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Role of Liming and Nitrogen Input in Determining Low Nitrogen Leaching Losses in Volcanic Soils

Marta A. Alfaro¹ · Laura Cardenas² · Francisco Salazar³ · David Hatch² · Luis Ramírez³

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Abstract

Purpose Livestock production based on grazing is the most important productive activity in southern Chile. Most grasslands are located on volcanic soils with unexplained low nitrogen (N) leaching losses.

Methods A cutting field experiment was carried out between February 2008 and March 2010 on an Andisol of the Osorno soil series, testing liming (0 or 2 t CaCO₃ ha⁻¹ yr⁻¹) and N rate applied as inorganic fertiliser applied in two equal dressings (0, 200, 400 kg N ha⁻¹ yr⁻¹). Measurements of pasture yield, N uptake and N leaching losses were complemented with mineralization studies under field and laboratory conditions, and soil N abiotic retention.

Results Total N leaching losses were greater in the 400 kg N treatments (90 kg N ha⁻¹, on average), with no differences between the liming treatments (10 kg N ha⁻¹, on average; $P > 0.05$). Nitrogen addition resulted in 50% and 92% more dry matter yield in treatments receiving 200 and 400 kg N than the control (N-0) treatment. Potential N mineralization varied between 206 and 282 mg N kg⁻¹ dry soil. The mineralization of more recalcitrant components of soil organic matter increased with both N and lime addition. Soil had a greater capacity to retain added N as ammonium (74% on average).

Conclusions The natural acidic soil conditions did not prevent nitrification, neither N leaching losses, probably associated to occurrence of N abiotic immobilization processes. Lime addition could influence N mineralization in the long term, while N addition would influence it in both short and long term.

Keywords Volcanic soil · Lime · N fertiliser application · N retention · Mineralization

1 Introduction

The most important agricultural activity in southern Chile is livestock production based on direct grazing (dairy and beef), generating 85% and 50% of the national milk and beef production (INE 2007). Grasslands used for livestock production in the region are located mainly on volcanic ash soils (andisol, ultisol; Alfaro et al. 2008a, b). They are characterised by low nutrient availability, high phosphorus (P) fixation capacity, high organic matter (OM) content (e.g.

from 10% up to 40%), low pH and a pH-dependent cation exchange capacity (Escudey et al. 2001). Liming is thus a recommended practice in the area to reduce soil acidity effects on pastures and to improve animal production (Mora et al. 2006). In this area, the use of nutrients (nitrogen, N and P in fertilisers and animal feed) has increased over the last twenty years (Núñez et al. 2010; Alfaro et al. 2021). This, in turn, has resulted in higher stocking rates with consequential higher demand of forage for grazing animals. Increases in total production per hectare and total number of animals in production have resulted in more N fertiliser being applied, with likely negative impacts for the soil, water and air.

European soils that are acidic and relatively low to medium in organic matter content (5–10%) have a large potential for nitrification after the addition of N fertiliser, especially during the autumn, which results in a high potential for leaching losses (Cardenas et al. 2019). In volcanic soils with high organic matter content, N leaching losses are low in mown grasslands, and have little variation compared

✉ Marta A. Alfaro
marta.alfaro@agresearch.co.nz

¹ AgResearch Ltd, 10 Bisley Road, Hamilton
3214, New Zealand

² Rothamsted Research North Wyke, Sustainable Agriculture
Systems, Okehampton, Devon EX20 2SB, UK

³ Instituto de Investigaciones Agropecuarias, INIA Remehue,
PO Box 24-O, Osorno, Chile

to un-fertilised areas, despite heavy N applications (e.g. Salazar et al. 2012). This could be related to the natural acidic conditions of these soils, or the high organic matter content and the quality of this organic matter, affecting nitrification in the soil.

In forest soils of southern Chile that are high in organic matter and have low pH, leaching losses are also low being associated to both ammonium abiotic fixation in the clay fraction of the soil (Kizewski et al. 2019) and to the dissimilarity nitrate reduction to ammonium (DNRA) process (Huygens et al. 2007). There is little information available on these soil dynamics for grassland areas in southern Chile (Cardenas et al. 2013; Alfaro et al. 2018, 2022; Chinchilla-Soto et al. 2022). Previous studies have shown that N leaching losses from mown grassland soils are low at inorganic or organic N fertiliser application rates of 60–200 kg N ha⁻¹ yr⁻¹ (dairy slurry, fish sludge), and that these losses are similar to those measured in control areas with no N addition (Alfaro et al. 2006b, 2008a, b, 2009; Núñez et al. 2010; Salazar et al. 2012).

Determination of the mechanisms affecting the low N leaching losses in andisols would contribute to the current scientific knowledge on N cycling in volcanic soils and improve the efficiency of N use in livestock systems. Thus, the objective of this study was to determine N leaching losses in a volcanic soil receiving high N inputs as inorganic fertiliser, with and without lime application. We hypothesized that: (1) the natural acidic soil conditions prevent nitrification, (2) the high organic matter content of the soil and high microbial activity favour N immobilization, and (3) ammonium and/or nitrate are retained in the soil due to specific physico-chemical properties of the andisol studied.

2 Materials and Methods

2.1 Experimental Site

An experiment was carried out at the Instituto de Investigaciones Agropecuarias (INIA), Remehue Regional Research Centre (40°35' S, 73°12' W, 72 m above sea level), between February 2008 and March 2010. Soil at the site is an Andisol of the Osorno soil series (Typic Hapludands; Ciren 2003), which has 2% slope, > 1 m depth, 5.7 pH (water), 18% organic matter content and high Olsen P (c. 30 mg kg⁻¹) (0–10 cm depth, Table 1). The mean annual rainfall is 1,242 mm (40-year average).

Soil at the site has been under permanent grassland for the last 20 years. Typically, the main species in the pasture are *Lolium perenne* L., *Dactylis glomerata* L. and *Holcus lanatus* L. Legumes and weeds were eliminated from the experimental plots with the use of a selective herbicide (Picloram

240 g a.i. L⁻¹, 250 cm⁻³ 200 L⁻¹, January 2008). In winter each year, an insecticide was applied to avoid problems with black caterpillar (Lambda-cyhalothrin 25% g a.i., 150 cm⁻³ 200 L⁻¹) (21st of June 2008 and 23rd of July 2009).

2.2 Treatments

The effect of two factors on leaching was tested under cutting management: the application of a fixed amount of lime (0, 2 t CaCO₃ ha⁻¹ yr⁻¹), and varying inputs of inorganic N fertiliser (0, 200, 400 kg N ha⁻¹ yr⁻¹). The amount of lime to apply was determined according to soil chemical analysis, to ensure optimum conditions for nitrification in the soil (pH 6.5 or over). Plots (6 × 5 m) were distributed in a randomized design ($n=4$). Half of each plot was used to determine leaching losses and the other half for the mineralization determinations (described below). Sodium nitrate (NaNO₃, 16% N) was used as the N source to avoid N losses via ammonia (NH₃) volatilization. Nitrogen fertiliser was applied in two equal dressings, in autumn (7th and 19th of March 2008 and 2009, respectively), and in spring (3rd and 28th of September 2008 and 2009, respectively). Lime was overcast in summer to avoid interactions with any recent P fertiliser applications (30th and 29th of January 2008 and 2009, respectively). All plots received a basic fertiliser application in autumn each year consisting of 52 kg P ha⁻¹ yr⁻¹ (Triple superphosphate, 46% P₂O₅); 83 kg K ha⁻¹ yr⁻¹ (Potassium chloride, 62% K₂O); 23 kg Mg ha⁻¹ yr⁻¹ (Magnesium Oxide; 85% Mg) and 20 kg S ha⁻¹ yr⁻¹ (Gypsum, 18% SO₄). All fertilisers were surface applied by hand (7th and 19th of March 2008 and 2009, respectively).

2.3 Meteorology, Soil and Pasture Sampling, and Data Analysis

Daily rainfall and evapotranspiration data during the experimental period was recorded with an automatic weather station placed within 800 m of the experimental site.

Each year of the experiment, soil chemical characterization was carried out in treatment-bulked samples from individual samples taken randomly ($n=4$) in late summer each year (0–10 cm depth) from each replicate plot. The samples were air dried (30 °C) and passed through a 2 mm sieve prior to analysis. Soil parameters (P, N, S, water pH, organic matter, Ca²⁺, K⁺, Mg²⁺, Na⁺, cation exchange capacity, Al³⁺ and Al saturation index) were determined by the standard methodology compiled by Sadzawka et al. (2006). Briefly, soil pH was measured in water and CaCl₂ solution by potentiometry, organic matter concentration was estimated using a modified Walkley-Black method by wet digestion; exchangeable cations (Ca²⁺, K⁺, Mg²⁺ and Na⁺) and exchangeable Al were extracted with a solution of 1 M

Table 1 Soil chemical analysis at the experimental site in 2008 (31/07, initial), 2009 (20/02) and 2010 (2/02) for N fertiliser ($\text{kg N ha}^{-1} \text{ yr}^{-1}$) and lime application ($\text{t ha}^{-1} \text{ yr}^{-1}$) treatments (0–10 cm depth, $n=4$, \pm standard error of the mean)

Treatments	P Olsen (mg kg^{-1})	Available N	S	pH (Water)	Ca ($\text{cmol} + \text{kg}^{-1}$)	K	CEC	Al	Al saturation (%)	Organic matter ($\text{g } 100 \text{ g}^{-1}$)	Organic C (%)	Total N	Total C	C/N ratio
2008														
Parameter	12 \pm 0.8	17 \pm 2.2	8 \pm 0.1	5.7 \pm 0.01	2.0 \pm 0.05	1.3 \pm 0.13	3.1 \pm 0.19	0.3 \pm 0.02	6.8 \pm 0.71	1687 \pm 181.4	9.8 \pm 0.10	1.3 \pm 0.01	9.6 \pm 0.06	7 \pm 0.07
2009														
0 Lime–0 N	30 \pm 7.9	60 \pm 9.2	10 \pm 1.6	5.4 \pm 0.03	5.9 \pm 1.40	1.7 \pm 0.31	9.7 \pm 1.83	0.2 \pm 0.05	2.9 \pm 0.83					
0 Lime–200 N	18 \pm 3.4	73 \pm 10.3	7 \pm 2.3	5.5 \pm 0.07	5.9 \pm 0.84	1.0 \pm 0.10	9.0 \pm 1.00	0.2 \pm 0.05	2.9 \pm 0.86					
0 Lime–400 N	20 \pm 4.0	131 \pm 14.4	5 \pm 1.17	5.4 \pm 0.12	4.6 \pm 1.14	1.1 \pm 0.25	8.7 \pm 1.68	0.3 \pm 0.08	4.5 \pm 1.54					
2 Lime–0 N	24 \pm 6.3	56 \pm 10.0	9 \pm 1.4	5.6 \pm 0.05	7.8 \pm 0.75	1.5 \pm 0.29	10.9 \pm 0.82	0.1 \pm 0.03	1.4 \pm 0.38					
2 Lime–200 N	19 \pm 5.7	58 \pm 14.1	13 \pm 1.8	5.5 \pm 0.09	5.6 \pm 0.66	0.8 \pm 0.22	8.2 \pm 0.75	0.3 \pm 0.03	3.3 \pm 0.49					
2 Lime–400 N	19 \pm 5.1	123 \pm 17.6	5 \pm 2.3	5.5 \pm 0.07	6.3 \pm 0.99	1.0 \pm 0.25	10.5 \pm 1.18	0.2 \pm 0.04	2.0 \pm 0.74					
2010														
0 Lime–0 N	61 \pm 3.0	13 \pm 1.38	9 \pm 0.3	5.8 \pm 0.05	4.6 \pm 0.25	1.0 \pm 0.07	7.3 \pm 0.53	0.2 \pm 0.03	3.4 \pm 0.66					
0 Lime–200 N	44 \pm 0.8	14 \pm 1.3	6 \pm 0.6	6.2 \pm 0.02	4.1 \pm 0.15	1.0 \pm 0.07	7.8 \pm 0.23	0.2 \pm 0.02	2.3 \pm 0.33					
0 Lime–400 N	52 \pm 4.5	23 \pm 4.3	7 \pm 0.8	6.3 \pm 0.03	4.6 \pm 0.37	0.8 \pm 0.09	8.8 \pm 0.66	0.2 \pm 0.03	1.9 \pm 0.53					
2 Lime–0 N	54 \pm 2.6	20 \pm 1.8	6 \pm 2.5	6.6 \pm 0.06	16.2 \pm 1.04	1.0 \pm 0.09	19.1 \pm 1.01	0.1 \pm 0.01	0.4 \pm 0.03					
2 Lime–200 N	46 \pm 4.2	21 \pm 2.8	5 \pm 0.5	6.7 \pm 0.11	15.1 \pm 2.22	0.7 \pm 0.08	18.4 \pm 2.34	0.1 \pm 0.01	0.4 \pm 0.14					
2 Lime–400 N	45 \pm 1.6	24 \pm 3.8	8 \pm 0.8	6.8 \pm 0.03	15.2 \pm 1.1	0.7 \pm 0.08	19.4 \pm 1.29	0.1 \pm 0.01	0.4 \pm 0.07					
Overall														
0 Lime–0 N	45 \pm 7.1	36 \pm 9.9	9 \pm 0.8	5.6 \pm 0.07	5.2 \pm 0.71	1.4 \pm 0.20	8.4 \pm 0.99	0.2 \pm 0.03	3.1 \pm 0.50					
0 Lime–200 N	31 \pm 5.1	45 \pm 12.0	6 \pm 1.1	5.8 \pm 0.14	5.0 \pm 0.52	1.0 \pm 0.06	8.4 \pm 0.53	0.2 \pm 0.03	2.6 \pm 0.43					
0 Lime–400 N	36 \pm 6.5	76 \pm 21.6	6 \pm 1.0	5.8 \pm 0.19	4.6 \pm 0.56	1.0 \pm 0.13	8.8 \pm 0.84	0.3 \pm 0.05	3.2 \pm 0.89					
2 Lime–0 N	39 \pm 6.5	34 \pm 9.5	7 \pm 1.4	6.1 \pm 0.19	11.9 \pm 1.70	1.3 \pm 0.17	15.0 \pm 1.65	0.1 \pm 0.02	0.9 \pm 0.26					
2 Lime–200 N	32 \pm 5.8	40 \pm 9.5	9 \pm 1.7	6.1 \pm 0.22	10.3 \pm 2.10	0.8 \pm 0.11	13.3 \pm 2.24	0.2 \pm 0.04	1.9 \pm 0.59					
2 Lime–400 N	32 \pm 5.6	74 \pm 20.4	7 \pm 1.2	6.1 \pm 0.24	10.8 \pm 1.80	0.9 \pm 0.13	14.9 \pm 1.86	0.1 \pm 0.03	0.2 \pm 0.47					
Significance														
Year (Yr)	***	***	NS	***	***	**	***	***	***					
Lime (L)	NS	**	NS	***	***	NS	***	***	***					
Nitrogen (N)	**	***	NS	***	NS	**	*	NS	NS					
Yr x L	NS	*	*	***	***	NS	***	NS	NS					
Yr x N	NS	***	*	**	NS	NS	NS	NS	NS					
L x N	NS	NS	NS	*	NS	NS	NS	NS	NS					
Yr x L x N	NS	NS	NS	NS	NS	NS	NS	NS	NS					

Significant differences are shown as *: $P \leq 0.05$; **: $P \leq 0.01$; ***: $P \leq 0.001$; NS: not significant ($P > 0.05$)

ammonium acetate ($\text{NH}_4\text{C}_2\text{H}_3\text{O}_2$) at pH 7.0 and 2 M potassium chloride (KCl), respectively, and analysed by atomic absorption spectrophotometry (AAS). Sulfate (SO_4^{2-}) and phosphate (PO_4^{3-}) anions were extracted in a solution of 0.5 M sodium bicarbonate (NaHCO_3) at pH 8.5 and 1 M calcium dihydrogen phosphate ($\text{Ca}(\text{H}_2\text{PO}_4)_2$) and analysed by the Murphy and Riley method and turbidimetry, respectively (Tabatabai 1982).

Measurements of NO_3^- leaching were carried out using the ceramic suction cups technique described by Lord and Shepherd (1993), which has been shown to be suitable for freely draining soils (Webster et al. 1993), such as that used in this study. Ceramic cups were placed at a depth of 60 cm in the soil (three replicates per plot, $n = 12$ per treatment) at an angle of 30° to the vertical. A mixture of soil and water slurry was introduced around the ceramic cup and pipe to ensure a good contact between the soil and the cup, and to prevent the preferential flow of water down the tube. Only NO_3^- was considered for leaching losses analysis, as previous studies in this soil have shown that this is the main form of N lost in this water pathway ($>90\%$) (Alfaro et al. 2008a, b). Individual samples were collected by applying a suction of 0.7 bars to the suction cups and removing any sample that was collected overnight. Samples were taken periodically during autumn and winter each year, every *c.* 100 mm of incremental drainage, and placed in 125 ml polyethylene bottles. All individual collections were frozen until analysis for nitrate-N (NO_3^- -N) by ion chromatography (Standard Methods 2005). Drainage between the sampling periods was estimated by subtracting potential evapotranspiration figures from rainfall using meteorological data collected close to the experimental site. Evapotranspiration for the pasture was calculated using the Penman-Monteith method (Penman 1948). The amount of NO_3^- -N leached over the period (kg N ha^{-1}) was then calculated according to the trapezoidal rule proposed by Lord and Shepherd (1993), as.

$$\text{N leached} = 0.5 (c1 + c2) v / 100 \quad (1)$$

where N leached (kg N ha^{-1}) is estimated from $c1$ and $c2$ which are the NO_3^- -N concentrations (mg L^{-1}) on successive sampling periods and v is the drainage volume produced (mm) between these periods. The total for the drainage season was calculated by the sum of all the sampling occasions. The mean concentrations were obtained by dividing the total NO_3^- -N loss by the total drainage and multiplying by 100 (Lord and Shepherd 1993). Total N leaching losses were calculated as the product of drainage and N concentration in the respective samples.

Grass dry matter (DM) yield was measured by direct harvest when the pasture reached *c.* 15 cm height, leaving a residue of 5 cm. The fresh material was weighed and mixed,

and 200 g then oven dried at 60°C for 48 h or until constant weight to estimate the DM concentration of the pasture (Sadzawka et al. 2007). Dry matter yield was expressed as $\text{kg DM ha}^{-1} \text{ yr}^{-1}$. Dry samples were ground in a Wiley mill (1 mm sieve) for later analysis of total N (Kjeldahl; AOAC 1970). Using this information and the total DM yield, the total plant N uptake for the relevant period was calculated. This information was also used to estimate N productive efficiency, as $\text{kg DM kg}^{-1} \text{ N}$ applied.

The soil N supply in the control and $400 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ treatments (limed and non-limed) were determined from soil samples taken fortnightly for N mineralization determinations. The soil samples were incubated (20°C) for 14 days with acetylene according to the methodology described by Antil et al. (2001) and Hatch et al. (1990) (day 0= t_0). Once the incubation period was finished (t_1), 20 g of moist soil was extracted with 100 ml of 2 M KCl, filtered through Whatman paper N°1 and analyzed for available N according to Kamphake et al. (1967). The amount of NH_4^+ found in the soil solution at t_1 , minus that at t_0 , was considered as the amount of N mineralized over the period.

2.4 Laboratory Incubations

The field experiment was complemented with laboratory incubations with the objective of determining potential N mineralization and nitrification of soil (0–10 cm depth) under optimum temperature conditions (20°C), according to the methodologies of Stanford and Smith (1972), Lober and Reeder (1993) and Dusek (1995), respectively. The amount of NH_4^+ and NO_3^- physico-chemically retained were also determined according to the methods described by Lumbanraja and Evangelou (1994), Thompson and Blackmer (1992), and Wild (1972), to investigate if low N leaching losses were partially related to these processes. To achieve this, soil samples from plots of each treatment (same four replicates) were collected on the 25th of January 2010, dried at ambient temperature (25°C) and sieved passed a 2 mm sieved prior to the following determinations.

Potential N mineralization. According to Stanford and Smith (1972), the equivalent of 25 g of dry soil was incubated in 200 ml bottles for up to 20 weeks at 70% of water holding capacity (WHC) at 25°C . Extractions were carried out at days 0, 1, 3, 7, 14, 28, 56, 84, 112 and 140. At each time, mineral N was extracted using 50 ml of 2 M KCl, as described previously. Each week, all bottles were weighed, and the missing water replaced with deionized water. The first order kinetic model was fitted to the data to give the potential mineralizable N (N_0), the mineralization rate constant (k) and N_t , the N mineralised in the time interval t , where $N_m = N_0 (1 - e^{-kt})$.

Potential anaerobic N mineralization. We followed the methodology proposed by Lober and Reeder (1993). Briefly, 5 g of air-dried soil was weighed and placed in 60 ml plastic syringes to which 13 ml of deionised water was added and air removed using the plunger. The syringes were sealed and placed in an incubator at 40 °C for 7 days, after which available N in soil solution was extracted with 2 M KCl as described above.

Potential nitrification. According to Dusek (1995), the equivalent to 10 g of dry soil in each treatment were placed in an incubation container. An UK soil from the Rowden experiment in the South West of England was included as an extra soil type for this analysis, to test the methodology. Samples were maintained for a pre-incubation period of seven days at 50% WHC at 20°C. After the pre-incubation period had finished, 50 ml of solution containing 15 mM sodium chlorate and 2 mM ammonium sulphate were added to each container. Containers were transferred to a water bath at 25°C, and the suspensions shaken for 6 h (except for time 0). After this, 2 ml aliquots of the suspensions were taken at 1, 2, 3, 4, 5 and 6 h after shaking and centrifuged at 4000 g (6137 ppm) at 4 °C for 15 min. The supernatant was analyzed for nitrite (NO_2^-). The potential activity in nmoles $\text{NO}_2^- \text{-N g}^{-1} \text{ dry soil h}^{-1}$ can be calculated as the rate of NO_2^- accumulation using a simple linear regression. The amount of NO_2^- produced in each sampling was calculated from the concentration of NO_2^- found in the sample multiplied by the original liquid volume (soil water + buffer) and then divided by 10 g soil.

Physico-chemical retention. The adsorption of NO_3^- and NH_4^+ were determined by the methodologies suggested by Wild (1972), Lumbanraja and Evangelou (1994) and Thompson and Blackmer (1992), respectively. Briefly, 5 g of soil was mixed with 25 ml of 0.01 M KNO_3 or 0.01 M NH_4Cl . Samples were then centrifuged at 4000 rpm (26,000 g) for 10 min and the supernatant discarded. This stage was

repeated three more times ensuring the soil was well mixed every time. After this, soil was weighed (including the entrained water). The NO_3^- was displaced by washing four times with 25 ml 1 M NaCl, while NH_4^+ was displaced with KCl 1 M. The washings were combined and analyzed for NO_3^- and NH_4^+ . The amount of NO_3^- and NH_4^+ retained was calculated from the concentration of NO_3^- or NH_4^+ found in the washings (total volume of washings) multiplied by the original volume (total washings + entrained water) divided by 5 g soil.

2.5 Statistical Analyses

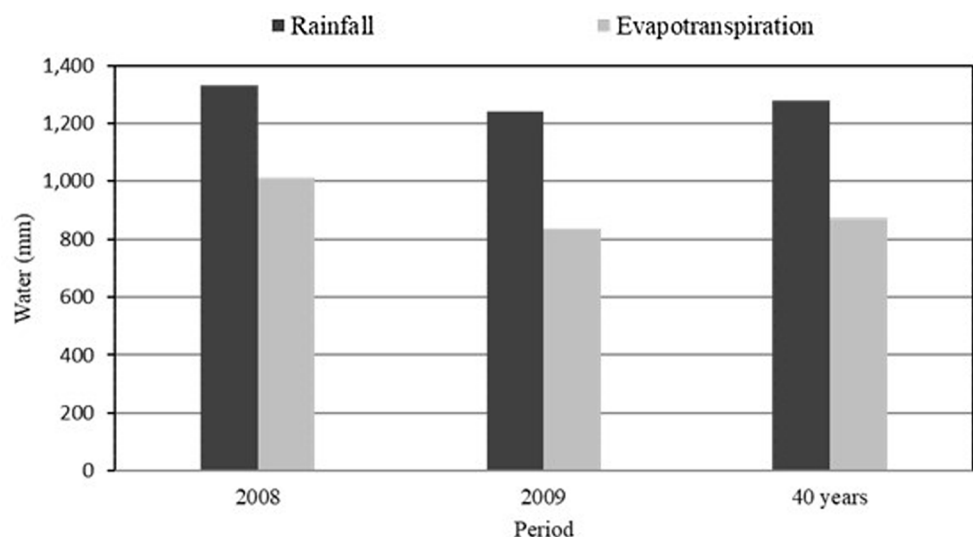
Treatments were distributed in the field in a randomized block design with four replicates as a combination of lime (0 and 2 t) and N application rate (0, 200, 400 kg N). For laboratory determinations, a randomized design with four replicates was used. Analysis of variance was used to compare treatments, considering specific treatments as fix effect. For the evaluations under field conditions, the year effect was also included in the interactions. Genstat 12.2 was used as the statistical package (Genstat 2003).

3 Results

3.1 Weather and Rainfall

During 2008 and 2009, total rainfall was 1330 and 1241 mm, similar to the long-term average. Evapotranspiration was greater during 2008 (1011 mm), compared to 836 mm for 2009, and 879 mm for the 40 year-average in the area, respectively) (Fig. 1). As drainage was calculated as the difference between rainfall and evaporation starting from the day when soil drained at 60 cm depth, the greater evapotranspiration in 2008 resulted in four leaching sampling dates, in

Fig. 1 Rainfall and evaporation values for 2008, 2009 and the 40 years average for INIA Remehue



comparison to the six samplings carried out in 2009, given 15% more drainage during 2009 than during 2008 (690.3 and 584.3 mm for 2009 and 2008, respectively).

3.2 Soil Characteristics

Initial soil chemical analyses showed high concentrations of organic matter (110 ± 7.2 g 100 g⁻¹), available N (17 ± 2.2 mg kg⁻¹) and K⁺ (1.3 ± 0.13 cmol + kg⁻¹), low concentrations of Olsen P (12 ± 0.8 mg kg⁻¹), S (8 ± 0.1 mg kg⁻¹), Ca²⁺ (2.0 ± 0.05 cmol + kg⁻¹) and Na⁺ (0.08 ± 0.01 cmol + kg⁻¹), and medium values of Mg²⁺ (0.4 ± 0.01 cmol + kg⁻¹) and Al³⁺ in soil solution (0.3 ± 0.02 cmol + kg⁻¹), with a slightly acidic pH in water (5.8 ± 0.03). The low availability of cations in soil solution resulted in a low CEC (3.8 ± 0.19 cmol + kg⁻¹) and a high Al saturation index ($6.8 \pm 0.71\%$).

The basal fertiliser strategy used as part of the experiment management resulted in increasing amounts of Olsen P over time, as results from the analysis conducted in 2009 and 2010 showed (Table 1; $P < 0.001$). Lime application did not affect Olsen P concentration ($P > 0.05$), but the addition of N resulted in a 27% reduction in the soil of the Olsen P, from 43 mg kg⁻¹ in the 0 N treatments to 33 mg kg⁻¹ in the +N treatments (averages of both years and lime applications; $P < 0.001$).

There was an interaction between the year of evaluation and the rate of lime application for available N and S concentrations in the soil ($P < 0.05$ in both cases), so that N and S concentrations decreased when 2 t of lime was applied, although for S this was only observed in the first year of application. Nitrogen concentration increased with increasing N rate, particularly during year two of evaluation ($P < 0.001$), and S only decreased with the highest N application in both years ($P < 0.05$).

There was no interaction between the year of evaluation and the rate of lime and N applied for the other soil parameters assessed. Nevertheless, lime application increased soil water pH with an additive effect over time, increasing from 5.4, as the average of lime treatments in the first year of evaluation, to 6.0 and 6.7 at the end of the second year of evaluation in the 0 and 2 t CaCO₃ ha⁻¹ application treatments, respectively ($P < 0.001$; Table 1). This was the result of an overcast application and Ca²⁺ and CEC increase over the two years of evaluation ($P < 0.001$) and a significant reduction of Al³⁺ in soil solution ($P < 0.001$), and thus, of the Al saturation index ($P < 0.001$; Table 1). Although pH increased during the second year of evaluation, the effect was lower with increasing N rates ($P < 0.01$).

Lime application alone did not affect K⁺ or Mg²⁺ concentration in soil solution ($P > 0.05$, data not shown) but reduced Na⁺ concentration from 1.00 to 0.89 cmol + kg⁻¹ (average of both years; $P < 0.001$). Nitrogen application alone did not

affect Mg²⁺ concentration in soil solution ($P > 0.05$, data not shown), but reduced K⁺ from 1.25 in the 0 N to 0.90 cmol + kg⁻¹ in the +N treatments (as averages of both rates and years of evaluation; $P < 0.01$). Additionally, increasing N rates increased Na⁺ concentration from 0.1 to 0.9 and 1.8 cmol + kg⁻¹ in the 0, 200 and 400 N treatments, respectively (averages of both years of evaluation; $P < 0.001$).

There was no interaction between the year of evaluation, lime and N application for N mineralization under field conditions ($P > 0.05$; Fig. 2a), neither amongst lime or N rate for cumulative N mineralization ($P > 0.05$). Mineralization was greater with the application of 400 kg N ha⁻¹ reaching up to 440 ± 28.6 kg N-NH₄⁺ ha⁻¹ ($P < 0.01$; Fig. 2b). Lime alone did not increase N mineralization ($P > 0.05$).

3.3 Nitrogen Leaching Losses

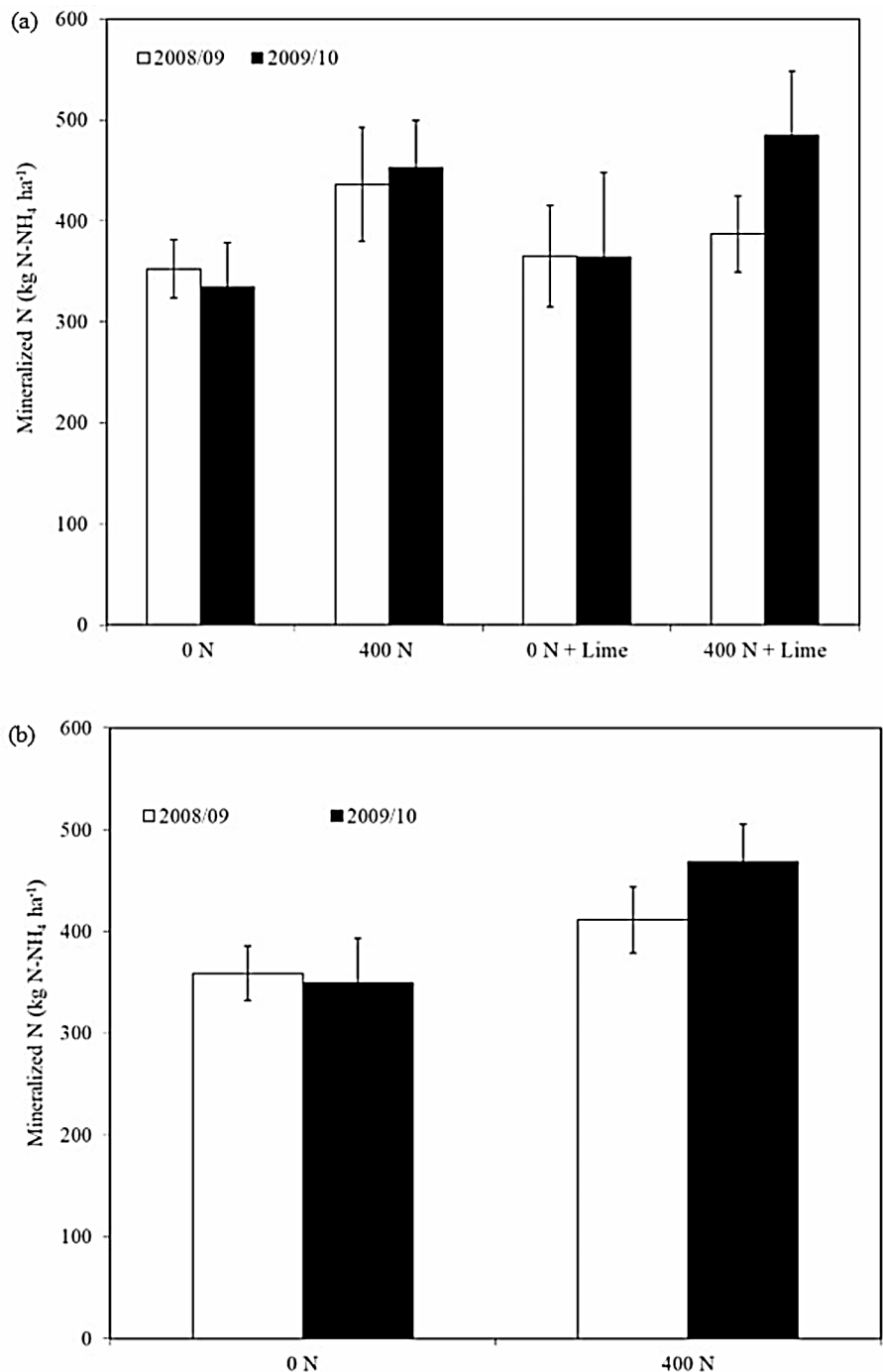
Losses were greater in the 400 kg N treatments during both years of evaluation, with no significant differences between limed and non-limed treatments ($P < 0.001$; Fig. 3). No interaction between the year of evaluation, lime and N addition, neither of lime x N were observed for the NO₃⁻ concentration in leachate samples ($P > 0.05$). Greater NO₃⁻ concentrations in leachates samples were measured in the 400 kg N treatment ($P < 0.05$), for both non-limed and limed plots in both years ($P < 0.001$), although in 2009 there was an increase in the average concentrations measured (Fig. 3) (Fig. 4).

3.4 Pasture Yield and N Plant Uptake

Yield. There was no interaction between the year of evaluation, lime application and N addition, and nor for lime x N application on pasture yield ($P > 0.05$; Table 2). Nitrogen addition increased yields, so that treatments receiving 200 and 400 kg N produced 50 and 92% more dry matter than the control treatment. The average for both years was $4,760 \pm 700$, $11,538 \pm 750$ and $7,703 \pm 290$ kg DM ha⁻¹ yr⁻¹ for the 400, 200 and control treatments, respectively. Liming increased yield by 11%, with the average for both years of $10,754 \pm 1,245$ and $11,914 \pm 1,373$ kg DM ha⁻¹ yr⁻¹ for non-limed and limed treatments, respectively. Annual pasture yield did not vary between years, so that no cumulative effect of N addition or liming on yield over time was observed.

Plant N uptake. Plant N uptake did not differ between years of evaluation, lime application and N addition, neither between lime x N application ($P > 0.05$; Table 2). Plant N uptake was 2.6 and 1.8 times greater in the 400 kg N treatment than in the 0 and 200 kg N (511 ± 26.5 , 352 ± 16.1 and 193 ± 12.8 kg N ha⁻¹ yr⁻¹ for the 400, 200 and 0 N treatments, respectively; $P < 0.001$). It was also 15% greater

Fig. 2 Nitrogen mineralization under field conditions ($\text{kg N-NH}_4 \text{ ha}^{-1}$) (a) for all treatments evaluated per season ($n=4$; $P>0.05$), and (b) average of N application treatments for all experimental period ($n=8$; $P<0.05$)

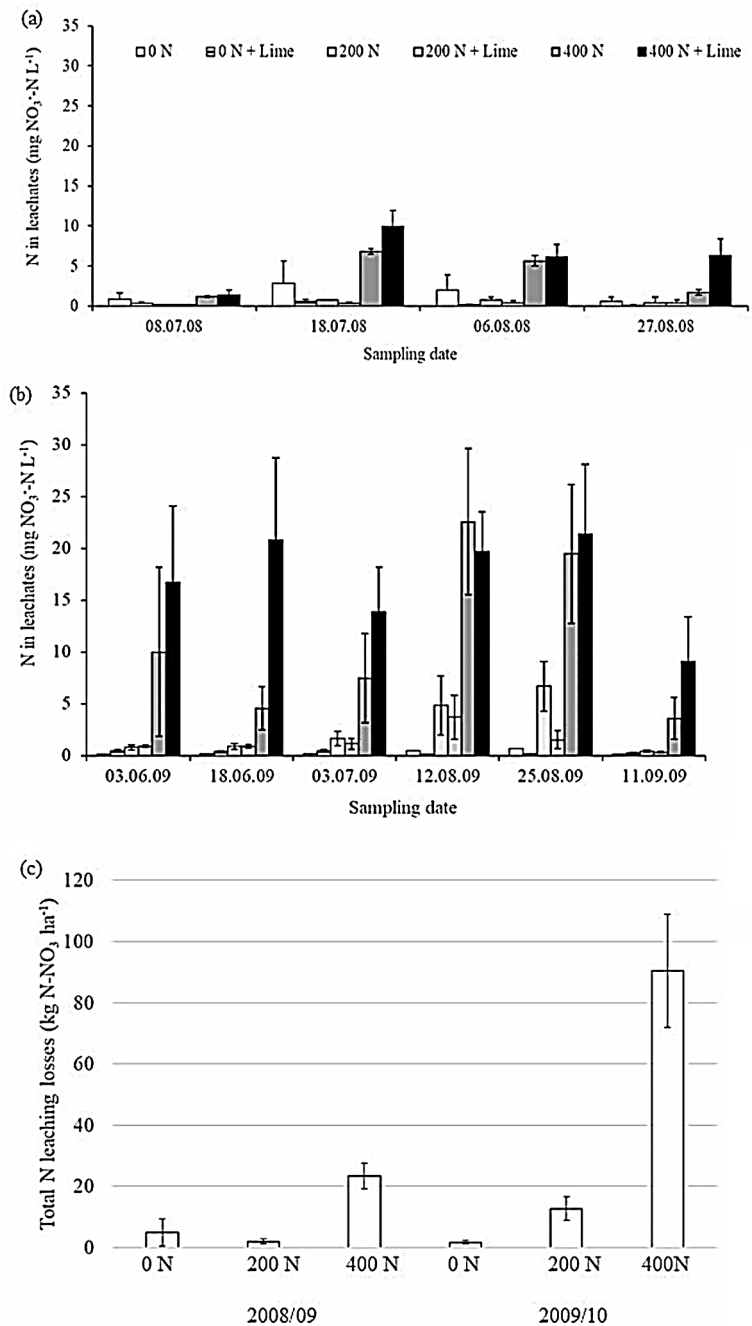


in the limed treatments at 376 ± 62.3 and $328 \pm 55.4 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ for limed and non-limed treatments, respectively ($P<0.001$). Nitrogen uptake was higher in 2008/09 than in 2009/10 (371 ± 61.5 and $333 \pm 57.1 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ for 2008/09 and 2009/10, respectively; $P<0.05$). The overall average N concentration in the forage varied between 1.5% (summer) to 4.4% (winter). Greater average N concentrations were found in the 400 kg N treatments ($P<0.001$).

3.5 Laboratory Incubations

Potential aerobic N mineralization. Potential N mineralization varied between 206 and 282 mg N kg^{-1} dry soil (Table 3). No interaction between lime and N addition was observed on this parameter ($P>0.05$). Nitrogen mineralization rate increased with N addition over the first 28 days ($P<0.05$) but no effect on total N mineralization was found (Table 3).

Fig. 3 Average nitrate concentrations in leachate samples collected in a permanent pasture during the drainage period of 2008 (a) and 2009 (b) and total leaching losses per N treatment (kg N ha⁻¹ yr⁻¹; average of years 2008 and 2009) (c) ($n=4$)



Potential anaerobic N mineralization. The results indicate that there was no interaction between lime and N input on potential anaerobic N mineralization ($P>0.05$), but potential anaerobic mineralization increased with both N and lime addition (Table 4).

Potential nitrification. The results from the UK Rowden soil gave a potential nitrification rate of 0.004 $\mu\text{g N kg dry soil}^{-1} \text{ h}^{-1}$. Variability between replicates was small (Fig. 5a). The rates from the Chilean volcanic soils did not show a linear response during the aerobic incubation, but instead a bell shape function was obtained (see example for

control in Fig. 5b). The functions fitted through these data were polynomial resulting in:

$y = -0.0008x^2 + 0.005x + 0.0021$ for the control; $y = -0.004x^2 + 0.0232x + 0.0746$ for the control + lime; $y = -0.0007x^2 + 0.0049x + 0.0163$ for the control + N, and $y = -0.0072x^2 + 0.0465x + 0.0326$ for the control + lime + N. The R^2 were 85%, 79%, 97% and 76% respectively.

Physico-chemical retention. The Chilean soil showed a greater capacity to retain N as NH_4^+ (74% on average) than NO_3^- . Results of this determination indicate that there was no interaction between lime and N addition for NH_4^+

Fig. 4 Dynamics of N-NH_4^+ (a) and N-NO_3^- (b) concentrations during the controlled aerobic incubation study for samples collected in July 2008

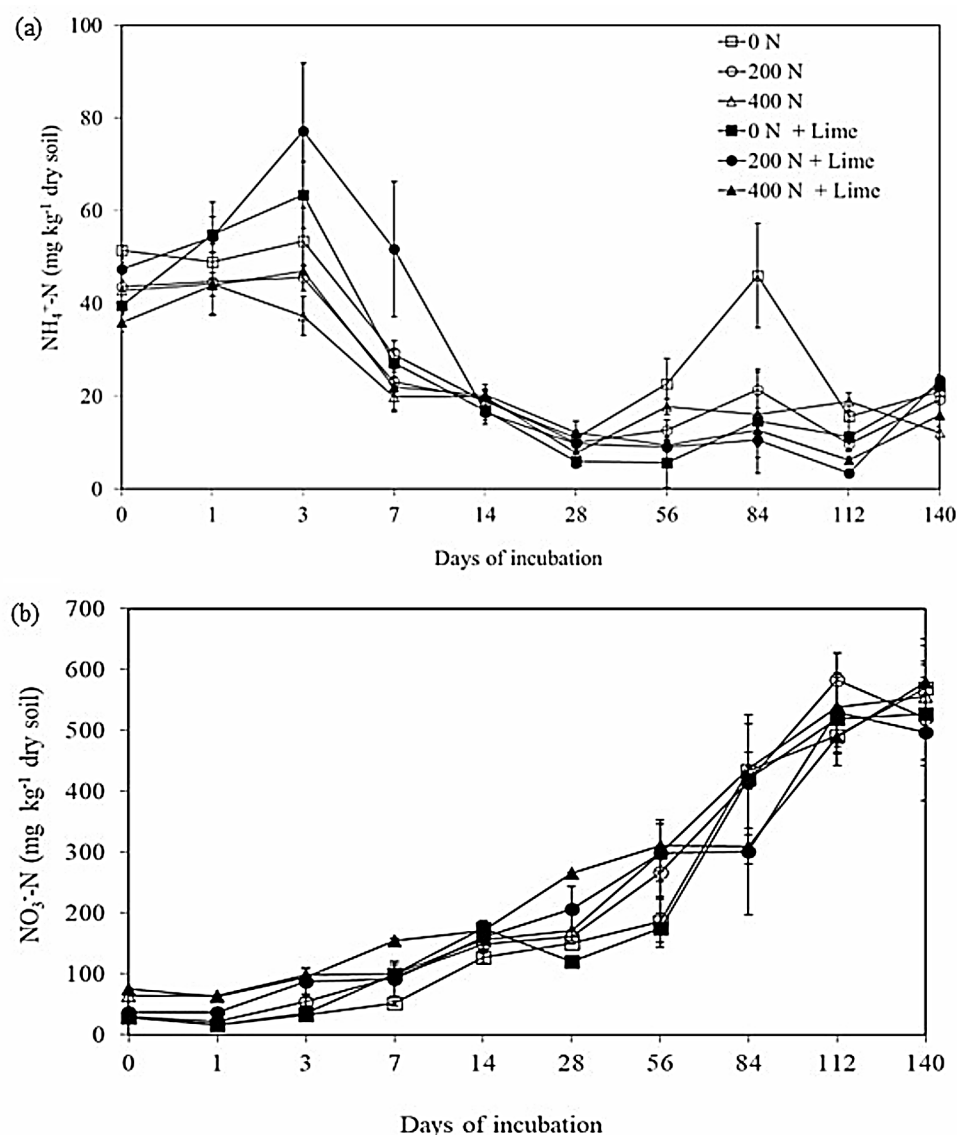


Table 2 Pasture yield ($\text{kg DM ha}^{-1} \text{yr}^{-1}$), and plant N uptake ($\text{kg N ha}^{-1} \text{yr}^{-1}$) in different N fertiliser ($\text{kg N ha}^{-1} \text{yr}^{-1}$) and lime application ($\text{t ha}^{-1} \text{yr}^{-1}$) treatments, for both years of evaluation ($n=4$; \pm standard error of the mean)

Parameter/year	Pasture yield ($\text{kg DM ha}^{-1} \text{yr}^{-1}$)	N uptake ($\text{kg N ha}^{-1} \text{yr}^{-1}$)
0 N	$7,703 \pm 381.5$ c	193 ± 12.8 c
200 N	$11,538 \pm 435.5$ b	352 ± 16.1 b
400 N	$14,760 \pm 517.1$ a	511 ± 26.5 a
Significance	$P < 0.001$	$P < 0.001$
0 Lime	$10,754 \pm 1,245.1$ b	328 ± 45.4 b
2 Lime	$11,914 \pm 1,372.6$ a	376 ± 52.3 a
Significance	$P < 0.05$	$P < 0.01$
2008/09	$11,527 \pm 1,244.3$	371 ± 51.5 a
2009/10	$11,140 \pm 1,416.2$	333 ± 47.1 b
Significance	NS	$P < 0.05$

NS: not significant

Table 3 Potential N mineralization in the N fertiliser ($\text{kg N ha}^{-1} \text{yr}^{-1}$) and lime application ($\text{t ha}^{-1} \text{yr}^{-1}$) treatments, using aerobic incubation ($n=4$; \pm standard error of the mean; $P > 0.05$). Sample collection in July 2008

Treatment	Potential N mineralization (mg N kg^{-1} dry soil)
0 N	216 ± 12.5
200 N	250 ± 28.8
400 N	264 ± 26.3
Significance	NS
0 Lime	234 ± 15.3
2 Lime	252 ± 29.8
Significance	NS

NS: not significant

or NO_3^- soil retention ($P > 0.05$), neither there was a significant effect of the N or lime treatments on N retention as NH_4^+ ($P > 0.05$). Lime application decreased NO_3^- retention ($P < 0.05$; Table 5).

Table 4 Potential N mineralization in the N fertiliser ($\text{kg N ha}^{-1} \text{ yr}^{-1}$) and lime application ($\text{t ha}^{-1} \text{ yr}^{-1}$) treatments under anaerobic conditions (\pm standard error of the mean; $n=8$ for N and $n=12$ for lime effect)

Treatment	Potential N mineralization ($\text{mg N-NH}_4^+ \text{ kg}^{-1}$ dry soil)
0 N	378 ± 41.3 b
200 N	455 ± 46.0 a
400 N	491 ± 40.4 a
Significance	$P < 0.01$
0 Lime	333 ± 13.5 b
2 Lime	549 ± 25.5 a
Significance	$P < 0.001$

4 Discussion

Total organic N and C were high (Table 1) reflecting OM accumulation in this volcanic soil, but C/N ratio was low. The soil also had a medium soil fertility level for grassland and animal production (Table 1). The low initial soil S concentration is typical in soils used for livestock production in the area (Alfaro et al. 2006a). These acidic soils also have low P availability and cation concentrations in soil solution, associated with soil P fixation by allophane and soil acidity, respectively (Escudey et al. 2001), which results in low pH and medium to high Al saturation indexes, as observed in the present study. This is associated to with annual rainfall and its effects on loss of cations losses from the soil and agricultural management such as the use of urea fertiliser as an N source (Mora et al. 2006).

The Al saturation index describes the relationship between Al^{3+} and cations in the soil solution ($\text{Al}^{3+} + \text{Ca}^{2+} + \text{Mg}^{2+} + \text{K}^+ + \text{Na}^+$), and it is related to Al^{3+} toxicity impacts on root growth (Mora et al. 1999), which affects both pasture yield and animal performance in southern Chile due to increases in foliar Al concentration (Mora et al. 2006). The application of CaCO_3 results in Ca^{2+} , OH^- and HCO_3^- formation due to the reaction of lime with soil water. The OH^- reacts with Al^{3+} in soil solution, while HCO_3^- reacts with H^+ to form CO_2 and H_2O which is then lost to the atmosphere, thus increasing pH (Bernier and Alfaro 2007). In our experiment, the pH in water reached up to 6.7 in the second year of application in the Lime treatments, which is considered appropriate for microbial activity. As a secondary effect, lime application reduced Al^{3+} availability in soil solution (Bossolani et al. 2020) and so decreased the Al saturation index in relation to the original soil condition, after the two years of continuous application, creating adequate soil fertility conditions for pasture yield (Mora et al. 2006). It also decreased Olsen P, N and SO_4^{2-} concentrations in the soil at the end of the experiment. This was probably not associated with PO_4^{2-} precipitation or the direct effect of CO_3^{2-} on NO_3^- and SO_4^{2-} leaching losses, as both P and lime were applied at different times of the year, to avoid

the displacement in soil concentrations, and NO_3^- losses are expected to be low in pastures in the area (Salazar et al. 2012), but as a result of increasing yields, as discussed ahead. This has been previously observed in pasture experiments on acidic soils in the current study, where lime addition increased pasture yield and crude protein concentration by 40 and 15%, respectively (Mora et al. 2006).

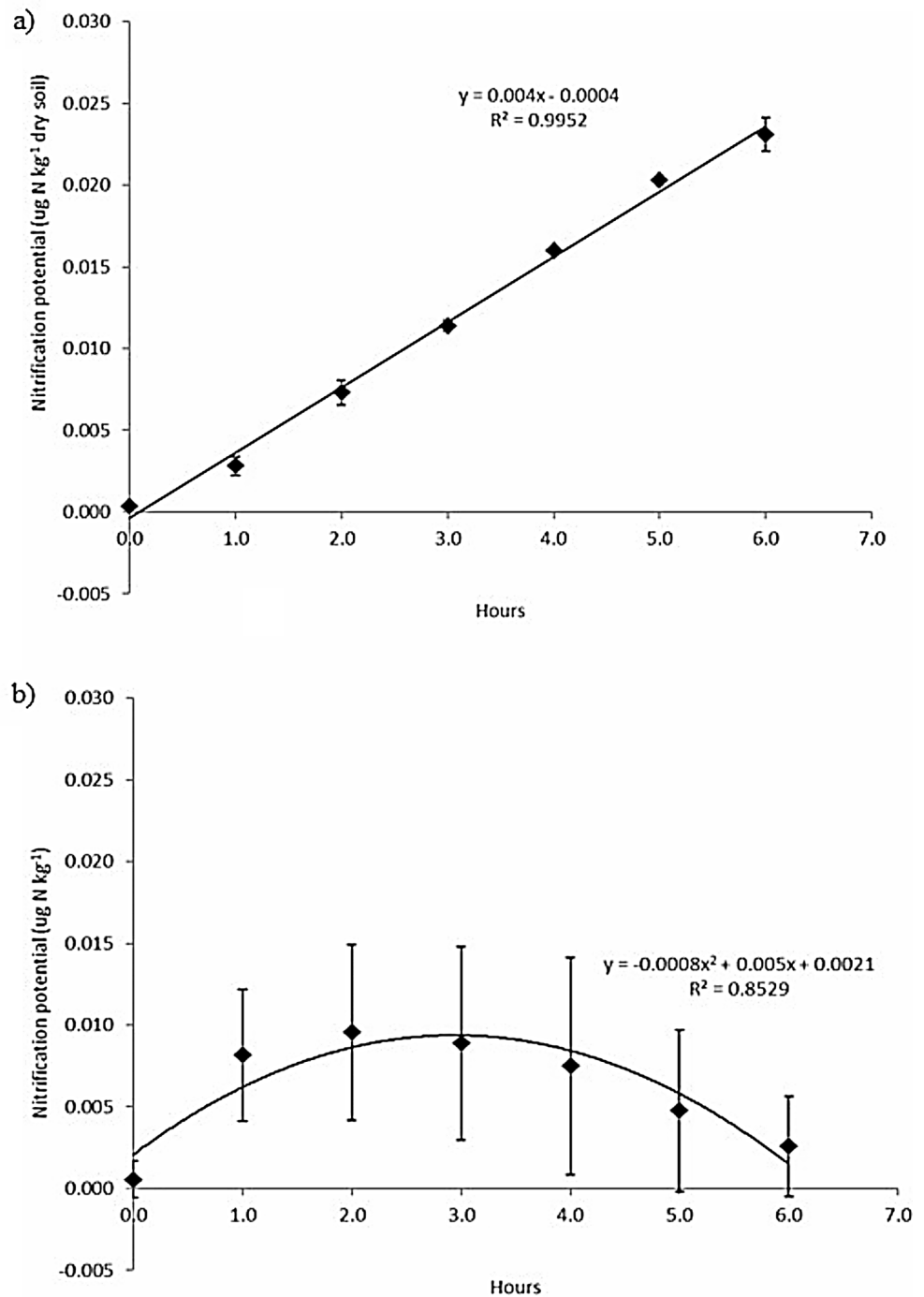
Nitrogen addition as NaNO_3 resulted in an increase of Na^+ concentration in soil solution and in an increase of soil pH, but in this case, no correction of Al saturation index was observed, as this is related to HCO_3^- activity in the soil, as discussed previously. In addition, as N input in fertiliser increase, soil exchangeable K^+ was reduced due the increase in yields and the related K plant uptake.

Soil nutrient parameters increased over time, even in the no N non-limed plots, probably as a result of differences in the sampling time, from July (winter) for the initial sampling, to February (Summer) months, as traditionally observed in pasture soils of southern Chile (Mora et al. 2006; Martinez-Lagos et al. 2015).

Average total NO_3^- leaching losses in our study were similar to those reported previously for cut swards in Chile (Salazar et al. 2010) and for a basin with extensive livestock production in Southern Chile (Godoy et al. 1997). Losses were lower than those reported for grazed swards in the study area by Alfaro et al. (2008a, b) in Chile; Ryden et al. (1984) in England and Ledgard et al. (1999) in New Zealand, despite a NO_3^- based fertiliser being used which is more prone to direct leaching than the urea or ammonium-based fertilisers in other studies. In addition, this could be related to higher N rates used in the present experiment and the potential impact this could have had on N accumulation in the soil over time (see ahead) due to differences in N uptake, which corresponded to 80% of the N applied for both N rates resulting in a N of c. 41 and c. 82 kg N ha^{-1} for the 200 and 400 N treatments, respectively.

The effect of N rate on N leaching losses seem to be accumulative, as NO_3^- -N concentrations and total N losses in the second year of evaluation were higher than those from the initial experimental year at both 200 and 400 kg N application rates, although none of the NO_3^- concentrations measured surpassed the 50 mg L^{-1} established as maximum threshold recommended to maintain adequate groundwater quality (EU 2006). In 2009, there were two more leachate sampling dates than in 2008 due to higher rainfall, which contributed one third of total N leaching losses that year. The second N dressing applied each September occurred after the last leaching sampling dates, so that N lost by leaching in this experiment would be mostly associated with the autumn N dressing (half the annual rate) and soil N from previous managements. Our results also suggest that N leaching losses were related to N input as fertiliser and not

Fig. 5 Nitrification potential in (a) Rowden soil and (b) Chilean soil ($\mu\text{g N kg dry soil}^{-1}$). Error bars are the standard deviation of the mean of four replicates



to the soil acidity conditions evaluated for this volcanic soil. The soil capacity to retain N (either as NH_4^+ or NO_3^-) would then become a key factor for the soil buffer capacity to avoid N leaching losses in this volcanic soil.

The lack of direct effect of lime application on NO_3^- leaching losses, is not consistent with data reported before for other volcanic soils under tropical conditions (Rosolem 2011), which shows that in a controlled corn greenhouse experiment using pots (0–40 cm depth), NO_3^- concentrations

increase in the upper soil layer (0–5 cm) would be the result of a greater nitrifying activity in the soil surface due to the increase in soil aeration generated by crop residue incorporation, in combination with CaO/MgO application. Further plant uptake would have prevented NO_3^- displacement to deeper soil layers. An interesting observation (Fig. 2) for the decreased leaching in 2008 after the first addition of lime to the 0 N control, would indicate the existence of a lag time between lime overcast application on N leaching losses,

Table 5 Soil N retention as either ammonium ($\text{NH}_4\text{-N}^+$) or nitrate ($\text{NO}_3\text{-N}$) (mg N kg^{-1} dry soil) from soil receiving N ($\text{kg N ha}^{-1} \text{ yr}^{-1}$) and lime application ($\text{t ha}^{-1} \text{ yr}^{-1}$) treatments (\pm standard error of the mean; $n=4$)

Treatment	$\text{NH}_4\text{-N}^+$ (mg N kg^{-1} dry soil)	$\text{NO}_3\text{-N}$
0 N	96 ± 5.5	35 ± 1.3
200 N	100 ± 4.2	35 ± 1.2
400 N	96 ± 1.8	32 ± 1.16
Significance	NS	NS
0 Lime	97 ± 3.6	$36 \pm 1.0 \text{ a}$
2 Lime	98 ± 2.6	$32 \pm 0.8 \text{ b}$
Significance	NS	$P < 0.01$

NS: not significant

associated to the require time for the lime to be slowly incorporated in the soil profile with rainfall (Bernier and Alfaro 2007) and, subsequently, impacts NO_3^- soil dynamics. This suggest that long-term research on the lime effect is warranted due to delays in lime effects developing and its full effects to become evident.

The total DM yield of all treatments in our study was higher than the results obtained by Teuber and Elizalde (1999) in a perennial ryegrass/white clover sward, and within the values reported by Lobos et al. (2016) and Flores et al. (2017). This could be related to the greater N addition used in our experiment (200, 400 kg N ha^{-1}) compared to that of Teuber and Elizalde (1999) (150 kg N ha^{-1}), and to differences in the sward composition in relation to the study by Lobos et al. (2016), with the inclusion of ryegrass with greater potential yields in the study by Flores et al. (2017). Pasture yield from the non-limed plots was higher than that reported by Mora et al. (2006) for swards growing in acidic soils in southern Chile (*c.* 8 t DM ha^{-1}) probably because in the previous study acidic soil conditions (water pH 5.1 and Al saturation index of 21%) were stronger than the ones reported here (pH 5.4 and Al saturation index of 6%), while DM yield from plots less acidic are lower than values reported previously for similar treatments (14 t DM ha^{-1} , Mora et al. 2006), probably due to the higher yield potential of a recently sown sward (two years old, ryegrass/white clover) in the previous study in comparison to the permanent pasture used in the current study.

Pasture yield did not significantly increase during the second year, so that no cumulative effect of N addition over time was registered. The plant N concentration values measured in this study were equivalent to typical crude protein concentrations for N fertilized pastures in the area (Anrique et al. 2008; Demanet et al. 2015), responding to N application rate, in agreement with Lobos et al. (2016) reaching *c.* 4% with *c.* 400 kg N ha^{-1} applied as fertiliser.

The field values of N mineralization measured here are within the range of data reported for the same soil series

previously by Alfaro et al. (2009) (209–308 kg N ha^{-1}) and by Lobos et al. (2016) (241–643 kg N ha^{-1}). The N mineralization values reported by Alfaro et al. (2009) are slightly lower than those measured in the current experiment, potentially due to the lower N rate applied in that experiment (67.5 kg N ha^{-1}), in comparison to the rates used in our study, and the expected priming effect of N application on soil biological processes (Cartes et al. 2009; Bossolani et al. 2020) including N mineralization, as discussed below.

The increase in N mineralization with increasing N rates in both the field and the anaerobic incubation experiments was probably the result of a priming effect associated to microbial biomass stimulation (Cartes et al. 2009; Qian and Schoenau 2002) which would occur after a short and rapid immobilization phase following N addition (Martínez-Lagos et al. 2015). In our study, because of the soil organic matter content of the soil and its volcanic nature, with amorphous clay mineral such as allophane, N adsorption could be favoured (Matus et al. 2008), avoiding further microbial processes and resulting in low N leaching (Salazar et al. 2012 and this study), and denitrification losses (Hube et al. 2016; Muñoz et al. 2011; Vistoso et al. 2012; Alfaro et al. 2021). Kizewski et al. (2019) using labelled N showed that direct soil organic matter NO_3^- immobilization, prior transformation to NO_2^- before immobilization into organic N, as well as NO_3^- conversion into dissolved organic nitrogen (DON) are the predominant abiotic process (66–100% and > 85% for immobilization and DON incorporation, respectively) preventing NO_3^- leaching losses, in agreement with results of our experiment, or further transformations such as denitrification. Our experimental data suggest that the soil capacity to conduct this abiotic immobilization has a limit, as a two-year application regime of 400 kg N ha^{-1} resulted in increasing leaching losses.

Work by Huygens et al. (2007, 2008) has shown that also biologically driven processes such as dissimilatory NO_3^- reduction to ammonium (DNRA) and to DON pool would result in low available N leaching losses in a volcanic rainforest soil of southern Chile, like that used in the present study. The combination of both abiotic and biological process ensures NO_3^- retention (Huygens et al. 2008) and would protect N for later microbial transformations as discussed previously, resulting in high N mineralization rates in more aggressive mineralization conditions such as those provided in the anaerobic incubation experiment. This would also explain the relatively high natural N concentration in pastures growing in these volcanic ash soils as results of our experiment show (> 4%N, except in summer), and in agreement with Anrique et al. (2008) and Demanet et al. (2015), as part of a continuous N plant supply and luxury N plant uptake, resulting in low animal N use efficiencies (Beltran et al. 2022).

The lack of interaction between lime application and N rates input on soil N mineralization in field conditions over the two years field experiment agrees with results from the potential aerobic incubation (see below).

Results of the potential mineralization study indicate an initial increase of NH_4^+ formation and decrease in NO_3^- concentrations in the soil in all treatments, with greater rates of transformation in the treatments receiving N fertiliser, as discussed previously. After day 28 of incubation, NH_4^+ decrease in soil solution until the end of the experimental period. Nitrate increased from day 3 onwards, with no differences between limed and non-limed treatments. Similar tendencies have been reported for another Chilean volcanic soil by Cartes et al. (2009). The values of mineralization from the aerobic incubation obtained here agree with previous values reported in Cardenas et al. (2013) for no N addition (215 mg kg^{-1} dry soil for the Osorno soil series).

The increase in potential anaerobic N mineralization of the more recalcitrant sections of soil organic matter with both N and lime addition (Table 4) suggests that lime addition could influence N mineralization in the long term, while N addition would affect it in both short and long term, as suggested previously.

Nitrogen retention in the soil was higher as NH_4^+ (74% of total N retained) than NO_3^- , because of the presence of clays particles negatively charged in this soil (Escudey et al. 2001) and the DNRA process, in agreement results of Huygens et al. (2008). Values reported previously for the same soil (Cardenas et al. 2013) were lower in the 0–10 cm soil layer for both NH_4^+ and NO_3^- being equivalent to 60 and 20 mg N kg^{-1} dry soil for NH_4^+ and NO_3^- , respectively. The results reported here may be the result of liming effects on NO_3^- displacement by the CO_3^{2-} ions provided by lime surface application, resulting in lower NO_3^- abiotic immobilization.

The results of the retention analyses were converted to total N retained in the soil considering a bulk density of 0.65 g cm^{-3} (0–10 cm depth), giving a value of 85 kg N ha^{-1} , on average, of which 63 and 22 kg N ha^{-1} were adsorbed as NH_4^+ and NO_3^- , respectively. This value seems low, although it would increase if the complete soil profile within the roots zone is considered, as data from Cardenas et al. (2013) suggest. In that study this value increased six times from 52 kg N ha^{-1} (0–10 cm) to 390 kg N ha^{-1} (0–60 cm).

Relying on mineralized N over N fertiliser application from a farmer's perspective, would have a direct economic benefit, while reducing risks of N losses to the wider environment. The main challenge for the use of this soil N remains in the distribution of this soil contribution over the year, as most of N mineralization is produced either during spring (35%) or summer (36%) due to either temperature,

soil moisture or oxygen constrains for microbial activity at other times of the year (Martínez-Lagos 2015).

Our results suggest that in the short term, the N rate applied in autumn would have a direct effect on N leaching losses during the following winter when N is applied as NO_3^- , while liming could increase these losses in the long term through an improvement in soil chemical conditions for microbial activity. Some of the excess N applied and not used for yield or plant uptake would remain in the soil as microbial biomass or immobilized by abiotic processes (protected), so that it would not be immediately available for further transformations and losses by nitrification or denitrification.

According to results of the retention studies, if N is applied in urea or NH_4NO_3 forms, its retention may increase contributing to lower N leaching and denitrification losses, as previous studies for the same soil showed (Hube et al. 2016). In that study, losses by denitrification increased only due to rainfall effects and were not directly associated to N input in urea fertiliser. It was suggested that rainfall provided the missing element (water) to activate soil N supply by mineralization or release due to plant growth. This would also agree with results by Alfaro et al. (2022) who showed that in a grassland volcanic soil, similar to that used in our study, N_2O losses were associated with soil water field pore space and temperature (0–10 cm), and not to the immediate availability of NO_3^- or NH_4^+ in the soil.

The processes evaluated here suggest that N use efficiency from fertiliser application in these soils could be increased. The use of NH_4^+ -based as well as enhanced fertilisers need to be further analysed in these soils to increase productivity and reduce environmental risks in the long term, also opening the question for other technologies that could be used to increase N use efficiency, such as the use of nano fertilisers applied either to soils or in combination with foliar application, as recent results have shown (Mejias et al. 2021; Hube et al. 2022). The contribution of soil mineralization to pasture growth can be further enhanced and understood, to match N fertiliser application to periods with low natural N soil supply. Further studies should also consider the role of soil organic matter fractions on N soil immobilization and that of soil microbial biomass on further N transformations.

5 Conclusions

Following a cutting regime, total N leaching losses were greater in the treatments receiving 400 kg N , with no differences between the limed and non-limed treatments ($P > 0.05$). Results showed that there was no interaction between lime and N input on soil N mineralization over the two-year field experiment ($P > 0.05$). Nitrogen fertilisation

increased N mineralization, as primer effect. The mineralization of more recalcitrant sections of soil organic matter increased with both N and lime addition.

No significant effect of the treatments on N retention as ammonium ($P > 0.05$) was observed, and lime application decreased soil nitrate retention ($P < 0.05$). Soil had a greater capacity to retain N as ammonium (74% on average) than nitrate.

Our results indicate that the natural acidic soil conditions (pH 5.4–5.7) did not prevent nitrification, neither leaching losses, probably associated to occurrence of N abiotic immobilization processes. They also suggest that liming could influence N mineralization, and potentially soil nitrification processes leading to increasing N leaching losses, in the long term, while N addition would have this effect in both short and long term.

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Author Contributions MAA and LC contributed to the study conception and design. Material preparation, field data collection and analysis were performed by MAA, FS, and LR. Laboratory incubation experiments and data analysis were performed by MAA, LC, and DH. The first draft of the manuscript was written by MAA and LC, and all authors commented on the manuscript.

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Data Availability The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

Declarations

Competing Interests The authors have no relevant financial or non-financial interests to disclose.

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