# Effects of *Eucalyptus* and *Pinus* Forest Management on Soil Organic Carbon in Brazilian Wooded-Savanna

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Additional information is available at the end of the chapter

## Abstract

Forestry has been recommended for carbon cycle management since it promotes carbon accumulation in soils and vegetation. Soil organic carbon (SOC) is fundamental to fertility and crop production in tropical soils and its conservation is critical to sustainable land management of neotropical savannas. Thirty to forty years of *Eucalyptus* and *Pinus* forestry in original Brazilian wooded-savanna affected forest floor layers, SOC and organic matter (OM) quality. *Eucalyptus* and *Pinus* showed higher forest floor carbon stocks than natural forest plots. On the surface soil layer, plantation effects on SOC were mediated by site-dependent factors. Below 10 cm, both plantations showed lower SOC than the native forest. The relationship between carbon and clay contents was significant in subsurface soil layers, suggesting that the particulate OM pool had been depleted by plantation activities. Plantations lead to soil OM replacement to a depth of 5 cm within 30 years. The new litter and OM in the plantations had lower quality (higher C:N ratios) than in natural forests. Our results indicate that particular care must be taken when choosing forest management practices in tropical-weathered soils because they can oxidize a significant part of the SOC pool with negative consequences to soil fertility and aggregate stability.

**Keywords:** carbon stocks, organic matter, litter, forest floor, plantation, afforestation, forestry, silviculture, subtropical forest, *Cerrado*

## 1. Introduction

Forest management is an available option for climatic change mitigation through carbon cycle management [1–3]. In that context, soil carbon is decisive in the long term [2] because soils contain two to three times the amount of carbon in vegetation [4] in chemical forms that are much more stable than in biomass, with residence times extending from decades to millions of years [5] (**Figure 1**). Soil organic carbon (SOC) is defined by climate, soil type, plant cover,

decomposer activity, perturbations and management [6]. Vegetation affects SOC because, through root and leaf production, it determines soil organic input quantity and quality, which are major decomposition control factors [7]. Soil type affects SOC decomposition and stabilization through drainage, structure, texture [8], the presence and type of clays, sesquioxides and other stabilizers [9]. Management practices affect disruption and aggregation of organic matter (OM), and thus, their influence on SOC is determinant [10] (**Figure 1**).

SOC is fundamental to fertility and crop production in tropical soils. In tropical savannas, most soils have a predominance of highly weathered clays; thus, they are acidic, low-fertility soils characterized by a low cation exchange capacity, low base saturation and high Al toxicity. Because the mineral fraction is dominated by low-activity clays, SOC is especially important in these soils, where OM is the main nutrient source for plants, soil fauna and microorganisms [11, 12]. SOC also plays a fundamental role in soil aggregation, and thus, it is essential for water supply and soil structure maintenance [13]. Hence, the conservation of SOC is critical to sustainable land management of neotropical savannas [14].

The Brazilian wooded-savanna's ecological complex (*Cerrado* biome) is important to national agricultural production, and it represents the national forestry core. Brazilian forestry generates approximately 4% of the total gross national product (www.ibge.gov.br). However, there are international concerns about the environmental impacts of exotic, fast-growing tree species on tropical soils [15, 16]. A significant part of the SOC pool is extremely sensitive to soil management or disturbance, and it can be easily lost with negative consequences to soil fertility and aggregate stability [14]. SOC losses can occur under intensive forest management, driven by enhanced oxidation due to soil preparation activities [17].
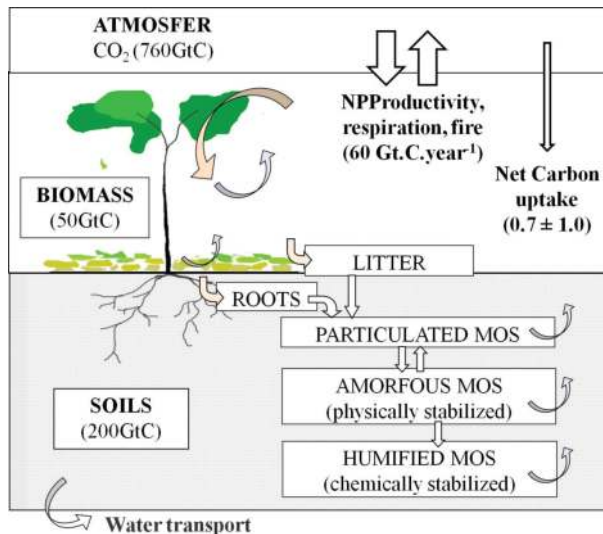


**Figure 1.** A general model of carbon in terrestrial ecosystems. Stocks are reported in Gt ($10^{15}$g) and fluxes in Gt year$^{-1}$. Note that arrows are not proportional to values. Values from [1].

Despite the importance of SOC to forest sustainable management in neotropical savannas, literature on the subject is scarce, and it mostly remains in thesis and regional papers where it is difficult to be consulted. Furthermore, available research papers conclude either that there is no effect or that contradictory effects are shown [18], as it has been recently appointed in a review of intensive logging effects on SOC [19]. The effects of *Eucalyptus* and *Pinus* forest management in SOC are not completely understood because the literature often reports contradictory results and conclusions. Although some authors have found significant SOC loss under *Eucalyptus* plantations compared to *Cerrado* [17], others found no significant change [20] or even increases in SOC [21, 22]. The same discrepancy is reported for *Pinus* stands, with depletion [14, 17, 21, 23], no change [20, 24] and increases [24], all being reported. These contradictory results can be explained by different experimental site conditions, such as soil type, stand age and tillage [17], as well as methodological differences, such as soil sampling depth.

Given the wide occurrence of forest plantations in tropical soils and the importance of SOC conservation to soil fertility, crop production and sustainable land management of neotropical savannas, we aimed to assess alterations in SOC content produced after 30 years of intensive management of *Eucalyptus* and *Pinus* on originally Brazilian wooded-savanna–covered areas. For this purpose, we have compared litter and soil C and N levels in planted and natural forests. To clarify previously reported contradictory results, we replicated our experimental work in four distant locations with the same experimental conditions.

## 2. Material and methods

### 2.1. Site description

We collected samples at four locations in the State of Sao Paulo, SE Brazil. They are approximately 100 km apart, located in the Luiz Antônio (21°61′S;47°75′W) (LZ), Mogi Guaçú (22°24′S;47°15′W) (MG), Pederneiras (22°34′S;48°89′W) (PD) and Itirapina (22°19′S;47°94′W) (IT) districts. At each location, we studied *Eucalyptus* and *Pinus* stands of similar ages as well as the nearest natural forest patches, which were considered to be the control areas. Thus, we defined 12 plots based on location and forest type (**Figure 2**).

Climate is characterized as tropical type II [25]. Climatic conditions are homogeneous among sites, with temperatures ranging between 19 and 22°C (mean annual T = 20°C) and with an annual mean precipitation of approximately 1200 mm year$^{-1}$. Rainfall is concentrated from October to March, and thus, there is a dry winter season between June and September when water demand exceeds water availability and soil moisture limits plant growth, biomass production, SOC decomposition and other ecosystem process.

Sites belong to the *Cerrado* biome, which is characterized by flat to gently undulating topography, with laterization as the main soil formation process. The studied soils were characterized as oxisols (*Humic Rhodic Haplustox*) at LZ, Inceptisols (*Oxic Dystrudepts*) at MG and PD and entisols (*Ustic Quartzipsamments*) at IT [26]. They represent a texture gradient, with average clay contents of 59% in LZ, 25% in MG, 15% in PD and 11% in IT. Altitudes range between 516 and 740 m. All sites are flat or have gentle slopes <7% and no signs of erosion were present.
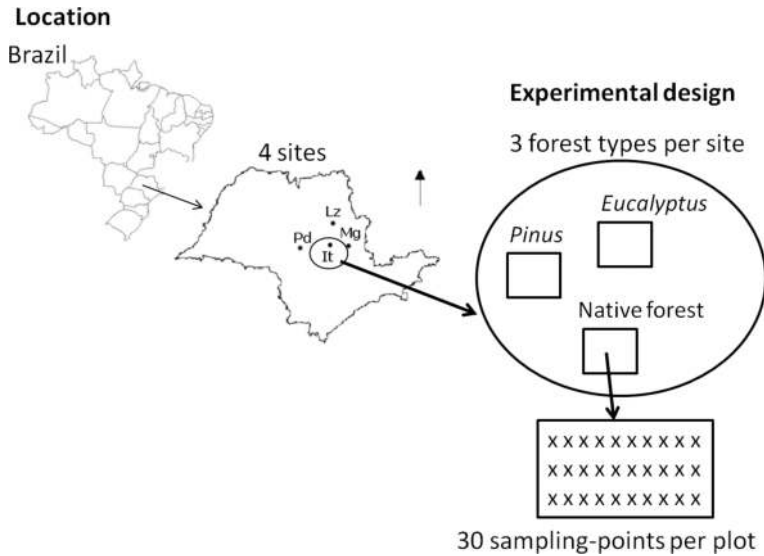
**Figure 2.** Location of the study area and experimental design.

The original *Cerrado* landscape consisted of grassland, savanna and wooded-savannas and dry seasonal forest patches. Control areas were located in remnants of Brazilian wooded-savanna (*cerradão*). They are characterized by closed canopies approximately 20 m tall. Leguminosae, Myrtaceae, Melastomataceae and Rubiaceae are the most represented families. There are no clear species dominance patterns, but some common species well-represented in the four sites are as follows: *Anadenanthera peregrina* var. *falcata* (Benth.) Altschul, *Qualea grandiflora* Mart., *Aspidosperma tomentosum* Mart., *Qualea multiflora* Mart. and *Roupala montana* Aubl.

All study stands are older than 30 years. They were planted in 1962, 1965, 1966, 1969 and 1972. Natural vegetation was first cleared, and after slash and burn of the original forest, a heavy disk plow was used to open seedling lines (~20 cm depth). Trees were planted manually. The used species were *Pinus elliottii* and *Eucalyptus citriodora* with the exception of the Itirapina site where *E. saligna* was the planted *Eucalyptus* species. No fertilizer or lime had ever been used at the sites, as they were planted for experimental purposes and for wood, not pulp, production. After 7–8 years of growth, 35% of trees are normally clear cut, which permits tree diameter to increase. The studied stands were clear cut about five times before the experimental work was performed. The length of rotation varied between 5 and 8 years according to market oscillations in the wood price, rather than planted species; therefore, there are slight differences on rotation times between sites, but they are not dependent on the planted species. At sampling time, the stands had closed canopies approximately 30 m tall.

One hundred trees per plot were measured for characterization of the vegetation. Mean basal area values ranged from 21 to 44 $m^2$ $ha^{-1}$ in the planted stands and from 23 to 43 $m^{-2}$ $ha^{-1}$ in the native forests. Vegetation densities (299–786 trees $ha^{-1}$ in plantations and 1200–1600 trees $ha^{-1}$ in native forests) and diameter distributions (70% of trees were smaller than 10-cm diameter

breast height in native forest) showed structural differences between natural and planted areas. Neither planted nor natural sites had been burnt in the last three decades.

## 2.2. Experimental design

Thirty sampling points were randomly selected at each plot (forest type × site) for soil and litter collection. The procedure was repeated four times at different locations in order to guarantee real independence of the observations (**Figure 2**). Our experimental work was then carried out following a randomized block design with sampling replication within the blocks [27], as we collected 30 samples inside each of the three forest type treatments (*Cerrado*, *Eucalyptus* and *Pinus*), and we replicated this complete design in four study locations (LZ, MG, PD and IT) corresponding to the four blocks. Sampling replication within the blocks permits us to test the block-treatment interaction. It is especially important when differences between blocks may be strong [27], as could be the present case.

## 2.3. Sampling and laboratory analysis

Collection was performed in 2004 at the end of the dry season during the maximum litter accumulation period. Forest floor samples were collected using a 25-cm$^2$ metal frame. All materials were collected, including not only litter but also fibric and humic horizons when present. Samples were oven dried to constant weight and ground for chemical analysis. C and N were determined by wet combustion [28] on 360 samples.

Soil samples were collected at three depths: 0–5, 10–25 and 35–50 cm. Pits were open to profile description and undisturbed soil sample collection. Undisturbed soil samples were collected using 5-cm diameter metal cores at four random replicates per plot. Then, soil bulk density was calculated as the oven-dry sample mass divided by the sampled volume for a total of 144 samples. Disturbed samples were collected with an auger at 30 sampling points per plot. After collection, soil samples were air dried and individually sieved through 2-mm mesh. SOC was determined by the Walkley & Black wet combustion method [29] following a tropical soil-adapted protocol [28] for a total of 600 samples. Since the studied soils are free of stones and gravel, corrections for those fractions were unnecessary. Texture was determined by the pipette method [28]. Sand (<2 mm to 64 μm), silt (<64 μm to 2 μm) and clay (<2 μm) fractions were determined for 216 samples.

Soil carbon stocks (Mg ha$^{-1}$) were calculated using bulk density (g cm$^{-3}$) and carbon content data (g kg$^{-1}$). Because we lack continuous sampling data, we used pedotransfer functions to estimate soil carbon stocks into the soil profile. Those functions, which related carbon content with soil depth or texture, can precisely calculate soil carbon stocks [30]. SOC exponentially decays with depth; therefore, most pedotransfer functions are based on the exponential model equation [22, 31, 32]. We used its more general form, which is:

$$y = a \exp(-bx), \tag{1}$$

where $y$ is carbon (kg m$^{-2}$) and $x$ is depth (cm). First, the exponential model parameters (a and b) were calculated using field data. Then, the model was integrated between the desired depths ($x_1$ and $x_2$, in cm), resulting in the function:

$$SOC_{est} = -a/b\left(\exp\left(-b\,x_1\right) - \exp\left(-b\,x_2\right)\right), \tag{2}$$

which estimates the SOC stock. We estimated the 0–30-cm depth carbon stocks following IPCC protocols [1–3].

Because the Kjeldahl acid digestion method loses accuracy when analyzing acidic, N-poor soils, we used a CN analyzer (Leco CN-2000) to determine soil C and N by dry combustion and gas chromatographic separation of 36 soil surface samples to obtain the A horizon C:N values. C:N ratios were then calculated for 720 samples; 36 samples from the soil surface (A horizon, 0–5 cm depth) and 360 from the forest floor.

### 2.4. Statistical analysis

We used general linear models to test the SOC content (g kg$^{-1}$) and C:N ratio responses to the three forest treatments (*Cerrado* considered as control, *Eucalyptus* and *Pinus*), which were considered as fixed factors, on four blocks (LZ, MG, PD and IT), taken as random factors. We used the AIC criteria to evaluate the interaction term significance [33]. Linear regression analysis was used to test the clay-SOC relationship. Data analysis and graphs were performed with R language [34].

## 3. Results and discussion

### 3.1. Forest floor

Forest floor layers under the native *Cerrado* forest consisted of fresh litter (Oi horizon) with discontinuous points of humified material at waterlogged spots. Similar conditions were found in *Eucalyptus citriodora* stands. We observed a thick organic layer in the *Eucalyptus saligna* stand, which was formed from fresh litter (Oi), fragmented debris (Oe) and a dark color–layer containing decomposed organic materials (Oa). Despite the sharpness of the transition between the organic and the A mineral horizon, this forest floor type can be considered a moder horizon because of its structure and morphological characteristics [35]. We also found clear forest floor layering under *Pinus elliottii*. Recognizable horizons were formed by freshly fallen (Oi horizon) and fragmented needles (Oe horizon). Humified material was scarce, and the transition between organic and A mineral horizons sharp; thus, these horizons are considered mor type according to Ponge's criteria [35]. Forest floor layer morphology differences between planted stands and natural forests could be explained on the basis of litter quality and soil fauna activities. High litter quality and activity of soil fauna incorporating SOC into the mineral soil lead to less organic material accumulation and most likely to faster nutrient cycling in the native forest, whereas in planted stands, lixiviation may be the principal cause of SOC incorporation into the surface A horizon, without almost any soil fauna intervention.

Forest floor organic carbon stocks were one to two times larger under the *Pinus* and *Eucalyptus* stands than under the native *Cerrado* forest (**Table 1**). Litter collected at the planted stands had higher carbon concentration and C:N ratios, leading to higher organic material accumulation

| Treatment | | Litter[a] | | | | Soil[b] |
|---|---|---|---|---|---|---|
| | | Mass (Mg ha⁻¹) | OC (g kg⁻¹) | OC (Mg ha⁻¹) | C:N | C:N |
| Ce | LZ | 8.194 (2.405) | 416.1 (28.7) | 3.523 (1.04) | 33 (3.91) | 15 (0.27) |
| Eu | LZ | 10.014 (3.129) | 440.2 (11.4) | 4.340 (1.39) | 84 (12.93) | 21 (1.13) |
| Pi | LZ | 10.322 (4.023) | 453.0 (14.2) | 4.697 (1.84) | 66 11.96) | 21 (0.63) |
| Ce | MG | 8.025 (2.699) | 429.8 (15.6) | 3.366 (1.20) | 43 (6.23) | 15 (0.60) |
| Eu | MG | 5.411 (1.664) | 438.5 (13.7) | 2.347 (0.75) | 61 4.40) | 18 (0.70) |
| Pi | MG | 8.666 (3.855) | 439.2 (25.2) | 3.948 (1.78) | 65 (9.20) | 23 (1.07) |
| Ce | PD | 8.146 (2.143) | 399.8 (26.7) | 3.158 (0.94) | 33 (3.81) | 17 (1.50) |
| Eu | PD | 11.583 (3.183) | 428.7 (30.9) | 5.010 (1.48) | 71 (12.08) | 18 (0.35) |
| Pi | PD | 11.862 (4.027) | 450.9 (11.1) | 5.318 (1.84) | 94 (12.57) | 21 (3.20) |
| Ce | IT | 10.706 (2.981) | 439.2 (15.3) | 4.731 (1.32) | 36 (4.93) | 18 (1.61) |
| Eu* | IT | 19.160 (3.334) | 474.9 (2.8) | 9.049 (1.58) | 68 (4.78) | 26 (0.83) |
| Pi | IT | 16.328 (4.298) | 420.0 (39.6) | 6.339 (1.86) | 64 (12.64) | 21 (1.18) |

Average litter dry weight mass (Mg ha⁻¹), litter organic carbon (OC) concentration (g kg⁻¹) and stocks (Mg ha⁻¹) and litter and soil A horizon (0–5 cm) C:N.

[a]Mean values of 30 samples; standard deviations are in brackets.

[b]Mean values of 3 samples; standard deviations are in brackets.

**Table 1.** Forest floor parameters under the *Cerrado* native forest (Ce), *Eucalyptus citriodora* (Eu) (*E. saligna*) and *Pinus elliottii* (Pi) stands in four locations in SE Brazil (LZ, MG, PD and IT).

than in the native forest (**Table 1**). Similar results are common in the literature: several authors reported higher forest floor accumulation [17, 36–38] and higher litter C stocks [21, 38] in *Pinus* stands than in the native *Cerrado* forest, as well as moder horizon formation [21, 37]. Higher forest floor accumulation and litter C stocks have been also reported in *Eucalyptus* stands compared to native *Cerrado* forest, although results are not as consistent as for *Pinus* [21, 22]. Less litter accumulation under *Eucalyptus* than under the *Cerrado* forest was reported; however, results were dependent on soil type because the differences were significant on loamy oxisol but not on sandy entisol [17]. We hypothesize than the observed *E. saligna* thick organic layer is due to the planted species, which is considered to be a strong forest floor accumulator in the Brazilian *Cerrado* [22] and in other areas with similar conditions in Congo [39] and South Africa [40]. However, we cannot confirm this hypothesis with our data because of the lack of replication of the *E. saligna* stand.

Forest floor accumulation depends on the input/output balance, which is controlled by litter production and decomposition [41]. It is known that *Eucalyptus* and *Pinus* stands had higher litter production than the *Cerrado* [14, 36, 37] and that this litter has lower nutritional content in our study sites [42]. These two features can explain the organic material accumulation and organic layer formation. We suggest that decomposition is strongly inhibited by *Pinus* plantations. Other authors' results support this idea; slow decomposition rates and longer residence times were reported when comparing *Pinus* with the adjacent native *Cerrado* near the study areas [36, 37].

### 3.2. Soil organic carbon

Our results show that soil carbon is related to forest type, soil depth and texture.

SOC distribution is heterogeneous in the soil profile. SOC fitted a lognormal distribution, as expected from the literature [43, 44]. SOC heterogeneity decreased from surface to sub-surface soil layers. We found variation coefficients from 18 to 58% at the 0–5-cm soil layer and from 5 to 26% in layers below 10-cm soil depth. Standard deviations increased between 10 and 25 cm and 35–50 cm layers in four sites (*Eucalyptus* and *Pinus* stands at Luiz Antônio and *Cerrado* and *Eucalyptus* stands at Mogi Guaçú) (**Table 2**). Literature reports SOC varia-tion coefficients from 5 to 59% in semiarid croplands with different land use intensities [45]. Other authors also report higher organic carbon variability in the surface than in deep layers in fast growing species plantations and natural *Cerrado* soils [22, 30]. High variability at the surface layer was expected and it can be explained by high spatial variation in surface organic carbon determinants such as soil fauna activities and litterfall. We found some samples with an extremely high C concentration that probably corresponds to local characteristics, such as charcoal, pieces of a termite mound or anthill and localized OM accumulation. Such samples were found especially in the Itirapina *Eucalyptus saligna* stand and the same situation has been reported under similar soil and vegetation conditions near our study site [22]. Therefore,

| Treatment | | Soil organic carbon (g kg⁻¹) | | | Total SOC (Mg ha⁻¹) | |
|---|---|---|---|---|---|---|
| | | 0–5 cm[a] | 10–25 cm[b] | 35–50 cm[b] | 0–30 cm[c] | $R^2$ |
| Ce | LZ | 28.41 (06.12) | 14.59 (1.27) | 12.73 (1.04) | 55.8 | 0.64 |
| Eu | LZ | 34.61 (06.33) | 13.31 (1.27) | 11.40 (1.56) | 68.3 | 0.83 |
| Pi | LZ | 43.14 (17.01) | 11.8 (0.55) | 10.64 (0.78) | 56.3 | 0.50 |
| Ce | MG | 24.67 (06.18) | 13.2 (2.66) | 10.93 (2.83) | 56.9 | 0.51 |
| Eu | MG | 11.65 (02.79) | 9.65 (2.14) | 9.13 (2.21) | 40.8 | 0.16 |
| Pi | MG | 35.45 (14.73) | 8.90 (1.20) | 7.79 (0.56) | 54.3 | 0.66 |
| Ce | PD | 30.48 (15.31) | 8.08 (1.01) | 6.34 (0.70) | 59.7 | 0.64 |
| Eu | PD | 10.83 (01.91) | 6.92 (0.97) | 6.10 (0.79) | 39.6 | 0.53 |
| Pi | PD | 12.26 (03.88) | 5.64 (0.73) | 5.58 (0.56) | 36.0 | 0.47 |
| Ce | IT | 36.86 (19.66) | 8.43 (1.20) | 6.63 (0.62) | 54.4 | 0.53 |
| Eu* | IT | 65.99 (38.28) | 7.56 (1.81) | 6.40 (1.16) | 85.7 | 0.76 |
| Pi | IT | 15.58 (06.18) | 5.41 (1.16) | 5.06 (0.87) | 33.7 | 0.40 |

The uncertainty of the stocks estimations is expressed by the pedotransfer functions adjusted coefficients ($R^2$).

[a]Mean values of 30 samples; standard deviations are in brackets.

[b]Mean values of 10 samples; standard deviations are in brackets.

[c]Estimated through pedotransfer functions based on exponential model.

**Table 2.** Soil organic carbon (SOC) concentration (g kg⁻¹) and organic carbon stocks (Mg ha⁻¹) under the *Cerrado* native forest (Ce), *Eucalyptus citriodora* (Eu) (*Eucalyptus saligna*) and *Pinus elliottii* (Pi) stands in four study locations in SE Brazil (LZ, MG, PD and IT).

they probably represent the spatial variability of the reality. However, they must be retired to perform the tests of significance because they were three standard deviations larger than mean values and their inclusion on the data set will invalidate the analyses of variance [46].

SOC content decreased as the depth increased in the soil profile (**Table 2**); 81–78% of the SOC contained 50 cm depth was concentrated on the surface soil layer (data not shown). More SOC content on the surface than on deeper layers is due to soil organic inputs being mostly superficial. Furthermore, literature reports that SOC distribution in the soil profile is well explained by indirect exponential functions in temperate [31] and tropical forest soils [32]. In this study, pedotransfer functions based on the exponential model properly fitted the observed data, with coefficients varying between 0.40 and 0.83. Similar pedotransfer function coefficients, from 0.54 to 0.73, were reported in oxisols of the Western Brazilian Amazon [32]. These results reinforce the idea that exponential models are useful for carbon stock estimations [22, 30–32]. Even continuous sampling with volumetric cores could improve the models' fit, collection at three depths is considered appropriate to SOC stocks estimation [31] and it demands significantly less field effort. The estimated SOC stocks (**Table 2**) match the values reported for *Cerrado*, *Eucalyptus* and *Pinus* plantations growing on similar soil and climatic conditions [22, 24].

Our soil bulk density values (**Table 3**) also fit the ranges reported in other studies conducted at Brazil [17, 22, 24, 47]. We found soil bulk density increments in plantations compared with natural *Cerrado* forests at every studied depth in *Eucalyptus* stands. At *Pinus* stands,

| Treatment | | Bulk density (g cm$^{-3}$) | | |
|---|---|---|---|---|
| | | 0–5 cm[a] | 10–25 cm[b] | 35–50 cm[b] |
| Ce | LZ | 0.93 (0.02) | 0.85 (0.03) | 0.94 (0.09) |
| Eu | LZ | 1.20 (0.04) | 1.00 (0.06) | 0.96 (0.05) |
| Pi | LZ | 0.89 (0.05) | 1.01 (0.06) | 0.95 (0.03) |
| Ce | MG | 1.03 (0.09) | 1.23 (0.06) | 1.12 (0.09) |
| Eu | MG | 1.27 (0.06) | 1.32 (0.06) | 1.30 (0.05) |
| Pi | MG | 1.15 (0.05) | 1.10 (0.13) | 1.03 (0.12) |
| Ce | PD | 1.22 (0.02) | 1.40 (0.04) | 1.41 (0.02) |
| Eu | PD | 1.28 (0.03) | 1.41 (0.04) | 1.44 (0.03) |
| Pi | PD | 1.28 (0.06) | 1.49 (0.02) | 1.40 (0.08) |
| Ce | IT | 0.86 (0.07) | 1.30 (0.11) | 1.36 (0.01) |
| Eu[*] | IT | 1.00 (0.16) | 1.46 (0.07) | 1.38 (0.05) |
| Pi | IT | 1.15 (0.10) | 1.42 (0.04) | 1.42 (0.06) |

Higher values in plantations suggest compaction.

[a]Mean values of 30 samples; standard deviations are in brackets.

[b]Mean values of 10 samples; standard deviations are in brackets.

**Table 3.** Soil bulk density (g cm$^{-3}$) under the *Cerrado* native forest (Ce), *Eucalyptus citriodora* (Eu) ([*]*Eucalyptus saligna*) and *Pinus elliottii* (Pi) stands in four study locations in SE Brazil (LZ, MG, PD and IT).

we found four exceptions (the 0–5-cm depth soil layer at Luiz Antônio, the 10–25-cm and 35–50-cm depth soil layers at Mogi Guaçú and the 35–50-cm depth soil layer at Pederneiras stands) (**Table 3**). Increases in bulk density under *Eucalyptus* plantation in *Cerrado* soils are also reported in other studies [17, 22] suggesting soil compaction. Soil mass rather than soil volume is conserved [48]; thus, compaction could mislead SOC stock comparisons on a soil volume basis, because greater SOC stocks could be detected as a result of soil mass increments and not necessarily by carbon enrichment or gains into the solid phase of the soil. Significant differences on SOC accumulation between *Eucalyptus* and *Cerrado* soils working on a depth but not on a mass basis support this idea [22].

We found gains as well as losses of carbon stocks into the upper 0–30-cm depth mineral soil layer working on a soil volume basis. Under *Eucalyptus* plantation, we found carbon decreases at Mogi Guaçú and Pederneiras stands, but increases at Luiz Antônio and Itirapina stands. Under *Pinus* plantation, we found decreases of carbon on mineral soil, with a unique exception into the Luiz Antônio stand (**Table 2**). However, rather than providing clarity, traditional corrections for bulk density often obscure soil carbon comparison, because the corrected values reflect differences in both carbon content and soil mass [48]; indeed, proper corrections for bulk density require estimates of C stored in an equivalent soil mass [48], but we lack continuous sampling data to properly do those estimates. Detecting relatively small changes of large compartments may still prove difficult and it has been asserted that differences on carbon stocks under different land uses are usually small when compared with the uncertainties in their measurements [19, 49, 50]. Therefore, we decided to directly use the soil mass field data to a comparative approach between forest types, because the lack of information into the 5–10-cm and 25–35-cm soil layers is an error source that will mask slight changes in SOC.

Our SOC results fit the literature-reported values [17, 24]. As expected, forest type had significant effects on SOC concentration; however, responses to forest type cannot be described straightforwardly because they were dependent on soil depth and site (**Table 4**). There were strong SOC

| Soil layer | Factor | df | F | p |
|---|---|---|---|---|
| 0–5 cm | Site | 3 | 26.286 | *** |
| | Forest type | 2 | 9.642 | *** |
| | Site×forest type | 6 | 30.395 | *** |
| 10–25 cm | Site | 3 | 125.318 | *** |
| | Forest type | 2 | 44.318 | *** |
| | Site × forest type | 6 | 1.968 | ns |
| 35–50 cm | Site | 3 | 117.654 | *** |
| | Forest type | 2 | 19.321 | *** |
| | Site × forest type | 6 | 1.626 | ns |

Soil organic carbon (SOC) content (g kg$^{-1}$) is affected by site and forest type. The significant interaction term at 0–5-cm depth soil layer indicates that the effects of forest type on surface SOC vary between sites.

*** $p < 0.001$, ns nonsignificant at the 0.05 level.

**Table 4.** General linear model results.

differences according to forest type at the soil surface layer, but the net effect varied between sites, including losses, gains and even no significant changes compared to the reference native forest (**Figure 3**). These results suggest that there are site-dependent determinant factors affecting surface layer SOC. At deeper layers, we found the same pattern in all studied stands: SOC concentration decreased under plantations and SOC levels were lower in the *Pinus* than in the *Eucalyptus* stands (**Figure 3**). At the *Eucalyptus* plantations, mean SOC concentration had decreased about 15% at the 10–25-cm soil layer and 10% at the 35–50-cm layer, while, under the *Pinus* plantations, it had decreased about 28% at the 35–50-cm layer and 21% at the 35–50-cm layer (**Table 5**).

Meta-analysis results support our findings; soil carbon stocks decline about 13% after natural forests to plantation conversion [51]. Nevertheless, the net effect depends on the type of planted species; broad tree plantation placed onto prior native forests or pastures did not affect the SOC stock, whereas pine plantations reduced SOC stocks about 12–15% [51]. Moreover, most of the soil carbon was lost under softwoods plantations (particularly *Pinus radiata*), while SOC accumulation was greater under deciduous hardwood or N-fixers plantations, following afforestation [52]. Although a recent review has shown consistent long-term (>300 years) decreases in SOC when primary forests are logged and harvested, it has concluded that previous results indicating lack of depletion of SOC in mineral soils subjected to harvest may have been a function of their short time frames [19]. Indeed, the effects of *Eucalyptus* afforestation on SOC get stronger with plantation age [18]. Furthermore, *Eucalyptus* effects could be related to mean annual precipitation, since SOC gains in drier sites and SOC depletions in wetter ones had been reported [18].

Our results suggest that clay content regulates SOC responses to forest management. Surface SOC concentration changes in *Eucalyptus* stands were more pronounced in sandy than in clay soils. The higher the clay content, the weaker the SOC changes in *Eucalyptus* stands at the soil surface (**Table 5**). At deeper layers, SOC could be related to clay content because it accompanies the texture gradient below a 10-cm soil depth (**Figure 3**). Similar results were reported in the same biome [17]. In tropical sandy soils, there is almost no SOC-clay adsorption; thus, the adsorption of carbon compounds released during decomposition is very weak, and SOC is more susceptible to being lost by lixiviation than in clay soils. Usually, clay soils show higher SOC content than sandy soils, and they are more difficult to modify through forest management. It can be explained by management practices, which have a greater impact on the soil OM associated with the sand fraction than on the fractions bound to clay and silt [53]. Furthermore, vegetal residues usually decompose more rapidly in sandy than in clay soils [52].

Our contrasting surface SOC results could be explained by tillage and/or soil preparation differences between sites at the initial plantation time, as no evident differences exist in climate, main soil formation processes, stand age and management practices, and differences in clay content are not related to SOC in the surface layer (see Section 3.3). Small differences in soil tillage and management practices at soil preparation time can generate significant losses of SOC, and the patterns of loss and accumulation of SOC strongly vary according to location [54]. More than 30% of the forestland and 50% of the grassland surface SOC pool variation were attributed to site variables in Ohio in the Great Lakes region of the USA [55]. Different patterns of surface SOC dynamics at each study site in *Eucalyptus* and *Pinus* stands on *Cerrado* soils were also reported in a previous study [17]. It concluded that the dynamics of soil OM in these stands depends on a set of environmental conditions and management practices that
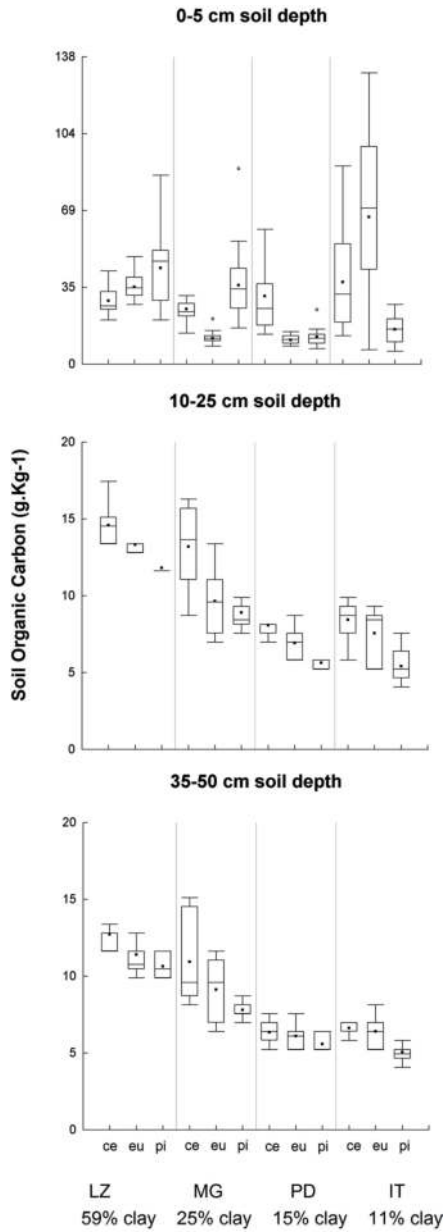
**Figure 3.** Soil organic carbon concentration (g kg$^{-1}$) as function of forest type in three soil layers at four studied locations in SE Brazil (LZ, MG, PD and IT). "Ce" depict data for *Cerrado* native forest; "eu" for *Eucalyptus* stands; "pi" for *Pinus* stands. Average soil clay content is reported for every location. Graph scale differs between surface and lower layers to facilitate interpretation. Effects of fast-growing species plantation on surface SOC depends on the site. Below 10-cm soil depth, plantations lead to SOC depletion. General linear model results are reported in **Table 4** and medium SOC changes in **Table 5**.

| Treatment | Soil layer | | | | | |
|---|---|---|---|---|---|---|
| | 0–5 cm | | | | 10–25 cm | 35–50 cm |
| | LZ (59%) | MG (25%) | PD (15%) | IT (11%) | All sites | All sites |
| *Eucalyptus* | 21.8% [ns] | −52.8% | −64.4% | 79.0% | −15.5% | −9.8% |
| *Pinus* | 51.8% | 43.7% | −59.8% | −57.7% | −28.3% | −20.6% |

Negative values indicate decreases and positive values increments. Mean site clay contents are reported in brackets.[ns]Nonsignificant at the 0.05 level.

**Table 5.** Average soil organic carbon (SOC) concentration changes according to forest type in different soil layers in four study sites in SE Brazil (LZ, MG, PD and IT).

cannot be expected to follow an obvious and simple general tendency, particularly at surface layers where SOC is very dynamic [17]. Initial decreases in SOC after pasture to plantation conversion has been also observed and they are often attributed to site preparation [52]; however, the same study suggested that the lack of inputs into the soil (because lack of vegetation in the first years after conversion) rather than the soil disturbance during site preparation is responsible for the observed decreases.

Site preparation activities could be responsible for SOC losses at our sites. At plantation time, burning harvest residue for site preparation was the common practice and then, the SOC decreases founded in planted stands below 10-cm soil depth may be related to enhance SOC oxidation at plantation time. Activities carried out during site preparation, such as natural vegetation clearing and plowing (up to ~20-cm depth) probably lead to net SOC losses because they increase aggregates disruption and aeration as well as increase the availability of native labile organic carbon for decomposers. The particulate SOC pool, which is very sensitive to management [56], could be easily oxidized at plantation time due to soil preparation activities. The SOC pool associated with clays, which is more resistant to disruption, could be retained in the mineral soil, forming 60–70% of the actual carbon pool in the studied soils (see Section 3.3). Species richness itself could partially explain higher carbon contents in the subsoil of *Cerrado* forest than in plantations, since the carbon concentration per unit clay or fine silt in the subsoil has been found to be 30–35% higher in mixed than in monospecific stands in natural broad-leaved forests in Germany [57]. Other literature has reported SOC depletion in *Eucalyptus* and *Pinus* [14, 17, 21, 23] stands compared with the native *Cerrado* forest. Conservative practices are important at this point; for example, the adoption of minimum tillage in soil preparation in Brazil has implied significant reductions of SOC losses under plantations in the last decade [47]. Forest management is highly relevant in tropical soils where 20–40% of total SOC is related to particulate OM [8], which is the most sensitive pool to decomposition losses, and it is mainly controlled by management [56]. Not only coarse particulate SOC, but also the fine occluded fraction could have been oxidized, even in the B horizons where SOC may be less stable than is often thought [58]. Despite the fact that tropical forest plantations are often supposed to, and even designed to, sequester SOC, our study shows that significant losses still occur. However, these losses may be reduced through less intensive practices of soil preparation, such as reduced or no tillage [6, 10, 14, 17].

## 3.3. SOC-clay relationship

The SOC-clay equations fitted linear regression models; however, the relationship differed with soil depth. Clay content explained 62% of the SOC content at the 10–25 cm depth and 75% at the 35–50 cm depth, but clay and SOC levels were not related at the surface layer (0–5 cm depth) (**Table 6**). The SOC-clay direct relationship can be explained by SOC stabilization. Carbon is adsorbed on clay surface exchange sites, where it is protected from decomposition, lixiviation and water transport losses. Mineral fractions <20 μm are responsible for SOC physical protection because of its occlusion into microaggregates [56]. Thus, the more abundant the clay, the more protected the SOC.

In Brazilian oxisols, clay content is considered as a major controlling factor of slow SOC cycling [59] and SOC accumulation is often higher in clay than in sandy soils [17, 60]. Therefore, clay content has a strong influence on soil carbon dynamics and storage in this type of soils. Although experimental results do not always confirm the linear relationship between carbon and fine soil mineral particles [61, 62], several studies support significant relationships either with clay [8, 20] or with clay + silt [9, 30, 43, 57]. The linear relationship is related to the number of adsorption sites on the clay mineral surface per unit soil weight or volume. This linear relationship has been also reported in soils dominated by low activity clays [8]. Other authors found strong texture effects on SOC in shallow and deep soil layers but not at the surface layer [30, 57, 63]. The unclear textural effect at the surface layer found in the *Cerrado* biome has been related to the high particulate OM content in this layer, which would mask the interaction of humic carbon with the soil mineral matrix [30]. This explanation is plausible for the null relationship found in the surface layer in our study.

The SOC differences between the surface and the lower layers may be due to the surface SOC pool mostly containing labile forms that originated from particulate OM, which is composed of organic fragments up to 20–50 μm [8]. Below a 10-cm soil depth, the SOC pool may contain more stable forms of 20 μm or smaller in size that could be stabilized through clay association. However, the relationship between physical protection and chemical quality is not as simple. SOC recalcitrance decreases with depth have been found in Mediterranean forest soils [58]. The authors found that recognizable plant fragments constituting the free-light SOC fraction were not necessarily the youngest fraction and that the nondecomposed fraction of SOC presented intermediate degrees of recalcitrance [9].

| Soil layer | Linear regression analysis | | |
|---|---|---|---|
| | **Adjusted model** | **$R^2$** | **p** |
| 0–5 cm | | | ns |
| 10–25 cm | $SOC = 5.96 + 0.012\ Clay$ | 0.62 | *** |
| 35–50 cm | $SOC = 4.74 + 0.011\ Clay$ | 0.75 | *** |

The SOC-clay relationship varies with depth, being nonsignificant at the surface layer, while clay content explains 60–75% of SOC content below 10-cm soil depth.

[***]p < 0.001, [ns] nonsignificant at the 0.05 level.

**Table 6.** Soil organic carbon (SOC) and clay contents (g kg$^{-1}$) linear regression analysis: adjusted models, regression coefficients ($R^2$) and significance level (p).

### 3.4. Organic matter quality: C:N ratios

We found higher C:N ratios in *Eucalyptus* and *Pinus* stands than in natural forests at the forest floor as well as into the surface A mineral horizon soil (**Figure 4**). Our data (**Table 1**) are in the range generally reported for tropical areas and forest [7, 64]. Commercial forest values are higher than those reported in the literature for the region [24, 37]. This discrepancy can be explained by the fertilization differences in stands managed for pulp or for timber production. *Eucalyptus* and *Pinus* stands increased mean C:N values about 1.9 times in the litter and 1.3 times in the A mineral soil (**Table 1**). OM C:N ratio increases are often related to the reduction in SOC quality, and they may have implications for nutrient cycling and fertility [64], especially in tropical soils where biotic factors have strong influences on decomposition dynamics [7].

*Eucalyptus* and *Pinus* are high nutrient-use efficient species [36, 37]; they produce high biomass growing in oligotrophic soils, but their leaves have low nitrogen concentrations. We know from a previous study that, at our study sites, litter nutrient content is significantly lower in planted stands than in the control native forests [42]. We suggest that this low-quality litter production leads to SOC quality degradation, which occurs gradually with litterfall increase during plantation development. New litter and the organic substances derived from decomposition are gradually incorporated into mineral soil where they will slowly replace the original SOC. The literature reports SOC replacement in *Cerrado* soils under *Eucalyptus* and *Pinus* culture [22, 38] and it reports higher soil C:N ratios in eight-year-old *Eucalyptus* and
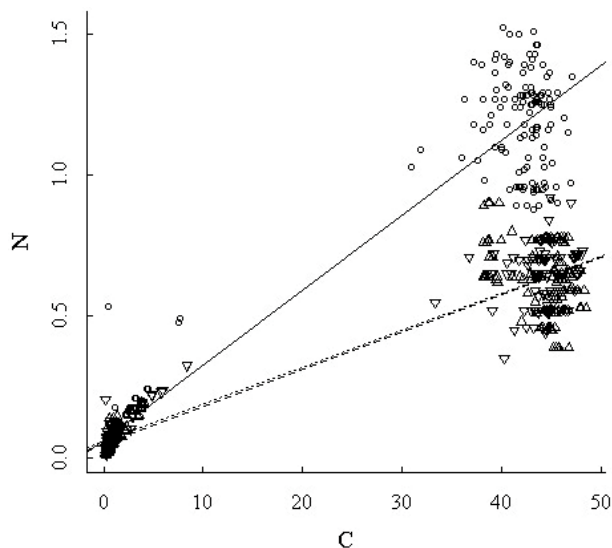


**Figure 4.** Organic matter C:N ratio as a function of forest type (circles and solid line depict data for *Cerrado* native Forest; triangles pointing down and dotted line for *Eucalyptus* stands; and triangles pointing up and dashed line for *Pinus* stands). Forest floor (higher C values on the right side of the plot) and upper A mineral horizons (lower C values on the left side of the plot) clearly differ in C and N contents. The increasing adjusted line slope indicates low C:N ratios. Plantations lead to significant C:N increases in the forest floor and mineral A horizons (p < 0.001).

*Pinus* stands than in the native forest in the same study area [20]. However, the reported differences were lower than ours, suggesting continuous longer term SOC replacement over time.

Our results indicate that SOC replacement occurred in 30 years in the A horizon to a depth of 5 cm in *Cerrado* soils transformed into *Eucalyptus* and *Pinus* plantations. Despite the fact that high litter C:N ratios could lead to the inhibition of decomposition [7], part of the decomposed OM is being incorporated into the mineral soil, as is shown by our topsoil C:N results. Changes in SOC C:N ratios may decrease the rates of nutrient cycling and affect the decomposer community, as indicated by diversity losses reported for various groups of soil fauna in SE Brazilian stands [65, 66].

## 4. Conclusions

*Eucalyptus* and *Pinus* plantations in the Brazilian wooded-savanna (*Cerrado*) lead to OM accumulation in the forest floor and higher litter carbon stocks, especially in *Pinus* stands, where mor humus-type was formed. *Eucalyptus* and *Pinus* litter showed high C:N ratios inducing OM accumulation compared to the native forest.

*Eucalyptus* and *Pinus* plantations significantly affected SOC concentration. In the surface soil layer, the effect on SOC was mediated by site-dependent factors, and a general pattern could not be identified. Gains, losses or no carbon changes with respect to the reference native *Cerrado* forest were observed at different sites under the same type of forestry. In lower layers (below a 10-cm soil depth), *Eucalyptus* and *Pinus* plantations lead to decreases in SOC concentration. In these subsoil soil layers, SOC was strongly related to clay content, suggesting that the particulate SOC pool was specifically oxidized by the soil preparation activities in plantations.

*Eucalyptus* and *Pinus* plantations lead to OM alterations in the forest floor and in the A mineral horizon. Plantations produced partial surface SOC replacement (down to a 5-cm soil depth) over 30 years. The new SOC has a higher C:N ratio that may cause SOC quality degradation.

According to our results, forest management practices may have strong implications for SOC pools that may offset the carbon biomass accumulation potential of plantations of fast-growing species, thus limiting their role in C sequestration and climate change mitigation. These implications are particularly critical in the case of the substitution of native forests by artificial plantations considering the possible negative consequences for biodiversity conservation.

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