Soil acidification and basic cation use efficiency in an integrated no-till crop–livestock system under different grazing intensities

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ABSTRACT

Under integrated crop–livestock production system (ICLS), the animal acts as a catalyzer, modifying and accelerating fluxes by ingesting forage nutrients (grazing) and returning them to the soil as urine and dung in a continuous process whose magnitude and direction depend on grazing intensity. Thus, ICLS may change soil acidification processes and rates. The objective of this research was to verify the influence of grazing intensities on soil acidification through measurements of the soil chemical attributes in the soil profile and the efficiency of basic cation use in a soybean–beef cattle integration system nine years after surface liming under long-term no-till conditions in southern Brazil. An experiment established in 2001 in a Rhodic Hapludox with soybean (summer) and a mix of black oat × Italian ryegrass (winter) succession was used. Treatments consisted of different grazing intensities during the winter season: intensive grazing (IG), moderate grazing (MG), and no-grazing (NG). The experiment was set up in a randomized complete blocks design with three replicates. To evaluate soil chemical attributes, soil was sampled up to 40 cm, in May 2010, nine years after lime application. To quantify basic cations budgets and efficiencies, the inputs and outputs of calcium, magnesium, and potassium, as well as their initial and final exchangeable soil stocks were evaluated. Areas under grazed treatments, regardless of the intensity (IG or MG), presented lower soil acidification. Calcium and magnesium budgets were negative under NG and positive under MG. Potassium budgets were always negative, regardless of the management system, due to soybean grain harvest exportation and non-productive outputs. The soybean–beef cattle integrated system, with either IG or MG, was more efficient in calcium and magnesium utilization to produce protein; however, grazing does not affect potassium use efficiency.

1. Introduction

Acid soils represent approximately four billion ha worldwide (30–40% of arable land), which are predominant in tropical and subtropical regions (Von Uexküll and Mutert, 1995). Despite having acidic soils, these regions have lately experienced significant development in agricultural systems because of a significant increase in food demand and commodity prices (Gilbert and Morgan, 2010). To obtain high yields in agricultural systems, acidic soils require many inputs (e.g., lime and fertilizers), which has created a conflict between food production models (systems) and food security and sustainability (FAO, 2012). In many places, the use of inputs has been unbridled, which has led to economic and environmental losses (Schmieman and Van Ierland, 1999; Guo et al., 2010). The search for sustainable food production systems that optimize the use of inputs rather than achieving short-term yield increases is imperative (Lal and Pierce, 1991). The ultimate goal to be measured in the long term is for food production systems to achieve nutrient surpluses (positive budgets) over time (Urquiaga et al., 1999). According to Fixen (2011), nutrient budgets define the direction of soil fertility and agroecosystem’s efficiency, functioning as a critical sustainability indicator.

No-tillage (NT) systems have long been considered an efficient strategy for sustainable agriculture in agroecosystems; in Brazil, NT systems account for more than 32 million ha for food production (FEBRAPPP, 2012). The soil in most of these areas only remains with crops that cover the soil and produces green manure (“cover crops”) during some period of the year. Most of these cover crops present high grazing potential (forage species). Therefore, the introduction of animals into these areas produces an
integrated crop–livestock system (ICLS) that can provide a more resilient and sustainable option for food production (Russelle et al., 2007).

Through grazing, animals act as catalysts that modify and accelerate the flow of nutrients by ingesting plant biomass and returning 70–95% of the plant nutrients to soil as urine and dung (Russelle, 1997), which is a continuous process whose magnitude and direction depend on the grazing intensity (Anghinoni et al., 2013). Such a process can contribute to a decrease in nonproductive losses from leaching or surface runoff because there is synergism between root growth and partial leaf thinning by grazing that stimulates resprouting (Moraes et al., 2013), resulting in continuous growth and demands for nutrient absorption. Therefore, modifying grazing intensity in ICLS can alter soil acidification processes and rates by affecting the chemical attributes in the soil profile over time.

There are three major pathways for soil acidification in agroecosystems: animal-induced, plant-induced and soil-induced. There is a concern regarding the acidification effects of animal manure, especially urine, and under intensive grazing systems, where urine can contribute to a cycle of approximately 180 kg N ha⁻¹ year⁻¹ (Ledgard et al., 1982). In a single urination, such a rate may be as high as 200–600 kg N ha⁻¹ in the urine patch (Black, 1992), with a fast downward flux through the soil macropores and leaches beyond the root uptake zone (Haynes and Williams, 1993). In this process, if the urine-N has gone through nitrification, which produces two H⁺ for each N molecule that is nitrified, then soil acidification will occur (Bolan and Hedley, 2003).

The most important plant-induced process in a long-term approach regards basic cation exportation by agricultural products (biomass or grains) (Bolan and Hedley, 2003). Overall, legume crops such as soybean (largely used in ICLS) result in higher soil acidification compared to other crops because they contribute to a higher uptake and exportation of basic cations (Ca, Mg and K) (Slattery et al., 1991). Furthermore, biological N fixation enables soybeans to become independent of soil solution N, which increases the susceptibility of N to the nitrate leaching that occurs by pairing with basic cations (Bolan and Hedley, 2003). Under ICLS, urine-nitrate enhance such processes, which might possibly lead to environmental damage when the urine/nitrate reaches surface and groundwater, and may cause a decrease in nutrient-use efficiency under such a food production system (Robertson and Vitousek, 2009). Nitrate leaching is also the main acidification process of soil-induced processes, and it is mediated by N fertilizer use (Bolan and Hedley, 2003), which is a predominant agricultural practice for pasture grasses.

Most of the agricultural research in tropical and subtropical regions have focused on developing methods to identify limiting requirements for soil correction and on determining the rates and application methods that result in higher crop response. Despite such efforts, few approaches have been developed to determine the processes and management practices activities that cause the return of soil acidic conditions. Little is known of the long-term effect of surface liming on ICLS under NT, its acidification processes after soil correction and how such processes affect nutrient-use efficiency. Under such a scenario, the objective of this study was to verify the influence of grazing intensities on soil acidification through measurements of soil chemical attributes along the soil profile and the efficiency of basic cation use in a soybean–beef cattle integration system under NT nine years after surface liming.

### 2. Materials and methods

#### 2.1. Historical characterization and treatments of the experimental area

This experiment reports on a long-term ICLS trial that has been conducted since 2001 at the Espinilho Farm (Agropecuaria Cerro Coroado), located in Sao Miguel das Missões in Rio Grande do Sul State, Brazil (28° 57' 23" S latitude and 54° 21' 22" W longitude). The experimental area of approximately 22 ha is located at an altitude of 465 m in the Brazilian subtropics, which has a warm humid summer (Cfa) climate according to the Köeppen classification. The average temperature is 19 °C, and the yearly average precipitation is 1850 mm. The area is characterized by a declivity of 0.02–0.10 m m⁻¹, and the soil is a clayey Oxisol (Rhodic Hapludox–Soil Survey Staff, 1999), which is a deep, well-drained and dark-red soil with a clayey texture (540, 270 and 190 g kg⁻¹ of clay, silt and sand, respectively). Kaolinite and hematite are the predominant minerals in the clay and iron oxide fractions.

The experimental area has been under NT management since 1993. Prior to the trial establishment, the soil was analyzed (November 2000) (Table 1). The cattle began grazing in June of 2001 in a black oat (Avena strigosa cv. Iapar 61)+Italian ryegrass (Lolium multiflorum “common”) mixed pasture system. The soybean–beef cattle integration consisted of grazing cycles from May to November (winter season) and soybean cropping (Glycine max cv. Iguacu in the first three seasons and cv’s. Nidera RR in the remaining ones) from November to May (summer season). Black oat was seeded each year (45 kg ha⁻¹), and Italian ryegrass was established by natural reseeding. At the end of the winter season, the area was desiccated with glyphosate (900 g a.i. ha⁻¹) and ethyl chlorimuron (37.5 g a.i. ha⁻¹), and in December of each year, soybean was seeded in rows spaced 45 cm apart at a density of 45 seeds m⁻². Seed inoculation was performed as recommended (specific product), agronomic management was conducted according to the technical recommendations (use of herbicides, insecticides, fungicides), and the soybean harvest occurred every May.

Treatments consisted of grazing intensities during the winter, which were determined by the grazing pasture height, in plots ranging from 0.8 to 3.6 ha. Grazing pasture heights were 10, 20, 30 and 40 cm with an additional reference treatment (non-grazed), organized in a randomized block design with three replications. For this study, intensive grazing (1G–10 cm pasture height),

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Table 1

Soil chemical attributes before no-till integrated crop–livestock system (soybean–beef cattle) establishment (November 2000).

<table>
<thead>
<tr>
<th>Layer (cm)</th>
<th>pH (H₂O)</th>
<th>OMᵃ</th>
<th>Caᵇ</th>
<th>Mgᶜ</th>
<th>Alᵈ</th>
<th>H + Al</th>
<th>Pᵉ</th>
<th>K⁺</th>
<th>Vᵀ</th>
<th>mᵀ</th>
</tr>
</thead>
<tbody>
<tr>
<td>0–5</td>
<td>4.9</td>
<td>42.2</td>
<td>62</td>
<td>13</td>
<td>3</td>
<td>87</td>
<td>13.4</td>
<td>240</td>
<td>48</td>
<td>4</td>
</tr>
<tr>
<td>5–10</td>
<td>4.6</td>
<td>34.8</td>
<td>48</td>
<td>18</td>
<td>6</td>
<td>97</td>
<td>9.8</td>
<td>119</td>
<td>41</td>
<td>9</td>
</tr>
<tr>
<td>10–15</td>
<td>4.6</td>
<td>25.5</td>
<td>41</td>
<td>22</td>
<td>7</td>
<td>97</td>
<td>5.2</td>
<td>88</td>
<td>40</td>
<td>11</td>
</tr>
<tr>
<td>15–20</td>
<td>4.6</td>
<td>25.5</td>
<td>40</td>
<td>11</td>
<td>1</td>
<td>101</td>
<td>3.7</td>
<td>55</td>
<td>34</td>
<td>17</td>
</tr>
</tbody>
</table>

ᵃ Soil organic matter.  
ᵇ Exchangeable (KCl 1 mol L⁻¹) Ca, Mg and Al.  
ᶜ Available P and K (Mehlich-1).  
ᵈ Base saturation.  
ᵉ Al saturation.
moderate grazing (MG—20 cm pasture height) and no-grazing (NG—reference treatment) were considered. The stocking rates for IG and MG treatments were, respectively, 1350 and 947 kg ha\(^{-1}\) of live weight (average of nine years—2001–2009).

Neutered male steers (crossbred Angus, Hereford and Nellore) approximately 12-months old entered the pasture system weighing approximately 200 kg to simulate a cattle fattening or finishing system. During the grazing cycle cattle feeding was forage-based, and they were only furnished with mineral salt. A continuous grazing system was adopted (with a minimum of three remaining steers = test steers), and grazing began when the forage height reached approximately 20 cm (approximately 1.5 Mg of dry matter ha\(^{-1}\)). Therefore, each grazing cycle was carried out from the first half of July to the first half of November. Pasture heights were controlled every 14 days by the Sward stick method (Barthram, 1986), which consists of a graduated stick measuring system with a “marker” that slides up and down until the first forage leaf blade is reached. In each plot, approximately 100 randomized readings (points) were conducted. The average pasture height resulted from managing the grazing intensity (stocks) by adding or removing steers from each plot as required.

2.2. Liming and fertilizations

At the beginning of the experiment, after the first grazing season and before the first soybean cropping (in November of 2001), surface liming was performed for the entire area at a rate of 4.5 Mg ha\(^{-1}\). Lime presented a total neutralization relative power (TNRP) of 62%. Fertilization over time was the same in all treatments and based on the Soil Chemistry and Fertility Comission of the Rio Grande do Sul and Santa Catarina States (COFES RS/SC, 2004); after consideration of soil analysis, and fertilization consisted of N applications during pasture season and P and K applications during soybean season (Table 2) to achieve yields of approximately 4.0–7.0 Mg ha\(^{-1}\) of forage dry matter and 4.0 Mg ha\(^{-1}\) of soybean grain.

2.3. Data used from previous studies

To produce a detailed approach, especially regarding soil basic cation budgets and use efficiencies, data from additional studies that had been carried out in the same experiment were required. Therefore, the information from four studies were used to construct a long-term approach. The first study reported on soil chemical and physical attributes during the experiment establishment (Cassol, 2003); the second study reported on steer carcass yields (Lopes et al., 2008); the third study was on animal manure production and distribution (Silva, 2012); and the last study reported on nutrient cycling and decomposition rates of green and animal manure (Assmann, 2013);

2.4. Sampling and analysis

The soil was sampled in May 2010 following the soybean harvest. Sampling was performed up to a 40 cm depth to evaluate the chemical attributes in the soil profile, and samples were stratified in twelve layers: 0–2.5, 2.5–5, 5–7.5, 7.5–10, 10–12.5, 12.5–15, 15–17.5, 17.5–20, 20–25, 25–30, 30–35 and 35–40 cm. Four trenches were opened in each plot, and the soil samples were shoveled (0–20 cm layer) and augured (20–40 cm layer), and a ruler and a spatula were used for stratification of the layers described above.

The samples were carried in plastic bags to the Soil Fertility Research Laboratory of the Federal University of Rio Grande do Sul (UFRGS), dried, crushed (2 mm mesh) and kept in plastic pots. The soil samples were analyzed according to the method of Tedesco et al. (1995) for the water pH (pH-H2O, 1:1 ratio) and SMP index (to estimate the potential acidity of H+Al); exchangeable Ca, Mg and Al (KCl 1 mol L\(^{-1}\)); and available K (Mehlich-1). The soil organic carbon (SOC) was quantified by dry combustion with a Shimadzu TOC-V CSH in the Environmental Biogeochemistry Laboratory of UFRGS.

2.5. Basic cation budgets and use efficiencies

For basic cation budgets, exchangeable Ca, Mg and K contents in the 0–40 cm soil layer were analyzed (Tedesco et al., 1995) in November 2001 [data from Cassol (2003)] and May 2010. The average soil bulk density of all of the treatments at the beginning of the experiment [1.35 kg dm\(^{-3}\); reported by Cassol (2003)] were used to calculate the soil exchangeable Ca, Mg and K stocks based on the equivalent soil mass method (Ellert and Betanny, 1995).

To calculate the Ca inputs via phosphate fertilizer, Ca concentrations of 16 and 10% in single superphosphate and triple superphosphate were considered. The Ca and Mg liming inputs were based on a lime analysis (30% CaO and 19% MgO). For K, only the amount of added K based on the K\(_2\)O applied rates was considered.

The system outputs were based on nine soybean seasons (2001/02–2009/10) and eight grazing seasons (2002–2009). The soybean yields were determined each year according to a 10 m (five random 2 m spots in each sowing row) manual harvest in each plot. The live weight production of the steers (animal output = meat + bones + bristle) for each grazing cycle was calculated by weighing animals before entering and after leaving the grazed areas. The soybean Ca, Mg and K grain contents were analyzed according to the method of Tedesco et al. (1995), and the average data were used to calculate the soybean outputs based on the following amounts: 2.77 g Ca kg\(^{-1}\), 2.80 g Mg kg\(^{-1}\) and 17.97 g K kg\(^{-1}\). Animal output was determined according to Price and Schweigert (1994) and the used values were 1.55 g Ca kg\(^{-1}\), 1.39 g Mg kg\(^{-1}\) and 1.71 g K kg\(^{-1}\).

### Table 2

<table>
<thead>
<tr>
<th>Nutrient</th>
<th>Fertilizer rate (kg ha(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Soybean season</strong></td>
<td></td>
</tr>
<tr>
<td>P(_2)O(_5)^a</td>
<td>60</td>
</tr>
<tr>
<td>KCl^b</td>
<td>90</td>
</tr>
<tr>
<td><strong>Pasture season</strong></td>
<td></td>
</tr>
<tr>
<td>N^c</td>
<td>45</td>
</tr>
</tbody>
</table>

\(^a\) The source used was single superphosphate in the first soybean season (2001/02) and the second forage season (2002); and for the remaining years, triple superphosphate was applied.

\(^b\) The source used was potassium chloride.

\(^c\) The source used was urea. N rates higher than 45 kg ha\(^{-1}\) were separated into two top-dressings at 30 and 60 days after forage establishment.
Equation 1 determined the non-productive losses (NPL) of the exchangeable basic cations by leaching and surface runoff (erosion):

$$\text{NPL} = (S_{\text{initial}} + R_{\text{initial}} + I_{\text{lime}} + I_{\text{fertilizer}}) - (O_{\text{soybean}} + O_{\text{animal}} + R_{\text{final}} + S_{\text{final}})$$

where $S =$ the soil stocks (initial = November 2001 and final = May 2010); $R =$ the remaining nutrients in the residue at soil sampling (initial = November 2001 and final = May 2010); $I =$ the inputs (lime = liming and fertilizer = fertilization); and $O =$ the outputs (soybean = soybean grains and animal = meat, bones and bristle).

The estimation of the Ca, Mg and K contents in the remaining residue (pasture, dung and soybean shoots) was based on the residue production during November 2001 and May 2010 and decomposition rates reported by Assmann (2013). Pasture residue production was determined in November 2001 and 2009 by cutting and sampling 0.25 m² areas (three random samples per plot); for soybean, 4.5 m² areas (five random samples of 0.9 m²) were cut and sampled in full bloom (R2 reproductive stage). Animal dung dry matter production was determined using data from Silva (2012). The final cation budgets resulted from subtracting the system stocks (soil exchangeable stocks + nutrient remaining in residue) in May 2010 from the initial stocks in November 2001.

The basic cation efficiencies for protein production (use efficiency) were calculated using the soybean N content according to the Kjeldahl method (Tedesco et al., 1995). The N content was multiplied by 5.71 (Merrill and Watt, 1973), which resulted in protein values between 38 and 42% over time. For meat, the considered carcass yield was 53.3% (Lopes et al., 2008), and a protein value of 20% was used (Price and Schweigert, 1994). Equation 2 determined the Ca, Mg and K use efficiency (UE):

$$\text{UE} = \frac{\text{Prot}_{\text{total}}}{O_{\text{total}}}$$

where Prot$_{\text{total}}$ = the total protein produced throughout nine years from soybean grains and cattle meat and O$_{\text{total}}$ = all of the system outputs throughout the nine years (soybean, animal and non-productive).

2.6. Statistical analysis

Data were submitted to an analysis of variance (ANOVA), and in cases of significant differences ($p < 0.05$), the averages were
compared by Tukey’s test ($p < 0.05$). The following statistical models were used for the ANOVA:

**a) Chemical attributes in the soil profile:**

$$Y_{ijk} = \mu + B_i + C_j + \text{Error}(ij) + L_k + \text{Error}(ik) + G_{ij}L_k + \text{Error}(ijk)$$

where $\mu$ = the overall experiment average; $B_i$ = the blocks ($i = 1, 2, 3$); $C_j$ = the grazing intensity ($j = 1, 2, 3$); $L_k$ = the soil layer ($k = 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12$); and Error = the experimental error.

Regarding the soil attributes graphs, when interaction between the treatment (grazing intensities) and depth (soil layers) occurred (two-way ANOVA, “Treat. vs Depth” variation source), the principal effect (only “Treat.” or only “Depth”) was ignored and a bar was placed to indicate the least significant difference (LSD) comparison of treatments at each depth. When the effect was principal (only “Treat.” in the average of all of the depths or only “Depth” in the average of all of the treatments), the LSD value was informed and letters were used to distinguish the means (lowercase letters in the vertical direction for depths and uppercase letters in the horizontal direction for treatments). The same letters or absence of statistical information indicated that no significant difference ($p > 0.05$) was detected.

### 3. Results and discussion

#### 3.1. Soil acidity attributes

In May 2010, nine years after the surface liming, the soil pH-H$_2$O (Fig. 1a) did not differ among grazing intensities ($p > 0.05$). The soil pH showed decreasing stratification along the soil profile, from 5.0 in the upper layer (0–5 cm) to a minimum pH value of 4.4 in the deepest evaluated layer (35–40 cm). In soils under NT conditions, the organic material is returned to the surface (litter) and soil acidity stratification is common (Sá et al., 2009). McLaughlin et al. (1990) reported that lower soil acidity in the upper layers results from the return of organic anions through plant and animal residues, organic anion oxidation in the surface soil and legume proton excretion in the deeper layers. This behavior is accentuated especially under low-diversity intensive food production systems where there is a lack of fauna for soil mobilization.

The base and aluminum saturation (Fig. 2b and c) followed the soil pH patterns, with stratification occurring along the profile. However, these attributes varied in the soil profile among the treatments ($p < 0.05$) without interaction ($p > 0.05$) between the soil depth and treatments. Grazing intensity (IG or MG) did not affect these attributes; however, the presence or absence of animals during the winter did affect such attributes. In the treatments with and without grazing, the average (0–40 cm) base saturation was 45 and 35%, and the average aluminum saturation was 27 and 44%, respectively. Therefore, the soil profile under ICLS was less acidic as compared to the NT-only treatment with cover crops during winter (NG).

Because treatments did not affect the soil pH-H$_2$O (Fig. 1a) but the other acidity attributes were affected (Fig. 1b and c), the different relationships among these attributes based on the grazing intensity were verified (Fig. 2). In areas under long-term NT in the Brazilian subtropical region, the relationship between soil pH and base saturation indicated that a pH-H$_2$O of 6.0 corresponded to a base saturation of 80% [CQFS RS/SC, 2004]. In the current results, if the linear models are adjusted for such a relationship between soil pH and base saturation and curve points are extrapolated (Fig. 2a), 80% base saturation is attained at a pH of 5.0 under the ICLS (IG or MG) and of 5.5 under NT with winter cover crops (NG). However, the common model for this relationship is a polynomial, since from a pH-H$_2$O of 5.5 the curve slope (change in base saturation) tended to decrease. Because the pH reached a maximum value of 5.2, the linear models were the best fit ($R^2 < 0.90$). However, considering the pH range in this study (4.4–5.2), when beginning at a pH of 4.5, each decimal increase in pH corresponded to a higher increase in base saturation under the ICLS (10.7 and 10.2% for IG and MG, respectively) compared to NG (5.4%) (Fig. 2a).

A reverse relationship was observed between pH-H$_2$O and aluminum saturation (Fig. 2b), wherein each decimal increase in pH led to a decrease of 11.7, 11.2 and 6.6% in aluminum saturation for IG, MG and NG, respectively. Furthermore, aluminum saturation below 10%, which was regarded as the threshold value for avoiding soybean phytotoxicity [CQFS RS/SC, 2004] was reached at a pH of 4.8 under the ICLS treatments (despite grazing intensity); under NG, however, the same value was reached at a pH of 5.1. These results indicated that at the same soil pH, soil under an ICLS was less acidic compared to NG with higher base saturation and lower aluminum saturation.

#### 3.2. Soil basic cations, aluminum and cation exchange capacity

The ICLS treatments resulted in higher exchangeable Ca and Mg contents along the soil profile (Fig. 3a and b). However, the available K (Fig. 3c), regardless of grazing intensity, was only higher under the ICLS in the 0–2.5 cm layer, and the value did not differ ($p > 0.05$) from NG below this layer.

Ca$^{2+}$, Mg$^{2+}$ and K$^{+}$ represented the soil basic cations in the calculations. The predominant cations in the soil sorptive complex were Ca$^{2+}$ and Mg$^{2+}$ (approximately 90% of CEC$_{TOT}$) as a result of higher-strength bonds in the solid phase following the hydrotropic series [Sparks, 2003]. Thus, these cations determined the base saturation behavior in the soil profile (Fig. 1b), despite K (Fig. 3c) not following such a pattern ($p < 0.05$) at depths below 2.5 cm.
Approximately 80% of the K ingested by bovines is returned to the system as urine in a concentrated form and increase K’ leaching in the soil (Haynes and Williams, 1993), combined with the binding preference in soil sorptive complex of Ca$^{2+}$ and Mg$^{2+}$.

Soil acidification below a pH of 5.0 was followed by aluminum solubilization from soil minerals. As a result, base saturation decreased and aluminum saturation increased because Al$^{3+}$ is preferably bonded in the soil solid phase, which favored basic cation leaching (Ca$^{2+}$, Mg$^{2+}$ and K$^{+}$) (Sparks, 2003). In this trial, the exchangeable Al$^{3+}$ contents (Fig. 3d) were lower under the ICLS (IG or MG) up to 15 cm deep with the exception of the top layer (0–2.5 cm), in which differences among treatments were not verified ($p > 0.05$). Such a pattern indicated that the higher aluminum saturation in the soil profile under NG (Fig. 1c) was a result of lower Ca$^{2+}$ and Mg$^{2+}$ contents in the soil profile (0–40 cm, Fig. 3a and b) and higher Al$^{3+}$ contents up to 15 cm (Fig. 3d). The introduction of species or modifications in the soil management, such as grazing (Lorenz and Rogler, 1967) to stimulate continuous shoot growth and root development in the deeper soil layers help to decrease the soil acidification rates (Coventry et al., 2003).

The effective cation exchange capacity (CEC) (Fig. 4a) and CEC at a pH of 7.0 (Fig. 4b) presented a similar behavior to what was observed for Ca$^{2+}$ and Mg$^{2+}$ (Fig. 3a and b), with higher values in the grazed areas (IG or MG) compared to the NG treatment. In addition, behavior in the solid phase can be related to the liquid phase (soil solution), with lower base saturation and CEC in the NG area possibly leading to lower ionic force in the soil solution, which can increase the aluminum activity and phytotoxic effects (Anghinoni and Salet, 1998).

### 3.3. Relationship between soil organic carbon and cation exchange capacity

The soil mineral composition under tropical and subtropical regions strengthens the relationship between soil CEC and SOC because the organic matter in highly weathered soils is largely responsible for negative charges, which are pH-dependent (Sparks, 2003). Under an ICLS, however, the pattern of such relationships has not been determined. Nine years after the experiment began, the treatments affected the SOC ($p < 0.05$) up to a 10 cm depth, with lower values under the IG treatment compared to the other treatments (Fig. 5). As previously discussed, the CEC (Fig. 4) was higher in the grazed treatments (IG or MG) compared to the NG treatment (reference area), even with similar pH values (Fig. 1a).

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Fig. 3. Calcium (a), magnesium (b), potassium (c) and aluminum (d) contents in the soil profile after nine years of no-till integrated crop-livestock system (soybean–beef cattle), with different grazing intensities. Tukey test ($p < 0.05$): upper case letters distinguish treatments in the average of depths; lowercase letters distinguish depths in the average of treatments; the bar indicates LSD comparing treatments in each depth; equal letters or absences of statistical information indicate that there was no difference.
Thus, the same SOC contents result in different effective CEC according to grazing intensity, with IG > MG > NG (Fig. 6).

Such behavior can be explained by the addition of different residues forms in each treatment, especially animal manure in the grazed systems. The soluble organic carbon from animal dung has a molecular weight approximately three times higher than that of plant residues (Iymuremye and Dick, 1996) because of the amount of functional groups; therefore, its reactivity is increased as well. In the same experiment, Silva (2012) reported that the surface covered by dung plates was almost 2% under IG (10 cm pasture height), and compared to plant residues, such a value may seem small. However, because of its disposition and fibrous contents (cellulose, hemicellulose and lignin), the decomposition of the recalcitrant fraction of dung have a half-life that ranges from 225 to 485 days (Assmann, 2013). The high reactivity of dung’s organic carbon (because of functional groups such as carboxylic) is associated with its slow decomposition, which impacts soils over time and can contribute to a higher ratio of CECs in an ICLS for the same SOC content (Fig. 6). Furthermore, leaf thinning by grazing can stimulate root production (Lorenz and Rogler, 1967) and increase the root exudation (Jones, 1998) and rhizospheric soil (Nicol et al., 2003). The continuous root exudation can also contribute to higher organic carbon reactivity under grazed systems.

3.4. Basic cation budgets

To best understand the processes and causes of soil acidification and chemical attributes nine years after surface liming under ICLS conditions, Ca, Mg and K budgets were calculated (Tables 3, 4 and 5). The added amounts (inputs) of these three nutrients were the same for the three treatments (systems). After nine pasture-soybean seasons, however, different management strategies led to different outputs.

Higher acidification under the ICLS as a result of two output sources (soybean and animals) were expected; however, acidification by animal outputs was very low (Tables 3, 4 and 5), which confirmed the role of animals as catalyzing and cycling agents (Anghinoni et al., 2013; Moraes et al., 2013). Therefore, by the
additional nutrient inputs can also improve nutrient availability and lead to higher productivity (Fig. 1) because the accumulated yield of nine years did not differ (p > 0.05) between grazing treatments (Fig. 7). According to Coventry and Slattery (1991): grain and animal outputs weakly contribute to the soil acidification process. Productive exportations only corresponded to 4.5, 7.5 and 2.5% of the Ca outputs (Table 3) and 6.8, 8.1 and 4.8% of the Mg outputs (Table 4) for IG, MG and NG, respectively. The low exportation of Ca (Table 3) and Mg (Table 4) via animals and soybean compared to the exchangeable stocks of the nutrients requires attention. Such outputs over nine years represented a small decrease of approximately 0.8 and 1.4 mmol·kg⁻¹ of soil exchangeable Ca²⁺ and Mg²⁺. The non-productive outputs (leaching and surface runoff) of Ca and Mg were higher under NG compared to IG and MG (Tables 3 and 4), and such losses are able to explain the higher Ca and Mg contents (Fig. 4a and b) along the soil profile under the ICLS compared to the NG treatment, which also resulted in higher CEC (Fig. 4) and base saturation (Fig. 1b) and in lower aluminum saturation in depths up to 40 cm (Fig. 1c).

The Ca budgets were negative under the IG treatment, significantly negative under the NG treatment (Table 3) and positive under the MG treatment. For Mg, only the grazed areas (ICLS) presented surpluses (positive budgets) (Table 4). High leaf thinning intensity can lead to lower pasture growth rates, which can decrease the nutrient uptake. Thus, grazing intensity is an important factor for determining grazing acidification rates (Unkovich et al., 1998). Although intensive grazing results in higher meat production (Fig. 7), this management approach also increases soil acidification through animal urine (Orr et al., 2011). Approximately 15% of the N in urine can be lost from the soil surface through macropore drainage (Williams et al., 1990). As a result of the higher amounts of soil exchangeable Ca²⁺ compared to Mg²⁺ and K⁺ (Fig. 3a, b and c; Tables 3 and 4), there was a higher probability of pairing Ca²⁺ to NO₃⁻ and other anions, such as SO₄²⁻ and Cl⁻ (provided by residue decomposition and fertilizers), which could lead to leaching in the soil profile.
Soil acidification in agricultural systems is strictly dependent on the water dynamics in the soil–plant–atmosphere system. Basic cation losses through leaching occur when cations are paired with anions, which are usually nitrate (NO$_3^-$) anions because they prevail in the soil solution; therefore, such losses depend on the movement of water throughout the soil (Addiscott, 2004). Nitrate generally exhibits low interaction with soil minerals. Moreover, the predominance of negative charges in the soil colloidal surface leads to NO$_3^-$ leaching to deeper layers. At these layers, it is unavailable for root extraction and N cycle is not completed. Thus, this process results in a proton surplus in the NO$_3^-$ original layer in addition to the loss of a basic cation (e.g., Ca$^{2+}$, Mg$^{2+}$ or K$^+$), which increases the surface soil acidification (Bolan and Hedley, 2003). According to Black (1992) it is extremely difficult to develop management strategies that control nitrate leaching and prevent soil-surface acidification. Furthermore, leaching from agricultural systems is a major contributor to nitrate accumulation in the surface and groundwater (Robertson and Vitousek, 2009), which leads to environmental damage. Therefore, measuring non-productive basic cation outputs is an easy way of comparing different agriculture systems based on environmental issues.

Furthermore, most of the water that percolates along the soil profile is responsible for leaching, as it flows around aggregates, with the water inside the layer remaining immobile. Nitrate and other solutes move slowly by diffusion from inside the aggregates to the external soil solution (Addiscott, 2004). Thus, systems that increase soil aggregation can help prevent nitrate leaching issues. In this regard, Souza et al. (2010) reported an increase in soil aggregation with the ICLS treatment and especially with the MG treatment compared to the NG treatment. As soil aggregation evolves with time, the magnitude of its impacts is also expected to increase.

Other studies that employed this approach concluded that the introduction of species or modifications in soil management that allow for continuous shoot growth and root development in deeper layers, such as in grazed systems, promote decreases in soil acidification rates (Convery et al., 2003), which can mitigate environmental problems based on nutrient leaching. In this context, Syswerda et al. (2012) verified that NT systems operating in the Midwestern USA that employed cover crops and reduced inputs could help to reduce nitrate leaching.

The K contents in soybean grains are six times higher than those of Ca and Mg; therefore, its productive outputs correspond to 46, 48 and 47% of the total outputs for IG, MG and NG (Table 5). Most of the fertilized K was exported as soybean grains, which resulted (Table 5) in negative K budgets as a result of non-productive losses despite the adopted system. Regarding the adopted systems (grazed or non-grazed), K fertilization requires more attention under continuous soybean crop.

The impacts of K outputs under ICLS were previously reported by Ferreira et al. (2009). As discussed, most of the animal-ingested K returned to the system as urine in a concentrated form and at a low soil volume (Haynes and Williams, 1993), which increased K leaching. Under an intensive grazing system with approximately three steers ha$^{-1}$ (with 450 kg of live weight each), it is estimated that 230 and 180 kg ha$^{-1}$ year$^{-1}$ of K and N, respectively, return to the pasture as urine (Ledgard et al., 1982). Such inputs, however, occur at a flow rate of 0.2 L s$^{-1}$ (Goodall, 1951), and under urine patches, the flow may reach 600 kg N ha$^{-1}$ (Black, 1992). Therefore, plants do not recycle the majority of nutrients that return through animal urine (Haynes and Williams, 1993). However, NG areas with no leaf thinning stimulus reached maturation earlier (Aguinaga et al., 2008). Therefore, because K is not a plant structural component, it becomes “free” after the maturation stage is reached (Taiz and Zeiger, 2004) and returns to the soil system through precipitation (plant washing). Furthermore, because it pairs with NO$_3^-$ or other anions, K leaching can also be a source of K loss.

Even with the grazing stimulus for root development in deeper layers, the diminishing rates of cation leaching, which occurred with Ca$^{2+}$ and Mg$^{2+}$ (Tables 3 and 4), and urine dynamics (Table 5) result in similar unproductive outputs (losses) of K under grazed (ICLS) and non-grazed (NT) systems. Therefore, adequate grazing management with moderate grazing intensities appears to be the best management strategy to equilibrate these two processes (urine losses and early culture maturation) (Table 5).

### 3.5. Basic cation use efficiency

Higher Ca and Mg outputs under NG due to non-productive losses resulted in lower basic cation protein production efficiency (Fig. 8). Despite the lack of protein production during the winter under such management, the total protein production compared to that of grazed areas (ICLS) did not differ (p > 0.05) (Fig. 7), which demonstrated that the lower Ca and Mg use efficiencies (Fig. 8) resulted from their global budgets (Tables 3 and 4). On average, each kg of Ca added to the soil in treatments with ICLS with IG or MG or without animals (NG = NT with winter cover crops) resulted in 9.5 and 3.7 kg of protein, respectively, and each kg of Mg resulted in 29.6 and 13.0 kg, respectively. Treatments did not affect (p > 0.05) the K use efficiency (Fig. 8), and each kg of K
added to the soil resulted in an average production of 14.4 kg of protein.

In addition to adopting MG under an ICLS, which is regarded as the best ICLS management option, other mitigation strategies may be adopted to decrease non-productive nutrient outputs during the winter season. Of particular interest is the use of slow release N fertilizers to decrease losses and acidifying potential; in such fertilizers, N release rate of the fertilizer granules follows the plant uptake rates (Hirel et al., 2011), which increases the N efficiency and decreases NO3− leaching (Tang and Rengel, 2003).

Di and Cameroon (2004) observed lower Ca2+, Mg2+ and K+ leaching losses in the presence of nitrification inhibitors, which helped retain N in negatively charged soils by retaining N in the ammonical form (NH4+). Therefore, N is available for a longer period for pasture uptake and results in higher biomass production. In this study (Di and Cameroon, 2004), Ca losses were approximately 120 kg ha−1 year−1 compared to 170–190 kg ha−1 year−1 without N fertilizer inhibitors. In the present trial, the values obtained for IG, MG and NG were 172, 101 and 279 kg ha−1 year−1, respectively. Such results reinforce that a well managed ICLS is an efficient strategy to reduce soil leaching that does not require specific expensive fertilizers.

4. Conclusions
Despite grazing intensity, after nine years of surface liming under an integrated crop–livestock system (soybean–beef cattle), soil acidification was lower in grazed areas, with higher base saturation and lower aluminum saturation in the whole soil profile analyzed (0–40 cm), compared to the non-grazed areas. However, calcium surpluses were observed only in integrated areas with moderate grazing; while magnesium surpluses occurred with intensive or moderate grazing. The negative budgets were result of higher non-productive losses, especially under non-grazed conditions. Regarding potassium, all managements resulted in negative budgets, due to higher soybean grain exportation and non-productive losses (leaching). Therefore, the integrated production system was more efficient in the utilization of calcium and magnesium to produce protein, and the potassium use efficiency was not affected by the animal grazing.

Acknowledgements
We would like to thank Adao Luís Ramos dos Santos for the support provided for the laboratorial analysis and field activities. We also thank the National Council for the Development of Science and Technology (CNPq) and the Coordination for the Improvement of Higher Education Personnel (CAPES) for financial and scholarship support.

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