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Research article

Carbon storage and ecological characteristics of restored and natural mixed deciduous forests in western Thailand

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Abstract

Importance of the work: Assessment of carbon storage and ecological characteristics in mixed deciduous forests is crucial for understanding forest succession and mitigating global warming.

Objectives: To examine differences in carbon storage and ecological characteristics between mining-disturbed and undisturbed mixed deciduous forests in Ratchaburi province, western Thailand.

Materials and Methods: In 2022, 12 plots (each 20 m × 20 m) were established in natural (NF) and restored (RF) mixed deciduous forests in Ratchaburi province, Thailand. Within each plot, all trees having a diameter at breast height ≥ 4.5 cm were inventoried. Ecological parameters (density, basal area, importance value index, diversity) and forest carbon stocks (aboveground biomass, soil carbon) were assessed.

Results: In total, 658 trees were identified, representing 67 species from 22 families. The composition differed between the two studied forests, with 48.06% similarity in species. The Shannon-Wiener diversity was 2.90 in the RF and 3.36 in the NF. The mean basal area (± SD) was 231.67 ± 71.15 m^2/ha in the RF and 416.41 ± 261.59 m^2/ha in the NF. Mean tree height (± SD) was 10.51 ± 3.72 m in the RF and 13.97 ± 4.41 m in the NF. Mean tree density (± SD) was 1,779.17 ± 729.97 trees/ha in the RF and 962.50 ± 147.27 trees/ha in the NF. The mean total carbon stock (± SD) was higher in the NF (366.27 ± 76.51 Mg C/ha) than in the RF (194.14 ± 45.80 Mg C/ha).

Main finding: The restored forest had lower carbon storage and ecological parameters, indicating long-term mining impacts. However, the restored forest still served as a carbon reservoir, contributing to the reduction of atmospheric CO_2 levels and preserving biodiversity after prolonged restoration.

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Introduction

Forests cover 31% of the global land area (FAO, 2020), with nearly one-half being tropical forests, which are characterized by complex structures, great biodiversity and high carbon stocks (Baker and Bunyavejchewin, 2017; Sullivan et al., 2017). Tropical forests provide essential ecosystem services such as climate regulation, water cycling, and food resource production (Good et al., 2015; Lewis et al., 2015). In particular, they serve as major global carbon reservoirs, storing an estimated 200–300 Pg of carbon and contributing up to 25% of global terrestrial carbon (Avitabile et al., 2016; Mitchard, 2018). Via photosynthesis, forest trees absorb atmospheric carbon dioxide (CO_2), which is stored in their aboveground biomass (AGB) and in the soil. Quantification of carbon storage is essential for understanding the carbon dynamics of forest ecosystems, in particular AGB and soil organic carbon (SOC), which are important components of the forest carbon pool.

Mixed deciduous forests (MDFs) are the predominant deciduous forest type in Thailand (Chaiyo et al., 2012; Thammanu et al., 2021). Other research has highlighted the MDFs as a major carbon sink for the country (Kaewkrom et al., 2011; Chaiyo et al., 2012; National Park Research Center, 2019; Chanlabut and Nahok, 2022). However, these forests often face anthropogenic disturbances, such as frequent fires and land use changes, resulting in major degradation and CO_2 emissions that threaten biodiversity and ecosystem services (Kachina et al., 2017; Asanok et al., 2020).

Approximately 20% of the western region of Thailand is covered by MDFs. Notably, the MDFs in Suan Phueng district, Ratchaburi province have been recognized as a natural corridor connecting two major forest complexes: the Western Forest Complex and the Kaeng Krachan Forest Complex (Chanlabut and Nahok, 2022). Furthermore, this area serves as a bridge for fauna and flora moving between several biogeographical regions, including Indo-Chinese, Sino-Malayan, Indo-Burmese, and Eastern Indian. Historically, these MDFs in Ratchaburi have experienced substantial encroachment and degradation due to intensive mining activities (Lehmann and Mahawat, 1989; Schwartz et al., 1995). Mining activities are known to cause long-term environmental impacts on forest ecosystems (Irzon et al., 2018). Studies in Indonesia have shown that tin mining leads to edaphic degradation, affecting soil structure, texture and organic matter (Oktavia et al., 2015; Sukarman et al., 2020). However, since mining ceased in the 1980s (Thoburn, 1994), many former mining areas have gradually transitioned back to forest. Despite a prolonged recovery period, information on carbon storage and the ecological characteristics of MDFs, remains limited, particularly in former mining areas. Further, ecological succession in restored forests can be evaluated through improvements in biodiversity, biomass, soil organic matter, and the reappearance of rare and keystone species that define the target ecosystem (Elliott et al., 2013). Such information is essential to understand the current carbon storage capacity of these forests and their structural characteristics.

To fill this gap, the current research aimed to examine carbon storage and the ecological characteristics of mining-disturbed MDFs in western Thailand and to compare them with undisturbed MDFs. Restoring forests on degraded lands can enhance carbon sequestration in both the biomass and soil (Elliott et al., 2013). The current study hypothesized that MDFs previously disturbed by mining activities would have lower carbon storage and altered forest structural attributes compared to undisturbed MDFs. The findings from this study should enhance understanding of the variations in ecological parameters and forest carbon stocks under different disturbance regimes within MDFs.

Materials and Methods

Site description

The study was conducted in the MDFs within the Suan Phueng Nature Education Park, Suan Phueng district, Ratchaburi province, western Thailand, covering approximately 200 km^2. It is mostly characterized by mountainous terrain with elevations in the range 200–400 m above mean sea level and is part of the Tenasserim Hills. This region experiences a tropical climate, with a mean annual temperature of 29.0°C and a range of 17.9–39.1°C and the mean annual precipitation is 1,226.9 mm. There is a distinct wet season from May to October and a reduced rainfall period between June and July (Chanlabut and Nahok, 2022). This area was partially subjected to mining, logging and encroachment from 1913 to 1999. Although it has been restored and has been undergoing natural regeneration for decades, traces of mining disturbances remain (Lehmann and Mahawat, 1989; Schwartz et al., 1995).

This study focused on two different locations: restored mining-disturbed MDFs and undisturbed MDFs (Fig. 1). These sites are located in two different sub-watersheds of the Phachi watershed, a sub-basin of the Mae Klong Basin. The restored forest is located in the Huai Phak watershed in an area that has been abandoned from tin mining for decades and natural restoration has occurred. The undisturbed MDFs are located in the Huai Khok Mu watershed, alongside the route to the Huai Khok Mu viewpoint. This area shows no evidence of mining disturbance and serves as a reference site for natural forest conditions. These two sites were selected to examine the carbon storage capacity and ecological characteristics of mining-disturbed MDFs and to compare them with undisturbed MDFs. The mining-disturbed MDF site was referred to as the restored forest (RF), while the undisturbed MDF site was designated as the natural forest (NF), as shown in Fig. 1.

Data collection

The data collection was carried out in April 2022. In total, six plots (each 0.04-ha; 20 m × 20 m) were randomly established to study carbon storage and ecological characteristics in each of the two MDF sites. Data collection consisted of soil samples and trees with a diameter at breast height (DBH) ≥ 4.5 cm. The inventoried tree data were used to analyze aboveground carbon storage and ecological characteristics. The soil samples were used to analyze soil carbon storage. All trees with DBH ≥ 4.5 cm were identified. The DBH was measured using a diameter tape (Lufkin W606PM) and the tree height was measured using the principle of triangulation with a clinometer (Suunto PM-5/360 PC). At the center of each quadrat, a sub-plot for soil collection was established using a triangular sampling design. Three replicate soil samples were obtained at two depth levels (0–30 cm and 30–100 cm). The three replicates for each depth were homogenized to form composite samples, which were then bagged for soil carbon content measurement. Undisturbed soil core samples were collected at the center of the triangular plot using 260 cm^3 stainless steel tubes (6 cm in diameter) to preserve the soil structure for subsequent bulk density determination.

Ecological characteristic analysis

Ecological characteristics for each MDF site were examined using tree data from the six plots, measuring parameters consisting of tree density, basal area, importance value index (IVI) and diversity indices (Gardener, 2014; Krebs, 2014). Species diversity was calculated using the Shannon-Wiener diversity index (Shannon, 1948), while the Simpson index measured species dominance (Simpson, 1949). The Sorensen index evaluated similarity between the two forest communities (Sørensen, 1948). The IVI was calculated to determine the relative ecological importance of species in communities using the equation following Curtis and McIntosh (1951)

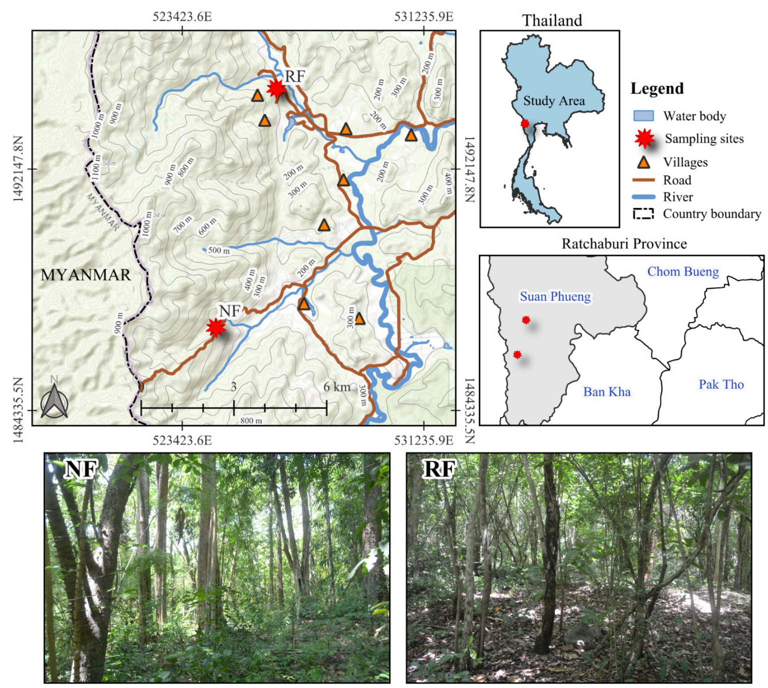

Fig. 1 Location of study sites and their appearance, where NF = natural forest site and RF = restored forest site following mining disturbance

Carbon storage and CO_2 absorption analysis

The total carbon stock of the MDFs was quantified by estimating the carbon stocks in the aboveground biomass and soils which are the main components of carbon storage in forest ecosystems (Lal, 2005). The soil carbon stock of the forest was estimated using the soil organic carbon content obtained based on the wet oxidation method (Walkley and Black, 1934) and soil bulk density was obtained using the core method (Blake and Hartage, 1986). Then, the soil carbon stock was calculated following Ellert et al. (2000).

The AGB estimates were derived from allometric equations that relate to tree height and diameter. The allometric equations were developed by Ogawa et al. (1965), as shown in equations 1–4:

$$AGB = W_s + W_b + W_l \quad (1)$$

$$W_s = 0.0396 \times (D^2H)^{0.9326} \quad (2)$$

$$W_s = 0.003487 \times (D^2H)^{1.027} \quad (3)$$

$$W_l = \left(\frac{28.0}{W_{tc}} + 0.025\right)^{-1} \quad (4)$$

where *AGB* represents the aboveground biomass, W_s represents the stem biomass weight, W_b represents the branch biomass weight and W_l represents the leaf biomass weight, with all parameters measured in kilograms. W_{tc} represents the sum of W_s and W_b. The weight of each component was calculated based on the DBH (*D*, measured in centimeters) and the height of the tree (*H*, measured in meters).

The aboveground biomass carbon stock (AGBC) was obtained by multiplying the AGB by a conversion factor of 0.47, which was developed by the Intergovernmental Panel on Climate Change (2006).

The capability of a forest to absorb atmospheric CO_2 is crucial for mitigating climate change (Kobayashi, 2004). The potential of atmospheric CO_2 absorption through photosynthesis ($MgCO_2eq$) was calculated by multiplying by a conversion factor of 44/12 (Intergovernmental Panel on Climate Change, 2006).

Statistical analysis

The normality assumption and equality of variances were assessed using the Shapiro-Wilk test. Mean values of ecological parameters (density, basal area, tree height, AGB, AGBC) were compared using Student's t test at the 95% confidence interval. Results were presented as mean ± SD values.

Results and Discussion

Ecological characteristics

Species composition and diversity

The quantitative ecological parameters are summarized in Table 1. In total, 658 trees from 67 species and 22 families were recorded in both forests. The RF had 427 trees (47 species, 18 families) with the Fabaceae (34.04%) being the most abundant. The NF had 231 trees (43 species, 19 families) with the Fabaceae (18.60%) being again the most abundant. The RF had a lower basal area and mean tree height than the NF but had a higher tree density ($p < 0.05$).

Table 1 Quantitative ecological characteristics of natural (NF) and restored forest (RF) communities

Ecological parameter	Forest community	
	NF	RF
Number of species	43	47
Number of families	19	18
Sorensen index (%)	48.06	
Shannon index	3.36	2.90
Simpson index	0.05	0.12
Density (trees/ha)	962.50 ± 147.27^b	1,779.17 ± 729.97^a
Basal area (m^2/ha)	416.41 ± 261.59^a	231.67 ± 71.15^b
Tree height (m)	13.97 ± 4.41^a	10.51 ± 3.72^b

Mean ± SD values with different lowercase superscripts are significantly ($p < 0.05$) different.

The Shannon-Wiener index was lower in the RF than in the NF. This lower index in the restored forest was likely due to the past several decades of mining activities (Nero, 2021), as species diversity is often linked to disturbance levels (Zambrano et al., 2019). The Simpson index of both communities was below 0.2, indicating no dominant species in either forest community (Chettri et al., 2023). This lack of dominance supported an even distribution of species, contributing to a more diverse and resilient ecosystem (Baliton et al., 2020). The Sorensen index showed a 48.06% overlap between the NF and RF communities, indicating that almost one-half of the species were shared between the two communities, as well as reflecting that the recovery in the restored forest was to approximately one-half of its original natural species composition. However, the RF might have retained some of its original species composition, indicating the ability of certain species to thrive after disturbance. Chazdon (2008) suggested that forests recovering from mining activity often exhibit different species compositions compared to undisturbed natural forests. The differences in ecological parameters suggest important differences in the community structure and species composition between the two communities.

Importance value index

The IVI of both communities is shown in Table 2. In the NF, *Parkia sumatrana* had the highest IVI, followed by several other species: *Harrisonia perforata*, *Oroxylum indicum* and *Croton* spp. In the RF, *Cleistanthus gracilis* had the highest IVI, followed by *Lagerstroemia floribunda*, *Vitex quinata* and *Croton persimilis*. The RF was predominantly characterized by smaller trees, while the NF site was dominated by larger ones. The difference in dominant species between the two MDF communities indicated their distinct species composition and ecological structure. These variations were likely influenced by varying environmental conditions, disturbance regimes and successional stages. The dominance of *C. gracilis* on the RF site suggested that it was a pioneer species in the secondary succession of this restored forest. Observation during the survey revealed that it is a shrub, mostly dense and distributed in the understory of taller trees, such as *L. floribunda*. *C. gracilis* could be further classified as shade-tolerant, indicating that the restored forest site was undergoing late successional stages (Matsuo et al., 2021). In contrast, the NF was dominated by trees with large basal areas, including *P. sumatrana*, *H. perforata*, *O. indicum*, *Croton* spp. and *C. persimilis*. These large dominant trees are characteristic of natural forest (Marod, 1999). These differences in species composition may also have implications for the ecosystem functions and services provided by these two forest communities. Research has suggested that ecosystem functions, including biotic structure and biomass, are significantly related to species composition (Carrick and Forsythe, 2020). Thus, species composition could contribute to differences in litter decomposition rates, which in turn influence carbon storage in both biomass and soil.

Table 2 Importance value index of top-10 tree species for natural (NF) and restored forest (RF) sites

Forest site	Relative (%)			IVI	%IVI
Species	R_F	R_D	R_A		
NF					
Parkia sumatrana	5.15	10.82	5.75	21.73	7.24
Harrisonia perforata	5.15	5.19	10.55	20.90	6.97
Oroxylum indicum	2.06	6.06	10.68	18.80	6.27
Croton spp.	4.12	7.79	6.84	18.76	6.25
Croton persimilis	3.09	5.63	6.61	15.33	5.11
Sterculia guttata	5.15	6.06	3.51	14.72	4.91
Garuga pinnata	4.12	4.76	4.51	13.39	4.46
Pterocarpus macrocarpus	4.12	6.49	2.45	13.07	4.36
Lagerstroemia floribunda	4.12	3.90	2.10	10.12	3.37
Cleistanthus gracilis	2.06	2.60	5.21	9.86	3.29
Sum of 10 species	39.18	59.31	58.19	156.67	52.22
Sum of total species	100.00	100.00	100.00	300.00	100.00
RF					
Cleistanthus gracilis	5.56	29.27	21.83	56.66	18.89
Lagerstroemia floribunda	6.67	10.07	15.09	31.83	10.61
Vitex quinata	4.44	9.84	6.33	20.62	6.87
Croton persimilis	4.44	3.75	3.78	11.97	3.99
Dalbergia cana	4.44	3.28	2.85	10.57	3.52
Cleistanthus papyraceus	4.44	3.98	2.08	10.51	3.50
Vitex pinnata	1.11	3.28	4.70	9.09	3.03
Spondias bipinnata	4.44	1.17	3.01	8.63	2.88
Caesalpinia sappan	2.22	2.58	3.53	8.33	2.78
Fernandoa adenophylla	3.33	2.58	2.00	7.91	2.64
Sum of 10 species	41.11	69.79	65.21	176.11	58.70
Sum of total species	100.00	100.00	100.00	300.00	100.00

R_F = Relative Frequency; R_D = Relative Density; R_A = Relative Abundance; IVI = Importance Value Index.

Diameter at breast height distribution

The distribution of DBH for both communities are presented in Fig. 2A. Both communities showed a reversed J-shape that differed markedly in their DBH class percentages. The RF had a higher proportion of smaller DBH classes, particularly 4.5–20 cm (accounting for 93% of the individuals) versus 67% in the NF. This indicated a secondary growth stand in the RF due to past disturbance, leading to a dense population of small trees (Lin et al., 2018). Conversely, the NF had a significantly higher percentage of large trees (DBH > 21 cm) at 33% compared to 7% in the RF. The trees with DBH ≥ 60 cm were also more prevalent in the NF (2% versus 0.23% in the RF). Trees with their DBH in the 51–60 cm and ≥ 71 cm classes were exclusively found in the NF. Differences in the DBH distributions between communities reflect varied forest structures, influenced by differing times since major disturbances (Johnson and Weigel, 1990). Typically, larger DBH trees indicate older, mature stands (O'Hara and Nagel, 2013). However, in the RF disturbed by mining, the large trees had been replaced by smaller ones.

Aboveground biomass carbon

The AGB for both communities is summarized in Table 3. The results showed differences in AGB and AGBC between the NF and RF ($p < 0.05$), aligning with studies that natural forests have higher biomass and carbon storage. For example, Pan et al. (2011) reported that mature forests sequestered more carbon due to their complex structure and species diversity, unlike recently restored forests.

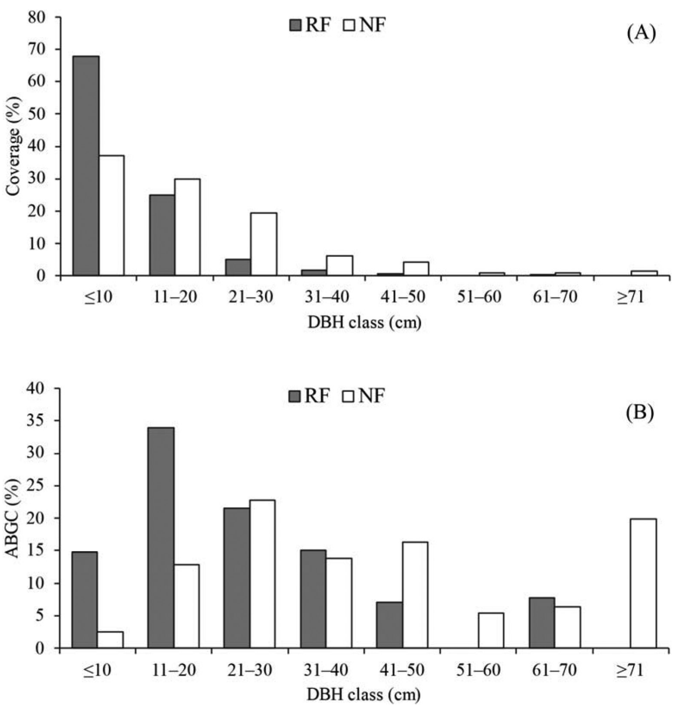

Fig. 2 Distribution of diameter at breast height (DBH) in natural forest (NF) and restored forest (RF): (A) percentage coverage; (B) percentage of aboveground carbon (ABGC) by DBH class

The differences in AGB and AGBC were due to distinctions in the DBH and height—key variables affecting aboveground biomass (Pothong et al., 2022). In the NF, DBH was in the range 4.9–94.0 cm with a mean of 19.58 ± 9.85 cm, whereas in the RF, it was in the range 4.5–67.0 cm with a mean of 11.72 ± 4.26 cm. The height of the NF trees was in the range 6.3–21.6 m with a mean of 14.18 ± 2.84 m, whereas in the RF, it was in the range 5.9–21.4 m with a mean of 11.23 ± 2.53 m. NF trees had higher values for both DBH and height compared to RF trees ($p < 0.05$). The larger and taller trees indicated greater maturity than the smaller ones.

The species storing both AGB and AGBC in the two communities are summarized in Table 4. The AGBC of the two forest communities had different species compositions. In the NF, *O. indicum*, *H. perforata* and *Lithocarpus* sp. were dominant compared to *C. gracilis*, *L. floribunda* and *V. quinata* in the RF, indicating differing species contributions to AGBC in each community.

The AGB results in the current study area were notably different compared to other regions within the country. For example in the same province, the AGB of a natural mixed deciduous forest in the Mae Nam Phachi Wildlife Sanctuary was estimated at 77.54 ± 55.16 Mg/ha (Chaiyo et al., 2012). Although this value was reported in 2012 and may not directly correspond to the current findings, it is crucial to recognize that the species composition and ecological variables in that study also differed from those in the current study. In the mixed deciduous forests of the Ban Mae Chiang Rai Lum Community Forest in Lampang province, northern Thailand, the AGB was reported at 69.67 ± 21.83 Mg/ha (Thammanu et al., 2021), which is lower than the value observed for the restored forest site in the current study (Table 3). This difference could be attributed to variations in the DBH size of the trees between the communities.

Table 3 Comparison of above ground biomass (AGB) and ABG carbon (ABGC) for natural (NF) and restored forest (RF) sites

Parameter	Forest site	
	NF	RF
AGB (Mg/ha)	245.61 ± 163.18^{a}	119.91 ± 43.47^{b}
AGBC (Mg C/ha)	115.44 ± 76.69^{a}	56.36 ± 20.43^{b}

Mean ± SD values with different lowercase superscripts are significantly ($p < 0.05$) different.

Table 4 Top-10 ranked species in natural (NF) and restored forest (RF) sites by above ground biomass (AGB) and ABG carbon (ABGC)

Forest site	Species	AGB (Mg/ha)	ABGC (Mg C/ha)	% of ABGC total
NF	*Oroxylum indicum*	27.38	12.87	11.15
	Harrisonia perforata	26.52	12.46	10.80
	Lithocarpus sp.	18.74	8.81	7.63
	Caesalpinia sappan	17.57	8.26	7.15
	Croton spp.	16.64	7.82	6.78
	Croton persimilis	16.38	7.70	6.67
	Cleistanthus gracilis	13.43	6.31	5.47
	Parkia sumatrana	13.29	6.25	5.41
	Garuga pinnata	11.09	5.21	4.52
	Sterculia guttata	8.08	3.80	3.29
	Other species	76.48	35.94	31.14
RF	*Cleistanthus gracilis*	25.12	11.81	20.95
	Lagerstroemia floribunda	19.23	9.04	16.03
	Vitex quinata	6.73	3.17	5.62
	Acacia auriculiformis	6.02	2.83	5.02
	Vitex pinnata	5.81	2.73	4.85
	Croton persimilis	4.51	2.12	3.76
	Caesalpinia sappan	4.23	1.99	3.53
	Spondias bipinnata	4.06	1.91	3.38
	Peltophorum dasyrrhachis	3.77	1.77	3.15
	Dalbergia cana	3.50	1.64	2.92
	Other species	36.93	17.36	30.80

Soil carbon

Soil organic carbon content

The SOC contents at 0–30 cm and 30–100 cm depths for both communities are presented in Fig. 3A. The NF had a higher mean SOC at 0–30 cm (22.63 ± 8.31 g C/kg) than at 30–100 cm (17.60 ± 5.82 g C/kg; $p < 0.05$). In contrast, the RF had consistent SOC levels across the soil depth (14.72 ± 0.54 g C/kg and 14.96 ± 0.35 g C/kg, respectively; $p > 0.05$). The SOC contents at a soil depth of 0–100 cm is presented in Fig. 3B. The mean SOC content in the NF was 20.12 ± 7.33 g C/kg, while in the RF it was 14.80 ± 0.48 g C/kg ($p < 0.05$). These findings highlighted the significant variations in the SOC contents between the two communities, indicating differing amounts of SOC between the two soil depths in the NF, while similar amounts were observed in the RF. Overall, there was a significant difference in SOC between the two forest communities.

The similar SOC contents in the two soil depths in the RF suggested that disturbed forests may have a more uniform soil carbon distribution due to disturbances homogenizing the soil properties between soil profiles. However, the higher SOC content in the topsoil is usually observed in natural forest (Jobbágy and Jackson, 2000).

The difference in the SOC content between the two forest communities could be attributed to the disruption of the natural soil profile and vegetation in the RF, which has implications for the long-term accumulation and stabilization of soil organic matter (Silveira, 2005). Furthermore, undisturbed natural forests often have higher SOC levels due to the accumulation of organic matter over time and minimal disturbance to soil structure (Xu et al., 2022).

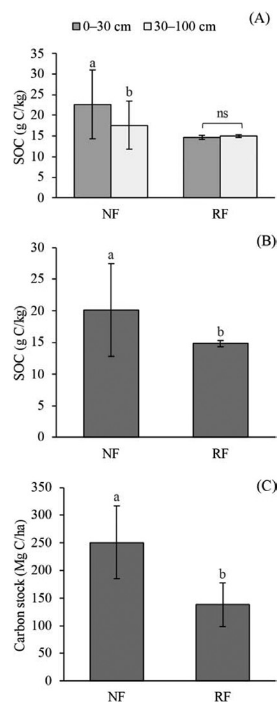

Fig. 3 Soil carbon in natural forest (NF) and restored forest (RF): (A) soil organic carbon (SOC) at two depth intervals; (B) SOC at 0–100 cm; (C) SOC stock, where different lowercase letters above columns indicate significant ($p < 0.05$) differences, ns indicates no significant difference and the error bars indicate ± SD

Soil organic carbon stocks

The SOC stocks are shown in Fig. 3C, being substantially higher in the NF ($p < 0.05$), at 250.83 ± 65.72 Mg C/ha (170.21–368.70 Mg C/ha), compared to the RF at 137.78 ± 39.56 Mg C/ha (77.40–172.41 Mg C/ha). The SOC stock in the RF was approximately 55% lower than that stored in the RF, which could be attributed to the loss of soil organic matter and the disruption of carbon cycling during the mining process, which reduced the ability of the soil to store and sequester carbon. As observed by Ingram et al. (2015), mining activities lead to significant losses of SOC. The lower SOC levels in the RF than in the NF suggested that the mining disturbance has had long-lasting impacts on forest soil carbon stocks.

Litter production and decomposition are essential ecosystem processes that play a crucial role in nutrient cycling within mixed deciduous forests (Marod et al., 2017). Slow decomposition rates lead to the accumulation of organic matter in the soil (Krishna and Mohan, 2017). Wongprom et al. (2022) showed that restored forests have higher litter decomposition rates than natural forests. Although the current study did not investigate the litter decomposition rates between the two communities, it could be hypothesized that the decomposition ability may differ between restored and natural sites. Furthermore, litter decomposition rates could vary depending on the species composition (Marod et al., 2017), which differed between the two forest communities in the current study. Therefore, the difference in SOC stocks between the two communities could be attributed in part to variations in the litter decomposition rates and species composition, which in turn affected the rate of litter decomposition.

Total carbon stock and CO_2 absorption

The total carbon stocks and CO_2 absorption are summarized in Table 5. Both parameters were different between the two forest communities ($p < 0.05$). The differences in total carbon stocks and CO_2 absorption between the NF and RF may have been due to variations in the community structure, aboveground biomass, and soil carbon stock. Elsewhere, it has been suggested that natural forests store more carbon than restored forests, with Pan et al. (2011) reporting that mature forests had higher carbon stocks due to their well-developed structure and resulting higher biomass. In particular, natural forests are characterized by a higher proportion of large, mature trees that can contribute significantly to aboveground biomass, which in turn acts as a substantial carbon sink by sequestering atmospheric CO_2. In contrast, restored forests mostly consist of younger, smaller trees that have not yet developed the extensive biomass necessary for high carbon storage. Mildrexler et al. (2020) demonstrated that larger trees with greater DBH values could store more carbon per area, despite having a lower stocking density.

In the current study, the total carbon stock of the RF was approximately 50% lower than that of the NF, indicating that restored forests could achieve biomass carbon levels comparable to those of natural forests (Kavinchan et al., 2015). However, the SOC and total carbon stock levels in restored forest were reported to have recovered to levels still less than one-half of those in an unmined reference site (Avera et al., 2015). Soil carbon may remain lower in restored forests due to soil disturbance (Wang et al., 2017), suggesting that soil carbon recovery may lag behind biomass carbon recovery (Zhang et al., 2019).

While the current study did not directly investigate the relationship between species diversity and carbon storage in forest communities, other research in tropical forests has indicated that species diversity could enhance the aboveground biomass through functional diversity and dominance (Poorter et al., 2015; Mensah et al., 2016). This suggests that the higher species diversity in the NF likely contributed to its greater biomass and AGBC than in the RF.

In conclusion, there were differences in ecological characteristics and potential for carbon storage between natural and restored mixed deciduous forests. The variation in carbon storage and CO_2 absorption between the two communities was attributed to differences in species composition, DBH distribution, species diversity and soil carbon accumulation. This variation was also a result of the long-term impacts of mining activities, which may require many decades for restored sites to approach the ecological function of undisturbed natural forests. However, the restored forest may provide opportunities for future CO_2 absorption as the younger regenerating vegetation continues its rapid growth, while its soil may require more time to restructure to its natural state. Finally, the natural forest acted as a more efficient carbon sink, underscoring its greater potential role in mitigating climate change through enhanced atmospheric CO_2 sequestration.

Table 5 Comparison of total carbon stocks and CO_2 absorption in natural (NF) and restored forest (RF) sites

Parameter	Forest site	
	NF	RF
Total carbon stock (Mg C/ha)	366.27 ± 76.51^{a}	194.14 ± 45.80^{b}
CO_2 absorption (Mg CO_2eq)	1,342.99 ± 280.54^{a}	711.85 ± 167.93^{b}

CO_2eq = carbon dioxide equivalent.

Mean ± SD values with different lowercase superscripts are significantly ($p < 0.05$) different.

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Conflict of Interest

The authors declare that there are no conflicts of interest.

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