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Soil sustainability challenges in eucalypt afforestation of floodplain grasslands in Paraguay's Humid Chaco: Carbon and phosphorus dynamics

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A B S T R A C T

The floodplains of the Humid Chaco (HC) ecoregion in Southern Paraguay comprise an area of great environmental value, which contributes to maintaining regional climate stability while supporting a rich biodiversity of flora and fauna. Throughout the region, soils play a fundamental role in C sequestration and water regulation but are strongly influenced by the water table. In recent years, different afforestation plans have led to thousands of hectares of land being covered with eucalyptus plantations, involving intensive drainage and tillage that strongly affect the water table. The intensive afforestation could potentially result in the loss of large amounts of soil organic matter (SOM), influencing nutrient cycling and ecosystem viability in the long term. The present study aimed to evaluate the impact of *Eucalyptus camaldulensis* plantations on natural grasslands in the HC, focusing on SOM quantity and quality and P dynamics. A representative area of the HC, including *E. camaldulensis* plantations of various ages (ranging from 2 to 8 years) and contiguous native grasslands were selected for study. Soil samples (0–5, 5–10, 10–20, 20–30 cm) were obtained to determine total C and N concentrations, and C:N ratio, and then SOM quality (by DSC-TGA, and solid-state ^{13}C CP-MAS NMR spectroscopy) and the different P forms present (by ^{31}P NMR spectroscopy) were analysed in the upper soil layer (0–5 cm). The intensive site preparation prior to plantation establishment initially caused a notable reduction (50 %) in soil organic carbon (SOC) and large increases in the carbon-to-nitrogen (C:N) ratio at all depths studied but especially in upper soil layers. These parameters gradually recovered as the trees grew, although the values did not return to the original levels measured in native grasslands. Thermal analysis and ^{13}C NMR spectroscopy revealed initial loss of labile and recalcitrant SOM compounds, with more noticeable loss of the most labile fractions. More than 40 % of the P in the natural grasslands was present as organic P forms (monoesters), indicating the high potential for P immobilization and preservation. Afforestation led to a substantial loss of soil extractable P, which affected both organic (monoesters and diester) and inorganic (orthophosphate) forms. This could have a significant impact on the short and medium-term P reserves, inducing P limitation and threatening ecosystem sustainability. Considering the short and medium-term impacts observed after the transformation of natural grasslands into *E. camaldulensis* plantations, strategic planning, establishing buffer zones with natural vegetation and applying proper soil management are urgently required to preserve SOM and nutrients in the floodplains of the Humid Chaco.

1. Introduction

Soil organic matter (SOM) is essential in terrestrial ecosystems, as it maintains soil fertility and contributes to global C cycling (Gerke, 2022). The soil in floodplains can be permanently or seasonally saturated with water, as these low-lying areas adjacent to rivers or other water bodies are subject to flooding. Floodplains are among the most productive ecosystems worldwide, providing a wide range of benefits and ecosystem services, such as water storage and purification, filtration of agricultural and urban pollutants, flood mitigation and C sequestration

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(Zedler and Kercher, 2005).

In recent years, artificial drainage systems have been constructed in different parts of the world during the transformation of floodplains into farmland or forest plantations with fast-growing species and the expansion of industrial and urban zones (Schoenholtz et al., 2001; Woziwoda and Kopeć, 2014; Larned et al., 2020). Intensive site preparation can have severe consequences, mainly by lowering the water table and deepening the aerobic zone, facilitating the decomposition of readily degradable compounds that had accumulated under previous anoxic conditions (Mastný et al., 2016). Drainage and site preparation can also greatly increase the emission of different greenhouse gases (GHG), such as CH_4 and CO_2, from the soil (Cooper et al., 2021). The decomposition or mineralization of SOC in floodplains and wetlands has thus garnered increasing interest in recent years (Sihi et al., 2016; Chen et al., 2018).

In terms of chemical quality, the high rate of SOM decomposition that usually occurs in drained ecosystems is related to the high carbohydrate content (readily degradable), which is accumulated during the previous anoxic conditions (Mastný et al., 2016; Busman et al., 2023). Aeration of soil following drainage may trigger a *priming effect* (Kuzyakov, 2010; Guenet et al., 2018), which is generally responsible for accelerating SOM decomposition. Several studies have suggested that the priming effect may play a particularly important role in drained soils (Ding et al., 2023; Li et al., 2023a). The accelerated decomposition can therefore also alter the chemical structure of the remaining SOM, with implications for soil microbial activity (Gregorich et al., 2015). Increased SOM recalcitrance and the subsequent reduced microbial activity in the medium term are commonly observed after drainage of soil (Gao et al., 2014; Mastný et al., 2016).

Phosphorus (P) dynamics, closely linked to SOM, are also essential for the conservation of floodplain soils due to the significant impact on plant communities and freshwater quality. In addition to physic-chemical mechanisms such adsorption/desorption and precipitation/dissolution (also influenced by pH and soil mineralogy), P cycling is regulated by microbial mineralization and immobilization processes (Lehmann and Kleber, 2015; Hawkins et al., 2022). It is generally considered that the high SOM contents and associated microbial activity in floodplains favour microbial P immobilization. Soluble inorganic P species (primary orthophosphate, $H_2PO_4^{-4}$) are transformed into organic P species (monoesters and diester), which serve as dynamic reserves that ensure the long-term P supply necessary for plant growth (Condron et al., 2005). Phosphorus immobilization process is complex and dynamic, governed by the P demand of plants and microbes, and influenced by environmental conditions (climate, nutrient availability) (Augusto et al., 2017). This mechanism is of particular importance in floodplains as it helps to prevent or reduce the amount of P entering floodwaters, thereby mitigating the risk of eutrophication and hypoxia in aquatic systems (Arenberg et al., 2020). Various studies across different ecosystems have shown that P dynamics are disrupted by deforestation, intensive agriculture (Stutter et al., 2015; Souza-Alonso et al., 2025) and other types of disturbance (Turrión et al., 2010; Merino et al., 2019), all of which contribute to SOM loss.

The Humid Chaco ecoregion, a subregion of the Gran Chaco, occupies an area of 500,000 km^2 across Argentina, Paraguay and Bolivia. This ecoregion is also a unique area of high biodiversity due to the high temperatures and water availability, the inaccessibility of the area and the fact that it acts as a biological corridor between biomes (Mereles et al., 2019). In the floodplains of the Humid Chaco, temporary anaerobic conditions and soil acidity promote medium-to-high SOC concentrations which are higher than in arid savannas (Médanos and Cerrado ecoregion) but lower than in the highly productive, humid-tall dense forest typical from the Alto Paraná and Amambay ecoregions in Paraguay (Encina-Rojas et al., 2023).

Despite the important environmental functions (biodiversity, underground water reserve, C sink) of the seminatural grasslands of the Humid Chaco, these systems are currently undergoing a profound transformation (Varela and Cirignoli, 2018). The region has experienced continuous and severe pulses of deforestation, agricultural expansion and habitat fragmentation in the last few decades (Grau et al., 2015; Henderson et al., 2021). Moreover, the conversion of thousands of hectares of grasslands into *Eucalyptus camaldulensis* plantations (estimated rate of 15,000 ha yr^{-1} since 2015), aimed at producing wood for energy purposes, is currently projected in the Humid Chaco (INFONA, 2023a, 2023b), with few regulatory policies including conservation actions (Szulecka and Zalazar, 2017). This trend in land conversion is also observed throughout the region, in neighbouring countries such as Uruguay and Argentina (Jobbágy et al., 2022).

In addition to the direct impact on flora and fauna, afforestation with water-demanding species such as eucalypts can significantly affect soil compaction (Certi et al., 2015, Tassinari et al., 2019) or alter the hydrology of the entire system, posing a risk to the water balance (Christina et al., 2017; Farley et al., 2005). In a global analysis, Farley et al. (2005) pointed out that grassland afforestation with eucalypts led to remarkable streamflow and run-off losses of up to 70 %, which were also maintained in the long term. Souza-Mattos et al. (2019) estimated a 100 mm/yr decline in groundwater levels following the conversion of Brazilian pastureland to eucalypt plantations. Substituting native grasslands by *Eucalyptus grandis* and *Pinus pinaster* plantations in Uruguay, also lowered soil moisture retention capacity in forested areas (González-Sosa et al., 2024)

Site preparation for establishing *E. camaldulensis* plantations involves drainage and ploughing, potentially lowering the water table level and increasing soil aeration, thereby facilitating deeper penetration of roots (Kleinman et al., 2015; Muñoz et al., 2024). A global meta-analysis (Berthrong et al., 2009) indicated that afforestation with eucalyptus plantations leads to soil acidification and significant decreases in exchangeable cations (Ca, K, Mg). Previous reports have confirmed that the already established *E. camaldulensis* plantations cause soil acidification, SOM decline, depletion of nutrient reserves (mainly N) and exchangeable cations (P, K and Ca) together with increased bulk density (Villalba-Martínez et al., 2024). These changes pose a significant risk of altering the ecosystem functionality and even the future sustainability of the plantations.

However, the effect of transforming humid grassland of the Humid Chaco areas into *E. camaldulensis* plantations can be responsible for changes on the quality of SOM and P. On the basis of previous findings, we hypothesize that the new soil conditions shall be accompanied by notable alterations in SOM quality, increasing the proportion of labile C forms and reducing more stable, recalcitrant components. Additionally, considering P biogeochemistry in relation to SOM and C dynamics, we also hypothesized that changes in SOM quantity and quality will affect the different P forms present in the soil. In this respect, soil removal and drainage may negatively affect the stability of organic P forms, reverse P immobilization and increasing the proportion of inorganic P that could be eventually loss. Given the rapid conversion of floodplain grasslands and its potential consequences due the important role of these ecosystems, this study evaluates the ecological consequences of *E. camaldulensis* afforestation by quantifying its impacts on two critical soil stability indicators: (1) SOM, regarding C storage and SOM quality) and (2) phosphorus cycling. Through a chronosequence analysis (2–8 year plantations vs. native grasslands), we discuss if the current practices can maintain soil functionality and the potential long-term degradation risks after afforestation of the Paraguay's Humid Chaco floodplains.

2. Material and methods

2.1. Experimental location and general characteristics of the study area

This study was conducted in the eastern region of Paraguay (Fig. 1), specifically in the Department of Caaguazú, located within the Humid Chaco ecoregion. The study area is also characterized by a subtropical

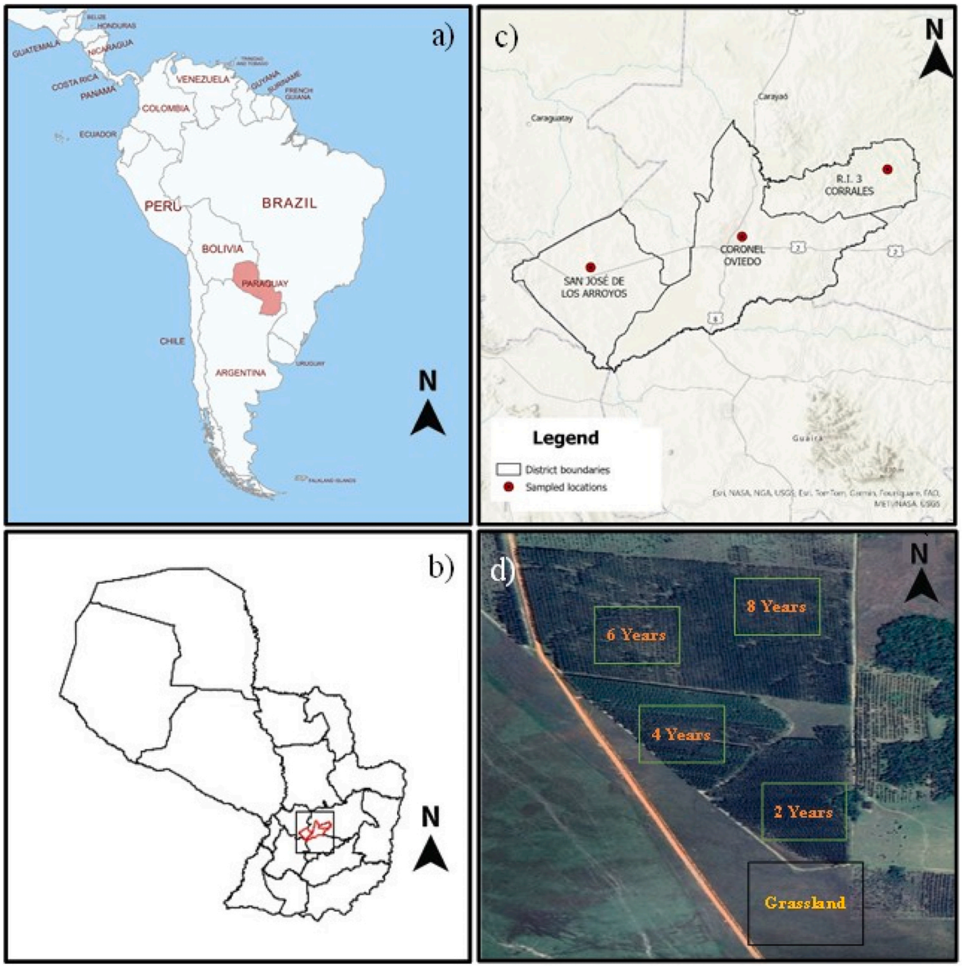

Fig. 1. a) Map of the South American continent indicating the location of Paraguay (in red), b) map of Paraguay showing the different provinces and the study area (black square, area in red), c) Location of sampling sites including *Eucalyptus camaldulensis* plantations (red points) and d) detail of one of the study sites (Coronel Oviedo) showing the experimental design.

climate, according to the Köppen climate classification, with annual precipitation reaching 1600 mm and an average temperature of 26 °C. In the area, the Paraguay River and its tributaries create a network of floodplains and wetlands that are essential for recharge of the Guaraní Aquifer (Ávila-Torres et al., 2018), one of the worlds largest freshwater reserves (Araujo et al., 1999). The Humid Chaco includes a mosaic of different ecosystems such as grasslands, savannas, wetlands and forests, with vast areas of permanently or seasonally water-saturated soils.

The region is characterized by extensive floodplains, harbouring numerous wetlands and lakes formed on alluvial deposits or within valleys. The vegetation primarily consists of seminatural grasslands, dominated by grasses such as *Paspalum notatum* and *Andropogon lateralis* (Ávila-Torres et al., 2018). A more detailed description of the plant communities is available from a study conducted in the northern Iberá wetlands, a nearby area located in the Corrientes province, Argentina (Corriale et al., 2013). In terms of management, vegetation growth is controlled through low-pressure grazing and low-intensity burning during autumn and winter. The moisture conditions in these floodplains are defined by annual cycles of flooding and drying, including a distinct dry season. The floodplain soils are thus affected by the high-water table under temporary anaerobic conditions and are mineral-based, with no peat formation. Fluvisols and Ultisols with gleyic properties (IUSS, 2015) predominate throughout the region. Due to the high precipitation and the influence of the shallow groundwater, these soils are therefore saturated throughout most of the year, especially in the months with high precipitation (August to December). These soils are characterized by slow drainage, high SOM concentrations, high acidity (pH = 4–5), low CICe (usually $\leq$ 4 $cmol_c\ kg^{-1}$) and a sandy texture.

In recent years, different afforestation plans have led to thousands of hectares of land being covered with eucalyptus plantations for energy purposes, involving intensive drainage and tillage. In terms of site preparation, the steps followed for the establishment of eucalyptus plantations are generally common. Thus, soil labour is carried out by first draining the plots followed by intensive tillage (Figure S1). Mouldboard ploughing is generally conducted to a depth of 0.30 m and subsoiling to a depth of 0.40 m, to improve rooting and accelerate drainage. Individual *E. camaldulensis* seedlings were then planted during the months of high rainfall in ridges of 0.60 m in height, with a density of 4 m between rows and 4 m between plants with a total of 625 plants per ha on average. During planting, a base fertilization of 60 kg ha^{-1} of P_2O_5 and a top dressing of 100 kg ha^{-1} of urea (45 %) were applied to each individual plants.

2.2. Plot selection

For our purposes, the experimental design was replicated in 3 different sites with *E. camaldulensis* plantations: San José de los Arroyos (25°36'4.31"S; 56°40'24.31"O), Coronel Oviedo (25°35'3.82"S; 56°30'15.36"O) and Corrales (25°24'15.37"S; 56°21'31.23"O). Each site contained one control plot (native grassland) and 3–4 treatment plots (*Eucalyptus* plantations of different ages). In each site, different plots of variable size, between 20 and 100 ha in size, were selected on the basis of the age of plantation: 2, 4, 6 and 8 years old stablished on seminatural grasslands plot. The large plot sizes reflected real-world plantation management units. Seminatural grasslands (without afforestation) contiguous (< 50 m apart) to *E. camaldulensis* plantations were also included to serve as control plots. The maximum growing time (8 yr) for *E. camaldulensis* was established on the basis of the average harvesting age for this species in this region.

In each plot, soil samples were collected within three 15 m × 20 m grids that were randomly placed in homogenous areas (verified by soil texture, pH, and baseline SOM measurements), separated by more than 100 m, and away from the margins of the plot to avoid a potential edge effect. This design ensured that each grid represented replicate sampling within a single age class or control. The three sites were selected based on rigorous criteria to ensure comparability: 1) Each study site was separated by a distance of less than 30 km, 2) all sites shared an homogenous landscape with identical topography (floodplain grasslands), 3) with same soil type (Fluvisols and Ultisols with gleyic properties), 4) uniform management history (low-intensity grazing and seasonal burning prior to afforestation), and 5) standardized *Eucalyptus camaldulensis* plantation site preparation practices (drainage, tillage, fertilization, and planting density). All these parameters were verified through field surveys and land-use records, which ensured that the chronosequence approach reflects temporal changes in soil properties. Randomization was applied within each site's plantation plot to mitigate spatial bias, aligning with the statistical assumptions of our model analysis.

2.3. Sampling design and processing

Soil sampling was carried out during the autumn of April 2021. Soil was collected using a hand auger at three different places within each 15 × 20 grid (> 100 apart) in homogeneous areas to reduce experimental error. The soil was collected in the centre of the planting rows in all cases, in the space between four *E. camaldulensis* trees, in areas with evident homogeneity and avoiding the presence of rocks or other natural artifacts. Leaf layer (if present) and organic debris were gently removed and the topsoil (0–30 cm) was sampled for chemical and bulk density determinations. At each sample plot, we collected 9 soil cores (3 cores x 3 random points) per depth interval (0–5, 5–10, 10–20 and 20–30 cm). Then, 3 composite samples, each comprising three subsamples, were obtained for each soil depth per grid.

Once in the laboratory, composite soil samples were mixed carefully, dried at room temperature and sieved (2 mm). Fine plant root and organic residues were removed by hand from the sieved soil samples. Each composite sample was analyzed for SOC, N (all depths, n = 180). Thermal analysis (DSC-TGA, n = 60) and NMR analysis (n = 5) were carried out only in the most superficial fraction (0–5 cm). For NMR analyses, a mixture of the three replicates generated in each plot was used.

2.4. Differential scanning calorimetry (DSC) and thermogravimetric analysis (TGA)

All of the soil samples were analyzed by differential scanning calorimetry and thermogravimetry (using a TGA-DSC thermogravimetric analyzer, Mettler Toledo). Air-dried soil samples (4 mg) from the plantations of different ages were placed in aluminium pans under dry air (under O_2 flux; flow rate, 50 mL min^{-1}) and scanned at a rate of 10 °C min^{-1}. The temperature ranged between 50 and 600 °C. The heat of combustion (Q, in J per gram) was determined by integrating the DSC curves (in Wg^{-1}) over the exothermic region (150–600 °C). The SOM concentration was calculated as the difference in weight of the soil burned at 150 and 550 °C. The thermogravimetic curves were obtained by the first derivative of TG curves (DTG) using the STARe Evaluation Software (Mettler Toledo, v17.0). DSC curves were analyzed by dividing the area under the curve into three main groups (W_1, W_2, and W_3) representing different degrees of resistance to thermal oxidation (Merino et al., 2014, 2016): 200–375 °C (W_1, considered as labile organic matter and mainly formed by carbohydrates and other aliphatic compounds); 375–475 °C (W_2, recalcitrant organic matter, including lignin or other polyphenols); and 475–550 °C (highly recalcitrant organic matter that included polycondensed aromatic forms). The resulting partial heats of combustion were designated Q_1, Q_2 and Q_3. The T50w, i.e. the temperature at which 50 % mass loss of the SOM was released, was also determined.

2.5. Solid state ^{13}C CP-MAS NMR spectroscopy

Soil samples from the plantations of different ages were analyzed by solid-state ^{13}C NMR spectroscopy. The samples were demineralized (5–7 times) with constant shaking in 10 % (w/w) hydrofluoric acid (HF) for 2 h each time. The spectrometer (Agilent Varian VNMRS-500-WB) was operated at proton resonance frequency of 500 MHz, and the zirconia rotor volume was 160 µL. Carbon chemical shifts were referenced to the carbon methylene signal of solid adamantane at 28.92 ppm. Cross-Polarization Magic Angle Spinning (1D CPMAS) analysis of the soil samples was carried out under the following conditions: contact time, 1 ms; inter-scan delay, 1 s (a proton T1 experiment was performed to check the suitability of this time); and MAS rate, 12 kHz. The number of scans was ca. 10,000–35,000. The cross-polarization time was set at 1 ms. For integration, each spectrum was divided into four regions representing different chemical environments of a 13 C nucleus: alkyl C (0–45 ppm), O-alkyl C (45–110 ppm), aromatic C (110–160 ppm) and carbonyl C (160–210 ppm). The NMR spectra were processed with MestreNova software 8.1.0 (Mestrelab Research Inc., Santiago de Compostela, Spain) to quantify the area under the signals.

2.6. Liquid state ^{31}P NMR spectroscopy

Phosphorus fractions were extracted using the method described by Cheesman et al. (2010) and modified by Noack et al. (2012). Briefly, 1 g of each soil sample was extracted with 30 mL of 0.25 M NaOH and 50 mM EDTA for 16 h. The extracts were then centrifuged for 10 min at 6500 rpm. One mL of 50 mg L^{-1} methylene diphosphonic acid (MDPA) solution was added as an internal standard to 20 mL of each extract, frozen at −80 °C, and lyophilized. Aliquots of 300 mg of each lyophilized extract were then redissolved in 0.3 mL of deuterium oxide (D_2O) and 2.7 mL of a solution containing 1.0 M NaOH and 0.1 M EDTA. The dissolved extracts were then placed in a 5 mm NMR tube for analysis. Spectra were acquired at 25 °C in a Varian VNMRS-500-WB NMR spectrometer at a ^{31}P frequency of 202.296 MHz. The recovery delay (0.5 s) was established to optimize and reduce the time of experiments, and the other conditions were as follows: 90° pulse, 6 µs; acquisition time, 0.2 s; and broadband 1 H decoupling. For each sample, 100,000 scans were acquired. The spectra obtained had a line broadening of 2 Hz. Spectroscopic signals were assigned to the different P compounds following Newman and Tate (1980) and Turner et al. (2003): orthophosphate (around 5.3 ppm), orthophosphate monoesters (3–5.1 ppm), pyrophosphates (-5.5 ppm) and orthophosphate diesters (-2–0 ppm). The MDPA internal standard appeared at 16.5 ppm. Peak signal areas in the ^{31}P NMR spectra were distinguished by integration of spectra. The spectra were processed with MestreNova software, version 8.1.0 (Mestrelab Research Inc., Santiago de Compostela, Spain).

2.7. Statistical analysis

For the analysis of soil parameters (C, N and C:N) a linear mixed model (LMM) was used, using as variables the *E. camaldulensis* planting year (5 levels= meadows, 2, 4, 6 and 8 years of planting), soil depth (4 levels= 0–5, 5–10, 10–20, 20–30 cm), as fixed effects, and plot nested within site as random effects. Data were log-transformed when necessary to meet normality assumptions (confirmed by Kolmogorov-Smirnov tests) and homoscedasticity (Levene's test).

Differences between the different plantation ages in relation to thermogravimetric parameters (W_1, W_2, W_3, T50w) were explored using one-way analysis of variance (ANOVA). When significant differences ($P < 0.05$) were detected in LMM or ANOVA, Tukey's HSD was used as post-hoc test for pairwise comparisons. The data obtained by integration of the areas of interest for each ^{13}C NMR or ^{31}P NMR spectra (0–5 cm layer only, pooled by treatment within sites) were analyzed through hierarchical clustering using Euclidean distance as the construct measure. In addition, relationships between C-compounds, P-compounds and the different physic-chemical variables in seminatural grasslands and afforested areas were also evaluated using principal component analysis (PCA) to simplify the interpretation of the results. Varimax rotation was selected, and KMO and sphericity were tested using Bartlett test. All statistical tests were performed using IBM SPSS Statistics version 23.0 (IBM SPSS Inc., Chicago, IL, USA) software for Windows.

3. Results

3.1. Changes in soil organic carbon (SOC), N and C:N in afforested grasslands

The effects *of E. camaldulensis* plantation age, soil depth, and their interaction (time*depth) on soil organic carbon (SOC), total Nitrogen (N) and C:N in soil was analyzed using linear mixed model (LMM) (Table 1). Thus, a significant decrease in SOC occurred immediately after the establishment of *E. camaldulensis* plantations (Fig. 2); thus, within the first 2 years after plantation establishment, the SOC concentration decreased significantly, by more than 50 %, from 3.7 % in the seminatural grasslands to 1.7 % in young (2 years) *E. camaldulensis* plantations. This effect was observed at all depths, but the magnitude of the effect decreases with depth. After this initial decrease, the SOC level gradually recovered, increasing up to 2.5 % 8 years after plantation establishment. Despite the gradual recovery, by the end of the 8-yr rotation period, the SOC values were still significantly lower than observed in seminatural grasslands. The total N concentration in grasslands was 0.6 %. After the establishment of *E. camaldulensis* plantations, suffered a significant decrease (83 %), from 0.6 % in seminatural grasslands to 0.1 % in recently transformed plantations. As in the case of SOC, reduction in N concentration was observed at all depths, also decreasing in magnitude with depth. Unlike SOC, the total N progressively accumulated but showed no signs of recovery throughout the 8-year rotation period, increasing only up to 0.16 % in the mature plantation (8 years), which still reflects a decrease of 80 % relative to N concentration found in seminatural grasslands. As a result of changes observed in SOC and total N, the C:N ratio increased greatly, from C:N = 6 in the grassland to C:N = 25 immediately after site preparation. The C:N ratios in older forest age classes ranked below those of younger forest age classes, though differences were not significant (Fig. 2).

Table 1
Model output for soil organic carbon (SOC), total Nitrogen (N) and C/N in soil, evaluating the effects *of E. camaldulensis* plantation age, soil depth, and their interaction (time*depth) using linear mixed model (LMM).

Variable	Factor	g.l	F value	Sign. (*P* value)
	Age	4,56	100.38	< 0.001
SOC	Depth	3,56	79.97	< 0.001
	Age*depth	12,56	4.97	< 0.001
	Age	4,56	362.07	< 0.001
N	Depth	3,56	45.92	< 0.001
	Age*depth	12,56	35.65	< 0.001
	Age	4,56	1037.10	< 0.001
C/N	Depth	3,56	9.42	< 0.001
	Age*depth	12,56	9.72	< 0.001

3.2. Effects of afforestation on SOM biochemistry quality

3.2.1. Differential scanning calorimetry (DSC) and thermogravimetric analysis (TGA)

The derivative thermogravimetric (DTG) curves (0–5 cm) of the native grassland soils (Fig. 3) revealed two exothermic peaks at 290–360 and 410–500 °C, with a dominance of compounds that can be considered to be within the labile fraction (< 375 ºC). In these soils, the most labile SOM fraction (W_1, weight lost at <375 °C, Fig. 3) made up > 52.9 % of the total SOM, whereas the recalcitrant fraction (W_2 and W_3 weight lost at >375 °C, Fig. 3) represented 47.1 % of the total. In these soils, the T50w values were low (342 °C) (Fig. 4**a**), reflecting a high proportion of SOM labile compounds. After establishment of the *E. camaldulensis* plantations (2 yr), a decrease in the total area (indicating a lower SOM concentration) and a "flattening" of the two exothermic peaks (Fig. 3) were observed, suggesting a large loss of both labile and recalcitrant OM compounds (>50 %, similar to observed in SOC concentration) and an increase in the T50w (356 ºC). Nevertheless, SOM fractions remained in similar proportions (W_1= 56 % and W_2+W_3 = 44 %; Fig. 4**b**). The highest t50w value (373 ºC) was not reached until 4 years after plantation establishment. The thermogravimetric curves corresponding to soils of older plantations (6, and 8 years), showed two distinct clear peaks around 270 ºC (labile OM) and 475 ºC (recalcitrant SOM), more marked than the peaks in the spectra of native grassland soils. This increase reflects the gradual recovery of SOM. However, both peaks were narrower, constrained within regions of narrower temperature, probably suggesting that the SOM entering the system was simplified (only *E. camaldulensis* litter is deposited through this period).

3.2.2. Solid state ^{13}C CP-MAS NMR spectroscopy

The distribution of C-compounds in grassland soils indicated that O-alkyl compounds were the dominant C- group (46 %), followed by alkyl-C and aromatic-C compounds, which were present in similar proportions (25 % and 22 %, respectively) (Table 2**,** Fig. 5**a**). The proportion of carboxyl-C was lowest (7 %). Dendrogram comparing the different afforestation years based on ^{13}C NMR spectral signals clearly separated 2-year plots from the main group of treatments (Fig. 6**b**). The PCA analysis provided 2 main principal components (PC1 and PC2) that explained 81.78 % of the total variance (Fig. 6**c**). Thus, PC1 clearly separated grasslands from the plots transformed in *E. camaldulensis* plantations, with a remarkable positive association with main physic-chemical parameters such as SOC, N or P. Then, PC2 showed a clear separation between 2-years plots with positive values associated with more recalcitrant C groups as carboxyl-C and aromatic-C. Coinciding with the important loss of SOM after site preparation, the soil samples obtained in the 2-year-old plantation underwent slight decreases in thermolabile C groups (alkyl, O-alkyl), while more recalcitrant groups (aromatic-C and carboxyl-C) increased slightly. Immediately after plantation establishment, alterations in the O-alkyl and aromatic-C fractions led to temporal increases in aliphaticity (alkyl-C/O-alkyl) and aromaticity (Aromatic-C/ alkyl-C + O-alkyl + Aromatic-C). After the initial decrease, the different C fractions returned to proportions similar to those found in seminatural grasslands throughout development of the plantation.

3.3. Extractable P and P forms determined by ^{31} P NMR spectrometry

The initial concentration of extractable P in the seminatural grasslands (164 mg kg^{-1}) underwent a notable decrease following site preparation and afforestation (Table 3**,** Fig. 5**b**). During the 8-yr of the forest rotation, more than 70 % of the initial P concentration was lost. The highest levels of inorganic P (orthophosphate, 54 %) and organic P (45 %, mainly monoesters) were found in the grasslands. Afforestation led to a large decrease in both inorganic and organic P forms. Although both types of P underwent large decreases throughout the rotation, the decrease in the proportion of orthophosphate was remarkable after 2

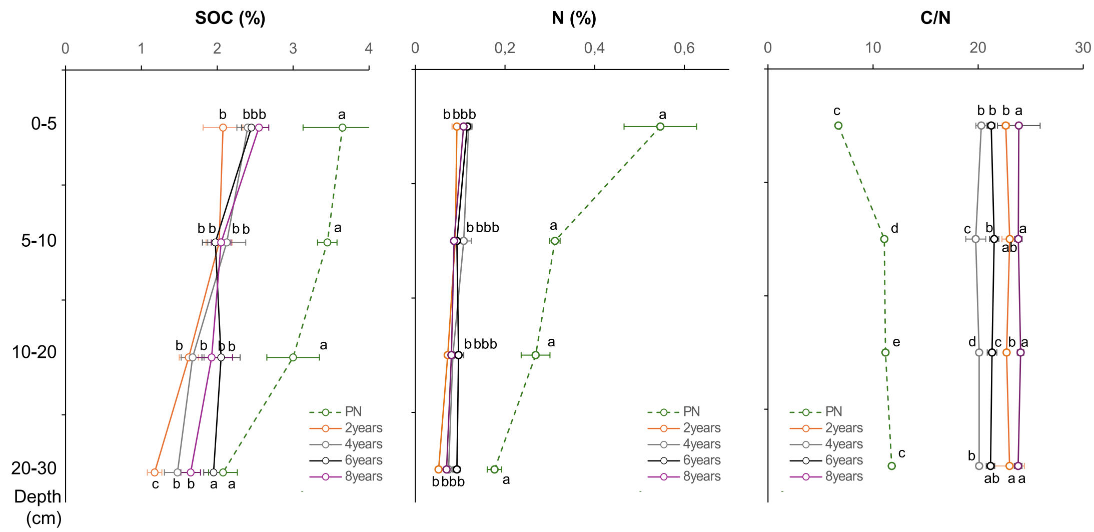

Fig. 2. Mean (±SD) of soil organic carbon (SOC), total nitrogen (N) and C/N ratio in soil samples (0–5, 5–10, 10–20 and 20–30 cm) under natural grasslands and *E. camaldulensis* plantations. Different letters indicate significant differences at $P < 0.05$ (post hoc Tukey's HSD test).

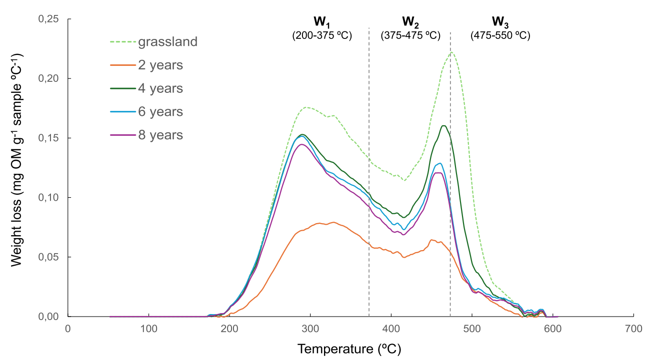

Fig. 3. Derivative thermogravimetric (TG) curves for soil samples (0–5 cm) under semi-natural grasslands and *E. camaldulensis* plantations of different ages (2, 4, 6, and 8 years). W_1, W_2, and W_3 represent the proportions of SOM weight loss at < 375 °C (labile), 375–475 °C (recalcitrant), and 475–550 °C (highly recalcitrant C). TG curves were constructed on the basis of three replicates.

years (46 %) and also at the end of the rotation (63 %) (Table 3). The monoester: diester ratio decreased gradually over the years of planting.

In this case, dendrogram comparing the different afforestation years based on ^{31}P NMR spectral signals clearly separated grasslands from the main group of afforested areas (Fig. 7). AS in the case of ^{13}C-RMN, the two main principal components (PC1 and PC2) explained a representative 91.66 % of the total variance (Fig. 7). Again, PC1 clearly separated grasslands from the plots transformed in *E. camaldulensis* plantations, with a remarkable positive association with the same physic-chemical parameters (SOC, N or P) but also with orthophosphate. Then, PC2 showed a gradual separation between treatments.

4. Discussion

The findings of our study demonstrate that the large-scale conversion of floodplain grasslands to Eucalypt plantations in the Humid Chaco affect negatively to soil properties, important for the ecosystem sustainability. In a rapid vision, our analysis reveals three key patterns in soil responses after grassland afforestation. The afforestation of native grasslands to *E. camaldulensis* plantations led to immediate and substantial declines in key parameters for soil conservation: young

Table 2
The proportion (%) of the different C-compounds identified using ^{13}C NMR in natural grasslands and *E. camaldulensis* plantations.

	Grassland	2 years	4 years	6 years	8 years
Alkyl (0–45 ppm)	25	24	23	24	26
O-alkyl (45–110 ppm)	46	39	49	48	48
Aromatic-C (110–160 ppm)	22	28	20	20	19
Carboxyl-C (160–210 pm)	7	9	8	8	7
Aliphaticity (alkyl-C/O-alkyl)	0.54	0.61	0.47	0.5	0.54
Aromaticity (Aromatic-C/ alkyl-C + O-alkyl + Aromatic-C)	0.24	0.31	0.22	0.22	0.19

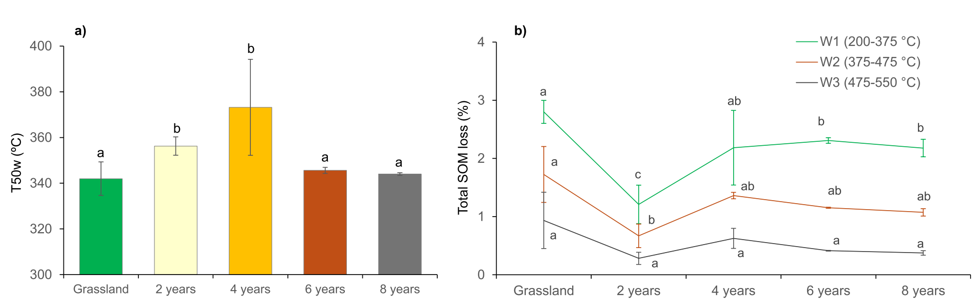

Fig. 4. Means (±SD) of a) T50w values and b) proportion (%) of soil organic matter (SOM) weight loss in each group of resistance to thermal oxidation (W_1, W_2, W_3) in soils samples (0–5 cm) under natural grasslands and *E. camaldulensis* plantations of different ages. W_1, W_2 and W_3 represent the proportions of SOM weight loss at < 375 °C (labile), 375–475 °C (recalcitrant) and 475–550 °C (highly recalcitrant C). Different letters indicate significant differences at $P < 0.05$ using ANOVA and Tukey's HSD as the post-hoc test.

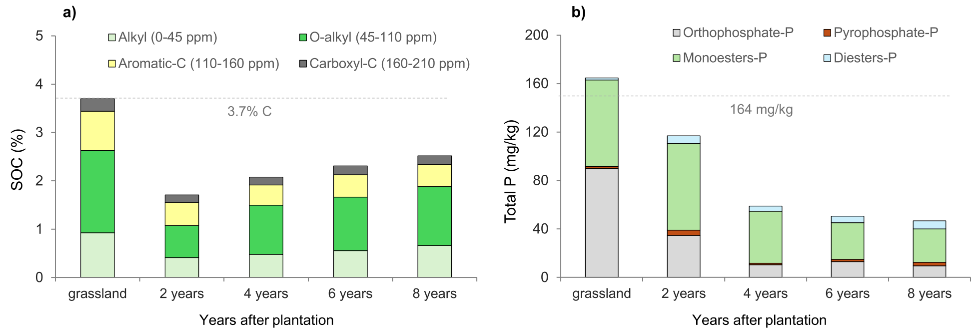

Fig. 5. a) Proportions of the different groups of C-compounds (Alkil-C, O-alkyl-C, Aromatic-C, Carboxyl-C) in each plantation age with respect to the SOC (%) and b) percentage of the P-compounds (orthophosphate, pyrophosphate, monoesters-P and diester-P) in each plantation age with respect to total P.

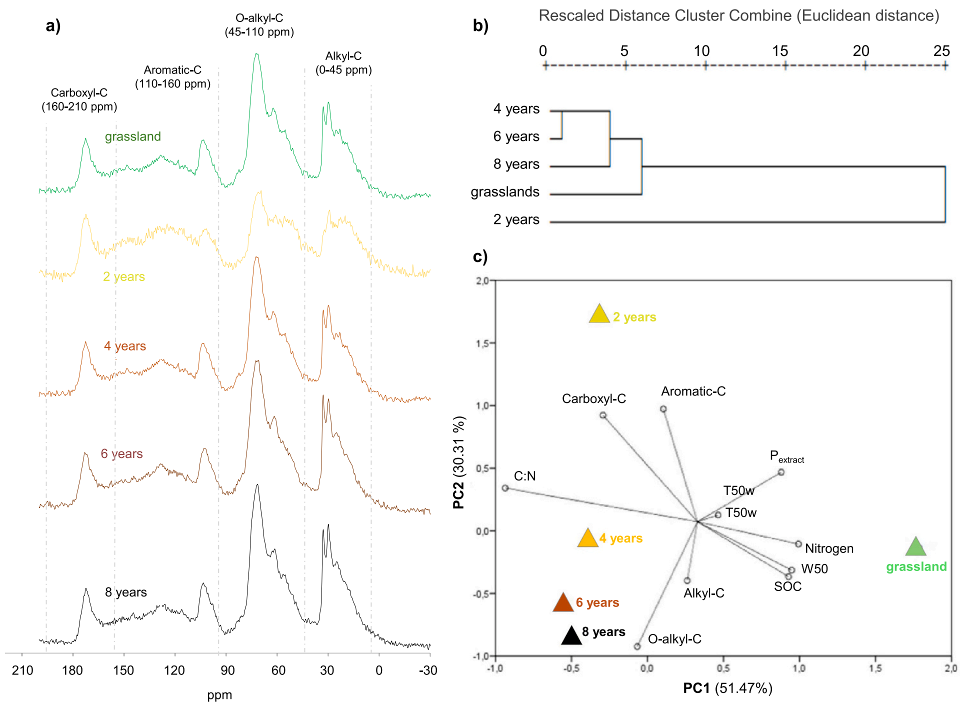

Fig. 6. a) Solid-state ^{13}C CP NMR spectra of the uppermost soil layer (0–5 cm) of the natural grassland and different years of *E. camaldulensis* afforestation; b) Dendrogram of different afforestation years based on ^{13}C NMR spectral data using average linkage (between groups) and Euclidean distance (ED); c) PCA biplot of the spectral data together with physico-chemical parameters analyzed.

Table 3
The proportion of the different P-forms in the uppermost mineral soil (0–5 cm) identified using ^{31}P NMR in natural grasslands and *E. camaldulensis* plantations established on natural grassland.

	Grassland	2 years	4 years	6 years	8 years
Orthophosphate-P (%)	54.5	29.6	17.71	25.76	20.54
Monoesters-P (%)	43.4	61.1	72.92	59.85	58.93
Diester-P (%)	1.01	5.6	7.29	10.61	14.29
Pyrophosphate-P (%)	1.01	3.7	2.08	3.79	6.25
Monoesters:Diester	43.0	10.9	10.0	5.6	4.1
Po:Pi	0.8	2	4.05	2.38	2.73
Extractable P (mg kg^{-1})	164.7	117.0	58.9	50.5	46.6
C:P	149	117	49	55	78

plantations (2-year-old) showed 50–83 % reductions in soil SOC and total N, along with 70 % loss of extractable P relative to controls (grasslands). Although these parameters exhibited gradual recovery with plantation age, values remained significantly below grassland levels even at the end of rotation (8 years), with SOC and N reaching only 68 % and 27 % of original values, respectively. Thermal and NMR analyses showed parallel trends. Initial SOM losses affected both labile (W_1, O-alkyl C) and recalcitrant (W_3, aromatic C) fractions, followed by partial recovery, likely dominated by tree-derived inputs. With respect P, the forms shifted from the dominance in organic-monoester in grasslands to depleted pools in all forms. This suggests long-term P limitation risks for forest sustainability and grasslands. In this section we will elucidate the key biogeochemistry mechanisms driving these impacts, identify how current management practices exacerbate degradation in soil parameters, and finally propose targeted interventions to mitigate these effects.

4.1. SOM and C:N ratios following afforestation of seminatural grasslands in floodplain soils

Before establishment of the *E. camaldulensis* plantations, the SOC concentrations in the native grasslands ranged from 2 % to 5.3 %, values that are much higher than those determined in upland soils in the region (Villalba-Martínez et al., 2024). In this respect, the organic input provided by root litter highlights the SOC sequestration potential of these

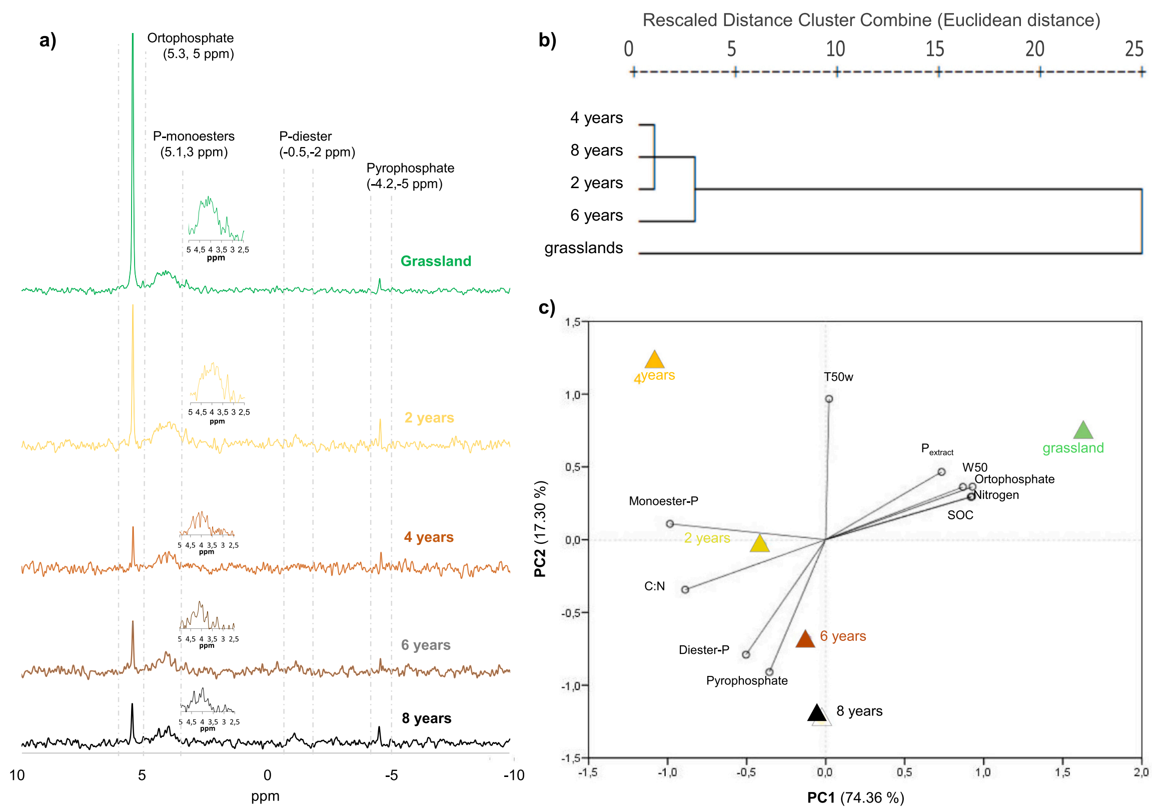

Fig. 7. a) Liquid-state ^{31}P NMR spectra of the uppermost soil layer (0–5 cm) of the natural grassland and different years of *E. camaldulensis* afforestation; b) Dendrogram of different afforestation years based on ^{31}P NMR spectral data using average linkage (between groups) and Euclidean distance (ED); c) PCA biplot of the spectral data together with physico-chemical parameters analyzed.

highly valuable ecosystems, making the adoption of proper measures essential to its conservation (Bai and Cotrufo, 2022). However, the high SOC concentration of these floodplains is mainly due to inhibition of SOM mineralization, as a consequence of the soil being periodically affected by anaerobic conditions. Following afforestation, the C and SOM concentration decreased significantly especially in the uppermost soil layer (0–5 cm and 5–10 cm), by 50 % within the first 2 years. Although the SOM increased gradually, a net loss of 37 % in SOM concentration occurred by the end of this first rotation (after 8 years). The observed SOM loss is significantly higher than some estimates (10–20 %) for natural soils under afforestation across a wide range of environments and soil types (Guo and Gifford, 2002; Cai et al., 2025). However, is also worth mentioning that although SOC losses are commonly reported after conversion of native vegetation to *Eucalyptus* spp. plantations, Fialho and Zinn (2014) indicted that gains and losses are equally distributed and, therefore, a general effect can be difficult to predict. The large loss of SOM due to mineralization may be due to several factors. First, lowering of the water table due to drainage and tillage increases soil oxygenation and makes SOM more accessible to microbial activity (Mastný et al., 2016). The favourable climate conditions in the region (1600 mm of precipitation, average temperature, 26 °C) further accelerate SOM decomposition. Different studies have also highlighted significant loss of SOM during land transformation and early plantations stages, even in upland soils, attributed to the high nutrient demand and also to the reduced C input from litter (Pérez-Cruzado et al., 2012, Cai et al., 2025). In *Eucalyptus* plantations, besides the effect on water availability, acidification or soil exchangeable cations (Farley et al., 2005; Berthrong et al., 2009), the different species of this genus typically have high lignin content, suggesting a limited contribution to SOM quality (Rencoret et al., 2011). As the forest plantation matures, C input from litter generally exceeds decomposition rates, leading to increased SOC (Pérez-Cruzado et al., 2012, Cai et al., 2025), as observed in the present study.

In terms of depth, similar values observed in the different fractions (especially noticeable in the case of C:N) are likely related with the initial soil ploughing, which reaches up to 30 cm depth, turning and mixing the soil and homogenizing soil nutrient distribution, at least to the depth considered in our study (30 cm). In the case of the C:N ratio in the grassland soils was approximately 6, consistent with low values observed in undisturbed riparian soils (Kumar, 2024). In the present study, the C:N ratio increased following afforestation, coinciding with the loss of SOM. Besides the evident information on C and N values, C:N ratio also informs us about litter decomposition and the formation of stable SOM (Xia et al., 2021). During wetland drainage, both C and N are predominantly lost; stored C is released to the atmosphere as CO_2 and N mineralization increased (Berglund and Berglund, 2010; Kalisz et al., 2021). Although drainage generally leads to a reduction in the C:N ratio (He et al., 2019), we must consider that the turning and mixing of soil during land preparation for planting has a major influence on the observed increase in C:N values. Furthermore, changes in the C:N ratio could be influenced by soil biological processes such as N fixation, immobilization and mineralization, which should also be considered (Jiao et al., 2016; Kumar, 2024).

4.2. Changes in SOM quality following afforestation of seminatural grasslands in floodplain soils

The present study revealed that labile organic compounds predominate in the SOM of seminatural grassland soils. Thermal analysis (DSC-TGA) and ^{13}C NMR spectroscopy revealed low T50w values (around 340 ºC) and low alky/O-alkyl ratio values (around 0.5), respectively. These findings are consistent with the distribution of C-compounds in tropical grasslands (Abe et al., 2009) and wetland soils and suggest a high potential for SOM degradation under aerobic conditions, such as those produced after soil drainage (Cooper et al., 2021; Busman et al., 2023).

Following intensive site preparation and the subsequent establishment of *E. camaldulensis* plantations (2 years after site preparation), a relatively greater loss of the most labile compounds (W_1 and O-alkyl fractions), which include C-compounds highly available to microorganisms (Baldock and Skjemstad, 2000), was detected by thermal analysis and ^{13}C NMR spectroscopy. A similar increase in thermal stability was reported in tropical wetlands converted to oil palm plantations after drainage and tillage (Cooper et al., 2021). The differences in the degradation of labile SOM are also consistent with the findings of other studies on degraded wetland soils (Gao et al., 2014; Mastný et al., 2016; Busman et al., 2023). Additionally, changes in SOM composition following land-use changes, due to aeration or ploughing or to intensive use such as those observed in natural systems transformed to intensive agriculture (or vice versa), have been documented in different upland environments (Boeni et al., 2014; Merino et al., 2023; Wang et al., 2023).

The SOM content partly recovered during growth of the plantation, mainly due to the increases in O-alkyl and alkyl-C compounds. During grassland recovery, the increase in SOM is driven by inputs of labile C from herbaceous litter (>10 Mg ha^{-1} yr^{-1}), root litter inputs (up to 0.95 Mg C ha^{-1} yr^{-1}) and root exudates (Guo and Gifford, 2002). Although grass is often present under *E. camaldulensis* plantations, afforestation leads to soil compaction (from 0.98 to 1.3 gr cm^{-3}) (Villalba-Martínez et al., 2024), restricting root growth and reducing litter input. In our opinion, the slight but gradual increase in SOM content is due to the litter biomass provided by the *E. camaldulensis* trees. However, although deposited continuously, *Eucalyptus* litter is decomposed slowly and presents recalcitrant components (acid leachates, high polyphenolic and lignin content) that ultimately affect the quality of the SOC (Zinn et al., 2002). In this sense, it is also worth highlighting the contribution of fine roots (and also deeper roots) to the increase in SOC. Thus, different studies in eucalyptus plantations showed a notable contribution of C from fine roots to the total C content of the soil, although the effect of root contribution can be considered as marginal compared to leaf litter inputs (Cao et al., 2020). In this respect, the thermograms showed a gradual increase in SOM but also revealed a narrowing (lower temperature range) of each of the peaks associated with both the W_1 and W_2-W_3 fraction, probably related to the homogeneity of C-compounds provided by *Eucalyptus* monocultures.

In addition to the enhanced soil aeration following site preparation and the favourable climatic conditions that promoted SOM decomposition, the mechanism referred to as the priming effect (Kuzyakov, 2010; Guenet et al., 2018) may be particularly relevant in these drained soils (Ding et al., 2023; Li et al., 2023a). Site preparation, involving draining and ploughing, led to the accumulation of large amounts of partly decomposed organic matter (grass litter) on the surface under anaerobic conditions in the soil. The decomposing litter would become highly available as an energy source for microorganisms, favoured by the new oxidizing soil environment caused by the lowered water table and improved soil aeration that favour plant establishment during afforestation. However, as eucalyptus plants grow long, and short rotation turns accumulates, soil compaction increases progressively (Cambi et al., 2015; Tassinari et al., 2019). As indicated above, initial soil conditions would stimulate microbial activity, accelerating the decomposition of both litter and SOM, and even including more stable organic matter compounds.

While some studies have shown that particulate OM contributes significantly to the priming effect (Perveen et al., 2019), other studies have highlighted the important role of mineral-derived protection (Vestergård et al., 2016; Chen et al., 2022). The sandy texture and the low charge of the clay minerals in native soils in the present study probably led to weak binding between minerals and SOM (Giannetta et al., 2019). Under aerobic conditions, fungi and Gram-positive bacteria are thought to utilize derivatives from Gram-negative bacteria to break down newly formed particulate organic matter (Li et al., 2023a). Given the allegedly low levels of particulate SOM (physically unprotected), the substantial SOM loss could facilitate the decomposition of the less accessible mineral-associated OM, which is typically considered protected from microbial decomposition by physicochemical processes (Lützow et al., 2006). Different studies have pointed out that part of the mineral-associated OM may undergo relatively rapid turnover (Giannetta et al., 2019; Yu et al., 2022). Moreover, the breakdown of aggregates during soil preparation would even increase the exposure of physically protected SOM to microbial activity, further accelerating its decomposition (Giannetta et al., 2019; Jin et al., 2021).

4.3. Extractable P and the evolution of soil P forms after grassland transformation

One of most notable effects of grassland afforestation in the Humid Chaco was the rapid and significant decrease in soil extractable P. A previous study already showed a dramatic reduction in soil extractable P reserves of more than 90 %, in grasslands afforested with *E. camaldulensis* (Villalba-Martínez et al., 2024). Although the magnitude of the reduction of P reserves is slightly lower in our study (72 % after 8 years of eucalyptus planting), it is highly likely that soil P is following the same path, with a loss that can be directly attributed to plant uptake. However, the possibility of P reservoirs depletion with an increased risk of P loss to water bodies via soil removal and drainage after intense site preparation should not be discarded (Arenberg et al., 2020). Plants in general, but particularly fast-growing species such as *Eucalyptus* spp., are characterized by the acquisition of large amounts of P and other nutrients, especially during early stages (Jobbágy and Jackson, 2004; Laclau et al., 2013; Bassaco et al., 2018). Thus, P deficiency is a common limitation for the initial development of *Eucalyptus* spp. plantations (Tng et al., 2014). In fact, a generic recommendation for the initial establishment of *Eucalyptus* is the application of inorganic fertilization at plantation establishment (Fernández et al., 2018; Foltran et al., 2019).

In the present study, the ^{31}P NMR spectra showed substantial changes in the distribution of the different P forms after site preparation and afforestation. In the native grasslands, organic P made up around 45 % of the extractable P, slightly below the normal range which is generally about 3:1 ratio favouring organic P (Condron et al., 2005; Turrión et al., 2010; García-Oliva et al., 2018). After afforestation, the organic P fraction was greatly reduced, with losses attributable to the increased mineralization of SOM triggered by the increased aeration caused by drainage and tillage. The loss of SOM possibly reduced the microbially-mediated process of P immobilization, fostering P mineralization. In this respect, the loss of SOM usually implies a decrease in the presence of organic P (Herlihy and McGrath, 2007; Stutter et al., 2018; Souza-Alonso et al., 2025). Some studies have even reported mineralization rates of organic P of between 5 and 20 kg P ha^{-1} yr^{-1}, which generally coincides with the P depletion observed in the grasslands under study (McDowell and Stewart, 2006).

Despite the possibly high mineralization rate of the organic P fraction, the inorganic P did not increase after afforestation. On the contrary, a notable decrease in orthophosphate (until year 4) that was sustained in time was also observed over time. The accumulation of large amounts of P in the tree biomass observed in previous studies (Villalba-Martínez et al., 2024) suggests that most of the mineralized P may be up taken during the forest growth. However, we should not rule out the possible transfer of P to aquatic environments with potential consequences such as freshwater eutrophication (Weihrauch and Weber, 2020). It is also possible that part of the inorganic P in these acid soils may be adsorbed by Al or Fe, leading to irreversible sequestration of P. The latter process may even be favoured by the decrease in SOM and increased acidity observed in these soils (Hawkins et al., 2022). Regarding the organic P forms, the slight increase in diester compounds after afforestation may be related to increased soil acidity, which prevents degradation of such compounds (Martín-Sanz et al., 2024).

Monoester compounds are usually present in a higher proportion than diester compounds (3:1 ratio) in most soils. The higher charges stabilize monoesters by association with metal oxides, organic matter and clays, which confer resistance to microbial activity (Condron et al., 2005, Turner, 2008). Nevertheless, the values of the different P fractions must be considered relative to the total P values. Thus, the proportions provide an erroneous picture, as in this case, the dominance of the organic fraction is not related to that of P immobilization but is due to the uptake of inorganic P by the *E. camaldulensis* plantation.

4.4. Contributing to grassland sustainability across the Humid Chaco

A previous study revealed the large impacts on SOM and nutrient dynamics of the transformation of native grasslands into *E. camaldulensis* plantations within the Humid Chaco (Villalba-Martínez et al., 2024). Various planning strategies and silvicultural practices that could potentially be implemented to reduce soil nutrient depletion and to maintain ecosystem sustainability and long-term functioning e.g., decreasing the planting density, extending the duration of *E. camaldulensis* rotation, or the application of organic fertilizers were already outlined. These practices should focus on preserving SOM content and quality while promoting more efficient P use in the ecosystem.

In line with the previous findings, the present study shows that depletion of SOM following seminatural grassland afforestation has direct implications for soil quality and soil fertility. In regard to hydrological processes, alterations in water flow, increased flood risk, prolonged drought or even changes in water quality are foreseeable (Mitsch and Gosselink, 2015; Couwenberg et al., 2010). This concern about hydrological impacts align with other research evaluating the impact of converting natural grassland to fast growing tree species plantations. Eucalyptus afforestation in the Brazilian Guarani aquifer and Argentina Pampas reduced groundwater recharges compared to grasslands (Souza-Mattos et al., 2019; Engel et al., 2005). In Paraguay's Humid Chaco the lowering of water table depth can occur in response to the drainage and the increased plantation-driven evapotranspiration. In Chile (Álvarez-Garretón et al., 2019) showed that replacing native forests or grasslands with pine or eucalypts plantations significantly reduced annual streamflow, especially in drier basins. This is of particular importance considering the vast area of grasslands and floodplains that will be further affected by drainage and tillage through the ecoregion, due to their potential for conversion to intensive forest production areas (INFONA, 2023a, 2023b). These data show that policy strategies recognizing the high value of wetland ecosystem services are necessary to protect of hydrological corridors. Other consequences, typical of transformed areas but outside of the scope of this study - including soil erosion and increases in greenhouse gas emissions - could eventually be observed.

Given the detrimental effects highlighted by these studies, the designation of extensive floodplain areas for afforestation should be reconsidered. Limiting intensive site preparation practices such as drainage and tillage in these valuable ecosystems is an urgent priority. Establishing buffer zones with natural vegetation can help filter run-off, reduce nutrient leaching and provide habitats for wildlife. The height of the water table exerts significant control on SOM dynamics and, therefore, on GHG emissions (Liu et al., 2024). Indeed, a moderate decrease of 10 cm in the water table results in an increase of 0.75 Mg C ha^{-1} yr^{-1} in CO_2 emissions (Li et al., 2023b). Therefore, implementing measures to rewet drained soils to promote floodplain recovery should be considered.

Considering P limitation in the Humid Chaco and the notable difference between P consumption and available soil reserves, removal of P through tree harvesting could also jeopardize soil P reserves in the short to medium term. The depletion of both inorganic and organic P forms in less than one decade after afforestation indicates that natural biogeochemical processes may not be sufficient to replenish the P extracted by tree plantations. Our estimates show that P losses (together with other nutrients) due to whole tree harvesting, and even conventional harvesting (stem wood and bark) are comparable to or exceed soil available reserves In the *E. camaldulensis* plots under study, removal of soil P through harvesting at the end of one single rotation ranged from 20 to 75 kg ha^{-1}. This range accounts for only wood harvesting (leaving bark, branches and leaves) or whole tree harvesting, respectively. These values indicate that P removal represents between 40 (wood and bark harvesting) and 90 % (whole tree harvesting) of the soil extractable P reserves, with an evident risk of limiting the capacity of soils to support future tree or grass production (Villalba-Martínez et al., 2024). The latter point will be particularly important if the introduction of cattle is considered. A similar pattern of P limitation is often observed in intensively managed agricultural soils, which require important P inputs through fertilization to maintain the crop production (Sharpley, 1995).

In light of the study findings, in order to ensure the long-term P availability for the new forest systems, whole-tree harvesting should be avoided. Logging residues should be properly managed and treated to retain most of the P (and other nutrients) within the system. Fertilization could also be considered to replenish the exported P; however, this practice must be carefully evaluated in these soils (which are strongly affected by the water table) due to the risk of eutrophication.

5. Conclusions

The data obtained in this study show that the afforestation of floodplains in the Humid Chaco, previously occupied by seminatural grasslands, with intensively managed plantations of *E. camaldulensis* involves significant soil disturbance during soil preparation (e.g. drainage, tillage). Soil disturbance has led to SOM loss and altered nutrient dynamics, potentially compromising the sustainability of system and its valuable ecosystem functions. Results were in line with the general trend reporting larger decreases of C and N in shallow horizons (especially $<$10 cm) after land use change to eucalyptus afforestation, while the effect is progressively diluted in deeper layers ($>$30 cm, not included in our study). The findings also indicated that mineralization rates increased greatly following site preparation.

The use of advanced instrumental techniques (thermal analysis and ^{13}C CPMAS NMR) revealed loss of large amounts of labile of SOM compounds during the initial years, with subsequent increases in C:N ratios. All of the SOM compounds, including recalcitrant (and possible mineral associated OM) compounds, were degraded during this process. The rapid degradation may be attributed to the soil properties (sandy texture, low cation exchange capacity) and favourable environmental conditions. By the end of the rotation, the SOM content and quality had partially recovered. Nevertheless, subsequent rotations could lead to gradual depletion and degradation of the SOM. On the other hand, the ^{31}P NMR spectrometric analysis revealed that both organic and inorganic P forms were gradually depleted during the first rotation of *E. camaldulensis* plantations. This was attributed to the accelerated mineralization of the organic P, combined with the high uptake of mineral P, released during the decomposition of the organic P. Due to the removal of large amounts of P after tree harvesting, this practice would lead to the depletion of soil P reserves, compromising P availability in the short to medium term. Given the vast area designated for afforestation in the region, as well as the importance and representativeness of humid grasslands at the global level, the adoption of proper measures for the conservation and sustainable management of these type of ecosystems is therefore essential.

CRediT authorship contribution statement

Pablo Souza-Alonso: Writing – review & editing, Writing – original draft, Validation, Supervision, Software, Investigation, Formal analysis, Data curation. **Carlos J. Villalba-Martínez:** Writing – review & editing, Writing – original draft, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Agustín Merino:** Writing – review &

editing, Writing – original draft, Validation, Supervision, Methodology, Investigation, Formal analysis, Conceptualization. **Jorge D. Etchevers:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Investigation, Conceptualization. **Verónica Piñeiro:** Writing – review & editing, Resources, Methodology, Investigation, Data curation.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2025.123034.

Data availability

Data will be made available on request.

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