NITROGEN SOIL BUDGETS IN CONTRASTING DAIRY GRAZING SYSTEMS OF SOUTHERN CHILE, A SHORT-TERM STUDY

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ABSTRACT

In recent years, the intensification of livestock production in Southern Chile has resulted in a high potential for environmental damage through nitrogen (N) losses, creating the need for the evaluation of N flows from these systems. The aim of the research was to determine N budgets and N use efficiency in two grazing systems in Southern Chile. For this, inputs and outputs were measured during one year on two grazing systems (heavy grazing, HG; and light grazing, LG). Also, a control treatment with no grazing (C) was considered. The annual N soil budget was determined by the difference between all N inputs (Σ inputs) and all N outputs (Σ outputs). The results of the experiment indicate that HG treatments received the biggest N input (427, 359 and 288 kg N ha^{-1} yr^{-1} for HG, LG and C treatment, respectively), however this treatment also had the biggest N losses (406 kg N ha^{-1} yr^{-1}), with a nitrogen recovery efficiency from fertilizer of 71%. In addition, herbage dry matter yield (DM) was greater in the HG than in the LG and C treatments (10.4; 8.1 and 7.1 t N ha^{-1} yr^{-1}, respectively). Also, N concentration in the forage was higher in this treatment (2.9%) than in the LG (2.7%) and C (2.5%) treatments. The results indicate that HG increases N use efficiency in pastures in Southern Chile, increasing the herbage production and quality, but also increasing the potential for N losses to the wider environment. Farmers should consider this when choosing the appropriate grazing system.

Keywords: Nitrogen budgets, grazing, nitrogen use efficiency, nitrogen losses, pasture.

INTRODUCTION

The Araucania (37º to 39º S and 70 LW) and Los Lagos (39° to 44° S and 71 to 74 W) regions in Southern Chile have 7.5 and 9.2% of the total area for livestock production in Chile, respectively, producing 70% of the country's milk and 60% of it's meat (INE, 2007). Livestock production in the area is based on direct grazing on permanent pastures (Alfaro and Salazar, 2005). Over the recent years, these livestock systems have intensified through the use of increasing amounts of nitrogen (N) fertilizer, which can lead to N losses, potentially affecting the wider environment (Alfaro and Salazar, 2005).

Nitrogen can be lost from the pasture system through ammonia (NH_3) emissions, through other gaseous N emissions (dinitrogen, nitric oxide and nitrous oxide; primarily products of denitrification) and through nitrate or ammonium leaching (Addiscott, 2006). The fluxes of N from the pasture depend on environmental and soil conditions (Jarvis and Ledgard, 2002) and N losses also reduce the productive efficiency of the pasture (Ryden, 1986; Jarvis, 1993; Jarvis, 1997).

Nitrogen budgets have been used in New Zealand as a tool for the evaluation of the environmental impact of N fertilizer application in grazed areas (Ledgard *et al.,* 1999). In The United Kingdom they have also been used by Jarvis (1993) to compare N use efficiency between systems. Nitrogen budgets are also used as indicators to assess changes in soil fertility and to quantify soil nutrient status (Lesschen *et al.,* 2007).

In Chile, few N budget studies have been carried out in grazed systems. However, the importance of the different N processes in livestock farms has been reported. Alfaro *et al.* (2005b, 2007, 2009) obtained good results estimating N budgets to determine the soil fertility conditions, the nutrient use efficiency and the potential environmental impact of the grazing activity.

The quantification of the N inputs and outputs in grazing systems of Southern Chile is therefore of interest, in order to assess the best grazing management options to increase pasture production with reduced N losses to the environment. The aim of the research was to quantity N soil budgets and N use efficiency (NUE) in two grazing systems in the dairy production area of southern Chile.

MATERIALS AND METHODS

Experimental site and treatments

The study was carried out between the 21st September 2005 and the 23rd September 2006, on a permanent pasture (*Lolium perenne* L., *Festuca arundinacea* Schreb., *Dactylis glomerata* L., and *Trifolium repens* L.) on an Andisol of Southern Chile (38°50' S, 72°42' W, 70 m.a.s.l). The soil at the site belongs to the Freire soil Series (Typic Placudands; CIREN, 2003) and had a silty loam texture at the 0-20 cm soil layer. The soil chemical properties were determined according to the method described by Sadzawka *et al.* (2000). At the start of the experiment, the soil had a mean Olsen-P concentration of 16 mg kg^{-1} and an adequate nutritional level for grassland production (Table 1).

Two contrasting grazing treatments together with a no grazing or control (C) were selected from the field assay described in Núñez *et al.* (2010). Briefly, as follows: Herbage availability (as determined using a rising plate meter at the start of the grazing) was used as criteria for the two grazing treatments (Table 2). Grazing intensities were estimated by the residual pasture post-grazing height, this defined the 'heavy grazing' (HG) or 'light grazing' (LG) treatment as described by Núñez *et al.* (2010).

The number of grazings per treatments was eight and 10 for the LG and HG treatments, respectively. The average animal intake was 10.3 and 8.1 t DM ha^{-1} yr^{-1} in the HG and LG treatments, respectively. The grazing times were 26.7 and 20.2 h for the HG and LG treatments, respectively (Núñez, 2008). In the C treatment, grass was cut seven times during the experimental period when it reached 1400, 1400, 1150 or 1150 kg DM ha^{-1} depending on the season (Table 2).

Treatments were organized in a randomized block design (*n*=3), with each paddock being 165 m^2 in area. Paddocks were grazed with six Holstein-Friesian non-lactating dairy cows (mean liveweight of 400 kg), with an annual average stocking rate of 2.1 LU ha^{-1} yr^{-1}.

Table 1. Soil chemical properties by treatments during the period April 2004 to September 2006, Maquehue Station (0-10 cm). Values given are means for the period, with the standard error of the mean in parentheses (n = 3)*.

Property	Initial value **	Treatments***		
		C	HG	LG
pH (H_2O)	5.5 (0.07)	5.5 (0.06)	5.6 (0.09)	5.5 (0.07)
Olsen-P (mg kg^{-1})	16 (0.68)	15.8 (0.63)	14.8 (1.11)	15.5 (0.29)
K (mg kg^{-1})	250 (36.42)	355 (21.24)	352 (65.53)	229 (22.49)
OM (g kg^{-1})	120 (0.72)	132 (0.75)	117 (0.75)	111 (0.65)
Total N (g kg^{-1})	5.1 (0.18)	5.3 (0.13)	5.2 (0.16)	5.3 (0.31)
TOC (g kg^{-1})	69.6 (2.11)	76.6 (3.11)	67.7 (2.54)	64.7 (0.67)
K (cmol(+) kg^{-1})	0.6 (0.09)	0.9 (0.06)	0.9 (0.17)	0.6 (0.06)
Na (cmol(+) kg^{-1})	0.3 (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)	7.7 (0.46)
Mg (cmol(+) kg^{-1})	1.4 (0.11)	1.6 (0.14)	1.8 (0.12)	1.8 (0.08)
Al (cmol(+) kg^{-1})	0.2 (0.06)	0.3 (0.06)	0.3 (0.07)	0.5 (0.06)
Σ Bases (cmol(+) kg^{-1})	9.3 (0.42)	8.4 (0.34)	10.1 (0.42)	10.2 (0.51)
ECEC (cmol(+) kg^{-1})	9.5 (0.41)	8.6 (0.33)	10.4 (0.44)	10.6 (0.47)
Al saturation (%)	2.3 (0.64)	2.8 (0.59)	3.3 (0.66)	4.3 (0.67)

*Average annual soil analysis; ** April 2004; *** September 2006; ECEC, effective cation exchange capacity; OM, organic matter; TOC, total organic carbon; C, no grazing; HG, heavy grazing; LG, light grazing.

Between grazing periods, cows were maintained in a pasture outside of the experimental area.

Urea-N fertilizer was applied to all plots (230 kg N ha^{-1} yr^{-1}) distributed in five dressings (15th October and 15th November of 2005, 4th April, 8th May and 17th August (2006), with 46 kg N applied on each occasion. A split dressing (coincidentally with the urea application) of potassium magnesium sulphate ("Sulpomag") was also applied (22% K_2O, 18% MgO, 21.5% S, 2.5% Cl) at 100 kg ha^{-1} on each occasion. Triple super

Table 2. Dry matter availability (kg DM ha^{-1}) criteria for the starting and stopping of grazing for each treatment.

Season	Start/stop of grazing	Herbage availability (kg DM ha^{-1}) per treatment		
		C	HG	LG
Spring	Start	No grazing	2200	2600
	Stop	1400†	1200	1600
Summer	Start	No grazing	2000	2400
	Stop	1400†	1200	1600
Autumn	Start	No grazing	1500	1800
	Stop	1150†	1000	1300
Winter	Start	No grazing	1500	1800
	Stop	1150†	1000	1300

†Herbage removed by cutting; DM, dry matter; C, no grazing; HG, heavy grazing; LG, light grazing.

Phosphate (46% P_2O_5) was applied in autumn (4th April 2006) and winter (17th august 2006) at a rate of 200 kg ha^{-1}. Lime ("Magnecal 15") and gypsum (fertile gypsum, 18% S) were applied in March 2006 at a rate of 1000 and 500 kg ha^{-1}, respectively.

Weather conditions at the experimental site were registered at the Maquehue Station, placed within 1 km distance of the experimental site (Meteorology Direction of Chile 2005-2006). Total rainfall for the period was 1607 mm, with a daily mean 2.1-5.6 mm for the grazing season. Daily average was 5.6 mm d^{-1} in winter and 2.1 mm d^{-1} in summer. Average air temperature was greater in summer (14.6°C), with a range of -4 to 35.5°C. Average soil temperature was 18.2°C, ranging from 6 to 21°C. Soil moisture varied during the year, ranging from 8 to 80%. Wind speed varied between 0.1 and 16.2 m s^{-1}.

Nitrogen inputs

Four sources of N inputs were considered: atmospheric deposition, applied fertilizer (urea), N biological fixation (NBF) and recycling (N in dung, urine and plant). Deposition of inorganic N (availability NO_3^--N and NH_4^+-N) from rainfall was measured using collectors located in each paddock. The rainwater was regularly collected (approximately each 7-14 days) and analyzed (12-24 h immediately after the sampling) for available nitrate and ammonium concentration by the extraction and distillation analysis (Kjeldahl method; Sadzawka *et al.*, 2000). The cumulative N deposition was calculated from the N concentration determined in the sample and the volume of rainfall collected for each sampling period.

The chemical fertilization included the inorganic N added in fertilizer. Nitrogen biological fixation was estimated considering the proportion of white clover (*Trifolium repens L.*) in the pasture, as described by Ledgard *et al.* (2001).

Nitrogen recycling from animal excreta was estimated from the grazing time, the paddock area, the N concentration in dung and urine, and the frequency and volumes of deposition events (urine and dung). Frequency and volumes of urine and dung deposition events were obtained from a separated parallel experiment (Núñez*,* 2008). The N recycled by the plants was estimated considering a 75% grazing efficiency (consumed dry matter). The total N recycling was determined by adding the N recycled via animal (dung, urine) and plant (kg N ha^{-1} yr^{-1}).

Nitrogen outputs

Outputs considered were N losses (N_2O emission, NH_3 emission, N leaching) and N plant uptake.

Nitrogen losses. Nitrous oxide emission from soil was estimated using the factors proposed by the National Greenhouse Gas Inventory Committee (NGGIC, 2005) and the Intergovernmental Panel on Climate Change (IPCC, 1997) for three sources: fertilizer application, recycled N in dung and recycled N in urine (fertilizers (£ (%)=1.25, dung (£ (%)=0.5 and urine (£ (%)= 0.4 for dry cows, respectively).

Ammonia emissions were determined using static chambers of the same design as those used by Saggar *et al.* (2004) for measuring nitrous oxide emissions. The chambers were of PVC construction, with a diameter of 250 mm, a height of 210 mm and a removable lid with an airtight seal. The concentration of ammonium-N in the acid samples was determined spectrophotometrically using the indophenol blue method (Searle, 1984). Cumulative emissions for each season were obtained as the product of the total number of days during the season with the mean of the measured daily emission rates (kg N ha^{-1} yr^{-1}).

Nitrogen leaching losses were determined using lysimeter plots. Each lysimeter consisted of a PVC cylinder of 75 mm diameter and 550 mm of length. A nylon membrane (pore size of <0.02 mm) separated the soil column in the upper 450 mm of the cylinder from a leachate collection volume in the lower 100 mm (Núñez, 2008). A tube travelling from the leachate collection area to the soil surface allowed regular sampling.

The volume of leachate in each lysimeter was measured every 14 d and sub-samples were taken to determine nitrate and ammonium concentrations. These concentrations were measured by extraction and distillation according to the methodology described by Sadzawka *et al.* (2000). Nitrogen leaching losses were calculated as the N concentration determined in the leachates in each lysimeter and the volumen of leachate collected for each sampling period (kg N ha^{-1} yr^{-1}).

Plant uptake. Dry matter production was estimated cutting an area of 0.1132 m^2 in the pasture before and after grazing and when required in the control treatment. Samples were dried at 65°C for 48°C or until constant weight. Nitrogen plant uptake was determined in sub samples from those indicated previously and N concentration was determined by the digestion, distillation and titillation methodology according to Sadzawka *et al*. (2007).

Total plant uptake was calculated from N concentration in the forage and the annual production per treatment.

Nitrogen soil budget

The N budget was determined as the difference between N inputs (Σ inputs) and N outputs (Σ outputs).

Nitrogen use efficiencies (NUEs)

Two types of efficiencies were calculated: (1) Nitrogen recovery efficiency (NRE): a) NRE (%) N inputs = (N uptake by the plants/N inputs total)*100 and b) NRE (%) of fertilizer = (N uptake by the plants/ applied nitrogen fertilizer)*100; (2) Agronomic nitrogen use efficiency (ANUE): a) ANUE N inputs (kg DM ha^{-1} / kg N ha^{-1}) = produced kg DM / kg N total in the system) and b) ANUE N inputs fertilizer (kg DM ha^{-1} / kg N ha^{-1}) = produced kg DM / kg applied N fertilizer).

Statistical analysis

ANOVA was used to verify the effect of the treatments in the different parameters described previously used the JMP 5.0.1.2 software (SAS Institute, USA, 2002). Statistical differences of means (95% significance level) were distinguished using and mean separation Tukey's multiple range test ($P \leq 0.05$).

RESULTS AND DISCUSSION

Nitrogen inputs

The contribution of NBF was greater in the HG and LG treatments with 34 to 13 kg N ha^{-1} yr^{-1}, respectively, than the C treatment with 9.5 kg N ha^{-1} yr^{-1} ($P \leq 0.05$; Table 4). This can be related to the greater proportion of legumes in the grazed treatments with 429 (HG) and 192 (LG) kg DM ha^{-1} yr^{-1} than the control with 144 kg DM ha^{-1} yr^{-1} (Table 3).

Table 3. Dry mater production, nitrogen concentration, and legume production under different grazing strategies during the period 2005-2006. Values given are means with the standard error in parentheses (n = 3).

Parameter	Treatments		
	C	HG	LG
Total dry matter yield (kg DM ha^{-1} yr^{-1})	7107c (302.5)	10383a (284.6)	8174b (171.4)
Nitrogen concentration (%)	2.47b (0.253)	2.90a (0.195)	2.76a (0.195)
Crude protein (%)	15.42b (1.194)	18.15a (0.988)	17.21a (0.996)
Crude protein yield (kg CP ha^{-1} yr^{-1})	1095.8c (51.25)	1885.6a (82.01)	1406.8b (81.17)
Clover* annual average (%)	2.03b (0.684)	4.13a (1.826)	2.35b (0.985)
Clover** (%)	0.7-4.2	0.3-13	0.1-5.5
Total clover production (kg DM ha^{-1} yr^{-1})	144.3c (20.50)	428.8a (49.26)	192.1b (30.35)

Different letters within rows indicate significant differences ($P \leq 0.05$). DM, dry mater production; CP, crude protein; C, no grazing; HG, heavy grazing; LG, light grazing. * *Trifolium repens L.;* ** Values minima and maxima.

The total N contribution to the system was 288.1, 427.4 and 358.7 kg N ha^{-1} yr^{-1} for the C, HG and LH treatments, respectively ($P \leq 0.05$; Table 4). Nitrogen input in fertilizer application represented 54-64% of the total N input, reaching up to 80% of the total inputs in the C treatment). The contribution plant recycling and animals recycling represented between 30 and 37% of the N inputs in the HG and LG grazing, respectively (Table 4). The N recycled by the incorporation of plant and animal residues (plant + dung + feces) was superior in the HG compared to LG and C treatments, respectively ($P \leq 0.05$). The lowest N input was due to rainfall (1.3-1.6 % of the total), with a total contribution of 4-5 kg N ha^{-1} yr^{-1} (Table 4).

Nitrogen losses

Nitrogen leaching losses and gas emissions varied between 67-102 kg N ha^{-1} yr^{-1} and were greater in the HG treatment ($P \leq 0.05$; Table 4). These losses represented 26%, on average, of the N outputs in the livestock systems. The accumulated N losses during the year were significantly greater in the HG treatment ($P \leq 0.05$), with 58.7 kg N ha^{-1} yr^{-1} compared to LG and the C treatments, which did not differ among them. The annual losses of NH_3 were 31.2, 39.9 and 37.9 kg ha^{-1} yr^{-1} in C, HG, LG, respectively, with no differences between grazing treatments ($P \leq 0.05$). Nitrous oxide emissions represented between 1.3 to 1.4% of the N applied as fertilizer. The annual N leached represented between 13 and 14.5% of the N outputs of the system (Table 4), with the biggest loss in HG (14.5%).

Plant uptake

The dry matter production during the year varied between 7 and 10 t DM ha^{-1}, according to treatment. The average foliar N was among 2.47 to 2.90%, depending of the grazing system and the control treatment (Table 3).

Nitrogen budgets

The N budgets were positive in all three treatments (Table 4) varying between 22 and 63 kg N ha^{-1} yr^{-1}, being smaller in HG and bigger in LG and C treatment. However, the HG treatment showed the lowest surplus with 22 kg N ha^{-1} yr^{-1}, being followed by the LG treatment (63.2 kg N ha^{-1} yr^{-1}). The surplus in the LG treatment differed from that of the control ($P \leq 0.05$).

Nitrogen use efficiencies (NUEs)

Nitrogen recovery efficiency (NRE) and agronomic N use efficiency (ANUE) were greater in the HG than in the LG and the C treatment ($P \leq 0.05$; Table 4). This was related to a greater DM production and plant uptake in this treatment (Table 3).

DISCUSSION

Nitrogen inputs

Nitrogen deposition in grazing areas of Southern Chile has been reported as low as 3.3 kg N ha^{-1} yr^{-1} (Godoy *et al*., 2001) and much lower values (Godoy *et al*., 2003). In addition, Boeckx *et al*. (2004) indicate that in pristine forests from southern Chile, N deposition varies between 0.2-3.5 kg ha^{-1} yr^{-1}. Alfaro *et al*. (2005a) and Oyarzún *et al*. (2002) reported values of 3 and 5 kg N ha^{-1} yr^{-1}, respectively, similar to the values determined in this experiment. The differences shown in the values of nitrogen deposition (4.6 to 4.9) per treatments can be attributed to analytic error (instrumental), since rainfall distribution was uniform between treatments and N concentration was similar.

Table 4. Nitrogen soil budgets (kg N ha^{-1} yr^{-1}) and nitrogen use efficiencies (NUEs) for the different treatments during the period 2005-2006. Values given are means with the standard error in parentheses (n = 3).

Inputs and outputs		Treatments	
	C	HG	LG
Nitrogen inputs			
Atmospheric deposition*	4.6a (0.73)	4.9a (0.67)	4.7a (0.73)
Fertilizer	230a (0.00)	230a (0.00)	230a (0.00)
NBF	9.5c (1.07)	33.9a (0.78)	13.5b (0.43)
Recycling (dung + urine + plant)	44.0c (0.90)	158.6a (4.03)	110.5b (1.04)
Total N inputs	288.1c (3.46)	427.4a (3.46)	358.7 b (4.17)
Nitrogen outputs			
Denitrification**	2.9a (0.000)	3.2a (0.004)	3.1a (0.001)
Volatilization***	31.2b (0.71)	39.9a (0.38)	37.9a (0.42)
Leaching*	33.2b (0.95)	58.7a (2.45)	32.1b (1.59)
Total N losses	67.3b (0.99)	101.8a (2.55)	73.1b (1.98)
Plant uptake	175.7c (5.8)	303.7a (12.25)	222.4b (3.88)
Total N outputs	243.0c (6.16)	405.5a (9.85)	295.5b (2.13)
Nitrogen surplus	45.1b (1.87)	21.9c (7.10)	63.2a (5.83)
Efficiency type			
NREF (%)	76.4c (10.06)	132.0a (8.54)	96.7b (12.97)
NRE inputs (%)	61.0b (5.46)	71.1a (5.33)	62.0b (7.35)
ANUE fertilizer (kg DM ha^{-1} / kg N ha^{-1})	30.9b (4.14)	45.1a (5.47)	35.5b (6.41)
ANUE inputs total (kg DM ha^{-1} / kg N ha^{-1})	24.7a (3.35)	24.3a (1.93)	22.8a (3.62)

Different letters among rows indicate significant differences (*P*≤0.05). C, no grazing; HG, heavy grazing; LG, light grazing; * NO_3^- -N + NH_4^+-N; **N_2O-N; *** NH_3 -N; NRE, nitrogen recovery efficiency; NREF, nitrogen recovery efficiency fertilizer; ANUE, agronomic nitrogen use efficiency. Different letters among the rows indicate significant differences (*P*≤0.05).

The fertilization of the pasture used in the present experiment for dairy production was high compared to the fertilization applied in beef production systems of Southern Chile, this is 150-200 kg N ha^{-1} yr^{-1} (Alfaro *et al*., 2006; Alfaro *et al*., 2007).

The contribution of NBF was greater in the HG and HL, respectively (Table 4). This can be related to the greater proportion of legumes in the grazed treatments (Table 3), because this practice favors light penetration to the pasture, which in turn, favor light incidence on the clover growing points (Ledgard *et al*., 2001).

The contribution of NBF in HG was about 8% compared to a 3.8% and 3.3% in the LG and C treatments, respectively. The results obtained in HG are comparable to the ones obtained by Urzúa (2005) in permanent pastures of Southern Chile (30 kg N ha^{-1} yr^{-1}). In natural pastures, fixation is usually lower (15 kg N ha^{-1} yr^{-1}), because of lower proportions of clover in the pasture, in agreement with the results obtained in the LG treatment.

Greater amounts of N recycled to the pasture, as feces and urine, were found in the HG treatment ($P \leq 0.05$), with values of 83 kg N ha^{-1} yr^{-1}, compared to the 55 kg N ha^{-1} yr^{-1} recycled in the LG treatment. This could be attributed to a greater number of grazing registered during the season in this treatment, which resulted in a greater grazing period, a greater DM animal intake, a greater number of dung and urine patches and thus, a greater N return to the soil, in agreement with Reyes *et al*. (2000). These N inputs to the pasture are vital for an optimal production and also as a compensation process of N losses by plant uptake, animal uptake, gaseous N and N leaching.

The N recycled by the incorporation of plant and animal residues (plant + dung + feces) was superior in the HG compared to the LG and C treatments ($P \leq 0.05$). This is attributed to the dry matter yield of this treatment, and the volume of residue generated. Recycling values measured in the present experiment are lower than those reported by Haynes and Williams (1993), Whitehead (2000) and Ledgard *et al.* (1998, 1999). This could be attributed to the higher stocking rate used in the New Zealand experiments and the animal handling used in our case, where animals were mostly managed outside the experimental paddocks, so that the rumination of the ingested grass was deposited in other areas.

Nitrogen losses

The annual losses of NH_3 were greater in the grazing systems (37.9-39.9 kg N ha^{-1}) than in the control (31.2 kg N ha^{-1}) (Table 4). Jarvis (1993) indicates that the NH_3 emission in England was 46 kg N ha^{-1}. Similarly, Ledgard *et al.* (1998), in New Zealand, reported losses of 41 kg N ha^{-1} yr^{-1} with a dose of 200 kg N ha in pasture under irrigations. However, Jarvis and Ledgard (2002) in comparative study in dairy farms from UK and New Zealand system reported great differences in NH_3 losses attributed mainly to the cows winter housing. While in UK the NH_3 emission reached 57.2 kg ha^{-1} in New Zealand the NH_3 losses was 24 kg ha^{-1}. Comparing the results obtained in this experiment with the results reported by Ledgard *et al*. (1998, 1999) and Jarvis and Legard (2002), a similarity was observed in the emissions NH_3 produced in dairy pastures with New Zealand studies. The results suggest that a change in the grazing intensity, independently of the stocking rate, has an effect in the levels of NH_3 emissions and therefore in a reduction in the quantity of available N for plant uptake. This has an effect in turn on the quantity and quality of the pasture.

Volatilized ammonia represented between 10 and 13% of the N total outputs in the pasture during the year. In addition, NH_3 losses represented between 13 and 17% of the N applied as urea, in agreement with the 20% indicated by Sommer and Jensen (1994) for fertilizer applied in grassland soils. Ammonia emissions in grazing systems in template areas represented between 8 and 9% of the total N inputs to the system (Bouwman *et al*., 2005), which also matches with this experiment.

The accumulated N losses during the year were significantly different (Table 4). Results of the present experiment suggest that grazing has a direct effect on N leaching losses from the pasture, in agreement with Ryden *et al*. (1984). The annual losses of leached N obtained in this experiment are low compared to what reported Di and Cameron (2004) and Ledgard *et al*. (1999) in irrigated pastures managed under grazing in New Zealand and Australia, respectively, because of differences in stocking rates and N application as fertilizer (both greater in the New Zealand studies).

The losses produced by leaching could be influenced by edaphic factors, local site history, in agreement with Ledgard *et al*. (1998) and the amount of rainfall, and also by the applied dose of fertilizer, in agreement with Mora *et al*. (2007). Another factor that affects leaching is the stocking rates used (Ledgard *et al*., 1999), but not in our case, due to the stocking rate was used was the same in both grazing systems. Alfaro *et al*. (2005b, 2006, 2007) indicate that the amount of OM, soil type and fertilizer application rate influence the volume of N leached; also N mineralization rate from urea could affect the amount of N leaching losses (Cartes *et al*., 2009).

The losses of nitrous oxide from urine, dung and fertilizer sources were low because the low percentage assigned to the N recycled as feces and urine in the methodology used (Table 4). In experiments carried out by Phillips *et al*. (2007), where the same estimation factors of N_2O were used, similar emission values were reported for the fertilizers (2.81 kg N_2O ha^{-1} yr^{-1} with a rate of 225 kg N ha^{-1} yr^{-1} applied) in temperate pastures, but emission values from urine and feces were higher (1.9-2.4 and 0.5-0.6 kg N_2O ha^{-1} yr^{-1} for urine and feces, respectively. This could be because the previous study used a stocking rate 2.4 times greater than that in the present study (300 dairy cows over 24 h at intervals of 14–21 days; Phillips *et al*. 2007).

Plant uptake

The pasture DM yield varied between 7 and 10 t DM ha^{-1}. The results indicate that the pasture yield varied with the grazing systems (Table 3). Greater production was obtained in the HG treatment ($P \leq 0.05$; Table 3). This means that a more intensive grazing results in a higher DM production, in agreement with McKenzie *et al*. (2006a). This treatment had the bigger yield results and the biggest N plant uptake N, but also had the higher N losses, having therefore a bigger potential for environmental pollution.

The herbage quality of the pasture was affected by the grazing systems applied; being in the two cases higher for HG and lower in the LG treatment (Table 3).

In grazed pasture, N concentration increased significantly with respect to the no grazed pasture (C), reaching in HG 1885.6 kg of crude protein per hectare per year. Although with did not observe differences in the protein concentration among HG and LG, the protein produced is clearly greater in HG due to its higher DM yield. In this sense, McKenzie *et al*. (2006a, 2006b) demonstrated that an intense and frequent grazing modifies the structure and botanic composition of the pasture, increasing its quality. Similar results were shown by Ru and Fortune (1996, 2000) in a template pasture under irrigation.

Nitrogen budgets

The treatments with less N plant uptake had a greater soil N surplus, being this the main factor that affect N accumulation in the soil, in agreement with Alfaro *et al.* (2003). The LG and the C treatments were have less efficient in converting soil N into DM (Table 4), with a NRE by the plants of 62%, on average, from the total inputs, compared to HG with 71%. A percentage of NRE by the plants ≥ to 65% of the N inputs to the system is considered optimum (Baligar *et al*., 2001; Delgado, 2002) depending on the crop systems and N rate applied. The global average reported by NRE is 50% (Baligar *et al*., 2001; van Es and Delgado, 2004). Considering this, grazing HG and LG were optimally efficient compared to the ranges established in literature.

Alfaro *et al.* (2005a) in a field study in a permanent pasture of Southern Chile reported negative balance results of N for the different treatments. The experiment used 3.5 calves as stocking rate (Holstein-Friesian with 212 kg initial). However, this N balance did not include the plant recycling, and the N fertilization applied was low (67.5 kg N ha^{-1}). In this case, the N deficit was associated to the lack of consideration of the OM mineralization and to a low fertilization. Therefore, this balance could be positive if it would be considered these variables and with this the results would have had the same tendency than that of our experiment.

Nitrogen use efficiencies (NUEs)

Nitrogen recovery efficiencies were greater in the grazing treatments. This was related to a greater DM production and plant uptake in this treatment (Table 3). This was due to the fact that an intense grazing reduces the amount of dead matter and senescence, facilitating the growth of new tillers, and therefore, a faster recovery of the pasture, which in turn, results in a greater N plant uptake and use. The efficiencies have relation with the total N inputs, however, the ANUE were not statistically different between treatments in agreement with the reported by (Baligar *et al*., 2001). These efficiencies are high compared to grazing systems from the United Kingdom, where 41-56% was reported during a period of seven years of evaluation (Leach *et al*., 2004). In this sense, Ledgard *et al.* (1999), in New Zealand pastures, reported a efficiency (N in product/ N inputs) of 30%, 20% in England (Jarvis, 1993; Ledgard *et al*., 1999), 14% in Holland and 23% in Switzerland (Ledgard *et al*., 1998) in pasture template. This way, the NRE of the pasture is higher because the N efficiency was not calculated based on the production of milk, meat or crop, which explains the differences between the efficiencies shown. As it was discussed in the section about the nitrogen surplus was bigger in the treatments with smaller NRE and smaller in the most intense grazing, this suggests that the plants used the N more efficiently on HG; however this increases the possibilities of N losses in this grazing type and therefore a bigger negative impact.

The agronomic efficiency of the pasture depends on the applied N to the plants from the soil and on the environmental conditions, in agreement with Whitehead (2000). The values showed by ANUE from the fertilizer applied is similar to that reported by Whitehead (2000), fluctuating between 20-30 kg DM ha^{-1} yr^{-1} at a dose of 250-400 kg N ha^{-1} yr^{-1}.

According to these results, the HG treatment would the best grazing management for dry matter yield, quality and NUE. Nevertheless, this treatment also showed the greater N losses to the environment, in comparison to the LG treatment which was similar to the C treatment.

Alternatives to reduce the potential environmental impact of this grazing management would be to adjust fertilizer N input to reduce N surpluses, to avoid N fertilizer application during rainy periods, to match N pasture demand with N fertilizer application and to avoid overgrazing in winter.

CONCLUSIONS

The heavy grazing (HG) treatment had larger nitrogen recovery efficiency (NRE) than the LG and control treatments (71, 62 and 61% for the HG, LG and C treatments, respectively).

Heavy grazing system facilitated the faster pasture recovery, which in turn, resulted in a greater dry matter production. Thus, the HG treatment produced a 27% more of dry matter production than the LG treatment (10383 and 8174 kg DM ha^{-1} yr^{-1} for both treatments, respectively).

A positive N budget in the treatments suggests N accumulation in the soil at the end of the grazing season because of the high N inputs, especially in the HG treatment (427 kg N ha^{-1}). Consequently, the highest input of N in HG system resulted in N total losses greater by 40% than LG and 52% than no grazed systems. According to our results, the HG treatment would the best grazing management for dry matter yield, quality and nitrogen use efficiency. Thus, we recommend that farmers should consider these environmental constrains when implementing intensive grazing strategies.

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