**Original Research Article**

**Rangeland Grazing Management in Argentine Patagonia**

**Running title:** Rotational Grazing in Patagonia

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**Novelty statement:**

* The humid Magellanic grass steppe static rotational grazing can increase soil water infiltration rates and soil N
* Continuous grazing has a greater proportion of bare soil, and grasses and graminoids
* Intensive grazing negatively influence soil water, soil, and vegetation parameters
* Rotational grazing has a potential to store more biomass than continuous grazing

**Abstract**

Rotational grazing is recommended as a management tool to sustain productivity and improve soil health of permanent grassland ecosystems. The aim of this project was to assess whether rotational grazing was a beneficial alternative to continuous grazing for the local environment, when using a moderate or high grazing intensity in the Argentine Patagonia. The parameters investigated were: 1) Soil water infiltration and soil water retention capacity. 2) Soil organic matter, soil N, soil erosion, and soil respiration. 3) Plant composition profiles, ANPP, biomass above- and below-ground, physical plant characteristics, and species diversity. In the humid Magellanic grass steppe static rotational grazing was found to increase soil water infiltration rates, soil N, proportions of forbs and shrubs, root/aerial plant ratio, number of plant species, and proportion of dead plant. Continuous grazing had a greater ANPP, proportion of bare soil, and grasses and graminoids. In Silvopastoral Andean vegetation, rotational grazing resulted in increased root biomass, root/aerial plant ratio, and proportion of forbs compared to continuous grazing, which was found to increase soil organic matter, soil N, plant length, root depth, aerial plant biomass, soil respiration, proportion of bare soil and dead plants compared to rotational grazing. Intensive grazing negatively influenced soil water, soil, and vegetation parameters. The results indicated that rotational grazing has a potential to store more biomass than continuous grazing.

**Keywords:** Grassland management, soil health, Patagonia, grazing

**Introduction** The Pategonia is a sparsely populated region of over 1 million km2 in Chile and Argentine. The landscape of Agentine Patagonia is arid (Aagesen, 2000) and the vegetation cover of grasslands varies from 60% or more to less than 10% in the most arid areas. Southern Patagonia is dominated by extensive livestock production systems with a restricted grass growth production period of 5 to 7 months due to low winter temperatures and water stress (Aagesen, 2000). The most frequently used management system in Patagonia is continuous grazing with fixed stocking rates in paddocks varying from 1,000 to 20,000 ha with only a few farmers practicing rotational grazing systems (Ormaechea and Peri, 2015).

The southern part of Patagonia (Santa Cruz and Tierra del Fuego provinces) is predicted to experience temperature increases of 2-30 C in the next 65 years. This increase will have a critical effect on the desertification of ecosystems in the region (Peri, 2011). Valle et al. (1998) mapped most of the Patagonian region according to the level of desertification and found that 9.3% was undergoing light desertification, 17.1% moderate, 35.4% moderate to severe, 23.3% severe, 8.5% very severe, and only 6.4% of the region’s land shows no signs of desertification.

The nutrient pools are relatively small in these arid rangelands, and any decline in nutrient stocks will have an impact on the annual plant productivity. A decrease in the aboveground biomass will involve a decline in both soil carbon and nitrogen. Soil nitrogen loss may be due to either nitrogen lost in surface runoff and vegetation removal by livestock or both (Gallardo and Schlesinger, 1992).

 Sheep rearing has, through history, been thought to cause the reduction of vascular plant diversity through extinction of preferred forage species (Bertiller and Bisigato, 1998). Invasion of shrubs has resulted in a significant loss of nutrient-rich topsoil (Aagesen, 2000). Grazing is also deemed responsible for a trampling effect that has destroyed the soil crust components (Scutari et al., 2004), and increased soil compaction, which in turn may be the cause of decreased infiltration and increased runoff (Schlesinger et al., 2000). Increased runoff creates faster flows in waterways, with more soil being lost, and sediment loads increased.

In the ecosystems of Patagonia, soil carbon represents 79-90% of the total carbon pool, depending on plant and environmental conditions (Peri, 2011). Peri et al. (2015) found that, across Patagonia, there is a significant difference in the soil respiration rates of grasslands with different vegetation composition. A higher soil respiration was seen in grasslands with trees than in those with only grasses and forbs. In addition, Peri et al. (2015) found that long-term intensive grazing decreases the soil respiration rate in grassland ecosystems. However, a thorough characterization and relationship between grazing intensity, grazing management systems and indicators of ecosystem health has never been established for this area. Therefore, a study was conducted to establish the relationship between indicators of ecosystem health, grazing intensity, and management systems in Southern Patagonia.

**Materials and methods**

The data was collected from 2012-2015 in Southern Patagonia in the province Santa Cruz, in permanent plots established as a part of PEBANPA network (Biodiversity and Ecological long-term plots in Southern Patagonia) (Peri et al., 2016). The following factors were measured: number of animals per ha, soil organic matter, soil nitrogen, soil respiration, vegetation types, annual net primary production (ANPP), soil erosion, aerial plant biomass, root biomass, root/aerial plant ratio, root length, plant diameter at base, plant length, proportion of dead plants, soil water infiltration, soil water retention capacity, number of plant species, and dominant plant community. Each measured factor had 3-5 replicates per year, for 4 consecutive years.

 **Study sites**

The study sites included four ecological areas between the latitudes 48o and 55oS: Dry Magellanic grass steppe, humid Magellanic grass steppe, Mata Negra Matorral thicket, and silvopastoral Andean vegetation. Hereafter called: dry grass steppe, humid grass steppe, matorral thicket, and Andean vegetation. The four systems are grazed at three grazing intensity categories: intensive, moderate, and low. The grazing intensity categories are determined by the National Agricultural Technology Institute (INTA), and are based on forage capacity of the ecological areas. The values for low, moderate and intensive grazing are, therefore, not equal in the four ecological areas. One site at each ecological area is managed with a 6 month rotational grazing management system, at the, predefined moderate grazing intensity, while the other sites practice continuous grazing. However, only data from the Andean and Humid grass steppe ecosystems were available for comparisons between grazing systems.

FIGURE 1 here

The dry and humid grass steppe covers 3 million ha. with grasses and shrubs as the dominating plant types. The dominant tussock species of these ecosystems are *Stipa chrysophylla* and *Festuca pallescens*, commonly associated with cool season *Poa dusenii* and *Carex andina* short grasses (Peri, 2011), and these grasses cover 85% of the area (Peri and Bloomberg, 2002).

Matorral thicket consists of shrubland, dominated mainly by *Junellia tridens*. Matorral thicket covers 2.8 million ha. in between the grasslands. Water is the most important factor regulating primary production in this area (Food and Agriculture Organization of the United Nations, 2005). Shrublands play an important role in the southern Patagonian landscapes by providing a large number of important ecosystem services, such as soil fertility and richness, as well as the bulk of the biomass in the understory plant communities (Soliveres and Eldridge, 2014).

Andean vegetation consists of deciduous *Nothofagus antarctica* forests used for silvopastoral systems with livestock feeding on natural grasslands that grow in the understory of thinned forests. The forest is thinned by sheep producers to maximize understory forage production. Forest thinning is a matter of balance since too much thinning of the forest may increase the evaporation and decrease the forage production, while too little thinning will hamper light penetration through the canopy and reduce forage production (Peri, 2011).

 **Climate conditions**

The annual precipitation across the region varies from 4000 mm at the foot of the eastern Andes to 150 mm in the central plateau 180 km east of the mountains (Soriano, 1983). The east coast is dominated by moist air from the Atlantic sea with annual precipitation evenly distributed (200-220 mm), in contrast to the seasonal winter rainfall in the remaining region (Soriano et al., 1980; Paruelo et al., 1998). The climate of the region is generally dry, cold and windy. The windy season is from November to March with south winds and frequent windstorms occurring in the summer and spring months, with intensities up to 120 km/h (Peri and Bloomberg, 2002).

**Data collection**

 **Soil and water samples**

The *soil water retention capacity* (mm/cm) was measured in the top layer of the soil by taking soil profiles from 0-30 cm length. The profiles were air dried and sieved (< 2mm) prior to the determination of water retention curves with plates as described by Richards (1948). This method determines the value of soil moisture depending on the matric potential (-1500, -300, -100, -33 and -10 kPa). Gravimetric moisture in the soil at field capacity (-10kPa) and gravimetric moisture at permanent wilting point (-1500 kPa) were calculated from the water retention curves to determine the available water retention capacity where: Available retention capacity = field capacity - permanent wilting point.

The *soil water infiltration* data was measured with the double-ring infiltrometer method (ASTM International D3385-09, 2009). The method consists of driving two open cylinders, one inside the other, into the ground, partially filling the rings with water and then maintaining the liquid at a constant level. The volume of liquid added to the inner ring to maintain a constant level, is the measure of the volume of liquid that infiltrates the soil. The volume that infiltrates the soil during timed intervals is converted to an incremental infiltration velocity, expressed in cm/h and plotted versus elapsed time (measured every 15 min, for one hour). The average incremental infiltration velocity of the test is equivalent to the infiltration rate. It is important to notice that the soil of the matorral thicket is sandier than the other soils, and the infiltration rate is therefore expected to be higher than in the other ecological areas.

To characterize *soil properties*, five random soil cores were taken (0.20 m) at each study site. Soil organic carbon was measured to determine the soil organic matter, by the traditional wet digestion method (Allison, 1960). Soil organic nitrogen was measured by the use of a LECO auto-analyser (LECO corporation, 2016).

 **Vegetation measurements**

The *characteristics of the vegetation* were estimated in three randomly selected linear transects of 20 m at each study site using the point method (Levy and Madden, 1933). A frame with a row of 10 steel pins spaced at 2 cm intervals was used. At each study site, each transect was divided into 20 cm intervals and vegetation cover was noted at each point on the frame at each 20 cm interval. All vegetation present was reported as: Dwarf shrubs, shrubs, forbs, grasses and graminoids, litter, and bare soil. In this way, 100 points were recorded for each transect.

The *annual above ground net primary production* (ANPP) of grasses and graminoids (expressed as g C m-1yr-1) was estimated after maximum plant growth which occurs in December-January. This was done by clipping all vegetation within a 0.2 m2 square plot in three permanent enclosures (1.5 x 1.2 m) that were randomly distributed in each site. The clipped vegetation was stored in airtight boxes to avoid respiration losses. The samples were dried in an oven at 600C for at least 24 h, weighed and biomass produced per ha calculated.

The level of *soil erosion* was determined by the Grassland Regeneration and Sustainability Standard 2.0 (GRASS) protocol scoring system (Borrelli et al., 2013).

*Soil respiration* (from roots and microorganisms) was measured at each site in the spring (November). CO2 resulting from soil respiration was measured using the soda lime method (Edwards, 1982). This method has previously been used successfully in Southern Patagonian ecosystems (Peri et al., 2015). Five random sampling stations were chosen each year at each study site. At each station, white opaque plastic respiration chambers (10 cm height x 21 cm diameter) with sealed lids, installed at a depth of 1 cm were inverted over open jars (diameter of 7 cm) containing 50 g of previously dried (1050C, 24 h) and weighed soda lime granules. The area inside the chamber was cleaned of all organic living matter. After 24 h, the soda lime jars were capped, dried at 1050 C for 24 h, and reweighed. Two blank chambers per site also with sealed lids, were installed to account for the gains in CO2 that occurred during oven drying. Soil respiration (gCO2h-1m-2) was calculated by correcting the measured CO2 for losses due absorption by soda lime upon drying (1.69) (Keith and Wong, 2006).

 **Species specific vegetation sampling and biomass determination**
 At each site, 6 plants of the grass species *Poa spiciformis* were harvested during the growth period (November-December) with a 10 cm diameter auger. The grass was cored at the centre to a depth of 20 cm. The diameter at the canopy and crown height (cm) of each individual grass sample was measured before harvesting, on site. Upon arrival at the lab, the percentage of the cored plant that was dead was measured by visual comparison on a 0.5 by 0.5 cm scaled transparent graph. Then, after removing soil and insects, the grass samples were separated into green leaves, dry leaves, and apical meristems, and maximum root length measured. Roots and apical meristems were dried and weighed. The leaf area of the grasses was determined (cm2) by scanning the total harvest of green leaves per plant, using a flat plate scanner.

 **Data analysis**
 Data was analysed using the software R (R Development Core Team, 2012) with the nlme package for linear models (Pinheiro et al, 2016). Vegetation profiles were plotted using Excel. All analyses were performed under the assumption that the data follows a multivariate normal or elliptical distribution. Summary statistics give an overview of the dataset by ecological area. A hierarchical cluster analysis was undertaken to investigate the correlations between the measured factors (Wickham and Francois, 2015).
Vegetation composition profiles were made to investigate botanical composition sensitivity to the measured grazing intensity, within each ecological area, assuming that the differences are due to grazing. Additionally, vegetation profiles were used to compare rotational and continuous grazing management at moderate grazing intensity in humid grass steppe and Andean vegetation. A linear regression analysis was made to determine if the measured factors were influenced by the ecological areas and the grazing intensities. The natural log of the plant percentages was used to remove the implications of the lower limit boundary of the data. The following linear regression was used:

y(*ijk*) = μ+αi + βj + αβij+ εijk [1]

where yijk denotes the ijkth observation, α(*i,*=1-4) ecological area, β(j=1-3) grazing intensity, and αβij  the interaction between ecological area and grazing intensity. ε (*i,jk*) ∼ N (0, σ2) distributed residuals. Interactions were found between area and grazing intensity for almost all measured factors, and therefore the following model was used for each ecological area:

y*ij* = μ+αi + εij [2]

where yij denotes the ijth observation in a given ecological area, α(*i,*=1-3) the grazing intensity, and εij the residual error with εij ∼ N (0, σ2). When the analyses were statistically significant, the post-hoc Tukey's multiple-comparison procedure test in R, using the multcomp package, was used for separation of the means (Hothorn et al. 2008).

A two-sample T-test was used to determine if significant differences existed between the means of the measured factors for rotational and continuous grazing management in Andean vegetation and humid grass steppe, where the following hypotheses were made:
 H0:μ1 = μ2 HA:μ1 ≠ μ2

**Results**

## **Exploratory data analysis**

The summary statistics revealed that the Andean vegetation had the highest grazing intensity (mean: 0.48 ewe/ha/yr) and lowest measured erosion (mean: 1.6%). Matorral thicket had the lowest grazing intensity (mean: 0.28 ewe/ha/yr), yet the highest erosion (mean: 12%). Andean vegetation had the greatest soil water retention capacity (mean: 3.05 mm/cm), soil respiration (mean: 1 gC/m2/h), aerial plant biomass (mean: 8.2 g), root mass (mean: 4.3 g), root length (mean: 13.4 cm), and plant length (mean: 7 cm). In contrast, the matorral thicket had the least soil respiration (mean: 0.4 gC/m2/h), and dry grass steppe, the least amount of aerial plant (mean: 5.4 g), root mass (mean: 2.2 g), root length (mean: 10.9 cm), and plant length (mean: 2.3 cm). Matorral thicket had the largest proportion of dead plants (mean: 23 %) and Andean vegetation the least (mean: 2.6 % ).

TABLE 1 here

The dendrogram (Figure 2) shows four correlation groups of the measured factors: 1) ANPP, soil water retention capacity, soil N, plant length, soil organic matter and soil respiration. 2) % cover by forbs, root/aerial plant ratio, % cover by grasses and graminoids, number of plant species, root biomass, and root length. 3) % cover by shrubs, proportion of dead plants, % cover by dwarf-shrubs, % cover by bare soil, and soil erosion. 4) % Litter cover, soil water infiltration, aerial plant, and diameter at base. Group 1 and 2 are more closely related to each other than to group 3 and 4, and group 3 and 4 are more closely related to each other than to group 1 and 2.

FIGURE 2 here

**Vegetation composition**

The vegetation profile for each area and grazing intensity are shown in Figure 3 a-d. These profiles show the enormous differences in the systems.

A larger percentage of bare soil and dwarf-shrubs cover was found with increasing grazing intensity in all ecological areas (Figure 3a-d). The increase in the proportion of bare soil with increasing grazing intensity is relatively proportional to the decrease in grasses and graminoids, while the proportion of forbs, dwarf shrubs and shrubs and litter cover stayed relatively constant. This was not the case in Andean vegetation where the percentage of litter cover decreased from to 75 % to 15 % when low was compared to moderate grazing. Humid grass steppe was the only vegetation system that had an increase of litter cover with increased grazing intensity, from 2% to 9 % when low is compared to intensive grazing.

FIGURE 3 here.

**Linear regression analysis**

Significant grazing intensity differences were found for 16 and 17 factors respectively for dry grass steppe and humid grass steppe, and for 16 factors in matorral thicket and Andean vegetation (Table 2). Model [2] was therefore sufficient to describe the differences for 76% and 81% of the measured factors (16/21 and 17/21).  All factors, except dwarf-shrubs, showed significant differences according to grazing intensity in at least one ecological area, and soil organic matter, soil N, bare soil, grasses and graminoids, ANPP, soil erosion, aerial plant, root, root length, diameter at base, proportion of dead plant, and number of plant species showed significant difference within all four ecological areas by grazing intensity.

However, the differences were not consistent. Intensive grazing was significantly different from moderate grazing for 13 factors in dry grass steppe, 11 factors in humid grass steppe, 15 factors in matorral thicket, but only 10 factors in Andean vegetation. Dry grass steppe, humid grass steppe, and Matorral thicket showed significant difference between moderate and intensive grazing in the following factors: organic matter, soil respiration, bare soil, ANPP, soil erosion, aerial plant, root biomass, root depth, proportion of dead plant, and no. of plant species. Only percentage bare soil, shrubs, soil erosion, aerial plant mass, root length, grams root to grams aerial plant ratio, diameter at base, plant length, proportion of dead plant, and number of plant species were significantly different between moderate and intensive grazing in the Andean ecosystem.

TABLE 2 here

## **Rotational grazing management compared to continuous grazing**

Less grasses and graminoids were detected in the continuous compared to the rotational grazing management system (Figure 4), respectively 81 versus 71 % in humid grass steppe, and 78 versus 52 % in Andean vegetation. More forbs were found in the rotational grazing sites at both ecological areas and more bare soil in the continuous grazing sites.

FIGURE 4 here

**Differences between grazing management**

Significant differences according to grazing management for 10 of the 21 measured factors were seen in the humid grass steppe ecosystem (Table 3). Seven of these factors (soil water infiltration, soil N, forbs and shrubs cover, root/aerial plant ratio, proportion of dead plants, and number of plant species) were greater for rotational grazing and three for continuous (bare soil, grasses and graminoids, and ANPP).

Table 3 here

Eleven factors were significantly different by grazing management in the Andean vegetation (Table 3). The three factors that were found greater for rotational management included forbs cover (%), root biomass (g), and root/aerial plant ratio (g/g) and eight factors were greater in continuous grazing management (soil organic matter, soil N, soil respiration, proportion of bare soil, aerial plant biomass, root and plant length, and portion of dead plants).

**Discussion**

**Ecological areas**

The differences found in the four ecological areas in this project, regardless of grazing intensity, are demonstrated in Figure 2. Matorral thicket was found to be different from the other systems in both vegetation type, soil measurements, and soil degradation. This was due to more shrubs, less soil organic matter, more bare soil, and soil erosion. Herrick (2000) demonstrated that shrubs tend to produce larger amounts of standing dead foliage and dead root biomass than grasses. This leads to greater amounts of above- and below ground organic matter, which enhances the soil and water infiltration and improves soil fertility. Matorral thicket was found, in this study, to have less soil organic matter (Table 1) than the other ecological areas. However, the matorral thicket had the highest infiltration rate of the four ecological areas (Table 1). This is likely due to the sandy soils in this system which will influence the soil physical parameters and hydrological properties (Blackburn, 1975).

 The Andean vegetation system included native forest vegetation, which may be the reason for the greater level of soil organic matter, soil infiltration, litter cover, soil water retention capacity, aerial plant production, plant length, root biomass and length, ANPP and the lower degree of soil erosion. The greater amount of litter cover, soil organic matter and the lower level of bare soil can be explained by the plant and litter cover that enhances soil infiltration rates and decreases evaporation, which ensures soil moisture is retained after each precipitation event (Sacks et al., 2014). This increases soil microbial activity, which promotes soil stability, preserves plant nutrients and availability, increases plant-growing conditions, and leads to incorporation of more organic matter into the soil (Teague et al., 2011).

In this study matorral thicket with shrub vegetation had the lowest soil respiration rate followed by dry grass steppe, humid grass steppe, and Andean vegetation. Annual net plant productivity, soil water retention capacity, and soil respiration were found to be correlated (Figure 2). This is in line with Cao et al. (2004) linking soil respiration to the level of vegetation cover and ANPP, based on the influence of root respiration, where Buyanovsky and Wanger (1983), demonstrated a correlation of moisture content in the soil and soil respiration.

**Grazing intensities**

A general increase in the measured indicators suggest that increasing ecosystem health occurred with the increase from light to moderate grazing. This is in contrast to the change in the indicators suggesting a decline in ecosystem health that was generally seen when comparing moderate to intensive grazing. However, this was not consistent within each ecosystem.

Aagesen (2000) and Basher and Webb (1997) found in Pategonia and New Zealand respectively that grazing intensity that removes large amounts of grasses, leads to bare soil patches and plant death. In both ecosystems, bare soil patches were invaded by less preferred forage species such as dwarf-shrubs and shrubs. This was seen when sheep started feeding on the tussocks when no preferred species are available, which usually occurs in the winter period in Argentine Patagonia. When the preferred grasses are removed and grazing on the base of the tussock begins, the tussocks form pedestals due to wind erosion. This exposes the roots, causing plant death and increased soil erosion (Aagesen 2000; Basher and Webb 1997). Results of this study are in an agreement with Aagesen (2000) and Basher and Webb (1997), in that soil erosion, bare soil, dwarf-shrubs, shrubs, and proportion of dead plant were closely correlated (Figure 2).

In this study, it was found that the change from light to moderate grazing generally stimulated plant growth which in turn stimulated aerial plant, root biomass, and diameter of individual grass plants. The increased plant growth stimulated ANPP, soil organic matter, soil N, and in turn reduces soil respiration. These differences, seen after 4 years, indicate that the moderate grazing intensity has benefits to the ecosystem health.

The differences seen between moderate and intensive grazing in dry grass steppe, indicated an apparent reduction of overall ecosystem health with a reduction of litter cover, forbs, aerial plant, root biomass, root length, and diameter of individual grass plants. The reduced plant growth is probably the cause of the reduced soil organic matter, soil N, and ANPP. The increased grazing intensity apparently reduced the number of plant species and resulted in increased bare soil, soil erosion, and standing dead grass.

Changing from low to moderate grazing in the humid grass steppe apparently stimulated ecosystem health by increasing grasses and graminoids growth, aerial plant biomass, root biomass, and diameter of individual grass plants. This in turn, may be the cause of the increased soil N, ANPP, and decreased soil respiration. However, an increased proportion of dead grass plant was measured at the increased grazing intensity. A decline in ecosystem health was seen from moderate to intensive grazing with an increased proportion of bare soil, more soil erosion, and proportion of dead grass plants. The increased grazing intensity from light to moderate to intensive was not enough to encourage the sheep to remove the litter biomass, which increased over grazing intensity. The increase of bare soil from moderate to intensive grazing lead to additional indicators of reduced ecosystem health, with a reduction of soil organic matter, ANPP, aerial plant and, root biomass, root and plant length, and the number of plant species.

Low to moderate grazing in matorral thicket stimulated forbs and plant length, which stimulated ANPP. The increased grazing lead to a decrease of grasses and graminoids and an increased proportion of soil erosion, which lead to a reduction of soil organic matter, soil respiration, aerial plant, root biomass, and diameter of individual grass plants. Moderate to intensive grazing intensity lead to a decrease in grasses and graminoids, aerial plant, plant length, root biomass, root length, diameter of individual grass plants, number of plant species, and an increase of bare soil and soil erosion. The lower plant production and cover lead to reduced ANPP, soil organic matter, soil N, and soil respiration.

Low to moderate grazing in Andean vegetation stimulated root length and plant species diversity at the forest floor. However, this might be due to thinning in the silvopastoral system, which allows for more plant diversity through more light and photosynthesis at the forest floor. A negative effect was that the increased grazing intensity also lead to increased soil erosion. These changes resulted in reduced soil organic matter, soil N, litter cover, grasses and graminoids, shrubs, ANPP, aerial plant, and plant length. Moderate to intensive grazing lead to a reduction of aerial plant, root length, plant length, and number of plant species as well as an increase of bare soil, soil erosion, and standing dead grass.

This is in line with the results of Bertiller and Bisigato (1998), who found a reduction in number of plant species and changes in plant composition from grasses to shrubs and bare soil with increased grazing disturbance in Patagonia. The cover composition change found in this study are in line with the results of Aagesen (2000), who also documented an increase in shrubs and dwarf-shrubs cover and decrease in grasses with increased sheep grazing intensity in Patagonia. However, the decrease of grasses was not applicable to the Andean vegetation in this study, because the forest is selectively thinned when it is to be used for grazing to promote a higher forage production for silvopastoral use (Peri et al., 2016). The removal of trees increased the amount of grass cover from low to moderate grazing, but decreased from moderate to intensive.

This can be explained by light normally being the primary limiting factor for plant growth (Seastedt and Knapp, 1993), and the forest and sward canopy therefore limit the light penetration to the understory. Both thinning the forest and low intensity grazing can remove this light impedance and allow plant growth. This, in turn, allows above- and belowground biomass accumulation with water retention and nitrogen accumulation. When light is not a limiting factor, because the top part of the vegetation has been removed by grazing, nitrogen becomes the limiting factor instead (Blair, 1997). The highest levels of nitrogen are therefore most commonly measured in un-defoliated, or very lightly defoliated grasslands (Teague et al., 2011). This is also the case for this study where the highest nitrogen measurements are found in low grazing followed by moderate and intensive grazing.

 Intensive grazing in this study was associated with negative impacts in all factors (Table 2), that showed significant difference, compared to moderate grazing. The negative impacts that are seen with the use of intensive grazing in this study can, as in agreement with Teague et al. (2008), be attributed to a degree of overgrazing where the plants are exposed to multiple severe defoliation without sufficient time to recover between the events. This can then lead to a decline in plant productivity and root biomass, which is in line with the study of Briske et al. (2008). Thus, if livestock regularly removes large amounts of plant biomass and litter, a degradation spiral can be initiated, especially in the most used patches. The degradation spiral is characterised by replacement of taller perennial grasses by shorter grasses, annual grasses and forbs, and finally bare ground (Teague et al., 2004). This effect may be what is visible in Figure 2 and Table 2, where the proportion of grasses decreases but bare soil increases with grazing intensity as well as forbs.

Several of the measured factors in this study are related to soil function, which is important since maintaining a normal soil function in rangeland ecosystems is critical for the overall health of the ecosystem (Barrett, 2001). Barrett (2001) demonstrates that it is only possible to maintain a normal soil function if the soil has an adequate plant and litter cover to provide protection from soil loss, and thereby allows soil microorganisms to perform optimally. Soil respiration is therefore, not an absolute indicator of ecosystem health, as a decrease can be considered a health indicator under growth conditions, but may also be an indication of poor health during conditions of drought and loss of biomass.

A correlation between the factors bare soil, soil erosion, and proportion of dead plant, and the factors litter cover, aerial plant, and soil water infiltration was found in this study (Fig 2). It was demonstrated that these two groups of factors are closer correlated than to the rest of the factors. This is because they depend on each other for soil function. Asner et al. (2003) found that bare soil can be seen as an indicator for soil function and for the risk of erosion. The risk of erosion increases if the soil cover is insufficient to disperse raindrops before they reach the soil (Schlesinger et al., 2000). The increased soil temperature and soil loss leads to negative effects on infiltration rates, soil evaporation, nutrient retention, and therefore the general biological functions that contribute to ecosystem function (Peri et al., 2015).

A decrease of infiltration rate and soil respiration with increased grazing intensity was found in this study (Table 2). One reason for this may be that the soil function can be inhibited by excessive trampling during heavy livestock use of an area (Asner et al., 2003). This can lead to soil degradation by increased soil compaction, which can elevate penetration resistance (Herrick, 2000). It is difficult to tell from our study if this is the reason for the decreased infiltration rate, but the decreased respiration is in line with the results found by Peri (2015) and Cao et al. (2004), who found soil respiration to decrease with increased grazing intensity in Patagonia.

This study demonstrates that grazing intensity has an influence on the plant composition in all the investigated ecological areas and that intensive grazing is associated with negative impacts in all measured factors when compared to a moderate grazing intensity.

**Grazing management strategies**

The comparison of rotational and continuous grazing (Table 3) showed significant differences for 10 factors in humid grass steppe, and 11 factors in Andean vegetation. The rotational grazing resulted in increased negative ecosystem health indicators for humid grass steppe (increase in proportion of shrubs and dead plants) but none for Andean vegetation and increased positive indicators in humid grass steppe (increased proportion of forbs, soil N, species diversity, soil water infiltration rate, and root/aerial plant rate) and Andean vegetation (increase in proportion of forbs, increased root biomass, and root/aerial plant ratio), compared to continuous grazing.

Continuous grazing also led to, more negative health indicators for humid grass stepper (increased proportion of bare soil) and Andean vegetation (increased proportion of bare soil, and proportion of dead plant) as well as increased positive indicators for them both (humid grass steppe: ANPP, increased proportion of grasses and graminoids. Andean vegetation: increased soil organic matter soil N, plant length, root length, aerial plant, and soil respiration) compared to rotational grazing.

The results for soil organic matter, soil water infiltration rate, and soil water retention capacity are in contrast to the study of Weber (2011), who found rotational grazing enhanced soil organic matter and soil-water content. This is supported by the study of Teague et al. (2011), who found rotational grazing in semi-arid rangeland to decrease impact on soil physical properties and infiltration rates compared to continuous grazing at the same stocking rate. On the other hand, Carter et al. (2014) found no differences between rotational grazing and continuous grazing in terms of soil organic matter, soil water infiltration in soil, or soil erosion. There is, therefore, not enough concurrence in the literature to determine if the findings in this study are in line with the literature.

 The lower levels of bare soil and proportion of dead plant found in this study with the use of rotational grazing are consistent with the study of Teague (2011) and Teague et al. (2010), who found rotational grazing to maintain plant cover, decrease bare soil paths and soil erosion, provide lower soil temperatures, and increase soil carbon compared to continuous grazing at the same stocking rate. The lower proportion of dead plant in this study, however, only applies to Andean vegetation since continuous grazing had a lower proportion of dead plant in humid grass steppe. The level of soil erosion and litter cover was not significantly different with the two management strategies in either humid grass steppe nor Andean vegetation.

This study found that rotational grazing influences several factors positively in humid grass steppe where the effect in Andean vegetation is limited to less bare soil and a lower proportion of dead plant (Table 3). In the comparison of continuous grazing and rotational grazing (Table 3) it was found that lands managed with rotational grazing had a plant composition with less grasses and graminoids and bare soil, but more forbs and litter cover. Teague et al. (2011) found rangelands managed with rotational grazing to have a higher proportion of desirable grasses and a lower proportion of less desirable grasses and forbs than lightly stocked continuous grazing. In this study, the percentage of grasses has not been differentiated into desired and less desired grasses, and both humid grass steppe and Andean vegetation were found to have a higher proportion of forbs when rotational grazing is compared to continuous grazing. The results of this analysis are therefore in contrast to the results of Teague et al. (2011).

Carbon content in soils can be seen as an indicator for soil health, plant production, water catchments, and even more importantly, as a sink for atmospheric carbon to offset climate changes (Janzen, 2004). The management and use of rangelands is therefore crucial for the land’s ability to sequester and retain organic carbon. Management that increases plant productivity increases carbon inputs to the soil, and decreases soil exposure to erosion and sunlight, allows higher levels of carbon accumulation in the soil (Parton et al., 1987). This analysis cannot clearly determine if there are higher levels of carbon in lands managed with rotational grazing compared to continuous grazing. However, the soil organic matter content showed a difference for Andean vegetation where continuous grazing had the highest content, but no difference was found in humid grass steppe, and the results are therefore inconclusive.

The decreased level of bare soil with the use of rotational grazing in both humid grass steppe and Andean vegetation may indicate a positive influence on carbon sequestration and retention of organic carbon. Jones and Donnelly (2004) found that soil carbon availability is regulated by plant production and the amount of plant litter cover to provide physical protection of the soil. This analysis did not find a significant difference for litter cover but only a tendency for differences in the vegetation profiles (Figure 3 a-d), but the decreased level of bare soil can be a reason to believe that the rotational grazed lands may be able to sequester more carbon.

In this analysis, it was found that rotational grazing in Andean vegetation resulted in increased root biomass compared to continuous grazing but in the humid grass steppe no significant difference was found. Sacks et al. (2014) found that increased root biomass growth causes stronger and more drought resistant plants. Wang and Fang (2009) found respiration produced primarily by roots and soil organism to be the primary pathway for CO2 fixed by plants to return to the atmosphere (Wang and Fang, 2009). Increased root biomass can therefore help to a greater carbon fixation. This may indicate that rotational grazing in Andean vegetation, which has significantly more root biomass than continuous grazing, is able to fixate more carbon.

**Conclusion**

The following conclusions were made 1) Success of a static rotational or continuous management system to sustain and improve soil health is dependent on the ecosystem. 2) Intensive grazing influences the measured parameters negatively for soil water, soil, and vegetation compared to moderate and low grazing. 3) Rotational grazing management influences humid Magellanic grass steppe and Silvopastoral Andean vegetation differently.

**Implications**

Successful grazing management in Argentine Patagonia is a complex mixture of grazing intensity and grazing system that must be suited to the specific ecosystem. Multiple indicators of ecosystem health, as defined in this study should be monitored in order to evaluate a management strategy. The long-term goals of the local people, food needs, and environmental concerns must be balanced in the short-term management plans.

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Aagesen, D. 2000. Crisis and conservation at the end of the world: Sheep ranching in Argentine Patagonia. Environ. Conserv. 27:208-215.

Allison, L.E. 1960. Wet-combustion apparatus and procedure for organic and inorganic carbon soil. Soil Sci Soc Am J. 24:36-40.

Asner, GP., Borghi, C.E. and Ojeda, R.A. 2003. Desertification in Central Argentina: Changes in ecosystem carbon and nitrogen from imaging spectroscopy. Ecol. Appl. 13:629-648.

ASTM International D3385-09, 2009. Standard test method for infiltration rate of soils in field using double-ring infiltrometer. DOI: 10.1520/D3385-09.

Barrett, J.R. 2001. Livestock farming: Eating up the environment? Environ. Health Perspect. 109:312-317.

Basher, L. and Webb, T. 1997. Wind erosion rates on terraces in the Mackenzie basin. J. R. Soc. N. Z. 27:499-512.

Bertiller, M.B. and Bisigato, A. 1998. Vegetation dynamics under grazing disturbance. The state-and-transition model for the Patagonian steppes. Ecología Austral. 8:191-199.

Blackburn, W. 1975. Factors influencing infiltration and sediment production of semiarid rangelands in Nevada. Water Resour. Res. 11:929-937.

Blair, J.M. 1997. Fire, N availability, and plant response in grasslands: A test of the transient maxima hypothesis. Ecology. 78:2359-2368.

Borrelli, P., Boggio, P., Sturzenbaum, P., Paramidani, M., Heiken, R., Pague, C., Stevens, M. and Nogués, A. 2013. Grassland regeneration and sustainability standard (GRASS). 2.0: The nature conservancy and Ovis 21, Argentina.

Briske, D.D., Derner, J.D., Brown, J.R., Fuhlendorf, S.D., Teague, W.R., Havstad, KM., Gillen, R.L., Ash, A.J. and Williams W.D. 2008. Rotational grazing on rangelands: Reconciliation of perception and experimental evidence. Rangeland Ecol Manag. 61:3-17.

Buyanovsky, G. and Wagner, G. 1983. Annual cycles of carbon dioxide level in soil air. Soil Sci. Soc. Am. J. 47:1139-1145.

Cao, G., Tang, Y., Mo, W., Wang, Y., Li, Y. and Zhao, X. 2004. Grazing intensity alters soil respiration in an alpine meadow on the Tibetan plateau. Soil Biol Biochem. 36:237-243.

Carter, J., Jones, A., O'Brien, M., Ratner, J. and Wuerthner. G. 2014. Holistic management: Misinformation on the science of grazed ecosystems. Inter J Biodiv. 2014:1-10.

Edwards, N. T. 1982. The use of soda-lime for measuring respiration rates in terrestrial systems. Pedobiologia. 312-330.

Food and Agriculture Organization of the United Nations 2005. Grasslands of the world. 34.

Gallardo, A. and Schlesinger, W.H. 1992. Carbon and nitrogen limitations of soil microbial biomass in desert ecosystems. Biogeochemistry. 18:1-17.

Herrick, J E. (2000). Soil quality: An indicator of sustainable land management? Appl Soil Ecol. 15:75-83.

Hothorn, T., Bretz, F. and Westfall, P. 2008. Simultaneous Inference in General Parametric Models. Biometrical J. 50: 346-363.

Janzen, H.H. 2004. Carbon cycling in earth systems - a soil science perspective. Agric Ecosyst Environ. 104:399-417.

Jones, M.B. and Donnelly, A. 2004. Carbon sequestration in temperate grassland ecosystems and the influence of management, climate and elevated CO2. New Phytol. 164:423-439.

Keith, H. and Wong, S C. 2006. Measurement of soil CO2 efflux using soda lime absorption: Both quantitative and reliable. Soil Biol Biochem. 38:1121-1131.

LECO Corporation, 2016. TruMac carbon/nitrogen/protein/sulfur in macro organic samples. Assessed 22.02.2015. Available online. < https://www.leco.com/products/analytical-sciences/nitrogen-protein-analyzer/928-series-cns>.

Levy, E.B. and Madden, E.A. 1933. The point method of pasture analysis. New Zeal J Agr. 20:267-279.

Ormaechea, S. and Peri, P. 2015. Landscape heterogeneity influences on sheep habits under extensive grazing management in Southern Patagonia. Livestock Res Rural Dev. 27:1-11.

Parton, W.J., Schimel, D.S., Cole, C. and Ojima, D. 1987. Analysis of factors controlling soil organic matter levels in great plains grasslands. Soil Sci. Soc. Am. J. 51:1173-1179.

Paruelo, J.M., Jobbágy, E.G. and Sala, O.E. 1998. Biozones of Patagonia (Argentina). Ecología Austral. 8:145-153.

Peri, P.L. and Bloomberg, M. 2002. Windbreaks in Southern Patagonia, Argentina: A review of research on growth models, windspeed reduction, and effects oncrops. Agrofor. Syst. 56:129-144.

Peri, P.L. 2011. Carbon storage in cold temperate ecosystems in Southern Patagonia, Argentina, in Atazadeh, E., (Ed.), Biomass and Remote Sensing of Biomass. InTech Publisher, Croacia, pp. 213-226. DOI: 10.5772/20294.

Peri, P.L., Bahamonde, H. and Christiansen, R. 2015. Soil respiration in Patagonian semiarid grasslands under contrasting environmental and use conditions. Journal of Arid Environments; 2. 119:1-8.

Peri, P.L., Bahamonde, H., Lencinas, M.V., Gargaglione, V., Soler, R., Ormaechea, S. and Pastur, G.M. 2016. A review of Silvopastoral systems in native forest of *Nothofagus antarctica* in Southern Patagonia, Argentina. Agrofoestry Systems. 90: 933-960.

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D. and R Core Team 2016. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-131. URL [https://CRAN.R-project.org/package=nlme](https://cran.r-project.org/package%3Dnlme).

R Development Core Team 2012. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL [http://www.R-project.org.](http://www.r-project.org./)

Richards, L. 1948. Porous plate apparatus for measuring moisture retention and transmission by soil. Soil Sci. 66:105-110.

Sacks, A. D., Teague, R., Provenza, F., Itzkan, S. and Laurie, J. 2014. Restoring atmospheric carbon dioxide to pre-industrial levels: Re-establishing the evolutionary grassland-grazer relationship. Geotheropy. 27: 1-92.

Schlesinger, W.H., Ward, T.J. and Anderson, J. 2000. Nutrient losses in runoff from grassland and shrubland habitats in Southern New Mexico: II. field plots. Biogeochemistry. 49:69-86.

Scutari, N.C., Bertiller M.B. and Carrera, A.L. 2004. Soil-associated lichens in rangelands of north-eastern Patagonia. Lichen groups and species with potential as bioindicators of grazing disturbance. The Lichenologist. 36:405-412.

Seastedt, T.R. and Knapp, A.K. 1993. Consequences of non-equilibrium resource availability across multiple time scales: The transient maxima hypothesis. Am. Nat. 141:621-633.

Soliveres, S. and Eldridge, D.J. 2014. Do changes in grazing pressure and the degree of shrub encroachment alter the effects of individual shrubs on understorey plant communities and soil function? Funct. Ecol. 28:530-537.

Soriano, A., Sala, O.E. and Leon, R.J.C. 1980. Vegetacion actual y vegetacion potencial en el pastizal de coiron amargo (stipa spp.) del sw. de chubut. Boletin De La Socialidad Argentina De Botanica. 21:309-314.

Soriano, A. 1983. Deserts and semi-deserts of Patagonia, in West, N. E., (Ed.) Ecosystems of the World - Temperate Deserts and Semi-Deserts, Elsevier Scientific, Amsterdam, The Netherlands, pp. 423-460.

Teague, W.R., Dowhower, S.L. and Waggoner, J.A. 2004. Drought and grazing patch dynamics under different grazing management. J. Arid Environ. 58:97-117.

Teague, W.R., Provenza, F., Norton, B., Steffens, T., Barnes, M., Kothmann M. and Roath, R. 2008. Benefits of multi-paddock grazing management on rangelands: Limitations of experimental grazing research and knowledge gaps. Grasslands: Ecology, Management and Restoration. Hauppauge, NY, USA: Nova Science Publishers. 41-80.

Teague, W.R., Dowhower, S.L., Baker, S.A., Ansley, R.J., Kreuter, U.P., Conover, D.M. and Waggoner, J.A. 2010. Soil and herbaceous plant responses to summer patch burns under continuous and rotational grazing. Agric., Ecosyst. Environ. 137:113-123.

Teague, W.R., Dowhower, S.L., Baker, S.A., Haile, N., DeLaune, P.B. and Conover, D.M. 2011. Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie. Agric., Ecosyst. Environ. 141:310-322.

Valle, H.F., Elissalde, N.O., Gagliardini, D.A., and Milovich, J. 1998. Status of desertification in the Patagonian region: Assessment and mapping from satellite imagery. Arid Soil Res. Rehabil. 12:95-122.

Wang, W. and Fang, J. 2009. Soil respiration and human effects on global grasslands. (special issue: Changes in land use and water use and their consequences on climate, including biogeochemical cycles.). Global Planet. Change. 67:20-28.

Weber, K.T. and Gokhale, B.S. 2011. Effect of grazing on soil-water content in semiarid rangelands of Southeast Idaho. J. Arid Environ. 75:464-470.

Wickham, H. and Francois, R. 2015. dplyr: A Grammar of Data Manipulation. R package version 0.4.3. [https://CRAN.R-project.org/package=dplyr](https://cran.r-project.org/package%3Ddplyr)

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Table 1: Summary statistics of vegetation and soil characteristics of selected ecosystems in the Argentine Patagonia

|  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| Dry Grass Steppe | Grazing intensity | Soil water infiltration | Soil water retention capacity | Soil organic matter | Soil N | Soil respiration | Bare soil | Litter cover | Grasses and graminoids | Forbs | Shrubs |
| Min. | 0.18 | 1.55 | 1.25 | 2.9 | 0.1 | 0.4 | 4.4 | 3.2 | 52 | 2.5 | 0.6 |
| Mean | 0.32 | 1.68 | 1.59 | 3.5 | 0.2 | 0.6 | 12.8 | 10.8 | 65 | 5.8 | 1.1 |
| Max | 0.51 | 1.81 | 1.95 | 4.4 | 0.3 | 0.7 | 24.7 | 20.1 | 72 | 9.8 | 1.8 |
| SD  | 0.15 | 0.08 | 0.23 | 0.42 | 0.04 | 0.09 | 7.64 | 6.07 | 6.53 | 2.26 | 0.41 |
|   | Dwarf shrubs | ANPP | Soil erosion | Aerial plant | Root | Root/ | Root depth | Diameter at base | Plant length | Proportion of dead plant | No. plant species |
| aerial plant ratio |
| Min. | 1.1 | 5.1 | 0 | 4.2 | 1.2 | 0.3 | 6.8 | 4.5 | 1.8 | 1.9 | 15 |
| Mean | 4.7 | 8.8 | 6 | 5.4 | 2.2 | 0.4 | 10.9 | 5.4 | 2.3 | 8.5 | 23.2 |
| Max | 15.9 | 16.3 | 16.2 | 7 | 3.2 | 0.5 | 14.5 | 7.1 | 2.5 | 19 | 30 |
| SD  | 4.66 | 4.58 | 6.90 | 0.84 | 0.69 | 0.08 | 2.74 | 0.75 | 0.19 | 6.85 | 5.38 |
| Humid Grass Steppe | Grazing intensity | Soil water infiltration | Soil water retention capacity | Soil organic matter | Soil N | Soil respiration | Bare soil | Litter cover | Grasses and graminoids | Forbs | Shrubs |
| Min. | 0.10 | 0.74 | 1.73 | 3.5 | 0.5 | 0.5 | 0.1 | 1.9 | 66 | 4.4 | 0.1 |
| Mean | 0.42 | 1.40 | 1.99 | 4.7 | 0.2 | 0.7 | 3.4 | 5.5 | 77 | 8.7 | 1.8 |
| Max | 0.78 | 2.10 | 2.31 | 5.7 | 0.3 | 0.9 | 12.5 | 9.5 | 91 | 19.2 | 6.2 |
| SD  | 0.25 | 0.54 | 0.17 | 0.62 | 0.03 | 0.13 | 4.63 | 2.76 | 8.53 | 5.41 | 2.11 |
|   | Dwarf shrubs | ANPP | Soil erosion | Aerial plant | Root | Root/aerial plant ratio | Root depth | Diameter at base | Plant length | Proportion of dead plant | No. plant species |
| Min. | 0.1 | 7.1 | 0 | 3.9 | 2.2 | 0.5 | 9.9 | 3.1 | 1.2 | 1.6 | 18 |
| Mean | 3.2 | 13.3 | 2.5 | 6 | 3.8 | 0.65 | 14 | 4.9 | 4.4 | 5.9 | 33 |
| Max | 7.9 | 21.1 | 8.8 | 10.1 | 6.0 | 0.82 | 16.6 | 7.6 | 7.9 | 16.9 | 42 |
| SD  | 2.49 | 5.58 | 3.36 | 1.98 | 1.18 | 0.09 | 1.73 | 1.40 | 2.15 | 5.35 | 8.21 |
| Matorral Thicket | Grazing intensity | Soil water infiltration | Soil water retention capacity | Soil organic matter | Soil N | Soil respiration | Bare soil | Litter cover | Grasses and graminoids | Forbs | Shrubs |
| Min. | 0.15 | 2.36 | 0.98 | 1.89 | 0.1 | 0.3 | 16 | 3.8 | 18 | 0.5 | 17.7 |
| Mean | 0.28 | 2.69 | 1.33 | 2.5 | 0.1 | 0.4 | 27 | 6.6 | 33 | 4.3 | 22.6 |
| Max | 0.49 | 3.08 | 1.64 | 3.1 | 0.2 | 0.57 | 36 | 10.2 | 45 | 7.8 | 28.1 |
| SD  | 0.16 | 0.22 | 0.20 | 0.35 | 0.03 | 0.09 | 7.24 | 2.10 | 9.78 | 2.75 | 3.86 |
|   | Dwarf shrubs | ANPP | Soil erosion | Aerial plant | Root | Root/aerial plant ratio | Root depth | Diameter at base | Plant length | Proportion of dead plant | No. plant species |
| Min. | 0.3 | 2.2 | 0 | 3.5 | 1.1 | 0.3 | 9.3 | 4.1 | 2.1 | 12 | 13 |
| Mean | 7.2 | 4.7 | 12 | 6.8 | 2.6 | 0.4 | 12.6 | 6.1 | 3.1 | 23 | 18.4 |
| Max | 15.3 | 8.4 | 33 | 10.9 | 4.5 | 0.5 | 16.7 | 8.9 | 3.9 | 39 | 23 |
| SD  | 4.99 | 2.27 | 13.91 | 2.49 | 1.28 | 0.06 | 2.08 | 1.49 | 0.60 | 9.15 | 3.81 |
| Andean Vegetation | Grazing intensity | Soil water infiltration | Soil water retention capacity | Soil organic matter | Soil N | Soil respiration | Bare soil | Litter cover | Grasses and graminoids | Forbs | Shrubs |
| Min. | 0.0 | 1.98 | 2.61 | 3.1 | 0.2 | 0.5 | 0.3 | 11 | 11 | 0.8 | 0.0 |
| Mean | 0.48 | 2.26 | 3.05 | 5.2 | 0.5 | 1.0 | 4.3 | 30 | 55 | 10.9 | 0.1 |
| Max | 0.85 | 2.83 | 3.80 | 7.2 | 0.8 | 1.6 | 10.6 | 77 | 79 | 29.9 | 0.6 |
| SD  | 0.31 | 0.255 | 0.36 | 1.24 | 0.19 | 0.35 | 3.29 | 26.96 | 23.43 | 10.05 | 0.21 |
|   | Dwarf shrubs | ANPP | Soil erosion | Aerial plant | Root | Root/aerial plant ratio | Root depth | Diameter at base | Plant length | Proportion of dead plant | No. plant species |
| Min. | 0.5 | 7 | 0.0 | 5.9 | 2.5 | 0.34 | 10.1 | 4.4 | 3.4 | 0.8 | 13 |
| Mean | 3.3 | 17 | 1.6 | 8.2 | 4.3 | 0.6 | 13.4 | 6.0 | 7.0 | 2.6 | 10.4 |
| Max | 8.6 | 5.4 | 5.4 | 11.7 | 7.3 | 1.0 | 17.7 | 8.4 | 10.1 | 7.3 | 28.0 |
| SD  | 2.51 | 9.71 | 2.07 | 1.84 | 1.50 | 0.25 | 2.22 | 1.18 | 2.05 | 2.52 | 5.48 |

Table 2: Linear model significance for effect of grazing intensity on measured factors within ecosystem. L = Low grazing intensity. M= Moderate grazing intensity. I= Intensive grazing intensity.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
|  | **Dry grass Steppe** | **Humid grass steppe** | **Matorral thicket** | **Andean vegetation** |
|  | **L** | **M** | **I** | **P** | **L** | **M** | **I** | **P** | **L** | **M** | **I** | **P** | **L** | **M** | **I** | **P** |
| **Soil water infiltration (cm/h)** | 1.72 | 1.70 | 1.62 | 0.357 | 1.86a | 1.07ab  | 0.78 b | 0.022 | 2.79 | 2.71 | 2.57 | 0.514 | 2.50 | 2.14 | 2.26 | 0.080 |
| **Soil water retention capacity (mm/cm)** | 1.70 | 1.65 | 1.40 | 0.247 | 2.09a | 1.97ab | 1.79b | 0.043 | 1.46 | 1.40 | 1.14 | 0.114 | 3.37 | 2.96 | 2.82 | 0.170 |
| **Soil organic matter (%)** | 3.44b | 3.96a | 3.20b | \*0.003 | 5.20a | 4.80a | 3.85b | \*<0.001 | 2.80a | 2.50b | 2.10c | \*<0.001 | 6.54a | 5.80b | 4.80b | 0.007 |
| **Soil N (%)** | 0.17b | 0.24a | 0.14c | \*<0.001 | 0.21b | 0.25a | 0.19ac | <0.001 | 0.16a | 0.15a | 0.11b | \*<0.001 | 0.75a | 0.54b | 0.44b | <0.001 |
| **Soil respiration (g C/m2/h)** | 0.65a | 0.55b | 0.44c | \*<0.001 | 0.85a | 0.62b | 0.53c | \*<0.001 | 0.52a | 0.44b | 0.33c | \*<0.001 | 1.18 | 1.41 | 0.71 | 0.102 |
| **Bare soil (%)** | 5.90b | 10.17b | 22.47a | \*<0.001 | 0.39b | 1.87b | 10.90a | \*<0.001 | 19.27b | 25.90b | 34.50a | \*0.004 | 2.87b | 4.77b | 8.33a | \*0.012 |
| **Litter cover (%)** | 17.90a | 9.90b | 4.50c | \*<0.001 | 2.47b | 7.33ab | 8.17a | 0.019 | 8.40 | 6.17 | 5.23 | 0.162 | 74.17a | 15.60b | 12.47b | <0.001 |
| **Grasses and graminoids (%)** | 68.80a | 66.90ab | 57.50b | 0.020 | 88.20a | 81.33b | 69.03b | 0.004 | 42.47a | 35.00b | 20.90c | \*<0.001 | 12.13b | 76.13a | 66.40a | <0.001 |
| **Forbs (%)** | 3.27b | 8.03a | 6.13a | 0.004 | 5.48 | 6.00 | 5.82 | 0.150 | 0.97b | 6.03a | 5.80a | 0.006 | 8.27 | 1.47 | 7.03 | 0.580 |
| **Shrubs (%)** | 0.83 | 1.40 | 1.13 | 0.274 | 0.30 | 0.67 | 1.30 | 0.217 | 25.20 | 19.50 | 23.03 | 0.199 | 0.03b | 0.03b | 0.47a | \*<0.001 |
| **Dwarf-shrubs (%)** | 2.30 | 3.60 | 8.27 | 0.288 | 3.16 | 2.80 | 4.78 | 0.452 | 3.70 | 7.40 | 10.03 | 0.272 | 2.53 | 2.00 | 5.30 | 0.299 |
| **ANPP (gC/m2)** | 6.10b | 14.80a | 5.50b | \*<0.001 | 8.40b | 19.50a | 7.90b | \*<0.001 | 4.10b | 7.50a | 2.50c | \*<0.001 | 7.80a | 32.30b | 12.80b | 0.015 |
| **Soil erosion (%)** | 0.00c | 3.00b | 15.00a | \*<0.001 | 0.00c | 1.00b | 8.00a | \*<0.001 | 0.00c | 5.07b | 29.80a | \*<0.001 | 0.00c | 0.50b | 5.00a | \*<0.001 |
| **Aerial plant (g)** | 4.75b | 6.38a | 5.10b | \*<0.001 | 9.13a | 5.17b | 4.27c | \*<0.001 | 9.76a | 6.61b | 4.08c | \*<0.001 | 10.90c | 8.37b | 6.48a | \*<0.001 |
| **Root (g)** | 1.33c | 2.92a | 2.22b | \*<0.001 | 5.42a | 3.15b | 2.56c | \*<0.001 | 4.18a | 2.35b | 1.23c | \*<0.001 | 3.80b | 4.08a | 2.74a | <0.001 |
| **Root/aerial plant ratio** | 0.29a | 0.46b | 0.43b | <0.001 | 0.59 | 0.61 | 0.60 | 0.061 | 0.43a | 0.36b | 0.30c | \*<0.001 | 0.35b | 0.49a | 0.42b | \*0.002 |
| **Root depth (cm)** | 13.21a | 12.02a | 7.42b | \*<0.001 | 15.42a | 14.10a | 11.70b | \*<0.001 | 14.50a | 13.08a | 10.30b | \*<0.001 | 11.20b | 15.40a | 12.10b | \*<0.001 |
| **Diameter at base (cm)** | 4.96b | 6.18a | 4.92bc | \*0.002 | 7.06a | 3.46b | 4.40b | <0.001 | 7.75a | 6.08b | 4.55c | \*<0.001 | 7.65a | 5.86b | 4.75c | \*<0.001 |
| **Plant length (cm)** | 2.29 | 2.16 | 2.31 | 0.389 | 5.05a | 7.18a | 1.74b | \*<0.001 | 3.19b | 3.63a | 2.35c | \*<0.001 | 9.20a | 8.05b | 6.80ac | \*0.009 |
| **Proportion of dead plant (cm2)** | 5.70b | 2.20c | 17.60a | \*<0.001 | 4.5b | 1.80c | 14.70a | \*<0.001 | 34.20a | 13.50c | 20.60b | \*<0.001 | 1.10b | 1.50b | 6.80a | \*<0.001 |
| **No. of plant species** | 28a | 25b | 17ac | \*0.002 | 36a | 34a | 20b | \*<0.001 | 21a | 20a | 14.00b | \*0.009 | 16b | 27a | 15b | \*<0.001 |

Table 3: Differences between rotational and continuous grazing in the humid grass steppe and Andean ecosystems. Positive differences (RG-CG) signify rotational grazing management has the highest value and negative that continuous grazing management has the highest value. Significant levels shaded.

|  |  |  |
| --- | --- | --- |
|   | **Humid grass steppe** | **Andean vegetation** |
|   | **RG *m***  | **CG *m*** | **RG-CG** | ***μ* π−ϖα*λ*υε** | **RG *m***  | **CG *m*** | **RG- CG** | ***μ* π−ϖα*λυ*ε** |
| Soil water infiltration (cm/h) | 1.92 | 1.06  | 0.86 | 0.005 | 2.33 | 2.14  | 0.19 | 0.61 |
| Soil water retention capacity (mm/cm) | 2.10 | 1.97  | 0.13 | 0.3 | 3.05 | 2.96  | 0.09 | 0.994 |
| Soil organic matter (%) |  4.90 | 4.80  | 0.1 | 0.269 | 3.5 | 5.8  | -2.3 | <0.001 |
| Soil N (%) | 0.26 | 0.25  | 0.01 | 0.008 | 0.3 | 0.5  | -0.2 | <0.001 |
| Soil respiration (g C/m2/h) | 0.65 | 0.62  | 0.03 | 0.735 | 0.6 | 1.4 | -0.8 | <0.001 |
| Bare soil (%) | 0.4 | 1.9  | -1.5 | 0.046 | 0.8 | 4.8  | -4 | 0.001 |
| Litter cover (%) | 4.0 | 7.3  | -3.3 | 0.232 | 16.6 | 15.6 | 1 | 0.122 |
| Grasses and graminoids (%) | 71.0 | 81.3  | -10.3 | 0.024 | 52.8 | 76.1 | -23.1 | 0.907 |
| Forbs (%) | 17.4 | 6.0  | 11.4 | 0.004 | 26.7 | 1.5 | 24.94 | 0.001 |
| Shrubs (%) | 5.0 | 0.7  | 4.3 | 0.039 | 0.0 | 0.0  | 0 | 0.052 |
| Dwarf-shrubs (%) | 2.1 | 2.8  | -0.7 | 0.154 | 3.3 | 2.0  | 1.3 | 0.994 |
| ANPP (gC/m2) | 17.4 | 19.5  | -2.1 | 0.038 | 16.4 | 32.3 | -15.8 | 0.759 |
| Soil erosion (%) | 1.0 | 1.0  | 0 | 0.153 | 1.1 | 0.5  | 0.6 | 0.386 |
| Aerial plant (g) | 5.6 | 5.2  | 0.4 | 0.299 | 6.9 | 8.4  | -1.5 | 0.007 |
| Root (g) | 4.3 | 3.2  | 1.1 | 0.171 | 6.7 | 4.1  | 2.6 | <0.001 |
| Root/aerial plant ratio | 0.78 | 0.61  | 0.17 | <0.001 | 0.96 | 0.49  | 0.47 | <0.001 |
| Root depth (cm) | 14.7 | 14.1  | 0.6 | 0.122 | 15.1 | 15.4 | -0.3 | 0.041 |
| Diameter at base (cm) | 4.8 | 3.5  | 1.3 | 0.749 | 5.9 | 5.9  |  0.0 | 0.664 |
| Plant length (cm) | 3.7 | 7.2  | -3.5 | 0.219 | 4.1 | 8.1  | -4 | <0.001 |
| Proportion of dead plant (cm2) |  2.7 | 1.8  | 0.9 | 0.012 | 0.9 | 1.5  | -0.6 | 0.006 |
| No. Plant species | 40.7 | 34.0  | 6.7 | 0.003 | 39.9 | 27.0  | 12.9 | 0.074 |