UNIVERSIDADE FEDERAL DE ALFENAS

BIANCA DINIS DA SILVA

FIRE DISTURBANCE AND HABITAT LOSS SHAPE DIETARY TRAIT SHIFTS IN TROPICAL FOREST BIRD COMMUNITIES THROUGH ENVIRONMENTAL FILTERING AND LIMITING SIMILARITY

ALFENAS/MG
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Dissertação apresentada como parte dos requisitos para obtenção do título de Mestre em Ciências Ambientais pela Universidade Federal de Alfenas. Área de concentração: Ciências Ambientais.

Orientadora: Érica Hasui.

ALFENAS/MG
2025

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Silva, Bianca Dinis da.

Fire disturbance and habitat loss shape dietary trait shifts in tropical forest bird communities through environmental filtering and limiting similarity / Bianca Dinis da Silva. - Alfenas, MG, 2025.

43 f. : il. -

Orientador(a): Érica Hasui.

Dissertação (Mestrado em Ciências Ambientais) - Universidade Federal de Alfenas, Alfenas, MG, 2025.

Bibliografia.

1. Atlantic Forest. 2. Avian. 3. Landscape. 4. Fire Ecology. 5. Trait-based responses. I. Hasui, Érica, orient. II. Título.

Ficha gerada automaticamente com dados fornecidos pelo autor.

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“Fire disturbance and habitat loss shape dietary trait shifts in tropical forest bird communities through environmental filtering and limiting similarity”

A Banca examinadora abaixo-assinada aprova a Dissertação apresentada como parte dos requisitos para a obtenção do título de Mestre em Ciências Ambientais pela Universidade Federal de Alfenas. Área de concentração: Ciências Ambientais.

Aprovada em: 26 de fevereiro de 2025.

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AGRADECIMENTOS

À minha família, minha base e meu alicerce, por sempre me mostrarem que a educação salva, transforma e é o caminho mais potente para a liberdade. Agradeço profundamente por todo amor, apoio e incentivo incondicional ao longo dessa caminhada. Cada palavra de carinho e cada gesto de cuidado fizeram toda a diferença.

Aos meus amigos mais íntimos — vocês sabem quem são —, obrigada por estarem ao meu lado nos momentos de angústia e nas pequenas conquistas do dia a dia. Suas palavras de apoio, escuta atenta e presença constante foram essenciais para que eu seguisse em frente com leveza e coragem.

À minha orientadora, Prof.ª Dr.ª Érica Hasui, expresso minha mais profunda gratidão pela orientação dedicada, pelos ensinamentos generosos, pela paciência constante e pelas contribuições valiosas que foram essenciais para a construção deste trabalho. Seu olhar atento, crítico e ao mesmo tempo sensível foi fundamental não apenas para o desenvolvimento desta pesquisa, mas também para o meu crescimento acadêmico e pessoal. Espero, sinceramente, um dia refletir, mesmo que em parte, a profissional inspiradora e admirável que você é.

Aos colegas de pesquisa, agradeço pelas discussões instigantes, pelas trocas de ideias, pela ajuda no campo e pelas reflexões conjuntas que enriqueceram não apenas esta dissertação, mas também minha formação como pesquisadora.

Por fim, mas não menos importante, agradeço profundamente à educação pública brasileira, que me acolheu aos quatro anos de idade e, desde então, tem sido alicerce fundamental em minha trajetória e em cada conquista. Sou igualmente grata às políticas afirmativas de cotas para estudantes de escolas públicas, que abriram caminhos mais justos para o ingresso na universidade. Às bolsas de Iniciação Científica do CNPq e à bolsa de mestrado da CAPES, meu sincero reconhecimento — sem esses investimentos, eu não teria chegado até aqui. Sou um pequeno fruto de um projeto educacional comprometido com transformação e equidade, e esta dissertação representa um singelo retorno à sociedade por todo o investimento depositado em mim.

O presente trabalho foi realizado com apoio da Coordenação de Aperfeiçoamento de Pessoal de Nível Superior - Brasil (CAPES) - Código de Financiamento 001.

RESUMO

Incêndios florestais e a perda de habitat são fatores-chave nas mudanças de biodiversidade em florestas tropicais, mas seus efeitos combinados sobre características funcionais ainda são pouco compreendidos. Neste estudo, investigamos como a severidade do fogo, a extensão queimada e a cobertura florestal afetam conjuntamente as características alimentares de aves em 15 paisagens da Mata Atlântica, no Brasil, cada uma contendo áreas pareadas de floresta queimada e não queimada (total de 30 unidades amostrais). As aves foram amostradas por meio de contagens por ponto padronizadas, e a composição funcional foi avaliada com base em dados contínuos de dieta, utilizando médias e variâncias ponderadas pela comunidade (CWM e CWV). As métricas foram calculadas com uma abordagem de reamostragem não paramétrica, e os efeitos das variáveis ambientais foram testados por meio de modelos lineares generalizados mistos (GLMMs). Nossos resultados mostram que tanto a média quanto a variabilidade das características alimentares foram afetadas pela perturbação causada pelo fogo, com respostas fortemente moduladas pela cobertura florestal. Observamos efeitos sinérgicos e não aditivos, especialmente nas interações entre severidade/extensão do fogo e cobertura de habitat. Sob alta severidade do fogo e baixa cobertura florestal, ocorreu convergência de características, com menor variabilidade no consumo de frutos, néctar e invertebrados, sugerindo forte filtragem ambiental. Essas condições favorecem espécies generalistas e reduzem a presença de especialistas associados a micro-habitats estruturais complexos. Por outro lado, o consumo de sementes apresentou padrões opostos: tanto a média quanto a variância aumentaram em florestas queimadas, indicando maior diversidade de estratégias alimentares e possível atuação de processos como similaridade limitante. Esse padrão pode estar associado ao aumento na disponibilidade de sementes após o fogo, devido à regeneração de espécies pioneiras e herbáceas. Em paisagens com maior cobertura florestal, também foi observado aumento na variabilidade do consumo de invertebrados sob extensões moderadas de fogo, indicando que ambientes mais heterogêneos podem favorecer diferentes tipos de forrageamento. Em conjunto, nossos resultados indicam que processos de filtragem ambiental e divergência funcional não são mutuamente exclusivos, mas coexistem ao longo de gradientes de perturbação, dependendo do traço funcional analisado e do contexto da paisagem. Este estudo reforça o valor de abordagens baseadas em características para detectar respostas ecológicas além da simples contagem de espécies, revelando mecanismos mais profundos de montagem de comunidades. Destacamos, ainda, a importância de integrar estratégias de manejo do fogo com a conservação da cobertura florestal para promover a resiliência funcional das comunidades em florestas tropicais propensas a incêndios.

Palavras-chave: Mata Atlântica; aves; paisagem; ecologia do fogo; respostas baseadas em características.

ABSTRACT

Fire and habitat loss are major drivers of biodiversity change in tropical forests, yet their combined effects on functional traits remain poorly understood. We investigated how fire severity, fire extent, and forest cover jointly influence bird dietary traits across 15 Atlantic Forest landscapes in Brazil, each containing paired burned and unburned forest sites (30 sampling units total). Birds were surveyed using standardized point count methods, and trait composition was evaluated through community-weighted means (CWM) and variances (CWV) of continuous dietary data, calculated with a nonparametric bootstrapping approach. We used generalized linear mixed models (GLMMs) to test the effects of fire and forest cover on trait metrics. Both CWM and CWV were influenced by fire disturbance, with responses strongly modulated by forest cover, revealing synergistic, non-additive effects. Trait convergence—such as reduced variability in fruit, nectar, and invertebrate consumption—was observed under high fire disturbance and low forest cover, suggesting strong environmental filtering. In contrast, increased trait variance in seed consumption in burned forests pointed to limiting similarity under enhanced post-fire resource diversity. These findings show that filtering and divergence are not mutually exclusive but coexist along disturbance gradients, depending on trait and landscape context. Our results highlight the value of trait-based approaches to detect functional responses beyond species richness and reinforce the need to integrate fire management with habitat conservation to maintain ecological resilience in fire-prone tropical forests.

Keywords: Atlantic forest; avian; landscape; fire ecology; trait-based responses

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1 INTRODUÇÃO GERAL

Nas últimas décadas, os incêndios florestais têm aumentado em regiões tropicais, como a Mata Atlântica, impulsionados principalmente por mudanças no uso da terra e mudanças climáticas (Pausas & Keeley, 2021; Williams *et al.*, 2019). Além disso, a Mata Atlântica é um bioma que já sofre há séculos com o desmatamento e fragmentação de habitat (Rezende *et al.*, 2018). Embora os impactos isolados do fogo ou do desmatamento sobre a fauna sejam relativamente conhecidos, os efeitos combinados desses distúrbios sobre a estrutura funcional das comunidades biológicas permanecem pouco explorados (Driscoll *et al.*, 2021). Considerando que o fogo pode atuar como um filtro ambiental seletivo e que a cobertura florestal influencia a disponibilidade e diversidade de recursos, compreender suas interações é essencial para prever respostas ecológicas em cenários de mudança ambiental. A relevância do estudo está ancorada na escassez de abordagens integradas que considerem simultaneamente os efeitos do fogo e da cobertura florestal sobre a estrutura funcional das comunidades, especialmente em ecossistemas megadiversos e ameaçados como a Mata Atlântica.

Nossa pesquisa parte da hipótese de que distúrbios ambientais intensos — representados por alta severidade de fogo e baixa cobertura florestal — podem promover comunidades funcionalmente homogêneas, dominadas por espécies generalistas, como resultado de fortes filtros ambientais (Keddy, 1992; Lasky *et al.*, 2013). Especificamente, esperamos que os valores médios ponderados por abundância (CWM) e as variâncias funcionais (CWV) de traços dietários sejam negativamente influenciados pelo aumento do distúrbio, refletindo convergência funcional nas comunidades de aves.

Com base nessa hipótese, nossos objetivos centrais do trabalho foram: (1) investigar como a severidade do fogo, sua extensão e a cobertura florestal interagem para moldar os traços alimentares de aves tropicais, e (2) identificar padrões de convergência ou divergência funcional em paisagens com diferentes históricos de distúrbio. Ao abordar essas questões, nosso estudo contribui para o avanço do conhecimento em ecologia funcional e do fogo, oferecendo subsídios para estratégias de manejo e conservação em florestas tropicais sujeitas a regimes crescentes de distúrbios antrópicos.

2 INTERACTIVE EFFECTS OF FIRE AND HABITAT LOSS SHAPE BIRD DIETARY TRAIT STRUCTURE IN TROPICAL FORESTS

2.1 Introduction

In recent decades, the number and intensity of wildfires have increased significantly worldwide, driven by climate change and anthropogenic pressures such as land-use transformations (Pausas & Keeley, 2021; Williams *et al.*, 2019). This escalation affects not only fire-prone regions but also ecosystems historically unadopted to such disturbances, including tropical forests. These forests are particularly vulnerable, as fire events can reduce ecosystem resilience and potentially shift them toward degraded or savanna-like states (Sansevero *et al.*, 2020).

The increasing susceptibility of tropical forests to fires reflects both their limited capacity to tolerate such disturbances and the compounding effects of landscape fragmentation. Fires consume leaf litter, increase mortality of seedlings and small trees, and open the canopy—conditions that promote further drying and fuel accumulation. When intense, fires can even damage large trees by affecting their root systems (Barlow *et al.*, 2002; Flores *et al.*, 2014; Uhl & Kauffman, 1990). This facilitates the establishment of pioneer plants, further increasing flammability (Sansevero *et al.*, 2020). Fragmented landscapes, with irregular edges and low forest cover, are particularly prone to ignition and spread (Singh & Huang, 2022). However, despite their potential to amplify disturbance effects, the interactive roles of fire and habitat loss in shaping community structure remain poorly understood (Driscoll *et al.*, 2021).

Fires in tropical forests not only cause direct mortality but also alter habitat structure and resource availability (Chalmandrier *et al.*, 2013; Smith & Lyon, 2000;

Hidasi-Neto *et al.*, 2012; Sitters *et al.*, 2016). These changes act as environmental filters, favoring species with traits suited to altered conditions while excluding others. Functional guilds respond divergently to these filters, reflecting differences in ecological requirements and adaptive strategies (Godoy *et al.*, 2025). For instance, insectivorous and canopy-dwelling birds are typically more sensitive to canopy loss and structural simplification (Barlow & Peres, 2004a), whereas granivores, frugivores, and nectarivores may benefit from the proliferation of early successional plants (Barlow & Peres, 2004b). Consequently, fire disturbances can shift community composition by filtering traits and relaxing constraints on species similarity.

Despite this complexity, most studies focus on taxonomic or functional diversity indices (Barlow & Peres, 2004a, 2004b; Hidasi-Neto *et al.*, 2012; Trouvé *et al.*, 2020; Bregman *et al.*, 2016), which may obscure how specific traits drive community responses. Aggregated indices often mask the mechanisms underlying community shifts (Poff, 1997; Southwood, 1997), limiting our ability to detect whether trait convergence (due to filtering) or divergence (due to **limiting similarity**) prevails. Examining individual traits offers a more mechanistic understanding. Dietary specialization, for instance, is highly responsive to disturbances such as fire severity, extent, and forest loss (Henle *et al.*, 2004; Driscoll *et al.*, 2020). Specialists are more likely to decline in degraded landscapes, whereas generalists persist due to broader niche breadths (Keddy, 1992; Lasky *et al.*, 2013).

Based on context, we aimed to address how bird communities respond to fire severity, fire extent, and forest cover in Atlantic Forest landscapes. Specifically, we assessed how these disturbances shape community structure through dietary trait-based responses. We used the Community-Weighted Mean (CWM) and Community-Weighted Variance (CWV) to assess shifts in dietary specialization along gradients of fire and forest loss. In particular, CWM reflects the dominance of trait values (i.e., prevalence of generalists vs. specialists), while CWV quantifies trait variability, indicating the range of strategies coexisting within communities (Muscarella & Uriarte, 2016).

We predicted that both CWM and CWV will decline with increasing disturbance, indicating trait convergence and dominance of generalists (Figure 1). We also expected that fire and forest loss would interact synergistically to intensify environmental filtering, reducing both trait variability and the presence of dietary specialists. In particular, species with narrow dietary niches—such as frugivores, insectivores, and nectarivores—should decline in disturbed landscapes (Maitner *et al.*, 2023). These predictions align with filtering theory, which suggests that only species able to cope with reduced resource diversity and competition will persist (Kubota *et al.*, 2018).

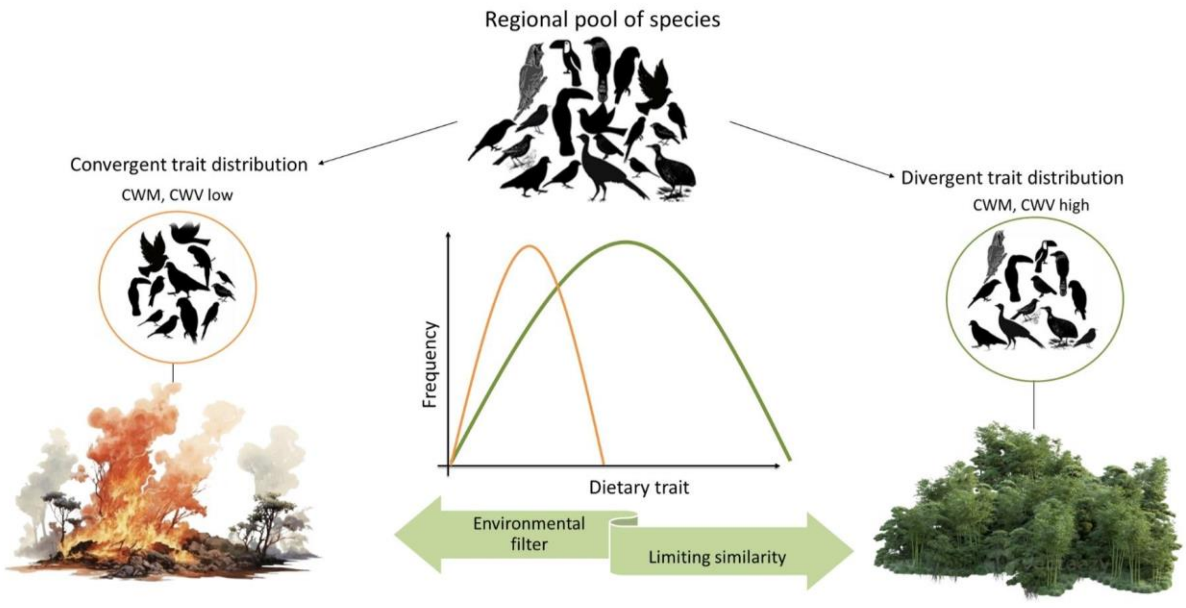

Figure 1. Conceptual framework illustrating the effects of fire disturbance on dietary trait distribution in bird communities. We hypothesize that increasing disturbance (e.g., fire) promotes environmental filtering (orange), leading to a convergent trait distribution with lower community-weighted mean (CWM) and trait variance (CWV), reflecting generalist-dominated, functionally homogeneous assemblages. This is illustrated by the shift toward the left in dietary trait values (orange curve). In contrast, low-disturbance environments such as intact forests may foster limiting similarity (green), promoting a divergent trait

distribution with higher CWM and CWV, as niche differentiation allows the coexistence of specialists with varied dietary strategies (green curve). The central species pool illustrates the potential diversity filtered under different disturbance regimes.

2.2 Methods

2.2.1 Study area

We conducted this study in the Cantareira-Mantiqueira Ecological Corridor region in the southeastern portion of the Atlantic Forest, Brazil (Figure 2). This region exhibits altitudinal variations ranging from 700 to 1700 meters (Oliveira-Filho & Fontes, 2000), with a climate classified as Cwa according to the Köppen classification, described as a humid subtropical climate with hot, rainy summers and dry, mild winters (Alvares *et al.*, 2013).

Currently, the Atlantic Forest has approximately 36% natural vegetation cover (Vancine *et al.*, 2024). Most of the remaining Atlantic Forest areas consist of small, isolated fragments suffering from varying degrees of anthropogenic disturbances (Ribeiro *et al.*, 2009). These high levels of forest fragmentation significantly impact on fire dynamics, increasing the fire susceptibility to both natural and human-caused fires and reducing the natural regeneration power (Guedes *et al.*, 2020; Singh & Huang 2022). Fire typically spreads into forest remnants from nearby land uses, such as pastures, contributing to the spread of fires within forests (Guedes *et al.*, 2020). This study, therefore, encompasses landscapes exhibiting a gradient of forest cover with varying levels of fire disturbance.

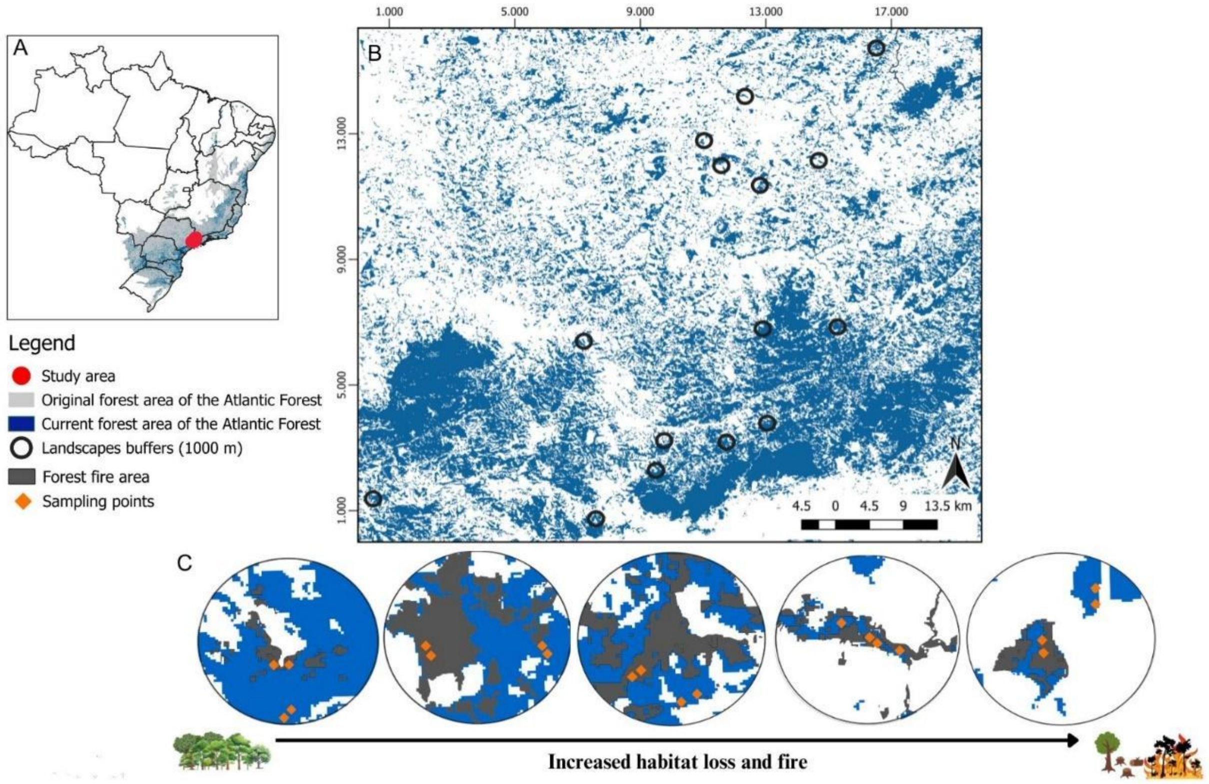

Figure 2. Study Area: The Cantareira-Mantiqueira Ecological Corridor, located in the southeastern region of the Atlantic Forest, Brazil (A). The map (B) illustrates the distribution of the 15 study landscapes along a gradient of habitat loss and fragmentation. In (C), examples of landscapes across a gradient of post-fire regeneration ages and habitat loss are shown, with four sampling points established in each landscape, arranged in pairs within both unburned and burned forests.

2.2.2 Sampling design and predictor's metrics

To identify and select burned landscapes and calculate fire metrics and forest cover, we performed four steps:

1. **Fire occurrence data.** We accessed the fire occurrence database provided by the National Institute for Space Research (INPE), which detects and monitors wildfires using thermal radiation emitted by fire, integrating data from multiple satellites (NOAA-18 and 19, METOP-B and C, TERRA and AQUA, NPP-Suomi, and NOAA-20). From this database, we selected fire occurrences recorded in 2014, 2020, and 2021, the years with the highest incidence of wildfires within the study area, focusing on fire occurring within forest patches. These years

were grouped into a single "burned forest" category, as an exploratory analysis revealed no significant differences in bird diversity across these years (see more in *reference omitted for double-blind review*). This grouping aligns with research suggesting that bird community recovery after fires may extend up to 10 years (Mestre *et al.*, 2013).

2. **Burned scar identification and fire metrics calculation**. To delineate fire extent and severity, we identified burn scars using Landsat 8 level-2 images with 30-meter spatial resolution and low cloud cover, obtained from the United States Geological Survey (USGS) EarthExplorer platform (http://earthexplorer.usgs.gov). For each fire event, we selected one pre-fire and one post-fire image to calculate the Normalized Burn Ratio (NBR), a spectral index widely used to assess vegetation health and burn severity. NBR is calculated using the near-infrared (NIR) and shortwave-infrared (SWIR) bands of satellite images, with higher NBR values indicating healthy, unburned vegetation, while lower values suggest areas with reduced vegetation cover or fire damage. By comparing NBR values before and after fire events, we obtained a quantitative measure of fire impact, allowing the detection of changes in vegetation structure and cover associated with each burn scar.
3. **Field validation of burn scars.** To ensure accurate classification of burned areas and to validate remote sensing data, we conducted field inspections in all selected landscapes. We looked for physical evidence of past fires such as charred tree trunks, ash and charcoal deposits on the forest floor, and fire scars on vegetation. Additionally, we interviewed local residents and landowners to gather information about the timing, extent, and characteristics of fire events. This step was essential for confirming the presence of past fires and minimizing

misclassification errors, thereby strengthening the reliability of our fire severity and extent metrics.

4. **Landscape Selection and Forest Cover Delineation.** Combining INPE and USGS maps, we identified 15 burned landscapes. For each landscape, we delineated 1 km buffers around the fire center to calculate landscape metrics. This scale was selected based on evidence from previous studies conducted in the region, which demonstrated that birds respond to similar scales when assessing taxonomic, functional, and phylogenetic diversity (Barros *et al.*, 2019; Adorno *et al.*, 2021; Manzoli *et al.*, 2024). The landscapes were separated by at least 2 km to ensure sampling independence and capturing a gradient of fire extent and severity. Forest cover within each landscape was manually delineated using 2022 Google Earth imagery (Google, 2022), with values ranging from 10% to 85% across the landscapes (Table S1, Supplementary Material). Finally, we calculated the fire extent in forest remnants within the same landscapes, which values ranged from 1.2% to 33.8% (Table S1).
5. **Fire Severity Assessment.** Within each burn scar, we calculated the average ΔNBR for all pixels within forest habitat boundaries, as well as for unburned forests within the landscape. Following Key and Benson (2006), positive values of ΔNBR indicate burned areas, with higher values corresponding to more severe fires and extensive fire damage; negative values represent unburned vegetation or areas of regrowth, with lower negative values linked to dense, intact vegetation. In our study, fire severity values ranged from -0.003 to 0.168 capturing a gradient of fire impacts (Table S1).

2.2.3 Bird community sampling

To assess bird communities across the studied landscapes, we established four sampling points within each landscape—two located in burned forests and two in unburned forests. We positioned the points randomly while ensuring spatial independence by maintaining a minimum distance of 200 meters between them. This spacing helped prevent overlapping detections and double counting, while also capturing the heterogeneity within each landscape (Vielliard *et al.*, 2010).

At each sampling point, we conducted bird surveys using the point count method within a 50-meter radius, following the protocol outlined by Sutherland (2006). We started the surveys at 5:30 a.m. and concluded by 10:00 a.m., conducting each survey over a 20-minute period. This schedule allowed us to maximize bird detectability during peak activity periods, especially in species-rich environments where cryptic or rare species might be present (Sutherland, 2006). We recorded all birds observed or heard within the points, but we excluded individuals that were merely flying over without interacting with the forest habitat. To account for temporal variation in bird activity and increase sampling robustness, we revisited each point on two separate days between October 2022 and January 2023. This approach resulted in four survey repetitions per forest type within each landscape (2 points × 2 visits x 20 min), totaling 80 minutes of observation effort per forest type.

We defined each forest type (burned or unburned) within a given landscape as a distinct sampling unit, considering each landscape as comprising two sampling units. In total, we surveyed 30 sampling units across the study area.

2.2.4 Sample Coverage

To assess the completeness of our bird community inventories and ensure robust estimates of species richness and trait composition, we calculated the sample coverage for each landscape. Sample coverage, as proposed by Chao and Jost (2012), estimates the proportion of the total number of individuals in a community that belong to the species detected in the sample. It provides a measure of inventory completeness, which is essential for minimizing sampling bias and improving the accuracy of community-level inferences (Chao & Jost, 2021).

To assess the completeness of our bird inventories, we estimated sample coverage using the `` `iNEXT3D()` `` function (Chao & Jost, 2021), which calculates the proportion of the total community captured in each sample based on incidence frequencies. Sample coverage values ranged from 0.76 to 0.98 across landscapes, with a mean (± SD) of 0.87 ± 0.06, indicating that our sampling effort was generally sufficient to capture most of the bird communities present.

Landscapes with relatively lower sample coverage (see Table 1) were considered with caution in the interpretation of community-level patterns. Importantly, all landscapes were sampled using a standardized protocol with equal effort across sites, minimizing systematic biases. By incorporating sample coverage estimates, we ensured a transparent evaluation of inventory quality and strengthened the robustness of our comparisons between burned and unburned forest patches.

Table 1. Sample coverage SC(n) estimates for bird assemblages across landscapes and environments (burn and unburned).

Assemblage	N	S.obs	SC(n)	SC(2n)
P04B	91	31	0.9261	0.9981
P04NB	114	50	0.8541	0.9876
P06B	60	24	0.9210	0.9967
P06NB	119	42	0.8919	0.9685
P07B	111	46	0.8488	0.9633
P07NB	93	48	0.8007	0.9803
P11B	116	46	0.8818	0.9895
P11NB	95	31	0.8961	0.9689
P14B	114	36	0.8961	0.9769
P14NB	86	40	0.7931	0.9243
P15B	68	34	0.7383	0.8806
P15NB	87	30	0.9446	0.9977
P16B	47	21	0.8802	0.9938
P16NB	64	30	0.7539	0.9102
P22B	58	20	0.9173	0.9924
P22NB	81	23	0.9901	1.0000
P23B	97	39	0.8479	0.9694
P23NB	101	40	0.8232	0.9191
P30B	163	47	0.9706	1.0000
P30NB	133	46	0.9189	0.9936
P31B	100	38	0.8408	0.9038
P31NB	130	37	0.9318	0.9908
P42B	82	35	0.8450	0.9756
P42NB	109	36	0.9006	0.9807
P43B	76	30	0.8853	0.9900
P43NB	123	39	0.9364	0.9959
P47B	153	48	0.9494	0.9996
P47NB	84	31	0.8472	0.9483
P48B	102	32	0.9236	0.9951
P48NB	114	36	0.9318	0.9973

2.2.5 Estimating species abundance

Due to the high presence of rare species and the likelihood of non-detection during individual surveys, we applied a hierarchical N-mixture model using the `unmarkedFrameOccu` function available in the `unmarked` package in R (Fiske & Chandler, 2011; Kellner *et al.*, 2023). This method estimates both species abundance and detection probability simultaneously, addressing the challenge associated with imperfect detection.

In the N-mixture model, species abundance was assumed to follow a Poisson distribution, while detection probability was modeled as a function of environmental and survey-specific covariate (i.e., landscape code). By incorporating the four repetitions per sampling unit into the analysis, the model accounts for variability in detection across sampling occasions, providing a more accurate estimate of true species abundance. To improve the reliability of abundance estimates, we included only species with observed abundance values equal to or greater than 1 and detection probabilities of at least 0.1. This threshold, based on recommendations by Royle *et al.* (2005), minimized biases associated with rare detections and enhanced the robustness of abundance estimates. This approach allowed us to accurately estimate species abundance and detection probabilities for each forest type, enabling a detailed assessment of the structure and dynamics of bird communities across burned and unburned forest landscapes.

2.2.6 Dietary traits

We employed a dataset as sources of dietary information for recorded species (Wilman *et al.*, 2014). This dataset provides estimates of the relative consumption of different diet categories, including invertebrates, fruits, nectar, and seeds, expressed as percentages ranging from 0% to 100%. These dietary categories offer a detailed perspective on species habits and help assess the role of environmental filtering in shaping species composition (Ulrich *et al.*, 2018).

For this study, we considered each dietary category as a distinct trait and analyzed the relative consumption of fruits, invertebrates, seeds, and nectar. Due to the limited number of carnivorous species records, we excluded this category from statistical analyses to ensure data robustness and consistency.

2.2.7 Calculation of traits

To assess changes in dietary specialization across fire gradients of characteristics and forest cover we calculated two metrics: Community-Weighted Mean (CWM) and Community-Weighted Variance (CWV). CWM represents the average trait value weighted by species abundance, indicating whether communities are dominated by generalists or specialists (Maitner *et al.*, 2023; Muscarella & Uriarte, 2016). A decline in CWM a shift toward generalist-dominated communities, potentially resulting from reduced resource diversity or simplified foraging niches (Kubota *et al.*, 2018). CWV quantifies the variability of trait values within communities, offering insights into the breadth of dietary strategies and functional redundancy (Maitner *et al.*, 2023; Muscarella & Uriarte, 2016). Lower CWV under disturbance conditions is typically associated with trait convergence and environmental filtering (Cornwell & Ackerly, 2009; Enquist *et al.*, 2015), whereas higher CWV may reflect niche differentiation and indicate more functionally diverse and potentially resilient communities (MacArthur & Levins, 1967; Grime, 2006).

We estimated CWM and CWV using a nonparametric bootstrapping approach, following the approach outlined by Maitner *et al.*, (2023). Bootstrapping involves resampling the observed data with replacement to generate replicated distributions, thereby incorporating uncertainty into trait-based analyses. Specifically, we resampled trait values in proportion to the estimated abundance of each species within each sampling unit.

For these calculations, we used two main datasets: (1) a table containing species-specific abundance estimates for each sampling unit, which served as weights to reflect each species' contribution to community-level trait structure; and (2) a dataset containing trait values for each species, necessary to associate functional characteristics with the corresponding abundance data. These two components are essential for trait-based metrics such as CWM and CWV, which depend on the weighted distribution of trait values within communities. We implemented the bootstrapping procedure using the traitstrap package in R (Maitner *et al.*, 2021, 2023). Each sampling unit was linked to a corresponding landscape code to account for the nested sampling design. We performed 50 iterations of resampling, drawing 15 trait values per iteration based on abundance weights—this number corresponded to the minimum total abundance observed across sampling units. For each resampling iteration, we calculated CWM and CWV, and we used the average values across iterations as the final estimates for each forest type within each landscape.

2.2.8 Statistical analysis

We employed Generalized Linear Mixed Models (GLMM) to test the effects of forest cover, fire extent, and fire severity on variation in community-weighted mean (CWM) and community-weighted variance (CWV). All models were fitted using the 'lmer' function from the 'lme4' package (Bates *et al.*, 2015). For each dietary category (fruits, invertebrates, nectar, and seeds), we initially constructed global additive models that included all three continuous predictor variables—forest cover (%), fire extent (ha), and fire severity (mean ΔNBR)—as well as the categorical variable forest type (burned vs. unburned) (see Table S1 in the Supplementary Material). To account for the nested sampling design, in which each landscape contained two sampling units, we included landscape identity as a random effect in all models (Table 2). We verified that the predictor variables were not strongly correlated (Pearson's r < 0.7), ensuring their joint inclusion in the models would not lead to multicollinearity issues.

Table 2. Fixed and random effects included in each global model. CWM diet = average trait consumption across the community. CWV diet = within-community trait variation. Forest cover = Representing the percentage of forest unburned in the landscape (%). Fire extent = Representing the extension of burned forest cover (ha). Fire severity = ΔNBR represents the change in NBR values (vegetation cover) between pre- and post-fire imagery. Habitat = representing burned and unburned forests.

GLMM structure		
Response variable	Fixed effect	Random effect
CWM diet ~	forest cover + fire extent + fire severity + habitat +	(1|landscape)
CWV diet ~	forest cover + fire extent + fire severity + habitat +	(1|landscape)

To reduce model complexity and avoid overfitting, we used the 'dredge' function from the 'MuMIn' package (Bartón, 2022) to generate all possible model subsets based on the global model, limiting the number of predictors to a maximum of two per model. We assessed models performance using the Akaike Information Criterion corrected for small samples (AICc; Akaike, 1974). We initially selected models based on ΔAICc, retaining those with values < 2, which are considered to have substantial empirical support (Burnham & Anderson, 2002). In addition, we applied an additional selection filter using the Akaike weights (Wi), retaining only models with a weight greater than 0.1, which represent higher relative support.

After selecting the best additive models based on AICc, we constructed interaction models using the same variables from the top-performing additive models.

We then compared the performance of these interaction models with their respective additive models using the ΔAICc and model weights, calculated with the AICtab function from the ‘bbmle’ package (Bolker, 2017). This approach allowed us to evaluate whether including interactions among predictor variables improved model performance, ensuring that the best model structures were selected based on the same AICc criterion.

Finally, we tested the normality of residuals for each model using the Shapiro-Wilk test. Models with p-values > 0.05 were considered to have normally distributed residuals. All statistical analyses were performed in R (R Core Team, 2023)

2.3 Results

Across the 15 landscapes, we recorded 181 bird species from 42. Tyrannidae was the most species-rich family (30 species), followed by Furnariidae and Thraupidae (14 each). Species richness was similar between forest types, with 149 species in unburned and 148 in burned forests. Exclusive species also occurred in comparable numbers (32 in burned, 33 in unburned). Among the species exclusive to burned forests, we observed a higher dietary specialization in some groups. Notably, we identified the following specialized species (consumption ≥ 70% in a single category): 1) Seed consumption trait: *Sicalis flaveola* and *S. luteola*; 2) Insect consumption trait: *Heliobletus contaminatus* and *Nyctibius griseus;* and 3) Fruit consumption trait: *Tersina viridis* and *Odontophorus capueira.* Among the species exclusive to unburned forests, we also found species with higher dietary specialization (consumption ≥ 70% in a single category), including 1) Insect consumption trait: *Cryptopezus nattereri* and *Myiornis auricularis;* 2) Nectar consumption trait: *Heliothryx auritus* and *Phaethornis eurynome*;

and 3) Fruit consumption trait: *Amazona amazonica* and *Euphonia violacea*; and *Ramphastos dicolorus.*

Regression models revealed contrasting responses of CWM and CWV across dietary traits in relation to forest cover, fire extent and fire severity (Table S3, Supplementary Material). Forest cover appeared in six of the top-ranked models, followed by fire severity (five models), forest type (three), and fire extent (two). Notably, four models included interaction terms between fire variables and forest cover—particularly for fruit and nectar consumption traits—indicating that the effects of fire were modulated by the landscape forest cover.

Both fruit and nectar consumption traits were affected by interactions with forest cover, although the specific fire variable involved—fire severity or fire extent—differed between traits and between CWM and CWV models. For fruit consumption, CWM was best explained by an interaction between fire extent and forest cover: average fruit consumption was lowest in landscapes with high fire extent and low forest cover, and increased with greater forest cover (Figure 3a). In contrast, CWV for fruit consumption responded to an interaction between fire severity and forest cover, with trait variance increasing with forest cover under low fire severity, but this effect was diminished under high severity (Figure 3b).

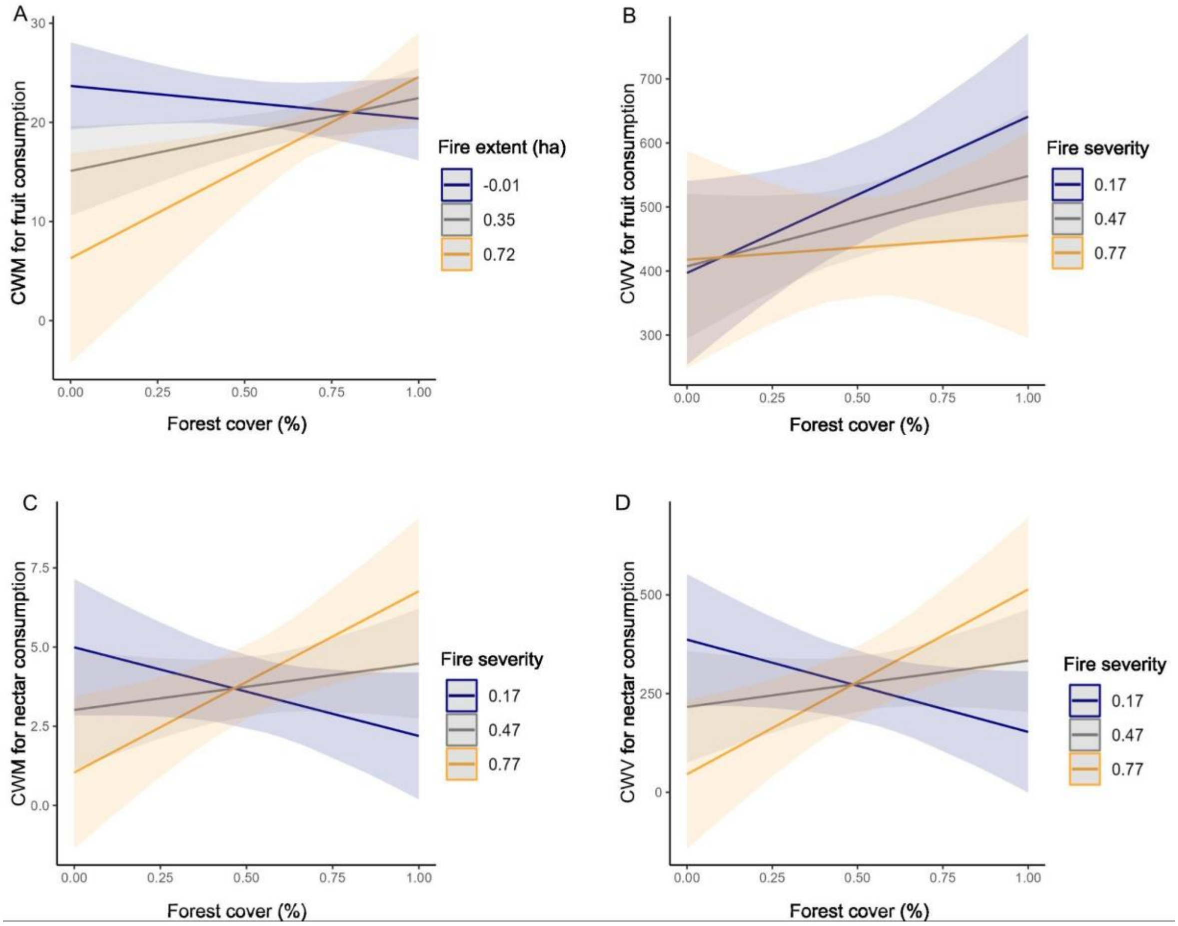

Figure 3. Effect of forest cover, fire severity, and fire extent on the fruit and nectar diet of birds in the Atlantic Forest, Brazil. The definitions of the variables can be found in the metadata. To visualize the effects of continuous fire metrics, we rescaled the data into three levels based on standard deviation values, represented by different colors: orange indicates one level above the standard deviation, gray represents the mean value, and blue corresponds to one standard deviation below the mean value. (A) Interactive model between forest cover and fire extent, showing that forest cover strongly increases average fruit consumption in the community, but this effect weakens as the fire extent increases. (B) Variance in fruit consumption rises with forest cover but diminishes under high fire severity. (C) Community-weighted mean (CWM) nectar consumption was influenced by the interaction between fire severity and forest cover, with highly burned landscapes exhibiting lower average nectar consumption, particularly in areas with high forest cover. (D) Trait variability (CWV) for nectar consumption followed a similar pattern, suggesting that high forest cover buffers the effects of severe fire, promoting dietary diversity, whereas in less forested landscapes under mild fire conditions, both mean nectar consumption and trait variability increased.

For nectar consumption, both CWM and CWV were best explained by interactions between fire severity and forest cover (Figures 3c and 3d). In landscapes with high fire severity and low forest cover, average nectar consumption and trait variability were both low. As forest cover increased under high fire severity, both CWM and CWV increased. Under low fire severity, however, increasing forest cover was associated with declines in both metrics.

In contrast to fruit and nectar traits, invertebrate consumption showed simpler patterns. CWM was best explained by an additive model including fire severity and forest type (Figure 4a). Average invertebrate consumption was higher in unburned forests compared to burned ones, and increased with fire severity. CWV was best explained by an interaction between fire extent and forest cover (Figure 4b). Trait variance was lowest in landscapes with low forest cover and high fire extent. In contrast, under low fire extent, CWV remained relatively high across the forest cover gradient.

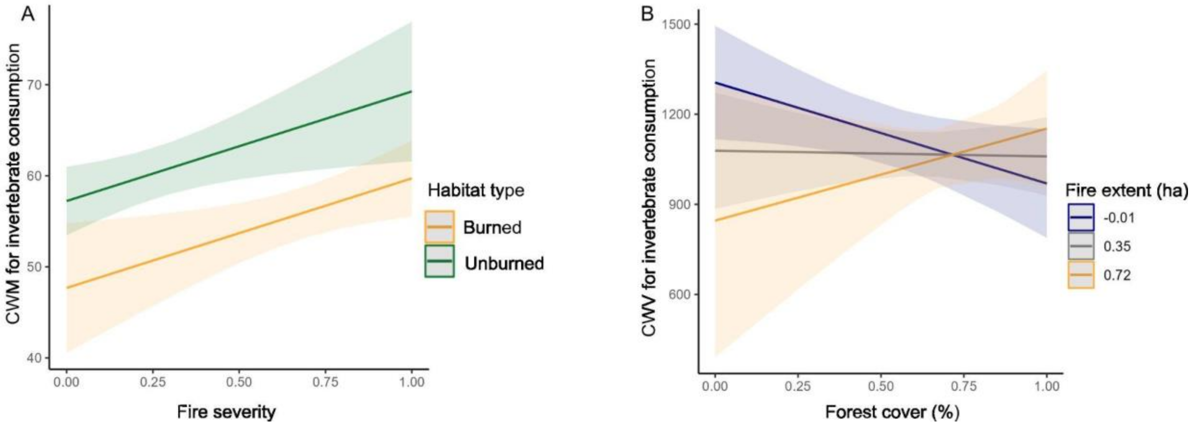

Figure 4. Effects of fire disturbance and forest cover on invertebrate consumption. (A) Community-weighted mean (CWM) invertebrate consumption was higher in unburned forests and areas with lower fire severity, suggesting that specialized insectivores are more sensitive to habitat disturbances. (B) Invertebrate trait variability (CWV) was influenced by the interaction between fire extent and forest cover, with higher fire extent in poorly forested landscapes leading to reduced trait variance, indicating stronger environmental filtering in these conditions.

For seed consumption, both CWM and CWV were best explained by forest type alone (Figures 5a and 5b). In both cases, values were higher in burned forests compared to unburned forests, indicating that seed consumption was more prevalent and more variable in burned landscapes.

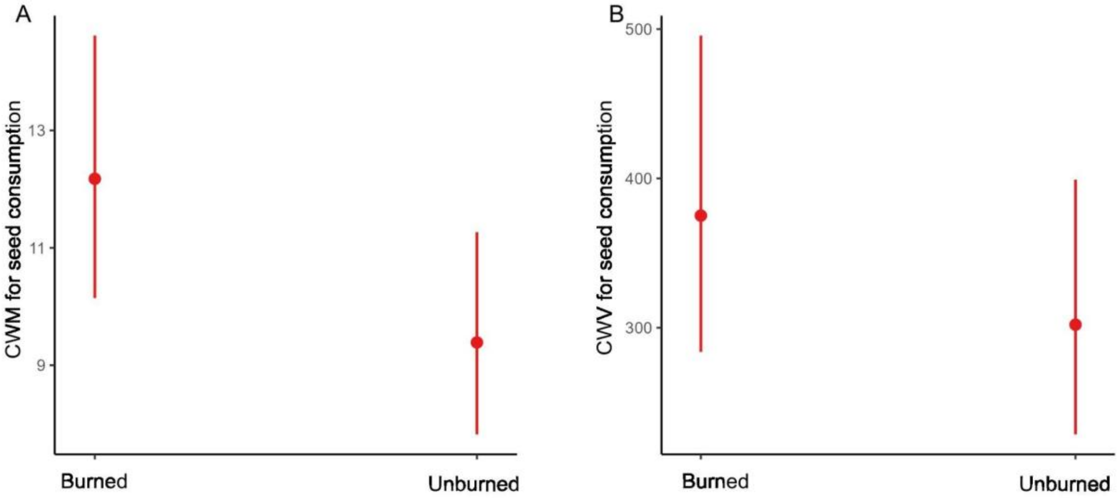

Figure 5. Effects of forest burning on seed consumption. (A) Community-weighted mean (CWM) seed consumption was higher in burned forests, indicating an increased presence of seed-consuming birds in post-fire environments. (B) Trait variability (CWV) for seed consumption was also greater in burned forests, suggesting a wider range of dietary strategies.

2.4 Discussion

Understanding how fire disturbance and habitat loss interact to shape ecological communities is critical, especially in tropical forests where both pressures are intensifying. While fire and deforestation are well-established drivers of biodiversity loss, their combined effects remain poorly understood (Cadotte *et al.*, 2015; Sitters *et al.*, 2016). Here, we present one of the first empirical assessments of how these disturbances jointly influence community-level dietary traits in tropical bird assemblages.

Our results showed that both the CWM and CWV of bird dietary traits were influenced by fire disturbance, with responses strongly modulated by forest cover. While some dietary components—such as fruit and nectar consumption—were shaped by interactions between fire variables and forest cover, others—such as seed and invertebrate consumption—were more directly associated with forest type or displayed simpler patterns. These findings demonstrate that fire and habitat loss do not act independently but instead interact to shape community functional structure, often producing amplified effects under specific environmental conditions.

Dietary trait consumption: influence of fire disturbance and forest cover

Both fruit and nectar consumption traits responded strongly to interactions between fire disturbance and forest cover. These interactions varied depending on the trait and response metric, but consistently highlighted the role of forest cover in modulating fire impacts.

For fruit consumption, CWM increased with forest cover, but this effect was contingent upon fire extent —communities in heavily burned and poorly forested landscapes showed notably low average fruit consumption. This pattern reflects strong environmental filtering, where species with low fruit consumption dominate in the most disturbed conditions. These results align with previous studies showing that environmental filtering becomes more prominent in structurally simplified habitats, such as post-fire or fragmented forests (Coster *et al.*, 2015; Fraaije *et al.*, 2015; Nimmo *et al.*, 2019). In fact, disturbance often promotes trait convergence, favoring generalist species while excluding specialists (Morante-Filho *et al.*, 2015). Conversely, intact forests with low fire impact and high forest cover tend to support a more diverse and temporally stable supply of fruit resources. These conditions promote trait divergence and niche partitioning based on fruit type, phenology, and foraging behavior (Hasui *et al.*, 2007). Species such as *Euphonia pectoralis, Ramphastos dicolorus*, and *Pyroderus scutatus* depend on specific fruiting plants and complex vegetation structure (Marjakangas *et al.*, 2020; Rogers *et al.*, 2021). However, post-fire simplification of vegetation reduces these niche opportunities, leading to generalist dominance and reduced trait diversity (Barlow & Peres, 2006; Coster *et al.*, 2015).

Nectar consumption traits also responded to the interaction between fire severity and forest cover, but showed a distinct pattern compared to other dietary components. In landscapes with high fire severity and low forest cover, both average nectar consumption (CWM) and trait variability (CWV) were low, indicating that nectarivorous species were largely filtered out under these compounded disturbances. However, in similarly fire-affected landscapes with greater forest cover, both metrics increased, suggesting that habitat amount can buffer the negative effects of fire and allow for greater functional diversity in nectar use. This pattern contrasts with findings from previous studies, where nectarivores are often reported as resilient or even positively associated with disturbed environments (Gray *et al.*, 2007; Neu *et al.*, 2023; Glass & Arcilla 2024).

Our results indicated that fire alone may not fully account for changes in nectarivorous traits, but that its impact is strongly modulated by landscape context. In highly disturbed yet well-forested landscapes, nectarivores may persist or even diversify due to retained habitat complexity and floral resource availability (Edwards *et al.*, 2014, Glass *et al.*, 2024, Tobias, 2015). Conversely, in similarly burned areas with low forest cover, the combined effects of habitat loss and fire disturbance may surpass ecological thresholds for nectarivore persistence. However, this result should be interpreted with caution, since nectar consumption remained relatively low at the community level (CWM < 8%; Table S2), reflecting the limited number of species that rely heavily on nectar.

Invertebrate consumption was influenced by both fire severity and forest type. CWM increased with fire severity, suggesting that bird communities in more intensely burned areas may include more insects in their diets. However, average invertebrate consumption was consistently higher in unburned forests compared to burned ones (Figure 4a), likely reflecting the absence of specialized insectivores that rely on structurally complex microhabitats such as dense leaf litter, fallen logs, and understory vegetation—features often degraded by fire (Barlow *et al.*, 2002; Şekercioğlu *et al.*, 2002). In particular, species such as C*ryptopezus nattereri, Drymophila squamata*, and *Conopophaga lineata*, which depend on the forest floor and understory, were recorded exclusively in unburned forests, underscoring their sensitivity to fire. Their absence in burned landscapes suggests strong environmental filtering, leading to reduced dietary specialization and a convergence around more generalist foraging strategies.

In contrast, trait variability (CWV) for invertebrate consumption was influenced by an interaction between fire extent and forest cover (Figure 4b). In landscapes with low forest cover and high fire extent, CWV was low, indicating trait convergence likely driven by environmental filtering. However, in more forested landscapes—even under extensive fire—trait variability increased, indicating that habitat amount supports a broader spectrum of invertebrate consumption levels within the bird community. This pattern may reflect the persistence of species with varying degrees of dietary dependence on invertebrates, including those capable of exploiting fire-modified habitats such as standing deadwood and exposed trunks, which may harbor wood-boring prey (Schepps *et al.*, 1999; Craig, 2012; Lorenz *et al.*, 2015; Hutto *et al.*, 2016). These findings may indicate that insectivorous birds exhibit heterogeneous responses to disturbance, with some species declining while others may persist or benefit depending on landscape context.

Seed consumption showed a distinct pattern compared to other dietary traits, with both the community-weighted mean (CWM) and trait variance (CWV) significantly higher in burned than in unburned forests (Figures 5a and 5b). This indicates that post-fire environments support a greater relative abundance and broader variation in seed consumption across bird species, including granivores such as *S. flaveola* and *S. luteola*, which were found exclusively in burned areas. This pattern may be linked to fire-induced changes in seed availability and vegetation regeneration. For instance, studies have shown that fire can stimulate seed germination (Tang *et al.*, 2003; Williams, 2000; Baker *et al.*, 2022), favor the growth of ruderal and pioneer plants, and enhance seedling recruitment (Gray et al., 2007). In tropical forests, such post-fire dynamics often lead to an increase in herbaceous species and early-successional trees that produce accessible and abundant seeds (Abedi *et al.*, 2022), which may benefit a variety of bird species with seed-based diets.

In fact, in degraded forests, such as those that have experienced fire events, there may be a wider variation in the degree to which species depend on seeds, leading to an increase in CWV. This pattern is consistent with community assembly processes driven by limiting similarity, where species partition available seed resources to reduce competition (Grime, 2006; MacArthur & Levins, 1967). However, fire does not universally promote dietary differentiation. Under more extreme fire regimes, resource depletion or simplification may lead instead to trait convergence, favoring generalists capable of persisting on a narrower resource base (Enright *et al.*, 2015; Keeley *et al.*, 2011). Thus, the spatial context and intensity of fire disturbance likely determine whether post-fire communities exhibit greater diversification or convergence in seed consumption traits.

Overall, our findings highlighted the complex and trait-specific ways in which fire disturbance and forest cover jointly shape the functional structure of bird communities.

Patterns of trait convergence, such as reduced variability in fruit, nectar, and invertebrate consumption, were prominent under high fire disturbance and low forest cover—conditions where environmental filtering is expected to dominate. In contrast, increased trait variance in seed consumption in burned areas suggests that limiting similarity may operate when post-fire regeneration enhances resource diversity. These results align with the idea that filtering and divergence are not mutually exclusive processes, but rather coexist along disturbance gradients and interact with landscape structure (Ulrich *et al.*, 2018; Barrett *et al.*, 2023).

By incorporating a trait-based approach, our study provides a more nuanced understanding of how community assembly mechanisms respond to combined fire disturbance and habitat loss in tropical forests. The contrasting responses among dietary traits emphasize that disturbance effects are not uniform across ecological dimensions, but depend on the interaction between disturbance type, trait identity, and landscape context (Barrett *et al.*, 2023). Ultimately, the interaction between fire regime and forest cover emerges as a critical driver of functional structure in bird communities, with implications for how we interpret and manage ecological resilience in fire-affected tropical landscapes.

Limitation and future studies

To estimate the community-weighted means (CWM) and community-weighted variances (CWV), we used the abundance of species within each sampling point, weighted by the species traits. However, this approach did not account for individual variability within species traits. Future studies should incorporate individual trait variation to provide a more nuanced understanding of community dynamics and trait distribution. Including individual-level trait data could improve the accuracy of CWM and CWV estimates, offering deeper insights into the effects of environmental and dispersal filtering on community structure (Abedi *et al.,* 2022).

Conservation implications

From a conservation perspective, our findings highlight the critical role of forested landscapes in buffering the impacts of fire on bird communities. In particular, landscapes presenting greater forest cover were more likely to support higher levels of trait specialization and variability, even under fire disturbance, suggesting that maintaining forest amount is key to promoting ecological resilience. Moreover, synergistic effects of fire and forest loss caution against addressing these threats in isolation. Conservation strategies in fire-prone tropical regions should therefore integrate fire management with habitat preservation, particularly in fragmented landscapes where the interaction between disturbances is more likely to erode functional diversity and disrupt ecosystem processes, including key services such as seed dispersal, pollination, and insect population regulation (Adedoja *et al.*, 2019; Robinson *et al.*, 2013). These results underscore the importance of maintaining unburned refuges as part of an effective conservation strategy (Adedoja *et al.*, 2019; Robinson *et al.*, 2013).

Furthermore, the increasing frequency and severity of fires under climate change scenarios demand urgent, integrated policy responses (Nitschke & Innes, 2006; Pivello *et al.*, 2021). Our findings emphasize that mitigating the compounded effects of fire and habitat loss requires not only ecological knowledge, but also landscape-scale planning and political commitment to protect remaining forest patches and implement adaptive fire management in tropical regions.

2.5 Conclusion

Our study provides one of the first empirical assessments of how fire disturbance and habitat amount jointly shape the functional trait structure of tropical bird communities. Both community-weighted means (CWM) and variances (CWV) responded to fire, with forest cover modulating their effects and revealing non-additive interactions. Environmental filtering dominated under high fire and low forest cover, while greater habitat availability was associated with increased trait diversity, suggesting a role for limiting similarity.

Fruit and nectar consumption showed similar patterns: both traits declined under combined high fire and low forest cover, but increased where habitat was retained, highlighting the buffering role of forested areas. In contrast, seed consumption was higher and more variable in burned forests regardless of forest cover, possibly reflecting altered resource dynamics that promote dietary diversification. These results underscore the value of trait-based approaches in understanding biodiversity responses to multiple stressors and emphasize the need to integrate fire management with habitat conservation to maintain ecological resilience in tropical landscapes.

Acknowledgements

We extend our sincere gratitude to Caio Tavolaro Melo, Ana Luiza Martins de Macedo and Lucas Silva Azeredo for their invaluable assistance during fieldwork. This study was financed in part by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior – Brasil (CAPES) – Finance code 001. Furthermore, we acknowledge the financial support provided by the Sao Paulo Research Foundation - FAPESP, the Minas Gerais Research Foundation - FAPEMG, the National Council for Scientific and

Technological Development - CNPq. Their support was instrumental in the completion of this research.

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Supplementary Material

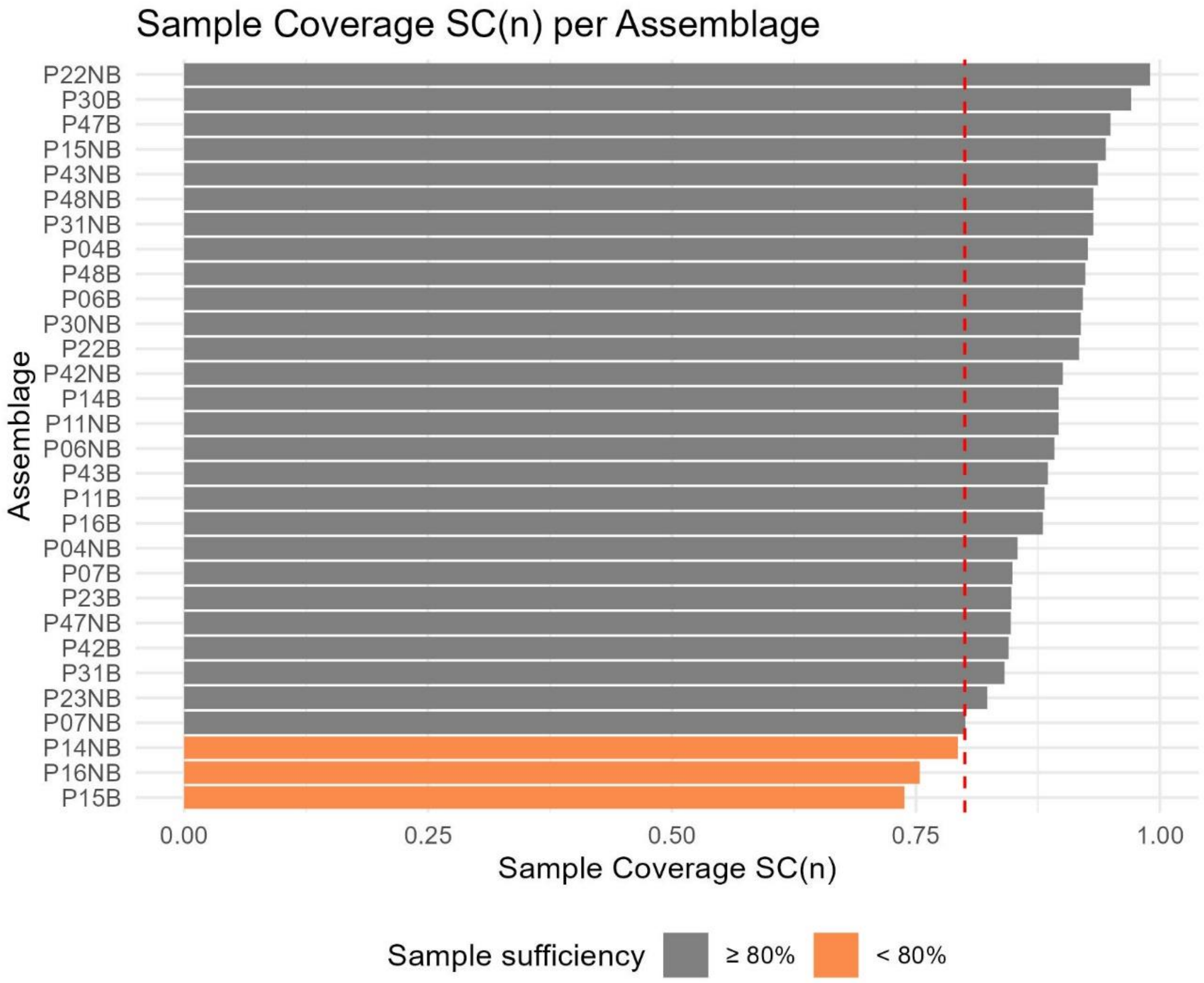

Figure S1. Sample coverage [SC(n)] estimates for bird assemblages across landscapes and fire conditions (NB = unburned forest, B = burned forest). SC was estimated for each assemblage (landscape × environment combination) using the iNEXT.3D package. The red dashed line indicates the completeness threshold (SC = 0.75) adopted in this study. Assemblages with SC(n) ≥ 0.80 (gray bars) were considered to have sufficient sampling completeness, while those with SC(n) < 0.80 (orange bars) showed lower completeness. Most assemblages exceeded the 80% threshold, ensuring robust representation of bird communities, although three assemblages fell below the desired level.

Table S1. Landscape characteristics, geographic data, fire history, forest cover, fire extent and fire severity in 15 Atlantic Forest landscapes, with paired burned (B) and unburned (NB) sites." (continua)

Landscape	Landscape sampling site code	habitat	forest_cover	fire_extent	fire_severity	longitude	latitude	elevation_m	fire_data_lastfire
P04	P04B	burn	73.8500	33.8051	0.1105	-46.4565	-23.2033	934	2020
P04	P04NB	nun_burn	73.8500	33.8051	0.0135	-46.4442	-23.2032	828	2020
P06	P06B	burn	85.4600	6.9522	0.0912	-46.6874	-23.3903	1098	2014
P06	P06NB	nun_burn	85.4600	6.9522	0.0078	-46.6870	-23.3953	1098	2014
P07	P07B	burn	74.8500	2.0464	0.0490	-46.5414	-23.3250	890	2014
P07	P07NB	nun_burn	74.8500	2.0464	-0.0135	-46.5430	-23.3281	1009	2014
P11	P11B	burn	61.0800	1.2373	0.0629	-46.7692	-23.4566	996	2014
P11	P11NB	nun_burn	61.0800	1.2373	0.0084	-46.7628	-23.4526	954	2014
P14	P14B	burn	78.3200	27.2145	0.1184	-46.5478	-23.2041	1086	2021
P14	P14NB	nun_burn	78.3200	27.2145	0.0388	-46.5515	-23.2079	1032	2021
P15	P15B	burn	52.2700	2.4519	0.1676	-47.0520	-23.4234	780	2014
P15	P15NB	nun_burn	52.2700	2.4519	0.1183	-47.0517	-23.4206	760	2014
P16	P16B	burn	35.9800	7.2616	0.0931	-46.7867	-23.2199	812	2014
P16	P16NB	nun_burn	35.9800	7.2616	0.0152	-46.7878	-23.2188	801	2014
P22	P22B	burn	54.7200	21.1749	0.0743	-46.4040	-22.8389	1108	2021
P22	P22NB	nun_burn	54.7200	21.1749	0.0099	-46.4045	-22.8379	1095	2021
P23	P23B	burn	18.3500	10.1808	0.0770	-46.5718	-22.9020	791	2021
P23	P23NB	nun_burn	18.3500	10.1808	0.0459	-46.5685	-22.9034	792	2021
P30	P30B	burn	27.7300	3.4938	0.1529	-46.6000	-22.9956	826	2014
P30	P30NB	nun_burn	27.7300	3.4938	0.0250	-46.6084	-22.9891	943	2014
P31	P31B	burn	41.2800	5.1767	0.1015	-46.5489	-23.0166	965	2014
P31	P31NB	nun_burn	41.2800	5.1767	0.0270	-46.5463	-23.0148	933	2014
P42	P42B	burn	55.8700	27.4583	0.1467	-46.6775	-23.3520	807	2021
P42	P42NB	nun_burn	55.8700	27.4583	0.0789	-46.6786	-23.3557	812	2021
P43	P43B	burn	59.4600	32.4952	0.1639	-46.5999	-23.3550	997	2021
P43	P43NB	nun_burn	59.4600	32.4952	0.1102	-46.5932	-23.3567	994	2021
P47	P47B	burn	22.3000	7.6916	0.1467	-46.6271	-22.9598	860	2020

Table S1. Landscape characteristics, geographic data, fire history, forest cover, fire extent and fire severity in 15 Atlantic Forest landscapes, with paired burned (B) and unburned (NB) sites.” (conclusão)

Landscape	Landscape sampling site code	habitat	forest_cover	fire_extent	fire_severity	longitude	latitude	elevation_m	fire_data_lastfire
P47	P47NB	nun_burn	22.3000	7.6916	0.0188	-46.6213	-22.9561	873	2020
P48	P48B	burn	10.4600	2.1718	0.0854	-46.4764	-22.9857	859	2014
P48	P48NB	nun_burn	10.4600	2.1718	-0.0026	-46.4791	-22.9823	840	2014

Table S2. Community-weighted dietary trait metrics (CWM and CWV) for frugivorous, insectivorous, nectarivorous, and granivorous in 15 Atlantic Forest landscapes, with paired burned (B) and unburned (NB) sites. (continua)

Landscape sampling site code	CWM frugivorous	CWM insectivorous	CWM nectarivorous	CWM granivorous	CWV frugivorous	CWV insectivorous	CWV nectarivorous	CWV granivorous
P04B	25.0670	50.6450	5.6120	12.5820	544.6508	1258.0070	350.8038	386.7623
P04NB	26.5800	57.1920	3.8190	8.3400	661.2646	1099.7218	298.0396	294.6393
P06B	14.7830	57.4150	4.4730	14.8130	417.2856	941.5126	323.9745	450.1118
P06NB	24.3320	59.9630	1.1830	10.6650	652.1005	1232.6041	81.5050	392.6232
P07B	20.9920	51.1800	5.5820	11.9070	401.2748	1178.3976	416.2714	303.3169
P07NB	20.8660	57.7450	5.8660	8.5360	686.0697	1141.7424	413.4289	292.5358
P11B	24.3560	51.2290	2.9400	13.6690	599.0575	1000.6990	218.1248	471.9709
P11NB	26.8600	53.4380	0.6160	11.8970	837.6604	986.7604	5.7774	418.5359
P14B	22.3130	57.3800	5.9820	10.2900	675.8873	1069.3508	475.7263	294.5121
P14NB	19.9360	57.7190	2.6830	11.2340	465.9996	1195.9350	187.9159	373.4804
P15B	20.6640	58.3580	6.0360	8.3590	514.2331	790.3917	422.3851	228.1604
P15NB	22.1170	61.2570	5.2550	4.6550	695.7572	1062.6360	369.9961	105.2827
P16B	17.3760	54.1330	3.9780	16.4100	212.8739	917.8360	342.2515	546.7448
P16NB	25.0770	44.6540	7.1660	15.3310	405.4167	1329.9865	515.8551	512.1298
P22B	19.4110	52.6620	0.2930	20.3320	585.1100	1126.1272	2.8438	795.3956
P22NB	14.9420	70.6040	0.0000	10.6650	371.6270	928.7685	0.0000	330.3181
P23B	23.1230	54.0560	1.7730	12.4700	478.5404	1115.9379	156.4195	493.0599

Table S2. Community-weighted dietary trait metrics (CWM and CWV) for frugivorous, insectivorous, nectarivorous, and granivorous in 15 Atlantic Forest landscapes, with paired burned (B) and unburned (NB) sites. (conclusão)

Landscape sampling site code	CWM frugivorous	CWM insectivorous	CWM nectarivorous	CWM granivorous	CWV frugivorous	CWV insectivorous	CWV nectarivorous	CWV granivorous
P23NB	20.7360	59.1030	1.7150	10.4230	501.9279	1160.0733	117.1395	432.6244
P30B	26.3510	52.0570	2.5820	11.2800	486.5926	1224.6366	115.7989	254.5407
P30NB	16.0850	63.8630	5.2690	7.1940	351.6378	1003.5990	425.6640	208.7819
P31B	20.1000	53.0090	3.9040	14.2190	414.5618	1169.8810	279.0359	539.5877
P31NB	17.4140	67.4690	2.4280	9.0270	488.2918	1065.9864	179.1753	306.9727
P42B	13.5960	67.6570	5.5250	6.9050	274.9724	857.0586	417.2575	205.2069
P42NB	17.6870	66.3440	4.6820	8.2510	377.8351	821.2813	385.3543	201.5265
P43B	11.6160	66.5540	6.9120	7.9210	161.9493	1037.6578	574.7092	246.1653
P43NB	20.8230	64.4940	0.0000	10.0560	546.5861	1168.6413	0.0000	299.0712
P47B	16.3390	57.0730	1.6960	17.6990	407.5111	1331.0337	100.4581	761.7095
P47NB	14.0920	63.8780	5.4810	8.9080	398.5217	1377.8306	434.2378	398.6659
P48B	24.2720	54.2750	2.6200	10.9840	415.6937	1322.4135	147.4033	230.4925
P48NB	19.6630	57.4410	6.4690	9.8920	458.2069	1273.9690	490.0067	268.4340

Table S3. Generalized linear models (GLM) using AICc-based model selection and weigth for bests models for diets types. The DeltaAICc is the difference in AICc values compared with the estimated best model (with lowest AICc), which allows the ranking of models from best (top of the table) to worst. The AICc weight is the estimated probability that a model is the best model in the set. (continua)

id	Response variable	Model structure	cond((Int))	habitat_burn	habitat_nunburn	forest_cover	severity	fire_extent
1	CWM Frugivorous	fire_extent*forest_cover+(1|landscape)				-3.004		-23.813
2	CWV Fugivorous	severity*forest_cover+(1|landscape)				302.51	34.46	
3	CWM Insectivorous	severity + habitat + (1|landscape)	+		9.545		12.03	
4	CWV Insectivorous	fire_extent*forest_cover+(1|landscape)				-327.1		-629.7
5		severity*forest_cover+(1|landscape)				-48.8	42.88	
6	CWM Granivorous	habitat + (1|landscape)	+		-0.26002			
7	CWV Granivorous	habitat+(1|landscape)	+		-0.21663			
8	CWM Nectarivorous	severity*forest_cover+(1|landscape)				-5.215	-6.591	
9	CWV Nectarivorous	severity*forest_cover+(1|landscape)				-432.77	-568.51	

Table S3. Generalized linear models (GLM) using AICc-based model selection and weigth for bests models for diets types. The DeltaAICc is the difference in AICc values compared with the estimated best model (with lowest AICc), which allows the ranking of models from best (top of the table) to worst. The AICc weight is the estimated probability that a model is the best model in the set. (conclusão)

forest_cover:severity	forest_cover:fire_extent	df	logLik	AICc	delta	weight	Residual (Shapiro-Wilk normality test)	*Family*
	29.581	6		162.8	0	0.622	W = 0.95276, p-value = 0.2003	lmer
-343.54		6		348.7	0	0.996	W = 0.97297, p-value = 0.6232	lmer
		5	-86.071	184.6	0	1	W = 0.94805, p-value = 0.1499	lmer
880.2		6		351.1	0	0.661	W = 0.98249, p-value = 0.8871	lmer
-294.94		6		352.9	1.7	0.281	W = 0.95731, p-value = 0.264	lmer
		4		146	0	0.63	W = 0.97986, p-value = 0.8219	Gamma(link = "log")
		4	-179.445	368.5	0	0.731	W = 0.98149, p-value = 0.8636	Gamma(link = "log")
14.209		6		125.9	0	0.9705	W = 0.92195, p-value = 0.03015	lmer
1170.23		6		353.4	0	0.989	W = 0.96102, p-value = 0.3288	lmer