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Soil organic carbon sequestration under *Araucaria angustifolia* plantations but not under exotic tree species on a mountain range

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ABSTRACT: Plantation forests can be efficient C sinks in biomass and soil organic carbon (SOC), but the latter depends on many factors, including climate. Tropical humid, mountain areas have cooler temperatures, slowing microbial decomposition, and thus can store considerable SOC. However, the effects of forest plantations on SOC of these montane areas are still poorly studied. Here, we aimed to assess changes in SOC, and related soil properties, after conversion of native rainforest to plantations of five tree species, with rotation cycles varying from 7 to 30 years, on the Mantiqueira Range, Minas Gerais, Brazil. We measured SOC contents and stocks (0.00-0.40 m layer) under a native montane rainforest (control) and plantations of *Eucalyptus, Pinus, Cunninghamia, Cupressus* and *Araucaria*, all planted in 3 × 3 m spacing, at an altitude of ca. 1,300 m, marked by humid and cool climate, where SOC contents are naturally high. Soil organic carbon varied from 55 g kg^{-1} under *Eucalyptus* to 105 g kg^{-1} under *Araucaria* (0.00-0.05 m layer), decreasing in depth (0.20-0.40 m) to the still high values of 20-40 g kg^{-1}. Soil organic carbon stocks for the top 0.20 m were also high, reaching ca. 140 Mg ha^{-1} under *Araucaria*, significantly higher value than the native forest (ca. 90 Mg ha^{-1}, $p<0.05$), which did not differ from the other species. Soil organic carbon stocks were not affected in the 0.20-0.40 m soil layer, whereas soil structure patterns changed under some species, without however resulting in bulk density changes, and pH decreased under *Araucaria*. Such data showed large SOC stocks under montane native forests can not only be preserved upon conversion to forest plantations, but considerable SOC sequestration can be achieved in 30-years rotation cycles plantations of indigenous *Araucaria angustifolia*, marked by more open canopies and greater understory biomass.

Keywords: carbon sequestration, soil organic matter, land use change, tropical soils.

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INTRODUCTION

Soil organic carbon (SOC) has been proposed as a promising strategy for significant CO_2 sequestration into a terrestrial C pool, on a decadal timeline, using plant biomass production, its natural microbial alteration and incorporation within the soil mineral matrix (e.g., Lal et al., 2018) as soil organic matter. In all soils, SOC contents result from interaction of the five factors of soil formation (climate, organisms, relief, parent material and time), as described by Jenny (1941), but also from edaphic factors such as clay content, mineralogy and structure. In agricultural areas, SOC sequestration potential must always be considered as an additional benefit to food and fiber production, and has initially been studied in sites converted from conventional to conservation tillage methods for annual crops.

Growing attention has been given to perennial crops, including pastures, fruit trees, and plantation forests (Ledo et al., 2020), which can incorporate considerable amounts of C into soils, while requiring much less intensive management systems and minimum disruption of soil structure. Among these options, plantation forests have the additional advantage of high potentials for C sequestration also in wood and aboveground biomass, especially when fast-growth tree species are planted in the humid tropics with short (5-7 years) rotation cycles. However, in the humid tropics, aboveground C stocks under plantation forests are often surpassed by SOC stocks, since these soils can have significant SOC contents (3 % or more) and can be very deep (e.g., >5 m) (Chiapini et al., 2023). Since SOC exists in a more advanced decomposition stage than aboveground biomass and wood, it has a potentially longer residence time than wood and biomass C, SOC can be a more stable global C pool, and thus has been the subject of numerous studies in plantation forest ecosystems (Laganiére et al., 2010).

Abundant biomass inputs under plantation forests in the humid tropics suggest SOC sequestration in soils underneath can also be considerable, but, in fact, there is wide variation regarding how and how intensely SOC is affected in these areas. Some researchers report considerable SOC losses after conversion to fast-growth tree plantations (Zinn et al., 2002), whereas other studies found no significant changes (Zinn et al., 2011, 2014), or conversely, significant SOC gains (Zinn et al., 2014). Such variability is mostly due to different experimental conditions (e.g., climate, slope, soil texture, etc.), silvicultural factors (e.g., soil preparation, tree species and varieties, stand age), and antecedent land use (e.g., native vegetation or degraded cropland) among the studied sites. All of these are critical factors affecting the direction (i.e., gain or loss) and intensity of SOC change, and the variability in the conclusions listed in the literature poses reasonable questions on whether tropical plantation forests can actually sequester significant C in soils, when compared to wood and aboveground biomass. In addition, the required silvicultural practices, rapid growth and intense SOC input under plantation forests often promote changes in other soil properties, such as decreased pH and exchangeable bases and increased Al^{3+} availability, whereas bulk density can increase or decrease, where densities under pristine vegetation were respectively low as in Oxisols with granular structure, or high as in Ultisols with block structure (Zinn et al., 2002, 2011, 2014).

The gold standard to assess a net SOC sequestration after conversion to plantation forests should ideally be their comparison with the same soils under pre-existent native vegetation. This is because converting croplands or degraded pasturelands, typically heavily depleted in SOC, into plantation forests will always show considerable SOC increases that cannot be considered a net C sequestration, since it did not account for former SOC losses in cultivated areas where local SOC dynamics was disturbed. Even when only native vegetation controls are used to assess SOC changes, the results can vary considerably: in a meta-analysis about Brazilian soils after conversion to afforestation with *Eucalyptus* sp., Fialho and Zinn (2014) reported SOC changes followed a normal distribution centered around zero, i.e., there were no significant net changes in SOC for

Brazil as a whole, for either a standardized 0.00-0.20 or 0.00-0.40 m layer. Although these results may lead to the broad conclusion that SOC is generally not affected by eucalypt plantations, on a site-by-site basis, SOC changes can actually be intense.

Identifying specific practices and environmental settings that favor SOC sequestration (or loss) is critical to better supporting planning and decision-making. One situation that can promote SOC sequestration is the establishment of forest plantations on high elevation areas, where cooler temperatures limit organic matter decomposition and soil respiration (Okello et al., 2023). This trend has recently received more attention in the tropics, where higher SOC stocks have been reported at higher elevations along mountain ranges (Tashi et al., 2016; Njeru et al., 2017; Atourakai et al., 2023; Okello et al., 2023). These studies were mostly focused on soils under native vegetation, but recent data have shown that increasing elevations can favor SOC sequestration under coffee stands, when compared to native forests, along a mountain range in Brazil (França et al., 2023). Although there is scant evidence in the literature regarding the effect of elevation on SOC dynamics upon land use change, if preliminary evidence has shown SOC sequestration for perennial crops such as coffee, plantation forests with much higher biomass input rates can present an even higher potential, which deserves investigation.

This study assessed changes in SOC, bulk density and other critical soil properties after the conversion of native forests to plantations of five forest tree species with different rotation cycles (7, 15 and 30 years) and origins (exotic vs. indigenous), at similar elevation (ca. 1,300 m) in the Mantiqueira Range in Southeastern Brazil. We tested the hypothesis that, due to cooler temperatures and thus slower organic decomposition, SOC changes upon conversion of native forests to plantation forests in these tropical humid mountains are mostly positive, i.e., SOC sequestration will ensue.

MATERIALS AND METHODS

Study area

The Mantiqueira Range is the inland rim of the *Além Paraíba* horst and graben fault system, a remarkable feature near the triple border among the States of Minas Gerais, São Paulo and Rio de Janeiro, in Southeastern Brazil (Figure 1a). Briefly, local lithology comprises Neoproterozoic (ca. 850 M. a.) orthogneiss and migmatite, i.e., the area can be considered a granite/gneiss complex (Codemig, 2003). These crystalline rocks typically weather, where slope is montane or hilly, into Ultisols and Inceptisols (*Argissolos* and *Cambissolos*, in the Brazilian Soil Classification) of loamy texture, with moderate to low fertility (Minas Gerais, 2010). Forest plantations were established for pulp production in the decade of 1940 near Monte Verde-Camanducaia, in the southernmost part of Minas Gerais, at elevations between 1,200 and 1,700 m a.s.l. Mean annual temperature and precipitation, measured at an automated climate station at elevation 1,300 m, are respectively 17.4 °C and 1,960 mm (Climate Data, 2023). The warmest month (February) has a mean temperature of 19.9 °C and the coolest (July) of 13.9 °C, and even the driest winter months have a mean precipitation of ca. 40 mm. These mean temperatures are ca. 2-4 °C lower relative to lowland areas in similar latitudes, and thus, most of these plantations were established with cold-tolerant tree genera and species not commonly used in most of Brazil, where climate is much warmer.

The sampled plantation stands were chosen in order to be the closest possible to the native forest control, and at the most possibly similar altitude and slopes (Figure 1b). The stands of *Araucaria angustifolia* (Bertol.) Kuntze, a conifer indigenous to southern and southeastern Brazil and currently considered endangered (Bertini et al., 2021), were planted in the 1950's directly after clearance of the native rainforest, and have been harvested in 30-year rotations. The same 30-year rotation cycle was used for the exotic *Cunninghammia lanceolata* (Lambert) Hooker stands, planted in 1964, and *Cupressus* *lusitanica* Miller, planted in 1995, both immediately after clearance of native vegetation. Shorter-rotation exotic tree species were also established on land formerly planted with the exotic conifers: *Eucalyptus maidenii* (F. Muell) was planted in 1992 and harvested after 7-yr cycles, whereas *Pinus patula* Schiede ex Schltdl. & Cham. was planted in 2004 on 15-year cycles. More specific data on studied sites are given in table 1. In all cases, tree seedlings were planted with a 3 × 3 m spacing among pits, which were excavated with hoes and spades, i.e., no mechanical cultivation practices resulting in widespread disruption of soil structure were ever employed. No liming and chemical or organic fertilizations were performed. Sheet or rill erosion were not observed in any site.

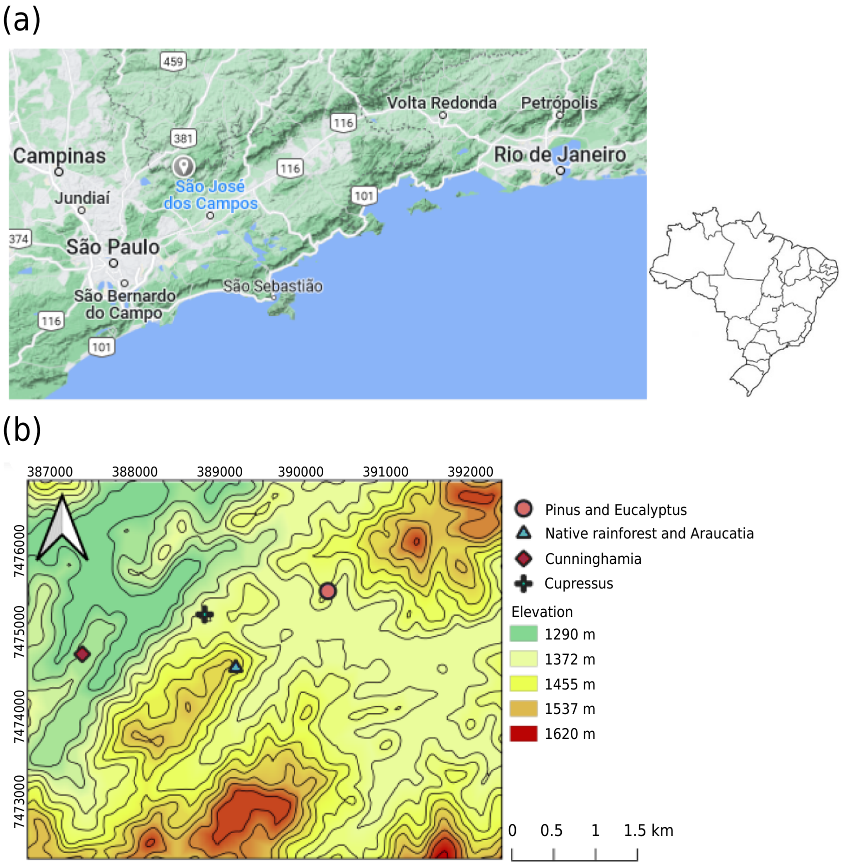

Figure 1. Studied sites in the Mantiqueira Range, southeastern Brazil: (a) approximate situation near São José dos Campos (from Google Maps), between the major cities of Rio de Janeiro and São Paulo; (b) Digital elevation model showing the location of sampled stands (note that two stands for Pinus and Eucalyptus are so close that appear as one single point; the same for Native Forest and Araucaria).

Table 1. Geographical and other data of the studied tree plantation stands

Forest type	Latitude S, longitude W	Altitude	Slope	Understory	Litter layer
		m	%		Mg ha^{-1}
Araucaria	22° 49’ 574”, 46° 04’ 466”	1,359	15	Very thick, very open canopy	15.5
Cunninghamia	22° 49’ 517”, 46° 05’ 510”	1,342	35	None, too dark under canopy	5.7
Cupressus	22° 49’ 366”, 46° 04’ 595”	1,377	14	Very sparse	10.4
Pinus	22° 49’ 279”, 46° 04’ 769”	1,350	8	Very sparse	19.4
Eucalyptus	22° 49’ 279”, 46° 04’ 769”	1,350	9	Thick	17.9
Native rainforest	22° 49’ 574”, 46° 04’ 466”	1,370	18	Thick	8.3

Soil sampling and analyses

At each treatment (five tree plantations and one native forest preserve as control) site, soils were sampled in three replications, each situated within a polygonal plantation stand, in order to allow true replication. Due to the hilly terrain, each stand had different sizes and shapes, and near the center of each stand, we excavated soil pits at least 0.50 m deep. The native forest stands are very close to the *Araucaria* stands, separated only by an earthen road, as well as the *Pinus* and *Eucalyptus* stands (Figure 1b). The distance between the native forest control and *Cupressus*, *Cunninghamia* stands never exceeded 2 km.

Undisturbed soil samples were collected directly from the pit walls, in the pre-established soil layers of 0.00-0.05, 0.05-0.10, 0.10-0.20, and 0.20-0.40 m, using cylindrical metal cores (0.05 m dia.). Soil bulk samples were taken from pit walls at the same layers with metal tools, air-dried for five days, and dry-sieved <2 mm. In order to provide a micromorphological perspective of soil composition, structure and changes thereof, undisturbed soil samples were taken from the top 0.10 cm in Kubiena boxes from all plots, carefully transported, and air-dried for three months.

Soil particle-size distribution was determined by the pipette method after dispersion of a 50-g fine earth sample in NaOH 1 mol L^{-1}, slow shaking for 16 h, collection of sand with a 53 µm sieve, and then siphoning to separate clays from the suspension (Gee and Bauder, 1986). Soil clays obtained as above from the native forest plot were mounted on oriented glass slides for X-ray diffraction on a Phillips PW1840 (Lelyweg, Holland) apparatus, using CoKα radiation. Soil bulk density was determined by the core method after oven-drying at 105 °C for 24 h (Grossman and Reinsch, 2002). Soil chemical characterization was performed as described in Teixeira et al. (2017). Briefly, soil pH in water was determined in a 1:2.5 soil:water suspension ratio. Exchangeable Ca^{2+}, Mg^{2+} and Al^{3+} were extracted with 1 mol L^{-1} KCl solution; Ca^{2+} and Mg^{2+} were determined by atomic absorption, and Al^{3+} contents by titration with NaOH solution 0.025 mol L^{-1} using bromothymol blue. Available K^+ and P were extracted with the Mehlich-I solution (0.05 mol L^{-1} HCl and 0.0125 mol L^{-1} H_2SO_4); K^+ was determined by flame photometry and P by colorimetry. Potential or total soil acidity (H+Al) was assessed by extraction with Ca-acetate buffered at pH 7.0 and estimated by the change in a previously set pH. With these data, the following soil fertility indices were calculated: cation exchange capacity (CEC pH 7.0 = $Ca^{2+}+Mg^{2+}+K^++H+Al$); and sum of base cations as $Ca^{2+}+Mg^{2+}+K^+$.

Soil organic C contents were determined in all soil bulk samples by dry combustion in a Vario TOC Cube (Elementar, Hanau, Germany). With such data and bulk density values, SOC stocks were calculated for each treatment and replicate for the standardized, combined 0.00-0.20 and 0.00-0.40 m soil layers as:

$$SOC_{stock} = \sum_i^n volume \times \rho b \times \frac{SOC_{content}}{100} \qquad \text{Eq. 1}$$

in which: SOC_{stock} is in Mg ha^{-1}; n is the number of pre-established soil layers included; $\sum_i^n volume$ is the total soil volume per standardized layer in m^3 ha^{-1}; ρb is soil bulk density in Mg m^{-3}; and $SOC_{content}$ is in %. The SOC contents in the sand fraction, obtained after sieving (during the particle-size distribution analysis, oven-drying and grinding with mortar and pestle), were determined by dry combustion as described above for two selected layers (0.00-0.05 and 0.10-0.20 m). With these data, the C pool as particulate organic matter was calculated as the percent fraction of total SOC, as proposed by Zinn et al. (2011). The litter layer was sampled using a 0.5 × 0.5 m wooden square frame and knife randomly located in each plot, oven-dried and weighed.

The experimental design used was completely randomized, with six treatments corresponding to the native forest and five tree species, in four soil layers and three replicates, totaling 72 soil samples. All results were treated by analysis of variance, and when a significant effect was detected, the Tukey-Kramer test of means at $p<0.05$ was applied to compare treatments, always for the same soil layer, using the statistical program JMP 5.1. The same design and analysis were done for the SOC stocks at the standard layers of 0.00-0.20 and 00.00-0.40 m, as well as for the particulate organic matter C.

Undisturbed, air-dried soil samples in Kubiena boxes were oven-dried for three days at 40 °C and one day at 80 °C, then impregnated with epoxy resin, hardened for 4 h at 100 °C and cured for another 4 h at 140 °C. The blocks were cut in a diamond disk saw, polished and mounted on glass slides with Hillquist® epoxy resin in the 7A/3B, then cut and lapped to a 30 µm thickness. Images of the glass slides showing soil structure patterns were recorded with a desk scanner, and thin sections were photographed with a digital camera coupled to a petrographic microscope.

RESULTS AND DISCUSSION

Soil characterization

The studied soils are considerably deep for the hilly terrains, exceeding 1 m in all sampled areas, and mean clay contents ranged from 250 to 540 g kg^{-1} (Table 2). The lowest clay contents occurred in the *Cunninghamia* plots, where the slope was steeper (35 %), and the highest clay contents were noted under the *Eucalyptus* and *Pinus* stands, which had the less pronounced slopes (8-9 %). The soil under native forest presented a structure consisting of small angular blocks, marked by planar voids (fissures) and peds with sharp to smooth outlines (Figure 2). According to X-ray diffraction data (Figure 3), at the 0.00-0.05 m soil layer, clay mineralogy is marked by wide predominance of gibbsite over poorly crystalline kaolinite, or more likely halloysite, and goethite also occurs. The same occurs at the 0.20-0.40 m soil layer, but kaolinite (or halloysite) is slightly more crystalline and abundant, and traces of 2:1 minerals occur, probably of illite and/or vermiculite. The sand fraction is dominated by angular, poorly sorted quartz (Figure 4), resulting from direct alteration of the underlying granite/gneiss. Muscovite is also visible in thin sections (Figures 4a, 4c and 4e), as well as minor amounts of coarse K-feldspars [not shown, see images in Fialho (2012)]. When combined, these data are indicative of advanced weathering stage on well-drained sites that are also subject to rejuvenation by erosion, and consistent with a tropical, montane humid climate. The predominance of gibbsite in clays was not expected in tropical montane soils on granitic rocks and can probably be ascribed to kaolinite deterioration under high moisture and effective drainage.

The soil under native forest is strongly acid (pH 4.7-5.1), with considerable exchangeable Al^{3+} greatly exceeding the sum of bases, especially in the top 0.10 m, which is a common pattern for montane forest vegetation under humid climate (e.g., Bertini et al., 2021). The SOC contents in this soil ranged from 66 g kg^{-1} at the 0.00-0.05 m to 19 g kg^{-1} at the 0.20-0.40 m soil layer, which are very high compared with data from other forests on coarser-textured soils at a similar elevation of 1,260 m (França et al., 2023), or in subtropical Brazil, at elevations ca. of 1,120 m (Bini et al., 2013). Such high values can be ascribed to thick forest cover, higher humidity and lower temperatures in the area of the present study, and to the halloysite + gibbsite clay suite. Available soil P is also low, another common condition in Brazilian acid soils.

Soil structural patterns and chemical properties as affected by plantation forests

Since the forest plantations were planted without site preparation with plows and only manual pitting with hoes for the seedlings, which is done only after 7, 15 or 30 years after harvesting and before new plantings, it was expected soil structure would change little, but this was not the case. The angular blocks noted in the native forest actually increased

Table 2. Mean soil chemical and physical characterization data and SOC per sampled layer

Forest type	pH(H_2O)	P[1]	Al^{3+}	Sum of bases	CEC pH 7	Clay	Sand	BD	SOC
		mg dm^{-3}	——— cmol$_c$ dm^{-3} ———			——— g kg^{-1} ———		g dm^{-3}	g kg^{-1}
				0.00-0.05 m					
Araucaria	4.6b[2]	2.8bc	2.9a	2.1ab	27.9a	420	450	0.78a	104a
Cunninghamia	4.9a	4.4a	2.1a	0.7b	17.2bc	250	550	0.68a	83ab
Cupressus	5.1a	2.0b	2.2a	0.6b	12.9c	420	360	0.94a	63b
Eucalyptus	5.0a	1.6c	1.9a	0.4b	15.7bc	390	460	1.02a	55b
Pinus	5.1a	3.2ab	1.0a	3.6a	17.9b	470	440	0.86a	75ab
Native	4.7b	2.6bc	1.9a	0.8b	18.5b	350	490	0.87a	66b
				0.05-0.10 m					
Araucaria	4.8b	1.6a	2.2a	0.5a	21.2a	400	490	0.90ab	77a
Cunninghamia	5.1a	2.3a	2.1ab	0.8a	16.2ab	250	550	0.79b	65ab
Cupressus	5.3a	1.8a	1.3ab	0.4a	9.0c	400	390	1.09ab	39c
Eucalyptus	5.2a	1.4a	1.3ab	0.3a	11.3bc	390	490	1.21a	36c
Pinus	5.1a	1.8a	1.0b	0.8a	18.4ab	450	430	1.10a	48bc
Native	4.8b	1.6a	1.4ab	0.3a	14.1bc	350	490	1.09ab	42c
				0.10-0.20 m					
Araucaria	5.0c	1.0b	2.7a	0.3a	17.4a	430	450	0.93a	29b
Cunninghamia	5.2b	2.1a	1.8a	0.5a	12.9b	260	530	0.97a	48ab
Cupressus	5.4a	1.1b	0.5a	0.4a	6.5c	390	400	1.15a	66a
Eucalyptus	5.2b	1.1b	1.0a	0.3a	10.1b	410	470	1.21a	41b
Pinus	5.1bc	1.6ab	0.9a	0.4a	10.9b	450	440	1.16a	32b
Native	5.0c	0.8b	1.0a	0.3a	12.7b	340	510	1.21a	28b
				0.20-0.40 m					
Araucaria	5.1b	0.4b	1.0b	0.2a	12.6a	440	440	1.06b	19b
Cunninghamia	5.3b	1.3a	1.7a	0.5a	10.8ab	270	540	0.96b	20ab
Cupressus	5.5a	0.8ab	0.3c	0.3a	5.5c	370	370	1.34a	34a
Eucalyptus	5.2b	0.5b	0.4c	0.2a	6.3c	420	450	1.17ab	20b
Pinus	5.3b	0.6b	0.4c	0.3a	5.7c	460	420	1.20ab	30ab
Native	5.1b	0.7ab	0.4c	0.2a	7.6bc	310	530	1.33a	19b

[1] Extraction was performed using Mehlich-I. [2] Means in a column followed by the same letter compare different forest types for the same layer (Tukey-Kramer's test at $p<0.05$). BD: bulk density.

in size after conversion to plantation forests, tending to a massive soil microstructure under *Eucalyptus* (Figures 2 and 4b), which has the shortest rotation age at 7 years, and thus the greatest number of soil preparation and harvesting operations. A similar trend occurred under *Cupressus* stands, planted on a much longer (30 years) rotation, but is marked by very thin litter layers (Table 1) due to the small size and prompt decay of its needles, which render the soil mostly bare or covered only by mosses and twig litter, apparently not favoring soil faunal activity which would promote a more granular structure pattern. An opposite trend occurred under 30-year rotation *Araucaria* and especially *Cunninghamia* stands, where soil structure underwent a loosening process, i.e., a change to a fine granular type (Figures 2 and 3). These soils under *Cunninghamia* and *Araucaria* also presented a much darker soil matrix than the others (Figures 2, 3e and f), due to the pigmentation promoted by increased SOC contents (Table 2). Finally, the soil under *Pinus* also presented an intermediate structural pattern, with a transition from subangular blocks with much larger planar voids to a granular type (Figure 4d) with

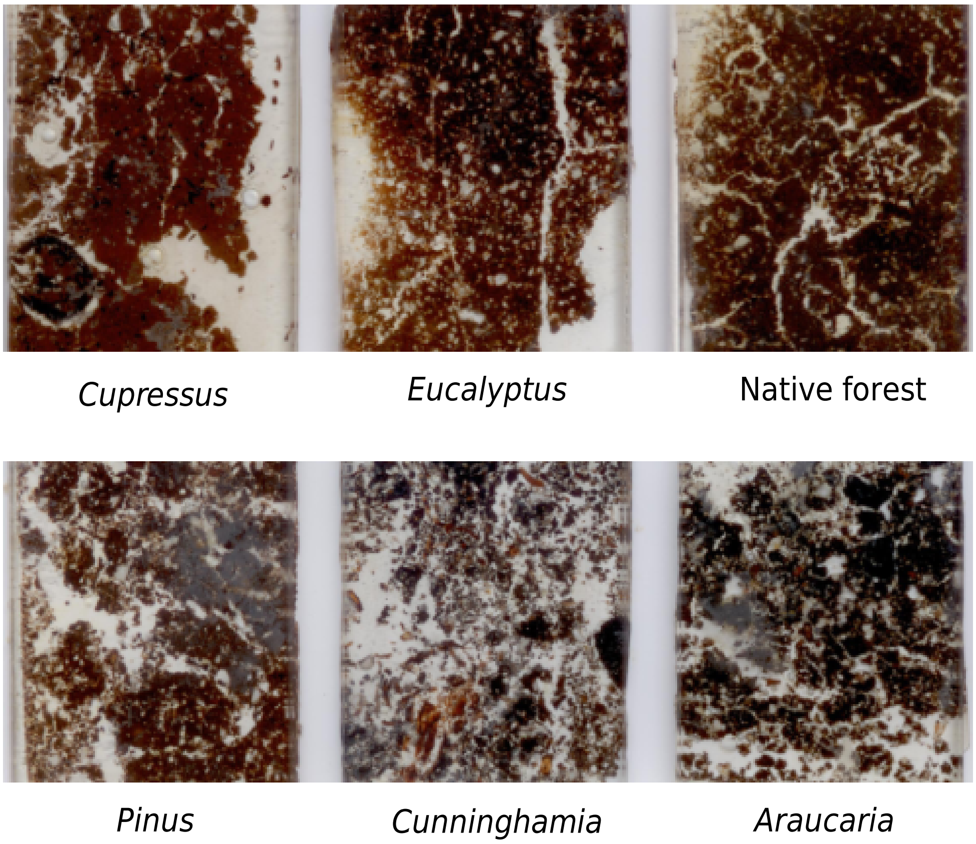

Figure 2. Glass slides (2.8 cm wide) showing semi-thin (ca. 0.2 mm thick) sections with soil structure patterns under different tree plantations and undisturbed native forest. Note finer soil aggregates under *Pinus*, *Araucaria* and especially *Cunninghamia* stands, and dark soil color under *Araucaria*. Note also coarse root remains under *Cunninghamia* and *Cupressus*.

packing voids, i.e., a partial loosening of structure, favoring water infiltration. A similar trend of loosening after native forest conversion to *Pinus* plantations in other locations was reported earlier and ascribed to increased root and faunal activity (Zinn et al., 2014). Nonetheless, these soil structural changes did not result in significant changes in bulk density between the native forest and plantation forests for the top 0.20 m

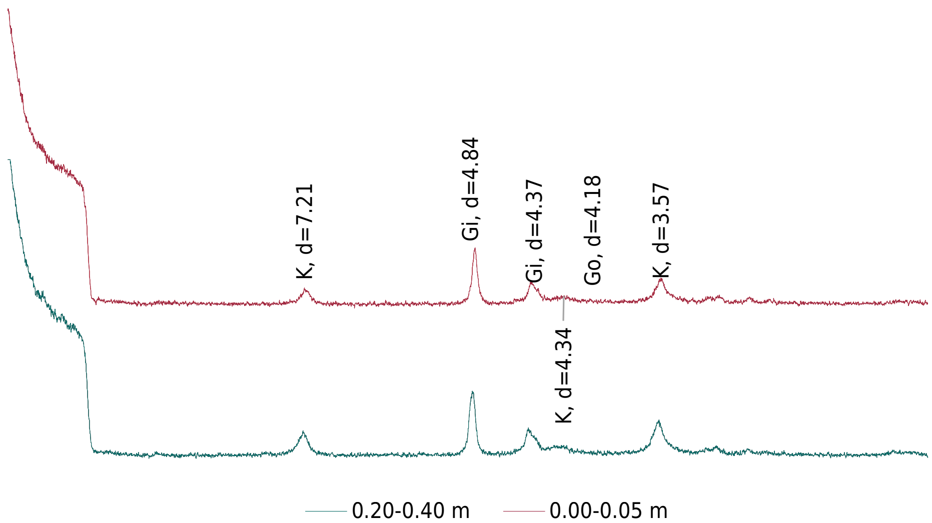

Figure 3. X-ray diffraction patterns for oriented clay from the native forest stands at two soil layers. K: kaolinite; d: basal distances; Gi: gibbsite; Go: goethite.

(Table 2), although bulk densities under *Araucaria* and *Cunninghamia* were significantly lower at the 0.20-0.40 m soil layer, compared with the native forest. These apparently contradictory trends are explained by the fact that, although the soil structural pattern has changed, i.e., towards a more granular type or greater blocks, it did not result in significant changes in total soil porosity.

The native forest and *Araucaria* plots had soil pH values slightly but significantly lower than the plantation forests for the top 0.20 m, which were, however, not associated with increased Al^{3+} activity (Table 2). The increased soil acidity under *Araucaria* was possibly caused by higher amounts of protonated carboxyl groups in soil organic matter due to higher SOC contents (see next), as suggested by higher values of CEC at pH 7.0. In Ultisols, soil organic matter not only increases the acid buffering capacity, but also hampers the production of exchangeable and soluble Al, besides inhibiting the mobilization of Al (Li et al., 2022), which can explain the pattern observed here. Contents of base cations were generally not affected by forest plantations, with the exception of higher values under *Pinus* at the 0.00-0.05 m soil layer, which can reflect strong uptake from deeper layers and cycling by litterfall.

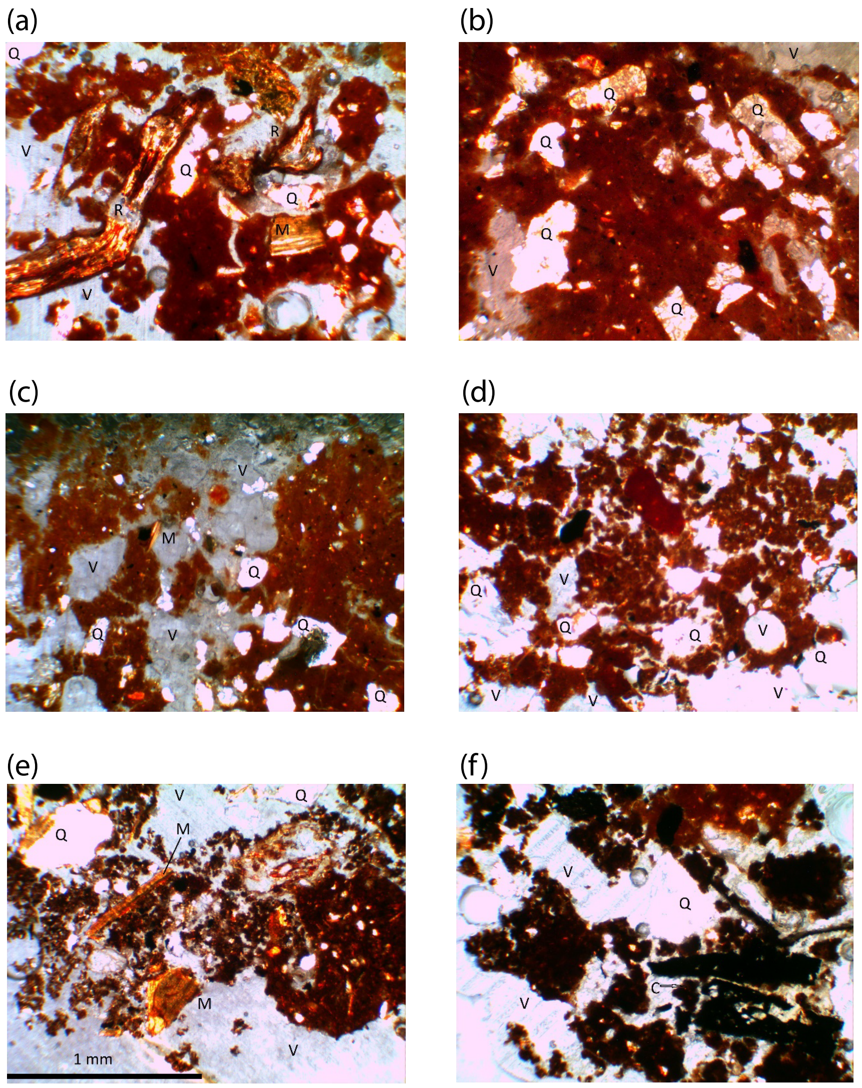

Figure 4. Microstructure and porosity patterns in soils under: (a) *Cupressus*; (b) *Eucalyptus*; (c) native forest; (d) *Pinus*; (e) *Cunninghamia*;, and (f) *Araucaria* stands. All images 2.3 mm wide, taken between partly crossed polars. Q: quartz grains; V: voids; M: mica; R: roots; C: charcoal. Note much darker aggregates under *Araucaria* stands (f), and denser microstructure under *Eucalyptus* (b) and *Cupressus* (a).

Soil organic carbon (SOC) as affected by plantation forests

Soil organic carbon contents were affected by plantation forests differently along the four sampling depths (Table 2). At the 0.00-0.05 m, SOC contents were considerably higher (+57 %) under *Araucaria* than under the native forest control, which did not differ significantly from the other tree species. However, at the 0.05-0.10 m, SOC contents were at least 50 % higher under both *Araucaria* and *Cunninghamia* than in the native forest control, whereas for the 0.10-0.20 and 0.20-0.40 m soil layers, only *Cupressus* presented higher SOC contents than the control. Such variability along sampling layers can be due to differential allocation of root or litter inputs among the plantation forests, but the difficulty in interpretation is easily solved by comparing SOC stocks for the standardized layers of 0.00-0.20 and 0.00-0.40 m (Table 3).

The 0.00-0.20 m soil layer is most easily and routinely affected by soil preparation, litter and root inputs, and thus, it is commonly sampled to assess SOC changes due to agricultural and forest management (Fialho and Zinn, 2014). In the present study, the native forest presented a SOC stock for the 0.00-0.20 m soil layer of 85.5 Mg ha^{-1}, values very similar to those under *Eucalyptus* and *Cupressus*. This value can be considered very high in comparison to lowland Brazilian soils at similar latitudes: to put it in perspective, it is similar to those for various native forest soils at 0.00-0.40 m at an elevation of ca. 900 m and MAT of 19.3 °C (Araujo et al., 2017), and exceeds SOC stocks for the top meter (0.00-1.00 m) for sandy and loamy soils under native woodland savannas at altitudes of 550 m and with mean annual temperature of 22.8 °C (Zinn et al., 2011). However, the *Araucaria* plantation presented a statistically significant, much higher SOC stock of 134 Mg ha^{-1} for the 0.00-0.20 m soil layer, which amounts to a total net SOC sequestration of ca. 50 Mg ha^{-1}, when compared to the native forest control. The other four plantation forest stands did not differ significantly among themselves and the control.

The 0.00-0.40 m soil layer can be considered as the maximum depth potentially reached by intensive mechanical soil preparation. Regarding SOC, it is less affected by management practices than the 0.00-0.20 m soil layer. In fact, for the 0.00-0.40 m soil layer, there were no significant differences among the different forest types, although the same 50 Mg ha^{-1} increase under *Araucaria* compared to the native forest control was noted. The fact this difference was no longer significant is due to the much larger background SOC stock for comparison (for instance, for the native rainforest control, SOC stocks in 0.00-0.40 m are >50 % higher than for the 0.00-0.20 m), resulting in the 50 Mg ha^{-1} increase not to be distinguished as significant against the larger pooled variance and means. In addition, the data of our study show the relevant SOC sequestration was restricted to the top 0.20 m layer, where most of the organic inputs and faunal activity are likely to occur.

In any case, the considerable SOC sequestration under *Araucaria* reported in the present study is a novel and auspicious fact to those interested in sequestering SOC as a by-product of wood production or preserving the genetic pool of a conifer indigenous to Brazil and other subtropical countries. However, this may not be the case everywhere, since *Araucaria*

Table 3. Soil organic carbon stocks as affected by plantation forests, according to soil layer

Forest type	0.00-0.20 m	0.00-0.40 m
	Mg ha^{-1}	
Araucaria	134.1a[(1)]	202.4a
Cunninghamia	97.1ab	156.0a
Cupressus	81.2b	132.3a
Eucalyptus	88.8b	134.1a
Pinus	104.8ab	153.5a
Native	85.5b	153.5a

(1) Means in a column followed by the same letter compare different forest types for the same soil depth (Tukey-Kramer's test at p<0.05).

Table 4. Percent of total SOC as particulate organic matter (sand-sized SOC)

Forest type	0.00-0.05 m	0.10-0.20 m
	%	
Araucaria	3.82a	3.41b
Cunninghamia	4.93a	8.10a
Cupressus	2.99a	4.96ab
Eucalyptus	4.25a	4.89ab
Pinus	4.50a	4.44ab
Native	3.53a	5.54ab

Means in a column followed by the same letter compare different forest types for the same depth (Tukey-Kramer's test at $p<0.05$)

trees planted in low-altitude, warm areas such as Australia can actually lose considerable amounts of SOC, compared to native vegetation (e.g., Richards et al., 2007), so the altitudes involved in the present study appear to be critic. There are only a few studies assessing SOC changes due to land use change at highlands or mountain ranges (e.g., Njeru et al., 2017). However, Lemma et al. (2006) reported a strong SOC sequestration (up to 63 Mg ha^{-1} for a 0.00-0.50 m soil layer) under 20-yrs exotic *Cupressus* and *Pinus* plantations, in the Ethiopian highlands (2,100-2,340 m a.s.l.). Such values are comparable to those reported here, and support the idea high altitudes favor SOC sequestration, provided forests can still grow. In addition, few data are available for *Araucaria* plantations in Brazil or elsewhere, but the SOC contents under *Araucaria* in the present study greatly exceed those reported under native *Araucaria* forests in the coldest mountain range of southern Brazil (Oliveira et al., 2015), perhaps because temperatures are too low, or cloud cover is too high, in that area. Furthermore, SOC contents under long-term (32 years) *Araucaria* stands were not found to be significantly different from those of native forests in clayey soils in southern, subtropical Brazil, at climatic conditions similar to those of the present study but at a slightly lower elevation (1,120 m) (Bini et al., 2013). Both comparisons suggest well-managed *Araucaria* plantations in humid, cool mountain ranges in the tropics are highly sustainable in terms of combining wood production and SOC sequestration. It was not clear from our results, or the literature, the reason why *Araucaria* was the only species to promote net SOC sequestration, but it may be due to its more open canopy, allowing for more understory growth and C input (Table 1).

Finally, it is remarkable POM-C (i.e., sand-sized SOC) comprised <5 % of total SOC at the 0.00-0.05 m, and only slightly more at the 0.10-0.20 m, with no significant differences between the native forest control and plantation forests (Table 4). These values were slightly lower compared to reports for other native forest soils also in the state of Minas Gerais, at different locations with similar high elevations of ca. 1,200 m (Zinn et al., 2018; França et al; 2023) and ca. 900 m. (Araujo et al., 2017), but much lower than the 25-40 % reported for warmer savanna soils at 550 m (Zinn et al., 2011). These remarkable results suggest a very high stage of humification for the whole of SOC for forest soils in the present study. In addition, this suggests a strong stabilization of large amounts of humified SOC through binding to the halloysite + gibbsite clays, and not to inherently low POM-C contents in the study area. This trend is relevant since POM-C is typically more susceptible to decomposition by soil food web or loss to wildfire than mineral-associated SOC, and consistent with high stability of SOC pools in the studied ecosystems.

CONCLUSIONS

We tested the hypothesis that SOC can be effectively sequestered in tropical humid, cool mountain climates due to lower temperatures and considerable biomass production. This hypothesis was confirmed for *Araucaria angustifolia* stands, which presented a significant ~50 Mg ha^{-1} SOC sequestration for the 0.00-0.20 m soil layer over the native vegetation, which can be considered a gold standard in proving a net SOC sequestration in an ecosystem. Antecedent SOC contents in the native forest were already very high compared to most forest and savanna ecosystems in Brazil, so there was an inherent concern these areas could show considerable SOC losses after land use change. However, the data reported here showed that SOC was preserved in the *Cunninghammia lanceolata*, *Cupressus lusitanica*, *Eucalyptus maidenii* and *Pinus patula* plantations studied, which suggests a high confidence in the environmental sustainability of wood production in these montane areas, even if *Araucaria* is not the species planted. Our soil characterization and SOC size fractionation studies suggest such high SOC stability can be ascribed to the fact that a large (>90 %) part of total SOC was highly humified and effectively bound to the halloysite +gibbsite clays of the studied soils, a trend that deserves to be studied elsewhere.

DATA AVAILABILITY STATEMENT

All raw data can be made available upon request to the corresponding author.

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