NITROGEN, PHOSPHORUS AND POTASSIUM LOSSES IN A GRAZING SYSTEM WITH DIFFERENT STOCKING RATES IN A VOLCANIC SOIL

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ABSTRACT

In Chile there is little information on nutrient losses in livestock systems. The experiment was carried out between 2004 and 2006. Two stocking rates (3.5 and 5.0 steers ha^{-1}) were tested under rotational grazing with Black and White Friesian steers on a permanent pasture (67.5 kg N and 40 kg P ha^{-1} yr^{-1}). To quantify surface runoff losses (N, P, K), three surface lysimeters (5 x 5 m) per treatment were established. N and K leaching losses were estimated with ceramic cups. Runoff and leachate samples were individually analyzed for available and total N, reactive (RP) and total P, and K. Dissolved organic N (DON) and organic P (OP) were estimated as the difference between total and available forms. The stocking rate did not increase total N, P and K losses ($P > 0.05$). Losses in surface runoff were < 0.5 kg N, < 0.05 kg P and < 0.6 kg K ha^{-1} yr^{-1}, respectively, due to the low amount of runoff measured. Nitrogen leaching losses were high (11 up to 71 kg ha^{-1} yr^{-1}) and K leaching losses were low (3 to 5 kg ha^{-1} yr^{-1}). Nitrogen in runoff was mainly lost as DON (50%). Nitrogen leaching losses were mainly as nitrate (70%). Phosphorus was lost as RP (70%). Thus, stocking rates of 5.0 steers ha^{-1} are plausible, but fertilizer application should be avoided in rainfall periods during the year to reduce incidental nutrient losses.

Key words: eutrofication, water quality, grazing, Andisol, beef production.

INTRODUCTION

The Los Lagos Region of southern Chile has suitable climatic conditions and soil types for cattle production. Volcanic soils are widespread in this area, being characterised by low nutrient availability, high P fixation capacity, high organic matter (OM) content and a pH-dependent cation exchange capacity (Escudey *et al.*, 2001). Volcanic soils fix between 85 and 90% of the P applied as inorganic fertilizer (Escudey *et al.*, 2001) because they are acidic, having Al^{+3} ion and hydrous oxides, which are very efficient in adsorbing $H_2PO_4^-$ ions (Morgan, 1997). This fact has created the perception that low P losses to waters can be expected in the area.

The use of N and P in fertilizers and animal feed in the area has increased over the last 10 years (Alfaro and Salazar, 2005), being these elements the main cost of fertilizer application. This has resulted in greater stocking rates being used in direct grazing. Because of the need to increase total production per hectare and total number of animals, it is expected that the environmental risk due to livestock production will increase.

In developed countries the environmental impact of livestock systems has been widely studied, because of the important role of this activity on water, soil and air pollution (Jarvis and Oenema, 2000). Despite the importance of livestock production in southern Chile and that N, P and K are strategic for grassland production in the area, there is little information about the contribution of these nutrients to water sources

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from beef production systems in the region. Lysimeter studies have been carried out under laboratory and field conditions (Alfaro *et al.*, 2006b) to evaluate N and P leaching losses after pasture cutting, but this does not reproduce exactly the conditions created by grazing. The effect of grazing on P losses has been reported previously (Alfaro and Salazar, 2007), but no comparison between stocking rates has been made. In addition, K leaching losses have not been measured under grazing conditions.

The objective of this study was to quantify N and K losses in runoff and leaching, and P losses in runoff in beef production systems with two different stocking rates in southern Chile.

MATERIALS AND METHODS

The experiment was carried out from 30 March 2004 to 31 October 2006 at the National Institute for Agricultural Research (INIA), Remehue Research Centre (40º35' S, 73º12' W). The soil at the site is an Andisol of the Osorno soil series (Typic Hapludands; CIREN, 2003), which has 6% slope, more than 1 m depth, high OM and available P (P Olsen) concentrations (Table 1). According to a weather station placed within 1 km distance, the 30 yr average rainfall for the area is 1 284 mm yr^{-1}.

In this study two stocking rates (3.5 and 5.0 steers ha^{-1}) were tested on two closed systems of 2.8 and 3.6 ha each, respectively. Grazing was carried out with Black and White Friesian steers (50-75% Holstein) with initial live weights of 220 ± 0.5 kg (2004), 244 ± 0.5 kg (2005) and 219 ± 0.0 kg (2006). Rotational grazing was used on a 25 yr old permanent pasture. The main species in the pasture were ryegrass (*Lolium perenne* L.), orchad grass (*Dactylis glomerata* L.) and yorkshire-fog (*Holcus lanatus* L.).

Treatments were fertilized in autumn 2004 (27 April) with 45 kg N ha^{-1} (urea, 46% N) and spring 2004 (10 September), with 22.5 kg N ha^{-1} (sodium nitrate, 16% N) and 40 kg P ha^{-1} (triple superphosfate, TSP, 46% P_2O_5). During autumn 2005 and 2006 (18 March and 21 April) treatments were fertilized with 45 kg N ha^{-1} (sodium nitrate) and in spring 2005 and 2006 (31 August each year), with 22.5 kg N ha^{-1} (sodium nitrate) and 40 kg P ha^{-1} (TSP).

Rainfall

Daily rainfall and evapotranspiration during the experimental period were recorded with an automatic weather station placed within 1 km distance of the experimental site.

Soil

Soil samples (n = 16) were taken randomly from all treatments in autumn and spring each year at 0-10 and 10-20 cm depth. Samples were bulked into two replicates for nutrients determination. P Olsen and other nutrients were determined by the methodology described by Sadzawka (1990). Each autumn, soil cores (0-5 cm depth, n = 15) were taken using the cylinder methodology (Rowell, 1997) to determine the effect of grazing on soil bulk density.

Table 1. Initial and final soil chemical analysis and bulk density at the experimental site for the 3.5 and 5.0 steers ha^{-1} treatments. 0-20 cm depth, 25/03/2004 and 02/03/2006. Average of two replicates (± standard error of the mean).

Stocking rate	3.5 steers ha^{-1}		5.0 steers ha^{-1}	
Year	2004	2005	2004	2005
Parameter				
P Olsen, mg kg^{-1}	28 ± 1.5	30 ± 6.9	27 ± 0.8	21 ± 5.1
Organic matter, %	18 ± 0.02	16 ± 1.3	15 ± 0.24	15 ± 1.4
CEC, cmol(+) kg^{-1}	10.3 ± 0.25	9.0 ± 1.34	10.3 ± 0.16	9.1 ± 1.37
S, mg kg^{-1}	2 ± 0.5	6 ± 1.3	2 ± 0.03	6 ± 2.1
Al saturation, %	1.5 ± 0.01	1.9 ± 0.60	2.0 ± 0.14	2.5 ± 1.09
Bulk density at 0-5 cm, g cm^{-3}	0.49 ± 0.005	0.65 ± 0.006	0.49 ± 0.008	0.66 ± 0.026

CEC: cation exchange capacity.

Nitrogen losses
To quantify N losses in surface runoff, three surface lysimeters (5 x 5 m) were established in each closed treatment, according to the methodology described by Alfaro and Salazar (2007). The accumulated runoff was measured three times per week with the use of a graduated collector. Runoff samples were collected in 125 mL polyethylene bottles. All individual collections were chemically analyzed and no filtering was needed.

Runoff samples were stored at 4 ºC until analysis for available N. Nitrate was measured using the salicylic acid method (Robarge *et al.,* 1983), and ammonium was determined through the indophenol methodology (Mulvaney, 1996). Total N was determined with the macro-Kjeldahl method (Method 10071 test 'N Tube; Hach, 2000a) and organic N was calculated as the difference between total N and the sum of the available N forms. Total N losses were calculated as the product of drainage and N concentration in the respective runoff samples.

Nitrogen leaching losses were determined 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). Ceramic cups were placed at 60 cm depth in the soil (three replicates per surface lysimeter, n = 9 per treatment) at an angle of 30° to the vertical. To protect the top of the cups from grazing animals, exclusion cages 0.09 m^2 (30 x 30 x 30 cm) were placed over them. In the first year, leachate samples were taken periodically every 2 wk during the drainage season, in the second and third year samples were taken every 100 mm of drainage. Samples were stored at -15 ºC until analysis for ammonium and nitrate.

Drainage between sampling periods was estimated by subtracting potential evapotranspiration data from rainfall using meteorological information. Evapotranspiration of the sward was calculated using the Penman-Monteith method (Penman, 1948). The amount of ammonium and nitrate leached over the period was then calculated according the trapezoidal rule proposed by Lord and Shepherd (1993). Leaching samples were analyzed for available N ($N\text{-}NO_3^-$ and $N\text{-}NH_4^+$) using the methodology described previously. Total N losses were calculated as the product of drainage and N concentration in the respective samples. Total N losses for the experimental period were calculated as the sum of N losses in runoff and N leaching losses.

Phosphorus losses
To quantify P losses in surface runoff, three surface lysimeters were used following the methodology described for N determinations. Reactive P (RP) was measured using the ascorbic acid method (Clesceri *et al.*, 1998), and total P was determined through the digestion with acid persulphate (method 8190; Hach, 2000b). Organic P (OP) was estimated as the difference between TP and RP for each sample, so that this data may also include particulate P. Total P losses were calculated as the product of drainage and P concentration in the respective samples.

Potassium losses
During 2005 and 2006 K losses in surface runoff and leaching were measured using the methodology of surface lysimeters and ceramic cups described previously. Potassium concentration was determined by atomic absorption spectrometry, according to Clesceri *et al.* (1998). Total K losses were calculated as the product of drainage and K concentration in the respective samples. Total K losses for the experimental period were calculated as the sum of K losses in runoff and K leaching losses.

Statistical analysis
Analysis of variance (ANOVA) was used to compare both treatments over the years in a randomized block design. Genstat 7.1 was used as statistical package (Payne *et al.,* 2003).

RESULTS

Weather and rainfall
During 2004, total rainfall was 1 231 mm, similar to that of an average year for the area. Both 2005 and 2006 had a surplus equivalent to 195.3 and 176.0 mm of rainfall, respectively, in relation to a 30 yr average. Average rainfall over the drainage period was 49 and 52% greater during 2005 and 2006 in relation to 2004, respectively. The greater rainfall of 2005 and 2006 had an impact on drainage values of those years, so that they were greater than drainage for 2004, with 633.8, 940.5 and 854.7 mm for 2004, 2005 and 2006, respectively.

Soil
Soil chemical analyses showed a good soil fertility level for grassland and animal production; however, S soil content was low (Table 1). Soil bulk density increased over the evaluation period, but there were no significant differences between treatments at the end of the experimental period ($P \leq 0.05$).

Water pathways

Results showed that the main pathway for water movement in Andisols was leaching, with 99% of total drainage ($P \leq 0.05$). No difference in the contribution of surface runoff to the total drainage was found between the two treatments ($P > 0.05$).

Nitrogen losses

Nitrate average concentration in surface runoff samples was higher than the 11.3 mg N L^{-1} established by the European Union (European Community, 1991) for water consumption. Ammonium concentration in surface runoff samples was over the Chilean Directive for quality of surface continental waters DS 87/01 (Nissen *et al*., 2000) in more than 80% of the total samples analysed. Organic N concentrations were similar to those of nitrate and ammonium in runoff samples. None of these concentrations was different between treatments in the overall analysis carried out at the end of the experiment ($P > 0.05$; Table 2).

Concentrations of NO_3^--N at each sampling date for leaching did not exceed the EU limit of 11.3 mg L^{-1} during the three years of evaluation. Annual mean values of NO_3^--N concentrations were below 7.5 mg L^{-1} for the two treatments, and below the EU limit. No significant differences were observed between treatments for mean NO_3^--N and NH_4^+-N concentrations ($P > 0.05$) for 2004, 2005 and 2006.

Cumulative N losses due to inorganic N leaching were variable for different years and ranged from 11 to 70 kg ha^{-1} yr^{-1}, where the highest values were observed in 2006.

Phosphorus losses

The animal stocking rate did not increase the overall RP and OP concentrations in runoff samples ($P > 0.05$). Average values for the three year period were 2.8 ± 0.85 and 2.4 ± 0.63 mg RP L^{-1} for the 3.5 and 5.0 steers ha^{-1}, respectively, while OP values were 0.6 ± 0.12 and 0.5 ± 0.13 mg OP L^{-1}, for each treatment, respectively (Table 3).

Overall P losses in runoff were low (< 0.05 g ha^{-1} yr^{-1}) and did not differ between treatments ($P > 0.05$; Table 3).

Potassium losses

Average K concentration in runoff samples was high and greater in the 3.5 steer ha^{-1} treatments, in relation to the 5.0 steer ha^{-1} treatment with 52 and 23 mg L^{-1}, respectively ($P \leq 0.05$; Table 4). This concentration increased over the spring in both treatments, even when no K fertilizer was applied (Figure 1). Average K concentration in leachate samples was low and did not differ between treatments being 0.35 mg L^{-1} on average for both treatments ($P > 0.05$; Table 4).

Table 2. Average N concentration in surface runoff and leaching samples and N losses in paddocks grazed by 3.5 and 5.0 steers ha^{-1} during 2004-2006 (± standard error of the mean).

Stocking rate	3.5 steers ha^{-1}			5.0 steers ha^{-1}		
Year	**2004**	**2005**	**2006**	**2004**	**2005**	**2006**
Average surface runoff concentrations, mg L^{-1}						
$N\text{-}NH_4^+$	37 ± 10.7a	10 ± 1.5b	20 ± 4.4 a	17 ± 5.0b	17 ± 3.2 a	12 ± 2.6b
$N\text{-}NO_3^-$	52 ± 19.9a	19 ± 4.6a	19 ± 4.6a	34 ± 11.4b	21 ± 6.1 a	11 ± 3.1b
Organic N	19 ± 12.9a	20 ± 3.7a	23 ± 5.4a	13 ± 6.7a	25 ± 3.8 a	15 ± 3.0b
Average leachate concentrations, mg L^{-1}						
$N\text{-}NH_4^+$	0.02 ± 0.006a	0.1 ± 0.03a	0.1 ± 0.04a	0.03 ± 0.016a	0.1 ± 0.04a	0.1 ± 0.08a
$N\text{-}NO_3^-$	1.7 ± 1.20a	7.5 ± 0.89a	4 ± 1.1a	4.7 ± 2.05a	2.0 ± 1.06a	3 ± 1.4a
Total N losses, kg ha^{-1}						
$N\text{-}NH_4^+$ in surface runoff	0.03 ± 0.002	0.02 ± 0.004	0.12 ± 0.025	0.01 ± 0.001	0.03 ± 0.006	0.11 ± 0.021
$N\text{-}NO_3^-$ in surface runoff	0.06 ± 0.003	0.05 ± 0.008	0.07 ± 0.014	0.03 ± 0.002	0.02 ± 0.009	0.08 ± 0.021
Organic N in surface runoff	0.05 ± 0.017	0.11 ± 0.017	0.21 ± 0.045	0.03 ± 0.005	0.14 ± 0.022	0.16 ± 0.036
$N\text{-}NH_4^+$ in leaching	0.2 ± 0.04	0.5 ± 0.26	0.7 ± 0.36	0.2 ± 0.10	0.6 ± 0.38	1.2 ± 0.75
$N\text{-}NO_3^-$ in leaching	10.9 ± 7.63	70 ± 8.3	40 ± 10.6	29.7 ± 13.00	19 ± 9.9	27 ± 12.7
Total N losses	11.1	70.7	41.1	30.0	19.8	28.6
Overall N losses, kg ha^{-1}		41 ± 20.4a			26 ± 12.8a	

Different letters in columns indicate significant differences between treatments according to Tukey test ($P \leq 0.05$).

Table 3. Average P concentrations and P losses in surface runoff samples collected in paddocks grazed by 3.5 and 5.0 steers ha^{-1}, during 2004-2006 (± standard error of the mean).

Stocking rate	3.5 steers ha^{-1}			5.0 steers ha^{-1}		
Year	2004	2005	2006	2004	2005	2006
Average surface runoff concentrations and range, mg L^{-1}						
Reactive P	1.1 ± 0.31a	3 ± 1.1a	3 ± 1.1a	1.7 ± 0.93a	2 ± 0.3a	2 ± 0.3a
Organic P	0.4 ± 0.13a	1 ± 0.1a	1 ± 0.3a	0.2 ± 0.08a	1 ± 0.2a	0.4 ± 0.08a
Total P losses in surface runoff, g ha^{-1}						
P-PO_4^{-2} in surface runoff	3.5 ± 0.25	4 ± 1.3	24 ± 6.4	1.5 ± 0.10	3 ± 0.7	19 ± 4.5
Organic P in surface runoff	0.5 ± 0.25	3 ± 0.5	3 ± 0.5	0.3 ± 0.04	16 ± 3.2	2 ± 0.4
Total P losses, g ha^{-1} yr^{-1}	4 ± 0.3	7 ± 0.9	27 ± 3.4	2 ± 0.1	19 ± 1.2	21 ± 2.5
Overall P losses, g ha^{-1}		13 ± 4.2a			14 ± 0.6a	

Different letters in columns indicate significant differences between treatments according to Tukey test (P ≤ 0.05).

Table 4. Average K concentrations and K losses in surface runoff and leaching samples collected in paddocks grazed by 3.5 and 5.0 steer ha^{-1}, during 2005 and 2006 (± standard error of the mean).

Stocking rate	3.5 steers ha^{-1}		5.0 steers ha^{-1}	
Year	2005	2006	2005	2006
Average surface runoff concentrations and range, mg L^{-1}				
K	22 ± 2.4 a (3-125)	83 ± 19.9 a (1-740)	23 ± 2.8 a (1-138)	23 ± 3.6 b (1-170)
Average leachates concentrations and range, mg L^{-1}				
K	0.5 ± 0.04 a (0.05-1.47)	0.4 ± 0.13 a (0.1-0.8)	0.3 ± 0.02 b (0.10-0.89)	0.4 ± 0.01 a (0.2-0.9)
Total K losses, kg ha^{-1}				
K in surface runoff	0.2 ± 0.009	0.6 ± 0.039	0.2 ± 0.005	0.3 ± 0.025
K in leachates	4.7 ± 1.47	3.3 ± 1.47	2.8 ± 0.24	3.2 ± 0.18
Total	4.9	3.9	3.0	3.5
Overall K losses, kg K ha^{-1}	4.4a		3.2a	

Different letters in columns indicate significant differences between treatments according to Tukey test (P ≤ 0.05).

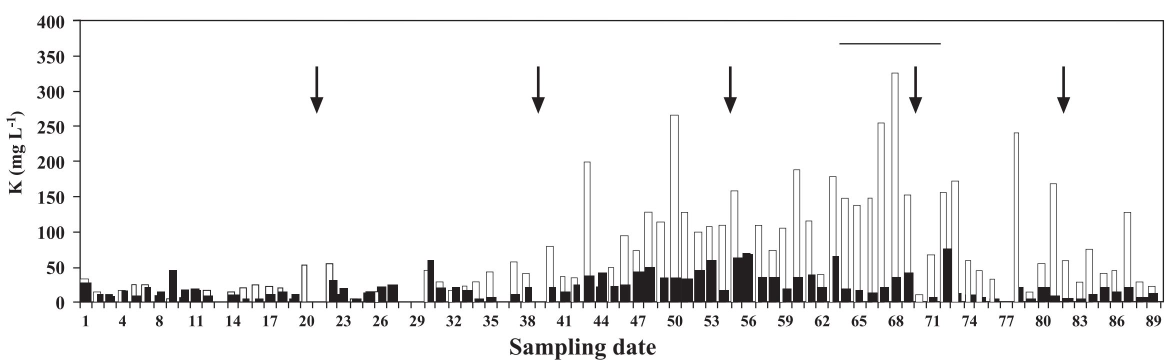

Figure 1. Potassium concentration over time (mg L^{-1}) in runoff samples collected from beef production systems with 3.5 (☐) and 5.0 (■) steers ha^{-1}. Sampling period between 02/01/2006 and 31/10/2006. (↓) grazing, (—) Potential spring mineralization period.

Potassium was mainly lost by leaching (92% on average). The overall losses were low (3 and 5 kg ha^{-1} yr^{-1}), with no differences between treatments ($P \leq 0.05$; Table 4).

DISCUSSION

Soil had an adequate P content and low acidity conditions. Sulphur (S) concentration in the soil was low, situation characteristic in soils used for livestock production in the area (Alfaro *et al.*, 2006a). There was not a significant variation of soil chemical conditions over the years, excepting for S that increased when soil samples were taken at the end of summer, probably because of soil OM mineralization (Alfaro *et al.*, 2006a). Soil bulk density results can be related to the effect of animal pouching over the years, especially during winter months (Heathwaite *et al.*, 1996).

Drainage data confirmed that in Andisoils leaching is the most important pathway for nutrient transfer and loss, especially for N, which agrees with results of Ledgard *et al.* (1999) for leaching experiments carried out in dairy systems on similar soils. It is expected that surface runoff will have a strong effect on nutrients losses only at critical times such as those of heavy rainfall occurring after fertilizer application or when the soil is dry. This agrees with results obtained by other authors for different soil types and nutrients such as N (Scholefield *et al*., 1993), P (Turner and Haygarth, 2000) and K (Alfaro *et al*., 2004).

Runoff results can be related to the low bulk density in the topsoil, so that pores are distributed vertically in this soil type, favouring vertical infiltration capacity, which resulted in a low tendency to generate surface runoff, in agreement with Dorel *et al.* (2000).

Nitrogen losses

Total N losses obtained were low compared with those measured in dairy systems on similar soils in New Zealand, using ceramic cups (Ledgard *et al*., 1999). In that experiment, leaching losses varied between 20-74 kg N ha^{-1} yr^{-1} without N fertilizer application, and increased up to 101 kg N ha^{-1} yr^{-1} after the application of 200 kg N as mineral fertilizer. The differences in N loss between both experiments can be explained by the higher amount of N recycled in urine and faeces by dairy cows. Results obtained under Chilean conditions are similar to those reported by Scholefield *et al.* (1993) for beef cattle production systems on old pastures in south West England, where average losses over 7 yr were 38 kg N ha^{-1} yr^{-1}. The lack of significant differences between treatments of the present study could be related to the high variability of ceramic cup concentrations data within each treatment, as a reflection of the animal grazing and with it, of the recycling effect in dung and urine. Also, these data could be related to differences detected in the animal behaviour, since in the 3.5 steer ha^{-1} treatment animals spent more time in the experimental plots because there was no need for feed searching, in contrast to the 5.0 steer ha^{-1} treatment, where the lower grass availability resulted in a greater animal movement within all grazing area, resulting in a more even distribution of urine and faeces over the total area, and with it, in a lower N soil accumulation in the experimental lysimeters.

The high available N concentration in runoff samples represents a risk for incidental surface water N pollution of water bodies located close to grazing areas in Southern Chile, with potential negative effects on other activities such as aquaculture and tourism, even though N losses in surface runoff were less than 0.5 kg N ha^{-1}.

No overall difference was found between treatments for the total N lost in runoff at the end of the experimental period ($P > 0.05$; Table 2). Of total N lost in runoff, ammonium and nitrate losses represented on average 26 and 23%, with no differences between treatments, as the result of urine lost in runoff immediately after excretion in each grazing and N lost after nitrate fertilizer application in spring time. DON represented 52% of total N lost in runoff, with no significant difference between treatments ($P > 0.05$). This could be related to the high soil OM content in the topsoil of the Osorno soil series (Table 1), and the expected high soil biomass activity registered at the site, which could be more relevant for N losses in runoff than animal and fertilizer management. These results agree with data from Jarvis (2002) and Alfaro *et al*. (2006b), so that in none intensively grazed pastures, DON could contribute with as much as 50% of the total N lost.

Cumulative N leaching losses due to inorganic N were variable for different years and ranged from 11 to 70 kg ha^{-1} yr^{-1}, when the highest values were observed in 2005 and 2006. High losses evaluated in 2005 and 2006 were the result of high nitrate concentration in leachates, and especially higher drainage during these years, being over 200 mm than in 2004. For the three evaluation years, a high proportion of the N was leached as NO_3^--N, with NH_4^+-N averaging less than 12% of the total inorganic N losses, in agreement with data reported by Webster *et al.* (1993) and Salazar *et al*. (2005). Dissolved organic N in leachates samples varied from 20 to 22 (2005) and from 9 to 12 (2006) kg ha^{-1} yr^{-1}, which represented 22 to 49% and 24 to 26% of the total N losses, for 2005 and 2006, respectively. According to this, DON is the most important N form in N leaching losses after nitrate.

Phosphorus losses

Average P concentrations measured in runoff samples (RP and OP) were much greater than the 50 and 25 mg total P L^{-1} established as eutrophication limit caused by anthropogenic influence for rivers and lakes, respectively (Leinweber *et al*., 2002). Peaks of RP concentration were measured associated to spring P fertilizer application during all years and were greater than those reported by McColl *et al*. (1977) for grasslands on volcanic soils. The high RP concentrations after TSF addition was probably due to the direct transport of fertilizers granules in runoff after the application and because TSP granules were rapidly solubilized by surface runoff (Heathwaite *et al*., 1998).

Peaks of OP concentration were measured during spring and probably were related to the flush of OM mineralization produced at that time of the year, in agreement with Turner and Haygarth (2000).

The stocking rate treatment did not increase P losses in runoff ($P > 0.05$). Overall P losses were low compared with results of P transfer from grazed land in Europe (Haygarth and Jarvis, 1997) and New Zealand (McColl *et al*., 1977). Values found in the present study are lower than those for P transfer found in forestry catchments in Chile (650 g ha^{-1} yr^{-1}; Oyarzun *et al.*, 1997). This can be associated mainly to the low amount of surface runoff produced in both treatments. Parallel studies carried out in the area have shown that for P loss, field slope is more relevant than animal management (Alfaro *et al.*, 2006c). Added to this, winter grazing did not increase significantly P concentration in runoff.

Greater P losses were measured during 2005 and 2006 (Table 3), given by the greater average P concentration in runoff samples after P fertilizer application in spring, because of the greater rainfall, as discussed previously. Total losses were mainly found as RP losses (70% on average), in agreement with Sharpley and Rekolainen (1997), so that P lost from volcanic soils was mainly dissolved P found in soil solution. Organic P losses represented only 30% of the total P lost in runoff.

Because of the high RP concentrations measured in this study in runoff samples collected from a volcanic soil with high P Olsen, best management practices (BMP) in relation to the timing of P fertilizer application should be adopted in the area, avoiding P fertilizer addition during winter period or high rainfall events. Also, environmental indexes for soil P content should be determined for Chilean conditions, as a complement for the traditional Olsen P soil test.

Potassium losses

Average K concentration in runoff samples increased over the spring period, probably because the increase in soil OM mineralization processes and the increase in fresh OM decomposition, in agreement with results for DON and OP data in the present study and with results of Alfaro *et al*. (2003b) for K dynamics (Figure 1).

Overall K losses were low, probably because K was not added as inorganic K. Data of the present study were lower than the 9 kg K ha^{-1} yr^{-1} found by Alfaro *et al.* (2003a) in permanent pastures managed under a grazing-cutting regime receiving 50 kg K ha^{-1} yr^{-1}. Leaching studies carried out in New Zealand reported K leaching losses between 9 and 19 kg K ha^{-1} yr^{-1} in similar soils to that used in this study (Williams *et al.,* 1990), with differences probably related to the use of milking cows in the study carried out in New Zealand, as discussed previously.

Our results indicate that it is possible to increase the stocking rate without increasing significantly nutrient losses; however, it is important to develop and adapt BMP in grazing systems to reduce N and P transfer from the grazing areas to streams and surface waters. The maintenance of buffer areas with no grazing or fertilizer application around the water bodies, the use of feed supplements to reduce the pressure on the winter sward production, and the avoidance of fertilizer applications during periods of heavy rainfall could be of high relevance to reduce the loss of N and P by runoff in grazing systems of southern Chile, in agreement with Heathwaite *et al.* (1996), specially in years of greater rainfall such as those of “El Niño” phenomenon.

CONCLUSIONS

The increase in the stocking rate from 3.5 to 5.0 steers ha^{-1} did not increase the overall N, P and K losses associated to water movement in the soil.

Losses in surface runoff were low: < 0.5 kg N ha^{-1} yr^{-1}, < 0.05 kg P ha^{-1} yr^{-1} and < 0.6 kg K ha^{-1} yr^{-1}, due to the low amount of runoff measured, because the high infiltration capacity of the Andisol in the topsoil layer.

Nitrogen leaching losses were high, ranging from 10 to 70 kg N ha^{-1} yr^{-1}. Potassium leaching losses were low varying between 3 and 5 kg K ha^{-1} yr^{-1}.

Nitrogen lost in runoff was mainly as DON (50%). Nitrogen lost by leaching was mainly as nitrate (70%). Phosphorus was mainly lost as reactive P (70%).

These results showed that stocking rates of 5.0 steers ha^{-1} (228 kg of initial live weight) are plausible in beef grazing systems of Southern Chile, but that an adequate nutrient management is required at all times during the year to reduce incidental N and P losses in runoff.

R E S U M E N

Pérdidas de nitrógeno, fósforo y potasio de un sistema pastoril con distinta carga animal en un suelo volcánico. Marta Alfaro[1 *], Francisco Salazar[1], Sergio Iraira[1], Nolberto Teuber[1], Dagoberto Villarroel[1], and Luis Ramírez[1]. En Chile existe poca información sobre la pérdida de nutrientes en sistemas ganaderos. El experimento se efectuó entre el 2004 y el 2006. Se evaluaron dos cargas animales (3,5 y 5,0 terneros ha^{-1}) bajo pastoreo rotativo con terneros Holstein Friesian sobre pradera permanente (67,5 kg N y 40 kg P ha^{-1} año^{-1}). Para cuantificar las pérdidas por arrastre superficial (N, P, K), se instalaron tres lisímetros superficiales (5 x 5 m) por tratamiento. Las pérdidas de N y K por lixiviación se estimaron con cápsulas cerámicas. Las muestras de arrastre superficial y lixiviado se analizaron individualmente para N disponible y total, P reactivo (RP) y total, y K. El nitrógeno orgánico disuelto (DON) y el P orgánico (OP) se estimaron como la diferencia entre los valores totales y los disponibles. La carga animal no incrementó las pérdidas de N, P y K (P > 0,05). Las pérdidas por arrastre superficial fueron < 0,5 kg N, < 0,05 kg P y < 0,6 kg K ha^{-1} año^{-1}, respectivamente, debido al escaso arrastre superficial medido. Las pérdidas de N por lixiviación fueron altas (11 a 71 kg ha^{-1} año^{-1}) y las de K bajas (3 a 5 kg ha^{-1} año^{-1}). El N en arrastre superficial se perdió principalmente como DON (50%). El N lixiviado se perdió principalmente como nitrato (70%). El P se perdió como RP (70%). Los resultados indican que cargas animales de 5,0 terneros ha^{-1} pueden utilizarse, pero debe evitarse la fertilización en períodos de lluvia con el fin de reducir pérdidas incidentales de nutrientes.

Palabras clave: eutrofización, calidad de agua, pastoreo, Andisol, producción de carne.

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