rainfall significantly exceeds evapotranspiration. Rainfall records for the 2 years before sampling indicate that drainage would have occurred during the October of sampling (35 mm), and would have moved the wetting front to a depth of 200-250 mm, and also in winter (May to July) when some 180 mm of drainage would have moved nitrate accumulated in the topsoil over summer-autumn to a depth of 1600-1800 mm. These events are evident in profiles in Figures 1 and 2b, the first causing the high value in the 150-300 mm section, which is still in the zone of active uptake, and the second in the high values between 900 and 1800 mm, particularly in the RG and SS pastures (Figure 1). Most of the profile sampled represents drainage for one year only, as the heavy drainage (480 mm) of the previous year (May to September) would have effectively removed the previous year's nitrate from the profile measured.

Figure 1 clearly shows that the nitrate-N concentrations in the soil solution under the pastures to 2.1 m was 2.5 times higher for SS than RG and CG ($P<0.001$). Calculations of the annual loss of nitrate below the uptake zone (300 to 1600-1800 mm) were of a similar pattern (SS = 37, RG = 17, CG = 16 kg N/ha/yr), although the values are considerably less than the 60-80 kg N/ha/yr reported by Field *et al.* (1985) for a rotationally sheep grazed pasture on an adjacent site.

As the same stocking rate was used on all treatments, equal quantities of herbage utilisation (11.5 t DM/ha/yr) and return of N in excreta (460 kg/ha less 40 kg N/ha lost in animal products = 420 kg N/ha/yr) would occur. Nitrogen fixation measurements showed similar inputs (RG = 88; SS = 70; CG = 72 kg N/ha/yr), as did changes in soil total N over the 10 years (RG +0.042%; SS + 0.041%; CG + 0.044% N). Consideration of the partial mass balance (Ball 1982) would indicate that total N losses woul also be similar, in which case the losses of ammonia to the atmosphere must have been higher in RG and CG to offset the greater leaching under SS.

Differences in pasture structures may offer an explanation (Table 1). With the rapid removal of the herbage canopy, the high proportion of exposed bare soil (Hay *et al.* 1989) developed higher soil temperatures in the RG system, providing ideal conditions for ammonia volatilisation. High rates of ammonia volatilisation have been measured in warm dry summer-autumn conditions under rotational grazing at Palmerston North (Ball & Ryden 1984), which suggest there would be a large proportion of N losses under RG from this source. By contrast, the substantial cover provided by the high density greater biomass SS pasture, could act as an absorbing surface for ammonia, and reduce maximum soil surface temperatures (by 6 °C, Table 1). Mitchell (1957) also reported a 15 °C differential between open RG swards and short dense swards. It seems that SS swards reduced ammonia volatilisation, retaining a higher proportion of the urinary N in the soil-plant system, and that the excess not absorbed by the

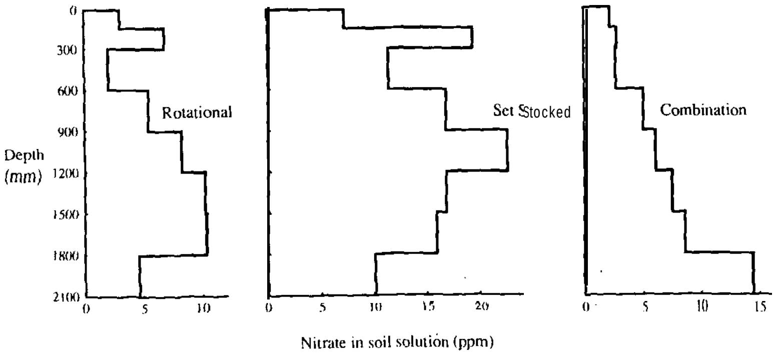

Figure 1 Soil nitrate profile below pastures grazed by sheep either rotationally, set stocked, or a combination of both.

Table 1 Differences in mean annual pasture characteristics and soil surface temperatures in autumn, induced by defoliation management.

	Rotational grazing	Set stocking	Combination grazing
Residual grass biomass (kg/ha)	1885c*	3520a	2680b
Residual clover biomass (kg/ha)	460a	260c	330b
Ryegrass tiller density (m²)	53ooc	10940a	8270b
Clover growing points (m²)	3170a	1830b	2310b
Bare ground (%)	19a	1c	6b
Peak soil surface temperatures (1 March) °C	48	42	-

* Values followed by different letters are significantly different at the $P<0.05$ level (applies to Table 2).

Table 2 Effect of grazing management on mean soil properties.

	Rotational grazing	Set stocking	Combination grazing
Microbial biomass N (ppm dry soil), O-7.5 cm	97.2b	110.6a	111.1a
Microbial biomass C(μg/g dry soil), O-7.5 cm	747b	843a	899a
Total soil organic matter (%), O-15 cm	7.24c	7.68b	8.07a
Soil organic matter light fraction (%), O-7.5 cm	0.131c	0.169a	0.146b

plants was converted to nitrate, particularly in summer-autumn when plant growth may be restricted by water shortage, and was lost in subsequent drainage events.

Mean populations of soil organisms (Carran 1983) and soil organic matter content (Table 2) were higher in SS and CG, suggesting a greater potential to immobilise N than in RG. However the relative effect of these factors in the high N loading of the urine patch is not known, but would appear to be small and have little relation to mineral-N losses. The importance of these factors needs further detailed assessment.

The nature of the nitrate profile for CG would indicate that ammonia volatilisation from the predominantly rotational grazing period of the system was the major influence. Any effect of set stocking in spring was of little consequence, probably because most of any nitrate formed was taken up during this period of rapid growth, and that drainage events did not occur during the previous spring. During the summer-autumn period when ammonia volatilisation is at its peak, the CG management is under rotational grazing. Most of the losses for any management system occur as a result of accumulation of summer-autumn surpluses of nitrate, with management determining only the extent of loss (Ball *et al.* 1985).

Pasture species

Under SS, cocksfoot did not persist and the pasture became ryegrass dominant within the first year. As a result the nitrate for both pasture types was similar (Figure 2a). Under rotational grazing, cocksfoot maintained a strong contribution to production (RG 57%, CG 34%) and the nitrate loss was approximately half that of ryegrass-based pastures (cocksfoot 13 kg N/ha, ryegrass 27 kg N/ha) ($P<0.01$).

Several factors could be operating involving the quality of organic matter being produced from root

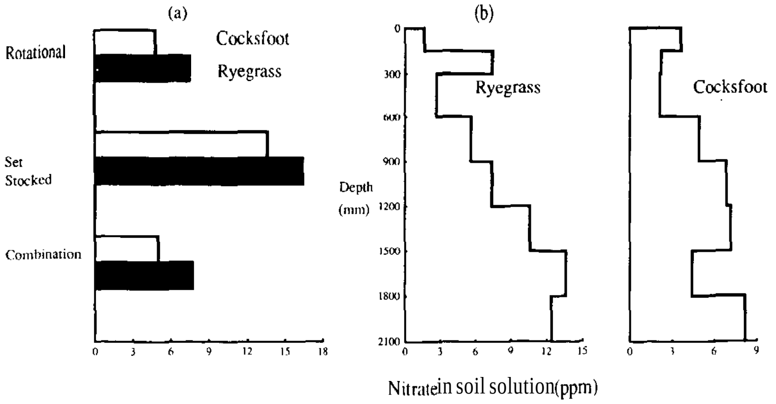

Figure 2 The effect of pasture species mixtures on soil nitrate leaching: (a) for all grazing managements (b) on the nitrate profile for the rotationally grazed managements (RG and CG).

and shoot residues, and the quantity of urinary-N cycling. Cocksfoot root material has been shown to be of lower quality (low N, high CN rates) than ryegrass (Whitehead 1970), and if the same is true of shoot residues, then the immobilisation potential of the organic matter produced would be higher (and mineralisation potential lower) under cocksfoot pastures than ryegrass. This is supported in principle by the small but consistently lower soil nitrate levels found in the top 75 mm of soil under cocksfoot pastures over several years (5.6 cf 4.2 ppm for ryegrass and cocksfoot respectively, $P<0.001$), similar to the difference shown in the top 300 mm of the leaching profile (Figure 2b). However, further work is required on these quality aspects

Implications

The implications of this study are that while the form of grazing management may not greatly influence total losses of N to the environment, it can influence the avenue by which N enters the wider environment. Critical aquatic environments where water quality is of prime importance, may be better protected by modification of grazing management towards rotational grazing, and the use of pastures based on grass species other than ryegrass. This provides another management option, along with reduced stocking rates (Field & Ball 1982), to reduce leaching of nitrate N into ground water.

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