Carbon Balance in Soils under Conifers and Broadleaved Species within La Sierra, Dominican Republic

Santiago W. Bueno-López [1,*], Luis R. Caraballo-Rojas [1] and Esclaudys Pérez-Gonzales [2]

1   Vice-Rectory of Research and Innovation, Silvicultural, Environmental and Climate Change Research Program, Pontificia Universidad Católica Madre y Maestra, Santiago de los Caballeros 51043, Dominican Republic; lcaraballo@pucmm.edu.do
2   Department of Natural Resources, Universidad ISA, La Herradura, Santiago de los Caballeros 51000, Dominican Republic; eperez@isa.edu.do
*   Correspondence: swbueno@gmail.com; Tel.: +1-917-349-5455

Abstract: Our research assesses the effects of four forest species, namely, *Swietenia macrophylla* King, *Swietenia mahagoni* (L.) Jack., *Pinus occidentalis* Swartz, and *Pinus caribaea* Morelet var. Caribaea, on the soil and litter organic carbon (C) stocks, C dioxide equivalent balance (BCO_2 Eq.) diurnal, and periodic dynamics beneath these species. Reforestation projects in the study region cover 1200, 543, 770, and 1152 hectares, respectively, with these four species being the most relevant in reforestation projects within the country. To determine the BCO_2 Eq. per unit area, we compared the greenhouse gas (GHG) fluxes of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) expressed as CO_2 Eq. units with the organic C stocks found in the mineral soil to a depth of 30 cm and in the forest litter. In four measurement periods over 18 months, we conducted field measurements in sixteen stands, four per species. Our results indicate that *S. mahagoni* emitted the lowest CO_2 Eq., while *S. macrophylla* released the highest amount into the atmosphere. At the end of the 18 months, BCO_2 Eq. from *S. macrophylla* soils was 299.70 metric tons ha^{-1} $year^{-1}$, while for *P. occidentalis*, *P. caribaea*, and *S. mahagoni*, the corresponding quantities were 103.64, 146.41, and 72.34, respectively. All species showed a general upward pattern in soil respiration from September 2020 to March 2022. The average CO_2 Eq. flux rates to the atmosphere were approximately 65.4, 51.1, and 75.9 percent higher in *S. macrophylla* soils compared to the respective rates of *P. occidentalis*, *P. caribaea*, and *S. mahagoni*.

Keywords: carbon dioxide equivalent; CO_2 equivalent fluxes; forest soils; organic carbon stocks; terrestrial ecosystems

1. Introduction

"Soil Carbon Balance" refers to the amount of C stored or released from soil over time through organic matter input and decomposition [1]. Forest soils are a significant C reservoir [2]. When organic matter input exceeds decomposition, soil acts as a C sink, mitigating climate change [3]. However, when losses exceed input, the soil becomes a source of C, contributing to global warming. Deforestation, degradation, or poor management practices can disturb the soil C balance, releasing net C into the atmosphere [4].

Soil organic carbon (SOC) sequestration is affected by various factors such as the climate, soil type, tree species, soil management, and chemical composition of soil organic matter [4,5]. The dominant tree species determines these factors. Understanding the interactions of SCB with these factors is crucial for sustainable land use and mitigating the effects of climate change.

Chemical elements also interact with SOC in complex ways influenced by soil type, climate, land use, and management practices [1]. Balanced nutrient inputs optimize SOC levels, enhancing soil health and productivity [1]. Adequate nitrogen availability can stimulate microbial activity, which helps in the decomposition of organic matter and the formation of stable SOC compounds. Similarly, phosphorus availability and management

can significantly influence SOC through its effects on plant growth, microbial activity, and soil structure [2].

Calcium, magnesium, and potassium support microbial communities that stabilize SOC, while nitrogen, phosphorus, and potassium improve plant growth and organic matter inputs [3]. All these elements play critical roles in SOC dynamics [4], enhancing soil structure, aiding microbial activity, and stabilizing SOC. Iron forms stable complexes with organic matter, while high aluminum levels can reduce plant and microbial growth, lowering SOC [5]. Trace elements such as copper, zinc, and manganese influence organic matter decomposition and SOC stabilization [6]. The soil pH, cation exchange capacity, and texture significantly impact SOC levels [1]. An optimal pH supports plant growth and microbial activity, while a higher CEC and clay content improve organic matter retention and SOC stabilization [4].

The accumulation of soil C is also influenced by tree species productivity, leaf litter quality and quantity, nitrogen, and C deposition [7–9]. Litter and the underground necromass are the main contributors to soil C, with varying capacities among forest species [10]. Soil C dynamics can be modified through species selection, thinning, harvesting, and fertilization [11]. Annual leaf-fall and herbaceous material contribute the most C to the soil. Reforestation of former cropland generally increases soil C stocks, while former grasslands and peatlands may not show significant changes or could even experience a decrease in soil C stocks [11].

Understanding soil carbon dynamics and its role in mitigating global warming is fundamental. However, in tropical regions, studies on soil C are relatively scarce compared to temperate soils. The lack of data poses a problem because below-ground measurements are needed to create complete C budgets for terrestrial ecosystems at local, regional, and global scales [12]. Soil C balances help identify how and where C is being stored, as well as how C storage may change as forests recover from past disturbances or transition into different forest types due to climate changes [13]. Tree species differences in productivity, litter quality and quantity, canopy structure, and nitrogen deposition are key to understanding their contributions to the C budgets inherent to their forest ecosystems [9]. Measuring the changes in soil C is challenging due to the high spatial variability and slow accumulation processes [14]. Soil respiration to the atmosphere, a component of C loss, remains one of the least understood aspects of the terrestrial C cycle. Both components are influenced by numerous factors that vary significantly in time and space, leading to imprecise reporting estimates for forests [14].

Forest ecosystems capture and store large amounts of carbon (C) more effectively than any other land use [15]. Soils in these ecosystems serve as crucial C sinks, absorbing CO_2 and storing it as soil organic matter [16], making it essential to acquire accurate and comparable data on soil C stocks and GHG emissions. Forest soils hold the largest terrestrial reserve of C globally [17]. Still, their C storage capacity is declining at an estimated annual loss of 75,000 million tons of soil globally [18].

In natural forests, soil C is usually in equilibrium, but deforestation or reforestation disrupts this balance. Each year, an estimated 15 to 17 million hectares are deforested, primarily in the tropics [18]. These activities often result in the loss of organic C, leading to significant CO_2 emissions.

The aim of this study was to (1) measure SOC stocks in forest litter and mineral soil up to a depth of 30 cm and quantify the fluxes of the three most significant GHGs (CO_2, CH_4, and N_2O); (2) compare these fluxes and stocks expressed in CO_2 Eq. units to determine the BCO_2 Eq. per unit area and estimate its magnitude; (3) examine the temporal dynamics in CO_2 Eq. stocks and temporal and diurnal dynamics in fluxes; and (4) assess the effects of four tree species (*Swietenia macrophylla* King, *Swietenia mahagoni* (L.) Jack., *Pinus occidentalis* Swartz, and *Pinus caribaea* Morelet var. Caribaea) on these variables. We hypothesize that organic C stocks in soils under broadleaved species are greater than under coniferous species.

2. Materials and Methods

The data collected for this study come from sixteen stands, four each for the species *Swietenia macrophylla* King, *Swietenia mahagoni* (L.) Jack., *Pinus occidentalis* Swartz, and *Pinus caribaea* Morelet var. Caribaea. *S. macrophylla*, *S. mahagoni*, *P. occidentalis*, and *P. caribaea* are located in La Sierra, Dominican Republic, with ages between 8 and 40 years (Figure 1). La Sierra is located between UTM coordinates 251,748 m E–325,795 m E and 2,116,888 m N–2,156,996 m N. It has an area of 1800 km^2, where slopes range from zero to 70 percent; the altitude above sea level varies from 400 m to 1600 m; the average annual temperature is 24 °C, with a variation between the maximum and minimum of less than 10 °C; and the average annual precipitation range is between 800 and 1600 mm [19].
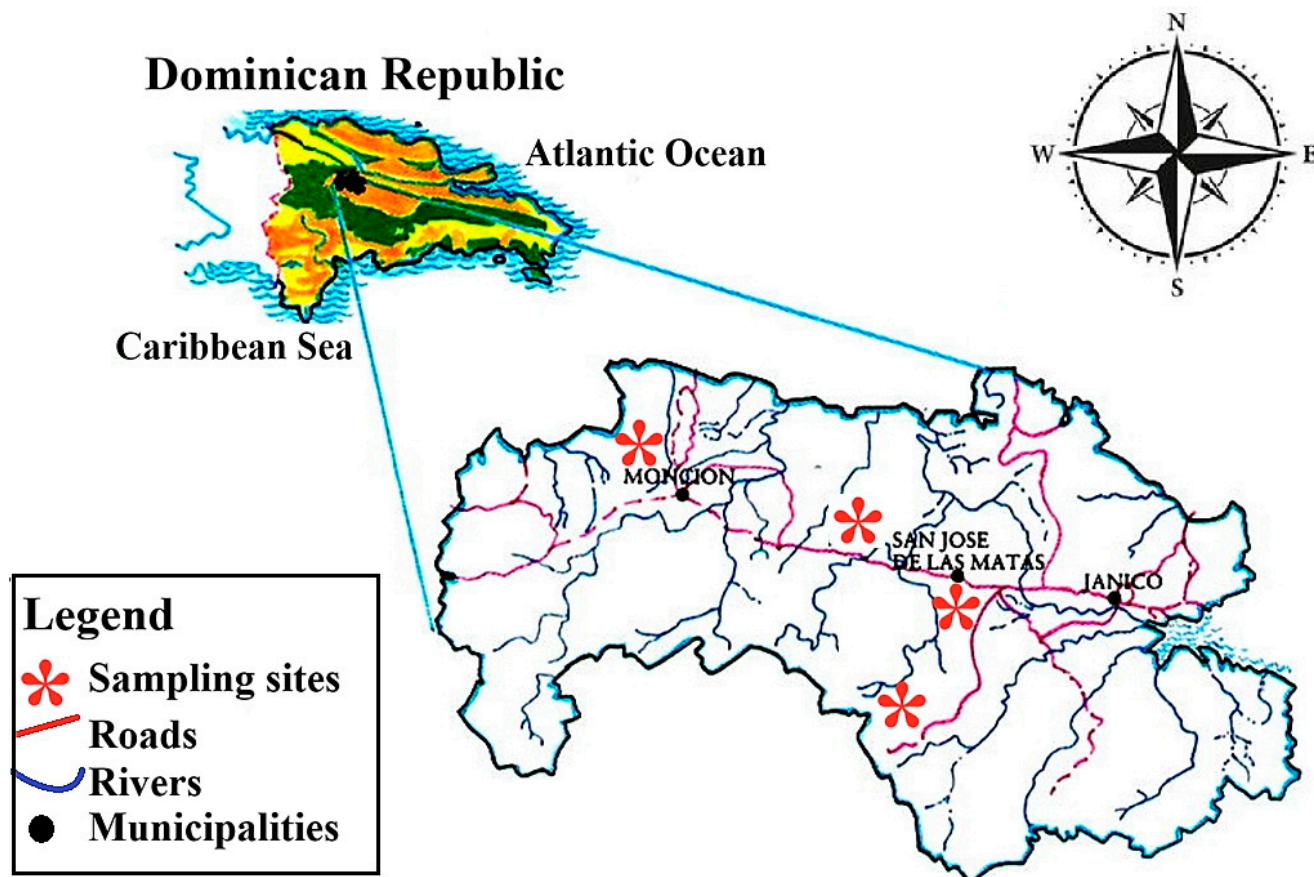
Figure 1. Distribution of sampling units (forest stands) for *P. caribaea*, *P. occidentalis*, *S. macrophylla*, and *S. mahagoni* within La Sierra, Dominican Republic.

The sampling units (SUs) are over igneous and metamorphic rock [20], primarily derived from basalt, gabbro, and shale and, to a lesser degree, quartz sandstone in the northern edge of the study zone. The topography is hilly to very hilly. The World Reference Base for Soil Resources does not provide a comprehensive and unique classification of the soil characteristics and features in the Dominican Republic. Still, soils in the study area could be classified as Leptosols [21]. Based on the Soil Taxonomy System (US Soil Conservation Service), they correspond to Entick Hapludolls, typical Ustorthents, and typical Haplustolls [22]. Soils in the study area have minimal horizon development, having mostly soil horizon sequences of Oi-Ab-Et-R, Oi-Ag-R, and Ap-Bw-R [22].

Table 1 includes basic information about the above-ground forest cover characteristics for each of the 16 stands studied, including the tree height, normal diameter (at breast height), stand density, and stand age. Understory species were scarce in the forest stands evaluated. In *S. macrophylla* and *S. mahagoni*, the understory contained the herbaceous species *Cestrum* spp., *Capraria biflora*, *Eugenia monticola*, and *Gynerium sagittatum*. Conifer

stands contained scarce herbaceous species (*Melinis minutiflora* P. Beauv., *Parthenium hysterophorus* L., and *Cyathea* spp.). In the westernmost location of the study zone, stands of *P. caribaea* and *P. occidentalis* contained some small shrubs (*Eysenhardtia polystachya* (Ortega) Sarg. and *Pithecellobium unguis-cati* L. Benth).

Table 1. Summary of the forest inventory in the 16 sampling units, including tree height (H), normal diameter at breast height (d.b.h.), stand density (trees per hectare, N), and stand age.

Forest Stand	Species	Age	d.b.h.	H	Stand Density
(No.)		(years)	(cm)	(m)	(N)
1		12	15.06	10.50	486
2	*S. macrophylla*	12	17.00	13.00	1171
3		8	11.85	9.75	743
4		12	17.96	13.25	657
1		25	13.71	14.25	400
2	*P. occidentalis*	30	18.25	15.50	457
3		35	30.25	23.00	229
4		40	25.21	19.00	543
1		18	17.52	19.25	657
2	*P. caribaea*	25	24.31	22.50	914
3		28	31.50	23.25	171
4		38	26.00	24.75	600
1		30	19.05	11.33	600
2	*S. mahagoni*	29	14.75	9.00	1657
3		40	37.64	17.00	286
4		40	14.96	9.75	686

2.1. Soil Sampling to Confirm Parent Material and Physicochemical Properties of the Soil

Temporary plots of 350 m^2 were deployed in each of the sixteen SUs. Litter and mineral soil samples to a depth of 30 cm were carefully packed in sealed plastic bags, labeled, and registered in a database for further laboratory analysis. Independent soil samples were taken at the beginning and end of the study to assess physicochemical properties. Additional independent samples taken at 30 and 50 cm depth in each SU were used to determine soil texture and verify similar edaphic parental material for the species.

The following methods were used for the physicochemical properties and texture determination by Junta Agroempresarial Dominicana and Ward Laboratories in Kearney, NE, USA (http://www.wardlab.com, accessed on 5 September 2024): pH in water (Extractor solution with KCl1N) [23]; Extractable acidity-Al + H+ (BaCl2-TEA, pH 8.2) [24]; Electrical conductivity (Saturation extract) [25]; Organic matter (Loss on Ignition) [26]; Nitrogen (Combustion) [27]; Phosphorus, Extractable Ca, Extractable Mg, and Extractable K (Mehlich 3-ICP) [28]; Cation Exchange Capacity, Ca/Mg, Ca/K, Mg/K, and (Ca + Mg)/K (Summation) [29]; Ca Saturation, Mg Saturation, K Saturation, and Al Saturation (Base Saturation by CEC–8.2: Sum of Cations) [24]; Copper and Zinc (EPA Method 3050B/3051 + 6010) [30]; Manganese and Iron (Mass Spectrophotometry) [24]; Clay, Silt, and Sand (Particle size analysis) [31]; and Texture (Soil Texture Triangle) [24].

2.2. Determination of OC Reserves in Litter

Throughout the study, we collected litter and small branches from the soil surface in each SU during four periods (September 2020, April and October 2021, and March 2022). Within each, we randomly selected three 1 m^2 plots for collection, resulting in 64 composite samples (CSs). We disregarded larger branches, weeds, and grass. All the material from the three plots was weighed in the field, completely mixed, and considered as litter biomass in field conditions.

The selected sample selection period did not follow a discernible pattern. Regretfully, the forest stands studied were located on privately owned land with limited access due

to ownership rights. Our observational study was hindered by many forest owners' busy schedules and overseas residences, making it challenging to access these properties within appropriately set time frames.

From each of the 64 CSs, we collected 2 kg subsamples, placed and sealed them in plastic bags, and transported them to our laboratory for dry biomass determination. Subsamples were dried at 110 °C for 24 h until they reached a constant weight. The relationship between the fresh weight and oven-dried weight was determined. Two oven-dried subsamples from each CS were sent to Ward Laboratories in Nebraska, USA (http://www.wardlab.com, accessed on 5 September 2024), for OC determination by infrared detection [32]. Organic C stocks were calculated for each of the 128 subsamples by factoring in the litter biomass under field conditions, oven-dried weight/fresh weight ratio, and OC concentration as:

$$OCR_{litter} = LBFC \times \frac{ODWL}{GWL} \times \frac{OCC}{100}$$

where

OCR_{litter}: OC stocks in litter (t. ha^{-1});
$LBFC$: litter biomass in field conditions (t ha^{-1});
$ODWL$: oven–dried weight of CS;;
GWL: green weight of CS;
OCC: OC concentration (%).

2.3. Determination of SOC Reserves in Mineral Soil

During four evaluation periods, mineral soil was randomly sampled at three depths (0–10 cm, 10–20 cm, and 20–30 cm) in each SU to estimate SOC. A composite sample (CS) was formed for each specific depth, resulting in 192 samples. Each collection of samples was conducted with a minimum distance of 1 m between each sampling point to avoid proximity to previously sampled areas. The SOC concentration was determined by the Junta Agroempresarial Dominicana (JAD) soil laboratory, using the Walkley–Black method [33]. Soil samples were dried at 50 °C and ground in a mechanical mortar and pestle. Finally, they were passed through a 2 mm sieve screen.

Measurements of soil bulk density (BD) in each of the 16 SUs in the four evaluation periods were made by collecting three independent soil samples at each soil depth being evaluated, using an AMS sliding hammer ($\varnothing$ interior = 4.8 cm; V = 182.77 cm^3). There were no conspicuous stones or rock fragments in the first 30 cm of the soil of the sampled areas, as confirmed by visually analyzing soil profile pits in the study area, built while preliminarily exploring the forest stands studied. This can be attributed to the past intensive agricultural practices practiced before afforestation that effectively removed such materials [34]. A total of 192 samples were processed in the laboratory, and the BD was determined by the fresh volume and oven-dried weight (110 °C for 24 h) ratios.

The OC concentration from the 192 CSs was reported as a percentage. This percentage was transformed into concentration units (g C kg^{-1}) and converted into a significant estimate by multiplying this concentration by the BD (kg. m^{-3}) and the volume of soil contained in one hectare (10,000 m^2) and the depth of each layer evaluated (0.10 m), to obtain SOC stocks (t. CO ha^{-1}). SOC reserves were calculated for each fixed depth of 0.10 m as:

$$SOC_{mineral\ soil} = BD \times OCC \times VS_{(i)}$$

where

$SOC_{mineral\ soil}$: SOC reserves (g CO ha^{-1}), converted to t CO ha^{-1}, multiplying by 10^6 factor (amount of g in one metric ton);
BD: Soil Bulk Density (kg. m^{-3});
OCC: Soil OC Concentration (g C kg^{-1});
$VS_{(i)}$: Soil volume in one hectare with 0.10 m thickness (1000 m^3);
i: 0.10 m = Thickness of each layer evaluated.

2.4. CO_2 Equivalent Fluxes from the Soil

Throughout all four measurement periods, we monitored CO_2-equivalent (CO_2 Eq.) fluxes from the ground using a G2508® spectrometer (Picarro Inc. in Sunnyvale, CA, USA), which can simultaneously measure CO_2, N_2O, and CH_4 fluxes. The spectrometer is coupled to a multiplexer and three automatic chambers (Eosense Environmental Gas Monitoring, Dartmouth, NS, Canada). The system's integrated software records the gases every 10 min. Fluxes can be recorded in multiple units, including t. ha^{-1} year^{-1}.

Over 10 h, CO_2-Eq. fluxes were measured in the 16 different SUs. A total of 2587 measurements were taken and averaged to obtain 640 hourly averages (10 h × 4 stands × 4 species × 4 periods). To express fluxes in CO_2 Eq. units, the CO_2, N_2O, and CH_4 fluxes were converted according to IPCC standards for each gas (CO_2 = 1; N_2O = 298; and CH_4 = 24) over a 100-year time horizon [35]. We also considered the molar mass ratio of C, obtained by dividing the CO_2 molar mass = (12.0107 + (15.9994 × 2) = 44.0095 g/mol by the molar mass of C = 12.0107, equaling 3.67.

Along with the GHG flux measurements, we also measured the "In Situ" soil temperature (°C) and humidity (%); these variables have been described as having a major influence on GHG fluxes [36]. Measurements for these two variables were taken every half hour during the 10 diurnal hours of assessment. Soil temperature was measured with a Hanna Digital HI98501® (Smithfield, RI, USA) thermometer with ±0.2 °C accuracy and a penetration of 10.6 cm. Soil humidity was measured to a depth of 76.2 cm with an Aquaterr Model 300® (0%–100% saturation), Aquaterr Instruments and Automation, LLC, Costa Mesa, CA, USA.

2.5. Carbon Dioxide Equivalent Balance

The BCO_2 Eq. for each SU was calculated by comparing data from organic C stocks (t. CO_2 Eq. ha^{-1}) and flux outputs (t. CO_2 Eq. ha^{-1} year^{-1}). The ecosystem boundary considered for our BCO_2 Eq. assessment included litter in the soil surface and mineral soil to a 30 cm depth. Specific components were excluded from flux estimation, as they had a negligible impact on the loss of organic C from the soil. These components are as follows: non-CO_2 losses, such as C monoxide, fluxes of volatile organic compounds, and herbivores.

To calculate BCO_2 Eq., we used a commonly recognized empirical model for predicting changes in SOC stocks [37], where negative values indicate a decrease in atmospheric C. The model is provided below to determine the change in C reserves.

$$\Delta CO_2\ Equivalent = \frac{(RC_0 - FC_0) - (RC_T - FC_T)}{T}$$

where

$\Delta CO_2\ Equivalent$ = Periodic change in SOC reserves (t CO_2 Eq. ha^{-1});
RC_0 = SOC reserves at moment 0 (t CO_2 Eq. ha^{-1});
FC_0 = CO_2 Eq. fluxes at moment 0 (t ha^{-1} year^{-1});
RC_T = SOC reserves at moment T (t CO_2 Eq. ha^{-1});
FC_T = CO_2 Eq. fluxes at moment T (t ha^{-1} year^{-1});
T = Time (years) between first and last assessment of interest.

2.6. Calculations and Statistical Analysis

We used SPSS Version 25.0 [38] to conduct all statistical analyses. Descriptive statistics were applied to calculate the means, ranges, standard error, standard deviation, and percentiles (10th and 90th) based on four SUs per species. We also tested Pearson's bivariate correlation among selected variables [39]. Unless expressly stated otherwise, our accepted probability level was set at α = 0.05.

Each of the four stands per species was considered a sampling unit to measure variation in the variables of interest. The normality assumptions and outliers were assessed for the primary variables, forest litter biomass, C content of litter, bulk density, and C

concentration of soils at each depth. The soil GHG flux data are generally not normally distributed, necessitating careful statistical approaches and consideration when analyzing and interpreting the results [40]. If the normality assumption happens to be violated and outliers are detected in the soil flux data, we would proceed to employ nonparametric tests such as the Freeman test, which is equivalent to a repeated-measures ANOVA (RMA), and the Wilcoxon test, which is equivalent to post hoc tests.

Primary variables were checked for normality using the Shapiro–Wilk test [41]. Using univariate ANOVA, we evaluated the relationship between soil temperature ($^\circ$C) and humidity (%) with periodical CO_2 Eq. fluxes. To assess the variation in organic C stocks in litter and mineral soil and CO_2 Eq. fluxes, considering forest type (broadleaves and conifers) as a fixed factor, independent sample *t*-tests ($\alpha = 0.05$) assuming or not equal variances were employed.

One-way RMA ($\alpha = 0.05$) were employed to analyze the effects of the species on the primary variables (bulk density and C concentration in soils at each evaluated depth, biomass, and C concentration in litter) and secondary variables (total and layer SOC stocks, C stocks in litter, and C and CO_2 Eq. stocks in litter and soil as a whole). If significant effects were found, pairwise comparisons through Bonferroni's [42] adjustment post hoc test ($p \leq 0.05$) were conducted. If the assumption of sphericity was met, the within-subjects tests were reported based on statistics assuming sphericity; on the contrary, we proceeded to use the corrections for the adjustment of degrees of freedom computed by SPSS, namely, Greenhouse–Geisser [43], or Huynh–Feldt [44]. The Greenhouse–Geisser procedure estimates epsilon ($\hat{\varepsilon}$) to correct the degrees of freedom of the F-distribution.

We evaluated the relationship between soil temperature ($^\circ$C) and humidity (%) with periodical CO_2 Eq. fluxes using univariate ANOVA. To assess the variation in organic C stocks in litter and mineral soil and CO_2 Eq. fluxes, considering forest type (broadleaves and conifers) as a fixed factor, independent sample *t*-tests ($\alpha = 0.05$) assuming or not equal variances were employed.

3. Results and Discussion

The effects of stand characteristics may affect C stocks. Stand age can significantly affect soil C storage across different forest types, but contradictory reports have been reported. In *Picea asperata* forests in China, soil carbon storage tends to decrease with increasing stand age, with younger stands showing higher total C storage than older stands [45]. Negative correlations between age and C storage have also been found on *Pinus koraiensis* [46].

Although age differences were substantial between species, and biomass generally increases with stand age [47], the species did not exhibit statistically significant changes in litter biomass (t ha^{-1}) and litter organic C stock (t ha^{-1}) contents. Other factors also exert an influence. The average age difference between the stands of *S. macrophylla* (11 years) and *S. mahagoni* (35 years) forest was almost 24 years; nonetheless, the average differences in d.b.h. and total tree height were only 6.13 cm and 0.15 m, respectively. This is due to the aggressive growth pattern of *S. macrophylla*, which can reach considerable allometric dimensions in a few years compared to the growth patterns of *S. mahagoni* [48].

Tree size as reflected by the allometric variables height and d.b.h. might indirectly influence SOC through its association with tree growth and biomass, but the specific impact of these variables on SOC is not straightforward [49]

Increased stand density generally leads to higher SOC storage [50]. However, the effect can vary with tree species; for example, under spruce, both the carbon and nitrogen contents increase with density, while under larch and pine, the effect is less pronounced [51]. In our study, both *S. mahagoni* and *S. macrophylla* had the largest stand density, with 807 and 764 trees per hectare, respectively. They also had more organic C stock in litter and mineral soil pools, with 62.37 and 46.3 t C ha^{-1}, respectively.

Higher stand densities sometimes reduce CO_2 emissions derived from SOC, particularly in deeper soil layers [50]. Our results contradict this statement by Sun et al. [50] because emissions were higher in the stands of *S. mahagoni*, which had the highest density.

The primary variables litter biomass, C content of litter, and C concentration and BD of soils at each depth were assessed for normality and outliers (Table 2). There were no outliers, and the data were normally distributed at each time point, as assessed by boxplots [52] and the Shapiro–Wilk test ($p > 0.05$), respectively (Figure 2).

Table 2. Shapiro–Wilk [41] normality test results for the primary variables litter biomass, C content of litter, and C concentration and bulk (apparent) density of soils at each depth for each species.

Species	Variable	Statistic	df	Sig.
S. macrophylla	Litter biomass (t ha^{-1})	0.954	16	0.561
P. occidentalis		0.900	16	0.079
P. caribaea		0.854	16	0.016
S. mahagoni		0.963	16	0.708
S. macrophylla	C content in litter (%)	0.907	16	0.105
P. occidentalis		0.917	16	0.153
P. caribaea		0.946	16	0.426
S. mahagoni		0.888	16	0.051
S. macrophylla	C concentration in coil at 0–10 cm depth (g kg^{-1})	0.958	16	0.635
P. occidentalis		0.963	16	0.722
P. caribaea		0.974	16	0.892
S. mahagoni		0.934	16	0.283
S. macrophylla	C concentration in soil at 10–20 cm depth (g kg^{-1})	0.940	16	0.350
P. occidentalis		0.858	16	0.018
P. caribaea		0.941	16	0.357
S. mahagoni		0.943	16	0.390
S. macrophylla	C concentration in soil at 20–30 cm depth (g kg^{-1})	0.818	16	0.005
P. occidentalis		0.908	16	0.107
P. caribaea		0.926	16	0.214
S. mahagoni		0.945	16	0.417
S. macrophylla	Soil bulk density at 0–10 cm depth (kg m^{-3})	0.934	16	0.284
P. occidentalis		0.935	16	0.289
P. caribaea		0.929	16	0.235
S. mahagoni		0.967	16	0.793
S. macrophylla	Soil bulk density at 20–30 cm depth (kg m^{-3})	0.958	16	0.618
P. occidentalis		0.937	16	0.311
P. caribaea		0.969	16	0.826
S. mahagoni		0.956	16	0.584
S. macrophylla	Soil bulk density at 20–30 cm depth (kg m^{-3})	0.973	16	0.891
P. occidentalis		0.967	16	0.780
P. caribaea		0.933	16	0.270
S. mahagoni		0.967	16	0.784
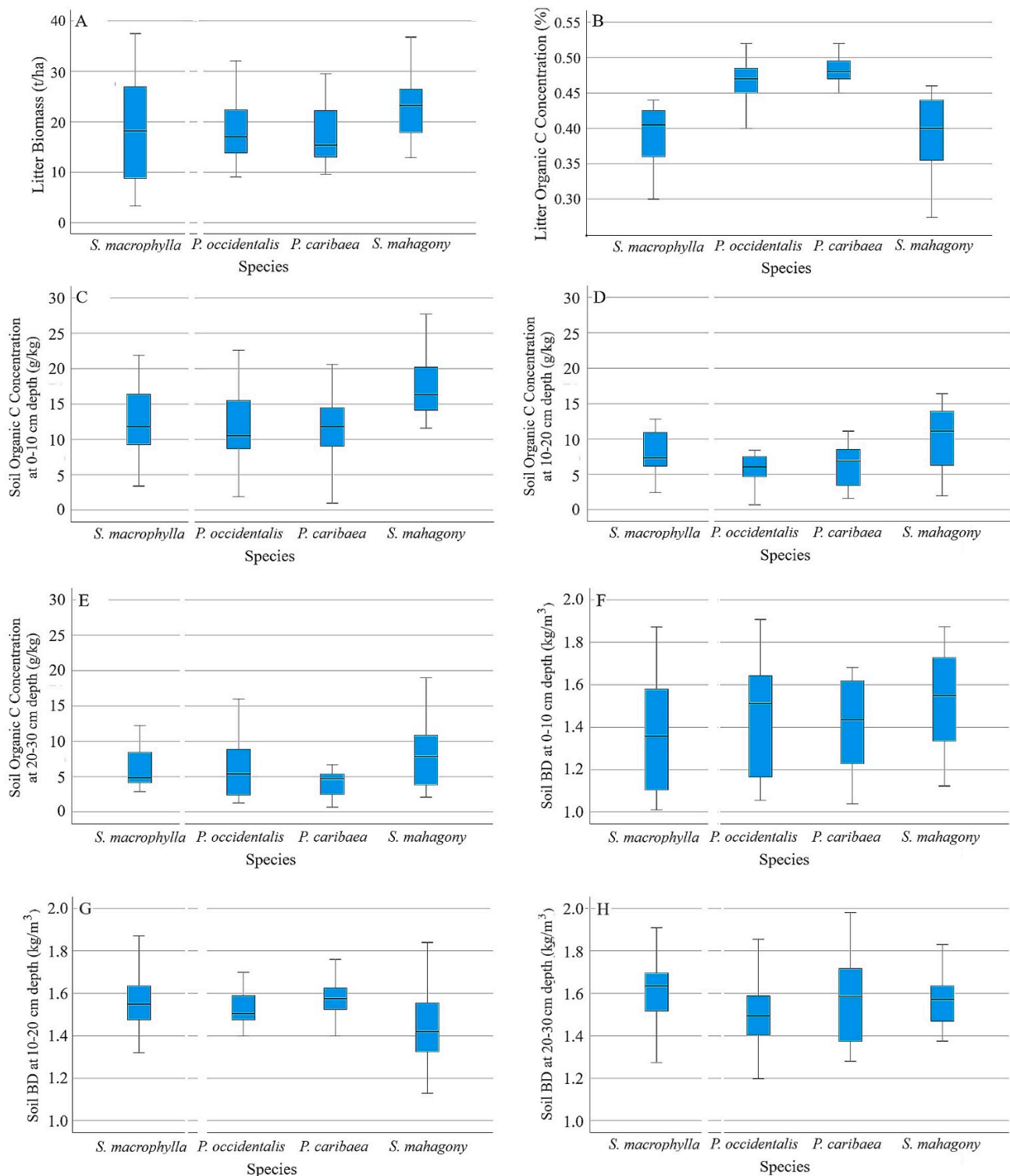
Figure 2. Boxplots showing the absence of outliers for the primary variables litter biomass panel (**A**), C content of litter panel (**B**), SOC concentration at the three depths assessed (0–10 in panel (**C**), 10–20 in panel (**D**), and 20–30 cm in panel (**E**)), and respective soil bulk density at each assessed depth (0–10 in panel (**F**), 10–20 in panel (**G**), and 20–30 cm in panel (**H**)).

3.1. Physicochemical Status of Soils

The average fertility levels observed at the beginning and end of the study, along with the soil texture evaluated to confirm similar edaphic parent material in the studied areas are shown in Table 3. The methods employed by the independent labs to evaluate each variable,

along with references, are also included. The results show that most physicochemical properties had higher values in *S. mahagoni* stands. Additionally, all soils under the species, except for *S. mahagoni* (sandy loam), had a loamy sand texture.

Table 3. Results of the analyses to characterize the average physicochemical properties and the type of parent material of the soils for each of the species.

Physicochemical Property (Method)	Unit	*S. macrophylla*	*P. occidentalis*	*P. caribaea*	*S. mahagoni*
pH in water (a)		5.20	4.39	4.26	5.12
Electrical conductivity (b)	mS/cm	0.19	0.09	0.09	0.54
Organic matter (c)	%	2.45	2.18	2.15	2.61
Nitrogen (d)	%	0.12	0.11	0.11	0.13
Phosphorus (e)	mg/kg	29.80	4.87	7.72	65.68
Exchangeable acidity (Al + H+) (f)	cmol(+)/kg^{-1}	1.07	19.50	1.88	1.53
Exchangeable Ca (e)	cmol(+)/kg^{-1}	19.78	17.36	8.79	22.77
Exchangeable Mg (e)	cmol(+)/kg^{-1}	10.41	13.98	13.60	14.90
Exchangeable K (e)	cmol(+)/kg^{-1}	0.13	0.13	0.15	0.99
Cation Exc. Capacity (g)	cmol(+)/kg^{-1}	29.13	50.96	24.41	39.80
Ca Saturation (h)	%	58.30	43.37	37.94	56.45
Mg Saturation (h)	%	38.16	30.19	46.12	37.85
K Saturation (h)	%	0.39	0.50	0.65	1.88
Al Saturation (h)	%	4.20	25.94	7.80	5.10
Ca/Mg (g)	Proport.	1.80	1.61	0.76	1.55
Ca/K (g)	Proport.	158.96	177.09	92.36	40.53
Mg/K (g)	Proport.	114.34	149.41	208.37	31.83
(Ca + Mg)/K (g)	Proport.	273.30	326.52	300.73	72.36
Copper (i)	mg/kg	2.09	3.53	2.12	4.38
Manganese (j)	mg/kg	64.40	72.19	64.81	93.04
Iron (j)	mg/kg	124.66	148.87	143.37	190.99
Zinc (i)	mg/kg	1.83	1.57	0.69	3.22
Clay (k)	%	4.50	4.00	4.00	6.50
Silt (k)	%	16.00	14.00	12.50	18.00
Sand (k)	%	79.50	82.00	83.50	75.50
Texture (l)		SL	LS	LS	LS

Methods: (a) Extractor solution with KCl1N; (b) Saturation extract; (c) Loss on ignition; (d) Combustion; (e) Mehlich 3 (ICP); (f) BaCl2-TEA, pH 8.2; (g) Summation; (h) Base saturation by CEC–8.2 (Sum of Cations); (i) EPA method 3050B/3051 + 6010; (j) Mass spectrophotometry; (k) Particle size analysis; (l) Soil Triangle method. Abbreviations: Exc. = Exchange; Ca = Calcium; Mg = Magnesium; K = Potassium; Al = Aluminum; Proport. = Proportion; mg/kg = milligrams per kilogram; mS/cm = milliSiemen per centimeter; Meq = Milliequivalents per 100 g of soil; SL = Sandy loam; LS = loamy sandy.

The levels of physical properties and chemical elements in soils that are considered adequate vary depending on the element and its role in plant growth [53,54]. However, the optimal content of these chemical and physical elements for C balance in forest soils is not clearly defined. It may vary depending on specific environmental conditions and land-use practices [55]. Key elements include phosphorus (30–50 mg/kg), nitrogen (0.25%–0.50%),

calcium (5–10 meq/100 g), magnesium (>35 mg/kg), potassium (40–80 mg/kg), copper (2–50 mg/kg), manganese (2–25 mg/kg), and iron (20–30 mg/kg) [53].

We found practical guidelines regarding status levels for the other properties, considering the goal of maintaining healthy forest soil. These guidelines provide clear and actionable levels for the pH in water (5.5–6.5) [56], electrical conductivity (EC) (1.10–5.70 mS/cm) [57], organic matter (>2%) [58], Cation Exchange Capacity (CEC) (10–30 cmol(+)/kg^{-1}) [54], magnesium saturation (10% to 40%) [54], potassium saturation (1%–8%) [54], aluminum saturation (5.5%–17%) [59], zinc (1–200 mg/kg) [60], clay (12.5%–20%) [61], silt (25%–40%) [61], and sand (25%–40%) [61].

Compared to these guidelines, the pH, EC, and nitrogen levels in the studied soils are below the appropriate threshold in all species. Phosphorus, potassium, and potassium saturation are also low except in *S. mahagoni*. Calcium saturation and zinc are low in *P. caribaea,* and aluminum saturation is low under the broadleaf species. Logically, these soils have a sandy texture and are therefore low in clay and silt content. The magnesium in all species is above the appropriate threshold. Calcium is above as well, except in *P. caribaea*. The potassium is high in *S. mahagoni*, and the CEC is above levels in *P. occidentalis* and *S. mahagoni*. Magnesium saturation is high in *P. caribaea* and aluminum saturation in *P. occidentalis*.

Most soil microbes thrive in a pH range of 6–7 [58], being optimal between 5.5 and 7 to support higher biomass production, leading to more significant organic C inputs to the soil. The average soil pH in our study was below the threshold of 5.5 indicated by [56]. These authors found a negative correlation between the soil organic C concentration and pH expressed in logarithmic units. A higher C concentration is associated with lower pH levels and sand content. Microbial activity outside this range can decrease, leading to a slower decomposition of organic matter and potentially higher SOC levels [58]. Soils with a high EC might have a high pH, which can affect the stability and decomposition of organic matter [58], leading to less microbial activity and affecting organic C accumulation. Increased nitrogen availability can lead to the formation of more stable organic C compounds through microbial processes. However, excessive nitrogen can sometimes result in increased C losses through leaching and GHG emissions [62]. Adequate phosphorus can stimulate microbial activity, potentially increasing the organic matter decomposition rate and soil organic C formation [60].

3.2. Organic C in Forest Litter

The mean values, standard errors, and percentiles (10th and 90th) for the litter variables dry biomass (t ha^{-1}), C concentration (%), C stocks (t ha^{-1}), and CO_2 Eq. (t ha^{-1}) for the species are presented in Table 4. For the entire evaluation period, the mean ($\pm$ standard error) dry biomass content in litter ranged from 18.38 $\pm$ 1.73 in *P. occidentalis* to 23.07 $\pm$ 1.70 t ha^{-1} in *S. mahagoni*. The mean organic C content (%) in litter biomass showed a range of mean ($\pm$ standard error) of 0.39 $\pm$ 0.01 (%) in both broadleaved species and 0.48 $\pm$ 0.02 in *Pinus caribaea*. Organic C stocks in the litter were largest in *S. mahagoni* (8.99 $\pm$ 0.71 t ha^{-1}), decreasing to 8.65 $\pm$ 0.83 t ha^{-1} (*P. occidentalis*), 8.43 $\pm$ 0.76 t ha^{-1} (*P. caribaea*), and 6.91 $\pm$ 0.95 t ha^{-1} (*S. macrophylla*). These values expressed in CO_2 Eq. units (t ha^{-1}), were respectively 32.99 $\pm$ 2.59, 31.75 $\pm$ 3.06, 30.93 $\pm$ 2.78, and 25.38 $\pm$ 3.50 t ha^{-1}. The lower pH levels in coniferous forest soils (average pH 4.32) compared to broadleaf forests (average pH 5.16) (Table 2), may result in a higher retention of fresher leaves in the soil under conifers [63].

Other studies have reported higher and lower values for litter biomass, showing that different species have different rates of litterfall and decomposition and, therefore, varying impacts on C dynamics [63]. Lee et al. [63] reported litter organic C stocks of 3.28 $\pm$ 0.13 t ha^{-1} for broadleaved and 4.63 $\pm$ 0.18 t ha^{-1} for conifers in South Korea. Our estimated values are also higher for conifers (8.54 $\pm$ 0.55 t ha^{-1}) than broadleaved (7.95 $\pm$ 0.61 t ha^{-1}). They are also higher than those reported by Lee et al. [63] by 83.17% and 59.38%, respectively. Cha et al. [64] suggest that the greater difficulty in decomposition

of the litter layer in coniferous forests compared to broadleaf forests could explain the observed differences in litter biomass.

Table 4. Mean and percentile values (10 and 90) for dry biomass, organic C content, C stocks, and CO_2 Eq. in the forest litter.

	Units	Stats	*S. macrophylla*	*P. occidentalis*	*P. caribaea*	*S. mahagoni*
Forest litter dry biomass	t ha^{-1}	N	16	16	16	16
		Mean	18.55	18.38	17.45	23.07
		Standard Error	2.63	1.73	1.56	1.70
		10th Percentile	3.69	10.13	11.32	13.69
		90th Percentile	33.19	31.52	27.06	34.75
C content	%	Mean	0.39	0.41	0.48	0.39
		Standard Error	0.01	0.01	0.02	0.01
		10th Percentile	0.31	0.41	0.46	0.30
		90th Percentile	0.44	0.50	0.52	0.45
C stocks	t ha^{-1}	Mean	6.91	8.65	8.43	8.99
		Standard Error	0.95	0.83	0.76	0.71
		10th Percentile	1.41	4.36	5.29	4.52
		90th Percentile	12.40	15.03	14.64	12.50
CO_2 Eq. stored in litter	t ha^{-1}	Mean	25.38	31.75	30.93	32.99
		Standard Error	3.50	3.06	2.78	2.59
		10th Percentile	5.17	16.00	19.42	16.57
		90th Percentile	45.52	55.18	55.33	45.86

3.3. SOC Stocks in Mineral Soils

The results of the assessment of BD (kg m^{-3}), organic C concentration (g kg^{-1}), and SOC stocks (t ha^{-1}) at three different soil layers, that is, P1 (0–10 cm), P2 (10–20 cm), and P3 (20–30 cm), are reported hereafter. The BD (kg m^{-3}) means (±standard errors) showed a steadily monotonic increase for *S. macrophylla* (1377.56 ± 70.93 to 1608.19 ± 42.33) and *P. caribaea* (1412.50 ± 52.62 to 1585.19 ± 56.44) with soil depth. Under *P. occidentalis*, BD was highest at P2 (1533.75 ± 23.00). Under *S. mahagoni*, BD was lowest at P2 (1440.63 ± 47.81) and highest at P1 (1528.13 ± 56.06). BD increased by 16.73%, 3.01%, and 12.23%, respectively, between P1 and P3 in *S. macrophylla*, *P. occidentalis*, and *P. caribaea*. In *S. mahagoni*, BD decreased by 1.34% between P1 and P3 (Table 5). Other researchers have reported contradictory findings on soil BD patterns. Some authors reported that it decreases with depth [65], while others reported increases with depth [66]. Wu et al. [67] reported that the bulk density decreased by 4.3% at deeper soil layers compared to surface layers in his study. Azeez et al. [68] noted BD to increase at 20–40 and 40–60 cm depths under some tree plantations.

The mean average (±standard error) organic C concentration (g kg^{-1}) in *S. mahagoni* at P1 was 12.45 ± 1.43, and it decreased monotonically thereafter at P2 (8.05 ± 0.78) and P3 (6.30 ± 0.76). The respective values for *P. occidentalis* were P1 (12.19 ± 1.39), P2 (5.65 ± 0.61), and P3 (6.26 ± 1.08). In *P. caribaea*, the observed results were P1 (11.41 ± 1.32), P2 (6.53 ± 0.96), and P3 (4.03 ± 0.47). Finally, in *S. mahagoni*, the realized values were P1 (17.38 ± 1.14), P2 (10.32 ± 1.13), and P3 (8.19 ± 1.16). These values in *S. mahagoni* were the highest among the species at each soil depth evaluated and had the highest variability.

SOC stocks (t ha^{-1}) varied across soil depths, with the highest average (±standard error) values found at P1 and the lowest at P3. The SOC mean (±standard error) stock values (t ha^{-1}) in the 30 cm evaluated were 39.39 ± 3.65, 35.60 ± 2.23, 33.06 ± 2.31, and 53.38 ± 3.91 for *S. macrophylla*, *P. occidentalis*, *P. caribaea*, and *S. mahagoni*, respectively. Expressed in CO_2 Eq. units, these values (t ha^{-1}) are 144.56 ± 13.38, 130.66 ± 8.20, 121.34 ± 8.49, and 195.90 ± 14.35.

Table 5. Mean values, standard errors, and percentiles (10 and 90) for organic carbon concentration (OCC), soil bulk density (BD), and organic carbon stock (OCS) at the three assessed soil depths under the four species.

| | Soil Layer (cm) | | OCC (g C kg^{-1}) | | | | BD (kg m^{-3}) | | | | OCS (t C ha^{-1}) | | | |
| | | | | Standard | | | | Standard | | | | Standard | | |
Species	Depth	N	Mean	Error	P$_{10}$	P$_{90}$	Mean	Error	P$_{10}$	P$_{90}$	Mean	Error	P$_{10}$	P$_{90}$
S. macrophylla	0–10	16	12.45	1.43	3.75	21.76	1377.56	70.93	1012.40	1827.20	16.86	2.08	6.46	29.89
	10–20	16	8.05	0.78	2.89	12.24	1552.50	37.63	1327.00	1800.00	12.54	1.26	4.54	19.61
	20–30	16	6.30	0.76	3.74	11.93	1608.19	42.33	1339.80	1869.40	9.99	1.12	5.32	17.27
P. occidentalis	0–10	16	12.19	1.39	5.26	21.73	1459.44	69.40	1071.40	1886.30	17.56	2.14	6.66	31.44
	10–20	16	5.65	0.61	0.70	8.05	1533.75	23.00	1421.00	1693.00	8.64	0.94	1.10	12.65
	20–30	16	6.26	1.08	1.65	13.62	1503.31	41.85	1265.20	1807.40	9.41	1.67	2.35	21.58
P. caribaea	0–10	16	11.41	1.32	1.91	18.78	1412.50	52.62	1090.80	1675.70	16.52	2.08	2.31	29.68
	10–20	16	6.53	0.96	1.88	10.89	1578.13	23.77	1449.00	1753.00	10.22	1.45	3.02	19.65
	20–30	16	4.03	0.47	0.98	6.35	1585.19	56.44	1292.60	1925.70	6.32	0.79	1.83	11.32
S. mahagoni	0–10	16	17.38	1.14	11.74	24.69	1528.13	56.06	1201.40	1838.70	26.15	1.56	18.00	37.74
	10–20	16	10.32	1.13	3.61	15.91	1440.63	47.81	1144.00	1784.00	14.31	1.43	5.68	22.19
	20–30	16	8.19	1.16	2.59	15.43	1507.63	28.43	1427.50	1736.90	12.92	1.81	3.62	23.65

With a few exceptions, the organic C concentration and SOC stocks decreased with soil depth, while the soil BD increased. The SOC sequestration potential generally differs from sites and soil depths [69]. Wei et al. [70] reported more significant SOC accumulation in the 0–10 cm depth than in the 10–80 cm depth. Huang et al. [71] also reported that SOC decreases with soil depth and is significantly higher in the 0–10 cm soil layer than in the other soil layers. The 0–10 cm layer usually has the highest SOC concentration because of the direct input of organic materials and because decomposition processes are also more active near the surface due to the higher microbial activity and better aeration [71].

Our study found that 48% of the average SOC stocks (t ha^{-1}) in the soil below all species were stored in the upper mineral layer (0–10 cm), followed by 24% in the second layer (10–20 cm), and 24% in the third layer (20–30 cm). According to Wei et al. [70], forest-derived SOC in afforested soils accounted for 52%–86% of the total OC in the 0–10 cm depth, 36% to 61% of the total SOC in the 10–20 cm depth, and 11%–50% of the total SOC in the 20–80 cm depth. In 30-year-old *Pinus elliotti* Engelm., *Schima superba* Gardner & Champ., and *Pinus massoniana* Lamb. plantations in subtropical China, the 0–15 cm soil layer accounted for 32.1% to 34.1% of the total SOC density. It was significantly higher than the other soil layers [69].

The observed SOC stocks in our study are low compared to studies in temperate latitudes. For instance, Uri et al. [72] reported 83.67 t ha^{-1} in the top 30 cm of soils under *Pinus sylvestris* L. in Estonia, and Berhongaray et al. [10] found SOC stocks in short-rotation woody crops with average values of 140 t ha^{-1} in Belgium. The means, standard errors, and percentiles (10th and 90th) for mineral soil variables are reported in Table 5.

3.4. Organic C Stocks in Both Pools (Litter and Soil) by Species

Organic C stocks in both the litter and soil were higher under *S. mahagoni*, realizing an average (±standard error) of 62.37 ± 37 t ha^{-1}. Following a decreasing order, the averages (±standard errors) for the rest of the species were 46.30 ± 3.83 t ha^{-1} (*S. macrophylla*), 44.25 ± 2.03 t ha^{-1} (*P. occidentalis*), and 41.49 ± 2.57 t ha^{-1} (*P. caribaea*). The corresponding averages (±standard errors) expressed in CO_2 Eq. units were 228.88 ± 15.29 t ha^{-1}, 169.93 ± 14.05 t ha^{-1}, 162.41 ± 8.46 t ha^{-1}, and 152.27 ± 9.41 t ha^{-1}.

3.5. Periodic Dynamics of CO_2 Eq. Stocks

The periodic dynamics of stocks in both litter and soil pools, summed up and expressed in CO_2 Eq. units, were examined using one-way RMA for each species, considering the four measurement periods (September 2020, April 2021, October 2021, and March 2022). The sphericity assumption was met for these four variables as assessed by Mauchly's test (Table 6). Post hoc analysis with a Bonferroni adjustment (Table 7) revealed that in both litter and soil pools considered together, there were statistically significant differences in CO_2 Eq. stocks under *P. caribaea* ($p = 0.001$). However, no differences were observed in the CO_2 Eq. stocks of *S. macrophylla* ($p = 0.172$), *P. occidentalis* ($p = 0.151$), and *S. mahagoni* ($p = 0.161$) stands.

CO_2 Eq. stocks in *P. caribaea* were highest in March 2022 (183.24 ± 17.97 t ha^{-1}), followed by September 2020 (172.13 ± 11.93 t ha^{-1}) and October 2021 (141.55 ± 13.56 t ha^{-1}), and were lowest in April 2021 (112.15 ± 9.85 t ha^{-1}). CO_2 Eq stocks under *P. caribaea* were statistically different between September 2020 and April 2021 ($p = 0.008$) and March 2022 and April 2021 ($p = 0.033$) (Table 7). Figure 3 illustrates the changes in CO_2 Eq stocks in litter and soils under different species during the four evaluation periods.

Table 6. One-way repeated-measures ANOVA results of statistically significant variables, including Mauchly's sphericity and within-subjects tests.

| Variable | Units | Species | df | Mauchly's Test | | | Within-Subjects Test | | | | Eta |
				X^2	*p*-Value	(E)	F	df1	df2	*p*-Value	w^2
Litter biomass	(t ha^{-1})	ALL	5	9.890	0.079	NA	2.887	3	45	0.560	0.161
Litter OC concentration	(%)	ALL	5	9.137	0.104	NA	29.026	3	45	0.000	0.659
Litter C stocks	(t ha^{-1})	ALL	5	10.362	0.066	NA	1.911	3	45	0.141	0.113
Soil BD (00–10 cm depth)	(kg m^{-3})	ALL	5	0.598	0.988	NA	1.059	3	45	0.376	0.066
Soil BD (10–20 cm depth)	(kg m^{-3})	NA	5	11.499	0.043	0.704	2.831	2.113	45	0.071	0.159
Soil BD (20–30 cm depth)	(kg m^{-3})	ALL	5	7.779	0.170	NA	1.182	3	45	0.327	0.073
Soil C concentration (00–10 cm depth)	(g C kg^{-1})	ALL	5	7.197	0.207	NA	4.923	3	45	0.005	0.247
Soil C concentration (10–20 cm depth)	(g C kg^{-1})	ALL	5	2.331	0.802	NA	10.777	3	45	0.000	0.418
Soil C concentration (20–30 cm depth)	(g C kg^{-1})	ALL	5	6.402	0.270	NA	4.147	3	45	0.011	0.217
SOC stocks (00–10 cm)	(t ha^{-1})	ALL	5	11.334	0.046	0.770	6.316	2.309	45	0.003	0.296
SOC stocks (10–20 cm)	(t ha^{-1})	ALL	5	1.453	0.919	NA	6.089	3	45	0.001	0.289
SOC stocks (20–30 cm)	(t ha^{-1})	ALL	5	6.938	0.226	NA	4.383	3	45	0.009	0.226
Total SOC stocks	(t ha^{-1})	ALL	5	13.359	0.021	0.655	13.920	1.965	45	0.000	0.481
Soil and litter organic C stocks	(t ha^{-1})	ALL	5	14.084	0.015	0.680	12.166	2.040	45	0.000	0.448
C stocks per period	(t ha^{-1})	*S. macrophylla*	5	3.091	0.505	NA	2.089	3	9	0.172	0.411
C stocks per period	(t ha^{-1})	*P. occidentalis*	5	5.734	0.400	NA	2.268	3	9	0.151	0.432
C stocks per period	(t ha^{-1})	*P. caribaea*	5	4.905	0.492	NA	14.71	3	9	0.001	0.831
C stocks per period	(t ha^{-1})	*S. mahagoni*	5	5.738	0.412	NA	2.172	3	9	0.161	0.423

Abbreviations and symbols: df = degrees of freedom; X^2 = Chi Square; E = Greenhouse–Geisser Epsilon; F = F statistic; w^2 = Eta Partial Square; OC = organic carbon; C = carbon; BD = soil bulk density; NA = not applicable; SOC = soil organic carbon; ha = hectare; t = metric tons.

Table 7. Bonferroni post hoc test results of statistically significant variables from the one-way repeated-measures ANOVA.

Variable	Units	Species	Variable 1	Variable 2	Mean Diff.	Std. Error	*p*-Value
Litter OC concentration	t ha^{-1}	ALL	*S. macrophylla*	*P. occidentalis*	−0.076	0.011	0.000
			S. macrophylla	*P. caribaea*	−0.095	0.011	0.000
			P. occidentalis	*S. mahagoni*	0.075	0.015	0.001
			P. caribaea	*S. mahagoni*	0.094	0.014	0.000
Soil C concentration (0–10 cm)	g kg^{-1}	ALL	*S. macrophylla*	*S. mahagoni*	−4.925	1.093	0.003
			P. occidentalis	*S. mahagoni*	−5.187	1.680	0.045
			P. caribaea	*S. mahagoni*	−5.969	1.940	0.046

Table 7. *Cont.*

Variable	Units	Species	Variable 1	Variable 2	Mean Diff.	Std. Error	*p*-Value
Soil C concentration (10–20 cm)	g kg^{-1}	ALL	*S. macrophylla*	*P. occidentalis*	2.396	0.782	0.047
			P. occidentalis	*S. mahagoni*	−4.664	0.992	0.002
			P. caribaea	*S. mahagoni*	−4.093	0.951	0.004
SOC stocks (0–10 cm)	t ha^{-1}	ALL	*S. mahagoni*	*S. macrophylla*	9.289	1.987	0.002
			S. mahagoni	*P. occidentalis*	8.596	1.804	0.002
			S. mahagoni	*P. caribaea*	9.628	2.684	0.016
SOC stocks (10–20 cm)	t ha^{-1}	ALL	*S. mahagoni*	*P. occidentalis*	5.673	1.274	0.003
Total SOC stocks	t ha^{-1}	ALL	*S. macrophylla*	*S. mahagoni*	−13.989	3.019	0.002
			P. occidentalis	*S. mahagoni*	−17.775	3.523	0.001
			P. caribaea	*S. mahagoni*	−20.314	4.120	0.001
Soil and litter organic C stocks	t ha^{-1}	ALL	*S. mahagoni*	*S. macrophylla*	16.064	3.641	0.003
			S. mahagoni	*P. occidentalis*	18.114	4.028	0.003
			S. mahagoni	*P. caribaea*	20.876	4.542	0.002
C stocks per period	t ha^{-1}	*P. caribaea*	September 2020	April 2021	59.977	5.175	0.008
			April 2021	March 2022	−71.090	9.808	0.033

Abbreviations and symbols: Diff. = difference; Std. = standard; OC = organic carbon; C = carbon; SOC = soil organic carbon; ha = hectare; t = metric tons.
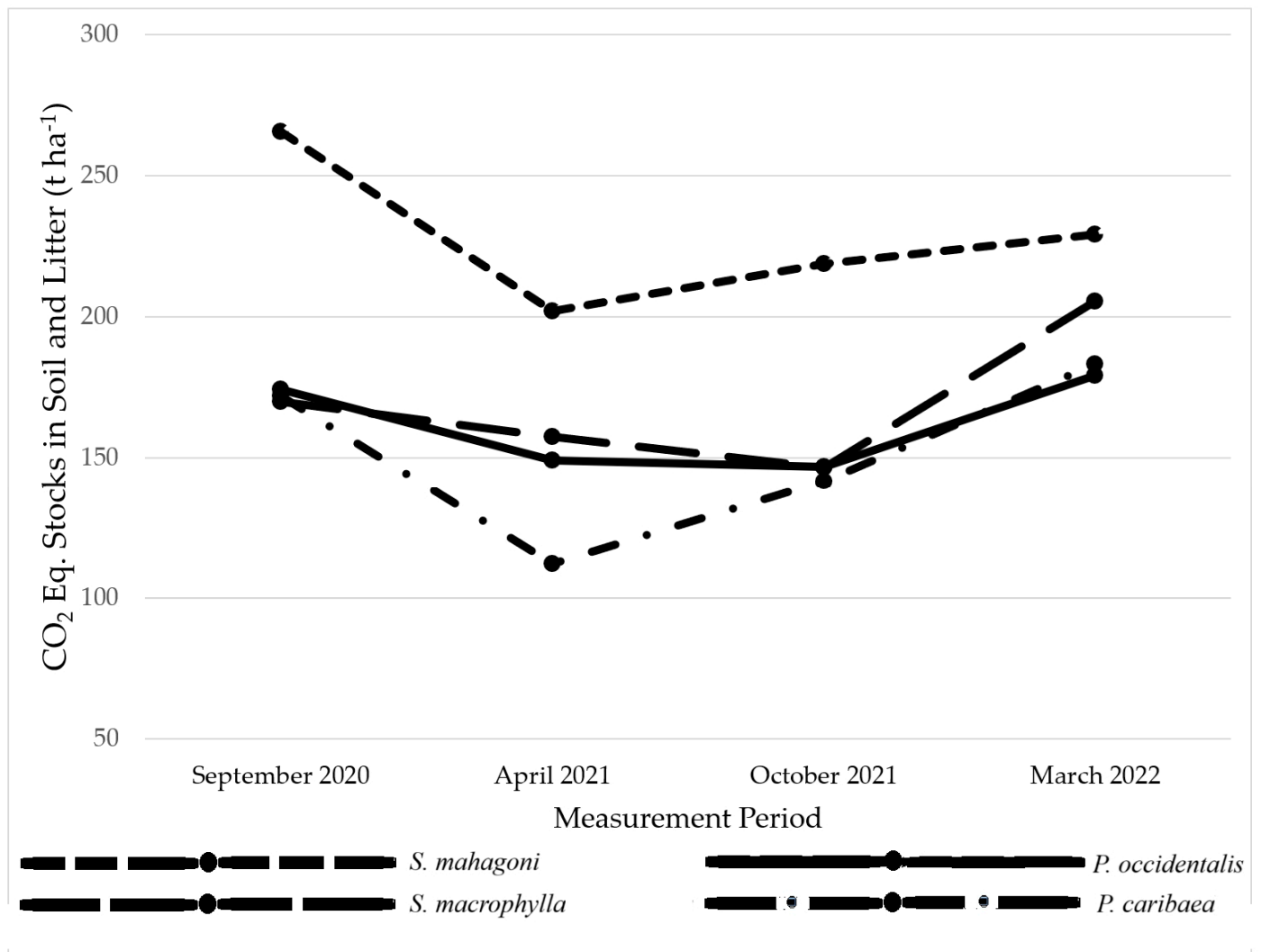
Figure 3. Periodic dynamics of CO_2 Eq. stocks in litter and soils (30 cm) under *S. macrophylla*, *P. occidentalis*, *P. caribaea*, and *S. mahagoni* in the four measurement periods (September 2020, April 2021, October 2021, and March 2022).

3.6. One-Way RMA Analyses for Litter and Mineral Soil Variables

One-way RMA was conducted to determine whether the species had a statistically significant effect on litter and mineral soil variables. As assessed by Mauchly's test [73], the assumption of sphericity was met for the three litter variables dry biomass content ($p = 0.079$), organic C concentration ($p = 0.104$), and organic C stocks ($p = 0.066$). The sphericity assumption was met in the mineral soil for all variables except for BD at P2 ($p = 0.04$), SOC at P1 ($p = 0.04$), total SOC stocks (30 cm) ($p = 0.024$), and the summed-up pool considering litter and soil ($p = 0.02$). The Epsilon ($\hat{\varepsilon}$) statistic was used to correct the degrees of freedom of the F statistic, as calculated according to Greenhouse and Geisser (1959). The realized Epsilon values were 0.704, 0.770, 0.655, and 0.680, respectively.

In the litter pool, the within-subjects test (Table 6) reveals that the species did not elicit statistically significant changes in the biomass content ($p = 0.56$) and organic C stocks ($p = 0.14$). Still, their effect was statistically significant for the organic C content ($p < 0.001$). Bonferroni's adjusted post hoc tests (Table 7) indicate statistically significant differences in the litter organic C concentration between *S. macrophylla* and both *P. occidentalis* ($p < 0.001$) and *P. caribaea* ($p < 0.001$), as well as between *S. mahagoni* and *P. occidentalis* ($p = 0.001$) and *P. caribaea* ($p < 0.001$).

The within-subjects test (Table 6) results also reveal statistically significant differences for the soil organic C concentration (g kg^{-1}) at P1 ($p = 0.005$), P2 ($p < 0.001$), and P3 ($p = 0.011$); for SOC stocks (t ha^{-1}) at P1 ($p = 0.003$), P2 ($p < 0.001$), and P3 ($p = 0.009$); and total SOC stocks (30 cm) at P1 ($p < 0.001$). The summed-up soil and litter organic C stocks were also statistically significant ($p < 0.001$) among species, and CO_2 Eq. periodic stocks were statistically significant in P. caribaea ($p = 0.001$).

In the soil pool variables, Bonferroni's adjusted post hoc tests (Table 7) revealed statistically significant differences in the organic C concentration at P1 between *S. mahagoni* and *S. macrophylla* ($p = 0.035$), *P. occidentalis* ($p = 0.045$), and *P. caribaea* ($p = 0.046$); and at P2 between *S. macrophylla* and *P. occidentalis* ($p = 0.047$), *S. mahagoni* and *P. occidentalis* ($p = 0.002$), and *S. mahagoni* and *P. caribaea* ($p = 0.004$).

SOC stocks at P1 differed significantly between *S. mahagoni* and *S. macrophylla* ($p = 0.002$), *S. mahagoni* and *P. occidentalis* ($p = 0.002$), and *S. mahagoni* and *P. caribaea* ($p = 0.016$); and at P2 between *S. mahagoni* and *P. occidentalis* ($p = 0.003$). Total SOC stocks were statistically different between *S. mahagoni* and *S. macrophylla* ($p = 0.002$), *P. occidentalis* ($p = 0.001$), and *P. caribaea* ($p = 0.001$). The summed-up soil and litter C stocks differed statistically between *S. mahagoni* and *S. macrophylla* ($p = 0.003$), *P. occidentalis* ($p = 0.003$), and *P. caribaea* ($p = 0.002$). The CO_2 Eq. periodic stocks were statistically significant in *P. caribaea* between September 2020 and April 2021 ($p = 0.008$) and April and March ($p = 0.033$).

All means for the soil variables were higher in *S. mahagoni*. The average total organic C stocks were higher under this species, highlighting its prominence in capturing and storing organic carbon within the study area.

3.7. Organic C Stocks in Different Forest Types

Two forest types were studied: coniferous (*P. occidentalis* and *P. caribaea*) and broadleaved (*S. macrophylla* and *S. mahagoni*). In the mineral soil (30 cm), the average (±standard error) SOC stocks were 25.98% higher for broadleaved species (46.38 ± 2.91 t ha^{-1}) as compared to conifer species (34.33 ± 1.60 t ha^{-1}). In the mineral soil, Lee et al. [63] reported 44.11 ± 1.54 t ha^{-1} for broadleaved and 33.96 ± 1.62 t ha^{-1} for conifers. Cook et al. [74] reported values of 38.3 ± 1.9 t ha^{-1} and 36.0 ± 1.6 t ha^{-1} for broadleaf and coniferous OC stocks, respectively. Subashree et al. [75], who studied tropical coniferous and broadleaf forests, reported stocks distributed as follows: 44.2% in the 0–10 cm layer, 32.0% in the 10–20 cm layer, and 23.8% in the 20–30 cm layer. Schulp et al. [9], as cited in Garrett et al. [76] and Augusto et al. [77], found higher SOC stocks to a depth of 20 cm under coniferous forests (76.44 t C ha^{-1}) than under broadleaved forests (67.45 t C ha^{-1}). This trend was also reported by Laganière et al. [78], who found higher SOC stocks at 15 cm soil depth under *Picea mariana* Mill. (46.3 t C ha^{-1}) than under *Populus tremuloides* Michx (34.7 t C ha^{-1}). In agreement with our findings, Wang et al. [79] found SOC stocks in the top 10 cm of soil under *P. massoniana*, which were lower than under the species *Castanopsis hystrix* Miq., *Michelia macclurei* Dandy, and *Mytilaria laosensis* Lecomte, reporting 29.2 t C ha^{-1} and 32.6 t C ha^{-1}, respectively. Vesterdal et al. [80] found consistent effects of species on SOC.

Combining both reservoirs, the average (±standard error) organic C stock was higher in broadleaved species by 26.76% (54.34 ± 3.13 t ha^{-1}) than in conifers (42.87 ± 1.71 t ha^{-1}). Our results are consistent with previous studies by Vesterdal et al. [80] and Lee et al. [63], which also found higher OC stocks in coniferous litter and higher OC stocks in the mineral soil under broadleaf species. This trend is attributed to the slower rate of decomposition in conifers and the more active soil fauna in broadleaf species, leading to a greater incorporation of organic matter in soil aggregates [9,64].

The results of the independent samples *t*-tests indicate that there were significant differences in the SOC stocks among forest types (t (48.09) = 3.63, $p = 0.001$) and organic C stocks considering both litter and soil as a whole (t (48.09) = 3.21, $p = 0.002$), with respective mean differences on the order of 12.05 and 11.46 t ha^{-1}. There were no statistically significant differences in litter C stocks among conifers and broadleaves (t (62) = −0.71, $p = 0.480$).

3.8. CO$_2$ Equivalent Fluxes from the Soil under the Species

All species except *S. mahagoni* absorbed GHGs from the atmosphere. During 160 h of evaluation, diurnal soil respiration (t CO$_2$ Eq. ha^{-1} year^{-1}) was on average ($\pm$ standard error) -211.62 ± 42.09 (*S. macrophylla*), -158.72 ± 47.23 (*P. occidentalis*), -365.73 ± 47.23 (*P. caribaea*), and 48.25 ± 38.46 (*S. mahagoni*). *P. caribaea* absorbed the highest amount of CO$_2$ Eq., followed by *S. macrophylla* and *P. occidentalis*. Species significantly affected CO$_2$ Eq. fluxes as assessed by the nonparametric Friedman Test [81] ($p < 0.001$). The Wilcoxon signed-rank test [82] for multiple comparisons adjusted with the calculated Bonferroni [41] significance level ($0.05/6 = 8.33 \times 10^3$) indicated statistically significant differences between CO$_2$ Eq. fluxes from *S. mahagoni* and *S. macrophylla* ($p < 0.001$), *S. mahagoni* and *P. occidentalis* ($p = 0.001$), *S. mahagoni* and *P. caribaea* ($p < 0.001$), and *P. occidentalis* and *P. caribaea* ($p = 0.002$).

Several factors may have contributed to the variability in soil CO$_2$ flux, including edaphic properties and high CO$_2$ status. Eq. fluxes from *S. mahagoni* may be attributed to one of those properties. However, the effects of tree species on soil CO$_2$ flux are not always definite, as site conditions, elevation, climate, and forest management decisions also play significant roles in determining soil carbon dynamics [9]. Wei et al. [83] stated that there appears to be an inverse relationship between soil EC and CO$_2$ fluxes. We found a contradictory trend. The levels of EC found in this study are much higher for *S. mahagoni* (0.54 mS/s) than for the other species (Table 2).

Wu et al. [84] reported emissions on the order of 15.93, 16.45, and 14.06 t CO$_2$ Eq. ha^{-1} for the conifers *Larix gmelinii* Rupr., *P. sylvestris* var. mongolica, and the broadleaved species *Betula platyphylla* Sukaczev, respectively. Uri et al. [72] reported 6.0 t C ha^{-1} yr^{-1} (22.02 t CO$_2$ Eq. ha^{-1} yr^{-1}) for *P. silvestris* in a temperate forest in Estonia, and Pastore et al. [85] reported 8.1 t ha^{-1} yr^{-1} (29.73 t CO$_2$ Eq. ha^{-1} yr^{-1}) for temperate coniferous stands in Ireland. Gahagan et al. [86] reported fluxes of approximately 6.72 and 6.95 t ha^{-1} yr^{-1} (24.66 and 25.51 t CO$_2$ Eq. ha^{-1} yr^{-1}) for hardwood and *Pinus resinosa* Ait. temperate stands, respectively, in Northern Michigan, USA.

Another study by Klotz and Torres [34] provided the quantitative status of edaphic properties such as soil EC in the same forest stands. We analyzed the data reported, and the only statistically significant variable among these species was EC (mS/cm). The Bonferroni post hoc test (Table 7) indicated that EC was significantly higher in soils under *S. mahagoni* than in soils under *P. occidentalis* ($p = 0.007$) and *P. caribaea* ($p = 0.002$). Klotz and Torres [34] reported that the average EC levels were 0.02 mS/cm for *P. occidentalis* and *P. caribaea* and 0.04 mS/cm for *S. mahagoni*. These levels are well below the lower limit of 1.10 mS/cm recommended by [57] for healthy soils. On the other hand, soils with a high EC might have a high pH, which can affect the stability and decomposition of organic matter [1], leading to less microbial activity and affecting organic C accumulation. The relationship between EC and CO$_2$ Eq. fluxes is not straightforward and is often overshadowed by factors such as temperature and moisture content [36].

3.9. Diurnal Dynamics of CO$_2$ Equivalent Fluxes

CO$_2$ Eq. diurnal flux averages from *S. mahagoni* soils were higher in 7 of the 10 h evaluated, being a source into the atmosphere in 5 of those 7 h. *P. occidentalis* CO$_2$ Eq. diurnal averages were higher in 5 of the 10 h and a source in 2 h. CO$_2$ Eq. diurnal averages from *S. macrophylla* also released GHGs in 2 h, whereas *P. caribaea* soils always behaved as a sink, having the lowest CO$_2$ Eq. diurnal averages in 6 of the 10 h assessed.

Without discrimination per species, no statistically significant CO$_2$ Eq. diurnal flux averages were found between the first hour of measurement (7:00–8:00) and the remaining nine. CO$_2$ Eq. diurnal flux averages in the second hour of measurement were statistically significantly different from the corresponding measurements from the fourth hour to the tenth but not the third hour, with *p* values from Bonferroni post hoc comparisons being lower than 0.006. Fluxes from the third hour differed statistically from the fourth to the tenth ($p < 0.007$). The rest of the comparisons were not statistically significant. The CO$_2$ Eq. diurnal flux average dynamics per species are shown in Figure 4.
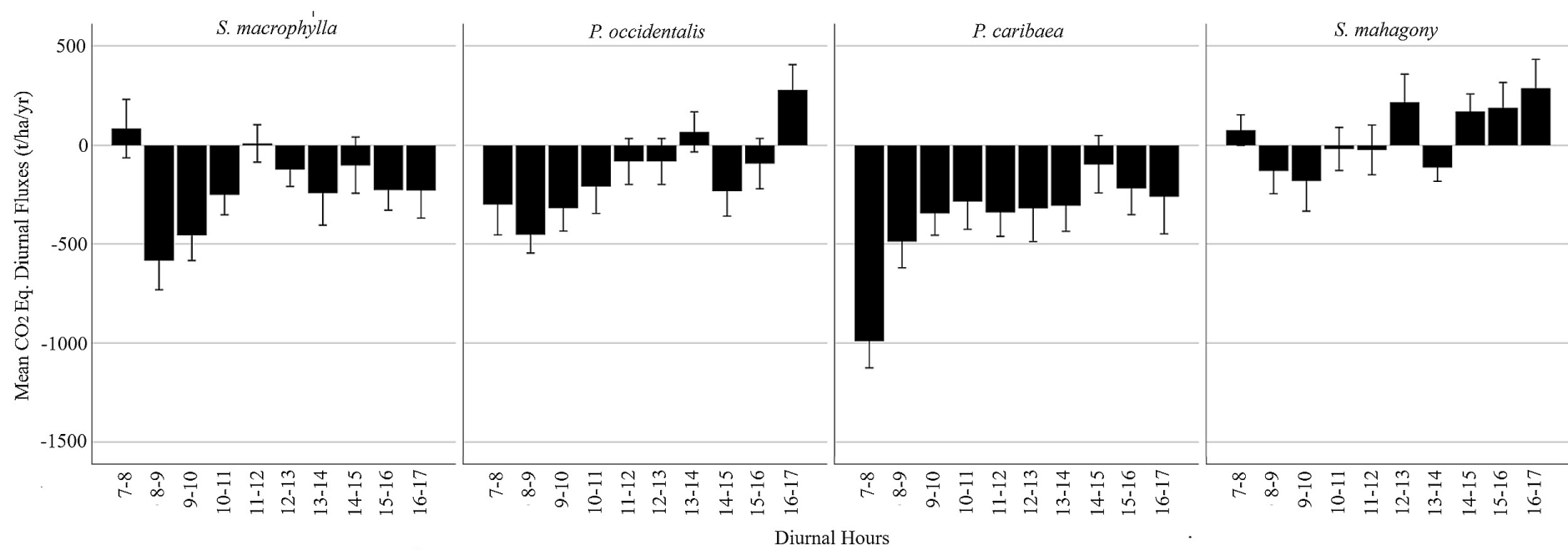
Figure 4. Diurnal dynamics (10 h) of CO_2 equivalent fluxes in soils under four stands of *S. macrophylla*, *P. occidentalis*, *P. caribaea*, and *S. mahagoni*.

3.10. Periodic Dynamics of CO_2 Equivalent Fluxes

The periodic dynamics of CO_2 Eq. flux averages (t ha^{-1} year^{-1}) per species reached maximum values in March 2022 for all species. Means ($\pm$standard errors) were 21.74 ± 37.12, -25.20 ± 80.24, -189.39 ± 63.64, and 110.15 ± 27.89 for *S. macrophylla*, *P. occidentalis*, *P. caribaea*, and *S. mahagoni*, respectively. In October 2021, CO_2 Eq. flux averages were the second highest for all species except *P. caribaea*, and in September 2021, the third largest CO_2 Eq. flux averages were observed. On average, both broadleaved species were sources during the four measurement periods, while conifers always acted as a sink. *S. mahagoni* behaved as a source in all periods except in October 2021. Considering all species, the CO_2 Eq. flux average showed a monotonic increase from September 2020 (-257.72 ± 46.12 t ha^{-1} yr^{-1}) to March 2022 (-20.68 ± 29.15 t ha^{-1} yr^{-1}).

Using the nonparametric Friedman and Wilcoxon tests, with the latter test's significant values adjusted by the Bonferroni correction, statistically significant differences were assessed for CO_2 Eq. average fluxes per period and species. In *S. macrophylla*, there were statistically significant differences ($p < 0.001$) in periodical CO_2 Eq. average fluxes, with fluxes in September 2020 being significantly lower than in October 2021 ($p < 0.001$) and fluxes in March 2022 being higher than fluxes in September ($p < 0.001$) and April 2021 ($p = 0.002$). In addition, in *p. occidentalis*, there were statistically significant differences ($p < 0.001$) in periodical CO_2 Eq. average fluxes. Fluxes in April were significantly lower than in October 2021 ($p = 0.003$) and in March 2022 ($p < 0.001$). No periodical statistically significant differences were found in *P. caribaea* ($p = 0.109$) and *S. mahagoni* ($p = 0.534$). CO_2 Eq. periodical flux averages dynamics per species are shown in Figure 5.

The quantitative status of the edaphic properties of soil humidity and temperature measured "In Situ" provided some insights regarding the periodical behavior of CO_2 Eq. flux averages. No significant correlations existed for the species between CO_2 Eq. flux averages and soil humidity. Average soil temperatures (°C) in September 2020, April 2021, October 2021, and March 2022 were 20.23 °C, 16.43 °C, 23.16 °C, and 27.27 °C. Soil temperatures in *S. macrophylla* were also highest in March 2022, with an average ($\pm$standard error) of 28.38 ± 0.69 °C, and lowest in April 2021 (16.88 ± 1.03 °C).
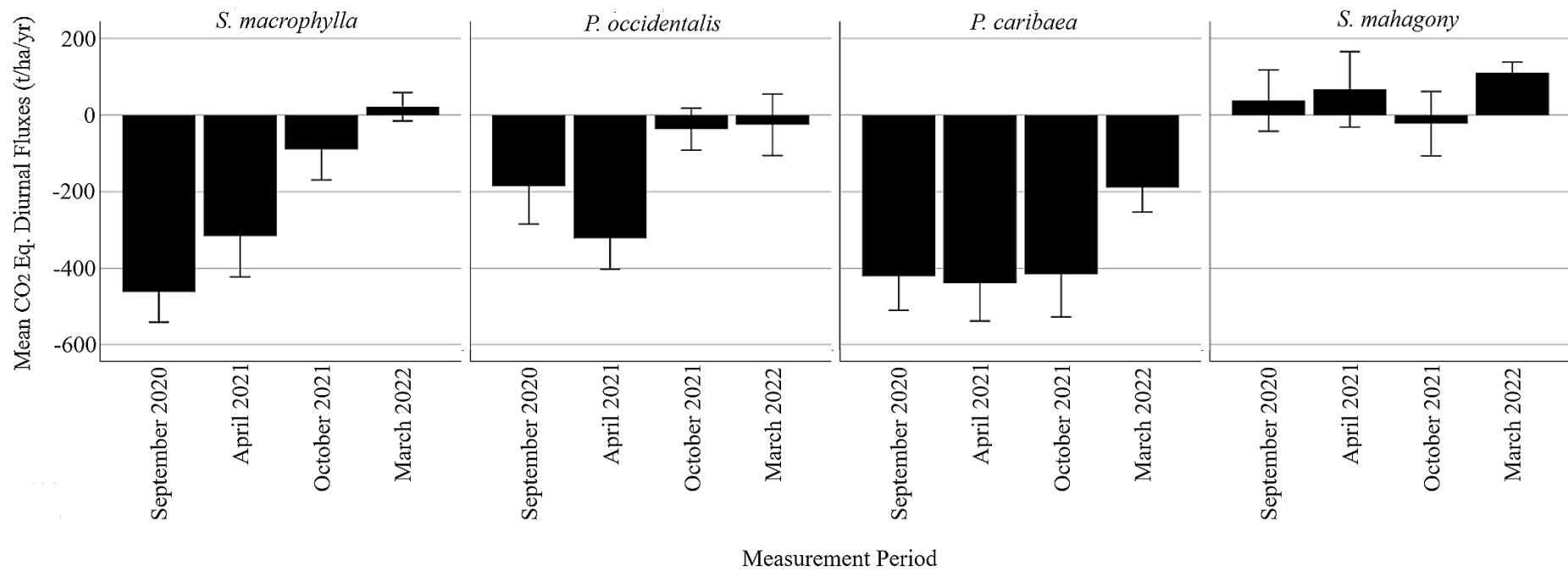
Maximum CO_2 Eq. flux averages were realized for all species in March 2022. In *S. macrophylla*, these two variables were positively correlated (r = 0.502, $p = 0.05$). No significant correlations existed between CO_2 Eq. flux averages and soil temperatures for the rest of the species. Higher temperatures generally increase microbial activity and root respiration, positively influencing the soil's GHG fluxes [79].

3.11. CO_2 Equivalent Balance

On average, all species released BCO_2 Eq. to the atmosphere during the assessed period (1.5 years). The soil BCO_2 Eq. ranged from 72.34 t ha^{-1} in *S. mahagoni* to 299.70 t ha^{-1} in *S. macrophylla*. *P. occidentalis* forest stands were second in releasing, and *P. caribaea* was third (Table 8). Although *S. mahagoni* behaved as a source in all periods, except in October 2021, organic C stocks were significantly higher in soils plus litter under this species.

Table 8. Carbon balance at soil scale, between average reserves (litter and soil) and CO_2 equivalent fluxes in soils under the species *S. macrophylla*, *P. occidentalis*, *P. caribaea*, and *S. mahagoni*, in the 1.5-year period between the first and last assessment.

Species	Reserves	Flux	Balance	Reserves	Flux	Balance	ΔCO_2 Equivalent
	(t CO_2 Eq. ha^{-1})	(t CO_2 Eq. ha^{-1} year^{-1})	(t CO_2 Eq. ha^{-1})	(t CO_2 Eq. ha^{-1})	(t CO_2 Eq. ha^{-1} year^{-1})	(t CO_2 Eq. ha^{-1})	(t CO_2 Eq. ha^{-1})
		September 2020			March 2022		1.5 Years
S. macrophylla	170.02	−0.463	633.30	205.49	21.74	183.75	299.70
P. occidentalis	174.33	−185.47	359.80	179.13	−25.20	204.33	103.64
P. caribaea	172.13	−420.12	592.25	183.24	−189.39	372.64	146.41
S. mahagoni	265.59	37.97	227.62	229.26	110.15	119.10	72.34

Figure 5. Periodic dynamics of CO_2 equivalent fluxes in the soils under four stands of *S. macrophylla*, *P. occidentalis*, *P. caribaea*, and *S. mahagoni*.

Harris et al. [87] estimate that global forests were a net C sink of $-7.6\,\mathrm{E} + 09 \pm 4.9\,\mathrm{E} + 10\,\mathrm{t}\,CO_2\,\mathrm{Eq.\,yr^{-1}}$, reflecting a balance between gross C removals ($-1.56\,\mathrm{E} + 10 \pm 4.9\,\mathrm{t}\,CO_2\,\mathrm{Eq.\,yr^{-1}}$) and gross emissions from deforestation and other disturbances ($8.1\,\mathrm{E} + 09 \pm 2.5\,\mathrm{E} + 09\,\mathrm{t}\,CO_2\,\mathrm{Eq.\,yr^{-1}}$). Rubaiyata et al. [88] found CO_2 Eq. uptakes of approximately $-6.58\,\mathrm{E^{-2}}$ tons per hectare in temperate forest stands in Goettingen, Germany.

In old-growth tropical forests in Malaysian Borneo, a region that is a global hotspot for emission from forest degradation, annual soil respiration was equal to the organic C inputs into the soil with differences between respiration and inputs on the order of 0.66 t CO_2 Eq. ha^{-1} year^{-1} [89]. Betula pendula forest stands in Estonia's pole, and the middle-aged and premature stages within, exhibited soil organic C budgets of -0.33, -2.13, and -2.97 t CO_2 Eq. ha^{-1} year^{-1}, respectively [90]. Overall, forests worldwide are estimated to absorb about 7.6 billion metric tons of CO_2 annually, acting as a net C sink of roughly 1.5 times the annual emissions from the entire United States [91]. The calculations for BCO_2 Eq. made for the four species were as follows:

S. macrophylla

$$\Delta CO_2 Eq._{SM} = \frac{\left\{170.02\left(\mathrm{t\,ha^{-1}}\right) - (-0.46)\left(\mathrm{t\,ha^{-1}year^{-1}}\right)\right\} - \left\{205.49\left(\mathrm{t\,ha^{-1}}\right) - (21.74)\left(\mathrm{t\,ha^{-1}year^{-1}}\right)\right\}}{1.5\;a\tilde{n}os}$$

$$\Delta CO_2 Eq._{SM} = 633.30 - 183.75/1.5 = 299.70\;\mathrm{tCO_2\,Eq.ha^{-1}}$$

P. occidentalis

$$\Delta CO_2 Eq._{SM} = \frac{\left\{174.33\left(\mathrm{t\,ha^{-1}}\right) - (-185.47)\left(\mathrm{t\,ha^{-1}year^{-1}}\right)\right\} - \left\{179.13\left(\mathrm{t\,ha^{-1}}\right) - (-25.20)\left(\mathrm{t\,ha^{-1}year^{-1}}\right)\right\}}{1.5\;a\tilde{n}os}$$

$$\Delta CO_2 Eq._{PO} = 359.80 - 204.33/1.5 = 103.64\;\mathrm{tCO_2\,Eq.ha^{-1}}$$

P. caribaea

$$\Delta CO_2 Eq._{SM} = \frac{\left\{172.13\left(\mathrm{t\,ha^{-1}}\right) - (-420.12)\left(\mathrm{t\,ha^{-1}year^{-1}}\right)\right\} - \left\{183.24\left(\mathrm{t\,ha^{-1}}\right) - (-189.39)\left(\mathrm{t\,ha^{-1}year^{-1}}\right)\right\}}{1.5\;a\tilde{n}os}$$

$$\Delta CO_2 Eq._{PC} = 592.25 - 372.64/1.5 = 146.41\;\mathrm{tCO_2\,Eq.ha^{-1}}$$

S. mahagoni

$$\Delta CO_2 Eq._{SM} = \frac{\left\{265.59\left(\mathrm{t\,ha^{-1}}\right) - (37.97)\left(\mathrm{t\,ha^{-1}year^{-1}}\right)\right\} - \left\{229.26\left(\mathrm{t\,ha^{-1}}\right) - (110.15)\left(\mathrm{t\,ha^{-1}year^{-1}}\right)\right\}}{1.5\;a\tilde{n}os}$$

$$\Delta CO_2 Eq._{SMg} = 227.62 - 119.10/1.5 = 72.34\;\mathrm{tCO_2\,Eq.ha^{-1}}$$

Limitations arose during the course of our research. Unfortunately, we could not conduct a full experiment because our work was on private property, so we were restricted to an observational study. Our observations were limited to daytime measurements due to vandalism in the area. Time was another constraint in developing the study. The measurements were taken over a short period because of ownership restrictions. For instance, understanding the influence of atmospheric and soil variables on greenhouse gas fluxes and organic carbon stocks is crucial. However, due to time constraints, we could not take the necessary measurements to establish a proper relationship between these variables. Readers need to be aware of this limitation.

In addition to addressing the limitations mentioned earlier, future research should prioritize overcoming the lack of comprehensive data on below-ground processes, such as decomposition, humification, and leaching. There are uncertainties in estimating forest

carbon potentials, and there is a need for improved data sampling in tropical regions for soil carbon. It has been emphasized that forest conservation and sustainable management are crucial for maintaining and enhancing soil carbon stocks as part of policy integration to promote nature-based climate solutions.

4. Conclusions

Averaged total organic C stocks were higher under *S. mahagoni*, realizing an average (±standard error) of 62.37 ± 4.17 t ha^{-1}. This highlights the prominence of *S. mahagoni* in capturing and storing organic carbon within the study area. Following a decreasing order, the averages (±standard errors) for the rest of the species were 46.30 ± 3.83 t ha^{-1} (*S. macrophylla*), 62.37 ± 4.17 t ha^{-1} (*P. occidentalis*), and 62.37 ± 4.17 t ha^{-1} (*P. caribaea*). The average flux rates of CO_2 Eq. fluxes to the atmosphere were approximately 65.4, 51.1, and 75.9 times higher in *S. macrophylla* soils compared to the respective rates of *P. occidentalis*, *P. caribaea*, and *S. mahagony*. *P. caribaea* stands had the lowest absolute magnitude of CO_2 Eq. fluxes (t ha^{-1} year^{-1}). A positive C balance indicates C being released into the atmosphere. Based on this fact, all species released CO_2 Eq. to the atmosphere, with *S. mahagoni* being the species with the lowest release amounts (72.34 t of CO_2 Eq. ha^{-1} yr^{-1}). At the other extreme, we found that *S. macrophylla* released 299.70 t of CO_2 Eq. ha^{-1} yr^{-1}.

All species showed a general upward pattern in soil respiration from September 2020 to March 2022. Maximum CO_2 Eq. flux averages were realized for all species in March 2022. CO_2. Eq. fluxes in *S. macrophylla* were significantly lower in September 2020 than in October 2021 and March 2022. Soil temperatures in *S. macrophylla* were also highest in March 2022, with an average (±standard error) of 28.38 ± 0.69 °C. Additionally, in *P. occidentalis*, there were statistically significant differences in periodical CO_2 Eq. average fluxes. In the soils of this species, fluxes in April were significantly lower than in October 2021 and March 2022.

The hypothesis regarding OC stocks in soils under broadleaved and coniferous species was confirmed. Coniferous forests had a higher organic C content in litter (8.54 ± 0.55 versus 7.95 ± 0.61 t ha^{-1}), while deciduous forests had higher SOC levels in the mineral soil (46.38 ± 2.91 versus 34.33 ± 1.59 t ha^{-1}). Combining both reservoirs, the average (±standard error) organic C stock was higher in broadleaved species by 26.76% (54.34 ± 3.13 t ha^{-1}) than in conifers (42.87 ± 1.71 t ha^{-1}), with this difference being significant.

Despite limitations in quantification due to unavoidable errors in calculations and measurement accuracy, the methods used to estimate organic C stocks and CO_2 Eq. fluxes can be applied to other ecosystems in future research. Our findings provide comprehensive information on soil organic C stocks, helping to understand the OC storage dynamics in these forest species. This understanding is crucial for reforestation efforts and CC mitigation in the Dominican Republic.

Quantifying the soil C budget in tropical forests is a complex task. However, we hope this research has provided some insights into the effects of forest species on the C balance within the soil. Tropical forests are among the most C-abundant ecosystems in the world, with significant C stored in both the above-ground biomass and soil. This essential soil C pool represents forest ecosystems' most significant C pool. However, accurately determining soil C budgets is challenging, particularly in tropical forests, where resources at the disposal of researchers are scarce. More field data on C budgets and stocks in tropical forests are still needed, especially following land-use changes. Long-term evaluations using both model simulations and field observations are crucial for understanding the effects of climate and land-use conversion on C budgets in tropical forest ecosystems.

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