

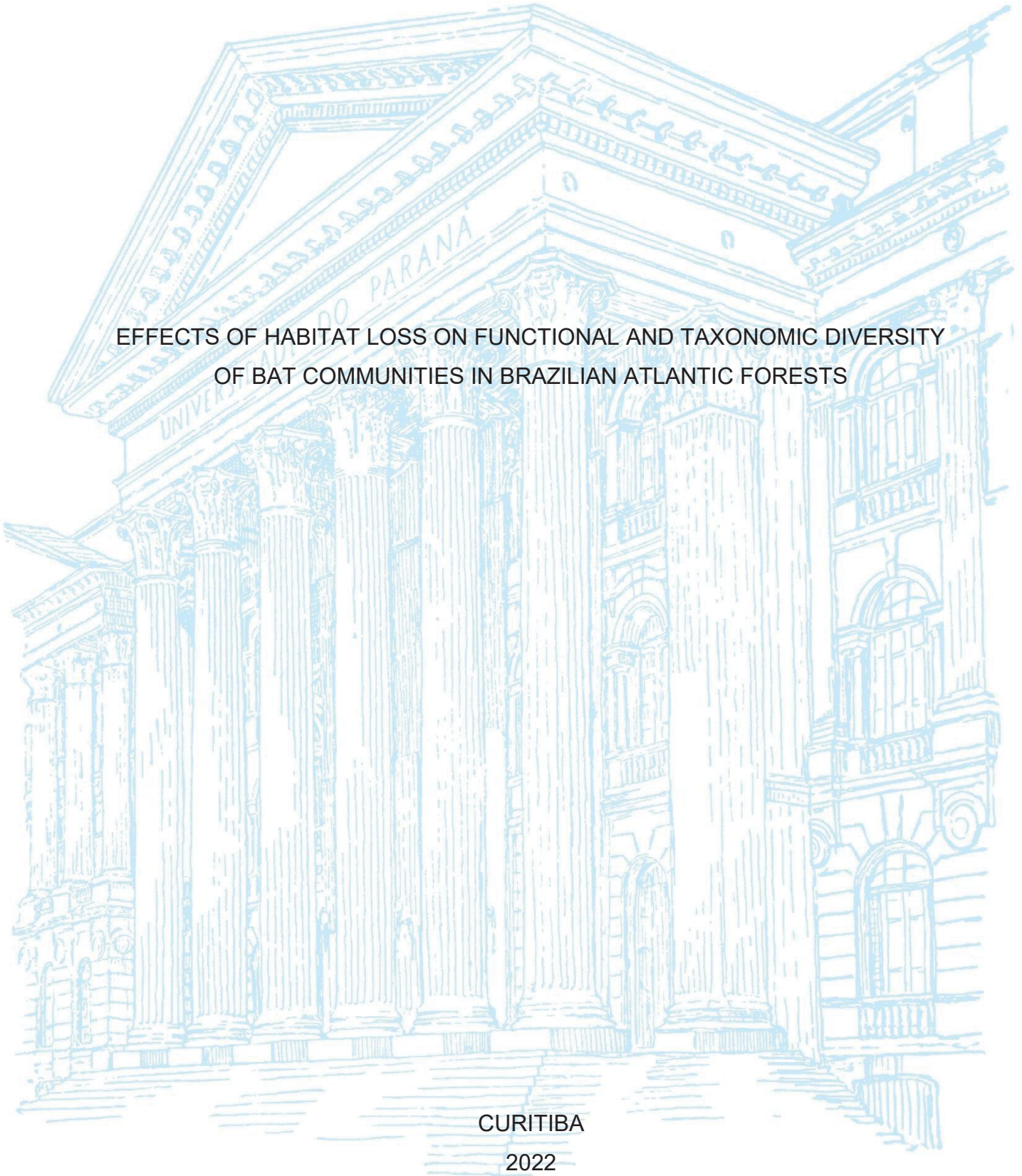
UNIVERSIDADE FEDERAL DO PARANÁ

MARIA FERNANDA FONSECA

EFFECTS OF HABITAT LOSS ON FUNCTIONAL AND TAXONOMIC DIVERSITY
OF BAT COMMUNITIES IN BRAZILIAN ATLANTIC FORESTS

CURITIBA

2022



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OF BAT COMMUNITIES IN BRAZILIAN ATLANTIC FORESTS

Dissertação apresentada ao curso de Pós-Graduação em Ecologia e Conservação, Setor de Ciências Biológicas, Universidade Federal do Paraná, como requisito parcial à obtenção do título de Mestre em Ecologia e Conservação.

Orientador: Prof. Dr. Fernando de Camargo Passos.

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A Deus, meu Senhor e Criador, e aos meus pais e irmã que sempre me incentivaram.

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“Cada descoberta nova da ciência é uma porta nova pela qual encontro mais uma vez Deus, o autor dela.”

— *Albert Einstein*

RESUMO

Perda e fragmentação do *habitat* estão entre as principais causas da perda de biodiversidade no planeta, sendo um tema de pesquisa importante para a conservação. No entanto, a maioria dos estudos sobre este tema tem focado apenas nos efeitos sobre riqueza, abundância e composição de espécies das comunidades, negligenciando os impactos da perda do *habitat* sobre a dimensão funcional da biodiversidade. O presente estudo objetivou analisar os efeitos do tamanho da área sobre a diversidade taxonômica e funcional de comunidades de morcegos da Mata Atlântica no Brasil. Os dados foram obtidos através de uma revisão de literatura, na qual foi coletada informações sobre riqueza e composição de espécies, abundância, tamanho da área e atributos funcionais das espécies de morcego. Foram calculadas as relações entre área e diferentes componentes da diversidade taxonômica (riqueza de espécies, índice de diversidade de Shannon e índice de equabilidade de Pielou) e diversidade funcional (riqueza funcional, equabilidade funcional e divergência funcional). Como esperado, as diversidades taxonômica e funcional foram afetadas pelo tamanho da área. Modelos lineares generalizados mostraram que riqueza de espécies, diversidade de Shannon, riqueza funcional e equabilidade funcional aumentaram com a área, enquanto o índice de Pielou e divergência funcional não foram afetados pela área. Além disso, foi observado que áreas menores que 100 ha parecem limitar a ocorrência de algumas espécies ameaçadas. Os resultados deste estudo estão de acordo com a relação espécie-área e dão suporte a relação diversidade funcional-área, indicando um efeito negativo da perda de *habitat* nas comunidades de morcegos da Mata Atlântica brasileira. Estes resultados destacam a importância de analisar a dimensão funcional da diversidade para alcançar uma compreensão mais completa das respostas da biodiversidade à transformação do *habitat*, e a importância de proteger grandes áreas de Mata Atlântica (> 100 ha) para garantir a permanência de espécies de morcegos ameaçadas e uma comunidade de morcegos diversa.

Palavras-chave: Chiroptera. Biologia da conservação. Área. Atributos funcionais. Relação espécie-área. Relação diversidade funcional-área.

ABSTRACT

Habitat loss and fragmentation is one of the greatest causes of biodiversity loss in the planet, representing an important research topic in conservation. However, most studies on this topic have focused only on the effects on richness, abundance and species composition of communities, missing the impacts of habitat loss on functional dimension of biodiversity. In this study we aimed to analyze the effects of the area size on taxonomic and functional diversity of bat communities of the Atlantic Forest in Brazil. We obtained data through a literature review, getting information on species richness and composition, abundance, area size and functional traits. We calculated the relationship between area and different components of taxonomic diversity (species richness, Shannon's diversity index and Pielou's evenness index) and functional diversity (functional richness, functional evenness and functional divergence). As expected, taxonomic and functional diversity was affected by area. Generalized linear models show that species richness, Shannon diversity, functional richness and functional evenness increased with area size, while Pielou's index and functional divergence were not affected by area. Furthermore, we found that areas smaller than 100 ha appeared to limit the occurrence of some threatened species. Our results are in accordance with the species - area relationship and supports the functional diversity - area relationship, indicating a negative effect of habitat loss on bat communities in the Brazilian Atlantic Forest. These results highlight the importance of analyzing the functional dimension of diversity to assess a more complete comprehension of biodiversity responses to habitat transformation, and the importance of protecting large areas of Atlantic Forest (> 100 ha) to ensure the persistence of threatened bat species and a diverse bat community.

Keywords: Chiroptera. Conservation biology. Area size. Functional traits. Species-area relationship. Functional diversity-area relationship.

LISTA DE FIGURAS

FIGURE 1 – GEOGRAFIC DISTRIBUTION OF STUDIES	27
FIGURE 2 – OCCURENCE OF THREATENED AND NOT-THREATENED SPECIES ACCORDING WITH AREA SIZE.....	27
FIGURE 3 – OCCURENCE OF THREATENED SPECIES ACCORDING WITH AREA SIZE.....	28
FIGURE 4 – RELATION BETWEEN AREA AND DIVERSITY PARAMETERS.....	31

LISTA DE TABELAS

TABLE 1 – FUNCTIONAL TRAITS	24
TABLE 2 – MODEL SELECTION TABLE	29
TABLE 3 – SUMMARY OF GLM RESULTS	30

SUMMARY

	INTRODUÇÃO GERAL	13
1	INTRODUCTION	18
2	MATERIAL AND METHODS	22
2.1	Data collection.....	22
2.2	Species traits.....	23
2.3	Data analysis.....	24
3	RESULTS	26
4	DISCUSSION	32
5	CONCLUSION	37
	CONCLUSÃO	38
	REFERENCES	39
	SUPPORTING INFORMATION	47
	S1 TABLE	47
	S2 TABLE.....	59

INTRODUÇÃO GERAL

A perda e a fragmentação do habitat têm sido apontadas como as principais causas do declínio da biodiversidade global e da degradação ambiental (PEREIRA et al., 2010; HADDAD et al., 2015; MEYER; STRUEBIG; WILLIG, 2016), tornando-se uma das principais preocupações em biologia de conservação. A conversão de habitats naturais em terras agrícolas tem sido o principal agente de transformação d habitat, e os cenários de biodiversidade global indicam que a área florestal continuará a ser reduzida nas próximas décadas (SALA et al., 2000; PEREIRA et al., 2010). A fragmentação do habitat, uma das consequências da perda do habitat, pode reduzir a biodiversidade em até 75% e afetar as funções do ecossistema (HADDAD et al., 2015). Embora a fragmentação possa ocorrer naturalmente no ambiente – como resultado de diferenças de topografia, solo e microclima levando muitas vezes a uma maior biodiversidade nessas áreas (SOUZA et al., 2019) – a fragmentação antrópica apresenta vários efeitos negativos. Algumas atividades humanas ocorrem em áreas amplas com frequência relativamente alta, removendo habitats naturais e sua complexidade, e as espécies não têm tempo suficiente para se adaptar às novas condições ambientais, podendo se extinguir localmente (COLLINGE, 2009).

As ameaças apresentadas pela perda e fragmentação do habitat são especialmente preocupantes em áreas de hotspots de biodiversidade, que contêm 44% de todas as plantas e 35% de todas as espécies de vertebrados em todo o mundo (MYERS et al., 2000). A Mata Atlântica, um desses hotspots, é um dos biomas mais degradados do planeta e o bioma mais desmatado do Brasil, com mudanças no habitat acontecendo em ritmo muito acelerado (MYERS et al., 2000; GALINDO-LEAL; CÂMARA, 2005; RIBEIRO et al., 2009). Considerando sua cobertura original de aproximadamente 150 milhões de hectares, a Mata Atlântica era uma das maiores florestas tropicais das Américas. No entanto, resta apenas 11,7% da vegetação original, com a maior parte deste remanescente existindo como pequenos fragmentos (< 50 ha) e floresta secundária. Este alto nível de perda e fragmentação do habitat é uma grande ameaça à biodiversidade atlântica. Ainda assim, pequenos fragmentos não devem ser negligenciados no esforço de conservação, pois esses fragmentos constituem 83,4% do remanescente florestal (RIBEIRO et al., 2009).

A relação espécie-área (SAR) é uma das poucas leis universais em ecologia, sendo amplamente documentada (LOMOLINO, 2000). De acordo com esta lei, a diversidade de espécies aumenta com o aumento da área, e devido ao crescente nível de redução do habitat em todo o mundo, desde 1980 a relação espécie-área tem sido amplamente utilizada na biologia da

conservação para determinar o tamanho ideal da área de reservas naturais e prever extinções decorrentes da perda de habitat (CONOR; MCCOY, 2013; FATTORINI; ULRICH; MATTHEWS, 2021). A SAR foi muito importante no desenvolvimento da teoria do equilíbrio da biogeografia de ilhas por MacArthur e Wilson (1967) (TJØRVE; MATTHEWS; WHITTAKER, 2021), que propõe que a riqueza de espécies em cada ilha é determinada pelo equilíbrio entre extinção e colonização dependendo da área e do isolamento.

Embora um esforço considerável tenha sido feito para entender os efeitos da perda e fragmentação do habitat sobre a biodiversidade, a maioria dos estudos se concentrou exclusivamente na dimensão taxonômica da diversidade (FAHRIG, 2003; BERNARD; FENTON, 2007; SILVA, 2018). As medições clássicas da biodiversidade, que incluem riqueza e índices de diversidade de espécies, assumem que todas as espécies contribuem de forma semelhante para o funcionamento do ecossistema, fornecendo assim uma perspectiva incompleta da complexidade da biodiversidade e uma compreensão enganosa das consequências da mudança do habitat (CISNEROS; FAGAN; WILLIG, 2014). Do ponto de vista funcional, comunidades com riqueza de espécies semelhante podem apresentar diversidade funcional diferente. Por isso é importante considerar a dimensão funcional da diversidade, pois ela quantifica a distribuição dos atributos funcionais em uma comunidade (VILLÉGER; MASON; MOUILLOT, 2008). Atributos funcionais são características morfológicas ou fisiológicas que influenciam diretamente o desempenho individual (VIOLE et al., 2007), sendo importantes impulsionadores do funcionamento do ecossistema. Assim, a perda de biodiversidade pode levar à perda de diversidade funcional, o que potencialmente afeta o funcionamento do ecossistema (MAYFIELD et al., 2010).

As pesquisas sobre diversidade funcional têm crescido grandemente nas últimas décadas, resultando em uma variedade de definições e formas de medi-la (PETCHEY; GASTON, 2006). Devido à complexidade da dimensão funcional vários índices foram propostos, mas não houve consenso sobre quais índices quantificar para acessar a diversidade funcional de uma comunidade (VILLÉGER; MASON; MOUILLOT, 2008; MOUCHET et al., 2010). A maioria desses índices são altamente correlacionados e podem ser influenciados pela riqueza de espécies (MOUCHET et al., 2010). De fato, não existe um índice único para medir a diversidade funcional como um todo (MASON et al., 2005). Da mesma forma que a diversidade taxonômica, a diversidade funcional possui diferentes componentes, e cada um deles é medido separadamente. Assim, em analogia aos componentes da diversidade de espécies, Mason et al. (2005) propuseram que a diversidade funcional pode ser decomposta em três componentes primários: riqueza funcional, equabilidade funcional e divergência funcional.

Visto que a maioria dos índices quantifica apenas uma dessas facetas da diversidade funcional, existem alguns critérios a serem seguidos ao escolher os melhores índices para avaliar a diversidade funcional: os índices devem ser capazes de incorporar vários atributos, avaliar atributos funcionais ponderados pela abundância de espécies e, juntos, devem medir todas as facetas da diversidade funcional (VILLÉGER; MASON; MOUILLOT, 2008; MOUCHET et al., 2010). Villéger, Mason e Mouillot (2008) propuseram um conjunto de três índices que atendem a todos esses critérios: FRic (riqueza funcional) que mede o volume do espaço funcional ocupado pela comunidade, FEve (equabilidade funcional) que mede quão uniformemente as abundâncias das espécies estão distribuídas neste volume, e FDiv (divergência funcional) que mede quão distantes as espécies mais abundantes estão do centro do espaço funcional. Essas medidas multivariadas são complementares e independentes umas das outras. Além disso, FDiv e FEve são independentes da riqueza de espécies.

Diferentemente, a riqueza funcional está fortemente e positivamente relacionada à riqueza de espécies. À medida que o número de espécies aumenta, mais atributos diferentes são adicionados ao espaço funcional. No entanto, essa correlação não é linear, pois as novas espécies adicionadas eventualmente apresentarão atributos redundantes (VILLÉGER; MASON; MOUILLOT, 2008). Sendo assim, podemos supor intuitivamente que a relação diversidade funcional-área (FDAR) é semelhante à relação espécie-área (SAR), que afirma que a riqueza de espécies aumenta à medida que a área aumenta. Existem poucos estudos avaliando a relação diversidade funcional-área e os resultados são variáveis, dependendo das métricas utilizadas. Wang et al. (2013), por exemplo, relataram uma forte relação positiva entre área e riqueza funcional de comunidades arbóreas em florestas temperadas, enquanto Karadimou et al. (2016) encontraram padrões diferentes, também para comunidades vegetais, dependendo da faceta da diversidade funcional analisada. A compreensão dessas relações (SAR e FDAR) em diferentes táxons pode fornecer informações adicionais importantes para uma ampla gama de estratégias de gestão ambiental.

Vários estudos têm explorado como a perda e a fragmentação da floresta tropical afetam a riqueza e a abundância de morcegos (por exemplo, BERNARD; FENTON, 2007; MEYER; KALKO, 2008; ALURRALDE; DÍAZ, 2021). No entanto, poucos estudos analisaram as consequências sobre a diversidade funcional (por exemplo, CISNEROS; FAGAN; WILLIG, 2014; GARCÍA-MORALES et al., 2016; FARNEDA; MEYER; GRELE, 2020). Os morcegos são um dos táxons mais importantes ecologicamente entre os mamíferos. Com 1.451 espécies reconhecidas em todo o mundo, elas representam quase um quinto da diversidade de mamíferos na Terra (FRICK; KINGSTON; FLANDERS, 2019; SIMMONS;

CIRRANELLO, 2021). A alta diversidade e abundância dos morcegos neotropicais os tornam importantes componentes da fauna tropical. Eles apresentam uma grande variedade de características morfológicas, uso do habitat, hábitos alimentares e nichos tróficos, que se refletem nos múltiplos papéis ecológicos que eles desempenham no ecossistema, como polinização, dispersão de sementes e controle de insetos (KALKO; HANDLEY; HANDLEY, 1996; MUSCARELLA; FLEMING, 2007; KALKA; SMITH; KALKO, 2008; KUNZ et al., 2011). Apesar da alta mobilidade dos morcegos, com potencial para voar longas distâncias e entre manchas florestais, sua diversidade de atributos funcionais sugere diferentes vulnerabilidades à perturbação do habitat entre as espécies, tornando-os um ótimo táxon modelo para avaliar as respostas à modificação do habitat (MEYER; STRUEBIG; WILLIG, 2016). Além disso, os morcegos são considerados bons bioindicadores, pois respondem a várias mudanças de habitat induzidas pelo homem, como perda e fragmentação do habitat (JONES et al., 2009).

Alguns estudos apontaram a perda do habitat como o principal processo, após o isolamento, levando a resposta de morcegos filostomídeos à fragmentação (por exemplo, MEYER; KALKO, 2008), enquanto outros estudos não encontraram relação positiva entre a riqueza de espécies e o tamanho da floresta per se (por exemplo, FARIA, 2006). Considerando a relação pouco clara entre riqueza de espécies de morcegos e área, e a relação ainda não avaliada entre área e diversidade funcional de morcegos, no presente estudo avaliou-se a resposta taxonômica e funcional de comunidades de morcegos à perda do habitat no bioma Mata Atlântica no Brasil, baseado na premissa da relação espécie-área e da relação diversidade funcional-área. Como a diversidade funcional contabiliza alguns aspectos da biodiversidade negligenciados por medidas clássicas, avaliamos a influência da área em diferentes componentes da diversidade taxonômica (riqueza de espécies, índice de diversidade de Shannon e índice de equabilidade de Pielou) e diversidade funcional (FRic, FEve e FDiv).

Esperava-se que a diversidade taxonômica e funcional fosse afetada negativamente pela redução da área. Especificamente, esperava-se encontrar 1) relação linear positiva entre área e riqueza, bem como 2) entre área e diversidade de Shannon. 3) Para o índice de Pielou esperava-se uma relação negativa com a área, pois em áreas maiores há mais chance de se amostrar espécies raras, reduzindo a equabilidade da distribuição das abundâncias das espécies. Para os componentes da diversidade funcional, esperava-se 4) uma relação linear positiva entre área e riqueza funcional (FRic), uma vez que este índice está muito correlacionado com a riqueza de espécies e quanto mais espécies são incluídas, mais atributos são adicionados no espaço funcional. 5) Para a equabilidade funcional (FEve) esperava-se uma relação negativa

com a área, pois com o aumento da área, maior é a probabilidade de capturar espécies com atributos únicos, e espécies redundantes se tornam mais abundantes, diminuindo assim a equabilidade no espaço funcional. Por fim, esperava-se 6) um aumento da divergência funcional (FDiv) à medida que a área aumentasse, uma vez que áreas maiores têm maior probabilidade de ter mais habitats, e assim, a probabilidade de aumentar a abundância de espécies com atributos únicos é maior. Além de compreender como a perda do habitat afeta a diversidade de morcegos, pretende-se com este trabalho encontrar uma área mínima na qual a diversidade de morcegos possa ser mantida.

1 INTRODUCTION

Habitat loss and fragmentation have been identified as the main causes of decline in global biodiversity and environmental degradation (PEREIRA et al., 2010; HADDAD et al., 2015; MEYER; STRUEBIG; WILLIG, 2016), becoming one of the main concerns in conservation biology. The conversion of natural habitats to agricultural lands has been the main agent of habitat transformation, and global biodiversity scenarios indicate that forest area will continue to be reduced in the coming decades (SALA et al., 2000; PEREIRA et al., 2010). Habitat fragmentation, one of the consequences of habitat loss, can reduce biodiversity by up to 75% and affect ecosystem functions (HADDAD et al., 2015). Although fragmentation can happen naturally in the environment – as result of differences in topography, soil and microclimate leading many times to greater biodiversity in these areas (SOUZA et al., 2019) – anthropic fragmentation presents several negative effects. Some human activities occur in broad areas with relatively high frequency, removing natural habitats and their complexity, and species have no enough time to adapt to the new environmental conditions, and may become locally extinct (COLLINGE, 2009).

The threats presented by habitat loss and fragmentation are specially worrying in biodiversity hotspot areas, which contain 44% of all plant and 35% of all vertebrate species world-wide (MYERS et al., 2000). The Atlantic Forest, one of these hotspots, is one of the most degraded biomes in the planet and the most deforested biome in Brazil, with habitat changes happening in a very fast pace (MYERS et al., 2000; GALINDO-LEAL; CÂMARA, 2005; RIBEIRO et al., 2009). Considering its original coverage of approximately 150 million hectares, the Atlantic Forest was one of the largest tropical forests in the Americas. However, there is only 11.7% of the original vegetation left, with most of this remnant existing as small fragments (< 50 ha) and secondary forest. This high level of habitat loss and fragmentation is a great threat to Atlantic biodiversity. Yet, small fragments should not be neglected in conservation effort, since these fragments constitutes 83.4% of the forest remnant (RIBEIRO et al., 2009).

The species-area relationship (SAR) is one of the few universal laws in ecology, being largely documented (LOMOLINO, 2000). According to this law, species diversity increases with increasing area, and due to crescent level of habitat reduction around the world, since 1980 the species-area relationship has been largely used in conservation biology to determine the optimal area size of nature reserves and to predict extinctions resulting from habitat loss (CONOR; MCCOY, 2013; FATTORINI; ULRICH; MATTHEWS, 2021). The SAR was very

important on the development of the equilibrium theory of island biogeography by MacArthur and Wilson (1967) (TJØRVE; MATTHEWS; WHITTAKER, 2021), which proposes that the species richness in each island is determined by the equilibrium between extinction and colonization depending on area and isolation.

Although considerable effort has been done to understand the effects of habitat loss and fragmentation on biodiversity, most of the studies have focused exclusively on the taxonomic dimension of diversity (FAHRIG, 2003; BERNARD; FENTON, 2007; SILVA, 2018). The classical biodiversity measurements, that include species richness and species diversity indices, assume that all species contribute similarly to ecosystem functioning, thus providing an incomplete perspective of the complexity of biodiversity and a misleading comprehension of the consequences of habitat change (CISNEROS; FAGAN; WILLIG, 2014). From a functional perspective, communities with similar species richness can present different functional diversity. This is why is important to consider the functional dimension of diversity, as it quantifies the distribution of functional traits in an assemblage (VILLÉGER; MASON; MOUILLOT, 2008). Functional traits are morphological or physiological characteristics that directly influence individual performance (VIOLLE et al., 2007), being important drivers of ecosystem functioning. Hence, biodiversity loss may lead to loss of functional diversity, which potentially affects ecosystem functioning (MAYFIELD et al., 2010).

Research on functional diversity has grown enormously over the past few decades, resulting in a variety of definitions and ways of measuring it (PETCHEY; GASTON, 2006). Due to the complexity of functional dimension several indices were proposed, but there was no consensus on which indices quantify to access functional diversity of a community (VILLÉGER; MASON; MOUILLOT, 2008; MOUCHET et al., 2010). Most of these indices are highly correlated and may be influenced by species richness (MOUCHET et al., 2010). Indeed, there is not a unique index to measure functional diversity as a whole (MASON et al., 2005). The same way as taxonomic diversity, functional diversity has different components, and each of them are measured separately. Thus, in analogy with the components of species diversity, Mason et al. (2005) proposed that functional diversity can be decomposed in three primary components: functional richness, functional evenness and functional divergence.

Since most indices quantify only one of these facets of functional diversity, there are some criteria to follow when choosing the best indices to evaluate functional diversity: the indices must be able to incorporate multiple traits, evaluate functional traits weighted by species abundances, and together, they must measure all the facets of functional diversity (VILLÉGER; MASON; MOUILLOT, 2008; MOUCHET et al., 2010). Villéger, Mason and Mouillot (2008)

proposed a set of three indices that meets all of these criteria: FRic (functional richness) that measures the volume of functional space occupied by the community, FEve (functional evenness) that measures how evenly species abundances are distributed in this volume, and FDiv (functional divergence) that measures how distant the most abundant species are from the center of functional space. These multivariate measurements are complementary and independent of each other. Moreover, FDiv and FEve are independent of species richness.

Differently, functional richness is strongly and positively related to species richness. As the number of species increases, more different traits are added to the functional space. However, this correlation is not linear, since the new species will eventually present redundant traits (VILLÉGER; MASON; MOUILLOT, 2008). That being so, we can intuitively assume that the functional diversity-area relationship (FDAR) is similar to the species-area relationship (SAR), which states that species richness increases as area increases. There are few studies evaluating functional diversity-area relationship and results are variable, depending on the metrics used. Wang et al. (2013) for example have reported a strong positive relationship between area and functional richness of tree communities in temperate forests, while Karadimou et al. (2016) found different patterns, also for plant communities, depending on the facet of functional diversity analyzed. The understanding of these relationships (SAR and FDAR) in different taxa may provide important additional information for a wide range of environmental management strategies.

Several studies have explored how tropical forest loss and fragmentation affects richness and abundance of bats (e.g. BERNARD; FENTON, 2007; MEYER; KALKO, 2008; ALURRALDE; DÍAZ, 2021). However, only a few studies have analyzed the consequences on functional diversity (e.g. CISNEROS; FAGAN; WILLIG, 2014; GARCÍA-MORALES et al., 2016; FARNEDA; MEYER; GRELE, 2020). Bats are one of the most ecologically important taxa among mammals. With 1,451 recognized species world-wide, they represent nearly a fifth of mammalian diversity on Earth (FRICK; KINGSTON; FLANDERS, 2019; SIMMONS; CIRRANELLO, 2021). Neotropical bats high diversity and abundance makes them important components of tropical fauna. They display a variety of morphological traits, habitat use, feeding habits and trophic niches, which are reflected in the multiple ecological roles they play in the ecosystem, such as pollination, seed dispersal and insect control (KALKO; HANDLEY; HANDLEY, 1996; MUSCARELLA; FLEMING, 2007; KALKA; SMITH; KALKO, 2008; KUNZ et al., 2011). In spite of bats high mobility, with potential to fly over long distances and among forest patches, their diversity of functional traits suggests different vulnerability to habitat disturbance among species, making them a great model taxon for evaluating responses

to habitat modification (MEYER; STRUEBIG; WILLIG, 2016). Furthermore, bats are considered good bioindicators as they respond to several human-induced habitat changes, such as habitat loss and fragmentation (JONES et al., 2009).

Some studies have pointed habitat loss as the main process after isolation underlying phyllostomid bat response to fragmentation (e.g. MEYER; KALKO, 2008), while others found no positive relation between species richness and forest size per se (e.g. FARIA, 2006). Considering the unclear relationship between bat species richness and area, and the still not evaluated relationship between area and functional diversity of bats, in the present study we assessed the taxonomic and functional response of bat assemblages to habitat loss in the Atlantic Forest biome in Brazil, based on the premise of species-area relationship and functional diversity-area relationship. As functional diversity accounts for some aspects of biodiversity overlooked by classical measures, we evaluated the influence of area on different components of taxonomic diversity (Species richness, Shannon diversity index, and Pielou evenness index) and functional diversity (FRic, FEve and FDiv).

We expected that taxonomic and functional diversity would be negatively affected by area reduction. Specifically, we expected 1) positive linear relation between area and richness as well as 2) between area and Shannon diversity. 3) For Pielou index we expected a negative relation to area, since in greater areas there is more chance of sampling rare species, reducing the equability of distribution of species abundances. For the components of functional diversity, we expected 4) positive linear relation between area and functional richness (FRic), since this index is very correlated with species richness and the more species are included, more traits are added in the functional space. 5) For functional evenness (FEve) we expected a negative relationship with area, since with the increase in area, greater the probability of sampling species with unique traits and redundant species become more abundant, decreasing evenness in functional space. Finally, we expected 6) increase in functional divergence (FDiv) as area increases, since greater areas have more probability of having more habitats, thus the probability of increasing the abundance of species with unique traits is higher. Besides comprehending how habitat loss affects bat diversity, we aim with this work to find a minimal area at which bats diversity can be maintained.

2 MATERIAL AND METHODS

2.1 DATA COLLECTION

To obtain data on the composition of bat communities in the Atlantic Forest in Brazil we collected information from the Atlantic Bats dataset (MUYLAERT et al., 2017) – a compilation of information on bat communities from the Atlantic Forests of South America, and through a comprehensive search in Google Scholar, Web of Science and Scielo online databases (accessed in 2020). In these online databases we performed a topic search using combinations of the keywords: “bat”, “diversity”, “community”, “assemblage”, “tropical forest”, “Atlantic Forest” and “Brazil”. The search was made in English and Portuguese and no restriction to publication year was used. The publications identified in the search were subsequently screened for suitability for the analysis based on the article's title, abstract, and text. In addition, the publications were selected based on the following criteria: 1) thesis and studies published in an indexed, peer-reviewed scientific journal; 2) carried out in the Atlantic Forest biome in Brazil; that provided 3) a list of bat species with capture number, and 4) information on area size and sampling effort. Furthermore, to avoid biases in the results, we selected only studies 5) with a sampling period of at least six months, and 6) that used ground-level mistnets as a method of capturing bats. The data from the Atlantic Bats dataset were filtered following the same criteria, with exception of some data that was obtained from not published thesis.

From the selected publications we elaborated a dataset containing the name of the authors, year of publication, size of study area in hectares, geographic coordinates, list of species and abundances, sampling effort, and species classification according to the red list of the Brazilian threatened species (ICMBio, 2018).

We followed the taxonomic arrangement of Nogueira et al. (2014) and the updated checklist of Brazilian bats (GARBITO et al., 2020) with the following changes in name of species: 1) we used *Gardnerycteris crenulatum* instead of *Mimon crenulatum*, following Hurtado and Pacheco (2014); 2) specimens of *Myotis dinellii* from Brazil are now recognized as *Myotis albescens* (MORATELLI et al., 2019); 3) the subspecies of *Tonatia saurophila* occurring in Brazil was elevated to species *Tonatia maresi*, so all specimens of *T. saurophila* collected in Brazil were considered *T. maresi* (BASANTES et al., 2020). We also considered the species *Choeroniscus minor* as *Dryadonycteris capixaba* in the Atlantic Forest, since *C. minor* was assumed to be restricted to the Amazon basin (ROCHA et al., 2014), and

Lonchophylla bokermanni was considered *Lonchophylla peracchii*, since *L. bokermanni* is restricted to Cerrado (DIAS; ESBÉRARD; MORATELLI, 2013). Specimens identified by the authors at the generic level were not included in the database (e.g. *Artibeus* sp.).

2.2 SPECIES TRAITS

Depending on the specific aims of a particular study, sets of different traits can be chosen for a particular measure of functional diversity (PETCHEY; GASTON, 2006; CASTILLO-FIGUEROA; PÉREZ-TORRES, 2021). To represent quantitatively bats functional diversity, we used three species traits that are associated with the ecological differentiation of the Neotropical bat community (GARCÍA-MORALES et al., 2016; SORIANO, 2000): 1) weight, based on the average body weight of each species, 2) size, measured as mean forearm length, and 3) diet, based on the main food item of each species (Table 1). Diet and body mass reflect the type and quantity of resources consumed by a species, and body size is associated with population size, life history and home range. Thus, trophic habit and animal body size are relevant parameters of the niche, reflecting the different ecological processes each bat species participates in (CHILLO; OJEDA, 2012; MEYER et al., 2008; SORIANO, 2000).

In bats, wing morphology provides a general view of their ecology and behavior, as it is an important predictor of foraging habitat and strategy, dispersal ability and home range (NORBERG; RAYNER, 1987; ARITA; FENTON, 1997). However, due to scarcity of information on wing morphology data, such as aspect ratio and wing load, for most of the species in this study (~70% of the species), we did not consider this trait in the analysis. So, in order to bring more information on the niche of the species, we considered two categories of insectivorous bats: aerial insectivores, which forage preferably in uncluttered areas capturing their preys while flying, and gleaning insectivores, which are more well adapted to narrow habitats, gathering their food on leaves or on the ground (CASTILLO-FIGUEROA; PÉREZ-TORRES, 2021; SORIANO, 2000). Therefore, in regard to diet the bats in this study were classified in eight categories: frugivores, nectarivores, aerial insectivores, gleaning insectivores, omnivores, carnivores, piscivores, and sanguinivores.

Most of the trait information was obtained from the book “Morcegos do Brasil” (REIS et al., 2007), that contain overall information on the bat species occurring in Brazil, and from the Ecological Register database (www.ecoregister.org, accessed on June, 2021), a repository of published ecological data of plant and animal species around the world (ALROY, 2019). The

information that was not found on these sources was obtained from Encyclopedia of Life (eol.org, accessed on July, 2021), that provides global access to knowledge about life on Earth, Atlantic Mammal traits data set (GONÇALVES et al., 2018) and Elton traits data paper (WILMAN et al., 2014). The body weight and forearm values for each species were set as a mean of the values obtained from all of the sources. The information of species' traits not available on the literature was inferred based on other species of the genus.

Table 1 Functional traits used to determine the functional diversity of bat communities.

Functional trait	Attribute	Data type	Value
Weight	Body weight	Numerical	Mean for the species (g)
Size	Forearm	Numerical	Mean length for the species (mm)
Diet	Main food item	Categorical	Frugivore, nectarivore, aerial insectivore, gleaning insectivore, omnivore, carnivore, piscivores, and sanguivore

2.3 DATA ANALYSIS

We used richness, Shannon diversity index (H') and Pielou evenness index (J') to assess the taxonomic diversity of each community. Shannon's index characterizes the species diversity in a community considering the relative abundance of the present species. Pielou's evenness index (J') is derived from Shannon's index and measures how uniformly the abundance of individuals among species are distributed in the community (PIELOU, 1966). These indices were calculated through the package Vegan (OKSANEN et al., 2020) on R software.

To quantify functional diversity, we chose the independent indices proposed by Vileger, Mason and Mouillot (2008): FRic (functional richness), that corresponds to the volume of functional space occupied by the species in the community; FEve (functional evenness), that quantifies the regularity of the distribution of abundance in functional space; and FDiv (functional divergence), that measures how species diverge in the distribution of abundance in the functional space. These indices are complementary, and account for the three components of functional diversity (MASON et al., 2005; MOUCHET et al., 2010). Moreover, unlike other indices, they can be used for several traits and account for the abundance of species in a given location (VILLEGER; MASON; MUILLOT, 2008).

The ecological distance between species was measured using Gower's coefficient, since this measure is the most suitable one when using various types of values, such as continuous and categorical variables (PAVOINE et al., 2009). The resulting functional dissimilarity matrix was used to calculate the functional diversity indices with the function dbFD of the R package FD (LALIBERTÉ; LEGENDRE; SHIPLEY, 2014).

To test the hypothesis that taxonomic and functional diversity increases with area size, we performed generalized linear models (GLM) for each diversity metric, with each index (richness, H' , J' , FRic, FEve, FDiv) as response variable in function of area size. We assumed Gaussian error distribution for species richness (square root transformed), Shannon, and Pielou indices given the normal distribution of these variables according to the Shapiro-Wilk normality test ($P > 0.05$), and Gaussian distribution for FRic, FEve and FDiv as well. We also evaluated the influence of sampling effort on species number testing for the interaction between area and sampling effort in each model. The best model for each metric was selected using the R package MuMIn (BARTON, 2020). Logarithmic transformations were performed on area and sampling effort to normalize values. All analysis were performed using R v.4.1.1 software (R-Development Core Team, 2021, available from: <https://www.r-project.org>).

3 RESULTS

The literature review resulted in a total of 73 studies carried out in 91 sites of the Atlantic Forest in Brazil (S1 Table), from this total 48 studies were obtained from the Atlantic Bats dataset (MUYLAERT et al., 2017), and 24 from online databases. The sampled sites ranged from 1 to 209,345 ha, and reached 13 of the 17 Brazilian states where the Atlantic Forest occurs (Figure 1). A total of 45,852 individuals from 91 bat species, 49 genera and 7 families were obtained. Among these species, three are classified as vulnerable (*Eptesicus taddeii*, *Lonchorhina aurita* and *Natalus macrourus*) according to the red list of the Brazilian threatened species (ICMBio, 2018). The family Phyllostomidae was the most representative, with 53 species sampled (56.9% of the total species), followed by Vespertilionidae with 16 species (17.2%), Molossidae with 13 (13.9%), Emballonuridae with 6 (6.5%), Noctilionidae and Thyropteridae with 2 species each (2.2%) and Natalidae with only one species (1.1%) (S2 Table).

Threatened species had smaller range of occurrence when compared with not threatened species, being captured in areas from 26 to 1689.73 ha (Figure 2). *Eptesicus taddeii* specifically occurred in areas ranging from 41 to 37,644 ha, *Lonchorhina aurita* in areas from 26 to 41,704 ha, and *Natalus macrourus* was captured in two areas, one with 100 ha and other with 1,689.73 ha. (Figure 3). Therefore, an area of at least 100 ha seems to limit the occurrence of threatened species.

Figure 1 Geographic distribution of the 91 study sites (red dots) included in the analysis. The gray area represents the original Atlantic Forest coverage, and the green area represents its remnants.

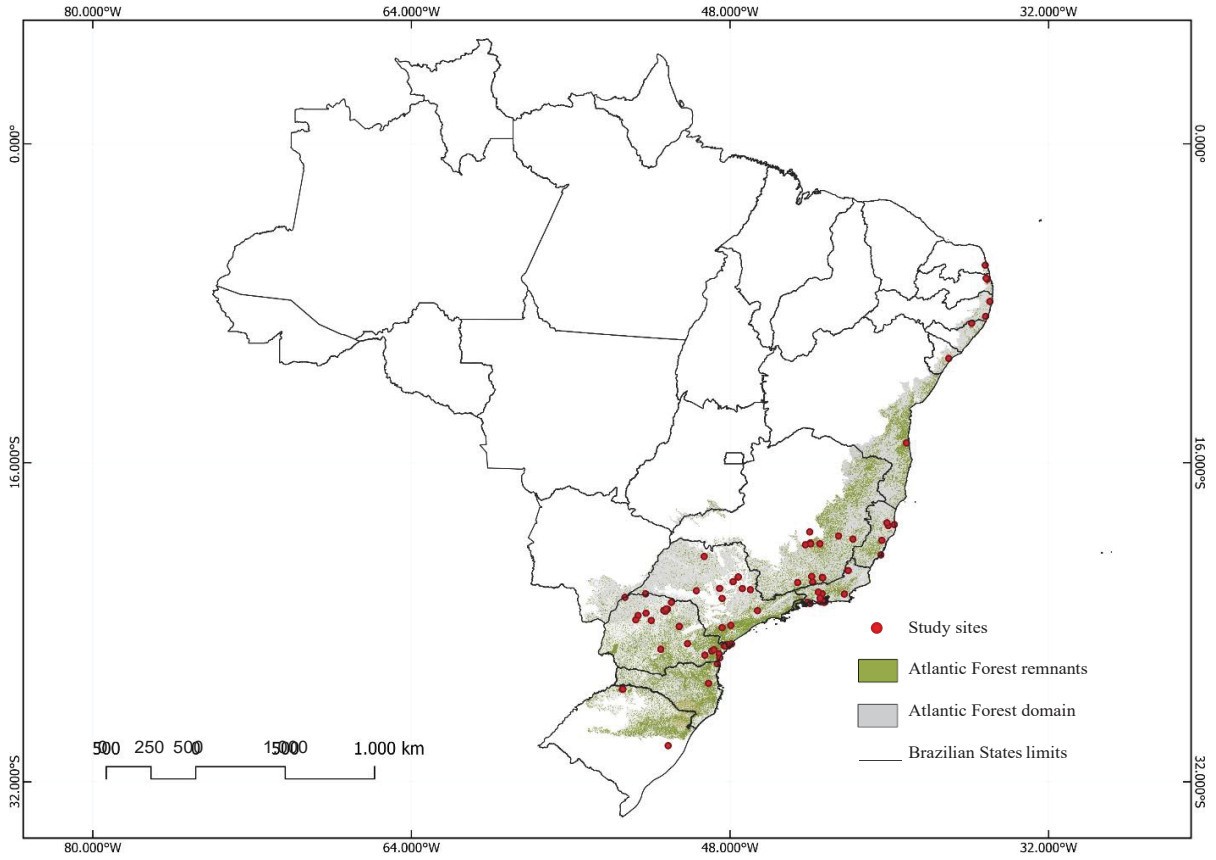


Figure 2 Occurrence of threatened and not-threatened bat species according with area log. Median of not threatened species: 5.4 (231.06 ha). Median of threatened species: 6.3 (573 ha).

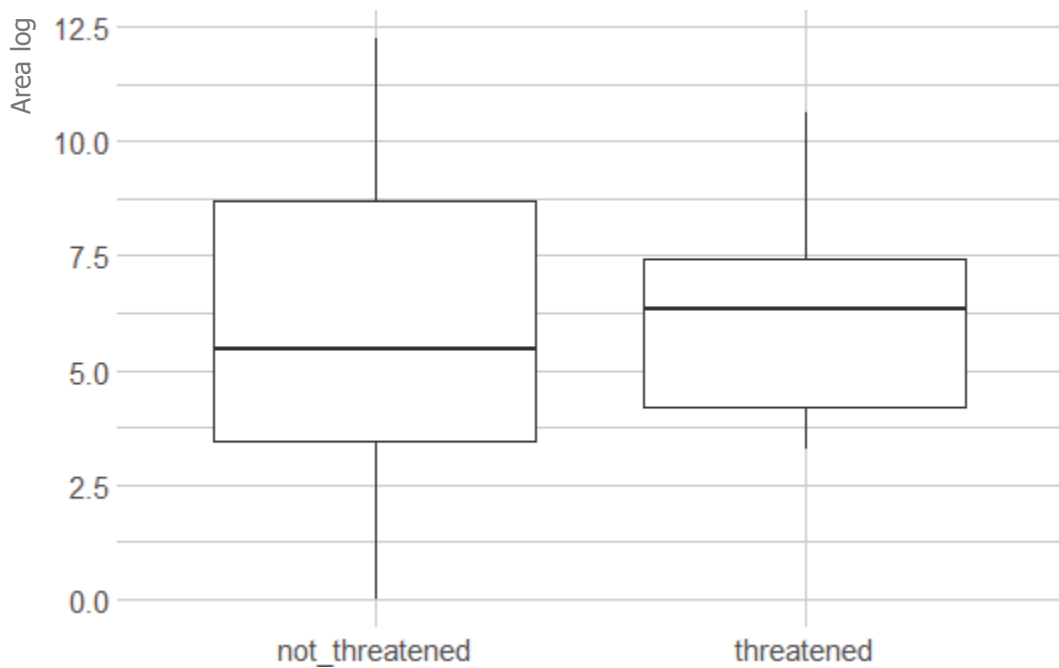
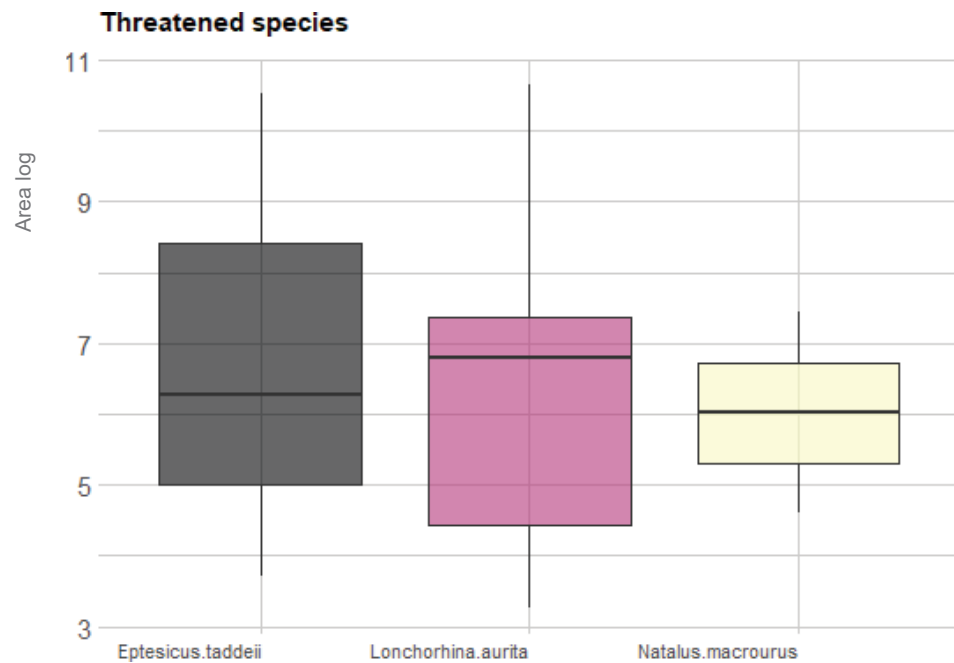


Figure 3 Occurrence of vulnerable bat species (*Eptesicus taddeii*, *Lonchorhina aurita* and *Natalus macrourus*) according with area log.



Overall, the linear models with sampling effort as an explanatory variable provided a better fit (higher weight and lower ΔAIC) to Shannon index, species richness and functional richness metrics when considering only area as explanatory variable. However, the sampling effort term was not significant in any case (Table 2). For Pielou index and FDiv the null model was selected as the best model, indicating that area and sampling effort do not explain the variation in Pielou index and FDiv.

Table 2 Model selection table of Shannon index, species richness and functional richness GLMs. The models tested considered only area as explanatory variable, only sampling effort (except for FRic), area and sampling effort as independent variables and their interaction. The best models selected are the ones with higher weight and $\Delta AIC < 2$ (values in bold). The values of area and sampling effort were log transformed.

Model selection table - Shannon										
	(Int)	area.log	s.effort.log	area.log:s.effort.log	family	df	logLik	AICc	delta	weight
H1	10.320	0.04540	0.03298		gaussian(identity)	4	-44.756	98.1	0.00	0.665
H2	0.9244	0.06621	0.04359	-0.002014	gaussian(identity)	5	-44.746	100.4	2.30	0.210
H3	11.880		0.04098		gaussian(identity)	3	-47.555	101.5	3.35	0.125
H	14.200	0.04464			gaussian(identity)	3	-61.156	128.6	30.46	0.000
H.null	16.600				gaussian(identity)	2	-64.616	133.4	35.24	0.000
Model selection table - Richness										
	(Int)	area.log	s.effort.log	area.log:s.effort.log	family	df	logLik	AICc	delta	weight
S1	2.636	0.11620	0.04141		gaussian(identity)	4	-93.297	195.2	0.00	0.741
S2	2.810	0.08258	0.02426	0.003257	gaussian(identity)	5	-93.291	197.5	2.31	0.233
S3	3.036		0.06191		gaussian(identity)	3	-97.773	201.9	6.70	0.026
S	3.200	0.10700			gaussian(identity)	3	-124.128	254.5	59.32	0.000
S.null	3.776				gaussian(identity)	2	-129.033	262.2	66.99	0.000
Model selection table - FRic										
	(Int)	area.log	s.effort.log	area.log:s.effort.log	family	df	logLik	AICc	delta	weight
fric1	0.3426	0.02212	-0.0002651		gaussian(identity)	4	1.155	6.3	0.00	0.375
fric2	0.8726	-0.08048	-0.0525700	0.009928	gaussian(identity)	5	2.085	6.8	0.46	0.298
fric	0.3561	0.01868			gaussian(identity)	3	-0.404	7.1	0.78	0.254
fric.N	0.4564				gaussian(identity)	2	-2.734	9.6	3.30	0.072

Shannon index ($p = 0.021$), species richness ($p = 0.003$), functional richness ($p = 0.03$) and functional evenness ($p = 0.003$) were positively related with area size (Table 3; Figure 4), corroborating our general hypothesis that diversity increases with area, and supporting the species-area relationship and functional diversity-area relationship.

Table 3 Summary of generalized linear models results. Taxonomic diversity parameters (Shannon index, Pielou index and species richness) and functional diversity parameters (FRic, FEve and FDiv) were tested as response variables, in function of area. Sampling effort was important for some selected models, but had not significant p-value. The values of area and sampling effort were log transformed.

	Estimate	S.E.	t value	Pr(> t)		Estimate	S.E.	t value	Pr(> t)
Shannon					FRic				
Intercept	1.03191	0.25631	4.026	0.000147***	Intercept	0.3425645	0.1330224	2.575	0.0122*
Area	0.04540	0.01922	2.362	0.021090*	Area	0.0221158	0.0099748	2.217	0.0300*
Sampling effort	0.03298	0.02318	1.422	0.159560	Sampling effort	-0.000265	0.0120327	-0.022	0.9825
Null deviance, d.f.				16.650, 69	Null deviance, d.f.				4.2619, 69
Residual deviance, d.f.				14.722, 67	Residual deviance, d.f.				3.9654, 67
AIC				97.512	AIC				5.689
Pielou					FEve				
Intercept	0.620716	0.023972	25.893	<2e-16 ***	Intercept	0.342620	0.027468	12.473	<2e-16***
Area	0.002967	0.003899	0.761	0.449	Area	0.013207	0.004468	2.956	0.00399**
Null deviance, d.f.				1.1026, 90	Null deviance, d.f.				1.5806, 90
Residual deviance, d.f.				1.0958, 89	Residual deviance, d.f.				1.4393, 89
AIC				-137.88	AIC				-113.1
Richness					FDiv				
Intercept	2.63584	0.51278	5.140	2.57e-06***	Intercept	0.806401	0.025931	31.098	<2e-16***
Area	0.11624	0.03845	3.023	0.00354**	Area	-0.002626	0.004218	-0.622	0.535
Sampling effort	0.04141	0.04638	0.893	0.37514	Null deviance, d.f.				1.2883, 90
Null deviance, d.f.				68.563, 69	Residual deviance, d.f.				1.2827, 89
Residual deviance, d.f.				58.924, 67	AIC				-123.58
AIC				194.59					

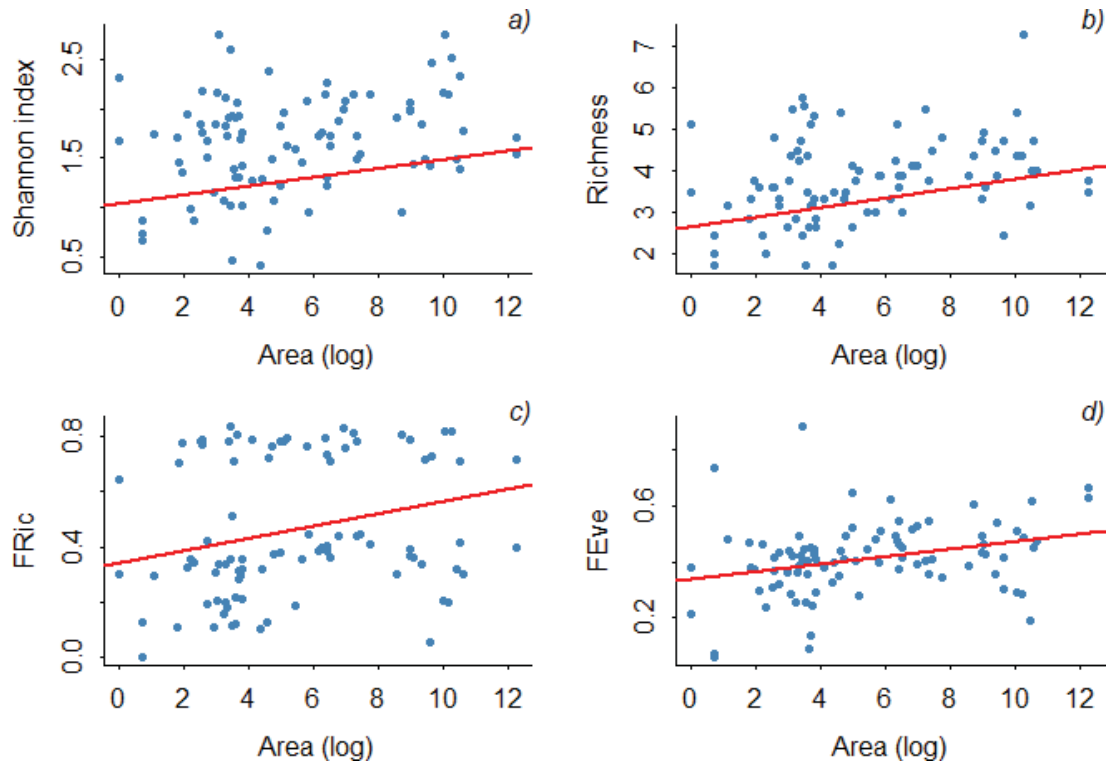
Significance codes

* P<0.05

** P<0.01

*** P<0.001.

Figure 4 Relation between area size and taxonomic diversity parameters – a) Shannon index and b) species richness, and the functional diversity parameters – c) FRic and d) FEve



4 DISCUSSION

In this study the data collection resulted in a good amount of information from bat communities along all the Atlantic Forest extent. However, many studies were not included in our database due to absence of simple information as sampling area size and the number of individuals captured of each species, and for some other studies we had to estimate the sampling effort. Moreover, there are few studies that account for bats morphological traits, for example, data on wing morphology measurements were very difficult to find. In our search, we found only nine studies, including one book chapter, that provided information on wing load and aspect ratio for the species in this study. Even so, these data were missing for 39 of the species, precluding the use of these traits on the analysis. These results bring up the importance of including more detailed sampling information on publications and we encourage more studies using bats functional traits to a better understanding of their ecosystem functions and responses to anthropogenic disruptions.

All of the studies selected used mist nets at ground level – the most common method of capturing bats, sampling mainly species that forage up to 3 m in the forest strata. Besides, some species are able to detect and avoid mist nets, such as active insectivorous bats (LAVAL; FITCH, 1977; TRAJANO, 1984; KALKO; HANDLEY; HANDLEY, 1996). Hence, species of the family Phyllostomidae are better represented in this study, since this family is easily sampled with mist nets. Although non-phyllostomids are not well represented, we included information of all species captured in the studies selected. We understand that this may cause a taxonomic bias on our understanding of the effect of area size on bat communities.

The contribution of sampling effort on species number in models which species richness were relevant (Shannon index, species richness and functional richness) evinces the species-sampling effort relationship, which says species number depends on sampling intensity. Similarly, the species-area relationship predicts an increase in species richness as the area increases (MACARTHUR; WILSON, 1967; CONNOR; MCCOY, 1979; ROSENZWEIG, 1995; AZOVSKY, 2011). Therefore, the interaction between sampling effort and area in this study suggest that these two variables are a better predictor of species richness than the area itself, enlightening the importance of a good sampling design.

Bats are valuable indicators of ecosystem quality, playing important roles in pollination, seed dispersion and insect control (KUNZ et al., 2011). In the past decades, the interest on the consequences of land-use changes on bats has increased in Brazil, mainly on the Atlantic Forest biome (MENDES; SRBEK-ARAÚJO, 2020). Some studies have focused on bat

responses to habitat loss and fragmentation with respect to changes in bat richness and abundance in different biomes (BERNARD; FENTON, 2007; MUYLAERT; STEVENS; RIBEIRO, 2016) and even in the Atlantic Forest (FARIA, 2006), and functional diversity has been used to explore the responses of bat assemblages to habitat loss, fragmentation and land-use transformation in the Neotropical region (FARNEDA et al., 2015; GARCÍA-MORALES et al., 2016; and FARNEDA; MEYER; GRELLE, 2020). Here, we evaluated the effects of habitat loss and fragmentation on taxonomic and functional diversity of bats in the Brazilian Atlantic Forest. Our results reveal that both taxonomic and functional diversity decreases with decreasing area, corroborating with the species-area relationship, one of the most known and documented patterns in ecology (LOMOLINO, 2000), and supporting the functional diversity-area relationship.

Different from Faria (2006) that found no evidence of area effect on bat species richness in Atlantic Forest remnants in north-eastern Brazil, we found that Shannon diversity and species richness of Atlantic bat communities are positively related with area as we expected. In larger areas there is greater probability of having more habitats, resources and abundance of the species using it (CONNOR; MCCOY, 2013), so we expect to have more species per area unit. This species-area relationship has also been observed in studies with small mammals (PARDINI, 2004), medium to large size mammals (SILVA, 2018), and birds (OLIVEIRA, 2018) in fragmented Atlantic Forest landscapes, corroborating the theory of Island Biogeography. In different Neotropical areas, larger fragments harbored greater bat species diversity than smaller ones (REIS; MULLER, 1995; GARCÍA-MORALES et al., 2016). Muylaert, Stevens and Ribeiro (2016) estimated a threshold of 47% of remaining forest needed to maintain bat richness. Below this threshold bat diversity declines. This value is higher than the thresholds found for other groups (30% for small mammals and 30 to 50% for understory birds) suggesting that bats are sensible to forest cover reduction despite being a group with high mobility.

Greater bat richness in smaller areas compared to larger ones was found by Reis et al. (2000), and some studies found no significant effect of area (BERNARD; FENTON, 2007; FARIA, 2006) and disturbance (ALURRALDE; DÍAZ, 2021) on species richness. Therefore, these studies show that not only sites with high percentage of continuous forest, but also small areas of well-preserved primary forest have an irreplaceable conservation value for bats. In general, the quality of the area is more important than the size itself to harbor a high diversity (REIS et al., 2000). In the same way, areas with moderate fragmentation with low contrast good-quality matrix may sustain a high bat species diversity (ESTRADA; COATES-

ESTRADA, 2002; BERNARD; FENTON, 2007; FARNEDA; MEYER; GRELLE, 2020; ALURRALDE; DÍAZ, 2021), indicating the importance of improving matrix quality for bats conservation.

Functional diversity was also affected by area size. Functional richness (FRic) and functional evenness (FEve) were positively related with area, while functional divergence (FDiv) was not. This result suggests that, as found in other studies, the response of functional diversity to habitat loss depends on the functional metrics evaluated (GARCÍA-MORALES et al., 2016; KARADIMOU et al., 2016; OLIVEIRA, 2018), emphasizing the importance of considering multiple indices that accounts for each facet of functional diversity of a community. As greater areas can harbor greater species richness, the same pattern can be inferred for functional richness, since these indices are highly correlated (VILLÉGER; MASON; MOUILLOT, 2008). With the emergence of new habitats, different species with unique traits may colonize the area and occupy new niches, increasing functional richness (KARADIMOU et al., 2016). This pattern was seen in this study, indicating that functional richness would increase with area.

Regarding functional evenness, it can be compared with Pielou index, as it measures the regularity of the distribution of species abundance across the functional space. Therefore, a community with similar distribution of individuals among the same functional group and similar distance between groups presents high FEve (VILLÉGER; MASON; MOUILLOT, 2008). We hypothesized that, as species number increases with area, more rare species are captured (increasing the functional richness) as well as more individuals of common species, resulting in a skewed species abundance distribution (VILLÉGER; MASON; MOUILLOT, 2008; KARADIMOU et al., 2016). However, our results show a different pattern. We found that functional evenness was positively related with area, suggesting that the species added as area increased were functionally redundant. In other words, the new species recorded are more similar to the existing species in the trait space, making the abundance in the functional groups more evenly distributed. With higher regularity in abundance among functional groups, the community becomes more stable against functional loss, since if one species disappears, other species with similar functional role could potentially compensate in abundance the absent species (FONSECA; GANADE, 2001). Different from FEve, Pielou's index did not present significative relation with area.

As species accumulate in the community with increasing area, some traits become dominant. The functional divergence represents how the distance between the most abundant functional types and the center of the functional trait range variates. If the species with extreme

trait values are more abundant in the community, the FDiv value is high and tends to decrease as area increases. Whereas if the species with trait values close to the center of the functional space are the most abundant in the community, the FDiv value is low and tends to increase with area (VILLÉGER; MASON; MOUILLOT, 2008; KARADIMOU et al., 2016). We expected finding greater functional divergence in greater areas, nevertheless we found no significant relation of FDiv with area size in this study. A possible prevalence of the same dominant species and their abundances in most of the sampled areas may explain this result. For example, the species *Artibeus lituratus*, *Carollia perspicillata* and *Sturnira lilium* were the most abundant species in several sites (e.g. FÉLIX, 2003; PASSOS et al., 2003; REIS et al, 2003; MELLO, 2009). This may have precluded great variations in FDiv.

The responses of bats to habitat alteration tend to be highly species-specific (KLINGBEIL; WILLIG, 2009; MEYER; STRUEBIG; WILLIG, 2016), depending on the characteristics of each bat species, such as body size, wing morphology, resource use, and foraging and echolocation strategies (SORIANO, 2000; FARNEDA et al., 2015). Relative wing loading, body mass and trophic level are very good predictors of bats vulnerability to habitat fragmentation. Animalivorous and heaviest bats are more affected by fragmentation, rarely persisting in small fragments (< 100 ha), whereas abundances of frugivores remain stable or increase. Habitat-specialist species with lower relative wing loading are also negatively affected (MEYER et al. 2008; KLINGBEIL; WILLIG, 2009; FARNEDA et al. 2015).

Many species included in this study (22 out of 93 species) were present only in areas greater than 100 ha. Most of them are aerial insectivorous species belonging to the families Emballonuridae and Molossidae. We may think that this functional group is greatly affected by area size. However, as we predicted ground mist nets are a selective method that cannot sample the whole bat community evenly, due to potential vertical stratification of certain species (GREGORIN et al., 2017). Many aerial insectivores, like molossids and emballonurids for example, forage in open areas and above canopy, hindering its capture by mist nets. Additionally, insectivorous bats tend to be underestimated with this method since they detect and avoid the mist nets more often than frugivores (TRAJANO, 1984; KALKO; HANDLEY; HANDLEY, 1996; MACSWINEY; CLARKE; RACEY, 2008). Hence, greater the area and sampling effort, greater the chance to capture aerial insectivorous bats and rare species.

The nectarivores *Lonchophylla mordax* and *Lonchophylla peracchii* were also found only in areas larger than 100 ha, the last being found only in one area of 15,000 ha. Animalivore and nectarivore guilds tend to be positively related to higher amounts of native vegetation (MUYLAERT; STEVENS; RIBEIRO, 2016). This may explain why these species were found

in such great areas. We also found some area limitations regarding threatened species. Among them *Natalus macrourus*, classified as a vulnerable species, was the most sensible to habitat loss. This insectivorous bat is cave specialist. However, recent changes in the Brazilian cave protection law reduced the conservation priority of caves in Brazil, easing the occurrence of irreversible negative impacts in caves of utmost relevance (Brasil, 2022). Combined with mining activities and habitat loss, these factors constitute the main threats for this species (DELGADO-JARAMILLO; BARBIER; BERNARD, 2018). Besides that, in this study *N. macrourus* was only found in areas greater than 100 ha, suggesting that great areas are needed to the effective conservation of this species.

Eptesicus taddeii is also classified as a vulnerable species. Endemic of the Atlantic Forest, this species has a narrow distribution with most of its registers on mixed ombrophilous forest, a physiognomy of the Atlantic Forest (MIRANDA et al., 2010). The constant reduction of the coverage of this biome together with a distribution of *E. taddeii* associated with primary or regenerating forests, consists the main threats of this species (BERNARD; NASCIMENTO; AGUIAR, 2013; PINNA; BERNARD & ESCARLATE-TAVARES, 2018). In our study this insectivorous bat occurred in areas ranging from 41 to 37,644 ha, indicating sensibility of this species to very small areas. *Lonchorhina aurita* is a vulnerable species with broad distribution. In this study it was recorded in areas ranging from 26 to 41,704 ha, however it is a rare species. Just as *Natalus macrourus*, *L. aurita* is restrictedly associated with caves, that is used as shelter. The destruction of caves by mining activity associated with species rarity and sensibility to habitat degradation consists the main threats of this insectivorous species (MEDEIROS et al., 2018). The extinction risk of these cave specialist bats will become substantially greater after the change in Brazilian cave protection law, highlighting the urgency of protection of karstic habitats and caves in Brazil, as well as forest areas of different sizes.

5 CONCLUSION

The positive relation between taxonomic and functional diversity and area size found in this study reveals the sensibility of Atlantic bat communities to habitat loss and consequently to fragmentation. Our results highlight the importance of protecting areas larger than 100 ha to ensure the conservation of threatened bat species in the Atlantic Forest biome. The preservation of large continuous forests is invaluable to species conservation, however, large areas of intact forest are very rare in the Atlantic Forest, whereas most of its remnants is present in small fragments (< 50 ha). Consequently, the preservation of small areas is also important for conservation of Atlantic bat communities. In a recent review, Arroyo-Rodríguez et al. (2020) designed an optimal landscape scenario to conciliate anthropogenic habitat use and wildlife preservation containing at least 40% of forest cover, with higher percentage needed in the tropics. Of this percentage, 10% should be a single (or few) large forest patch, and 30% should be several evenly dispersed small fragments embedded in high-quality matrix. This landscape scenario represents a feasible solution for bats conservation in such fragmented biome.

This study presents a general view of bats response to area size in the Atlantic Forest, and the inclusion of the functional dimension in the analysis resulted in a more detailed understanding of bats response to habitat loss, reinforcing that greater areas hold communities with more functions and more stability against extinction. Nevertheless, we highlight the importance of analyzing the response of each bat guild to evaluate which functional traits are more affected by area reduction, and to consider different scales. We also encourage research assessing bats functional traits to bring more accurate knowledge on the functional diversity of this group. Moreover, studies combining different sampling methods, like acoustic sampling, telemetry and canopy mist-nets, are required to fill the gaps in knowledge of bat species that are not captured with mist nets.

CONCLUSÃO

A relação positiva entre a diversidade taxonômica e funcional e a área encontrada neste estudo revela a sensibilidade das comunidades de morcegos da Mata Atlântica à perda de habitat e conseqüentemente à fragmentação. Os resultados obtidos destacam a importância de proteger áreas maiores que 100 ha para garantir a conservação de espécies de morcegos ameaçadas no bioma Mata Atlântica. A preservação de grandes porções de floresta contínua é inestimável para a conservação das espécies, no entanto, grandes áreas de floresta intacta são muito raras na Mata Atlântica, visto que a maioria de seus remanescentes está presente em pequenos fragmentos (< 50 ha). Conseqüentemente, a preservação de pequenas áreas também é importante para a conservação das comunidades de morcegos da Mata Atlântica. Em uma revisão recente, Arroyo-Rodríguez et al. (2020) projetaram um cenário de paisagem ideal para conciliar o uso antropogênico do habitat e a preservação da vida selvagem, contendo pelo menos 40% de cobertura florestal, com maior porcentagem necessária nos trópicos. Dessa porcentagem, 10% deve ser um único (ou poucos) fragmento grande de floresta e 30% deve ser vários pequenos fragmentos uniformemente dispersos incorporados em uma matriz de alta qualidade. Este cenário de paisagem representa uma solução viável para a conservação de morcegos em um bioma tão fragmentado como a Mata Atlântica.

Este estudo apresenta uma visão geral da resposta dos morcegos ao tamanho da área na Mata Atlântica, e a inclusão da dimensão funcional na análise resultou em um entendimento mais detalhado da resposta dos morcegos à perda do habitat, reforçando que áreas maiores abrigam comunidades com mais funções e maior estabilidade contra a extinção. No entanto, destaca-se a importância de analisar a resposta de cada guilda de morcegos para avaliar quais atributos funcionais são mais afetados pela redução da área, e também considerar diferentes escalas. Incentiva-se também pesquisas avaliando características funcionais de morcegos para trazer um conhecimento mais preciso sobre a diversidade funcional desse grupo. Além disso, estudos combinando diferentes métodos de amostragem, como amostragem acústica, telemetria e redes de neblina de dossel, são necessários para preencher as lacunas do conhecimento de espécies de morcegos que não são capturadas com redes de neblina.

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SUPPORTING INFORMATION

S1 Table. List of the 91 sites of the Atlantic Forest used in the analysis, including author name, year of publication, state where the sampling was made, area size (in hectares) and species richness for each site. The sites are ranked from the smallest to the largest area.

Area_ID	Author	Year	State	Area(ha)	Richness
1	Nobre, P. H. <i>et al.</i>	2009	MG	1.00	12
2	Esbérard, C. E. L.	2003	RJ	1.00	26
3	Bernardi, I. P. <i>et al.</i>	2012	RS	2.00	6
4	Reis, N. R. dos <i>et al.</i>	2003	PR	2.00	4
5	de Barros, R. S. M. <i>et al.</i>	2006	MG	2.00	3
6	de Carvalho, M. C.	2008	SP	3.00	10
7	Sipinski, E. A. B. & Reis, N. R. dos	1995	SC	6.00	8
8	Reis, N.R. dos. & Muller, M.F.	1995	PR	6.20	11
9	Gallo, P. <i>et al.</i>	2010	PR	7.00	14
10	Bosco-Breviglieri, C. P.	2008	SP	8.00	13
11	de Barros, R. S. M. <i>et al.</i>	2006	MG	9.00	6
12	Pereira, A. D. <i>et al.</i>	2019	PR	10.00	4
13	Xavier, B. da S. <i>et al.</i>	2018	MG	12.02	13
14	Barbosa, G. P. <i>et al.</i>	2018	SP	12.70	13
15	Esbérard, C. E. L.	2003	RJ	13.00	23
16	Muylaert, R. L. <i>et al.</i>	2014	SP	15.00	10
17	Mello, R. M. <i>et al.</i>	2016	MG	15.00	11
18	França, D.S. <i>et al.</i>	2013	RJ	19.00	7
19	Pedro, W. A. <i>et al.</i>	2001	SP	20.00	14
20	Munster, L. C.	2011	PR	21.00	19
21	Peracchi, A. L. & Albuquerque, S. T.	1993	ES	22.00	30
22	Ortêncio-Filho, H. <i>et al.</i>	2005	PR	25.00	8
23	de Oliveira, C. C. N and Pontes, A. R. M	2010	PE	26.00	20
24	Esbérard, C. E. L. & Bergallo, H. G.	2005	RJ	27.00	18
25	Scultori, H. C. S.	2010	PR	28.00	18

26	Esbérard, C. E. L.	2003	RJ	29.00	22
27	Lourenço, E. C. <i>et al.</i>	2010	RJ	31.00	33
28	Pereira, A. D. <i>et al.</i>	2019	PR	31.00	6
29	Esbérard, C. E. L.	2003	RJ	32.00	31
30	Zanon, C. M. V. & Reis, N. R.	2007	PR	33.00	3
31	Bernardi, I. P. & Passos, F. C.	2012	RS	35.00	12
32	Esbérard, C. E. L.	2003	RJ	36.00	19
33	Brito, J. E. C. <i>et al.</i>	2010	PR	37.00	7
34	Reis, N. R. dos. <i>et al.</i>	2003	PR	39.00	26
35	Bianconi, G. V. <i>et al.</i>	2004	PR	40.00	10
36	Miranda, J. M. D. & Zago, L.	2015	PR	41.00	10
37	Brito, J. E. C.	2011	PR	43.00	11
38	de Sá Neto, R. J.	2003	AL	44.00	28
39	Silveira, M. <i>et al.</i>	2011	SP	45.00	8
40	Bernardi, I. P. & Passos, F. C.	2012	RS	45.00	7
41	Reis, N.R. dos. & Muller, M.F.	1995	PR	60.00	11
42	de Barros, R. S. M. <i>et al.</i>	2006	MG	78.00	3
43	Félix, J. S.	2003	PR	85.47	12
44	Mello, R. M. <i>et al.</i>	2016	MG	94.00	5
45	Esbérard, C. E. L. <i>et al.</i>	2010	RJ	100.00	29
46	Pereira, A. D. <i>et al.</i>	2019	PR	110.00	11
47	Rosa, S. D.	2004	PR	118.51	12
48	Pires, D. P. de S. & Fabián, M. E.	2013	RS	142.00	7
49	Pedrozo, A. R. <i>et al.</i>	2016	SP	142.88	17
50	da Silva, L. A. M. & de Farias, A. M. I.	2004	PE	157.00	14
51	Barros, M. A. S. <i>et al.</i>	2017	RN	174.95	16
52	Rogeri, P. K	2016	SP	231.06	9
53	Pereira, A. D. <i>et al.</i>	2019	PR	288.00	9
54	Rogeri, P. K	2016	SP	331.24	15
55	Nunes, H. L.	2013	PB	338.80	15
56	Graciolli, G. & Bianconi, G. V.	2007	PR	470.00	11
57	Mendes, P. <i>et al.</i>	2014	ES	573.00	26

58	Rubio, M. B. G. <i>et al.</i>	2014	PR	530.00	18
59	Talamoni, S. <i>et al.</i>	2013	MG	600.00	15
60	Talamoni, S. <i>et al.</i>	2013	MG	600.00	15
61	Talamoni, S. <i>et al.</i>	2013	MG	602.00	13
62	Reis, N. R. dos. & Muller, M.F.	1995	PR	680.00	15
63	Pereira, A. D. <i>et al.</i>	2019	PR	680.00	9
64	Aguiar, L. M. S. & Marinho-Filho, J.	2007	MG	886.00	17
65	Ortêncio-Filho, H. <i>et al.</i>	2014	PR	1035.00	17
66	Talamoni, S. <i>et al.</i>	2013	MG	1048.60	17
67	Luz, J. L. <i>et al.</i>	2011	RJ	1361.00	30
68	Oprea, M. <i>et al.</i>	2009	ES	1500.00	14
69	Brito, D. de V. & Bocchiglieri, A.	2012	SE	1520.00	14
70	Zeppelini, C. G. <i>et al.</i>	2017	PB	1689.73	20
71	de Oliveira, N. Y. K.	2010	PR	2253.00	23
72	Althoff, S. L.	2007	SC	5296.16	15
73	Baptista, M. & Mello, M. A. R. (2001); Mello, M. A. R. (2009)	2001	RJ	6100.00	19
74	Reis, N. R. dos. <i>et al.</i>	2006	PR	7883.00	11
75	Bertola, P. B. <i>et al.</i>	2005	SP	7916.00	22
76	Novaes, R. L. M. <i>et al.</i>	2017	RJ	8000.00	24
77	da Silva, J. R. R. <i>et al.</i>	2013	PR	8716.13	13
78	Dias, D. <i>et al.</i>	2002	RJ	11556.65	20
79	Falcão, F. D. C. <i>et al.</i>	2003	MG	12403.00	15
80	Faria, D. M.	2006	BA	15000.00	6
81	Alves, L. A.	2008	SP	15100.00	22
82	Dornelles, G.D. P. & Graciolli, G.	2017	SP	22500.00	19
83	Gnocchi, A. P. <i>et al.</i>	2019	ES	22711.00	29
84	Dias, D. & Peracchi, A. L.	2008	RJ	26260.00	19
85	Pimenta, V. T	2013	ES	27946.00	53
86	Reis, N. R. dos. <i>et al.</i>	2003	SP	33845.33	10
87	Gregorin, R. <i>et al.</i>	2017	MG	35974.00	16
88	Cláudio, V. C. <i>et al.</i>	2020	SP	37644.00	22

89	Passos, F. C. <i>et al.</i>	2003	SP	41704.00	16
90	Geraldes, M.	1999	SP	209345.00	14
91	Geraldes, M.	1999	SP	209345.00	12

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S2 Table. Description of species present in this study, including conservation status according with the national list (ICMBio, 2018), diet classification and sites where they were registered. For Area_ID information see S1 Table. LC = least concern, NT = near threatened, VU = vulnerable, DD = data deficient.

Species	Status	Diet	Area_ID
Emballonuridae			
<i>Centronycteris maximiliani</i>	LC	aerial insectivore	85
<i>Peropteryx kappleri</i>	LC	aerial insectivore	73
<i>Peropteryx leucoptera</i>	LC	aerial insectivore	51,55
<i>Rhynchonycteris naso</i>	LC	aerial insectivore	85
<i>Saccopteryx bilineata</i>	LC	aerial insectivore	83,85
<i>Saccopteryx leptura</i>	LC	aerial insectivore	50,55,67,70,85
Molossidae			
<i>Cynomops abrasus</i>	LC	aerial insectivore	27,29
<i>Cynomops planirostris</i>	LC	aerial insectivore	85
<i>Eumops auripendulus</i>	LC	aerial insectivore	29
<i>Eumops perotis</i>	LC	aerial insectivore	79
<i>Molossops neglectus</i>	LC	aerial insectivore	31,75,77
<i>Molossops temminckii</i>	LC	aerial insectivore	60,66
<i>Molossus aztecus</i>	DD	aerial insectivore	87
<i>Molossus molossus</i>	LC	aerial insectivore	2,11,15,27,29,36,49,60,61,66,67,76,79,83,85
<i>Molossus rufus</i>	LC	aerial insectivore	27,29,67,73,83,85
<i>Nyctinomops laticaudatus</i>	LC	aerial insectivore	27,66,85
<i>Nyctinomops macrotis</i>	LC	aerial insectivore	27,29
<i>Promops nasutus</i>	LC	aerial insectivore	51
<i>Tadarida brasiliensis</i>	LC	aerial insectivore	29,79
Natalidae			
<i>Natalus macrourus</i>	VU	aerial insectivore	45, 70
Noctilionidae			
<i>Noctilio albiventris</i>	LC	aerial insectivore	65

<i>Noctilio leporinus</i>	LC	piscivore	2,27,67,73,81
Phyllostomidae			
<i>Anoura caudifer</i>	LC	nectarivore	1,2,6,7,10,11,13,14,15,16,17,18,20,21,24,25,26,27,29,3 2,34,38,42,43,45,46,49,54,57,58,59,60,61,62,64,66,67,7 1,72,74,75,76,78,79,81,82,84,88,89,90
<i>Anoura geoffroyi</i>	LC	nectarivore	2,13,15,16,18,20,25,27,37,38,47,58,59,66,68,71,72,75,7 6,79,81,82,83,84,85,88,89,90,91
<i>Artibeus cinereus</i>	DD	frugivore	2,9,13,14,15,16,17,18,20,21,23,24,25,26,29,31,34,35,36 ,37,38,46,47,48,50,51,52,55,63,68,69,70,71,72,76,80,81 ,82,83,84,85,88,89,90,91
<i>Artibeus fimbriatus</i>	LC	frugivore	1,2,3,7,9,10,13,14,15,16,17,18,19,20,21,22,23,24,25,26, 27,29,31,32,33,34,35,36,37,38,40,43,45,46,47,52,53,57, 58,63,64,65,67,69,71,72,75,76,77,78,81,82,83,84,85,87, 88,89,90,91
<i>Artibeus gnomus</i>	DD	frugivore	82,85
<i>Artibeus lituratus</i>	LC	frugivore	1,2,3,5,6,7,8,9,10,11,12,13,14,15,16,17,18,19,20,21,22, 23,24,25,26,27,28,29,30,31,32,33,34,35,37,38,39,40,41, 42,43,44,45,46,47,48,49,50,51,52,53,54,55,56,57,58,59, 60,61,62,63,64,65,67,68,69,70,71,72,73,74,75,76,77,78, 79,81,82,83,84,85,86,87,88,89,90,91
<i>Artibeus obscurus</i>	LC	frugivore	2,10,15,20,21,23,24,25,26,27,29,32,37,38,45,47,49,50,5 5,57,59,64,65,66,67,70,71,72,73,75,76,77,78,80,81,82,8 3,84,85,86,87,88,89,90,91
<i>Artibeus planirostris</i>	LC	frugivore	2,6,7,8,9,10,15,20,21,23,24,26,27,28,29,32,33,34,35,38, 41,43,45,50,51,52,53,54,55,57,62,65,66,67,69,70,71,72, 77,85,86,87,89
<i>Carollia brevicauda</i>	LC	frugivore	21,60,71,73,83
<i>Carollia perspicillata</i>	LC	frugivore	1,2,4,6,7,8,9,10,11,12,13,14,15,16,17,18,19,20,21,22,23 ,24,25,26,27,29,32,33,34,35,37,38,39,41,42,43,44,45,46 ,49,50,51,52,53,54,55,57,58,59,60,61,62,63,64,65,66,67 ,68,69,70,72,73,74,75,76,77,78,79,80,81,82,83,84,85,86 ,87,88,89,90,91
<i>Chiroderma doriae</i>	LC	frugivore	2,7,8,15,17,19,20,23,24,25,26,27,29,32,34,38,45,47,54, 57,67,68,69,71,72,76,78,81,82,89,91
<i>Chiroderma villosum</i>	LC	frugivore	2,21,24,26,27,29,32,35,38,50,65,66,67,68,70,85
<i>Chrotopterus auritus</i>	LC	carnivore	1,3,6,8,9,10,19,21,22,31,34,35,37,38,43,44,45,49,60,62, 65,67,72,73,78,81,85,87,89,90

<i>Desmodus rotundus</i>	LC	sanguivore	1,4,6,8,9,10,13,15,16,17,19,20,21,23,24,26,27,29,30,32,33,34,35,38,45,48,50,51,52,54,56,57,58,59,60,61,64,65,66,67,68,70,71,72,73,75,76,78,79,81,82,83,84,85,87,88,89
<i>Diaemus youngi</i>	LC	sanguivore	27,34,85
<i>Diphylla ecaudata</i>	LC	sanguivore	15,23,24,25,32,34,37,38,45,58,64,66,78,82,84,88
<i>Dryadonycteris capixaba</i>	DD	nectarivore	21,64,85,87
<i>Gardnerycteris crenulatum</i>	LC	gleaning insectivore	21,64,73,83,85
<i>Glossophaga soricina</i>	LC	nectarivore	2,6,10,14,15,18,19,20,21,23,25,26,27,29,32,34,38,39,45,47,48,49,50,51,52,54,55,57,58,59,60,61,64,66,67,68,69,70,71,73,74,75,76,78,81,82,83,84,85,86,87,90
<i>Glyphonycteris daviesi</i>	LC	gleaning insectivore	85
<i>Glyphonycteris sylvestris</i>	LC	gleaning insectivore	19,85
<i>Lampronycteris brachyotis</i>	LC	gleaning insectivore	21,25,70,82,85
<i>Lichonycteris degener</i>	DD	nectarivore	38
<i>Lonchophylla mordax</i>	LC	nectarivore	80
<i>Lonchophylla peracchii</i>	LC	nectarivore	15,24,26,27,32,45,67,73,76,78,84,85
<i>Lonchorhina aurita</i>	VU	gleaning insectivore	23,38,58,67,70,89
<i>Lophostoma brasiliense</i>	LC	gleaning insectivore	21,45,51,85,87
<i>Lophostoma silvicolum</i>	LC	gleaning insectivore	23,38,55,70
<i>Micronycteris hirsuta</i>	LC	gleaning insectivore	21,45,80,83,85
<i>Micronycteris megalotis</i>	LC	gleaning insectivore	2,8,9,15,21,23,24,26,27,32,34,35,41,43,45,49,57,58,59,62,66,67,73,75,78
<i>Micronycteris microtis</i>	LC	gleaning insectivore	19,25,46,76,83,84,85,88,89
<i>Micronycteris minuta</i>	LC	gleaning insectivore	1,15,21,45,76,78,83,85
<i>Micronycteris schmidtorum</i>	LC	gleaning insectivore	70,85
<i>Mimon bennettii</i>	LC	gleaning insectivore	1,2,4,13,20,32,71,74,78,81,84,88,91
<i>Phyllostomus discolor</i>	LC	omnivore	9,10,16,21,23,32,39,49,50,51,54,55,58,60,64,65,68,69,70,85
<i>Phyllostomus elongatus</i>	LC	omnivore	23,38
<i>Phyllostomus hastatus</i>	LC	omnivore	9,10,13,14,15,21,23,26,27,34,41,43,46,49,50,51,54,55,58,65,66,67,68,73,76,78,83,85,86

<i>Platyrrhinus incarum</i>	LC	frugivore	38,85
<i>Platyrrhinus lineatus</i>	LC	frugivore	1,2,3,5,6,8,9,10,11,13,14,15,16,17,19,22,23,24,26,27,29,32,34,38,39,41,43,44,45,46,47,49,50,51,52,54,55,58,59,60,61,62,64,65,66,67,68,69,70,72,73,75,76,78,79,81,82,83,84,85,87,90
<i>Platyrrhinus recifinus</i>	LC	frugivore	2,17,19,20,21,25,26,27,29,37,45,54,58,67,71,76,78,83,84,85,86,88,89
<i>Pygoderma bilabiatum</i>	LC	frugivore	1,3,6,7,8,17,19,20,21,22,24,25,26,29,31,33,34,36,38,39,40,41,43,45,46,47,49,53,56,57,59,61,62,63,65,67,68,71,72,73,75,76,77,79,82,84,86,88,89,91
<i>Rhinophylla pumilio</i>	LC	frugivore	21,23,38,70,80,83,85
<i>Sturnira lilium</i>	LC	frugivore	1,2,3,4,5,6,7,8,9,10,12,13,14,15,16,17,18,19,20,21,22,23,24,25,26,27,28,29,30,31,32,33,34,35,36,37,38,39,40,41,43,44,45,46,47,48,49,50,51,52,53,54,55,56,57,58,59,60,61,62,63,64,65,66,67,69,70,71,72,73,74,75,76,77,78,79,81,82,83,84,85,86,87,88,89,91
<i>Sturnira tildae</i>	LC	frugivore	20,21,25,37,52,54,57,67,71,72,75,76,81,82,85,88,89
<i>Tonatia bidens</i>	LC	omnivore	2,15,20,24,26,27,29,32,38,45,58,67,71,76,78,81,82,84,85,90
<i>Tonatia maresi</i>	LC	gleaning insectivore	38,55,69,70,83,85
<i>Trachops cirrhosus</i>	LC	carnivore	20,21,23,27,37,38,45,51,55,58,67,69,71,81,82,83,85,87,88,91
<i>Trinycteris nicefori</i>	LC	gleaning insectivore	21, 38,69,85
<i>Uroderma magnirostrum</i>	LC	frugivore	21,68,85
<i>Vampyressa pusilla</i>	LC	frugivore	1,2,13,15,17,19,20,21,22,24,25,26,27,29,32,34,35,38,39,41,43,45,47,49,54,57,59,64,66,67,71,72,76,78,79,81,82,83,84,85,86,87,88,90
<i>Vampyrodes caraccioli</i>	LC	frugivore	25,27
Thyropteridae			
<i>Thyroptera tricolor</i>	LC	aerial insectivore	67,71
<i>Thyroptera wynneae</i>	DD	aerial insectivore	83,87
Vespertilionidae			
<i>Eptesicus brasiliensis</i>	LC	aerial insectivore	2,15,27,29,34,45,56,60,61,73,74,75,76,77,79
<i>Eptesicus diminutus</i>	LC	aerial insectivore	12,13,14,28,31,34,40,46,53,57,62,63,75,83,90
<i>Eptesicus furinalis</i>	LC	aerial insectivore	2,28,29,31,34,36,40,46,56,62,63,77,83,85

<i>Eptesicus taddeii</i>	VU	aerial insectivore	36,57,88
<i>Histiotus velatus</i>	LC	aerial insectivore	14,15,26,29,34,36,45,48,54,56,59,62,74,75
<i>Lasiurus blossevillii</i>	LC	aerial insectivore	9,27,29,34,36,41,47,51,56,59,60,62,65,74,77,79,85
<i>Lasiurus ebenus</i>	DD	aerial insectivore	88
<i>Lasiurus ega</i>	LC	aerial insectivore	2,27,29,45,49,61,65,85
<i>Myotis albescens</i>	LC	aerial insectivore	13,14,29,31,49,75,85
<i>Myotis izecksohni</i>	DD	aerial insectivore	36
<i>Myotis lavalii</i>	DD	aerial insectivore	51
<i>Myotis levis</i>	LC	aerial insectivore	2,9,34,48,56,71,75,81,85
<i>Myotis nigricans</i>	LC	aerial insectivore	2,8,9,11,14,15,16,26,27,28,29,31,34,41,45,47,48,49,50, 53,55,56,57,59,60,61,62,63,65,67,68,69,70,73,74,75,76, 77,79,81,83,85,87,88,91
<i>Myotis riparius</i>	LC	aerial insectivore	2,14,26,27,29,31,40,45,57,63,67,71,75,76,85,88
<i>Myotis ruber</i>	LC	aerial insectivore	31,34,36,53,56,61,62,71,74,75,77,81,88,90,91
<i>Rhogeessa io</i>	DD	aerial insectivore	34