EFFECT OF THE STOCKING RATE AND LAND SLOPE ON NITROGEN LOSSES TO WATER ON A GRAZED PASTURE OF SOUTHERN CHILE

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ABSTRACT

Most of the studies on nitrogen (N) leaching have been carried out on cut grass, and there is a lack of information on beef grazed grasslands. The objective of this study was to quantify N runoff and leaching losses in beef production systems with two different immediate stocking rates (63 and 191 Holstein Friesian steers ha^{-1} day^{-1}) and two land slopes (4 and 12%). Runoff and leachate samples were analyzed from 2004 to 2006 for total N, nitrate and ammonium. No significant differences for the total N losses were found between treatments (p > 0.05), which were low ranging from 0.9 to 26.8 kg N ha^{-1} yr^{-1}. The main pathway for the losses was leaching, which contributed >99% of the total N lost. The main form of N leaching was nitrate-N (>84%). Nitrate-N concentration in runoff samples was high, averaging 14 to 31 mg L^{-1}. We suggest that these low N losses could be related the low N fertiliser inputs in the pasture and to the N adsorption properties of volcanic soils and, so that further research is required on this subject.

Keywords: grazing, beef production, nitrogen leaching, runoff, volcanic soils.

INTRODUCTION

Cattle production is an important economic activity in southern Chile (39º to 44º S; 71º to 73º W), which is based principally on natural and improved pastures. Most of the cattle herd is concentrated in this area, which produces 70% of the country's milk and 50% of its meat (ODEPA, 2006, 2007; Anrique, 1999). In recent years, cattle production has intensified, through increasing stocking rates (from 0.42 AU ha^{-1} up to 1 or more AU ha^{-1}) (Smith *et al.*, 2002) and use of purchased fertilisers and concentrates. Nowadays, Chile is an exporting country for milk and meat production, which is expected to increase in the future.

Globally, there is an increasing concern about the impact of farm intensification on the environment, which in European countries has resulted in specific legislation, as well as guidelines for good agricultural practices (EC, 1991; Ignazi, 1996; Archer and Marks, 1997; Neeteson, 2000).

Farming is recognised as a major source of nitrate (NO_3^-) in ground and surface waters worldwide (e.g. European Environment Agency, 1995; Powlson, 2000). It has been estimated that agriculture contributes between 37 and 82% of the nitrogen (N) input into surface waters of Western Europe (waters of Western Europe (Isermann, 1990). Within agriculture, cattle production systems are considered one of the most important contributors due to their low efficiency of N use (Isermann, 1990).

In cattle production systems one of the most important pathways of N losses is leaching (e.g. Alfaro *et al.*, 2008), with leaching losses being greater from grazed pasture than from a cut sward, and likely to equal or exceed the range observed in arable production systems (Ryden *et al.*, 1984).

In New Zealand on grazed pastures with milking cows and no N fertiliser application, Ledgard *et al.* (1999) showed N losses ranging from 20 up to 74 kg N ha^{-1} yr^{-1}. Betteridge *et al.* (2004) measured N leaching from pasture grazed by beef cattle at 12 to 17 kg NO_3^--N ha^{-1} yr^{-1}. In Chile most studies have been done on cut grass, using lysimeters (Nissen and Daroch, 1991; Salazar, 2002) with only few studies published evaluating N losses due to leaching and/or runoff (Alfaro *et al.*, 2005; Alfaro *et al.*, 2008; Núñez *et al.*, 2010). The latter study reported N losses of 33 and 59 kg N ha^{-1} yr^{-1}. The objective of this study was to quantify N losses in runoff and leaching in beef production systems with two immediate stocking rates and two land slopes on a volcanic soil.

MATERIALS AND METHODS

The experiment was carried out between 2004 and 2006 at the National Institute for Agricultural Research (INIA), Remehue Research Centre (40º35'S, 73º12'W). The soil is an Andisol of the Osorno soil series (Typic Hapludands; CIREN, 2003), which at the experimental site has more than 1 m depth, loamy texture within the 0-60 cm layers, high organic matter (170 g kg^{-1}) and available phosphorus (Olsen P, 25 mg kg^{-1}) concentrations, and low aluminum saturation index. According to the meteorological station at the site, the 30 years average rainfall for the area is 1,260 mm yr^{-1} and a mean ambient temperature of 11.3°C (7.2 to 15.6°C).

In this study, animals were managed under rotational grazing in closed systems (2 ha each) on a permanent pasture 25 years old that had been always used for grazing with beef cattle. The grass area was divided in 54 equal grazing strips, offering three grazing strips (1254 m^2) every 3 days or one grazing strip (418 m^2) per day, which was equivalent to two different immediate stocking rates (number of animals per area unit per day) of 63 steers and 191 steers ha^{-1} day^{-1}, respectively. These treatments were evaluated on a 4% slope field and the 63 steers ha^{-1} day^{-1} treatment was also evaluated in a 12% slope field. Grazing was carried out with Holstein-Friesian steers with initial live weights of 212 ± 9.9 kg, 173 ± 23.0 kg and 248 ± 12.0 kg for 2004, 2005 and 2006, respectively so that the average stocking rate at the beginning of each experimental season was 1.3 AU ha^{-1}. Treatments were fertilized each year in autumn with 45 kg N ha^{-1} (urea, 46% N, 1st year; 2nd and 3rd year , sodium nitrate, 16% N) and in the spring with 25 kg N ha^{-1} (sodium nitrate) and 29 kg P ha^{-1} (triple superphosfate, 46% P_2O_5). Detailed information on systems description and management can be found in Alfaro *et al.* (2009).

To quantify N losses in surface runoff, three surface lysimeters (5x5m) were established in each treatment, according to the methodology described by Alfaro and Salazar (2007), and surface runoff was collected three times per week. The accumulated surface runoff was measured at each sampling date. Leaching losses at 60 cm depth were estimated using ceramic cups (Webster *et al.*, 1993), with three cups per surface lysimeter (n=9 per treatment). Samples were collected fortnightly in the first year and after each 100 mm of drainage in the 2nd and 3rd year. Drainage for the period was calculated as the difference between rainfall and evapotranspiration for each sampling period, according to Lord and Shepherd (1993). Total rainfall and evapotranspiration for the period was registered with an automatic weather station placed at the experimental site (1 km distance). Leachate samples were frozen until analysis for available N (NO_3^- and NH_4^+). Runoff samples were stored at 4ºC until analysis for available N. Nitrate was measured using the salicylic acid method (Robarge *et al*, 1983), and ammonium was determined through the indophenol methodology (Mulvaney, 1996) using and authomated sample analyser (Skalar SA 1050). The limit of determination for NO_3^- and NH_4^+ determination was 0.2 and 0.1 mg L^{-1}, respectively. On runoff samples, total N was determined using the Ntube test method 10071 (®Hach, 2000) and the dissolved organic N (DON) was estimated as the difference between total N and available N in the samples.

Total N losses were calculated as the product of drainage and N concentration in the respective samples. Total N losses for the experimental period were calculated as the sum of N losses in runoff and N leaching losses.

Analysis of variance (ANOVA) was used to compare nitrate and ammonium concentrations, surface runoff losses, leaching losses and overall N losses between the treatments tested. Genstat 7.1 was used as the statistical package.

RESULTS AND DISCUSSION

Rainfall, evaporation and drainage

Rainfall, evaporation and drainage data from April to December for 2004, 2005 and 2006, and a 30 year average for the INIA-Remehue meteorological station are presented in Figure 1. For this period of evaluation, the year 2004 had a similar rainfall to the 30 year average, however, 2005 and 2006 had a higher rainfall, being 195.6 and 176.0 mm higher than the 30 year average. Evaporation was similar for 2005 and 2006 compared to the 30 year average, but in 2004 was lower than the average (-47.2 mm). According to this information, estimated drainage for 2005 and 2006 were 306.7 and 220.9 mm higher than for 2004. The high drainage observed in the second and third experimental seasons had an impact on N leaching losses, because this parameter, with N concentration, determines N losses.

Results also showed that the main pathway of water movement was leaching through the soil, accounting for 99% of the total volume of drainage water collected ($p < 0.05$), and that the immediate stocking rate and slope tested did not affect the proportion of water loss due to leaching or runoff ($p > 0.05$). This data are in agreement with Alfaro *et al.* (2005, 2009) for a study carried out in the same area with beef cattle (6% field slope) and with Ledgard *et al.* (1999) for grazing cow systems on a similar soil type in New Zealand.

High water runoff volumes were collected in wet years, and were important in critical periods such as the start of autumn when the soil was dry after summer or during winter with waterlogged conditions. The impact of water runoff on N losses to water would

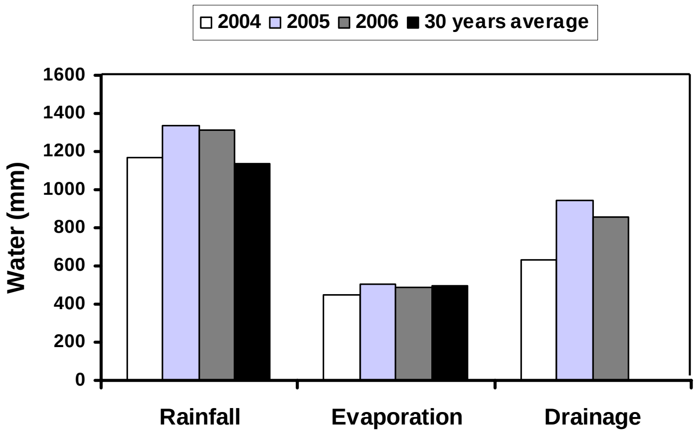

Figure 1. Rainfall, evaporation and drainage from April to December for 2004, 2005 and 2006, compared with the 30 year average. INIA-Remehue meteorological station.

be more important when autumn or spring fertilization of grass matches high rainfall periods, such as those observed in spring time during 2005 and 2006.

Nitrogen runoff losses

The field slope affected either organic N, ammonium N or nitrate N concentration with high values observed in each year for the 12% slope compared to 4% slope treatments (Table 1; $p \leq 0.05$). On the other hand, grazing intensity affected nitrate concentrations only during year 1, which increased (Table 1; $p \leq 0.05$). There was no interaction between the stocking rate and the field slope for N losses in runoff ($p > 0.05$).

There were differences between years for N losses due to runoff, following the order 2006 > 2005 > 2004 ($p < 0.01$), which was associated to differences in rainfall between years. During the three years, nitrate-N concentration in runoff exceeded the Chilean and EC limit for drinking water (11.3 mg L^{-1}; EC, 1991).

In addition, ammonium concentrations also were higher than the values established in the Chilean normative for surface water (1 mg L^{-1}, DS 87/01). High N losses were observed during winter and after spring fertilizer applications, especially in rainfall periods, so that N transport in runoff from grazing areas could be important in paddocks located next to surface waters, affecting the use of water for other activities such as aquaculture and tourism (Alfaro *et al.,* 2005). Good agriculture practices could be used to reduce the risk of water pollution due to N runoff. Buffer strips have been shown to be effective in protecting surface waters (e.g. Hickey and Doran, 2004). In addition, it is important to match N fertilization rates with N requirements by grass to avoid surpluses as these systems tend to accumulate N, as soil and gate budgets have shown (Alfaro *et al*., 2009).

Average data for the three years of this study showed that only the field slope affected total N losses due to runoff ($p \leq 0.05$). This would be related to both the direct transport of dissolved nitrate in the runoff water and the effect of soil particle transport which resulted in high DON

Table 1. Average nitrogen concentration in surface runoff samples (mg L^{-1}) and total losses in surface runoff (kg N ha^{-1}) for treatments with different stocking rates and field slopes, 2004-2006 (± sem).

Treatment	**63 steers ha^{-1} d^{-1}, 4% slope**	**191 steers ha^{-1} d^{-1}, 4% slope**	**63 steers ha^{-1} d^{-1}, 12% slope**
2004			
Average surface runoff concentrations and range (mg L^{-1})			
$N\text{-}NH_4^+$	27 ± 11.3 **a**	30 ± 13.9 **a**	43 ± 15.7 **a**
$N\text{-}NO_3^-$	26 ± 13.6 **b**	61 ± 23.9 **a**	52 ± 19.9 **a**
Organic N	15 ± 6.1 **b**	14 ± 3.7 **b**	77 ± 9.7 **a**
Total N losses (kg ha^{-1})			
$N\text{-}NH_4^+$ in surface runoff	0.01 ± 0.001 **a**	0.03 ± 0.003 **a**	0.04 ± 0.005 **a**
$N\text{-}NO_3^-$ in surface runoff	0.003 ± 0,0.0016 **a**	0.004 ± 0.0022 **a**	0.006 ± 0.0047 **a**
Organic N in surface runoff	0.007 ± 0.004 **c**	0.03 ± 0.003 **b**	0.17 ± 0.037 **a**
Total	**0.02 b**	**0.06 b**	**0.22 a**
2005			
Average surface runoff concentrations and range (mg L^{-1})			
$N\text{-}NH_4^+$	11 ± 1.2 **b**	9 ± 1.4 **b**	25 ± 3.5 **a**
$N\text{-}NO_3^-$	17 ± 3.0 **a**	13 ± 4.5 **a**	32 ± 7.4 **a**
Organic N	17 ± 4.09 **b**	11 ± 2.30 **b**	38 ± 5.81 **a**
Total N losses (kg ha^{-1})			
$N\text{-}NH_4^+$ in surface runoff	0.04 ± 0.002 **b**	0.06 ± 0.016 **b**	0.09 ± 0.016 **a**
$N\text{-}NO_3^-$ in surface runoff	0.07 ± 0.007 **a**	0.09 ± 0.021 **a**	0.06 ± 0.012 **a**
Organic N in surface runoff	0.07 ± 0.013 **b**	0.14 ± 0.024 **b**	0.26 ± 0.041 **a**
Total	**0.18 c**	**0.29 b**	**0.41 a**
2006			
Average surface runoff concentrations and range (mg L^{-1})			
$N\text{-}NH_4^+$	20 ± 2.8 **a**	12 ± 4.7 **a**	17 ± 2.7 **a**
$N\text{-}NO_3^-$	10 ± 3.0 **a**	27 ± 8.1 **a**	20 ± 2.9 **a**
Organic N	22 ± 3.8 **a**	17 ± 7.4 **a**	24 ± 5.1 **a**
Total N losses (kg ha^{-1})			
$N\text{-}NH_4^+$ in surface runoff	0.15 ± 0.020 **a**	0.10 ± 0.040 **a**	0.19 ± 0.035 **a**
$N\text{-}NO_3^-$ in surface runoff	0.08 ± 0.022 **b**	0.27 ± 0.064 **a**	0.23 ± 0.043 **a**
Organic N in surface runoff	0.15 ± 0.005 **b**	0.13 ± 0.032 **b**	0.21 ± 0.008 **a**
Total	**0.38 c**	**0.50 b**	**0.63 a**

Different letters in columns show significant differences among treatments ($p \leq 0.05$)
± sem: standard error of the mean

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Treatment	63 steers ha^{-1} d^{-1}, 4% slope	191 steers ha^{-1} d^{-1}, 4% slope	63 steers ha^{-1} d^{-1}, 12% slope
Average 2004-06			
Average surface runoff concentrations and range (mg L^{-1})			
N-NH_4^+	20 ± 3.7 **a**	18 ± 4.1 **a**	34 ± 5.7 **a**
N-NO_3^-	14 ± 1.8 **c**	23 ± 4.0 **b**	31 ± 8.7 **a**
Organic N	18 ± 2.6 **b**	14 ± 2.6 **b**	47 ± 8.6 **a**
Total N losses (kg ha^{-1})			
N-NH_4^+ in surface runoff	0.07 ± 0.020 **b**	0.07 ± 0.015 **b**	0.12 ± 0.023 **a**
N-NO_3^- in surface runoff	0.06 ± 0.011 **b**	0.14 ± 0.037 **a**	0.11 ± 0.031 **a**
Organic N in surface runoff	0.07 ± 0.021 **b**	0.10 ± 0.021 **b**	0.21 ± 0.022 **a**
Total	**0.20 b**	**0.31 b**	**0.44 a**

Different letters in columns show significant differences among treatments ($p \leq 0.05$)

±sem: standard error of mean

losses (Jarvis, 2002). Ammonium losses were high in the 12% slope treatment probably due to direct runoff of urine when the soil was saturated. Studies carried out by Preedy *et al.* (2001) in UK showed similar results after manure application to grass.

Grazing intensity did not significantly affect DON and ammonium losses, however, higher losses of nitrate were observed in the higher stocking rate treatment ($p \leq 0.05$), which could be attributed to the higher amount of excreta under the increased stocking rate in agreement with data reported by Haynes and Williams (1993).

Nitrogen leaching losses

Field slope and grazing intensity did not significantly affect NO_3-N and NH_4-N concentration and total N losses ($p > 0.05$; Table 2). Nitrogen leaching losses were greater in 2005 and 2006 than in 2004 because the drainage was higher for these two years. In addition, N losses at the end of the drainage period (spring period) represented between 58% to 82% of the total losses for 2005 and 2006, so that on rainy springs N input as fertilizer had an important effect on N losses, which should be considered in cattle production systems with more N intensive use, or where soil N levels are high.

Nitrate-N concentrations were low in the three drainage seasons, being below 10 mg L^{-1} for all treatments and evaluation years (Figure 2), with higher concentrations observed at the end of the drainage period. These values were below the EC limit for drinking water (EC, 1991). Average concentrations were less than 2.8 mg L^{-1} and 0.06 mg L^{-1} for NO_3-N and NH_4-N, respectively (Table 2).

Nitrogen losses occur mainly as NO_3^-, representing from 84 to 99% of the total N leached. Similar results have been reported by Ledgard *et al.* (1999) for a study carried out in New Zealand on volcanic soils. Ammonium nitrogen is easily transformed in the soil to nitrate, which is the main N form in soils (e.g. Di and Cameron, 2002). However, studies have also showed that N could be leached as organic form (Hawkins and Scholefield, 2000; Murphy *et al.*, 2000).

Table 2. Average N concentration in leachates (mg L^{-1}) and total N leaching losses (kg N ha^{-1}) for treatments with different immediate stocking rates and field slopes, 2004-2006 (± sem).

Treatment		**63 steers ha^{-1} d^{-1}, 4% slope**	**191 steers ha^{-1} d^{-1}, 4% slope**	**63 steers ha^{-1} d^{-1}, 12% slope**
Average leachate concentrations and range (mg L^{-1})				
$N\text{-}NH_4^+$	2004	0.03 ± 0.006 **a**	0.04 ± 0.006 **a**	0.04 ± 0.001 **a**
	2005	0.01 ± 0.006 **a**	0.02 ± 0.016 **a**	0.02 ± 0.014 **a**
	2006	0.13 ± 0.062 **a**	0.02 ± 0.006 **a**	0.01 ± 0.004 **a**
$N\text{-}NO_3^-$	2004	0.3 ± 0.10 **a**	0.9 ± 0.32 **a**	0.1 ± 0.02 **a**
	2005	1.7 ± 1.04 **a**	2.8 ± 0.51 **a**	0.2 ± 0.09 **a**
	2006	2.3 ± 0.34 **a**	2.2 ± 1.85 **a**	2.5 ± 1.21 **a**
Total N losses (kg ha^{-1})				
$N\text{-}NH_4^+$ in leachates	2004	0.2 ± 0.04 **a**	0.2 ± 0.04 **a**	0.2 ± 0.01 **a**
	2005	0.1 ± 0.05 **a**	0.2 ± 0.15 **a**	0.2 ± 0.13 **a**
	2006	1.1 ± 0.53 **a**	0.2 ± 0.05 **a**	0.1 ± 0.04 **a**
	Average	0.5 ± 0.21 **a**	0.2 ± 0.06 **a**	0.2 ± 0.05 **a**
$N\text{-}NO_3^-$ in leachates	2004	2.0 ± 0.64 **a**	5.4 ± 2.00 **a**	0.9 ± 0.14 **a**
	2005	16.3 ±9.82 **a**	26.8 ± 4.83 **a**	1.5 ± 0.83 **a**
	2006	19.8 ± 2.88 **a**	18.8 ± 15.79 **a**	21.5 ± 10.38 **a**
	Average	12.7 ± 4.02 **a**	17.0 ± 5.64 **a**	8.0 ± 3.59 **a**
Total (kg N ha^{-1})				
	2004	2.2 ± 0.61 **a**	5.7 ± 1.96 **a**	1.1 ± 0.15 **a**
	2005	16.4 ±9.81 **a**	27.0 ± 4.85 **a**	1.7 ± 0.81 **a**
	2006	20.9 ± 3.16 **a**	18.9 ± 15.80 **a**	21.6 ± 10.39 **a**
	Average	13.2 ± 3.95 **a**	17.2 ± 5.69 **a**	8.1 ± 3.55 **a**

Different letters in columns show significant differences among treatments ($p \leq 0.05$)
±sem: standard error of mean

Measurements in this study indicate some organic N leaching losses, however, there was a high variability between samples, and data are not presented in this paper.

Cumulative annual N losses due to inorganic N leaching were small and ranged from *c.* 0.9 to 26.8 kg ha^{-1} yr^{-1}. No significant differences were observed between treatments ($p > 0.05$; Table 2). There was a high variability among the leaching samples within treatments and years. This situation is common when N is evaluated in soil and especially on grazing systems due to patches of urine and faeces in the pasture (Haynes and Williams, 1993).

Nitrogen leaching values determined in the present study were higher than those reported by Nissen and Daroch. (1991), Misselbrook *et al.* (1996), Salazar (2002) and Salazar *et al.* (2005) for cut grass, and lower than values reported by

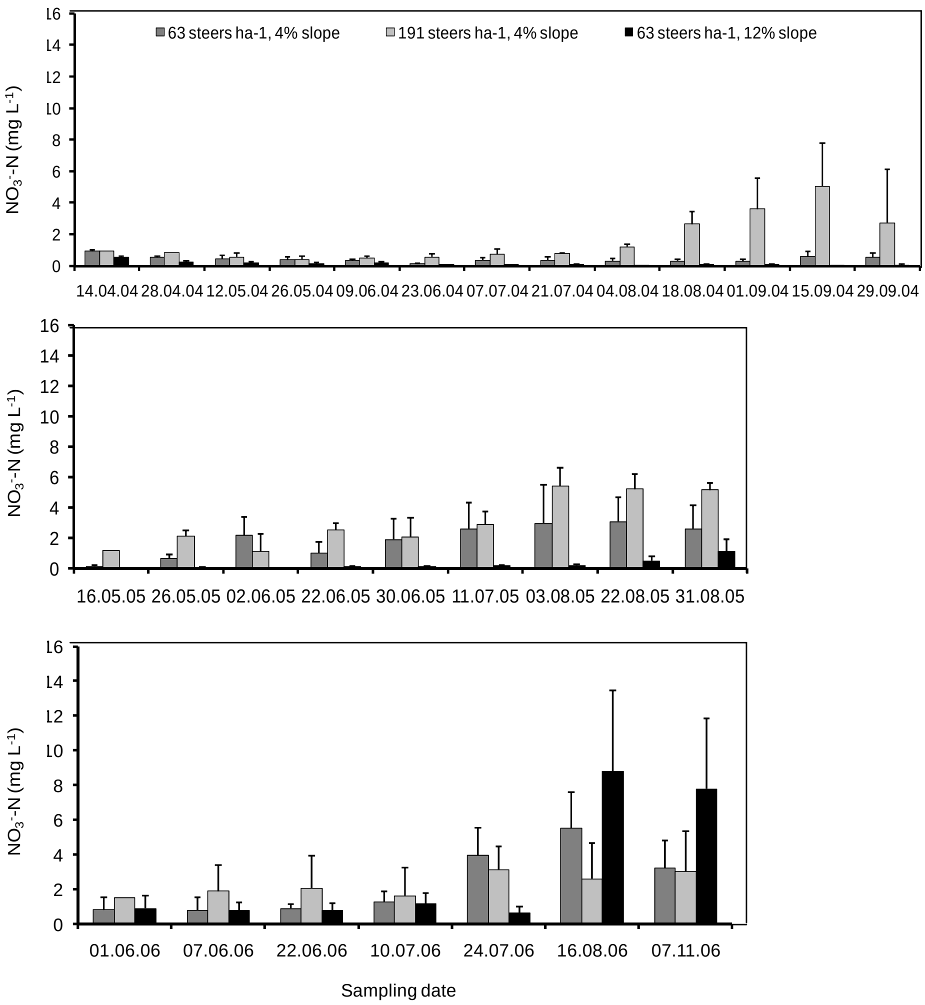

Figure 2. Concentration of NO_3^--N in leachate samples for the different treatments and sampling dates in 2004, 2005 and 2006.

de Klein and Ledgard (2001), Cameron *et al.* (1992) and Cameron and Di (2004) for grazed pastures in New Zealand. Many studies have shown that N leaching losses are higher in crops and in grazed pastures compared to cut grasslands. An evaluation carried out by Ryden *et al.* (1984) showed 5.6 times higher losses in a grazed pasture compared to a similar pasture under cutting. Higher N losses due to leaching from grazed grassland compared to cut grassland could be explained by the high concentration of N (600 – 1,000 kg N ha^{-1}) under urine or faeces patches (Haynes and Williams, 1993), which are well above the needs of the plants and so prone to leaching losses.

The low N leaching values determined in the present study compared with published data elsewhere on grazed pastures, could be associated to differences in soil type, animal type and intensification of the cattle production system evaluated, especially on N use. Most of the published studies abroad have been carried out using dairy cows, with high stocking rates (e.g. 2.5 cows ha^{-1}; de Klein and Ledgard, 2001). In addition, N fertilizer rates in these experiments are higher than the one used in the present study, which has an important effect on N losses (Ledgard *et al.*, 1996).

Recent studies have shown that forest volcanic soil of Southern Chile have very specialised microbial and abiotic retention processes which can reduce the risk for N leaching, despite the high N turnover rates determined in this soil (Huygens *et al.*, 2008). Studies in these ecosystems indicated very high gross NH_4^+ production fluxes, but also a suppression of autotrophic nitrification as a result of strong competition for available NH_4^+ by abiotic immobilization processes and NH_4^+ assimilating heterotrophic microorganisms (Huygens *et al.* 2007). More studies are required to understand the role of these processes on N dynamics in grazed soils of southern Chile.

Results from the present study suggest that livestock production systems in southern Chile could be intensified through the use of increasing stocking rates and fertilizers inputs, but the matching between agronomic managements and intensive rainfall should be considered as a strong constraint for N losses to the wider environment. In addition, other pathways of N losses should be considered, such as losses as ammonia and nitrous oxide emissions, for future studies. Both pathways could be relevant for both N use efficiency and greenhouse gasses generation and global warming.

CONCLUSIONS

Results from this study showed that field slope and immediate stocking rate did not affect total N leaching losses significantly ($p > 0.05$). However, these factors affect N concentration in runoff water, where higher values were observed in 12% slope and 191 steers ha^{-1} compared with 4% slope and 63 steers ha^{-1}. Nitrogen losses due to runoff were low (<0.65 g ha^{-1} yr^{-1}), however, in some sampling periods high concentrations were observed which overpass the EC concentration limit for drinking water. In the 4% slope treatment runoff losses were as nitrate, DON and ammonium and in the 12% slope treatment N was mainly transferred as dissolved organic N. Nitrogen was mainly lost by leaching, nor by runoff (>99% of total N losses), so that losses in this pathway ranged from 0.9 to 26.8 kg ha^{-1} yr^{-1}, where nitrate accounted for >84% of the N losses. Nitrate-N concentrations in leachates were low, with annual averages lower than 2.8 mg L^{-1}. There was a year effect on N losses due to leaching, which can be explained for higher drainage in 2005 and 2006 compared to 2004. Late rainfall in late winter and spring had an important impact on N losses, representing from 58% to 82% of the total N losses. We suggest that low leaching losses reported in this study could be related to the low N fertiliser inputs and the potential role of N adsorption in the volcanic soils. Further research is required to determine the contribution of this process to N cycling in grassland soils of southern Chile.

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