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NITROGEN LOSSES UNDER DIFFERENT CATTLE GRAZING FREQUENCIES AND INTENSITIES IN A VOLCANIC SOIL OF SOUTHERN CHILE

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ABSTRACT

Chilean livestock production systems have intensified over the last years, with increasing amounts of N fertilizer inputs creating the potentiality for environmental damage through N pollution of water and air, so that alternative production strategies have been developed to reduce such environmental impacts. This study assesses N losses under different grazing frequencies and intensities on permanent pasture (*Lolium perenne* L., *Festuca arundinacea* Schreb., *Dactylis glomerata* L., and *Trifolium repens* L.) on an Andisol in Southern Chile. Four grazing strategies were evaluated: frequent-heavy (FH), frequent-light (FL), infrequent-heavy (IH), infrequent-light (IL), and a no grazing control (C) treatment, and each with three replicates in a randomized complete block design. Results of the experiment indicate that N leaching losses were greater in the FH treatment (58.7 kg available N ha^{-1}; $p < 0.05$) and with most of the leaching occurring in spring (39%). On average, N ammonia (NH_3) losses were 10% greater in the frequent grazing treatments in relation to the infrequent grazing treatments, since there were no significant differences ($P \leq 0.05$) among individual grazing events for FH, FL and IH. Results indicate that grazing frequency affects leaching losses while grazing intensity affects ammonia emissions from the grassland. Grazing with dairy cows in Southern Chile should consider this environmental constraint to ensure sustainable production over time.

Key words: ammonium, grazing, leaching, nitrate, NH_3 volatilization.

INTRODUCTION

Chile is a country with great potential for livestock production based on grazing, however, due to the intensification of these productive systems in recent years, there has been increase of N fertilizer applications and stocking rates employed (Demanet *et al.,* 2006a). This has resulted in potentially high negative environmental impacts due to the inefficiency of ruminants to transform N from herbage into milk or meat (Hoekstra *et al.,* 2007).

Inappropriate grazing strategy and fertilizer application management could exacerbate N losses to water, soil, and air (Shepherd and Lord, 2004). These losses from pasture can be due to N leaching (Delaby *et al.;* 2000; Jarvis and Ledgard, 2002; Di and Cameron, 2004) and gaseous emissions such as ammonia volatilization (NH_3) (Jarvis, 1997).

Nitrogen leaching losses from grazing pastures have been previously reported (Jarvis, 1993; Jarvis and Ledgard, 2002; Eckard *et al.,* 2003; Oenema, 2006; Dueri *et al.,* 2007). Di and Cameron (2004) reported N leaching losses of 85 kg N ha^{-1} yr^{-1} in pastures, although higher values have been reported by Ledgard *et al.* (1999) in seeded pastures with 74-204 kg N ha^{-1} in the first year and 20-146 kg N ha^{-1} in the second year. Low losses have been reported and associated to low rainfall inputs, for example, on a Swiss dairy farm (38 kg N ha^{-1} yr^{-1} with *c*. 900 mm; Dueri *et al.,* 2007) and in Southern Australia (26-33 kg N ha^{-1} yr^{-1} with 750 mm; Pakrou and Dillon, 2004).

Ammonia emissions from pastures have been widely studied (Misselbrook *et al.,* 2004; Saggar *et al.,* 2004b; Misselbrook *et al.,* 2005a; 2005b). They have ranged from 19-28 kg N ha^{-1} under grazing in Australia (Eckard *et al.,* 2003) up to 47-57 kg N ha^{-1} yr^{-1} in the Netherlands from grass/fertilizer-N areas (Schils *et al.,* 2005) and dairy farms in New Zealand (Jarvis and Ledgard, 2002). Management of these pastures used N fertilizers and irrigation, as well as an animal grazing system.

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There are few studies in Chile about N leaching losses in grazed pastures. Alfaro *et al*. (2007) reported losses ranging from 3-70 kg N ha^{-1} yr^{-1} in accordance with stocking rates and grazing strategies used. Leaching losses between 13-67 kg nitrate (NO_3^-) ha^{-1} yr^{-1} have also been reported with dairy slurry application (Alfaro *et al.,* 2006). The potential for N leaching losses with irrigation after inorganic fertilizer addition has been estimated as 88-90 kg N ha^{-1} in the area (Mora *et al.,* 2007), ranging from 4-90 kg N ha^{-1}. Most of these losses (90%) are nitrate (Mora *et al.,* 2004; Alfaro *et al.,* 2005). Regarding ammonia volatilization, Vidal and Chamorro (2005) reported losses of 12-19 kg NH_3 ha^{-1} in wheat (*Triticum aestivum* L.), whereas Salazar *et al*. (2007) reported 10 kg NH_3 ha^{-1} yr^{-1} in grazing systems with urea fertilizer application (100 kg N ha^{-1}).

Soil and nutrient losses in these experiments were relatively low in all grazing strategies given a combination of good cover, low slope, and low rainfall intensities (O'Reagain *et al.,* 2005). Therefore, different types of pasture management (frequency and intensity) may affect N losses in the cover. Thus, the objective of this study was to evaluate the effect of different types of grazing management on nitrate and ammonium (NH_4^+) leaching losses, as well as NH_3 emissions in Southern Chile.

MATERIALS AND METHODS

Experimental site and treatments

The study was carried out in 2005-2006, spring (21 September to 21 December), summer (21 December to 22 March), autumn (22 March to 21 June), and winter (21 June to 23 September) on permanent pasture (*Lolium perenne*, *Festuca arundinacea*, *Dactylis glomerata,* and *Trifolium repens*) on an Andisol of the Freire series (Typic Placudands; CIREN, 2003), in Southern Chile (La Araucanía Region, Cautín Province; 38° S, 72° W, 70 m.a.s.l.). The soil has a silt loam texture (0-20 cm) and its bulk density varies in depth from 0.82 to 1.32 g cm^{-3} (Mella and Kuhne, 1985).

Weather data at the experimental site were collected from the Maquehue Station, located 1 km from the experimental site (Dirección Meteorológica de Chile 2005-2006). Total rainfall for the 12-month period was 1607 mm with a daily mean of 2.1-5.6 mm for the grazing season. The daily mean was 5.6 mm d^{-1} in winter and 2.1 mm d^{-1} in summer. Mean air temperature was greater in summer, 14.6 °C (-4 to 35.5 °C) while mean soil temperature was 18.2 °C (6 to 21 °C) during the period under study. Soil moisture varied between 8 and 80% during the year while wind speed varied from 0.1-16.2 m s^{-1}.

Soil chemical properties were determined in accordance with the method described by Sadzawka *et al*. (2000) at 0-10 cm. At the start and end of the experiment, the soil had a mean Olsen-P concentration of 16 mg kg^{-1} and 15.4 mg kg^{-1}, respectively, and an adequate nutritional level for grassland production. Organic matter (OM) content had a mean of 13%, total N 5.1 to 5.3 g kg^{-1}, and pH (water) 5.5 (Table 1).

Table 1. Soil chemical properties of each treatment during the period from April 2004 to September 2006, Maquehue Station (0-10 cm; n = 3). Values in parentheses are the mean standard error.

Property	Initial value (April 2004)	Treatments (September 2006)				
		C	FH	FL	IH	IL
pH (H_2O)	5.5 (0.07)	5.5 (0.06)	5.6 (0.09)	5.5 (0.07)	5.5 (0.09)	5.5 (0.07)
Olsen-P, mg kg^{-1}	16 (0.68)	15.8 (0.63)	14.8 (1.11)	14.5 (0.96)	14.5 (1.19)	15.5 (0.29)
K, mg kg^{-1}	250 (36.42)	355 (21.24)	352 (65.53)	372 (97.31)	247 (46.87)	229 (22.49)
OM, g kg^{-1}	120 (0.72)	132 (0.75)	117 (0.75)	113 (1.65)	114 (0.87)	111 (0.65)
Total N, g kg^{-1}	5.1 (0.18)	5.3 (0.13)	5.2 (0.156)	5.4 (0.16)	5.3 (0.09)	5.3 (0.31)
TOC, g kg^{-1}	69.6 (2.11)	76.6 (3.11)	67.7 (2.54)	68.1 (2.42)	65.9 (2.90)	64.7 (0.67)
K, $cmol_{(+)}$ kg^{-1}	0.6 (0.09)	0.9 (0.06)	0.9 (0.17)	0.9 (0.25)	0.6 (0.12)	0.6 (0.06)
Na, $cmol_{(+)}$ kg^{-1}	0.3 (0.01)	0.1 (0.01)	0.1 (0.01)	0.1 (0.01)	0.1 (0.01)	0.1 (0.02)
Ca, $cmol_{(+)}$ kg^{-1}	7.0 (0.29)	5.7 (0.18)	7.2 (0.22)	6.8 (0.49)	7.8 (0.21)	7.7 (0.46)
Mg, $cmol_{(+)}$ kg^{-1}	1.4 (0.11)	1.6 (0.14)	1.8 (0.12)	1.7 (0.08)	2.0 (0.13)	1.8 (0.08)
Al, $cmol_{(+)}$ kg^{-1}	0.2 (0.06)	0.3 (0.06)	0.3 (0.07)	0.4 (0.05)	0.4 (0.03)	0.5 (0.06)
Σ Bases, $cmol_{(+)}$ kg^{-1}	9.3 (0.42)	8.4 (0.34)	10.1 (0.42)	9.6 (0.66)	10.5 (0.41)	10.2 (0.51)
ECEC, $cmol_{(+)}$ kg^{-1}	9.5 (0.41)	8.6 (0.33)	10.4 (0.44)	10.0 (0.64)	10.9 (0.39)	10.6 (0.47)
Al saturation, %	2.3 (0.64)	2.8 (0.59)	3.3 (0.66)	3.8 (0.71)	3.4 (0.37)	4.3 (0.67)

Values in parentheses are the mean standard error.
ECEC: Effective cation exchange capacity; OM: organic matter; TOC: total organic carbon; C: no grazing control; FH: frequent-heavy; FL: frequent-light; IH: infrequent- heavy; and IL: infrequent-light.

Table 2. Herbage availability criterion at the beginning and end of each grazing season for each treatment.

Season	Beginning and end of grazing	Treatments: Herbage availability (kg DM ha^{-1}) C	FH	FL	IH	IL
Spring	Beginning	No grazing	2200	2200	2600	2600
	End	1400[1]	1200	1600	1200	1600
Summer	Beginning	No grazing	2000	2000	2400	2400
	End	1400[1]	1200	1600	1200	1600
Autumn	Beginning	No grazing	1500	1500	1800	1800
	End	1150[1]	1000	1300	1000	1300
Winter	Beginning	No grazing	1500	1500	1800	1800
	End	1150[1]	1000	1300	1000	1300

[1]Herbage cut in the season, DM: dry matter; C: no grazing control; FH: frequent-heavy; FL: frequent-light; IH: infrequent- heavy; and IL: infrequent-light.

Four grazing treatments were evaluated as well as a no grazing control (C). Treatments differed in grazing frequency and intensity: frequent-heavy (FH), frequent-light (FL), infrequent-heavy (IH), and infrequent-light (IL). Herbage availability (as determined by a rising plate meter at the start of grazing) was used as a criterion to determine grazing frequency management (Table 2): 1) infrequent treatments (IH and IL) had greater herbage availability; and 2) frequent treatments (FH and FL) had lower herbage availability before grazing. This type of grazing nomenclature is similar to the one used by McKenzie *et al*. (2006a; 2006b), where animal entrance and exit to the pasture is based on the herbage availability.

Grazing intensity was measured by the height of post-grazing residual pasture, thus 'heavy' grazing treatments (FH and IH) and 'light' grazing treatments (FL and IL) meant a higher and lower cattle herbage consumption, respectively. Therefore, time animals spent in the paddock was different among these treatments. The total number of grazing events per treatment was 10, 11, 7, and 8 for FH, FL, IH, and IL, respectively. A greater number of grazing events occurred in spring (2-4 per treatment) whereas the lowest number of grazing events was found in the summer (1-2) given pasture dry matter availability.

Longer grazings were recorded in spring (12-18 h) and shorter ones in winter (0.9-2.2 h). Inter-grazing time was shorter in spring (22-38 d) and longer in winter (27-77 d). Frequent grazing treatments had shorter shifts (22 to 40 d for spring and winter, respectively) than infrequent grazing treatments (30 to 77 d for spring and winter, respectively). Grazing frequency and intensity can affect the N input pathways of dung and urine in the pasture, and therefore, can affect N loss processes in the system (Haynes and Williams, 1993).

Herbage was removed from the no grazing control (C) by cutting the grass when it had reached an availability of 1400, 1400, 1150, and 1150 kg DM ha^{-1} for the spring, summer, autumn, and winter seasons, respectively (Table 2), so as to generate similar plant uptake conditions as those recorded in grazing treatments.

Three 165 m^2 replicate paddocks were established for each treatment in a randomized complete block design. Paddocks were grazed with six Holstein-Friesian non-lactating dairy cows, 400 kg mean live weight. Mean stocking rate was 2.1 livestock units (LU) ha^{-1} yr^{-1}. Cows were maintained in pasture outside of the experimental area between grazing.

Urea-N fertilizer (46% N) 46 kg N $ha^{-1}$was applied twice to all plots in spring (15 October and 15 November 2005), once in autumn (4 April and 8 May 2006), and once in winter (17 August 2006). Along with urea, a split dressing of potassium magnesium sulfate ("Sulpomag") was applied at a rate of 100 kg ha^{-1} (22% K_2O, 18% MgO, 21.5% S, 2.5% Cl, and 0.5% maximum moisture). Triple super phosphate (46% P_2O_5), 200 kg ha^{-1}, was applied once in autumn (4 April 2006) and once in winter (17 August 2006). Magnecal (lime, 15% MgO) and gypsum (18% S) were applied at a rate of 1000 and 500 kg ha^{-1}, respectively in March 2006 to prevent soil acidity problems.

Nitrogen content in dung and urine

Dung and urine samples were directly collected from each cow and a composite sample (200 g and 150 mL, respectively) for each paddock was analyzed. Measurements of dung and urine pH were analyzed immediately after collecting the samples and these were stored at 4 °C for further analyses. Total hydrolysable N in dung was determined by distillation with NaOH, Kjeldahl digestion with H_2SO_4, and a catalytic mixture of K_2SO_4 (Stevenson, 1982). Inorganic N concentration in dung was measured by extraction with KCl and distillation (Sadzawka *et al.,* 2001). Inorganic N (NO_3^- and NH_4^+) concentration in urine was measured by distillation with MgO and Devarda alloy (Al, Cu, and Zn) and sulfamic acid (Stevenson, 1982). Total N in urine was measured by Kjeldahl's digestion method (Sadzawka *et al.,* 2001). Total N inputs via dung and urine for each treatment were estimated from: time animals were in the paddock, paddock area, N concentration in dung and urine, as well as frequency and volumes of depositions (urine and dung). Frequency and volumes of urine and dung depositions were determined from separate measurements (24 h each, one in spring and another in autumn) with three 450 kg live weight cows.

Ammonia emission measurements

Ammonia emissions were determined with static chambers to measure nitrous oxide emissions (design used by Saggar *et al.,* 2004a). Chambers were constructed with PVC with a 250 mm diameter, 210 mm height, and a removable lid with an airtight seal. Three replicate chambers were distributed at random in each grazed plot. They were inserted 110 mm into the soil. Measurements were taken 1 d before grazing and for 4 d after. Chambers were closed for a period of 4.5 h, between 1200 and 1630 h at each measurement according to the kinetics of ammonia emission previously determined for this soil under grazing during spring and summer (unpublished data). The highest emissions were obtained between 1200-1630 h, representing a mean of 54% of total emission for the 14-h period as in Bussink (1992), Ruess and McNaughton (1988) for soil under grazing management.

The gas sample was taken during each measurement at the beginning and end of each period from the chamber headspace via a septum in the chamber lid. The gas sample (10 mL) was immediately introduced into a gas sampling tube containing 5 mL of sulfuric acid (0.05 M) which was shaken to ensure ammonia absorption in the acid. Ammonium-N concentration in the acid samples was determined spectrophotometrically by the indophenol blue method (Searle, 1984). The ammonia emission rate (*F*) was determined according to:

$$F = V(C_{4.5} - C_0)/(t * A)$$

where V (m^3) is the chamber headspace volume, C ($\mu g\ m^{-3}$) ammonia-N concentration in the chamber headspace at 0 or 4.5 h, t (h) sampling duration, and A (m^2) area enclosed by the chamber. Cumulative emissions for each season were obtained as the product between the total number of days during the season and the mean of the measured daily emission rates.

Measurement of leaching losses

Available N (ammonium and nitrate) leaching losses (0-0.55 m) were determined with lysimeter plots (three per grazed paddock). Each lysimeter consisted of a PVC cylinder with a 75 mm diameter and 550 mm length. A nylon membrane (pore size < 0.02 mm) separated the soil column in the top 450 mm of the cylinder from a collected leachate volume in the bottom 100 mm. Lysimeters were established 4-5 d before each grazing season and remained in the field for the full period. A tube connecting the leachate collection volume section and the soil surface allowed regular sampling. Leachate volume in each lysimeter was measured every 14 d and sub-samples were taken to determine nitrate and ammonium concentrations. These were measured by extraction and distillation according to the methodology described by Sadzawka *et al*. (2000). Available N leaching losses were calculated from N concentration determined in leachates in each lysimeter and leachate volume collected for each sampling period.

Soil inorganic N

Soil samples were taken 3 d before and 3 d after each grazing event (0-10 cm). Four soil cores were taken each time from each paddock at a 10 cm depth. The number of samples was 72, 78, 90, 60, and 66 for C, FH, FL, IH, and IL, respectively, varying with the number of grazing events per season. Available N was determined by KCl (2 M) extraction and distillation described by Sadzawka *et al*. (2000) for the Kjeldahl method.

Statistical analysis

The effect of treatments on N losses by ammonia emissions and leaching was verified by ANOVA and JMP 5.0.1.2, respectively (SAS Institute, 2002). Mean statistical differences (95% significance level) were calculated by Tukey's multiple range test ($P \leq 0.05$).

RESULTS

Nitrogen inputs via dung and urine

The highest N returns to the paddock via dung and urine occurred during spring (32-55 kg N ha^{-1}) and the lowest during winter (2-6 kg N ha^{-1}) which agreed with variations in N concentration in dung and urine (Table 3). The highest inputs were found in heavy grazing treatments with approximately 83 kg N ha^{-1} yr^{-1} in FH and 86 kg N ha^{-1} yr^{-1} in IH. The lowest inputs occurred in light grazing treatments with approximately 68 kg N ha^{-1} yr^{-1} in FL and

Table 3. Nitrogen inputs to pasture via dung and urine (n = 3). Values in parentheses are the mean standard error.

Treatments	Total N concentration in dung[1]	Total N available N in dung[1,2]	Total N concentration in urine	Available N concentration in urine	Total available N deposited		Total N deposited	
					Dung	Urine	Dung	Urine
	g kg^{-1}		g L^{-1}		kg ha^{-1}			
Spring								
FH	29.7 (0.62)	0.17 (0.004)	7.86 (0.417)	0.94 (0.050)	0.076	4.19	13.3	34.9
FL	26.4 (0.28)	0.15 (0.002)	6.81 (0.377)	0.82 (0.045)	0.072	3.90	12.6	32.5
IH	25.2 (0.54)	0.15 (0.003)	7.45 (0.123)	0.75 (0.012)	0.082	4.07	13.8	40.7
IL	25.0 (0.26)	0.15 (0.003)	6.01 (0.356)	0.54 (0.032)	0.057	2.05	9.5	22.8
Summer								
FH	18.4 (0.74)	0.11 (0.004)	5.80 (0.401)	0.58 (0.040)	0.027	1.12	4.5	11.2
FL	18.0 (0.48)	0.10 (0.003)	4.68 (0.896)	0.47 (0.089)	0.017	0.64	3.1	6.4
IH	16.3 (1.31)	0.09 (0.008)	4.27 (0.675)	0.32 (0.050)	0.013	0.36	2.3	4.8
IL	18.6 (1.50)	0.11 (0.009)	6.69 (1.115)	0.67 (0.111)	0.013	0.61	2.1	6.1
Autumn								
FH	25.2 (0.50)	0.15 (0.003)	10.33 (0.317)	1.65 (0.051)	0.020	1.80	3.4	11.2
FL	35.5 (0.58)	0.20 (0.003)	7.49 (0.162)	0.97 (0.021)	0.023	0.88	4.0	6.8
IH	31.0 (1.42)	0.18 (0.008)	7.71 (0.644)	1.02 (0.085)	0.036	1.66	6.3	12.6
IL	29.6 (0.41)	0.17 (0.002)	8.22 (1.279)	1.27 (0.198)	0.016	0.96	2.8	6.2
Winter								
FH	26.6 (0.20)	0.15 (0.001)	4.29 (0.635)	0.32 (0.047)	0.010	0.33	1.8	2.5
FL	24.1 (0.76)	0.14 (0.004)	6.88 (0.136)	0.62 (0.012)	0.004	0.13	0.8	1.9
IH	27.1 (0.20)	0.16 (0.001)	5.07 (0.067)	0.38 (0.005)	0.012	0.21	2.1	3.4
IL	26.1 (0.15)	0.15 (0.001)	5.99 (0.104)	0.54 (0.009)	0.011	0.40	1.9	3.6

[1]Available N: NO_3^- + NH_4^+; C: no grazing control; FH: frequent-heavy; FL: frequent-light; IH: infrequent-heavy; and IL: infrequent-light. [2]Based on dry matter (DM).

55 kg N ha^{-1} yr^{-1} in IL ($P \leq 0.05$). Annual contribution of recycled N via dung and urine was always significantly higher in heavy grazing than in light grazing treatments (Table 3).

Ammonia emissions

Cumulative seasonal and annual emissions were greater than grazing treatments and no grazing control with 36-41 and 31 kg NH_3 ha^{-1} yr^{-1}, respectively (Table 4; $P \leq 0.05$). There were no differences between grazing treatments in summer and winter (Table 4). In autumn, frequent grazing (FH and FL) had significantly greater losses than infrequent grazing treatments (IH and IL) ($P \leq 0.05$).

Estimated emissions throughout the year were greater in FH and FL than in IH and IL treatments with mean net emissions above no grazing control of 9.5 and 5.8 kg N ha^{-1} for frequent and infrequent treatments, respectively. Our results showed that shorter shifts between grazing events resulted in greater emissions, independent of grazing intensity. Total N volatilized per treatment was 31.2, 39.9, 41.4, 36.1, and 37.9 kg ha^{-1} for C, FH, FL, IH, and IL treatments, respectively (Table 4; $p < 0.05$).

Nitrogen leaching losses

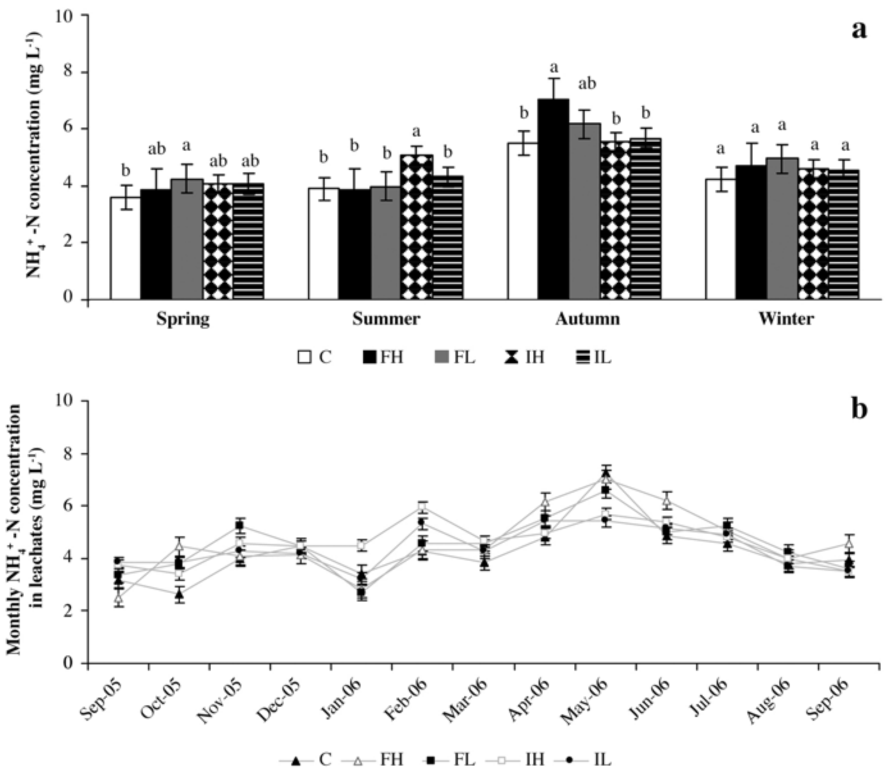
Ammonium (NH_4^+) concentration in leachates at a depth of 55 cm was lower than nitrate (NO_3^-) in those samples during the four grazing seasons, varying between 3.56 to 6.97 mg L^{-1} (Figure 1a) with a maximum peak in winter (June) of 8 mg L^{-1} (Figure 1b). Nitrate concentration varied between 7 and 33 mg L^{-1} during the four seasons in the frequent-heavy treatments (Figure 2a). The highest nitrate concentration was found in spring in the heavy grazing treatment (FH) with a peak of 50 mg L^{-1}. Nitrate concentrations were lower in summer and increased towards autumn and winter (Figure 2b).

Total N leaching losses per treatment were 33.2, 58.7, 25.8, 37.6, and 32.1 kg N ha^{-1} for C, FH, FL, IH, and IL treatments, respectively ($P \leq 0.05$; Table 4), indicating that grazing has a direct effect on N leaching losses from pasture, in agreement with Ryden *et al*. (1984).

Table 4. Ammonia emissions and nitrogen leaching losses from paddocks under different grazing regimes.

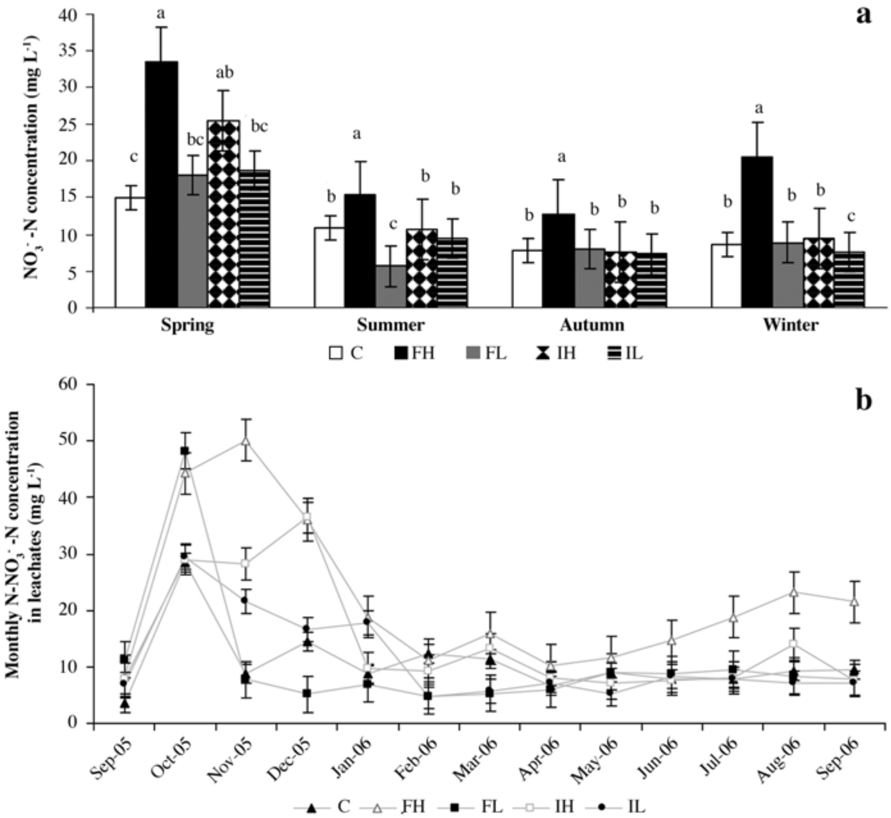
	Nitrogen losses from treatments				
	Ammonia emissions				
Season	**C**	**FH**	**FL**	**IH**	**IL**
	kg ha^{-1}				
Spring	7.5c	10.0a	10.2a	8.4b	9.5ab
Summer	8.8b	10.4a	11.2a	10.1a	10.6a
Autumn	9.7c	12.1a	12.7a	11.4b	11.4b
Winter	5.2b	7.3a	7.3a	6.1a	6.4a
Total	31.2d	39.9ab	41.4a	36.1c	37.9bc
	Nitrogen leaching losses				
Spring	11.5bc	22.9a	5.8c	14.6b	13.2b
Summer	6.0b	8.2a	3.9d	4.6cd	5.3bc
Autumn	7.2c	9.8a	7.5c	8.3b	5.4d
Winter	8.5c	17.9a	8.6c	10.1b	8.2c
Total	33.2c	58.7a	25.8d	37.6b	32.1c

Different letters within rows indicate significant differences according to Tukey's multiple range test ($P \leq 0.05$).
C: no grazing control; FH: frequent-heavy; FL: frequent-light; IH: infrequent- heavy; and IL: infrequent-light.



Treatments: C: no grazing control; FH: frequent-heavy; FL: frequent-light; IH: infrequent-heavy; and IL: infrequent-light ($P \leq 0.05$).

Figure 1. a) Concentration of NH_4^+-N in leachates collected from grazed permanent pasture during spring, summer, autumn, and winter of 2005-2006; and b) monthly mean of NH_4^+-N concentration in leachates collected from grazed permanent pasture from September 2005 to September 2006.

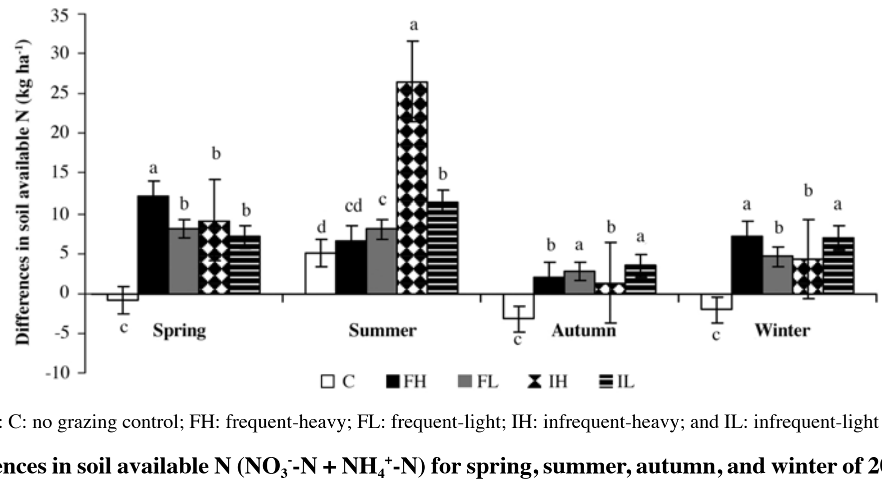


Treatments: C: no grazing control; FH: frequent-heavy; FL: frequent-light; IH: infrequent-heavy; and IL: infrequent-light ($P \leq 0.05$).

Figure 2. a) Concentration of NO_3^--N in leachates collected from grazed permanent pasture during spring, summer, autumn, and winter of 2005-2006; and b) monthly mean of NO_3^--N concentration in leachates collected from grazed permanent pasture from September 2005 to September 2006.

Nitrogen availability in the soil

Soil available N (NH_4^+-N and NO_3^--N) varied between treatments for seasons. The highest availability was found in summer (up to 29 kg available N ha^{-1} for IH, $P \leq 0.05$), and decreasing in winter (13 kg N ha^{-1}; $P \leq 0.05$) with differences among treatments. Soil available N increased after grazing stopped in all seasons (Figure 3).



Treatments: C: no grazing control; FH: frequent-heavy; FL: frequent-light; IH: infrequent-heavy; and IL: infrequent-light ($P \leq 0.05$).

Figure 3. Differences in soil available N (NO_3^--N + NH_4^+-N) for spring, summer, autumn, and winter of 2005-2006 before and after grazing in permanent pasture in Chile. Sampling points for each date and treatment = 3. Sampling dates by period varied from 66-90 in accordance with the number of grazing events.

DISCUSSION

Nitrogen inputs via dung and urine

Higher inputs were found in heavy grazing treatments. probably because plants were in a vegetative state during autumn and spring with a high leaf-stem relationship and low dead matter content producing a higher N concentration related to greater N animal intake and excretion. Annual recycled N contribution via dung and urine was always higher in heavy grazing than in light grazing treatments (Table 3). Results obtained by Reyes *et al.* (2000) showed that increasing grazing intensity increased N deposition on the soil. A similar situation occurred in this study in FH and IH treatments. Thus, intensive grazing produced a higher N contribution compared to a low stocking rate (Reyes *et al.,* 2003). A longer grazing period resulted in higher excretion frequency and area covered by excreta, thus greater distribution resulting in higher quantity of recycled N in intensive grazing treatments, in agreement with Haynes and Williams (1993) and McGechan and Topp (2004).

Urine-N contribution varied between 69-72% of total N deposition, depending on treatments and grazing time and was higher under heavy grazing. Urine-N was greater than dung-N because available N in urine was 10-13% of total N concentration, as previously described by Haynes and Williams (1993) in livestock grazing systems. Nitrogen concentrations measured in this study were similar to those reported by Silva *et al.* (2005) for a dairy herd in New Zealand (7.3-10 g L^{-1} of total N). Higher N concentration in dung and urine were found in autumn and spring when forage had a greater N concentration since there is a significant and positive linear relationship between animal N intake, and N excretion in dung and urine (Steinshamn *et al.,* 2006).

Ammonia emissions

The highest emission rate occurred in mid-spring and early autumn due to greater air temperatures which increased soil temperature at a 5-cm soil depth (data not shown) while decreasing soil moisture content favoring ammonia emissions, in agreement with Potter *et al.* (2003), Misselbrook *et al.* (2006) and Beuning *et al.* (2007). Mean air temperature was higher in summer 14.6 °C, (-4 to 35.5 °C), while mean soil temperature was 18.2 °C, (6 to 21 °C) during the period of study.

Ammonia emissions measured in the present experiment are in agreement with de Klein (2001) and de Klein and Ledgard (2001), who found emissions in New Zealand between 11 to 70 kg N ha^{-1} yr^{-1} in temperate weather and fertilizer application under irrigation. In these pastures N fertilizers were applied as urea. Our ammonia loss measurements are also similar to the values reported by Jarvis and Ledgard (2002) in the United Kingdom and New Zealand under intensive dairy and by Eckard *et al.* (2003) in Australia under grazing. Ammonia emissions were 10% greater on average in treatments managed with frequent grazing (FH and FL) than infrequent grazing (IH and IL), and 30% greater than emissions from C treatment due to the number and duration of grazing events.

In an evaluation of NH_3 emissions measured before and after grazing for each season, a tendency of greater volatilization was found after grazing as a consequence of animals depositing N on the surface of the unprotected soil which is subject to losses, except in the control, due to the absence of N recycling from dung and urine and plant uptake consumption of available soil N. Therefore, losses of NH_3 in pasture will depend on the characteristics of the system (Bouwman *et al.*, 2005), especially on grazing management. Hence, ammonia emissions were higher during and immediately after a grazing event, in agreement with Rotz (2004). Salazar *et al.* (2007) showed that ammonia losses in permanent pasture in Chile are about 10% of applied N (100 kg N ha^{-1} yr^{-1}). In our study, losses represented 13-18% of applied N as urea fertilizer. Losses also varied according to grazing frequency and intensity, in agreement with Bouwman *et al.* (2005). The lack of large differences between control and grazing treatments could be related to soil OM content since this characteristic can mask treatment effects such as fertilizer application, and probably grazing according to Alfaro *et al.* (2006).

Experiment results demonstrated that a higher frequency of cattle grazing with shorter shifts between grazings produced higher grassland N volatilization, probably because of a reduction in grass cover at the end of each grazing in these treatments, as well as the application of 92 kg ha^{-1} of N as urea (Table 4) in spring grazing. Volatilization N losses in Southern Chile were also high and should therefore be considered in grazing strategy decisions, along with grazing frequency and intensity based on herbage availability and stocking rates.

Nitrogen leaching losses

Nitrate concentrations were lower in summer, increasing towards autumn and winter (Figure 2b). Both mean and peak concentration values were greater than the threshold established for drinking water in the United States (10 mg L^{-1} for nitrate) and the European Union (11.3 mg L^{-1}) (Pervanchon *et al.,* 2005), thus implying risk for underground water pollution of grazed areas in Southern Chile, especially since this productive area is undergoing an intensification process by increasing stocking rates and N input in fertilizers. Characteristics of this soil (Table 1) are evidence that Andisol from Chile has a high NO_3^-

leaching capacity producing base loss, that is, NH_4^+. On the other hand, the type of lysimeter used should be considered along with its possible obstacles since these ion concentrations were not measured in underground water sources.

Monthly fluctuation of leached NH_4^+ showed a similar tendency during the year, less than 6 mg L^{-1}, except in the months of May and June 2006 (Figure 1b). However, NO_3^- concentration oscillated between 7 and 33 mg L^{-1} during the four seasons (Figures 2b).

Both ammonium and nitrate concentration values (Figure 1a and 2a) in this experiment were greater than those reported by Alfaro *et al*. (2005) for grazed paddocks in Southern Chile (0.02-0.04 and 1.7-1.2 mg L^{-1}, respectively). This could be explained by a greater amount of N (230 kg N ha^{-1} yr^{-1}) used in this study compared to 65 kg N ha^{-1} yr^{-1} in the previous experiment, as well as our higher stocking rates (4 and 2 LU ha^{-1}, on average, for this and the previous experiment, respectively). Cows in this experiment probably also had higher diet N inputs than the steers used by Alfaro *et al*. (2005; 2007) who obtained higher results of N excretion and N recycling to the soil as discussed previously, and thus, a higher ammonium concentration in the leachates. Also, in the present experiment, urea was used as fertilizer source, while Alfaro *et al*. (2005) used sodium nitrate as fertilizer, which resulted in greater ammonium concentrations in the leachates samples in the present experiment. These results also agree with those reported by Eckard *et al*. (2003) and Mora *et al*. (2007). Values obtained in the present experiment are low in comparison to values obtained in an irrigated pasture in New Zealand (1.9-142 mg NT L^{-1}; Barlow *et al.,* 2007) which can be related to both, the greater N amounts applied as fertilizer (120, 196, 230, 240, 248, 258 kg N ha^{-1}, predominantly with urea) and irrigation which increases N leaching (Smith and Monaghan, 2003; Cameron and Di, 2004; Barlow *et al.,* 2007). However, all doses of N in this experiment are the same, reason why this factor cannot be attributed to leachate losses given different grazing intensities and frequencies.

The highest nitrate loss occurred in the FH treatment, because of the greater animal forage intake during grazing, and because in this treatment animals remained more time on the pasture, resulting in a greater dung and urine deposition, and thus, in a greater input of organic N (Table 3) to the soil, as discussed previously. Also, the more heavy grazing treatments were associated to lower grass cover (Menneer *et al.,* 2005), which in association to a rainfall excess resulted in greater N leaching losses. During the experimental period, rainfall was 39% more than the mean (1150 mm) for the area (UCT, 2007).

Alfaro *et al*. (2005) showed lower total N leached values in Southern Chile than those obtained in our investigation fact which could be related to the lower N input and stocking rates used in that study, as well as differences in soil types. Soil in this study had lower OM (13%) thus, lower ion retention than in the previous study (18% OM) which could have resulted in higher N leaching losses. This may also have an effect on soil physical properties (texture, structure, and bulk density). Although the field management conditions and dose of fertilizers applied according by Alfaro *et al*. (2005) were different.

The spring season recorded the greatest N loss, independent of the grazing system, probably because at this time there was a greater number of grazing events and more N was applied as fertilizer (92 kg N ha^{-1}) in relation to other seasons (without significance analysis among seasons). Furthermore, rainfall in spring and winter (379 and 549 mm, respectively) was 58% of the total for the year (Table 4), resulting in a greater losses from soil available N, in agreement with Alfaro *et al*. (2007). Nitrogen was mainly lost as nitrate during all seasons ($P \leq 0.05$).

Nitrogen was mainly lost as nitrate during all seasons ($P \leq 0.05$), in agreement with Alfaro *et al*. (2005) and Wachendorf *et al*. (2005). The nature of both ions might explain these differences, since nitrate is readily susceptible to leaching because it is negatively charged, and therefore is not adsorbed by the clay and organic colloids, which are also negatively charged. On the dressing, the ammonium ion is positively charged and tends to be retained by cation exchanges (Whitehead, 2000).

Values reported in this investigation are lower than those reported as potential N losses in the area by Mora *et al*. (2007) on Andisol and estimated in irrigated and grazed pasture (88-90 kg N ha^{-1}) with 300 kg N ha^{-1} input as fertilizer and irrigated pastures in New Zealand (Di and Cameron, 2004; Ledgard *et al*., 1999). However, they are similar to those reported by Mora *et al*. (2004) and Alfaro *et al*. (2007) in non-irrigated and grazed pastures on volcanic soils in Chile.

Results of the present experiment showed that nitrate leaching was mainly affected by grazing intensity (FH and IH), this have been attributed to the N input in recycling and plant uptake through forage harvest by the animal like it was discussed previously.

Soil N availability

Soil available N increased after grazing stopped in all seasons (Figure 3), probably related to rapid mineralization of animal excrements, which in turn, enhances nutrient availability (Kooijman and Smit, 2001). However, this increased potential N losses by volatilization and leaching in the following seasons because of available N accumulation in the soil profile. In the non-grazed paddocks, soil available N decreased due to plant uptake and lack of recycled N.

Results showed that intensive grazing resulted in a greater proportion of bare soil, according with Demanet *et al*. (2006b) and Teuber *et al*. (2006), so that soil available N, this is, N coming from fertilizers or animal recycling can be lost by both, ammonia emissions and leaching processes, in agreement with Jarvis (1997).

In spring time, N availability in the soil was lower than in summer and autumn, and was related to greater dry matter availability and plant uptake at that time (Mora *et al.,* 2007; Bardgett *et al.,* 2007; Robson *et al.,* 2007). Furthermore, this was related to a greater number of grazing events per treatment and season since grazing produces changes in the quantity of N present in soil through recycling nutrients via dung and urine in accordance with the results of this study and those of Watson and Poland (1999). The greatest amount of soil N found in the IH treatment for all seasons (Figure 3) was related to a lower dry matter production, and plant uptake in that treatment resulting in higher soil available N (Figure 3). Changes in soil N balance were produced after grazing, in accordance with Watson and Poland (1999).

Our results showed that grazing frequency seems to control ammonia emissions while grazing intensity affects N leaching losses. Environmental conditions could increase or reduce N losses, that is, summer conditions with high temperatures and low soil moisture content resulted in higher ammonia emissions (autumn and summer). Winter conditions, i.e., high rainfall and soil moisture content resulted in greater N leaching losses (spring and winter) as in Alfaro *et al*. (2005; 2007). However, intensive grazing results in greater forage and animal production (McKenzie *et al*., 2006a; 2006b; Pavlů *et al.,* 2006) suggest that intensive dairy production in grazed systems in Southern Chile should consider these constraints in order to develop and adjust the Best Management Practices (BMP) for fertilizer application and grazing management to minimize negative environmental impacts.

CONCLUSIONS

Grazing strategy affected N dynamics in volcanic soil. Treatments managed under frequent grazing were 10% more, on average, than treatments managed under infrequent grazing, and 30% more than emissions from the no grazing control. Treatments grazed with the frequent heavy criterion (FH and IH) had 40% more N lost by leaching than light treatments (mean FL and IL) and 31% more than the control without grazing (C). Results showed that a higher grazing intensity resulted in greater N leaching losses (59 kg ha^{-1} yr^{-1}), and greater grazing frequency resulted in higher ammonia losses (40-41 ha^{-1} yr^{-1}).

From the agricultural point of view, the heavy frequency strategy is the most recommendable in terms of quantity and quality forage produced. However from an environmental stand point this system resulted in greater N losses so this constraint should be considered in dairy production in Southern Chile.

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RESUMEN

Pérdidas de nitrógeno bajo diferentes frecuencias e intensidades de pastoreo en un suelo volcánico del sur de Chile. Los sistemas chilenos de producción ganadera se han intensificado en los últimos años con el uso creciente de fertilizantes que aportan nitrógeno (N), creando el potencial de daño ambiental a través de la contaminación del agua y el aire con N, de manera que se han diseñado estrategias alternativas de producción con el objetivo de reducir este potencial impacto. El presente estudio busca determinar las pérdidas de N bajo diferentes frecuencias e intensidades de pastoreo en una pradera permanente (*Lolium perenne* L., *Festuca arundinacea* Schreb., *Dactylis glomerata* L., y *Trifolium repens* L.) en un Andisol del sur de Chile. Se evaluaron cuatro estrategias de pastoreo: frecuente-intenso (FH), frecuente-laxo (FL), infrecuente-intenso (IH), infrecuente-laxo (IL), y un control (C) o tratamiento sin pastoreo, cada uno con tres repeticiones en un diseño de bloques completos al azar. Las pérdidas de N por lixiviación fueron mayores en el tratamiento FH (59 kg N disponible ha^{-1}; $P \leq 0,05$) donde la mayor parte de las pérdidas ocurrieron en la primavera (39%). En promedio, las pérdidas de amoniaco (NH_3) fueron 10% más altas en los tratamientos frecuentes en relación a los tratamientos infrecuentes, ya que entre los pastoreos individuales FH, FL e IH no hubo diferencia significativa ($P \leq 0,05$). Los resultados indican que la frecuencia de pastoreo afecta las pérdidas por lixiviación mientras que la intensidad del pastoreo afecta las emisiones de amoniaco desde la pradera. El pastoreo con vacas de lechería en el sur de

Chile debería considerar estas limitaciones ambientales para asegurar una producción sustentable en el tiempo.

Palabras clave: amonio, pastoreo, lixiviación, nitrato, volatilización de amoniaco.

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