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Articles

Structure and dynamics of *Minquartia guianensis* Aublet in the Tapajós National Forest, Belterra – PA

Estrutura e dinâmica de *Minquartia guianensis* Aublet na Floresta Nacional do Tapajós, Belterra – PA

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ABSTRACT

In the Amazon, many species are not managed due to a lack of ecological studies that would assist in the formulation of appropriate forest management systems. Thus, the objective of this study is to evaluate the structure and dynamics of *Minquartia guianensis* before and after timber harvesting and thinning of non-commercial species in an area of terra firme forest in the Tapajós National Forest, state of Pará. In the experimental area, all trees with DBH ≥ 5 cm were measured and identified in 60 plots of 0.25 ha each. The plots were monitored through measurements in 1981, 1983, 1987, 1989, 1995, 2008, and 2012. The horizontal structure, diameter distribution, and basal area were analyzed. Dynamics were assessed using mortality and recruitment rates. The application of forest management techniques based on the characteristics of the species can promote the natural regeneration of *M. guianensis* trees, as was verified in the treatments. Even with the reduction in the density of trees of the species, the population maintained the positive dynamics of trees changing diameter classes and accumulating in the upper classes. The species was among those with the highest importance value in the Tapajós National Forest throughout the monitoring period. Although *M. guianensis* was not extracted from the area, it showed recovery capacity regarding basal area even after disturbances such as harvesting and thinning.

Keywords: Terra firme forest; Ecology; Species management; Acariquara

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RESUMO

Na Amazônia muitas espécies não são manejadas, devido à falta de estudos ecológicos que auxiliem na formulação de sistemas de manejo florestal adequados. Assim, o objetivo do estudo é avaliar a estrutura e a dinâmica de *Minquartia guianensis* antes e após a colheita da madeira e o desbaste de espécies não comerciais em uma área de floresta de terra firme na Floresta Nacional do Tapajós, estado do Pará. Na área experimental foram medidas e identificadas todas as árvores com DAP ≥ 5 cm em 60 parcelas de 0,25 ha cada. O monitoramento das parcelas foi realizado por meio de medições nos anos 1981, 1983, 1987, 1989, 1995, 2008 e 2012. Foram analisadas a estrutura horizontal, a distribuição diamétrica e a área basal. A dinâmica foi avaliada por meio das taxas de mortalidade e de recrutamento. A aplicação de técnicas de manejo florestal com base nas características da espécie pode promover a regeneração natural de árvores de *M. guianensis*, como foi verificado nos tratamentos. Mesmo com a redução da densidade de árvores da espécie, a população manteve a dinâmica positiva de passagem de árvores entre classes de diâmetro e acúmulo nas classes superiores. A espécie esteve entre aquelas com maior valor de importância na Floresta Nacional do Tapajós ao longo de todo o tempo de monitoramento. *M. guianensis*, embora não tenha sido extraída da área apresentou capacidade de recuperação em área basal mesmo após distúrbios, como a colheita e os desbastes

Palavras-chave: Floresta de terra firme; Ecologia; Manejo por espécies; Acariquara

1 INTRODUCTION

Forest management represents an alternative strategy for the rational use of forest resources, as it combines economic development with ecological sustainability, enabling the supply of tropical timber to consumer markets while reducing environmental impacts (Souza; Soares, 2013).

Its ecological foundations are related to the autecology of species and their interactions with the ecosystem, including fauna and flora, whereas its economic foundations are associated with technological characteristics, product commercialization, and the financial and social viability of the activity (Hirai; Carvalho; Pinheiro, 2008).

Despite the importance of forest management, many species in the Amazon remain unmanaged due to the lack of ecological studies that can support the development of appropriate forest management systems. Commercial species harvested under forest management practices therefore require particular attention to prevent negative impacts on their populations. However, potential timber species

that have not yet been adequately studied should also be investigated, including those classified according to their conservation status by the National Center for Flora Conservation (CNCFlora) as NE— (Not Evaluated) and by the IUCN Red List as LC— (Least Concern), with the aim of providing information that can support managers and current legislation in the development of appropriate management systems (Vinhote et al., 2020). Among the species that are still not adequately managed is *Minquartia guianensis* Aublet, commonly known as acariquara. This species belongs to the family Coulaceae and occurs widely in Amazonian forests (Ter Steege et al., 2013).

Previous studies have highlighted the economic and ecological importance of this species (Andrade et al., 2017), which has considerable market demand due to the technological properties of its wood, particularly its mechanical resistance and resistance to attack by xylophagous organisms (Ogeron et al., 2018). The wood is used to manufacture utility poles, fence posts, stakes, and railway sleepers (Ogeron et al., 2018), as well as in house structures and rustic furniture because of its aesthetic properties (Burakouski et al., 2021). Its fruits are consumed by wildlife, and its leaves are used in traditional medicine (Ruiz et al., 2011). However, most trees of *M. guianensis* in natural populations do not reach large diameters, which prevents the species from being included in forest management plans, since Brazilian legislation routinely permits the harvesting only of trees with a diameter at breast height (DBH) of ≥ 50 cm.

In Brazil, environmental legislation establishes fixed harvesting rates without considering the structural characteristics of managed forests (Brasil, 2009; Braz et al., 2018). The search for legal approaches that enable the sustainable use of timber resources is based on knowledge of locally important commercial tree species; therefore, floristic and phytosociological studies of forest vegetation are essential (Freitas; Vinhote; Pinto, 2022).

The study of population dynamics provides information on species-specific parameters, such as (i) diameter growth rates, (ii) mortality rates, and (iii) recruitment rates, as well as on patterns of competition and species distribution within the

forest community (Martins et al., 2018). These data contribute to the monitoring and assessment of forest recovery processes (Lima et al., 2018). In this way, they support sustainable forest management practices and help prevent the local extinction of species with low abundance (Silva et al., 2015).

Knowledge of diameter structure and, particularly, diameter growth patterns is essential for understanding the capacity of a given species to replenish its population during the period defined by the established cutting cycle, thereby ensuring the sustainability of forest production (Braz et al., 2018). Therefore, studies investigating forest species under different management conditions provide valuable information on population dynamics. In this context, the objective of this study was to evaluate the forest structure and population dynamics of *M. guianensis* before and after timber harvesting and thinning of non-commercial species in an upland terra firme forest area within the Tapajós National Forest, western Pará, Brazil.

2 MATERIALS AND METHODS

2.1 Study area

The study area is located in the Tapajós National Forest (FLONA Tapajós), near Km 114 of the BR-163 highway (Santarém–Cuiabá Highway), representing one of the regions where *Minquartia guianensis* occurs in the Amazon. The Tapajós National Forest is a federal Conservation Unit (CU) covering approximately 527,000 hectares and located in western Pará State, Brazil. According to the Köppen climate classification adapted by Martorano, Nechet and Pereira (1993), the regional climate is predominantly classified as Am3, characterized by annual rainfall ranging from 2,000 to 2,500 mm and a moderate dry season, with some months recording precipitation below 60 mm.

From January to July, mean temperatures range around 25.2°C, while from July to December they are approximately 26.0°C (Martorano et al., 2021). The predominant

soil type is Dystrophic Yellow Latosol, which is deep, acidic, and has a heavy texture (Oliveira Júnior et al., 2015). The vegetation is classified as terra firme Dense Ombrophilous Forest, characterized by the presence of large trees, abundant woody climbers, and epiphytes (IBGE, 2012).

2.2 Historical Background of Study Area

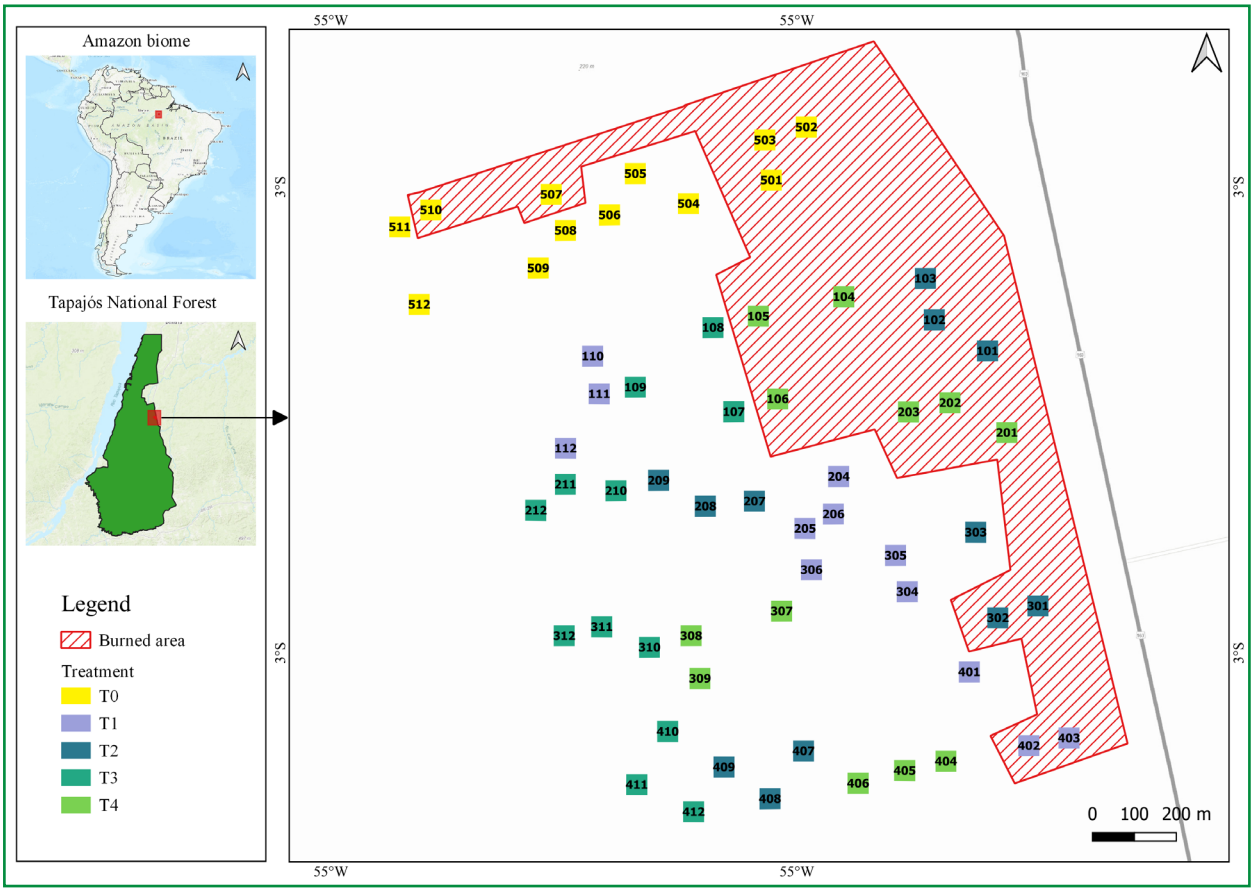
In 1981, a silvicultural experiment was established over an area of 180 ha, including timber harvesting and the application of post-harvesting silvicultural treatments. Planned logging was conducted in 1982, involving the harvesting of 38 species, with an average of 12.5 trees ha^{-1} and a harvested volume of 90 m^3 ha^{-1} (Francez et al., 2024). This volume exceeds the maximum harvesting intensity currently permitted under Brazilian legislation, which is 30 m^3 ha^{-1} (Brasil, 2009). At the time of harvesting, in 1982, legislation established that selective logging should be applied to mature individuals with a minimum cutting diameter of 45 cm, although no specific harvesting intensity was defined. Although *Minquartia guianensis* was abundant in the study area, it was not harvested during the logging operation.

The experimental design consisted of randomized blocks with four replicates. Each block covered 36 ha, with 9 ha allocated to each treatment. To monitor floristic structure and dynamics, in 1981, 36 ha of mature forest and 144 ha of logged forest were selected, totaling 180 ha within the experimental area. Sixty permanent sample plots, each measuring 0.25 ha (50 × 50 m), were established, with 48 plots allocated to the logged forest and 12 plots to the unlogged forest. Plot establishment followed simple random sampling principles. Each plot was subdivided into 25 subplots measuring 10 × 10 m, in which all trees with a diameter at breast height (DBH) ≥ 5 cm were measured, identified, and numbered, following the methodology described by Carvalho (1992).

Tree measurements were conducted in 1981, 1983, 1987, 1989, 1995, 2008, and 2012. However, monitoring of the unlogged forest plots, corresponding to the control

treatment (T0), began in 1983. Silvicultural treatments were applied to the logged forest plots at the end of 1993 and beginning of 1994 (T1, T2, T3, and T4). A forest fire in 1997 affected five plots in the control area (T0) and thirteen plots in the logged areas (T1, T2, T3, and T4) (Figure 1).

Figure 1 – Spatial distribution of permanent sample plots in the experimental area, logged forest, and control area in the Tapajós National Forest, Pará, Brazil



Source: Authors (2025)

Tree identification in the forest was performed by parataxonomists. Botanical material from unidentified trees was collected and subsequently submitted for identification at the Embrapa Eastern Amazon Herbarium in Belém, Pará, Brazil.

The unlogged plots, i.e., the “control” plots, were designated as treatment T0, whereas the remaining plots were assigned to treatments T1, T2, T3, and T4, which were implemented at the end of 1993 and beginning of 1994 (Table 1).

Table 1 –Distribution of treatments applied in the experimental area established in the Tapajós National Forest, Pará, Brazil

Treatment	Description
T0	Control.
T1	Harvesting of commercial timber species with a diameter at breast height (DBH) ≥ 45 cm, with no subsequent intervention.
T2 T3 T4	Harvesting of commercial timber species with DBH ≥ 55 cm, followed by girdling and killing of non-commercial species, resulting in 20%, 40% and 60% reductions in the original basal area, respectively.

Source: Authors (2025)

2.2 Data analysis

Population structure was assessed based on the following variables: absolute density (trees ha^{-1}) and relative density (%), absolute dominance (m^2 ha^{-1}) and relative dominance (%), and absolute and relative frequency (%). The Importance Value Index (IVI) was calculated as the arithmetic mean of the relative values of abundance, dominance, and frequency, resulting in an index ranging from 0 to 100.

Diameter distribution was analyzed based on absolute density (trees ha^{-1}) across diameter classes with a class width of 10.0 cm. Basal area (G) was calculated as the sum of the cross-sectional areas of all trees within each plot and subsequently expressed per hectare. Tree density was calculated as the total number of trees per unit area, expressed in hectares.

Tree recruitment rates (% $year^{-1}$) and mortality rates (% $year^{-1}$) were calculated. Mortality rates were determined based on the number of trees alive in the first measurement and the number of surviving trees recorded in the subsequent remeasurement, considering individuals with DBH ≥ 5 cm. Recruitment was defined as the entry of a tree into the population after reaching a DBH of 5 cm and being recorded for the first time in the population.

For the statistical analysis, data on tree density and basal area were tabulated for the plots surveyed in 1981, 1983, 1987, 1989, 1995, 2008, and 2012. To characterize

the population structure of *M. guianensis*, tree density (trees ha^{-1}) and basal area (m^2 ha^{-1}) were evaluated. Tree density refers to the number of living trees recorded in the plots during a given inventory. Basal area was calculated as the sum of the cross-sectional areas of each tree in each measurement year. Density and basal area values were extrapolated to a per-hectare basis. Recruitment (RR) and mortality (MR) rates were calculated for the following monitoring intervals: 1981–1983, 1983–1987, 1987–1989, 1989–1995, 1995–2008, and 2008–2012.

To assess the effects of silvicultural interventions, time, and forest fires on tree density, basal area, mortality rate (MR), and recruitment rate (RR), generalized linear mixed models (GLMMs) were used, in which the linear predictor included both fixed and random effects. This approach offers the advantage of avoiding biases associated with variable transformations resulting from logarithmic discrepancies, while also allowing more precise estimates of the response variables (Hess et al., 2015). This approach was selected because the data had a hierarchical structure and did not meet the assumptions of conventional linear models, particularly residual normality and homoscedasticity. Therefore, the Gamma distribution with an inverse link function was adopted, as it is more appropriate for positive data with variance increasing with the mean. This choice improved model flexibility and enhanced the robustness of the resulting inferences. This modeling strategy increased model flexibility and the robustness of the statistical inferences. Subsequently, an Analysis of Deviance was performed to assess the significance of the variables included in the model, allowing subsequent mean comparisons.

The GLMMs were fitted in R (version 4.4.1) using the glmer() function from the lme4 package (Bates et al., 2015). The fixed-effects structure included the main effects of treatment, time, and fire occurrence, with the aim of assessing the relative contribution of different forest management intensities, temporal variation associated with long-term monitoring, and fire disturbances to vegetation structure. Treatments were categorized into a control level and four additional levels representing gradients

of silvicultural intervention applied to the sampling units. Time was treated as a categorical factor, allowing comparisons among different points throughout the historical measurement series. Fire was modeled as a binary variable (presence/absence), representing an accidental burning event that affected a non-uniform subset of the experimental plots.

The random-effects structure included a random intercept for block (1 | Block) to account for unexplained variation associated with plot location. In addition, a random term for the block × treatment interaction (1 | Block:Treatment) was included, allowing the management effect to vary among blocks due to environmental heterogeneity. This structure prevented pseudoreplication and ensured the independence of observations. Alternative models, with and without interactions among fixed effects, were also tested (Equation 5, for models without interactions), and the best model was selected based on the Akaike Information Criterion (AIC) and likelihood ratio tests (LRT). The final model included only the main effects because it provided the best balance between model fit and parsimony. Marginal (R^2m) and conditional (R^2c) coefficients of determination were also calculated to assess the proportion of variance explained by the fixed effects and by the fixed and random effects combined, respectively.

$$g(\mu_{ijk}) = \beta_0 + \beta_1 \,.\, Tratamento_j + \beta_2 \,.\, Tempo_k + \beta_3 \,.\, Fogo_{ijk} + u_i + u_{ij} \quad (1)$$

where: $g(\mu_{ijk}) = E[Y_{ijk}]$ is the expected value of the response Variable; g (.) is the inverse link function; β_0 is the overall intercept; β_1 to β_3 are the coefficients of the fixed effects; $u_i \sim N(0, \sigma^2_{Bloco})$ is the Random effect of block; $u_i \sim N(0, \sigma^2_{Bloco:\,Tratamento})$ is the Random effect of the block x treatment interaction.

Before model validation, collinearity among predictors was assessed using the Variance Inflation Factor (VIF), and models with VIF > 10 were discarded. Model validation included graphical analysis of residuals, assessment of variance homogeneity, and evaluation of the normality of random effects, ensuring that the model assumptions were adequately met.

To assess temporal differences within each treatment, paired contrasts were performed with adjustment for multiple comparisons using the Multivariate t (MVT) method, implemented through the *EMMEANS* package (Lenth et al., 2018). The MVT approach is particularly appropriate for unbalanced longitudinal data, in which the number of observations differs among time points, and for data with a dependence structure resulting from repeated measurements. It also provides stringent control of Type I error in multiple comparisons (Rhee et al., 2022).

The data obtained from measurements of the permanent plots were digitized and organized using the Tropical Forest Monitoring (MFT) program, developed by Embrapa Eastern Amazon, and subsequently exported to spreadsheets for statistical analysis. Analyses were performed using Microsoft Office Excel (2010) and R version 4.4.1 (R Development Core Team, 2024).

3 RESULTS AND DISCUSSIONS

3.1 Population Structure, Diameter Distribution, and Basal Area

In the experimental area, among the plots unaffected by the 1997 fire, tree density in T2 was significantly higher than that in the other treatments throughout the monitoring period. In the plots affected by the fire, tree density in treatment T1 was lower than that in T2 and T4, whereas T0 did not differ from T1 and T4 throughout the monitoring period (Table 2).

The temporal analysis of tree density within treatments from 1981 to 2012 revealed no significant differences among measurement years. There was no significant change in tree density within the treatments before and after the fire, indicating that fire may not have been an important source of variation in tree density. Statistical analysis (Table 2) indicated that tree density of the species was not significantly affected by logging, either immediately after harvesting or 29 years after timber extraction.

Table 2 – Mean number of *M. guianensis* trees ha^{-1} in plots unaffected by and affected by fire under treatments T0, T1, T2, T3, and T4 during the 1981–2012 monitoring period in the Tapajós National Forest, Pará, Brazil

Treatment	Unaffected by fire						
	1981	1983	1987	1989	1995	2008	2012
T0	-	9,14 abA	10,29 abA	9,14 abA	10,29 abA	10,29 abA	10,86 abA
T1	10,40 aA	9,20 aA	9,20 aA	9,60 aA	8,80 aA	8,00 aA	8,00 aA
T2	25,33 cA	24,00 cA	24,00 cA	24,00 cA	23,33 cA	21,33 cA	22,00 cA
T3	9,33 aA	9,33 aA	8,67 aA	9,33 aA	8,00 aA	8,67 aA	8,67 aA
T4	10,00 bA	9,33 bA	8,67 bA	9,33 bA	8,00 abA	8,67 bA	8,67 bA
Treatment	**Affected by fire**						
	1981	**1983**	**1987**	**1989**	**1995**	**2008**	**2012**
T0	-	8,80 abA	8,80 abA	8,80 abA	9,60 abA	8,80 abA	8,80 abA
T1	12,00 aA	6,00 aA	8,00 aA	8,00 aA	8,00 aA	8,00 aA	8,00 aA
T2	15,20 cA	15,20 cA	16,80 cA	16,00 cA	16,00 cA	12,80 cA	12,00 cA
T3	-	-	-	-	-	-	-
T4	14,00 bA	13,33 bA	14,67 bA	14,00 bA	15,33 bA	8,67 bA	9,33 bA

Source: Authors (2025)

In where: Treatment T3 had no plots affected by the fire. Means followed by the same lowercase letter within a column indicate that treatments did not differ significantly from one another, whereas means followed by the same uppercase letter within a row indicate that monitoring periods did not differ significantly according to the MVT test at the 5% probability level.

It is important to note that, in absolute terms, the number of *M. guianensis* trees in plots unaffected by fire decreased between the first (1981) and second (1983) measurements in the logged treatments: T1 (−11.53%), T2 (−5.26%), and T4 (−6.70%). In fire-affected plots, reductions were observed in T1 (−50.00%) and T4 (−4.79%). Although the study species was not harvested, its population decreased in the years following timber harvesting activities. However, the reduction in density was not statistically significant and appears to have remained within the range of natural variation observed among the plots (Table 2).

The analysis showed that, in plots unaffected by fire, by the end of the monitoring period (2012), tree density corresponded to 81.25%, 61.54%, 84.21%, 60.71%, and 60.00% of the initial density observed in T0, T1, T2, T3, and T4, respectively. In fire-affected plots, density in 2012 corresponded to 63.64%, 50.00%, 63.16%, and 57.14%

of the initial density recorded in 1981 for T0, T1, T2, and T4, respectively. In the final measurement year, a reduction in tree density was observed in treatments T1, T2, T3, and T4 in both fire-unaffected and fire-affected plots (Table 2).

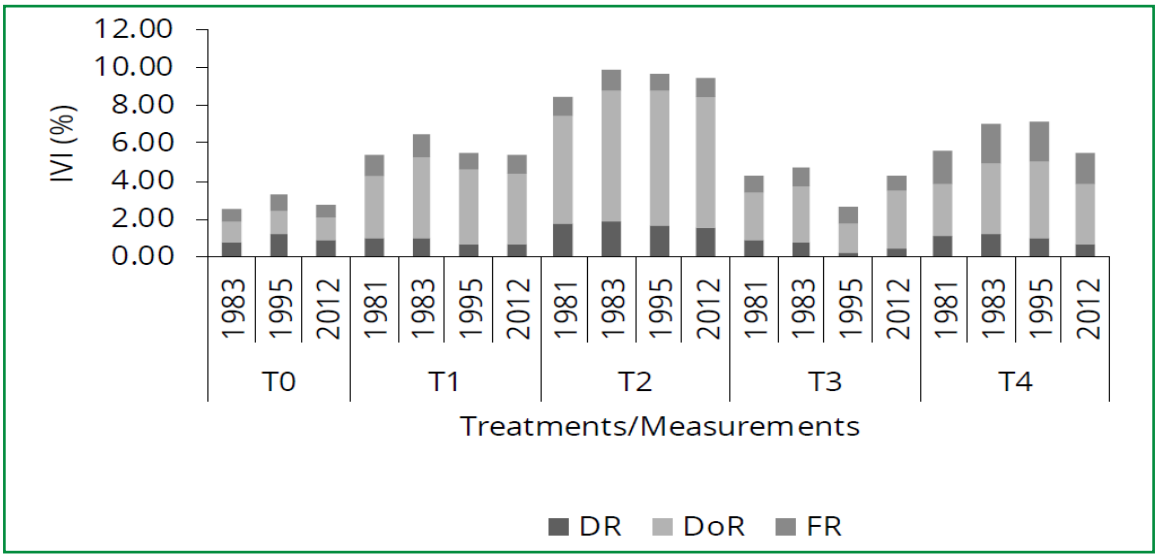
Duque et al. (2003), studying a topographic gradient in the Colombian Amazon, reported high densities of *M. guianensis* across all environments, including valley landscapes, steep slopes, and plateaus, and consequently classified the species as a generalist. Camargo and Ferraz (2004) and Miron et al. (2021) suggested that this broad distribution is likely associated with the species' relatively high tolerance to variations in environmental conditions at both local and regional scales, as well as with its seed dispersal agents, including birds, monkeys, and bats, which facilitate long-distance seed dispersal.

Because frequency, density, and dominance are indicators of the ecological—and often economic—importance of a species within a forest (Silva et al., 2021), several studies have highlighted *M. guianensis* among the ecologically important species of the Amazon region. In the present study, *M. guianensis* ranked among the 20 species with the highest Importance Value Index (IVI) throughout the monitoring period, including in the logged plots.

Over the 31-year monitoring period, the increase in IVI observed in treatments T1 and T2, with the species moving from 10th to 9th place in T1 and from 5th to 1st place in T2, as well as the relative maintenance of its IVI in T3 and T4, demonstrates the species' ability to adapt following disturbances in the study area. This contrasts with other species that may have declined in IVI rank because they were less resistant or less adapted to fire (Figure 2).

This adaptation of *M. guianensis* may indicate that the species benefits from the application of silvicultural treatments, including thinning or liberation treatments targeting other species. This suggests potential opportunities for managing and directing *M. guianensis* populations toward timber production. Braga et al. (2024) recommended the application of post-harvesting thinning to stimulate the growth of *M. guianensis*, thereby promoting faster recovery of its woody biomass stock.

Figure 2 – Importance Value Index (IVI) of *M. guianensis* under treatments T0, T1, T2, T3, and T4 in 1981, 1983, and 2012 in the experimental area at Km 114, Tapajós National Forest, Pará, Brazil

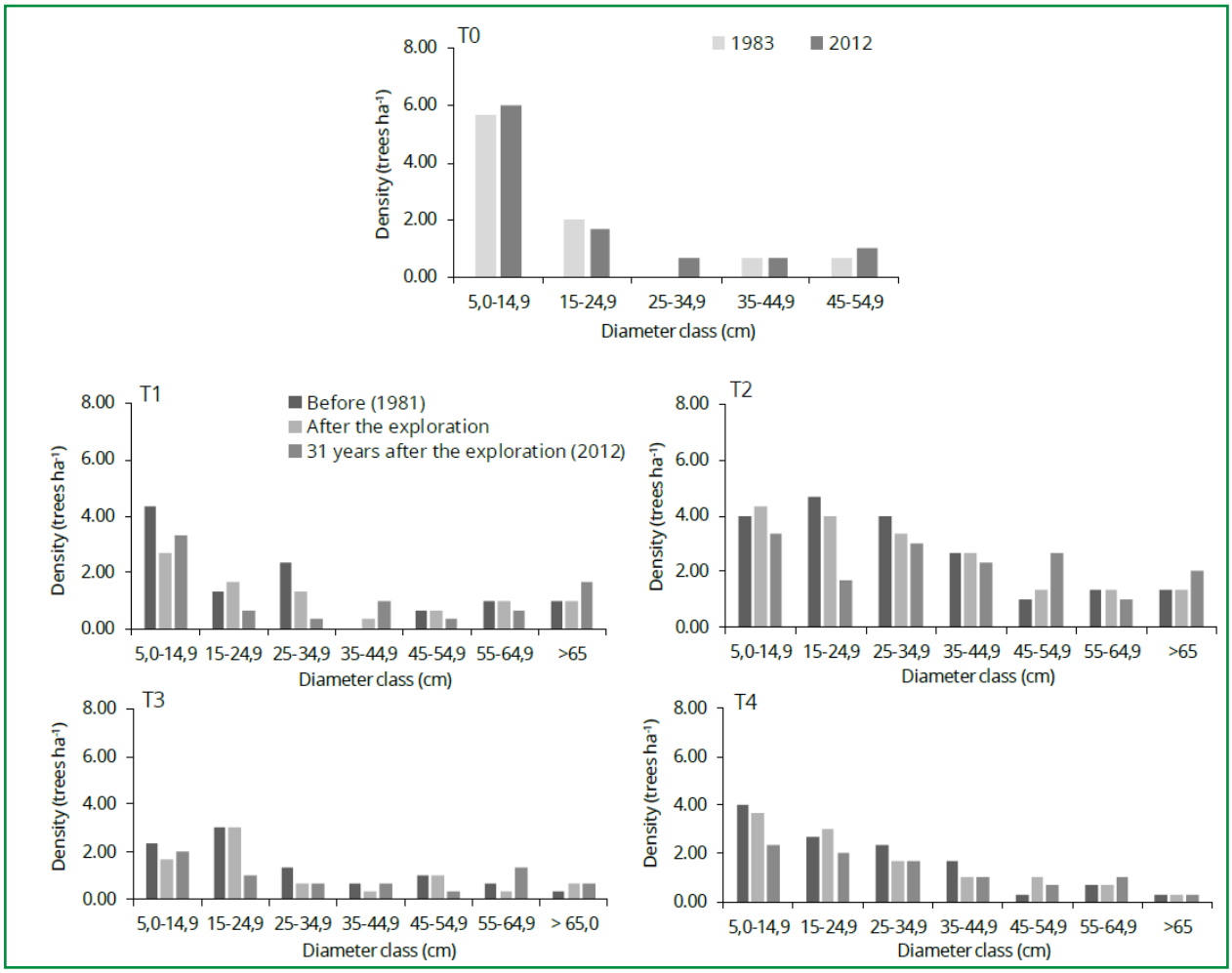


Source: Authors (2025)

Despite the turnover of trees over time and the reduction in population density in T1, T2, T3, and T4, the species increased its absolute dominance in all treatments, except T4, where absolute dominance remained close to the value observed at the beginning of monitoring, 0.84 and 0.83 $m^2\ ha^{-1}$ in 1981 and 2012, respectively. This finding indicates the ability of *M. guianensis* to benefit from canopy gaps created by disturbances such as timber harvesting and thinning, which may promote diameter growth. In the study by Natividade (2018), *M. guianensis* was among the 10 species with the highest IVI in the Tapajós National Forest. Vieira et al. (2015), in a managed forest in the municipality of Placas, Pará, Brazil, reported a density of 7.7 trees ha^{-1} and an IVI of 1.78% for *M. guianensis*. The species ranked third among those with the highest IVI in the results reported by Silva et al. (2021), also in the Tapajós National Forest, and fifth in terms of IVI in the study by Lima et al. (2019), conducted in the Iratapuru River Sustainable Development Reserve, located between the municipalities of Laranjal do Jari, Mazagão, and Pedra Branca do Amapari, Amapá State, Brazil.

Regarding the diameter distribution of *M. guianensis*, following timber harvesting, the diameter classes below 15 cm exhibited higher mortality, resulting in a reduction in tree density within these classes (Figure 3).

Figure 3 – Density (trees ha^{-1}) by diameter class (cm) under the different treatments in 1981, 1983, and 2012 in the experimental area at Km 114, Tapajós National Forest, Pará, Brazil



Source: Authors (2025)

The effects of operational activities, such as road and log-yard construction, tree felling, logging operations, and post-harvesting tree falls, promote canopy-gap formation and increase light availability within the forest. These disturbances may result in increased mortality, particularly in the smaller diameter classes, and reduced recruitment.

In the plots under the evaluated treatments, the diameter distribution results revealed a shift in the population toward larger diameter classes over the 31-year monitoring period. This shift is evident from the accumulation of individuals in the larger diameter classes, such as the 45–54.9 cm classes in T0 and T2, the 55–64.9 cm classes in T3 and T4, and the DBH ≥65 cm classes in T1, T2, and T3. All treatments showed an accumulation of at least 30% more individuals in the three largest diameter classes compared with the beginning of the monitoring period.

As the canopy closes over time, light availability decreases, consequently reducing the recruitment of new trees into the smallest DBH classes. However, *M. guianensis* did not show a deficit in these classes, even after the observed losses, indicating continued regeneration under anthropogenically disturbed conditions. The application of silvicultural treatments between 1993 and 1994, although potentially causing damage to residual trees due to the falling of girdled trees, appears to have contributed to the transition of trees among diameter classes, as the upper classes were continuously replenished through these transitions.

The establishment and development of young adult trees of forest species within a given microhabitat vary according to topography and edaphic factors, as these characteristics influence resource availability and forest disturbance dynamics (Zuleta et al., 2020). Nevertheless, the application of forest management techniques based on species-specific characteristics may promote the natural regeneration of *M. guianensis*. This can be observed in treatments T1, T2, T3, and T4, which, despite having undergone high-intensity logging, maintained positive population dynamics, characterized by the progression of trees among diameter classes and accumulation in the larger classes. However, it is important to emphasize that the accumulation of large trees may lead to future mortality among smaller individuals, as observed by Mory and Jardim (2001), particularly in the smaller diameter classes, due to competition with larger trees. According to Silva et al. (2019), as young plants compete with one another or with plants of other species, mortality may increase during this stage, depending on species-specific characteristics, as a result of natural selection during the transition to adulthood.

M. guianensis exhibited a high number of trees in the smallest diameter classes, indicating the maintenance of regeneration and the continued persistence of the species. These results corroborate the findings of Orellana et al. (2014), who reported that this pattern is consistent with natural forest dynamics, in which relatively few trees attain large diameters due to intense competition for light, water, and nutrients with other species throughout their life cycle.

3.2 Basal Area Recovery

Regarding the basal area of *M. guianensis*, no significant differences among the mean values of treatments T3 and T4 were observed during the 1981–2012 period in the plots unaffected by fire. Treatment T0 exhibited significantly lower basal area, whereas T2 exhibited significantly higher basal area than the other treatments. In the fire-affected plots, significant differences were observed between T1 and T4, with the same pattern observed for T0, which had significantly lower basal area, and T2, which had significantly higher basal area (Table 3).

The temporal analysis of basal area within each treatment across the measurement years revealed no significant differences among monitoring periods. The results indicate that neither logging impacts nor the occurrence of fire significantly altered the basal area of *M. guianensis* populations among the different treatments. These findings corroborate those of Andrade et al. (2020a), who, in a study conducted in the same experimental area, observed that *Rinorea guianensis* Aublet and *Protium apiculatum* Swart were resilient to fire-induced stress, even in areas of dense ombrophilous forest where, according to Barlow et al. (2016), fires are uncommon. Andrade et al. (2020a) suggested that these results demonstrate that fire severity was insufficient to impair the ecosystem's natural capacity to recover from disturbance.

Table 3 – Mean basal area values ($m^2\ ha^{-1}$) in plots unaffected by and affected by fire under treatments T0, T1, T2, T3, and T4 during the 1981–2012 period in the Tapajós National Forest, Pará, Brazil

Treatment	Unaffected by fire						
	1981	1983	1987	1989	1995	2008	2012
T0	-	0,34 abA	0,36 abA	0,36 abA	0,39 abA	0,43 abA	0,44 abA
T1	1,13 bA	1,11 bA	1,17 bA	1,20 bA	1,26 bA	1,19 bA	1,23 bA
T2	2,61 cA	2,63 cA	2,81 cA	2,90 cA	3,04 cA	3,15 cA	3,25 cA
T3	0,73 aA	0,81 aA	0,79 aA	0,87 aA	0,90 aA	1,02 aA	1,05 aA
T4	0,85 aA	0,79 aA	0,80 aA	0,81 aA	0,75 aA	0,69 aA	0,69 aA
Treatment	**Affected by fire**						
	1981	**1983**	**1987**	**1989**	**1995**	**2008**	**2012**
T0	-	0,32 abA	0.33 abA	0,34 abA	0,36 abA	0,29 abA	0,32 abA
T1	0,75 bA	0,75 bA	0.80 bA	0,83 bA	0,87 bA	0,92 bA	0,93 bA
T2	1,27 cA	1,25 cA	1,33 cA	1,36 cA	1,43 cA	1,43 cA	1,45 cA
T3	-	-	-	-	-	-	-
T4	1,32 aA	1,31 aA	1,39 aA	1,43 aA	1,52 aA	1,31 aA	1,33 aA

Source: Authors (2025)

In where:Treatment T3 had no plots affected by fire. Means followed by the same lowercase letter within a column indicate that treatments did not differ significantly from one another, whereas means followed by the same uppercase letter within a row indicate that periods did not differ significantly according to the MVT test at the 5% probability level.

When the entire *M. guianensis* population was analyzed, with all plots combined regardless of whether they were affected by fire, treatments T1, T2, T3, and T4 had basal areas of 1.07, 1.83, 0.73, and 0.84 $m^2\ ha^{-1}$, respectively, before logging (1981). One year after timber harvesting (1983), the corresponding basal areas were 1.05, 1.84, 0.68, and 0.81 $m^2\ ha^{-1}$, respectively. Over the 31 years following harvesting, T1 recovered an additional 0.11 $m^2\ ha^{-1}$ beyond the original total population basal area. For DBH classes <55 cm, basal area decreased from 0.35 $m^2\ ha^{-1}$ in 1981 to 0.25 $m^2\ ha^{-1}$ in 2012, representing a 28.57% reduction from the initial value (Figure 3). However, in diameter classes ≥ 55 cm, basal area recovery was sufficient to compensate for the original basal area lost due to logging impacts, with an increase of 30.99%, corresponding to an additional 0.22 $m^2\ ha^{-1}$ of basal area.

Treatment T2 showed a total increase of 0.40 m^2 ha^{-1} between 1981 and 2012. In the DBH <55 cm classes, basal area increased by 0.14 m^2 ha^{-1}, from 0.99 to 1.13 m^2 ha^{-1}, representing a 14.14% increase. In the diameter classes **≥** 55 cm, basal area increased by 0.26 m^2 ha^{-1}, corresponding to a 30.95% increase relative to the original value. Treatment T3 showed an increase of 0.14 m^2 ha^{-1} between 1981 and 2012. In the DBH <55 cm classes, basal area decreased by 0.20 m^2 ha^{-1} (46.51%), from 0.43 m^2 ha^{-1} in 1981 to 0.23 m^2 ha^{-1} in 2012, whereas the DBH ≥55 cm class accumulated 0.35 m^2 ha^{-1}, increasing from 0.30 m^2 ha^{-1} in 1981 to 0.65 m^2 ha^{-1} in 2012. In T4, overall basal area decreased by 0.01 m^2 ha^{-1}. The DBH <55 cm class showed a reduction of 0.12 m^2 ha^{-1} (22.22%), whereas the DBH ≥55 cm class accumulated 0.11 m^2 ha^{-1} (36.67%).

The post-harvesting dynamics favored the accumulation of basal area in large trees. This accumulation resulted from the transition of trees among diameter classes and the growth of residual trees. Increased light availability resulting from canopy openings may have promoted diameter growth in the remaining trees. Jennings et al. (2001) observed that low-intensity canopy disturbances may contribute to increased growth increments and survival of shade-tolerant species due to increased light availability.

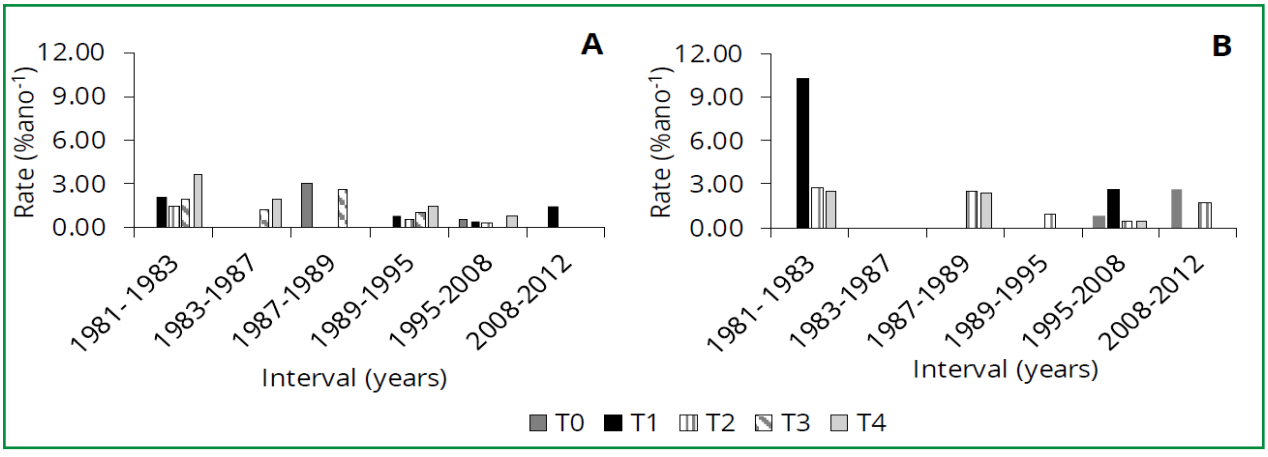
3.3 Mortality and Recruitment Rates

Mortality of *M. guianensis* trees in the study area was considered a rare event. Throughout the entire study period, tree mortality occurred in only half of the time intervals evaluated. In plots unaffected by fire, the estimated annual mortality rates differed significantly between treatments T1 and T2 during 1981–1983, 1989–1995, and 1995–2008. In fire-affected plots, T1 differed significantly from T2 during 1981–1983 and 1995–2008. No significant differences were recorded among the remaining treatments. Significant differences were observed among measurement periods within treatments; however, no effect of fire on the annual mortality rates of *M. guianensis* trees was detected.

In the study by Francez et al. (2024), which analyzed the dynamics of *Carapa guianensis* Aublet in the same study area, higher mortality was recorded during the period immediately following logging (1983–1989), due to the removal of large trees, which primarily affected smaller-diameter trees. During the second period (1995–2003), natural mortality was concentrated among small-diameter trees, regardless of the silvicultural treatment applied.

In the plots unaffected by fire, of all trees alive at the beginning of monitoring (1981 or 1983), 18.75%, 38.46%, 15.79%, 39.29%, and 40.00% had died by the end of the monitoring period (2012) in T0, T1, T2, T3, and T4, respectively. In fire-affected plots, these percentages were 36.36%, 50.00%, 36.84%, and 42.85% in T0, T1, T2, and T4, respectively (Figure 4).

Figure 4 – Annual mortality rates in fire-unaffected (A) and fire-affected (B) plots for trees with DBH ≥ 5 cm under the different treatments from 1981 to 2012 in the Tapajós National Forest, Pará, Brazil



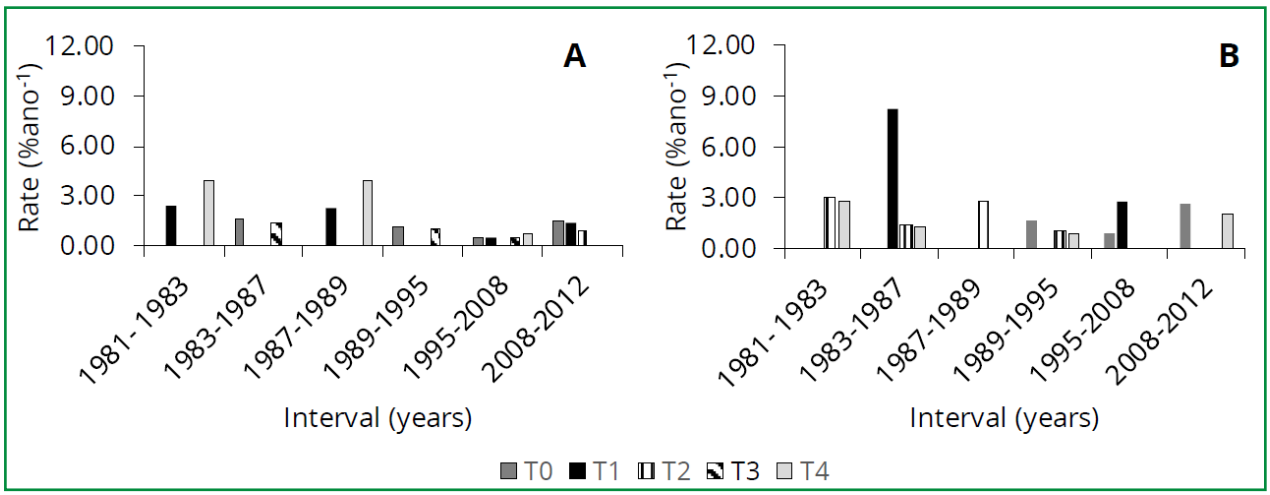
Source: Authors (2024)

Annual mortality rates in tropical forests, whether managed or unmanaged, vary considerably but generally range from 1% to 3% per year (Dobbertin et al., 2001; Kohler et al., 2001; Likoski; Vibrans; Silva, 2022). Silva et al. (2015) evaluated the dynamics of a dense ombrophilous forest and found mortality rates ranging from 2.5% to 3.8%

during 2005–2010. Silva et al. (1995) reported mortality rates of 2.6%, 2.4%, and 2.2% at five, six, and eleven years after harvesting, respectively, in the Tapajós National Forest.

The estimated annual recruitment rates in fire-unaffected plots during 1983–1987 indicated that treatment T0 differed significantly from T3, whereas during 2008–2012, T2 differed significantly from T0 and T1. In fire-affected plots, during 1983–1987, recruitment in T1 differed significantly from T2 and T4. No significant differences were recorded among the remaining treatments (Figure 5). Thus, logging and fire appear to have affected tree recruitment in both fire-affected and fire-unaffected plots.

Figure 5 – Annual recruitment rates in fire-unaffected (A) and fire-affected (B) plots for trees with DBH ≥5 cm under the different treatments from 1981 to 2012 in the Tapajós National Forest, Pará, Brazil



Source: Authors (2024)

At the end of the 31-year monitoring period, recruitment rates showed no marked fluctuations (Figure 5), corroborating the findings of Avila et al. (2017), who demonstrated that following selective logging, the effects of disturbance on recruitment rates are temporary and short-lived. Regarding the management of the species, the mortality and recruitment rates indicate that, for *M. guianensis*, the effects of selective logging and silvicultural treatments were temporary.

Tree mortality may be explained by losses associated with timber harvesting in 1982. Harvesting intensity is a key factor determining forest dynamics, with high-intensity harvesting increasing mortality, recruitment, and growth rates of residual trees (Amaral et al., 2019; Andrade et al., 2020b). Young trees are less resistant to disturbances than large trees because they are more strongly affected by logging operations (Andrade et al., 2020b).

The mortality observed in the treatments, particularly during 1995–2008, may have been associated with severe drought events related to El Niño episodes in combination with the application of silvicultural treatments. Antezana-Vera and Marenco (2021), when evaluating the effects of climatic variability on sap flow rates (SFR) in *M. guianensis* trees in Manaus, observed that droughts associated with events such as El Niño may negatively affect the species, which is characteristic of humid environments. They concluded that although SFRs tend to increase during the dry season, tree growth decreases during the driest months, potentially reflecting a decline in stomatal conductance. However, the findings of Antezana-Vera and Marenco (2021) do not allow us to conclude that climatic events occurring between 1995 and 2008 affected the *M. guianensis* population.

Regarding forest management, Braga et al. (2024) suggested that timber harvesting following a Sustainable Forest Management Plan (SFMP) should be conducted in previously selected operational units, considering species groups based on population density and wood density, or by estimating growth rates. Furthermore, Braz et al. (2012) and Braga et al. (2024) indicated that each SFMP should adopt a site-specific strategic methodology based on the particular characteristics of each species, thereby establishing harvesting scenarios that maximize timber production without compromising the future forest stock.

Therefore, when developing appropriate management systems, it is important that the minimum cutting diameter (MCD) and cutting cycle be established for each species and site, taking into account forest structure, mortality and recruitment rates, and the ecophysiological requirements associated with local site conditions (Braz et al., 2012; David et al., 2019). Based on these considerations and the results presented here,

the characteristics of *M. guianensis* can be used to define tree-selection criteria aimed at enabling sustainable management of the species and, consequently, increasing the availability of its timber products in the market.

Importantly, the establishment of a species-specific MCD for *M. guianensis* could increase its relevance in forest management areas, considering the species› economic potential for commercialization in forest-based and community enterprises in the Brazilian Amazon. This approach could increase the value of standing forests while contributing to the reduction of illegal timber extraction.

4 CONCLUSIONS

The *M. guianensis* population exhibited dynamic changes in density, basal area, and frequency. Nevertheless, species-specific management and the application of silvicultural treatments are recommended to promote natural regeneration and the growth of younger individuals.

Other species declined in their IVI rankings, allowing *M. guianensis* to remain among the 10 species with the highest IVI in the community, demonstrating its resilience to stress caused by fire and anthropogenic disturbances.

Despite the reduction in tree density, the population maintained positive dynamics, characterized by the transition of trees among diameter classes and their accumulation in the larger classes. Although *M. guianensis* was not harvested from the study area, the species demonstrated the capacity to recover basal area increment following disturbances such as timber harvesting and thinning.

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