

Impact of Grazing and Silvopastoral Systems on Carbon and Nitrogen in Sodic Soils of the Dry Chaco

Natalia Banegas

`banegas.natalia@inta.gob.ar`

Instituto de Investigación Animal del Chaco Semiárido (IIACS), Instituto Nacional de Tecnología Agropecuaria (INTA)

Daniel Dos Santos

Instituto de Biodiversidad Neotropical (IBN, CONICET-UNT)

Emilce Viruel

Instituto de Investigación Animal del Chaco Semiárido (IIACS), Instituto Nacional de Tecnología Agropecuaria (INTA)

Néstor Ignacio Gasparri

Instituto de Ecología Regional (IER, CONICET-UNT)

Research Article

Keywords: Productive Systems, Sodic soil, Soil Organic Carbon, Total Nitrogen, Tree canopy

Posted Date: September 6th, 2024

DOI: <https://doi.org/10.21203/rs.3.rs-4888294/v1>

License:   This work is licensed under a Creative Commons Attribution 4.0 International License.

[Read Full License](#)

Additional Declarations: No competing interests reported.

Version of Record: A version of this preprint was published at Agroforestry Systems on June 28th, 2025. See the published version at <https://doi.org/10.1007/s10457-025-01245-1>.

Abstract

Grazing and silvopastoral systems represent effective strategies for enhancing soil organic carbon (SOC) and total nitrogen (TN) availability in sodic soils. This study conducted a comprehensive assessment over a 6-year period to evaluate alternative cattle production methods aimed at increasing SOC and TN levels across various soil depths. Mineral-associated organic carbon (AOC) and particulate organic carbon (POC) fractions were analyzed to elucidate the dynamics of SOC. The experimental plots, totaling 9 hectares each, included pure pasture (PP), silvopastoral systems under tree canopy (SPS-UC), and silvopastoral systems between tree canopies (SPS-BC), all cultivated with *Chloris gayana* cv Epica INTA-Pemán. Trees of *Neltuma alba* (syn. *Prosopis alba*) were planted in the silvopastoral area in 1998. Statistical analyses focused on evaluating the impacts of these treatments, temporal effects, and their interactions on SOC, POC, AOC, and TN across four measurement points. Significant differences were observed in the distribution of SOC, POC, AOC, and TN between PP and SPS systems. Notably, SPS-BC exhibited the lowest SOC and TN values. Both PP and SPS-UC showed increases in SOC within the top 50 cm of soil depth, primarily attributed to elevated AOC levels. These findings underscore the potential of grazing and silvopastoral systems in increase soil fertility by increments in soil organic matter to mitigate sodic soil limitations. Moreover, the study highlights the necessity for further research in silvopastoral systems, with a high possibility in implementation for livestock production in Dry Chaco, to investigate how different tree configurations influence SOC and TN dynamics in these soils.

1. Introduction

Soil salinization and alkalinization are significant threats to global soil resources and rank among the most common processes of land degradation. Approximately 210 million hectares of soil worldwide are classified as sodic or alkali (Levy and Shainberg 2005). In Argentina, saline and sodic conditions affect about 8.5 million hectares, spanning both arid and semiarid environments. This makes Argentina the country with the largest area of saline and sodic soil in South America, and the third largest globally, following Russia and Australia. It is estimated that 53,139 hectares in Argentina are specifically classified as sodic soils. The presence of sodium in these soils negatively impacts their infiltration capacity, increases their susceptibility to water and wind erosion, and exposes more soil organic matter to decomposition (Taleisnik and López-Launestein 2011). Furthermore, there is documented evidence showing that both salinization and sodification have occurred due to changes in land use and deforestation in the Dry Chaco region of Argentina (Jobbágy et al. 2021; Maertens et al. 2022). For areas with such soil limitations, which are prone to severe degradation, it is crucial to evaluate practices that enhance production capacity while simultaneously improving soil conditions and reducing degradation risks.

Sodic soils are marked by elevated pH levels throughout their profile, high exchangeable sodium content, low organic matter, and sparse natural vegetation (Kaur et al. 2002). In these less fertile agricultural regions, sustainable livestock production practices offer a viable alternative with notable environmental and economic advantages (Singh 1995; Kaur et al. 2002). The introduction of perennial grasses,

afforestation, agroforestry, and silvopastoral systems can enhance the biological, physical, and chemical properties of sodic soils. These practices boost microbial activity, increase nitrogen availability, reduce soil erosion, and promote the formation and sequestration of soil organic matter (Cardinael et al. 2017; Godde et al. 2020; Fernández et al. 2020).

Soil organic carbon (SOC) is crucial for maintaining soil resilience, supporting ecosystem services, and mitigating climate change (Li et al. 2020). Increasing SOC stocks, or SOC sequestration, involves adding soil organic matter (SOM), stabilizing SOM into long-lasting SOC pools, and conserving SOC by reducing carbon loss pathways. Both aboveground and belowground biomass contribute to SOM, while microbial interactions play a key role in either stabilizing SOM or causing its loss, as well as influencing greenhouse gas emissions based on grazing and land management practices (Sarkar et al. 2020).

Quantifying both labile and recalcitrant SOC pools is crucial for understanding carbon storage, stabilization, and retention mechanisms within the soil ecosystem (Zimmermann et al. 2007; Basak et al. 2021). The labile pool, characterized by shorter residence times and rapid oxidation, plays a significant role in the soil food web, energy flow, and nutrient cycling. In contrast, the recalcitrant pool consists of compounds that are physically and biochemically protected, decomposing slowly due to microbial activity, thus having longer residency times and enhancing carbon sequestration (Zimmermann et al. 2007; Basak et al. 2021).

Gaining a deeper understanding of how different cattle production strategies impact sodic soils in the Dry Chaco area, particularly regarding SOC and total nitrogen (tN) stocks and the dynamics of organic carbon fractions, is vital for landowners. This knowledge is essential for the effective technical and political implementation of sustainable land management practices. Furthermore, the insights generated can inform the proper management of sodic soils to ensure sustainable cattle production strategies.

In this study, we evaluate the effects of different cattle production systems (silvopastoral vs. grazing) on SOC and tN content at different soil depths in the Dry Chaco's sodic soils. We also assess whether these management strategies influence SOC fractions, and thereby, the stability of SOC over time.

2. Material and Methods

2.1. Site

This study was conducted at the Animal Research Institute of Semiarid Chaco, located in Leales, southeastern Tucumán province, Argentina (27° 11' S and 65° 17' W, Fig. 1). The institute is situated in the depressed saline plain area at an elevation of 335 meters above sea level. The region experiences a subtropical sub-humid climate with a marked dry season from April to October. The mean annual precipitation is 880 mm, concentrated mainly from October to March, and the mean annual temperature ranges from 13°C in July to 25°C in January. The soil type falls under the Sub-group Fluvaquent Haplustolls of the Mollisols Order, according to the US Soil Taxonomy System (Soil Survey Staff 2014).

The experiment was conducted in an 18-hectare field, with 9 hectares implemented as Silvopastoral System (SPS) and the other remaining 9 hectares used as Grazing System (Pure Pasture - PP). The entire area was cultivated with the African grass *Chloris gayana* cv Epica INTA-Pemán in 2010. In the SPS, *Neltuma alba* (syn. *Prosopis alba*) trees were planted in 1998, with an arrangement of 10 x 10 m that resulted in a density of 100 trees/ha. Both systems were rotationally grazed by Braford heifers from June of one year (average body weight of 160 kg) to March of the following year (average body weight of 280 kg), spanning a period of 10 months. During the winter and spring months (June to November), deferred forage was utilized, supplemented with an energetic and proteic diet at a rate of 0.06% of body weight. The length of stay in the paddocks during the winter period was determined based on forage availability. In the summer period, the rotational grazing schedule was determined based on the accumulation of growing degree days (GDD), with an optimal threshold of 400 GDD established for both systems (Martínez Calsina et al. 2015).

2.2. Experimental design and soil samples

A completely randomized design was employed, with six paddocks established in each system. Soil samples were collected at different depths: 0–20 cm, 20–50 cm, and 50–100 cm in March of 2011, 2013, 2015, and 2017. Six soil subsamples were collected per plot for each depth and combined to obtain a composite sample for chemical determinations. The samples were air-dried and sieved through a 2 mm mesh to separate plant material. Additionally, soil bulk density samples were taken at the same times and depths using the cylinder method (Blake and Hartge, 1986). These samples were oven-dried at 105°C for 48 hours and weighed without coarse particles (> 2 mm). Bulk density (g cm^{-3}) was calculated as the ratio of the dry mass to the cylinder volume.

For the SPS, soil samples were taken at two sites: i) under the tree canopy and ii) in the equidistance of four trees (i.e the site with less influence of trees). In consequence, the evaluated conditions (treatments) were: 1) Pure pasture (PP), 2) Silvopastoral system under tree canopy (SPS-UC), and 3) Silvopastoral system between tree canopies (SPS-BC).

2.3. Soil determinations

SOC and tN stocks were assessed in this study. SOC was determined using the wet oxidation method of Walkley-Black (Nelson and Sommers 1982). Particulate organic carbon (POC) and mineral-associated organic carbon (AOC) were determined by dispersion and sieving following the method of Cambardella and Elliot (1992). The content of tN was determined using the Kjeldahl method (Bremmer and Mulvaney 1982).

To estimate SOC in tons per hectare, a volumetric conversion using bulk density and analyzed depth was employed to determine the weight of the topsoil. The SOC content, expressed as a percentage, was then converted to a measure of mass (Ellert and Bettany 1995). For the SPS plots, OC stocks were calculated (Mg C ha^{-1}) by combining the SOC stocks under the tree canopy and between the tree canopies, considering their respective relative surface areas (Cardinael et al. 2017).

2.4. Statistical analyses

All graphics and statistical analyses were performed with R software version 4.2.2 (R Core Team 2022). We assessed the effects of different conditions (treatment) on soil parameters at different soil depth layers separately. The statistical analyses aimed to evaluate the significance of treatment, time, and their interaction on SOC, POC, AOC, and tN over the four measurement periods (March 2011, 2013, 2015, and 2017).

A mixed-effects model was utilized, employing the 'lmer' function from the 'lme4' package in R (version 1.1, CRAN), with the following formula:

```
lmer(Soil indicator ~ Treatment * Time + (1 | EU), data = dataset_atdepthx)
```

Here, 'Soil indicator' represents the specific soil parameter being analyzed. A random effect for the experimental unit (EU) was included to account for potential correlations within the data.

Finally, the overall stock of SOC and tN was compared before and after the complete experimental period. We performed a one-way analysis of covariance (ANCOVA) to assess the impact of the treatments (i.e., PP, SPS-UC, and SPS-BC). The dependent variable was the post-treatment content of soil components (SOC or tN), measured at specific soil depths, with the covariate being the pre-treatment assessment of the same variable.

3. Results

We fitted a linear mixed model using Restricted Maximum Likelihood estimation (REML) and the nloptwrap optimizer to predict soil parameters based on Treatment and Time. The model was specified as follows: Soil Parameter ~ Treatment * Time + (1|EU), where EU represents the experimental units included as random effects. Soil parameters encompass measurements of SOC, its fractions POC and AOC, and tN at various soil depths over multiple years. The estimates, confidence intervals, and p-values for each fixed effect are summarized in Table 1. The model's intercept is reported alongside this section, corresponding to the average score of responses for the PP treatment at the onset of the field experiment measurements (i.e. Time = 0). Significance level was set at $\alpha = 0.05$. Figure 1 represents the scatterplots of data set with fixed-effects regression lines.

3.1. SOC and tN content at different depths

The fixed effects of the model demonstrated significant impacts on SOC measurements in the topsoil layer (0–20 cm) (Fig. 1a, top left). The model's overall explanatory power is substantial, with the portion attributed to fixed effects alone (marginal R^2) at 0.90. The model's intercept was $1.17 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [1.12, 1.22]). Over time, SOC increased significantly for the baseline condition (beta = $0.03 \text{ g } 100 \text{ g}^{-1} \text{ soil } \text{y}^{-1}$, 95% CI [0.02, 0.04]), showing no difference from the rate of change in SPS-UC. In contrast, the interaction term SPS-BC:Time had an average value of -0.05 units compared to PP, representing a

significant, steepest negative effect on SOC. This behavior can be also expressed in terms of average rate of change in $\text{g C kg}^{-1} \text{ soil y}^{-1}$.

Concerning tN (Fig. 1d, bottom left), the model's explanatory power related to fixed effects alone was 0.94, with an intercept at $0.09 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [0.09, 0.10]). Time demonstrated a statistically significant positive effect ($\beta = 2.75\text{e-}03 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$), fostering tN levels in both PP and SPS-UC systems. However, the interaction term SPS-BC:Time had a contrasting negative impact ($\beta = -4.09\text{e-}03 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$) on the baseline condition, resulting in the lowest tN content at the end of the experimental period.

At the intermediate soil layer (20–50 cm) for SOC, the model exhibited substantial explanatory power ($R^2 = 0.93$). We observed significant differences in initial conditions between SPS-UC and SPS-BC compared to PP, with average variations of $0.23 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$ and $0.17 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$, respectively. Time exhibited a positive, albeit more moderate, effect ($\beta = 0.02 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$). Notably, the interactions of Treatment with Time revealed significant effects: SSP-UC demonstrated a positive impact ($\beta = 0.03 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$), while SSP-BC exhibited a negative effect ($\beta = -0.03 \text{ g } 100 \text{ g}^{-1} \text{ y}^{-1}$) on SOC concentration, as compared to the slope fitted for PP (Fig. 1a, center). For tN, the model's explanatory power related to the fixed effects alone (marginal R^2) is 0.91. The model's intercept is at $0.04 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [0.04, 0.05]). PP showed a consistent positive change over time ($\beta = 2.20\text{e-}03 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$). The interaction effects with Time were significant and varied for the other treatments: SSP-UC:Time ($\beta = 1.65\text{e-}03 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$) indicated a strongest positive change, whereas SPS-BC:Time ($\beta = -2.83\text{e-}03 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$) demonstrated a negative change regarding PP (Fig. 1d, center).

To deepest layer (50–100 cm), neither Treatment nor Time significantly influenced the measured variables (SOC, POC, AOC, tN) based on the models fitted (Fig. 1a-e, right). For SOC, the linear mixed model showed a marginal R^2 of 0.13 and intercept at $0.31 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [0.26, 0.36]). Regarding POC, the marginal R^2 was 0.26 and intercept at $0.12 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [0.09, 0.14]). For AOC, the model's marginal R^2 was 0.10 and intercept at $0.20 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [0.15, 0.24]). In the case of tN, the marginal R^2 was 0.07 and intercept at $0.02 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [0.02, 0.03]).

3.2. Net change in SOC and tN stocks

We performed a one-way ANCOVA, examining the influence of the system (PP, SPS-UC, SPS-BC) on post-treatment soil component content (SOC or tN) at a specific depth, using the pre-treatment assessment as a covariate. A preliminary analysis examined the homogeneity-of-slopes assumption, checking if the relationship between the covariate and the dependent variable differed significantly across various combinations of soil components and depths. Results showed no significant differences in this relationship based on the independent variable after comparison of nested models using the F test (always $P > 0.05$). Figure 2 displays the results of separate ANCOVA analyses, examining the relationship

between pre- and post-treatment concentrations of OC and tN at different soil depths. Each panel in the figure corresponds to a distinct combination of soil depth and parameter, with data points representing actual observations and lines depicting model fits. The presence of the dashed line, representing the identity line, allows for the visual assessment of deviations. In the context of Fig. 2, if there is no difference between pre and post values, one would expect the points to align along the identity line, representing the scenario where x equals y .

Revisiting the previous point, ANCOVA was employed to examine outcomes associated with SOC and tN concentrations, utilizing pre-treatment values and system as predictor variables. This analysis was performed for each of the three soil depth layers. In the model for SOC at the topsoil layer (Fig. 2, upper left panel), both the pre-treatment scores and the system were found to be significant (adjusted $R^2 = 0.96$). Here, results highlight a direct but non-significant relationship between the pre- and post-treatment (beta = 0.26, $p = 0.25$). The model indicated a significant positive effect of SPS-UC (beta = $0.21 \text{ g } 100 \text{ g}^{-1} \text{ soil}$, $p = 0.027$) and a significant negative effect of SPS-BC (beta = $-0.19 \text{ g } 100 \text{ g}^{-1} \text{ soil}$, $p = 0.004$) compared to PP. Similarly, the impact of pre-treatment scores on post-treatment ones was found to be non-significant (beta = $0.17 \text{ g } 100 \text{ g}^{-1} \text{ soil}$, $p = 0.57$) in the model assessing SOC content at the intermediate soil deep layer (adjusted $R^2 = 0.96$). Here, the SPS-UC system demonstrated a significant increase (beta = $0.36 \text{ g } 100 \text{ g}^{-1} \text{ soil}$, $p = 4.3\text{e-}04$) regarding PP. A comparable effect was also observed in the model for SOC at the deepest soil layer (SPS-UC: beta = $0.05 \text{ g } 100 \text{ g}^{-1} \text{ soil}$, $p = 6.3\text{e-}03$). This last model performed excellently well (adjusted $R^2 = 0.99$) and retrieved a significant positive relationship between pre- and post-treatment concentrations of SOC (beta = 0.57, $p = 1.5\text{e-}03$).

In contrast to previous models, tN concentrations generally exhibit a negative yet non-significant correlation between pre- and post-treatment observations (Fig. 2, right column). For the topsoil layer, the ANCOVA model performed well (adjusted $R^2 = 0.97$). The SPS-BC system had a significant positive impact (beta = $0.03 \text{ g } 100\text{g soil}^{-1}$, $p = 5.4\text{e-}03$), while the SPS-EC system displayed a significant negative effect (beta = $-0.02 \text{ g } 100\text{g}^{-1} \text{ soil}$, $p = 4.1\text{e-}05$) compared to PP. Moving on to the intermediate soil deep layer, the ANCOVA model (adjusted $R^2 = 0.96$) indicated the same trend of significant positive effect for SPS-UC (beta = $0.03 \text{ g } 100\text{g soil}^{-1}$, $p = 4.0\text{e-}4$), and negative for SPS-BC (beta = $-0.01 \text{ g } 100\text{g soil}^{-1}$, $p = 7.6\text{e-}03$). Finally, for the deepest soil layer, the ANCOVA model also performed well (adjusted $R^2 = 0.91$). Again, SPS-UC displayed a significant positive effect (beta = $0.01 \text{ g } 100\text{g soil}^{-1}$, $p = 5.0\text{e-}04$) whereas a significant negative effect for the SPS-BC system (beta = $-4.2\text{e-}03 \text{ g } 100\text{g soil}^{-1}$, $p = 7.3\text{e-}03$).

Between 2011 and 2017, throughout the duration of our experiment, the average SOC stock accumulation showed notable variations (Fig. 2; Table 1).

3.3. SOC Dynamics: POC and AOC at different depths

For POC, the model revealed significant impacts of factors (Fig. 1b, left). Its total explanatory power was substantial, with the portion related to fixed effects alone (marginal R^2) at 0.48. The baseline scenario scored an average of $0.63 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [0.59, 0.69]), remaining stable over time. SPS-UC:Time

($\beta = -0.01 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$) and SPS-BC:Time ($\beta = -0.02 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$) had statistically significant negative effects. Regarding AOC (Fig. 1c, left), the model's explanatory power related to fixed effects alone (marginal R^2) was 0.93, with an intercept of $0.54 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [0.50, 0.58]). AOC increased significantly over time in the reference category PP ($\beta = 0.02 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$, 95% CI [0.01, 0.03]), similar to the effect observed in SPS-UC. In contrast, the rate of change for SPS-BC was significantly lower than that of PP, with a difference of -0.03 units.

At 20–50 cm of soil depth, the model of POC demonstrated a strong explanatory power of 0.93. PP exhibited consistent behavior over the years, while SPS-UC:Time showed a significant positive effect ($\beta = 0.03 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$) and SPS-BC:Time exhibited a significant negative impact ($\beta = -0.01 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$) (Fig. 1b, center). For AOC, the model exhibited substantial explanatory power (conditional $R^2 = 0.89$). The model's intercept was found to be $0.25 \text{ g } 100 \text{ g}^{-1} \text{ soil}$ (95% CI [0.22, 0.28]). It was observed that Treatment SPS-UC displayed a consistent positive trend over time, akin to PP, indicating an increase of approximately 0.02 units per year. In contrast, the term SPS-BC:Time had a significant negative effect ($\beta = -0.02 \text{ g } 100 \text{ g}^{-1} \text{ soil y}^{-1}$), indicating no significant change over time in comparison to the other treatments (Fig. 1c, center).

4. Discussion

The stratification patterns observed in SOC, POC, AOC, and tN distribution within grazing and silvopastoral systems are attributed to increased inputs from aboveground and belowground sources, root turnover, and reduced soil disturbance (Howlett et al. 2011; de Souza Almeida et al. 2021). During the trial period, Silvopastoral Systems (SPS-UC and SPS-BC) showed the highest values of these variables at soil depths of 0–20 cm and 20–50 cm. This trend can be linked to the age and growth of *Neltuma alba* (syn. *Prosopis alba*), planted in 1998, which by evaluation had matured over 14 years, resulting in extensive aboveground and belowground development. The deeper, thicker roots facilitated organic material incorporation into the soil (Valenzuela Que et al. 2022). Deep-rooted trees also accessed soil nutrients, intercepted leached nutrients, promoted nutrient recycling (via litter and fine root turnover), and enhanced nutrient use efficiency, contributing to system sustainability (Zhu et al. 2020; Valenzuela Que et al. 2022). Additionally, native herbaceous species likely contributed residues to the silvopastoral systems compared to tropical pasture (*Chloris gayana* cv Finecut), sown in 2010, a year before initial sampling. Over subsequent years, increased litter deposition, root development, and turnover contributed significantly to organic carbon dynamics (Battisti et al. 2018; de Abreu et al., 2020). Systems with perennial pastures typically exhibit significant changes in organic matter during the first three to five years after establishment, driven by root biomass and exudates supporting soil microbes and organic matter formation (Sanford 2014).

Consistent with previous studies, the SPS-BC treatment showed the lowest SOC and tN values. This could be due to soil C and N heterogeneity across distances from tree planting lines, and variations in residue input quantity and quality (Upson et al. 2016; Guillot et al. 2019; de Abreu et al. 2020). Viruel et

al. (2022) also found similar patterns in regional systems, linking differences between grass, tree rows, and tree canopy to resource dynamics that create fertility islands and unique soil microenvironments (Cubillos et al. 2016; Xu et al. 2017; Chandregowda et al. 2018). Microbial communities under tree canopies regulate nutrient availability and facilitate soil processes, potentially influencing organic material mineralization. Additionally, continuous grazing in tree rows (SPS-BC) may lead to mechanically compacted clods that inadequately protect SOC, enhancing its mineralization from aggregates (Kuzyakov 2002; Wiesmeier et al. 2012). Increased grazing pressure in grassy areas between trees could further contribute to nutrient loss (Allington and Valone 2014; Magliano et al. 2017).

At depths of 50–100 cm, SPS-UC showed higher SOC and tN content, which remained stable across all treatments over time. The elevated tN levels in SPS-UC at shallower depths could be attributed to nitrogen-rich leaf deposition from *Neltuma alba* (syn. *Prosopis alba*), along with root recycling and nodular nitrogen turnover from biological fixation (Burle et al. 2003; Banegas et al. 2019). The C:N ratio values ranged from 11.9 to 13.7 across all depths and treatments (data not shown). Our findings indicated a decreasing C:N ratio over time at depths of 0–20 cm and 20–50 cm, suggesting increased carbon decomposition likely due to higher nitrogen inputs (Stevenson et al. 2023).

Understanding organic carbon fractions is crucial for elucidating organic matter formation and stabilization dynamics (Villarino et al. 2021). In our study, the initial increase in SOC within the top 50 cm of soil depth in SPS-UC and PP corresponded to gains in active organic carbon (AOC). Initially, at 0–20 cm depth, AOC comprised 45.6%, 55.6%, and 44.9% of SOC for PP, SPS-UC, and SPS-BC, respectively. By the end of the evaluation, these proportions had increased to 51.1% for PP and 63.8% for SPS-UC, while remaining constant at 46.1% for SPS-BC. Similarly, at 20–50 cm depth, these fractions were 48.7%, 46.7%, and 32% for PP, SPS-UC, and SPS-BC, respectively, increasing to 62.3%, 42.6%, and 38% by the evaluation's end. Villarino et al. (2021) note that AOC persists relatively long in soil but requires substantial nitrogen inputs for formation and has limited storage capacity. Under the tree canopy (SPS-UC), the abundance of biomass inputs and nitrogen increase over time likely promote AOC formation, thereby enhancing SOC content. Consistently, Cá et al. (2022) found that SOC sequestration in silvopastoral systems is tied to organic matter inputs from litterfall, root exudation, and turnover beneath tree canopies, fostering nutrient and organic matter accumulation and stimulating microbial activity. Additionally, legume-grass associations in SPS-UC may alter litter carbon-to-nitrogen ratios, accelerating decomposition rates, enhancing microbial activity efficiency, promoting nutrient cycling, and forming more stable organic compounds (Cotrufo et al. 2013; Liang et al. 2017; Bai and Cotrufo 2022).

For PP, the observed increases in SOC can be attributed to several factors (Tonucci et al. 2011). Firstly, grasses are known for their extensive root systems, which significantly contribute to soil carbon accumulation. In grassland ecosystems, approximately 60% of net primary productivity is allocated belowground (Bai and Cotrufo 2022). Banegas et al. (2020) reported root biomass values ranging from 14,747 to 15,408 kg ha⁻¹ in the upper 40 cm of soil under *Chloris gayana* cv Finecut. Fisher et al. (1994) documented carbon sequestration rates by grasses ranging from 3 to 14 Mg C ha⁻¹ y⁻¹, with 75% of this carbon accumulating below 20 cm depth, where it is less susceptible to oxidation. Root carbon inputs

are incorporated into SOC more effectively than aboveground inputs due to their chemical composition and close interaction with soil microorganisms. On average, root carbon inputs exhibit a SOC stabilization efficiency five times greater than aboveground carbon inputs (Bai and Cotrufo 2022). Secondly, grass roots generally have faster turnover rates compared to tree roots (Fujisaka et al. 1998), contributing to increased carbon accumulation in pasture soils (Tonucci et al. 2011). Some studies suggest that plants allocating more carbon to roots play a crucial role in soil carbon sequestration, particularly in the formation of active organic carbon (AOC) (Bai and Cotrufo 2022). Lastly, grassland-derived soils are reported to have a higher potential for carbon stabilization, possibly because carbon derived from pastures is rapidly associated with the fine soil fraction (Collins et al. 2000; Tonucci et al. 2011).

Conversely, the observed decline in SOC over time in SPS-BC was linked to decreases in particulate organic carbon (POC) content in the top 50 cm of soil depth. POC forms through the fragmentation of plant and microbial residues, consisting of lightweight fragments composed of large polymers. Its presence in the soil is strongly influenced by residue quantity and decomposition rate (Carter et al. 1996; Villarino et al. 2021; Bai and Cotrufo 2022). Consistent with this, SPS-BC exhibited lower grass productivity and soil cover (data not shown), contributing to the decline in SOC observed.

Tree planting and silvopastoral systems are recognized as effective strategies for improving saline and sodic soil conditions (Gupta et al. 2019; Sileshi et al. 2020). Introducing trees into vegetation management in sodic areas offers several benefits. The deep root systems of trees contribute to lowering the water table by absorbing water deeply, aiding in the gradual leaching of salts from the topsoil. Combining trees with pasture enhances carbon input and promotes microbial activity, facilitating the accumulation of soil organic carbon. Our study in the Dry Chaco region, which faces natural sodicity and salinity limitations exacerbated by human activities like deforestation and inefficient water management, consistently showed these beneficial effects. Implementing silvopastoral systems with *Neltuma alba* (syn. *Prosopis alba*) emerges as a promising approach to restore and mitigate one of the region's primary land degradation processes.

5. Conclusion

This study highlights the significant potential of grazing and silvopastoral systems for SOC and tN sequestration within the 100 cm soil profile in sodic soils of Dry Chaco. The accumulation predominantly occurred within the first 50 cm of soil in both systems. However, the distribution of SOC and tN in silvopastoral systems varied based on the sample site, emphasizing the importance of site-specific factors in influencing carbon and nitrogen dynamics. These findings underscore the potential of grazing and silvopastoral systems to enhance soil fertility by increasing soil organic matter, thereby mitigating the limitations of sodic soil. Moreover, the study highlights the need for further research into silvopastoral for implementation in livestock production in the Dry Chaco. This research should focus on how different tree configurations influence soil organic (SOC) and total Nitrogen (tN) dynamics, ultimately improving land management practices for carbon and nitrogen sequestration in sodic soils.

Declarations

6.Acknowledgements

PNSUELO-1134042. Aprovechamiento de residuos para aumentar el reciclado en el suelo. Sumideros de carbono y emisiones del suelo.

PNPA-11260714. Pasturas Ecoeficientes y de Bajo Carbono en Ganadería.

PE-E1-I015-001. Sistemas Silvopastoriles integrados hacia un manejo sustentable.

References

1. Allington GRH, Valone TJ (2014) Islands of fertility: A byproduct of grazing? *Ecosyst*, 17: 127–141. <https://doi.org/10.1007/s10021-013-9711-y>
2. Banegas NR, Corbella R, Viruel E, Plasencia A, Roig B, Radrizzani A (2019) *Leucaena leucocephala* introduction into a tropical pasture in the Chaco region of Argentina. Effects on soil carbon and total nitrogen. *Trop Grassl* 7:295–302. [https://doi.org/10.17138/tgft\(7\)295-302](https://doi.org/10.17138/tgft(7)295-302)
3. Banegas N, Dos Santos DA, Guerrero Molina F, Albanesi A, Pedraza R (2020) Glomalin contribution to soil organic carbon under different pasture managements in a saline soil environment. *Arch Agron Soil Sci* 68: 340-354. <https://doi.org/10.1080/03650340.2020.1834536>
4. Basak N, Mandal B, Datta A, Chandra Kundu M, Kumar Rai A, Basak P, Mitran T (2020) Stock and stability of organic carbon in soils under major agro-ecological zones and cropping systems of sub-tropical India. *Agric Ecosyst Environ* 312: 107317. <https://doi.org/10.1016/j.agee.2021.107317>.
5. Battisti LFZ, Schmitt Filho AL, Loss A, Sinisgalli PADA (2018) Soil chemical attributes in a high biodiversity silvopastoral system. *Acta Agronómica* 67: 486-493. <https://doi.org/10.15446/acag.v67n4.70180>
6. Blake GR, Hartge KH (1986). Bulk density. In: Klute, A., (Ed.), *Methods of Soil Analysis, Part 1—Physical and Mineralogical Methods*, 2nd Edition, Agronomy Monograph 9, American Society of Agronomy. *Soil Sci Soc Am*, pp 363-382
7. Bremner JM, Mulvaney CS (1982) Nitrogen total. In: Page, A.L., Miller, R.H. and Keeney, D.R. (Eds.), *Methods of soil analysis: Part II. Chemical and Microbiological Properties-Agronomy Monograph No. 9*. American Society of Agronomy *Soil Sci Soc Am*. pp 371-378
8. Burle STM, Shelton HM, Dalzell SA (2003) Nitrogen cycling in degraded *Leucaena leucocephala-Brachiaria decumbens* pastures on an acid infertile soil in south-east Queensland, Australia. *Trop Grassl* 37:119–128
9. Cá J, Ferreira Lustosa Filho J, Rocha da Silva N, Tavares de Castro CR, Senna de Oliveira T (2022) C and N stocks in silvopastoral systems with high and low tree diversity: Evidence from a twenty-two

- year old field study. *Sci Total Environ* 833: 155298. <https://doi.org/10.1016/j.scitotenv.2022.155298>.
10. Cambardella CA, Elliott ET. 1992. Particulate soil organic-matter changes across a grassland cultivation sequence. *Soil Sci Soc Am J* 56:777-783.
<https://doi.org/10.2136/sssaj1992.03615995005600030017x>
 11. Cardinael R., T. Chevallier, A. Cambou, C. Béral, B. Barthès, C. Dupraz, C. Duran, E. Kouakoua , C. Chenu. 2017. Increased soil organic carbon stocks under agroforestry: A survey of six different sites in France. *Agriculture, Ecosystems and Environment* 236: 243–255.
<https://doi.org/10.1016/j.agee.2016.12.011>
 12. Carter MR (1996) Analysis of soil organic matter storage in agroecosystems. In: Carter, M.R., Stewart, B.A., (Eds.), *Structure and organic matter storage in agricultural soil*. CRC Press, pp 3–14.
 13. Chandregowda MH, Murthy K, Bagchi S (2018) Woody shrubs increase soil microbial functions and multifunctionality in a tropical semi-arid grazing ecosystem. *J Arid Environ* 155: 65–72.
<https://doi.org/10.1016/j.jaridenv.2018.02.006>
 14. Collins HP, Elliott ET, Paustian K, Bundy LG, Dick WA, Huggins DR, Smucker AJM, Paul EA (2000) Soil carbon pools and fluxes in long-term corn belt agroecosystems. *Soil Biol Bioch* 32:157–168.
[https://doi.org/10.1016/S0038-0717\(99\)00136-4](https://doi.org/10.1016/S0038-0717(99)00136-4)
 15. Cotrufo MF, Wallenstein MD, Boot CM, Denef K, Paul E (2013). The Microbial Efficiency-Matrix Stabilization (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: do labile plant inputs form stable soil organic matter? *Global Change Biol* 19:988–995.
<https://doi.org/10.1111/gcb.12113>
 16. Cubillos AM, Vallejo VE, Arbeli Z, Terán W, Dick R, Molina CH, Molina E, Roldan F (2016) Effect of the conversion of conventional pasture to intensive silvopastoral systems on edaphic bacterial and ammonia oxidizer communities in Colombia. *Eur J Soil Biol* 72: 42–50.
<https://doi.org/10.1016/j.ejsobi.2015.12.003>
 17. de Abreu LHG, de Freitas IC, Santana PHL, de Almeida Barbosa DL, Tuffi Santos LD, Santos MV, Arruda Sanglard D, Almeida Frazao L (2020) Variation in soil carbon, nitrogen and microbial attributes within a silvopastoral system in the Brazilian Cerrado. *Agrofor Syst* 94: 2343–2353.
<https://doi.org/10.1007/s10457-020-00554-x>
 18. de Souza Almeida LL, Frazão LA, Lessa TAM, Fernandes LA, de Carvalho Veloso AL, Lana AMQ, de Souza IA, Pegoraro RF, Ferreira EA (2021) Soil Carbon and Nitrogen Stocks and the Quality of Soil Organic Matter under Silvopastoral Systems in the Brazilian Cerrado. *Soil Tillage Res* 205, 104785.
<https://doi.org/10.1016/j.still.2020.104785>
 19. Ellert BH, Bettany JR (1995) Calculation of organic matter and nutrients stored in soils under contrasting management regimes. *Can J Soil Sci* 75:529-538. <https://doi.org/10.4141/cjss95-075>
 20. Fernández P, le Polain de Waroux Y, Jobbágy E, Loto D, Gasparri NI (2020) A hard-to-keep promise: Vegetation use and aboveground carbon storage in silvopastures of the Dry Chaco. *Agric Ecosyst Environ* 303: 107117. <https://doi.org/10.1016/j.agee.2020.107117>

21. Fisher MJ, Rao IM, Ayarza MA, Lascano CE, Sanz JI, Thomas RJ, Vera RR (1994) Carbon storage by introduced deep-rooted grasses in the South American savannas. *Nature* 371:236–238. <https://doi.org/10.1038/371236a0>
22. Fujisaka S, Castilla C, Escobar G, Rodrigues V, Veneklaas EJ, Thomas R, Fisher M (1998) The effects of forest conversion on annual crops and pasture: Estimates of carbon emissions and plant species loss in a Brazilian Amazon colony. *Agric Ecosyst Environ* 69:17–26. [https://doi.org/10.1016/S0167-8809\(98\)00091-7](https://doi.org/10.1016/S0167-8809(98)00091-7)
23. Godde CM, de Boer I, Ermgassen EZ, Herrero M, van Middelaar C, Muller A, Rös E, Schader C, Smith P, van Zanten H, Garnett T (2020) Soil carbon sequestration in grazing systems: managing expectations. *Clim Chan*, 161: 385-391. <https://doi.org/10.1007/s10584-020-02673-x>
24. Guillot E, Hinsinger P, Dufour L, Roy J, Bertrand I (2019) With or without trees: resistance and resilience of soil microbial communities to drought and heat stress in a Mediterranean agroforestry system. *Soil Biol Biochem*, 129:122–135. <https://doi.org/10.1016/j.soilbio.2018.11.011>
25. Gupta SR, Dagar JC, Jangra R, Gaur A (2019). Tree-based Systems for Enhancing Environmental Services of Saline Environments. In: Dagar, J., Yadav, R., Sharma, P. (eds) *Research Developments in Saline Agriculture*. Springer, Singapore. https://doi.org/10.1007/978-981-13-5832-6_16
26. Howlett DS, Mosquera-Losada MR, Nair PKR, Nair VD, Rigueiro-Rodriguez A (2011) Soil carbon storage in silvopastoral systems and a treeless pasture in northwestern Spain. *J Environ Qual* 40: 825–832. <https://doi.org/10.2134/jeq2010.0145>.
27. Jobbágy EG, Giménez R, Marchesini V, Diaz Y, Jayawickreme DH, Nosetto MD (2021) Salt Accumulation and Redistribution in the Dry Plains of Southern South America: Lessons from Land Use Changes. In: Taleisnik, E., Lavado, R.S. (Eds) *Saline and Alkaline Soils in Latin America*. Springer, Cham. https://doi.org/10.1007/978-3-030-52592-7_3
28. Kaur B, Gupta SR, Singh G (2002) Bioremediation of a sodic soil by silvopastoral systems in northwestern India. *Agrofor Syst* 54:13–20 (2002). <https://doi.org/10.1023/A:1014221306004>
29. Kuzyakov Y (2002) Review: Factors affecting rhizosphere priming effects. *J Plant Nutr Soil Sci* 165: 382-396. [https://doi.org/10.1002/1522-2624\(200208\)165:4<382::AID-JPLN382>3.0.CO;2-#](https://doi.org/10.1002/1522-2624(200208)165:4<382::AID-JPLN382>3.0.CO;2-#)
30. Levy GI, Shainberg I (2005) Sodic Soils. In: *Encyclopedia of Soils in the Environment*. Elsevier, pp 504-513. <https://doi.org/10.1016/B0-12-348530-4/00218-6>
31. Li Y., Liu X, Xu W, Bongers F, Bao W, Chen B, Chen G, Guo K, Lai J, Lin D, Mi X, Tian X, J Wang, Yan J, Yang B, Zheng Y, Ma K (2020) Effects of diversity, climate and litter on soil organic carbon storage in subtropical forests. *For Ecol Manage* 476: 118479. <https://doi.org/10.1016/j.foreco.2020.118479>.
32. Liang C, Schimel JP, Jastrow JD (2017) The importance of anabolism in microbial control over soil carbon storage. *Nat Microbiol* 2: 17105. <https://doi.org/10.1038/nmicrobiol.2017.105>
33. Magliano PN, Fernández RJ, Florio EL, Murray F, Jobbágy EG (2017) Soil physical changes after conversion of woodlands to pastures in dry Chaco Rangelands (Argentina). *Rangeland Ecol Manage* 70: 225–229. <https://doi.org/10.1016/j.rama.2016.08.003>.

34. Maertens M, De Lannoy GJM, Vincent F, Massart S, Giménez R, Houspanossian J, Gasparri I, Vanacker V (2022) Spatial patterns of soil salinity in the central Argentinean Dry Chaco. *Anthropocene* 37: 100322. <https://doi.org/10.1016/j.ancene.2022.100322>.
35. Nelson DW, Sommers LE (1982) Total carbon, organic carbon and organic matter. *Methods of soil analysis, Part II, Chemical and Microbiological Properties*, 2nd Edition. American Society of Agronomy. Soil Sci Soc Am, pp 539-579
36. R Core Team. 2022. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <https://www.R-project.org/>
37. Ripple WJ, Smith P, Haberl H, Montzka SA, McAlpine C, Boucher DH (2014) Ruminants, climate change and climate policy. *Nat clim change* 4: 2-5. <https://doi.org/10.1038/nclimate2081>
38. Sanford G. 2014. Perennial grasslands are essential for long term SOC storage in the mollisols of the North Central USA. In: *Soil Carbon*. Springer International. Springer, Cham, pp 281-288. https://doi.org/10.1007/978-3-319-04084-4_29
39. Sarkar R, Corriher-Olson V, Long C, Somenahally A (2020) Challenges and Potentials for Soil Organic Carbon Sequestration in Forage and Grazing Systems. *Rangeland Ecol Manage* 73: 786-795. <https://doi.org/10.1016/j.rama.2020.04.002>
40. Shahid SA, Al-Shankiti A (2013) Sustainable food production in marginal lands—Case of GDLA member countries. *Int Soil Water Conser Res* 1: 24-38. [https://doi.org/10.1016/S2095-6339\(15\)30047-2](https://doi.org/10.1016/S2095-6339(15)30047-2)
41. Sileshi GW, Mafongoya PL, Nath AJ (2020) Agroforestry Systems for Improving Nutrient Recycling and Soil Fertility on Degraded Lands. In: Dagar, J.C., Gupta, S.R., Teketay, D. (eds) *Agroforestry for Degraded Landscapes*. Springer, Singapore. https://doi.org/10.1007/978-981-15-4136-0_8
42. Singh G (1995) An agroforestry practice for the development of salt lands using *Prosopis juliflora* and *Leptochloa fusca*. *Agrofor Syst* 29: 61-75. <https://doi.org/10.1007/BF00711282>
43. Soil Survey Staff (2014) *Keys to Soil Taxonomy*. 12th ed. Washington D.C., USA: USDA-Natural Resources Conservation Service.
44. Stevenson A, Zhang Y, Huang J, Hu J, Paustian K, Hartemink AH (2023) Rates of soil organic carbon change in cultivated and afforested sandy soils. *Agric Ecosyst Environ* 360: 108785. <https://doi.org/10.1016/j.agee.2023.108785>.
45. Taleisnik E, López-Launestein D (2011) Leñosas perennes para ambientes afectados por salinidad. Una sinopsis de la contribución argentina a este tema. *Ecología Austral*, 21: 3-14 (http://www.scielo.org.ar/scielo.php?pid=S1667-782X2011000100002&script=sci_arttext&tlng=en)
46. Tonucci RG, Ramachandran Nair PK, Nair VD, Garcia R, Bernardino FS (2011) Soil Carbon Storage in Silvopasture and Related Land-Use Systems in the Brazilian Cerrado. *J Environ Qual* 40:833–841. <https://doi.org/10.2134/jeq2010.0162>
47. Twisk J, Bosman L, Hoekstra T, Rijnhart J, Welten M, Heymans M (2018) Different ways to estimate treatment effects in randomized controlled trials. *Contem Clin Trials Comm*10: 80-85. <https://doi.org/10.1016/j.conctc.2018.03.008>

48. Upson MA, Burgess PJ, Morison JIL (2016) Soil carbon changes after establishing woodland and agroforestry trees in a grazed pasture. *Geoderma*, 283:10–20.
<https://doi.org/10.1016/j.geoderma.2016.07.002>
49. Valenzuela Que FG, Villanueva-López G, Alcudia-Aguilar A, Medrano-Pérez O, Cámara-Cabrales L, Martínez-Zurimendi P, Casanova-Lugo F, Raj Aryal D (2021) Silvopastoral systems improve carbon stocks at livestock ranches in Tabasco, Mexico. *Soil Use Manage* 38:1237–1249.
<https://doi.org/10.1111/sum.12799>
50. Villarino SH, Pinto P, Jackson R, Piñeiro G (2021), Plant rhizodeposition: A key factor for soil organic matter formation in stable fractions. *Sci Adv* 7:eabd3176.<https://doi.org/10.1126/sciadv.abd3176>
51. Viruel E, Fontana CA, Bassi D, Puglisi E, Radrizzani A, Martínez Calsina L, Banegas N, Cocconcilli P (2022) Silvopastoral systems in dry Chaco, Argentina: Impact on soil chemical parameters and bacterial communities. *Soil Use Manage* 2020;00: 1-13_<https://doi.org/10.1111/sum.1265>
52. Wiesmeier M, Steffens M, Mueller CW, Kölbl A, Reszkowska A, Peth S, Horn R, Kögel-Knabner I (2012) Aggregate stability and physical protection of soil organic carbon in semi-arid steppe soils. *Eur J Soil Sci* 63: 22-31. <https://doi.org/10.1111/j.1365-2389.2011.01418.x>
53. Xu S, Silveira M, Inglett KS, Sollenberger LE, Gerber S (2017) Soil microbial community responses to long-term land use intensification in subtropical grazing lands. *Geoderma* 293: 73–81.
<https://doi.org/10.1016/j.geoderma.2017.01.019>
54. Yongfei Bai Y, Cotrufo MF (2022) Grassland soil carbon sequestration: Current understanding, challenges, and solutions. *Science* 377: 603-608.<https://doi.org/10.1126/science.abo2380>
55. Zhu X, Liu W, Chen J, Bruijnzeel LA, Mao Z, Yang X, Cardinael R, Meng F, Sidle RC, Seitz S, Nair VD, Nanko K, Zou X, Chen C, Jiang XJ (2020) Reductions in water, soil and nutrient losses and pesticide pollution in agroforestry practices: a review of evidence and processes. *Plant Soil*, 453: 45–86.
<https://doi.org/10.1007/s11104-019-04377-3>
56. Zimmermann M, Leifeld J, Schmidt MWI, Smith P, Fuhrer J (2007) Measured soil organic matter fractions can be related to pools in the RothC model. *Eur J Soil Sci* 58: 658–667.<https://doi.org/10.1111/j.1365-2389.2006.00855.x>

Table 1

Table 1 is available in the Supplementary Files section.

Figures

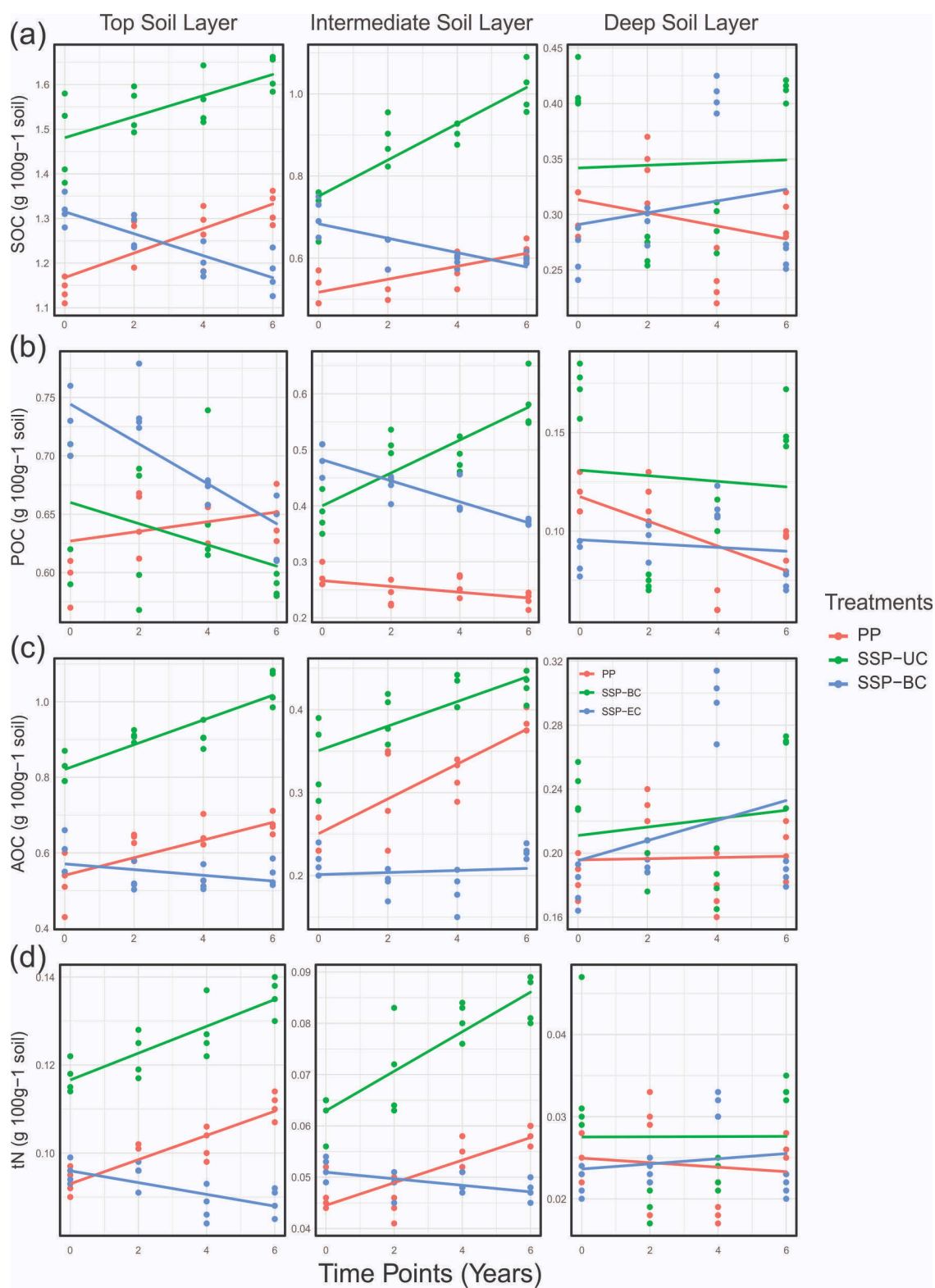


Figure 1

Scatterplots with fixed-effects regression lines. Panels are organized by soil components (rows) and soil depths (columns). Lines and points of identical colors represent corresponding experimental systems. PP: Pure Pasture; SPS-UC: Silvopastoral System Under Tree Canopy; SPS-BC: Silvopastoral System Between Tree Canopies.

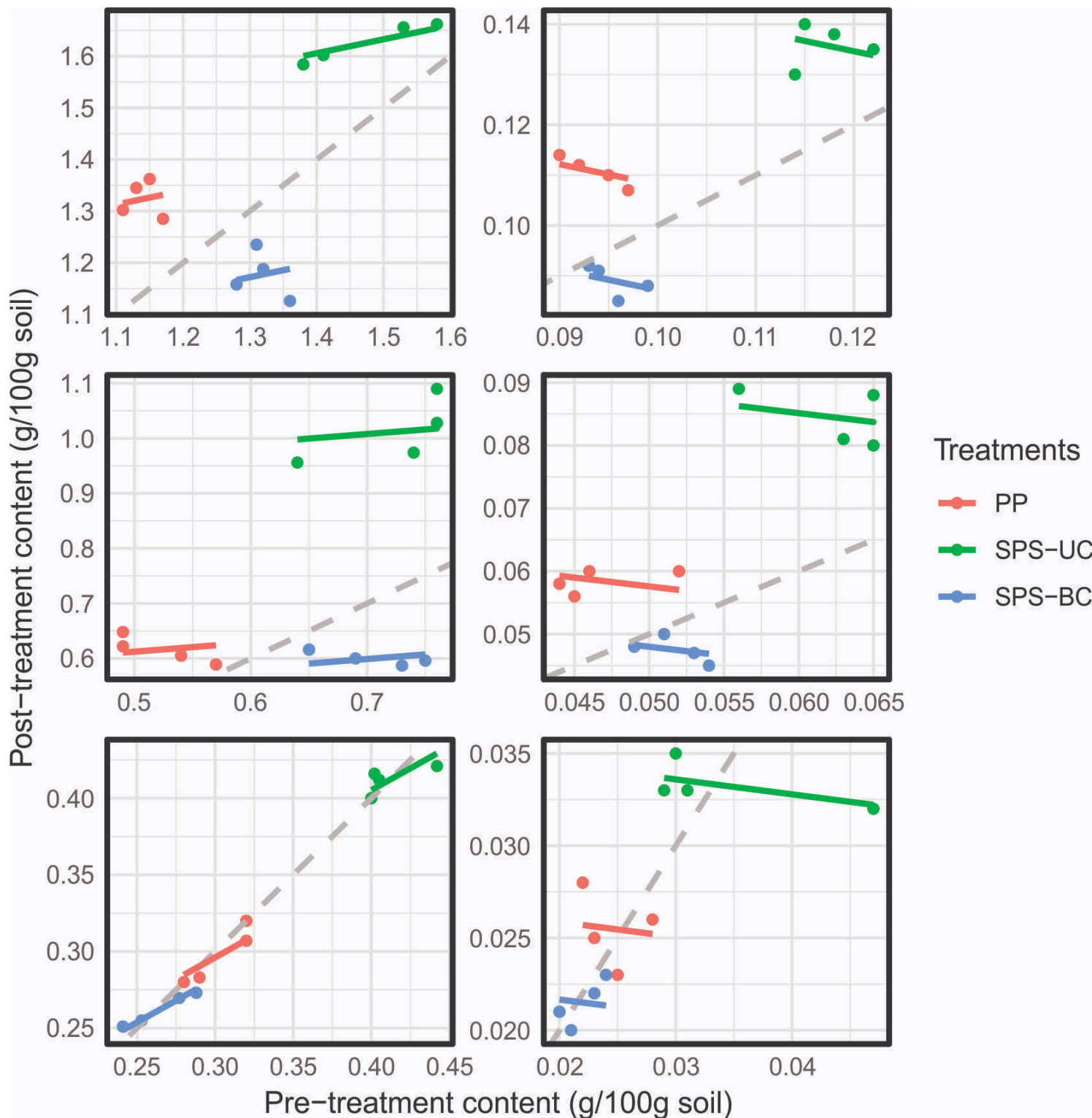


Figure 2

Relationship between pre- and post-treatment concentrations of soil parameters across various soil depths. Outcomes of ANCOVA analysis segmented by soil components (Soil organic carbon to the left, total nitrogen to the right) and the three levels of soil depth. Data points depict actual observations, while lines represent the fits provided by the ANCOVA model. The dashed line indicates the identity line.

Different treatments are distinguished by color. PP: Pure Pasture; SPS-UC: Silvopastoral System Under Tree Canopy; SPS-BC: Silvopastoral System Between Tree Canopies.

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- [Table1MainmanuscriptBanegasetal.xlsx](#)