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Rare tree species in the Atlantic Rainforest: Integrating ecological, spatial and taxonomic analyses

Otávio Miranda Verly
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Thesis submitted to the Forest Science Graduate Program of the Universidade Federal de Viçosa in partial fulfillment of the requirements for the degree of *Doctor Scientiae*.

Adviser: Carlos M. M. Eleto Torres

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Assent:

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To my parents, friends, teachers and the Nature...
... I dedicate this with love.

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“O correr da vida embrulha tudo, a vida é assim: esquenta e esfria, aperta e daí
afrouxa, sossega e depois desinquieta. O que ela quer da gente é coragem.”
(Guimarães Rosa - Grande Sertão: Veredas)

ABSTRACT

VERLY, Otávio Miranda, D.Sc., Universidade Federal de Viçosa, August, 2025. **Rare tree species in the Atlantic Rainforest: Integrating ecological, spatial and taxonomic analyses**. Adviser: Carlos Moreira Miquelino Eleto Torres.

The Brazilian Atlantic Rainforest, rich in endemic species, is severely fragmented and faces increasing pressure from human activities, which affect not only its structure and composition but also the ability to accurately monitor and conserve its biodiversity. To address persistent knowledge gaps, we analyzed long-term data across multiple forest fragments differing in land-use history (LUH) and environmental conditions. This research explores how anthropogenic and environmental factors, such as climate, soil, topography, and LUH, modulate taxonomic and functional diversity, forest dynamics, and the distribution of rare species in the Atlantic Rainforest. In Chapter 1, we investigated how environmental variables and LUH influence tree community diversity. Using forest inventory data from 136 plots across nine forest fragments, we calculated taxonomic and functional diversity indices and built linear mixed-effects models (LMMs) to evaluate the contribution of environmental and anthropogenic factors. Land-use history was included as a random effect, while climatic, topographic, and edaphic variables were grouped as fixed effects. Our results demonstrated that both diversity dimensions were highly heterogeneous across fragments. Taxonomic indices showed similar patterns, with greater explanation by precipitation and temperature. Functional indices were more influenced by climatic and anthropogenic variables, although random effects remained predominant in explaining total variance. In chapter 2, it was assessed how diversity, along with environmental and anthropogenic conditions, influences forest dynamics, particularly changes in abundance and aboveground biomass over time. We used 20 years of forest inventory data from 53 plots in four fragments, encompassing 24,379 individual measurements across 6,838 stems. We categorized species into abundance classes (common, intermediate, and rare) and examined net dynamics of abundance and biomass for each group using LMMs. Our findings revealed distinct demographic responses among abundance classes: common species declined in abundance but increased in biomass, while rare species increased in both. Explanatory variables differed among groups, with diversity and LUH being the most consistent drivers. Notably, rare species were more sensitive to edaphic gradients, whereas common species responded more strongly to climatic factors such as precipitation seasonality. These patterns underscore the importance of maintaining diversity to sustain forest biomass growth and resilience under changing environmental conditions. Chapter 3 focused on the distribution of rare and threatened species. Using data from 137 plots sampled over 30 years, we identified 17 new occurrences and reviewed 243 historical records of 12 rare species. These species showed limited representation in existing botanical collections, with several inconsistencies in taxonomic identification and geolocation. We emphasize the need for continued botanical sampling, especially in poorly explored mountainous regions, and for improved data curation in biodiversity repositories. Accurate information is essential for identifying conservation priorities and informing policy decisions to safeguard these vulnerable forest landscapes. Finally, in chapter 4, we propose and describe *Myrcia magnipunctata* (Myrtaceae), a new, rare and threatened species, characterized by sparsely branched treelets until then restricted to a fragment of Semideciduous Seasonal Forest in the municipality of Coronel Fabriciano.

Keywords: abundance and biomass dynamics; community assembly; digital biodiversity repositories; environmental filtering; forest fragments; human-modified landscape; linear mixed-effects models; rare species conservation

RESUMO

VERLY, Otávio Miranda, D.Sc., Universidade Federal de Viçosa, agosto de 2025. **Espécies arbóreas raras da Mata Atlântica: Integrando análises ecológicas, espaciais e taxonômicas**. Orientador: Carlos Moreira Miquelino Eleto Torres.

A Mata Atlântica brasileira, rica em espécies endêmicas, é severamente fragmentada e enfrenta crescente pressão de atividades humanas, que afetam não apenas sua estrutura e composição, mas também a capacidade de monitorar e conservar com precisão sua biodiversidade. Para abordar lacunas persistentes de conhecimento, analisamos dados de longo prazo em múltiplos fragmentos florestais com diferentes históricos de uso da terra (LUH) e condições ambientais. Esta pesquisa explora como fatores antropogênicos e ambientais, como clima, solo, topografia e LUH, modulam a diversidade taxonômica e funcional, a dinâmica florestal e a distribuição de espécies raras na Mata Atlântica. No Capítulo 1, investigamos como variáveis ambientais e o LUH influenciam a diversidade da comunidade arbórea. Utilizando dados de inventário florestal de 136 parcelas em nove fragmentos florestais, calculamos índices de diversidade taxonômica e funcional e construímos modelos lineares de efeitos mistos (LMMs) para avaliar a contribuição de fatores ambientais e antropogênicos. O histórico de uso da terra foi incluído como um efeito aleatório, enquanto variáveis climáticas, topográficas e edáficas foram agrupadas como efeitos fixos. Nossos resultados demonstraram que ambas as dimensões de diversidade eram altamente heterogêneas entre os fragmentos. Os índices taxonômicos mostraram padrões semelhantes, com maior explicação pela precipitação e temperatura. Os índices funcionais foram mais influenciados por variáveis climáticas e antropogênicas, embora os efeitos aleatórios tenham permanecido predominantes na explicação da variância total. No capítulo 2, foi avaliado como a diversidade, juntamente com as condições ambientais e antropogênicas, influencia a dinâmica florestal, particularmente as mudanças na abundância e biomassa acima do solo ao longo do tempo. Utilizamos 20 anos de dados de inventário florestal de 53 parcelas em quatro fragmentos, abrangendo 24.379 medições individuais em 6.838 troncos. Categorizamos as espécies em classes de abundância (comum, intermediária e rara) e examinamos a dinâmica líquida de abundância e biomassa para cada grupo usando LMMs. Nossas descobertas revelaram respostas demográficas distintas entre as classes de abundância: espécies comuns declinaram em abundância, mas aumentaram em biomassa, enquanto espécies raras aumentaram em ambos. As variáveis explicativas diferiram entre os grupos, com diversidade e LUH sendo os impulsionadores mais consistentes. Notavelmente, espécies raras foram mais sensíveis a gradientes edáficos, enquanto espécies comuns responderam mais fortemente a fatores climáticos, como a sazonalidade da precipitação. Esses padrões ressaltam a importância de manter a diversidade para sustentar o crescimento da biomassa florestal e a resiliência sob condições ambientais mutáveis. O capítulo 3 focou na distribuição de espécies raras e ameaçadas. Usando dados de 137 parcelas amostradas ao longo de 30 anos, identificamos 17 novas ocorrências e revisamos 243 registros históricos de 12 espécies raras. Essas espécies apresentaram representação limitada nas coleções botânicas existentes, com várias inconsistências na identificação taxonômica e geolocalização. Enfatizamos a necessidade de amostragem botânica contínua, especialmente em regiões montanhosas pouco exploradas, e de melhor curadoria de dados em repositórios de biodiversidade. Informações precisas são essenciais para identificar prioridades de conservação e informar decisões políticas para salvaguardar essas paisagens florestais vulneráveis. Por fim, no capítulo 4, propomos e descrevemos *Myrcia magnipunctata* (Myrtaceae), uma espécie nova, rara e ameaçada, caracterizada por arvoretas esparsamente ramificadas, até então restrita a um fragmento de Floresta Estacional Semidecidual no município de Coronel Fabriciano.

Palavras-chave: dinâmica de abundância e biomassa; montagem de comunidade; repositórios digitais de biodiversidade; filtragem ambiental; fragmentos florestais; paisagem modificada pelo homem; modelos lineares de efeitos mistos; conservação de espécies raras

SUMMARY

GENERAL INTRODUCTION

Globally, forest ecosystems have undergone fragmentation, habitat degradation, and widespread biodiversity loss due to anthropogenic activities such as deforestation and urban expansion (Haddad et al., 2015; Sefidi and Copenheaver, 2020; Sefidi et al., 2021; Boulton et al., 2022). In the Neotropics, the Brazilian Atlantic Rainforest is an example of an ecosystem with alarming levels of degradation. Considered a global biodiversity hotspot (Myers et al., 2000; Rezende et al., 2018; Joly et al., 2019), this biome is characterized by high species richness and high rates of endemism (de-Lima et al., 2020, 2024). Historically, the Atlantic Rainforest has experienced intense land-use changes (dos-Santos et al., 2020), with less than 30% of the original territory still forested (MapBiomas, 2024; Vancine et al., 2024), and only 12.4% of this area consisting of primary forest remnants (SOS Mata Atlântica and INPE, 2024). These remnants still face serious threats, including deforestation, urbanization, and agricultural expansion, which lead to habitat fragmentation (Gelli et al., 2023). Consequently, most of the Atlantic Rainforest landscape is dominated by isolated secondary forest fragments, with 97% of them being smaller than 50 ha (Joly et al., 2014; Rosa et al., 2021; Vancine et al., 2024), resulting in most of its forests being located less than 100 m from the edges (Ribeiro et al., 2009; Vancine et al., 2024).

These external pressures, as well as internal pressures imposed by different management practices (Dieler et al., 2017; Arcanjo et al., 2023), alter the structural pattern of forests through their impact on the populations of both common and rare species (Hordijk et al., 2024). Chronic exposure to anthropization (de-Aguiar et al., 2025), or the combination of these factors and extreme environmental conditions may lead to population declines or even extinction of endemic or rare species (Idárraga-Piedrahíta et al., 2022), or collapsing demographic fluctuations even in dominant species with wide distributions (Stanke et al., 2021).

Although anthropogenic factors are currently a leading factor in the regulation of forest ecosystems, environmental factors are natural drivers of forest dynamics, shaping their structure, diversity and distribution of rare species. Topographic (Körner, 2007; Zhang et al., 2016, 2013; Nüchel et al., 2019) and edaphic properties drive vegetation structure, diversity and composition (Zhang et al., 2016; Chen et al., 2018; Pelletier et al., 2019; Ouyang et al., 2021). Topography, in particular, plays a critical role by creating fine-scale environmental heterogeneity that shapes species richness and community assemblages (Wang et al., 2016; Rodrigues et al., 2020). Climatic factors, especially temperature and precipitation (Antúnez et al., 2017; Chen et al., 2018; Pelletier et al., 2019), control species growth and distribution by imposing ecophysiological limits to their development (Salomón et al., 2022; Mirabel et al., 2023; Liu et al., 2024). With the worsening of climate change, forest alteration processes are intensifying (Aleixo et al., 2019; Senf et al., 2020; Chen et al., 2024), threatening the survival of vulnerable species (Körner, 2007; Diem et al., 2018; Boulton et al., 2022).

Ultimately, anthropogenic factors interact with environmental ones, resulting in structurally distinct forest communities that harbor different levels of diversity, influencing the turnover of species, individuals, and biomass (Sfair et al., 2016; Teixeira et al., 2020; Sui et al., 2025). Consequently, diversity functions as a mediating factor between environmental conditions and forest dynamics (Dilnessa et al., 2023).

In this context, common species and rare species respond differently to environmental conditions and anthropogenic pressures, leading to distinct forest dynamics depending on the proportion of these groups (Köhler and Huth, 2004; Stanke et al., 2021; Hordijk et al., 2024). Common species are generally more limited by climatic variables such as rainfall patterns, as they have high photosynthetic capacity (dos-Santos et al., 2019), and require high water availability for their rapid growth (Pulla et al., 2021; Li et al., 2023). In contrast, rare species are more sensitive to anthropogenic activities, with reductions in their abundance and severely impaired recruitment (Köhler and Huth, 2004). Furthermore, they are highly dependent on acidity (Šantrůčková et al., 2019; Wells and Sawyer, 2022; da-Rocha et al., 2024) and fertility conditions (Madrigal-González et al., 2023; Yao et al., 2025), which in turn are strongly affected by land-use history (Lishan and Regasa, 2022; Wells and Sawyer, 2022; Baláš et al., 2024).

Despite its remarkable biodiversity, composed of hundreds of rare and threatened species, sampling in tropical ecosystems such as the Atlantic Rainforest is challenging, particularly in remote or sparsely populated regions (Hortal et al., 2015; Hopkins, 2019; Stropp et al., 2020; Hughes et al., 2021). Botanical collections remain scarce in critical areas like mountainous regions (de-Araujo and Ramos, 2021). These sampling gaps impair biodiversity indicators, delay species discovery, and restrict the delineation of their distributions (Canhos et al., 2015; Hortal et al., 2015; Oliveira et al., 2016). To overcome these limitations, in recent decades, digital repositories of floristic data have emerged from institutional collaborations (Canhos et al., 2015; Hortal et al., 2015; Lannuzel et al., 2022). This improves access to herbarium records and overcomes barriers to information sharing (Canhos et al., 2022; SpeciesLink network, 2024), although some remain limited by institutional networks (Marsico et al., 2020; de-Araujo et al., 2022). Even though they are not free from technical inconsistencies, integrating national and global platforms helps reduce sampling gaps (de-Araujo and Ramos, 2021; Heberling et al., 2021; de-Araujo et al., 2022), especially because many species are known only from herbarium specimens (Oliveira et al., 2016; Marsico et al., 2020). This is particularly critical for rare species, where identification challenges and limited collections create a feedback loop that hinders current and future taxonomic progress (Hortal et al., 2015). Furthermore, in the Atlantic Rainforest, as tropical forests exhibit the highest species richness and proportion of rare species worldwide (Hordijk et al., 2024), and there is a tendency for a positive relationship between diversity and the provision of ecosystem services (Gamfeldt et al., 2013; Cavanaugh et al., 2014; Snäll et al., 2021).

Among these services, carbon storage is particularly important in tropical forests (Steur et al., 2022; Drüke et al., 2023; Liu et al., 2024), especially their secondary formations, which are important carbon sinks, contributing to the fight against anthropogenic climate change (Rodrigues et al., 2023; Altomare et al., 2025). Thus, determining which and how biotic and abiotic environmental factors modulate forest dynamics is essential for the conservation of these ecosystems, with a focus on how different dimensions of diversity, the abundance and biomass of common and rare species, and the occurrence and distribution of rare species are affected, as well as on taxonomic investigations that may reveal scientifically undescribed species. Therefore, this research was divided into four chapters to achieve the proposed objectives: (Chapter I) determine how anthropogenic and environmental variables (topographic, edaphic, and climatic) modulate taxonomic and functional diversity; (Chapter II) investigate how all these variables impose, directly and indirectly through their effects on diversity, different dynamic processes on common and rare species; (Chapter III) make new records of rare species and map them along with historical records, assessing the adequacy of their botanical collections in digital biodiversity repositories; and (Chapter IV) describe a new rare and threatened species of the genus *Myrcia* (Myrtaceae).

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CHAPTER I:

LAND USE HISTORY AND ENVIRONMENTAL FILTERING DRIVE TAXONOMIC ANDFUNCTIONAL DIVERSITY IN ATLANTIC RAINFOREST FRAGMENTS

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Research article

Land use history and environmental filtering drive taxonomic and functional diversity in Atlantic Rainforest fragments

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ABSTRACT

The Brazilian Atlantic Rainforest is a fragmented biodiversity hotspot where anthropogenic and environmental drivers shape species composition, abundance, and dimensions of tree diversity. This study addresses how anthropogenic disturbance and edaphoclimatic and topographic environmental factors influence taxonomic and functional diversity in Atlantic Rainforest. We used forest inventory data from 136 plots across nine fragments with different land-use histories. Environmental variables and taxonomic and functional diversity indices were obtained at the plot level. We explored the relationships between the variable sets and diversity by building linear mixed-effects models (LMM), in which land-use history was included as a random effect in all models, while the remaining variables were grouped as fixed effects. The fragments were dissimilar, and taxonomic and functional diversity indices values were heterogeneous. The LMM showed varied performance among the variable groups and diversity indices, with greater influence from random effects, while climatic and anthropogenic models stood out among the fixed effects. Taxonomic indices showed similar patterns, with greater explanation by precipitation and temperature, while other models had low explanatory power. Functional indices were more influenced by climatic and anthropogenic variables, although random effects remained predominant in explaining total variance. Anthropogenic variables related to land-use history and time since abandonment, as well as climatic variables related to temperature, precipitation, and water deficit, are the main drivers of taxonomic and functional diversity in Brazilian Atlantic Rainforest fragments.

1. Introduction

Forest ecosystems around the world have been suffering from anthropogenic activities, such as deforestation for agriculture and urban expansion, which drive fragmentation and widespread biodiversity loss, along with the destruction of their habitats (Haddad et al., 2015; Sefidi and Copenheaver, 2020; Sefidi et al., 2021; Boulton et al., 2022). The Brazilian Atlantic Rainforest is a striking example of a global biodiversity hotspot (Myers et al., 2000; Rezende et al., 2018; Joly et al., 2019), characterized by higher species richness and high rates of endemism (de-Lima et al., 2020; de-Lima et al., 2024). Historically, this biome has experienced intense land use changes (dos-Santos et al., 2020), and currently, only 28.1 % of the original territory remains forested (MapBiomas, 2024), and just 12.4 % of this area consists of primary forest remnants formations (SOS Mata AtlânticaINPE, 2024). Therefore, most of the Atlantic Rainforest landscape is dominated by isolated secondary forest fragments, often confined to small and isolated fragments (Joly et al., 2014; Rosa et al., 2021).

Several environmental variables, such as topographic (Körner, 2007; Zhang et al., 2016; Nüchel et al., 2019) and edaphic properties drive vegetation structure, diversity and composition (Zhang et al., 2016; Chen et al., 2018; Pelletier et al., 2019; Ouyang et al., 2021).

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Topography, in particular, plays a critical role by creating fine-scale environmental heterogeneity that shapes species richness and community assemblages (Wang et al., 2016; Rodrigues et al., 2020). Altitudinal variation also influences forest composition, with montane forests dominated by species adapted to cooler, wetter conditions, such as Araucaria forests in the south, contrasting with lowland semi-deciduous and evergreen forests (Joly et al., 2014; Rodrigues et al., 2020). Furthermore, topography affects soil moisture and nutrient availability, which in turn modulate species distributions and successional pathways (Zhang et al., 2016; Pelletier et al., 2019). Climatic factors, especially temperature and precipitation (Antúnez et al., 2017; Chen et al., 2018; Pelletier et al., 2019), as well as extreme events such as droughts and storms (Crockett and Westerling, 2018; Senf et al., 2020; Bauman et al., 2022; Villanova et al., 2024), also control species growth and distribution by imposing ecophysiological limits to their development (Salomón et al., 2022; Mirabel et al., 2023; Liu et al., 2024). With the worsening of climate change, forest alteration processes are intensifying (Anderegg et al., 2020b; Senf et al., 2020; S. Chen et al., 2024), threatening the survival of vulnerable species (Körner, 2007; Diem et al., 2018; Boulton et al., 2022) and compromising the provision of essential ecosystem services (Cavanaugh et al., 2014; Anderegg et al., 2020a; Inague et al., 2021).

Although natural environmental gradients influence species distribution, the Brazilian Atlantic Rainforest illustrates how anthropogenic pressures in highly fragmented landscapes can drastically alter ecosystem dynamics (Tavares-Júnior et al., 2020; Torres et al., 2023; da-Rocha et al., 2024). Extensive deforestation for selective logging, livestock, agriculture, such as sugarcane and coffee plantations, has replaced diverse forests with monocultures, disrupting seed dispersal and reducing forest coverage (Chen et al., 2018; Ouyang et al., 2021). Fire regimes associated with pasture management further degrade sensitive taxa and promote the growth of fire-tolerant vegetation (Dawe et al., 2022). Proximity to urban and agricultural areas intensifies these disturbances, with many forest fragments experiencing increased edge effects, higher mortality rates, and invasion by non-native (Boulton et al., 2022; Torres et al., 2023). As a result, the remaining isolated secondary forests exhibit reduced structural and functional diversity, including significant declines in large-seeded tree species essential for ecosystem resilience and carbon storage (Haddad et al., 2015; Nüchel et al., 2019; Dasgupta et al., 2022).

The impacts of anthropogenic activities on tree communities may be more pronounced than the influence of intrinsic environmental factors (Flores et al., 2024). Therefore, understanding the drivers of tree species composition and diversity is fundamental for promoting effective forest resource management and conservation (Thammanu et al., 2021). One way to measure and understand these impacts is by investigating the relationships between these factors and the different diversity dimensions of tree communities. The biodiversity dimensions include phylogenetic, taxonomic, and functional diversity, each reflecting distinct aspects of ecosystem structure and functioning, and responding differently to environmental and anthropogenic influences (Gregorius, 2009; Nakamura et al., 2020; Castillo-Campos et al., 2024). Taxonomic diversity, which considers species identity and abundance, is typically assessed through species richness and evenness indices, and remains the most traditional approach to comparing biodiversity across communities (Simpson, 1949; Pielou, 1975; Su, 2016). Functional diversity reflects the range and distribution of species' ecological traits, such as morphological, physiological, or phenological characteristics, that influence ecosystem processes and determine species' responses to environmental variation and disturbance (Botta-Dukát, 2005; Schleuter et al., 2010; Legras et al., 2018). Together, these dimensions provide a more comprehensive understanding of biodiversity patterns and their ecological implications.

The investigation and understanding of the influence patterns of environmental and anthropogenic factors on the different dimensions of diversity in highly threatened and diverse environments such as the Atlantic Rainforest may ensure the conservation of these landscapes, facilitating the proposal of environmental policies and preferential practices for sustainable management (Sefidi and Copenheaver, 2020; Dasgupta et al., 2022). Therefore, identifying the drivers that promote or limit diversity is crucial for quantifying changes in forest resilience and its role in the global carbon balance and climate change mitigation (Madrigal-González et al., 2020; Boulton et al., 2022; S. Chen et al., 2024; Flores et al., 2024). Mainly because more diverse forests provide greater availability of ecosystem services (Gamfeldt et al., 2013; Cavanaugh et al., 2014; Snäll et al., 2021), particularly the carbon storage (Steur et al., 2022; Drüke et al., 2023; Liu et al., 2024).

In this study, we examine how different dimensions of tree diversity (taxonomic and functional) respond to anthropogenic and environmental variables across nine fragments of Atlantic Rainforest. Given the inherent heterogeneity among forest fragments, we anticipate that the effects of these variables will vary spatially. In general, we expect (i) anthropogenic variables (fragment size and forest cover) affect negatively diversity, especially species richness (Cardelús et al., 2019), and that edge effects drive richness and functional diversity declines (Ibáñez et al., 2017); (ii) the modified Shannon-Weaver index is more affected by climatic and soil factors (Paz et al., 2021; Gianasi et al., 2024); and, (iii) elevation negatively affects functional diversity (Oseguera-Olalde et al., 2022).

2. Material and methods

2.1. Study area

We used data from 136 plots installed in nine fragments of Atlantic Rainforest distributed across the eastern portion of Minas Gerais, Brazil (Fig. 1). These fragments are heterogeneous in their environmental conditions of climate, topography and soil, as well as in landscape features and forest characteristics such as structure and floristic composition (Table 1; for further details, see Torres et al., 2023). All fragments are areas of environmental conservation, consisting of secondary forests regenerated after various land uses, such as selective logging, silviculture, and agriculture (Torres et al., 2023).

2.2. Forest variables

2.2.1. Community composition and structure

We conducted forest inventories in all areas, considering palms, shrubs, and trees with a Diameter at Breast Height (DBH) of 1.3 m or greater, with a minimum diameter of 5.0 cm. The species list was compiled based on the classification of angiosperm families from the APG IV (APG, 2016), and the scientific nomenclature used followed the "Flora e Funga do Brasil" (2025) and Plants of the World Online (POWO, 2025). We compiled the number of individuals, number of stems, basal area, and species richness for each plot. We also investigated species composition dissimilarity using Bray–Curtis index, which is useful for quantifying compositional dissimilarity between two different sites based on both presence-absence and species abundance values (Bray and Curtis, 1957; Ricotta and Podani, 2017).

2.2.2. Diversity indices

We adopted a plot-level approach to calculate all variables reflecting taxonomic and functional diversity (Castillo-Campos et al., 2024), in order to assess how anthropogenic and environmental variables influence the tree diversity. Taxonomic diversity was assessed through the first two Hill's true diversity values (Hill, 1973; Jost, 2006), and functional diversity was evaluated through functional richness and functional diversity (Mouchet et al., 2010). The Hill's true diversity values (^{q}D) are diversity measures distributed along a scale of diversity orders, where the exponent q defines the sensitivity of the measure to species abundance (Jost, 2006). We obtained two values with different diversity

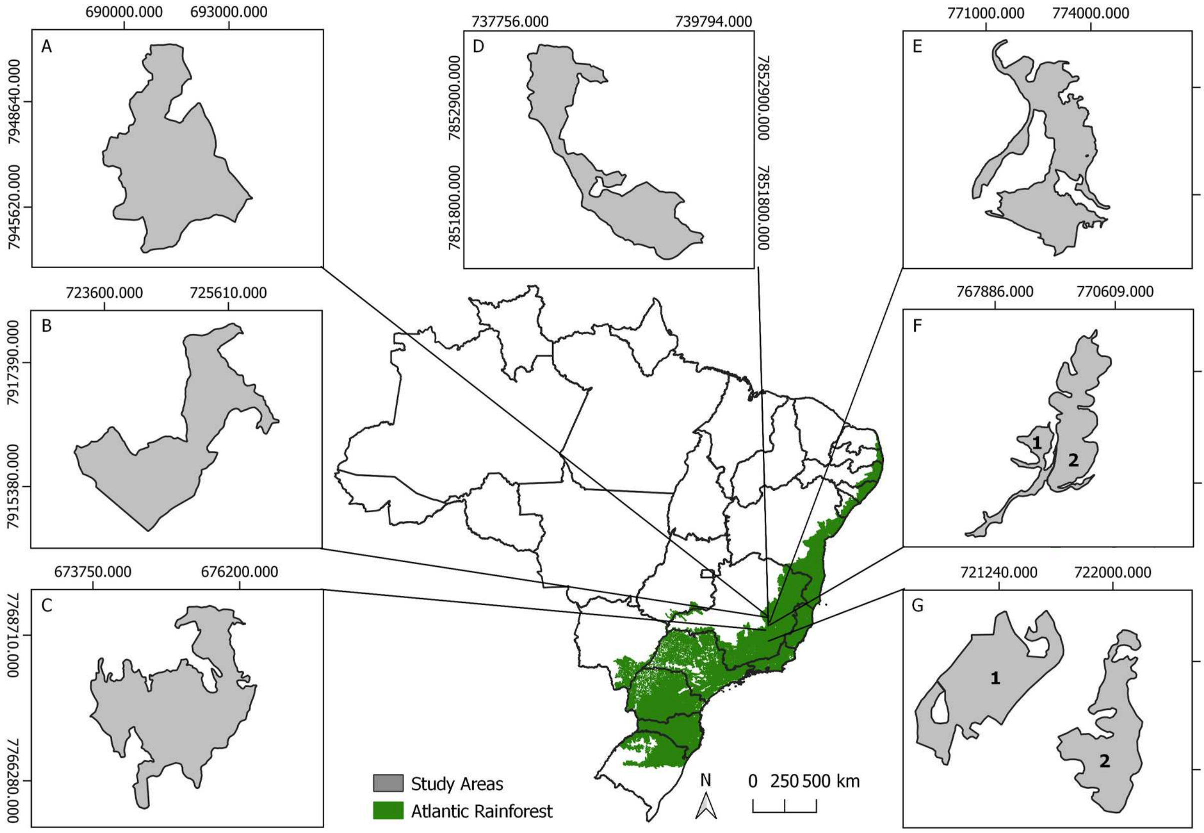

Figura. 1. Locations of the nine studied Atlantic Rainforest fragments located in Minas Gerais, Brazil. **A.** Rio Guanhães; **B.** Cachoeira das Pombas; **C.** Chapadão; **D.** São José; **E.** RPPN Fazenda Macedônia; **F1.** Ipaba Mata2; **F2.** Ipaba Mata1; **G1.** Mata da Silvicultura; **G2.** Mata da Garagem.

Table 1
Characteristics of the nine Atlantic Rainforest fragments in Minas Gerais, Brazil, included in this study.

Fragment	Acronym	Municipality	Area (ha)	Plots	Plots Establishment	Tree Richness	Maximum Elevation	Year of Inventory
Chapadão	CAL	Catas Altas	417.0	15	Systematic	223	860	2023
Mata da Garagem	GAR	Viçosa	31.5	10	Systematic	92	727	2023
Cachoeira das Pombas	GUA	Guanhães	360.2	20	Random	182	1169	2022
Ipaba Mata1	IP1	Caratinga	271.2	16	Random	129	299	2022
Ipaba Mata2	IP2	Caratinga	79.4	6	Random	106	310	2022
RPPN Fazenda Macedônia	MAC	Ipaba	867.0	23	Systematic	206	347	2021
Mata da Silvicultura	MSI	Viçosa	46.2	20	Random	155	744	2024
Rio Guanhães	SAB	Sabinópolis	1748.0	15	Systematic	152	904	2023
São José	SJO	Coronel Fabriciano	100.6	12	Random	246	929	2022

orders: species richness (^{0}D) and the exponential of the Shannon-Weaver Diversity index [Exp (H')], which emphasizes the number of common or abundant species (^{1}D) (Chao et al., 2014; Hsieh et al., 2016).

We assessed functional diversity using two metrics: functional richness (FRic), which is independent of species abundance, and Rao's quadratic entropy (RaoQ), which is weighted by abundances. FRic is a measure that represents the functional space occupied by the community (Cornwell et al., 2006), and is defined by the intersection of functional traits in a multidimensional space (Villéger et al., 2008; Laliberté and Legendre, 2010; Mouchet et al., 2010), reflecting the amount of available resources in the community (Mason et al., 2005; Córdova-Tapia and Zambrano, 2015). In contrast, Rao's quadratic entropy (Rao, 1982) estimates functional diversity from the sum of pairwise functional distances among species, weighted by their relative abundance (Mouchet et al., 2010). This index may decrease even with higher species richness, as it is influenced by functional redundancy and by the dominance of species with similar traits (Botta-Dukát, 2005; Ricotta and Szeidl, 2009; Castillo-Campos et al., 2024).

Functional diversity measures were calculated based on four morpho-ecological traits that represent different aspects of ecosystem functioning, using a previously adapted approach (Castillo-Campos et al., 2024). We consulted widely used specialized literature for the Atlantic Rainforest (e.g. Lorenzi, 2010; 2009; 2009b), botanical descriptions available on Flora e Funga do Brasil (2025), and when data were lacking, we supplemented them with field observations. Finally, we compiled, for each species, the reproductive traits: i) fruit type and ii) seed dispersal syndrome (anemochory, autochory, and zoochory); and the ecological traits: iii) leaf phenology (evergreen, deciduous, and semi-deciduous), and iv) successional group (pioneer, early secondary, late secondary, and climax; we consider some alien species as pioneers) (Gómez-Ortiz et al., 2019). We noted the absence of consensus regarding fruit type nomenclature. Therefore, we relied on extensive botanical glossaries (Brasil, 2009; Gonçalves and Lorenzi, 2011), and developed our own classification system (Table S1).

A functional dissimilarity matrix was built using Gower distance on a matrix of species and their functional traits (Yixuan Li et al., 2022).

Next, we transformed categorical traits into discrete numbers and standardized them (mean = 0, variance = 1) so that all would carry equal weight in estimating the functional volume (Villéger et al., 2008; Laliberté and Legendre, 2010). Then, we added small perturbations (1^{-10000}) to the standardized values to avoid multidimensional null sums from highly collinear traits (Cornwell et al., 2006). We also calculated community-weighted means (CWM) for each plot using scalar values ranging from acquisitive to conservative strategies for the successional group (1 – pioneers, including exotic species; 2 – early secondaries; 3 – late secondaries; and 4 – climax) and for deciduousness (1 – deciduous; 2 – semi-deciduous; and 3 – evergreen) to investigate trait responses to environmental variation. Indices were calculated using the “dbFD” function of the “FD” package (Laliberté et al., 2014) in the R Studio environment (RStudio Team, 2020).

2.3. Anthropogenic and environmental drivers

2.3.1. Anthropogenic variables

We used forest structure attributes and landscape variables to investigate anthropogenic effects on the different dimensions of diversity. For this, we obtained, at the plot level, the basal area extrapolated per hectare (BA), used as a proxy for forest age, as it increases significantly along the successional trajectory from young to mature forests (Menezes and Melo, 2019; van-Breugel et al., 2019; Verly et al., 2023), and influences community demographic processes (Rozendaal and Chazdon, 2015).

The landscape variables included: land-use history (LUH), time since abandonment (time since the end of previous anthropogenic uses), forest cover, forest size, and distance to the edge. To obtain these variables, we classified land-use history of each of 136 plots into three categories: agricultural production (12), deforestation (13), and selective logging (111). We compared land-use information gathered from neighboring landowners and from the property deeds, with aerial photographs (from 1960 to 1996) and Landsat satellite images from 1985 to improve our understanding of land-use changes and determine the time since abandonment of the fragments. Forest cover was calculated for different years (1985, 2002, and 2017) and for circular buffers with different radii (500, 1000, and 2000 m) around each plot (Torres et al., 2023). These data were retrieved from MapBiomas (2024), which classifies land cover and land use based on Landsat imagery with a resolution of 30 × 30 m, processed in the Google Earth Engine platform. We used QGIS 3.28 Firenze (QGIS Development Team, 2025) to manipulate the buffers, integrate spatial information, and analyze fragmentation.

2.3.2. Soil and topography variables

We collected composite soil samples from each plot at depths of 0–20 cm and 20–40 cm. We then performed chemical analyses: soil active acidity in H_2O (pH), exchangeable acidity (Al^{3+}), total acidity ($H^{+}+Al^{3+}$), exchangeable cations (Ca^{2+}, K^{+}, and Mg^{2+}), total and effective cation exchange capacity (T and t), sum of exchangeable bases (SB), base saturation (V), available phosphorus (P) and remaining phosphorus in solution (P-Res), aluminum saturation index (m), and soil organic matter (SOM). Analyses were performed at the Soil Fertility Laboratory of the Federal University of Viçosa (UFV), Brazil, following standard methods (Teixeira et al., 2017).

Topographic variables were assessed through calculations of elevation, declivity percentage, and aspect (exposure face angle) of each plot (Torres et al., 2023; da-Rocha et al., 2024). For this, we used a Digital Terrain Model (DTM) for the study area. The processing was performed in R version 2024.12.0 (RStudio Team, 2020), using the "terra" package (Hijos, 2024). Initially, the DTM was clipped based on the boundaries of the sample plots. Then, elevation, declivity percentage, and aspect were calculated, and the mean values for each plot were extracted.

2.3.3. Climate variables

We used data recorded by the weather station closest to each forest fragment to analyze different climatic metrics. We followed previous studies that used different sets of these variables and modeled different aspects of the forest community (Torres et al., 2023; da-Rocha et al., 2024). Therefore, we calculate the mean temperature (Temp) and total precipitation of the inventory year (Precp), of the previous year (Precp-1), the mean of the two (Precp-2), three (Precp-3), and four (Precp-4) preceding years, total precipitation in the three driest months of the inventory year (Precp-Dry), and the number of months with precipitation <100 mm (Less100).

We also estimated the climatic water deficit (CWD), as water stress is one of the main drivers associated with tree recruitment and mortality in the Atlantic Rainforest (Torres et al., 2023). Furthermore, it can incorporate the effects of anthropogenic variables on the water balance, reflecting drought conditions more accurately than conventional climatic variables, such as total annual precipitation (Chave et al., 2014), and therefore may also be considered an anthropogenic variable. This variable represents the water balance between potential evapotranspiration (PET) and real evapotranspiration (AET) and is used as a proxy for drought conditions. We follow a previously proposed calculation protocol (Lutz et al., 2010), and used the CWD and AET functions (Redmond, 2022) in the R Studio environment (RStudio Team, 2020). The CWD calculation, in addition to local precipitation and temperature, includes the declivity, aspect and location (geographic coordinates) of the plot. The index is interpreted such that the more negative the CWD values, the higher the water stress, whereas values close to zero indicate the absence of water stress (Poorter et al., 2017). As the effects of water stress vary and can have delayed impacts on tree development and mortality (Anderegg et al., 2020b; Bauman et al., 2022; S. Chen et al., 2024), we also calculated the mean CWD for one (CWD-1), two (CWD-2), three (CWD-3), and four (CWD-4) years prior to the measurement year, in addition to the CWD of the inventory year.

2.4. Data analysis

2.4.1. Variables selection

We tested our initial variables to reduce the dimensionality of variables to be included in the statistical models for predicting diversity. First, we conducted a Spearman correlation analysis for the set of soil variables (28). The variables measured in the 0–20 cm layer was highly correlated ($r \geq 0.80$) with their respective measurements in the 20–40 cm layer (Fig. S1). Therefore, we used only the values from the 20–40 cm depth layer in the subsequent steps, as they were slightly better correlated with species richness. We then eliminated collinear variables and selected only P, K, Mg, pH, Al^{3+}, t and SOM.

We also performed Spearman correlation for the forest cover data from different monitoring years (1985, 2002, and 2017) and buffer distances from the plot’s centroid (500, 1000, and 2000 m). As the nine year-ratio combinations were highly correlated ($r \geq 0.65$), we selected forest cover from 2002 within a 2000 m buffer, as it was better correlated with species richness (Fig. S2). Among the climatic variables, we selected Temp, Precp, Precp-4, Precp-Dry, and CWD-1. Temp was well correlated with Less100 and all CWD indices ($r \geq -0.69$). The Precp was not well correlated with any other climatic variable ($r < \pm 0.60$), while Precp-4 was strongly correlated ($r = 0.90$) only with Precp-3 (Fig. S3). Precp-Dry was highly correlated ($r = -0.85$) with Precp-2. CWD-1 was correlated ($r \geq 0.74$) with all CWD values from previous years and from the year of the forest inventory. Although Temp and CWD-1 were strongly correlated ($r = -0.74$), we retained both variables due to their ecological relevance.

2.4.2. Community comparations

After calculating all diversity indices per plot, we conducted statistical comparisons between forest fragments to investigate differences in diversity among them. First, we tested the normality of the different diversity metrics using the Shapiro-Wilk test. Then, we performed one-way ANOVA with Tukey’s test and compared the means of the functional diversity index RaoQ and the community-weighted means of the ecological groups (CWM-EG) and deciduousness (CWM-Dec), since these showed normal distribution ($p \geq 0.133$). For species richness (^{0}D), exponential Shannon-Weaver index (^{1}D), and functional richness (FRic), comparisons were conducted using the nonparametric Wilcoxon test, as the data did not meet normality assumptions ($p \leq 0.05$). Analyses were performed using R Studio software (RStudio Team, 2020).

2.4.3. *Taxonomic and functional diversity modeling*

We used linear mixed-effects models (LMM) to investigate the combined effects of anthropogenic and environmental variables on taxonomic and functional diversity in Atlantic Rainforest fragments in Southeastern Brazil. This approach was chosen because LMM is flexible in handling unbalanced variables. We built individual models for each group of fixed effect variables. The anthropogenic model included fragment size, edge distance, forest cover in a 2000 m buffer from the plot centroid in 2002, and plot basal area extrapolated to a hectare. Environmental models included: i) topographic variables (elevation, declivity percentage, aspect), ii) edaphic variables (pH, P, K, Mg^{2+}, Al^{3+}, effective cation exchange capacity, and organic matter from the 20–40 cm soil layer), and iii) climatic variables (mean annual temperature of the forest inventory year, precipitation in the inventory year and the four-year mean preceding it, the sum of precipitation in the three driest months of the inventory year, and the CWD from the previous year). Land use history and time since abandonment were used as random effect variables in all models, due to the unique historical and successional characteristics of each fragment. Models were built using the “lme4” package (Bates et al., 2015) in the R Studio environment (RStudio Team, 2020). We evaluated the confidence interval distribution graphically to determine the significance of the fixed variables in each model. Model fit performance was compared using the Akaike Information Criterion (AIC) (Burnham and Anderson, 2002), where models with lower AIC values were considered better fitting. The contribution of fixed and random variables was assessed using marginal and conditional R^2 values. Marginal R^2 corresponds to the variance explained only by fixed effects, while conditional R^2 represents the variance explained by both fixed and random effects (Nakagawa and Schielzeth, 2013).

3. Results

3.1. *Patterns of diversity and dissimilarity*

We recorded 11,333 stems, distributed across 78 families, 256 genera, and 673 species, of which 30 were identified only to the genus level, 12 to the family level, and 26 remained unidentified. The fragments SJO, CAL, and IP2 showed the highest values of taxonomic diversity and internal heterogeneity for these indices, but only the first two were statistically distinct from the other fragments (Table 2). In contrast, IP1 had the lowest values for these indices and for functional richness (FRic), differing statistically from the other fragments.

SAB exhibited the highest FRic and functional diversity (RaoQ), but showed the lowest community-weighted mean (CWM) for the ecological group (CWM-EG), differing significantly from the other fragments. The forest in MAC was also statistically higher for RaoQ and CWM-EG, but together with GUA, it was statistically lower for the community-weighted mean of deciduousness (CWM-Dec). Notably, the SJO fragment, which was statistically lower in RaoQ, showed the highest CWM-Dec, differing from all other fragments. The mean richness (^0D) was 32 species, and at the plot level ranged from 13 to 67 species, in the MAC and CAL fragments, respectively (Table S2). Only 9 species (1.34 %) occurred in all fragments, and 373 (55.42 %) were recorded in only one of them.

The fragments had a mean dissimilarity of 0.68, and their plots did not form clear clusters. However, the SJO fragment nested 10 of its 12 plots, while GAR and MSI grouped most of their plots in a mixed group (Fig. 2). SJO was the most floristically distinct fragment, being most similar to CAL (0.74) and most dissimilar to the others (≥0.80) (Table S3). IP2 was also highly dissimilar (≥0.77), being floristically similar (≤0.66) only to its neighboring IP1 and MAC. The strength and significance of the correlations between taxonomic and functional diversity indices varied (Table S4). Taxonomic diversity indices were highly correlated with each other ($r = 0.86$; $p < 0.001$). Except for the community-weighted mean for deciduousness (CWM-Dec) ($r = -0.68$; $p < 0.001$), Rao’s functional diversity (RaoQ) was weakly correlated with all other metrics ($r \leq \pm 0.43$), although this relationship was significant ($p < 0.001$) only with species richness (^0D) and functional richness (*FRic*).

3.2. *General anthropogenic and environmental effects*

The performance of the linear mixed effects models varied across groups of variables and diversity indices, explaining between 14.4 % and 55.5 % of the variability when including both fixed and random effects, and between 1.2 % and 32.8 % when including only fixed effects (Table S5). Overall, the interaction of the random variables LUH and abandonment time had a greater influence on the models than the fixed variables, especially for CWM-EG and CWM-Dec. The model including climatic variables showed the strongest fixed effects for half of the diversity indices, particularly for ^{0}D and ^{1}D , and was dominated by

Table 2

Means (± standard deviation) of taxonomic (richness – ^{0}D and exponential of the Shannon-Weaver index – ^{1}D) and functional (functional richness – FRic, Rao’s functional diversity – RaoQ, and the community-weighted mean for the ecological group – CWM-EG and deciduousness – CWM-Dec traits) diversity, in different Atlantic Rainforest fragments, Minas Gerais, Brazil.

Fragment	Diversity Index											
	Taxonomic				Functional							
	$^0\boldsymbol{D}$		$^1\boldsymbol{D}$		*FRic*		*RaoQ*		*CWM-EG*		*CWM-Dec*	
CAL	40.47 ± 10.11	a	29.37 ± 8.51	a	37.66 ± 20.66	abc	4.80 ± 1.05	bc	1.64 ± 0.19	bc	2.30 ± 0.16	bc
GAR	30.10 ± 4.93	abc	21.47 ± 3.09	abc	35.45 ± 5.95	abc	5.49 ± 0.79	ab	1.81 ± 0.22	abc	2.13 ± 0.19	cde
GUA	31.47 ± 6.26	abc	20.29 ± 5.59	bc	37.72 ± 15.41	ab	5.57 ± 0.80	ab	1.69 ± 0.33	bc	2.02 ± 0.30	e
IP1	24.12 ± 5.68	c	15.94 ± 4.19	c	19.66 ± 10.94	c	5.47 ± 0.83	ab	1.66 ± 0.23	bc	2.19 ± 0.16	cde
IP2	34.50 ± 11.48	abc	26.73 ± 9.88	abc	42.23 ± 22.36	abc	4.73 ± 0.38	bc	1.97 ± 0.19	ab	2.32 ± 0.19	abcd
MAC	26.61 ± 7.32	bc	20.42 ± 7.04	bc	42.16 ± 19.71	ab	6.01 ± 0.78	a	2.02 ± 0.23	a	2.02 ± 0.22	e
MSI	31.20 ± 5.11	ab	20.02 ± 5.35	bc	30.06 ± 14.16	bc	4.85 ± 0.64	bc	1.77 ± 0.09	bc	2.45 ± 0.13	ab
SAB	31.80 ± 4.46	ab	23.27 ± 4.35	ab	51.10 ± 18.88	a	6.13 ± 1.08	a	1.61 ± 0.17	c	2.03 ± 0.21	de
SJO	43.08 ± 13.62	a	33.98 ± 11.96	a	33.37 ± 16.61	abc	3.88 ± 1.02	c	1.79 ± 0.20	abc	2.56 ± 0.15	a
Total	32.00 ± 9.35		22.67 ± 8.30		36.34 ± 18.24		5.31 ± 1.06		1.77 ± 0.25		2.20 ± 0.27	

stNotes: Different letters in the same column indicate means significantly different ($p < 0{,}05$) for the Wilconxon (^0D – *FRic*) and Tukey (*RaoQ* – *CWM-Dec*).

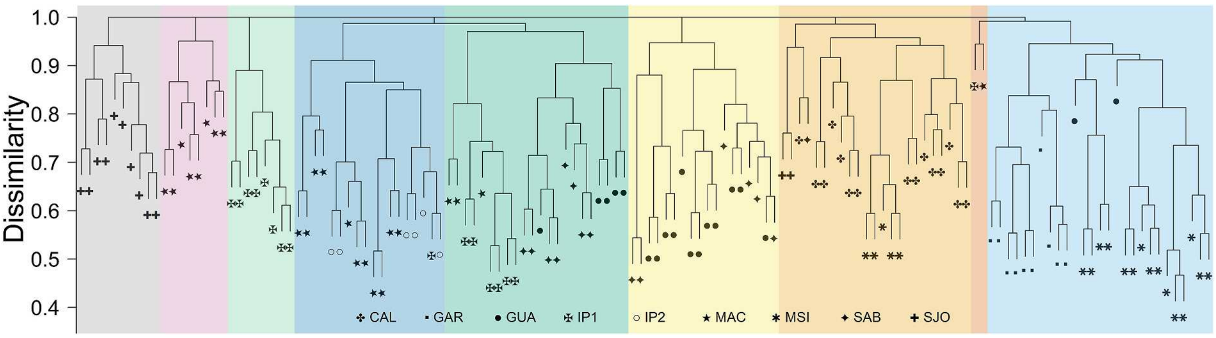

Fig. 2. Floristic clustering dendrogram using the Bray-Curtis dissimilarity index for nine Atlantic Forest fragments, Minas Gerais, Brazil.

temperature (Temp) and precipitation (Precp) from the inventory year (Fig. 3). The anthropogenic model showed stronger fixed effects for the remaining indices, especially for FRic. In this model, forest size and basal area were the most important variables, although random effects were also significant in explaining the variance of RaoQ and CWM-Dec. Despite their weak fixed effects and dominance of random effects, the edaphic models showed better fit, as indicated by lower AIC values.

3.3. Taxonomic diversity

The behavior of the variables and models was similar between ^{0}D and ^{1}D, but the explanatory power of the fixed variables was greater for ^{0}D in all models (Fig. 3; Table S5). The highest marginal R^2 was observed for the climatic models for both ^{0}D (0.328) and ^{1}D (0.266). These two diversity indices were positively affected by Precp and negatively by Temp, while Precp-Dry negatively affected only ^{0}D. The other models were minimally influenced by fixed variables (marginal $R^2 < 0.15$). The fixed effect of the anthropogenic model was significantly affected by the positive relationship of BA with both indices. The same occurred for elevation in the topographic model. Although much smaller than the random effect, the fixed effect in the edaphic model was dominated by pH and Al^{3+} for both ^{0}D and ^{1}D.

3.4. Functional diversity

Following the pattern of taxonomic diversity indices, CWM-EG

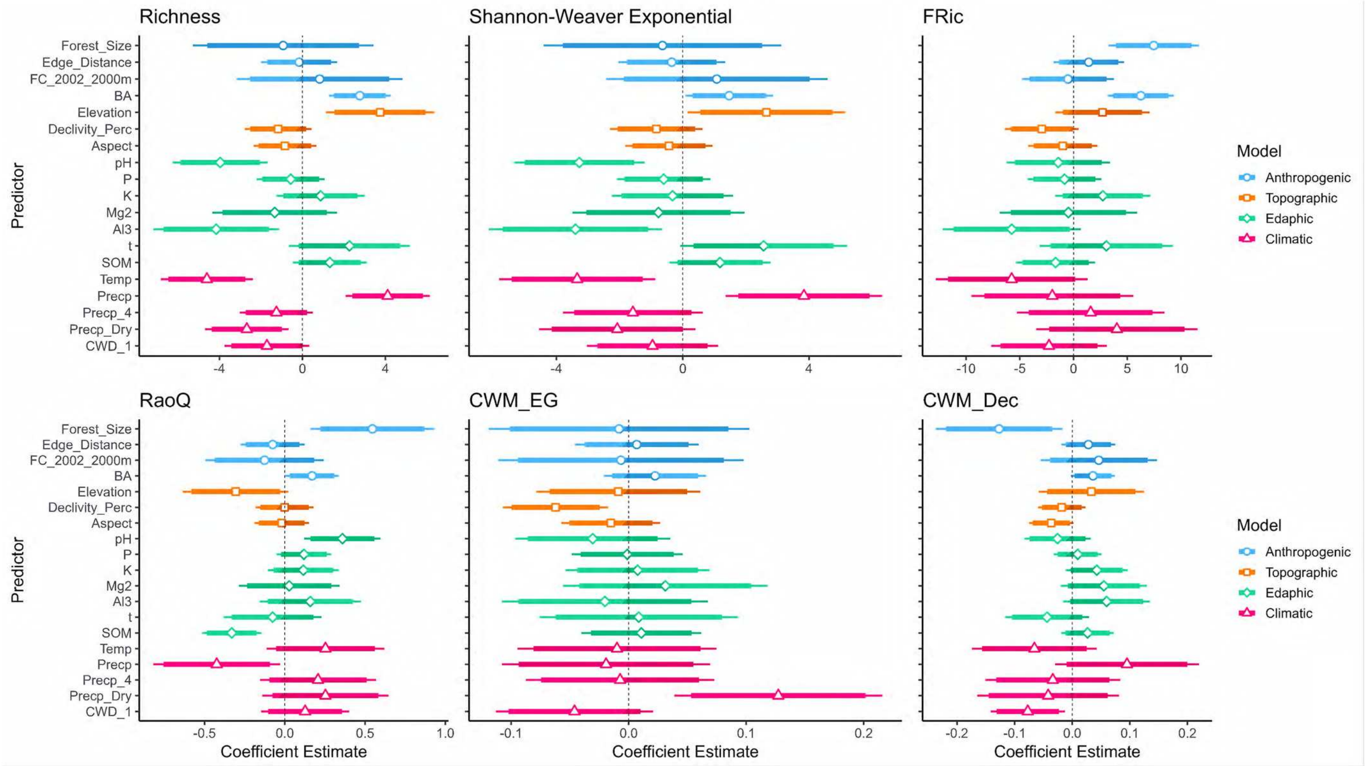

Fig. 3. Effects of anthropogenic and environmental variables on the taxonomic (richness and exponential of the Shannon-Weaver Index) and functional indices (functional richness – FRic, Rao's functional diversity – RaoQ, and the community-weighted mean for the ecological group – CWM-EG and Deciduousness – CWM-Dec traits) diversity in nine Atlantic Forest fragments, Minas Gerais, Brazil. Standardized coefficients with 95 % confidence intervals are indicated. The vertical line in each graph indicates zero. Variables containing zero within the confidence intervals are considered insignificant. FC_2002_2000m – forest cover in a 2000 m buffer from the plot centroid in 2002; BA – basal area of the plot extrapolated to hectare; Precp – total precipitation in the year of the forest inventory; Precp_4 – Mean precipitation in the four years preceding the inventory; Precp_Dry – precipitation of three driest months of the inventory year; CWD_1 – climatic water deficit one year before the census; and soil variables in the 20–40 cm layer: pH; cations (P, K and Mg^{2+}); exchangeable acidity (Al^{3+}); effective cation exchange capacity (t) and organic matter (SOM).

showed a stronger fixed effect from climatic variables, dominated by a significantly positive relationship with Precp-Dry (Fig. 3). FRic, RaoQ, and CWM-Dec showed stronger fixed effects from anthropogenic variables, but random effects were responsible for most of the variance explained in RaoQ. Overall, the models did not fit well for explaining FRic, as they showed low total R^2 and high AIC values (Table S5). Among anthropogenic variables, BA positively influenced both FRic and RaoQ, while forest size was also positively related to these indices but negatively to CWM-Dec. Declivity percentage was significant only for CWM-EG, negatively affecting this index. In the climatic model, Precp and CWD-1 negatively influenced RaoQ and CWM-Dec, respectively. The edaphic model had significant variables only for RaoQ, with pH showing a positive effect and SOM a negative one.

4. Discussion

Many studies have analyzed the effect of anthropogenic and environmental variables on species diversity, but most have focused in taxonomic diversity indices, such as species richness and the Shannon-Weaver index (Bentsi-Enchill et al., 2022). Aiming to explore the existing gaps in the drivers of community assembly, we present a joint analysis of how different variables from these groups of factors interact with taxonomic and functional diversity, based on 136 Atlantic Forest plots in Southeastern Brazil. The nine communities we evaluated were dissimilar, with highly variable species compositions and diversity indices (Table 2, S2, S3; Fig. 2). We found that climatic and anthropogenic variables had significant fixed effects, but the random effect of land-use history (LUH) and abandonment time stood out in explaining the variability of the different diversity indices (Figs. 3–5; Table S5). Our findings contribute to elucidating the environmental filtering mechanisms driven by anthropogenic and environmental factors in the assembly of tree-communities in the Atlantic Rainforest. This contribution is especially important for the conservation of hyperdiverse tropical forests, as tree diversity is fundamental for their ecological stability and buffering of extreme climatic events (Johnson et al., 2012; Gao et al., 2021; Ouyang et al., 2021; Schnabel et al., 2025). Furthermore, diverse and adequately managed tropical forests are capable of maintaining their patterns of functional diversity and primary productivity through short-term rotation of functional traits (Carreño-Rocabado et al., 2012).

4.1. Patterns of diversity and dissimilarity

Spatially isolated but environmentally similar sites tend to host taxonomically distinct but phylogenetically and functionally convergent communities (de-Paula et al., 2020). Our findings partially contradicted this statement, as we observed that the nine forest fragments we evaluated were taxonomically dissimilar (Fig. 2; Table S3), in agreement with the first aspect. However, we also observed contrasting patterns of functional diversity between some fragments, especially for IP1 and SJO, which showed opposing values for these indices (Table 2), suggesting that the functional breadth across fragments varied. This difference is possibly related to land-use histories, as IP1 underwent deforestation, while SJO is a steeper and less accessible area that was subjected to less impactful selective logging interventions. Furthermore, the difference in species composition results from distinct regional and local environmental conditions, which filter taxonomic and functional composition, leading to fragments with many exclusive species and few shared among themselves (Baynes et al., 2016). Complementarily, environmental heterogeneity and land-use history of each fragment also explain the divergence among diversity indices, as they drive different successional trajectories, which tend to result in mature forests with higher taxonomic diversity but greater functional homogeneity (Pinho et al., 2018).

The correlation between species richness (^{0}D) and the exponential of

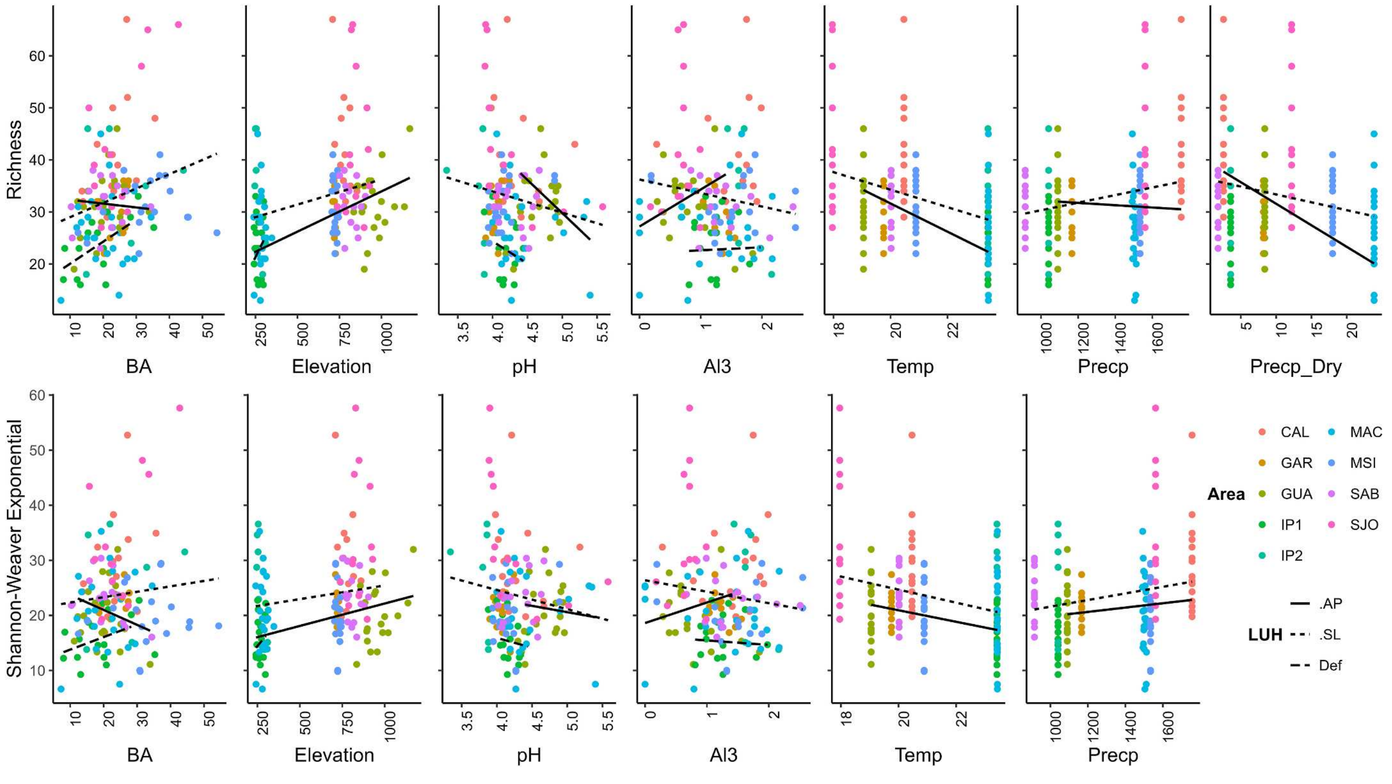

Fig. 4. Bivariate relations of taxonomic diversity indices (richness and exponential of the Shannon-Weaver index) with basal area of the plot extrapolated to hectare (BA), elevation, total precipitation in the year of the forest inventory (Precp), pH, exchangeable acidity (Al^{3+}) and precipitation of three driest months of the inventory year (Precp_Dry) in nine Atlantic Forest fragments, Minas Gerais, Brazil. Lines are the different land use history: AP – Agricultural production; Def – Deforestation and SL – Selective logging.

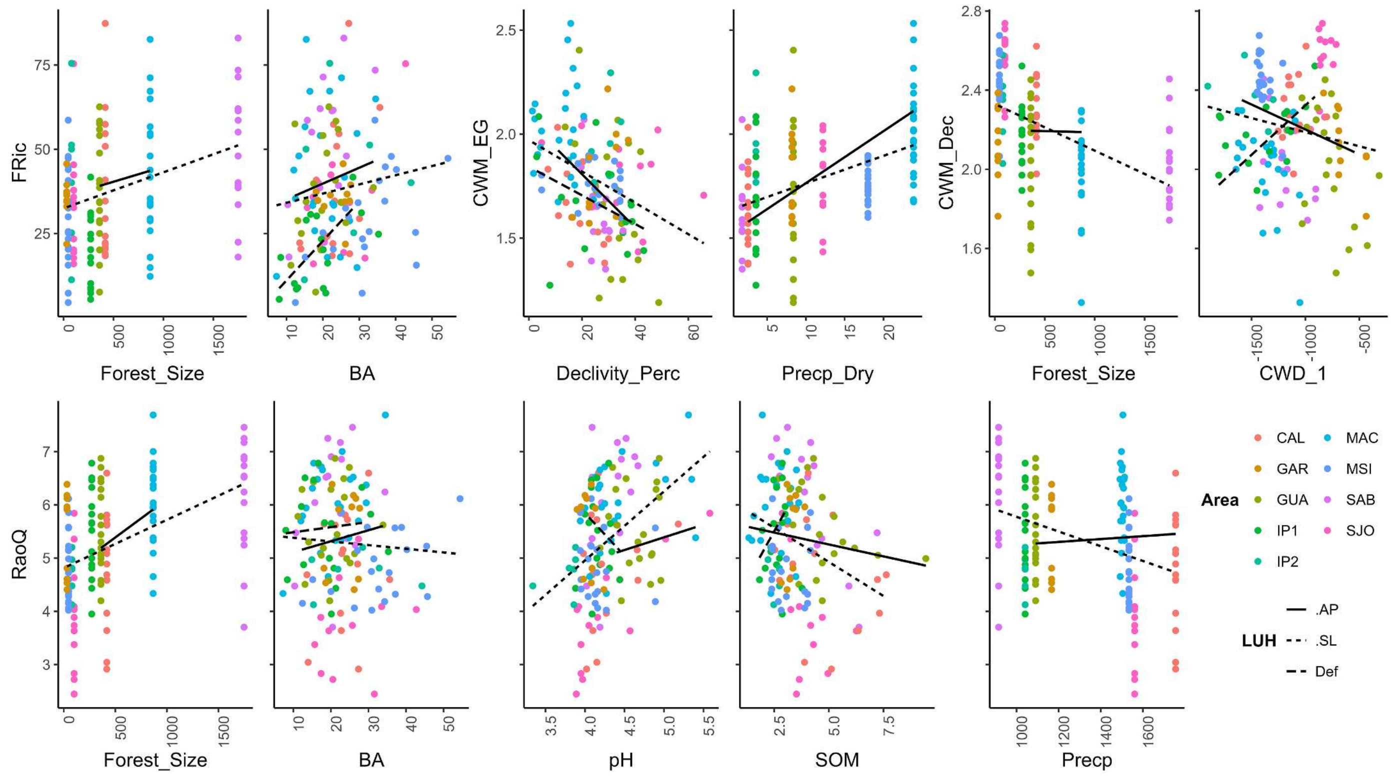

Fig. 5. Bivariate relations of functional diversity indices (functional richness – FRic, Rao's functional diversity – RaoQ, and the community-weighted mean for the ecological group – CWM-EG and Deciduousness – CWM-Dec traits) with forest size (ha), basal area of the plot extrapolated to hectare (BA), declivity percentage, precipitation of three driest months of the inventory year (Precp_Dry), climatic water deficit one year before the census (CWD-1), total precipitation in the year of the forest inventory (Precp), pH and organic matter (SOM) of soil in the 20–40 cm layer in nine Atlantic Forest fragments, Minas Gerais, Brazil. Lines are the different land use history: AP – Agricultural production; Def – Deforestation and SL – Selective logging. Apagar o eixo Y repetido em RaoQ, olhar o script do CAP 2.

the Shannon-Weaver index (^{1}D) was significantly strong (0.86; $p <$ 0.001), but correlations between these indices and others were weak or not significant (Table S4), revealing little dependence of the functional spectrum on species diversity. We observed that functional diversity indices, especially RaoQ, were not strongly correlated with the other diversity metrics (Table S4), particularly for SJO, which exhibited the highest taxonomic diversity but the lowest RaoQ (Table 2). This finding supports previous observations that functional diversity is more influenced by anthropogenic land-use factors and environmental variables such as nutrient availability, rather than species richness (Busch et al., 2018; Pinho et al., 2018).

We also observed that equivalent values of ^{0}D, at the plot level, represented highly variable (Table S2) and poorly correlated ($r \geq -0{,}32$; $p < 0{,}001$) (Table S4) functional richness (FRic). These results reflect a documented pattern in the relationship between taxonomic and functional diversity in tropical forests across different biomes in the Americas, Africa, and Asia (Cavanaugh et al., 2014). This occurs due to the higher abundance of functionally distinct and locally adapted species, which lead to variable weighting of functional diversity (Spasojevic et al., 2018; Madrigal-González et al., 2020), here measured through Rao's quadratic entropy (RaoQ). The relationship between species richness and functional richness is key to understanding the role of species numbers in ecosystem processes (Córdova-Tapia and Zambrano, 2015). Therefore, assessing functional diversity is essential to predict community responses to environmental changes, as functionally diverse species assemblages may be more stable over time, through fluctuating abundances, ensuring that these forests can persist even under anthropogenic pressures (Spasojevic et al., 2018).

4.2. *General anthropogenic and environmental effects*

Taxonomic and functional diversity indices were variably affected by the anthropogenic and environmental factors investigated. Overall, anthropogenic and climatic factors were more significant in explaining the variability in taxonomic and functional diversity (Fig. 3; Table S5). However, despite weaker fixed effects, the edaphic models showed the best fit, with higher conditional R^2 and lower AIC (Table S5), indicating a dependence of soil variables on LUH groupings and abandonment time (Teixeira et al., 2020; Bentsi-Enchill et al., 2022). We observed that different LUH exhibited variable influence on the diversity indices, with stronger effects in areas previously used for agriculture (Figs. 4 and 5). Although selective logging also contributes to forest structure (Baynes et al., 2016; Fulé et al., 2023), we found that its effects on diversity indices were generally milder when compared to agricultural cultivation or deforestation. Anthropogenic and environmental variables tend to similarly drive taxonomic diversity patterns within the same ecosystem (Idárraga-Piedrahíta et al., 2022). However, our results reaffirm the findings of a previous study, in which the gradient of anthropogenic pressure was strongly correlated with variations in tree species composition, but soil variables were also associated with these changes, with pH being the most important factor (Bentsi-Enchill et al., 2022). We also found a strong influence of the interaction between LUH and abandonment age on functional diversity in our study sites, demonstrating that functional patterns are highly dependent on forest succession processes, as observed to the Atlantic Rainforest in the Brazilian northeastern (Pinho et al., 2018).

Among anthropogenic fixed effects, basal area (BA) was significant for four of the six diversity indices tested (Figs. 3–5). This reinforces the role of forest age as a key factor in Atlantic Rainforest diversity, as BA is a proxy for forest age (Pinho et al., 2018; van-Breugel et al., 2019; Verly et al., 2023). Although each community may experience distinct successional trajectories (Sfair et al., 2016; Arroyo-Rodríguez et al., 2017; Teixeira et al., 2020), using BA as a determining factor of diversity can be an effective alternative when forest age is not available However, contrary to expectations, edge effects were not a significant predictor for any of the observed diversity indices. In most of our areas, no clear effect of edge distance on tree mortality was found, which was attributed to the presence of mature forest plots near the edge (Torres et al., 2023). Additionally, edge effects can penetrate up to 300 m into the forest interior (Chaplin-Kramer et al., 2015), and the majority of our plots (119) are within this range. Although there is evidence that fragmentation reduces functional diversity through species dominance (Baynes et al., 2016), there is a debate about the role of fragmentation in modulating diversity at different landscape scales (Gonçalves-Souza et al., 2025). In the Atlantic Rainforest, divergent human disturbances can result in edge plots with higher taxonomic and functional diversity than forest interior communities, showing that even fragmented landscapes have high conservation value (Sfair et al., 2016). These patterns support our findings of no clear relationship with edge effects, as anthropogenic pressures do not exert consistent influence over diversity dimensions, even when species composition differs between edge and interior plots.

The influence of climatic variables fixed effects in our models was mainly attributed to temperature and precipitation in the year of the forest inventory (Fig. 3). Precipitation is recognized as an important predictor of richness and diversity in the Atlantic Rainforest (Paz et al., 2021). This is because historically milder and more stable climatic conditions tend to result in more diverse communities, as climate acts as an environmental filter in the processes of speciation, selection, and species distribution (Santos et al., 2021; Gianasi et al., 2024), contributing to taxonomic, phylogenetic, and functional forest composition (Aguirre-Gutiérrez et al., 2020; Ying Li et al., 2022). In contrast, CWD-1 was significant only for the community-weighted mean of deciduousness (CWM-Dec) in our communities. Nevertheless, the effect of evapotranspiration and water deficit is recognized as a driver of demographic processes (Torres et al., 2023) and of different dimensions of diversity (Zhu et al., 2019; Gao et al., 2021), in which drier regions are less functionally diverse (Aguirre-Gutiérrez et al., 2022). This occurs because soil water availability and air humidity regulate tree leaf phenology, controlling foliage flushing and senescence, so that deciduous species adapt better to drier conditions, while evergreen species may be more abundant under more favorable conditions (Godlee et al., 2024; Rankine et al., 2024).

Our findings highlight the climatic effect on environmental filtering and community assembly in tree communities and point to potential changes in patterns and diversity losses in the Atlantic Rainforest, due to reduced richness and functional homogenization (Inague et al., 2021). This is because anthropogenic climate change has led to various effects on forest growth and diversity across the globe (Inague et al., 2021; Hammond et al., 2022; Salomón et al., 2022; Mirabel et al., 2023; Fortier et al., 2024). This negative effect is more pronounced in less diverse communities (Spasojevic et al., 2018; Madrigal-González et al., 2020), which tend to exhibit lower niche space occupancy (Ying Li et al., 2022) and consequently greater taxonomic and functional homogenization (Aguirre-Gutiérrez et al., 2020).

4.3. *Taxonomic diversity*

The anthropogenic, edaphic, and climatic variables that were significant showed similar effects for ^0D and ^1D (Figs. 3 and 4), suggesting that these factors influence both species establishment and the extent of their populations. Contrary to our initial expectation that fragment size and forest cover would affect ^0D, we found that BA, a proxy for forest age, was the only anthropogenic fixed effect variable with significant influence. The positive effect we observed for the taxonomic diversity indices is a convergent pattern for the Atlantic Rainforest (Teixeira et al., 2020). Therefore, our results are in line with previous studies, in which species diversity is inversely related to the anthropization gradient (Bentsi-Enchill et al., 2022). This pattern of influence appears to be more evident for the historical effect of land use, but not for current pressures, since we did not observe significant effects of edge distance on our taxonomic diversity values.

We expected that elevation would not be a significant variable for taxonomic diversity, but we observed a positive effect regardless of LUH, while declivity percentage and aspect were not relevant topographic variables (Zhu et al., 2019; Rahman et al., 2023). Taxonomic diversity may show different responses to elevation, with divergent patterns depending on vegetation type and the availability of resources and moisture along the altitudinal gradient (Zhu et al., 2019; Galván-Cisneros et al., 2023; Sanjeewani et al., 2024). Edaphic variables constitute a group of environmental factors strongly related to ecological filtering, acting on the differentiation of floristic composition in plant communities (van-Breugel et al., 2019; Rahman et al., 2023). However, more mature forests vegetation tend to exhibit weaker responses to soil fertility conditions (van-Breugel et al., 2019). In the Atlantic Rainforest, soils generally take about 20 years of succession to approximate primary forest standards for many variables (Teixeira et al., 2020). This pattern would explain the significant effects observed only for acidity-related variables (pH and Al^{3+}) in our areas, which have undergone at least three decades of abandonment.

As expected, climatic variables were significant for taxonomic diversity, since both temperature and precipitation showed divergent yet similarly related effects for ^{0}D and ^{1}D (Figs. 3 and 4). We found that cooler mean temperatures were associated with higher taxonomic diversity values. Long periods of low temperatures tend to reduce ^{0}D, as trees tend to lower their physiological activities, which reduces their growth or their chances of establishment (Liu et al., 2023; Li et al., 2024; Chen et al., 2025). Therefore, there is a temperature range in which its relationship with plant diversity is positive (Sanjeewani et al., 2024), but high temperatures may also reduce tree diversity in tropical ecosystems (Fortier et al., 2024). Thus, even though our fragments show higher ^{0}D and ^{1}D values in cooler areas, sharp temperature changes may result in reduced taxonomic diversity through the dominance of more adaptive species groups (Tian et al., 2023; Sanjeewani et al., 2024). Regarding precipitation, our results aligned with patterns observed in other forested areas, as it is a general trend that higher rainfall increases the availability and uptake of resources for the establishment of different species and functional profiles (Ying Li et al., 2022; Sanjeewani et al., 2024). Unexpectedly, precipitation of three driest months of the inventory year (Precp-Dry) was significant only for ^{0}D, showing a negative relationship, suggesting that the intensity of the dry season restricts the establishment of certain species. A possible explanation is that areas with more intense dry periods were colonized by a greater number of drought-adapted species (Bunker and Carson, 2005).

In our study, we addressed the effects individually, but in reality, all these factors act interactively in community assembly (van-Breugel et al., 2019), with elevation playing an aggregate role among the other effects (Galván-Cisneros et al., 2023; Silva et al., 2023). A previous study found that BA has a positive effect on taxonomic diversity regardless of the topographic position of the community (van-Breugel et al., 2019), indicating that ecological succession is an important predictor that is not constrained by location along this gradient, which was corroborated by our results. On the other hand, that study found that edaphic variables may have contrasting effects at different topographic positions. This was also supported by our observations, as pH and Al^{3+}, which were significant for ^{0}D and ^{1}D, showed high dispersion (Fig. 4) and were not correlated with elevation ($r \leq \pm 0.16$).

4.4. Functional diversity

Our results demonstrate that species composition and abundance are highly variable at both regional and local scales (Table 2; Fig. 2), confirming that neutral theories regarding environmental effects on community assembly are of limited applicability to tropical forests (Ding et al., 2019; Pos et al., 2019). In this context, functional diversity results from a sophisticated mechanism of species adaptation to environmental variation, which regulates the spatiotemporal dynamics of community assembly (Pinho et al., 2018; Spasojevic et al., 2018; Madrigal-González et al., 2020). More extreme environmental conditions tend to lead to functionally poorer communities (Gianasi et al., 2024), generally composed of a considerable share of species specialized in local environmental conditions (Idárraga-Piedrahíta et al., 2022). In contrast, communities experiencing more stable environmental conditions and greater resource availability tend to show higher population evenness among functional profiles (Ying Li et al., 2022).

We found that the interaction between LUH and the abandonment time of the areas explains much of the variability in FRic and RaoQ for most models (Table S5). The observed influence of abandonment time aligns with the idea that successional recovery trajectories impose temporal environmental filters, progressively favoring traits associated with competitive dominance, shade tolerance, and conservative strategies (Chazdon, 2008; Rozendaal et al., 2019). Communities exposed to chronic anthropogenic pressures experience losses in their capacity to support the establishment of certain species, thereby reducing long-term functional richness and diversity (Ribeiro et al., 2019). The time since anthropogenic activities ceased also influences functional composition patterns in the Atlantic Rainforest (Pinho et al., 2018), supporting our findings that land use and abandonment time affect the forest functional diversities. The anthropogenic factors, fragment size and BA were significant in explaining the variability in FRic and RaoQ (Figs. 3 and 5). This is likely because larger and older forests (with higher BA) offer more favorable environmental conditions than smaller and younger forests, thus promoting functional evenness (de-Paula et al., 2020; Ying Li et al., 2022), as species from more advanced successional stages tend to be less responsive to environmental variables (van-Breugel et al., 2019). The same concept may be applied to our finding that RaoQ was lower at higher SOM levels, as more mature forests exhibit higher levels of organic matter (or carbon) in the soil (Li et al., 2013). These trait shifts reflect successional niche replacement and contribute to the functional signature of secondary forests (Chazdon, 2008; Rozendaal et al., 2019).

We also found significant effects of soil pH, which was positively correlated with RaoQ, suggesting that soil acidity increases the dominance of certain functional profiles in the tree community, thereby increasing functional diversity. Mean RaoQ values were lower in areas previously used for agriculture (Fig. 5), suggesting that more intense anthropogenic pressures may lead to higher pH values and to edaphic conditions less favorable for species colonization (Bentsi-Enchill et al., 2022). RaoQ was the most responsive functional diversity index to the set of variables we evaluated (Figs. 3 and 5). This index is recognized as a more appropriate measure, when compared to other functional diversity metrics, because it captures functional dissimilarity between species and the weight of each functional profile, when weighted by the abundance of species (Botta-Dukát, 2005). Therefore, identifying models and variables that influence this index is essential to understanding the functional patterns of a community.

The community-weighted mean for the ecological group (CWM-EG) and deciduousness (CWM-Dec) were also strongly affected by the interaction between LUH and abandonment time (Table S5). This is possibly related to the influence of anthropogenic pressure and abandonment time on the modulation of edaphic conditions and ecological succession, and consequently, on patterns of change in vegetative functional trait values in the Atlantic Rainforest (Pinho et al., 2018). Although soil variables did not have significant effects, the edaphic model for CWM-Dec showed the highest conditional R^2, reflecting these variables' dependence on LUH, particularly pH and nutrient availability (Pinho et al., 2018; Bentsi-Enchill et al., 2022). Among the fixed variables, declivity percentage and Precp-Dry were the only ones with significant influence on CWM-EG (Figs. 3 and 5). The influence of declivity on functional traits may be less evident, its effect is important for community differentiation (Rahman et al., 2023). A study conducted in inselberg areas of the Atlantic Rainforest found that less steep plots had a predominance of species with acquisitive leaf traits (de-Paula et al., 2020). We observed a similar pattern, as in our fragments, less steep plots showed a greater abundance of pioneer and early secondary plants (Fig. 5), which are acquisitive functional profiles related to fast growth and tend to decrease with successional advance (Ali et al., 2017; Teixeira et al., 2020).

Although a previous study in the same Atlantic Rainforest region found that elevation, temperature, and water availability were important predictors of vegetative functional traits, such as leaf and petiole size (Silva et al., 2023), we observed that only forest size and CWD-1 were significant fixed variables for CWM-Dec. The effect of CWD-1 differed among LUH classes, with greater dominance of evergreen species at higher water deficits in areas previously used for selective logging and agriculture, and surprisingly more evergreen communities at lower deficits in plots previously deforested (Fig. 5). There is a tendency for conservative strategies in vegetative functional traits (evergreen leaves) associated with lower water availability (Silva et al., 2023). These unexpected patterns likely result from divergent adaptive strategies among species, which exhibit different degrees of deciduousness along a water availability gradient (Idárraga-Piedrahíta et al., 2022; Rankine et al., 2024). In this context, fragment size also acts as a regulator of local moisture through microclimatic conditions (Schnabel et al., 2025; Steinparzer et al., 2025). The influence of these climatic variables is especially important in the context of conservation ecology, as projections show that more severe climate change could drastically increase functional homogenization and alter the community-level distribution of certain functional traits (Aguirre-Gutiérrez et al., 2020, 2022; Inague et al., 2021; Erdős et al., 2024).

We found that several environmental factors shape plant community filtering and assembly. Therefore, the formation of communities filtered by specific combinations of environmental factors may reduce functional diversity among species, as only those with adaptive characteristics to particular environments are able to establish (Vahdati et al., 2017; Erdős et al., 2024). We expected that increased elevation would act as a factor intensifying these filters for community functional assembly, but our results did not support the initial expectation that higher-altitude environments would host functionally more homogeneous communities (Ding et al., 2019; Oseguera-Olalde et al., 2022). It is possible that our plots are within an optimal altitudinal range, as the effect of elevation gradients in tropical forests is discontinuous, with a gradual reduction in diversity above certain elevation thresholds, as energy and moisture availability peak at mid-elevations (Zhu et al., 2019).

Tree communities with higher functional diversity are more resilient to environmental stress, as niche complementarity among species results in greater occupation of the possible functional space (Abrams, 1995; Fridley, 2001; Comita et al., 2010; Johnson et al., 2012; Madrigal-González et al., 2020), ensuring greater resource availability. Richer and more diverse forests tend to exhibit higher primary productivity, and consequently a significant stock of woody biomass (Madrigal-González et al., 2020; Bayat et al., 2021; Gao et al., 2021). This pattern is even more pronounced in functionally diverse communities, where tree species also promote greater carbon and nitrogen allocation in the soil (X. Chen et al., 2024). Therefore, maintaining diversity patterns in tropical forests ensures the perpetuation of these ecosystems as global carbon sinks (Ruiz-Jaen and Potvin, 2010; Madrigal-González et al., 2020). This highlights the role of hyperdiverse forests in mitigating the effects of anthropogenic climate change (Ruiz-Jaen and Potvin, 2010;

Aguirre-Gutiérrez et al., 2020; Schnabel et al., 2025; Steinparzer et al., 2025).

Our findings reinforce the critical role of land-use legacy and environmental heterogeneity in shaping multiple dimensions of tree diversity. By integrating functional and taxonomic perspectives, this study highlights the need for nuanced conservation strategies that account for historical trajectories and local-scale environmental filters. Therefore, our results can inform the creation or improvement of public policies and management practices for conservation, offering insights to the establishment of protected areas that should prioritize forests least affected by anthropogenic activities or secondary forests originating from less intense LUH, which harbor greater diversity. Future studies should expand to include phylogenetic dimensions and explore how projected climate change may interact with land-use legacies to further reshape community composition in tropical forests.

CRediT authorship contribution statement

Otávio Miranda Verly: Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Pedro Manuel Villa:** Methodology, Formal analysis, Data curation. **Marcelo Vitor Gualberto Santos Chaves:** Formal analysis, Data curation. **Samuel José Silva Soares da-Rocha:** Formal analysis, Data curation. **Luiz Claudio Medeiros Cabral-da-Silva:** Data curation. **Klisman Oliveira:** Writing – review & editing. **Maria Paula Miranda Xavier Rufino:** Writing – review & editing. **Samuel Braz Vieira:** Data curation. **D'lano Figueiredo Teixeira Sathler:** Writing – review & editing, Data curation. **Carlos Moreira Miquelino Eleto Torres:** Writing – review & editing, Validation, Supervision, Resources, Project administration, Methodology, Funding acquisition, Conceptualization.

Declaration of generative AI and AI-assisted technologies in the writing process

Statement: During the preparation of this work the author(s) used ChatGPT (version GPT-4.5) in order to translate, review the text, search for synonyms and improve the fluidity of writing. After using this tool/service, the authors reviewed and edited the content as needed and took full responsibility for the content of the published article.

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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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2025.126830.

Data availability

Data will be made available on request.

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Supplementary Materials

Table S1. Fruit classification system developed for Atlantic Rainforest species and based in Brasil (2009) and Gonçalves & Lorenzi (2011).

Type/Subtype	Dehiscence	Description
Dry fruit		
Achene		Originating from a unilocular ovary, seed attached to the pericarp by only one point.
Typical achene	I	There is no special modification in the pericarp.
Cypsela	I	Originating from the inferior ovary. Accessory structures originating from the perianth extend in the apex of the pericarp.
Capsule		It has two or more carpels and presents several subtypes, which differ by the type of opening.
Ceraceous	D	Incomplete opening due to separation of the pericarp layers, like siliques.
Circumcised/Pyxid	D/S	Transverse opening at the apical end, dividing into lid/operculum and remainder of the fruit/amphora.
Schizocarp	D/I	Originating from a syncarpous ovary, whose carpels separate upon maturation, carpels as mericarps.
Fissuricidal	D	Opening into several lateral longitudinal slits.
Loculicidal	D	Opening in the wall of the septum, along the dorsal bundle, exposing the seeds.
Poricidal	D	Pore opening at the apex of the fruit.
Samaroid/Pseudosamara	I	Originating from two or more carpels, with wings on all carpels developed at fruiting.
Septicidal	D	Opening by the junction of the carpels, opening from the base to the apex or vice versa.
Septifragal	D	Opening breaking the welding of the carpels and placenta, maintaining the internal part of the septa.
Silique	D	Opening from the base to the apex, maintaining the false central septum, where the seeds are attached.
Velatidium	D	Ovary inferior, pericarp with loculicidal or septifragal dehiscence. Hypanthium remains intact.
Caryopsis		Pericarp completely united to the seed coat of the single seed on its entire surface..
Typical (Naked) caryopsis	I	Typical pattern, without specialized structures for dispersal
With modifications	I	With modifications such as glumes (smooth or spiny), awns or other adherent involucres
Craspedius	I	Originating from a superior ovary and unicarpellar. At maturity, it segments transversely, and the articles are dispersed, leaving behind a framework formed by the fusion of the carpels and their dorsal veins.
Cryptoloment	D	The outer layer of the pericarp separates into two valves; the inner, indehiscent layer is segmented into carcerules.
Cryptosamara	I	The outer layer of the pericarp separates into two valves; the inner one is indehiscent, forming a wing.
Follicle	D	Monocarpellar gynoecium, superior, lateral or apical opening by the welding of the carpals.
Legume	D/I	Monocarpellar gynoecium, superior, lateral opening by the welding of the carpals and along the rib.
Typical legume	D	Monocarpellar gynoecium, superior, lateral opening by the welding of the carpals and along the rib.
Bacoid	I	Indehiscent, fleshy fruits, with a slightly to very thickened pericarp and an endocarp consisting only of the inner epidermis, undifferentiated, but not woody, scleroderm or coriaceous; usually with many seeds, but few and even one is not rare.

Drupaceous	I/D	Indehiscent or late dehiscent, seeds surrounded by a coriaceous or hardened endocarp, and generally concreted with the tegument. Like a drupe.
Nucoid/Chamber	I/S	Indehiscent or late dehiscent pericarp, originating from a single carpel. Free seeds surrounded by juicy pulp.
Samaroid	I	Fruit originating from an ovary with one or more carpels with a wing for each developed carpel.
Loment	I?	Originating from the superior ovary, monocarpellar, transverse segmentation into articles.
Samara		Monocarpellar/pseudomonocarpellar ovary, with one wing
Amphinuclear	I	Outer layer of the pericarp fused. Does not separate into two valves, with circular wing.
Paranuclear	I	Outer layer of the pericarp fused. Does not separate into two valves, with a wing on only one side.
Fleshy fruit		
Aggregate		Development of ovaries from a single flower, which grow as a single structure (as distinct from a compound fruit).
Apocarpous	I/D	The ovaries develop independently but are connected at a central point.
Hipanthium/ Rose hip	I	Small achenes within a fleshy receptacle.
Conocarp/Polyachene	I	Formed by the congrowth peduncles.
Syncarpic	I	Fusion of apocarpous ovaries.
Berry/Bacoid		Fleshy fruit, with succulent mesocarp and endocarp, generally with several seeds.
Typical berry	I	Fleshy pericarp, relatively uniform, large number of seeds. Variable shape and texture.
Acrossarcidium/Acrossarc	I	Without hardened endocarp, surrounded by fleshy exocarp
Amphissarcide	I	Fleshy pericarp, non-individualized locules, large seeds, juicy and placental pulp.
Balustrade	I	Fleshy pericarp, fleshy mesocarp and endocarp, multiple seeds with sarcotesta.
Campomanesioid	I	Fleshy pericarp, numerous, narrow locules, with juicy pulp
Hesperidium	I	Multiseeded, originating from a superior ovary, several carpels not completely fused.
Melanidium	I	Multiseeded, superior ovary, fleshy pericarp, parietal placentation, seeds with arils.
Pepo	I/D	Multiseeded, inferior ovary, seeds embedded in juicy pulp
Pome	I	Originating from an inferior ovary, with perianth surrounded by the fleshy receptacle.
Solanide	I	Fleshy pericarp, two or more locules and with the central cavity filled with seeds.
Bacaceous	I	It has a thin pericarp, septate or not locule, with few seeds and no pulp.
Baccidium	I	Thin and even membranous epicarp. The seeds are surrounded by pulp, without being in locules.
Drupaceous	I	Seeds enclosed in a leathery or hardened endocarp, the pyrene.
Artocarpus	I	Structure formed by the entire gynoecium or part of it, which surrounds the fruit, usually drupaceous.
Betulidium	I	Dry. Inferior ovary, with alar expansions originating from the hypanthium.
Drupe	I	Monospermic.
Philotrimide	D	Epicarp opens into several valves, mesocarp fleshy or woody.
Nut	I/D	Furnished with a structure originating from the receptacle, forming a dome.
True nut	I/D	Furnished only by the receptacle, forming a dome.
Acorn	I	Dry pericarp with fleshy/attractive part corresponding to the peduncle or hypanthium.

Nucule/Atrocarpaceous	I	Accessory structure, usually the persistent calyx, for dispersal.
Nuculanic	I	Two seeds or more.
Tryma	D	Dehiscent variation of a drupaceous fruit, characterized by the opening of the epicarp.
Compound/Multiple		Fruits (fruticles) are united at maturity to form a single structure, the pseudofruit.
Drupaceous	I	Succulent structure, like a set of small drupes (called drupeoles) united or arranged together.
Siconium	I	Concave and fleshy receptacle, formed by numerous fruits in its internal part.
Sorosis	I	Several flowers/fruits grouped into a single structure, ripening simultaneously.

Note: D – dehiscent; I – indehiscent; S – semidehiscent.

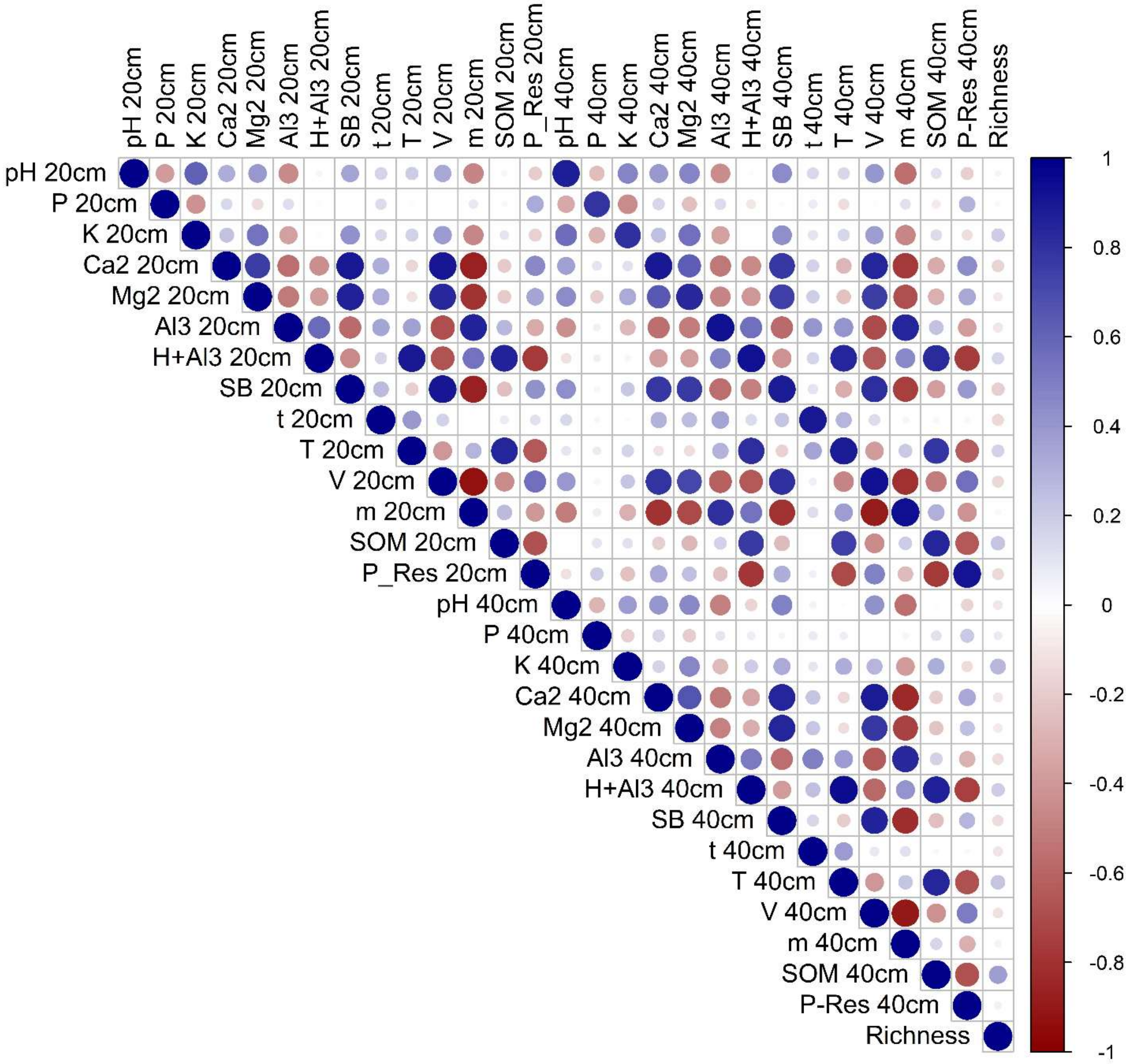

Figure S1. Spearman correlations between soil variables at different depths (0–20 cm and 20–40 cm); and richness. Dark blue circles indicate positive correlations, and dark red circles indicate negative correlations. The size of the circle indicates the strength of the correlation. For the abbreviation of soil variables see Material and Methods.

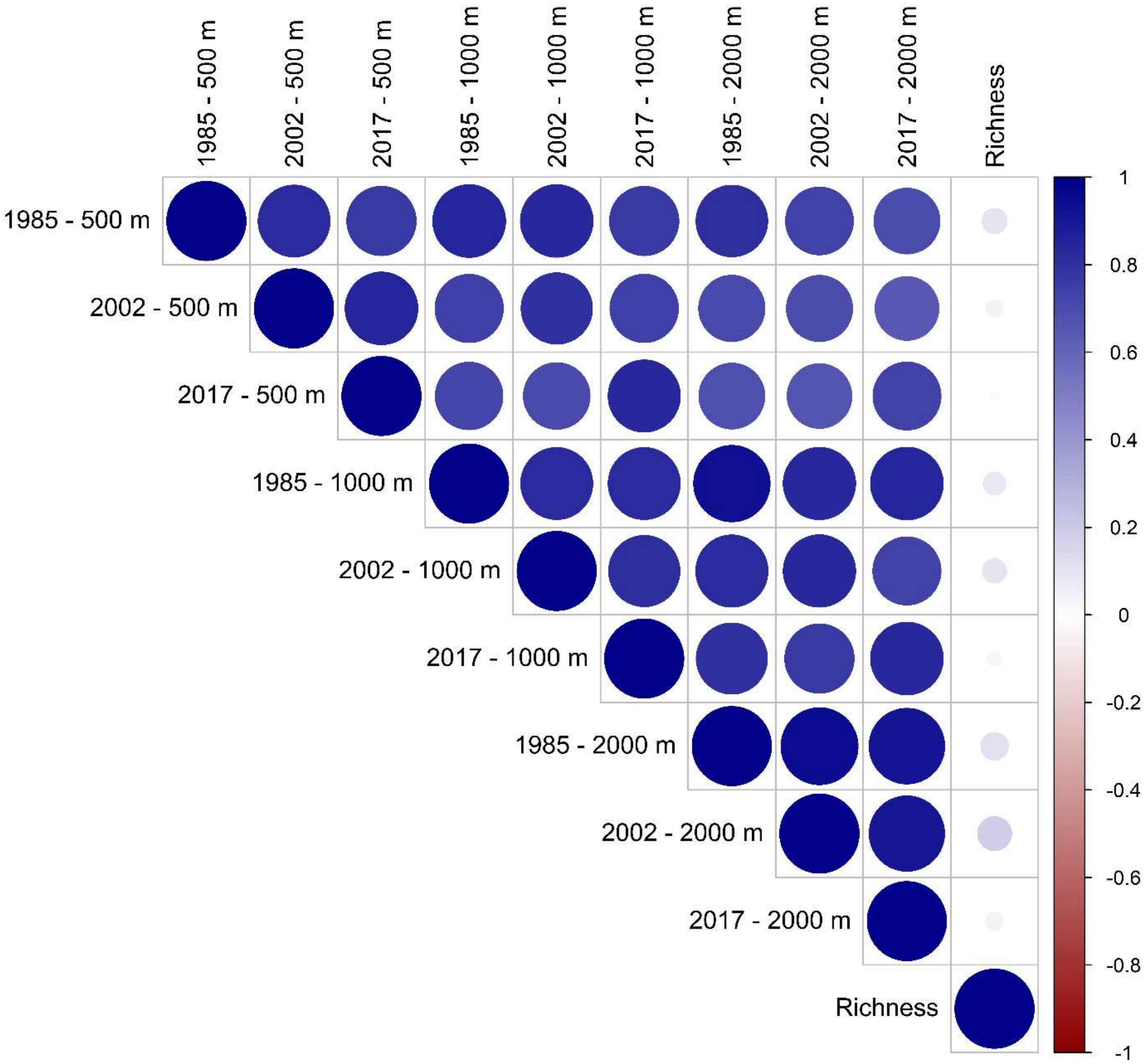

Figure S2. Spearman correlations between forest cover area using radii of 500, 1000, and 2000 m, for the year 1985, 2002 and 2017; and richness. Dark blue circles indicate positive correlations, and dark red circles indicate negative correlations. The size of the circle indicates the strength of the correlation.

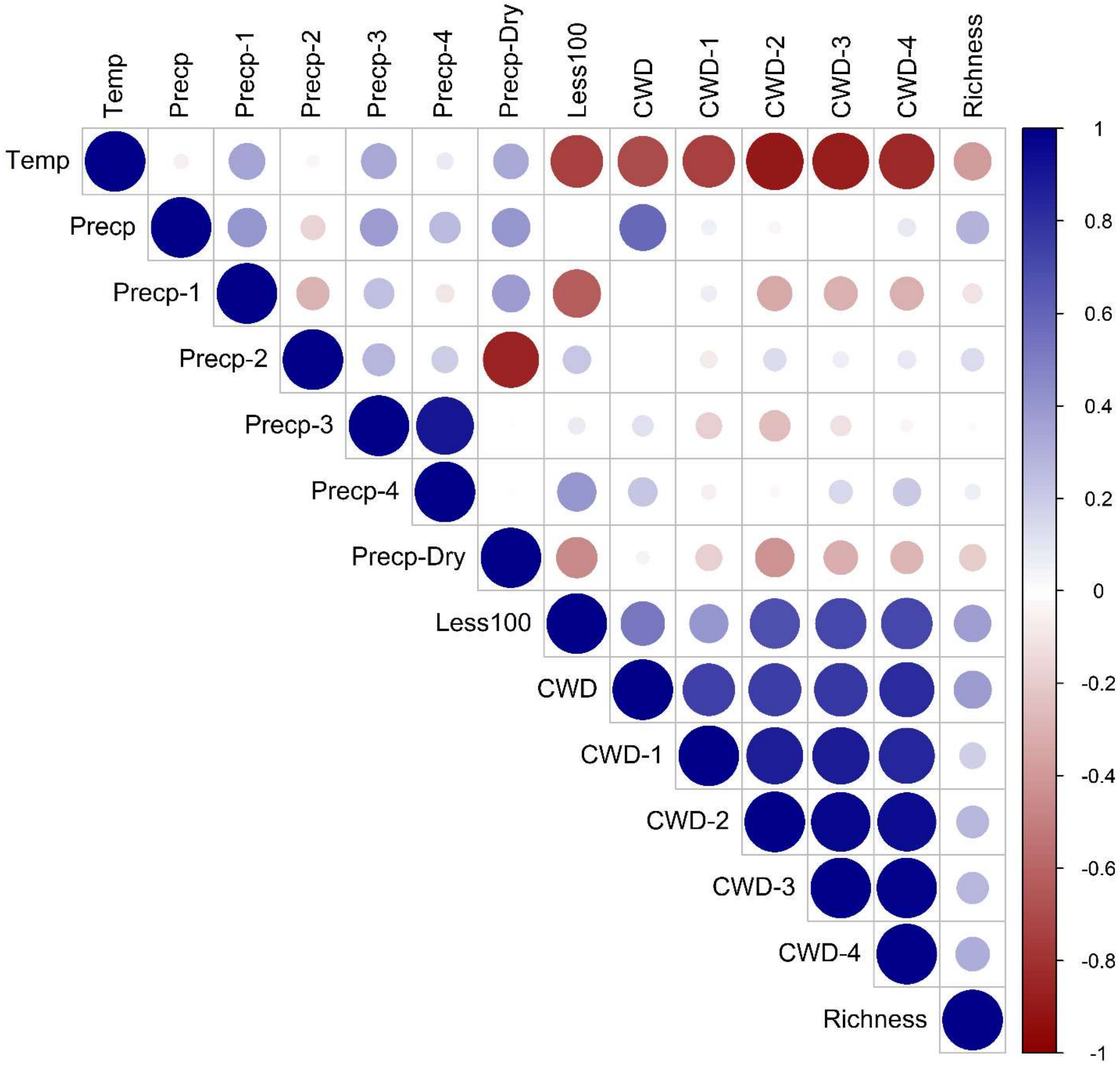

Figure S3. Spearman correlations between climatic variables of temperature, precipitation, and climatic water deficit (CWD); and richness. Blue circles indicate positive correlations, and red circles indicate negative correlations. The size of the circle indicates the strength of the correlation. For the abbreviation of soil variables see Material and Methods.

Table S2. Taxonomic (richness - ^{0}D and exponential of the Shannon-Weaver Index - ^{1}D) and functional (functional richness - FRic, Rao's functional diversity - RaoQ, and the community weighted mean for the ecological group – CWM-EG and Deciduousness – CWM-Dec traits) diversity values per plot, in different Atlantic Rainforest fragments, Minas Gerais, Brazil.

Local	**Plot**	**Diversity Index**					
		Taxonomic		**Functional**			
		^{0}D	**^{1}D**	**FRic**	**RaoQ**	**CWM-EG**	**CWM-Dec**
CAL	1	50	38,30	57,40	5,08	1,61	2,22
CAL	2	34	24,30	22,23	4,69	1,67	2,36
CAL	3	35	26,28	18,42	3,96	1,83	2,43
CAL	4	52	33,78	19,01	2,92	1,60	2,62
CAL	5	41	29,81	22,02	3,64	1,87	2,48
CAL	6	35	27,07	50,87	5,81	2,03	2,47
CAL	7	48	34,93	62,41	4,60	1,65	2,20
CAL	8	39	30,42	54,62	5,72	1,38	2,20
CAL	9	29	19,77	24,21	5,62	1,47	2,27
CAL	10	32	25,76	20,45	3,04	1,81	2,43
CAL	11	43	32,40	40,87	5,64	1,53	2,20
CAL	12	32	21,63	20,99	4,58	1,38	2,28
CAL	13	34	22,58	34,64	5,16	1,75	2,19
CAL	14	36	20,72	29,42	6,60	1,57	1,98
CAL	15	67	52,74	87,33	4,89	1,50	2,16
GAR	1	36	27,42	21,87	4,41	1,97	2,39
GAR	2	32	21,56	37,40	4,81	2,22	2,32
GAR	3	36	24,08	39,42	5,94	1,89	1,97
GAR	4	35	22,52	34,71	4,61	1,65	2,31
GAR	5	32	20,48	34,81	4,54	1,57	2,19
GAR	6	30	23,28	45,69	5,96	1,67	2,14
GAR	7	27	21,26	36,71	6,12	1,60	2,04
GAR	8	22	16,88	33,06	6,39	1,60	2,06
GAR	9	26	17,89	34,45	6,18	1,94	1,76
GAR	10	25	19,34	36,40	5,91	2,00	2,07
GUA	1	36	23,85	40,77	6,49	1,57	1,97
GUA	2	25	16,88	28,83	4,51	1,76	2,21
GUA	3	34	24,32	34,45	6,43	1,92	1,97
GUA	4	35	17,58	28,23	5,08	2,07	2,38
GUA	5	27	13,37	34,29	6,70	1,73	1,73
GUA	6	19	15,33	15,60	5,78	1,50	1,83
GUA	7	35	25,38	58,35	6,14	1,72	2,11
GUA	8	31	17,12	54,08	6,29	1,40	1,59
GUA	9	30	11,11	38,63	5,62	1,21	2,45
GUA	10	32	19,79	57,53	6,29	2,40	2,04
GUA	11	41	27,73	62,60	5,44	1,66	2,27
GUA	12	29	18,43	34,17	4,91	2,03	2,31
GUA	13	28	16,86	20,14	4,58	1,30	1,71
GUA	14	31	19,91	25,03	5,30	1,96	2,25
GUA	15	46	31,98	55,90	4,99	1,95	2,19
GUA	16	31	22,25	39,82	5,05	1,89	2,36
GUA	17	36	27,96	31,84	4,20	1,52	1,90
GUA	18	30	22,35	48,86	6,87	1,30	1,61
GUA	19	22	13,36	7,64	5,19	1,19	1,48
IP1	1	23	14,68	8,73	3,95	1,44	2,19
IP1	2	30	21,06	16,24	4,43	1,67	2,12
IP1	3	33	22,74	31,15	5,82	1,80	2,26
IP1	4	27	12,88	24,81	5,12	1,45	2,27
IP1	5	20	12,38	20,22	5,65	2,06	2,31
IP1	6	17	12,22	5,49	5,52	1,58	2,04
IP1	7	26	18,75	28,97	6,47	1,66	2,29
IP1	8	28	18,05	30,68	6,78	1,81	2,20

Local	Plot	Diversity Index					
		Taxonomic		Functional			
		^0D	^1D	FRic	RaoQ	CWM-EG	CWM-Dec
IP1	9	23	16,96	17,92	4,59	1,58	2,28
IP1	10	16	11,00	7,34	4,81	1,53	2,32
IP1	11	23	15,70	8,54	4,88	1,42	2,52
IP1	18	17	12,38	10,18	5,97	1,70	2,09
IP1	19	16	9,27	8,93	6,51	1,27	1,89
IP1	20	32	22,20	23,38	5,00	1,89	2,17
IP1	21	26	14,58	30,22	5,67	1,68	1,97
IP1	22	29	20,13	41,73	6,33	2,08	2,03
IP2	12	38	31,52	40,10	4,48	1,92	2,38
IP2	13	35	28,74	51,26	4,81	2,29	2,58
IP2	14	46	34,62	50,03	4,12	1,95	2,35
IP2	15	18	13,80	11,28	4,97	2,06	2,19
IP2	16	46	36,59	75,49	4,79	1,80	2,42
IP2	17	24	15,11	25,22	5,20	1,77	2,03
MAC	1	32	25,14	67,18	6,48	2,23	1,87
MAC	2	45	35,26	54,06	4,65	2,02	2,30
MAC	3	29	20,77	82,60	6,78	1,88	2,03
MAC	4	21	14,17	28,77	5,70	2,20	2,29
MAC	5	39	30,39	67,25	6,02	2,12	2,11
MAC	6	21	14,83	14,92	6,04	1,94	1,94
MAC	7	33	28,02	33,90	5,33	1,79	2,25
MAC	8	23	17,36	33,40	7,00	1,97	2,14
MAC	9	25	18,00	33,33	5,09	2,15	2,07
MAC	10	21	13,89	25,84	6,66	2,01	1,69
MAC	11	34	27,86	54,69	6,37	2,32	2,13
MAC	12	25	20,96	34,23	6,71	1,96	2,00
MAC	13	27	19,44	43,83	5,70	2,07	1,88
MAC	14	13	6,63	12,30	4,34	1,76	1,33
MAC	15	29	25,36	64,94	7,69	2,11	2,13
MAC	16	31	27,10	35,37	5,79	1,90	1,98
MAC	17	29	19,44	29,44	5,96	2,53	1,68
MAC	18	25	19,09	56,68	6,31	1,90	2,10
MAC	19	28	24,27	48,02	5,94	1,81	2,11
MAC	20	26	22,94	71,21	6,48	1,67	2,07
MAC	21	14	7,50	18,01	5,38	2,45	2,30
MAC	22	22	13,33	17,93	5,33	1,68	2,00
MAC	23	20	17,94	41,70	6,52	2,09	2,06
MSI	1a	34	21,55	43,97	5,16	1,62	2,43
MSI	1b	30	18,97	40,73	4,73	1,86	2,37
MSI	2a	41	29,44	44,80	4,32	1,75	2,44
MSI	2b	26	16,65	4,54	4,15	1,87	2,35
MSI	3a	27	21,34	20,35	5,86	1,60	2,44
MSI	3b	34	26,89	29,97	5,14	1,72	2,38
MSI	4a	30	23,29	25,27	4,37	1,68	2,52
MSI	4b	38	26,03	32,55	4,06	1,68	2,42
MSI	5a	24	18,98	18,01	4,60	1,74	2,39
MSI	5b	31	15,26	25,58	4,42	1,89	2,54
MSI	6a	30	17,95	21,05	4,99	1,88	2,60
MSI	6b	29	18,85	15,63	4,28	1,90	2,68
MSI	7a	29	17,79	25,33	5,22	1,88	2,58
MSI	7b	26	18,10	47,32	6,12	1,76	2,44
MSI	8a	28	9,83	23,42	4,02	1,71	2,59
MSI	8b	22	10,03	7,31	4,75	1,66	2,48
MSI	9a	37	16,71	47,89	5,58	1,79	2,39
MSI	9b	35	21,04	34,17	5,58	1,82	2,45
MSI	10a	36	22,47	58,63	4,18	1,73	2,54
MSI	10b	37	29,26	34,61	5,57	1,80	2,07

Local	Plot	Diversity Index					
		Taxonomic		Functional			
		^{0}D	^{1}D	FRic	RaoQ	CWM-EG	CWM-Dec
SAB	1	38	29,90	55,09	6,73	1,53	2,00
SAB	2	37	30,38	62,04	6,54	1,57	2,06
SAB	3	34	26,28	58,60	6,04	1,84	2,09
SAB	4	33	23,82	71,44	6,90	1,54	1,82
SAB	5	23	18,80	38,57	7,25	1,65	1,74
SAB	6	31	16,06	73,46	6,24	1,65	2,20
SAB	7	35	23,68	39,26	5,37	1,35	1,85
SAB	8	34	17,85	47,99	5,24	1,54	1,99
SAB	9	31	21,81	33,56	4,48	2,07	2,36
SAB	10	31	23,11	40,11	6,52	1,54	1,90
SAB	11	28	18,70	82,98	7,46	1,53	2,04
SAB	12	25	22,06	18,03	5,48	1,39	2,20
SAB	13	37	29,07	61,59	6,86	1,69	1,81
SAB	14	27	24,20	61,39	7,17	1,59	1,89
SAB	15	33	23,35	22,44	3,70	1,70	2,46
SJO	1	31	26,10	45,41	5,84	1,85	2,29
SJO	2	30	21,77	34,23	5,36	1,71	2,53
SJO	3	41	29,35	42,41	4,63	1,48	2,26
SJO	4	66	57,65	75,37	4,03	2,04	2,57
SJO	5	65	45,61	37,86	4,09	1,96	2,63
SJO	6	58	48,15	17,72	2,45	2,02	2,74
SJO	7	27	19,30	20,16	3,63	1,43	2,53
SJO	8	50	43,43	32,69	3,38	1,67	2,56
SJO	9	42	32,43	23,42	2,72	1,93	2,71
SJO	10	35	30,19	16,01	2,83	1,85	2,66
SJO	11	33	23,59	19,40	3,74	1,66	2,64
SJO	12	39	30,14	35,80	3,89	1,86	2,65

Table S3. Floristic dissimilarity values using the Bray-Curtis index among nine Atlantic Rainforest fragments, Minas Gerais, Brazil. Dark blue indicates lower floristic dissimilarity, and dark red indicates high dissimilarity between communities.

Fragment	CAL	GAR	GUA	IP1	IP2	MAC	MSI	SAB	SJO
CAL	-	0,72	0,72	0,85	0,89	0,82	0,68	0,64	0,74
GAR		-	0,74	0,74	0,77	0,72	0,65	0,70	0,83
GUA			-	0,76	0,87	0,79	0,76	0,56	0,84
IP1				-	0,62	0,56	0,83	0,76	0,86
IP2					-	0,66	0,84	0,87	0,88
MAC						-	0,77	0,74	0,88
MSI							-	0,74	0,81
SAB								-	0,81
SJO									-

Table S4. Spearman correlations between taxonomic (richness - ^{0}D and exponential of the Shannon-Weaver Index - ^{1}D) and functional (functional richness - FRic, Rao's functional diversity - RaoQ, and the community weighted mean for the ecological group – CWM-EG and deciduousness – CWM-Dec traits) diversity values. The upper triangle shows the correlation values, while the lower triangle presents the p-values of significance.

Diversity Index	**Phylogenetic**		**Functional**			
	^{0}D	**^{1}D**	**FRic**	**RaoQ**	**CDM-EG**	**CDM-Dec**
^{0}D	1	0.86	0.48	-0.31	0.03	0.32
^{1}D	<0.001	1	0.50	-0.20	0.09	0.23
FRic	<0.001	<0.001	1	0.43	0.10	-0.21
RaoQ	<0.001	0.018	<0.001	1	-0.01	-0.68
CDM-EG	0.764	0.295	0.241	0.893	1	0.16
CDM-Dec	<0.001	0.007	0.015	<0.001	0.060	1

Table S5. Performance (marginal and conditional R^2 values, and Akaike Information Criterion - AIC) of linear mixed models explaining taxonomic (richness and exponential of the Shannon-Weaver Index) and functional (functional richness - FRic, Rao's functional diversity - RaoQ, and the community weighted mean for the ecological group – CWM-EG and deciduousness – CWM-Dec traits) diversity, based on different groups (anthropogenic, topographic, edaphic, and climatic) of predictor variables.

Diversity Index	**Model Metrics**	**Model**			
		Anthropogenic	**Topographic**	**Edaphic**	**Climatic**
Richness	R^2 marginal	0.079	0.146	0.083	0.328
	R^2 conditional	0.458	0.373	0.425	0.356
	AIC	991.757	984.298	933.156	984.410
Exponential Shannon	R^2 marginal	0.031	0.088	0.067	0.266
	R^2 conditional	0.385	0.361	0.386	0.369
	AIC	971.735	959.359	907.085	961.116
FRic	R^2 marginal	0.222	0.036	0.109	0.061
	R^2 conditional	0.263	0.144	0.334	0.288
	AIC	1179.157	1189.241	1122.989	1195.866
RaoQ	R^2 marginal	0.192	0.078	0.158	0.156
	R^2 conditional	0.414	0.401	0.434	0.376
	AIC	414.802	406.358	353.065	420.387
CWM-Ecological Groups	R^2 marginal	0.012	0.071	0.021	0.235
	R^2 conditional	0.337	0.290	0.337	0.391
	AIC	63.613	38.816	24.797	59.147
CWM-Deciduousness	R^2 marginal	0.190	0.035	0.062	0.136
	R^2 conditional	0.517	0.522	0.555	0.535
	AIC	30.869	16.596	-7.175	34.633

CHAPTER II:

DECOUPLING ABUNDANCE AND BIOMASS IN SECONDARY ATLANTIC RAINFOREST: DIFFERENTIAL RESPONSES OF RARE AND COMMON TREE SPECIES TO ENVIRONMENTAL DRIVERS

To be submitted to the **Journal of Environmental Management**

Decoupling abundance and biomass in secondary Atlantic Rainforest: Differential responses of rare and common tree species to environmental drivers

Abstract

In the fragmented landscape, such as Brazilian Atlantic Rainforest, anthropogenic and environmental factors shape tree diversity. These three factors modulate the dynamics among species with different population proportions. We explored how anthropogenic and environmental conditions, as well as diversity, influence forest dynamics in the Atlantic Rainforest. We used 20 years of forest inventory data from 53 plots in four fragments with different land-use histories. Environmental variables and taxonomic and functional diversity indices were obtained at the plot level. We also calculated net values of abundance (NDA) and biomass (NDB) dynamics for different species abundance classes (SAC): common, intermediate, and rare. We explored the relationships between the variables sets and NDB and NDA of each SAC by building linear mixed-effects models (LMM), in which land-use history (LUH) was included as a random effect, while the other variables were grouped as fixed effects. Over 20 years, we conducted 24,379 measurements on 6,838 stems, with continuous increases in biomass, basal area, and diversity in most areas, despite local fluctuations in stem density. A total of 514 species were recorded, rotating between 423 (2002) and 440 (2022), with Myrtaceae, Fabaceae, and Lauraceae standing out. The SAC contributed distinctly to abundance and biomass. The richest forests had dominance distributed among a greater number of species, and poorer forests concentrated it in fewer species, a pattern maintained for 20 years. The abundance of common species decreased, but their biomass increased; rare species increased in both. The LMMs varied in performance across variable groups and dynamic components. For the NDB and dynamics of intermediate species, the models were not very accurate. Diversity and landscape models were the most explanatory, dominated by the random effect of LUH. At the fixed effects level, in general, common species responded to temperature, dry-season precipitation, and diversity; rare species to anthropogenic landscape and soil variables. We revealed that different tree abundance classes respond differently to environmental and historical factors and highlight the importance of conserving rare species and maintaining diversity to ensure forest biomass stability and growth in the face of climate change and anthropogenic pressures.

Keywords: community assembly, environmental filtering, forest fragments, human-modified landscape, linear mixed-effects models, rare species conservation.

1. Introduction

The Brazilian Atlantic Rainforest is a global biodiversity hotspot (Myers et al., 2000; Rezende et al., 2018; Joly et al., 2019), with higher species richness and high rates of endemism (de-Lima et al., 2020, 2024). Historically, this biome has experienced intense land use changes (dos-Santos et al., 2020), and currently, less than 30% of the original territory remains forested (MapBiomas, 2024; Vancine et al., 2024), and just 12.4% of this area consists of primary forest remnants formations (SOS Mata Atlântica and INPE, 2024). These remnants still face serious threats, including deforestation, urbanization and agricultural expansion, which lead to habitat fragmentation (Gelli et al., 2023). Therefore, most of the Atlantic Rainforest landscape is dominated by isolated secondary forest fragments, with 97% of them being smaller than 50 ha (Joly et al., 2014; Rosa et al., 2021; Vancine et al., 2024), making most of its forests <100 m from the edges (Ribeiro et al., 2009; Vancine et al., 2024).

Although the Brazilian Atlantic Rainforest is a critical South American example, forest ecosystems have suffered from anthropogenic changes on a global scale. Activities such as deforestation for agriculture and urban expansion lead to fragmentation, habitat destruction, and widespread biodiversity loss (Haddad et al., 2015; Sefidi and Copenheaver, 2020; Sefidi et al., 2021; Boulton et al., 2022). These external pressures, as well as external pressures imposed by different management practices (Dieler et al., 2017; Arcanjo et al., 2023), alter the structural pattern of forests through their impact on the populations of both common and rare species (Hordijk et al., 2024). Chronic exposure to anthropization (de-Aguiar et al., 2025), or the combination of this factors and extreme environmental conditions may lead to population declines /extinction of endemic or rare species (Idárraga-Piedrahíta et al., 2022), or collapsing demographic fluctuations even in dominant species with wide distributions (Stanke et al., 2021).

Climatic factors are the main drivers of tree species dominance and rarity worldwide (Hordijk et al., 2024). In this context, temperature and precipitation play a crucial role in modulating forest community diversity and structure (Antúnez et al., 2017; Chen et al., 2018; Pelletier et al., 2019), imposing ecophysiological limits to their development (Salomón et al., 2022; Mirabel et al., 2023; Liu et al., 2024). Furthermore, extreme events such as droughts and storms also control species growth and distribution (Crockett and Westerling, 2018; Senf et al., 2020; Bauman et al., 2022; Villanova et al., 2024). Studies have supported the hypothesis that climate variations may lead to greater species turnover among sites, as specific sets of species become established under particular environmental conditions, thereby promoting taxonomic and phylogenetic diversity and reducing the nesting of tree communities (Idárraga-Piedrahíta et al., 2022).Therefore, worsening climate change will result in the intensification of forest dynamics processes (Aleixo et al., 2019; Anderegg et al., 2020b; Senf et al., 2020; Chen et al., 2024), threatening the survival and distribution of rare and vulnerable species (Körner, 2007; Diem et al., 2018; Boulton et al., 2022) and compromising the provision of essential ecosystem services (Cavanaugh et al., 2014; Anderegg et al., 2020a; Inague et al., 2021).

Edaphic variables are the second most important driver of species rarity or abundance globally (Hordijk et al., 2024). This is because edaphic properties drive forest structure, diversity and composition (Chen et al., 2018; Pelletier et al., 2019; Ouyang et al., 2021). Additionally, factors such as topography play a fundamental role in structuring the tree community (Körner, 2007; Zhang et al., 2016, 2013; Nüchel et al., 2019), fine-scale environmental heterogeneity that shapes species richness and community assemblages (Wang et al., 2016; Rodrigues et al., 2020). Topography modulates species distributions and the dynamics of successional pathways through a second-order effect on soil moisture and nutrient availability (Zhang et al., 2016; Pelletier et al., 2019). Finally, all these environmental factors interact, resulting in different successional trajectories for forests (Sfair et al., 2016; Teixeira et al., 2020; Sui et al., 2025). This leads to structurally distinct communities that shelter varying levels of diversity (see Chapter 1), which modulates species, individual, and biomass turnover. Therefore, diversity acts as an intermediate factor between environmental conditions and forest dynamics (Dilnessa et al., 2023), such that more diverse forests tend to be greater productivity (Abrams, 1995; Comita et al., 2010; Madrigal-González et al., 2020; Li et al., 2024).

Tropical forests, such as the Atlantic Rainforest, represent the highest species richness and proportion of rare species worldwide (Hordijk et al., 2024), and there is a tendency for a positive relationship between diversity and the provision of ecosystem services (Gamfeldt et al., 2013; Cavanaugh et al., 2014; Snäll et al., 2021). Among these services, carbon storage is particularly important in tropical forests (Steur et al., 2022; Drüke et al., 2023; Liu et al., 2024), as they represent the largest global stocks of biomass and carbon (Wagner et al., 2016). In addition to these stocks, secondary formations of these forests are important carbon sinks, especially in the face of anthropogenic climate change (Rodrigues et al., 2023; Altomare et al., 2025). Thus, determining which and how biotic and abiotic environmental factors modulate the population and biomass dynamics of common and rare species is essential to maintaining this ecological role. Furthermore, investigating these interactions under different scenarios of anthropogenic pressure adds a robustness layer to the ecological approach needed to effectively manage and conserve forest resources (Thammanu et al., 2021). This is particularly important because, in some scenarios, the human activities effects on forests may surpass the environmental factors influence (Guerra et al., 2013; Danneyrolles et al., 2019; Flores et al., 2024). Studies that address these aspects in highly threatened and biodiverse ecosystems, such as the Atlantic Rainforest, can support the conservation of these landscapes by proposal of scientifically based environmental policies and management practices (Sefidi and Copenheaver, 2020; Dasgupta et al., 2022). Ultimately, understanding these relationships is critical for determining the stability and patterns of diversity and structural dynamics in these forests, driven by the dynamics of species with different levels of abundance, which directly impacts their role in the global carbon balance (Madrigal-González et al., 2020; Boulton et al., 2022; Chen et al., 2024; Flores et al., 2024).

Our objective was to examine how the dynamics of different species abundance classes (common, intermediate, and rare) respond to anthropogenic, abiotic (landscape, climate, and soil), and biotic (taxonomic and functional diversity) variables in four Atlantic Rainforest fragments over 20 years of forest succession. So, we addressed the following research questions: (i) how do different structural and diversity metrics behave throughout forest dynamics? (ii) What are the patterns of abundance and biomass dynamics of common, intermediate and rare species? (iii) How do environmental, anthropogenic and forest diversity factors drive the abundance and biomass dynamics of common and rare species?

2. Material and Methods

2.1. Study area

We used data from 53 plots (10x50m) installed in four fragments of Atlantic Rainforest distributed across the eastern portion of Minas Gerais, Brazil (Figure 1). These fragments are heterogeneous in their environmental conditions of climate, topography and soil, as well as in landscape features and forest characteristics such as structure and floristic composition (Table 1; for further details, see Torres et al., 2023). All fragments are areas of environmental conservation, consisting of secondary forests regenerated after various land uses, such as selective logging, silviculture, and agriculture (Torres et al., 2023).

Table 1. Characteristics of the nine Atlantic Rainforest fragments with different land-use history (AP – Agricultural production; SL – Selective logging, and Def – Deforestation) in Minas Gerais, Brazil, included in this study.

Fragment	**Acronym**	**Municipality**	**Area (ha)**	**Plots**	**Tree Richness**	**Maximum Elevation**	**Land-use History (%)**		
							AP	**SL**	**Def**
Cachoeira das Pombas	GUA	Guanhães	360.2	19	226	1,169	40	60	0
Ipaba Mata1	IP1	Caratinga	271.2	16	174	299	0	18.75	81.25
Ipaba Mata2	IP2	Caratinga	79.4	6	140	310	0	100	0
São José	SJO	Coronel Fabriciano	100.6	12	304	929	0	100	0

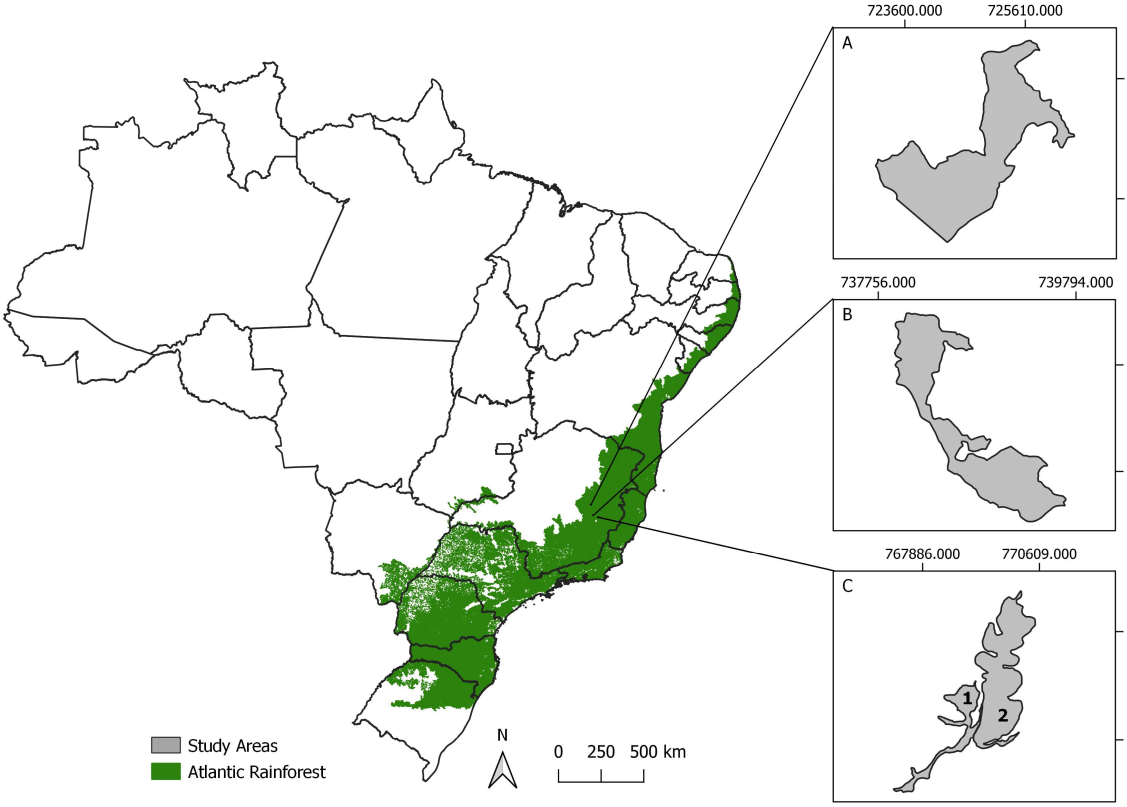

Figure. 1. Locations of the four studied Atlantic Rainforest fragments located in Minas Gerais, Brazil. **A.** Cachoeira das Pombas; **B.** São José; **C.1.** Ipaba Mata2; **C.2.** Ipaba Mata1.

2.2. Forest variables

2.2.1. Forest inventory

Five forest inventories were conducted in all plots during 20 years of monitoring (2002; 2007; 2012; 2017 and 2022), considering palms, shrubs, and trees within Diameter at Breast Height (DBH) at 1.3 m ≥ 5.0 cm. We measured the circumference of each stem using a graduated tape measure and the height with a Vertex IV digital hypsometer. In addition, we made species identifications *in situ*, and when this was not possible we collected botanical samples for later identifications, with the support of specialized literature and the help of botanical experts. The abundance and the species list for each inventory year were compiled based on the classification of angiosperm families from the APG IV (APG, 2016), and the scientific nomenclature used followed the "Flora e Funga do Brasil" (2025) and Plants of the World Online (POWO, 2025).

2.2.2. Biomass determination

From the dendrometric data, we calculated the total stem volume with bark for each stem using the following equation, adjusted for secondary forests in Minas Gerais state (Amaro, 2010).

$$VS_{WB} = 0.00007 * DBH^{2.204301} * Ht^{0.563185}$$

where: VS_{WB}: volume of the stem with bark (m^3); *DBH*: diameter at breast height (cm); and Ht: total height of the individual (m).

Above-ground biomass for stems with bark (AGB) was calculated based on stem volume and basic wood density (BWD). We used regional, national, and international databases to obtain species-specific BWD values (Table S1). When values were unavailable for a given species, we used the average of its congeners or family, following a predefined prioritization scheme. Finally, we obtained summed values of AGB for each species, plot, and monitoring year.

2.2.3. Species abundance classification

To assess how different anthropogenic, environmental, and diversity-related variables of the forest fragments influence the net dynamics of abundance and AGB (hereafter NDA and NDB, respectively) across different population abundance classes, we classified the species based on their contribution to cumulative relative abundance (CRA). For this, we relied on studies that classified species according to the representativeness of their populations within the community (citation). There are many definitions of dominant and rare species at different spatial scales and suitable for different communities (Rabinowitz, 1981; Caiafa and Martins, 2010; Avolio et al., 2019; Hordijk et al., 2024; Silva et al., 2024). Finally, we then listed the species in descending order of relative abundance and defined three species abundance classes (hereafter SAC): (i) common – those comprising the first 50% of the CRA; (ii) intermediate – species comprising the next 30% of the CRA (up to 80% of total CRA); and (iii) rare – those representing the remaining 20% of the CRA. In cases of equal relative abundance, we used the species' relative biomass as a tiebreaker. Species that entered the community during intermediate or final surveys were considered rare, as they consistently represented minor population inputs.

2.2.4. Diversity indices

We adopted a plot-level approach to calculate taxonomic and functional diversity that could mediate dynamic processes of species rarity and rotation, and biomass in each year of inventory. Taxonomic diversity was assessed through species richness and the Shannon-Weaver diversity index (H'). We assessed functional diversity through functional richness (FRic), which is independent of species abundance, and Rao's quadratic entropy (RaoQ), which is weighted by abundances. This indices represents the functional space occupied by the community (Cornwell et al., 2006) in a functional-traits multidimensional space (Villéger et al., 2008; Laliberté and Legendre, 2010; Mouchet et al., 2010), and the functional diversity from the sum of pairwise functional distances among species, weighted by their relative abundance (Mouchet et al., 2010), respectively.

Functional diversity measures were calculated based on four morpho-ecological traits that represent different aspects of ecosystem functioning, using a previously adapted approach (Castillo-Campos et al., 2024). We consulted widely used specialized literature for the Atlantic Rainforest (e.g. Lorenzi, 2008, 2009a, 2009b), botanical descriptions available on Flora e Funga do Brasil (2025), and when data were lacking, we supplemented them with field observations. Finally, we compiled, for each species, the reproductive traits: i) fruit type and ii) seed dispersal syndrome (anemochory, autochory, and zoochory); and the ecological traits: iii) leaf phenology (evergreen, deciduous, and semi-deciduous), and iv) successional group (pioneer, early secondary, late secondary, and climax; we consider some alien species as pioneers) (Gómez-Ortiz et al. 2019). As scientific literature does not present a consensus regarding the nomenclature of fruit types, we used our own classification system developed previously (Table S1, Chapter 1).

A functional dissimilarity matrix was built using Gower distance on a matrix of species and their functional traits (Yixuan Li et al., 2022). Next, we transformed categorical traits into discrete numbers and standardized them (mean = 0, variance = 1) so that all would carry equal weight in estimating the functional volume (Villéger et al., 2008; Laliberté and Legendre, 2010). Then, we added small perturbations (1^{-10000}) to the standardized values to avoid multidimensional null sums from highly collinear traits (Cornwell et al., 2006). We also calculated community-weighted means (CWM) for each plot using scalar values ranging from acquisitive to conservative strategies for the successional group (1 – pioneers, including alien species; 2 – early secondaries; 3 – late secondaries; and 4 – climax) and for deciduousness (1 – deciduous; 2 – semi-deciduous; and 3 – evergreen) to investigate trait responses to environmental variation. Indices were calculated using the “dbFD” function of the “FD” package (Laliberté et al., 2014) in the R Studio environment (RStudio Team, 2020).

2.3. Anthropogenic and environmental drivers

2.3.1. Landscape variables

We used forest structure attributes and landscape variables to investigate anthropogenic effects on the different dimensions of diversity. For this, we obtained, at the plot level, the basal area extrapolated per hectare (BA), used as a proxy for forest age, as it increases significantly along the successional trajectory from young to mature forests (Menezes and Melo, 2019; van-Breugel et al., 2019; Verly et al., 2023), and influences community demographic processes (Rozendaal and Chazdon, 2015).

The landscape variables included: land-use history (LUH), time since abandonment (time since the end of previous anthropogenic uses), forest cover, forest size, and distance to the edge. To obtain these variables, we classified the LUH into four categories: deforestation, agricultural production, silviculture, and selective logging. We compared land-use information gathered from neighboring landowners and from the property deeds, with aerial photographs (from 1960 to 1996) and Landsat satellite images from 1985 to improve our understanding of land-use changes and determine the time since abandonment of the fragments. Forest cover was calculated for different years (1985 and 2002) and for circular buffers with different radii (500, 1000, and 2000 m) around each plot (Torres et al., 2023). These data were retrieved from MapBiomas (MapBiomas, 2024), which classifies land cover and land use based on Landsat imagery with a resolution of 30 × 30 meters, processed in the Google Earth Engine platform. We used QGIS 3.28 Firenze (QGIS Development Team, 2025) to manipulate the buffers, integrate spatial information, and analyze fragmentation.

Topographic features were assessed through calculations of elevation of each plot (Torres et al., 2023; da-Rocha et al., 2024). For this, we used a Digital Terrain Model (DTM) for the study area. The processing was performed in R version 2024.12.0 (RStudio Team, 2020), using the "terra" package (Hijos, 2024). Initially, the DTM was clipped based on the boundaries of the sample plots. Then, elevation was calculated, and the mean values for each plot were extracted.

2.3.2. Soil variables

We collected composite soil samples from each plot at depths of 0–20 cm and 20–40 cm. We then performed chemical analyses: soil active acidity in H_2O (pH), exchangeable acidity (Al^{3+}), total acidity (H^{+}+Al^{3+}), exchangeable cations (Ca^{2+}, K^{+}, and Mg^{2+}), total and effective cation exchange capacity (T and t), sum of exchangeable bases (SB), base saturation (V), available phosphorus (P) and remaining phosphorus in solution (P-Rem), aluminum saturation index (m), and soil organic matter (SOM). Analyses were performed at the Soil Fertility Laboratory of the Federal University of Viçosa (UFV), Brazil, following standard methods (Teixeira et al., 2017).

2.3.3. Climate variables

We used data recorded by the weather station closest to each forest fragment to analyze different climatic metrics and calculated different climate variables for each inventory year. We followed previous studies that used different sets of these variables and modeled different aspects of the forest community (Torres et al., 2023; da-Rocha et al., 2024). Therefore, we calculate the mean temperature (Temp) and total precipitation of the inventory year (Precp), of the previous year (Precp-1), the mean of the two (Precp-2), three (Precp-3), and four (Precp-4) preceding years, total precipitation in the three driest months of the inventory year (Precp-Dry), and the number of months with precipitation <100 mm (Less100).

We also estimated the climatic water deficit (CWD), as water stress is one of the main drivers associated with tree recruitment and mortality in the Atlantic Rainforest (Torres et al., 2023). Furthermore, it can incorporate the effects of anthropogenic variables on the water balance, reflecting drought conditions more accurately than conventional climatic variables, such as total annual precipitation (Chave et al., 2014), and therefore may also be considered an anthropogenic variable. This variable represents the water balance between potential evapotranspiration (PET) and real evapotranspiration (AET) and is used as a proxy for drought conditions. We follow a previously proposed calculation protocol (Lutz et al., 2010), and used the CWD and AET functions (Redmond, 2022) in the R Studio environment (RStudio Team, 2020). The CWD calculation, in addition to local precipitation and temperature, includes the declivity, aspect and location (geographic coordinates) of the plot. The index is interpreted such that the more negative the CWD values, the higher the water stress, whereas values close to zero indicate the absence of water stress (Poorter et al., 2017). As the effects of water stress vary and can have delayed impacts on tree development and mortality (Anderegg et al., 2020b; Bauman et al., 2022; Chen et al., 2024), we also calculated the mean CWD for one (CWD-1), two (CWD-2), three (CWD-3), and four (CWD-4) years prior to the measurement year, in addition to the CWD of the inventory year.

2.4. Data analysis

2.4.1. Variables selection

We tested our initial variables to reduce the dimensionality of variables to be included in the statistical models for predicting diversity. First, we conducted a Spearman correlation analysis for the set of soil variables (28). The variables measured in the 0–20 cm layer was highly correlated ($r \geq 0.75$) with their respective measurements in the 20–40 cm layer (Figure S1). Therefore, we used only the values from the 20–40 cm depth layer in the subsequent steps, because the variables in this layer were less correlated with each other. We then eliminated collinear variables and selected only pH, P, P-Rem, Al^{3+}, $H+Al^{3+}$, SB and t.

We also performed Spearman correlation for the forest cover data from different monitoring years (1985 and 2002) and buffer distances from the plot's centroid (500, 1000, and 2000 m). As the six year-ratio combinations were better correlated between the years of observation ($r \geq 0.65$), so we selected forest cover from 2002 within a 2000 m buffer, as it was less correlated with all the others (Figure S2). Among the climatic variables, we selected Temp, Precp, Precp-4, Precp-Dry, and CWD-1. Temp was well correlated and all CWD indices ($r \geq -0.75$). The Precp was not well correlated with any other climatic variable ($r < \pm 0.38$), while the precipitation of the other previous years was highly correlated ($r \geq 0.72$) with each other (Figure S3), and for this reason we selected only Precp-4. Precp-Dry was highly correlated ($r = 0.80$) only with Less100. CWD-1 was correlated ($r \geq 0.74$) with all CWD values from previous years and from the year of the forest inventory, in addition, it was identified as a determining factor in recruitment, mortality and growth in our study areas (Torres et al., 2023). Although Temp and CWD-1 were strongly correlated ($r = -0.80$), we retained both variables due to their ecological relevance. We also performed this analysis for diversity indices and removed species richness, as it was collinear to the other indices (Figure S4).

2.4.2. Net dynamics modeling

We used linear mixed-effects models (LMM) to investigate the combined effects of anthropogenic and environmental variables on NDA e NDB in Atlantic Rainforest fragments in Southeastern Brazil. This approach was chosen because LMM is flexible in handling unbalanced variables. We built models for each group of fixed effect variables. The landscape model included fragment size, edge distance, forest cover in a 2000 m buffer from the plot centroid in 2002, time since abandonment, plot basal area extrapolated to a hectare, and the elevation. Edaphic models included pH, P, Al^{3+}, $H^{+}+Al^{3+}$, SB and P-Rem. To the climatic model we use Temp, Precp, Precp-4, Precp-Dry and CWD-1. Finally, the diversity model included H', FRic, RaoQ, CWM-EG and CWM-Dec. Land-use history and year of inventory were used as random effect variables in all models, due to the unique historical and successional characteristics of each fragment. Models were built using the "lme4" package (Bates et al., 2015) in the R Studio environment (RStudio Team, 2020). We evaluated the confidence interval distribution graphically to determine the significance of the fixed variables in each model. Model fit performance was compared using the Akaike Information Criterion (AIC) (Burnham and Anderson, 2002), where models with lower AIC values were considered better fitting. The contribution of fixed and random variables was assessed using marginal and conditional R^2 values. Marginal R^2 corresponds to the variance explained only by fixed effects, while conditional R^2 represents the variance explained by both fixed and random effects (Nakagawa and Schielzeth, 2013).

3. Results

3.1. Structure and diversity patterns

Over five forest inventory events conducted during 20 years of monitoring, we performed 24,379 measurements on a total of 6,838 stems, of which 3,000 (43.9%) were alive and measured in all years. In the first inventory (2002) we recorded 4,724 stems, and 20 years later, 4,777 stems were measured. All fragments showed an increase in stem density until 2012, followed by a decline in the 2017 inventory (Table 2; Figure S5). These decreases, the IP1 and IP2 areas had a negative stem density balance in 2022, unlike GUA and SJO, which showed a positive balance over the 20 years analyzed. Basal area, biomass, and the standard deviation of DBH (SDD) increased continuously across all areas, except for IP2, which experienced a slight decline in basal area and biomass between 2012 and 2017.

In 2002, we found a total of 423 species across all areas, distributed among 187 genera and 67 families (Table S2). By 2022, these numbers had increased to 440 species, 193 genera, and 68 families. Over the 20 years of monitoring, 363 species remained in the communities, 60 were lost, and 78 were recruited, in addition to 13 species present only in intermediate inventories, totaling 514 recorded taxa. The richest families were Myrtaceae (67), Fabaceae (57), and Lauraceae (52), which together accounted for 34.24% of the total recorded species. This high richness was driven by the genera *Ocotea* (27) in Lauraceae, and *Myrcia* (25) and *Eugenia* (23) in Myrtaceae, while in Fabaceae the richest clade was *Inga* (11). A total of 56.7% of the genera were monospecific in our fragments. At the fragment and plot levels, the areas showed a continuous increase in species richness, except for IP1, which peaked in 2012 but had lower richness in 2022 compared to 2002 (Table 2; Figure S5). Regarding the Shannon-Weaver Index (H'), although all plots showed an increase (Figure S5), at the fragment level, IP1 showed a slight decline, while SJO had the highest H' values throughout all monitoring years (Table 2). The GUA and SJO fragments showed the highest species richness and community H', and were also the forests that maintained the most consistently expressive increases in biomass even in intervals of reduced abundance (Table 2; Figure S5).

Table 2. Means (± standard deviation) dynamics of the structure (Ab – abundance; BA – basal area; AGB – above-ground biomass for stems with bark; SDD – Standard Deviation of DGH) and diversity (S – Richness and H'– Shannon-Weaver Index) of tree species, during 20 years of monitoring (2002-2022), in four Atlantic Rainforest fragments in Minas Gerais, southeastern Brazil (GUA – Cachoeira das Pombas; IP1 – Ipaba Mata1; IP2 – Ipaba Mata2; SJO – São José).

Fragment	Year	Ab (Stems ha^{-1})	BA (m^{2} ha^{-1})	AGB (Mg ha^{-1})	SDD (cm)	S	H'
GUA	2002	1.988.4±576.1	15.39±4,70	53.29±22,28	4.65±1,87	175	4.34
	2007	2.101.1±648.8	17.68±4,79	62.13±23,91	4.94±1,99	182	4.36
	2012	2.140.0±655.8	19.47±5,23	71.35±27,25	5.18±2,02	184	4.37
	2017	2.053.7±631.5	20.09±6,23	74.73±31,42	5.46±2,04	183	4.45
	2022	1.995.8±614.4	20.26±5,68	83.67±34,52	5.57±2,11	182	4.45
IP1	2002	1.718.8±447.4	14.99±5,97	47.52±29,82	4.58±1,34	130	3.99
	2007	1.847.5±428.1	16.86±5,35	53.57±28,76	4.59±1,50	130	3.96
	2012	1.815.0±509.8	17.26±6,03	58.71±31,96	4.81±1,70	135	3.96
	2017	1.692.5±454.7	16.90±7,12	59.94±37,50	5.18±1,88	132	3.96
	2022	1.697.5±432.0	18.80±7,62	74.85±45,36	5.62±1,94	129	3.94
IP2	2002	1.796.7±309.9	22.84±6,40	92.38±36,36	7.23±3,08	104	3.94
	2007	1.863.3±313.0	25.39±7,17	104.038±40,89	7.51±3,08	109	4.01
	2012	1.880.0±261.1	27.11±7,93	115.468±45,72	7.83±3,08	106	4.00
	2017	1.743.3±199.8	25.92±9,78	111.05±54,99	8.00±3,55	104	3.99
	2022	1.640.0±226.6	26.29±11,31	123.89±70,74	8.51±4,08	106	4.01
SJO	2002	1.536.7±423.9	20.22±9,34	81.31±52,42	6.71±1,64	222	4.85
	2007	1.626.7±386.0	22.45±9,13	90.09±54,02	6.90±1,55	235	4.91
	2012	1.680.0±281.7	24.17±7,45	97.08±47,05	7.00±1,40	236	4.90
	2017	1.638.3±325.5	24.84±6,94	100.37±46,06	7.22±1,33	235	4.93
	2022	1.718.3±364.5	24.67±8,00	108.82±49,74	7.11±1,34	246	4.98

3.2. Dynamics of abundance classes

The number of species in each class of abundance (SAC), and the relative contribution of these groups to abundance and AGB, differed among fragments in the first inventory (2002) (Figure 2; Table S3). The average proportion of common, intermediate and rare species richness was 7.25, 17.91 and 74.84%, respectively. In general, the fragments with lower species richness (IP1 and IP2) had their dominance concentrated in a few species, while the richest forests (GUA and SJO) distributed their dominance in a larger group of species (Figure 2; Table S3), a pattern that was maintained throughout the analysis period (Figure S6). Common and intermediate species contributed slightly less to AGB (points below the bars), while rare species contributed relatively more (points above the bars), except in IP2, where *Marlimorimia contorta* (DC.) L.P.Queiroz & P.G.Ribeiro (Fabaceae) accounted for 36.94% of AGB. This pattern of contribution remained unchanged after 20 years of monitoring, despite intense transitions of species between the SACs, particularly in the richest and most diverse fragment in SJO (Table S3).

The contribution of SAC to total abundance and AGB varied at the plot level, with both positive and negative values across all SAC in different plots (Table S4) but remained consistent across fragments. The abundance of common species declined in all fragments, with this effect becoming more pronounced after the 2012 inventory (Figure 3). Intermediate species showed more variable trends, with declining in IP1 and IP2, increasing in GUA, and remaining stable in SJO. Rare species, on the other hand, showed an increase in abundance across all fragments, especially in SJO. In terms of biomass, common species tended to remain stable, while intermediate and rare species showed continuous increases in all areas, except in IP2, where there was a marked increase in biomass for common species and a sharp decline in biomass for intermediate species after 2012.

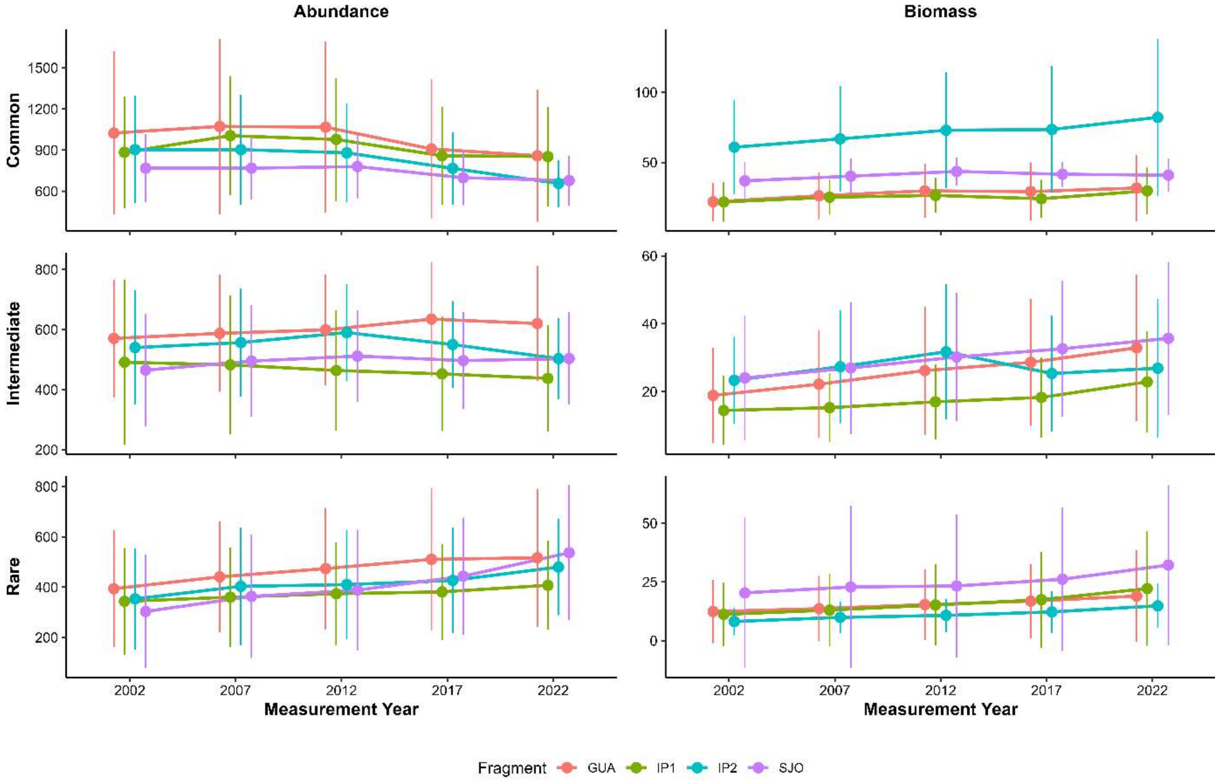

Figure 3. Abundance (stems ha^{-1}) and biomass (above-ground biomass for stems with bark) (Mg ha^{-1}) dynamics of different abundance classes during 20 years of monitoring (2002-2022), in four Atlantic Rainforest fragments in Minas Gerais, southeastern Brazil.

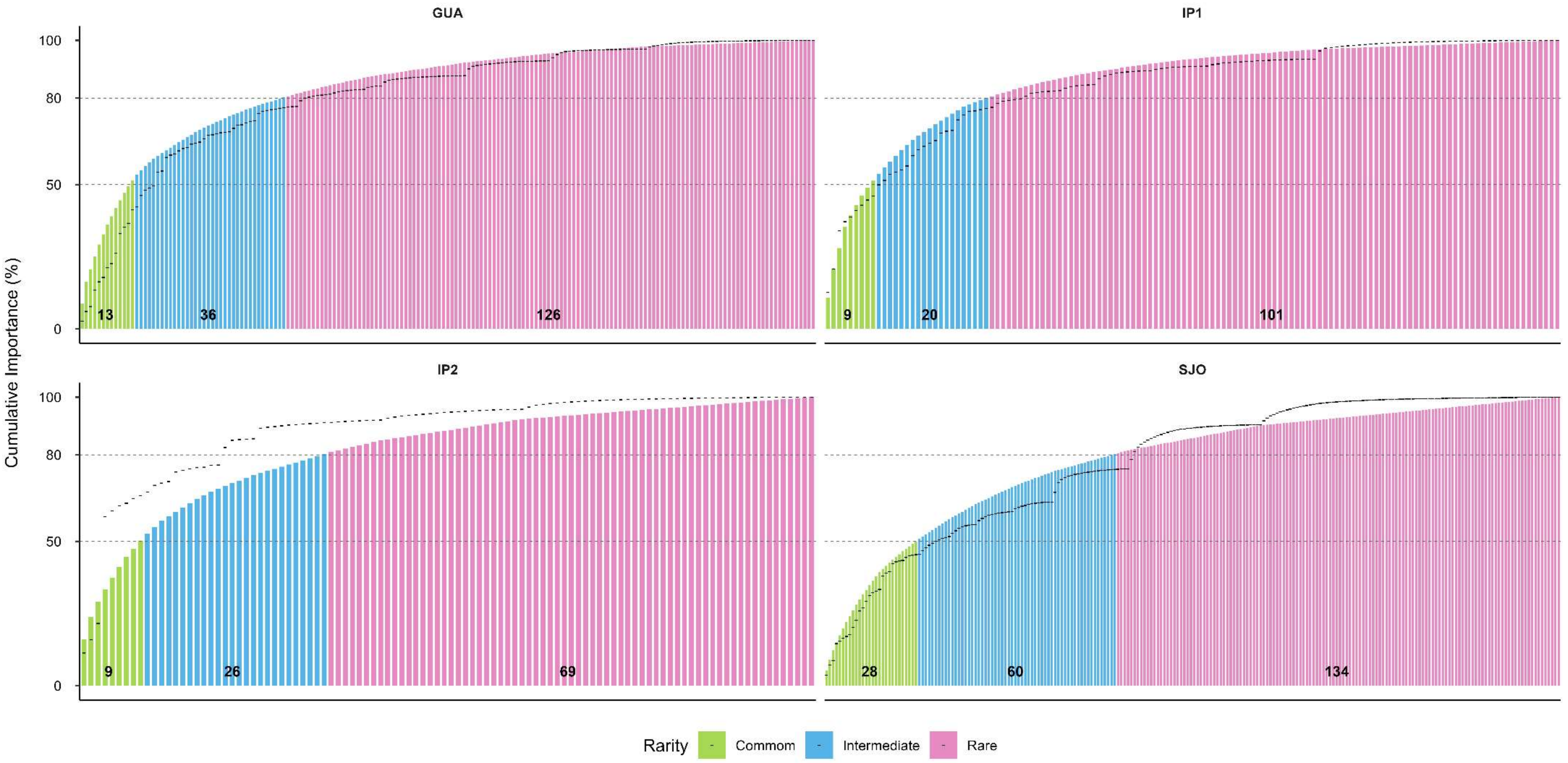

Figure 2. Distribution of tree species into abundance classes defined by their contribution to cumulative relative abundance in the first survey (2002), and their contribution to biomass, in four Atlantic Rainforest fragments in Minas Gerais, southeastern Brazil. Colored bars represent each species' cumulative contribution to relative abundance, classified as common (green, up to 50%), intermediate (blue, 50–80%), and rare (pink, >80%). Dotted lines indicate each species' corresponding contribution to cumulative relative biomass. Numbers within the bars indicate the number of species in each abundance class.

3.3. General anthropogenic and environmental effects

The performance of the linear mixed effects models varied across groups of variables and dynamic components, with conditional R^2 ranging from 0.037 to 0.435 of the variability when including both fixed and random effects, and between 0.003 and 0.154 when including only fixed effects (marginal R^2; Figure 4; Table S5). Overall, the influence of the random variable LUH explained most of the variability in abundance (NDA) and biomass (NDB) for all abundance classes. This effect was most evident in the models for NDA and NDB for common species, which presented the highest conditional R^2 when compared to intermediate and rare species. Some exceptions were NDA and NDB for rare and intermediate species in the landscape and edaphic models. At model level, the diversity model showed the strongest explanatory power, with the highest conditional R^2 for NDA common (0.435) and NDA intermediate (0.253), while the landscape model performed better for NDB common (0.292). For NDA rare, the climatic model showed the highest conditional R^2 (0.194). The climatic model had the strongest fixed effects for NDA common (marginal R^2 = 0.154), and the edaphic model for NDA rare (marginal R^2 = 0.116).

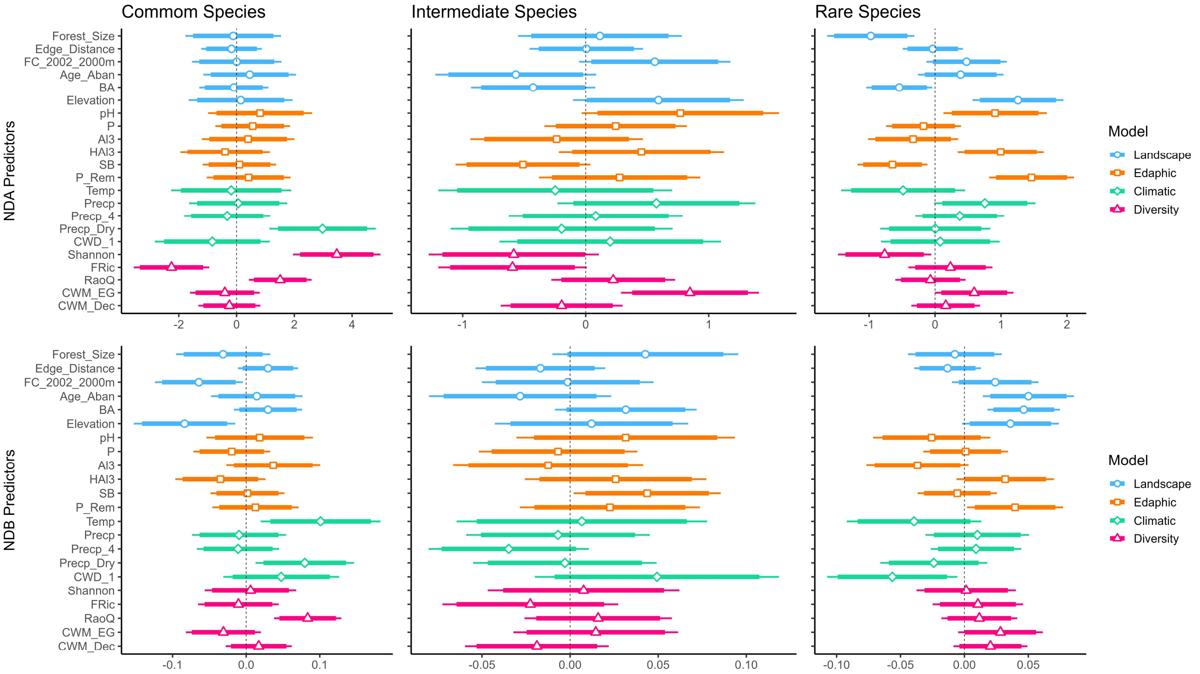

Figure 4. Effects of anthropogenic and environmental variables on the abundance (NDA) and above-ground biomass for stems with bark (NDB) net dynamics of different abundance classes in four Atlantic Forest fragments, Minas Gerais, Brazil. Standardized coefficients with 95% confidence intervals are indicated. The vertical line in each graph indicates zero. Variables containing zero within the confidence intervals are considered insignificant. FC_2002_2000m – forest cover in a 2000 m buffer from the plot centroid in 2002; BA – basal area of the plot extrapolated to hectare; soil variables in the 20-40 cm layer: pH, P cations, exchangeable acidity (Al^{3+}), total acidity ($H^{+}+Al^{3+}$), sum of exchangeable bases (SB) and remaining phosphorus in solution (P-Rem); Temp – annual mean temperature; Precp – total precipitation in the year of the forest inventory; Precp_4 – Mean precipitation in the four years preceding the inventory; Precp_Dry – precipitation of three driest months of the inventory year; CWD_1 – climatic water deficit one year before the census; and diversity indices: Shannon – Shannon-Weaver diversity index, FRic – functional richness, RaoQ – Rao's quadratic entropy, CWM_EG – community-weighted means of ecological groups, and CWM_Dec – community-weighted means of deciduousness.

3.4. Predictors of abundance net dynamic

The behavior of the variables and models was heterogeneous among the different SAC (Figure 4; Table S5). The highest marginal R^2 values were observed for the climate model for common species (0.154), and for the edaphic (0.116) and landscape (0.112) models for rare species. However, the models with the highest conditional R^2 were dominated by random effects, particularly the diversity model for common (0.435) and intermediate (0.253) species, and the climate model for rare species (0.194). The effects of anthropogenic and environmental variables varied according to land-use history (LUH), shown by the differences in slope of the trend lines. Overall, the common species NDA were positively affected by the precipitation of the three driest months of the inventory year (Precp-Dry), species (H') and functional (RaoQ) diversity, and negatively by functional richness (FRic) (Figure 5). Intermediate species showed a positive response to CWM-EG. The dynamics of rare species were characterized by negative relationships with fragment size, basal area (BA), exchangeable bases (SB) and H', while elevation, pH, total acidity (HAl3) and remaining phosphorus in solution (P-Rem) had positive effects.

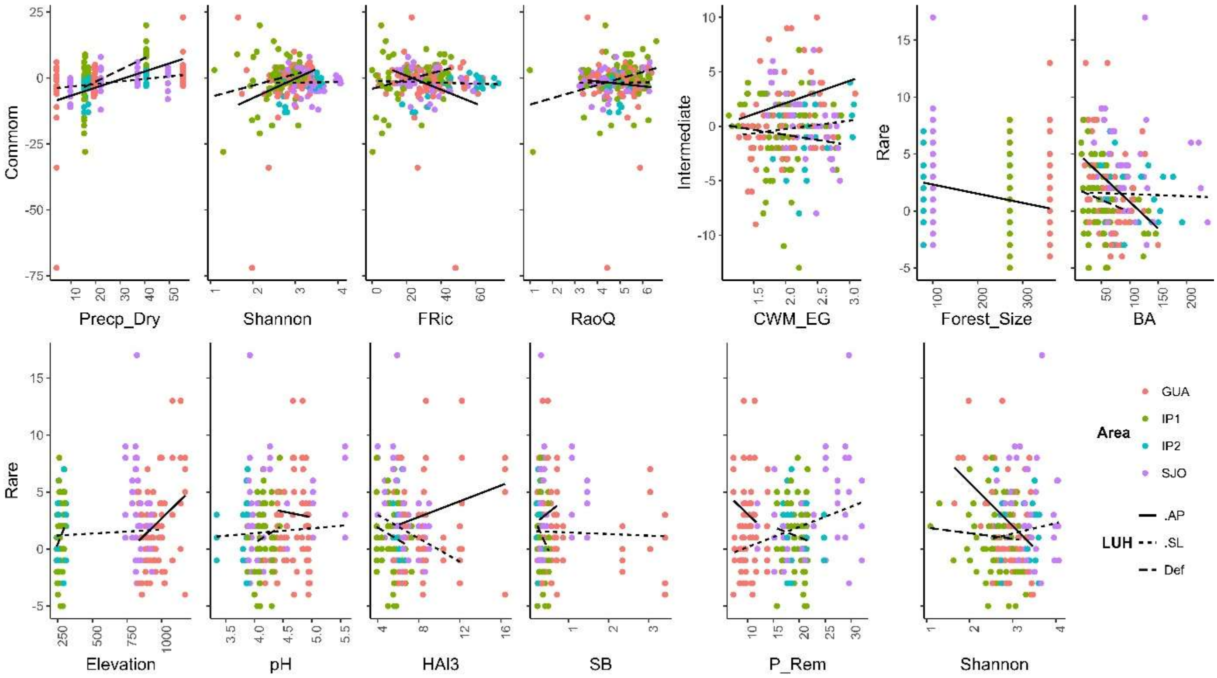

Figure 5. Bivariate relations of abundance (NDA) net dynamics of different abundance classes the variables significant by linear mixed-effects models (LMM): forest size, basal area of the plot extrapolated to hectare (BA), elevation, pH, total acidity ($H^{+}+Al^{3+}$), sum of exchangeable bases (SB), remaining phosphorus in solution (P-Rem), precipitation of three driest months of the inventory year (Precp_Dry), Shannon-Weaver diversity index (Shannon), functional richness (FRic), Rao's quadratic entropy (RaoQ), community-weighted means of ecological groups (CWM_EG), and community-weighted means of deciduousness (CWM_Dec), in different land use histories (AP – Agricultural production; SL – Selective logging, and Def – Deforestation).

3.5. Predictors of biomass net dynamic

Overall, the models were less explanatory for NDB (Table S5), with fewer significant variables compared to NDA (Figure 4). The highest marginal R^2 values were observed for the landscape model for common species (0. 101) and the climatic model for rare species (0.092). The model with the highest conditional R^2 were the landscape model for common (0.292) and rare (0.115) species, and the edaphic model for intermediate species (0.103). The NDB of common species was negatively related to forest cover and elevation, while temperature, Precp-Dry, and RaoQ showed positive relationships (Figure 6). SB was the only significant fixed variable for intermediate species, with a positive relationship. The NDB of rare species was positively related to abandonment time, BA, and P-Rem, while CWD-1 showed a negative relationship, although the trend was positive in areas with agricultural LUH.

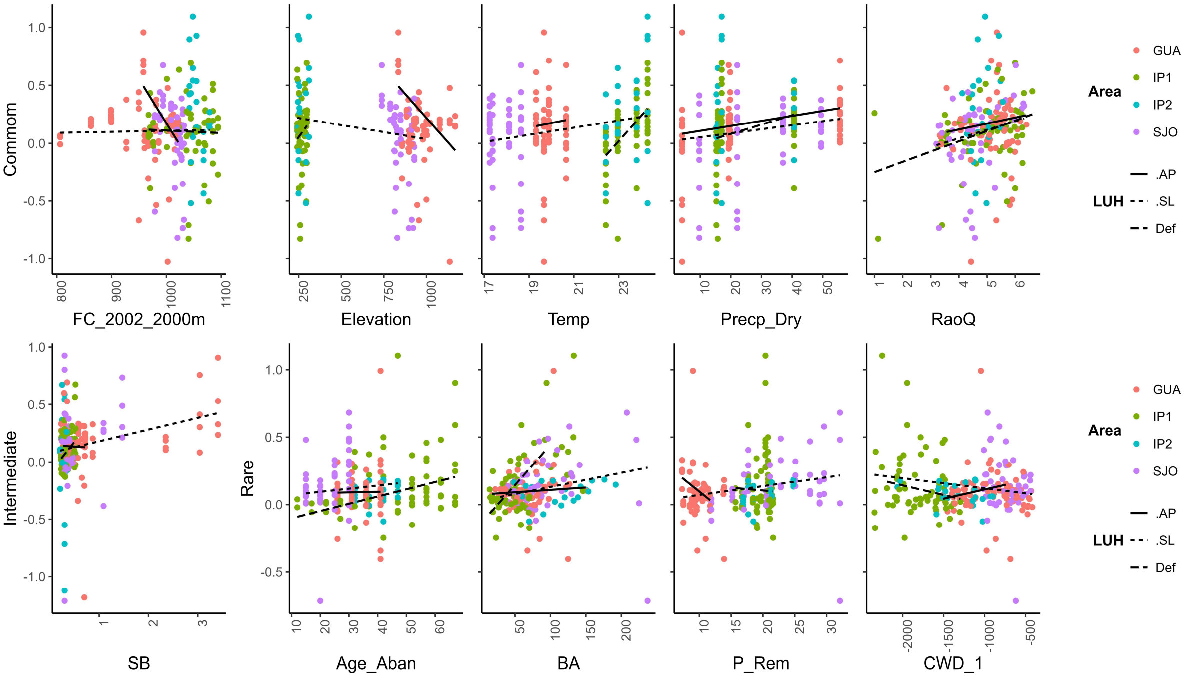

Figure 6. Bivariate relations of above-ground biomass for stems with bark (NDB) net dynamics of different abundance classes with the variables significant by linear mixed-effects models (LMM): forest cover in a 2000 m buffer from the plot centroid in 2002 (FC_2002_2000m), time since abandonment, basal area of the plot extrapolated to hectare (BA), elevation, sum of exchangeable bases (SB), remaining phosphorus in solution (P-Rem), annual mean temperature (Temp), precipitation of three driest months of the inventory year (Precp_Dry), climatic water deficit one year before the census (CWD_1), community-weighted means of ecological groups (CWM_EG), and community-weighted means of deciduousness (CWM_Dec), in different land use histories (AP – Agricultural production; SL – Selective logging, and Def – Deforestation).

4. Discussion

Studies have addressed the abundance and biomass (AGB) dynamics in the Atlantic Rainforest (Teixeira et al., 2009, 2020; da-Cunha et al., 2021; Vibrans et al., 2022; Gelli et al., 2023; Luz et al., 2023; Vancine et al., 2024), but our study is pioneering in addressing the unfolding of these dynamic processes in three species abundance classes (common, intermediate, and rare) (SAC). We explored the gaps related to the response of abundance and AGB of locally common and rare species to anthropogenic, environmental, and forest diversity factors as agents of community modulation. We employed linear mixed-effects models (LMM) to explore the influence of these factors on the abundance (NDA) and AGB (NDB) net dynamics in the SACs over 20 years of succession in 53 plots of four forest fragments of the biome in Southeastern Brazil. During monitoring, the forest fragments showed general increases in stem abundance until 2012, and a decline in subsequent inventories. However, basal area (BA), AGB, species richness and diversity (H'), and community-weighted means for ecological group and deciduousness increased uninterruptedly, while richness (FRic) and diversity (RaoQ) showed distinct successional trajectories among the fragments (Table 2; Figure S5). We discovered that the rare species relative contribution is proportionally higher for AGB than for abundance (Figure 2), and that despite intense transitions among SAC, this pattern remained over 20 years of monitoring (Figure S6). Common species tend to reduce their population expressiveness as rare species increase in abundance, but both groups continue increasing their contribution to AGB. Even with the heterogeneous performance of the LMM, we found that the random variable of land-use history (LUH) explained most of the variance in NDA and NDB (Table S4), but that some fixed effect drivers were also significant (Figures 4, 5, 6). Different factors were significant, but taxonomic and functional diversity indices were consistent drivers of NDA across the different SAC, with more pronounced effects when LUH was agriculture and milder when plots experienced selective logging. However, the relationships with NDB were less evident, with lower conditional and marginal R^2 for the different models and less expressive relationships with the different predictors (Table S4; Figures 4, 6). Our results expand the understanding of environmental filtering mechanisms mediated by anthropogenic and environmental factors in the modulation of the common, intermediate, and rare species dynamics. This approach is innovative for considering the distinct responses of abundance classes and is relevant to support practices aiming at maintaining tree diversity and the ecological stability of tropical forests in the face of extreme climatic disturbances (Gao et al., 2021; Ouyang et al., 2021; Klockow et al., 2023; Schnabel et al., 2025).

4.1. Structure and diversity patterns

There was an increase in stem abundance in all fragments until the 2012 inventory, followed by a decline in 2017, which was recovered by some forests (GUA and SJO), but not in others (IP1 and IP2) (Table 2). Secondary forests may experience distinct successional trajectories due to intrinsic characteristics and the environmental and anthropogenic factors to which they are exposed (Guerra et al., 2013; Arroyo-Rodríguez et al., 2017; Vibrans et al., 2022). In the Atlantic Rainforest, fluctuation in abundance is well documented (da-Rocha et al., 2020; Rodrigues et al., 2023; Verly et al., 2023), and is related to the dynamic equilibrium of tropical forests, where internal events such as canopy gap openings can promote these variations (Laurance, 2004; Campanello et al., 2007; Thier and Wesenberg, 2016), in addition to the fact that species turnover is spatiotemporally variable (de-Toledo et al., 2025). However, the reduction in stem number was synchronized among all fragments, which is likely related to severe drought events (Klockow et al., 2023), such as the one that occurred in 2014, and documented as causing high mortality in the study areas (da-Rocha et al., 2020; Torres et al., 2023). Even with the fluctuation in stem abundance, only the IP2 fragment did not show a constant increase and positive balance for BA, AGB, and Standard Deviation of DGH (SDD), which are structural indicators of forest maturation (da-Rocha et al., 2020; Vibrans et al., 2022).

Species richness grew steadily in all areas during the 20 years of monitoring, which is a pattern in Seasonal Semideciduous Forests of the Atlantic Rainforest, and is also an indication of the maturation of tropical forests (Oliveira et al., 2019; da-Rocha et al., 2020). Our results also point to a floristic composition pattern consistent with other forest formations of the biome, where Myrtaceae, Fabaceae, and Lauraceae are recognized as some of the richest families of the Brazilian flora and of the Atlantic Rainforest (Stehmann et al., 2009; BFG, 2015), being sampled with greater richness in other fragments of this vegetation (de-Figueiredo et al., 2015; da-Rocha et al., 2017). At the genus level, these families were represented by *Ocotea*, *Myrcia*, *Eugenia* and *Inga*, which are clades routinely reported among the richest in the biome (de-Figueiredo et al., 2015; da-Rocha et al., 2017).

Although structural patterns have consistently increased, the different diversity indices show distinct successional trajectories among the fragments, with SJO showing the greatest gain in taxonomic diversity, while IP2 stood out for high FRic and IP1 with an increase in RaoQ. The divergence between taxonomic and functional indices suggests distinct environmental filtering processes among the fragments, which may be modulated by the intensity of disturbance in each environment (Biswas and Mallik, 2011; Cadotte et al., 2011; Thier and Wesenberg, 2016). Even with distinct successional trajectories in our fragments, all represent conserved forests with expressive taxonomic and functional diversity, as well as a growing accumulation of biomass. All plots have been abandoned for at least three decades (Chapter 1), confirming the importance of these fragments in hosting sensitive species, since in the Atlantic Rainforest there is a tendency for young forests (< 20 years) to be converted into agricultural land (Teixeira et al., 2009). Furthermore, previous surveys confirmed that many locally rare species are endemic to the biome and are listed in some extinction threat category (da-Rocha et al., 2017; Luz et al., 2023), and may often be overlooked due to non-sighting (Terra et al., 2020). Therefore, determining the rare species contribution to forest structure and the factors that modulate their dynamics is fundamental to support monitoring, conservation, and management policies of fragments aimed at protecting rare and endemic species (Thier and Wesenberg, 2016; Luz et al., 2023).

4.2. Abundance and biomass dynamic of abundance classes

We found that rare species comprised a substantial portion of richness (68.00–81.87%), while common species comprised at most 20.18% of the species. In the Atlantic Rainforest, rare species are recognized for corresponding to most of the richness in different types of forest formations (Caiafa and Martins, 2010; Oliveira et al., 2019; Terra et al., 2020; de-Toledo et al., 2025). We observed that in poorer forests a smaller species-set is responsible for most of the population dominance, corroborating studies carried out in the biome (Rodrigues et al., 2019), and a global pattern of richness inversely related to dominance (Hordijk et al., 2024). Although each community may experience distinct successional trajectories (Sfair et al., 2016; Arroyo-Rodríguez et al., 2017; Teixeira et al., 2020), our results evidenced a community tendency of increase in abundance and AGB at the fragment level (Figure 3, S5), but a heterogeneity of responses among plots (Table S3), which is linked to the natural behavior of Atlantic Rainforest dynamics (Vibrans et al., 2022). This internal variation demonstrates distinct response patterns among the different study areas, with areas such as SJO frequently presenting more extreme values (Figure 3).

Our results revealed population stability of intermediate species, but a consistent decline among common species in contrast to the gradual rare species increase (Figure 3). Common species are recognized for representing a large part of stem abundance and biomass stocks (Tabarelli et al., 2010; Oliveira et al., 2019), regulating the net balance of these components through their high representativeness in the mortality and recruitment dynamics in the Atlantic Rainforest (de-Figueiredo et al., 2015; Batista et al., 2016). We noticed intense NDA for some common species, with drastic population reductions of *Xylopia sericea* A.St.-Hil. (Annonaceae) in the IP1 fragment, *Mabea fistulifera* Mart. (Euphorbiaceae) in IP2, and *Eremanthus erythropappus* (DC.) MacLeish (Asteraceae) in GUA (Table S2). Although studies predict an increase in dominance with forest maturation (Hordijk et al., 2024), in the Atlantic Rainforest common species tend to reduce their population expressiveness along succession (Batista et al., 2016). Therefore, the population declines of these species are justified by the successional demographic dynamics itself, especially because they are pioneer species (Tabarelli et al., 2010), which tend to be replaced by late-successional species (Poorter and Garnier, 2007;

Wright et al., 2010), decreasing their contribution to the total community stock of biomass and carbon (de-Figueiredo et al., 2015).

Complementarily, the positive rare species NDA is explained by the reduction of common ones, since at regional scales the fluctuation in abundance between these two groups is inversely related (Hordijk et al., 2024). Furthermore, the abundance of each rare species is highly dependent on neighboring conspecific density, an effect not evident among common species (Johnson et al., 2012; Pulla et al., 2021). Therefore, while the common species NDA is driven by community maturation and rotation with late-successional rare species, the rare species NDA is limited by intraspecific competition. Although NDA between common and rare species was divergent, all SAC showed consistently positive values of NDB over the 20 years of monitoring. This pattern is recognized among secondary fragments of Seasonal Semideciduous Forest (de-Figueiredo et al., 2015). Therefore, monitoring dynamics can better inform estimates of resource changes in forest ecosystems (Stanke et al., 2021; Klockow et al., 2023).

4.3. Predictors of abundance and biomass dynamic

Species from different abundance classes (common, intermediate, and rare) respond in distinct ways to environmental variables, diversity indices, and different LUHs (Figure 5, 6). In general, common species showed more evident responses and seem more influenced by climatic factors, while rare species respond more strongly to topographic (elevation) and edaphic (pH and H^{+}+Al^{3+}) factors. Common species contribute most to forest dynamics (Batista et al., 2016), facilitating the detection of deterministic processes in these fluctuations. However, land-use history modifies these relationships, creating distinct response trajectories that impact the common and rare species dynamics in different ways and intensities (Köhler and Huth, 2004; Stanke et al., 2021; Hordijk et al., 2024). Much of this alteration is explained by the increase in acidity and nutrient depletion of soil used for agriculture and silviculture (Lishan and Regasa, 2022; Wells and Sawyer, 2022; Baláš et al., 2024). In this sense, we observed milder effects of predictor variables on dynamics when the LUH was selective logging. In our areas, taxonomic and functional diversity were also strongly affected by more intensive LUHs (Chapter 1), mainly due to the negative effect of agricultural cultivation on soil properties (Pinho et al., 2018; Bentsi-Enchill et al., 2022; Fachin et al., 2024).

Additionally, selective logging in tropical forests favors the dominance of certain species and reduces the abundance of rare species, especially because they are more sensitive to disturbances and have severely compromised recruitment (Köhler and Huth, 2004). Furthermore, more disturbed forests may be more vulnerable to the invasion of alien species but tend to be less exploited due to the scarcity of commercially attractive species (Aynekulu et al., 2016). In the Atlantic Rainforest, a previous study showed that selective logging is a more recurrent exploratory activity in more mature and diverse forests, due to the greater availability of trees and variety of species, and intensified by ease of access, with greater harvests near edges (da-Cunha et al., 2021). However, the research also found that this activity causes little expressive impact (<5%) on community abundance and biomass, and that the most abundant species were the most exploited, probably due to the greater availability of stems for cutting. Thus, our plots subjected to selective logging, which were abandoned more than three decades ago (Chapter 1), were able to reduce the anthropogenic effect on their structural patterns, mainly because niche complementarity in more diverse forests reduces their sensitivity to environmental stress (Chisholm et al., 2013; Madrigal-González et al., 2020; Bayat et al., 2021; Gao et al., 2021).

Among the anthropogenic fixed effects, we also found that BA had a negative effect on the rare species NDA (Figure 5). In our areas, lower relative recruitment and growth were observed in plots with higher basal area and in less disturbed forests (Torres et al., 2023), suggesting some saturation in stem occupation and reduced resource availability for the establishment of new rare trees, since more mature forests harbor higher proportions of rare species (Oliveira et al., 2019). On the other hand, both time since abandonment and basal area (BA) were positively related to the rare species NDB (Figure 6). The synchronized response of time since abandonment and BA is ecologically fundamental, as BA is a proxy for forest age (Pinho et al., 2018; van-Breugel et al., 2019; Verly et al., 2023). This reinforces the role of forest succession as a modulator of diversity and population dynamics of rare Atlantic Rainforest species (Chapter 1). However, the global trend evidenced by a recent study is that in more mature forests there is a reduction in the expressiveness of rare species due to the dominant species stems growth (Hordijk et al., 2024). In fact, what we observed was a reduction in abundance, but an increase in biomass throughout succession regardless of SAC, which could be linked to the greater propensity for mortality of small trees in tropical forests (Ma et al., 2016; da-Rocha et al., 2020; Pulla et al., 2021), maintaining the largest stems alive and growing, providing a significant contribution to biomass.

Edge effects have a major influence on community structuring in the Atlantic Rainforest, potentially outweighing the effects imposed by environmental characteristics such as the topographic gradient (Lopes et al., 2009; Guerra et al., 2013; Thier and Wesenberg, 2016). But unexpectedly, among the landscape variables related to fragment anthropogenesis, distance to edge was not an explanatory variable for the dynamics of any SAC. This result is likely related to the fact that tree mortality in our areas is not clearly influenced by distance to the edge, which was attributed to the presence of mature forest plots near the edge (Torres et al., 2023). Moreover, all our plots are less than 300 m from the edge, a range where edge effects can penetrate the forest and be easily detected (Lopes et al., 2009; Chaplin-Kramer et al., 2015). However, we found that the forest cover from 2002 within a 2000 m buffer was negatively related to the common species NDB. Therefore, biomass growth was greater in plots surrounded by a lower proportion of forests, and consequently under a greater edge effect. This result supports studies that found that small fragments and forest areas along the edge have a higher proportion of abundant early-successional species, and that common tropical species tend to grow faster when under edge effect (Thier and Wesenberg, 2016; Albiero-Júnior et al., 2021).

Unexpectedly, fragment size was inversely related to the rare species NDA, especially for the GUA fragment (Figure 5). Larger fragments tend to have recruitment rates higher than mortality (Batista et al., 2016), and greater species rarity (Lovejoy et al., 1986; Silva et al., 2024; Gonçalves-Souza et al., 2025). However, as our plots were installed in a strip of greater edge influence, where wind effects and fluctuations in sunlight, temperature, and humidity on the fragment margins could favor the establishment of common (pioneer) species to the detriment of rare (late-successional) ones regardless of their total size (Laurance, 2004; Tabarelli et al., 2010; Lima et al., 2015; Magnago et al., 2015; Thier and Wesenberg, 2016). A first-order effect of higher elevation, coupled with lower temperatures, could also explain this unexpected result (Vieira et al., 2011; Alberton et al., 2023; Basnet et al., 2025), as the forests of GUA and SJO are the highest and with the lowest temperatures (Figure 5, 6), which could lead to higher mortality or reduced recruitment of these species (Pulla et al., 2021; Treml et al., 2022). The coupling between elevation and temperature may also explain the positive effect of elevation on the rare species NDA and negatively with the common species NDB, suggesting that the altitudinal gradient operates different filters among species, regulating both the rare species establishment and the growth of common ones. This divergent effect was demonstrated in our areas in an independent SAC evaluation, which revealed that there is optimal relative growth at intermediate elevations (Torres et al., 2023), where species of both SACs can develop without a negative influence of high elevations for common species and low elevations for rare ones. Elevation may also mediate forest dynamics through its effect on taxonomic and functional diversity (Rodrigues et al., 2020; Jiang et al., 2023).

Our results revealed a pronounced effect of taxonomic and functional diversity variables in the diversity models to explain forest dynamics, especially NDA (Figure 4, 5). This can be explained by the partitioned effects of biotic and abiotic variables, which contribute with different intensities and pathways to biomass production in tropical forest ecosystems, with biotic factors dominating biomass dynamics in the Atlantic Rainforest (Alberton et al., 2023). Therefore, the patterns of mortality, recruitment, and growth of survivors differ among forests with different intrinsic structural and diversity characteristics, leading to heterogeneous fluctuations in abundance and biomass (Vibrans et al., 2022).

Species diversity (H') was positively related to the NDA of common species and to that of rare species when LUH was selective logging, but negatively when the plots experienced deforestation or agricultural use (Figure 5). This is a contrasting pattern, as there is a positive relationship between species richness, and consequently diversity, and the number of rare species in tropical forests (Hordijk et al., 2024). Therefore, little or no effect of taxonomic diversity on the common species dynamics would be expected, and a higher rare species NDA in plots with higher H', which was observed only in plots subjected to selective logging. Thus, secondary Atlantic Rainforest forests originating from more intensive LUH present successional pathways shaped by these land uses and by their impact on population dynamics, which tends to limit the rare species recruitment. Among the functional diversity metrics, there were antagonistic effects on the common species NDA, with FRic affecting it negatively and RaoQ being positively related. However, these variables presented a decoupled effect on the common species NDB, with a positive effect only from RaoQ (Figure 6). We noted a divergent trend between these indices in our areas, in which areas that increased in RaoQ declined in FRic, and vice versa (Figure S5), justifying the opposing effect on the common species NDA. This contrasting effect is an indication of functional redundancy (Sui et al., 2025), which occurs in more mature communities with more stable environmental conditions and greater resource availability, which tend to present greater population uniformity among functional profiles, reducing the existence of rare profiles that drive RaoQ (Ying Li et al., 2022). However, a more pronounced effect of these variables would be expected for the rare species NDA, since there is a positive relationship between the increase in rarity and functional diversity (Sui et al., 2025). Finally, CWM-EG was the only significant variable for the intermediate species NDA, with a negative effect only for deforestation LUH and consistently positive for agricultural production (Figure 5). This shows that species with intermediate abundance belong to later-successional groups and suggests that the species recruitment in this group is higher when the forest grows from a more intensive LUH. Furthermore, we previously found a strong relationship between higher CWM-EG and less intense drought periods, especially for previously farmed areas (Chapter 1), pointing to CWM-EG as an intermediate factor coupled to the climatic constraints imposed by Precp-Dry for the intermediate species NDB (Alberton et al., 2023). This reinforces the increasing importance of functional diversity, as well as phylogenetic diversity, as drivers promoting biomass during forest maturation (Yuan et al., 2016).

We previously revealed that taxonomic diversity (species richness and exponential of Shannon-Weaver index) in our areas is higher in areas with lower mean annual temperature (Chapter 1). However, the common species NDB is positively related to temperature (Figure 6). Studies suggest that biomass production is lower in colder areas due to reduced photosynthetic activity (Malhi et al., 2010; Vieira et al., 2011; Malhi et al., 2017). This effect may be more pronounced in common pioneer species, such as *Anadenanthera peregrina* (L.) Speg. and *Apuleia leiocarpa* (Vogel) J.F.Macbr. (Fabaceae), and *Toulicia stans* (Schott) Radlk. (Sapindaceae), which were highly representative in AGB in our fragments (Table S2), as they exhibit higher metabolic rates due to rapid growth (Huante and Rincón, 1997; Li et al., 2023), and reductions in photosynthesis result in significant declines in biomass accumulation (Oberbauer et al., 1985; Kitao et al., 2021).

Common species showed a negative response (decline in abundance) under drought events, while rare species were not clearly affected, and the recruitment and stem growth rates were maintained (Figure 3). Although the increase in annual precipitation reduces the magnitude of species dominance worldwide (Hordijk et al., 2024). Specifically, we observed that among the climatic variables, Precp-Dry was positively correlated with the common species NDA and NDB for all LUH (Figure 6), demonstrating that water stress during the dry season regulates both recruitment and growth of these species. We also observed higher rare species NDB at more negative values of CWD-1, except when LUH was agriculture, suggesting that rare species in less disturbed forests are less sensitive to water deficit. Although the biotic effects of diversity are active forest dynamics modulators, abiotic factors also regulate biomass through first-order effects on tree growth, and second-order effects on diversity (Chapter 1) and leaf phenology (Alberton et al., 2023). In this context, hydroclimatic control regulates the structural characteristics and diversity of the forest, and ultimately, the productivity of these tropical ecosystems (Guan et al., 2015; Alberton et al., 2023). Therefore, certain species and seasonal tropical formations reduce their productivity below a precipitation threshold that determines greater water stress during the dry season (Guan et al., 2015; Vico et al., 2015; Wagner et al., 2016; Vico et al., 2017; Alberton et al., 2023). Moreover, high mortality rates of pioneer species have already been associated with extreme drought events in Seasonal Semideciduous Forests of the Atlantic Rainforest (da-Rocha et al., 2020). Therefore, the response of common species (mostly pioneers) NDA and NDB to the intensity of the dry season may be amplified by the acquisitive strategy of most species in this group, which have high photosynthetic capacity (dos-Santos et al., 2019), and therefore require high water availability for their rapid growth, being less resistant to water scarcity (Pulla et al., 2021; Li et al., 2023).

Among the diversity metrics, CWM-Dec was the only one that did not show a significant relationship with the dynamics of any SAC. This is contrary to what could be expected, since hydrological regimes are associated with the leaf senescence and renewal dynamics in different forest ecosystems in the tropics, which in turn are related to biomass development and growth (Alberton et al., 2023). Although our approach does not include monitoring of phenological cycles and their synchrony, CWM-Dec is a weighted representation of species deciduousness. Thus, this metric could be related to NDB, since evergreen species tend to show greater net gain of biomass and carbon (Vico et al., 2017), and are more concentrated in the interior of the fragments (Tabarelli et al., 2010). However, the balance between species with different phenological strategies in the community composition is essential to ensure more stable biomass stocks, since evergreen species are more sensitive to longer dry seasons (Vico et al., 2015).

We observed contrasting patterns in the influence of edaphic variables on forest dynamics across different LUH (Figures 5, 6), confirming the impact of anthropogenic activities on soil characteristics and forest succession after abandonment (Pinho et al., 2018; Bentsi-Enchill et al., 2022; Wells and Sawyer, 2022; Baláš et al., 2024; Fachin et al., 2024), and the effect of soil on the structuring of tree abundance and rarity (Madrigal-González et al., 2023; Hordijk et al., 2024). Our results indicate a positive influence of remaining phosphorus (P-Rem) on the rare species NDA and NDB, especially when LUH was selective logging. This element has been documented as one of the most related to photosynthetic activity of late-successional species in the Atlantic Rainforest, and particularly important for their growth during the dry season (dos-Santos et al., 2019). This coupled climate–soil effect is especially relevant during intense drought events, when younger trees with less developed root systems have reduced nutrient uptake capacity (He et al., 2024), leading to a decrease in tree density, which can ensure milder changes in soil acidity and moisture (Lin et al., 2023). Our areas showed an increase in taxonomic diversity parameters and in CWM-EG over the 20 years of monitoring (Figure S5), evidencing that more mature forests present a higher population proportion of late-successional species, caused by the turnover between pioneer and late species, through the arrival of new late species with small populations (rare) (de-Figueiredo et al., 2015; de-Toledo et al., 2025). This establishes a trend in which rare species belong to later successional groups. Therefore, the positive effect of P on the rare species dynamics in forests that experienced selective logging is explained by the more advanced successional stage provided by less intensive LUH.

The rare species NDB (Figure 6) was also related to the edaphic factors of pH and total acidity ($H^{+}+Al^{3+}$), and to fertility (SB), which was also related to the intermediate species NDA (Figure 5). Soil acidity is often associated with limitations in the forest species establishment (Šantrůčková et al., 2019; Wells and Sawyer, 2022; da-Rocha et al., 2024). Furthermore, acidity related to aluminum hampers nutrient absorption by plants (Bonomelli and Artacho, 2021), which may increase the fall of certain tree species in the Atlantic Rainforest (de-Lima et al., 2025). This justifies the relationship between acidity and abundance dynamics, but especially biomass, indicating that even in nutritionally adequate soils, unsuitable pH and Al values can limit growth. Soil fertility is a modulating factor of tree species abundance on a global scale (Madrigal-González et al., 2023), and is linked to a greater presence of certain rare species (Yao et al., 2025) and higher turnover of individuals, which also drives higher diversity (Pos et al., 2019), justifying the relationships of SB with the intermediate and rare species dynamics. This effect was notably stronger (greater slope) for the intermediate species NDB and for the rare species NDA when LUH was agriculture, and milder when it was selective logging. This suggests that the effect of fertility on dynamics is lower in forests and species that are later successional (often rare), since vegetation in more mature forests shows weaker responses to fertility (van-Breugel et al., 2019). However, the pioneer species establishment and growth, which are often common (Tabarelli et al., 2010), are more strongly limited by water availability (Pulla et al., 2021; Li et al., 2023), justifying the strong positive effect of SB on the intermediate species NDA and the rare species NDB in plots with LUH of agriculture, and milder effects in more mature plots subjected to selective logging.

Our study fills gaps regarding which environmental factors have different effects on the dynamics of species groups (common, intermediate, and rare), demonstrating distinct environmental requirements and different responses to historical disturbances. In particular, rare species were able to maintain population growth even under the influence of intense drought events, while common species declined, although all increased biomass. Therefore, the management of forest fragments must consider these different response patterns when proposing environmental policies and implementing management and conservation practices, especially aiming to protect rare species, which contribute proportionally more to biomass. Adopting this approach is essential in a scenario of recent projections that foresee drastic reductions in precipitation in the Atlantic Rainforest and an increase in water scarcity (Costa et al., 2024). These more severe first-order future impacts are imposed by changes in the hydrological regime, and second-order impacts by alterations in the forest's intrinsic diversity patterns (Alberton et al., 2023). The influence of these climatic factors associated with anthropogenic uses is expected to intensify processes of species migration and local extinction in the future, through the species population dynamics across different abundance classes (Fridley, 2001). Furthermore, the influence of LUH may lead to more severe fragmentation patterns that compromise the genetic populations dynamics, which also modulates community dynamics in the long term (Dauby et al., 2014; Freire et al., 2019). Our results also show that forests with lower diversity are more susceptible to environmental pressures, whereas more diverse forests can maintain biomass growth even in situations of reduced abundance. This reinforces the role of hyperdiverse forests in ecosystem stability, the global carbon cycle, and in coping with anthropogenic climate change (Ruiz-Jaen and Potvin, 2010; Aguirre-Gutiérrez et al., 2020; Schnabel et al., 2025; Steinparzer et al., 2025), in addition to the need to maintain taxonomic and functional diversity as regulatory factors of forest growth and dynamics (Ruiz-Jaen and Potvin, 2010; Cadotte et al., 2011; Madrigal-González et al., 2020; Li et al., 2024).

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Declaration of generative AI and AI-assisted technologies in the writing process

During the preparation of this work the author(s) used ChatGPT (version GPT-4.5) to translate, review the text, search for synonyms and improve the fluidity of writing. After using this tool/service, the authors reviewed and edited the content as needed and took full responsibility for the content of the published article.

Data availability

Data will be made available on request.

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Supplementary Matirials

Table S1. Priority order of databases for obtaining basic wood density (BWD) values.

Priority for obtaining BWD values	Reference
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3	Oliveira, G.M.V., 2014. Densidade da madeira em Minas Gerais: amostragem, espacialização e relação com variáveis ambientais. 125 f. Tese (Doutorado em Ciências Florestais) - Universidade Federal de Lavras, Lavras. http://repositorio.ufla.br/jspui/handle/1/4880.
4	Rolim, S.G., Piotto, D., 2018. Silvicultura e tecnologia de espécies da Mata Atlântica. Rona Editora.
5	Lima, J.A. d.-S., 2009. Biomassa arbórea e estoques de nutrientes em fragmentos florestais da APA Rio São João: O efeito da fragmentação sobre a Mata Atlântica da Baixada Litorânea Fluminense. 180 f. Tese (Doutorado em Ecologia e Recursos Naturais) - Universidade Estadual do Norte Fluminense, Campos dos Goytacazes.
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12	Lorenzi, H., 2009. Árvores Brasileiras: Manual de identificação cultivo de plantas arbóreas nativas do Brasil, v. 3, 1th ed. Instituto Plantarum, Nova Odessa.
13	Carvalho, P.E.R., 2003. Espécies arbóreas brasileiras, v. 1. Brasília - DF: Embrapa Informação Tecnológica; Colombo - PR: Embrapa Florestas.
14	Zanne, A.E., Lopez-Gonzalez, G., Coomes, D.A., Ilic, J., Jansen, S., Lewis, S.L., Miller, R.B., Swenson, N.G., Wiemann, M.C., Chave, J., 2009. Global wood density database. https://doi.org/10.5061/DRYAD.234.
14.a	Species mean for values from Tropical South America in Zanne et al., 2009.
14.b	Species mean for all values in Zanne et al., 2009.
14.c	Genus mean of at least three values from previous literature.
14.d	Genus mean for values from Tropical South America in Zanne et al., 2009.
14.e	Genus mean of less than three values from previous literature.
14.f	Value of a congener cited in some previous literature when the genus has no other species on the list or for Tropical South America in Zanne et al., 2009.
14.g	Genus mean for all values in Zanne et al., 2009.
14.h	Family mean of at least three values from the literature.

14.i	Family mean for tropical South American values in Zanne et al., 2009.
14.j	Family mean of less than three values from literature.
14.k	Family mean for all values in Zanne et al., 2009.
14.l	Mean for all previous values.

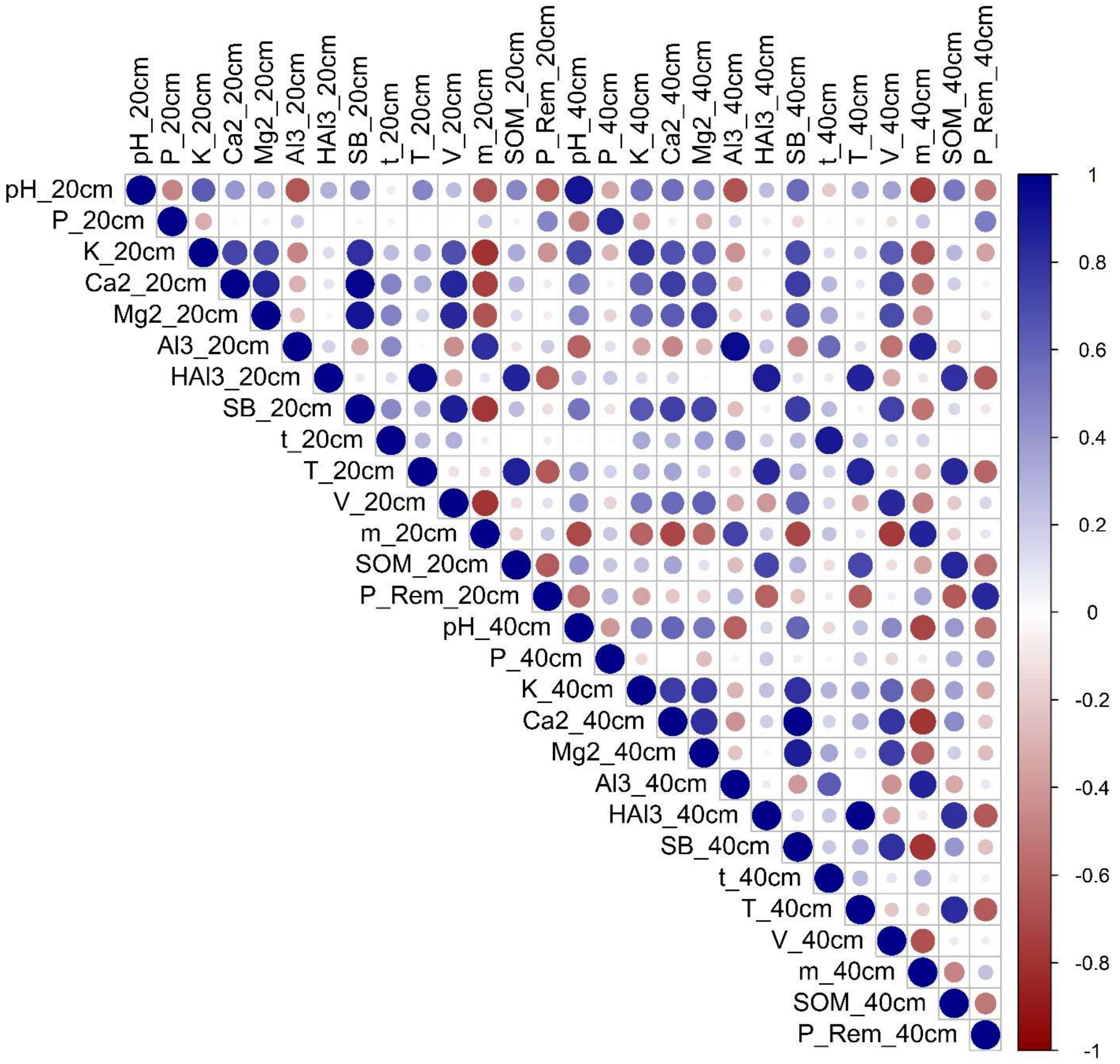

Figure S1. Spearman correlations between soil variables at different depths (0–20 cm and 20–40 cm); and richness. Dark blue circles indicate positive correlations, and dark red circles indicate negative correlations. The size of the circle indicates the strength of the correlation. For the abbreviation of soil variables see Material and Methods.

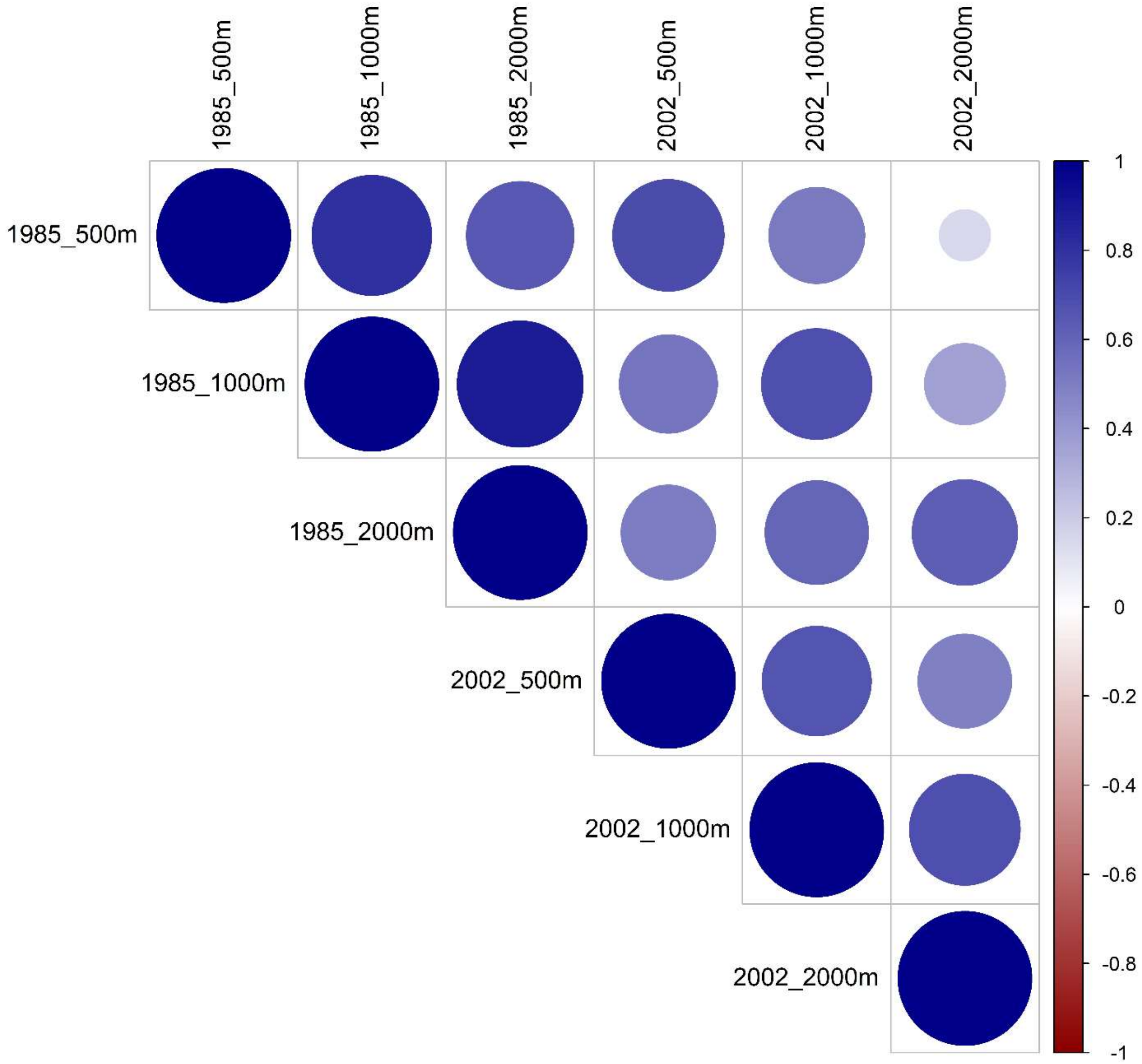

Figure S2. Spearman correlations between forest cover area using radii of 500, 1000, and 2000 m, for the year 1985, 2002 and 2017; and richness. Dark blue circles indicate positive correlations, and dark red circles indicate negative correlations. The size of the circle indicates the strength of the correlation.

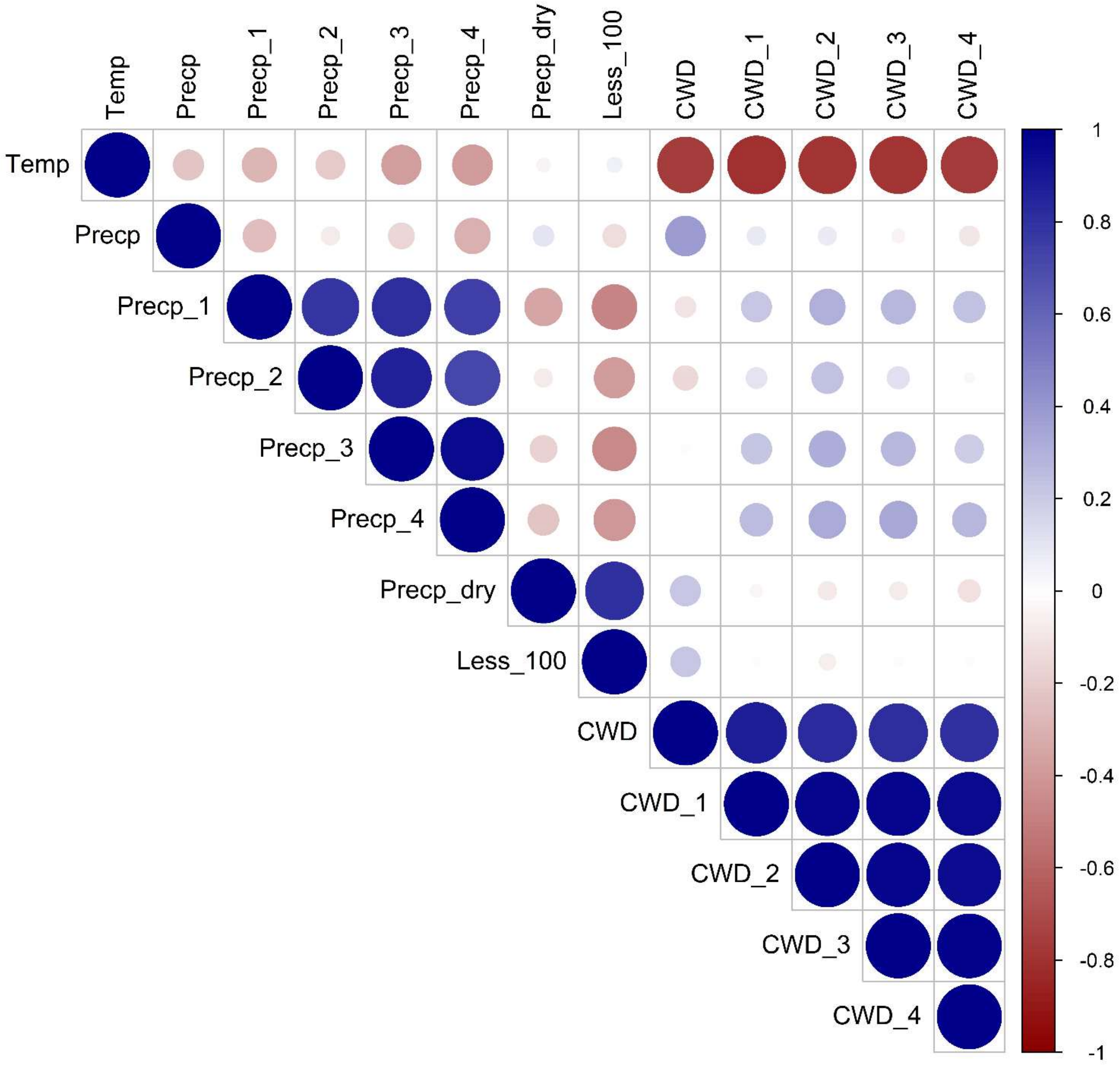

Figure S3. Spearman correlations between climatic variables of temperature, precipitation, and climatic water deficit (CWD); and richness. Blue circles indicate positive correlations, and red circles indicate negative correlations. The size of the circle indicates the strength of the correlation. For the abbreviation of soil variables see Material and Methods.

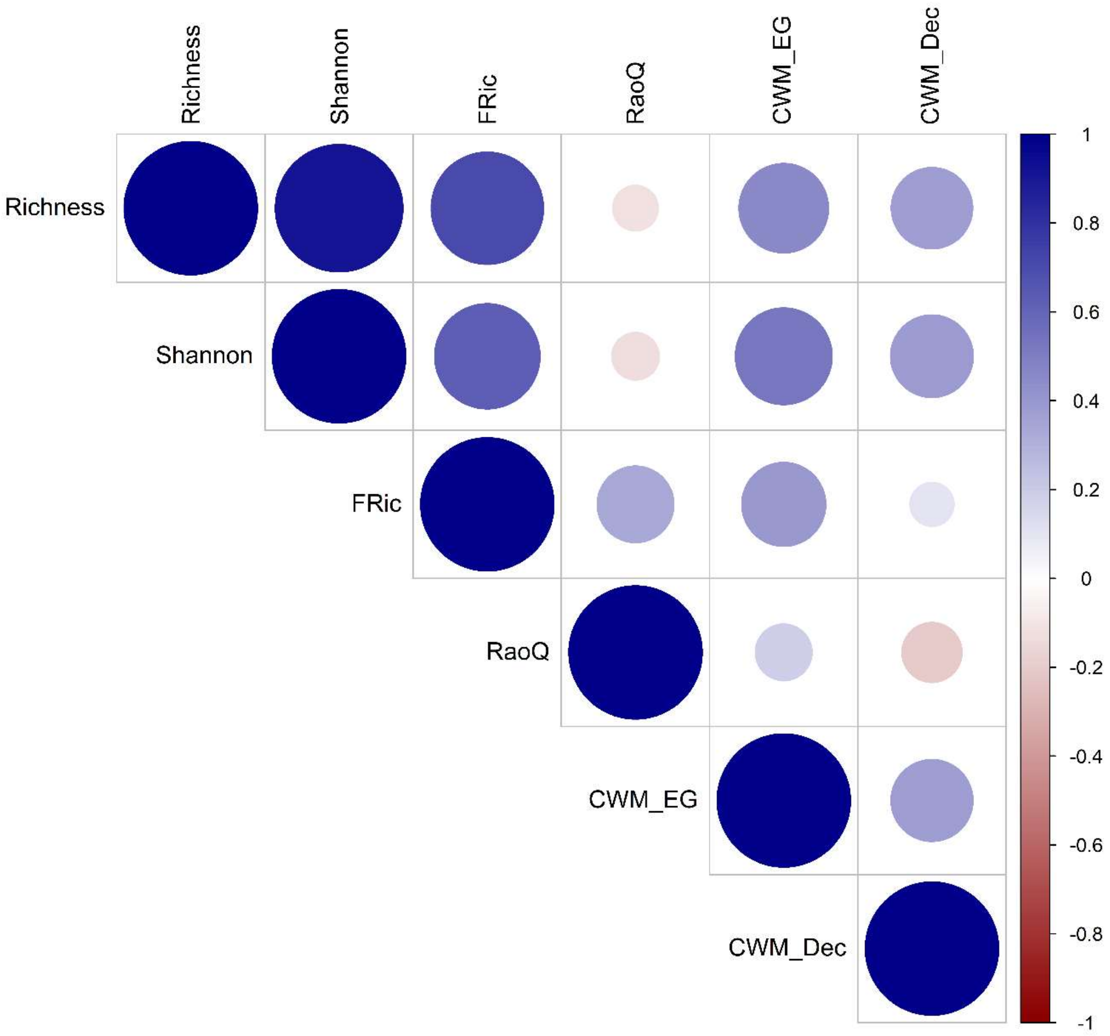

Figure S4. Spearman correlations between taxonomic (Richness and Shannon) and functional (FRic, RaoQ, CWM-EG and CWM-Dec) diversity indices. Blue circles indicate positive correlations, and red circles indicate negative correlations. The size of the circle indicates the strength of the correlation. For the abbreviation of soil variables see Material and Methods.

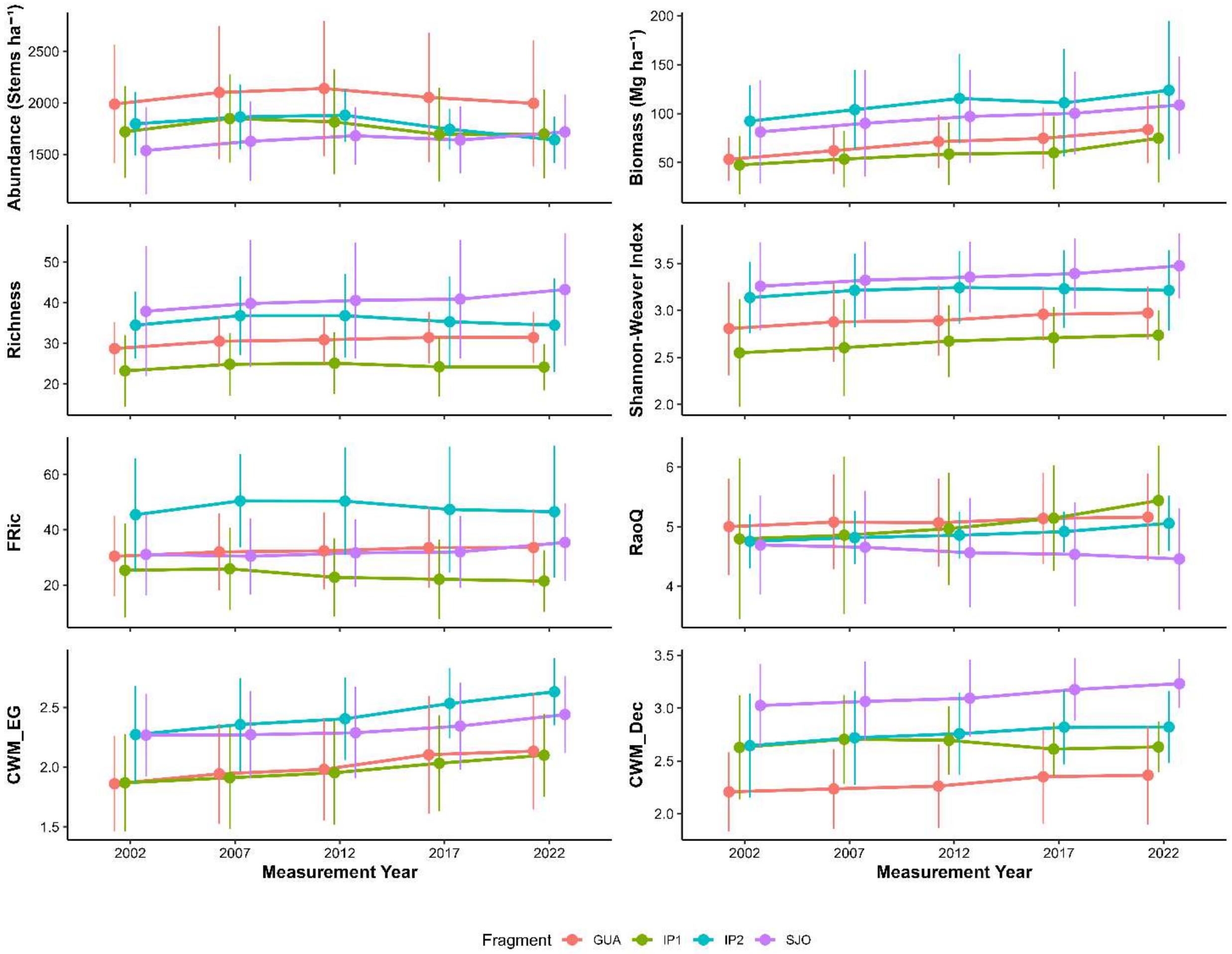

Figure S5. Abundance and biomass net dynamics of different rarity groups during 20 years of monitoring (2002-2022), in four Atlantic Rainforest fragments in Minas Gerais, southeastern Brazil.

Table S2. Abundance dynamics of tree species, during 20 years of monitoring (2002/2007/2012/2017/2022), in four Atlantic Rainforest fragments in Minas Gerais, southeastern Brazil (GUA – Cachoeira das Pombas; IP1 – Ipaba Mata1; IP2 – Ipaba Mata2; SJO – São José).

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Achariaceae Harms	**2**	**3**	**3**	**3**	**3**	**1**	**1**	**2**	**1**	**3**	**1**	**1**	**1**	**1**	**1**	**2**	**2**	**1**	**1**	**1**
Carpotroche brasiliensis (Raddi) A Gray	2	3	3	3	3	1	1	2	1	3	1	1	1	1	1	2	2	1	1	1
Anacardiaceae R.Br.	**9**	**9**	**11**	**12**	**12**	**55**	**58**	**60**	**59**	**73**	**10**	**9**	**9**	**5**	**5**	**17**	**17**	**16**	**17**	**13**
Astronium fraxinifolium Schott	5	5	5	5	5	36	39	44	46	56	5	5	5	4	4					
Astronium graveolens Jacq.						1	1	1	1	1										
Tapirira guianensis Aubl.	3	3	2	2	2	17	17	14	11	15	5	4	4	1	1	17	17	16	17	13
Tapirira obtusa (Benth.) J.D.Mitch.	1	1	4	5	5															
Thyrsodium spruceanum Benth.						1	1	1	1	1										
Annonaceae Juss.	**78**	**94**	**96**	**95**	**104**	**196**	**250**	**208**	**108**	**74**	**19**	**18**	**17**	**12**	**11**	**57**	**59**	**60**	**64**	**66**
Annona cacans Warm.	1	1	1	1	1						2	2	2	1	1	18	16	15	15	11
Annona dolabripetala Raddi	1	2	2	2	2	5	5	5	5	4						7	8	7	4	3
Annona sylvatica A.St.-Hil.	7	7	8	7	7	16	12	7	2	1										
Guatteria australis A.St.-Hil.												1	1	1	1	1	1	1	5	7
Guatteria campestris R.E.Fr.						2	2	3	5	3	15	13	12	8	6	7	7	7	7	6
Guatteria ferruginea A.St.-Hil.																2	2	3	3	4
Guatteria latifolia (Mart.) R.E. Fr.						1	1	1												
Guatteria oligocarpa Mart.																2	2	2	2	5
Guatteria rupestris Mello-Silva & Pirani	2	3	3	3	3															
Guatteria sellowiana Schltdl.	66	79	79	78	83											6	6	6	7	8
Guatteria villosissima A.St.-Hil.	1	2	3	3	7											7	7	8	8	8
Xylopia aromatica (Lam.) Mart.						35	58	64	73	50										
Xylopia brasiliensis Spreng.																5	8	9	10	10
Xylopia frutescens Aubl.																1	1	1	2	3
Xylopia sericea A.St.-Hil.				1	1	137	172	128	23	16	2	2	2	2	3	1	1	1	1	1
Apocynaceae Juss.	**28**	**33**	**34**	**31**	**30**	**9**	**10**	**8**	**8**	**8**	**3**	**3**	**3**	**4**	**4**	**1**	**1**	**1**	**1**	**3**
Aspidosperma compactinervium Kuhlm.	7	10	11	11	11															
Aspidosperma Mart. & Zucc. 01-GUA	1	1	1	1	1															
Aspidosperma olivaceum Müll.Arg.	8	9	9	9	9															

Botanical Family/ Species	Fragment/Measurement Year																			
	GUA					IP1					IP2					SJO				
	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22
Aspidosperma parvifolium A.DC.						5	5	4	4	4										1
Aspidosperma cf. *spruceanum* Benth. ex Müll.Arg.																1	1	1	1	1
Aspidosperma subincanum Mart.																				1
Himatanthus bracteatus (A. DC.) Woodson	5	5	5	2	2	3	4	3	3	3	3	3	3	4	4					
Tabernaemontana laeta Mart.						1	1	1	1	1										
Tabernaemontana solanifolia A.DC.	7	8	8	8	7															
Aquifoliaceae Bercht. & J.Presl	**16**	**17**	**14**	**16**	**16**	**2**	**2**	**2**	**2**	**2**	**1**	**1**				**13**	**10**	**9**	**10**	**10**
Ilex affinis Gardner																1				
Ilex cerasifolia Reissek	2	2	2	3	3											2	2	1	2	2
Ilex conocarpa Reissek	6	6	3	3	3															
Ilex dumosa Reissek											1	1				1				
Ilex integerrima (Vell.) Reissek	2	3	3	3	3											5	5	5	5	5
Ilex L. 01-GUA/SAB	6	6	6	7	7															
Ilex L. 01-IP1						2	2	2	2	2										
Ilex paraguariensis A.St.-Hil.																1	1	1	1	1
Ilex theezans Mart. ex Reissek																3	2	2	2	2
Araliaceae Juss.	**4**	**4**	**4**	**5**	**4**	**3**	**3**	**6**	**6**	**9**	**2**	**3**	**3**	**5**	**5**	**1**	**2**	**2**	**2**	**2**
Dendropanax cuneatus (DC.) Decne. & Planch.											1									
Didymopanax morototoni (Aubl.) Decne. & Planch.						3	3	6	6	9	1	3	3	5	5					
Didymopanax vinosus (Cham. & Schltdl.) Marchal	4	4	4	5	4											1	2	2	2	2
Arecaceae Schultz Sch.	**23**	**27**	**33**	**38**	**39**											**5**	**6**	**5**	**4**	**5**
Euterpe edulis Mart.	21	27	33	38	38											1	1	1	2	3
Geonoma schottiana Mart.																4	5	4	2	2
Syagrus romanzoffiana (Cham.) Glassman	2				1															
Asteraceae Bercht. & J.Presl	**171**	**165**	**145**	**29**	**23**	**11**	**13**	**6**	**4**	**6**						**11**	**17**	**20**	**17**	**19**
Austrocritonia angulicaulis (Sch.Bip. ex Baker) R.M.King & H.Rob.																		2	2	2
Eremanthus erythropappus (DC.) MacLeish	165	159	140	26	21															
Moquiniastrum polymorphum (Less.) G. Sancho	1	1	1																	
Piptocarpha macropoda (DC.) Baker	4	4	3	2	2	11	13	6	4	6						5	10	11	8	10

Botanical Family/ Species	Fragment/Measurement Year																			
	GUA					IP1					IP2					SJO				
	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22
Vernonanthura divaricata (Spreng.) H.Rob.	1	1	1	1												6	7	7	7	7
Bignoniaceae Juss.	**55**	**56**	**60**	**69**	**71**	**4**	**5**	**6**	**7**	**8**	**4**	**4**	**4**	**4**	**4**	**20**	**20**	**22**	**24**	**25**
Adenocalymma macrophyllum (Cham.) DC.									1	1										
Cybistax antisyphilitica (Mart.) Mart.	3	3	3	4	4															
Handroanthus chrysotrichus (Mart. ex DC.) Mattos		1	1			1	1	1	1	1						1	1	1	1	1
Handroanthus serratifolius (Vahl) S.Grose						1	1	1	1	1	3	3	3	3	3	3	3	3	3	3
Jacaranda macrantha Cham.	44	44	47	54	55											10	8	9	9	10
Sparattosperma leucanthum (Vell.) K.Schum.						2	3	3	3	4	1	1	1	1	1	6	8	9	11	11
Zeyheria tuberculosa (Vell.) Bureau ex Verl.	8	8	9	11	12			1	1	1										
Bixaceae Kunth						**2**	**3**	**2**	**2**	**2**	**2**	**2**	**3**	**2**	**1**					
Bixa atlantica Antar & Sano						2	3	2	2	2	2	2	3	2	1					
Burseraceae Kunth	**5**	**6**	**6**	**8**	**7**						**4**	**4**	**3**	**4**	**6**	**5**	**5**	**5**	**5**	**5**
Protium atlanticum (Daly) Byng & Christenh.											2	2	2	3	3					
Protium heptaphyllum (Aubl.) Marchand											2	2	1	1	3	3	3	3	3	3
Protium warmingianum Marchand	4	5	5	7	7															
Trattinnickia ferruginea Kuhlm.																2	2	2	2	2
Trattinnickia Willd. 01-GUA	1	1	1	1																
Calophyllaceae J.Agardh	**3**	**3**	**4**	**3**	**2**											**2**	**2**	**2**	**3**	**3**
Kielmeyera altissima Saddi	3	3	3	3	2											2	2	2	2	2
Kielmeyera cf. *membranacea* Casar.																			1	1
Kielmeyera variabilis Mart. & Zucc.			1																	
Cardiopteridaceae Blume																**1**	**3**	**3**	**4**	**6**
Citronella paniculata (Mart.) R.A.Howard																1	3	3	4	6
Caryocaraceae Voigt																**4**	**3**	**3**	**3**	**3**
Caryocar edule Casar.																4	3	3	3	3
Celastraceae R.Br.	**28**	**35**	**38**	**46**	**47**			**1**	**1**	**2**	**2**	**4**	**3**	**5**	**5**	**1**	**1**	**1**	**3**	**3**
Cheiloclinium cognatum (Miers) A.C.Sm.											1	1		1	2					
Monteverdia A. Rich. 01-IP1								1	1	1										
Monteverdia evonymoides (Reissek) Biral	4	4	4	5	5															
Monteverdia floribunda (Reissek) Biral	12	17	19	23	24															

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Monteverdia gonoclada (Mart.) Biral	12	14	15	18	18					1	1	3	3	4	3	1	1	1	1	1
Salacia elliptica (Mart.) G. Don																			2	2
Chrysobalanaceae R.Br.	**8**	**7**	**7**	**6**	**4**	**26**	**32**	**37**	**40**	**47**	**37**	**37**	**37**	**37**	**36**	**15**	**17**	**17**	**13**	**15**
Hirtella glandulosa Spreng.						9	12	11	10	10	12	8	7	5	5					
Hirtella gracilipes (Hook.f.) Prance						3	3	3	3	4										
Hirtella hebeclada Moric. ex DC.																7	8	8	7	8
Hirtella L. 01-GUA	2	2	2	1																
Leptobalanus octandrus (Hoffmanns. ex Roem. & Schult.) Sothers & Prance						1	1	3	3	3	3	3	3	3	3	3	3	3		
Licania Aubl. 01-IP1						1	1													
Licania cf. *cymosa* Fritsch																				1
Licania hypoleuca Benth.																				1
Licania kunthiana Hook.f.	3	4	4	4	4	12	15	20	24	30	22	26	27	29	28	1	2	2	2	2
Licania spicata Hook.f.	3	1	1	1												2	2	2	2	1
Parinari brasiliensis (Schott) Hook.f.																2	2	2	2	2
Clethraceae Klotzsch	**3**	**4**	**3**	**2**	**1**											**21**	**22**	**26**	**26**	**25**
Clethra scabra Pers.	3	4	3	2	1											21	22	26	26	25
Clusiaceae Lindl.											**1**					**1**	**1**	**1**	**1**	**1**
Garcinia gardneriana (Planch. & Triana) Zappi											1					1	1	1	1	1
Combretaceae R.Br.	**1**	**2**	**2**	**2**	**2**											**6**	**8**	**7**	**6**	**6**
Terminalia glabrescens Mart.																3	4	3	2	2
Terminalia cf. *januariensis* DC.		1	1	1	1											3	4	4	4	4
Terminalia L. 01-GUA	1	1	1	1	1															
Cordiaceae R.Br. ex Dumort.	**9**	**9**	**9**	**9**	**8**	**33**	**30**	**28**	**20**	**13**	**9**	**9**	**9**	**7**	**5**	**2**	**2**	**1**	**2**	**2**
Cordia fusca M.Stapf								1	1	1										
Cordia L. 01-IP1						2	1													
Cordia L. 01-SJO																1	1			
Cordia sellowiana Cham.	7	7	7	7	6	30	28	26	19	12	9	8	8	6	4	1	1	1	2	2
Cordia silvestris Fresen.												1	1	1	1					
Cordia trichoclada DC.						1	1	1												

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Cordia trichotoma (Vell.) Arráb. ex Steud.	2	2	2	2	2															
Cunoniaceae R.Br.																**2**	**2**	**3**	**3**	**3**
Lamanonia ternata Vell.																2	2	3	3	3
Cyatheaceae Kaulf.																				**2**
Cyathea corcovadensis (Raddi) Domin																				2
Dicksoniaceae M.R.Schomb.																**1**	**1**	**1**	**1**	**2**
Dicksonia sellowiana Hook.																1	1	1	1	2
Ebenaceae Gürke	**9**	**9**	**9**	**10**	**8**															
Diospyros lasiocalyx (Mart.) B.Walln.	9	9	9	10	8															
Elaeocarpaceae Juss.						**3**	**3**	**2**	**2**	**2**	**6**	**6**	**6**	**6**	**5**	**8**	**7**	**7**	**8**	**8**
Sloanea guianensis (Aubl.) Benth.																1	1	1	2	2
Sloanea hirsuta (Schott) Planch. ex Benth.											4	4	4	4	3	1	1	1	1	1
Sloanea retusa Uittien						3	3	2	2	2	2	2	2	2	2	6	5	5	5	5
Erythroxylaceae Kunth	**4**	**5**	**5**	**5**	**12**			**2**	**3**	**7**	**6**	**7**	**7**	**6**	**1**					
Erythroxylum citrifolium A.St.-Hil.					2															
Erythroxylum pelleterianum A.St.-Hil.	4	5	5	4	9			2	3	7	6	7	7	6	1					
Erythroxylum cf. *subracemosum* Turcz.				1	1															
Euphorbiaceae Juss.	**172**	**187**	**197**	**197**	**177**	**151**	**152**	**147**	**122**	**112**	**94**	**78**	**68**	**44**	**35**	**55**	**58**	**59**	**47**	**44**
Alchornea glandulosa Poepp. & Endl.						3	3	3	3							3	3	3	1	1
Alchornea triplinervia (Spreng.) Müll.Arg.																16	15	16	15	14
Aparisthmium cordatum (A.Juss.) Baill.											1					21	23	22	14	8
Croton floribundus Spreng.	16	16	13	13	11															
Croton urucurana Baill.	7	7	6	3	2						2									
Mabea fistulifera Mart.	144	159	173	176	159	148	148	143	118	111	86	73	65	39	29					
Manihot cf. *leptopoda* (Müll.Arg.) D.J.Rogers & Appan																		1	1	1
Maprounea guianensis Aubl.	4	4	4	4	4		1	1	1	1	2	2				15	17	17	16	20
Sapium glandulosum (L.) Morong	1	1	1	1	1															
Senefeldera verticillata (Vell.) Croizat											3	3	3	5	6					
Fabaceae Lindl.	**497**	**490**	**475**	**451**	**435**	**159**	**142**	**136**	**129**	**126**	**88**	**91**	**87**	**76**	**74**	**130**	**125**	**124**	**105**	**99**
Anadenanthera colubrina (Vell.) Brenan	4	4	4	2	2					2						26	20	18	11	10

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Anadenanthera peregrina (L.) Speg.	19	18	17	17	16														1	
Andira fraxinifolia Benth.											1	1	1	1	1	3	3	4	4	3
Andira ormosioides Benth.																2	2	2	2	2
Apuleia leiocarpa (Vogel) J.F.Macbr.	39	43	43	40	39	100	88	82	82	81	29	30	30	28	28	1	1	1		
Cassia ferruginea (Schrad.) Schrad. ex DC.	2	2	2	2	2	4	3	2	2	1	1	1				5	5	5	2	2
Copaifera langsdorffii Desf.	10	10	10	9	9	8	7	10	10	10						8	8	8	8	7
Dalbergia foliolosa Benth.	13	16	16	18	18											3	3	3	3	3
Dalbergia frutescens (Vell.) Britton	1	1	1	1	1															
Dalbergia nigra (Vell.) Allemão ex Benth.	11	11	10	7	8	4	4	1	1	1	8	7	5	5	7					
Deguelia hatschbachii A.M.G.Azevedo	4	4	4	5	4															
Diplotropis ferruginea Benth.	52	48	47	43	43															
Hymenolobium janeirense Kuhlm.	1	1	1	1	1											1	1	2	2	2
Inga capitata Desv.	1	1	1	2	2						4	4	4	4	3				1	1
Inga cylindrica (Vell.) Mart.	14	11	8	5	3	2	3	3	3	1	1	1	1	1		2	2	2	3	3
Inga flagelliformis (Vell.) Mart.						6	4	3	1	1										
Inga laurina (Sw.) Willd.												2	1							
Inga Mill. 01-GUA	1	1	1	1																
Inga Mill. 01-IP1						1														
Inga nobilis Willd.						7	7	7	6	2	1	1	1	1	1					
Inga striata Benth.	9	7	7	6	3															
Inga subnuda Salzm. ex Benth.	4	4	4	2	2											7	6	6	3	3
Inga thibaudiana DC.																	1	1	1	1
Inga vera Willd.						1	1	1												
Jupunba brachystachya (DC.) M.V.B.Soares et al.						2	2	2	2	1	1	1	1	1	1	2	2	2	2	2
Jupunba langsdorffii (Benth.) M.V.B.Soares et al.	2	2	2	2	2															
Leucochloron incuriale (Vell.) Barneby & J.W.Grimes	10	10	11	11	11															
Machaerium acutifolium Vogel											1	1	1	1	1				1	1
Machaerium brasiliense Vogel	65	68	71	70	64	3	4	5	4	4	5	4	4	2	2	2	2	1	1	1
Machaerium hirtum (Vell.) Stellfeld	17	16	7																	
Machaerium nyctitans (Vell.) Benth.	81	81	79	81	75											19	18	18	15	13

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Machaerium villosum Vogel	9	8	8	7	8															
Marlimorimia contorta (DC.) L.P.Queiroz & P.G.Ribeiro	1	1	1	1	1	7	7	8	7	9	23	23	23	19	18					
Melanoxylon brauna Schott	27	28	29	31	36	1	1	1	1	1	3	3	3	2	2	6	6	6	6	8
Muellera obtusa (Benth.) M.J. Silva & A.M.G. Azevedo	1	1	1	1	1															
Ormosia arborea (Vell.) Harms																1	2	2	2	2
Ormosia fastigiata Tul.											1	1	1							
Ormosia cf. *vicosana* Rudd																2	2	2	2	2
Piptadenia gonoacantha (Mart.) J.F.Macbr.	28	17	16	9	8											1	1	1	1	1
Plathymenia reticulata Benth.	13	15	14	14	14	1	1	1	1	1										
Platymiscium floribundum Vogel												1	1	1	1					
Platypodium elegans Vogel	7	9	8	6	7												1			
Pterocarpus rohrii Vahl											1	1	1	1	1					
Pterodon emarginatus Vogel																1	1	1		
Schizolobium parahyba (Vell.) Blake						1	1	1	1	1										
Senegalia polyphylla (DC.) Britton & Rose	5	4	3	3	2															
Senna macranthera (DC. ex Collad.) H.S.Irwin & Barneby					1											3	3	3	3	1
Senna multijuga (Rich.) H.S.Irwin & Barneby																6	6	6	5	4
Stryphnodendron polyphyllum Mart.				1	2	4	2	2	2	4					1	4	4	5	2	5
Swartzia apetala Raddi	18	21	26	31	31															
Swartzia flaemingii Raddi	4	4	4	4	4											1	1	1	1	1
Swartzia myrtifolia Sm.											1	2	2	2	2	2	2	2	2	2
Swartzia oblata R.S.Cowan																15	15	15	13	14
Swartzia cf. *pilulifera* Benth.						7	7	7	6	6	2	2	2	2	2					
Swartzia Schreb 01-IP2											1	1	1	1	1					
Tachigali friburgensis (Harms) L.G.Silva & H.C.Lima	6	5	5	5	4											4	4	4	4	4
Tachigali rugosa (Mart. ex Benth.) Zarucchi & Pipoly	18	18	14	13	11						4	4	4	4	2	3	3	3	4	1
Humiriaceae A.Juss.											**8**	**8**	**8**	**8**	**9**					
Humiriastrum dentatum (Casar.) Cuatrec.											8	8	8	8	9					
Hypericaceae Juss.	**17**	**18**	**17**	**14**	**15**	**2**	**2**	**1**			**2**	**2**	**1**			**29**	**26**	**22**	**11**	**10**
Vismia brasiliensis Choisy	17	18	17	14	15															

Botanical Family/ Species	Fragment/Measurement Year																			
	GUA					IP1					IP2					SJO				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Vismia guianensis (Aubl.) Choisy						2	2	1			2	2	1							
Vismia martiana Reichardt																29	26	22	11	10
Lacistemataceae Mart.	**8**	**10**	**11**	**13**	**11**	**8**	**9**	**10**	**11**	**10**	**1**	**1**	**3**	**3**	**3**	**2**	**2**	**2**	**5**	**7**
Lacistema pubescens Mart.	8	10	11	13	11	8	9	10	11	10	1	1	3	3	3	2	2	2	5	7
Lamiaceae Martinov	**64**	**57**	**49**	**38**	**30**	**11**	**10**	**10**	**6**	**6**	**4**	**4**	**4**	**5**	**5**	**3**	**5**	**5**	**5**	**4**
Aegiphila integrifolia (Jacq.) Moldenke						3	1	1									1			
Hyptidendron asperrimum (Spreng.) Harley	48	39	29	19	12											2	3	4	4	3
Vitex polygama Cham.	16	18	20	19	18															
Vitex sellowiana Cham.						8	9	9	6	6	4	4	4	5	5	1	1	1	1	1
Lauraceae Juss.	**41**	**45**	**47**	**49**	**47**	**143**	**154**	**159**	**165**	**168**	**53**	**60**	**62**	**57**	**45**	**52**	**58**	**59**	**63**	**67**
Aiouea saligna Meisn.																2	2	2	2	2
Aiouea tomentosa (Meisn.) R.Rohde	4	4	4	4	1															
Aniba firmula (Nees & Mart.) Mez						100	112	116	126	127	19	25	26	26	19					
Aniba cf. *intermedia* (Meisn.) Mez																2	2	2	2	2
Cinnamomum Schaeff. 01-SJO																1	1	1	1	1
Endlicheria glomerata Mez						1	1	1	1	1										
Endlicheria paniculata (Spreng.) J.F.Macbr.	2	2	2	1	1												1	1	1	1
Lauraceae Juss. 01-GUA	1	1	1	1	1															
Lauraceae Juss. 01-IP1						1														
Lauraceae Juss. 01-SJO																1	1	1		
Licaria armeniaca (Nees) Kosterm.																3	3	3	4	2
Licaria guianensis Aubl.	3	3	3	3	3											1	1	1	1	1
Mezilaurus crassiramea (Meisn.) Taub. ex Mez						4	4	4	4	4	2	2	2	2	1					
Nectandra lanceolata Nees											2	2	2	2	2					
Nectandra megapotamica (Spreng.) Mez																1	1	1	1	1
Nectandra oppositifolia Nees & Mart.	3	3	3	2	2	1					1								2	4
Nectandra Rol. ex Rottb. 01-IP1						2	1	2												
Nectandra Rol. ex Rottb. 01-IP2											5	3	3	1						
Nectandra Rol. ex Rottb. 01-SJO																1	1	1	1	1
Ocotea aciphylla (Nees & Mart.) Mez																2	3	3	3	5

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Ocotea Aubl. 01-IP1						6	6	4	3											
Ocotea Aubl. 02-IP1										1										
Ocotea citrosmoides (Nees) Mez						1	1	2	2	4										
Ocotea corymbosa (Meisn.) Mez	17	16	17	19	17						1	2	2	2	2	3	3	3	3	3
Ocotea dispersa (Nees & Mart.) Mez																2	2	2	2	2
Ocotea divaricata (Nees) Mez						1	1	1	1	1						5	6	6	7	7
Ocotea floribunda (Sw.) Mez		1	1	2	2															
Ocotea cf. *frondosa* (Meisn.) Mez																7	8	9	9	10
Ocotea glaziovii Mez																1	1	1	1	2
Ocotea glomerata (Nees) Mez																2	2	4	4	4
Ocotea indecora (Schott) Mez	1	1	1	1	1											1	1	1	1	1
Ocotea cf. *insignis* Mez																	1	1	1	1
Ocotea cf. *kostermanniana* Vattimo-Gil																1	1	1	1	1
Ocotea lanata (Nees & Mart.) Mez						2	1	1			4	3	3	1						
Ocotea cf. *longifolia* Kunth																1	1	1	1	1
Ocotea nectandrifolia Mez		1	1	1	2															
Ocotea nitida (Meisn.) Rohwer						5	8	9	11	13	5	7	8	8	6					
Ocotea nutans (Nees) Mez		1	1	1	1															
Ocotea odorifera (Vell.) Rohwer						10	10	10	9	7	13	14	14	14	13					
Ocotea cf. *percurrens* Vicent.						1	1	1	1	1										
Ocotea cf. *pomaderroides* (Meisn.) Mez																4	4	4	4	3
Ocotea puberula (Rich.) Nees	1	1	1	1	1															
Ocotea pulchella (Nees & Mart.) Mez	9	11	12	13	14															
Ocotea silvestris Vattimo-Gil					1															
Ocotea spectabilis (Meisn.) Mez										2										
Ocotea velloziana (Meisn.) Mez															1					
Persea cf. *caesia* Meisn.																1	1	1	1	1
Persea major (Meisn.) L.E.Kopp						5	5	5	5	5						3	3	3	3	3
Persea rigida Nees & Mart.																1	1	1	1	1
Phyllostemonodaphne geminiflora (Mez) Kosterm.											1	1	1			2	2	1		

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Rhodostemonodaphne anomala (Mez) Rohwer																1	2	1	1	1
Urbanodendron verrucosum (Nees) Mez						3	3	3	2	2		1	1	1	1	3	3	3	5	6
Lecythidaceae A.Rich.	**2**	**2**	**2**	**2**	**1**	**23**	**31**	**31**	**33**	**37**	**12**	**14**	**15**	**16**	**17**	**1**	**1**	**1**	**1**	**1**
Cariniana estrellensis (Raddi) Kuntze	2	2	2	2	1											1	1	1	1	1
Lecythis lurida (Miers) S.A.Mori						23	31	31	33	37	12	14	15	16	17					
Loganiaceae R.Br. ex Mart.																**1**	**1**	**1**	**1**	**2**
Strychnos gardneri A.DC.																1	1	1	1	2
Lythraceae J.St.-Hil.	**3**	**3**	**3**	**5**	**5**															
Lafoensia pacari A.St.-Hil.	3	3	3	5	5															
Malpighiaceae Juss.	**30**	**36**	**37**	**40**	**42**	**35**	**33**	**36**	**45**	**48**	**5**	**6**	**8**	**7**	**8**	**3**	**4**	**5**	**5**	**5**
Byrsonima cf. *alvimii* W.R.Anderson																1	1	2	2	2
Byrsonima cf. *cacaophila* W.R.Anderson																	1	1	1	1
Byrsonima cf. *chrysophylla* Kunth																1	1	1	1	1
Byrsonima crassifolia (L.) Kunth	26	30	31	35	36															
Byrsonima crispa A.Juss.						2	4	5	7	8	2	3	5	5	6					
Byrsonima ligustrifolia A.Juss.																1	1	1	1	1
Byrsonima Rich. ex Kunth 01-GUA		2	1																	
Byrsonima Rich. ex Kunth 01-IP2											1	1	1							
Byrsonima sericea DC.						33	29	31	38	40	2	2	2	2	2					
Heteropterys byrsonimifolia A.Juss.	4	4	5	5	6															
Malvaceae Juss.	**17**	**19**	**20**	**20**	**19**	**4**	**4**	**4**	**4**	**4**	**2**	**2**	**2**	**2**	**2**	**2**	**3**	**5**	**7**	**6**
Eriotheca macrophylla (K.Schum.) A.Robyns	6	8	8	8	8											1	1	1	1	1
Eriotheca pentaphylla (Vell.) A.Robyns																1	1	2	4	3
Eriotheca Schott & Endl. 01-IP1						2	2	2	2	2										
Eriotheca Schott & Endl. 01-IP2											1	1	1	1	1					
Luehea grandiflora Mart.	11	11	12	12	11	1	1	1	1	1	1	1	1	1	1					
Pseudobombax grandiflorum (Cav.) A.Robyns																	1	2	2	2
Sterculia apetala (Jacq.) H.Karst.						1	1	1	1	1										
Melastomataceae A.Juss.	**13**	**11**	**16**	**32**	**30**	**2**	**3**	**5**	**6**	**3**				**1**	**1**	**97**	**115**	**122**	**120**	**125**
Meriania claussenii (Naudin) Triana																22	21	20	18	15

Botanical Family/ Species	Fragment/Measurement Year																			
	GUA					IP1					IP2					SJO				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Meriania glabra (DC.) Triana																1	2	2	2	3
Miconia albicans (Sw.) Steud.									1											
Miconia brunnea DC.				1	1											1	1	1	1	1
Miconia buddlejoides Triana																13	20	21	18	19
Miconia cinnamomifolia (DC.) Naudin																4	4	5	7	7
Miconia corallina Spring																4	5	6	9	10
Miconia cubatanensis Hoehne																9	12	12	12	12
Miconia dodecandra Cogn.																8	9	13	18	21
Miconia holosericea (L.) DC.								1	1	1										
Miconia lepidota DC.																	3	7	6	7
Miconia ligustroides (DC.) Naudin				1	1															
Miconia petropolitana Cogn.																5	3	2	3	9
Miconia prasina (Sw.) DC.						1	1	1	1	1										
Miconia pusilliflora (DC.) Naudin						1	2	2	1									1		
Miconia Ruiz & Pav. 01-IP1									1	1										
Miconia sellowiana Naudin	12	10	15	29	27			1	1							3	4	4	2	2
Mouriri cf. *doriana* Saldanha ex Cogn.																1	2	2	2	2
Mouriri glazioviana Cogn.														1	1	2	2	2	2	2
Pleroma D.Don 01-SJO																4	4	3	1	
Pleroma estrellense (Raddi) P.J.F.Guim. & Michelang.				1	1															
Pleroma fissinervium Schrank et Mart. ex DC.																1	1	1	1	1
Pleroma francavillanum (Cogn.) P.J.F.Guim. & Michelang.																1	1	1	1	1
Pleroma granulosum (Desr.) D. Don	1	1	1													18	21	19	17	13
Meliaceae A.Juss.	**10**	**7**	**6**	**6**	**7**	**9**	**12**	**13**	**15**	**13**						**20**	**19**	**19**	**19**	**23**
Cabralea canjerana (Vell.) Mart.	1	1	1	1	1											5	5	5	6	6
Guarea F.Allam. ex L. 01-SJO																2	1			
Guarea guidonia (L.) Sleumer							1	1	1											
Guarea macrophylla Vahl						4	4	3	4	4										
Guarea pendula R.S.Ramalho, A.L. Pinheiro & T.D.Penn.						3	4	4	4	5						9	9	10	9	12
Trichilia catigua A.Juss.																4	4	4	4	5

Botanical Family/ Species	Fragment/Measurement Year																			
	GUA					IP1					IP2					SJO				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Trichilia lepidota Mart.						1	1	1												
Trichilia pallida Sw.	9	6	5	5	6	1	2	4	6	4										
Monimiaceae Juss.	**2**	**3**	**4**	**4**	**3**											**5**	**6**	**5**	**5**	**6**
Macropeplus ligustrinus (Tul.) Perkins																3	4	2	2	3
Mollinedia argyrogyna Perkins																		1	1	1
Mollinedia cf. *gilgiana* Perkins																1	1	1	1	1
Mollinedia oligantha Perkins	2	3	4	4	3															
Mollinedia salicifolia Perkins																1	1	1	1	1
Moraceae Gaudich.	**15**	**17**	**17**	**19**	**20**	**68**	**81**	**88**	**92**	**98**	**47**	**54**	**52**	**59**	**62**	**8**	**10**	**8**	**8**	**9**
Brosimum gaudichaudii Trécul						1	1	1	1	1										
Brosimum glaziovii Taub.																5	6	6	6	7
Brosimum guianense (Aubl.) Huber	6	6	7	8	8	46	60	67	74	79	42	45	44	50	53					
Ficus gomelleira Kunth						20	18	18	15	16										
Ficus insipida Willd.																1	1			
Ficus L. 01-GUA	1	1																		
Ficus L. 01-IP2											1	1								
Ficus L. 01-SJO																	1			
Ficus pertusa L.f.																1	1	1	1	1
Helicostylis tomentosa (Poepp. & Endl.) Rusby												1	1	1	1					
Maclura tinctoria (L.) D.Don ex Steud.	4	4	3	3	2															
Sorocea bonplandii (Baill.) W.C.Burger et al.	4	6	7	8	10	1	2	2	2	2	4	7	7	8	8	1	1	1	1	1
Myristicaceae R.Br.	**4**	**4**	**4**	**3**	**4**											**10**	**10**	**11**	**12**	**12**
Virola bicuhyba (Schott ex Spreng.) Warb.	1	1	1													10	10	11	12	12
Virola gardneri (A.DC.) Warb.	3	3	3	3	4															
Myrtaceae Juss.	**188**	**222**	**241**	**259**	**256**	**68**	**67**	**80**	**86**	**85**	**38**	**43**	**52**	**46**	**51**	**73**	**86**	**93**	**87**	**87**
Campomanesia guaviroba (DC.) Kiaersk.	7	5	6	5	5															1
Campomanesia hirsuta Gardner															1					
Campomanesia laurifolia Gardner											1	1	1	1	1					
Campomanesia Ruiz et Pav. 01-IP1						2	2	2	2											
Campomanesia schlechtendaliana (O.Berg) Nied.											1	1	1	1						

Botanical Family/ Species	Fragment/Measurement Year																			
	GUA					IP1					IP2					SJO				
	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22
Campomanesia xanthocarpa (Mart.) O.Berg																	1	1	2	2
Eugenia cf. *acutata* Miq.																1	1	1	1	1
Eugenia excoriata O.Berg						27	28	28	30	28					1					
Eugenia florida DC.						2	2	2	2	2										
Eugenia cf. *francavilleana* O.Berg	18	18	18	20	18															
Eugenia fusca O.Berg																5	5	5	3	3
Eugenia cf. *handroana* D.Legrand																2	2	2	2	2
Eugenia itajurensis Cambess.																1	1	1	1	1
Eugenia L. 01-GUA	1	3	5	5	6															
Eugenia L. 01-IP1						4	2	2	1											
Eugenia L. 01-IP2											1	1	1	1						
Eugenia L. 01-SJO																		1	1	1
Eugenia L. 02-IP1						1	1	1	1	1										
Eugenia L. 02-IP2											2	3	6	7	8					
Eugenia L. 03-IP2													1	1	1					
Eugenia L. 04-IP2														1	1					
Eugenia L. 05-IP2															1					
Eugenia leonorae Mattos											4	5	5	4	5					
Eugenia nutans O.Berg																1	1	1	1	1
Eugenia pisiformis Cambess.						2	3	3	3	3										
Eugenia pruniformis Cambess.									1	1		1	1	1	1					
Eugenia punicifolia (Kunth) DC.					1															
Eugenia reperta Sobral & Mazine											4	4	6	5	4					
Eugenia sonderiana O.Berg	6	8	8	10	10															
Myrcia amazonica DC.	82	105	113	124	123						1	1	1	1	1					
Myrcia anceps (Spreng.) O.Berg																1	1	1	1	1
Myrcia coelosepala Kiaersk.	8	10	15	20	22															
Myrcia crocea (Vell.) Kiaersk.													1	1	1					
Myrcia DC. 01-GUA	1																			
Myrcia DC. 01-IP2												1	1							

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Myrcia DC. 01-SJO																2	2	1		
Myrcia DC. 02-SJO																1	1	1	1	
Myrcia DC. 03-SJO																	1	1		
Myrcia DC. 04-SJO																	1	1	1	1
Myrcia cf. *eugeniopsoides* (D.Legrand & Kausel) Mazine.																				1
Myrcia excoriata (Mart.) E.Lucas & C.E.Wilson																1	1	1	1	1
Myrcia fenzliana O.Berg																			1	1
Myrcia guianensis (Aubl.) DC.			1	1	1	18	18	19	21	20	6	6	8	9	12	1	3	3	6	6
Myrcia cf. *laxiflora* Cambess.																				1
Myrcia loranthifolia (DC.) G.P.Burton & E.Lucas																2	2	2	4	5
Myrcia cf. *macrocarpa* DC.																		1	1	1
Myrcia multipunctata Mazine				1	1															
Myrcia mutabilis (O.Berg) N.Silveira			1	1	2															
Myrcia neolucida A.R.Lourenço & E.Lucas																1	1	1	1	1
Myrcia pseudosplendens Sobral & Mazine						2	2	2	3	5										
Myrcia splendens (Sw.) DC.	54	59	57	55	51	8	8	13	11	11	15	15	14	8	2	48	52	58	51	44
Myrcia tomentosa (Aubl.) DC.	8	7	7	5	4															
Myrcia undulata O.Berg																				2
Myrcia cf. *vittoriana* Kiaersk.																	1	1	1	1
Myrciaria disticha O.Berg																1	1	1	1	1
Myrciaria floribunda (H.West ex Willd.) O.Berg		1	2	3	3	1	1	7	10	12				1	5	1	1	2	2	2
Myrtaceae Juss. 01-IP1								1	1	1										
Myrtaceae Juss. 01-SJO																2	3	2	1	1
Myrtaceae Juss. 02-SJO																1	1	1		
Pimenta pseudocaryophyllus (Gomes) Landrum																1	1	1	1	1
Plinia peruviana (Poir.) Govaerts			1	1	1										1					
Psidium cattleyanum Sabine	2	3	3	3	3	1														
Psidium L. 01-IP2											2	2	2	1						
Psidium oblongatum O.Berg										1	1	2	3	3	5					2
Psidium rufum Mart. ex DC.		1	1	1	1												1	1	1	1

Botanical Family/ Species	Fragment/Measurement Year																			
	GUA					IP1					IP2					SJO				
	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22
Siphoneugena densiflora O.Berg	1	2	3	4	4															
Siphoneugena cf. *reitzii* D.Legrand																	1	1	1	1
Nyctaginaceae Juss.	**2**	**2**	**2**	**3**	**2**	**27**	**35**	**32**	**32**	**19**	**4**	**5**	**7**	**5**	**5**	**7**	**7**	**7**	**8**	**10**
Guapira hirsuta (Choisy) Lundell							1	1	1	1				1	1					
Guapira opposita (Vell.) Reitz	2	2	2	3	2	27	34	31	31	18	4	5	7	4	4	7	7	7	8	10
Ochnaceae DC.	**4**	**4**	**4**	**4**	**4**	**4**	**3**	**3**	**3**	**4**	**4**	**5**	**5**	**5**	**2**					**1**
Ouratea Aubl. 01-IP1/MAC						4	3	3	3	4										
Ouratea castaneifolia (DC.) Engl.																				1
Ouratea polygyna Engl.	1	1	1	1	1						4	5	5	5	2					
Ouratea semiserrata (Mart. & Nees) Engl.	3	3	3	3	3															
Pentaphylacaceae Engl.			**1**	**1**	**1**															
Ternstroemia brasiliensis Cambess.			1	1	1															
Peraceae Klotzsch	**2**	**2**	**3**	**11**	**11**	**5**	**7**	**8**	**10**	**15**	**1**	**2**	**2**	**6**	**10**	**45**	**44**	**41**	**40**	**45**
Chaetocarpus echinocarpus (Baill.) Ducke	1	1	1	3	3										1					
Pera glabrata (Schott) Baill.				6	6			1	1	2						1	1	1	1	2
Pera heteranthera (Schrank) I.M.Johnst.	1	1	2	2	2	4	5	5	5	7	1	2	2	2	2	8	7	7	6	6
Pogonophora schomburgkiana Miers ex Benth.						1	2	2	4	6				4	7					
Phyllanthaceae Martinov																**2**	**2**	**2**	**2**	**2**
Hieronyma oblonga (Tul.) Müll.Arg.																36	36	33	33	37
Margaritaria nobilis L.f.																2	2	2	2	2
Phytolaccaceae R.Br.						**1**														
Seguieria langsdorffii Moq.						1														
Piperaceae Giseke				**1**	**1**															
Piper amalago L.				1	1															
Primulaceae Batsch ex Borkh.	**4**	**6**	**6**	**7**	**7**			**1**	**1**	**1**	**1**	**1**	**1**	**1**	**1**	**7**	**7**	**8**	**9**	**7**
Geissanthus ambiguus (Mart.) G.Agostini								1	1	1	1	1	1	1	1					
Myrsine coriacea (Sw.) R.Br. ex Roem. & Schult.	1	1	1	1	1															
Myrsine gardneriana A.DC.	1	1	2	2	2															
Myrsine L. 01-GUA		2																		
Myrsine umbellata Mart.	2	2	3	4	4											7	7	8	9	7

Botanical Family/ Species	Fragment/Measurement Year																			
	GUA					IP1					IP2					SJO				
	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22
Proteaceae Juss.	**5**	**5**	**5**	**5**	**4**	**1**	**1**	**1**	**1**	**1**						**4**	**4**	**4**	**5**	**7**
Euplassa incana (Klotzsch) I.M.Johnst.	3	3	3	3	3															
Euplassa legalis (Vell.) I.M.Johnst.						1	1	1	1	1										
Euplassa semicostata Plana																			1	1
Roupala montana Aubl.	2	2	2	2	1											4	4	4	4	4
Roupala paulensis Sleumer																				2
Rosaceae Juss.						**1**	**1**													
Prunus myrtifolia (L.) Urb.						1	1													
Rubiaceae Juss.	**88**	**100**	**108**	**71**	**70**	**2**	**3**	**3**	**5**	**6**	**1**	**2**	**2**	**2**	**3**	**49**	**54**	**63**	**65**	**79**
Alseis involuta K.Schum.	6	6	5	6	7															
Amaioua guianensis Aubl.	4	12	17	22	23	1	1	1	1	1	1	1	1	1	1				1	2
Amaioua intermedia Mart. ex Schult. & Schult.f.																1	1	1	1	1
Bathysa australis (A.St.-Hil.) K.Schum.																1	1	1	1	1
Bathysa nicholsonii K.Schum.	11	11	15	15	15	1	1	1	1	1										
Cordiera A.Rich. ex DC. 01-IP1										1										
Cordiera concolor (Cham.) Kuntze									1	1										
Cordiera stipulacea C.H.Perss. & Delprete																1	1	1	1	1
Coussarea cf. *meridionalis* (Vell.) Müll.Arg.																3	3	4	4	4
Coussarea nodosa (Benth.) Müll.Arg.																3	4	7	8	13
Coutarea hexandra (Jacq.) K.Schum.			1	1	1															
Duroia valesca C.H.Perss. & Delprete												1	1	1	1					
Ferdinandusa edmundoi Sucre																14	16	17	19	21
Guettarda viburnoides Cham. & Schltdl.	10	10	10	9	8															
Ixora brevifolia Benth.	2	2	2	2	3															
Ladenbergia hexandra (Pohl) Klotzsch																2	2	2	2	2
Molopanthera paniculata Turcz.	53	57	56	13	10															
Palicourea sessilis (Vell.) C.M.Taylor																2	6	9	11	15
Posoqueria acutifolia Mart.																2	2	2	2	2
Psychotria carthagenensis Jacq.							1	1	2	2					1		1	2	3	3
Psychotria L. 01-SJO																10	7	5		

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Psychotria nuda (Cham. & Schltdl.) Wawra	1	1	1	1	1															
Schizocalyx cuspidatus (A.St.-Hil.) Kainul. & B. Bremer																9	9	11	11	12
Simira rubra (Mart.) Steyerm.																1	1	1	1	1
Simira sampaioana (Standl.) Steyerm.	1	1	1	2	2															
Simira cf. *walteri* Silva Neto & Callado																				1
Rutaceae A.Juss.	**7**	**7**	**8**	**9**	**10**	**18**	**18**	**16**	**17**	**18**		**1**	**1**	**1**	**1**	**12**	**9**	**8**	**7**	**5**
Dictyoloma vandellianum A.Juss.	1	1	1	1	1											2	2	2	2	
Hortia brasiliana Vand. ex DC.	2	2	2	2	2	16	16	14	14	15		1	1	1	1	3	3	3	3	3
Zanthoxylum fagara (L.) Sarg.																4	1			
Zanthoxylum rhoifolium Lam.	4	4	5	6	7				1	1						1	1	1		
Zanthoxylum riedelianum Engl.						2	2	2	2	2						2	2	2	2	2
Sabiaceae Blume						**1**	**1**	**1**	**1**	**1**										
Meliosma chartacea Lombardi						1	1	1	1	1										
Salicaceae Mirb.	**79**	**91**	**95**	**110**	**110**	**20**	**23**	**21**	**22**	**28**	**5**	**5**	**5**	**5**	**4**	**23**	**26**	**27**	**31**	**29**
Casearia arborea (Rich.) Urb.	12	14	16	21	23	5	8	9	13	20	3	3	3	3	3	5	6	4	4	2
Casearia decandra Jacq.	38	39	40	44	40	6	7	7	7	7										
Casearia grandiflora Cambess.																6	7	10	11	10
Casearia Jacq. 01-IP1						1	1	1	1											
Casearia lasiophylla Eichler																1	1	1	1	2
Casearia obliqua Spreng.	3	4	4	4	5															
Casearia rupestris Eichler																1	1	1	1	1
Casearia sylvestris Sw.	4	4	4	4	5											6	7	7	8	9
Casearia ulmifolia Vahl ex Vent.	3	5	6	7	7	8	7	4	1	1	1	1	1	1						
Piparea dentata Aubl.	19	25	25	30	30														1	1
Piparea multiflora C. F. Gaertn.											1	1	1	1	1	4	4	4	5	4
Sapindaceae Juss.	**110**	**110**	**118**	**121**	**115**	**108**	**130**	**146**	**156**	**159**	**15**	**15**	**15**	**14**	**14**	**18**	**20**	**26**	**30**	**37**
Allophylus edulis (A.St.-Hil. et al.) Hieron. ex Niederl.	2	3	4	4	4															
Allophylus petiolulatus Radlk.																		1	1	1
Cupania hirsuta Radlk.						27	28	32	29	19	2	2	2	2	2					
Cupania L. 01-SJO																		1		

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Cupania ludowigii Somner & Ferrucci						5	8	9	15	17	1	1	1	1	1	7	9	11	12	12
Cupania oblongifolia Mart.	2	3	3			26	27	26	27	24	8	6	4	2	1	1			2	1
Cupania vernalis Cambess.	5	6	12	13	13											2	2	2	2	2
Matayba elaeagnoides Radlk.																1	2	2	2	2
Matayba guianensis Aubl.	4	4	2	2	2															2
Matayba marginata Radlk.																5	5	5	5	5
Matayba mollis Radlk.	15	15	16	18	15															
Talisia Aubl. 01-GUA	2	1	1																	
Toulicia stans (Schott) Radlk.	80	78	80	84	81	50	67	79	85	99	4	6	8	9	10	2	2	4	6	12
Sapotaceae Juss.	**9**	**9**	**9**	**9**	**9**	**87**	**93**	**90**	**93**	**98**	**36**	**40**	**44**	**44**	**36**	**10**	**13**	**14**	**13**	**14**
Chrysophyllum gonocarpum (Mart. & Eichler ex Miq.) Engl.						2	2				3	3	3	2	2	1	2	2	2	2
Ecclinusa ramiflora Mart.						6	6	4	4	4	20	21	23	23	14	1	1	1	1	1
Micropholis gardneriana (A.DC.) Pierre													1	2	2	1	1	1	1	1
Pouteria Aubl. 01-IP2											1	1	1							
Pouteria filipes Eyma																		1	1	1
Pouteria gallifructa Cronq.	3	3	3	3	3															
Pouteria gardneri (Mart. & Miq.) Baehni	3	3	3	3	3															
Pouteria guianensis Aubl.						55	60	59	60	63	2	2	2	2	2					
Pouteria microstrigosa T.D.Penn.											2	3	3	3	3	1	1	1	1	1
Pouteria cf. *nordestinensis* Alves-Araújo & M.Alves						18	18	19	20	22	8	10	11	12	13					
Pouteria procera (Mart.) K.Hammer								1	1	1										
Pouteria ramiflora (Mart.) Radlk.						2	3	3	4	4						1	1	1	1	1
Pouteria cf. *synsepala* Popovkin & A.D.Faria																1	1	1	1	1
Pouteria torta (Mart.) Radlk.						4	4	4	4	4						3	3	3	2	2
Pouteria venosa (Mart.) Baehni	3	3	3	3	3															
Pradosia kuhlmannii Toledo																1	3	3	3	4
Simaroubaceae DC.	**2**	**3**	**3**	**3**	**3**															
Homalolepis floribunda (A.St.-Hil.) Devecchi & Pirani	2	3	3	3	3															
Siparunaceae (A.DC.) Schodde	**6**	**6**	**8**	**19**	**25**	**19**	**18**	**27**	**26**	**34**	**8**	**10**	**14**	**18**	**14**	**23**	**24**	**25**	**29**	**31**
Siparuna bifida (Poepp. & Endl.) A.DC.						1	1	1	1	1										

Botanical Family/ Species	**Fragment/Measurement Year**																			
	GUA					**IP1**					**IP2**					**SJO**				
	02	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**	**02**	**07**	**12**	**17**	**22**
Siparuna guianensis Aubl.	6	6	8	19	25	17	15	24	23	30	7	9	13	17	13	8	8	8	10	7
Siparuna reginae (Tul.) A.DC.						1	2	2	2	3	1	1	1	1	1	15	16	17	19	24
Solanaceae A.Juss.	**1**	**1**	**1**	**1**	**1**	**17**	**7**	**4**	**4**	**2**						**9**	**5**	**8**	**4**	**5**
Brunfelsia brasiliensis (Spreng.) L.B.Sm. & Downs								1	1	1										
Solanum L. 01-GUA	1	1	1	1	1															
Solanum L. 01-SJO																1				
Solanum pseudoquina A.St.-Hil.									1	1								1	2	2
Solanum swartzianum Roem. & Schult.						17	7	3	2							8	5	7	2	3
Strombosiaceae Tiegh.																**2**	**3**	**3**	**3**	**3**
Tetrastylidium grandifolium (Baill.) Sleumer																2	3	3	3	3
Styracaceae DC. & Spreng.	**2**	**2**	**2**	**2**	**2**											**1**	**1**	**1**	**1**	**1**
Styrax leprosus Hook. & Arn.																1	1	1	1	1
Styrax pohlii A.DC.	2	2	2	2	2															
Symplocaceae Desf.	**11**	**13**	**15**	**16**	**9**															
Symplocos estrellensis Casar.	3	3	4	4	4															
Symplocos Jacq. 01-GUA	1	1	1	1																
Symplocos pubescens Klotzsch ex Benth.	7	9	10	11	5															
Theaceae Mirb. ex Ker Gawl.	**4**	**6**	**7**	**9**	**9**											**3**	**3**	**3**	**3**	**3**
Laplacea fruticosa (Schrad.) Kobuski	4	6	7	9	9											3	3	3	3	3
Urticaceae Juss.	**3**	**1**				**35**	**23**	**6**	**3**	**3**	**2**	**1**				**14**	**11**	**10**	**10**	**8**
Cecropia glaziovii Snethl.						4	2									2	2	2	3	5
Cecropia hololeuca Miq.	3	1				9	8	2			2	1				9	6	5	4	1
Cecropia pachystachya Trécul						19	10	1												
Pourouma guianensis Aubl.						3	3	3	3	3						3	3	3	3	2
Vochysiaceae A.St.-Hil.	**1**	**2**	**2**	**5**	**5**											**1**	**1**	**1**	**2**	**7**
Qualea multiflora Mart.																1	1	1	1	2
Vochysia dasyantha Warm.																				2
Vochysia cf. *oppugnata* (Vell.) Warm.	1	2	2	5	5															
Vochysia schwackeana Warm.																			1	3
Undetermined	**10**	**7**	**7**	**3**	**3**	**1**	**5**	**3**	**3**	**3**	**1**	**1**	**1**		**1**	**5**	**5**	**3**	**2**	**2**

Botanical Family/ Species	Fragment/Measurement Year																			
	GUA					IP1					IP2					SJO				
	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22	02	07	12	17	22
Undetermined 01-GUA	1																			
Undetermined 02-GUA	4	1	2																	
Undetermined 03-GUA	3	3	3																	
Undetermined 04-GUA	1	1																		
Undetermined 05-GUA		1	1																	
Undetermined 06-GUA	1	1	1	1	1															
Undetermined 07-GUA				1	1															
Undetermined 08-GUA				1	1															
Undetermined 01-IP1							2													
Undetermined 02-IP1						1	3	3	3	2										
Undetermined 03-IP1										1										
Undetermined 01-IP2											1	1	1							
Undetermined 02-IP2															1					
Undetermined 01-SJO																2	1	1		
Undetermined 02-SJO																1	1			
Undetermined 03-SJO																	1			
Undetermined 04-SJO																1	1	1	1	1
Undetermined 05-SJO																1	1	1	1	1
Total Richness	**175**	**182**	**184**	**183**	**182**	**130**	**130**	**135**	**132**	**129**	**104**	**109**	**106**	**104**	**106**	**222**	**235**	**236**	**235**	**246**
Total Abundance	**1,889**	**1,996**	**2,033**	**1,951**	**1,896**	**1,375**	**1,478**	**1,452**	**1,354**	**1,358**	**539**	**559**	**564**	**523**	**492**	**922**	**976**	**1,008**	**983**	**1,031**

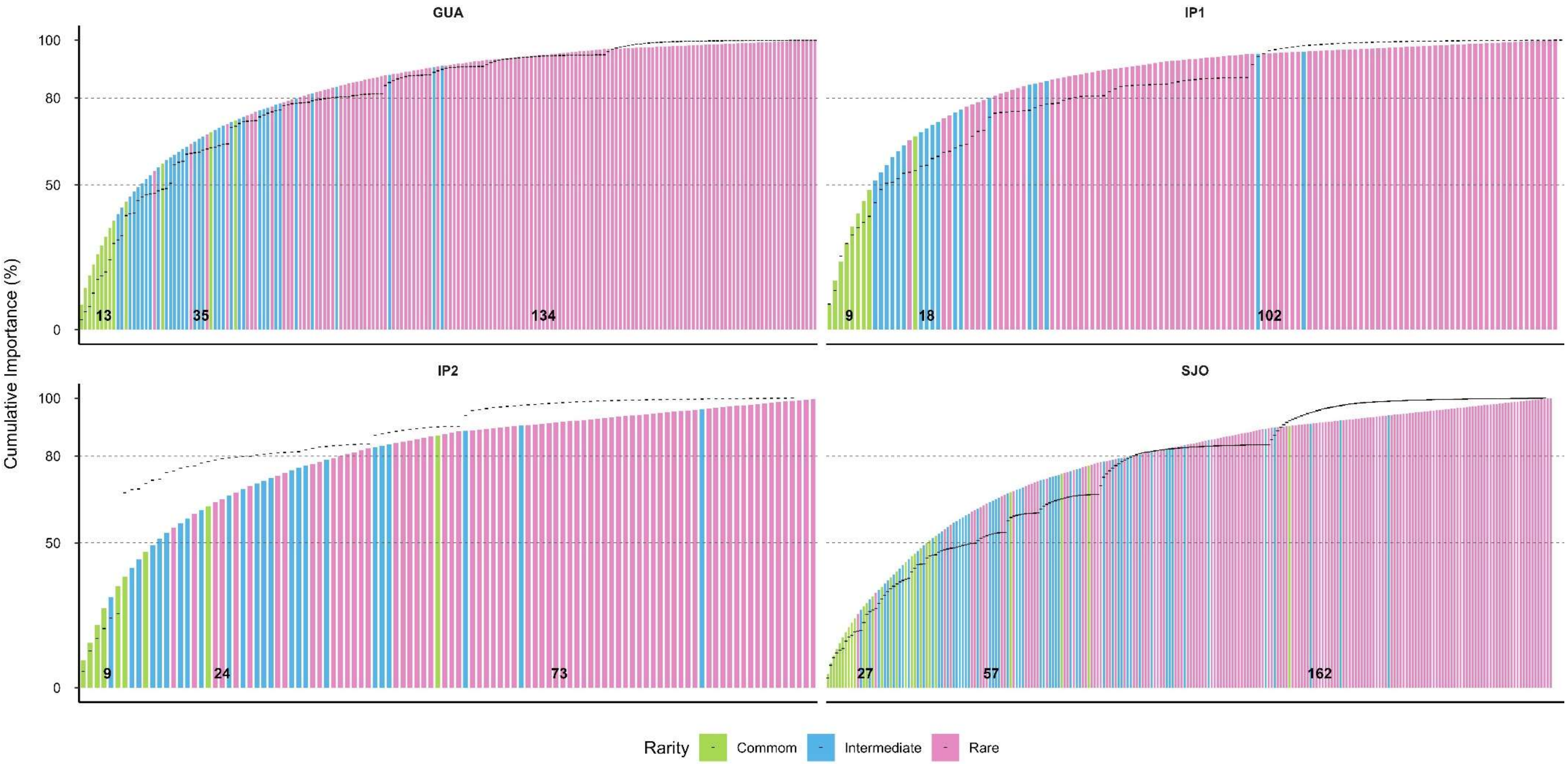

Figure S6. Transition of tree species between abundance classes and their population and biomass contributions in Atlantic Rainforest fragments after 20 years of forest dynamics (2002–2022), in four Atlantic Rainforest fragments in Minas Gerais, southeastern Brazil. Colored bars represent each species' cumulative contribution to total abundance, initially classified as common (green, up to 50%), intermediate (blue, 50–80%), and rare (pink, >80%). Dotted lines indicate each species' corresponding contribution to cumulative relative biomass. Numbers within the bars indicate the number of species in each abundance class.

Table S3. Richness distribution and dynamic in different abundance classes during 20 years of monitoring (2002-2022), in four Atlantic Rainforest fragments in Minas Gerais, southeastern Brazil.

Abundance Class	Fragment			
	GUA	IP1	IP2	SJO
Commom	**13 (6,31%)**	**9 (5,63%)**	**9 (6,87%)**	**28 (10,18%)**
Stayed	13	9	9	27
Left	-	-	-	1
Intermediate	**36 (17,48%)**	**20 (12,50%)**	**26 (19,85%)**	**60 (21,82%)**
Stayed	35	18	24	57
Left	1	2	2	3
Rare	**157 (76,21%)**	**131 (81,87%)**	**96 (73,28%)**	**187 (68,00%)**
Stayed	108	76	48	117
Left	18	25	21	17
Joined	26	26	25	45
Transitioned	5	4	2	8
Total	**206**	**160**	**131**	**275**

Table S4. Abundance and biomass net dynamics of different abundance classes during 20 years of monitoring (2002-2022), in four Atlantic Rainforest fragments in Minas Gerais, southeastern Brazil.

Fragment	Plot	Net Dynamic					
		Abundance			Biomass (Mg)		
		Common	Intermediate	Rare	Common	Intermediate	Rare
GUA	1	-7	2	5	0,8021	1,0153	0,1423
GUA	2	-8	11	14	0,2300	0,3519	0,2327
GUA	3	-22	4	4	0,2901	0,6485	0,0298
GUA	4	-7	8	12	-0,0441	1,5554	0,0341
GUA	5	-14	1	-3	0,4254	1,3194	0,1176
GUA	6	-4	-14	0	0,1553	-0,7666	-0,1019
GUA	7	-2	-5	-3	-0,3208	2,0008	0,2225
GUA	8	-6	-9	2	0,3516	0,3631	0,1311
GUA	9	0	-2	8	2,9564	0,1265	0,5676
GUA	10	-33	19	17	-0,0199	0,2920	0,2851
GUA	11	3	4	-2	0,6617	0,3909	0,0784

Fragment	Plot	Net Dynamic					
		Abundance			Biomass (Mg)		
		Common	Intermediate	Rare	Common	Intermediate	Rare
GUA	12	2	2	1	0,2868	0,6606	1,7636
GUA	13	16	4	-4	0,4709	1,7182	-0,0070
GUA	14	2	19	23	0,6747	0,2605	0,3481
GUA	15	5	9	16	0,6848	0,8311	0,9473
GUA	16	-55	6	25	-0,1780	0,3683	0,5169
GUA	17	-6	-3	5	0,7272	0,5207	0,2083
GUA	18	-12	-7	-1	1,0378	0,6340	0,3755
GUA	19	-9	-2	-2	0,1059	1,0492	0,3304
IP1	1	3	-7	1	0,4656	0,3678	0,3575
IP1	2	3	0	2	0,1660	0,4692	-0,2000
IP1	3	-5	7	-10	0,2490	0,3642	1,2144
IP1	4	-2	-13	1	0,7551	1,0154	2,1831
IP1	5	10	-6	-4	1,0916	0,7437	0,1208
IP1	6	-39	5	14	-1,1902	0,2857	0,2026
IP1	7	14	-27	-3	0,2741	-0,0673	0,9770
IP1	8	0	-1	4	0,2236	0,2307	0,2311
IP1	9	7	-18	15	0,2481	-0,0812	0,3886
IP1	10	-7	1	-2	0,2015	1,2313	-0,0020
IP1	11	-20	-3	8	0,0559	0,2697	0,0332
IP1	18	0	4	-10	0,7217	0,4031	-0,1362
IP1	19	0	4	8	0,9878	0,1142	-0,1832
IP1	20	9	14	16	0,7276	0,6150	1,1398
IP1	21	-3	-7	2	1,3487	0,0618	0,2465
IP1	22	5	3	9	0,0125	0,6838	2,2076
IP2	12	-6	3	6	2,1175	1,4282	0,6485
IP2	13	-4	6	1	1,9617	0,5914	0,3764
IP2	14	-10	3	10	-0,5510	0,0709	0,3596
IP2	15	-21	0	6	-0,1974	0,1436	0,1059
IP2	16	-2	-8	7	0,7131	-0,5399	0,4114
IP2	17	-31	-15	8	2,3378	-0,6228	0,0974

Fragment	Plot	Net Dynamic					
		Abundance			Biomass (Mg)		
		Common	Intermediate	Rare	Common	Intermediate	Rare
SJO	1	12	7	21	1,7832	0,4966	0,2682
SJO	2	-1	1	16	1,1628	1,7399	0,8692
SJO	3	11	17	23	0,5124	0,7134	0,8567
SJO	4	-5	-4	13	-0,6050	0,6513	0,4559
SJO	5	-15	-6	26	-0,0966	0,7143	1,0690
SJO	6	-1	-5	5	0,1410	0,7222	0,7386
SJO	7	-3	12	6	-0,5406	2,3631	-0,1007
SJO	8	-16	4	6	-0,3927	-1,0809	0,1780
SJO	9	-1	-1	9	-0,4284	0,0457	1,0902
SJO	10	-13	-3	1	0,0930	0,0118	0,3325
SJO	11	-8	4	5	0,9843	0,4572	1,0233
SJO	12	-14	-3	9	-0,1932	0,1644	0,3024

Table S5. Performance (marginal and conditional R² values, and Akaike Information Criterion - AIC) of linear mixed models explaining abundance (NDA) and biomass (NDB) net dynamics of common, intermediate and rare tree species, based on different groups (landscape, edaphic, climatic, and forestry diversity) of predictor variables.

Dynamic Component	**Model Metrics**	**Landscape**	**Model** **Edaphic**	**Climatic**	**Diversity**
NDA Common	R^2 marginal	0.003	0.009	0.154	0.085
	R^2 conditional	0.312	0.301	0.261	0.435
	AIC	1513.803	1452.416	1491.458	1440.424
NDA Intermediate	R^2 marginal	0.085	0.055	0.032	0.069
	R^2 conditional	0.123	0.119	0.208	0.253
	AIC	1169.537	1114.133	1163.447	1115.736
NDA Rare	R^2 marginal	0.112	0.116	0.076	0.031
	R^2 conditional	0.152	0.151	0.194	0.087
	AIC	1165.215	1103.921	1163.858	1131.792
NDB Common	R^2 marginal	0.073	0.026	0.092	0.057
	R^2 conditional	0.292	0.168	0.164	0.176
	AIC	182.127	129.837	163.323	124.383
NDB Intermediate	R^2 marginal	0.043	0.078	0.031	0.011
	R^2 conditional	0.076	0.103	0.063	0.037
	AIC	132.803	63.968	116.684	81.675
NDB Rare	R^2 marginal	0.101	0.055	0.035	0.064
	R^2 conditional	0.115	0.095	0.097	0.096
	AIC	-8.358	-60.014	-14.169	-59.575

CHAPTER III:

FROM HERBARIUM TO LANDSCAPE: NEW RECORDS AND MAPPING RARE AND THREATENED SPECIES OF BRAZILIAN ATLANTIC RAINFOREST

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From Herbarium to Landscape: New Records and Mapping Rare and Threatened Species of Brazilian Atlantic Rainforest

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ABSTRACT

The Atlantic Rainforest is a biodiversity hotspot with high endemism. Botanical sampling in its interior mountains is limited, affecting knowledge of locally endemic or undescribed species. The aim was to investigate the presence and distribution of rare and threatened species in different Atlantic Rainforest fragments. Atlantic Rainforest of Minas Gerais state, Brazilian southeastern. We used multi-level forest inventory data from 137 plots across nine Semideciduous Seasonal Forest fragments, sampled 1–9 times over 30 years. We selected species with ≤ 60 previous records, associating coordinates to plot distribution maps. We analyzed species' distribution in the biodiversity repositories SpeciesLink, JABOT, and GBIF to avoid omitting exclusive records on any of these platforms. We documented 17 new records and listed 243 previous records of 12 rare species in eight families. These species are endemic to the Atlantic Rainforest, with most showing some level of threat. The new records expanded species' occurrence zones, and *Homalolepis insignis* and *Rhodostemonodaphne anomala* were documented for the first time in Minas Gerais state, highlighting that collection in under-sampled regions is essential for improving species knowledge and reducing sampling bias. The previous record numbers varied across species, with misidentifications causing inconsistencies in occurrence records, particularly for *Didymopanax longipetiolatus*. Physical and digital collections need review to correct identification errors, synonyms used improperly, and imprecise coordinates. This information is crucial for identifying priority areas for conservation, especially rare and threatened species. A concerning lack of synchronization between scientific publications, biodiversity repositories, and government organizations may compromise policy development for environmental management and resource allocation to protect vulnerable areas.

1 | Introduction

The Atlantic Rainforest is a highly fragmented global biodiversity hotspot (Myers et al. 2000; Rezende et al. 2018; Joly et al. 2019). Currently, only 28.1% of its original extent remains covered by forest (MapBiomas 2024). This biome is characterized by high floristic richness, with 45% of species being endemic and 82% threatened (de Lima et al. 2020, 2024). Despite its remarkable biodiversity, sampling in tropical ecosystems like the Atlantic Rainforest remains challenging, particularly in remote or sparsely populated regions (Fernández et al. 2015; Hortal et al. 2015; Hopkins 2019; Stropp et al. 2020; Hughes et al. 2021).

Although the Atlantic Rainforest has widespread human occupation (Pinto and Voivodic 2021), botanical collections remain limited in mountainous regions (de Araujo and Ramos 2021). These inaccessible areas often harbor a high proportion of locally endemic species and numerous others yet to be described, many of which are at risk of being entirely lost due to anthropogenic activities such as mining and deforestation (Barlow et al. 2016; Oliveira et al. 2016; de Araujo and Ramos 2021; Lannuzel et al. 2022; Nery et al. 2023; Sant'Anna-Santos 2023). Furthermore, certain botanical clades are often overlooked, receiving limited attention in taxonomic and phylogenetic studies (Lombardi 2009; Hortal et al. 2015; Oliveira et al. 2016). These gaps hinder the accuracy of biodiversity indicators, limit the discovery and description of new species, restrict advancement in taxonomic knowledge, and challenge the precise delineation of distribution ranges, particularly for rare species (Canhos et al. 2015; Hortal et al. 2015; Oliveira et al. 2016).

In recent decades, initiatives from various scientific institutions have collaborated on big data initiatives to develop and enhance digital repositories of floristic biodiversity, based on the most relevant physical collections worldwide (Canhos et al. 2015; Hortal et al. 2015; Lannuzel et al. 2022). These platforms represent a new model for flora data management, providing valuable information to scientists and professionals working with diverse vegetation formations and taxonomic groups. Largely free to access, these repositories overcome traditional barriers to scientific knowledge sharing by eliminating the need for physical visits to botanical collections (Canhos et al. 2015). Although the volume of available data is substantial (Canhos et al. 2022; SpeciesLink network 2024), access to certain records may be limited to specific repositories, depending on the supporting institutional networks (Marsico et al. 2020; de Araujo et al. 2022).

Digital repositories are not free from inconsistencies and may reflect sampling biases. However, integrating data from multiple platforms can enhance sampling completeness and mitigate Wallacean shortfalls (de Araujo and Ramos 2021; Heberling et al. 2021; de Araujo et al. 2022). Integrating national and global databases can help address this shortfall, as national repositories often include records from small regional herbaria, thereby reducing field sampling gaps (Marsico et al. 2020; de Araujo et al. 2022). Furthermore, herbarium collections often represent the only source of data available for several species, and the information they contain is crucial to support studies and policies related to ecological and taxonomic knowledge, biogeographic inferences, and the development of conservation strategies (Oliveira et al. 2016; Marsico et al. 2020; de Araujo and Ramos 2021). For rare species, the limitations can be even more pronounced, as collections often face challenges with accurate taxonomic identification. This issue creates a cycle where the lack of collections delays the advancement of species knowledge, further complicating the accurate identification of previous records and the generation of new ones (Hortal et al. 2015).

Recent studies on the structure and dynamics of Atlantic Rainforest fragments (Nery et al. 2023; Rodrigues et al. 2023; Torres et al. 2023; Harper et al. 2024; Valente et al. 2024) have addressed various ecological aspects. However, these investigations often overlook demographically and ecologically emerging species with lower prominence. Consequently, the distribution of rare and endangered species remains poorly understood, along with the adequacy and accessibility of their collections hosted in big data e-infrastructures.

To address these gaps, our objective was to investigate the presence and distribution of rare and endangered species across different fragments of the Atlantic Rainforest in Minas Gerais, Brazil. Thus, we provide: (i) new records of these species, including herbarium deposit details; (ii) updated distribution maps incorporating new and previous species records; (iii) comments on previous records available in herbarium collections and digital biodiversity repositories; and (iv) an overview of how these new records may affect the conservation of rare and endangered species.

2 | Material and Methods

2.1 | Study Area

We used forest inventory data from 137 plots (0.05 ha) distributed across nine fragments of Seasonal Semideciduous Forest of the Atlantic Rainforest, located in eastern Minas Gerais, Brazil (Figure 1). These fragments, inventoried between 1 to 9 times, exhibit heterogeneity in size, floristic composition, soil traits, and topography (Table 1) (for more details, see Torres et al. 2023).

2.2 | Field Survey

We conducted multi-level forest inventories in all areas. In the plots, we considered the following inclusion levels for palms, shrubs, and trees: (I) diameter at breast height (DBH) ≥ 5 cm; (II) DBH < 5 cm and total height (Ht) ≥ 2 m; and (III) height from 0.3 m to < 2 m. When possible, we conducted botanical vegetative collections, and when available, collected fertile material from tree species, which are the primary focus of our study. Furthermore, we recorded and collected other life forms, including subshrubs, lianas, herbs, and ferns observed within the plots, along trails, or in other locations visited within the fragments. In total, we performed 1009 botanical collections (Tables S1 and S2), and all samples were oven-dried. Of these, 232 (22.99%) were prepared and incorporated into the herbarium collection of the herbarium of the Federal University of Viçosa (VIC). Due to the vast experience of the research team on the regional flora, especially the first author (OMV), who also participated in the field expeditions, many species were identified in situ. The species collected in the field were only those that presented reproductive organs. However, species that were not confirmed or identified in the field were collected whenever possible, for later verification and determination. This stage was supported by experts in various botanical groups, comparisons with specimens deposited in the VIC collection, and consultations of specialized literature. Over 30 years of monitoring, our

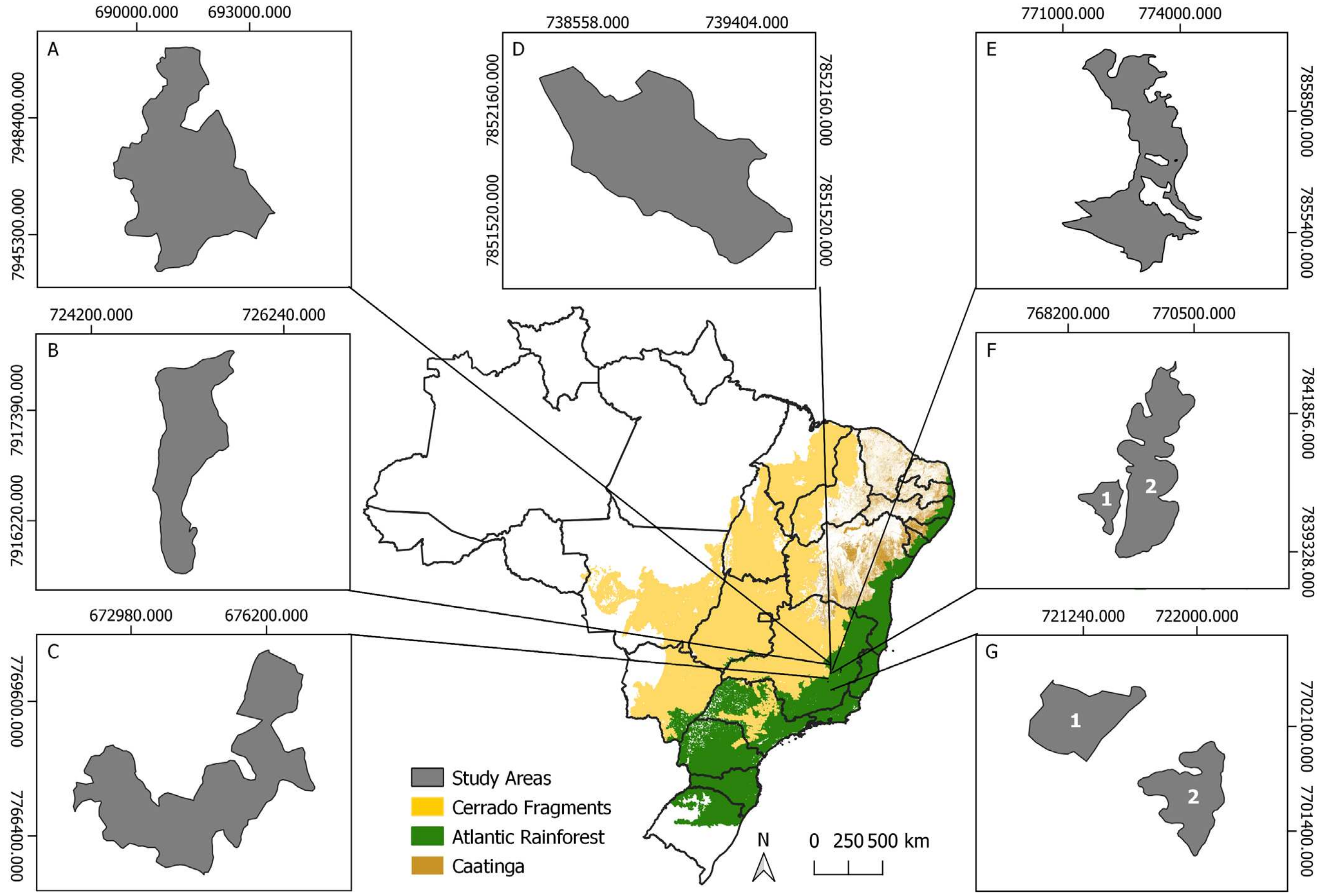


FIGURE 1 | Locations of the nine studied Atlantic Rainforest fragments located in Minas Gerais, Brazil. (A) Rio Guanhães; (B) Cachoeira das Pombas; (C) Chapadão; (D) São José; (E) RPPN Fazenda Macedônia; (F1) Ipaba Mata2; (F2) Ipaba Mata1; (G1) Mata da Silvicultura; (G2) Mata da Garagem.

TABLE 1 | Characteristics of the nine Atlantic Rainforest fragments in Minas Gerais, Brazil, included in this study.

Fragment/Area	Acronym	Municipality	Area (ha)	Plots	Tree richness	Maximum elevation	Inventory years
Chapadão	CAL	Catas Altas	417.0	15	242	860	2023
Mata da Garagem	GAR	Viçosa	31.5	10	119	727	1996; 1998; 2001; 2003; 2013; 2018; 2023
Cachoeira das Pombas	GUA	Guanhães	360.2	20	226	1169	2002; 2007; 2012; 2017; 2022
Ipaba Mata1	IP1	Caratinga	271.2	16	174	299	2002; 2007; 2012; 2017; 2022
Ipaba Mata2	IP2	Caratinga	79.4	6	140	310	2002; 2007; 2012; 2017; 2022
RPPN Fazenda Macedônia	MAC	Ipaba	867.0	23	254	347	2021
Mata da Silvicultura	MSI	Viçosa	46.2	20	182	744	1994; 1997; 2000; 2004; 2008; 2010; 2013; 2016; 2020
Rio Guanhães	SAB	Sabinópolis	1748.0	15	177	904	2023
São José	SJO	Coronel Fabriciano	100.6	12	304	929	2002; 2007; 2012; 2017; 2022

ongoing forest inventory efforts recorded 926 species of trees, shrubs, and palms across nine Atlantic Rainforest fragments, along with an extensive list of other life forms (data not presented here). The species list was compiled according to the APG IV classification (APG 2016), with scientific nomenclature following the "Flora e Funga do Brasil" (2025) and Plants of the World Online (POWO 2025) databases.

2.3 | Species Selection and Digital Repositories Survey

We carry out the process of selecting rare species from our field data and describe the subsequent search strategy used to gather and curate their existing records from digital biodiversity repositories. We focused on species with low population densities in our database, analyzing both tree and non-tree-shrub species recorded in each monitoring fragment. Initially, we examined the distribution of each species using the "state" filter in the national biodiversity repository SpeciesLink (Canhos et al. 2022; SpeciesLink network 2024) to assess the representativeness of collections in the state of Minas Gerais (MG). For species with no more than 20 records in the state, we conducted a national-level analysis, retaining only those with up to 80 national records.

After verifying the species distribution, we used SpeciesLink (SpeciesLink network 2024) to summarize all records for each species, grouping duplicates from each individual collection. SpeciesLink integrates about 600 datasets from over 140 Brazilian institutions, along with contributions from 37 international institutions (Canhos et al. 2015, 2022; SpeciesLink network 2024). Next, we accessed the JABOT platform (JBRJ 2024), which provides data from the collections of the Rio de Janeiro Botanical Garden. We further expanded our search to the GBIF repository (Global Biodiversity Information Facility), a global initiative for aggregating and distributing biodiversity data (Nelson and Ellis 2018; de Araujo et al. 2022). For each selected species, we retrieved all available collections by searching for accepted names and known synonyms across these three digital repositories, ensuring the inclusion of records that may have been recorded under outdated nomenclature. This approach was employed to avoid omitting records that might be exclusive to certain platforms (Heberling et al. 2021; de Araujo et al. 2022).

We also investigated records identified at the genus and family levels, particularly those collected from the Rio Doce basin, where all our study areas are located. This allowed us to identify probable species records that had not yet been classified at speciess level, which were then included in our collection list (Table S2). Additionally, we evaluated the notes available on herbarium specimens labels (Vieira et al. 2024), assessing the accuracy of species identifications (including their synonyms). We applied a conservative approach, excluding records with significant inconsistencies regarding morphological characteristics (either observed or described), collection locations, or records lacking sufficient detail. Finally, we selected species up to 50 records. We follow the herbarium acronyms recognized in the Index Herbariorum, both in searches in digital repositories and in mentions during the discussion (Thiers 2025).

2.4 | Collections Mapping

We associate geographic coordinates with most of the collections. When original coordinates were unavailable or inconsistent with the information on the exsiccate label, we inferred coordinates based on available data, provided that at least the municipality was known (Table S2, coordinates underlined). Next, we plotted the coordinates on thematic maps to illustrate the distribution of each species in the Atlantic Rainforest (Vieira et al. 2024), with a focus on highlighting new records in relation to the previously known occurrence area. We constructed the maps (Figures 2–7) using QGIS software v. 3.34 Prizren (QGIS Development Team 2024), supported by shapefiles provided by the IBGE portal (IBGE 2024) and INPE (Assis et al. 2019).

3 | Results

Our collection efforts resulted in 19 new records (15 herbarium deposits and four field observations; Table S2) of 12 rare species, distributed across seven of the nine surveyed areas, with no rare species recorded in the Viçosa (GAR and MSI) and CAL fragments. For these species, we identified 330 previous records, of which 243 were deemed valid. Records were excluded if they contained inconsistencies in identification or collection origin (71) or were from cultivated specimens (16) (Table S2). Myrtaceae Juss. was the most represented family with three species, followed by Lauraceae Juss. and Simaroubaceae DC. with two species each, while the remaining families were represented by a single species (Table 2). All recorded species are endemic to the Atlantic Rainforest, with the majority (8) classified as threatened. Remarkably, *Homalolepis insignis* (A.St.-Hil. & Tul.) Devecchi & Pirani and *Rhodostemonodaphne anomala* (Mez) Rohwer were documented for the first time in the state of Minas Gerais. All our records can be accessed online at the SpeciesLink repository (https://specieslink.net/search/) by inputting the deposit data as search terms.

4 | Discussion

This study investigated the presence and distribution of rare and endangered species across Atlantic Rainforest fragments in Minas Gerais, Brazil, aiming to address critical knowledge gaps. First, we provide the results of an intensive field survey that yielded 19 new occurrence records for 12 rare species (Table 2), with detailed collection information and herbarium deposits provided herein (Table S2). We generated updated distribution maps (Figures 2–7) by integrating these new findings with 243 validated historical records, effectively illustrating the expanded known ranges for these taxa. Previous records from major digital biodiversity repositories and herbarium collections were critically assessed, highlighting inconsistencies and the crucial need for data curation (Sections 4.1 and 4.2). Finally, we synthesize these findings to provide an overview of how both the new records and the evaluation of existing data influence the conservation perspectives for these rare and endangered species (Section 4.3).

FIGURE 2 | Legend on next page.

FIGURE 2 | New records for *Justicia minensis* Profice (Acanthaceae Juss.) (A) and *Didymopanax longipetiolatus* (Pohl ex DC.) Marchal (Araliaceae Juss.) (B–F), and their distributions in the Atlantic Rainforest (G). (A) Minas Gerais: Sabinópolis, *Verly 101*; (B) Minas Gerais: Coronel Fabriciano, *Verly 23*; (C–F) Minas Gerais: Coronel Fabriciano, *Verly 244* (in vivo). (C) Overview of the collected individual and the understory in the fragment area where the new population was recorded; (D) Branch apex, phyllotaxy; (E) Detail of buds and a flower after anthesis; (F) Detail of immature fruits. Scale: 50 mm.

TABLE 2 | New records and conservation status, assessed by the "National Center for Flora Conservation" (CNC), "Official National List of Threatened Flora Species" (LNFA), and the International Union for Conservation of Nature (IUCN), of rare and threatened species in Semideciduous Seasonal Forest fragments of the Atlantic Rainforest, Minas Gerais, Brazil.

		Conservation status		
Family/Species	**New record**	**CNC**	**LNFA**	**IUCN**
Acanthaceae Juss.				
Justicia minensis Profice	VIC056564	NE	NM	NM
Araliaceae Juss.				
Didymopanax longipetiolatus (Pohl ex DC.) Marchal	VIC056021 VIC058117	LC	NM	LC
Lauraceae Juss.				
Persea rigida Nees & Mart.	VIC057707	NT	NM	NT
Rhodostemonodaphne anomala (Mez) Rohwer	VIC057709	NE	EN	NM
Myrtaceae Juss.				
Eugenia leonorae Mattos	VIC057681 VIC057692	EN	EN	NM
Eugenia reperta Sobral & Mazine	VIC057776	NM	DD[a]	NM
Myrcia pseudosplendens Sobral & Mazine	VIC057680	NE	DD[b]	CR
Proteaceae Juss.				
Euplassa semicostata Plana	VIC057786	EN	EN	LC
Rubiaceae Juss.				
Duroia valesca C.H.Perss. & Delprete	VIC057686 VIC058115	VU	VU	VU
Simaroubaceae DC.				
Homalolepis floribunda (A.St.-Hil.) Devecchi & Pirani	VIC056446	CR	CR	CR
Homalolepis insignis (A.St.-Hil. & Tul.) Devecchi & Pirani	VIC056916 VIC058109	EN	EN	EN

Note: Conservation status: CR, critically endangered; DD, data deficient; EN, endangered; LC, least concern; NE, not evaluated; NM, not mentioned; NT, near threatened; VU, vulnerable.

[a] Conservation status by Sobral et al. (2022).

[b] Conservation status by Sobral et al. (2016).

4.1 | Species

4.1.1 | Acanthaceae Juss.

Justicia minensis Profice

Fig.: 2A; Brazil: Minas Gerais: Sabinópolis, *Verly 101*.

Justicia L. is the most species-rich genus in Acanthaceae (Graham 1988), with around 600 species distributed across all tropical regions worldwide, being highly diversified in the tropics and subtropics of South America (Ezcurra 2002). In Brazil, the number of described and accepted species within the genus has been increasing, driven by taxonomists efforts that have recently resulted in the description of numerous new species (Côrtes and Rapini 2010; da Silva, Gil, Reis, et al. 2019; da Silva, Gil and Kameyama 2019; Aoyama and Indriunas 2022). Additionally, taxonomic revisions of the *Justicia* complex and related genera have led to new combinations and, in some cases, an expanded circumscription of the clade, incorporating species previously assigned to other genera (Profice 2010; Côrtes and Rapini 2010, 2013; da Costa-Lima and Chagas 2019).

Consequently, the number of recognized species has increased from 128 in 2015 (Braz and Azevedo 2016) to 157, of which 94 are endemic to Brazil (Chagas and da Costa-Lima 2024).

Justicia minensis is one of these endemic species. It was originally described as *Beloperone lanceolata* Mart. ex Nees, in 1847 (IPNI 2024). The initial description was followed by two nomenclatural variations for its varieties: *B. lanceolata* var. *latifolia* Nees and *B. lanceolata* Mart. ex Nees var. *lanceolata*, currently considered as heterotypic and homotypic synonyms, respectively (Chagas and da Costa-Lima 2024). However, due to taxonomic inconsistencies and a lack of updated studies on the family in South America, a revised nomenclature was proposed in 2010 (Profice 2010), which is now widely accepted (Chagas and da Costa-Lima 2024).

The date of the first record of *Justicia minensis* remains uncertain, as none of the syntype materials or their duplicates include documented collection dates. The nomenclatural revision by Profice (Profice 2010) references deposit M0186141 for the syntype Minas Gerais: Mariana, Serra da Piedade, *and Martius Unn*. However, we identified two additional deposits at GZU (GZU000250464 and its fragment GZU000262390) corresponding to the same collection. Furthermore, deposit M0186140 is attributed to a Martius collection from the same region but features a questionable collection number (Table S2) and lacks a collection date. This deposit could potentially be a duplicate of Martius' syntype; however, since we found no further evidence, we considered it an independent record. The study also identifies two collections from Minas Gerais: Morro do Pilar, *Sellow 27,37* with a B deposit and a photograph at F, as syntypes. In the online repositories we reviewed, this collection is reported as "without number"; however, we believe it corresponds to the *Sellow 37* record, as K duplicates were deposited in 1967 (Table S2). Two other K deposits are labeled as Types (Chagas and da Costa-Lima 2024), which we associate with the *Sellow 27* record, as these were deposited in 1854. Based on these findings, we consider all K deposits to be duplicates of the B syntype.

In addition to these materials, another collection was made in Serra da Piedade (*Paula 1940*) and two in Rio de Janeiro: Petrópolis, *Campos-Góes & Constantino 330* and *Sucre 2520 & Braga 361* (Table S2). Furthermore, a recent study confirmed the species occurrence in Serra Negra, in the southeastern region of Minas Gerais near the border with Rio de Janeiro, based on two collections: one from 2009 (*Ribeiro et al. 203*) and another one from 2012 (*Salimena & Nobre 3552*) (Braz et al. 2022). In 2014, a new collection was made in the same region (*Salimena 3719*). Although its identification appears inconsistent when comparing the leaf shape (oval-lanceolate) with the syntypes (narrow-elliptic), we opted to consider it a valid record for the species.

We excluded two records from the state of Mato Grosso: Ribeirão Cascalheira, *Harley et al. 10581*, and Nova Xavantina, *Giulietti 283* (Table S2). These materials are likely not *J. minensis* since they were collected in a region far from the species known zone of endemism. Additionally, there is no information on the identifier, nor available photographic records for comparison with the type, syntypes, or our own records. The collection notes for the *Giulietti 283* material also describe the plant as collected in a "wet place" and having blue flowers, which contradicts the species known morphology and occurrence reports. We also found five collections/deposits (RB8610, RB25771, RB66169, RB330913, and RB352707) recorded as *J. minensis* in the Brazilian Biodiversity Information System (SiBBr 2024), but these actually correspond to *Justicia trifoliata* Vand. ex Roem. & Schult. (JBRJ 2024). This inconsistency is likely due to a delay in system updates, as the determinations were revised to *J. trifoliata* in May 2023 (JBRJ 2024). We also disregarded one record (*Sellow Unn*. - E00957412) that lacks collection location information, originally identified as *Beloperone amherstiae* Nees (a synonym of *Justicia brasiliana* Roth), and later as *J. minensis*, which we do not agree with. Due to the highlighted inconsistencies, the actual number of collections is imprecise. Thus, we consider only nine records, which we deem consistent, to construct the distribution map for the species (Table S2).

The known occurrence range of *J. minensis* is restricted to the states of Rio de Janeiro (RJ) (Rizzini 1954) and Minas Gerais (MG) (Braz et al. 2022; Chagas and da Costa-Lima 2024) (Figure 2D; Table S2). Our record, which seems to be the most recent one, extends the occurrence zone by approximately 80 km to the north (Figure 2D). The occurrence records of the species in Serra Negra were made in *campo rupestre* and hillside forest environments (Braz et al. 2022). Although the exact collection locations are not provided, the materials from Serra da Piedade are also from high-altitude rocky environments, which is the predominant vegetation in this region. Notably, the collection notes for one of the materials from Petrópolis (*Sucre 2520 & Braga 361*) mention that the plant was "growing in a quarry". In contrast, our record was made in a Seasonal Semideciduous Montane Forest, rather than a rocky environment. This new record not only extends the known occurrence zone but also highlights the species' presence in non-rocky, high-altitude habitats.

4.1.2 | Araliaceae Juss.

Didymopanax longipetiolatus (Pohl ex DC.) Marchal

Fig.: 2B-C; Brazil: Minas Gerais: Coronel Fabriciano, *Verly 23* and *244*.

Didymopanax is a group within the *Schefflera* J.R. Forst. & G.Forst. complex, which is represented by approximately 600 described species and possibly reaching 900 when including species yet to be described (Plunkett et al. 2005; Fiaschi et al. 2008; Frodin et al. 2010; Fiaschi and Plunkett 2018). Since the broad circumscription of *Schefflera*, previously the largest genus in Araliaceae, has proven to be polyphyletic, new genera have been proposed to accommodate morphologically and geographically distinct species (Plunkett et al. 2005, 2021; Fiaschi and Plunkett 2018; Freitas et al. 2020). *Didymopanax* Decne. & Planch was established as a separate genus to accommodate species from eastern South America, but its acceptance as a clade, as well as its extension, has been historically controversial (Fiaschi and Plunkett 2018; Frodin et al. 2010). Thus, numerous bicarpellate species (a diagnostic characteristic for *Didymopanax*) have been described under *Schefflera* (Fiaschi et al. 2020).

In addition to the species that were initially misclassified under other genera and later reassigned to *Didymopanax* (Fiaschi et al. 2020), new species continue to be described, primarily from the tropical regions of Brazil, especially the Amazon (Fiaschi et al. 2008; Fiaschi and Plunkett 2016). Recently, the genus *Didymopanax*, established almost two centuries ago (Decaisne and Planchon 1854) and later synonymized under *Schefflera*, was reestablished (Fiaschi et al. 2020). The most widely accepted circumscription of the genus (Sensu Frodin 1995) includes about 40 species, most of which are distributed in Brazil (Fiaschi and Plunkett 2018; Fiaschi et al. 2020). Additionally, new genera have been proposed to accommodate taxa that did not fit well within the broader groupings of *Didymopanax* or *Schefflera* (Plunkett et al. 2021).

Some species of the genus *Didymopanax* are widely distributed, such as *Didymopanax calvus* (Cham.) Decne. & Planch., and *Didymopanax morototoni* (Aubl.) Decne. & Planch. However, other taxa are microendemic, often represented by a single collection, as seen with *Didymopanax capixabus* (Fiaschi) Fiaschi & G.M. Plunkett and *Didymopanax plurispicatus* (Maguire, Steyerm. & Frodin) Fiaschi & G.M. Plunkett (Fiaschi and Plunkett 2018). Our survey suggests that *D. longipetiolatus*, while not a part of this latter group, is also not a species with widespread occurrence.

We listed 63 records of *Didymopanax longipetiolatus*, but we excluded 16 of them (Table S2) due to inconsistencies regarding their identification. These discrepancies were related to the occurrence area, morphological incompatibility observed in the photographs of the materials, or mismatched description of the plant habit. Most of these records are likely to correspond to species such as *Didymopanax angustissimus* Marchal and *Didymopanax calvus* (Cham.) Decne. & Planch., as the collection notes describe large trees. Misidentifications of *D. longipetiolatus* and *D. calvus* have been previously reported in the literature (Fiaschi and Pirani 2007), leading to improper occurrence records. Additionally, previous studies (Fiaschi and Frodin 2006; Fiaschi and Plunkett 2018) are the probable misidentification of records from Espírito Santo: Ibatiba, *Fiaschi et al. 3117* and *Hatschbach 46679*, which, according to these authors, might belong to an undescribed, closely related, taxon. We agree with this assessment and suggest that these records likely correspond to *Didymopanax racemiferus* (Fiaschi & Frodin) Fiaschi & G.M. Plunkett, a closely related species endemic to Espírito Santo (Fiaschi and Plunkett 2018; Fiaschi and Nery 2024). We also disregarded the records from Espírito Santo: Santa Maria de Jetibá, *Kollmann et al. 5800/5893/6057*, located near Ibatiba. Although still listed as *D. longipetiolatus* in online repositories (SpeciesLink network 2024), these materials were formally identified as *Schefflera racemifera* Fiaschi & Frodin [= *D. racemiferus*] in the recent revision of the group for the Neotropics (Fiaschi and Plunkett 2018).

Among the 47 records considered valid, only the *Pohl* (*Schott*) *5368* collection was excluded from the distribution map, as it lacked information about the collection site. Among the other records, the majority (31) were collected in Rio de Janeiro, forming a strip that extends from the municipalities of Magé (S) to Santa Maria Madalena (N) (Figure 2D). This occurrence zone includes the species first collection (*Burchell 2690*), which serves as the Type material for the description of the heterotypic synonym *Sciadophyllum burchellianum* Baill., as well as the most recent collection we are aware of (*Nunes et al. 1494*) (Table S2). In Minas Gerais, 15 records were made, 12 of which are from the Juiz de Fora and Descoberto region, with one from Ouro Preto and two from the Caratinga Biological Station (Table S2). These records are from a region close to where we made our own collection.

Additionally, we found a collection from Paraguay: Departamento Alto Paraná, *Stutz Unn.* (GG-109170/1), which clearly does not belong to the species, given its endemism in Brazil (Fiaschi and Nery 2024). The deposit W[1890]0002263 is associated with a poorly documented collection by *Riedel Unn.* (Global Biodiversity Information Facility (GBIF) 2024), but it is likely a duplicate of the record from Rio de Janeiro: Serra dos Órgãos, *Riedel 326*. Finally, we found the deposits W0057471 and W18900002263, which lacked information on the collection site, collector, or record year (GBIF 2024), and were therefore excluded from the mapping.

Although there are a reasonable number *D. longipetiolatus* collections compared to other species we investigated, its records are concentrated in a few locations, forming two known distribution centers (Figure 2D). Our records, however, do not belong to either of these clusters and extends the species occurrence zone by approximately 100 km northwest of the northernmost recorded collection (*Castro et al. 641*), bringing its distribution closer to the boundery between the Atlantic Rainforest and the Cerrado. The occurrence of *D. longipetiolatus* is higher in the Ombrophilous Forests of the Atlantic Rainforest in Rio de Janeiro and Minas Gerais (Fiaschi and Pirani 2007). Although its occurrence in Semideciduous Seasonal Forests has been reported (Fiaschi and Nery 2024), few collections document this occurrence. Our records expand its distribution into Montane Semideciduous Seasonal Forests, where we not only collected individuals but also observed an established population across much of the fragment's understory. The discovery of a new population, distant from known records, and the consequent expansion of its occurrence zone, is an important finding for ensuring the species' conservation. Furthermore, it may contribute to future studies aiming a better understanding of the evolutionary, morphological, and geographic relationships within this taxonomically controversial group.

4.1.3 | Lauraceae Juss.

Persea rigida Nees & Mart.

Fig.: 3A; Brazil: Minas Gerais: Coronel Fabriciano, *Verly 212*.

Persea Mill. comprises approximately one hundred species, most of which are found in the New World, with only *Persea barbujana* (Cav.) Mabb. & Nieto Fel. and *Persea indica* (L.) Spreng. originating from islands in the Eastern Atlantic (Rohwer et al. 2009; de Moraes and Vergne 2018; de Moraes 2022; de Moraes and Brotto 2024). Furthermore, *Persea americana* Mill., native to Mexico, is widely cultivated around the world (Berdugo-Cely et al. 2023). Despite this, new species continue to be described for South America (Rohwer and van-der Werff 2023), where some *Persea* species may an important role in highly threatened ecosystems (Rojo-Cruz et al. 2023; Zamorano et al. 2023). The delimitation between *Persea* and related genera is complex, and even with phylogenetic analysis, understanding these boundaries remains controversial, complicating the placement of species within this taxonomic cluster (Rohwer et al. 2009; Li et al. 2011).

In Brazil, 26 native *Persea* species are recorded, in addition to the cultivated *P. americana* (de Moraes and Trinca 2021). This includes taxa recently described from the Atlantic Rainforest (de Moraes and Trinca 2021; de Moraes and Brotto 2024). Of this total, 17 are endemic to Brazil (Flora e Funga do Brasil 2025), and among the 14 found in the Atlantic Rainforest, six are endemic to this biome, including *Persea rigida* Nees & Mart. and the microendemic and newly described *Persea quarciticola* P.L.R.Moraes & Brotto (de Moraes and Brotto 2024).

Persea rigida is considered rare in Brazil, with its occurrence historically referred only to São Paulo (Baitello et al. 2009), where its Type material (*Sellow 652*) was collected in 1833. In this state, the species was once considered "presumably extinct" (Secretaria do Meio Ambiente do Estado de São Paulo (SMA/SP) 1998; Baitello 2003), but in the last decade, its status was updated to "critically endangered" (SMA/SP 2016). In recent years, the species' occurrence has been expanded to Paraná (Brotto et al. 2019), with six records made between 2016 and 2017 (Table S2), along with a collection from 1972 (chronologically the second one) that was only correctly identified in 2019 (SpeciesLink network 2024). Additionally, two materials collected in Minas Gerais: Camanducaia, *Leitão-Filho & Barros 10683*; Varginha, *Naves 44*; and one from Santa Catarina: Indaial, *Korte & Kniess 6986*, have been recently correctly identified, further extending the occurrence of *P. rigida* to four states. Therefore, our record represents the third for Minas Gerais and the 13th for Brazil, expanding the species' range by ~370 km northeast of the northernmost collection (*Naves 44*) (Figure 3D).

Some *Persea* species face difficulties in sexual reproduction due to predation, low seed production, and germination, making interventions using genetic rescue techniques and in vitro vegetative propagation essential for their conservation (Meneguzzi et al. 2022; Rojo-Cruz et al. 2023). Furthermore, our record also expands the possibilities for conservation practices for *P. rigida*, as the discovery of new populations may reveal greater genetic diversity within the species.

Rhodostemonodaphne anomala (Mez) Rohwer

Fig.: 3B; Brazil: Minas Gerais: Coronel Fabriciano, *Verly 21*.

Rhodostemonodaphne Rohwer & Kubitzki is a South American genus with approximately 20 species (Rohwer 1993; Quinet and Andreata 2002; Madriñán 2004). Most species occur in Brazil, with the Amazon region being the center of diversity. However, only three species are found in the Atlantic Rainforest, two of which are endemic, while *Rhodostemonodaphne macrocalyx* (Meisn.) Rohwer ex Madriñán also occurs in the Caatinga (Madriñán 1996; Flora e Funga do Brasil 2025). *Rhodostemonodaphne anomala* is one of the endemic species of the Atlantic Rainforest, with a disjunct distribution across the states of Bahia and Rio de Janeiro (Flora e Funga do Brasil 2025). This species is considered very rare and has been documented in mature Ombrophilous Forest (Fernandez and Amorim 2020).

There are only eight previous records of *R. anomala* (Table S2). Four of these were collected in Bahia, three in Rio de Janeiro, and one with no specified location. The date and location of the first record (*Glaziou 13150*), which is also the Type material, remain uncertain, but indications suggest it was collected in 1911 in Petrópolis, Rio de Janeiro. In the following century, only five new collections were made. In 2016, after a gap of more than 20 years, a new record was made in Bahia: Salvador, *Almeida Unn*. In 2022, we made the first record of the species in the state of Minas Gerais. In addition to the individual we collected, we observed the presence of other adult individuals and young plants in the regenerating stratum. However, this is not the most recent known record, as a new collection was made in Rio de Janeiro: Guapimirim, *Brotto & Völtz 5534*, in 2023.

Our record for Minas Gerais is also the first for the species in a Montane Semideciduous Seasonal Forest, expanding its range and formation of occurrence. However, this record does not affect the species' disjunct distribution, as there are no known records connecting this population with the others. The population we recorded is ~600 km from the recognized range in Bahia and ~350 km from the records in Rio de Janeiro, making it the furthest record from the Atlantic coast (~265 km) (Figure 3D).

4.1.4 | Myrtaceae Juss.

Eugenia leonorae Mattos

Fig.: 4; Brazil: Minas Gerais: Caratinga, *Verly 56*; Ipaba, *Verly 175*.

Eugenia L. is the richest genus in Myrtaceae, comprising over 1000 species, primarily distributed in the Neotropics (Mazine et al. 2016; Valdemarin et al. 2019; POWO 2025). In Brazil, 414 species are accepted for the clade, with a high rate of endemism (300 species—72.5%) (Mazine et al. 2024). The species are distributed across all biomes in the country, with a center of diversity in the Atlantic Rainforest (Lucas and Bünger 2015). The number of species in the genus is still increasing, with numerous descriptions in recent years, mainly for the Atlantic Rainforest (Sobral et al. 2012, 2015, 2017; Bünger et al. 2018; Valdemarin et al. 2019; Sobral et al. 2021; Fernandes, Prieto, et al. 2024).

Eugenia leonorae is one of the many endemic species of the genus, occurring only in the Semideciduous Seasonal Forest of the Atlantic Rainforest in Minas Gerais, Rio de Janeiro (Mazine et al. 2024), and Bahia (according to herbarium records) (Table S2). Initially described as *Calycorectes schottianus* O.Berg in 1857, the species was transferred to the genus *Eugenia*. This transfer occurred because *Calycorectes* was considered a taxonomically uncertain group within the Myrtaceae family (Landrum and Kawasaki 1997), and only calyx characteristics are insufficient for genus delimitation (Giaretta et al. 2019). This taxonomic shift is supported by the phylogenetic circumscription of *Calycorectes* under *Eugenia* (Mazine et al. 2014, 2016). Consequently, several traditional species were reassigned to *Eugenia* (Giaretta et al. 2018).

The type material and its seven duplicates (*Schott 1044*) were collected in Rio de Janeiro in 1837, but the exact municipality or collection location was not provided. Five additional collections from Rio de Janeiro were made (Table S2): São Pedro da Aldeia, *Farney et al. 4999* and *Farney et al. 4441*; Nova Friburgo, *Peron 875*; Cabo Frio, *Rezende et al. 52*; and Maricá, *Souza 3168*. There is also a record from Bahia: Itambé, *Oliveira 389* and one from

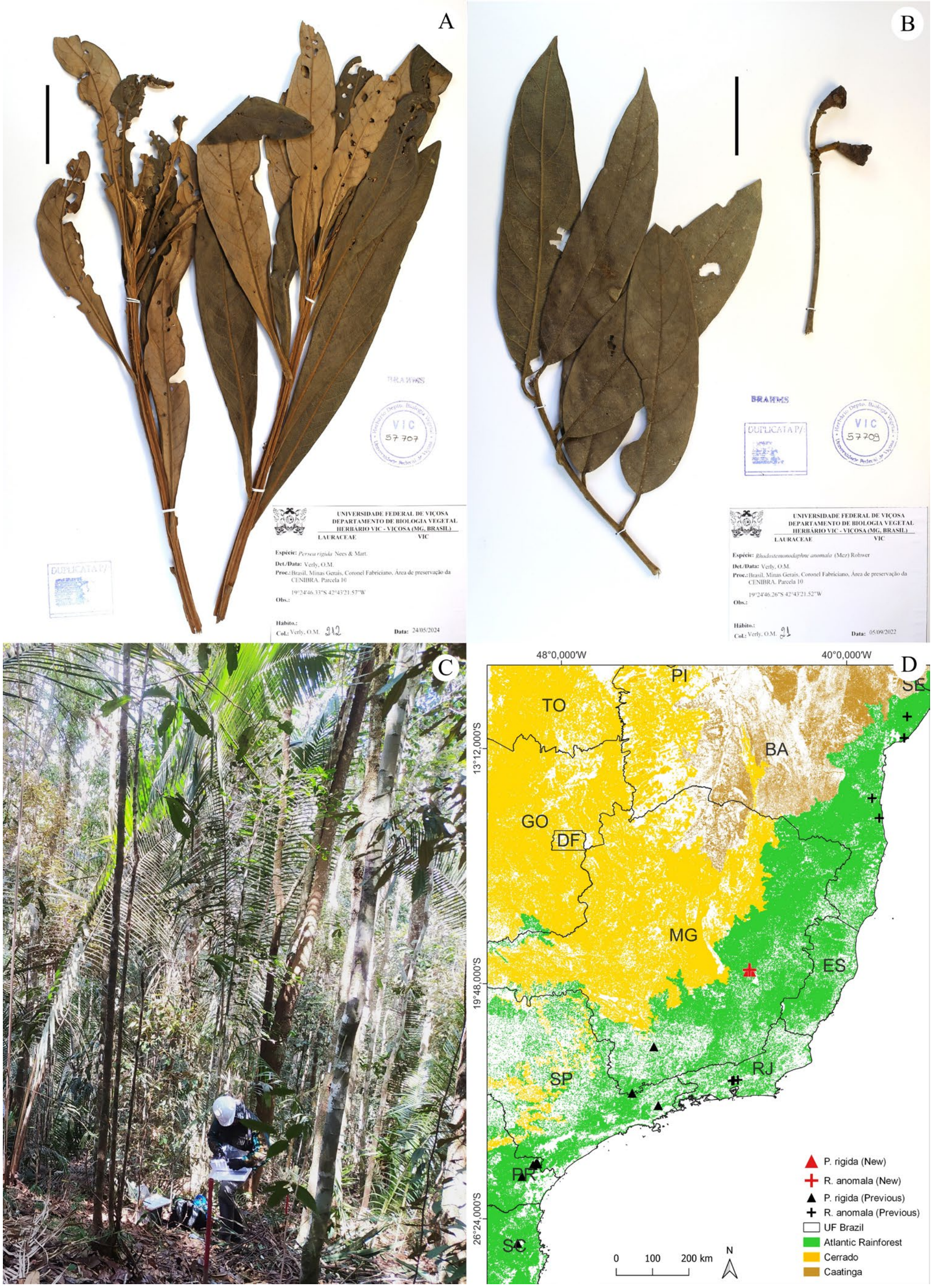
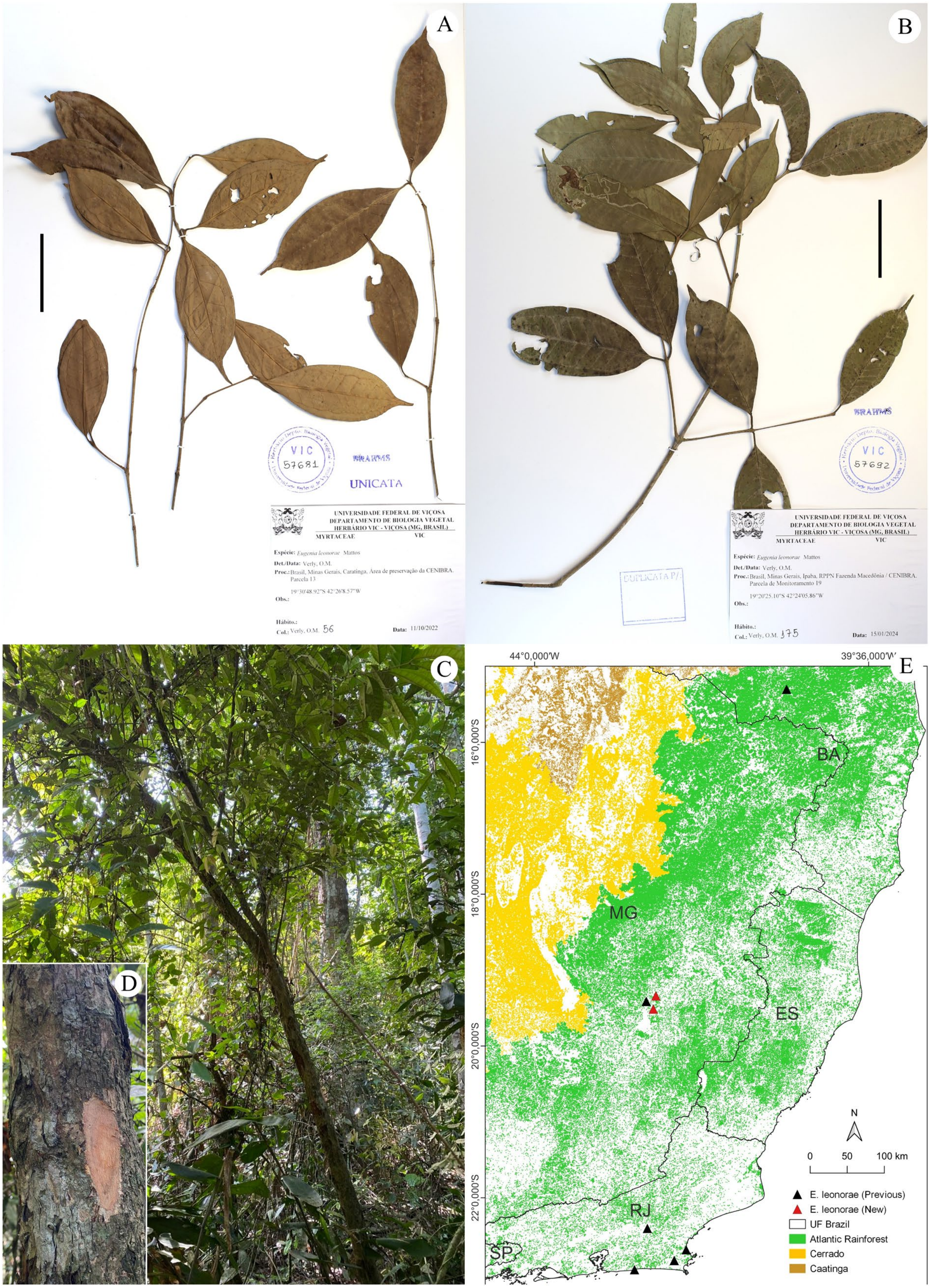


FIGURE 3 | New records for the Lauraceae *Persea rigida* Nees & Mart. (A) and *Rhodostemonodaphne anomala* (Mez) Rohwer (B, C), and their distributions in the Atlantic Rainforest (D). (A) Minas Gerais: Coronel Fabriciano, *Verly 212*; (B) Minas Gerais: Coronel Fabriciano, *Verly 21*; (C) Overview of the understory in the fragment area where the new population was recorded. Scale: 50 mm.

Minas Gerais: Santana do Paraíso, *Sobral 14571*. We included a collection from the same location as the last one (*Sobral 13571*), which is currently identified only at the genus level (SpeciesLink network 2024), but we identified it as *E. leonorae*.

We disregarded a collection from Rio de Janeiro: s. loc., *Sellow Unn*. (K000565054), as the material has two records (GBIF 2024), and in one of them (the only one with a photograph), the species mentioned is *Eugenia vattimoana* Mattos, indicating a clear inconsistency in the deposit and identification data. Cultivated individuals in Campinas, São Paulo (SP), resulted in 13 deposits (Table S2). Additionally, the collection from São Paulo: Valinhos, Estação Experimental de Valinhos (IF), *Hopkins* (*Gandolfi, S.*) *15618* also appears to come from a cultivated individual. While we recorded these deposits (Table S2), we did not consider their locations in the species' occurrence area (Figure 4D).

Our records, made in Caratinga and Ipaba, Minas Gerais, do not significantly expand the species' range, as the specimens were observed in the same region as the previous record for the state [~15 km to the northeast (*Verly 175*) and southeast (*Verly 56*)], in the municipality of Santana do Paraíso (Figure 4D). However, these new records demonstrate the existence of other



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DEPARTAMENTO DE BIOLOGIA VEGETAL
HERBÁRIO VIC - VIÇOSA (MG, BRASIL)
MYRTACEAE VIC

Espécie: *Eugenia leonorae* Mattos
Det./Data: Verly, O.M.
Proc.: Brasil, Minas Gerais, Caratinga, Área de preservação da CENIBRA. Parcela 13
19°30'48.92"S 42°26'8.57"W
Obs.:
Hábito.:
Col.: Verly, O.M. 56 Data: 11/10/2022

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DEPARTAMENTO DE BIOLOGIA VEGETAL
HERBÁRIO VIC - VIÇOSA (MG, BRASIL)
MYRTACEAE VIC

Espécie: *Eugenia leonorae* Mattos
Det./Data: Verly, O.M.
Proc.: Brasil, Minas Gerais, Ipaba, RPPN Fazenda Macedônia / CENIBRA. Parcela de Monitoramento 19
19°20'25.10"S 42°24'05.86"W
Obs.:
Hábito.:
Col.: Verly, O.M. 175 Data: 15/01/2024

FIGURE 4 | New records for *Eugenia leonorae* Mattos (Myrtaceae Juss.) (A–C) and its distribution in the Atlantic Rainforest (E). (A) Minas Gerais: Caratinga, *Verly 56*; (B) Minas Gerais: Ipaba, *Verly 175*; (C) Habitat of an adult individual (*Verly 175*) and the understory around the record. (D) Irregular outer bark and beige inner bark of an adult individual. Scale: 50 mm.

populations in the region, even if disjointed due to landscape fragmentation. Some of these populations are preserved in conservation areas, like the Private Reserve of Natural Heritage (RPPN) Fazenda Macedônia, where we observed numerous regenerating individuals of this species, which could be the focus of studies aimed at their conservation.

Eugenia reperta Sobral & Mazine

Fig.: 5A; Brazil: Minas Gerais: Caratinga, *Verly 219*.

Eugenia reperta was recently described, and the two collections used in its description, collected in Santana do Paraiso, MG, were the only previous records (Sobral et al. 2022), making it the species with the fewest collections among those we evaluated (Table S2). Although the study describing the species mentions the occurrence of other individuals in forest fragments within the municipality, no additional collections were made to document them. The study also highlights the insufficient sampling for the municipality (0.42 collections km^{-2}), suggesting that the scarcity of collections for the species may be due to a sampling bias.

The two earlier collections were initially identified as *Eugenia robustovenosa* Kiaerskou, a synonym for *Eugenia umbrosa* O.Berg, a species widely distributed in the Atlantic Rainforest (Sobral et al. 2022). This error was noted during the study for its description, but the records remain under the incorrect nomenclature in the national repository SpeciesLink (2024), revealing a concerning lack of synchronization between scientific publications, collections, and digital repositories. Even more critically, the taxon is still (September 2024) not listed in the country's largest digital catalog of plant species, Flora e Funga do Brasil (2025). The particularly alarming case of *E. reperta* in biodiversity repositories worsens the dissemination of the limited available information, jeopardizing its conservation.

Although our record was made shortly after the species was published (October 2022), it was only recently (August 2024) that we were able to precisely determine the species. Clearly, the previously mentioned inconsistencies contributed to the delay in this process. Nevertheless, this new collection expands the species' known range by ~22 km southeast of its previous records, marking the first occurrence for areas to the right side of the Rio Doce (Figure 5G).

Myrcia pseudosplendens Sobral & Mazine

Fig.: 5B-F; Brazil: Minas Gerais: Ipaba, *Verly 202*.

Myrcia DC. has undergone extensive studies in systematics and phylogeny in recent years (Lima et al. 2021). Considering the current circumscription (Lucas et al. 2018), which includes the former genera *Calyptranthes* Sw., *Gomidesia* O.Berg and *Marlierea* Cambess., it is the second largest genus of neotropical Myrtaceae in terms of species count, with nearly 800 species (Amorim and Alves 2015; Fernandes et al. 2021; POWO 2025). Brazil harbors significant diversity within this genus, especially in the Atlantic Rainforest, where approximately 260 species occur, with over 70% being endemic (Amorim and Alves 2015, 2016; Fernandes et al. 2015; Santos et al. 2024). The number of species within the genus continues to grow in the country, with the Atlantic Rainforest again standing out. In the past decade, the efforts of numerous taxonomists have led to the description of many species for this biome (Amorim and Alves 2015; Santos et al. 2015; Amorim and Alves 2016; Caliari et al. 2016; Sobral et al. 2016; Fernandes et al. 2020; Melo et al. 2020; Santos and Fernandes 2020; Fernandes et al. 2021; Fernandes, Gaem, et al. 2024; Gaem et al. 2024).

The three *Myrcia pseudosplendens* individuals collected and mentioned in the species description are from the municipalities of Marliéria and Santana do Paraíso (Sobral et al. 2016), located approximately 40 km apart, thus describing a very restricted occurrence area, as they were until then the only known records. We also identified the collection *Bortoluzzi et al. 352*, deposited in the VIC herbarium, which had previously been recorded under the former genus *Gomidesia* O.Berg, now synonymized with *Myrcia*. This collection was also made in Rio Doce State Park (PERD), in Marliéria, further confirming the species' presence within the limited occurrence. Our records made in Ipaba extend the occurrence area by ~15 km to the northeast of this known region (Figure 5G). Furthermore, our field observations in the Caratinga region have provided a better understanding of the species' restricted distribution. Importantly, the most significant outcome of these new records is the discovery of populations of the *M. pseudosplendens* in fragments of Seasonal Semideciduous Forest on the right bank of the Rio Doce, in the municipalities of Caratinga and Ipaba (Table S2).

Although the previously known populations and those recorded by us are in distinct forest fragments, they are all part of a forest massif that extends ~70 km along the Rio Doce, composed of important forest reserves such as the PERD and the RPPN Fazenda Macedônia. In the latter, we observed numerous regenerating individuals, indicating that *Myrcia pseudosplendens* is well established in the area. This finding underscores the potential for studies focusing on the species' population structure, regeneration dynamics, and long-term conservation strategies.

4.1.5 | Proteaceae Juss.

Euplassa semicostata Plana

Fig.: 6A; Brazil: Minas Gerais: Coronel Fabriciano, *Verly 206*.

Euplassa Salisb. ex Knight is a neotropical genus, with approximately 20 species distributed from the Andes mountainous regions to southeastern Brazil (Plana and Prance 1998, 2004). In Brazil, there are 15 species (Prance and Pirani 2024), with the main center of diversity for the genus being the mountainous systems in central Minas Gerais, in seasonal forest and rocky field ecosystems (Plana and Prance 1998; Centro Nacional de Conservação da Flora (CNCFlora) 2012), where regionally endemic species occur (Prance and Pirani 2024). *Euplassa semicostata* is one such species, initially considered endemic to the Espinhaço Range in Minas Gerais, but now also documented in two records from Chapada Diamantina, Bahia (Table S2) (Plana and Prance 1998; CNCFlora 2012; Fernandez and Moraes 2019). The species has a disjunct distribution in high-altitude areas,

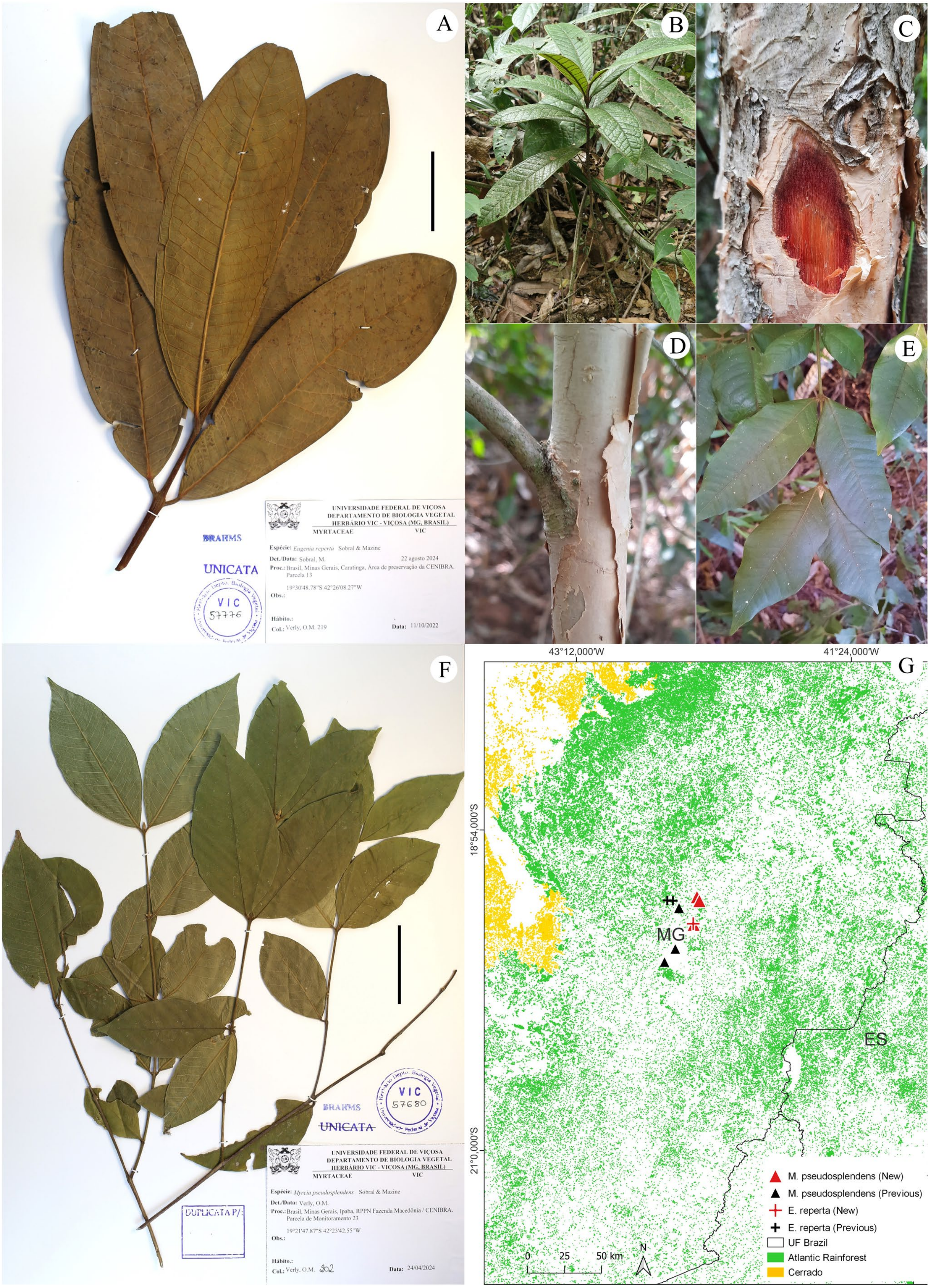

FIGURE 5 | Legend on next page.

FIGURE 5 | New records for the Myrtaceae *Eugenia reperta* Sobral & Mazine (A, B) and *Myrcia pseudosplendens* Sobral & Mazine (C–F), and their distributions in the Atlantic Rainforest (G). (A) Minas Gerais: Coronel Fabriciano, *Verly 219*; (B) Seedling approximately 30 cm tall; (C) Peeling outer bark and reddish inner bark of an adult individual; (D) Peeling outer bark in a juvenile individual; (E) Vegetative branch of a juvenile individual; (F) Minas Gerais: Coronel Fabriciano, *Verly 202*. Scale: 50 mm.

showing low density at its sites of occurrence (Versieux et al. 2011; CNCFlora 2012; Fernandez and Moraes 2019).

Affected by a recognized sampling deficiency (CNCFlora 2012; Caiafa et al. 2022), we summarized only 22 records of *E. semicostata*, with two lacking collection dates and one lacking location information. The first dated collection of *E. semicostata* (*Schwacke Unn.*) was made in 1893 in Minas Gerais: Ouro Preto: Serra de Saramenha. However, its type specimen (*Irwin et al. 28922*) was collected decades later (1971) in the Espinhaço Range (Table S2). After that, an average of one collection was made every 2 years until 2004, when two collections were made in Santana do Riacho municipality (*Ceccantini 2144/2151*). Then, there was a 19-year gap without new collections until 2023, when the record from Minas Gerais: Jequitinhonha, *Siqueira 1647* was made. In early 2024, the species was sighted and photographed in Antônio Dias municipality, *Kanouté 102*, in the Cocais das Estrelas region, the same area where we made our record in Coronel Fabriciano municipality, *Verly 206*, also in 2024. Therefore, along with Kanouté's observation, we expanded the species' known range by ~80 km east of its previously known areas in the Espinhaço Range and Serra do Caraça (Figure 6D).

Although it has been classified as "Endangered" (EN) for over a decade, efforts to study and preserve the species remain incipient, with no known conservation activities (Fernandez and Moraes 2019). The diffuse populations of *E. semicostata* are constantly threatened by fire and mining (Neves and Conceição 2010; Fernandez and Moraes 2019; Caiafa et al. 2022). In addition to anthropogenic pressures, individuals monitored over long periods did not flower, demonstrating that this species has a slow reproductive cycle (Caiafa et al. 2022). Due to their occurrence in areas of interest to mining companies, they have been the subject of studies on flowering and fruiting induced by growth regulators (Caiafa et al. 2022). These initiatives may contribute to its conservation, since state legislation mandates that for each tree of an EN species that is cut, 20 must be planted as environmental compensation (Secretária de Estado de Meio Ambiente e Desenvolvimento Sustentável e a Diretora-Geral do Instituto Estadual de Florestas, Minas Gerais (SEMAD/IEF/MG) 2021).

4.1.6 | Rubiaceae Juss.

Duroia valesca C.H.Perss. & Delprete

Fig.: 6B; Brazil: Minas Gerais: Ipaba, *Verly 48* and *242* (same individual).

Duroia L.f. is a small Neotropical genus with only 26 species, 24 of which are found in Brazil, mainly in the Amazon (Persson and Delprete 2010; The Brazil Flora Group (BFG) 2015; Nascimento et al. 2019), where most species have a very restricted distribution (Flora e Funga do Brasil 2025). The group is poorly studied, with little information available on its taxonomic relationships. However, the genus exhibits some peculiar and more widely reported ecological traits, such as the allelopathic effect of *Duroia hirsuta* (Poepp. & Endl.) K.Schum. (Campbell et al. 1989; Page et al. 1994) and its mutualism with ants (Pfannes and Baier 2002; Frederickson 2005; Frederickson and Gordon 2007, 2009; Báez et al. 2016; Salas-Lopez et al. 2016) or the importance of *Duroia velutina* Hook.f. ex K.Schum. in the diet of Amazonian primates (Barnett et al. 2012).

Duroia valesca, the only species of the genus in the Atlantic Rainforest and endemic to the biome (Flora e Funga do Brasil 2025), was also the last to be described in the group, over a decade ago (Persson and Delprete 2010). This species closely related to *D. velutina* and has been historically confused with it, leading to numerous misidentified collections from the biome (SpeciesLink network 2024). Identification conflicts are recurrent in the group, with incorrect identifications even involving paragenetic and congeneric species (Taylor 2002). Considering this, we consider all collections from the Atlantic Rainforest identified as *D. velutina* or only at the genus level up until our survey as valid records of *D. valesca*.

We listed 35 collections that presumably are *D. valesca* (Table S2), of which we disregarded two (*Pinheiro Unn.* and *Ramalho 1050*) from cultivated individuals in the arboretum of the Dendrology Sector of the Federal University of Viçosa. Although cultivated in an area of Atlantic Rainforest, the origin of these individuals is uncertain, with some indications that the seeds may have been sourced from the Amazon. The remaining collections (33) are distributed across three main forest reserves: (i) Reserva Natural Vale (10); (ii) Estação Biológica de Caratinga (5); and (iii) PERD (4) (Figure 6D). Some scattered collections in the state of Bahia form a zone of occurrence for the species between the municipalities of Camacã (S) and Uruçuca (N). Besides the first collection of the species, Bahia: Itamaraju, *Monteiro 23575*, made in 1971, remains the only record in the southern part of the state, about ~140 km away from the closest collections. Our record was made near PERD, in a contiguous but isolated forest fragment separated by the Rio Doce. Thus, we extended the occurrence range of *D. valesca* in this region by ~20 km northeast of the previous northernmost record at PERD (*Sucre et al. 10155*).

4.1.7 | Simaroubaceae DC.

Homalolepis floribunda (A.St.-Hil.) Devecchi & Pirani

Fig.: 7A; Brazil: Minas Gerais: Guanhães, *Verly 72*.

Homalolepis Turcz. is the largest genus in Simaroubaceae, with 28 species restricted to South America (Devecchi et al. 2018a;

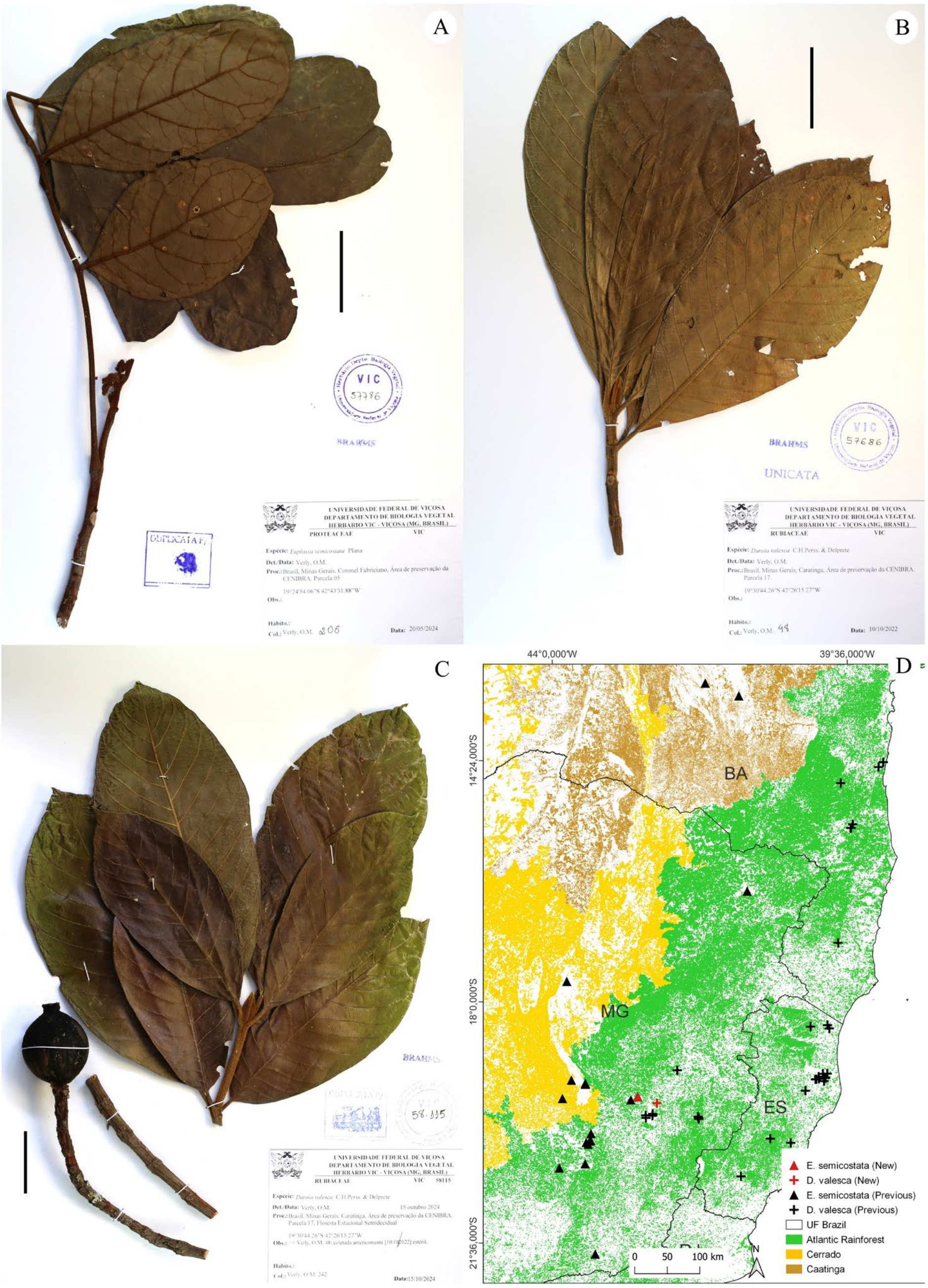

FIGURE 6 | New records for *Euplassa semicostata* Plana (Proteaceae Juss.) (A) and *Duroia valesca* C.H.Perss. & Delprete (Rubiaceae Juss.) (B, C), and their distributions in the Atlantic Rainforest (D). (A) Minas Gerais: Coronel Fabriciano, *Verly 206*; (B) Minas Gerais: Ipaba, *Verly 48*; (C) Minas Gerais: Ipaba, *Verly 242*. Scale: 50 mm.

Devecchi et al. 2022; POWO 2025). The genus was reestablished and received a new circumscription to include several species previously treated under *Simaba* Aubl., which was shown to be non-monophyletic (Devecchi et al. 2018c, 2018d), and was previously the largest genus in the family (Devecchi and Pirani 2016). The taxonomic distinction of species within this generic complex is challenging, requiring detailed studies of floral morphology and phylogeny to reveal evolutionary trends and relationships among the taxa (Alves et al. 2017; Devecchi et al. 2018b; Devecchi et al. 2018d; Alves et al. 2022). Molecular advances have elucidated the complex relationships of the clade and led to the description of new species (Devecchi and Pirani 2015; Devecchi et al. 2016; Devecchi, Thomas, and Pirani 2018a, 2018b). Some of these were previously considered as morphological variations within an initial taxon, where monophyly has not been proven (Devecchi et al. 2018b, 2018c).

An example is the species *Homalolepis floribunda* (= *Simaba floribunda* A.St.-Hil.) and *Homalolepis cuneata* (A.St.-Hil. & Tul.) Devecchi & Pirani (= *Simaba cuneata* A.St.-Hil. & Tul.), which are morphologically closely related (Devecchi et al. 2018c; Devecchi and Pirani 2020), wherein the former belongs to a sister lineage of the latter (Devecchi et al. 2018c). This similarity has led to questionable identifications of materials collected throughout the distribution range of these species. This occurred with the material from Minas Gerais: Barbacena, *Glaziou 12525*,

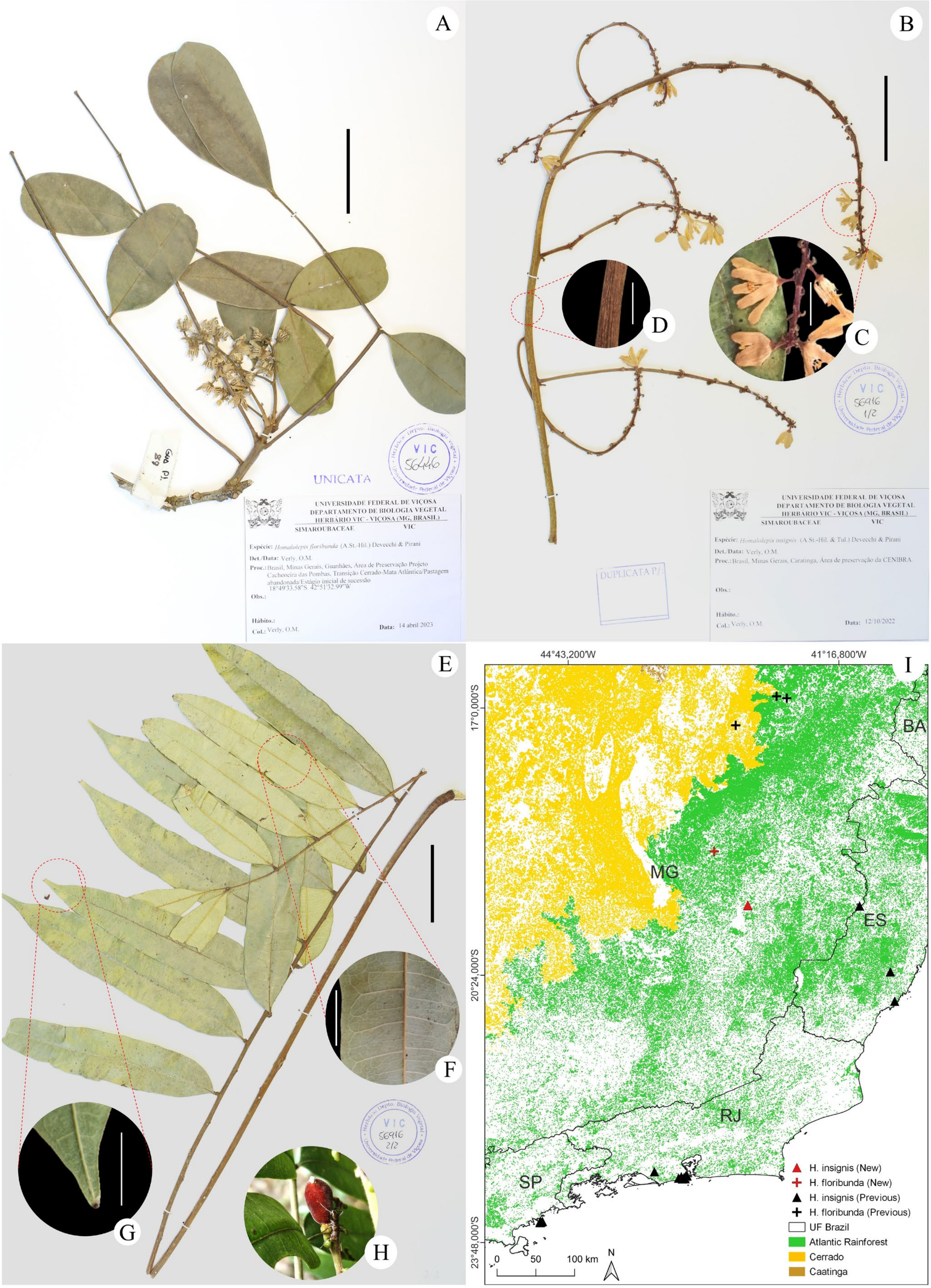

FIGURE 7 | Legend on next page.

FIGURE 7 | New records for the Simaroubaceae *Homalolepis floribunda* (A.St.-Hil.) Devecchi & Pirani (A) and *Homalolepis insignis* (A.St.-Hil. & Tul.) Devecchi & Pirani (B–H), and their distributions in the Atlantic Rainforest (I). (A) Minas Gerais: Guanhães, *Verly 72*; (B–G) Minas Gerais: Caratinga, *Verly 54*. (B–D) Inflorescence. (B) Part of the inflorescence with secondary and tertiary branching; (C) Flower details; (D) Details of secondary branching. (E–G) Leaf. (E) Whole leaf, leaflets showing both sides; (F) Detail of primary and secondary veins on the abaxial side of the leaflet; (G) Detail of the leaflet apex; (H) Developing fruit (in vivo). Scale: White 10 mm, black 50 mm.

deposited as *S. cuneata*, and confirmed in 1980 by Cavalcante. In 1989, the material was reevaluated by Franceschinelli and Thomas and identified as *S. floribunda*, who added a note that the circumscription adopted for the species included *S. cuneata*. Similar situations were observed for the collections: Espírito Santo: Governador Lindenberg, *Demuner et al. 2791*; Nova Venécia, *Amorim 7550*; Conceição da Barra, *Farney 4747*; and Linhares, *Silva 9*, among others (Table S2). In both cases, the correct identification is *H. cuneata*, and the records were excluded from our mapping process. Additionally, we disregarded two other collections from Espírito Santo: Linhares, *Sartori et al. 204*; and Presidente Kennedy, *Gomes 4261*; as a recent study reviewed the occurrence of Simaroubaceae for the state and did not report the occurrence of *H. floribunda*, representing other collections of *H. cuneata*.

H. floribunda is described as endemic to the state of Minas Gerais, being found in transitional vegetation between the Atlantic Rainforest and Cerrado, in enclaves of Seasonal Semideciduous Forest and Dry Deciduous Forest (Devecchi et al. 2018c). We also disregarded all these records from the states of Acre, Alagoas, Bahia, Espírito Santo, Paraíba, Rio de Janeiro, Rio Grande do Norte, and Sergipe (Table S2). Most collections from the Northeast, the probability is *Homalolepis bahiensis* (Moric.) Devecchi & Pirani and *Homalolepis ferruginea* (A.St.-Hil.) Devecchi & Pirani, according to the collection history of the group.

Such inconsistencies related to the correct identification of collections make the number of records of the species uncertain. These problems are especially concerning this genus, which includes several species classified as threatened (Devecchi et al. 2018a). A notable portion of these species is also microendemic (Devecchi et al. 2018c), with very restricted distribution and known records only for the type locality and its immediate surroundings (Devecchi and Pirani 2015; Devecchi et al. 2016; Devecchi et al. 2018a). Previously, *H. floribunda* was recognized only from the type locality, Minas Novas, and the surroundings of Araçuaí, municipalities in northeastern Minas Gerais (Flora e Funga do Brasil 2025). Our record for Guanhães, Minas Gerais, expands its distribution by ~360 km south of the previously known distribution area, but it remains restricted to Seasonal Semideciduous Forests near the Cerrado-Atlantic Rainforest ecotone (Figure 7H).

Homalolepis insignis (A.St.-Hil. & Tul.) Devecchi & Pirani

Fig.: 7B-G; Brazil: Minas Gerais: Caratinga, *Verly 54* and *236*.

Homalolepis insignis was also initially described under the genus *Simaba* Aubl., until the new circumscription of the complex reallocated it under *Homalolepis* Turcz. (Devecchi et al. 2018c, 2018d). During our research, 43 records for the species were found (Table S2). Two deposits, P00401122 and SPFW5101, were disregarded due to the lack of records of the collection location and the absence of photographic material for comparison. Most of the remaining records are dated to the 19th and 20th centuries, concentrated in the Tijuca National Park (Parna-Tijuca), Rio de Janeiro, and the Ilha Anchieta State Park (PEIA), São Paulo. These records align with the described occurrence region for the species, which was limited to the interior of coastal rainforests and sandy restingas (Devecchi and Pirani 2020) (Figure 7H), noting that the species is notably allopatric (Devecchi et al. 2018c).

The collection Minas Gerais: Serra do Caraça, *Glaziou 15889*, was previously cited as a possible record of this species; however, the identification was indicated as certainly incorrect (Devecchi et al. 2018c). Thus, our collections from the municipality of Caratinga are the first record of *H. insignis* for the state of Minas Gerais, expanding the species' range by ~150 km to the west. These new records and the recent collection from Espírito Santo: Baixo Guandú, *de-Marcelino 37*, which are also distant from the coastal strip, are the first records of the species outside its typical phytophysiognomy, occurring in areas of Seasonal Semideciduous Forest.

4.2 | Comments on Botanical Collections

During our investigation on digital biodiversity repositories, we observed several inconsistencies regarding the collections of the rare and threatened species under study. A primary issue was the occurrence of erroneous identifications or records identified only at the genus level. These misidentifications in herbarium collections can lead to inaccurate species distribution maps and hinder the description of new species (de Moraes and Brotto 2024). While many of the cited specimens (Table S2) may currently be correctly identified in physical collections, they have not yet been updated in digital repositories. This delay is likely due to structural and human resource limitations that prevent immediate synchronization, as observed in the collections associated with *Justicia minensis*. Therefore, we chose to list all records available in the digital repositories associated with the evaluated species. In doing so, this study also provides an indirect catalog of deposits that should be reviewed in physical collections and, when necessary, corrected in digital databases.

We repeatedly identified inconsistencies in records of *Duroia valesca*. Several collections were initially misidentified as *Duroia triflora* Ducke or *Duroia velutina* Hook.f. ex K.Schum. (SpeciesLink network 2024), or continue to be as such, as both species are restricted to the northern Brazilian Amazon (Flora e Funga do Brasil 2025). Similar issues were observed with *Didymopanax longipetiolatus* and *Homalolepis floribunda* further complicated by the outdated nomenclature, resulting in widespread reliance on synonyms rather than valid names for many records. These observations align with previous assessments of biodiversity repositories, which revealed that most taxonomic inconsistencies are attributed to the persistent use of scientific synonyms (Freitas et al. 2020; Greeff et al. 2022). Such inconsistencies are the critical need for ongoing curation of both physical and digital collections. Accurate nomenclatural updates are essential to rectify species distribution errors, particularly for those that are threatened, rare, or microendemic. By ensuring proper taxonomic adjustments in herbarium collections, researchers can delineate occurrence areas with greater precision, enhancing our understanding of species conservation statuses, therefore, facilitating the development of targeted conservation strategies (Freitas et al. 2020).

We also detected numerous inconsistencies in the georeferencing of collections. Most specimens collected before 1940 lack geographic coordinates or detailed information about the collection site. While newer records are generally more detailed, many still lack precise geographic data, which may be related to the absence of GPS equipment and the use of non-standardized methods by amateur collectors. Additionally, we observed that many records had their original coordinates altered during the digitization process of the physical collections. In these cases, erroneous coordinates inferences were often made, frequently defaulting to the central urban area of the municipality listed in the record. Accurate georeferencing is crucial for determining the precise occurrence areas of species, particularly for mapping microendemic species and updating their extinction risk categories (Cruz et al. 2022).

Although digital biodiversity repositories operate under ongoing uncertainty regarding their long-term support (Merali and Giles 2005) and are subject to the data limitations already mentioned, they must be consistently supported and improved by the scientific community through systematic studies that highlight their main gaps and propose interventions (Freitas et al. 2020; de Araujo et al. 2022; White et al. 2023). Maintaining and enhancing these repositories is vital for providing reliable data on species distribution and conservation status, which supports biodiversity metrics creation. These metrics, in turn, enable informed decision-making and policy development, which are fundamentally important in megadiverse countries like Brazil (Canhos et al. 2014; Pimm et al. 2014; Sousa-Baena et al. 2014; The Brazil Flora Group (BFG) 2018; Freitas et al. 2020). From this perspective, institutions and researchers should prioritize the organizational and nomenclatural updating of herbarium collections and the rapid processing of new records for rare and threatened species. All these improvements must be promptly reflected in digital collections to ensure the accuracy of the data shared at large scale (Greeff et al. 2022).

We listed 16 records collected from cultivated individuals of three species (Table S2), which were not included in our mapping. Among these, the records of Myrtaceae *Eugenia leonorae* (13) and *Myrcia pseudosplendens* (1) appear to be taxonomically consistent. On the other hand, we were unable to verify the two records for *D. valesca*, originating from the living collection of the arboretum of the Department of Dendrology at the Federal University of Viçosa. This occurred because the origin of the propagules that gave rise to the individuals is uncertain, potentially being of Amazonian origin, and they could not pertain to *D. valesca* (see comments on the species). Living collections cultivated in arboretum can be a suitable strategy for the *ex situ* conservation of rare plant species. However, the success of this approach can be limited for species with specific ecological requirements, particularly those unable to complete their reproductive cycles in cultivation (Гордеева 2021). Effective ex situ conservation of rare and threatened species relies on the establishment and management of well-documented, genetically diverse living collections, where the provenance of each plant is clearly known (Christe et al. 2014; Arnet et al. 2015; Cavender et al. 2015).

The species we evaluated had an average of 20.25 previous collections. Particularly, *Eugenia reperta* and *Myrcia pseudosplendens* were poorly represented, with two and four records, respectively. This aligns with previous studies that highlight the significant sampling gap in tropical flora, where hundreds of species are represented by just a single collection (de Araujo et al. 2022). Such a lack of botanical collections is critical, as it hinders the description of potentially threatened new species and limits taxonomic and ecological knowledge of already identified rare and threatened species (Gentry 1986; Oliveira et al. 2016; Leão et al. 2023). While the literature suggests that botanical collectors often prioritize rare species over common ones (ter Steege et al. 2011), our findings revealed an opposite trend. Most species in our database showed higher representation in herbarium collections when they were common in the evaluated areas. To address human biases in botanical collecting, which contribute to sampling (Wallacean) and taxonomic gaps, a promising approach is the integration of herbarium data with e-infrastructures that leverage community observations, as this citizen science strategy has the potential to enhance data coverage and reduce these gaps (White et al. 2023).

4.3 | Implications for the Conservation of Rare and Threatened Species

We present new records, with herbarized materials incorporated into the VIC herbarium collection, and field observations of rare, endemic, and threatened species from the Atlantic Rainforest (Table S2). Many endemic species in this biome exhibit strictly local or microregional distributions (de Moura et al. 2022; Leão et al. 2023; Sant'Anna-Santos 2023; Zavatin et al. 2023). This pattern is often attributed to ecological specialization, further exacerbated by the geographic isolation caused by the mountainous landscape (Werneck et al. 2011; de Moura et al. 2022; Nery et al. 2023). Conducting new collections of endemic species in poorly sampled areas is crucial for refining the delineation of their occurrence ranges and for reducing sampling biases (Werneck et al. 2011).

These records are essential for advancing species conservation, as the scarcity of collections and information limits the assessment of conservation status (Sobral et al. 2012; Sousa-Baena et al. 2014; de Moura et al. 2022). Sampling deficiency and lack of population studies are some of the factors that lead to a classification of DD (Data Deficient) according to the International Union for the Conservation of Nature (IUCN) conservation criteria (IUCN 2001; Canhos et al. 2014; IUCN 2019). Among the species we recorded, *Eugenia reperta* and *Myrcia pseudosplendens* were designated in this category during their description due to the lack of sampling effort within their occurrence range (Sobral et al. 2016, 2022) as well as the scarcity of known collections (Table S2). Therefore, new field campaigns should be conducted to improve the accuracy of these species distribution and support studies that determine their actual threatened categories (Werneck et al. 2011; Canhos et al. 2014; Cruz et al. 2022). These population and environmental studies should also be periodically conducted to reassess threat categories according to IUCN criteria, even for species previously evaluated (Foggi et al. 2015).

Despite having a minimally adequate dataset, species such as *Justicia minensis* have not yet been evaluated regarding their conservation status by the International Union for Conservation of Nature (IUCN 2024) or by the National Center for Flora Conservation (Canhos et al. 2014; CNCFlora 2024). Furthermore, we observed significant discrepancies among official scientific sources regarding the conservation status of plant species (CNCFlora 2024; see Table 2). For instance, *Persea rigida* was assessed as NT (Near Threatened) in 2021 (Amorim and Aragão 2021; Flora e Funga do Brasil 2025), but this has not yet been updated on the CNCFlora portal (2024), where the species remains as DD, based on a 2012 assessment. Additionally, *P. rigida* has not been included in the "Official National List of Species Threatened with Extinction" (LNFA), last updated in 2022 (Brasil 2022). Similarly, *Homalolepis insignis*, reassessed in 2019 as CR (critically endangered) (da Rosa and Fernandez 2019), continues to be listed as LC (Least Concern) on CNCFlora even though it was included in the latest LNFA update (Brasil 2022). Our newly recorded data and the correction of inconsistencies in the collections of rare species we evaluated will help ensure that they are properly evaluated. Additionally, legislative authorities must urgently consider incorporating these species into the official threatened species lists to facilitate the timely establishment of effective conservation policies.

Although ecological knowledge about rare and endangered species remains incipient, some technical documents indicate the use of species such as *Eugenia leonorae* for forest restoration in Conservation Units in the state of Rio de Janeiro (CNCFlora 2018). Recommendations for this purpose can be multifaceted, presenting positive aspects, such as the development of novel studies on the ecology and conservation of these species, or negative aspects, such as the excessive collection of seeds for seedling production. Therefore, new records of rare and endangered species are fundamental to support studies and methods for seedling production and other techniques for in situ or *ex situ* conservation (Kusuma et al. 2019; Lima et al. 2022; Gaier and Resasco 2023).

Myrtaceae is one of the most prominent families in the Atlantic Rainforest, where it is the center of diversity for *Myrcia*, its largest exclusively neotropical genus (dos Santos et al. 2021). It comprises numerous species of food and medicinal interest, which are locally exploited, even though commonly threatened (Farias et al. 2023). Thus, it constitutes an ecologically important group for threatened environments in the Atlantic Rainforest (Lucas and Bünger 2015; Staggemeier et al. 2017). Furthermore, the high rate of endemism in the Atlantic Rainforest establishes a high conservation value for its forest fragments (Rocha and Amorim 2012; Amorim and Alves 2015; Staggemeier et al. 2015). This has been demonstrated by our new records for the family, which we collected in highly conserved fragments of Semideciduous Seasonal Forest of the Atlantic Rainforest. These environmentally stable fragments are responsible for sustaining regional diversity and enhancing the phylogenetic understanding of endemic species and their relationships with taxa from other biomes (Staggemeier et al. 2015; dos Santos et al. 2021).

During our field expeditions, we also observed regenerating individuals of *Didymopanax longipetiolatus*, *E. leonorae*, *E. reperta*, *M. pseudosplendens*, and *Rhodostemonodaphne anomala*. These findings suggest that these forest fragments are ecologically stable, providing favorable conditions for the growth and persistence of these populations. This is particularly significant for rare and endemic species, which often require specific environmental conditions for germination and establishment and may encounter challenges in seedling recruitment even within their natural habitats (Wen and Yang 2021; Sánchez-Martín et al. 2023). Although multiple environmental factors drive plant population dynamics, the density of rare species may be more strongly influenced by anthropogenic factors, such as fire, than by edaphic characteristics, for example (Sritharan et al. 2023). Therefore, the presence of these species highlights ecologically complex and conservation-priority areas, critical for preventing the extinction of rare and threatened taxa (Moneo et al. 2022). Therefore, the conservation of critical habitats through conservation units, such as the RPPN Fazenda Macedônia, is essential to maintain the environmental conditions necessary for the survival and perpetuation of these threatened species.

Author Contributions

Otávio Miranda Verly: data curation (lead), formal analysis (lead), investigation (lead), methodology (lead), validation (lead), writing – original draft (lead). **Luiz Claudio Medeiros Cabral-da-Silva:** data curation (equal), investigation (equal), writing – original draft (equal). **Marcos Sobral:** formal analysis (equal), writing – review and editing (equal). **Klisman Oliveira:** data curation (equal), formal analysis (equal), writing – review and editing (equal). **Laura Beatriz Assis Teixeira:** writing – review and editing (equal). **Maria Paula Miranda Xavier Rufino:** writing – review and editing (equal). **Aline Ferreira de Mendonça:** data curation (equal). **Kesleyane Pereira Camilo:** writing – review and editing (equal). **Carlos Moreira Miquelino Eleto Torres:** conceptualization (equal), funding acquisition (equal), methodology (equal), project administration (equal), resources (equal), supervision (equal), validation (equal), writing – review and editing (equal).

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Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that supports the findings of the study are available as Tables S1 and S2. Other data will be made available on request.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section.

Supplementary Materials

Table S1. Botanical collections of different life forms, in different Atlantic Rainforest fragments, Minas Gerais, Brazil.

Fragment	**Richness**			**Botanical collections: total/deposited in herbarium/presented in this study**					
	Species	**Genera**	**Families**	**Trees**	**Shrubs**	**Palms**	**Lianas**	**Herbs**	**Total**
CAL	242	152	61	158/20/0	3/3/0	1/1/0	2/2/0	8/8/0	172/34/0
GAR	119	95	42	17/0/0	0/0/0	0/0/0	0/0/0	0/0/0	17/0/0
GUA	226	139	55	127/11/1	2/2/0	0/0/0	2/2/0	3/3/0	134/18/1
IP1	174	105	47	67/9/1	2/1/0	0/0/0	2/2/0	0/0/0	71/12/4*
IP2	140	98	42	51/14/4	9/6/2	0/0/0	3/3/0	2/2/0	65/25/6
MAC	254	159	51	57/19/2	14/11/0	0/0/0	5/5/0	6/5/0	82/40/2
MSI	182	125	50	93/14/0	12/7/0	0/0/0	0/0/0	2/2/0	107/23/0
SAB	177	111	50	96/8/0	10/7/0	0/0/0	3/3/0	8/8/1	117/26/1
SJO	304	157	60	210/28/3	16/11/2	1/1/0	4/4/0	13/10/0	244/54/5
Total	926	272	80	876/123/10	68/48/4	2/2/0	21/21/0	42/38/1	1,009/232/15(4*)

* Only human observation. See details in Tabel S2.

Table S2: Records in the online portals GBIF, JABOT, and SpeciesLink for extremely rare species (and their respective synonyms) of the Atlantic Rainforest collected in forest fragments in the state of Minas Gerais, Brazil, ordered by year of collection and collector.

Deposits (Related Codes)	**Country of Deposit**	**Collector (Collection Number)**	**Year of Collection**	**Municipality**	**State**	**Latitude Longitude**	**Location Notes**	**Comments**
Acanthaceae Juss. ***Justicia minensis* Profice** **Synonyms:** **basionym: *Beloperone lanceolata* Mart. ex Nees** **homotype: *Beloperone lanceolata* Mart. ex Nees var. *lanceolata*** **heterotypic: *Beloperone lanceolata* var. *latifolia* Nees**								
M0186140 (595705/265097)	Germany	Martius, C.F.P.v. (1218/660?)	-	Mariana	MG	19°49'19"S 43°40'38"W	Serra da Piedade.	-
GZU000250464$^{T/S}$ (GZU000262390) **M0186141$^{T/S}$ (595706/ 265098)**	Germany	Martius, C.F.P.v. (Unn.)	-	Mariana?	MG	19°49'19"S 43°40'38"W	Serra da Piedade. in sylvis ad Mariana; Provinciae Min. Ger. [Minas Gerais].	GZU000262390 is a fragment of GZU000250464.
B242734^{S} (F8932) **K000529339$^{S▼}$** **K000529343**	Germany United Kingdon	Sellow, F. (Unn./37?)	≤1854	Morro do Pilar	MG	19°13'23"S 43°22'47"W	-	Syntype B destroyed, photography F 8932!.
RB49364 (RB00598784)	Brazil	Campos-Góes, O.; Constatino, D. (330)	1944	Petrópolis	RJ	22°24'02"S 43°12'32"W	Maria Comprida.	-
P04561495 ** **NY00930019** **MO1113059**	France USA	Harley, R.M.; et al. (10581)	1968	Ribeirão Cascalheira	MT	13°22'56"S 50°51'09"W	Acampamento Base da Expedição. c 8 km ao S do acampamento base. Terreno pantanoso por córrego pelo Campo.	Collection outside the species' known occurrence zone. There is no indication of the identifier. There are no available photographs of materials for comparison.
IPA17592 **	Brazil	Giulietti, A.M. (283)	1968	Nova Xavantina	MT	14°48'43"S 52°17'56"W	Beira da estrada Barra do Garças - Xavantina. Lugar úmido, capoeira alta.	Collection outside the known range of occurrence of the species. Additionally, in the material description, blue flowers are mentioned, which is not consistent with the morphology of the species, and the description of the location as a "humid place" is

Deposits (Related Codes)	**Country of Deposit**	**Collector (Collection Number)**	**Year of Collection**	**Municipality**	**State**	**Latitude Longitude**	**Location Notes**	**Comments**
								not consistent with occurrence reports.
RB138680 (RB00598787) HB#	Brazil	Sucre, D. (2520); Braga, P.I.S. (361)	1968	Petrópolis	RJ	22°24'25"S 43°08'59"W	Estrada do contorno. Crescendo em pedreira.	-
BHCB6577	Brazil	Paula, -.; et al. (1940)	1985	Caeté	MG	19°49'48"S 43°40'26"W	Serra da Piedade. Transição para Cerrado.	-
CESJ054208	Brazil	Ribeiro, J.H.C.; et al. (203)	2009	Olaria	MG	21°57'60"S 43°53'60"W	Serra Negra, Serrinha, Sítio do Rinaldo Degredo. Poços do Sr. Altair.	-
CESJ062811	Brazil	Salimena, F.R.G.; Nobre, P.H. (3552)	2012	Olaria	MG	21°57'60"S 43°53'60"W	Sítio Degredo; Interior de mata, em encosta.	-
CESJ064076	Brazil	Salimena, F.R.G. (3719)	2014	Rio Preto	MG	21°58'55"S 43°55'16"W	RPPN São Lourenço do Funil, Campo rupestre arenoso à direita da sede.	It is probably not the species, as the leaves are smaller and oval vs. lanceolate in syntypes. It's probably the material cited in Braz et al. (2022).
VIC056564 *	Brazil	Verly, O.M. (101)	2023	Sabinópolis	MG	18°33'43"S 43°10'40"W	Área de preservação da CENIBRA. Floresta Estacional Semidecidual. Topo de morro.	-
Araliaceae Juss. ***Didymopanax longipetiolatus* Pohl ex DC.) Marchal** **Synonyms:** **basionym: *Panax longipetiolatus* Pohl ex DC.** **homotype: *Didymopanax longepetiolatum* (Pohl ex DC.) Marchal** ***Schefflera longipetiolata* (Pohl ex DC.) Frodin & Fiaschi** **heterotypic: *Sciodaphyllum burchellianum* Baill.**								
P02286451 K000588177 IAN 26607	France United Kingdon Brazil	Burchell, W.J. (2690)	1826	Magé	RJ	22°05'21"S 43°49'40"W	IBB/Magé, RJ, Frechal to Majé.	Heteronym Type.
M-0172434^{T} (544867/227518) G00329615^{H}	Germany Switzerland USA	Pohl, J.B.E. [Schott, H.W.?] (5368)	1828/9	-	-	-	-	The deposit/code G-DC-263365/1 is mentioned indicating Holotype, probably

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
H[NY]1582692^{Ph} **MO103155138** **MO103155139^{I}** **W0057469** **W0057470**	Austria							a new numbering for Holotype G.
K000588176 **P03379046** **P03379047**	United Kingdon France	Riedel, L. (326)	1832	Teresópolis?	RJ	22°26'41"S 43°01'46"W	Habitat in silvaticis umbosis provinciae Rio de Janeiro ad radicem montium Orgãos.	-
K#	United Kingdon	Miers, J. (4031)	1878	Teresópolis?	RJ	22°31'55"S 43°05'18"W	Organ Mountains.	Collection was mentioned in the Fiaschi and Pirani (2007) review.
R34677 (R010065344▼ R010065346)	Brazil	Palma, M. (Unn.)	1883	Teresópolis?	RJ	22°25'30"S 42°54'36"W	[Estrada do Rio] Socavão, Bananal.	-
R9803 **P03379045**	Brazil France	Glaziou, A.F.M. (20345a)	1892	Rio de Janeiro	RJ	22°57'13"S 43°18'04"W	Floresta da Tijuca, au Bico do Papagaio.	-
OUPR523 **	Brazil	Schwacke, C.A.W. (8229)	1892	-	MG	19°13'36"S 43°31'46"W	Serra do Cipó.	Disregarded because it does not come from known areas where the species occurs, in addition to not being mentioned in any of genera revision, nor presenting a photograph for comparison.
R34679 (R010065339▼ R010065342)	Brazil	Brade, A.C.; Santos-Lima., J.S. (11588)	1932	Santa Maria Madalena	RJ	21°59'14"S 41°52'50"W	Santo Antônio de Embu [Imbé].	-
SP52787 **IAC57383**	Brazil	Heringer, E.P. (1839)	1945	Juiz de Fora	MG	21°37'44"S 43°16'53"W	Fazenda do Sertão, via Cel. Pacheco.	-
CESJ#	Brazil	Kringer, -. (16511)	1956	Juiz de Fora	MG	21°37'44"S 43°16'53"W	-	Collection was mentioned in the Fiaschi and Pirani (2007) review.
RB378827 (RB01185081)	Brazil	Guedes, R.R.; Gonzaga, L.P. (10)	1979	Magé	RJ	22°33'32"S 43°03'59"W	-	-
RB270078 (RB01185071)	Brazil	de-Lima, H.C.; Martinelli, G. (1725)	1982	Silva Jardim	RJ	22°32'58"S 42°17'45"W	Reserva Biológica de Poço das Antas.	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
RB378441 (RB01185086)	Brazil	Guedes, R.R.; et al. (255)	1983	Magé	RJ	22°33'32"S 43°03'59"W	-	-
RB239672 (RB01185068)	Brazil	Guimarães, E. (1400); Mautone, L.	1983	Silva Jardim?	RJ	22°31'58"S 42°16'44"W	Reserva Biológica de Poço das Antas.	-
MBM 84409 **NY 617210** **US 3177909**	Brazil USA	Hatschbach, G. (46679)	1983	Ibatiba	ES	20°14'02"S 41°30'38"W	-	Comparing the photograph of this collection, we observe that it resembles *Didymopanax racemiferus* (Fiaschi & Frodin) Fiaschi & G.M. Plunkett, as the leaflets are narrower. Furthermore, Fiaschi and Plunkett (2018) point out the inconsistency of this identification.
BHCB 7423	Brazil	Andrade, P.M. & Lopes, M.A. (348)	1984	Caratinga	MG	19°43'44"S 41°49'24"W	Estação Biológica de Caratinga, Matão.	-
RB378829 (RB01185062)	Brazil	Guedes, R.R.; Gonzaga, L.P. (638)	1984	Magé	RJ	22°35'53"S 42°58'24"W	Distrito de Citrolândia.	-
RB378442 (RB01185124)	Brazil	Guedes, R.R.; et al. (943)	1985	Magé	RJ	22°33'32"S 43°03'59"W	-	-
RB721423 (RB01273274) **HB#** **IAN#**	Brazil	de-Oliveira, R.R.; et al. (1175)	1987	Rio de Janeiro	RJ	22°57'42"S 43°15'24"W	Alto da Boa Vista, Morro Queimado.	-
RB276453 (RB01185091)	Brazil	Farney, C. (1444)	1987	Santa Maria Madalena	RJ	21°58'00"S 41°58'00"W	Parque estadual do Desengano. Serra da Rifa, Fazenda da Rifa, Mata do Fulistão. Mata de Encosta.	-
RB260228 (RB01185059)	Brazil	Marquete, R.; et al. (9)	1987	Magé	RJ	22°35'53"S 42°58'24"W	Distrito de Citrolândia.	-
RB 315817 (RB00558530)	Brazil	de-Lima, H.C. (4247)	1991	Guapimirim	RJ	22°28'35"S 42°53'32"W	Estação Ecológica Estadual de Paraíso. Morro das Pacas.	-
RB254211 (RB00048540) **SPF176505**	Brazil	de-Lima, H.C. (1831)	1992	Silva Jardim	RJ	19°43'44"S 41°49'24"W	Reserva Biológica de Poço das Antas, margens do Rio São João.	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
HUFU16838 ** **(HUFU00001927)** **ESA013841** **HUFU16839** **IAC44324** **PE16257**	Brazil	Barreto, K.D.; et al. (1225)	1993	Campos do Jordão	SP	22°42'31"S 45°27'59"W	Horto Florestal.	Probability *Didymopanax calvus* (Cham.) Decne. & Planch., because comparing the photograph of the deposited material it is possible to observe the silvery indumentum on the abaxial side of the leaf. Collection note: "Tree, ca. 12 m".
BCTw18187 **	Brazil	Albuquerque-Junior, A.H. (Unn.)	1994	Teresópolis	RJ	22°24'44"S 42°57'56"W	-	We believe that this is not the *D. longipetiolatus*, since its size would make it difficult to collect a wood sample. Furthermore, there are many collections of *Didymopanax angustissimus* Marchal for the same municipality, which could lead to identification conflicts.
RB320370 **(RB00490308)** **CEPEC47098** **FCAB5362** **SPF182660**	Brazil	Farias, D.V.S. (147)	1994	Silva Jardim	RJ	22°33'15"S 42°19'02"W	Reserva Biológica de Poço das Antas. Área FP029.	-
RB310266 **(RB00327474)** **ESA123529**	Brazil	Braga, J.M.A. (1797)	1995	Silva Jardim	RJ	22°34'20"S 42°18'49"W	Reserva Biológica Poço das Antas. Trilha do Pau Preto, margem do Rio Pau Preto. Mata alterada de encosta.	-
RB511420 **(RB00635325)**	Brazil	Pessoa, S.V.A.; et al. (761)	1995	Santa Maria Madalena	RJ	21°57'02"S 41°59'22"W	Mata da Rifa, depois da posse do Sr. José Maria.	-
CESJ30998	Brazil	Salimena, F.R.G. (11)	2000	Descoberto	MG	21°25'14"S 42°57'08"W	Reserva Biológica da Represa do Grama.	-
CESJ31878	Brazil	Castro, R.M. (46)	2001	Juiz de Fora	MG	21°45'41"S 43°23'43"W	Reserva Biológica Municipal de Santa Cândida, em interior de mata, próximo a trilha.	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
CESJ34834 **SP357679** **(SP010788)**	Brazil	Castro, R.M.; et al. (641)	2001	Descoberto	MG	21°25'14"S 42°57'08"W	Reserva Biológica da Represa do Grama. Interior da mata.	-
BHCB7422	Brazil	de-Andrade, P.M.; Lopes, M.A. (511)	2001	Caratinga	MG	19°43'44"S 41°49'24"W	Estação Biológica de Caratinga, Matão.	-
SPF147893	Brazil	Fiaschi, P. (919)	2001	Nova Friburgo	RJ	22°19'00"S 42°16'60"W	-	-
SPF145571 **K#**	Brazil United Kingdon	Fiaschi, P.; Castro, R. (617)	2001	Descoberto	MG	21°25'14"S 42°57'08"W	Reserva Biológica da Represa do Grama.	-
CESJ31585	Brazil	Forzza, R.C.; et al. (1752)	2001	Descoberto	MG	21°25'14"S 42°57'08"W	Reserva Biológica da Represa do Grama.	-
CESJ37525 **BHCB63748** **MBM275478**	Brazil	Forzza, R.C.; et al. (2190)	2002	Descoberto	MG	21°25'14"S 42°57'08"W	Reserva Biológica da Represa do Grama. Interior da mata.	-
SPF146508 ** **MBML18499**	Brazil	Kollmann, L.; et al. (5800)	2002	Santa Maria de Jetibá	ES	20°02'36"S 40°41'50"W	Terreno de Olindo Krüger (área 1).	Initially (2004) identified as *Schefflera longipetiolata* (Pohl ex DC.) Frodin & Fiaschi [= *Didymopanax longipetiolatus* (Pohl ex DC.) Marchal], and then as *Schefflera plurifolia* Fiaschi & Frodin [= *Didymopanax plurifolius* (Fiaschi & Frodin) Fiaschi & G.M. Plunkett], both given by Fiaschi. However, the latter has an occurrence restricted to the Amazon. Subsequently, a review of the genus determined that as *Schefflera racemifera* Fiaschi & Frodin [= *Didymopanax racemiferus* (Fiaschi & Frodin) Fiaschi & G.M. Plunkett] (Fiaschi and Plunkett, 2018). The MBML

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
								duplicate still maintains the initial identification.
CESJ39811 **SPF177071** **MBM#**	Brazil	Assis, L.C.S. (734)	2003	Descoberto	MG	21°25'14"S 42°57'08"W	Reserva Biológica da Represa do Grama.	-
CESJ58395	Brazil	Nascimento, C.A.A. (4)	2003	Juiz de Fora	MG	21°46'50"S 43°22'28"W	Mata do Biotério da UFJF. Interior de mata.	-
RB402166 (RB01185137) **CEPEC139091**	Brazil	Forzza, R.C.; et al. (2890)	2004	Macaé	RJ	22°19'42"S 41°58'56"W	Ponte sobre o rio na saída para Tapera.	-
MBML19298	Brazil	Kollmann, L.; et al. (5893)	2003	Santa Maria de Jetibá	ES	20°02'36"S 40°41'50"W	Trilha Pau Pereira.	It is *Schefflera racemifera* Fiaschi & Frodin [= *Didymopanax racemiferus* (Fiaschi & Frodin) Fiaschi & G.M. Plunkett] (Fiaschi and Plunkett, 2018).
MBML19976 **	Brazil	Kollmann, L.; et al. (6057)	2004	Santa Maria de Jetibá	ES	20°02'36"S 40°41'50"W	-	Type material of *Schefflera racemifera* Fiaschi & Frodin [= *Didymopanax racemiferus* (Fiaschi & Frodin) Fiaschi & G.M. Plunkett] (Fiaschi and Plunkett, 2018).
HUFU43820 ** (HUFU00052493)	Brazil	Nakajima, J.N.; et al. (4152)	2005	Capitólio	MG	20°38'55"S 46°13'39"W	-	Comparing the deposited material, it does not appear to be *D. longipetiolatus*, due to the silvery indumentum on the abaxial side of the leaf.
OUPR44609	Brazil	Siqueira, P.C. (72)	2005	Ouro Preto	MG	20°20'06"S 43°31'18"W	Catarina Mendes. Floresta estacional semidecidual.	-
CESJ48599	Brazil	Feliciano, E.A. (5)	2006	Juiz de Fora	MG	21°43'22"S 43°22'54"W	Mata do Krambeck.	-
CEPEC113344 ** **SPF176379** **SPFw3612**	Brazil	Fiaschi, P.; et al. (3117)	2006	Ibatiba	ES	20°14'02"S 41°30'38"W	-	See the comments about *Hatschbach, G. (46679)*.
RB571777 (RB00772716)	Brazil	Finotti, R. (1701)	2007	Guapimirim	RJ	22°32'31"S 42°50'15"W	Propriedade do Sr. Eli, Areal Santa Helena.	-

Deposits (Related Codes)	**Country of Deposit**	**Collector (Collection Number)**	**Year of Collection**	**Municipality**	**State**	**Latitude Longitude**	**Location Notes**	**Comments**
RB475067 (RB00556759)	Brazil	Borges, R.; et al. (928)	2009	Guapimirim	RJ	22°33'55"S 42°51'38"W	Área de estudo da Embrapa Agrobiologia. Floresta ombrófila densa secundária sobre morrote.	-
HRJ11250 **	Brazil	Manão, C.Y.G.; et al. (897)	2009	Angra dos Reis	RJ	23°09'45"S 44°10'43"W	Ilha Grande, Trilha Caxadaço (vertente Dois Rios).	Comparing the deposited material, it does not appear to be *D. longipetiolatus*, as the base of the petiole does not form a sheath.
SPF212709 RB628165 (RB00983986) NY02641666	Brazil USA	Fiaschi, P. (3740)	2011	Silva Jardim	RJ	22°31'59"S 42°18'27"W	Reserva Biológica de Poço das Antas, Rodovia Silva Jardim-Casemiro de Abreu, ramal à dir. Ca. 6,5 km antes da sede. Estrada para barragem Juturnaíba, 3,5 km. Trilha Rodolfo Sul.	-
SPF212691 SPFw3636	Brazil	Fiaschi, P.; Conceição-Jr, J.B. (3723)	2011	Silva Jardim	RJ	22°31'59"S 42°18'27"W	Reserva Biológica de Poço das Antas, Rodovia Silva Jardim-Casemiro de Abreu, ramal à dir. Ca. 6,5 km antes da sede. Estrada para barragem Juturnaíba, 3,5 km. Trilha Rodolfo Sul.	-
CESJ48925	Brazil	Menini-Neto, L. (409)	2011	Juiz de Fora	MG	21°43'22"S 43°22'54"W	Mata do Krambeck. Interior de mata, sub-bosque.	-
SPF212693 NY02641493	Brazil USA	Fiaschi, P. (3769)	2012	Casimiro de Abreu	RJ	22°26'11"S 42°01'34"W	Reserva Biológica União. BR-101, ca. 3 km após sede do parque, em direção a Campos dos Goytacazes. Estrada da Macuca, rumo a Rio das Ostras, entrada à direita em frente ao cemiterio de Rocha Leão. Trilha em meio a	Initially identified as *Schefflera calva* (Cham.) Frodin & Fiaschi [= *Didymopanax calvus* (Cham.) Decne. & Planch.].

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
							Eucalyptus, ca. 2 km da estrada da Macuca, acesso à direita. Floresta Ombrófila Densa.	
RB766608 (RB01389084) RFFP18911	Brazil	Ribeiro, E.R. (297)	2012	Casimiro de Abreu	RJ	22°25'01"S 42°02'37"W	Reserva Biológica União. Trilha do Buracão e Trilha de acesso às torres.	-
RB590784 (RB00828161) NY03227437	Brazil USA	Bovini, M.G. (3843)	2013	Magé	RJ	22°34'42"S 43°01'46"W	Reserva Particular do Patrimônio Natural Campo Escoteiro Geraldo Hugo Nunes.	-
RB756261 ** (RB01378710) R230878 (R010071064)	Brazil	Verdi, C.; et al. (7488)	2017	Santa Maria Madalena	RJ	21°59'50"S 41°55'36"W	Parque Estadual do Desengano. Terras Frias. Ribeirão Vermelho, trilha para o local de captação de água da CEDAE [Companhia Estadual de Águas e Esgotos]. Floresta Ombrófila Densa em estágio médio de regeneração, encosta, solo predominantemente argiloso.	It is not *D. longipetiolatus*, because the individual collected is too large. Collection note: “Mother tree nº 102; CCH 180 cm.”.
RB756262 ** (RB01378711) FLOR68453 NY04334849	Brazil USA	Verdi, C.; et al. (7494)	2017	Santa Maria Madalena	RJ	21°55'48"S 41°55'31"W	Parque Estadual do Desengano. Terras Frias. Ribeirão Vermelho, trilha para o local de captação de água da CEDAE (Companhia Estadual de Águas e Esgotos). Floresta Ombrófila Densa em estágio médio de regeneração, encosta, solo predominantemente argiloso.	See the comments about *Verdi, C.; et al. (7488)*. Collection note: “Mother tree nº 102; CCH 126 cm; height 19m, stem 10m.”.
RB756274 ** (RB01378723)	Brazil	Verdi, C.; et al. (7518)	2017	Santa Maria Madalena	RJ	21°52'40"S 41°55'11"W	Tudelândia. Morumbeca dos Marreiros. Parque	See the comments about *Verdi, C.; et al. (7488)*.

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
RBR55642 **FLOR#** **NY#**	USA						Estadual do Desengano. Floresta Ombrófila Densa em estágio médio a avançado de regeneração, encosta, solo predominantemente argiloso.	Collection note: “Mother tree nº 122; CCH 130 cm”.
RB764179 ** **(RB01386598)** **HUEFS237402** **RBR46326** **R241086** **(R010069804/ R01006980)**	Brazil	Baez, C.; et al. (1627)	2018	Resende	RJ	22°20'10"S 44°32'18"W	Parque Estadual da Pedra Selada. Bosque do Visconde. Trilha da Família.	It is probably *D. calvus*, as we observed identification conflicts between this species and *D. longipetiolatus*, and the recorded size of the individual (CCH 106 cm) does not match the description of the species. In addition, there are collections for *D. calvus* in the municipality.
RB772754 **(RB01395398)**	Brazil	Fernandes, T. (288)	2018	Silva Jardim	RJ	22°38'06"S 42°26'11"W	Fazenda Santa Helena, mata no entorno do açude.	-
RB756263 ** **(RB01378712)** **FLOR#** **K001890401**	Brazil United Kingdon	Verdi, C.; et al. (7496)	2018	Santa Maria Madalena	RJ	21°55'48"S 41°55'30"W	Terras Frias. Ribeirão Vermelho, trilha para o local de captação de água da CEDAE. Floresta Ombrófila Densa em estágio médio de regeneração, encosta, solo predominantemente argiloso.	See the comments about *Verdi, C.; et al. (7488)*. Collection note: “Mother tree nº 105; CCH 89 cm; height 15m, stem 10m.”.
RB764195 ** **(RB01386614)** **FLOR68797** **SPF#**	Brazil	Baez, C.; et al. (1643)	2019	Resende	RJ	22°20'10"S 44°32'18"W	Parque Estadual da Pedra Selada. Bosque do Visconde. Trilha da Família. Floresta Ombrófila Densa em avançado estágio de regeneração; solo predominantemente argiloso.	See the comments about *Baez, C.; et al. (1627)*.

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
VIC056021 *	Brazil	Verly, O.M. (23)	2022	Coronel Fabriciano	MG	19°24'55"S 42°43'29"W	Área de preservação da Cenibra, Parcela 4.	-
RB 851603 **(RB01491963)**	Brazil	Nunes, D.; et al. (1494)	2023	Silva Jardim	RJ	22°32'06"S 42°16'12"W	REBIO [Reserva Biológica] de Poço das Antas. Na estrada do Aristides.	-
VIC058117 *	Brazil	Verly, O.M. (244)	2024	Coronel Fabriciano	MG	19°24'52"S 42°43'32"W	Área de preservação da Cenibra, Parcela 6.	-
Lauraceae Juss. ***Persea rigida* Nees & Mart.** **Synonyms:** **Uncertain Application Name: *Persea rigida* var. *claussenii* Meisn.**								
B100185203^{T} **B100185204** **GZU00249334** **K000602081** **MO1901658** **M0111053** **NY927583** **P01991984**		Sellow, F. (652)	1833	-	SP	-	-	-
NY504856 **S12-22325**	USA Sweden	Hatschbach, GG. (29764)	1972	Bocaiuva do Sul	PR	25°12'38"S 49°06'23"W	Mata pluvial encosta de morro. Rio Putunâ.	-
UEC147092 **(UEC097032)**	Brazil	Aguirre, G.H.; et al. (102A)	2005	São Luiz do Paraitinga	SP	23°13'12"S 45°17'02"W	Fragmento de Floresta Ombrófila Densa em paisagem rural	-
FURB34919 **IFFSC577**	Brazil	Korte, A.; Kniess, A. (6986)	2010	Indaial	SC	27°05'24"S 49°13'51"W	Parque Nacional da Serra do Itajaí. Floresta Ombrófila Densa; Estágio Avançado	-
ESAL25288	Brazil	Naves, R.P. (44)	<2013	Varginha	MG	21°34'08"S 45°24'29"W	Parque São Francisco	-
MBM407312 **EFC16268** **FUEL55360** **HRCB68199** **HUEM31508** **(HUEM000014290)**	Brazil	Brotto, M.L.; Lozano, E.D. (2360/2372)	2016	Adrianópolis	PR	24°50'59"S 48°42'19"W	Parque Estadual das Lauráceas	The collections were made on different dates.

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
MBM407580 **RB785634** **(RB01409441)** **SPSF51747** **VIES59463**								
HRCB69141	Brazil	de-Moraes, P.L.R.; et al. (5424)	2016	Adrianópolis	PR	24°51'22"S 48°43'04"W	Parque Estadual das Lauráceas	-
MBM407307	Brazil	Brotto, M.L.; Lozano, E.D. (2355)	2016	Tunas do Paraná	PR	24°53'45"S 48°47'37"W	Fazenda Primavera	-
MBM407586 **EFC16269** **HRCB68198** **SPSF51748** **VIES59459** **RB785637** **(RB01409444)**	Brazil	Brotto, M.L.; Lozano, E.D. (2378)	2017	Adrianópolis	PR	24°50'55"S 48°40'02"W	Parque Estadual das Lauráceas. Floresta Ombrófila Densa Montana	-
MBM409106 **EFC23007** **FURB60719** **HCF23466** **(HCF000023930)** **HRCB77819** **HRCB77848** **HUEFS110783** **RB786609** **(RB01410664)** **UPCB103224** **VIES61996**	Brazil	Barboza, E.; et al. (4581)	2017	Adrianópolis	PR	24°50'60"S 48°42'16"W	Parque Estadual das Lauráceas. Floresta Ombrófila Densa Montana	-
MBM408354 **EFC23002** **HUEFS261936** **UPCB103191** **RB788459** **(RB01412831)**	Brazil	Brotto, M.L.; Völtz, R.R. (2426)	2017	Tunas do Paraná	PR	24°54'08"S 48°48'11"W	Fazenda Primavera. Floresta Ombrófila Densa Montana	-
UEC196822 **(UEC137461)** **BM001118065**	Brazil	Leitão-Filho, H.F.; Barros, R.C. (10683)	1979	Camanducaia	MG	22°52'19"S 46°00'59"W	Nativo em mata nas proximidades da Vila Monte Verde	-

Deposits (Related Codes)	**Country of Deposit**	**Collector (Collection Number)**	**Year of Collection**	**Municipality**	**State**	**Latitude Longitude**	**Location Notes**	**Comments**
VIC057707 *	Brazil	Verly, O.M. (212)	2024	Coronel Fabriciano	MG	19°24'46"S 42°43'22"W	Área de preservação da CENIBRA. Parcela 10	-
***Rhodostemonodaphne anomala* (Mez) Rohwer** **Synonyms:** **basionym: *Nectandra anomala* Mez**								
K000642286^{T} **B100185180^{H}** **C#** **G00368976^{I}** **P00078422^{I}** **P00745446^{I}** **MO1700549**	United Kingdon Germany Denmark Switzerland France USA	Glaziou, A. F. M. (13150)	1911?	Petrópolis?	RJ	22°31'00"S 43°10'00"W	-	-
RB00129414 **(RB00129414)** **MO1700553** **MO101900150**	Brazil USA	Gois (Góes), O.C. (110)	1948	Petrópolis	RJ	22°31'48"S 43°12'40"O	Quitandinha.	-
ALCB15368 **(ALCB003053)**	Brazil	Gusmão, E.F.d. (Unn.)	1977	-	-	-	-	-
HUEFS1788	Brazil	Noblick, L.R.; et al. (2329)	1982	Camaçari	BA	12°18'00"S 38°18'00"W	Área controle da Caraíba Metais	-
CEPEC53603	Brazil	Thomas, W.W. (8282)	1991	Uruçuca	BA	14°35'35"S 39°17'04"W	7.3Km ao N de Serra Grande, na Rod. para Itacaré. Fazenda Lagoa. Parcela 32. Árvore # 1496.	-
CEPEC57574 **SPF168618** **MO1700544** **NY504989**	Brazil USA	Hage, J.L.; et al. (2357)	1993	Una	BA	15°09'00"S 39°04'00"W	Reserva Biológica do Mico-leão (IBAMA). Entrada no km 46 da Rod. BA-001 Ilhéus/Una.	-
HURB17264 **ALCB135750** **(ALCB064813)** **MBM426248**	Brazil	Almeida, T. (Unn.)	2016	Salvador	BA	12°53'54"S 38°26'00"W	Ipitanga, APA Joanes.	-
VIC057709 *	Brazil	Verly, O.M. (21)	2022	Coronel Fabriciano	MG	19°24'46"S 42°43'21"W	Área de preservação da CENIBRA. Parcela 10.	-
MBM444555	Brazil	Brotto, M.L.; Völtz, R.R. (5534)	2023	Guapimirim	RJ	22°30'00"S 43°03'02"W	Parque Nacional da Serra dos Órgãos. trilha para o	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
							pico Dedo de Nossa Senhora.	
Myrtaceae Juss. ***Eugenia leonorae* Mattos** **Synonyms:** **homotype: *Calycorectes schottianus* O.Berg**								
BR0000013090807 **	Belgium	Peckolt, T. (377)	-	-	-	-	-	-
B#^{T} **K000565053** **K000974369** **(V)F93810 (874604)** **W48330** **W48331(31471)** **W48332** **HUEFS87223**	Germany United Kingdon USA Austria Brazil	Schott, H.W. (1044)	1837	-	RJ	-	-	There is an inconsistency regarding the collection number, as some duplicates mention *Schott, H.W. (5830)* as the collection number.
K000565054 **	United Kingdon	Sellow, F. (Unn.)	1859	-	RJ	-	-	The same material appears twice in the GBIF portal. In one of them (the only one with a photographic record) the species is *Eugenia vattimoana* Mattos, with an evident inconsistency regarding its deposit data in the portal itself. Therefore, we disregarded this material.
SinBIOTA837	Brazil	Martins, F.R. (284)	1976	-	-	-	-	-
UEC11719 **	Brazil	Matthes, L.A.F. (7723)	1977	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 295-D	Cultivated.
UEC11717 **	Brazil	Matthes, L.A.F. (7736)	1977	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 21-B	Cultivated.
UEC11718 **	Brazil	Matthes, L.A.F. (7940)	1977	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 124-A	Cultivated.
UEC11724 **	Brazil	Matthes, L.A.F. (7941)	1978?	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 468-D	Cultivated.
UEC11720 **	Brazil	Matthes, L.A.F. (7947)	1978?	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 19-B	Cultivated.

Deposits (Related Codes)	**Country of Deposit**	**Collector (Collection Number)**	**Year of Collection**	**Municipality**	**State**	**Latitude Longitude**	**Location Notes**	**Comments**
UEC11722 **	Brazil	Matthes, L.A.F. (7949)	1978?	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 275-D	Cultivated.
UEC11721 **	Brazil	Matthes, L.A.F. (7950)	1978?	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 293-D	Cultivated.
UEC11723 **	Brazil	Matthes, L.A.F. (7951)	1978?	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 300-D	Cultivated.
UEC11726 **	Brazil	Matthes, L.A.F. (9258)	1978	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 847-D	Cultivated.
UEC11716 **	Brazil	Matthes, L.A.F. (9262)	1978	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás nº 988-A	Cultivated.
UEC34804 ** **INPA172212**	Brazil	Hopkins, M. (Gandolfi, S.) (15618)	1983	Valinhos	SP	22°56'26"S 46°59'48"W	Estação Experimental de Valinhos (IF)	Probably cultivated.
NY561486 **K001178053**	USA United Kingdon	Peron, M.V. (875)	1989	Nova Friburgo	RJ	22°24'14"S 42°30'50"W	Reserva Ecológica Municipial de Macaé de Cima	-
HUEFS201827 ** **S13-31361**	Brazil Sweden	Santin, D.A.; et al. (33602)	1993	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás	Cultivated.
UEC71314 ** **S13-31367**	Brazil Sweden	Santin, D.A.; et al. (33601)	1994	Campinas	SP	22°54'31"S 47°02'58"W	Coleção Bosque dos Jequitibás	Cultivated.
RB816847 (RB01444765)	Brazil	Farney, C.; et al. (4999)	2000	São Pedro da Aldeia	RJ	22°49'35"S 42°09'37"W	Serra de Sapiatiba	-
RB392878 (RB00451646) **BHCB96433 (BHCB000710)** **SPF#**	Brazil	Farney, C.; et al. (4441)	2001	São Pedro da Aldeia	RJ	22°49'35"S 42°09'37"W	Serra de Sapiatiba	-
RB364543	Brazil	Rezende, G.S.; et al. (52)	2001	Cabo Frio	RJ	22º41'05"S 42º00'12"W	Distrito Tamoios, Condomínio Florestinha. Mata de Restinga.	-
HUFSJ3164 **SP488226**	Brazil	Sobral, M. (14571)	2011	Santana do Paraíso	MG	19°24'40"S 42°31'45"W	Propriedade de Fernando Dantas dos Santos	-
HUFSJ1601	Brazil	Sobral, M. (13571)	2011	Santana do Paraíso	MG	19°24'49"S 42°31'44"W	Em interior da mata.	Our determination.
CEPEC150929 **RBR44166**	Brazil	Oliveira, N.E. (389)	2014	Itambé	BA	15°17'58"S 40°40'49"W	Conglomerado BA_1328-2-10-55	Collection note: “Tree, exudate not observed. Outer bark grayish and thin, inner

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
								bark yellowish, type: smooth, detachment: no detachment, odorless. Habit: Tree”.
VIC057681 *	Brazil	Verly, O.M. (56)	2022	Caratinga	MG	19°30'49"S 42°26'09"W	Área de preservação da CENIBRA. Próximo à Lagoa Silvana. Floresta Estacional Semidecidual.	-
RFFP21527	Brazil	Souza, M.C. (3168)	2023	Maricá	RJ	22°56'52"S 42º41'00"W	Ponta Negra, Morro da Sacristia, elevação circundante a Praia da Sacristia	-
VIC057692 *	Brazil	Verly, O.M. (175)	2024	Ipaba	MG	19°20'25"S 42°24'06"W	RPPN Fazenda Macedônia. Próximo ao Rio Doce. Topo de morro. Floresta Estacional Semidecidual.	-
***Eugenia reperta* Sobral & Mazine**								
HPL6987^{P} **BHCB109908**	Brazil	Tsuji, R.; Lorenzi, H. (1566)	2007	Santana do Paraíso	MG	19°21'49"S 42°34'07"W	Coletado em direção ao bairro Achados, em borda de estrada.	Identified as *Eugenia robustovenosa* Kiaersk. in the SpeciesLink repository.
HUFSJ3170^{T}	Brazil	Sobral, M. (14577)	2011	Santana do Paraíso	MG	19°21'40"S 42°36'18"W	Achado.	Identified as *Eugenia robustovenosa* Kiaersk. in the SpeciesLink repository.
VIC057776*	Brazil	Verly, O.M. (219)	2022	Caratinga	MG	19°30'49"S 42°26'08"W	Área de preservação da CENIBRA. Parcela 13.	-
***Myrcia pseudosplendens* Sobral & Mazine**								
UEC71315 **	Brazil	Santin, D.A.; et al. (33603)	1994	Campinas	SP	22°54'34"S 47°03'03"W	Coleção Bosque dos Jequitibás.	Cultivated.
VIC026258^{P} **HUFSJ8849**	Brazil	Lopes, W.P. (503)	1997	Marliéria	MG	19°45'57"S 42°37'25"W	Parque Estadual do Rio Doce. Região do vinhático.	-
VIC022615	Brazil	Bortoluzzi, R.L.C.B.; et al. (352)	1998	Marliéria	MG	19°40'47"S 42°33'11"W	Trilha da Garapa Torta – Parque Estadual do Rio Doce.	Our determination.
ESA087899 **	Brazil	Lucas, E.J.; et al. (165)	2003	Tunas do Paraná	PR	25°03'47"S 49°02'04"W	Distrito de Morro Grande. 1.074 m.	It's likely not the species. The notes in the record indicate that the species was not confirmed (cf.), and the

Deposits (Related Codes)	**Country of Deposit**	**Collector (Collection Number)**	**Year of Collection**	**Municipality**	**State**	**Latitude Longitude**	**Location Notes**	**Comments**
								collection was made in a region far from the species' endemic zone.
RB531572 (RB00679980 RB01074742) **BHCB#** **HUFSJ1583** **HUFSJ1600**	Brazil	Sobral, M. (13570)	2011	Santana do Paraíso	MG	19°24'40"S 42°31'43"W	Em interior da mata.	-
HUFSJ 3161 **BHCB#** **RB#**	Brazil	Sobral, M. (14568)	2011	Santana do Paraíso	MG	19°24'49"S 42°31'45"W	Em interior da mata.	-
Human observation *	-	Verly, O.M. (Unn.)	2022	Caratinga	MG	19°30'41"S 42°26'03"O	Área de preservação da CENIBRA, Parcela 05. Próximo à Lagoa Silvana. Floresta Estacional Semidecidual.	-
Human observation *	-	Verly, O.M. (Unn.)	2022	Caratinga	MG	19°30'55"S 42°25'57"O	Área de preservação da CENIBRA, Parcela 18. Topo de morro próximo à Lagoa Silvana. Floresta Estacional Semidecidual.	Photographed *in vivo* and *in sico*.
Human observation *	-	Verly, O.M. (Unn.)	2022	Caratinga	MG	19°31'10"S 42°25'58"O	Área de preservação da CENIBRA, Parcela 20. Topo de morro próximo à Lagoa Silvana. Floresta Estacional Semidecidual.	-
Human observation *	-	Verly, O.M. (Unn.)	2022	Caratinga	MG	19°31'10"S 42°25'58"O	RPPN Fazenda Macedônia, Parcela 20. Próximo ao Rio Doce. Topo de morro. Floresta Estacional Semidecidual.	-
VIC057680 *	Brazil	Verly, O.M. (202)	2024	Ipaba	MG	19°21'48"S 42°23'42"O	RPPN Fazenda Macedônia, Parcela 23. Próximo ao Rio Doce. Topo de morro. Floresta Estacional Semidecidual.	In addition to the photographed juvenile individual, other juvenile individuals were recorded in plots 05, 10, and 12.

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
Proteaceae Juss.								
***Euplassa semicostata* Plana**								
BHCB 53427	Brazil	da-Mota, R.C.; (204)	-	Catas Altas	MG	20°04'38"S 43°26'26"W	Serra do Caraça.	-
BHCB 107347	Brazil	Mendes, M.S.; Brina, A.E. (Unn.)	-	Congonhas	MG	20°28'42"S 43°53'41"W	Mineração Casa de Pedra, Corpo Norte - CSN	-
RB135686 (RB00322188) NY01021669	Brazil USA	Schwacke, P. (Unn.)	1893	Ouro Preto	MG	20°24'58"S 43°30'30"W	Serra de Saramenha.	-
UB223958^{H} MO1149541^{I} MO101742896$^{I\blacktriangledown}$ MO101742900^{I} NY579258^{I} NY579258^{I} US2709390^{I} K000634317	Brazil USA United Kingdon	Irwin, H.S.; et al. (28922)	1971	Barão de Cocais?	MG	17°41'40"S 43°46'60"W	Serra do Espinhaço lower slopes of Serra da Caraça, ca. 10km W. of Barão de Cocais. Cerrado. Locally common. Cerrado, sedge meadow and gallery forrest, gray sandy soil.	-
SPF37462	Brazil	Cordeiro, I. (6062)	1980	Santana do Riacho	MG	19°10'08"S 43°42'52"W	Km 131 ao longo da rodovia Belo Horizonte-Conceição do Mato Dentro. Capão-restos de mata (local seco).	-
SPF37466	Brazil	Pirani, J.R.; et al. (6930)	1981	Santana do Riacho	MG	19°10'08"S 43°42'52"W	Km 123 ao longo da rodovia Belo Horizonte-Conceição do Mato Dentro. Beira de mata mesófila.	-
SPF22258 MBM254459 NY626404	Brazil USA	Hensold, N.; et al. (2778)	1982	Catas Altas	MG	20°04'03"S 43°25'41"W	-	-
SPF76682	Brazil	Marcondes-Ferreira, W. (280)	1990	Santa Bárbara	MG	19°57'54"S 43°25'22"W	Caminho de asfalto até o encontro dos rios.	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
SPF100927	Brazil	Campos, M.T.V.A.; de-Souza, E.D.P. (13380)	1993	Santana do Riacho	MG	19°13'23"S 43°30'08"W	Serra do Cipó.	-
SPF100982	Brazil	Campos, M.T.V.A.; Roque, N. (13338)	1993	Santana do Riacho	MG	19°13'23"S 43°30'08"W	Serra do Cipó.	-
SPF106673	Brazil	Campos, M.T.V.A. (13644)	1994	Morro do Pilar	MG	19°13'47"S 43°30'24"W	Serra do Cipó.	-
SPF95494 **ESA024337** **SP311710** **NY626244** **NY626251**	Brazil USA	Sano, P.T.; et al. (14575)	1994	Abaíra	BA	13°14'35"S 41°43'01"W	Serra do Barbado. Ao longo do caminho da forquilha da Serra.	-
NY626252	USA	Riedel, L. (1449)	≤1998	-	-	-	Pr. Capanéma.	-
SPF169590	Brazil	Araújo-Nóbrega, M. (288)	2000	Ibicoara	BA	13°26'00"S 41°13'00"W	Chapada Diamantina.	-
BHCB67350	Brazil	da-Mota, R.C. (1166)	2000	Catas Altas	MG	20°04'38"S 43°26'26"W	Serra do Caraça.	-
BHCB52597	Brazil	de-Vasconcelos, M.F. (Unn.)	2000	Catas Altas	MG	20°07'25"S 43°27'53"W	Serra do Caraça.Trilha para o Pico do Inficionado, borda de mata.	-
CESJ36672 **SPF153077** **SPF168909**	Brazil	Pifano, D.S.; Castro, R.M. (329)	2002	Juiz de Fora	MG	21°45'53"S 43°21'29"W	Morro do Imperador.	-
BHCB86009 **CESJ45624** **SPF161520**	Brazil	Stehmann, J.R. (3498)	2003	Catas Altas	MG	20°05'58"S 43°29'38"W	RPPN Santuário do Caraça.	-
SPF167245 **SPFw898**	Brazil	Ceccantini, G.C.T. (2144)	2004	Santana do Riacho	MG	19°13'04"S 43°30'26"W	-	-
SPF167246 **SPFw905**	Brazil	Ceccantini, G.C.T. (2151)	2004	Santana do Riacho	MG	19°13'04"S 43°30'26"W	-	-
CVRD17361	Brazil	Siqueira, G.S. (1647)	2023	Jequitinhonha	MG	16°20'55"S 41°05'30"W	-	-
FDBK2	-	Kanouté, D.B. (102)	2024	Antônio Dias	MG	19°27'48"S 42°49'41"W	Sede Cenibra Cocais - Cocais da Estrela.	Human observation. Photographic only.
VIC057786 *	Brazil	Verly, O.M. (206)	2024	Coronel Fabriciano	MG	19°24'54"S 42°43'32"W	Área de preservação da CENIBRA. Parcela 05	-
Rubiaceae Juss.								

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
Duroia valesca **C.H.Perss. & Delprete**								
HST3814 **HUCPE122** **HUEFS176346** **RB174184** **(RB00714682)**	Brazil	Monteiro; M.T. (23575)	1971	Itamaraju	BA	17°07'11"S 39°43'58"W	Fazenda Palmeira.	-
RB160342 **(RB00333711)**	Brazil	Spada, J. (110)	1972	Linhares	ES	19°08'48"S 40°01'18"W	Reserva Florestal de Linhares - CVRD. Próximo Estrada 161 Talhão 601. Reserva Natural Vale do Rio Doce (Particular).	-
FCAB3171 **UB176257** **SP175513** **F1880936** **(V0263480F)**	Brazil	Sucre, D.; et al. (10155)	1973	Marliéria	MG	19°40'24"S 42°30'09"W	Reserva Florestal do Rio Doce, encontro do Rio Turvo com Rio Doce, Matas de encosta do.	-
RB237832 **(RB00333654)** **IAC46168** **SP216085**[I] **RB509655** **(RB00621166)** **MO101190665** **MO103139177**[I] **MO103139178**[H] **NY502122** **GB-0048953**	Brazil	Ramalho, R.S. (1050)	1977	Viçosa	MG	20°46'26"S 42°52'25"W	-	His record states that the seeds came from USA, however, there are doubts.
RB336660 **(RB00333999)**	Brazil	Mori, S.A.; dos-Santos, T.S. (Unn.)	1979	Camacã	BA	15°20'57"S 39°30'59"W	Ramal para a Torre da Embratel na Serra Boa, ao N de São João da Panelinha. Plantação de cacau.	-
CVRD837 [H] **IAC46168** **SP216085**[I] **RB509655**	Brazil	Silva, I.A. (281)	1981	Linhares	ES	19°09'19"S 40°03'35"W	Reserva Natural da Vale do Rio Doce, Estrada Gávea, 5.85 km.	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
(RB00621166) **MO101190665** **MO103139177**^{I} **MO103139178** ^{H} **NY502122** **GB-0048953**	USA Sweden							
CVRD1004^{P} **IAC46167** **SP269393**^{P} **RB509656** **(RB00621175)** **NY502121** **NY502124** **GB-0155243**	Brazil USA Sweden	Folli, D.A. (418)	1983	Linhares	ES	19°07'32"S 39°56'39"W	Reserva Florestal da CVRD, Estrada Paraju, ant. 211, km 0,172 lado esquerdo.	-
RB227188 ** **(RB00340087)** **INPA128660** **MBM91065** **SP197726**	Brazil	Pinheiro, A.L. (Unn.)	1983	Viçosa	MG	20°46'26"S 42°52'25"W	-	Cultivated in the arboretum of the Dendrology Sector of the University of Viçosa (probable origin of seeds: Amazon).
BHCB6368	Brazil	de-Andrade, P.M. (Unn.)	1985	Caratinga	MG	19°43'53"S 41°49'01"W	Estação Biológica de Caratinga.	Our determination.
BHCB7921 **BHCB9288** **ESA039633** **FUEL22845** **SPF122456** **SP316365** **MO3115227**	Brazil USA	Lopes, M.A.; de-Andrade, P.M. (836)	1985	Caratinga	MG	19°44'40"S 41°48'51"W	Estação Biológica de Caratinga, Fazenda Montes Claros.	-
BHCB7922 **BHCB9316** **SPF122457** **MO3114971**	Brazil USA	Lopes, M.A.; de-Andrade, P.M. (838)	1985	Caratinga	MG	19°44'40"S 41°48'51"W	Estação Biológica de Caratinga, Fazenda Montes Claros. Mata do Jaó.	-
BHCB17055 **MO3115361**	USA	Costa, L.V. (Unn.)	1989	Caratinga	MG	19°44'40"S 41°48'51"W	Estação Biológica de Caratinga, fazenda Montes Claros.	-
NY502123	USA	Thomas, W.W. (6911)	1991	Uruçuca	BA	14°25'23"S 39°03'42"W	7.3 km N of Serra Grande on road to Itacaré. Fazenda	Our determination.

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
							Lagoa do Conjunto Fazenda Santa Cruz.	
BHCB30714 **PERD110** **MO3115515**	Brazil USA	Costa, L.V.; et al. (Unn.)	1993	Marliéria	MG	19°43'54"S 42°36'09"W	Parque Estadual do Rio Doce, Águas Claras.	-
CVRD5009(P) **RB509653** **(RB00621144)** **NY502120**	Brazil USA	Folli, D.A. (2372)	1994	Linhares	ES	19°10'58"S 39°56'41"W	-	-
CEPEC95367 **RB551775** **(RB00718987)**	Brazil	Jardim, J.G.; et al. (1367)	1998	Almadina	BA	14º44'06"S 39º41'46"W	Rod. Almadina/ Ibitupã, entrada a ca. 5 km W da sede do município. Fazenda Cruzeiro do Sul. "Serra do sete-paus", ca. 8 km da entrada.	-
CVRD6234	Brazil	Folli, D.A. (3409)	1999	Linhares	ES	19°08'07"S 39°55'15"W	-	-
MBML-Herbario18491 **K001135815**		Kollmann, L.; et al. (5807)	2002	Santa Maria de Jetibá	ES	20°02'26"S 40°44'46"W	Belém, terreno de Olindo Krüger (área 1).	Our determination.
BHCB84997 **SPF163907**	Brazil	France, G.S. (424)	2003	Marliéria	MG	19°41'49"S 42°30'15"W	-	Our determination.
CVRD8453(P) **IAC46169** **RB509654** **(RB00621156)** **GB-0048952**	Brazil Sweden	Siqueira, G.S. (66)	2003	Linhares	ES	19°06'55"S 40°01'29"W	-	-
CVRD8815(P) **IAC46166** **RB509652** **(RB00621140** **RB00625157)** **GB-0032373**	Brazil Sweden	Folli, D.A. (4892)	2004	Linhares	ES	19°09'22"S 40°04'21"W	Reserva Natural Vale. Estrada Jueirana Vermelha.	-
PERD710	Brazil	Dias, L.G. (Unn.)	2005	Marliéria	MG	19°41'36"S 42°36'21"W	Parque Estadual do Rio Doce, Trilha da Lagoa Preta.	Our determination.
MBML-Herbario27599	Brazil	Demuner, V.; et al. (2843)	2006	Santa Leopoldina	ES	20°06'12"S 40°26'31"W	Colina Verde (Morro Agudo), lado esquerdo,	Our determination.

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
							Prop.: Israel Elias Ramos. Borda de mata.	
BHCB147340 (BHCB012127)	Brazil	Vidal, C. (763)	2007	Governador Valadares	MG	19°01'16"S 42°08'07"W	UHE Baguari, Fazenda Santa Luzia. Floresta estacional semidecidual.	Our determination.
CVRD12501 ESA106437 HPL13168	Brazil	Flores, T.B.; Folli, D.A. (206)	2009	Linhares	ES	19°09'22"S 40°04'21"W	Reserva Natural Vale. Estrada Jueirana Vermelha.	-
CVRD12493	Brazil	Folli, D.A. (6401)	2009	Linhares	ES	19°04'16"S 39°53'50"W	-	-
JPB60558	Brazil	Persson, C.; et al. (2211)	2014	Uruçuca	BA	14°29'42"S 39°07'55"W	Trail from headquarter of the Condurus state park, ca. 16 Km W of Serra Grande on road to Uruçuca.	Our determination.
JPB60578 GB-0155241	Brazil Sweden	Persson, C.; et al. (2231)	2014	Camacã	BA	15°24'21"S 39°32'45"W	Fazenda Santo Antônio, Fazenda Santo Antônio of Serra Bonita private Reserve (RPPN) 6 km NW of Camacan on road to jacareci, then on trail into Fazenda. Disturbed and intact tropical foreste with cacao plantation.; DESC: Tree 5 m, dbh 6 cm.	Collection note: "Sterile. Largest leaf with 20 secondary veins and stipules up to 8 cm long."
MO100867401 GB-0155242	USA Sweden	Persson, C.; et al. (2241)	2014	Caratinga	MG	19°42'49"S 41°49'30"W	RPPN Feliciano Manuel Abdala (Estação Biológica de Caratinga).	-
JPB60519	Brazil	Razafimandimbison, S.G.; Kainulainen, K. (1284)	2014	Camacã	BA	15°24'15"S 39°32'26"W	-	Our determination.
RB772990 (RB01395657)	Brazil	Grupo de Coletores do Núcleo Juçara (182)	2015	Conceição da Barra	ES	18º23'47"S 39º51'59"W	Base avançada. Núcleo Juçara, Matriz 182 - trilha córrego das pedras.	-
RB775367 (RB01398163)	Brazil	Grupo de Coletores do Núcleo Juçara (223)	2016	Conceição da Barra	ES	18º20'50"S 39º53'36"W	Córrego das preguiças. Núcleo Juçara, Matriz 223	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
							- Próximo à jazida de barro da Suzano.	
CAP9691	Brazil	Torres-Leite, F.; et al. (512)	2016	Castelo	ES	20°36'12"S 41°11'04"W	Parque Estadual Mata das Flores.	-
SAMES14843	Brazil	Oliveira. V.B. (Unn.)	2019	Rio Bananal	ES	19°19'37"S 40°13'20"W	Sub-bacia hidrográfica dos Pontões e Lagoas.	-
CVRD17020 **RB863007** **(RB01523228)**	Brazil	Siqueira, G.S. (1426)	2021	Linhares	ES	19°04'27"S 39°57'44"W	Aceiro Café, Macuco (m).	-
SAMES11520	Brazil	Menezes, L.F.T. (2952)	2022	Pinheiros	ES	18°22'11"S 40°08'39"W	Reserva biológica córrego do veado.	-
VIC057686 * **VIC058115 ***	Brazil	Verly, O.M. (48) Verly, O.M. (242)	2022 2024	Caratinga	MG	19°30'44"S 42°26'15"W	Área de preservação da CENIBRA. Parcela 17	-
Simaroubaceae DC. ***Homalolepis floribunda*** **(A.St.-Hil.) Devecchi & Pirani** **Synonyms:** **homotype: *Simaba floribunda* A.St.-Hil.** ***Aruba floribunda*** **(A.St.-Hil.) Kuntze** ***Quassia floribunda*** **(A.St.-Hil.) D.Dietr.** ***Quassia floribunda*** **(A.St.-Hil.) Noot.** ***Zwingera floribunda*** **(A.St.-Hil.) Spreng.** **heterotypic: *Zwingera discolor* Colla**								
BHCB41392 **LY0118480** **LY0118482** **LY0118483**	Brazil France	-	-	-	-	-	-	Records without any collection information.
K001252425 ** **NY02681297?**	United Kingdom USA	Langsdorff, G.H.v. (Unn.)	-	-	MG	-	-	This is *Homalolepis cuneata* (A.St.-Hil. & Tul.) Devecchi & Pirani.
CEPEC122682 **	Brazil	Martini, A.; et al. (Unn.)	-	Uruçuca	BA	14°29'59"S 39°06'54"W	Estrada de Serra Grande para Uruçuca. Área do inventário do plano manejo do Parque Estadual da Serra do Conduru. Mata da Torre do Celular.	Disregarded based on the occurrence zone described by Devecchi and Pirani, 2020. In addition, there are several collections of *Homalolepis bahiensis* (Moric.) Devecchi & Pirani and *Homalolepis ferruginea* (A.St.-Hil.)

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
								Devecchi & Pirani for the state.
W0059308 ** **W0059309**	Áustria	Schüch, R. (294)	-	Brasília	DF	-	-	Disregarded based on the occurrence zone described by Devecchi and Pirani, 2020.
K001252426 ** **P04876069** **P04876070** **LY0118483**	United Kingdom France	Glaziou, A. (2528)	1868	Rio de Janeiro	RJ	22°56'37"S 43°12'01"W	Caminho do Corcovado.	Disregarded based on the occurrence zone described by Devecchi and Pirani, 2020. Probaly, it is *Homalolepis cuneata* (A.St.-Hil. & Tul.) Devecchi & Pirani.
P06676790 ** **P06676791** **LY0118480** **LY0118484**	France	Glaziou, A. (17574)	1889	-	RJ	-	-	-
L.2129194 ** **(601380)**	Netherlands	Vincent, D. (1107)	1915?	-	-	-	-	-
L.2129194 **	Netherlands	Vicente, D. (1107)	1918?	-	-	-	-	-
K001252427 ** **K001252428** **USw9460**	United Kingdom USA	Krukoff, B.A. (5276)	1933	Sena Madureira	AC	09°11'01"S 68°43'21"W	Near mouth of Rio Macauhan, basin of Rio Purus.	See comments on *Schüch, R. (294)*.
RB47828 ** **(RB00382946)** **INPA#** **SPF204891**	Brazil	Nunes, G.M. (36)	1943	Rio de Janeiro	RJ	22°19'18"S 43°24'42"W	Avelar (Vila). Fazenda Pau Grande.	See comments on *Glaziou, A. (2528)*.
UB167228 ** **HB15997** **IPA14453** **SPF204898** **MO101006324** **Y02695274**	Brazil	Duarte, A.P. (5953)	1961	Porto Seguro?	BA	16°27'50"S 39°04'39"W	Entre (Arraial d'?) Ajuda e Porto Seguro.	Except UB167228, the duplicates are identified as *Simaba cuneata* A.St.-Hil. & Tul. [= *Homalolepis cuneata* (A.St.-Hil. & Tul.) Devecchi & Pirani].
IPA76602 ** **HST3406**	Brazil	Monteiro, M.T. (23423)	1970	Santa Cruz Cabrália	BA	16°16'41"S 39°01'29"W	Fazenda Embiruçú.	The duplicate was determined as *Trichilia lepidota* Mart. by J.R.Pirani in 1996.

Deposits (Related Codes)	**Country of Deposit**	**Collector (Collection Number)**	**Year of Collection**	**Municipality**	**State**	**Latitude Longitude**	**Location Notes**	**Comments**
CEPEC9444 ** **IPA44323** **SPF32636**	Brazil	Pinheiro, R.S. (2207)	1973	Maiquinique	BA	15°37'16"S 40°15'58"W	a 20 km Noroeste, Mata.	This is *Homalolepis cuneata* (A.St.-Hil. & Tul.) Devecchi & Pirani.
CEPEC7724 **	Brazil	Pinheiro, R.S. (1571)	1978	Una	BA	15°17'36"S 39°04'31"W	Rod. Una/Sta. Luzia.	See comments on *Schüch, R. (294)*.
CVRD65 ** **INPA167998** **NY02681291**	Brazil	Silva, I.A. (9)	1978	Linhares	ES	19°09'21"S 40°01'07"W	Reserva Florestal da CRVD.	See comments on *Schüch, R. (294)*.
R7961 ** **R71150**	Brazil	Glaziou, A. (12525)	<1980	Barbacena	MG	21°13'33"S 43°46'25"W	-	See comments on *Langsdorff, G.H.v. (Unn.)*.
MO101076492^{T} **P01817243** **P01817244**	USA France	Saint-Hilaire, A. (1630)	<1980	Minas Novas	MG	17°13'07"S 42°35'25"W	Villa do Fanado.	-
MOSS1845 ** **MOSS3573**	Brazil	de-Oliveira, O.F. (1643)	1981	São José de Mipibu	RN	06°04'00"S 35°14'30"W	Coletado em ambiente perturbado; Margem da BR 101; Granja Canaã. Solo arenoso.	See comments on *Martini, A.; et al. (Unn.)*.
UEC170297 ** **(UEC054377)**	Brazil	Sartori, A.L.B.; et al. (204)	1996	Linhares	ES	19°09'21"S 40°01'07"W	Reserva Florestal da CRVD, começo da estrada do Flamengo.	See comments on *Schüch, R. (294)*.
HPL8571 **	Brazil	Salviani, E.R. (162)	1998	Eunápolis	BA	16°22'39"S 39°34'49"W	Coletado ca, 4km da divisa entre os municípios de Eunápolis-Itabela.	See comments on *Schüch, R. (294)*.
MAC31559 **	Brazil	Batista, M.A. (Unn.)	2000	Irecê	BA	11°11'11"S 43°12'14"W	Santo Inácio, Gentil do Ouro.	See comments on *Martini, A.; et al. (Unn.)*.
BHCB73686 **	Brazil	de-Oliveira, P.P. (10660)	2002	Rio das Ostras	RJ	22°25'42"S 42°02'08"W	Reserva Biológica União.	Disregarded based on the occurrence zone described by Devecchi and Pirani, 2020. Probably, it is *Homalolepis cuneata* (A.St.-Hil. & Tul.) Devecchi & Pirani.
RB433248 ** **SPF200795**	Brazil	Fernandes, D.S.; et al. (867)	2003	Cabo Frio	RJ	22°36'14"S 42°00'46"W	2º distrito, Tamoios, Parque Ecológico do Mico Leão Dourado. Floresta de restinga, impactada por loteamento e mineração de areia.	See comments on *Langsdorff, G.H.v. (Unn.)*.

Deposits (Related Codes)	**Country of Deposit**	**Collector (Collection Number)**	**Year of Collection**	**Municipality**	**State**	**Latitude Longitude**	**Location Notes**	**Comments**
RB448611 ** (RB00486251)	Brazil	Rezende, G.S.Z. (232)	2003	Cabo Frio	RJ	22°36'14"S 42°00'46"W	Gargoá, dentro dos limites do Parque Ecológico Municipal do Mico Leão Dourado.	The initial and correct identification of the collection was as *S. cuneata*.
MAC20358 **	Brazil	Machado, M.A.B.L. (371)	2004	Coruripe	AL	10°12'56"S 36°17'56"W	Mata do Nola. Mata Atlântica em tabuleiro.	See comments on *Schüch, R. (294)*.
MAC21103 **	Brazil	Machado, M.A.B.L. (452)	2004	Coruripe	AL	10°12'56"S 36°17'56"W	Usina Coruripe - Fazenda Capiatã A.	See comments on *Schüch, R. (294)*.
SPF194158 ** MBML32752 NY#	Brazil	Demuner V.; et al. (2791)	2006	Governador Lindenberg	ES	19°08'54"S 40°27'27"W	Prop.: Fernando Nicolli	See comments on *Langsdorff, G.H.v. (Unn.)*.
MAC24000 **	Brazil	Lyra-Lemos, R.P.; Rodrigues, M.N.; Cavalcante, F. (9149)	2006	Feliz Deserto	AL	10°27'29"S 36°28'34"W	-	See comments on *Schüch, R. (294)*.
VIES31682 **	Brazil	Dias, H.M.; et al. (263)	2007	Caravelas	BA	17°43'55"S 39°15'57"W	-	The collection was determined by W. Thomas, who also determined the HUSEFS collection, both incorrectly. It is *Homalolepis cuneata* (A.St.-Hil. & Tul.) Devecchi & Pirani.
RB451102 ** SPF204513 VIES27621 NY02711703	Brazil USA	Farney, C. (4747)	2007	Conceição da Barra	ES	18°34'38"S 39°44'50"W	Conceição da Barra. Comunidade de Lajinha, Fazenda Rancho Tropical II, Mata de Restinga.	See comments on *Langsdorff, G.H.v. (Unn.)*.
MAC26101 **	Brazil	Lopes, L.(Unn.)	2007	Paripueira	AL	09°46'50"S 35°55'17"W	Margem da AL-101. Proximidades do Lot. Sonho Verde.	See comments on *Schüch, R. (294)*.
HPL11721 **	Brazil	Lorenzi, H. (6370)	2007	Lauro de Freitas	BA	12°53'40"S 38°19'38"W	Coletado no condomínio Buscavida.	See comments on *Martini, A.; et al. (Unn.)*.
HPL10904 **	Brazil	Lorenzi, H. (6375)	2007	Camaçari	BA	12°41'51"S 38°19'27"W	Coletado na estrada do coco. Logo após a ponte sobre o rio Jauípe (sentido Norte).	Disregarded based on the occurrence zone described by Devecchi and Pirani, 2020. In addition, *H. bahiensis* was collected in the same location (SPF216522) and in other

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
								regions of the municipality (ALCB024904).
RFFP20342 **	Brazil	-	2008	Angra dos Reis	RJ	23°10'22"S 44°10'44"W	Ilha Grande, Parque Estadual da Ilha Grande, Estrada Vila do Abraão a Vila Dois Rios, área do Projeto RAPELD - Módulo Leste, Parcela Curva Maravilha.	See comments on *de-Oliveira, P.P. (10660)*.
RB470422 ** **(RB00532060)** **CEPEC125332** **HCF000002838** **HCF14109** **HUEFS177514** **(HUEFS38756)** **SPF194243** **SPF206419** **MBML36753** **UPCB76380** **NY1184779**	Brazil USA	Amorim, A.M.A.; et al. (7550)	2008	Nova Venécia	ES	18°46'37"S 40°26'38"W	Área de Proteção Ambiental Pedra do Elefante.Trilha principal na mata do Fuxico. Floresta estacional semedicidual.	Except HCF14109 and SPF194243, the duplicates are identified as *S. cuneata*.
MAC34633 **	Brazil	Rodrigues, M.N. (2274)	2008	Barra de São Miguel	AL	10°23'60"S 36°31'18"W	-	Disregarded based on the occurrence zone described by Devecchi and Pirani, 2020. In addition, there are several collections of *Homalolepis cuneata* (A.St.-Hil. & Tul.) Devecchi & Pirani.
MAC40404 **	Brazil	Lyra-Lemos, R.P.; Mota, M.C.S.; N. Ramos; Lessa L. (12264)	2009	-	AL	-	-	See comments on *Schüch, R. (294)*.
MAC47695 **	Brazil	Cavalcante, F.; Bastos, A.M. (458)	2010	Arapiraca	AL	10°15'56"S 37°06'40"W	Povoado Bananeira.	See comments on *Rodrigues, M.N. (2274)*.
HUEFS183164 ** **SPF221516**	Brazil	de-Matos, E.N. 357)	2011	Caravelas	BA	17°43'55"S 39°15'57"W	Barra de Caravelas-Faz. Walace.	See comments on *Langsdorff, G.H.v. (Unn.)*.

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
ASE0027243 ** **SPF217234**	Brazil	Landim, M. (1595)	2011	Pirambu	SE	10°44'16"S 36°51'22"W	Restinga.	The main deposit was initially as *H. floribunda* (2012) and then as *H. ferruginea* (2015), while the ducplicata SPF was determined as *H. bahiensis* (2015), demonstrating that the material is not *H. floribunda*. In addition, there are several collections of *H. bahiensis* and *Homalolepis arenaria* (Devecchi & Pirani) Devecchi & Pirani for the municipality. Therefore, this collection probably is one of these species.
UFRN13669 ** **EAC61285** **SPF232942**	Brazil	São-Mateus, W.M.B.; Sulzbacher, M. (36)	2011	Tibau do Sul	RN	06°13'35"S 35°03'56"W	Santuário de Pipa, a margem da estrada Tibau do Sul-Praia da Pipa. Área antropizada em estado avançado de regeneração ca. 40 anos.	Initially (2017) the collection was determined by the collector as *S. cuneata*., while its duplicate SPF was determined as *H. bahiensis* by Devecchi, M.F.; then (2018) the duplicate EAC was again determined as *H. floribunda* by Sued, S. In our analysis, it is *H. bahiensis*, since there are other collections of this species in the municipality (IPA56303, IPA66753, IPA67206, RN6392 and VIES18652), in addition to collections identified at the genus level and morphologically related to *H. bahiensis*.
HUENF10469 **	Brazil	Souza, T.P. (28)	2011	Campos dos Goytacazes	RJ	21°47'49"S 41°27'14"W	Maciço do Itaoca.	See comments on *de-Oliveira, P.P. (10660)*.

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
MAC63710 ** **COR16256**	Brazil	Ferreira, E.S.; Oliveira, E.V.S. (159)	2012	Pirambu	SE	10°44'16"S 36°51'22"W	Rebio de Santa Isabel: Lagoa redonda, acesso pela RPPN Dona Benta e seu Caboclo. Restinga.	There are several collections of *H. bahiensis* and *H. arenaria* for the municipality. Therefore, this collection is one of these species.
VIES40079 **	Brazil	Gomes, J.M.L. (4261)	2012	Presidente Kennedy	ES	21°05'56"S 41°02'48"W	Restinga.	Other collections from this municipality [*Pereira, O.J.; et al., (5816)*, *Vinha, P.C. (844)* and *Souza, V.D. (345)]* are of *H. cuneata*, which led us to not consider this record as *H. floribunda*.
SPF204555	Brazil	Kubo, M.T. (156)	2012	Araçuaí	MG	16°50'58"S 42°04'13"W	Caatinga.	
RBR43361 **	Brazil	Alverga, T.P.P. (Unn.)	2013	Rio de Janeiro	RJ	23°02'51"S 43°31'45"W	Restinga.	See comments on *de-Oliveira, P.P. (10660)*.
UFP80241 **	Brazil	Farias, M.C.V.; et al. (417)	2013	Pirambu	SE	10°36'51"S 36°45'01"W	Povoado Alagamar, Assentamento São Sebastião. Tabuleiros costeiros.	See comments on *de-Ferreira, E.S.; Oliveira, E.V.S. (159)*.
RBR42976 **	Brazil	Alverga, T.P.P. (Unn.)	2014	Rio de Janeiro	RJ	23°02'51"S 43°37'48"W	Marambaia. R12/119, Itaguaí.	See comments on *de-Oliveira, P.P. (10660)*.
RBR43362 **	Brazil	Alverga, T.P.P. (Unn.)	2014	Rio de Janeiro	RJ	23°02'51"S 43°37'48"W	Marambaia. R20/14, Itaguaí.	See comments on *de-Oliveira, P.P. (10660)*.
RBR43363 **	Brazil	Soares, L.H. (Unn.)	2014	Rio de Janeiro	RJ	23°02'55"S 43°35'58"W		See comments on *de-Oliveira, P.P. (10660)*.
SPF216519 **NY02694870**	Brazil USA	Devecchi, M.F. (403)	2015	Araçuaí	MG	16°52'27"S 41°56'28"W	11 km na estrada para Caraí, pelo acesso na BR-367, distante 35 km de Itinga sentindo Araçuaí.	-
CSTR6019 **	Brazil	Kerpel, S. (Unn.)	2015	Maturéia	PB	07°15'59"S 37°20'57"W	Floresta Serrana de Altitude.	Collection note: "Tree ca. 4m tall".
VIC056446 *	Brazil	Verly, O.M. (72)	2023	Guanhães	MG	18°49'29"S 42°51'51"W	Área de Preservação Projeto Cacheoeira das Pombas.	-
***Homalolepis insignis* (A.St.-Hil. & Tul.) Devecchi & Pirani** **Synonyms:**								

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
homotype: *Simaba insignis* A.St.-Hil. & Tul. ***Quassia insignis* (A.St.-Hil. & Tul.) Noot.** **heterotypic: *Simaba glandulifera* Gardner** ***Aruba glandulifera* (Gardner) Kuntze** **heterotypic: *Simaba longifolia* Casar.**								
P00401122 **	France	-	-	-	-	-	-	Record without any collection information.
SPFW5101 **	Brazil	-	-	-	DF?	-	-	There are no records of collection of reproductive or sterile material. It is probably not the species.
MO101076568	USA	Guillemin, J.B.A. (1016)	-	-	-	-	-	-
MO(G)101076573^{I} **MO10306358^{H}**	USA	Casaretto, G. (1854)	-	-	-	-	-	-
NY02681290	USA	Burchell, W.J. (1165)	-	-	RJ?		-	-
S09-3474	Sweden	Mosén, H.J. (2448)	-	Rio de Janeiro	RJ	-	-	-
NY02681294 **NY02840231**	USA	Bowie, J.; Cunningham, A. (367)	1814	-	RJ?	-	-	-
NY02695258	USA	Riedel, L. (999)	1832	-	RJ	-	-	-
NY02681295	USA	Luschnath, B. (41)	1835	-	RJ	-	-	-
NY02681299	USA	Gardner, G. (19)	1836	Rio de Janeiro	RJ	22°57'06"S 43°12'44"W	In sylvis montis Corcovado circa aquaductum.	-
G00342782 **G00342783** **E296300▼** **E296302** **P01817251▼** **P01817253** **GH00044365** **GH00044366** **NY02286533** **NY02286534** **MO103063570^{L}** **MO103063573▼ ^{IL}**	Switzerland United Kingdom France USA	Gardner, G. (20)	1836	Rio de Janeiro	RJ	22°57'09"S 43°12'42"W	In woods aqueduct on the Corcovado.	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
MO103063581^{H} US1066236 F26530? ^{IL} **W0059306 (G-G-244460-1 G-G-244460-2)**	Austria							
P01817249^{T} P01817250	France	Guillemin, J.B.A. (840)	1839	Rio de Janeiro	RJ	22°57'02"S 43°13'24"W	Corcovado. Serra d'Estrella	-
P00394169	France	Claussen, M. (56)	1842	-	-		-	-
V0203656F (940036) V0389456F (940037) P06676763	USA France	Nadeaud, J. (6135)	1862	Rio de Janeiro	RJ	22°57'08"S 43°12'41"W	Corcovado.	-
P06676764	France	Glaziou, A.F.M. (2904)	1868	Rio de Janeiro	RJ	22°57'08"S 43°12'41"W	Corcovado.	-
P06676765	France	Glaziou, A.F.M. (6135)	1872	Rio de Janeiro	RJ	22°57'08"S 43°12'41"W	Corcovado.	-
R7958 P06676766▼ P06676768	Brazil France	Glaziou, A.F.M. (9384)	1878	Rio de Janeiro	RJ	22°56'53"S 43°12'34"W	Caixa d'água, ao Corcovado.	-
NY02681298 NY02682056 US1420691 US1420692 US01858067 US01858068 P06676769 P06676770	USA France	Miers, J. (3805)	1878	Rio de Janeiro	RJ	22°57'08"S 43°12'41"W	Corcovado.	-
RB1237 (RB00382986) BHCB41390? SPF204882	Brazil	Ducke, J. (Unn.)	1921	Rio de Janeiro	RJ	22°59'19"S 43°14'12"W	Dois Irmãos. Gávea.	-
HB269188 (RBcarpo2181 00768353) PACA-AGP119588	Brazil	Kuhlmann, J.G. (Unn.)	1930	Rio de Janeiro	RJ	22°59'24"S 43°16'02"W	-	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
RB82439 (RB00383028)	Brazil	Lages, C.A. (225)	1936	-	-	--	-	-
RB149256 (RB00383059) **SPF204902**	Brazil	Tatto, D.C.; et al. (686)	1940	Rio de Janeiro	RJ	22°53'57"S 43°37'25"W	Caminho do Morro do Papagaio.	-
V0203657F (839058) **W0059304** **W0059305**	USA	Helmreichen, V. (57)	<1943	Rio de Janeiro	RJ	22°58'49"S 43°19'00"W	-	-
RB48279 (RB00382971) **INPA#** **SPF204883**	Brazil	Kuhlmann, J.G. (6258)	1943	Rio de Janeiro	RJ	22°59'11"S 43°14'43"W	Alto da Gávea.	-
RB53045 (RB00382977) **SPF204896**	Brazil	Occhioni, P. (345)	1943	Rio de Janeiro	RJ	22°58'22"S 43°14'49"W	Matas da Vista Chinesa.	-
FCAB2497	Brazil	Capell, P. (Unn.)	1951	Rio de Janeiro	RJ	22°58'60"S 43°14'07"W	Gávea.	-
HB35371 **NY02681303**	Brazil USA	Trinta, Z.A. (1086); Fromm, E. (2162)	1964	Anchieta	ES	20°43'50"S 40°34'11"W	-	-
R71149	Brazil	Glaziou, A.F.M. (Unn.)	<1980	Rio de Janeiro	RJ	22°57'08"S 43°12'41"W	Corcovado.	-
RB696380 (RB01293853)	Brazil	Cominote, J. (105)	1985	Rio de Janeiro	RJ	22°57'47"S 43°16'55"W	Serra Carioca, próximo a estrada do Salgueiro.	-
SPF138516 **SP338919** **NY572006**	Brazil USA	Pirani, J.R. (4517)	1999	Ubatuba	SP	23°32'14"S 45°04'39"W	Parque Estadual da Ilha Anchieta, Trilha da Restinga da Praia das Palmas.	-
VIES16162	Brazil	Pereira, O.J.; Espindula, E. (6366)	2000	Domingos Martins	ES	20°21'21"S 40°37'27"W	Rio Jucu.	-
SPSF32763	Brazil	Duarte, R.M.R. (Unn.)	2001	Ubatuba	SP	23°31'57"S 45°02'47"W	Parque Estadual da Ilha Anchieta.	-
SPF235185	Brazil	Alves, G.G.N. (51)	2014	Ubatuba	SP	23°32'32"S 45°03'55"W	Parque Estadual da Ilha Anchieta.	-
SPF235207	Brazil	Alves, G.G.N. (71)	2014	Ubatuba	SP	23°32'25"S 45°03'51"W	Parque Estadual da Ilha Anchieta, Mata ao lado de alojamento.	-
SPF216193	Brazil	Devecchi, M.F. (269)	2014	Ubatuba	SP	23°32'30"S 45°04'19"W	Ilha Anchieta, Trilha para a Praia do Sul, ca. 500m depois da bifurcação da	-

Deposits (Related Codes)	Country of Deposit	Collector (Collection Number)	Year of Collection	Municipality	State	Latitude Longitude	Location Notes	Comments
							trilha para a Praia das Palmas.	
SPF216188 **SPF233641** **NY02694840**	Brazil USA	Devecchi, M.F. (270)	2014	Ubatuba	SP	23°32'30"S 45°04'19"W	Ilha Anchieta, Trilha para a Praia do Sul, ca. 500m depois da bifurcação da trilha para a Praia das Palmas.	-
SPF216189 **NY02694846**	Brazil USA	Devecchi, M.F. (271)	2014	Ubatuba	SP	23°32'23"S 45°04'07"W	Parque Estadual da Ilha Anchieta. Mata atrás dos alojamentos, no caminho que leva para as trilhas para as Praia das Palmas e do Sul.	-
SPF216508 **NY02694835**	Brazil USA	Devecchi, M.F. (388)	2015	Ubatuba	SP	23°32'48"S 45°04'36"W	Ilha Anchieta, Trilha para a Praia do Sul.	-
SPF216509 **NY02694853**	Brazil USA	Devecchi, M.F. (389)	2015	Ubatuba	SP	23°32'48"S 45°04'36"W	Ilha Anchieta, Trilha para a Praia do Sul.	-
VIC056916 *	Brazil	Verly, O.M. (54)	2022	Caratinga	MG	19°30'49"S 42°26'09"W	Área de preservação da CENIBRA. Parcela 13.	-
CVRD17732	Brazil	de-Marcelino, M.S. (37)	2023	Baixo Guandú	ES	19°31'08"S 41°00'56"W	Terreno do Moratti.	-
VIC058109 *	Brazil	Verly, O.M. (236)	2024	Caratinga	MG	19°30'25"S 42°26'12"W	Área de preservação da CENIBRA. Barranco da estrada que margeia a Lagoa Silvana.	-

Notes: * Our records; ** Records disregarded due to inconsistencies in information of collection, species identification, morphological description, field notes, occurrence area, etc.; ? - Probable information based on the annotations of the record; ≤ - There is only the year of determination of the collection, and therefore, the year of collection is equal to or earlier than this; # - Deposits mentioned but not located by us; ▼ - The interval from this to the next deposit number are duplicates; Underline - Approximate probable coordinates based on the collection notes. ^{T} – Type; ^{H} – Holotype; ^{I} – Isotype; ^{IL} – Isolectotype; ^{L} – Lectotype; ^{P} – Paratype; ^{S} – Syntype; ^{Ph} – Photographic record only.

CHAPTER IV:

Myrcia magnipunctata (MYRTACEAE), A NEW SPECIES FROM MINAS GERAIS, SOUTHEASTERN BRAZIL

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Myrcia magnipunctata (Myrtaceae), a new species from Minas Gerais, southeastern Brazil

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Myrcia De Candolle (1827: 406) is an American genus with 744 (Santos *et al.* 2024) to 788 species (POWO 2024). There are 410 species occurring in Brazil, from which 114 are recorded from the Brazilian state of Minas Gerais (Santos *et al.* 2025). Along the study of forest fragments in the southeastern portion of Minas Gerais there were collected some specimens of *Myrcia* with long and narrow leaves which we consider to represent an undescribed species. The generic concept of *Myrcia* in this paper follows Lucas *et al.* (2018) and the species is considered under the morphological species concept (McDade 1995). Morphological terms follow largely Gonçalves & Lorenzi (2011) and Harris & Harris (2001). Measurements are presented in the order length × width, except if stated otherwise; when only one set of measures is presented it refers to length. Herbarium acronyms follow Thiers (2024).

Myrcia magnipunctata Verly, Cabral-da-Silva & Sobral, *sp. nov.* Type:—BRAZIL. Minas Gerais: Coronel Fabriciano, área de preservação da Cenibra, parcela 10, 19°24.46'16" S, 42°43.21'42" W, 24 May 2024, *O.M. Verly 1* (holotype VIC!). Figures 1–2.

Diagnosis:—This species is morphologically close to *Myrcia espiritosantensis* B.S.Amorim (in Amorim & Alves 2016: 288; type images MBML barcode 00017196, RB barcode 00548991), from which it differs by the leaves with petioles up to 2 mm (*versus* petioles 5–7 mm) and blades 400–410 × 67–98 mm, 4.3–6.1 times longer than wide (*vs.* 165–300 × 45–102 mm, 2.9–3.7 times longer than wide) with 23–32 lateral veins at each side (*vs.* 16–18), bracteoles elliptic or ovate, to 2 mm, not concealing the buds before anthesis (*vs.* ca. 7 mm, concealing the buds before anthesis). It is also vegetatively strikingly alike *Myrcia megaphylla* M.F.Santos & Sobral (in Santos *et al.* 2015: 104, type images BHCB 035310, K 000752554, SPF 198095), also with oblong blades, but differs by the not monopodial growth (*vs.* monopodial growth in *M. megaphylla*), blades visibly pilose (*vs.* glabrous) and markedly punctate (*vs.* glandular dots inconspicuous), flowers with sepals about the same size (*vs.* unequal) and anthers with displaced thecae (*vs.* not displaced).

Description:—Trees 3–5 m height, rugose trunk, grey, longitudinally striate, very slender, less than 50 mm in diameter, scarcely branched, sometimes bending downwards. Twigs cylindrical, the internodes to 70 × 7 mm, rufescent, covered with simple erect trichomes 1.0–1.3 mm. Leaves opposite, subsessile, usually congested at the apex of short branches; petioles visible only from the abaxial side, to 2 × 3 mm, semiterete, with simple brown trichomes 0.3–0.5 mm; blades oblong, 400–410 × 67–98 mm, 4.3–6.1 times longer than wide (sometimes proximal leaves in a branch are smaller, to 140 × 50 mm), discolorous when dry, adaxial surface dull dark green or grey, with very scattered simple grey trichomes 0.2–0.3 mm on the surface, these better visible only under lenses, the indumentum denser and more visible along the midvein, abaxial surface light brown, densely covered with simple erect brown trichomes 0.5–0.7 mm, the indumentum denser along the veins; glandular dots 0.2–0.3 mm in diameter, very scattered, less than one per mm², scarcely perceptible adaxially (only when backlit), visible abaxially, darker than the surface and sometimes slightly raised above it; base cordiform, the lobes extending about 10 mm behind the insertion of the petioles; apex acute; midvein finely impressed adaxially and strongly raised abaxially; lateral veins 23–32 at each side, leaving the midvein at angles 45–80°, slightly raised adaxially, markedly raised abaxially; secondary lateral veins, that is, those between the lateral veins forming the intramarginal vein, and high order venation perceptible on both surfaces and with a thinner gauge than the main lateral veins; intramarginal veins two,

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HERBÁRIO VIC - VIÇOSA (MG, BRASIL)

MYRTACEAE **VIC** **57664**

Espécie: *Myrcia*

Det./Data: Sobral, M. & Verly, O.M.

Proc.: Brasil, Minas Gerais, Coronel Fabriciano, Área de preservação da CENIBRA. Parcela 10

19°24'46.16"S 42°43'21.42"W

Obs.:

Hábito.:

Col.: Verly, O.M. (1) **Data:** 24/05/2024

FIGURE 1. *Myrcia magnipunctata*. Holotype (scale: 50 mm).

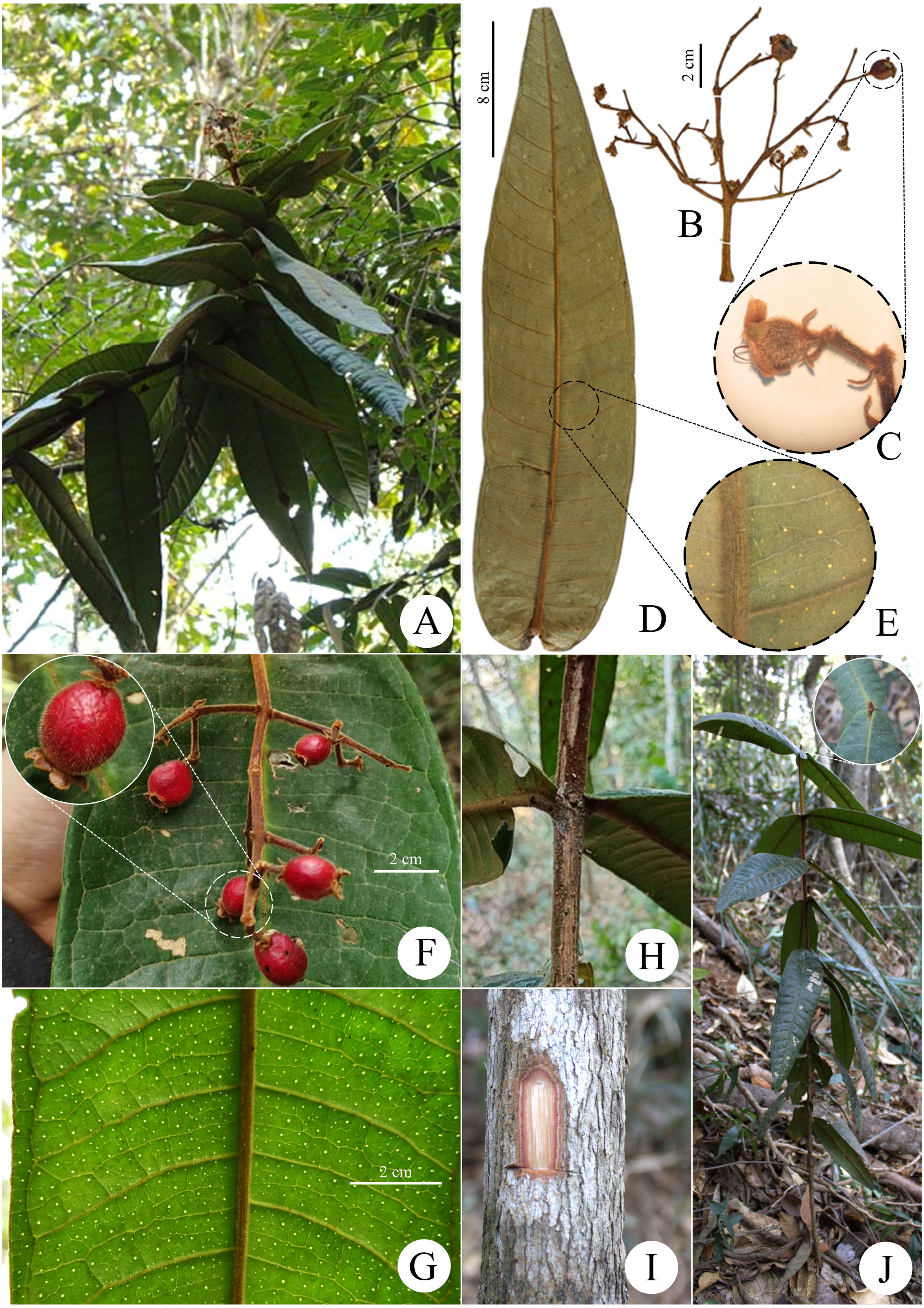

FIGURE 2. *Myrcia magnipunctata*. A) Adult individual in nature, about 4 m high, with arching branches; B) part of inflorescence with immature fruits; C) immature fruit; D) abaxial surface of an adult leaf; E) Detail of midvein, lateral veins and glandular dots in abaxial surface; F) mature fruits in nature, and a detail of them; G) glandular dots in nature; H) base of leaves and phyllotaxy in nature; I) adult stem in nature; J) juvenile individual in nature (not collected), with a detail of its apical bud. (A–E: *Verly 225*; F–H: *Verly 245*; I: *Verly 1*. Photos by O.M. Verly) .

2.5–3.1 mm and 0.4–0.5 mm from the slightly revolute margin. Inflorescences paniculiform, on the last node of the season branches, at the axiles of the distal leaves, 2–4 axes emerging from each axil, with 70–120 flowers each, densely covered with brown or rufescent trichomes 1.0–1.2 mm, the axes 95–135 × 1.5–3.0 mm, the peduncle 25–47 mm, the first order branches diminishing in length distally, the proximal ones 55–60 mm, the distal ones about 10 mm, the second order branches 5–15 mm, the third order branches 5–6 mm; bracts narrowly triangular, to 5 × 1–2 mm, persistent; flowers sessile usually in groups of three at the apex of the branches; bracteoles ovate or elliptic, 2 × 1 mm, occasionally deciduous before anthesis but usually persistent after it, with simple grey trichomes 0.2–0.5 mm; flower buds 4–5 mm × 3.0–3.5 mm in diameter, pilose, more densely so on the ovaries, the trichomes simple, grey or brown, 0.2–0.3 mm; sepals five, about the same size, widely ovate, 2.0–2.3 × 2 mm, pilose on both surfaces, abaxially with ascending simple trichomes 0.2–0.3 mm, adaxially with simple appressed trichomes to 0.1 mm; petals five, rounded, 2.5–3.0 mm in diameter (in bud), concave, scarcely glandulose, abaxially with trichomes to 0.2 mm, adaxially glabrous; stamens not counted, the filaments (in bud) about 2 mm, the anthers subglobose, 0.2–0.3 × 0.2 mm, the thecae slightly displaced; staminal ring (in bud) 2 mm in diameter, with white trichomes to 0.2 mm; hypanthium not extended above the ovary; style glabrous except for sparse trichomes to 0.2 mm at the base, to 3 mm (in bud), stigma punctiform; ovary with two locules and two ovules per locule. Fruits subglobose to slightly obpyriform or ellipsoid, 10–15 mm long × 10–12 mm in diameter, reddish to vinaceous when ripe, with scattered trichomes, crowned by the sepals, with scarce pulp, no more than 1 mm thick; seeds two to three, somewhat reniform, about 10 mm long, 6 mm wide and 3 mm thick, testa brown, shining, easily detachable; embryos with two foliaceous cotyledons and a visible hypocotyl.

Distribution, habitat, phenology:—*Myrcia magnipunctata* is presently known from a semideciduous forest fragment in the municipality of Coronel Fabriciano, in the Atlantic Forest biome in the southeastern portion of the state of Minas Gerais, where it grows as a small, slender treelet in the interior of forests; immature flowers were collected in May and mature fruits in October.

Affinities:—This species is related with *Myrcia espiritosantensis*, from the southeastern Brazilian state of Espírito Santo, with which it is compared in the diagnosis. Regarding its sectional placement, the anthers with thecae displaced in relation to each other suggests its inclusion in section *Gomidesia* (O.Berg) B.S.Amorim & E.Lucas (in Lucas *et al.* 2018: 9; basionym: *Gomidesia* O.Berg, 1855–1856: 5). It is also vegetatively alike *Myrcia megaphylla*, a species from rocky outcrops in the Campos Rupestres of Minas Gerais, with which it is also compared in the diagnosis; this species, nevertheless, seems to belong to *Myrcia* sect. *Myrcia* (cited by Santos *et al.* 2015 as "*Myrcia* clade 5" in reference to the then informal grouping of species of *Myrcia* proposed by Lucas *et al.* [2011: 922], later assigned to formal section in Lucas *et al.* 2018).

Conservation:—*Myrcia magnipunctata* is known from three individuals collected—and eight more, including seedlings, observed—along a tree inventory in a forest fragment in the municipality of Coronel Fabriciano. This may be an indicative of its rarity, although the collection site is not sufficiently surveyed in order to allow a proper evaluation of the its possible conservation status. Coronel Fabriciano has an area of 221 km² (IBGE 2024) from which there are recorded 166 plant collections (CRIA 2024), what points for an area still scarcely surveyed, with an average of 0.75 collections/km², still below the minimum average index of 1 collection/km² suggested by Campbell (1989). An evaluation of its conservation status via Geocat (Geocat 2025) resulted in an Extent of Occurrence of 0.087 km² and an Area of Occupancy of 4 km², suggesting a status of Critically Endangered (CR) according to IUCN guidelines (IUCN 2019). Nevertheless, considering the scarcity of knowledge of the area studied, additional information may modify this evaluation in a near future.

Etymology:—The epithet, meaning "with large punctuations" in Latin, alludes to the markedly visible glandular dots in the blades of the species.

Paratypes:—BRAZIL. Minas Gerais: Coronel Fabriciano, área de preservação da Cenibra, proximidades da parcela 10, 19°24.46'21" S, 42°43.21'42" W, 9 September 2024, *O.M. Verly 225* (VIC!); *ibidem*, área de preservação da Cenibra, projeto São José, Parcela 06, 19°24'52"S, 42°43', 33" W, 16 October 2024, *O.M. Verly 245* (VIC!).

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GENERAL CONCLUSION

We discovered that our fragments were dissimilar, with highly heterogeneous species compositions and taxonomic and functional diversity indices. Climatic and anthropogenic variables had significant fixed effects, but the random effect of land-use history (LUH) and time since abandonment stood out in explaining the variability of the different taxonomic and functional diversity indices. Our findings contribute to elucidating the environmental filtering mechanisms driven by anthropogenic and environmental factors in the assembly of tree-communities in the Atlantic Rainforest. This contribution is especially important for the conservation of hyperdiverse tropical forests, as tree diversity is fundamental for their ecological stability and buffering of extreme climatic events.

Over 20 years of monitoring, forest fragments showed increases in stem abundance until 2012, followed by declines in subsequent inventories. Basal area, aboveground biomass (AGB), species richness and diversity, and community-weighted means for ecological group and deciduousness increased steadily, while richness and functional diversity followed distinct successional trajectories among fragments. The relative contribution of rare species was proportionally higher for AGB than for abundance, and despite intense transitions between species abundance classes (SAC), this pattern remained over two decades. Common species tended to decline in population size as rare species increased in abundance, yet both groups continued to increase in AGB. Land-use history (LUH) explained most of the variance in net dynamic of abundance (NDA) and AGB (NDB). Taxonomic and functional diversity indices consistently drove NDA across SACs. Additionally, climatic factors related to precipitation mainly influenced the dynamics of common species, whereas rare species were more affected by edaphic gradients of acidity and fertility.

We made 19 new records of 12 rare species, distributed across seven botanical families. These species have historically been poorly sampled, with a good portion of their 330 previous records, containing inconsistencies in identification or collection origin. Remarkably, *Homalolepis insignis* (A.St.-Hil. & Tul.) Devecchi & Pirani and *Rhodostemonodaphne anomala* (Mez) Rohwer were documented for the first time in the state of Minas Gerais. Previous records from major digital biodiversity repositories and herbarium collections are subject to many inconsistencies and the crucial need for data curation. Our findings provide an overview of how both the new records and the evaluation of existing data influence the conservation perspectives for these rare and endangered species.

Myrcia magnipunctata is a new species, rare and threatened, characterized by sparsely branched treelets distributed in the subcanopy of a restricted portion of a fragment of Semideciduous Seasonal Forest in the municipality of Coronel Fabriciano, Minas Gerais. It is morphologically close to *Myrcia espiritosantensis* B.S.Amorim, from which it differs by the leaves with petioles up to 2 mm (*versus* petioles 5–7 mm) and blades 400–410 × 67–98 mm (*vs.* 165–300 × 45–102 mm). It is also vegetatively strikingly alike *Myrcia megaphylla* M.F.Santos & Sobral, also with oblong blades, but differs by the not monopodial growth (*vs.* monopodial growth), blades visibly pilose (*vs.* glabrous) and markedly punctate (*vs.* glandular dots inconspicuous).

Taken together, the findings presented in this thesis underscore the complex interactions between environmental gradients, anthropogenic legacies, and biodiversity patterns in the Atlantic Rainforest. By integrating long-term forest monitoring, modeling approaches, and taxonomic investigation, we provide a multifaceted contribution to understanding how diversity is regulated by environmental conditions, and how both interact to regulate forest dynamics, particularly through the roles of rare and common species. These findings are essential for advancing forest conservation strategies, improving biodiversity databases, and refining taxonomic knowledge in one of the most threatened tropical biomes. Ultimately, this work reinforces the urgency of protecting hyperdiverse ecosystems, not only for their intrinsic value but also for their critical role in ecosystem stability, rare species conservation and climate resilience.