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Changes in plant community attributes and carbon stocks in altimontane forest islands in the Atlantic Forest-Cerrado transition under different conservation conditions

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Abstract
Background: Ecotonal regions in tropical mountains, such as the Altimontane Forest Islands, are home to unique, highly biodiverse, and vulnerable plant communities. Human activities alter the functionality, structure, diversity, and provision of ecosystem services in these ecosystems. Uncertainties persist regarding their resilience to human impacts and the most effective ways to conserve them.
Questions: How do the species composition, vegetation structure, and diversity differ between protected and unprotected forest islands subject to anthropogenic disturbances? Are there differences in the carbon stocks of these forest islands?
Studied species: Vascular plants of the Cerrado-Atlantic Forest.
Study site: Forest Islands in Minas Gerais, Brazil.
Methods: We calculated alpha and beta diversity and estimated aboveground biomass and aboveground carbon stocks from tree samples (diameter at breast height ≥ 5 cm) collected in Forest Islands.
Results: We found that protected and unprotected areas had similar alpha diversity but differed in floristic composition. We observed high beta dissimilarity between protected and unprotected Altimontane Tropical Forest Islands (β*sor* = 0.93 and βbray = 0.94). Unprotected islands show greater species substitution (90 %), unimodal vertical structure, greater susceptibility to colonization by new generalist species, and lower carbon stock (80.70 Mg C ha^{-1}).
Conclusions: Changes in species composition, vegetation structure, and carbon storage were observed on unprotected islands, demonstrating the need to protect these unique ecosystems.
Keywords: alpha diversity, beta diversity, carbon sequestration, community composition, and structure.

Resumen
Antecedentes: Las regiones de ecotono en montañas tropicales, como las Islas de Bosque Altimontano, albergan comunidades vegetales únicas, altamente biodiversas y vulnerables. Las actividades humanas alteran la funcionalidad, la estructura, la diversidad y la provisión de servicios ecosistémicos en estos ecosistemas. Persisten incertidumbres sobre su resiliencia a los impactos humanos y las formas más efectivas de conservarlos.
Preguntas: ¿Cómo difieren la composición de especies, la estructura de la vegetación y la diversidad entre islas forestales protegida y no protegida sometidas a perturbaciones antropogénicas? ¿Existen diferencias en las reservas de carbono de estas islas forestales?
Especies de estudio: Plantas vasculares del Cerrado-Bosque Atlántico.
Sitio de estudio: Islas de Bosque en Minas Gerais, Brasil.
Métodos: A partir del muestreo de individuos arbóreos (DAP ≥ 5 cm) en las Islas de Bosque, se calculó la diversidad alfa y beta, y se estimó la biomasa aérea y el almacén de carbono aéreo.
Resultados: Se encontró que las áreas protegidas y no protegidas presentaban una diversidad alfa similar, pero una composición diferente. Se observó alta disimilitud de especies entre ambos tipos de área (β*sor* = 0.93 y β*bray* = 0.94). La isla no protegida mostró mayor recambio de especies (90 %), estructura vertical unimodal, mayor susceptibilidad a la colonización por nuevas especies generalistas y menores reservas de carbono 80.70 Mg C ha^{-1}.
Conclusiones: Los cambios en la composición de especies, la estructura de la vegetación y el almacenamiento de carbono en la isla no protegida refuerzan la necesidad de proteger estos ecosistemas únicos.
Palabras clave: composición y estructura de la comunidad, diversidad alfa, diversidad beta, almacén de carbono.

Tropical forests are essential ecosystems for maintaining biodiversity and regulating the global climate (Bañares-De-Dios *et al*. 2024). However, tropical forests face major challenges from socio-economic development and growing environmental degradation (Cowie 2022). A large portion of primary natural habitats in tropical and subtropical regions has already been altered, leading to declines in biodiversity and ecosystem services essential to life on Earth and human well-being (Fernandes *et al*. 2020, Cunha-Blum *et al*. 2025).

Despite covering only 25 % of the Earth's surface, mountains play a strategic role in species conservation, harboring around 87 % of the global biodiversity, with high levels of endemism due to their altitudinal and climatic variations (Humboldt & Bonpland 1805, Myers *et al*. 2000, Mittermeier *et al*. 2005, Bañares-de-Dios *et al.* 2024). On a global scale, the Espinhaço Range Biosphere Reserve (ERBR) is a hotspot, recognized as a megadiverse region in the Neotropic (Myers *et al*. 2000, Cunha-Blum *et al.* 2025). Despite occupying only 1 % of the Brazilian territory, ERBR is home to approximately 15 % of Brazil's vascular flora, and around 40 % of its species are endemic (Coelho *et al.* 2018a, Silveira *et al.* 2019, Fernandes *et al.* 2020, Lima *et al.* 2020).

Within ERBR, the Altimontane Seasonal Forests of the Atlantic Forest stand out, which are part of the Grassland-Savannah transition of Cerrado (Coelho *et al.* 2018a). Known as "Capões de Mata" or forest islands, these formations are adapted to the high environmental heterogeneity and unique conditions of Brazilian mountains (Costa *et al.* 2023, Ribeiro *et al.* 2018). Forest islands have emerged as endemic centers of animal and plant species in South America, acting as local and regional biodiversity centers (Myers *et al.* 2000, Mittermeier *et al.* 2005, Coelho *et al.* 2016, Neves *et al.* 2018, Costa *et al.* 2021b).

The impacts of human activities have put the resilience of ERBR forest islands to the test (Coelho *et al*. 2018a, Morellato & Silveira 2018). Thus, understanding these forest islands for their flora, their vegetation structure and their key ecosystem services crucial to humanity, like carbon (C) stocks is essential (Costa *et al.* 2021a, Gonçalves *et al.* 2023). After all, these studies help to understand local biodiversity and establish strategies to mitigate the effects of global climate change, especially in the Neotropical region (Cunha-Blum *et al.* 2025, Tassinari *et al.* 2025), making it possible to guarantee its conservation against growing human and environmental pressures.

Changes in floristic composition, vegetation structure and species diversity on forest islands might be a multifactorial process. These changes include the conservation and land use regime, as a consequence of recurrent anthropogenic actions and the effects of climate change (Costa *et al.* 2021b, 2023), which modify soil properties, dynamics, and ecological function of forest-island ecosystems (Coelho *et al.* 2018b). Consequently, such changes influence floristic composition, species diversity and carbon sequestration (Souza *et al.* 2020, Campos *et al.* 2021). In addition to these factors is the fragility of elevational environments, which are characterized by high spatial and temporal variability, mainly due to bioclimatic and geomorphological conditions (MMA 2016, Barral *et al.* 2023, Montes *et al.* 2023).

In the ERBR region, some forest islands are protected by conservation units (Coelho *et al.* 2018b). However, no studies to date have focused on comparing the provision of ecosystem services by vegetation in areas with different land use and conservation regimes in the region. Studies conducted in other phytophysiognomies systems have highlighted the importance of protected areas in preserving biodiversity and increasing carbon stocks, compared with disturbed areas (Françoso *et al.* 2015, 2016, Barral *et al.* 2023, Carvalho *et al.* 2024). Thus, it is essential to investigate the potential effectiveness of forest-island vegetation in providing ecosystem services, with the aim to promoting management and conservation practices that enhance their environmental sustainability in scenarios of climate change and different land uses (Coelho *et al.* 2018a).

In general, preserved forest islands exhibit high beta diversity (Coelho *et al*. 2017, Moura *et al.* 2021) and harbor environmentally sensitive species (*e.g*., *Richeria grandis* Vahl, *Tapirira guianensis* Aubl.) associated with higher soil moisture and preserved conditions (Coelho *et al*. 2016, 2018b, Pereira *et al.* 2017, Costa *et al*. 2023, Moura *et al.* 2025). On the other hand, forest islands exposed to frequent anthropogenic disturbances, particularly fires and grazing, show an increase in pioneer or disturbance-tolerant species (*e.g*., *Myrsine umbellata* Mart., *Myrcia splendens* (Sw.) DC), and a reduction in the vertical structure of the forest (Coelho *et al*. 2018a, Costa *et al*. 2022). Therefore, these changes lead to the replacement of plant species composition and a reduction in aboveground biomass and carbon stocks (Chave *et al*. 2014, Scolforo *et al.* 2016, Souza *et al*. 2022, Sullivan *et al*. 2025).

We evaluated the floristic composition, vegetation structure, species diversity and carbon stock in two communities in the Espinhaço Range Biosphere Reserve, Brazil, subject to different land use regimes, one inside a Conservation Unit and the other in an unprotected area. Our objective, within a case study framework, was to address the following questions: (i) How do the species composition, vegetation structure, and diversity differ between protected and unprotected forest islands subject to anthropogenic disturbances? And are there differences in the carbon stocks of these forest islands? We hypothesized that the floristic composition would differ between the two areas. On the protected forest island (PAI), species with greater moisture and soil conservation requirements are expected to be present, while in the unprotected area (UPA), with disturbances such as grazing and fire, pioneer and disturbance-tolerant species will be more frequent. In terms of species diversity and vegetation structure, we believe that the PAI will have a higher tree density and more diverse biomass, contributing to higher carbon stocks, while in the UPA, we expect homogenization of the vegetation, a reduction in biomass, and lower carbon stocks.

Materials and methods

Study areas. We conducted this study in two forest-island areas with different use and conservation regimes, each approximately 1,200 m² (0.12 ha), located in the Espinhaço Range Biosphere Reserve (ERBR) in Minas Gerais, Brazil (Figure 1). The PAI is within the Rio Preto State Park (RPSP) (18° 14’ 13” S, 43°19’ 37” W), where no significant disturbances have been recorded in the past 50 years. The UPA is located within the RPSP buffer zone (18° 14’ 29” S, 43° 18’ 37” W) and is subject to frequent disturbances, including cattle presence and forest fires (Coelho *et al*. 2018b, Ferreira *et al*. 2022, Mucida *et al*. 2022). Both areas are located at an average of 1,600 m asl.

The vegetation of these forest islands is classified as Altimontane Seasonal Semideciduous Forest, and, phytogeographically, they are part of an enclave of Atlantic Forest disjunctions within the Cerrado biome (Ab’Saber1967, Coelho *et al*. 2016, Gonçalves *et al*. 2020). The region’s climate is subtropical Altimontane (Cwb) (Köppen), with rainy summers and dry winters, with average temperatures ranging from 10 to 19 °C and annual rainfall from 850 to 1,500 mm (INMET 2023, Pinto *et al*. 2023, Barral *et al*. 2023). The region is geologically characterized by quartzite rocks and shallow soils (Bispo *et al*. 2015, Coelho *et al*. 2016). The forest islands occur predominantly on organic Histosols and hydromorphic soils in areas associated with the fragility of the quartzite and are always near water springs (Coelho *et al*. 2018a). In the two areas studied, the soils are classified as Histosols according to the USDA Soil Taxonomy (Soil Survey Staff 2010), specifically as the Typic Haplosaprists (Bispo *et al*. 2015), while in the World Reference Base for Soil Resources (WRB) they correspond to Ombric Sapric Histosols (IUSS Working Group WRB 2022).

Vegetation sampling. In both, the PAI by RPSP, and the adjacent UPA, ten 10 × 10 m plots (100 m² each) were installed sequentially one after the other, totaling a sampled area of 0.1 ha per island, following the guidelines of the Cerrado Permanent Plots Manual (Felfili & Rezende 2005). In the plots, all living trees with a diameter at breast height (DBH) ≥ 5 cm were measured. Trees with subdivided stems were included in the survey when the total basal area of one of the stems reached DBH ≥ 5 cm. The total height of the trees was estimated using a graduated botanical collecting pole (Felfili & Rezende 2005).

When possible, botanical identification was done *in situ*. Otherwise, botanical material was collected for later identification by comparison in herbaria, consultation with specialists and specialized literature. The botanical samples were deposited in the Jeanini Felfilli Dendrological Herbarium (HDJF) at Universidade Federal dos Vales do Jequitinhonha e Mucuri (UFVJM), Minas Gerais, Brazil.

We classified the species according to the *Angiosperm Phylogeny Group* (APG 2016) and updated taxonomic names in accordance to Flora e Funga do Brasil - Reflora (Flora e Funga do Brasil 2024). We consulted the National Center for the Conservation of Flora - CNCFlora (CNCFlora 2024) to assess the species’ threat categories. In this study, we adopted the threat categories defined by the World Conservation Union (IUCN 2024).

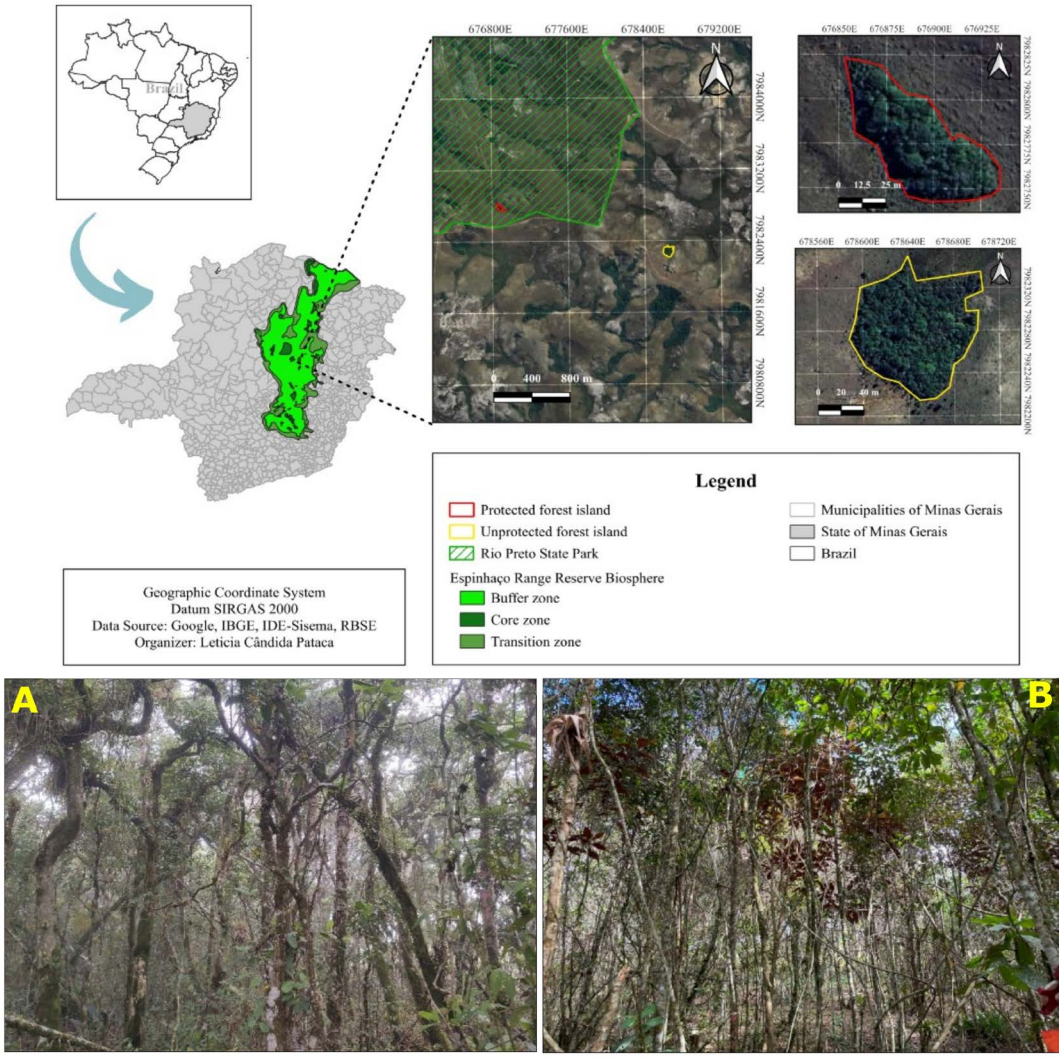

Figure 1. Geographical location of the forest islands (Capões de Matas) in the Rio Preto State Park-RPSP. (A) protected area and (B) adjacent unprotected area in the Espinhaço Range Biosphere Reserve-ERBR, Minas Gerais, Brazil.

Species diversity and vegetation structure. Taxonomic alpha diversity (α) was calculated by the effective number of species (Jost 2006). Three orders of taxonomic diversity were calculated: ${}^{0}q$, which equals to species richness; ${}^{1}q$, which weighs the number of species by their abundance in the community, and ${}^{2}q$, which gives greater weight to the abundance of the dominant species in the community. Alpha diversity (α) was calculated using the "hillR" package (Li 2018) in the R software (R Core Team 2024, version 4.3.3).

Beta diversity (β) was analyzed by calculating the Sørensen and Bray-Curtis dissimilarity indices (Magurran 2004). In addition, we examined the relationship between dissimilarity and species turnover in the plots, and how the latter relates to the shared richness of species and sites (nestedness; Baselga *et al*. 2012a, 2016). To analyze β diversity, the "betapart" package (Baselga *et al*. 2012b, Baselga 2016) was used in R. Vegetation structure was assessed using phytosociological parameters (Mueller-Dombois & Ellenberg 1976) and based on the distribution of trees in diameter and height classes. We adopted a fixed amplitude of 5 cm between diameter classes and of 3 cm for height classes.

Estimation of Above Ground Biomass (AGB) and Above Ground Carbon (AGC). To calculate AGB, we used the Pantropical allometric equation proposed by Chave *et al*. (2014): $AGB_{est} = 0.0673 \times (\rho D^2 H)^{0.976}$, est = ρ(σ = 0.357, AIC = 3130, df = 4002), where: ρ = wood density of each species (g/cm³), D = diameter at breast height (cm) and H = height (m). The wood density data for the species were obtained according to Oliveira Filho & Scolforo (2008) and ranged from 0.409 to 0.82 g/cm³. To obtain the AGC stored in the two communities (in hectares), the AGC values for each tree were calculated and then extrapolated to the entire community in Mg C ha^{-1} and multiplied by 0.47 (47 %) (CETEC 1995, IPCC 2014, Liu *et al*. 2023).

Data analysis. To compare patterns of species diversity between the two forest islands under contrasting conservation regimes, we applied generalized linear models (GLM) (Da Silva *et al.* 2022). We modeled abundance, *α* diversity (^0q, ^1q and ^2q), Sørensen turnover, dissimilarity, and nesting as response variables and the effect of areas as the explanatory variable. Abundance and richness (^0q) were modeled with Poisson, quasi-Poisson, or negative binomial distributions, depending on model fit. For diversity indices ^1q and ^2q, we tested Gaussian and Gamma distributions with a log link and used beta regression models for dissimilarity (turnover and nestedness) and *β* diversity (Sørensen and Bray-Curtis), using the 'betareg' package (Cribari-Neto & Zeileis 2010). In all analyses, the residuals were evaluated to determine the best distribution using the "performance" package (Lüdecke *et al*. 2021) in R.

We checked the distribution of vegetation structure (diameter and height classes) using the Shapiro-Wilk (W) and Kolmogorov-Smirnov (*D*) tests. To test differences in floristic composition between the areas, we applied a Permutational Multivariate Analysis of Variance (PERMANOVA) with Bray-Curtis dissimilarity and 1,000 permutations (Clarke 1993, Anderson 2001, 2006).

Results

Floristic composition and species diversity. We recorded 102 species belonging to 38 families and 36 genera, of which 47 species (20 families) occurred in the PAI and 55 species (18 families) in the UPA. The families with the highest richness and density in the PAI were Myrtaceae (10 species and 95 trees), Phyllanthaceae (1 and 31), Lauraceae (6 and 30) and Araliaceae (2 and 23). In total, these families accounted for 40.43 % of the species and 59.45 % of the trees recorded (Table S1). In the UPA, the most representative botanical families were Primulaceae (1 species and 60 trees), Myrtaceae (10 and 46), Melastomataceae (11 and 36), Lauraceae (10 and 35). Together, they accounted for 74.54 % of the trees and 58.18 % of the species in this area (Table S2).

In the evaluation of *α* diversity, no significant differences were observed between the areas for species richness, (^0q) ($\chi^2 = 0.034$, d.f. = 19, $P = 0.855$), nor alpha diversity: ^1q ($F = 0.522$, d.f. = 19, $P = 0.479$) and ^2q ($F = 0.79$, d.f. = 19, $P = 0.385$) (Figure 2).

However, species composition differed between the compared areas ($F = 7.251$, $R^2 = 0.287$, $P < 0.001$), and 28.7 % of this variation may be attributable to the use and conservation regimes of the forest islands. These data are corroborated by the high dissimilarity between both type of areas, as indicated by the Sørensen and Bray-Curtis indices (92.6 and 93.6 %, respectively), with turnover values exceeding 90 % and nestedness at just 1.0 % for both indices. We found significant difference in Sørensen's dissimilarity between the areas ($z = 2.47$, $P = 0.013$) and the Bray-Curtis *β* diversity value ($z = 2.08$, $P = 0.037$), with higher values in the UPA (Figure S1).

Ten species were shared between the PAI and UPA: *T. guianensis* Aubl., *Rhodostemonodaphne anomala* (Mez) Rohwer, *Ocotea pulchella* (Nees & Mart.) Mez, *Miconia flammea* Casar, *M. trianae* Cogn., *M. umbellata* Mart., *Myrcia venulosa* DC., *Styrax ferrugineus* Nees & Mart., *Cabralea canjerana* (Vell.) Mart., and *Guapira opposita* (Vell.) Reitz. Species abundance was directly affected by the conservation status of the area ($\chi^2 = 21.10$; d.f. = 19; $P = 0.014$), being higher in the PAI, suggesting that it accumulates a greater number of species. In contrast, the UPA showed lower abundance and less variability in the community (Figure S2).

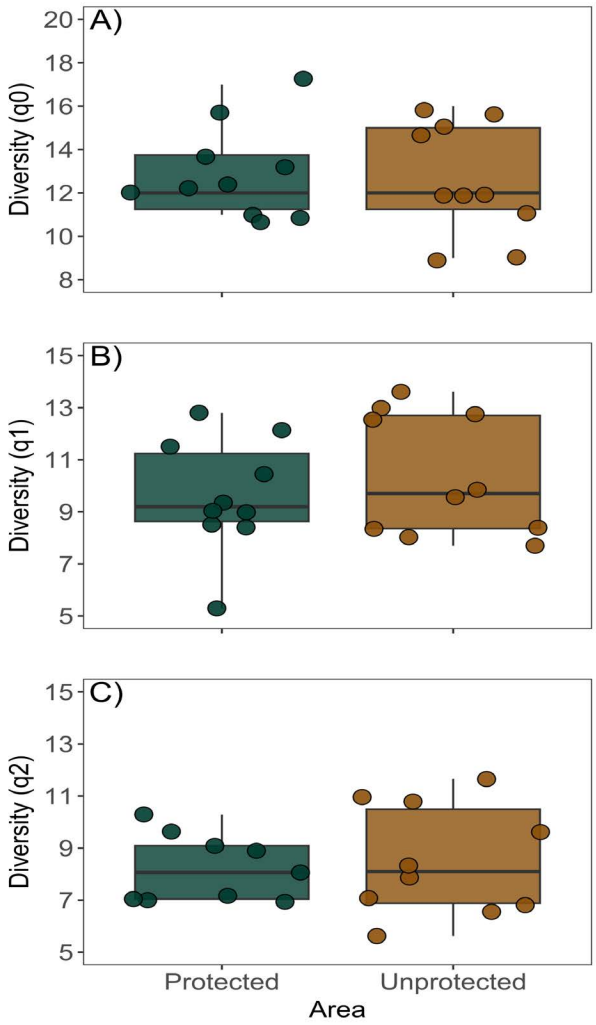

Figure 2. Alpha diversity values for tree vegetation (DBH ≥ 5 cm) sampled on protected and unprotected forest islands in the Espinhaço Range Biosphere Reserve, Minas Gerais, Brazil. A) q^0 = richness, B) q^1 = all species are weighted proportionally by their abundance in the community and C) q^2 = abundance of the dominant species.

Vegetation structure. We sampled a total of 545 trees, of which 301 (55.23 %) and 244 (44.77 %) were recorded in PAI and UPA, respectively. Regarding the total basal area of 93.4 m² ha^{-1}, there was a significant difference between the areas (D = 0.342, P < 0.001). The highest values were observed in the PAI, with an average basal area per tree of 0.022 ± 0.003 m², representing 70.3 % of the total value (65.7 ± 1.6 m²·ha^{-1}). In contrast, the UPA had a total basal area of 27.7 ± 1.8 m²·ha^{-1} (29.64 %), equivalent to an average basal area per tree of 0.011 ± 0.001 m². In the PAI, the species that contributed the most biomass were *R. grandis*, accounting for 28.3 % of the community's basal area (18.6 m² ha^{-1}), Myrtaceae sp., with 14.8 % (9.74 m² ha^{-1}), and an undetermined species, with 7.1 % (4.68 m² ha^{-1}) (Table S1). In the UPA, the species *M. umbellata* contributed 20.03 % (5.55 m² ha^{-1}) of the total average basal area of the community, followed by *C. canjerana* with 7.87 % (2.18 m² ha^{-1}) and *Cryptocarya aschersoniana* Mez with 6.82 % (1.89 m² ha^{-1}) (Table S2).

We found statistical differences in the distributions of height classes between both compared areas (D = 0.141 and P = 0.0089). In the PAI, the tree stratum ranged in height from 2 to 18 m, and in the UPA from 3 m to 27 (Figure 3A, B). However, in both areas we found that trees were concentrated in the 6-12 meter classes (6-9 m and 9-12 m), which represents 64.2 and 79.5 % trees in the PAI and UPA, respectively. In the last type of protected

area, only 5.3 % of the trees reached heights of more than 15 m. We detected a similar pattern in the diameter distributions, which differed between the PAI and UPA ($D = 0.28$ and $P < 0.001$). In the former, most trees concentrate between 5 and 20 cm (Figure 3C), whereas in the latter, 68.8 % of the trees had DBHs between 5 and 10 cm (Figure 3D).

Above-ground biomass (AGB) and above-ground carbon (AGC). The AGB and AGC values estimated for both areas combined were 471.85 Mg ha^{-1} and 221.77 Mg C ha^{-1}, respectively. In all metrics, the PAI had higher average values (AGB = 300.14 ± 25.3 Mg ha^{-1} and AGC = 141.06 ± 11.9 Mg C ha^{-1} (63.6 %) than the UPA (AGB = 171.70 ± 31.7 Mg ha^{-1} and AGC = 80.70 ± 14.9 Mg C ha^{-1} (36.4 %). We found significant differences between the areas for both AGB ($W = 16$, $P = 0.011$) and AGC ($W = 84$, $P = 0.0089$). The carbon stock in the PA was 60.4 Mg C ha^{-1} higher than in the UPA (Figure 4). The species that contributed the most AGB and AGC were Myrtaceae sp. and *R. grandis* in the PAI, and *M. umbellata*, *Byrsonima japurensis* A.Juss., and *B. chrysophylla* Kunth., in the UPA (Table 1).

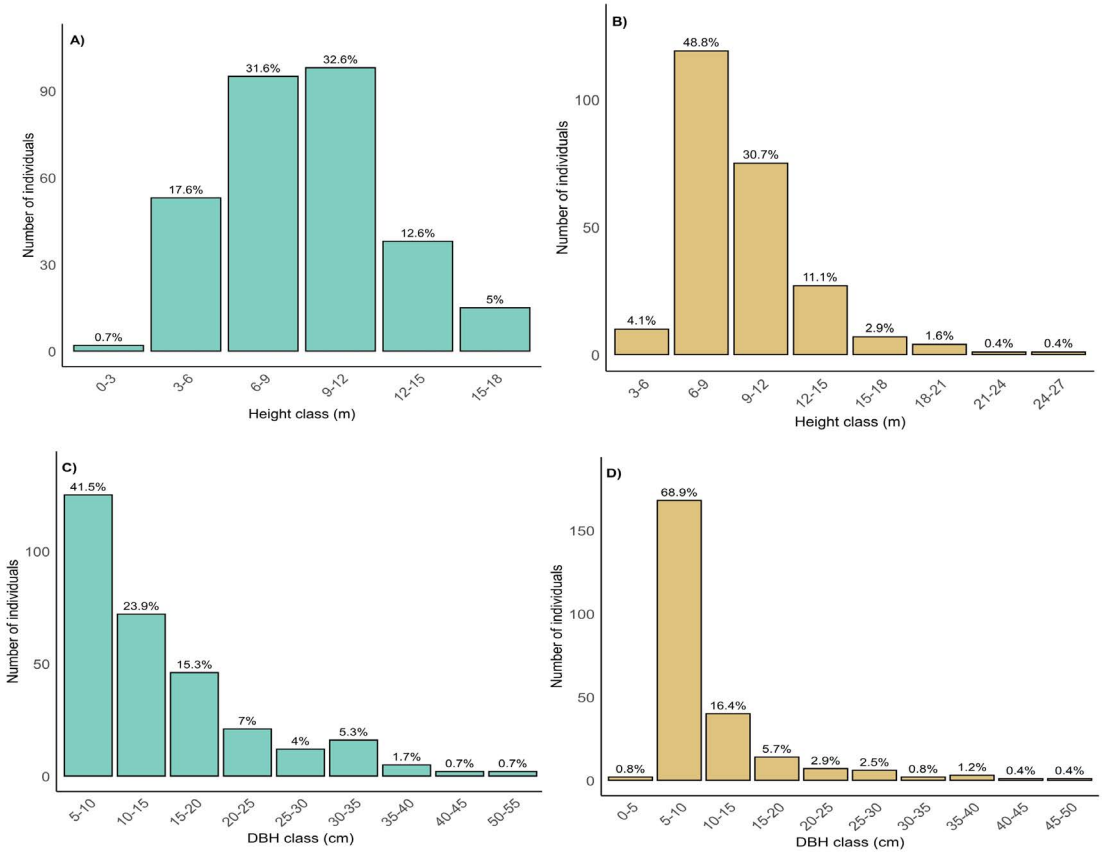

Figure 3. Distribution of trees (DBH ≥ 5 cm) in height and diameter classes between the protected (A and C) and unprotected (B and D) forest islands in the Espinhaço Range Biosphere Reserve, Minas Gerais, Brazil 2024.The values above the bars represent the percentage of each class.

Discussion

Floristic composition, α and β diversity, and vegetation structure. Our findings describe the complexity and uniqueness of the Altimontane Tropical Forest Islands, located in the Atlantic Forest-Cerrado transition, showing clear patterns of variation in floristic and structural characteristics, as well as in the provision of ecosystem services (carbon

stock), which differ in the conservation context. Based on the floristic composition, the altitudinal forest islands are essential elements of the Atlantic Forest-Cerrado biogeographic transition zone in the ERBR region. This is due to the high representation of families typical of elevational formations and forests of the Atlantic Forest biome (such as Myrtaceae, Lauraceae, and Primulaceae; see Tables S1 and S2; Magalhães *et al.* 2018, Costa *et al.* 2021a, 2023). This, in turn, suggests that elevation gradient and the associated abiotic conditions shape distinct communities (*e.g.*, *M. umbellata*, *C. canjerana*), which, in addition to sharing affinities with other biomes and/or specific environments, also harbor endemic plant species (Coelho *et al.* 2016, 2018b). These results reinforce the role of Brazilian Protected Areas in maintaining biodiverse communities (Françoso *et al.* 2015, Souza *et al.* 2020, Barral *et al.* 2023).

Environmental conditions imposed by high elevation, such as higher humidity, hydromorphic and dystrophic soils (Myers *et al.* 2000, Coelho *et al.* 2016, Costa *et al.* 2021a, Moura *et al.* 2021, Costa *et al.* 2022), may also explain the dominance of species such as *R. grandis*, Myrtaceae sp., and *T. guianensis* in the protected area. The increased presence of species such as *M. umbellata*, *C. aschersoniana*, and *Myrcia splendens* in the UPA seem to be linked to the ecological characteristics that enable them to colonize areas with more stressful environmental conditions. Species in Primulaceae, such as *M. umbellata* and *M. guianensis*, have been reported to show a range of adaptations to grazing and fire. Although described as fire-sensitive, these plants are resilient and are often abundant in fire-disturbed areas (Rios *et al*. 2018, Antunes *et al.* 2025). These adaptations include tolerance to soil compaction, chemical changes linked to ammonification from deposition of cattle manure and urine, tolerance to variations in soil moisture, and resistance to fire (Bauer *et al.* 2012, Amorim *et al.* 2019, Salomão *et al.* 2023).

Our results also indicate that, although the areas are subject to distinct levels of conservation, they exhibit a relatively similar pattern across the three orders of taxonomic diversity assessed. This is because a balanced abundance distribution, with few significant variations among species, yields low diversity. This, in turn, reduces the sensitivity of the diversity metrics (^1q -proportionally weighted by their abundance in the community, and ^2q - abundance of the dominant species in the community) to differences in abundance, bringing these indices closer to species richness

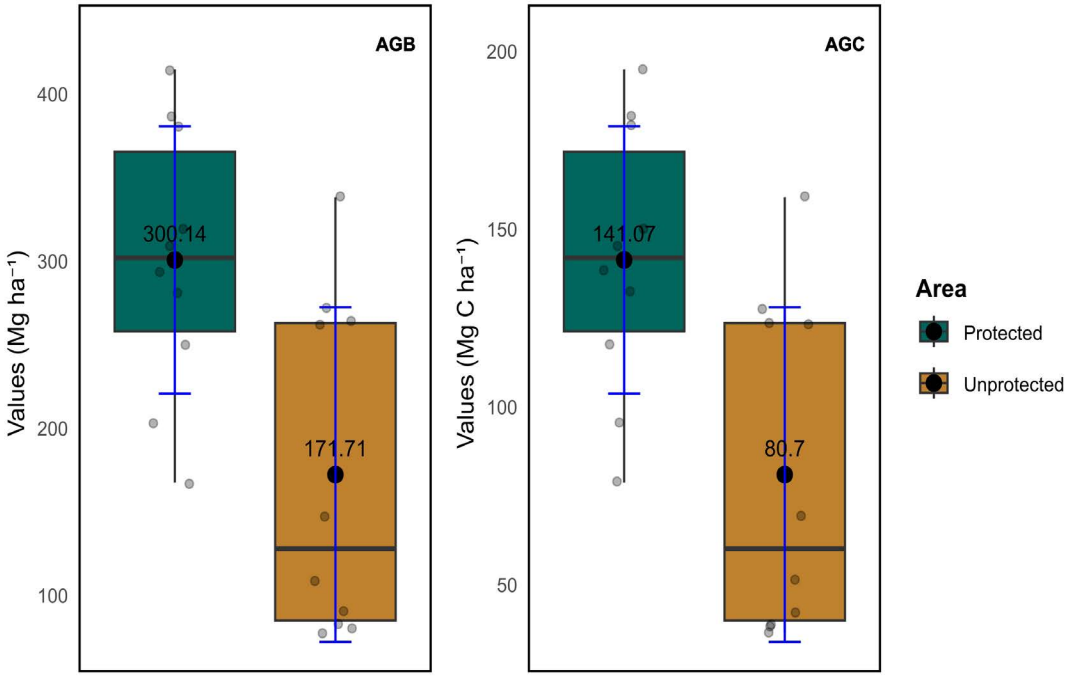

Figure 4. Aboveground biomass (AGB), aboveground carbon stock (AGC) and carbon dioxide equivalent (CO_2-eq) in unprotected and protected forest islands sampled in the Espinhaço Range Biosphere Reserve, Minas Gerais, Brazil.

(^0q). The richness of species and genera, combined with the balanced distribution of abundances, contributes to the homogeneity of the diversity indices, reflecting similar taxonomic patterns between the analyzed areas. The high level of species turnover recorded in the areas was considerably higher on the UPA (90 %) than on the PAI (78 %). This suggests that species composition is strongly influenced by the intensity of anthropogenic changes to abiotic conditions in the UPA. These altered conditions have, in turn, facilitated the colonization by generalist species adapted to disturbed environments (Donoghue 2008, Chao *et al.* 2014, Coelho *et al.* 2016, 2018b).

The environmental degradation observed on the unprotected island would also be the primary factor responsible for the high dissimilarity values (Sørensen: 0.92 and Bray-Curtis: 0.93) recorded in the areas. This is because such conditions influence the composition between areas, which was mainly attributed to species substitution, as observed by Fernandes *et al.* (2020). This shows that each area supports different floristic assemblages, depending on its state of land conservation and use (Baselga *et al.* 2012b, Bispo *et al*. 2016, Alvarado *et al*. 2017, Coelho *et al*. 2018a, Souza *et al.* 2020, Moura *et al*. 2021, Barral *et al*. 2023, Guerrini *et al.* 2024).

Thus, the influence of conservation status on the beta diversity of each forest island reinforces the idea that unprotected areas are subject to greater selective pressure (Coelho *et al.* 2018b, Costa *et al*. 2022, Bañares-de-Dios *et al*. 2024). This would explain, for example, the presence of pioneer species like *Eremanthus erythropappus* (DC.) MacLeish on the unprotected island. This species is widely distributed in Campos Rupestres (rupestrian grasslands), which feature shallow, quartzite soils with low water retention and fertility (Coelho *et al*. 2016, Pereira *et al*. 2017).

The observed species substitution pattern is consistent with environmental filtering (MacArthur & Wilson 1963, Rosindell *et al.* 2011). The abiotic conditions, such as reduced soil humidity, increased compaction, and greater luminosity, resulting from the presence of cattle and fire, led to a unique floristic composition, featuring species from the forest environment as well as plants from other environments, according to the microclimate variations existing on the UPA. In contrast, PAI maintain more stable environmental conditions, allowing species distributed in the area not to be replaced too intensively, as their environments are more similar (Coelho *et al*. 2016, 2018a, Pereira *et al.* 2017, Gonçalves *et al*. 2023).

This same aspect supports the low contribution of nestedness among the islands, suggesting that the communities are not subsets of each other, but are structured independently under different abiotic conditions, formed by species adapted to the environmental conditions present on each forest island. After all, there is a strong connection between the abundance of species and acidic, aluminum-rich, and less fertile soils on forest islands (Coelho *et al.* 2016, 2018b). Thus, we believe that these characteristics influence the floristic composition and structure of the vegetation, resulting in a low basal area and monodominance of some species, such as those found in the UPA: *M. umbellata*, *C. aschersoniana*, *M. splendens*, and *Miconia* sp.

Biomass and carbon stock. The above-ground biomass (AGB) and above-ground carbon (AGC) values obtained are higher than those reported by Carvalho *et al*. (2024), who recorded 10.47 Mg ha^{-1} of AGB and 5.23 Mg C ha^{-1} of AGC for another forest also located in a Conservation Unit. The higher AGB values in the PAI are attributed to the presence of trees with larger basal area, average total height, canopy density, and higher wood density (Coelho *et*

Table 1. Species with the highest contributions to AGB and AGC in protected and unprotected forest islands sampled in the Espinhaço Range Biosphere Reserve, Minas Gerais, Brazil.

Area	Species	Total AGB (Mg ha^{-1})	Total AGC (Mg C ha^{-1})
Protected	*Myrtaceae* sp.	59.69	28.04
Protected	*Richeria grandis*	55.35	26.57
Unprotected	*Myrsine umbellata*	43.07	20.24
Unprotected	*Byrsonima japurensis*	19.50	7.16
Unprotected	*Byrsonima chrysophylla*	15.17	7.03

al. 2018a, Lutz *et al*. 2018, Ali *et al*. 2020, Vargas-Larreta *et al.* 2021, Abbasi *et al*. 2022). These factors are directly linked to greater capacity to store biomass and carbon (Coelho *et al*. 2017, Lutz *et al*. 2018, Ali *et al.* 2020). This result highlights the importance of conserving forest islands, such as the one in the RPSP, not only for flora and biodiversity, but also for maintaining ecosystem services. However, further studies including a larger set of forest islands are needed to confirm these patterns.

In contrast, in the UPA, changes in floristic composition, species diversity, and vegetation structure, resulting from anthropogenic disturbances, reduce its carbon stock. Our results also indicate that anthropogenic disturbances, such as cattle presence and fire, on forest islands lead to homogenization of vegetation structure, resulting in smaller DBHs and greater dominance by some species. As a result, there is a reduction in biodiversity, tree biomass and carbon stocks (Souza *et al*. 2022, Barral *et al*. 2023, Kurtz *et al.* 2024, Antunes *et al*. 2025).

Recent research indicates that AGC accumulation is strongly influenced by species richness and composition (Bryant *et al*. 2024). These findings highlight the importance of species diversity in maximizing carbon stocks (Morandi *et al*. 2020, Pichler *et al*. 2022, Carvalho *et al*. 2024), as corroborated by our results. In our results, we found some carbon-hyperdominant species in both areas (*e.g*., Myrtaceae sp., *R. grandis*, *M. umbellata*, *B. japurensis*, and *B. chrysophylla*, among others) (Table 1). Other studies in Atlantic montane forests have also found species of Lauraceae, Melastomataceae, Myrtaceae and, Primulaceae to be carbon hyperdominant (Antunes *et al.* 2025, Souza *et al.* 2023). In other words, biodiverse ecosystems, such as the PAI, have a greater capacity to sequester carbon than ecosystems with ecologically dominant species that are shorter, have smaller diameters, and have lower wood densities, as was recorded on the UPA.

The AGC values recorded for the PAI are within the documented range for intermediate-elevation Atlantic mountain formations (1,600-1,700 m), such as 120.36 and 170.4 Mg C ha^{-1} (Souza *et al*. 2023), and also those reported by Antunes *et al*. (2025) of 54-199 Mg C ha^{-1}, and Altomare *et al*. (2025) with a maximum value of 165.48 Mg C ha-1. In contrast, the lowest stock observed in the UPA is close to the lowest values recorded in these studies, which are generally associated to low elevations, environments subject to anthropogenic disturbances, and soils with low phosphorus concentrations. Although we did not include the effects of other environmental variables, other studies have reported a direct relationship between elevation and soil fertility, as well as an inverse relationship between fire occurrence and AGC stocks (Antonelli *et al.* 2018, Souza *et al*. 2021, 2022, Griffiths *et al.* 2021, Antunes *et al*. 2025). These results reinforce the importance of Conservation Units (CUs) for protecting high-elevation forest islands, which are unique and fundamental environments for maintaining ecosystem services. Furthermore, they confirm our hypothesis that disturbed forests have a reduced capacity to sequester carbon, in agreement with previous studies (Gallo *et al.* 2009, Avigliano *et al.* 2019, Bryant *et al.* 2024).

This is the first study to describe the floristic composition and quantify vegetation structure, species diversity, above-ground biomass, and above-ground carbon in Altimontane forest islands of the Espinhaço Range in Brazil. This makes it clear that ERBR forest islands are a priority for future sustainability and conservation efforts (Barros *et al.* 2023). Finally, Protected Areas are especially important for the conservation and protection of tropical Altimontane environments, which remain largely neglected. These include forest islands, which are essential for providing ecosystem services, mitigating the global effects of climate change, and ensuring the quality of life and well-being of the human population. As this study was based on two forest islands, our results should be interpreted within the scope of a case study. Nevertheless, they provide valuable baseline information on the floristic composition, diversity, vegetation structure, and carbon stocks of Altimontane Forest Islands in the Espinhaço Range. These findings highlight the importance of expanding future research to include a larger number of forest islands across different conservation contexts and environmental gradients, to test the generality of the patterns observed here. Although we did not find differences in species diversity, our study demonstrated that the degradation of these ecosystems promotes the replacement of many native species of Altimontane Forest Islands. This result, therefore, highlights the need to protect these areas to conserve biodiversity and maintain the integrity of this unique and fragile ecosystem.

Supplementary material

Supplemental data for this article can be accessed here: https://doi.org/10.17129/botsci.3818

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