Forest Ecology and Management 562 (2024) 121937

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

Growth dynamics of an Amazonian forest: Effects of reduced impact logging and recurring atypical climate events during a 20-year study

Marcus Vinicio Neves d'Oliveira [a,*], Robert Pritchard Miller [b], Luis Claudio Oliveira [a], Evaldo Muñoz Braz [c], Fábio Thaines [d], Jaquelyne Lins Januário [e], Mario Humberto Aravena Acuña [f]

[a] Embrapa Acre, Rodovia BR-364, km 14, Rio Branco, Acre CEP 69900-056, Brazil
[b] Instituto Sociedade, População e Natureza, ISPN, Brasilia, DF, Brazil
[c] Embrapa Florestas, Estrada da Ribeira, Km 111, C. P. 319, Colombo, PR CEP 83411-000, Brazil
[d] Tecman-Florestas, Rua Copacabana, 148, Sala 204, Village W. Maciel, Rio Branco, AC CEP 69.918-500, Brazil
[e] University of Acre (UFAC), Forest Science Graduate Program (PPG-CIFLOR), Rodovia BR 364, Km 4 - Distrito Industrial, Rio Branco, AC CEP 69920-900, Brazil
[f] Forest Science, Post-doc, University of Paraná (UFPR), Rua dos Funcionarios 1540, Juvevê, Curitiba, PR, Brazil

A R T I C L E I N F O

Key words:
Permanent sample plots
Above-ground biomass
Forest management
Commercial timber species

A B S T R A C T

Although forest management with reduced impact logging (RIL) practices is regarded as a way to generate income from tropical forests without losing their overall conservation values, the behavior of forests following logging is a topic that has been little studied in the southwestern Brazilian Amazon (SWA), where the predominant forest type is open with low density of commercial timber species. This study examined the growth dynamics of a logged forest in the SWA through monitoring of permanent sample plots (PSP) over twenty years post logging. Following logging, the forest fully recovered its original tree density (439. ha^{-1}) and aboveground biomass (AGB) stocks (188.4 Mg. ha^{-1}) but did not recover extracted timber volume in the commercial size class (DBH > 50 cm, 28 m^3. ha^{-1} before logging and 22.6 m^3. ha^{-1} in 2022). Tree growth, recruitment and mortality rates presented a high fluctuation during the monitoring period, increasing immediately after logging (0.43 cm. yr.$^{-1}$, 4.1 % and 4.7 % respectively), declining five years after, (0.25 cm. yr.$^{-1}$, 0.7 % and 1.5 % respectively) and presenting a new peak sixteen years after logging (0.35 cm. yr.$^{-1}$, 3.9 % and 7.4 % respectively) which persisted to the end of the study period. Consequently, forest turnover was very high, with stand half-life and doubling life of 29.6 and 27.2 years, respectively. The results show that in terms of forest dynamics and production, the behavior of logged forests was also affected by atypical climate events, such as recurring droughts across Amazonia. Consequently, in the forest type studied, and under the prescribed logging cycle of 25 years, RIL alone was not sufficient to guarantee the sustainability of long-term production of the main timber species harvested. Following the current growth trend (0.46 m^3. ha^{-1}. yr.$^{-1}$ for trees above 10 cm DBH and 0.22 m^3. ha^{-1}. yr.$^{-1}$ for trees above 50 cm DBH) the time to recover the total commercial volume and commercial volume for trees above 10 cm DBH and above 50 cm DBH would be 34 and 45 years respectively. Our findings suggest the need for adoption of silvicultural treatments to increase forest productivity in areas under RIL in SWA.

1. Introduction

The conservation of tropical forests is of global importance due to their high biodiversity (Phillips et al., 1994) and role in carbon storage (Pan et al., 2011; Anderson-Teixeira et al., 2016; Riutta et al., 2018). Tropical forests have been highly impacted by climate events such as droughts, and anthropic disturbances such as fire, logging and, principally, clearing for agriculture, mining, dams, highways and other infrastructure projects, resulting in a mix of deforestation and forest fragmentation and degradation (Lewis et al., 2015). In Brazil, illegal or unregulated logging has resulted in several impacts to native forests: reduction in populations of commercial species, increased fire risk caused by logging debris and canopy opening, damage to future timber stocks by heavy equipment and illegal hunting by logging crews, among others, as well as being, in many cases, the first step in the process of conversion of forests to other land uses (Soares-Filho et al., 2006; Nepstad et al., 1999).

Different from other forms of land use, tropical forest management

* Corresponding author.
E-mail address: marcus.oliveira@embrapa.br (M.V. Neves d'Oliveira).

https://doi.org/10.1016/j.foreco.2024.121937
Received 28 December 2023; Received in revised form 22 April 2024; Accepted 23 April 2024
Available online 11 May 2024
0378-1127/© 2024 Elsevier B.V. All rights reserved.

(TFM), through the use of reduced-impact logging (RIL) practices, produces relatively low impact on the forest's structure (Sullivan et al., 2022), reducing environmental impacts (e.g. Putz et al., 2008 and 2012) and maintaining species composition (Bicknell et al., 2015), since most of the forest is not directly disturbed by logging (Putz et al., 2019, Braz and Mattos, 2015).

In Brazil, RIL was introduced by the end of the 70's (SUDAM/IBDF/PRODEPEF, 1978), but the first regulations for sustainable forest management plans (SFMP) were only established in 1998 (IBAMA, 1998). These regulations require SFMP as the standard practice to be applied in timber harvesting operations in native forests, whether on public lands, such as in logging concessions in national forests and community logging initiatives in the legal reserves of agricultural settlement projects and extractivist reserves, as well as on private lands. Regulations establish cycles of 20–35 years and logging intensities from 15 to 30 m^3. ha^{-1}, based on an assumed post-logging commercial timber volume increment of 0.86 m^3. ha^{-1}. $year^{-1}$. At the moment, there are 18 concession contracts for SFMP in national forests, totaling a little more than a million hectares (Rodrigues et al., 2020) and 268,000 ha of community forest management initiatives (Miranda et al., 2020). In general terms, however, on-the-ground monitoring of compliance with regulations is limited, and there are many cases of fraudulent SFMP, with inflated figures for the volume of prime timber species being used to give a cover of legality to timber illegally harvested from indigenous lands and conservation areas (Valdiones et al., 2022; IBAMA, 2018). These and other practices have been associated with organized crime and violence in rural areas (Greenpeace, 2017; Human Rights Watch, 2019). Nonetheless, regulation, inspection and bureaucratic requirements currently control SFMP in Brazil and tracking of the timber origin is beginning to be used, guaranteeing that, in principle, the production destined for export comes from SFMP (IBAMA, 2019).

TFM is generally accepted as an activity that balances economic development, biodiversity conservation and environmental services (Edwards, et al., 2014). The regulations that govern TFM in Brazil, with provisions for pre-harvest inventories, mapping of trees with Cartesian coordinates, planning of access roads, log landings and skidding trails, directional tree felling and preservation of seed trees, among others, in theory imply in much lower levels of ecological impacts and damage to remaining forest when compared to conventional logging operations (Miller et al., 2011). Nevertheless, there is no consensus on the sustainability of TFM, especially regarding the ability of logged forests to recover the timber volume removed (Piponiot, et al., 2019a; Macpherson et al., 2010; Sist et al., 2021) and above-ground biomass (AGB) losses (Hu et al., 2020; Zimmerman and Kornos, 2012) to levels close to that of the original forest.

Long-term studies on forest growth dynamics through the use of permanent sample plots (PSP) are recognized as the best way to understand forest responses to natural and anthropogenic disturbances, predict production, determine logging cycles and intensities, recommend silvicultural treatments and verify the sustainability of timber production in TFM (Fredericksen and Mostacedo, 2000; Fredericksen and Putz, 2003; Dauber et al., 2005; Wadsworth and Zweede, 2006; Sist and Ferreira, 2007; Villegas et al., 2008). These studies have gained particular importance in view of a broader scenario where atypical climate events have affected the species composition, AGB and carbon stocks dynamics of tropical forests around the world (Phillips et al., 2009 and 2010; Allen et al., 2010; Feeley et al., 2011; McDowell et al., 2018). Although tropical forests growth dynamics have been studied for decades (Sheil, 1998; Malhi et al., 2002; Lewis et al., 2004; Laurance et al., 2009), due to the difficulties in accessing remote forest areas, PSP establishment and monitoring, as well as the diversity of tropical forests ecosystems, these studies are still scarce and generally limited to small and easy to access areas (Lausch et al., 2017).

This paper presents the results of a 20-year study of forest growth dynamics in a little studied open forest type in the southwestern Brazilian Amazon (SWA). This forest typology extends over more than 20 million hectares in the SWA, covering parts of Acre and Rondônia states and the southwest of Amazonas state (ACRE, 2007). This region has been impacted by the implementation of ranches and colonization projects along the BR 364 road since the 70 s, making the region a hot spot for deforestation and legal and illegal logging.). The timber production of Acre and Rondônia together represents around 10.5 % (1.1 million m^3) of the total produced by the Brazilian Amazon (estimated at 10.5 million m^3) and Lábrea municipality, where the study was carried out, alone produces 1.2 % of this total. This timber production is supposed to come from areas with management plans approved by the government environmental agencies which require the use of RIL practices. The resulting wood products reach markets in other regions of Brazil, with select products such as decking and flooring going to international markets (IBGE, 2022). The study area was logged in 2002 and data were collected in PSP between 2002 and 2022 to analyze forest growth dynamics and structure parameters and to test the sustainability (senso Putz et al., 2022) of the silvicultural system applied, as to the recovery of the forest timber volume and AGB stocks over time.

2. Methodology

2.1. Site description

The study area of 600 ha comprises the 2001 annual production unit (APU) of the Iracema II Farm, a private forest located in the municipality of Lábrea in the State of Amazonas (Fig. 1).

The climate is classified as Aw (Köppen, tropical forest with dry season) with an annual precipitation of around 2000 mm and an average temperature of 25 °C. Distinct wet and dry seasons can be recognized, with the dry season occurring between the months of June and September, when operations related to forest management and logging are carried out. The predominant soils are podzolic and dystrophic red-yellow latosols with high clay content (Radambrasil, 1976). The relief is gently undulating with an average elevation of 170 m above sea level. The forest is predominantly open, with emergent trees and occurrence of bamboo thickets (*Guadua* spp.) and palm trees, and with transitions to denser closed canopy forest.

The logging followed the precepts of RIL, with mapping of trees on a Cartesian coordinate system during the pre-logging forest inventory (census of all tree species with DBH $\geq$ 30 cm) and prior planning of roads, log landings and skid trails. The silvicultural system adopted was the diameter limit cutting (O'Hara, 2014), which, following the Brazilian forest law, was 50 cm DBH. According to Brazilian legislation, areas around springs (60 m radius), rivers and streams (30 m buffer) were considered permanent preservation areas (PPA), and protected from tree cutting and heavy machine traffic. The logging operations were planned and executed through a technical cooperation between the forest company ST Manejo de Florestas and Embrapa. Forest operations (roads, log landings and skid trails) were previously planned based on the forest inventory census. The field teams were trained in directional tree felling and log skidding with a forest tractor (wheeled skidder) and logging was carried out under supervision of researchers. In the pre-logging forest inventory, 178 tree species were identified, of which 52 had commercial value in the local market. The total mean volume was 86 m^3 ha^{-1}, of which 35 m^3 ha^{-1} represented commercial timber species. The logging was of low intensity, with average log extraction around 9 m^3. ha^{-1} (1.3 trees per ha) with no further silvicultural treatments applied. All logging operations (road and log landing opening, tree felling and skidding) took place during the dry season between July and October 2002.

2.2. Permanent sample plots (PSP)

Monitoring was carried out by means of permanent sample plots (PSP) allocated within the forest management area, which was divided into 18 blocks of 30 ha each. Of these, three were randomly selected, and, in these blocks ten 1 ha (100×100 m) PSP were systematically

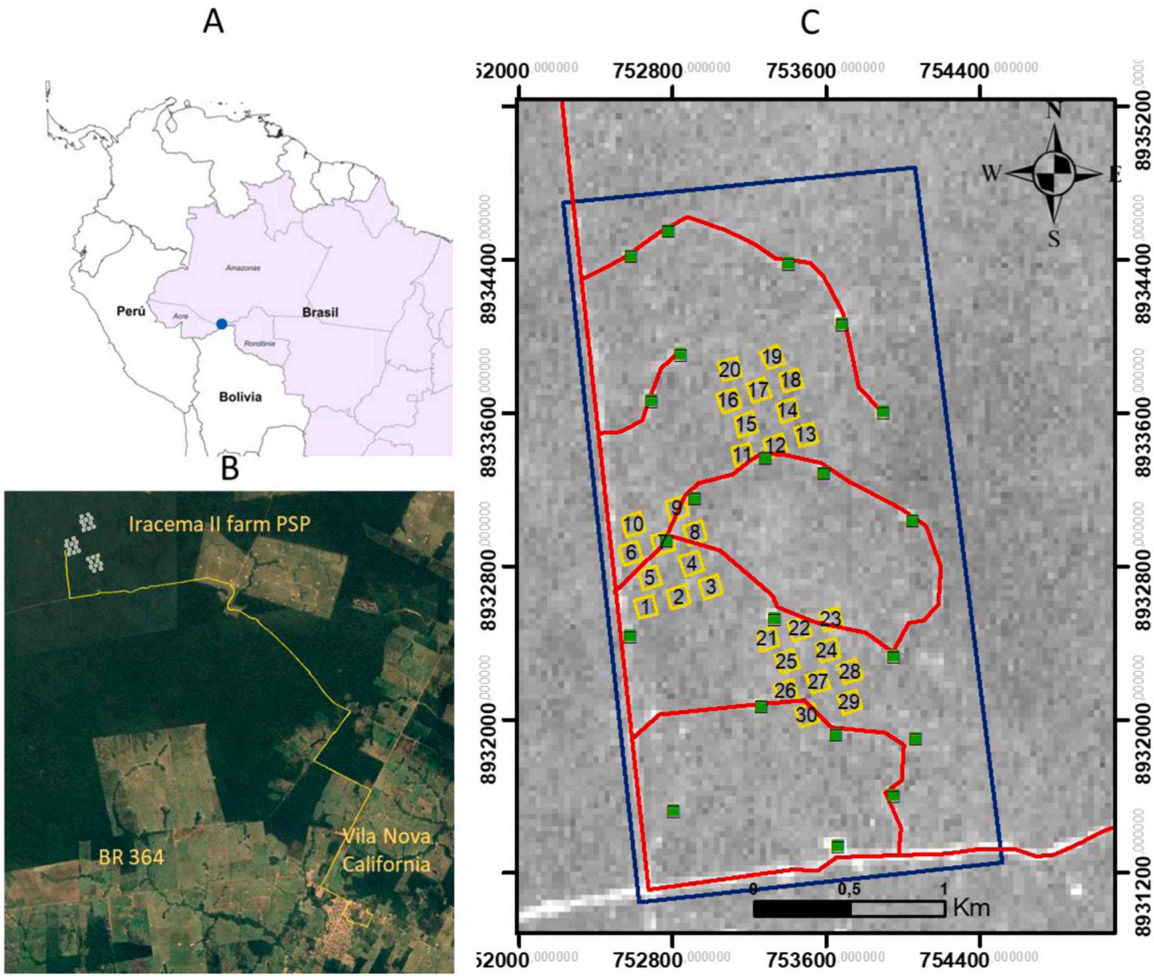

Fig. 1. Study area: (A) South America map showing the location of the Iracema II Farm in Amazonas state, southwestern Brazilian Amazon; (B) Google Earth image showing the study area (white polygons) and its access through Vila California, Rondônia state (yellow line) and (C) Landsat 7 image (LE07_L2SP_001067_20021006_20200916_02_T1_SR_B3) showing the Iracema II annual production unit (APU) after logging (red lines are forest roads and green polygons are log landings) and the location of the 30 permanent sample plots (PSP; numbered yellow polygons).

established, totaling 30 ha (Fig. 1-C). The PSP were established prior to logging, between May and July 2002. The plots were subdivided into 100 subplots (10×10 m) and all trees above 10 cm DBH were marked, measured and identified by parabotanists. Species were identified by association of common and scientific names based on previous forest inventories and identification consistency was checked along the 7 measurements made in the PSP from 2002 to 2022. During this period 306 species were identified by common name, of which 63.4 % could be identified at species level, 33.3 % at genus level and 3.3 % identified only by the common name. Wood density was obtained from the Global Wood Density Database (GWDD) (Zanne et al., 2009). Palm trees, which comprise approximately 5 % of stems over 10 cm DBH, were not an object of this study. In the PSP the following parameters were evaluated: growth, recruitment, and mortality. After PSP installation, six more measurements were made, in 2004, 2007, 2010, 2015, 2018 and 2022. During the 20 years of monitoring, four atypical climate events (long droughts) were observed in the region in 2005, 2011, 2015–2016 and 2018.

2.3. Size classes and species groups

According to Brazilian legislation, the minimum cutting diameter allowed in forest management plans in the Amazon is 50 cm (DBH). In compliance with this norm, trees below this diameter are classified as volume for future logging and commercial timber volume. This criterion was used to divide the sampled population into two size categories: i. equal to or below 50 cm DBH and ii. Above 50 cm DBH.

The population was further divided into two ecological groups: pioneers and non-pioneers. The definition of groups followed that proposed by Swaine and Whitmore (1988). The relative density of pioneer species was used as an environmental indicator of the disturbances produced by logging operations (Carvalho et al., 2017). The species classification was made according to the list defined for the region by d'Oliveira and Ribas (2011) (SM Appendix 1). The species were also divided into i. Commercial and ii. Not commercial, according to their use. The commercial classification considered the Acre, Rondônia and Amazonas states markets (d'Oliveira and Ribas, 2011, SM Appendix 1).

2.4. Forest demographic and growth dynamics parameters

The absolute (N. ha^{-1}) and relative densities (percentage of individuals relative to the total population) of the sampled trees were calculated, by species groups, throughout the repeated measurements in the PSP. Pioneer species were used as an indicator of disturbance in the forest (d'Oliveira and Ribas, 2011) and commercial species as an indicator of the recovery of timber stocks in logged areas.

All trees that reached DBH $\geq$ 10 cm DBH during the measurement time interval were considered as recruitment. The calculation of the recruitment rate was standardized as the division of the total number of plants entering a measurement by the number of adults in the previous measurement, divided by the time interval between the two measurements (Condit et al. 1995).

Mortality (M) was calculated according to Sheil et al. (1995): $M = 1 - (N1 / N0)^{1/t}$ where N0 and N1 are the total number of individuals existing in the first and last evaluation of the population, carried out in the time interval *t*, in years.

Dynamism was calculated as the average of mortality and recruitment rates (Phillips and Gentry, 1994). The half-life of the forest, defined here as the estimated time for a population to lose 50 % of its individuals due to mortality (Lieberman et al., 1985; Swaine et al., 1987; Cascante-Marin et al., 2011), was calculated by the formula Ln (0.5) / Ln (1 – (*M*/100)), where M is the annual mortality rate. Stand doubling time, defined as the time required for half of the forest population to consist of recruitments, was calculated by the formula Ln(0.5)/Ln (1-(*I*/100)), where *I* is the annual recruitment rate.

Annual mean DBH increment (MAI) was calculated by subtracting the values of the field measurement considered as initial from the measurement considered as final, divided by the time interval in years between these measurements. To avoid inconsistent measurements, MAI smaller than -0.1 cm. year^{-1} and larger than 2 cm. year^{-1}, were discarded. Trees were divided into ten 10 cm interval diameter classes, starting from 10 cm DBH and ending in the class of trees with DBH > 90 cm.

2.5. Forest structure parameters

The forest structural parameters of volume (V – m^3. ha^{-1}) and aboveground dry biomass (AGB – Mg. ha^{-1}) were calculated, for individual trees, by the formulas:

$$V = 0{,}000308 * (DBH)^{2{,}1988}$$

(FUNTAC, 1990)

$$AGB = (DBH^{2.671} * 0{,}064)/1000 \text{ (Melo, 2017)}$$

The Melo (2017) equation was used because it was developed in the Antimary State Forest, a close and structurally similar forest and so far the only equation available for the SW Amazon forests. The changes in AGB stocks were calculated throughout the repeated measurements in the PSP, considering separately: AGB loss by tree mortality and AGB gain by recruitment (trees that reached the minimum sampling size of 10 cm DBH) and growth of living trees remaining from the previous measurement. The net growth of AGB for the period (ΔAGB) was calculated by the formula ΔAGB = (AGB1 – AGB0) / *t*, where AGB0 and AGB1 are, respectively, the initial and final AGB of living trees of the period and *t* the time interval in years.

2.6. Statistical analyses

The experimental design adopted was complete randomized blocks, with the systematic distribution of plots to reduce the variance caused by random effects (natural causes) and capture fixed effects (logging, roads, logs landings and skid trails). The PSP systematic distribution also facilitated plot location (GPS was unavailable to us in 2002) and measurement. Repeated measures data analysis was used to compare trends in the studied variables over time, based on a general mixed model as expressed by the equation:

$$Y = X\beta + ZU + \epsilon$$

Where: X = matrix for fixed effects; β = vector of the fixed effects of unknown parameters; Z = matrix for random effects; U = vector of unobservable random effects; ϵ = vector of residual random errors. The repeated measures model was determined by equation:

$$y_{ijk} = \mu + \alpha_i + \tau_k + (\alpha * \tau)_{ik} + \epsilon_{ijk}$$

Where: yijk = aboveground biomass of tree j at time k in plot I; μ = overall mean; αi = fixed effect of logging; τk = fixed effect of year or period k; (α*τ)ik = fixed interaction effect for plot i at time k or plot i at period k; ϵijk = random error at time k in plot i or at period k in plot i.

To process the data, we used the MIXED procedure (SAS 9.2) with repeated measures. The KR (Kenward-Roger) option was employed to calculate degrees of freedom and compound symmetry for the covariance structure. When the overall F test was significant ($p < 0.05$), we used post-hoc least squares means (LS-means) tests with adjusted Tukey's test to determine significant differences ($p < 0.05$) among periods.

3. Results

3.1. Demographic and growth dynamics parameters

The mean tree density before logging (2002) was 421 ± 36 individuals per ha (DBH ≥ 10 cm). Tree density varied over the study period (Fig. 2 A). As expected, in measurements immediately after logging there was a decrease in tree density due to high mortality rates related to effects of logging operations (decreases of 4.9 % and 4.6 %, in years 2 and 5, respectively; Fig. 2 A and B, SM Table 1). In the fourth measurement, eight years after logging, there was an increase in tree density (2.3 %) due to a reduction in the effect of logging on mortality and an increase in recruitment (3.8 %) favored by gaps (Fig. 2 A and B). At the sixth measurement, 16 years after logging, there was a significant ($p<0001$) peak in mortality (3.8 %), similar to the post-logging period (2004–2007), producing a decrease in tree density. In the last two measurements (2018 and 2022), tree density showed a peak, produced by the decrease in mortality (from 3.9 % to 3.0 %) and a significantly higher ($p<0001$) increase in recruitment rate (from 2.6 % to 7.4 %). The mortality and recruitment rate calculated for the entire period were 2.3 % and 2.5 %, respectively. The forest showed high dynamism (2.4), short half-life (29.6 years) and doubling life (27.2 years) so that the number of survivors from the first measurement became similar to the accumulated number of dead and recruitment trees in the last measurement (Fig. 2 A). Until the fourth measurement (8 years after logging), mortality of larger trees (DBH > 50 cm) was significantly higher than the smaller trees, while after that mortality was not statistically different in the studied size classes until the last measurement (20 yr. after logging) when the smaller trees (10 cm ≥ DBH ≤ 50 cm) presented a significantly higher mortality rate (SM Table 1).

At the end of the study period, the mean tree density in the PSP was 5 % higher than that observed before logging. This increase was observed only in trees below 50 cm DBH. At the end of the monitoring period, although the average number of trees above 50 cm DBH was the same as at the beginning of the study (before logging), their percentage in the sampled population decreased over time, indicating that following logging, the forest presents a tendency towards higher tree density, but with smaller diameters (Table 1).

An increase in the relative density of pioneers was observed throughout the study period but this was not statistically significant. The density of pioneer species varied from 5.0 % before logging to 6.6 % in the last measurement (2022), 20 years after logging, with a peak of 6.9 % observed in the fourth PSP measurement (2015). This small variation during the monitoring time indicates that the gaps opened by logging operations (forest roads, log landings, felling gaps) were not sufficient in number and/or size to favor the establishment of pioneer species. When compared with non-pioneers, pioneer species presented a significantly higher mortality rate, from the eighth year after logging (2010) to the end of the study period. Pioneer species recruitment was not significantly different from non-pioneer species until the eighth year after logging, then significantly higher from the eighth to the eighteenth year and again not statistically different in the last measurement (20 yr. after logging) (SM Table 1). Throughout the monitoring time, no variation was observed in the trees' wood density in the PSP, indicating that logging did not favor the entry of species with lower wood density (SM Table 2).

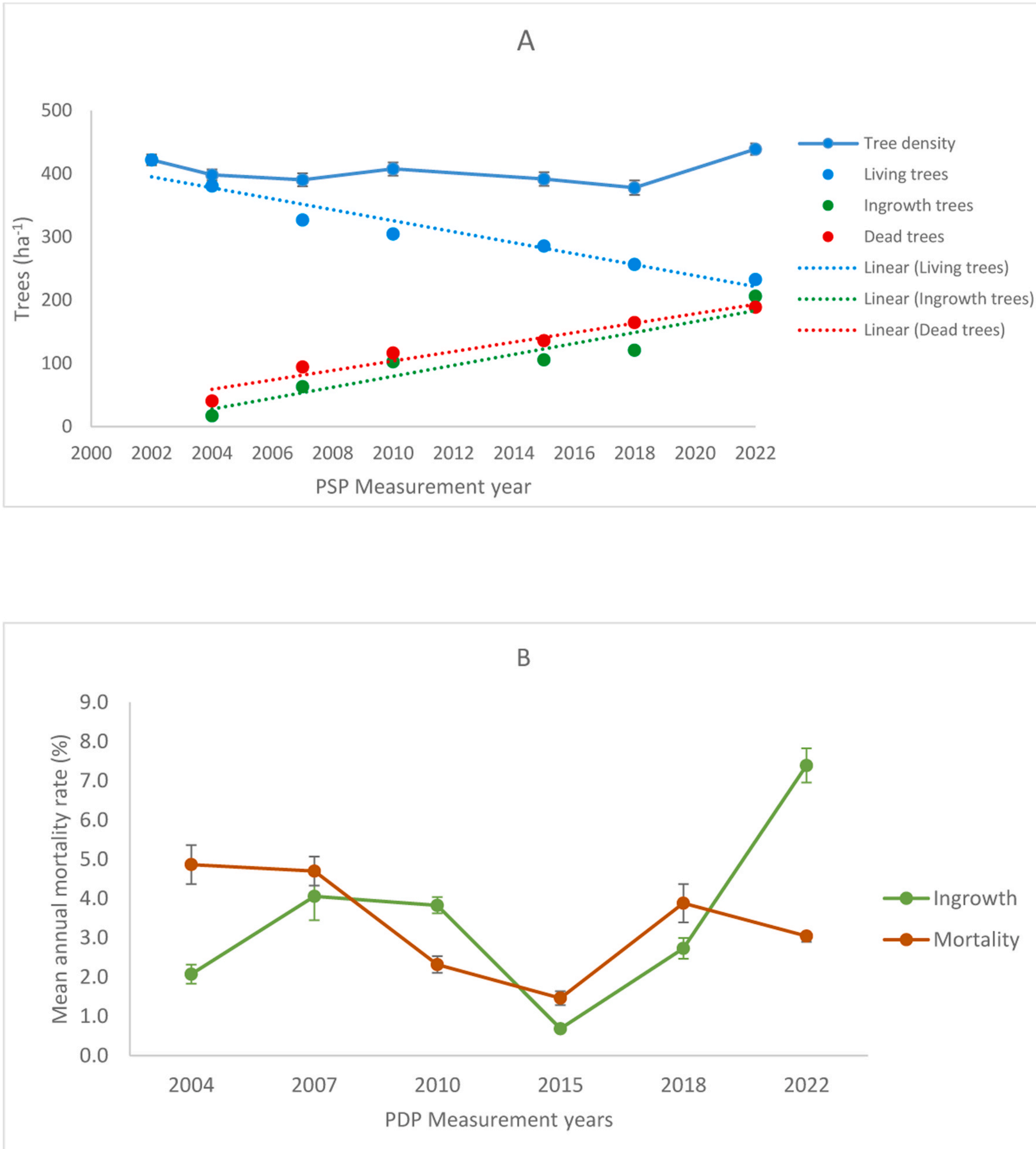

Fig. 2. (A) mean tree density (N. ha^{-1}) and accumulated survivors, dead and recruitment trees and (B) annual recruitment and mortality rates (%), throughout the measurement years in the Iracema II Farm permanent sample plots (PSP). Error bars are SE ($p<0.05$).

The absolute and relative density of trees of commercial species presented little variation and remained almost constant throughout the measurements, with means of around 80 individuals per hectare, equivalent to 20 % of the total population above 10 cm DBH (Table 1).

The highest mean annual increment (MAI; 0.43 ± 0.13 cm. $yr.^{-1}$) was observed two years after logging. From 2004 until 2015 the MAI showed a tendency to decrease, and, in the penultimate measurement, we observed another peak (0.35 ± 0.1 cm. $yr.^{-1}$) (Fig. 3A). The 2004 MAI was significantly higher ($p<0.001$) and the 2015 significantly lower ($p<0.001$) than in the other periods (Fig. 3A, SM Tables 3 and 4). The MAI according to DBH classes was significantly lower in the two first DBH classes (10–20 and 20–30 cm). In the other classes, MAI was significantly higher in the classes from 70 to 90 cm DBH, with a tendency to decrease for trees greater than 80 cm DBH (Fig. 3B, SM Tables 3 and 4).

3.2. Aboveground biomass (AGB) and bole volume

The AGB in PSP before logging was 188.3 ± 5.6 Mg. ha^{-1} (DBH ≥ 10 cm) and almost equally divided in the two size classes considered: DBH≥ 50 cm, 94.9 ± 8.7 Mg. ha^{-1} and 10 ≤ DBH < 50 cm, 93.4 ± 8.1 Mg. ha^{-1}. Logging impacts resulted in AGB losses until the fifth year after logging (2007). From the fifth year on, AGB stocks increased on average by 1.4 Mg. ha^{-1}. $yr.^{-1}$, mainly in the smaller tree size class, which was responsible for most of the observed AGB recovery during the monitoring time, recovering the pre-logging AGB stocks. At the end of the study period, trees in the smaller size class presented a mean AGB significantly higher (106.3 ± 2.4 Mg. ha^{-1}) than the DBH ≥50 cm trees. The greatest contribution to the AGB increase was the growth of the remaining trees (Fig. 4, SM Table 5).

Table 1
Mean absolute (N per ha) and relative (%) density of trees, and standard error (SE) according to size class and species group along measurement years in the Iracema II Farm permanent sample plots (PSP). Error bars are SE (p<0.05).

Year		10 ≤ DBH < 50 cm		DBH ≥ 50 cm		Pioneers		Commercial		All
		N	%	N	%	N	%	N	%	N
2002	Mean	405.7b	96.2	16.2a	3.8	20.2bc	5	87.3a	20.7	421.9b
	SE	8.5		0.7		1.4		3.9		8.5
2004	Mean	383.2 cd	96.2	14.9abc	3.7	19.2c	5	81.8b	20.5	398.2 cd
	SE	8.7		0.7		1.3		3.9		8.8
2007	Mean	376.4ed	96.4	14.0c	3.6	25.4bc	6.7	78.5b	20.1	390.4de
	SE	10.2		0.6		3.8		3.6		10.3
2010	Mean	393.6bc	96.6	14.0c	3.4	28.7a	7.2	80.7b	19.8	407.6bc
	SE	10.2		0.7		3.7		3.5		10.5
2015	Mean	377.0d	96.3	14.6c	3.7	25.3bc	6.6	78.5b	20.1	391.6cde
	SE	10.5		0.7		3.2		3.6		10.8
2018	Mean	362.3e	95.9	15.6ab	4.1	24.2bc	6.7	76.6b	20.2	377.9e
	SE	11.2		0.7		3.1		3.4		11.6
2022	Mean	422.7a	96.3	16.3a	3.7	27.0bc	6.4	87.2a	19.9	439.2a
	SE	8.6		0.7		3.3		3.2		9

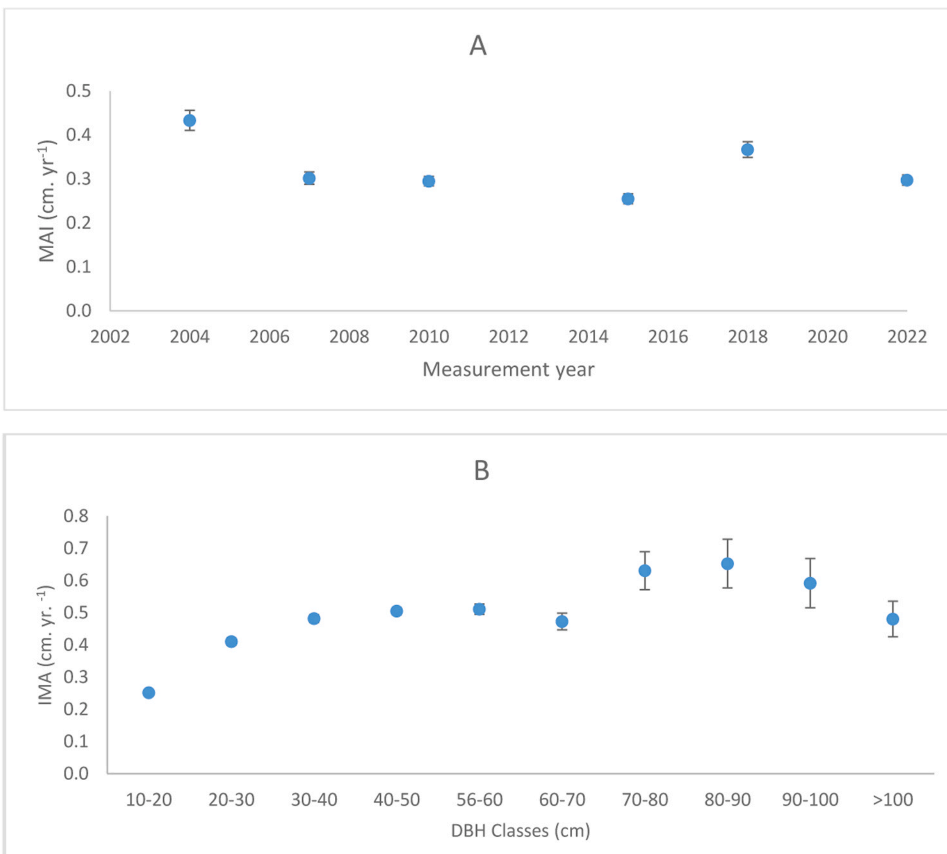

Fig. 3. Mean annual increment (MAI, cm. yr.$^{-1}$) per measurement year (A) and diametric class (B) of the trees (DBH ≥10 cm) in the PSP of the Iracema II farm. Error bars represent standard errors (p<0,05).

3.3. Total and commercial species volume

The mean volume (DBH ≥ 10 cm) before logging was 153.5 ± 3.5 m^3. ha^{-1} and after 20 years was 158.2 ± 4.5 m^3. ha^{-1}. Throughout the monitoring period, volume in the smaller size class (10 cm ≤ DBH < 50 cm) was always greater than in the commercial timber size class (DBH > 50 cm), which is typical for open forests in the region (e.g., d'Oliveira and Sant'Anna, 2003). Nevertheless, in this study, the volume stock of smaller trees presented a tendency to increase while the volume of trees above 50 cm DBH decreased. As expected, and already observed in AGB stocks, in the five years after logging there was a volume loss of around 20 m^3. ha^{-1} in the PSP. At the end of the monitoring period, although the total timber volume (DBH ≥ 10 cm) recovered and even showed a small increase, the volume of trees over 50 cm DBH remained lower than that observed before logging (Fig. 5A).

The volume of commercial species showed a similar behavior. Timber volume in the smaller size class (DBH < 50 cm) presented positive growth from the fourth measurement onwards (2010, eight years after logging) and at the end of the study time was equivalent to the initial volume (27.4 m^3. ha^{-1}). In the commercial timber class (DBH > 50 cm), even with the positive volume increase from 2010 onwards, at the end of the monitoring period the volume was still below that observed before

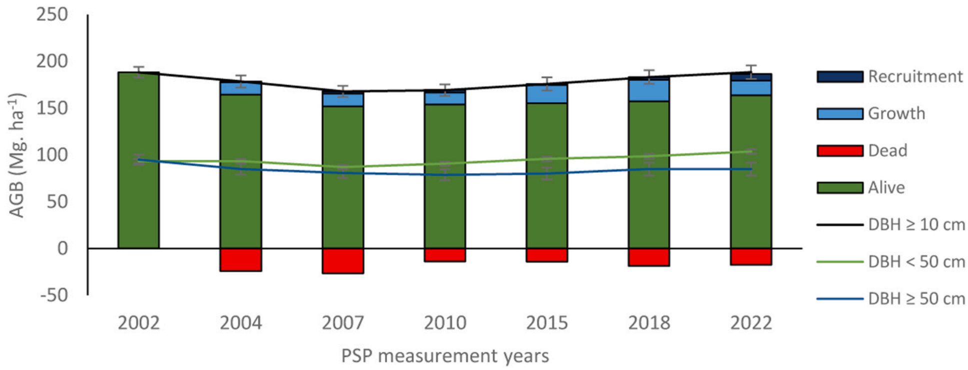

Fig. 4. Changes in aboveground biomass (AGB) in the permanent sample plots (PSP) of Iracema II Farm during the study period, expressed as mean AGB in Mg. ha^{-1}: Total ABG (black line) and AGB by size classes (DBH ≥ 10 cm < 50 cm, green line; DBH ≥ 50 cm, blue line); mean AGB gain or loss promoted by recruitment (dark blue bar); standing trees growth (light blue bar) and dead trees (red bar). Green columns represent the remaining (alive) standing trees AGB from the previous measurement.

logging (Fig. 5B, SM Table 6). Following the current trend (0.46 m^3. ha^{-1}. $yr.^{-1}$), we would expect the full recovery of the original commercial timber volume (trees above 10 cm DBH) in 2039, or twelve years beyond the prescribed logging cycle of 25 years. The recovery of trees above 50 cm was slower (0.22 m^3. ha^{-1}. $yr.^{-1}$) and would take 45 years to reach the original volume.

4. Discussion and conclusion

4.1. Demographic and growth dynamics parameters

Although high, even considering the use of RIL techniques and the low logging intensity, the initial mortality rates observed in the first two measurements of this study can be attributed to the immediate effects of logging, followed by a gradual reduction in mortality rates from the fifth year after logging onwards (Riutta, et al., 2018; Pearson et al., 2014). However, the mortality peak observed in 2018 (15 years after logging) indicates that other factors have been affecting the growth dynamics of forests in the region. Atypical climatic events have been reported as producing severe changes in the dynamics of forests (Phillips et al., 2009, 2010; McDowell et al., 2018; Vidal et al., 2016; Amaral et al., 2019; Bennett, et al., 2023) and the droughts observed in 2005 and 2011 probably contributed to increase post-logging mortality, with droughts in 2015–2016 (Nunes et al., 2021) and 2018 contributing as well to the observed mortality peak and the maintenance of a high mortality rate as measured in 2022 (e.g. Phillips et al., 2009, mortality up to two years after atypical climate events).

Atypical climatic events produce increases in mortality which are followed by a subsequent increase in recruitment (e.g. Sheil and Phillips, 1995). Openings in the forest canopy contribute to the establishment of new trees and to the diameter growth of small trees in the forest understory (Sullivan et al., 2022) so that they reach the minimum measurement diameter. In our study, mortality and recruitment were strongly related, and fluctuations in recruitment rates accompanied mortality. As a result, a large recruitment peak occurred in the last measurement making the recruitment rate for the entire period slightly higher than mortality. As result of the high recruitment and mortality rates, the forest also showed high dynamism, which is in line with other studies that demonstrate an acceleration of turnover rates in tropical forests (Lieberman, et al., 1985; Phillips et al., 1994; Cascante-Marin et al., 2011), especially in the Amazon (Lewis et al., 2004; McDowell, et al., 2018).

Low logging intensity usually limits the recruitment of pioneer species (e.g. Amaral et al., 2019; Hogan et al., 2018). With canopy closure, the forest environment becomes less favorable for the establishment and survival of these species, generally resulting in a decrease in recruitment and increase in mortality of pioneer species. In our study, the small variation in the relative density of pioneer species during the monitoring time indicates that the gaps opened by logging operations were not large enough to favor the establishment of pioneer species. Nonetheless, the initial high value for presence of pioneers is a characteristic of this regiońs open forests (Krueger, 2004; d'Oliveira et al., 2013).

4.2. Aboveground biomass (AGB)

The relatively low mean AGB before logging (when compared to other Amazon forests, e.g. Paracou in French Guiana, Baraloto et al., 2012 and Tapajós national forest, Castro et al., 2021), the similarity of AGB stocks in the two size classes (<50 cm or >50 cm DBH) and, consequently, the low logging intensity practiced are characteristics of the open forests of the SWA (Krueger, 2004; d'Oliveira et al., 2013; Rutishauer et al., 2015). Logging intensities are highly variable in tropical forests, producing different impacts on the dynamics of the residual forest (Medjibe et al., 2013; Hu et al., 2020; Sullivan et al., 2022). The effects of logging operations on the structure and dynamics of the forest vary, therefore, according to the intensity of logging and the forest management regime applied and usually tend to dilute over the cycle time (Blanc et al., 2009; Gourlet-Fleury et al., 2013; West et al., 2014). In our study, logging produced a low impact on the AGB stocks (mean AGB loss of 10 Mg. ha^{-1}), which fully recovered 20 years after logging. Our results point in the same direction as previous studies in tropical forests, in which, unless high logging intensities are applied (e.g. Butarbutar, et al., 2019), AGB stocks generally tend to recover quickly after logging (Gourlet-Fleury, 2013; Rutishauser et al., 2015; Riutta et al., 2018), with the dynamics of AGB stocks more likely to be driven by mortality than by growth of residual trees and recruitment (Johnson et al., 2016). Under the conditions of this study, the AGB recovery also supports the claim that Neotropical forests remain as significant carbon sinks even following logging (Brienen et al., 2015; Mills, et al., 2023). However, the post logging mean AGB increment (1.4 Mg. ha^{-1}. $year^{-1}$), although greater than previous studies in South American undisturbed forests (e.g. 0.74 +- 0.34 t. ha^{-1}. $yr.^{-1}$, Phillips et al., 1998) was much lower than that recorded for other logged forests in the Amazon (Piponiot et al., 2016; Castro et al., 2021).

In our study, productivity close to that of undisturbed forests can be attributed to the relatively low impact of forestry operations, low logging intensity and time since logging. Although our surveys did not specifically register the presence of bamboo thickets, we did not observe any evidence of increase (e.g. Bedrij et al., 2022) nor decrease (e.g. d'Oliveira et al., 2013) in bamboo populations after logging, which could be factors impacting forest dynamics.

On the other hand, the atypical mortality rates, with peaks outside the influence of logging operations, suggests that climate (e.g. Fauset et al., 2019) also played an important role. AGB dynamics depends

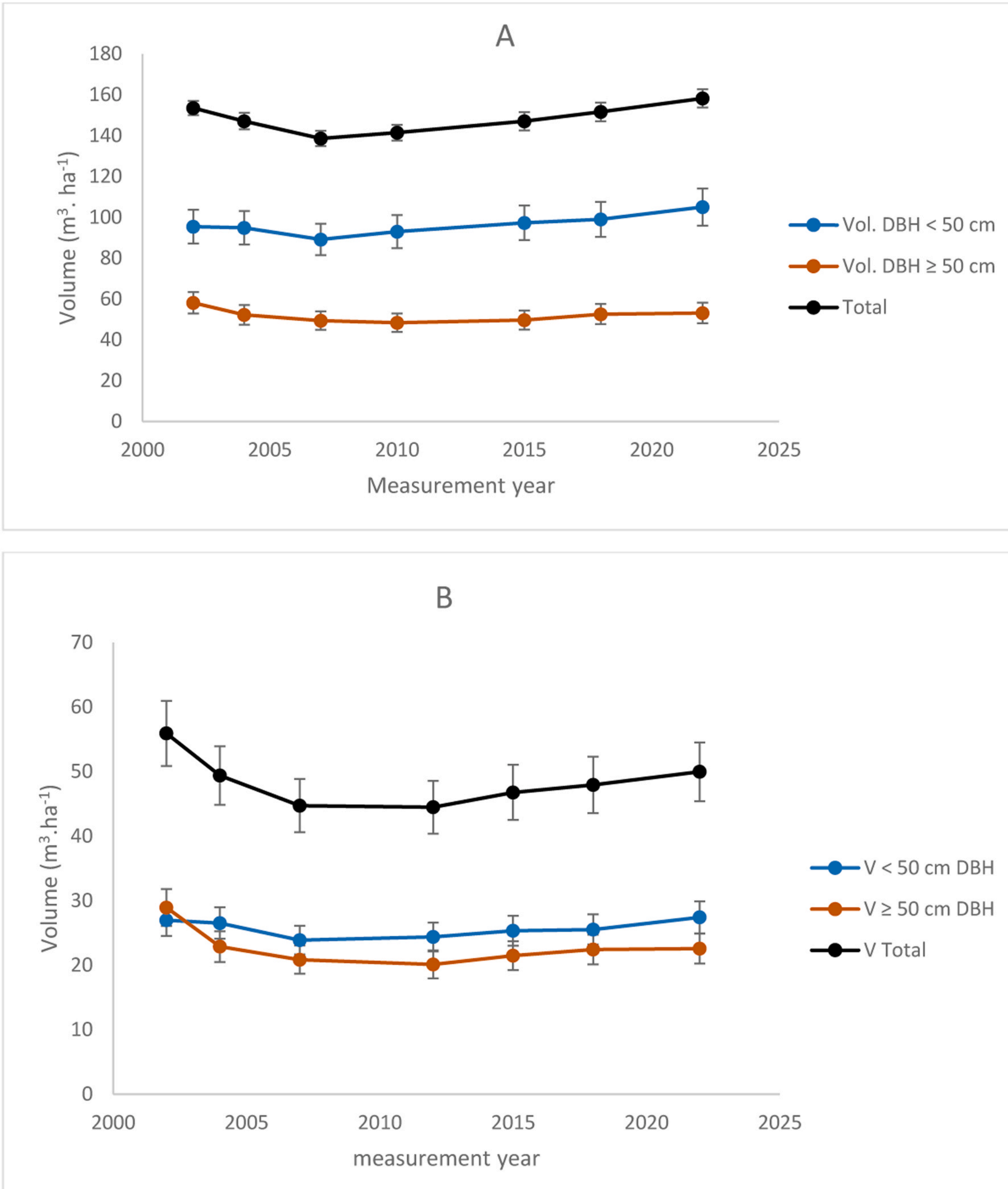

Fig. 5. Total (DBH ≥ 10 cm), commercial (DBH ≥ 50 cm) and future logging (V ≤ 50 cm) volume for all species (A) and commercial species (B) in the permanent sample plots (PSP) of Iracema II Farm. Error bars represent standard errors (p<0,05).

mainly on the residual standing trees ́ growth (Vidal et al., 2016; Roopsind, et al., 2018), such that losses produced by the mortality of large trees cannot be compensated by the recruitment, as suggested by the results of Fauset et al. (2019). This effect has been observed in Amazon as a whole, but is particularly high in SWA, and also reflects an estimated 30 % decrease in the Amazon Forest ́s carbon sequestration capacity (Brienen et al., 2015).

4.3. Volume of commercial species

Different than the AGB, timber stocks of commercial species did not recover at the same speed, mainly due to the significantly higher mortality of trees above 50 cm DBH in the first eight years after logging, producing losses much greater than what was compensated by the growth of remaining trees and recruitment. Slower recovery of timber stocks has been observed in other sites in the Amazon (Piponiot et al., 2019a,b), usually associated with high logging intensities and tree mortality (Castro et al., 2021; Reategui-Betancourt, et al., 2023). In a recent study, Sist et al. (2021) suggested that the sustainable timber production in the Brazilian Amazon can only be achieved combining long cycles (60 yr.) and light logging (10 m^3. ha^{-1}). However, in our study area, low logging intensity kept the forest close to its original volume, lowering volume increment along the monitoring time due to the decrease in the residual trees ́ growth produced by competition (e.g. the observed decrease of AMI in the last measurement as a result of higher tree density and AGB stocks, when compared to the original forest before logging) and tree mortality produced by natural causes. Considering this scenario, simply increasing the logging cycle would not be sufficient to guarantee timber stock recovery. Furthermore, increasing logging cycles or adopting additional costly silvicultural treatments could also compromise economical sustainability of forest management in a region where logging intensity is already low (Rockwell et al., 2014; d'Oliveira et al., 2013, Andersen et al., 2014). As for the expansion of cycles mentioned, some authors (Bick et al., 1998; Canetti, et al., 2021) have shown that there is a time interval (around 50 yr.) in which the maximum current annual increment and the mean annual increment of the population can be attained. After this point, the increment tends to stagnate when the population of each species is considered.

4.4. Tropical forest management implications

Understanding the recovery of carbon and biomass stocks after disturbances of anthropic origin is essential for the development of policies to mitigate the effects of climate change (Adinugroho et al., 2022). Our results agree with several previous studies based on long-term PSP in tropical forests (Laurance et al., 2004, 2009; Phillips et al., 2004) that suggest three large-scale ecological changes over the last five decades: (i) pantropical increase in stem turnover rates; (ii) neotropical increase in AGB; (iii) Amazon forests ́increase in recruitment and mortality rates (Lewis et al., 2004).

The discussion of the sustainability of tropical forest management (STFM) is complex, with results pointing in different directions (Zimmerman and Kornos, 2012; Gourlet-Fleury et al., 2013). However, STFM continues to be the only economic activity involving timber production that maintains the forest structure and biodiversity and allows its recovery to conditions close to the original forest following the cutting cycle (Baraloto et al., 2012; Bicknell et al., 2015; Putz et al., 2019).

When compared to other forms of land use, by preserving forest structure, RIL of Amazonian forests, when properly carried out, can be an important instrument towards preventing or mitigating global climate change (Ellis et al., 2019). Tropical forest management is a nature-based solution for land use and RIL is a way to balance income generation and conservation. Managed forests, in principle, are supposed to sustain a positive productivity along the logging cycle length. In this study, twenty years after logging, the observed full recovery of AGB stocks and higher tree density are evidence that by the end of the prescribed cycle forest productivity will be reduced by competition and forest dynamics will be led mainly by natural causes.

In such a scenario, although the commercial timber volume recovery will take longer (34 yr. to fully recover volume of commercial species (dbh>10) and 45 yr. for trees above 50 cm DBH), we suggest that at the end of the prescribed logging cycle (in 2027) the commercial timber stocks could support a second logging cycle with similar logging intensity (10 m^3. ha^{-1}) if a broader range of species were to be used (e.g. as suggested by Castro et al., 2021). Respecting the maintenance of seed trees, the logging of the available commercial volume and senescent growing decaying trees (Dawkins and Philip, 1998, e.g. trees DBH >80 cm) would generate income for the forest owners and increase forest productivity (e.g. promoting canopy release especially for trees DBH < 30 cm), and lead to a more productive third logging cycle.

With regard to the long-term plans for the study site, a second logging cycle is not planned, as the owners of the Iracema Farm chose to dedicate the area to the voluntary market of carbon credits.

CRediT authorship contribution statement

Jaquelyne Lins Januario: Writing – review & editing, Validation, Formal analysis, Data curation. **Fabio Thaines:** Writing – review & editing, Validation, Data curation, Conceptualization. **Mario Humberto Aravena Acuña:** Writing – review & editing, Formal analysis, Data curation. **Marcus V.N. d'Oliveira:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Luis Claudio Oliveira:** Validation, Supervision, Project administration, Methodology, Investigation, Formal analysis, Data curation. **Robert Pritchard Miller:** Writing – review & editing, Writing – original draft, Investigation, Formal analysis. **Evaldo Muñoz Braz:** Writing – review & editing, Writing – original draft, Investigation, Conceptualization.

Declaration of Competing Interest

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

Data Availability

A link to the data will be provide

Acknowledgments

We thank Embrapa Acre parabotanical team Airton Farias, Aldeci Oliveira, Paulo Machado and Manoel Freire for their support during fieldwork. We thank JBS and CIRAD for financial support for field work. We thank the Iracema II owners and manager (Cidão) for providing us with access to the property where the PSPs are situated. We thank Roberto Sgorla, ST Manejo Florestal manager, for support during field campaigns. J.L Januario also received support through a CAPES scholarship.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2024.121937.

References

ACRE, 2007. Base de dados do zoneamento ecológico e econômico do Estado do Acre. SEMA, Rio Branco.

Adinugroho, W.C., et al., 2022. Recovery of carbon and vegetation diversity 23 years after fire in a tropical dryland forest of Indonesia. Sustainability 14, 6964. https://doi.org/10.3390/su14126964.

Allen, C.D., et al., 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Ecol. Manag. 259, 660–684. https://doi.org/10.1016/j.foreco.2009.09.001.

Amaral, M.R.M., et al., 2019. Dynamics of tropical forest twenty-five years after experimental logging in Central Amazon. Forests 10, 89. https://doi.org/10.3390/f10020089.

Andersen, et al., 2014. Monitoring selective logging in western Amazonia with repeat lidar flights. Remote Sens. Environ. 151, 157–165. https://doi.org/10.1016/j.rse.2013.08.049.

Anderson-Teixeira, K.J., et al., 2016. Carbon dynamics of mature and regrowth tropical forests derived from a pantropical database (TropForC-db). Glob. Chang. Biol. 371, 1703. https://doi.org/10.1111/gcb.13226.

Baraloto, C., et al., 2012. Contrasting taxonomic and functional responses of a tropical tree community to selective logging. J. Appl. Ecol. 49, 861–870. https://doi.org/10.1111/j.1365-2664.2012.02164.x.

Bedrij, N.A., et al., 2022. Selective logging of a subtropical forest: long-term impacts on stand structure, timber volumes, and biomass stocks. Ecol. Manag. 518, 120290 https://doi.org/10.1016/j.foreco.2022.120290.

Bennett, A.C., et al., 2023. Sensitivity of South American tropical forests to an extreme climate anomaly. Nat. Clim. Chang. 13, 967–974. https://doi.org/10.1038/s41558-023-01776-4.

Bick, U., et al., 1998. Assessment and measurement of forestry key parameters for the evaluation of tropical forest management. Plant Res. Dev. v. 47 (48), 38–61.

Bicknell, J.E., et al., 2015. Reconciling timber extraction with biodiversity conservation in tropical forests using reduced-impact logging. J. Appl. Ecol. 52, 279–388. https://doi.org/10.1111/1365-2664.12391.

Blanc, L., et al., 2009. Dynamics of aboveground carbon stocks in a selectively logged tropical forest. Ecol. Appl. 19, 1397–1404. https://doi.org/10.1890/08-1572.1.

Braz, E.M., Mattos, P.P., 2015. Manejo de produção em florestas naturais da Amazônia: mitos e verdades. Nativa v. 3 (n. 4), 292–295.

Brienen, R.J.W., et al., 2015. Long-term decline of the Amazon carbon sink, 2015 Nature 519, 344–348. https://doi.org/10.1038/nature14283.

Butarbutar, T., et al., 2019. Carbon recovery following selective logging in tropical rainforests in Kalimantan, Indonesia. Ecosyst. 6, 36. https://doi.org/10.1186/s40663-019-0195-x.
Canetti, et al., 2021. A new approach to maximize the wood production in the sustainable management of Amazon forest. Ann. For. Sci. 78, 67. https://doi.org/10.1007/s13595-021-01079-8.
Carvalho, A.L., et al., 2017. Natural regeneration of trees in selectively logged forest in western Amazonia. Ecol. Manag. 392, 36–44. https://doi.org/10.1016/j.foreco.2017.02.049.
Cascante-Marin, A., et al., 2011. Tree turnover in a premontane neotropical forest (1998–2009) in Costa Rica. Plant. Ecol. 212, 1101–1108. https://doi.org/10.1007/s11258-010-9890-y.
Castro, T.C., et al., 2021. The continuous timber production over cutting cycles in the Brazilian Amazon depends on volumes of species not harvested in previous cuts. Ecol. Manag. 490, 119214 https://doi.org/10.1016/j.foreco.2021.119124.
Condit, R., et al., 1995. Demography and harvest potential of Latin American timber species: data from a large permanent plot in Panama. J. Trop. Sci. 7, 599–622 https://jtfs.frim.gov.my/jtfs/article/view/1856/1573. (accessed 15 Aug. 2023).
d'Oliveira, M.V.N., et al., 2013. Can forest management be sustainable in a Bamboo dominated forest? A 12-year study of forest dynamics in western Amazon. Ecol. Manag. 310, 672–679. https://doi.org/10.1016/j.foreco.2013.09.008.
d'Oliveira, M.V.N., Ribas, L.A., 2011. Forest regeneration in artificial gaps twelve years after canopy opening in Acre State Western Amazon. Ecol. Manag. 261, 1722–1731. https://doi.org/10.1016/j.foreco.2011.01.020.
d'Oliveira M.V.N., Sant'Anna, H. 2003, Inventário florestal e avaliação do avanço do desmatamento no Projeto de Colonização Pedro Peixoto. Embrapa Acre, série documentos 83, 47p. http://www.infoteca.cnptia.embrapa.br/infoteca/handle/doc/498961.
Dauber, E., et al., 2005. Sustainability of timber harvesting in Bolivian tropical forests. Ecol. Manag. 214, 294–304. https://doi.org/10.1016/j.foreco.2005.04.019.
Dawkins, H.C., Philip, M.S., 1998. Tropical Moist Forest Silviculture and Management: A History of Success and Failure. CAB International, Wallingford.
Edwards, D.P., et al., 2014. Selective-logging and oil palm: multitaxon impacts, biodiversity indicators, and trade-offs for conservation planning. Ecol. Appl. 24, 2029–2049. https://doi.org/10.1890/14-0010.1.
Ellis, P.W., et al., 2019. Reduced-impact logging for climate change mitigation (RIL-C) can halve selective logging emissions from tropical forests. Ecol. Manag. 438, 256–266. https://doi.org/10.1016/j.foreco.2019.02.004.
Fauset, et al., 2019. Individual-based modeling of Amazon forests suggests that climate controls productivity while traits control demography. Front. Earth Sci. 7 https://doi.org/10.3389/feart.2019.00083.
Feeley, K.J., et al., 2011. Directional changes in the species composition of a tropical forest. Ecology 92, 871–882. https://doi.org/10.1890/10-0724.1.
Fredericksen, T.S., Mostacedo, B., 2000. Regeneration of timber species following selection logging in a Bolivian tropical dry forest. Ecol. Manag. 131, 47–55. https://doi.org/10.1016/S0378-1127(99)00199-1.
Fredericksen, T.S., Putz, F.E., 2003. Silvicultural intensification for tropical conservation. Biodivers. Conserv. 12, 1445–1453. https://doi.org/10.1023/A:1023673625940.
Funtac, 1990. Estrutura do plano de manejo de uso múltiplo da floresta Estadual do Antimari. Acre State Technological Foundation, Rio Branco.
Gourlet-Fleury, S., et al., 2013. Tropical forest recovery from logging: a 24 year silvicultural experiment from Central Africa. Philos. Trans. R. Soc. Lond. B. 368, 1–10. https://doi.org/10.1098/rstb.2012.0302.
Greenpeace, 2017. Blood-Stained Timber: Rural Violence and the theft of Amazon timber. https://www.greenpeace.org.br/hubfs/Greenpeace_BloodStainedTimber_2017.pdf. (accessed 15 Aug. 2023).
Hogan, J.A., et al., 2018. Understanding the recruitment response of juvenile Neotropical trees to logging intensity using functional traits. Ecol. Appl. 28, 1998–2010. https://doi.org/10.1002/eap.1776.
Hu, J., et al., 2020. Above-ground biomass recovery following logging and thinning over 46 years in an Australian tropical forest. Sci. Total Environ. 734, 139098 https://doi.org/10.1016/j.scitotenv.2020.139098.
Human Rights Watch, 2019. Rainforest Mafias: How Violence and Impunity Fuel Deforestation in Brazil's Amazon. https://www.hrw.org/sites/default/files/report_pdf/brazil0919_web.pdf. (accessed 15 Aug. 2023).
IBAMA, 2018, Ibama identifica 22 pessoas envolvidas na exploração ilegal de ipê em terras indígenas no noroeste de MT: 12/11/2018 - https://www.ibama.gov.br/noticias/436-2018/1766-ibama-identifica-22-pessoas-envolvidas-na-exploracao-ilegal-de-ipe-em-terras-indigenas-no-noroeste-de-mt.
IBAMA, 2019, Produção madeireira de espécies nativas brasileiras: 2012 a 2017, Brasilia, 376p.
IBAMA/MMA. 1998. Manual de Padronização das Ações de Vistoria e Orientação Técnica dasAtividades Florestais. Diretoria de Gestão dos Recursos Naturais Renováveis (DIREN), InstitutoBrasileiro do Meio Ambiente e dos Recursos Naturais Renováveis (IBAMA)/ Ministério do MeioAmbiente (MMA). Brasilia.
IBGE, 2022, Produção da Extração Vegetal e da Silvicultura. https://www.ibge.gov.br/estatisticas/todos-os-produtos-estatisticas/9105-producao-da-extracao-vegetal-e-da-silvicultura?=&t=resultados.
Johnson, M.O., et al., 2016. Variation in stem mortality rates determines patterns of above-ground biomass in Amazonian forests: implications for dynamic global vegetation models. Glob. Chang. Biol. 22, 3996–4013. https://doi.org/10.1111/gcb.13315.
Krueger, W., 2004. Effects of future crop tree flagging and skid trail planning on conventional diameter-limit logging in a Bolivian tropical forest. Ecol. Manag. 188, 381–393. https://doi.org/10.1016/j.foreco.2003.08.006.
Laurance, W.F., et al., 2004. Pervasive alteration of tree communities in undisturbed Amazonian forests. Nature 428, 171. https://doi.org/10.1038/nature02383.
Laurance, S.G.W., et al., 2009. Long-term variation in Amazon forest dynamics. J. Veg. Sci. 20, 323–333. https://doi.org/10.1111/j.1654-1103.2009.01044.x.
Lausch, A., et al., 2017. Understanding forest health with remote sensing-part II—a review of approaches and data models. Remote Sens 9, 129. https://doi.org/10.3390/rs9020129.
Lewis, S.L., et al., 2004. Concerted changes in tropical forest structure and dynamics: evidence from 50 South American long-term plots. Philos. Trans. R. Soc. Lond. B. 359, 421–436. https://doi.org/10.1098/rstb.2003.1431.
Lewis, S.L., et al., 2015. Increasing human dominance of tropical forests. Science 349, 96250. https://doi.org/10.1126/science.aaa9932.
Lieberman, D., et al., 1985. Mortality patterns and stand turnover rates in a wet tropical forest in Costa Rica. J. Ecol. 73, 915–924 https://www.jstor.org/stable/2260157. (accessed 15 Aug. 2023).
Macpherson, A.J., et al., 2010. A Model for comparing reduced impact logging with conventional logging for an Eastern Amazonian forest. Ecol. Manag. 260, 2002–2011. https://doi.org/10.1016/j.foreco.2010.08.050.
Malhi, Y., et al., 2002. An international network to monitor the structure, composition and dynamics of Amazonian forests (RAINFOR). J. Veg. Sci. 13, 439–450. https://doi.org/10.1111/j.1654-1103.2002.tb02068.x.
McDowell, N., et al., 2018. Drivers and mechanisms of tree mortality in moist tropical forests. N. Phytol. 219, 851–869. https://doi.org/10.1111/nph.15027.
Medjibe, V.P., et al., 2013. Certified and uncertified logging concessions compared in gabon: changes in stand structure, tree species, and biomass. Environ. Manag. 51, 524–540. https://doi.org/10.1007/s00267-012-0006-4.
Melo, A.W.F., 2017. Alometria de árvores e biomassa florestal na Amazônia Sul-Ocidental. Tese de Doutorado, Instituto Nacional de Pesquisas da Amazônia, Manaus.
Miller, S.D. et al., 2011. Reduced impact logging minimally alters tropical rainforest carbon and energy exchange. PNAS, www.pnas.org/cgi/doi/10.1073/pnas.1105068108.
Mills, M.B., et al., 2023. Tropical forests post-logging are a persistent net carbon source to the atmosphere. e2214462120 Ecol. Env. Sci. 120. https://doi.org/10.1073/pnas.2214462120.
Miranda, et al., 2020. Manejo Florestal Sustentável em Áreas Protegidas de uso comunitário na Amazônia. Soc. e Nat. v. 32, 778–792. https://doi.org/10.14393/SN-v32-2020-51621.
Nepstad, D.C. et al., 1999. Flames in the rain forest: origins, Impacts and alternatives to Amazonian fires. The Pilot Program to Conserve the Brazilian Rain Forest, Brasília, DF.
Nunes, M.H., et al., 2021. Recovery of logged forest fragments in a human-modified tropical landscape during the 2015-16 El Niño. Nat. Commun. 12, 1526. https://doi.org/10.1038/s41467-020-20811-y.
O'Hara, K.L., 2014. Multiaged Silviculture: Managing for Complex Forest Stand Structures. Oxford University Press, https://doi.org/10.1093/acprof:oso/9780198703068.001.0001.
Pan, Y., et al., 2011. A Large and persistent carbon sink in the World's forests. Science 333, 988–993. https://doi.org/10.1126/science.1201609.
Pearson, T.R.H., et al., 2014. Carbon emissions from tropical forest degradation caused by logging. Environ. Res. Lett. 9, 034017 https://doi.org/10.1088/1748-9326/9/3/034017.
Phillips, O.L., et al., 1994. Dynamics and species richness of tropical rain forests. Proc. Natl. Acad. Sci. 91, 2805–2809. https://doi.org/10.1073/pnas.91.7.2805.
Phillips, O.L., et al., 1998. Changes in the carbon balance of tropical forests: evidence from long-term plots. Science 282, 439–442. https://doi.org/10.1126/science.282.5388.439.
Phillips, O.L., et al., 2004. Pattern and process in Amazon tree turnover, 1976–2001. Philos. Trans. R. Soc. B: Biol. Sci. 359, 381–407. https://doi.org/10.1098/rstb.2003.1438.
Phillips, O.L., et al., 2009. Drought sensitivity of the Amazon rainforest. Science 323, 1344–1347. https://doi.org/10.1126/science.1164033.
Phillips, O.L., et al., 2010. Drought–mortality relationships for tropical forests. N. Phytol. 187, 631–646. https://doi.org/10.1111/j.1469-8137.2010.03359.x.
Phillips, O.L., Gentry, A.H., 1994. Increasing turnover through time in tropical forests. Science 263, 954–958. https://doi.org/10.1126/science.263.5149.954.
Piponiot, C., et al., 2016. Carbon recovery dynamics following disturbance by selective logging in Amazonian forests. eLife Sci. J. 5, e21394 https://doi.org/10.7554/eLife.21394.
Piponiot, C., et al., 2019a. Optimal strategies for ecosystem services provision in Amazonian production forests. Res. Lett. 14, 124090 https://doi.org/10.1088/1748-9326/ab5eb1.
Piponiot, C., et al., 2019b. Can timber provision from Amazonian production forests be sustainable? Environ. Res. Lett. 14, 064014 https://doi.org/10.1088/1748-9326/ab195e.
Putz, F.E., et al., 2008. Reduced-impact logging: challenges and opportunities. Ecol. Manag. 256, 1427–1433. https://doi.org/10.1016/j.foreco.2008.03.036.
Putz, F.E., et al., 2012. Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. Cons. Lett. 5, 296–303.
Putz, F.E., et al., 2019. Intact forest in selective logging landscapes in the tropic. Front. Glob. Chang. 2, 30. https://doi.org/10.3389/ffgc.2019.00030.
Putz, E.F., et al., 2022. Sustained timber yield claims, considerations, and tradeoffs for selectively logged forests. PNAS Nexus 1, 1–7. https://doi.org/10.1093/pnasnexus/pgac102.
Radambrasil. 1976. Levantamento dos Recursos Naturais. Folha SC19, Rio Branco. Vol. 12, DNPM, MME. Rio de Janeiro, Brasil, 458pp.

Reategui-Betancourt, J.L., et al., 2023. Timber yield of commercial tree species in the eastern Brazilian Amazon based on 33 years of inventory data. For. Int. J. Res. 1–10. https://doi.org/10.1093/forestry/cpad043.

Riutta, T., et al., 2018. Logging disturbance shifts net primary productivity and its allocation in Bornean tropical forests. Glob. Chang. Biol. 24, 2913–2928. https://doi.org/10.1111/gcb.14068.

Rockwell, C.A., et al., 2014. Logging in bamboo-dominated forests in southwestern Amazonia: Caveats and opportunities for smallholder forest management. Ecol. Manag. 315, 202–210. https://doi.org/10.1016/j.foreco.2013.12.022.

Rodrigues, M.I., et al., 2020. Concessão florestal na Amazônia Brasileira. Ci. Fl. 30 (4), 1299–1308. https://doi.org/10.5902/1980509821658.

Roopsind, A., et al., 2018. Trade-offs between carbon stocks and timber recovery in tropical forests are mediated by logging intensity. Glob. Chang. Biol. 24, 2862–2874. https://doi.org/10.1111/gcb.14155.

Rutishauser, E., et al., 2015. Rapid tree carbon stock recovery in managed Amazonian forests. Curr. Biol. 25, R787–R788. https://doi.org/10.1016/j.cub.2015.07.034.

Sheil, D., et al., 1995. The interpretation and misinterpretation of mortality rate measures. J. Ecol. 83, 331–333. https://doi.org/10.2307/2261571.

Sheil, D. 1998. A half-century of permanent plot observation in Budongo forest Uganda: histories, highlights, and hypotheses, in: Dallmeier, F., Comiskey, J.A. (Eds.), Forest biodiversity research, monitoring and modeling: conceptual background and old world case studies. Unesco, The Pathernon Publishing Group, Paris, pp. 399-428.

Sheil, D., Phillips, O., 1995. Evaluating turnover in tropical forests. Science 268, 894. https://doi.org/10.1126/science.268.5212.894.a.

Sist, P., et al., 2021. Sustainability of Brazilian forest concessions. Ecol. Manag. 496, 119440 https://doi.org/10.1016/j.foreco.2021.119440.

Sist, P., Ferreira, F.N., 2007. Sustainability of reduced-impact logging in the Eastern Amazon. Ecol. Manag. 243, 199–209. https://doi.org/10.1016/j.foreco.2007.02.014.

Soares-Filho, B.S., et al., 2006. Modelling conservation in the Amazon basin. Nat. Lett. 440, 520–523. https://doi.org/10.1038/nature04389.

SUDAM/IBDF/PRODEPEF. 1978. Estudo de Viabilidade Técnico-Econômica da Exploração Mecanizada em Floresta de Terra Firme na Região de Curuá-Una. Belém. Projeto PNUD/ FAO/IBDF/BRA–76/027. 133 p.

Sullivan, M.K., et al., 2022. A decade of diversity and forest structure: post-logging patterns across life stages in an Afrotropical forest. Ecol. Manag. 513, 120169 https://doi.org/10.1016/j.foreco.2022.120169.

Swaine, M.D., et al., 1987. The Dynamics of tree populations in tropical forest: a review. J. Trop. Ecol. 3, 359–366. https://doi.org/10.1017/S0266467400002339.

Swaine, M.D., Withimore, T.C., 1988. On the definition of ecological species groups in Tropical rain forests. Vegetatio 75, 81–86. https://doi.org/10.1007/BF00044629.

Valdiones, A.P. et al., 2022. A evolução do setor madeireiro na Amazônia entre 1980 a 2020 e as oportunidades para seu desenvolvimento inclusivo e sustentável na próxima década. Belém, PA: Imazon: Imaflora: ICV: IDESAM.

Vidal, E., et al., 2016. Recovery of biomass and merchantable timber volumes twenty years after conventional and reduced-impact logging in Amazonian Brazil. Ecol. Manag. 376, 1–8. https://doi.org/10.1016/j.foreco.2016.06.003.

Villegas, Z., et al., 2008. Silvicultural treatments enhance growth rate of future crop trees in a tropical dry forest. Ecol. Manag. 258, 971–977. https://doi.org/10.1016/j.foreco.2008.10.031.

Wadsworth, H.F., Zweede, J.C., 2006. Liberation: acceptable production of tropical forest timber. Ecol. Manag. 233, 45–51. https://doi.org/10.1016/j.foreco.2006.05.072.

West, T.A.P., et al., 2014. Forest biomass recovery after conventional and reduced-impact logging in Amazonian Brazil. Ecol. Manag. 314, 59–63. https://doi.org/10.1016/j.foreco.2013.11.022.

Zanne, A.E. et al., 2009. Global wood density database. http://datadryad.org/repo/handle/10255/dryad.235. (accessed 15 Aug. 2023).

Zimmerman, B.L., Kornos, C., 2012. Prospects for sustainable logging in tropical forests. BioScience 62, 479–487. https://doi.org/10.1525/bio.2012.62.5.9.