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The influence of anthropogenic activities on threatened avian fauna and its implications for biodiversity conservation

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Influência de atividades antrópicas sobre a avifauna ameaçada de extinção e suas implicações para a conservação da biodiversidade

Ana Claudia de Almeida

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To the enchanting Brazilian birds,
a constant presence in my daily life
and for which I channel all my endeavors
towards conservation and protection.

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General abstract

Human pressures such as deforestation, fire, overexploitation, pollution, and mining are highly degrading natural ecosystems and leading to biodiversity loss worldwide. Tropical regions are suffering more from human impacts compared to temperate regions due to intense climate and land-use changes. In association with them, other extinction drivers have threatened tropical biodiversity, such as hunting and invasive species. Biodiversity loss can also disrupt networks in natural ecosystems, negatively impacting the provision of ecosystem services. To understand how human pressures impact biodiversity and ecosystem services provision in tropical areas, we can use birds as study models. Tropical birds are at high risk of extinction due to the ongoing degradation of tropical regions, which also endangers many ecosystem services they provide. Therefore, in this thesis I investigated the effects of future climate and land-use changes on potential distributions of threatened birds and how endemic and threatened bird defaunation impacts the provision of ecosystem services. In Chapter 1, I used an ecological niche modeling coupled with land-use model to evaluate the responses of a vulnerable bird (*Crax fasciolata*) to separate and combined climate and land-use changes under historical and future scenarios in Brazil. The results showed an increase in environmental suitability for *Crax fasciolata* in Brazil in the future, but with variation between domains and future scenarios. While future projections indicate an increase in climatic suitability, the looming threat of habitat loss presents a substantial challenge. Particularly in the Pantanal and Cerrado domains, the habitat loss can result in a significant decline in environmental suitability, thereby elevating the risk of extinction for this species since they are more abundant in these domains. In Chapter 2, I integrated climate-based models generated using ecological niche modeling with forest cover data to evaluate current and future potential distributions of endemic and threatened Atlantic Forest birds, and I identified priority areas for restoration and conservation for these birds. The results showed

that the majority of bird species are projected to experience a reduction in projected distributions in the near future. Additionally, the findings indicate that only a limited portion of priority areas for conservation lies within the extant protected areas. This situation heightens the vulnerability of these bird species to extinction. In Chapter 3, I investigated the impact of defaunation of these endemic and threatened Atlantic Forest birds on the ecological network structure and provision of ecosystem services in different scenarios. I have found that the network structure will remain stable due to functional redundancy, but medium to high rates of bird defaunation can cause substantial decreases in key ecosystem services. These findings highlight the urgent need for conservation efforts within the researched regions, particularly in domains experiencing severe degradation such as the Pantanal, and Brazil’s biodiversity hotspots: Atlantic Forest and Cerrado. I emphasize the critical necessity of expanding/creating protected areas, implementing natural vegetation restoration programs, strengthening control of deforestation, and eradicating overhunting and other threats to conserve endemic and threatened birds within Brazil and safeguard the ecosystem services they provide.

Resumo geral

As pressões humanas, como desmatamento, incêndios, superexploração de recursos, poluição e mineração, estão intensivamente degradando os ecossistemas naturais e causando a perda de biodiversidade em todo o planeta. As regiões tropicais estão sofrendo mais com os impactos humanos em comparação com regiões temperadas devido às intensas mudanças climáticas e do uso do solo. Em associação a estes, outros fatores causadores de extinção têm ameaçado a biodiversidade tropical, como a caça e a introdução de espécies exóticas. A perda de biodiversidade também pode perturbar as redes nos ecossistemas naturais, impactando negativamente na provisão de serviços ecossistêmicos. Para entender como as pressões humanas impactam a biodiversidade e os serviços ecossistêmicos em áreas tropicais, podemos utilizar as aves como modelos de estudo. Elas estão sob alto risco de extinção devido à contínua degradação das regiões tropicais, o que também ameaça muitos serviços ecossistêmicos que elas fornecem. Portanto, nesta tese, investiguei os efeitos das mudanças climáticas e do uso do solo futuras na distribuição potencial de aves ameaçadas e como a defaunação de espécies endêmicas e ameaçadas impactam a provisão de serviços ecossistêmicos. No Capítulo 1, utilizei a modelagem de nicho ecológico associada a um modelo de uso do solo para avaliar as respostas de uma ave vulnerável (*Crax fasciolata*) às mudanças climáticas e do uso do solo separadas e combinadas sob cenários histórico e futuros no Brasil. Os resultados mostraram um aumento na adequabilidade ambiental para *Crax fasciolata* no Brasil no futuro, mas com variação entre domínios fitogeográficos e cenários futuros. Embora as projeções futuras indiquem um aumento na adequabilidade climática, a ameaça iminente de perda de habitat representa um grande desafio. Particularmente no Pantanal e no Cerrado, a perda de habitat pode resultar em um declínio significante na adequabilidade ambiental, elevando o risco de extinção para a espécie devido à sua maior abundância nesses domínios. No Capítulo 2, integrei modelos climáticos gerados pela modelagem de nicho ecológico com dados de cobertura florestal para avaliar as distribuições potenciais atuais e futuras de espécies de aves endêmicas e ameaçadas de extinção na Mata Atlântica e identifiquei áreas prioritárias para restauração e conservação para estas aves. Os resultados mostraram que a maioria das espécies podem experienciar uma redução nas distribuições potenciais em um futuro próximo. Adicionalmente, os resultados indicam que apenas uma porção limitada das áreas prioritárias para conservação estão dentro de áreas protegidas existentes. Essa situação aumenta a vulnerabilidade dessas espécies de aves à extinção. No Capítulo 3, investiguei o impacto da defaunação dessas espécies endêmicas e ameaçadas da Mata Atlântica na estrutura da rede ecológica e na provisão de serviços ecossistêmicos em diferentes cenários. Os resultados mostraram que a estrutura da rede permanecerá estável devido à redundância funcional, mas taxas médias e altas de defaunação podem causar reduções substanciais de serviços ecossistêmicos essenciais. Esses resultados destacam a necessidade urgente de esforços de conservação nas regiões pesquisadas, particularmente em domínios com degradação severa, como o Pantanal, e os hotspots de biodiversidade do Brasil: Mata Atlântica e Cerrado. Enfatizo a necessidade de expandir/criar áreas protegidas, implementar programas de restauração de vegetação natural, fortalecer o controle do desmatamento e erradicar a caça e outras ameaças para conservar as aves ameaçadas e endêmicas no Brasil, além de proteger os serviços ecossistêmicos que elas fornecem.

General Introduction

Human activities exert significant pressures on natural ecosystems, posing a grave and imminent threat to the conservation of biodiversity (York et al. 2003, Crain et al. 2009). Habitat loss is considered the major cause of species extinction (Simberloff 1984, Sayer & Whitmore 1991, Pimm & Raven 2000), but many studies have shown that populations decline, and consequent species extinction, may be a result of the combined action of multiple stressors (e.g., McCallum 2012, Bellard et al. 2015, Díaz et al. 2019, Gonçalves et al. 2021). Factors such as the overexploitation of natural resources, pollution, and conversion of natural environments into anthropogenic areas can result in local extinctions and, consequently, reduce biodiversity (Pereira et al. 2012, Felipe-Lucia et al. 2015). The burning of fossil fuels and deforestation intensify climate change, interfering with the growth, metabolism, and reproduction of living beings, as well as the spread of diseases, either through the proliferation of pathogens, or by changes in immunity and consequent greater susceptibility to diseases (Lafferty 2009, McCallum 2012). In combination with these extinction drivers, factors such as species distribution and ecological traits, and habitat quality are also associated with biodiversity loss (Di Marco et al. 2014).

At global scale, there are more than 42,100 threatened species, representing about 28% of all assessed species (IUCN 2022). Although human pressures are occurring globally, tropical areas have experienced intense climate and land-use changes in the last decades (Lambin et al. 2003, Brodie et al. 2012, Hansen et al. 2013, Zeppetello et al. 2020). These extinction drivers, associated with overexploitation and invasive species, led to the highest vertebrate extinction rates in tropical areas (Dirzo et al. 2014, Ceballos et al. 2017). Moreover, specific regions within tropical zones, such as biodiversity hotspots, face a disproportionate weight of human pressures, jeopardizing the efficacy of biodiversity conservation efforts (Venter et al. 2016, Williams et al. 2020). But not only the biodiversity is affected by human pressures. Especially during the past 50 years, human activities have caused intense alterations in ecosystems, degrading many ecosystem services–provisioning, supporting, regulating, and cultural services–provided by biodiversity (Millennium Ecosystem Assessment 2005).

Although the extinction drivers are well known between scientists, there are still many gaps in knowledge about how human activities can impact biodiversity and ecosystem services in the future. A great tool to understand the impacts of human activities such as land-use and climate change on biodiversity is the ecological niche modeling. It allows us to assess how the environmental changes can determine the species distribution (based on gain or loss in quantity and quality of suitable areas) and plan effective conservation actions (Peterson et al. 2011, Gouveia et al. 2016, Vale et al. 2018a, Ramalho et al. 2021). Given that biodiversity loss significantly impacts the provision of ecosystem services, land-use and climate changes, associated with other extinction drivers such as overexploitation, diseases, and invasive species, are intrinsically related to ecosystem functioning and human welfare. To evaluate how anthropogenic actions can affect ecosystem services, we can simulate, using species distribution maps and putative ecosystem services attribution to species, different scenarios of biodiversity loss (Bogoni et al. 2020). The interactions between species and the ecosystem services can be measured by ecological network metrics, which will indicate how stable is the network in the provisioning of ecosystem services in response to defaunation (Dehling 2018, Bogoni et al. 2020).

Tropical birds can be study models to explore these human impacts on biodiversity and ecosystem services. First, since tropical areas are suffering from intense degradation and harbor the majority of bird species (87% of all bird species), tropical birds are under high risk of extinction, especially those living in biodiversity hotspots, which makes them subjects of many conservation efforts (Şekercioğlu et al. 2012, Tobias et al. 2013). Second, they are well studied, and many datasets are available from scientific community and citizen-scientist birdwatchers (Şekercioğlu et al. 2012). Third, bird defaunation can lead to decrease of important ecosystem services that ensure the proper functioning of the environment and human well-being, since they provide pollination, pest/weed control, seed dispersal, nutrient cycling, and biophilia through birdwatching (Whelan et al. 2008, Michel et al. 2020).

In this context, this thesis investigates the effects of future climate and land-use changes on potential distributions of threatened birds and how endemic and threatened bird defaunation impacts the provision of ecosystem services in a biodiversity hotspot. The thesis is structured into three chapters. In Chapter 1, I estimated the vulnerability of a threatened bird species (the Bare-faced Curassow, *Crax fasciolata*) to separate and combined climate and land-use changes. As this species has a broad geographical range, my goal was to comprehend its susceptibility across different domains in Brazil and to discuss environmental policies for each domain, seeking the conservation of *Crax fasciolata*. For this, I built climate-based models using ecological niche modeling and combined with a land-use model considering the Brazilian political context, and quantified the environmental suitability under historical and future scenarios. This chapter is already published on Biotropica (https://doi.org/10.1111/btp.13142). In Chapter 2, I chose endemic and threatened Atlantic Forest bird species to understand their responses to future environmental conditions. I integrated current and future climate-based models generated by ecological niche modeling with forest cover data. This approach also enabled to determine priority areas for conservation and restoration for these birds in an extremely degraded biodiversity hotspot. In Chapter 3, I evaluated the impacts of endemic and threatened Atlantic Forest birds defaunation on ecosystem services. I combined information on species geographic ranges and ecosystem services provided by them to build different defaunation scenarios and investigated the impacts using ecological network metrics. I understand that the findings of this study provide relevant information to develop robust environmental policies to guarantee the conservation of threatened bird species across different Neotropical domains, as well as the ecosystem services associated with them.

Chapter 1: Regional variation in responses of a vulnerable bird species to land-use and climate change[1]

[1] This chapter is published on Biotropica (https://doi.org/10.1111/btp.13142)

Abstract

Climate and land-use changes are expected to negatively affect many species and ecological processes, leading to biodiversity loss. However, some species can adapt to these changes. Wide-ranging species are expected to be less impacted by such changes, but they can occur in different domains with contrasting environmental conditions, resulting in different conservation statuses along their range. To understand whether a species will overall benefit or lose with global change, we evaluated the responses of a wide-ranging but a vulnerable bird (*Crax fasciolata*) to separate and combined effects of climate and land-use changes under different environmental policies in Brazil. Using ecological niche modeling and a land-use model within the Brazilian political context, we quantified climatic, habitat, and environmental suitability for *Crax fasciolata* under historical (2000) and future (2050) scenarios. Our findings showed that environmental suitability can increase for *Crax fasciolata* in Brazil in the future, but these effects vary according to the domain and the specific future scenario considered. Climatically suitable areas will increase in all scenarios, and those environmental scenarios that include better habitat conditions will provide more environmentally suitable areas for *Crax fasciolata*. However, this increase comes from newly suitable areas in the Atlantic Forest and the Amazon, while the Pantanal, the Caatinga, and the Cerrado will lose environmental suitability due to native vegetation loss. Despite the availability of these new areas, reduced landscape permeability may hinder *Crax fasciolata* from reaching them. This reinforces the urgent call for public policies for native vegetation protection, reforestation and effective deforestation control.

Introduction

Climate and land-use changes are among the main causes of biodiversity loss (Sala et al. 2000, Dirzo et al. 2014). Climate change can affect ecological dynamics and generate different species responses, such as shifts in geographic range, adaptation to new climatic conditions, or extinction if no adaptation is achieved (Berg et al. 2010, Bellard et al. 2012). Land-use changes (particularly habitat loss) often lead to population declines, species extinction, metacommunity fragmentation, and ecosystem functioning loss (Fahrig 1997, Thompson et al. 2017). Studies have shown the importance of evaluating the impacts of climate and land-use changes together since they are interrelated processes and their combination can cause significant shifts in suitable environments across many taxa (see Jetz et al. 2007, Loiselle et al. 2010, Gouveia et al. 2016, Ramalho et al. 2021). These shifts can be critical in both temperate and tropical regions (Asner et al. 2010, Hof et al. 2011, Newbold 2018). Especially in the last decades, tropical areas have been intensively impacted by climate and land-cover changes (Lambin et al. 2003, Brodie et al. 2012, Hansen et al. 2013, Zeppetello et al. 2020), which is worrying as tropical species may be more vulnerable to climate change impacts than temperate species (Araújo et al. 2013, Khaliq et al. 2014, Manes et al. 2021).

For tropical birds, the combination of climate change and habitat loss could drive the extinction of hundreds of species, including currently non-threatened species (Şekercioğlu et al. 2008, 2012). Environmental changes impact particularly the tropical birds that are non-migratory, forest-dependent, endemic, and have specialized habitat and food requirements (Newbold et al. 2013, Borges et al. 2019, Manes et al. 2021). In response to environmental changes, small-ranged species are expected to have a greater range contraction due to their high vulnerability compared to wide-ranging species (Jetz et al. 2007, Manes et al. 2021). However, wide-ranging species can occur in different domains exposed to different anthropogenic pressures and contrasting environmental conditions, which can impact species differently along their range (e.g., IUCN 2012). Although studies have projected shifts in wide-ranging bird species distributions (see Peterson et al. 2001, Reside et al. 2012, Sáenz-Jiménez et al. 2020), it is unknown how these species may respond to environmental changes across different domains.

In Brazil, there are six recognized domains with unique ecological characteristics: Amazon, Cerrado, Caatinga, Pantanal, Atlantic Forest, and Pampa (Câmara et al. 2015). Two of these domains are biodiversity hotspots – Atlantic Forest and Cerrado – having a high number of endemic and flagship species and being heavily impacted by human pressures (Mittermeier et al. 2005). For each domain, studies have simulated future scenarios exploring different simulations of land-use policies (Câmara et al. 2015, Soterroni et al. 2018) and greenhouse gas emissions (Salazar et al. 2007, Souza & Manzi 2014). In general, cropland and pasture tend to increase in the future in Brazil, mainly in pessimistic scenarios (i.e., with low compliance with the law; Câmara et al. 2015, Soterroni et al. 2018), increasing the pressure on species that depend on native vegetation. Regarding climate change, projections predict variation among domains and scenarios (Salazar et al. 2007, Souza & Manzi 2014). These climate alterations can lead to a replacement of existing formations, such as forests turned into savannas, severely impacting biodiversity (Salazar et al. 2007, Souza & Manzi 2014).

A good bird species to evaluate responses to environmental changes in different domains and environmental policies in Brazil is the Bare-faced Curassow (*Crax fasciolata*), a vulnerable species from the Cracidae family. This species is widely distributed in South America, encompassing central and southwest Brazil, eastern Bolivia, Paraguay, and northern Argentina (BirdLife International 2016), and inhabits humid, semideciduous, and gallery forests, as well as forest edges (del Hoyo 1994). In Brazil, this species is distributed in four domains: Amazon, Pantanal, Cerrado, and Atlantic Forest. Despite its wide distribution, characteristics such as non-migration and forest association make *C. fasciolata* highly vulnerable to environmental changes (BirdLife International 2016). The conversion of natural habitats to agriculture and pasture is one of the main threats to *C. fasciolata*, making this species highly vulnerable (Brooks 2006, Thornton et al. 2012, BirdLife International 2016). In addition, climate change is expected to modify the geographic range of many cracids, with high contraction in future scenarios, especially if dispersal is limited (Peterson et al. 2001). Considering these pressures and the wide range of *C. fasciolata*, combining future climate and land-use scenarios can aid in understanding how and where environmental changes might impact this species and guide more efficient conservation policies.

We evaluated *C. fasciolata* responses to the effects of climate and land-use changes in different future scenarios in Brazil, as well as in the different Brazilian domains. We aimed to estimate the vulnerability of a threatened species to climate and land-use changes (separate and combined) in the different Brazilian domains and to discuss Brazilian public policies associated with them. We expected that: (a) future climate change could decrease suitability for *C. fasciolata*, considering that a previous study on a closely related species (*C. rubra*) predicted declines in potentially suitable areas even in an optimistic scenario (Peterson et al. 2001); (b) future land-use changes could lead to higher decreases in suitable habitat areas in pessimistic scenarios than in an optimistic scenario since studies have predicted higher forest loss rates under scenarios with relaxed environmental laws (Câmara et al. 2015, Soterroni et al. 2018); and (c) combining climate and land-use changes, all future scenarios could show a decrease in the species' potential distribution in Brazil, with a higher decrease in pessimistic scenarios (i.e., scenarios that combine higher greenhouse gas emissions and no forest protection law). For Brazilian domains, we expected: (a) a high climatic suitability loss in the Cerrado and Caatinga considering the high increase in temperature and high decrease in precipitation predicted for these domains (Souza & Manzi 2014), and a smaller loss in the Amazon because even if precipitation decreases in this domain (Souza & Manzi 2014), suitable climate conditions can be maintained for the species since *C. fasciolata* inhabits areas influenced by drier conditions (i.e., forest formations in savannas); (b) a higher habitat suitability loss in the Pantanal and Cerrado due to large expansion in agricultural land, and an increase in habitat suitability in the Atlantic Forest promoted by forest restoration (Soterroni et al. 2018); and (c) a higher decrease in environmental suitability in the Cerrado, and smaller decreases in the Amazon and Atlantic Forest.

Material and methods

Study area and species database

The study area is 6,880,354 km² in extent (33º45'00" S to 3º40'12" N; 73º32'24" W to 38º52'48" W; Figure 1) and was delimited based on the species occurrence data (see details in the Climatic suitability subsection). It includes the six Brazilian domains: Amazon, Pantanal, Cerrado, Atlantic Forest, Caatinga, and Pampa (see Assis et al. 2019 for domain limits). The Brazilian Amazon Forest is the largest forest in Brazil, with moist evergreen dense forest comprising the largest portion of the native vegetation cover (Câmara et al. 2015). The Pantanal is the world's largest tropical wetland, being formed by permanent aquatic, periodically flooded, and permanently dry habitats, and includes forest woodland, open wood savanna, and grasslands (Junk et al. 2006, Evans & Costa 2013, Assine et al. 2015). The Cerrado has a high landscape heterogeneity, including savanna, semideciduous, and deciduous forests (Silva et al. 2006). The Atlantic Forest has wide latitudinal and longitudinal ranges, which produce different topographies, climatic zones, and forest composition, such as dense rainforest, open and mixed semideciduous, and deciduous forests (Ribeiro et al. 2009, Câmara et al. 2015). The Caatinga is a mosaic of scrub and fragmented

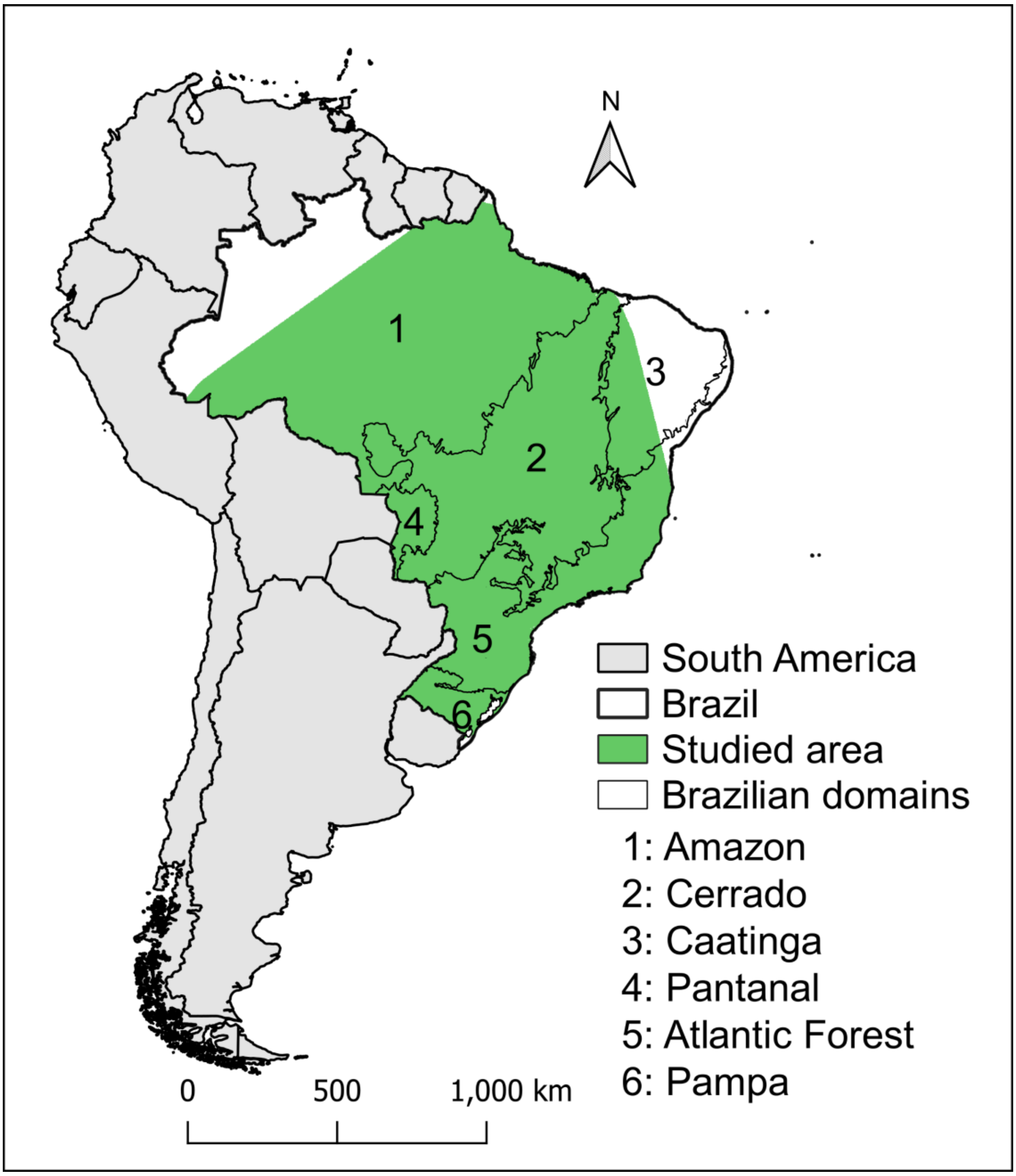

Figure 1. Studied area including the six Brazilian domains: Amazon, Cerrado, Pantanal, Caatinga, Atlantic Forest, and Pampa.

dry forests (Santos et al. 2011). The Pampa is the smallest domain in Brazil, composed of natural grasslands with sparse shrubs and tree formations (Roesch et al. 2009).

We gathered *C. fasciolata* occurrence records from the Global Biodiversity Information Facility (GBIF), Distributed Information System for Biological Collections (speciesLink), Brazilian Biodiversity Information System (SiBBr), and literature (Pivatto et al. 2006, Hasui et al. 2017, Nunes et al. 2018) sources, as well as personal observations, for a total of 10,896 records. We cleaned the occurrence records by removing duplicates, records with latitude and longitude equal to zero, records before 1970 (as historical variables from WorldClim version 2.1 consider climate data since 1970), and records beyond the species' published distribution range (according to BirdLife International 2016). We also performed geographical cleaning to reduce spatial autocorrelation using the 'spThin' R package (Aiello-Lammens et al. 2015). Considering that information about the dispersal of *C. fasciolata* is unknown, for geographical cleaning we excluded records separated by less than 5 km based on the average dispersal of the only species from *Crax* genus found in the literature (average dispersal post-release of 3-3.5 km for *C. blumenbachii*; Bernardo et al. 2011). We also performed environmental cleaning to avoid environmental biases. Environmental cleaning was carried out using the five least correlated bioclimatic variables selected to evaluate climatic suitability (see the Climatic suitability subsection). In this process, the environmental filters removed records with similar bioclimatic variable values in environmental space (following Varela et al. 2014). After the cleaning process, we retained 304 independent occurrence records.

Climatic suitability

We used Ecological Niche Modeling (ENM) to predict potentially suitable areas both in historical and future climate scenarios, providing continuous and binary maps of suitability. We constructed climate-based models under historical (near current, average for 1970-2000) and future (2050, average for 2041-2060) conditions. For the historical model, we downloaded 19 bioclimatic variables at 2.5 arc-minutes resolution (~5 × 5 km at the equator) from WorldClim version 2.1 (WorldClim 2020). The calibration and projection area of the model (Figure S1) refers to a minimum convex polygon (MPC) around the presence records, adding 20% of the polygon area as a buffer (see Barve et al. 2011). Based on this area, we extracted the values of the bioclimatic variables and calculated the Pearson's correlation using the 'corrplot' R package (Wei & Simko 2017) to select a subset of the bioclimatic variables. This resulted in five least correlated variables ($r \leq 0.6$): Temperature Seasonality, Mean Temperature of Wettest Quarter, Precipitation of Driest Month, Precipitation of Warmest Quarter and Precipitation of Coldest Quarter. For future models, we used data from Global Circulation Models (GCMs) from the Coupled Model Intercomparison Project Phase 6 (CMIP6). We selected the three best GCMs for South America considering Cannon (2020) and availability in WorldClim version 2.1 (WorldClim, 2020): MIROC6, BCC-CSM2-MR and IPSL-CM6A-LR. Based on current challenges to mitigate greenhouse gas emissions, and seeking to discuss conservation actions for future conditions, we chose two emission scenarios from the Shared Socioeconomic Pathways (SSPs): SSP5-8.5, the worst scenario, and SSP3-7.0, an intermediate scenario between a more optimistic and the worst scenario (Hausfather 2019). Here, SSP5-8.5 is called a pessimistic climate scenario and SSP3-7.0 a moderate climate scenario.

Considering that combining different algorithms improves predictions (Araújo et al. 2005), we used four niche modeling algorithms: Bioclimate analysis (BIOCLIM), which requires presence-only data, Maximum Entropy (MaxEnt), which requires presence and background data, and Random Forest (RF) and Support Vector Machine (SVM), which require presence and pseudo-absence data. We considered 10,000 background points (within the calibration area) for MaxEnt selected by random method (see Phillips & Dudík, 2008), 304 pseudo-absence points for Random Forest (same number as presence records) selected by "SRE" method, and 1,000 pseudo-absence points for SVM also selected by "SRE" method. "SRE" method is a random selection of points within the calibration area but outside the suitable area estimated by an envelope from the presence records (see Barbet-Massin et al. 2012). We generated pseudo-absence points using the 'biomod2' R package (Thuiller et al. 2020). We performed a 10-fold cross-validation, repeating the procedure 10 times for each algorithm. We applied the True Skill Statistic (TSS; Allouche et al. 2006) using a threshold value of 0.7 to obtain models with high accuracy (only replicas with TSS > 0.7 were selected; Table S1). For each algorithm, we normalized the resulting forecasts of climatic suitability values to range from 0 (low suitability) to 1 (high suitability). We built continuous ensemble models using a weighted average by TSS (Araújo et al. 2011, Barbet-Massin et al. 2012) and for ensemble binary models we applied the majority rule (Araújo & New 2007). We ran the analyses using the 'dismo', 'kernlab' and 'randomForest' R packages (Liaw & Wiener 2002, Karatzoglou et al. 2004, Hijmans et al. 2017). Since we aim to understand how the future scenarios can influence the climatically suitable areas for the species in the different domains and political contexts in Brazil, we cropped the climatic suitability maps to the extent of Brazil (Figure 1).

Habitat suitability

We quantified habitat changes using historical (2000) and future (2050) forest cover maps from the Global Biosphere Management Model (GLOBIOM). Since GLOBIOM maps are not available for 2050 for all regions where *C. fasciolata* occurs, we focused our study on Brazil using models within the Brazilian political context (GLOBIOM-Brazil). Thus, we considered the specificities of this country, such as the Brazilian Forest Code and the influence of agricultural demands on deforestation (Câmara et al. 2015, Soterroni et al. 2018). The maps have ~50 × 50 km resolution at the equator (Câmara et al. 2015, Soterroni et al., 2018). Considering the habitat preferences of *C. fasciolata* (semideciduous and gallery forests, and dense forest edges; del Hoyo 1994), we selected only native forests (Soterroni et al. 2018): primary forests, managed forests (native forests exploited in a sustainable way), and protected areas. These classes vary between dense and sparse forests. To evaluate habitat suitability, we transformed the original percentage values of forest cover to range from 0 (low suitability) to 1 (high suitability; following Ramalho et al. 2021).

We chose three land-use scenarios for future comparisons: Forest Code, Illegal Deforestation Control, and No Forest Code (Soterroni et al. 2018). We refer to the Forest Code as the optimistic habitat scenario, the Illegal Deforestation Control as the moderate habitat scenario, and the No Forest Code as the pessimistic habitat scenario. The optimistic habitat scenario is based on the enforcement of Brazilian Forest Code (Brasil 2012), which provides high environmental law enforcement, full control of illegal deforestation, amnesty of legal reserve debts for small farms before 2010, legal reserves recovery after 2020, and application of environmental reserve quotas (Câmara et al. 2015, Soterroni et al. 2018). In the pessimistic habitat scenario, no law enforcement is applied except for the Atlantic Forest, where there is a specific environmental law. The moderate habitat scenario is based on partial control of illegal deforestation, with enforcement increased by 50% upon the No Forest Control scenario, and best captures the conversion of native vegetation during the historical period (Soterroni et al. 2018).

Environmental suitability

To understand how the Brazilian scenarios can influence the availability of environments for the species, we combined effects of climate and habitat changes for historical and future scenarios. First, we resampled the habitat suitability maps (i.e., land-use, considering forest cover), downscaling them to the same resolution as the climatic suitability maps cropped to extent of Brazil. We then multiplied the climatic suitability maps with the habitat suitability maps, generating environmental suitability maps (following Ramalho et al. 2021). The percentage values of environmental suitability ranged from 0 (low suitability) to 1 (high suitability). We generated one historical (historical habitat + historical climate), and six future environmental suitability scenarios: optimistic habitat + moderate climate; moderate habitat + moderate climate; pessimistic habitat + moderate climate; optimistic habitat + pessimistic climate; moderate habitat + pessimistic climate; pessimistic habitat + pessimistic climate.

Data analysis

To calculate the areas predicted to expand or contract in the future climate scenarios for the projection area in South America, Brazil, and the six domains (Amazon, Atlantic Forest, Cerrado, Caatinga, Pampa, and Pantanal), we subtracted the future binary maps from the historical binary map. For these areas, we also calculated the extent of climatically suitable areas in both historical and future maps and estimated the relative percentage of losses and gains of climatic suitability in the future scenario. We then applied a paired *t*-test to compare mean differences between the historical and future scenarios using continuous maps. We also subtracted future maps from the historical map for both habitat and environmental changes in Brazil and the four domains and calculated the relative percentage of these changes. All analyses were performed using R version 4.0.2 (R Core Team 2020) and QGIS version 3.16.3 (QGIS Development Team 2021). The R routines that provide most of this study steps are available in GitHub (<https://github.com/anaalmeida8/Data_Crax_fasciolata.git>).

Results

Climatic suitability

The ensemble models showed a TSS value of 0.77, presenting a good predictive power (Table S2). Comparing the historical and future predictions using continuous maps (Figure 2), our results showed that climatic suitability increased in both moderate (mean difference = 0.027; $t = 133.4$; p-value <0.001) and pessimistic (mean difference = 0.024; $t = 115.9$; p-value <0.001) climate scenarios in Brazil. Binary maps showed that climatically suitable areas expanded by 2.6% and 5% in the moderate and the pessimistic climate scenarios, respectively (Figure 3, Table S3). For the domains, binary maps showed that climatically suitable areas expanded in the Atlantic Forest (approximately 19% in both moderate and pessimistic climate scenarios), in the Caatinga (approximately 51% and 48% in the moderate and the pessimistic scenarios, respectively), in the Pampa (approximately 80% and 87% in the moderate and the pessimistic scenarios, respectively), and in the Amazon (approximately 4% and 10% in the moderate and the pessimistic scenarios, respectively; Figure 3, Table S3). In contrast, we observed a contraction in climatically suitable areas in the Pantanal and the Cerrado. In the Cerrado, the predictions showed a loss of approximately 10% in the moderate scenario and 9% in the pessimistic scenario, while in the Pantanal, the loss varied from 4% in the moderate scenario to 10% in the pessimistic scenario (Figure 3, Table S3). Overall, the results were very similar between the moderate and the pessimistic climate scenarios for Brazil and the four domains (Figure 3). Results for climatic suitability in South America are in Supporting Information (Results S1, Figure S2 and Table S3).

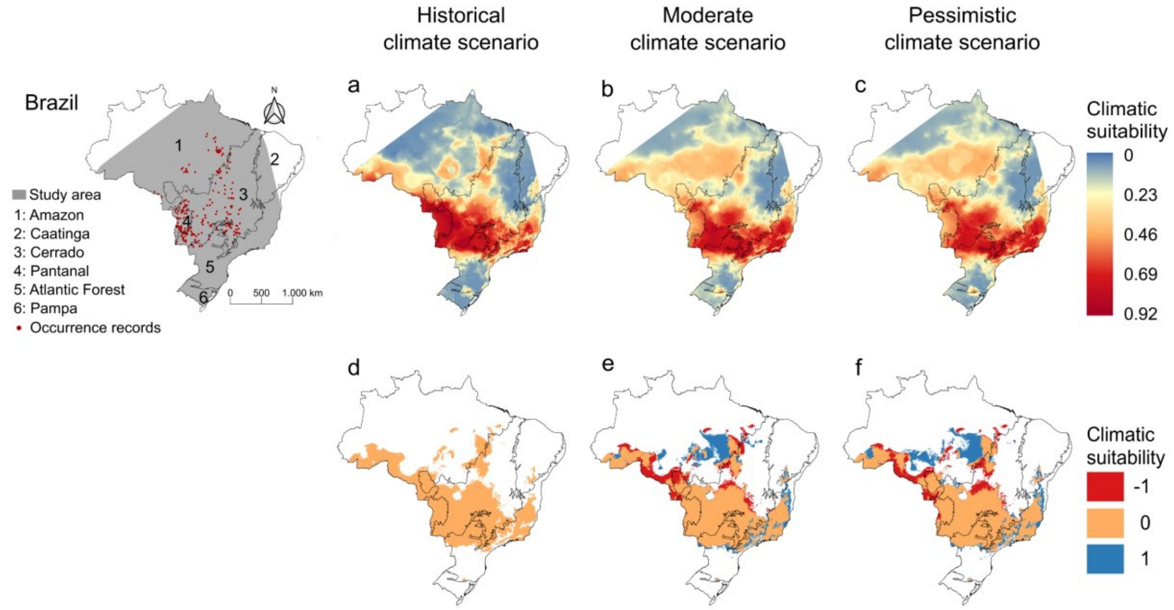

Figure 2. Climatic suitability for *Crax fasciolata* in historical (1970-2000) and future (2050) moderate and pessimistic climate scenarios in Brazil in continuous (a-c) and binary (d-f) maps. For continuous maps: climatic suitability values ranged from 0 (low suitability) to 0.92 (high suitability). For binary maps: red (-1) indicates climatic suitability loss; orange (0) indicates climatically suitable areas in the historical scenario (d) and unchanged climatically suitable areas in the future scenarios (e-f); and blue (1) indicates climatic suitability gain. Figure S2 presents the continuous and binary maps of South America with the whole projected area for the species.

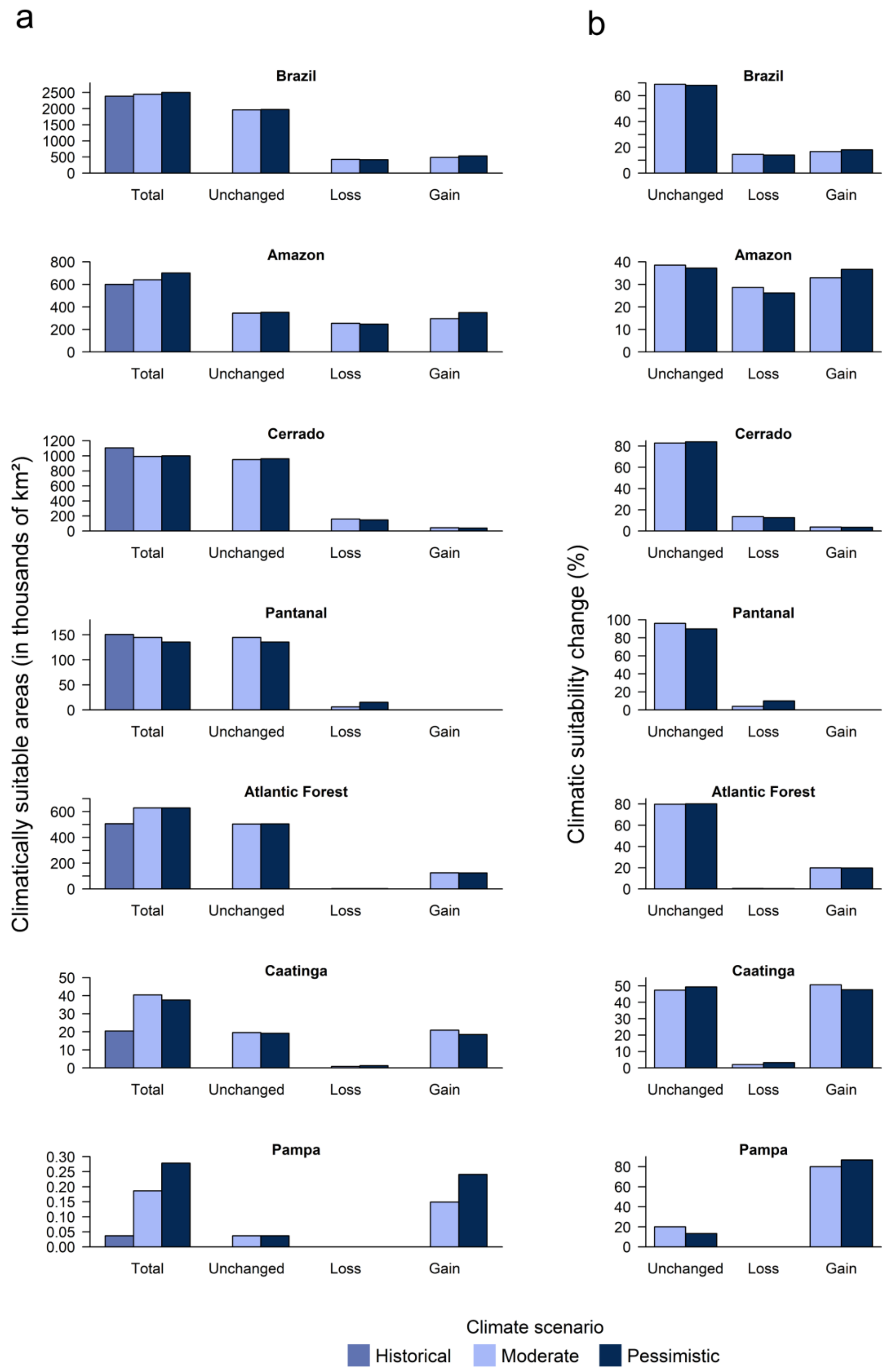

Figure 3. Historical (1970-2000) and future (2050) moderate and pessimistic climatically suitable areas in km² (a), and percent changes in future climatic suitability (b) in different regions.

Habitat suitability

Comparing the historical and future habitat scenarios, the suitable area was predicted to contract in all future scenarios in Brazil (Figure 4), but the contraction was higher in the pessimistic habitat scenario (Figure 5, Table S4). For domains, we observed contractions in the Amazon, Cerrado, Pantanal, Caatinga, and Pampa. The Cerrado and the Pantanal had the highest contraction rates, being higher in the moderate scenario in the Cerrado, and in the optimistic scenario in the Pantanal (Figure 5, Table S4). In contrast, we found that suitable habitat areas expanded in the Atlantic Forest, but in small proportion, with very similar rates among future scenarios (approximately 0.1% in all future scenarios; Figure 5, Table S4). The models predicted that changes in habitat suitability were stronger in the southern and eastern Amazon, and eastern Cerrado, reaching a loss of 87% of habitat suitability in the pessimistic scenario (Figure 4). In the Pantanal, while in the moderate and the pessimistic scenarios the habitat suitability changes were predicted to occur in the northern area, in the optimistic scenario the changes can occur in the northern and the central region (Figure 4).

Environmental suitability

Our results show that environmentally suitable areas expanded in all future scenarios in Brazil compared to the historical scenario, except for the pessimistic habitat + moderate climate scenario (Figure 6), in which we observed a contraction (Figure 7, Table S5). This expansion was greater considering the optimistic habitat + pessimistic climate scenario, and smaller considering the pessimistic habitat + pessimistic climate scenario. Environmentally suitable areas were predicted to expand in the Amazon, the Atlantic Forest, and the Pampa, and contract in the Cerrado, the Caatinga, and the Pantanal (Figure 7, Table S5). The projections showed a greater expansion in the optimistic habitat + pessimistic climate scenario and a smaller expansion in the pessimistic habitat + moderate climate scenario in the

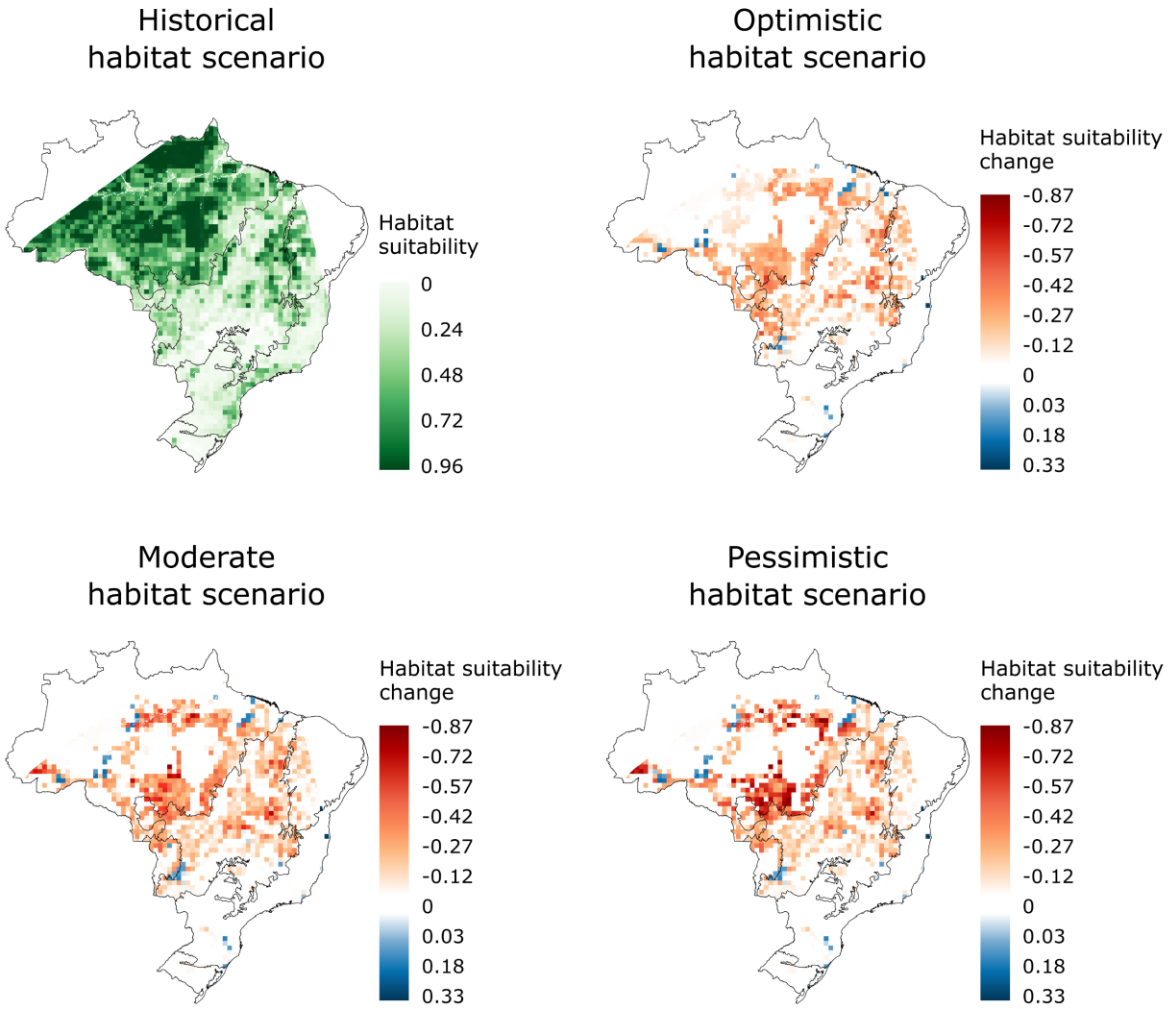

Figure 4. Habitat suitability for *Crax fasciolata* in historical (1970-2000) scenario, and changes in habitat suitability in future (2050) optimistic, intermediate, and pessimistic scenarios in Brazil. In the historical scenario: habitat suitability ranges from 0 (low suitability) to 0.96 (high suitability). In the future scenarios: red indicates habitat suitability loss; blue indicates habitat suitability gain; and white indicates no change.

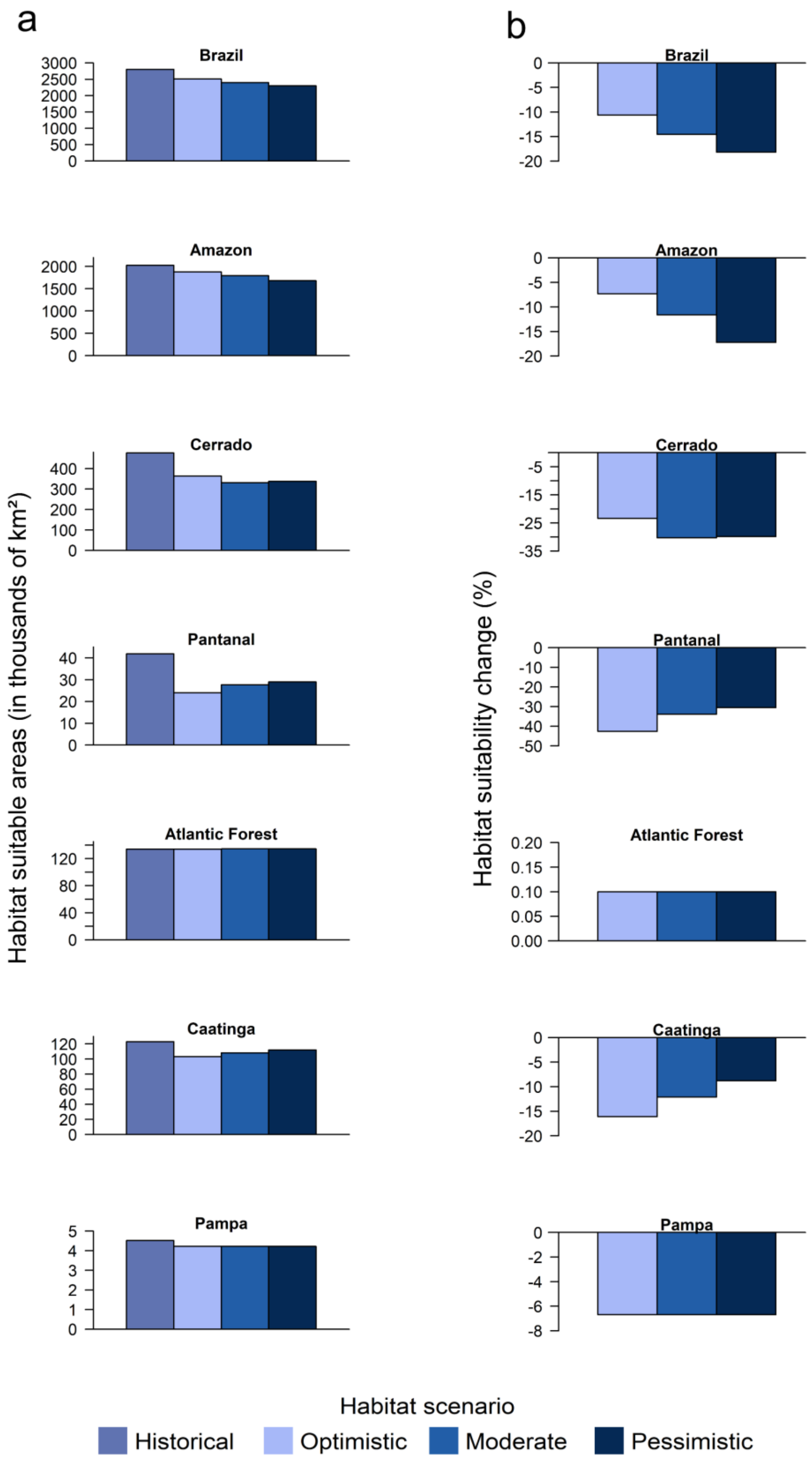

Figure 5. Historical (1970-2000) and future (2050) optimistic, moderate, and pessimistic habitat suitable areas in km² (a), and percent changes in future habitat suitability compared to the historical scenario (b) in different regions.

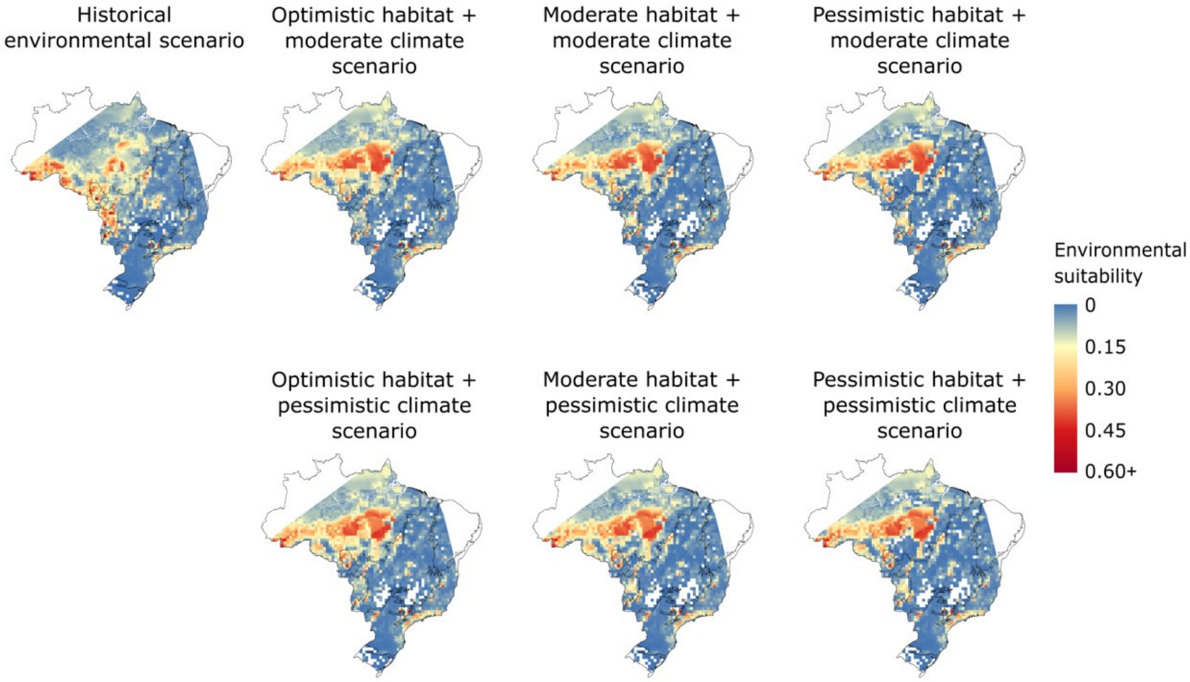

Figure 6. Environmental suitability for *Crax fasciolata* in historical (1970-2000) and future (2050) scenarios in Brazil. Environmental suitability ranges from 0 (low suitability) to 0.61 (high suitability).

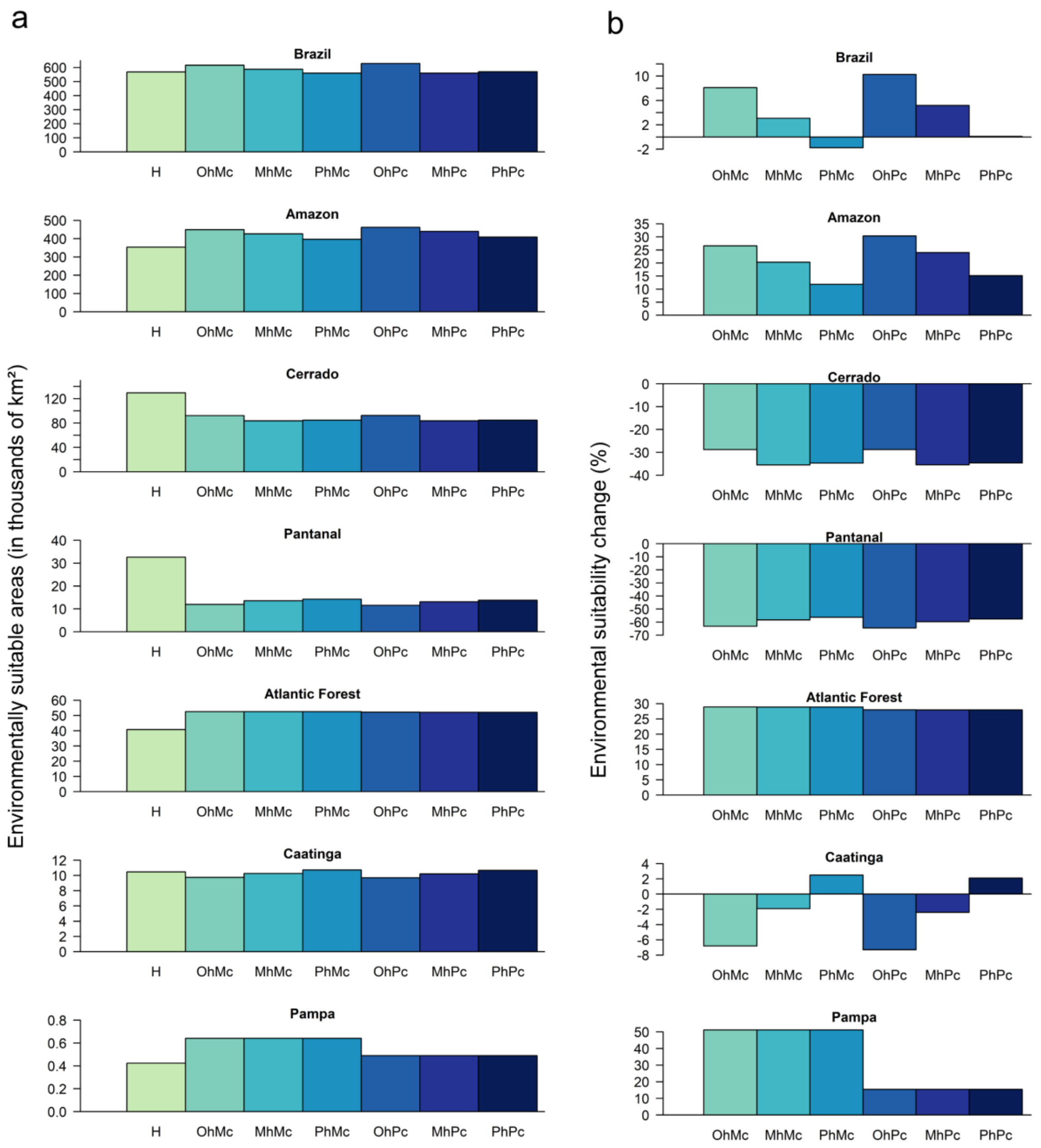

Figure 7. Historical (1970-2000) and future (2050) optimistic, moderate, and pessimistic environmentally suitable areas in km² (a), and percent changes in future environmental suitability compared to the historical scenario (b) in different regions. Environmental scenarios: H: historic; OhMc: optimistic habitat + moderate climate; MhMc: moderate habitat + moderate climate; PhMc: pessimistic habitat + moderate climate; OhPc: optimistic habitat + pessimistic climate; MhPc: moderate habitat + pessimistic climate; PhPc: pessimistic habitat + pessimistic climate.

Amazon, while in the Atlantic Forest the expansion was similar in all future scenarios. In the Pampa, we observed that the three scenarios with moderate climate had the same expansion values (51%), as well as the three scenarios with pessimistic climate (15%). Considering the Amazon domain and the extent of Brazil, habitat scenarios combined with the moderate climate scenario had lower environmental suitability than those combined with the pessimistic climate scenario, since the pessimistic scenario showed the best results for climatic suitability. For the Cerrado, the projections showed higher contractions in environmentally suitable areas in the moderate habitat + moderate climate and moderate habitat + pessimistic climate scenarios, while for the Pantanal and the Caatinga in the optimistic habitat + moderate climate and optimistic habitat + pessimistic climate scenarios (Figure 7, Table S5). In general, the models predicted that higher environmental suitability (above 0.45) was concentrated in the Amazon in the future (Figure 6).

Discussion

Our study is the first to evaluate separate and combined effects of climate and land-use changes on *C. fasciolata*, a vulnerable and charismatic species that occurs across Brazil. As expected, our results showed different future predictions of the species' distribution for each scenario, considering both the Brazilian extent and its different phytophysiognomic domains.

Evaluating climatic suitability, we found that in 2050 more climatically suitable areas could be available for *C. fasciolata* in Brazil, for both moderate and pessimistic scenarios. This result refutes our expectation that climate change could decrease suitable areas for the species. Climate change may not always be negative for birds (e.g., Peterson et al. 2001, Reside et al. 2012, Şekercioğlu et al. 2012), and its impacts evaluated in other cracids showed that responses can contract or expand geographic range depending on the species (Peterson et al. 2001). When different Brazilian domains were evaluated independently, however, our results showed different patterns for each domain. In the Amazon, Caatinga, Pampa, and Atlantic Forest domains, *C. fasciolata* presented a greater proportion of gain in suitable climatic areas. This response in the Amazon and Atlantic Forest may be associated with the "savannization" process expected to these domains due to climate change, in which humid forests are likely to be replaced by dry forests (Hutyra et al. 2005, Salazar & Nobre 2010, Scarano & Ceotto 2015, Sansevero et al. 2020). Although *C. fasciolata* is forest-dependent, this savannization process can benefit this species, as it currently inhabits dry forests in Cerrado and Pantanal, domains that often experience high variation in temperature and precipitation. In the Pampa, the increase in climatic suitability corresponds to a small area, and it is explained by the small variations in climate expected for this domain by 2050 (Souza & Manzi 2014). Since in the Caatinga it is expected an increase in temperature and a decrease in precipitation by 2050 (Souza & Manzi 2014), we expected a decrease in climatic suitability, but our results showed the opposite. It is possible that the presence of forests in the evaluated area can maintain and even increase climatic suitability for *C. fasciolata*, as observed by Ramalho et al. (2021) for amphibians. Contrastingly, in the Cerrado and Pantanal, *C. fasciolata* presented a greater proportion of loss in suitable climatic areas in future scenarios. For these domains, predictions suggest high increases in temperature and critical decreases in precipitation in these domains (Souza & Manzi 2014, Marengo et al. 2015, 2016), which can challenge thermoregulation in birds (Ruuskanen et al. 2021).

Considering habitat suitability, our results were consistent with the expectation of loss of suitable areas for *C. fasciolata* in Brazil for all future scenarios. Even accounting for the implementation of the Brazilian Forest Code (Brasil 2012), future projections show native vegetation loss in Brazil due to agricultural expansion and conversion of legal reserve surpluses (Fendrich et al. 2020, Soterroni et al. 2018). Among Brazilian domains, the Pantanal, followed by the Cerrado, showed the highest amount of habitat loss for all scenarios, as expected. This result can be explained by land-use prediction for these domains, in which agriculture and pasture activities are expected to increase (Soterroni et al. 2018, Guerra et al. 2020). This advance in pasture lands will also negatively affect Caatinga, but in smaller proportions than in Cerrado and Pantanal (Soterroni et al. 2018). However, the Caatinga showed a higher loss of habitat suitability in the optimistic scenario due to great stocks of reserves that can be legally converted into pasture and croplands (Câmara et al. 2015, Soterroni et al. 2018). For the Amazon and the Cerrado, the evaluated optimistic scenario still predicted deforestation, although in a smaller proportion than the moderate and the pessimistic habitat scenarios. Currently changes in political administration, such as lack of actions in deforestation control and dismantlement of sustainable development incentives and programs, triggered a vast deforestation in the Amazon in the last two years (Ferrante & Fearnside 2019; see estimates over the years in Trancoso 2021; but see Hurtt et al. 2011, Vale et al. 2021). Considering this, it is likely that species' response in the Amazon will follow the pessimistic habitat scenario if this relaxation of environmental laws continues for the next years. In contrast, for the Atlantic Forest, the optimistic, pessimistic and moderate habitat scenarios did not predict substantial changes. Studies suggested that, in the future, Atlantic Forest will remain almost unchanged due to additional law related to deforestation control (Atlantic Forest Law; Brasil 2006, Câmara et al. 2015), or that it is likely to show an expansion due to the enforcement and extensive ongoing reforestation programs (Rodrigues et al. 2009, Câmara et al. 2015, Crouzeilles et al. 2019).

The combined climate and habitat changes showed an increase in environmental suitability under all future scenarios for Brazil, except for the pessimistic habitat + moderate *climate* scenario. This result refutes our expectation that all scenarios could show a decrease in environmental suitability. Bird species with a wide distribution is likely to show minimal overall range loss under environmental changes (Travis 2003, Jetz et al. 2007), particularity that is an advantage for *C. fasciolata*. However, we observed that, in the Pantanal and the Cerrado, *C. fasciolata* can experience a drastic contraction in its potential distribution in all future environmental scenarios, which could make the species more threatened, especially because *C. fasciolata* is abundant only in the Pantanal (Ridgely 2010). In the Caatinga, the decrease in habitat suitability is the main driver in the reduction of environmentally suitable areas, mainly in the optimistic scenario and the implementation of the Brazilian Forest Code. In the Amazon, the Atlantic Forest, and the Pampa, the climatically suitable areas in 2050 can cover more habitat suitable areas, and consequently, new environmentally suitable areas may be available to be colonized by *C. fasciolata*. However, studies suggest that cracids have limited dispersal ability (Bernardo et al. 2011, Hosner et al. 2016), which may hinder the species from reaching suitable areas in the future in the Atlantic Forest and Amazon, as well as the suitable areas in the Caatinga and the Pampa (where the species does not currently occur), especially in the Pampa, which is very distant from the current ranges of the species.

Following our findings, we reinforce the importance of the Brazilian Forest Code to protect native vegetation, as well as the urgent call for environmental policies for reforestation and effective deforestation control (e.g., Rodrigues et al. 2009, Brazil 2017). In addition, conserving a large extent in areas with higher environmental suitability, such as in the Amazon and the Atlantic Forest, can help to maintain both viable populations and good habitat conditions to receive new individuals of *C. fasciolata* that can disperse from unsuitable areas (Hole et al. 2011, Borges & Loyola 2020, Rezende et al. 2020). But it is important to emphasize that the Atlantic Forest is highly fragmented due to the agricultural development and urbanization, which reduces landscape permeability (Ribeiro et al. 2009, Rosa et al. 2021), preventing *C. fasciolata* from reaching suitable areas in this domain. In this sense, restoring native vegetation and creating corridors in the Atlantic Forest to increase connectivity can facilitate *C. fasciolata* to reach and colonize the newly environmentally suitable areas. Lastly, we predicted that *C. fasciolata* can lose a drastic amount of environmentally suitable areas in the Pantanal and Cerrado, which deserve especial attention. Although we observed poor habitat conditions for the species in these two domains in 2050, we predicted that they can still maintain climatically suitable conditions. Thus, restoring natural vegetation increases environmental suitability in the Pantanal and Cerrado, which becomes a good strategy for *C. fasciolata* conservation in these domains.

Beyond the concern about habitat loss and climate change as the main drivers of bird extinction, hunting is a strong pressure currently threatening *C. fasciolata* (BirdLife International 2016) – also observed in other cracids (Brooks 2006, Bonfim et al. 2018, Rios et al. 2021). In addition, land-use changes at local scale (not evaluated in this study), such as patch isolation and edge-effects strongly threaten many species, as well as *C. fasciolata* (BirdLife International 2016), reducing habitat connectivity and resource acquisition, and changing microhabitat conditions (Andrén 1994, Lees & Peres, 2009, Ewers & Banks-Leite 2013). In combination, hunting, habitat loss (at national, regional and local scales), and climate change put cracids as the most threatened family of birds in the Neotropics (Brooks 2006). Although we did not evaluate hunting pressure, our study provides a primary evaluation of environmental impacts on *C. fasciolata*, which can help conservationists to identify the best management actions, creating strategies for species to adapt to new environmental conditions. And despite the limitation of carrying out the study with only one species, the study represents an important step because it tests, for the first time, how a wide-ranging species can respond in different ways to changes in climate and land-use depending on the domain in which it is distributed. This implies great importance for conservation, as this same approach can be used for other species categorized as threatened by the IUCN.

Conclusion

We found that environmental suitability for *C. fasciolata* can increase in Brazil in the future, but these effects vary according to the scenario and the domain. Although habitat suitable areas may decrease by 2050, climatically suitable areas can increase, and the combination of these factors can increase overall environmental suitability in most future scenarios. Scenarios that include strict environmental law enforcement will be the best for the conservation of *C. fasciolata*, but we must be careful when analyzing each domain separately. The Pantanal and the Cerrado deserve special attention since our projections show that *C. fasciolata* may experience the highest losses in these domains. Although new environmentally suitable areas may be available for *C. fasciolata* in all future scenarios compared to the historical scenario, some factors may hinder the species from reaching these areas, such as low dispersal ability, geographical barriers, and reduced landscape permeability (Schloss et al. 2012). Thus, conserving the remaining native vegetation and restoring deforested areas can help in persistence and survival of *C. fasciolata* in Brazil. More than that, implementing conservation strategies for this species helps in achieving other conservation goals, since *C. fasciolata* acts as an umbrella species: (a) as a frugivorous species, *C. fasciolata* can help in forest regeneration through seed dispersal (Brooks 2006); (b) due to its wide distribution, conserving its habitat contributes to conserve large and different ecosystems across Brazil, as well as to protect other threatened species living in the same regions.

Supporting information for chapter 1

Results S1. Results for climatic suitability in South America.

Comparing the historical and future predictions using continuous maps, our results showed that, in South America (the original potential species distribution of *Crax fasciolata*) climatic suitability decreased significantly in both moderate (mean difference = 0.008; t = 45.1; p-value <0.001) and pessimistic (mean difference = 0.004; t = 23.2; p-value <0.001) scenarios. Binary maps showed that climatically suitable areas in South America could contract by approximately 6% in both moderate and pessimistic climate scenarios (see Figure S2, Table S3).

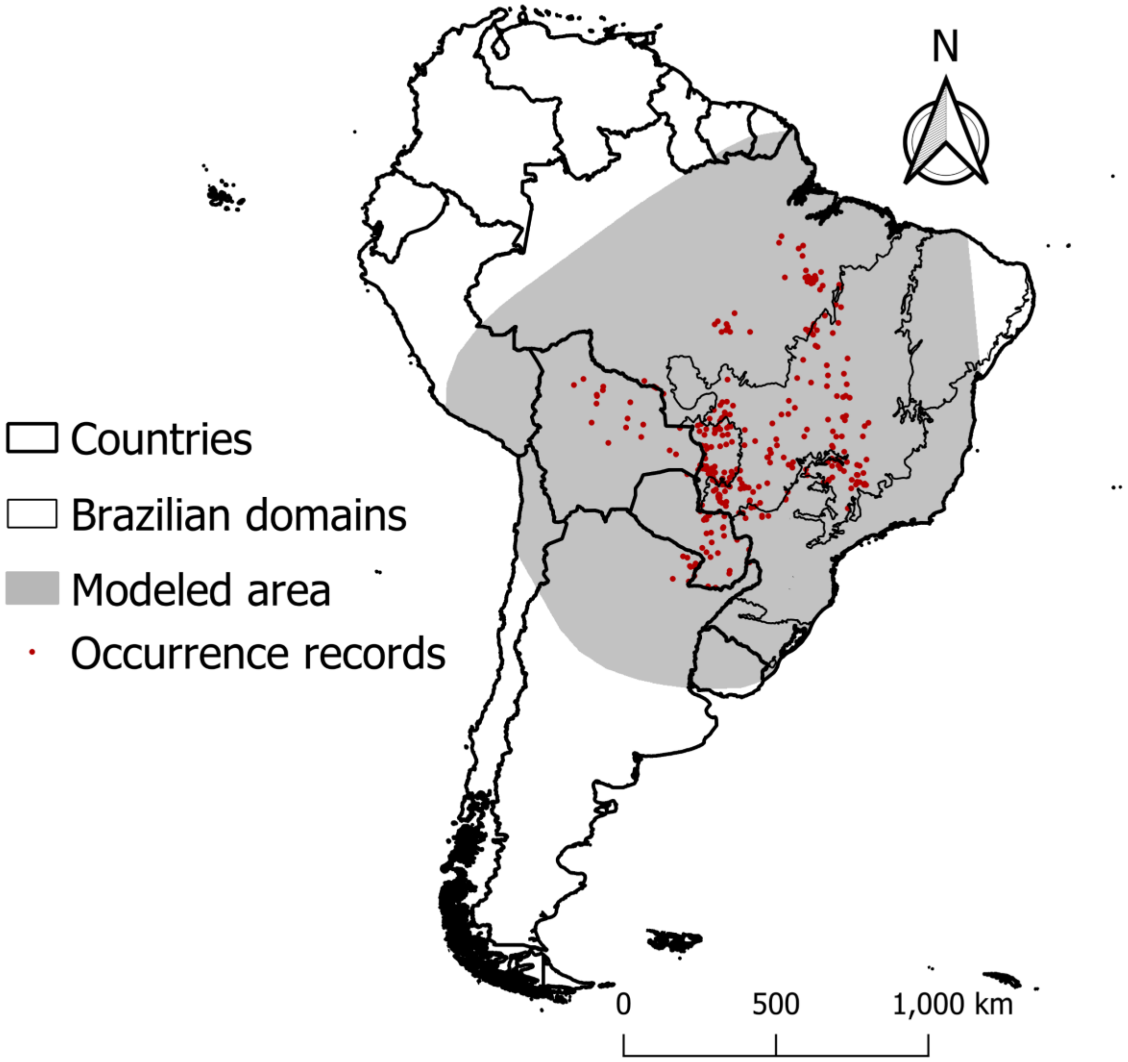

Figure S1. Distribution of occurrence records of *Crax fasciolata* in South America and area used for ecological niche modeling. We defined the area to be modeled as the minimum convex polygon around records + buffer of 20%. The 304 occurrence records are distributed as follows: (i) by country, 14 in Argentina, 22 in Bolivia, 243 in Brazil, and 25 in Paraguai; (ii) by Brazilian biome, 39 in the Amazon, 54 in the Pantanal, 139 in the Cerrado, and 11 in the Atlantic Forest.

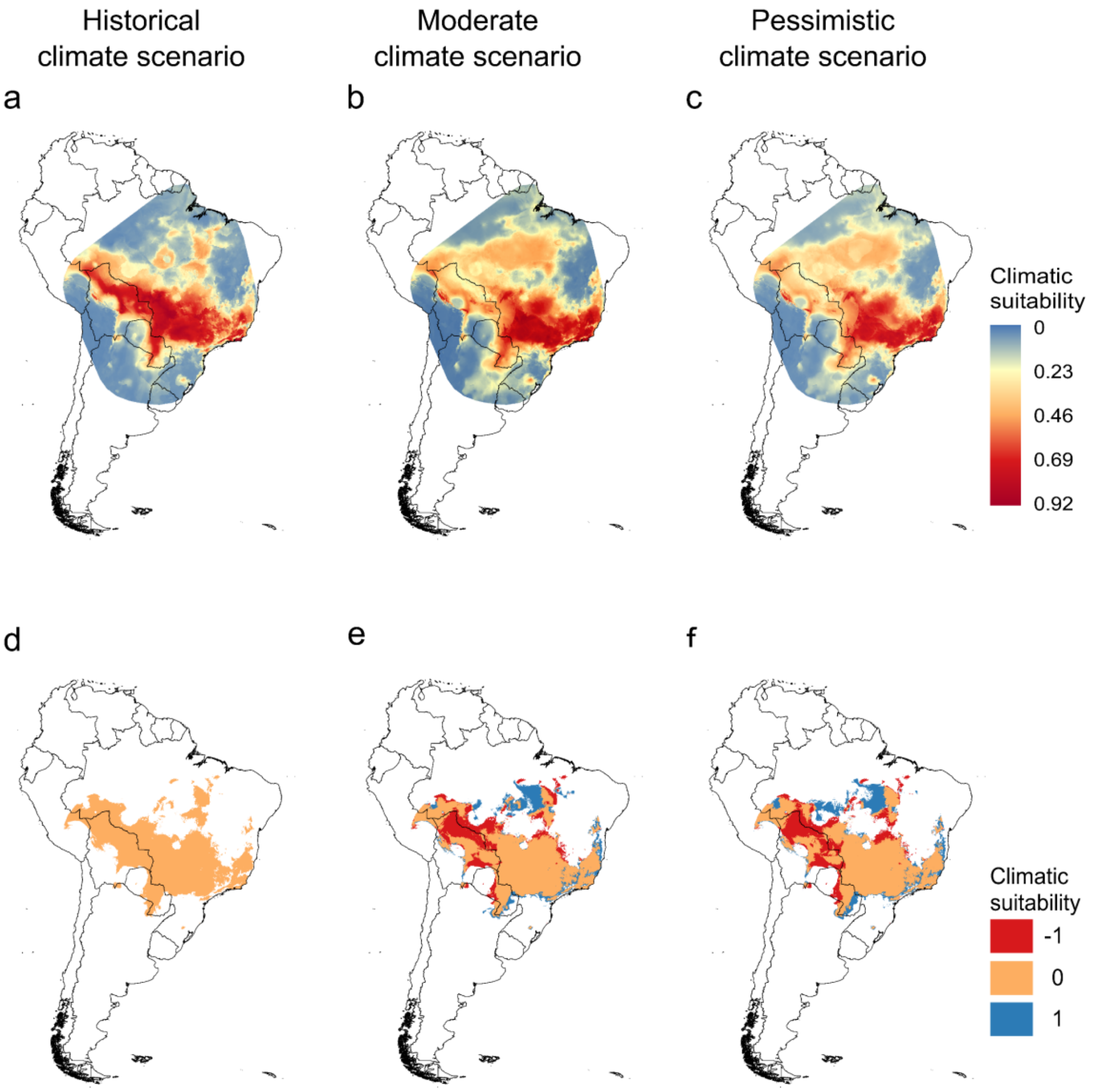

Figure S2. Climatic suitability for *Crax fasciolata* in historical (1970-2000) and future (2050) moderate and pessimistic climate scenarios in South America in continuous and binary maps. For continuous maps (a-c): climatic suitability ranges from 0 (low suitability) to 0.92 (high suitability). For binary maps (d-f): red (-1) indicates climatic suitability loss; orange (0) indicates climatically suitable areas in the historical scenario (d) and unchanged climatically suitable areas in the future scenarios (e-f); and blue (1) indicates climatic suitability gain.

Table S1. True Skill Statistic (TSS) values for each model generated by 10-fold cross-validation of the four niche modeling algorithms.

Model	**Replica**	**TSS value**
BIOCLIM	1	0.729
	2	0.761
	3	0.799
	4	0.755
	5	0.722
	6	0.729
	7	0.770
	8	0.817
	9	0.751
	10	0.771
Random Forest	1	0.842
	2	0.763
	3	0.816
	4	0.816
	5	0.842
	6	0.803
	7	0.750
	8	0.816
	9	0.776
	10	0.776
MaxEnt	1	0.783
	2	0.760
	3	0.757
	4	0.788
	5	0.811
	6	0.763
	7	0.783
	8	0.784
	9	0.774
	10	0.791
SVM	1	0.766
	2	0.735
	3	0.771
	4	0.722
	5	0.755
	6	0.712
	7	0.762
	8	0.784
	9	0.791
	10	0.739

Table S2. True Skill Statistic (TSS) values and uncertainties resulted from the individual models and the ensemble of the ecological niche modeling of *Crax fasciolata* for each scenario (historical and future moderate and pessimistic). TSS value of the ensemble was calculated by the average of the TSS of all replicas.

Model	**TSS (mean)**	**Uncertainty**	
		Scenario	**Standard deviation**
BIOCLIM	0.761	Historical	0.027
		Future moderate	0.022
		Future pessimistic	0.020
Random Forest	0.800	Historical	0.105
		Future moderate	0.106
		Future pessimistic	0.105
MaxEnt	0.779	Historical	0.070
		Future moderate	0.104
		Future pessimistic	0.106
SVM	0.754	Historical	0.078
		Future moderate	0.075
		Future pessimistic	0.078
Ensemble	0.773	Historical	0.070
		Future moderate	0.077
		Future pessimistic	0.077

Table S3. Climatically suitable areas in historical (1970-2000) and future (2050) scenarios, and changes in future climatic suitability in different regions compared to the historical scenario.

Region	**Climate scenario**	**Total suitable area (km²)**	**Unchanged suitable area (km²)**	**Unchanged suitability (%)**	**Decreased suitable area (km²)**	**Suitability loss (%)**	**Increased suitable area (km²)**	**Suitability gain (%)**
South America	Historical	3,283,364	-	-	-	-	-	-
	Moderate	3,051,757	2,494,687	63.0	788,676	21.4	557,070	15.6
	Pessimistic	3,055,622	2,441,714	65.4	841,649	20.3	613,901	14.4
Brazil	Historical	2,382,553	-	-	-	-	-	-
	Moderate	2,445,467	1,959,856	68.9	422,697	14.5	485,611	16.7
	Pessimistic	2,500,483	1,969,859	68.1	412,693	13.9	530,624	18.0
Amazon	Historical	599,649	-	-	-	-	-	-
	Moderate	639,978	344,310	38.5	255,159	28.6	295,668	32.9
	Pessimistic	700,544	351,297	37.2	247,542	26.2	348,617	36.6
Pantanal	Historical	150,698	-	-	-	-	-	-
	Moderate	144,635	144,615	96.0	6,083	4.0	20	0.0
	Pessimistic	135,425	135,425	89.9	15,273	10.1	0	0.0

Table S3. (continued)

Region	Climate scenario	Total suitable area (km²)	Unchanged suitable area (km²)	Unchanged suitability (%)	Decreased suitable area (km²)	Suitability loss (%)	Increased suitable area (km²)	Suitability gain (%)
Cerrado	Historical	1,106,382	-	-	-	-	-	-
	Moderate	991,755	948,044	82.8%	158,338	13.5%	43,711	3.7%
	Pessimistic	998,423	959,182	84.0%	147,199	12.6%	39,241	3.4%
Atlantic Forest	Historical	505,801	-	-	-	-	-	-
	Moderate	628,701	503,497	79.8%	2,304	0.4%	125,204	19.8%
	Pessimistic	628,383	504,311	80.1%	1,490	0.2%	124,072	19.7%
Caatinga	Historical	20,386	-	-	-	-	-	-
	Moderate	40,451	19,552	47.4%	834	2.0%	20,898	50.6%
	Pessimistic	37,648	19,157	49.3%	1,230	3.2%	18,492	47.6%
Pampa	Historical	37	-	-	-	-	-	-
	Moderate	186	37	20.0%	0	0.0%	149	80.0%
	Pessimistic	278	37	13.3%	0	0.0%	241	86.7%

Table S4. Habitat suitable areas in historical (2000) and future (2050) scenarios, and changes in future habitat suitability values in different regions compared to the historical scenario. Negative values indicate loss, while positive values indicate gains in habitat suitability.

Region	Land-use scenario	Habitat suitable area (km²)	Changes in habitat suitability (%)
Brazil	Historical	2,803,689	-
	Optimistic	2,507,086	-10.6
	Moderate	2,396,645	-14.6
	Pessimistic	2,295,441	-18.2
Amazon	Historical	2,025,713	-
	Optimistic	1,877,922	-7.3
	Moderate	1,791,698	-11.6
	Pessimistic	1,678,408	-17.2
Pantanal	Historical	41,823	-
	Optimistic	23,970	-42.6
	Moderate	27,613.	-33.9
	Pessimistic	28,990	-30.5
Cerrado	Historical	476,772	-
	Optimistic	363,854	-23.4
	Moderate	330,909	-30.3
	Pessimistic	337,624	-29.9

Table S4. (continued)

Region	Land-use scenario	Habitat suitable area (km²)	Changes in habitat suitability (%)
Atlantic Forest	Historical	134,032	-
	Optimistic	134,135	0.1
	Moderate	134,201	0.1
	Pessimistic	134,212	0.1
Caatinga	Historical	122,872	-
	Optimistic	103,030	-16.1
	Moderate	108,050	-12.1
	Pessimistic	112,036	-8.8
Pampa	Historical	4,518	-
	Optimistic	4,213	-6.7
	Moderate	4,213	-6.7
	Pessimistic	4,213	-6.7

Table S5. Environmentally suitable areas in historical (2000) and future (2050) scenarios, and changes in future environmental suitability values in different regions compared to the historical scenario. Negative values indicate loss, while positive values indicate gains in environmental suitability.

Region	**Scenario**	**Environmentally suitable area (km²)**	**Changes in environmental suitability (%)**
Brazil	Historical	567,851	-
	Optimistic habitat + Moderate climate	616,163	8.1
	Moderate habitat + Moderate climate	587,306	3.1
	Pessimistic habitat + Moderate climate	559,588	-1.8
	Optimistic habitat + Pessimistic climate	628,667	10.3
	Moderate habitat + Pessimistic climate	559,568	5.2
	Pessimistic habitat + Pessimistic climate	570,433	0.1
Amazon	Historical	354,040	-
	Optimistic habitat + Moderate climate	449,019	26.6
	Moderate habitat + Moderate climate	426,752	20.3
	Pessimistic habitat + Moderate climate	396,875	11.8
	Optimistic habitat + Pessimistic climate	462,448	30.4
	Moderate habitat + Pessimistic climate	439,963	24.0
	Pessimistic habitat + Pessimistic climate	408,699	15.2

Table S5. (continued)

Region	Scenario	Environmentally suitable area (km²)	Changes in environmental suitability (%)
Pantanal	Historical	32,614	-
	Optimistic habitat + Moderate climate	11,966	-63.3
	Moderate habitat + Moderate climate	13,554	-58.4
	Pessimistic habitat + Moderate climate	14,213	-56.3
	Optimistic habitat + Pessimistic climate	11,547	-64.6
	Moderate habitat + Pessimistic climate	13,118	-59.7
	Pessimistic habitat + Pessimistic climate	13,765	-57.7
Cerrado	Historical	129,548	-
	Optimistic habitat + Moderate climate	92,250	-28.8
	Moderate habitat + Moderate climate	83,577	-35.5
	Pessimistic habitat + Moderate climate	84,613	-34.7
	Optimistic habitat + Pessimistic climate	92,323	-28.8
	Moderate habitat + Pessimistic climate	83,643	-35.5
	Pessimistic habitat + Pessimistic climate	84,650	-34.7

Table S5. (continued)

Region	Scenario	Environmentally suitable area (km²)	Changes in environmental suitability (%)
Atlantic Forest	Historical	40,775	-
	Optimistic habitat + Moderate climate	52,557	28.9
	Moderate habitat + Moderate climate	52,533	28.9
	Pessimistic habitat + Moderate climate	52,540	28.9
	Optimistic habitat + Pessimistic climate	52,178	28.0
	Moderate habitat + Pessimistic climate	52,154	27.9
	Pessimistic habitat + Pessimistic climate	52,160	27.9
Caatinga	Historical	10,459	-
	Optimistic habitat + Moderate climate	9,740	-6.8
	Moderate habitat + Moderate climate	10,257	-1.9
	Pessimistic habitat + Moderate climate	10,716	2.5
	Optimistic habitat + Pessimistic climate	9,690	-7.3
	Moderate habitat + Pessimistic climate	10,208	-2.4
	Pessimistic habitat + Pessimistic climate	10,679	2.1

Table S5. (continued)

Region	Scenario	Environmentally suitable area (km²)	Changes in environmental suitability (%)
Pampa	Historical	424	-
	Optimistic habitat + Moderate climate	641	51.2
	Moderate habitat + Moderate climate	641	51.2
	Pessimistic habitat + Moderate climate	641	51.2
	Optimistic habitat + Pessimistic climate	490	15.4
	Moderate habitat + Pessimistic climate	490	15.4
	Pessimistic habitat + Pessimistic climate	490	15.4

Chapter 2: Extant protected areas are not enough to conserve endemic and threatened Atlantic Forest birds

Abstract

Climate and land-use changes are pivotal factors responsible for shifting ecological dynamics worldwide, leading to biodiversity loss, with tropical forests experiencing the most significant impact. In particular, the Neotropical Atlantic Forest biodiversity can suffer a contraction in suitable areas in the future due to intense deforestation and climate change, especially vulnerable groups with high rates of endemism, such as birds. Determining priority areas for conservation and restoration is a key action to mitigate the negative impacts of climate and land-use changes on biodiversity. In this study, we used ecological niche modeling to generate current and future climate models and integrated them with forest cover data. This allowed us to assess the present and future potential distribution of endemic and threatened Atlantic Forest bird species, enabling the identification of priority areas for conservation and restoration initiatives for these birds. We found that most bird species tend to lose suitable areas in the future. Also, a small portion of priority areas for conservation lies inside of extant protected areas, in both current and future scenarios, putting these bird species at a higher risk of extinction due to expected future climate and land-use changes. Moreover, a substantial proportion of areas demonstrating high climatic suitability in both current and future scenarios require reforestation efforts to achieve conservation targets. All these concerns highlight the urgent need for establishing new protected areas and implementing restoration actions in the Atlantic Forest.

Introduction

Human pressures are expanding and reaching even remote and protected areas on Earth and threatening global biodiversity (Venter et al. 2016, Williams et al. 2020). An intense degradation of forests caused by many human pressures (e.g., mining, conversion to agriculture, urbanization, fire, and pollution) has been detected across different biogeographic realms (Lewis et al. 2015, Grantham et al. 2020). Especially in tropical regions, land-use changes have increased fast in the last decades, which changes forest function and health, and poses high risks to the tropical biodiversity (Lambin et al. 2003, Lewis et al. 2015, Song et al. 2018, Hoang & Kanemoto 2021). In addition, tropical forests are vulnerable to climate change since increased temperature and droughts are stressors for plants and can lead to a replacement of forests by savannas, which present more heat-tolerant plant species (Souza & Manzi 2014, Corlett 2016). Climate change also negatively impacts other groups since it challenges thermoregulation in many tropical species, especially those with narrow thermal-tolerance ranges (Tewksbury et al. 2008, Brodie et al. 2012, Khaliq et al. 2014, Ruuskanen et al. 2021). In combination, climate and land-use changes can shift many ecological dynamics, such as trophic interactions (van der Putten et al. 2004), forest structure (Asner et al. 2010), species dispersion and distribution (Della Roca & Milanesi 2020, Ramalho et al. 2021, Koo & Park 2022), and carbon storage (Kaplan et al. 2012), being the main drivers of biodiversity loss (Sala et al. 2000, Mantyka-Pringle et al. 2015), which can be even more severe in tropical forests compared to temperate ecosystems (Asner et al. 2010, Lewis et al. 2015).

In particular, the Atlantic Forest–a threatened, and extremely vulnerable biodiversity hotspot in Brazil–can suffer intense deforestation and climatic impacts in the future (Asner et al. 2010). Combined climate and land-use changes can affect many groups in this realm, such as birds, amphibians, and mammals, causing contraction in suitable areas and even replacement of forest-dependent species by species that live in open savannas (Jetz et al.

2007, Loiselle et al. 2010, Gouveia et al. 2016, Sales et al. 2020, Ramalho et al. 2021). As one of the top five biodiversity hotspots of the world, the Atlantic Forest shows an elevated rate of species and endemism (da Fonseca et al. 2004, Jenkins et al. 2015, Marques & Grelle 2021), such as birds, with 223 endemic species (Vale et al. 2018b), out of which 65 are threatened (IUCN 2022, Ministério do Meio Ambiente 2022). The main threat to these birds listed in the IUCN Red List of Threatened Species is habitat loss, but climate change is also a critical hazard to some of them (IUCN 2022). Given the critical status of these species, which are on the brink of extinction, it is urgent to implement conservation measures for them.

To mitigate the negative impacts of climate and land-use changes on biodiversity it is crucial to determine priority areas for conservation and restoration (Crouzeilles et al. 2013, Zwiener et al. 2017, Vale et al. 2018a, Strassburg et al. 2020). A relevant tool to assess the locations that provide suitable habitats for the species is ecological niche modeling (Zwiener et al. 2017, Vale et al. 2018a). We can use correlative ecological niche modeling, combining species' occurrence records, environmental variables, and mathematical algorithms, to predict potential geographic distributions of species, enhancing our comprehension of patterns across current, past, and future periods (Peterson et al. 2011, Sillero et al. 2021). Some studies that used ecological niche modeling to evaluate the effects of environmental changes on birds in the Atlantic Forest have considered only climatic variables (Marini et al. 2010, Souza et al. 2011, Hoffmann et al. 2015, Vale et al. 2018a, Mota et al. 2022). The studies that combined climate models with landscape metrics are still scarce and did not evaluate priority areas for conservation actions or did not consider future changes (e.g., Loiselle et al. 2010, Oliveira-Silva et al. 2022). Planning areas for conservation and restoration must be done considering both current and future species' distribution to improve the effectiveness of conservation actions, since environmental changes can redistribute the species' ranges (Hannah et al. 2007, Loyola et al. 2013, Vale et al. 2018a). In addition, two of these studies only focus on identifying priority areas for conservation (Marini et al. 2010, Vale et al. 2018a), and to our knowledge, there is no study developed to identify priority areas for both conservation and restoration for birds in the Atlantic Forest. Furthermore, the studies that used ecological niche modeling to predict areas with high suitability for biodiversity, and thus identify priority areas for conservation, did not quantify the extent to which these areas are encompassed within extant Protected Areas. Most of the Atlantic Forest remains unprotected (Pacheco et al. 2018), and quantify the proportion of areas exhibiting high suitability for species that is covered by Protected Areas is crucial for assessing whether these extant reserves are efficient in safeguarding biodiversity (Zwiener et al. 2017).

Here, we combined ecological niche models and forest cover to evaluate the current and future potential distribution of endemic and threatened Atlantic Forest bird species. We aimed to identify priority areas for conserving these species, and also areas suitable for restoration efforts, which are important to enhance connectivity in the Atlantic Forest and facilitate species dispersion (Rezende et al. 2020). Furthermore, we aimed to assess how much of these areas overlap the Protected Areas in the Atlantic Forest. We expected that the extant Protected Areas would only encompass a small fraction of the regions exhibiting higher suitability in both present and future scenarios, which justifies the creation of new protected areas.

Material and Methods

Study area

We followed the Atlantic Forest delimitation called "Integrator Limit" proposed by Muylaert et al. (2018), which integrates four different maps for this realm. The study area is 1,619,514 km² in extent (33° 46' 0.7314" S to 2° 48' 31.2942" S; 57° 53' 35.0874" W to 28° 50' 9.2754" W), distributed in Brazil, Argentina and Paraguay (Figure 1). Due to its wide latitudinal and

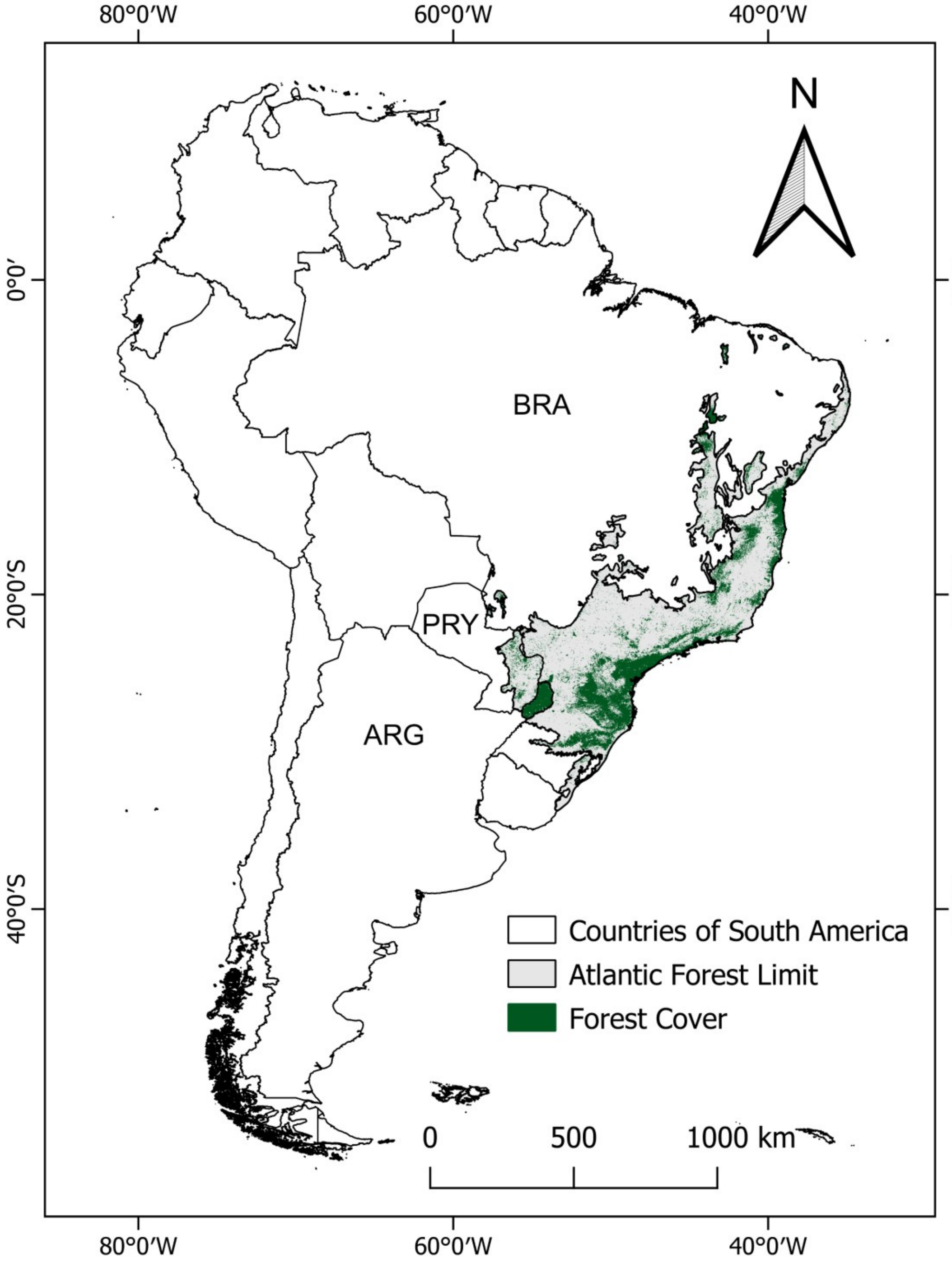

Figure 1. Distribution of the Atlantic Forest realm (following Muylaert et al. 2018). The Atlantic Forest comprises three countries: Brazil (BRA), Argentina (ARG), and Paraguay (PRY). Forest cover area comprises the set of forest classes obtained from MODIS Collection 6 (see "Percentage of forest cover" section).

longitudinal ranges, the Atlantic Forest presents highly heterogeneous environmental conditions, resulting in different phytophysiognomies, including dense and open forests, semidecidual forests, Araucaria mixed forest, restingas, and mangroves (Ribeiro et al. 2009, Câmara et al. 2015, Marques et al. 2021). The Atlantic Forest is highly fragmented due to increase in agricultural activities (Fonseca et al. 2009, Joly et al. 2014), and urbanization (Scarano & Ceotto 2015), where 83.4 % of the fragments are less than 50 ha (Ribeiro et al. 2009), putting many endemic species at risk (Rezende et al. 2018).

Species database

We followed Moreira-Lima (2013), Vale et al. (2018b), IUCN (2022), and Ministério do Meio Ambiente (2022) to select the endemic and threatened Atlantic Forest bird species, totalizing 65 species. We gathered the occurrence records from the Global Biodiversity Information Facility – GBIF (see Table S1 for DOI), Distributed Information System for Biological Collections – *species*Link (speciesLink 2021), Brazilian Biodiversity Information System – SiBBr (SiBBr 2021), VertNet (VertNet 2016), iNaturalist (iNaturalist 2021), eBird (eBird Basic Dataset 2022), and literature sources (see Hasui et al. 2017). We first cleaned the occurrence records by removing: (i) duplicated records; (ii) records with latitude and longitude equal to zero; (iii) records before 1970 (as WorldClim version 2.1 considers climate data for historical variables since 1970); (iv) imprecise occurrences (e.g., coordinates assigned to country capitals and centroids, and biodiversity institutions); (v) and records beyond the study area. Then, we left one record per cell as a geographical cleaning to reduce spatial autocorrelation (Rezende et al. 2020; see "Climatic-based models" section for cell-size resolution). Finally, we selected only the species with a minimum of 30 records after the cleaning process to build climatic models with high accuracy (see Stockwell & Peterson 2002, Sillero et al. 2021), totalizing 55 bird species (Table S2).

Climate-based models

Using Ecological Niche Modeling (ENM) we predicted potentially suitable areas in current and future climate scenarios, generating continuous and binary climatic suitability maps. We constructed correlative climate-based models under current (average for 1970–2000) and future (2070, average for 2061–2080) conditions. For the current model, we downloaded 19 bioclimatic variables at 30 arc-seconds resolution (~1 × 1 km at the equator) from WorldClim version 2.1 (Fick & Hijmans 2017). Then, we extracted the values of the bioclimatic variables and calculated the ecological-niche factor analysis (ENFA; Hirzel et al. 2002), selecting a subset of the bioclimatic variables. We obtained five axes (Figure 2) representing more than 90% of the cumulative proportion of variance (Table 1). Then, we selected the variables with higher percent contribution values in each axis (followed by a biological importance criteria): Mean Diurnal Range, Isothermality, Mean Temperature of Warmest Quarter, Precipitation of Wettest Quarter, and Precipitation of Driest Quarter (Table 1).

For the future model, we used data from Global Circulation Models (GCMs) from the Coupled Model Intercomparison Project Phase 6 (CMIP6). We selected the two best GCMs for South America considering Cannon (2020) and availability in WorldClim version 2.1 for 30 arc-seconds resolution: MIROC6 (Takemura 2019) and UKESM1-0-LL (O'Connor 2019). Considering challenges to mitigate greenhouse gas emissions, we chose the SSP3-7.0 scenario from the Shared Socioeconomic Pathways (SSPs). This scenario is based on the focus of countries on achieving energy and food security goals, intense consumption, the persistence of inequalities, high population growth in developing countries, and low international priority in conservation actions (O'Neill et al. 2016, Riahi et al. 2017).

We chose four algorithms for ENM, as combining multiple algorithms increases the accuracy of the predictions (Araújo et al. 2005): Bioclimate analysis (BIOCLIM) as a presence-only modeling method; Maximum Entropy (MaxEnt) as a presence-background

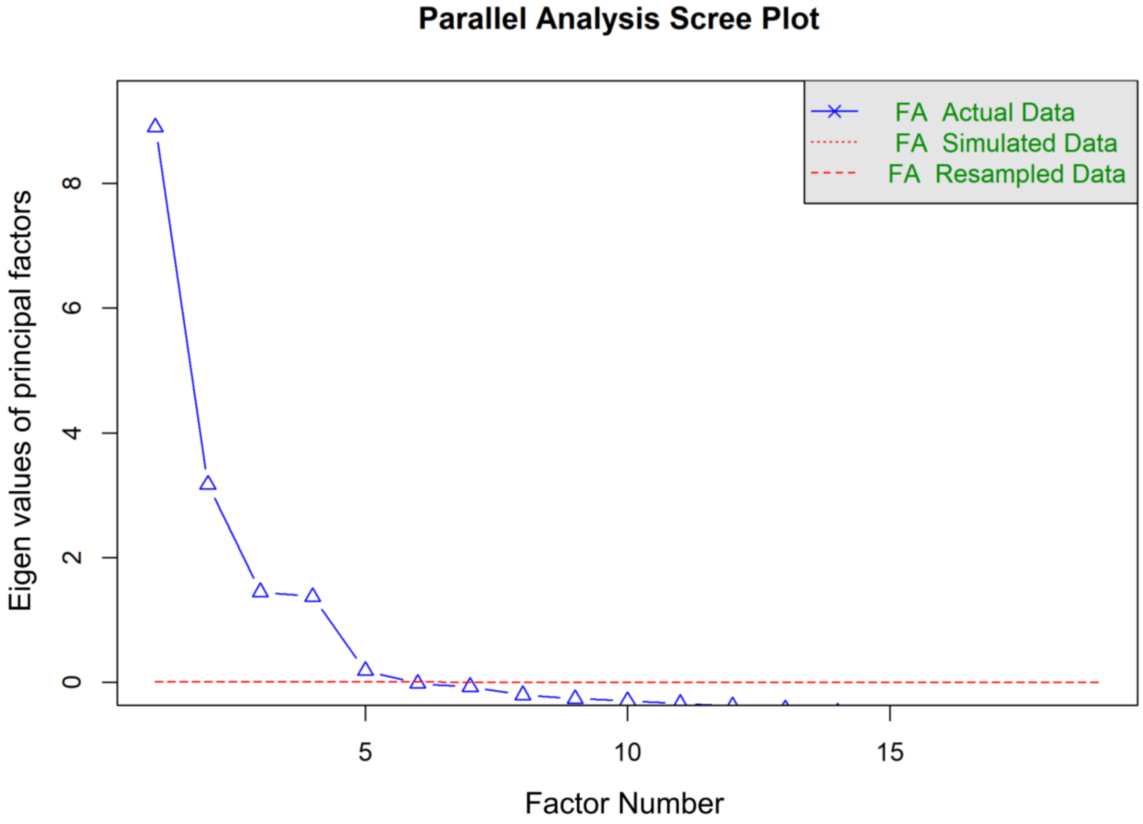

Figure 2. Scree plot for climatic variables axes selection. The number of triangles (in blue line) above the red dashed line (FA Resampled Data) indicates the number of axes that represent more than 90% of the cumulative proportion of variance.

Table 1. Loadings of factor analysis for each axis used in climatic variables selection. MR1 to MR5 are the axes selected by factor analysis, representing more than 90% of the cumulative proportion of variance. Selected variables are bolded.

Bioclimatic variables	**MR1**	**MR4**	**MR2**	**MR3**	**MR5**
BIO1 = Annual Mean Temperature	0.92	0.33	0.05	0.14	0.16
BIO2 = Mean Diurnal Range	0.02	0.28	0.08	**0.94**	0.15
BIO3 = Isothermality	0.32	0.65	0.03	0.01	**0.6**
BIO4 = Temperature Seasonality	0.32	0.66	0.05	0.34	0.54
BIO5 = Max Temperature of Warmest Month	0.93	0.11	0.15	0.29	0.06
BIO4 = Temperature Seasonality	0.77	0.25	0.13	0.5	0.27
BIO7 = Temperature Annual Range	0.19	0.23	0.04	0.89	0.31
BIO8 = Mean Temperature of Wettest Quarter	0.7	0.4	0.07	0.01	0.04
BIO9 = Mean Temperature of Driest Quarter	0.76	0.16	0.13	0.28	0.36
BIO10 = Mean Temperature of Warmest Quarter	**0.99**	0.04	0.11	0.03	0.07
BIO11 = Mean Temperature of Coldest Quarter	0.82	0.43	0.03	0.22	0.3
BIO12 = Annual Precipitation	0.22	0.77	0.57	0.08	0.1
BIO13 = Precipitation of Wettest Month	0.04	0.17	0.96	0.03	0.09
BIO14 = Precipitation of Driest Month	0.26	0.93	0.13	0.05	0.13
BIO15 = Precipitation Seasonality	0.22	0.85	0.31	0.03	0.26
BIO16 = Precipitation of Wettest Quarter	0.03	0.18	**0.98**	0	0.07
BIO17 = Precipitation of Driest Quarter	0.26	**0.93**	0.13	0.06	0.13
BIO18 = Precipitation of Warmest Quarter	0.34	0.03	0.69	0.17	0.38
BIO19 = Precipitation of Coldest Quarter	0.1	0.9	0.06	0.17	0.13
Proportion Variance	0.3	0.29	0.15	0.12	0.07
Cumulative Variance	0.3	0.59	0.74	0.86	0.93

modeling method; and Random Forest (RF), and Support Vector Machine (SVM) as presence-absence modeling methods. For RF and SVM, we followed Barbet-Massin et al. (2012) for pseudo-absences generation. We set 10 runs of 100 pseudo-absences using the "2º far" selection method for species with few records (except for *Celeus galeatus*, *Dysithamnus plumbeus*, and *Scytalopus iraiensis*, which 1,000 pseudo-absences selected by "SRE" method gave better results; Table S2). For species with more than 200 records, we set 1,000 pseudo-absences selected by the "SRE" method (Table S2). The "2º far" method is a random selection of points distant at least two degrees from any presence record. The "SRE" method creates an envelope around the presence records and selects randomly the points within the study area but outside the envelope (see Barbet-Massin et al. 2012). For MaxEnt, we selected 10,000 background points by the "random" method (Phillips & Dudík 2008) within the study area (except for *Amazona vinacea*, *Biatas nigropectus*, *C. galeatus*, *D. plumbeus*, *Hemitricchus furcatus*, *Iodopleura pipra*, *S. iraiensis*, and *Touit surdus*, which 1,000 background points selected by the "SRE" method gave better results; Table S2). We used the "biomod2" R package (Thuiller et al. 2020) to generate pseudo-absences. We performed 10-fold cross-validation, repeating the procedure 10 times for each algorithm. We applied the True Skill Statistic (TSS; Allouche et al. 2006), selecting only replicas with TSS ≥ 0.7 to get models with high accuracy (Table S3). We standardized, for each algorithm, the climatic suitability values of the predicted models, scaling them from 0 (low suitability) to 1 (high suitability). Then, we built continuous ensemble models using a weighted average by TSS (Araújo et al. 2011, Barbet-Massin et al. 2012), and binary ensemble models using the majority rule (Araújo & New 2007). We ran the modeling procedures using the "dismo", "kernlab", and "randomForest" R packages (Liaw & Wiener 2002, Karatzoglou et al. 2004, Hijmans et al. 2017). Using the binary maps, we calculated the extent of climatically suitable areas in current and future maps and the percentage of losses and gains of climatic suitability in the future scenario.

Percentage of forest cover

We downloaded a land-use raster map at 500 × 500 m resolution for year 2020 from MODIS Collection 6 (Friedl & Sulla-Menashe 2015), maintained by the NASA EOSDIS Land Processes Distributed Active Archive Center (LP DAAC). This land-use raster map covers the entire Atlantic Forest. We cropped the map to the study area and reclassified the land-use classes. Because most bird species analyzed are forest-dependent, we considered as forest the following classes: Evergreen Needleleaf and Broadleaf Forests, Deciduous Needleleaf and Broadleaf Forests, Mixed Forests, Closed and Open Shrublands, and Woody Savannas (Figure 1). Then, we upscaled the map to 1 km^2 resolution (same resolution of climatic suitability models) and obtained the percentage of forest cover by pixel. We transformed the percentage values of forest cover to range from 0 (low suitability) to 1 (high suitability).

Combining climate and forest maps

We combined the climatic models under current and future scenarios with the forest cover map. Considering that we aimed to identify priority areas for conservation and restoration efforts, we assumed no future changes in forest cover in the Atlantic Forest since previews models showed that with the implementation of the Forest Code and the Atlantic Forest Law in Brazil, the forest area will be maintained stabilized (Câmara et al. 2015). We used the EcoLand analysis, which consists in generating a scatter plot–with climate suitability values on X-axis and the landscape suitability values on Y-axis–and setting a threshold-dependence criterion for climate and landscape maps (Rezende et al. 2020, Santos et al. 2020; Sobral-Souza et al. 2021). First, we set the thresholds as low (≤ 0.5) and high (> 0.5) for both climatic and forest suitability for each species, focusing on identifying the appropriate management actions for each region (i.e., the areas for conservation and restoration efforts; see Rezende et al. 2020, Sobral-Souza et al. 2021). This resulted in four categories: (i) low climate and forest suitability (low priority for restoration and conservation); (ii) low climate and high forest suitability (climate change scenario); (iii) high climate and low forest suitability (high priority for restoration); and (iv) high climate and high forest suitability (high priority for conservation and creation of protected areas); see example for *Pipile jacutinga* in Figure 3.

To define the priority areas for conservation and restoration efforts considering all species, we combined all EcoLand maps for each scenario using the “mosaic” function from the “raster” R package (Hijmans 2023) to compute the max cell values in the overlapped layers. Using the max cell values guarantees that the pixel will be included in priority areas for conservation or restoration even if that pixel has good habitat quality for only one species (avoiding the decrease of cell values if the pixel is not good for the majority of the species). To understand if the predicted areas with high climate and high forest can expand or contract in the future EcoLand scenario (for each species separately and for the combination of all species), we calculated the area for this category and the percentage of losses and gains. We also performed Pearson’s correlation and linear regression to verify if the future expansion or contraction is related to the amount of highly suitable areas in the current scenario (i.e., if the loss or gain is related to wider or narrower current potential distribution). To understand which areas have highly priority for conservation, we only selected cells with both high climatic and forest suitability that overlap in both current and future maps.

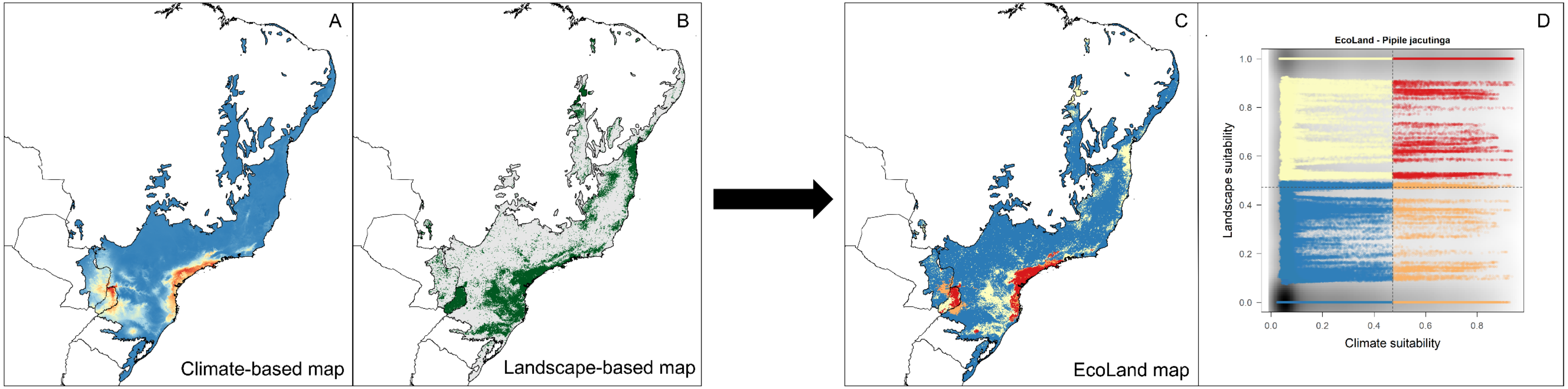

Figure 3. Example of the EcoLand analysis for *Pipile jacutinga*. We combined the climate-based map (A) and the landscape-based map, i.e., the forest cover map (B), generating the EcoLand map (C). We ran the EcoLand analysis creating a scatter plot (D), with climate suitability values on X-axis and the landscape suitability values on Y-axis, after setting a threshold-dependence criterion (values ≤0.5 mean low suitability and values >0.5 mean high suitability) for both climate and landscape maps. Categories in the scatter plot: low climate and forest suitability (in blue; lower priority for restoration and conservation); low climate and high forest suitability (in yellow; climate change scenario); high climate and low forest suitability (in orange; high priority for restoration); and high climate and high forest suitability (in red; high priority for conservation and creation of protected areas).

Finally, we downloaded, for the year 2023, the shapefiles of Protected Areas in Brazil from the National Register of Protected Areas – Ministry of the Environment (Ministério do Meio Ambiente 2023), and Argentina and Paraguay from the Protected Planet site (UNEP-WCMC & IUCN 2023). The Protected Areas include both private and public Conservation Units at municipal, state/provincial, and federal jurisdictions. These areas are classified in two main management categories: strict protection, with the main objective of nature conservation, allowing the indirect use of natural resources (such as tourism, scientific research, and environmental education) in some of them; and flexible or sustainable use, which enables the harmonization of nature conservation with the sustainable use of natural resources (see Brasil 2000, Paraguay 2001, Sistema Federal de Áreas Protegidas 2023). We calculated the extent of the Protected Areas covering the study area and the priority areas for conservation inside the extant Protected Areas for combined species' EcoLand maps for current and future scenarios, and the overlapped current and future areas. Using the same EcoLand analysis threshold for forest suitability (i.e., ≤0.5 for low and >0.5 for high suitability), we calculated the extent of good forest quality inside the Atlantic Forest for comparisons with our results. All analyses were performed using R version 4.3.0 (R Core Team 2023) and QGIS version 3.22 (QGIS Development Team 2022).

Results

Overall, modeling procedures provided ensemble models with good predictive power (TSS values varying from 0.80 to 0.99; Table S3). Our results showed that the potentially climatic suitable areas could increase from current to future scenarios only for three bird species (*Eleoscytalopus psychopompus*, *Formicivora erythronotos*, and *Pyrrhura leucotis*; Table S4). We also found that 45 bird species (~82%) may lose more than 50% of the climatic suitable areas by 2070, 36 species (~66%) may lose more than 70%, and 29 species (~53%) may lose more than 90%.

After combining the climatic and forest maps in the EcoLand map, we calculated the extent of suitable areas (i.e., high climatic and high forest suitability) for each bird species, which we thus considered priority areas for conservation. The expected arithmetic mean contraction in suitable areas by 2070 was 33.8%. We found that 26 bird species (~47%) may lose more than 50% of these areas, 21 species (~38%) may lose more than 70%, and 6 species (~11%) may lose more than 90% (Table S5). We have also found that nine species could gain suitable areas by 2070 (*Celeus galeatus*, *Celeus tinnunculus*, *Crax blumenbachii*, *E. psychopompus*, *F. erythronotos*, *Formicivora littoralis*, *Hemitriccus kaempferi*, *Platyrinchus leucoryphus* and *Pyriglena atra*). We found no correlation between the amount of suitable area in the current scenario and the percentage of loss or gain in suitable areas in the future scenario for each species ($t = 0.489$, $df = 53$, p-value = 0.627; Figure 4).

When we combined all species' EcoLand maps, our results showed that the birds could lose about 15% of the areas with high climatic and high forest suitability (Table S5). We verified that priority areas for conservation in the current scenario correspond to 231,229 km^2, and in the future scenario, the priority areas for conservation summed 195,576 km^2. Currently, the Protected Areas in the Atlantic Forest covers 8.3% (133,788 km^2) of the total area of the realm (1,619,514 km^2). Nevertheless, these areas encompass only a limited portion of the priority areas for conservation identified in this study. In the current scenario, only 23.4% of the priority areas for conservation (54,197 km^2) fall inside the extant Protected Areas, corresponding to 3.3% of the Atlantic Forest realm. In the future scenario, only 24.6% of the priority areas for conservation (48,076 km^2) remained inside the extant Protected Areas, corresponding to 2.9% of the realm.

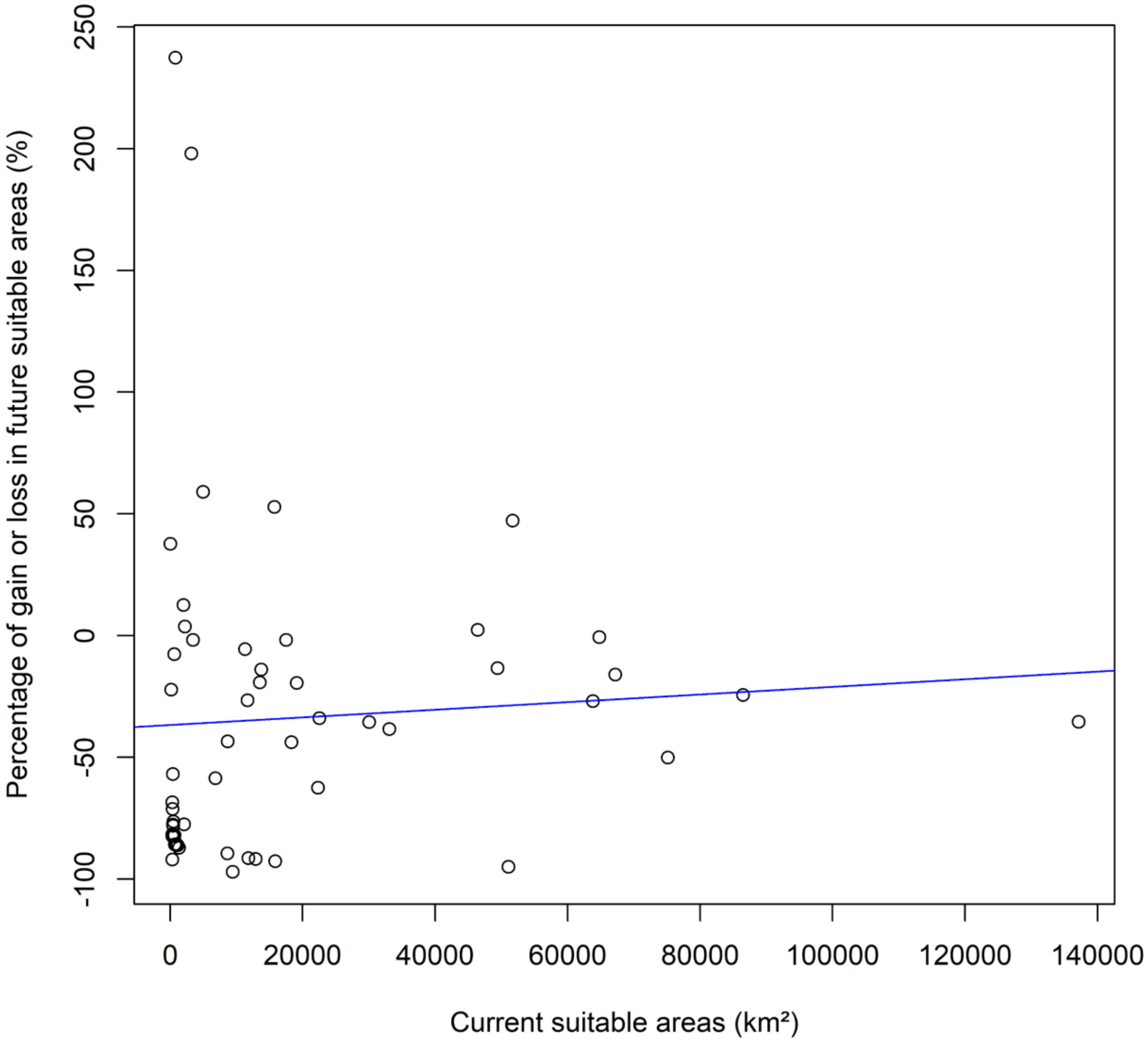

Figure 4. Pearson's correlation and linear regression between the amount of suitable area in the current scenario and the percentage of loss (negative values) or gain (positive values) in suitable areas in the future scenario for 55 endemic and threatened Atlantic Forest bird species (*t* = 0.489, df = 53, *p*-value = 0.627).

The EcoLand maps also revealed that the priority areas for restoration summed 444,227 km² in the current scenario and 235,843 km² in the future scenario. These results showed that 65.8% of the areas with high climatic suitability need to be reforested in the current scenario, and 54.7% in the future scenario. Currently, according our estimates (using the threshold of high forest suitability, i.e., >0.5), only 18.7% of the realm has a good forest quality (303,069 km²). We also found that priority areas for restoration might change from current to future scenarios since the climatic suitability can change between these periods, but priority areas for conservation remain consistent between the two scenarios (Figure 5).
By overlapping the current and future EcoLand maps for high climate and high forest suitability, we obtained the high-priority areas for the endemic and threatened Atlantic Forest bird species, corresponding to 192,385 km² in extent (Figure 6A). However, only 24.8% of this area (47,689 km²) was projected inside the extant Protected Areas (Figure 6B), corresponding to 2.9% of the Atlantic Forest realm.

Discussion

Our study evaluated for the first time the impacts of climate change associated with forest cover in the current and future potential distribution of 55 endemic and threatened Atlantic Forest bird species. We found that most bird species tend to lose suitable areas by year 2070 (~84%). We also found that, under both current and future scenarios, a small portion of priority areas for conservation falls inside of extant protected areas, while more than half of the climatically suitable area has low forest cover and needs urgent reforestation initiatives.

Currently, the IUCN Red List of Threatened Species considers only 5 out of the 55 species analyzed in this study threatened by climate change (IUCN 2022; Table S6). However, in our climate-based models, we have identified that most of the species may lose more than 50% of their climatically suitable areas in the future, which aligns with findings

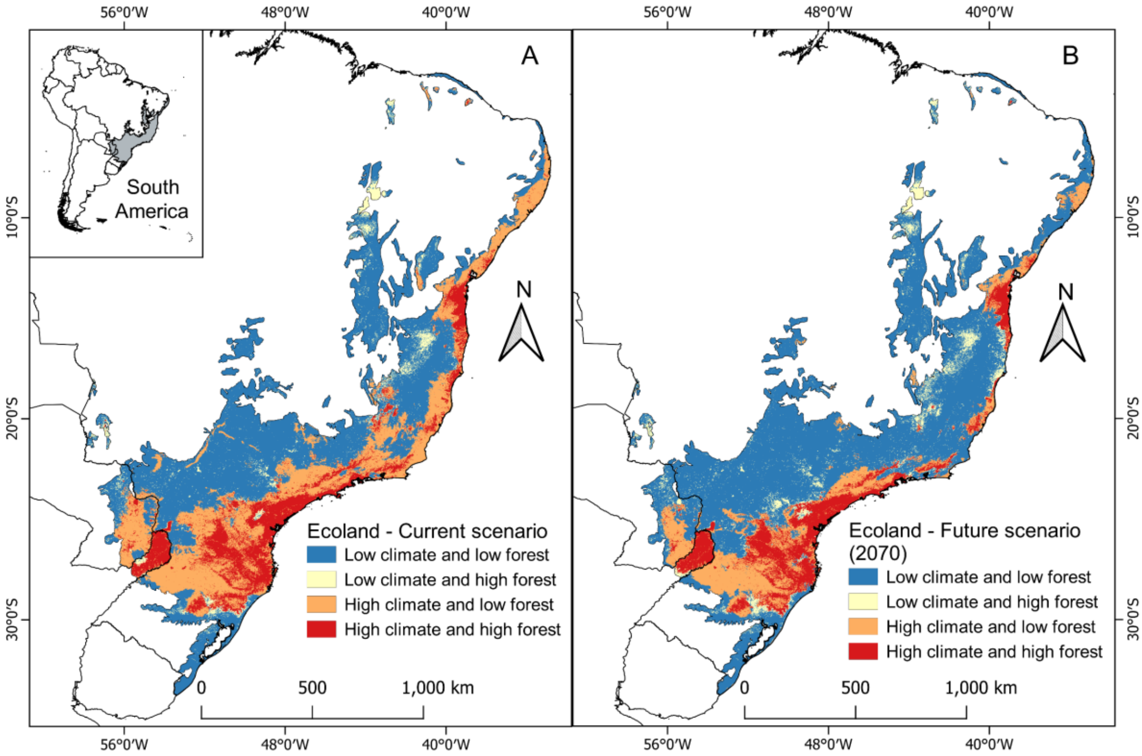

Figure 5. Environmental suitability for 55 endemic and threatened Atlantic Forest bird species within the Atlantic Forest realm, predicted using the combined climatic and forest maps through EcoLand analysis in current scenario (A), and future scenario (B). Each color represents the combination of the thresholds used in the analysis: low (≤0.5) and high (>0.5) climatic and forest suitability.

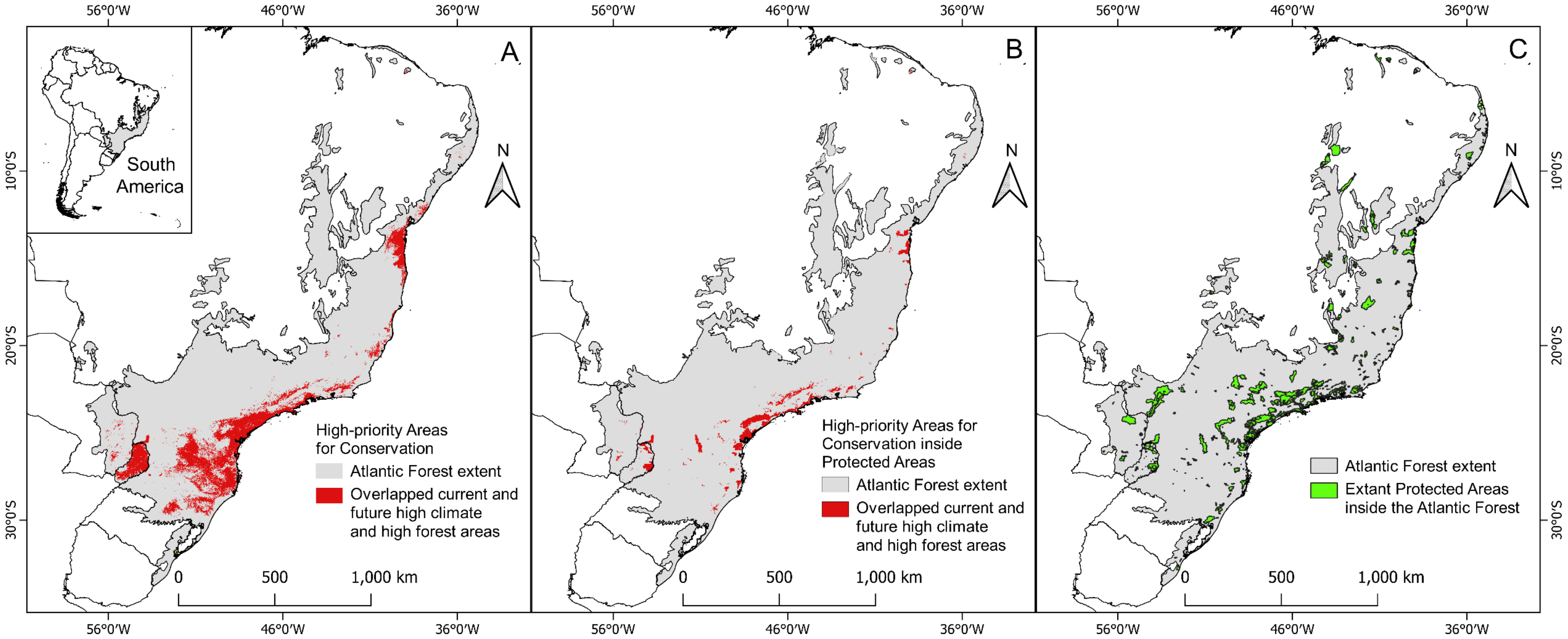

Figure 6. Overlapped current and future EcoLand maps for high climate and high forest suitability: (A) the high-priority areas for conservation for 55 endemic and threatened Atlantic Forest bird species; (B) the high-priority areas for conservation inside the extant Protected Areas; (C) Extant Protected Areas inside the Atlantic Forest.

from previous studies (Souza et al. 2011, Vale et al. 2018a), indicating that climate change is one of the main causes of bird extinction in the Atlantic Forest.

Despite these losses in projected climatically suitable area, our observations revealed an increase in suitable areas for nine species by 2070 when we combined climatic and forest maps to create the EcoLand map. This can be explained by the displacement of climatically suitable areas toward regions with greater amount of forest cover. But for one species, *E. psychopompus*, the areas with future environmental suitability expansion are located very far from its current distribution, which prevents the species from reaching these areas and coping with future environmental changes. Furthermore, our findings indicate that almost half of the species face the risk of losing more than 50% of their suitable habitats, underscoring the critical and pressing necessity for restoring the Atlantic Forest.

Although it is expected that bird species with broader distributions would experience minimal range loss under environmental changes, since larger ranges buffer the impact of these changes (Travis 2003, Jetz et al. 2007, Almeida et al. 2022), our results did not confirm this assumption. We observed the highest increases in future environmental suitability for two small-ranged species, *F. erythronotos* (237%) and *E. psychopompus* (198%). For other narrow-ranged species, the environmental suitability can increase, such as for *C. blumenbachii* (12.6%), *F. littoralis* (37.7%), and *P. atra* (3.7%), or slightly decrease, such as for *Rhopornis ardesiacus* (-7.6%) and *Formicivora paludicola* (-1.7%). Moreover, species currently exhibiting larger suitable areas are also susceptible to significant reductions in their future distributions, such as *Scytalopus iraiensis* (~95%) and *Sporophila frontalis* (~50%). These gains and losses of environmental suitability depend on how each species will respond to climate change since climatically suitable areas displacement can cover regions with greater or smaller amounts of forest cover. In addition, the differences in responses among species to environmental changes are expected considering the differences in ecological traits and niche breadth, which influence in future distribution expansion or contraction (Peterson et al. 2001, Ramalho et al. 2021).

Land-use changes can also negatively affect tropical regions, especially species with narrow distributions (Lambin et al. 2003, Jetz et al. 2007, Hansen et al. 2013). Additionally, deforestation poses a significant risk of triggering a mass bird extinction event within the Atlantic Forest (Brooks et al. 1999). The three countries where the Atlantic Forest occurs have strict laws to protect native forests and assure reforestation efforts: the “Zero Deforestation” Law in Paraguay (Paraguay 2004), the Native Forest Law in Argentina (Argentina 2007), and the Atlantic Forest Law in Brazil (Brasil 2006). Despite these laws, illegal deforestation still occurs in this realm, mainly in Paraguay and Argentina, due to lack of deforestation control, threatening biodiversity (Esquivel et al. 2019, Mohebalian et al. 2022). But Brazil also witnesses a concerning trend in deforestation, as revealed by the Annual Report on Deforestation in Brazil, from MapBiomas Alerta Project (MapBiomas 2023). The report exposed a staggering loss of 30,012 hectares within the Brazilian Atlantic Forest (corresponding to 1.5% of the total area deforested across the country). Disturbingly, when analyzing the trends over the past four years, the cumulative deforestation reached nearly 100,000 hectares (MapBiomas 2023). Furthermore, by 2050, it is expected cropland expansion by ~28% with the natural vegetation suppression and pasture replacement in the Brazilian Atlantic Forest, increasing remnants deforestation (Câmara et al. 2015, Soterroni et al. 2018). Although we did not consider land-use changes, the EcoLand map for all species showed a contraction in priority areas for conservation (i.e., areas with high climatic and high forest suitability) by 2070 (about 15%). These losses are located mainly in some portions of the Inland Atlantic Forest of Argentina, Brazil and Paraguay, and Bahia sub-regions in Brazil (see Ribeiro et al. 2009 for sub-regions). These concerns emphasize the crucial implementation of environmental public policies for reforestation and full deforestation control.

Regarding the priority areas for restoration, a substantial proportion of areas demonstrating high climatic suitability in both current and future scenarios require reforestation efforts to achieve conservation targets. These areas surround the priority areas for conservation in both current and future scenarios (Figure 5), and was intensively deforested to make a place for agriculture (see Diniz et al. 2022, MapBiomas, 2022). Considering that the realm is highly fragmented, with 83.4 % of the patches showing less than 50 ha (Ribeiro et al. 2009), the current landscape in the Atlantic Forest increases the risk of extinction of forest dependent and endemic birds since they are area sensitive and show higher abundances in larger patches than in small patches (Giraudo et al. 2008, Uezu & Metzger 2011, 2016). Moreover, the high degree of isolation of habitat patches hinders the dispersal of these sensitive species, rendering them reliant on the current occupied patches, intensifying their vulnerability to fragmentation, and consequently increasing the risk of extinction (Uezu & Metzger 2016). In addition, our results demonstrated that a good quality of forest cover (> 50% of forest amount) represents only 18.7% of the study area, indicating the urgent need for restoration efforts. Thus, promoting the reforestation of the areas with low forest suitability and high climatic suitability can expand environmentally suitable areas for the endemic and threatened Atlantic Forest birds, while also enhancing connectivity between patches, facilitating their dispersal (Borges & Loyola 2020, de la Sancha et al. 2021, Diniz et al. 2022).

We also observed a contraction in priority areas for restoration in the future, but even with this contraction, providing suitable areas by reforestation currently can mitigate local climate change impacts in the future by offering a buffer to climate variation (Ramalho et al. 2021). In Brazil, the Atlantic Forest Law implements reforestation enforcement (Brasil 2006), but it is still a challenge for this realm due to the lack of financial incentives to farmers associated with the high cost and adequate technology required for implementation and maintenance of reforestation (Pinto et al. 2014). Another impasse in the restoration of the Atlantic Forest is that the Brazilian Forest Code exempts the small farmers from the need to recover the legal reserve areas (Brasil 2012). Estimates revealed that if the small farmers restore the legal reserves, the forested area will expand in 4 million ha (Câmara et al. 2015). An increase in forest cover is expected in the next years in Brazilian Atlantic Forest with the Atlantic Forest Pact, an action that aims to restore 15 million ha by 2050 (Pacto pela Restauração da Mata Atlântica 2023). Thus, identifying the priority areas for restoration is essential to assist in decision-making, reduce costs, and improve the ecological effectiveness of reforestation, thereby supporting both existing and potential future initiatives.

The EcoLand map including all species shows that the priority areas for conservation are located in different sub-regions of the Atlantic Forest under both current and future scenarios: Bahia, Serra do Mar, and Araucaria Forests in Brazil (see Ribeiro et al. 2009 for sub-regions), and specific areas of the Inland Atlantic Forest in Argentina and Paraguay. Other studies have found similar patterns in Brazil for priority areas for the conservation of birds (Vale et al. 2018a), butterflies (Santos et al. 2020), and woody plants (Zwiener et al. 2017), but only considering the current scenario. In this study, we verified that a low percentage of these areas is protected along the Atlantic Forest in both current (~23%) and future (~25%) scenarios, as well as when overlapping the scenarios (~25%). These estimates represent only a minor fraction of the entire Atlantic Forest biome (approximately 3% for the current, future, and overlapping scenarios). This highlights that less than half of the extant Protected Areas, which constitute 8.3% of the Atlantic Forest biome, contain environmentally suitable areas for endemic and threatened Atlantic Forest bird species. Thus, our estimates show that the extant Protected Areas would not be enough to mitigate the risks of extinction of these birds, in agreement with other studies (Ferro et al. 2014, Giorgi et al. 2014, Silva et al. 2014, Pereira et al. 2016, Zwiener et al. 2017, Vale et al. 2018a, Williams et al. 2022). Moreover, suitable areas for these species outside the protected areas are also in danger, since forest cover interventions can occur both by the current environmental public policies–which allow some degree of deforestation–and changes in governance interests (Rezende et al. 2018, Ramos et al. 2023). Considering all these concerns together, we underscore the urgent call for the establishment of new protected areas.

Our approach considered climate change but a constant amount of forest cover over time, which does not show us where future land-use changes can occur based on current environmental public policies. Nevertheless, this approach allows us to assess which areas need to be conserved or restored to guarantee the persistence of endemic and threatened Atlantic Forest bird species in both current and future scenarios. Another limitation in this study was the exclusion of the analysis of ten bird species that showed < 30 records, which can interfere with the results and underestimate crucial sites for conservation or restoration (Zwiener et al. 2017), like Diamantina and São Francisco sub-regions, which are under intense degradation (Ribeiro et al. 2009, Silva & Barbosa 2017), and also harbor endemic and threatened species (Bornschein et al. 2007, Vasconcelos & D’Angelo Neto). However, our study was a first step to identifying priority areas for conservation and restoration efforts under both current and future scenarios, considering suitable areas for 55 endemic and threatened Atlantic Forest bird species. From our findings, we can delimit the main areas to carry out these efforts and draw more effective strategies on finer scales. By delimiting the priority sites for each strategy, even on a broader scale, local or regional studies can be carried out to assist decision-making, allowing cost reduction and increasing the effectiveness of conservation and restoration actions. Some examples are identifying, inside the priority areas for conservation, those areas that have a low economic interest, low population density (Vale et al. 2018a) or that could receive assistance to transform forest remnants (like some remnants with potential for legal reserve compensation) into private reserves in exchange for fiscal incentives for land-owners (Lowen et al. 1995, Crouzeilles et al. 2013, Pereira et al. 2016, Zwiener et al. 2017, Young & de Castro 2021). Because the Atlantic Forest is highly fragmented (Ribeiro et al. 2011), restoring natural vegetation is a fundamental strategy to complement conservation actions and protect biodiversity, especially when considering the habitat connectivity between Protected Areas (Travis 2003, Diniz et al. 2022, Vale et al. 2018a).

Conclusion

We found that suitable areas considering climate change and forest cover for endemic and threatened Atlantic Forest bird species can decrease by 2070. This loss varies across species, however, about half may lose more than 50% of the suitable areas in the future. We also found that over 70% of the priority conservation areas (i.e., areas with high climate and high forest suitability) in both current and future scenarios lie outside the extant Protected Areas across the Atlantic Forest in Brazil, Argentina, and Paraguay, which puts all these bird species at a higher risk of extinction due to expected future climate and land-use changes. All these concerns highlight the urgent need for establishing new protected areas in the Atlantic Forest to mitigate the impacts of environmental changes and enhance the survival prospects of these birds. Beyond the creation of new protected areas, reforestation of the target areas identified in this study can expand suitable areas, which allows the maintenance of viable populations for many species evaluated here, in addition to increasing connectivity between preserved patches and Protected Areas. It also positively affects other groups, including plants, vertebrates, and invertebrates, especially those groups that are forest-dependent and need larger areas for their movement (Rezende et al. 2018, Almeida et al. 2022, Diniz et al. 2022).

Thus, we emphasize the crucial implementation of environmental public policies for full deforestation control, creation of new Protected Areas and restoration of natural vegetation in specific areas of the Atlantic Forest to achieve threatened and endemic biodiversity conservation targets in the future.

Supporting information for chapter 2

Table S1. List of birds' occurrence records downloaded from Global Biodiversity Information Facility (GBIF).

Species	**GBIF source**
Acrobatornis fonsecai	https://doi.org/10.15468/dl.hhpuxa
Amazona pretrei	https://doi.org/10.15468/dl.wsfd6z
Amazona rhodocorytha	https://doi.org/10.15468/dl.4b963y
Amazona vinacea	https://doi.org/10.15468/dl.7dybyv
Anumara forbesi	https://doi.org/10.15468/dl.2qn8cj
Automolus lammi	https://doi.org/10.15468/dl.c4zba2
Biatas nigropectus	https://doi.org/10.15468/dl.w58ypf
Buteogallus lacernulatus	https://doi.org/10.15468/dl.qm8umw
Celeus galeatus	https://doi.org/10.15468/dl.psv7wc
Celeus tinnunculus	https://doi.org/10.15468/dl.pdntef
Cotinga maculata	https://doi.org/10.15468/dl.f3qaxz
Crax blumenbachii	https://doi.org/10.15468/dl.w58ypf
Dendrocincla taunayi	https://doi.org/10.15468/dl.2g8yzt
Dysithamnus plumbeus	https://doi.org/10.15468/dl.u2v65x
Eleoscytalopus psychopompus	https://doi.org/10.15468/dl.7wapku
Formicivora erythronotos	https://doi.org/10.15468/dl.rn6kf6
Formicivora littoralis	https://doi.org/10.15468/dl.5gabu3
Formicivora paludicola	https://doi.org/10.15468/dl.8z2gcg
Glaucis dohrnii	https://doi.org/10.15468/dl.af8xyv
Hemitriccus furcatus	https://doi.org/10.15468/dl.dxkhzk

Table S1. (continued)

Species	GBIF source
Hemitriccus kaempferi	https://doi.org/10.15468/dl.n39rj2
Hemitriccus mirandae	https://doi.org/10.15468/dl.gda237
Herpsilochmus pileatus	https://doi.org/10.15468/dl.9r2jvt
Iodopleura pipra	https://doi.org/10.15468/dl.nh6fkq
Leptodon forbesi	https://doi.org/10.15468/dl.rpfpfv
Lipaugus conditus	https://doi.org/10.15468/dl.z7ebpc
Myrmoderus ruficauda	https://doi.org/10.15468/dl.mxe5yb
Myrmotherula minor	https://doi.org/10.15468/dl.4vu6xe
Myrmotherula urosticta	https://doi.org/10.15468/dl.xmc598
Onychorhynchus swainsoni	https://doi.org/10.15468/dl.q8vnww
Phylloscartes beckeri	https://doi.org/10.15468/dl.vnekgx
Phylloscartes ceciliae	https://doi.org/10.15468/dl.da9v8w
Pionus reichenowi	-
Pipile jacutinga	https://doi.org/10.15468/dl.w58ypf
Platyrinchus leucoryphus	https://doi.org/10.15468/dl.q59wgy
Pyriglena atra	https://doi.org/10.15468/dl.mdw8qf
Pyriglena pernambucensis	https://doi.org/10.15468/dl.cwkkcm
Pyrrhura cruentata	https://doi.org/10.15468/dl.qjtmb5
Pyrrhura griseipectus	https://doi.org/10.15468/dl.bf3bfc
Pyrrhura leucotis	https://doi.org/10.15468/dl.2hbah4
Rhopornis ardesiacus	https://doi.org/10.15468/dl.ad48r7
Sclerurus cearensis	https://doi.org/10.15468/dl.f7bxv8
Scytalopus diamantinensis	https://doi.org/10.15468/dl.932a4a

Table S1. (continued)

Species	**GBIF source**
Scytalopus iraiensis	https://doi.org/10.15468/dl.sqn79w
Sporophila falcirostris	https://doi.org/10.15468/dl.u9fxhr
Sporophila frontalis	https://doi.org/10.15468/dl.gt23eb
Stilpnia peruviana	https://doi.org/10.15468/dl.qsbvtq
Synallaxis infuscata	https://doi.org/10.15468/dl.mmmvek
Tangara fastuosa	https://doi.org/10.15468/dl.musxc8
Terenura sicki	https://doi.org/10.15468/dl.6wffmk
Thalurania watertonii	https://doi.org/10.15468/dl.jxd2vr
Thripophaga macroura	https://doi.org/10.15468/dl.kjaefm
Touit surdus	https://doi.org/10.15468/dl.6e2c43
Xipholena atropurpurea	https://doi.org/10.15468/dl.6d9mdq
Xiphorhynchus atlanticus	https://doi.org/10.15468/dl.vet7ub

Table S2. Status, number of occurrence records, and number and method of pseudoabsence and background points selection for 55 endemic and threatened Atlantic Forest bird species. VU: Vulnerable; EN: Endangered; CR: Critically Endangered.

Species	**Status**	**Number of records**	**Pseudoabsences RF and SVM**		**Background MaxEnt**	
			Number	**Method of selection**	**Number**	**Method of selection**
Acrobatornis fonsecai	VU	50	10 runs of 100	2º far	10,000	random
Amazona pretrei	VU	176	10 runs of 100	2º far	10,000	random
Amazona rhodocorytha	VU	264	1,000	sre	10,000	random
Amazona vinacea	EN	708	1,000	sre	1,000	sre
Anumara forbesi	VU	58	10 runs of 100	2º far	10,000	random
Automolus lammi	EN	41	10 runs of 100	2º far	10,000	random
Biatas nigropectus	VU	277	1,000	sre	1,000	sre
Buteogallus lacernulatus	VU	454	1,000	sre	10,000	random
Celeus galeatus	VU	168	1,000	sre	1,000	sre
Celeus tinnunculus	VU	32	10 runs of 100	2º far	10,000	random
Cotinga maculata	CR	35	10 runs of 100	2º far	10,000	random
Crax blumenbachii	EN	50	10 runs of 100	2º far	10,000	random

Table S2. (continued)

Species	**Status**	**Number of records**	**Pseudoabsences RF and SVM**		**Background MaxEnt**	
			Number	**Method of selection**	**Number**	**Method of selection**
Dendrocincla taunayi	EN	53	10 runs of 100	2º far	10,000	random
Dysithamnus plumbeus	VU	59	1,000	sre	1,000	sre
Eleoscytalopus psychopompus	EN	32	10 runs of 100	2º far	10,000	random
Formicivora erythronotos	EN	49	10 runs of 100	2º far	10,000	random
Formicivora littoralis	EN	80	10 runs of 100	2º far	10,000	random
Formicivora paludicola	CR	32	10 runs of 100	2º far	10,000	random
Glaucis dohrnii	VU	93	10 runs of 100	2º far	10,000	random
Hemitriccus furcatus	VU	220	1,000	sre	1,000	sre
Hemitriccus kaempferi	VU	90	10 runs of 100	2º far	10,000	random
Hemitriccus mirandae	VU	80	10 runs of 100	2º far	10,000	random
Herpsilochmus pileatus	VU	169	10 runs of 100	2º far	10,000	random
Iodopleura pipra	EN	204	1,000	sre	1,000	sre
Leptodon forbesi	EN	91	10 runs of 100	2º far	10,000	random

Table S2. (continued)

Species	Status	Number of records	Pseudoabsences RF and SVM		Background MaxEnt	
			Number	Method of selection	Number	Method of selection
Lipaugus conditus	VU	30	10 runs of 100	2º far	10,000	random
Myrmoderus ruficauda	EN	91	10 runs of 100	2º far	10,000	random
Myrmotherula minor	VU	185	10 runs of 100	2º far	10,000	random
Myrmotherula urosticta	VU	136	10 runs of 100	2º far	10,000	random
Onychorhynchus swainsoni	VU	37	10 runs of 100	2º far	10,000	random
Phylloscartes beckeri	EN	44	10 runs of 100	2º far	10,000	random
Phylloscartes ceciliae	CR	54	10 runs of 100	2º far	10,000	random
Pionus reichenowi	VU	100	10 runs of 100	2º far	10,000	random
Pipile jacutinga	EN	335	1,000	sre	10,000	random
Platyrinchus leucoryphus	VU	172	10 runs of 100	2º far	10,000	random
Pyriglena atra	EN	57	10 runs of 100	2º far	10,000	random
Pyriglena pernambucensis	VU	34	10 runs of 100	2º far	10,000	random
Pyrrhura cruentata	VU	160	10 runs of 100	2º far	10,000	random
Pyrrhura griseipectus	EN	54	10 runs of 100	2º far	10,000	random

Table S2. (continued)

Species	Status	Number of records	Pseudoabsences RF and SVM		Background MaxEnt	
			Number	Method of selection	Number	Method of selection
Pyrrhura leucotis	VU	146	10 runs of 100	2º far	10,000	random
Rhopornis ardesiacus	EN	72	10 runs of 100	2º far	10,000	random
Sclerurus cearensis	VU	59	10 runs of 100	2º far	10,000	random
Scytalopus diamantinensis	EN	51	10 runs of 100	2º far	10,000	random
Scytalopus iraiensis	EN	75	1,000	sre	1,000	sre
Sporophila falcirostris	VU	344	1,000	sre	10,000	random
Sporophila frontalis	VU	544	1,000	sre	10,000	random
Stilpnia peruviana	VU	389	1,000	sre	10,000	random
Synallaxis infuscata	EN	70	10 runs of 100	2º far	10,000	random
Tangara fastuosa	VU	158	10 runs of 100	2º far	10,000	random
Terenura sicki	CR	53	10 runs of 100	2º far	10,000	random
Thalurania watertonii	EN	96	10 runs of 100	2º far	10,000	random
Thripophaga macroura	VU	86	10 runs of 100	2º far	10,000	random
Touit surdus	VU	224	1,000	sre	1,000	sre

Table S2. (continued)

Species	Status	Number of records	Pseudoabsences RF and SVM		Background MaxEnt	
			Number	**Method of selection**	**Number**	**Method of selection**
Xipholena atropurpurea	VU	126	10 runs of 100	2º far	10,000	random
Xiphorhynchus atlanticus	VU	163	10 runs of 100	2º far	10,000	random

Table S3. True Skill Statistic (TSS) values resulted from the individual models and the ensemble of the ecological niche modeling of 55 endemic and threatened Atlantic Forest birds for each scenario (current and future). TSS value of the ensemble was calculated by the average of the TSS of all replicas ≥0.7.

Species	**TSS values (mean)**				
	Bioclim	**RF**	**SVM**	**MaxEnt**	**Ensemble**
Acrobatornis fonsecai	0.80	0.91	0.80	0.96	0.87
Amazona pretrei	0.84	0.89	0.88	0.91	0.88
Amazona rhodocorytha	0.72	0.92	0.91	0.78	0.86
Amazona vinacea	0.79	0.90	0.85	0.81	0.84
Anumara forbesi	0.73	0.92	0.89	0.92	0.89
Automolus lammi	0.94	0.92	0.87	0.98	0.92
Biatas nigropectus	0.82	0.93	0.89	0.87	0.88
Buteogallus lacernulatus	0.71	0.93	0.91	0.82	0.86
Celeus galeatus	0.84	0.85	0.82	0.83	0.83
Celeus tinnunculus	0.88	0.94	0.86	0.94	0.91
Cotinga maculata	0.83	0.86	0.82	0.84	0.84
Crax blumenbachii	0.81	0.90	0.88	0.89	0.87
Dendrocincla taunayi	0.91	0.98	0.97	0.99	0.96
Dysithamnus plumbeus	0.77	0.89	0.86	0.79	0.84
Eleoscytalopus psychopompus	0.80	0.97	0.97	0.93	0.92
Formicivora erythronotos	0.85	0.96	0.94	0.98	0.94
Formicivora littoralis	0.88	0.99	0.99	0.99	0.97
Formicivora paludicola	0.86	0.99	0.99	0.99	0.96

Table S3. (continued)

Species	**TSS values (mean)**				
	Bioclim	**RF**	**SVM**	**MaxEnt**	**Ensemble**
Glaucis dohrnii	0.83	0.91	0.90	0.91	0.89
Hemitriccus furcatus	0.81	0.96	0.95	0.92	0.91
Hemitriccus kaempferi	0.89	0.99	0.99	0.99	0.96
Hemitriccus mirandae	0.88	0.93	0.96	0.99	0.94
Herpsilochmus pileatus	0.93	0.95	0.96	0.97	0.95
Iodopleura pipra	0.76	0.93	0.91	0.90	0.88
Leptodon forbesi	0.84	0.95	0.90	0.98	0.92
Lipaugus conditus	0.84	0.98	0.97	0.93	0.94
Myrmoderus ruficauda	0.78	0.86	0.85	0.87	0.84
Myrmotherula minor	0.74	0.91	0.86	0.85	0.85
Myrmotherula urosticta	0.89	0.94	0.90	0.95	0.92
Onychorhynchus swainsoni	0.77	0.85	0.80	0.84	0.82
Phylloscartes beckeri	0.86	0.87	0.89	0.95	0.90
Phylloscartes ceciliae	0.88	0.94	0.90	0.99	0.93
Pionus reichenowi	0.84	0.91	0.90	0.95	0.90
Pipile jacutinga	0.73	0.91	0.87	0.80	0.85
Platyrinchus leucoryphus	0.72	0.85	0.75	0.79	0.80
Pyriglena atra	0.85	0.95	0.98	0.98	0.94
Pyriglena pernambucensis	0.80	0.91	0.82	0.99	0.90
Pyrrhura cruentata	0.71	0.89	0.83	0.84	0.84
Pyrrhura griseipectus	0.87	0.98	0.97	0.97	0.95

Table S3. (continued)

Species	**TSS values (mean)**				
	Bioclim	**RF**	**SVM**	**MaxEnt**	**Ensemble**
Pyrrhura leucotis	-	0.88	0.81	0.78	0.82
Rhopornis ardesiacus	0.81	0.93	0.90	0.92	0.89
Sclerurus cearensis	0.80	0.98	0.98	0.97	0.94
Scytalopus diamantinensis	0.91	0.99	0.99	0.99	0.98
Scytalopus iraiensis	0.79	0.88	0.84	0.83	0.84
Sporophila falcirostris	-	0.87	0.81	0.74	0.83
Sporophila frontalis	0.79	0.91	0.90	0.86	0.86
Stilpnia peruviana	0.78	0.96	0.94	0.88	0.89
Synallaxis infuscata	0.85	0.94	0.89	0.99	0.92
Tangara fastuosa	0.89	0.91	0.92	0.97	0.93
Terenura sicki	0.88	0.96	0.97	0.99	0.95
Thalurania watertonii	0.89	0.94	0.94	0.98	0.94
Thripophaga macroura	0.84	0.91	0.82	0.92	0.87
Touit surdus	0.88	0.95	0.92	0.94	0.92
Xipholena atropurpurea	0.82	0.91	0.82	0.92	0.87
Xiphorhynchus atlanticus	0.90	0.94	0.94	0.98	0.94

Table S4. Climatically suitable areas in current (1970-2000) and future (2070) scenarios, and changes in future climatic suitability compared to the current scenario for 55 endemic and threatened Atlantic Forest bird species. Negative values indicate loss of climatic suitability, and positive values indicate gain of climatic suitability. Bolded results indicate loss of at least 90% of climatic suitability.

Species	**Current (in km²)**	**Future (in km²)**	**Changes (%)**
Acrobatornis fonsecai	36,662.2	4,904.5	-86.6
Amazona pretrei	70,036.8	31,539.6	-55.0
Amazona rhodocorytha	**157,264.1**	**11,843.3**	**-92.5**
Amazona vinacea	533,557.3	138,766.9	-74.0
Anumara forbesi	**20,614.5**	**138.6**	**-99.3**
Automolus lammi	**31,110.8**	**2,865.8**	**-90.8**
Biatas nigropectus	294,838.4	112,524.4	-61.8
Buteogallus lacernulatus	158,961.9	102,513.7	-35.5
Celeus galeatus	298,537	172,447.4	-42.2
Celeus tinnunculus	**27,516.7**	**767.8**	**-97.2**
Cotinga maculata	32,339.7	4,482.9	-86.1
Crax blumenbachii	**34,981.4**	**1,104.5**	**-96.8**
Dendrocincla taunayi	**11,744.6**	**26.4**	**-99.8**
Dysithamnus plumbeus	**116,723.5**	**8,160.7**	**-93.0**
Eleoscytalopus psychopompus	6,506.0	111,966.8	1,621.0
Formicivora erythronotos	5,269.0	8,471.0	60.8
Formicivora littoralis	2,079.6	700.4	-66.3
Formicivora paludicola	7,850.7	845.8	-89.2

Table S4. (continued)

Species	Current (in km²)	Future (in km²)	Changes (%)
Glaucis dorhnii	**38,194.8**	**67.7**	**-99.8**
Hemitriccus furcatus	11,4587.2	61,300.9	-46.5
Hemitriccus kaempferi	**8,205.8**	**379.0**	**-95.4**
Hemitriccus mirandae	**5,709.9**	**185.0**	**-96.8**
Herpsilochmus pileatus	**33,470.0**	**781.1**	**-97.7**
Iodopleura pipra	141,683.6	45,905.5	-67.6
Leptodon forbesi	**27,183.4**	**206.3**	**-99.2**
Lipaugus conditus	**6,609.0**	**12.7**	**-99.8**
Myrmoderus ruficauda	**50,449.5**	**263.2**	**-99.5**
Myrmotherula minor	76,086.5	32,260.0	-57.6
Myrmotherula urosticta	**36,167.4**	**205.8**	**-99.4**
Onychorhynchus swainsoni	46,391.5	15,832.8	-65.9
Phylloscartes beckeri	**37,817.3**	**951.1**	**-97.5**
Phylloscartes ceciliae	**11,713.6**	**89.5**	**-99.2**
Pionus reichenowi	56,250.5	21,660.4	-61.5
Pipile jacutinga	151,632.0	86,226.0	-43.1
Platyrinchus leucoryphus	159,952.3	109,946.9	-31.3
Pyriglena atra	13,320.6	6,845.2	-48.6
Pyriglena pernambucensis	**14,001.0**	**93.8**	**-99.3**
Pyrrhura cruentata	**111,558.0**	**10,410.5**	**-90.7**
Pyrrhura griseipectus	826.9	101.4	-87.7
Pyrrhura leucotis	150,564.2	165,731.6	10.1

Table S4. (continued)

Species	Current (in km²)	Future (in km²)	Changes (%)
Rhopornis ardesiacus	18,217.2	4,256.6	-76.6
Sclerurus cearensis	**2,651.5**	**43.5**	**-98.4**
Scytalopus diamantinensis	**4,167.7**	**286.4**	**-93.1**
Sclerurus cearensis	**2,651.5**	**43.5**	**-98.4**
Scytalopus diamantinensis	**4,167.7**	**286.4**	**-93.1**
Scytalopus iraiensis	**277,072.2**	**7,836.8**	**-97.2**
Sporophila falcirostris	370,416.8	138,086.9	-62.7
Sporophila frontalis	248,594.7	83,779.1	-66.3
Stilpnia peruviana	136,527.5	84,727.1	-37.9
Synallaxis infuscata	**16,240.4**	**93.8**	**-99.4**
Tangara fastuosa	**22,895.0**	**217.4**	**-99.1**
Terenura sicki	**9,336.1**	**23.0**	**-99.8**
Thalurania watertonii	**21,432.9**	**99.7**	**-99.5**
Thripophaga macroura	**46,771.8**	**2,821.9**	**-94.0**
Touit surdus	176,598.2	33,823.6	-80.8
Xipholena atropurpurea	**70,775.7**	**1,643.1**	**-97.7**
Xiphorhynchus atlanticus	**25,361.7**	**410.1**	**-98.4**

Table S5. Suitable areas (high climatic and high forest suitability) resulted from the EcoLand map in current (1970-2000) and future (2070) scenarios, and changes in future suitability compared to the current scenario for 55 endemic and threatened Atlantic Forest bird species. Negative values indicate loss of climatic suitability, and positive values indicate gain of climatic suitability. Bolded results indicate loss of at least 90% of climatic suitability.

Species	Current (in km²)	Future (in km²)	Change (%)
Acrobatornis fonsecai	**9,436.9**	**278.8**	**-97.0**
Amazona pretrei	13,740.9	11,835.2	-13.9
Amazona rhodocorytha	22,307.4	8,368.9	-62.5
Amazona vinacea	137,132.1	88,577.2	-35.4
Anumara forbesi	379.8	164	-56.8
Automolus lammi	440.2	96.2	-78.1
Biatas nigropectus	63,793.3	46,631.3	-26.9
Buteogallus lacernulatus	67,181.5	56,442.5	-16.0
Celeus galeatus	51,697.8	76,098.2	47.2
Celeus tinnunculus	15,709.5	24,021.7	52.9
Cotinga maculata	6,797.6	2,813.0	-58.6
Crax blumenbachii	1,981.4	2,232.0	12.6
Dendrocincla taunayi	372.4	83.4	-77.6
Dysithamnus plumbeus	8,663.7	4,904.7	-43.4
Eleoscytalopus psychopompus	3,176.8	9,468.1	198.0
Formicivora erythronotos	787.3	2,656.3	237.4
Formicivora littoralis	25.2	34.7	37.7

Table S5. (continued)

Species	Current (in km²)	Future (in km²)	Change (%)
Formicivora paludicola	3,407.8	3,349.3	-1.7
Glaucis dorhnii	**15,837.7**	**1,156.3**	**-92.7**
Hemitriccus furcatus	17,499.7	17,198.2	-1.7
Hemitriccus kaempferi	4,967.2	7,901.2	59.1
Hemitriccus mirandae	360.7	104	-71.2
Herpsilochmus pileatus	11,272.2	10,654.7	-5.5
Iodopleura pipra	22,510.6	14,873.4	-33.9
Leptodon forbesi	925.8	132.1	-85.7
Lipaugus conditus	1,298.4	166.9	-87.1
Myrmoderus ruficauda	641.6	115.8	-82.0
Myrmotherula minor	19,110.1	15,396	-19.4
Myrmotherula urosticta	**11,782.8**	**1,016.3**	**-91.4**
Onychorhynchus swainsoni	13,532.6	10,934.3	-19.2
Phylloscartes beckeri	2,060.8	463.5	-77.5
Phylloscartes ceciliae	318.4	56.3	-82.3
Pionus reichenowi	11,675.9	8,567.5	-26.6
Pipile jacutinga	64,778.2	64,406.5	-0.6
Platyrinchus leucoryphus	46,396.6	47,519.5	2.4
Pyriglena atra	2,189.2	2,269.7	3.7
Pyriglena pernambucensis	550.8	99.7	-81.9
Pyrrhura cruentata	18,292.1	10,277.6	-43.8
Pyrrhura griseipectus	153.4	119.3	-22.2

Table S5. (continued)

Species	**Current (in km²)**	**Future (in km²)**	**Change (%)**
Pyrrhura leucotis	30,016.5	19,352.4	-35.5
Rhopornis ardesiacus	587.0	542.6	-7.6
Sclerurus cearensis	312.9	98.9	-68.4
Scytalopus diamantinensis	**323.3**	**25.8**	**-92.0**
Scytalopus iraiensis	**51,068.2**	**2,605.6**	**-94.9**
Sporophila falcirostris	86,474.5	65,379.7	-24.4
Sporophila frontalis	75,117.3	37,468.1	-50.1
Stilpnia peruviana	49,403.2	42,838.4	-13.3
Synallaxis infuscata	439.2	78.4	-82.1
Tangara fastuosa	675.9	96.3	-85.8
Terenura sicki	291.3	53.7	-81.6
Thalurania watertonii	464.4	110.0	-76.3
Thripophaga macroura	8,607.2	344.4	-89.4
Touit surdus	33,071.1	20,357.9	-38.4
Xipholena atropurpurea	**12,881.8**	**1,071.4**	**-91.7**
Xiphorhynchus atlanticus	1,091.0	151.7	-86.1
All species	231,229.7	195,576.4	-15.4

Table S6. Main threats to 55 endemic and threatened Atlantic Forest bird species according to the IUCN Red List of Threatened Species (IUCN, 2022). VU: Vulnerable; EN: Endangered; CR: Critically Endangered.

Species	Status	Habitat	Main threats
Acrobatornis fonsecai	VU	Forest	Habitat loss
Amazona pretrei	VU	Forest	Habitat loss, illegal trade, hunting, virus disease
Amazona rhodocorytha	VU	Forest	Habitat loss, illegal trade, hunting, persecution/control
Amazona vinacea	EN	Forest	Habitat loss, illegal trade, hunting, persecution/control
Anumara forbesi	VU	Forest	Habitat loss, hunting, invasive species, diseases
Automolus lammi	EN	Forest	Habitat loss, pollution
Biatas nigropectus	VU	Forest	Habitat loss
Buteogallus lacernulatus	VU	Forest	Habitat loss, persecution/control
Celeus galeatus	VU	Forest	Habitat loss
Celeus tinnunculus	VU	Forest	Habitat loss
Cotinga maculata	CR	Forest	Habitat loss, illegal trade
Crax blumenbachii	EN	Forest	Habitat loss, hunting
Dendrocincla taunayi	EN	Forest	-
Dysithamnus plumbeus	VU	Forest	Habitat loss
Eleoscytalopus psychopompus	EN	Forest	Habitat loss

Table S6. (continued)

Species	Status	Habitat	Main threats
Formicivora erythronotos	EN	Shrubland	Habitat loss
Formicivora littoralis	EN	Forest	-
Formicivora paludicola	CR	Wetland	Habitat loss, invasive species, diseases
Glaucis dohrnii	VU	Forest	Habitat loss
Hemitriccus furcatus	VU	Forest	Habitat loss
Hemitriccus kaempferi	VU	Forest	Habitat loss, climate change
Hemitriccus mirandae	VU	Forest	Habitat loss, climate change
Herpsilochmus pileatus	VU	Shrubland	Habitat loss
Iodopleura pipra	EN	Forest	Habitat loss
Leptodon forbesi	EN	Forest	Habitat loss, illegal trade
Lipaugus conditus	VU	Forest	Habitat loss, climate change, disturbance (tourism)
Myrmoderus ruficauda	EN	Forest	Habitat loss
Myrmotherula minor	VU	Forest	Habitat loss
Myrmotherula urosticta	VU	Forest	Habitat loss
Onychorhynchus swainsoni	VU	Forest	Habitat loss
Phylloscartes beckeri	EN	Forest	Habitat loss, climate change
Phylloscartes ceciliae	CR	Forest	Habitat loss, hunting
Pionus reichenowi	VU	Forest	Habitat loss, hunting, illegal trade
Pipile jacutinga	EN	Forest	Habitat loss, hunting, illegal trade

Table S6. (continued)

Species	**Status**	**Habitat**	**Main threats**
Platyrinchus leucoryphus	VU	Forest	Habitat loss
Pyriglena atra	EN	Forest	Habitat loss
Pyriglena pernambucensis	VU	Forest	-
Pyrrhura cruentata	VU	Forest	Habitat loss, illegal trade
Pyrrhura griseipectus	EN	Forest	Habitat loss, illegal trade, invasive species, diseases
Pyrrhura leucotis	VU	Forest	Habitat loss, illegal trade
Rhopornis ardesiacus	EN	Woodland	Habitat loss, climate change
Sclerurus cearensis	VU	Forest	Habitat loss
Scytalopus diamantinensis	EN	Forest	Habitat loss
Scytalopus iraiensis	EN	Grassland	Habitat loss, invasive species, diseases
Sporophila falcirostris	VU	Woodland	Habitat loss, illegal trade
Sporophila frontalis	VU	Forest	Habitat loss, illegal trade
Stilpnia peruviana	VU	Woodland	Habitat loss, illegal trade
Synallaxis infuscata	EN	Forest	Habitat loss
Tangara fastuosa	VU	Forest	Habitat loss, illegal trade
Terenura sicki	CR	Forest	Habitat loss
Thalurania watertonii	EN	Forest	Habitat loss, illegal trade
Thripophaga macroura	VU	Forest	Habitat loss
Touit surdus	VU	Forest	Habitat loss
Xipholena atropurpurea	VU	Forest	Habitat loss, hunting, illegal trade

Table S6. (continued)

Species	**Status**	**Habitat**	**Main threats**
Xiphorhynchus atlanticus	VU	Forest	Habitat loss

Chapter 3: Defaunation of endemic and threatened birds leads to the loss of essential ecosystem services in the Atlantic Forest

Abstract

Birds play a crucial role in providing many ecosystem services (ES) that are essential for maintaining ecosystem functioning. However, human pressures such as land-use change, climate change, and overexploitation have caused declines in bird populations worldwide, particularly in tropical areas, which holds several endemic and vulnerable to extinction species. Bird defaunation thus can severely affect network structures, trophic cascade, and the provision of essential ES in tropical forests. Endemic and threatened bird species are facing an imminent risk of extinction due to many threats caused by human activities, which can impact the provision of ES. Using geographic ranges of 65 endemic and threatened Atlantic Forest bird species and ecological network metrics, we investigated the impact of bird defaunation on the ES within the Atlantic Forest using different defaunation scenarios (low, medium and high defaunation). Our findings showed that ecological network structure remained relatively stable in all defaunation scenarios, but defaunation led to a substantial depletion in ES (especially with medium to high defaunation rates), raising concerns about potential alterations in ecosystem functioning and ES provision. Conservation efforts, such as expanding protected areas, implementing forest restoration and deforestation control measures, and eradicating overhunting, are crucial to ensure endemic and threatened birds' survival and safeguard the ES they provide in the Atlantic Forest realm.

Introduction

Human activities have exerted an increasing impact on global biodiversity, leading to a significant imbalance within ecosystems and consequent biodiversity loss (Johnson et al. 2017, Williams et al. 2020). Human overpopulation and increased human activities have resulted in the most significant proportional decline of wilderness within tropical forests (Morris 2010, Williams et al. 2020). Along tropical regions, biodiversity faces significant threats primarily stemming from land-use changes (e.g., deforestation for agriculture and urbanization), overexploitation, invasive species and diseases, energy and mining, climate change, pollution, and transportation (Morris 2010, Antonelli 2022). Among these, land-use changes and direct overexploitation emerge as the dominant threats to tropical terrestrial ecosystems (Jaureguiberry et al. 2022). Over the past few decades, all these extinction drivers have severely impacted the Neotropics, a region renowned for its remarkable biodiversity, with estimates suggesting that approximately 33% of all species can be currently under threat (Antonelli 2022).

Terrestrial animals in tropical regions, particularly vertebrates, are currently experiencing high population declines in response to these extinction drivers (Dirzo et al. 2014, Ceballos et al. 2017). For example, large-bodied species seem to be more vulnerable considering their low fecundity and growth rate, large home ranges, and preference by hunters and poachers (Galetti & Dirzo 2013). Forest specialists are more affected by deforestation (particularly of old-growth forests) and habitat fragmentation than generalists (e.g., Carrara et al. 2015, Pinto Leite et al. 2018, Palmeirim et al. 2020). Small-ranged, and consequently, endemic species are more likely to have greater range contractions in response to climate and land-use changes compared with wide-ranging species (Jetz et al. 2007, Manes et al. 2021). Lastly, rare species have a high probability to be extinct due to a combination of factors:

narrow geographical range, few populations remaining, small population size, overhunting, low dispersal ability, and specialized niche (Işik 2011).

Extinction drivers have a far-reaching impact, not only resulting in the loss of species but also triggering a cascade of events that significantly affect ecosystem dynamics. Studies have demonstrated that defaunation (i.e., the loss of animal species and populations, and local declines in abundance) can negatively affect networks structure, trophic cascade, and essential ecosystem services, such as pollination, pest and disease control, nutrient cycling, and food resources (Morris 2010, Estes et al. 2011, Kurten 2013, Dirzo et al. 2014, Young et al. 2016, Bogoni et al. 2020). For example, the defaunation of large-bodied frugivores can decrease carbon storage in tropical forests, as well as alters forest dynamics by eliminating key bird-seed dispersal interactions, which can decrease plant community diversity (Harrison et al. 2013, Kurten 2013, Bello et al. 2015, Emer et al. 2019). Pollination, another crucial animal-plant interaction, is also threatened by anthropogenic stressors, leading to a global pollinator crisis that impacts not only natural ecological dynamics but also crucial aspects of human life, including crop production and other essential goods (Torezan-Silingardi et al. 2021). Furthermore, defaunation results in the extinction of evolutionary distinct interactions in a network and decreases phylogenetic diversity, with consequent loss of ecological functions (Emer et al. 2019).

In this way, ecological network analyses offer valuable insights into how defaunation impacts the provisioning of ecosystem services. An ecological network describes interactions among species of a community and with their environment in different ecological processes (Pascual & Dunne 2006, Dehling 2018). Measures such as connectance, degree distribution, nestedness, and modularity indicate how the ecological processes are organized in the network, as well as the network robustness, i.e., the tolerance of the network to perturbations and species extinction (Dunne et al. 2002, Dehling 2018). For instance, an increase in nestedness can be observed in response to defaunation, indicating that a few species provide a large number of ecosystem services, while many species provide only one or a few ecosystem services, thereby unbalancing the ecosystems (Bogoni et al. 2020). Additionally, the removal of highly connected species within an ecosystem can trigger secondary extinctions, resulting in the subsequent collapse of the food web (Dunne et al. 2002) and ecosystem services (Keyes et al. 2021).

Birds can be good models to investigate the impacts of defaunation on ecosystem services. Firstly, birds provide many ecosystem services, such as pollination, pest control, seed dispersal, ecosystem engineering, nutrient cycling, scavenging, and ecotourism (Michel et al. 2020). Secondly, extinction drivers such as land-use and climate changes, as well as hunting are threatening bird species worldwide (Szabo et al. 2012, BirdLife International 2023). For tropical birds, it is expected that climate and land-use changes provoke drastic contractions in their range sizes in the future (Jetz et al. 2007, Prieto-Torres et al. 2021). In addition, a recent meta-analysis showed that hunting pressure was responsible for a severe decrease in bird abundance in tropical areas (Benítez-López et al. 2017). As a result, these extinction drivers can contribute to increased bird defaunation, leading to a subsequent decline in the crucial ecosystem services provided by birds.

Defaunation can be severe in biodiversity hotspots, which harbor high diversity and endemism. A notable example is the Neotropic Atlantic Forest realm, boasting 223 endemic bird species (Vale et al. 2018b), out of which 65 are currently classified as threatened (see IUCN 2022, Ministério do Meio Ambiente 2022). The Atlantic Forest has a long history of degradation due to many human activities such as mining, wood exploitation, and cultivation of different crops (Lira et al. 2021). During the last decades, the main drivers of deforestation in the Atlantic Forest were urbanization, industrialization, and expansion of croplands, the last encouraged by mechanization and the international commodities market (Rezende et al. 2018, Faria et al. 2021, Lira et al. 2021). This intense land-use change has resulted in extensive forest loss, leaving behind fragmented remnants of native vegetation that are now highly isolated in numerous patches (Ribeiro et al. 2009, Rezende et al. 2018, Esquivel et al. 2019). Estimates revealed that there is only 7% of the Atlantic Forest's natural vegetation cover in Paraguay (Fleytas 2007), 28% in Brazil (Rezende et al. 2018), and 40% in Argentina (Cockle et al. 2010). But not only deforestation jeopardizes the endemic and threatened Atlantic Forest birds. They are suffering from multiple threats, primarily from habitat loss, but also from climate change, hunting, illegal trade, virus diseases, pollution, and invasive species (IUCN 2022).

In light of the critical situation that endemic and threatened Atlantic Forest birds are facing, there is a high risk of extinction of some of them in the near future. This potential loss of these bird species could have severe consequences, leading to a significant decline in the ecosystem services they provide. Thus, our primary objective was to comprehensively understand the impact of bird defaunation on the ecosystem services provision within the entire Atlantic Forest using different defaunation rates. We expected to observe substantial changes in the network structure as a result of bird defaunation, leading to a considerable proportion of ecosystem services being lost in this realm, mainly in scenarios with greater defaunation rates. By investigating these connections between bird species and ecosystem services, we seek to highlight the far-reaching implications of conserving endemic and vulnerable to extinction bird species.

Material and Methods

Study area

We considered the entire Atlantic Forest realm to investigate the impacts of bird defaunation on ecosystem services. We used the “Integrator Limit” delimitation proposed by Muylaert et al. (2018), which is 1,619,514 km² in extent (33° 46' 0.7314" S to 2° 48' 31.2942" S; 57° 53' 35.0874" W to 28° 50' 9.2754" W; Figure 1). The Atlantic Forest extends along a wide latitudinal and longitudinal gradient and is distributed in Argentina, Brazil, and Paraguay. It gives the Atlantic Forest a high climatic and forest diversity, which reflects in a high biodiversity and endemism (Ribeiro et al. 2009).

We divided the Atlantic Forest into hexagons of 10,000 ha ($1*10^8$ m²). Many studies have used this hexagon size within the Atlantic Forest to evaluate habitat suitability and landscape resilience (Pardini et al. 2010, Püttker et al. 2011, Crouzeilles et al. 2014, Tambosi et al. 2014, Strassburg et al. 2016). This size is large enough to include species with large dispersal abilities and generates a number of hexagons that allows robust analyses (Crouzeilles et al. 2014). We obtained a total of 15,900 hexagons within the Atlantic Forest delimitation.

Species database

We selected 65 endemic and threatened Atlantic Forest bird species (Table 1) following Vale et al. (2018b) and Prado et al. (2022). We primarily checked the status in the IUCN Red List of Threatened Species (IUCN 2022). For species that were not assessed by IUCN, we checked Brazil's Red List of Threatened Species (Ministério do Meio Ambiente 2022). Then, we downloaded the range maps available from the IUCN platform (IUCN 2022), and checked the distribution for each bird species using the Wikiaves platform (WikiAves 2023). For species with different distributions from WikiAves or without range map at IUCN, we built a minimum convex polygon (MCP) adding a buffer of 5 km around the occurrence records (Table S1). We selected a buffer of 5 km considering the mean dispersal ability estimated for medium to heavy bird species (~ 5 km) by Tourinho et al. (2022). To build the MCP, we gathered occurrence records from the Global Biodiversity Information Facility (GBIF; see

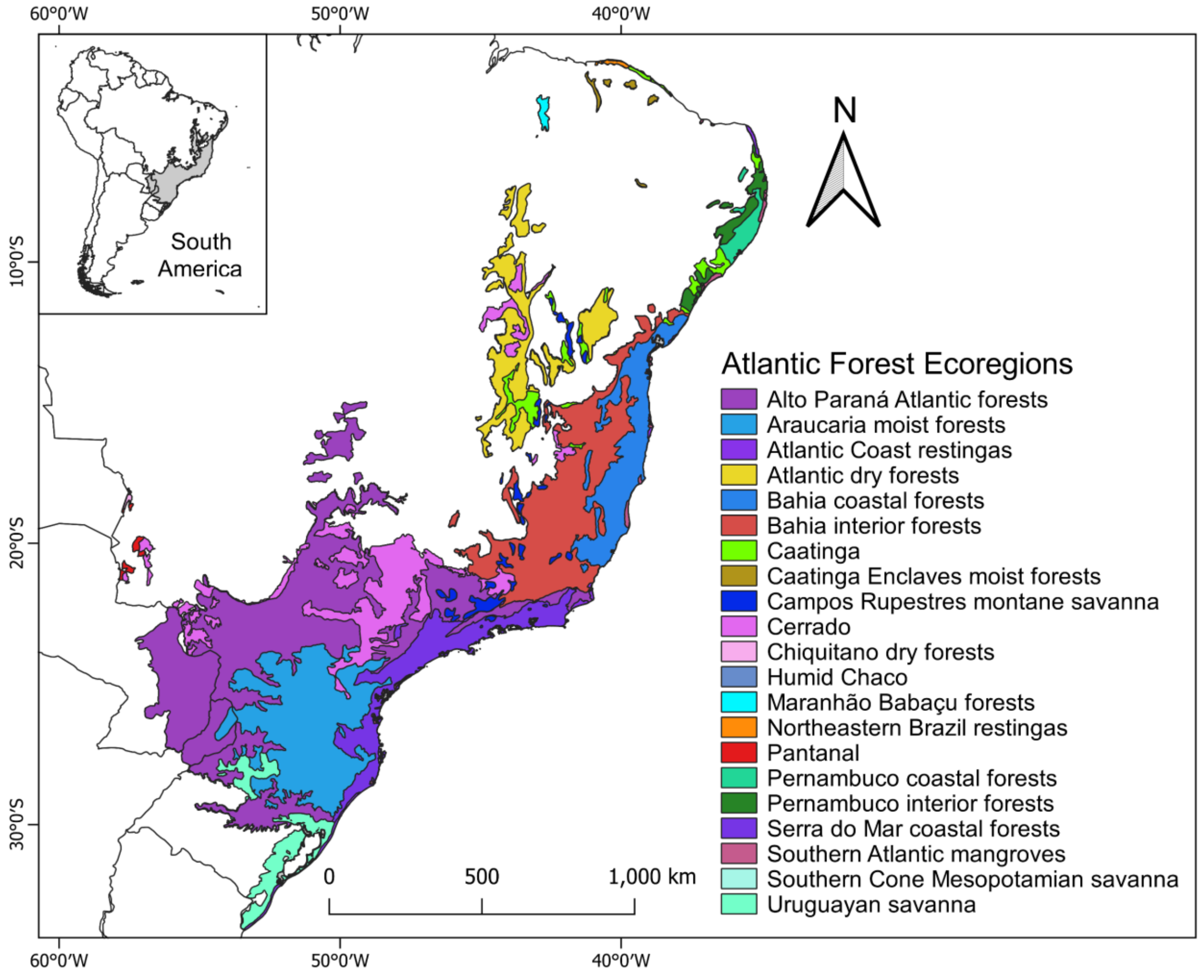

Figure 1. Distribution of the Atlantic Forest realm sensu Muylaert et al. (2018), and ecoregions sensu Olson et al. (2011).

Table S2 for DOI), Distributed Information System for Biological Collections – *species*Link (speciesLink 2021), Brazilian Biodiversity Information System – SiBBr (SiBBr 2021), VertNet (VertNet 2016), iNaturalist (iNaturalist 2021), eBird (eBird Basic Dataset 2022), and literature (Hasui et al., 2017). Based on the range maps, we built a matrix of presence and absence of the bird species within each hexagon.

Ecosystem services and matrix multiplication

Among 16 ecosystem services (ES) provided by Neotropical birds, as proposed by Michel et al. (2020), we specifically selected six: pollination, pest control, seed dispersal, birdwatching, food, and ecosystem engineering (Table 1). Additionally, we introduced two more ecosystem services to our study, namely, seed predation and biological control (Table 1), based on the control of weed plants by granivores and rodents by carnivores (see Whelan et al. 2008). We selected these ES considering the known traits and behaviors of the bird species, and which ES can most impact the ecosystem functioning and human well-being. We attributed the putative presence or absence of the ES based on body mass, diet, and behavioral traits present in the literature (Table 1). We also consulted an ornithologist (PhD. José Carlos Morante Filho) to review the ecosystem services attribution matrix. We then multiplied the presence-absence matrix of species per hexagon by the ES-by-species matrix to obtain a sum of ecosystem services (i.e., the number of species providing each ecosystem service) by hexagon (following Bogoni et al. 2020). After this process, we generated a baseline matrix with 193,900 observations (i.e., $\sum$ (species × ES × hexagons)).

Data analysis

We simulated three defaunation scenarios based on putative baseline matrix: low defaunation, medium defaunation, and high defaunation scenarios. The scenarios were

Table 1. Ecosystem services (ES) provided by 65 endemic and threatened Atlantic Forest bird species.

ES	Description	Source
Provision		
Food	Food resource for humans. We selected birds with body mass >200 g.	Michel et al. (2020); Tobias et al. (2022)
Regulation		
Pollination	Consumption of nectar. Species with a minimum of 90% of nectarivore diet.	Wilman et al. (2014)
Pest control	Consumption of invertebrates. Species with a minimum of 30% of invertivore diet.	Wilman et al. (2014)
Seed dispersal	Movement of the seeds away from the mother plant without embryo destruction. We considered species with a minimum of 30% of frugivore diet and confirmed dispersal of seed without embryo destruction.	Gaiotti et al. (2017); Liu & Lyons (2012); Tella et al. (2016); Wilman et al. (2014)
Seed predation	Consumption of seeds with embryo destruction. We considered species with a minimum of 70% of granivore diet or confirmed consumption of seeds with embryo destruction.	Coriolano (2015); Galleti & Rodrigues (1992); Kilpp et al. (2015); Pereira (2006); Wilman et al. (2014)

Table 1. (continued)

ES	Description	Source
Biological control	Consumption of vertebrates. We considered species with a minimum of 30% of carnivore diet.	Wilman et al. (2014)
Support		
Ecosystem engineering	We considered the feeding activity of woodpeckers and the excavation for nests by psittacids.	Michel et al. (2020); Vasconcelos et al. (2015)
Cultural		
Birdwatching	We considered all birds as potential species for birdwatching.	Michel et al. (2020)

defined using the IUCN threat status, which consists in randomly removing sets of bird species from hexagons (see Bogoni et al. 2020). For the low defaunation scenario, we removed: (i) 10% of all species classified as Vulnerable (VU); (ii) 20% of all species classified as Endangered (EN); and (iii) 50% of all species classified as Critically Endangered (CR), based on probability of extinction in the wild for each category defined by IUCN (2012). For the medium defaunation scenario, we removed: (i) 30% of all species classified as Vulnerable (VU); (ii) 50% of all species classified as Endangered (EN); and (iii) 85% of all species classified as Critically Endangered (CR). For the high defaunation scenario, we removed: (i) 50% of all species classified as Vulnerable (VU); (ii) 70% of all species classified as Endangered (EN); and (iii) 90% of all species classified as Critically Endangered (CR). We defined the defaunation rates for medium and high defaunation scenarios arbitrarily, but considering that the higher the degree of threat, the greater the probability of extinction (see IUCN 2012). We compared the species and ES spatial distributions between the baseline and low defaunation scenario, the low and medium defaunation scenario, and the medium and high defaunation scenario using the Kolmogorov-Smirnov (KS) test. We ran this analysis using the “syrjala” function from “ecespa” R package (de la Cruz 2008).

To investigate the defaunation effects on ecosystem services, we used the ecoregions proposed by Olson et al. (2001) inside the Atlantic Forest delimitation (Figure 1). These ecoregions represent distinct biotas nested within the realms and allow comparisons between species assemblages for conservation purposes (Olson et al. 2001). We considered as transition zones the ecoregions generally defined as belonging to the Caatinga, Cerrado, Pantanal, and Pampa realms. Using the hexagon centroids, we divided the hexagons into the ecoregions, delimiting the bird assemblages in each ecoregion. We used ecological bipartite networks to examine, in each ecoregion, the matrices of adjacency of ecosystem services generated in baseline and defaunation scenarios (see Boccaletti et al. 2006), i.e., the number of species in each ecoregion connected to any ecosystem service they provide (Bogoni et al. 2020).

To compare changes in robustness of the network of ecosystem services provided by species between baseline and defaunation scenarios, we obtained the following metrics: connectance; modularity; nestedness; ecosystem services average degree; and ecoregion average degree. Connectance is the proportion of realized ecological interactions among all possible interactions in the network (May 1973). Low connectance indicates that the network is highly sensitive to loss of interactions between species and ES, while high connectance indicates that the network is stable, preventing the ES loss (Bogoni et al. 2020). In a network, nodes (herein species per ecoregion and ecosystem service) can form groups representing subsets of nodes that have stronger connections within the group than with nodes outside the group. Modularity, or compartmentalization, is a measure that quantifies the probability of the nodes forming these groups, identifying the areas densely connected in a network (Boccaletti et al. 2006, Thébault 2013). Higher modularity indicates a more cohesive network and increases the persistence of a network to perturbations (Stouffer & Bascompte 2011, Bogoni et al. 2020). Nestedness is a structural pattern in networks that represents a hierarchical organization in which groups with less connected species/ecosystem services are a nested subset of the most connected (Almeida-Neto et al. 2008, Bogoni et al. 2020). Low nestedness values indicate a decrease in the overall amount of ES provided by birds, but high nestedness values indicate that the ES is provided unevenly among species (Bogoni et al. 2020). The average degree represents the mean number of interactions between species and their putative ecosystem service for each ES (ES average degree) and ecoregion (Ecoregion average degree) in the network, indicating the quantitative impact of defaunation scenarios on ecosystem services (Boccaletti et al. 2006, Bogoni et al. 2020). A higher average degree indicates a more complex and interconnected network of interactions among species and their putative ecosystem services. Conversely, a decline in this value is a clear indicator of potential ecosystem service losses resulting from defaunation. We ran the network analyses using the "bipartite" R package (Dormann et al. 2008). All analyses were performed using R version 4.3.0 (R Core Team 2023).

Results

Our results showed a defaunation rate of 17.1% for the low defaunation scenario, 42% for the medium defaunation scenario, and 60.4% for the high defaunation scenario. In the baseline scenario, the bird species are concentrated in the eastern Atlantic Forest (along Bahia and Pernambuco coastal forests), reaching a number of 24 species per hexagon. In the high defaunation scenario, it reaches a maximum of 16 species (Figure 2). We observed a significant difference in the species spatial distributions between the baseline and low defaunation scenario (KS = 0.006; p-value = 0.001), and between the low and medium defaunation scenario (KS = 0.011; p-value = 0.001), but no significant difference between the medium and high defaunation scenario (KS = 0.007; p-value = 0.06; Figure 2). The defaunation simulations decreased the ecosystem services from 193,900 links observed in the baseline matrix to 156,697 links in the low defaunation matrix (-19.2%), 106,753 links in the medium defaunation matrix (-44.9%), and 72,371 links in the high defaunation matrix (-62.7%). Ecosystem services losses can occur over the Atlantic Forest, but areas with fewer ecosystem services can lose all the services provided by the analyzed birds, in greater intensity as indicated in the high defaunation scenario (Figure 3). We observed a significant difference in the species spatial distributions between the baseline and low defaunation scenario (KS = 0.011; p-value = 0.001), between the low and medium defaunation scenario (KS = 0.029; p-value = 0.001), and between the medium and high defaunation scenario (KS = 0.033; p-value = 0.001; Figure 3). The main ES affected by defaunation was food in all

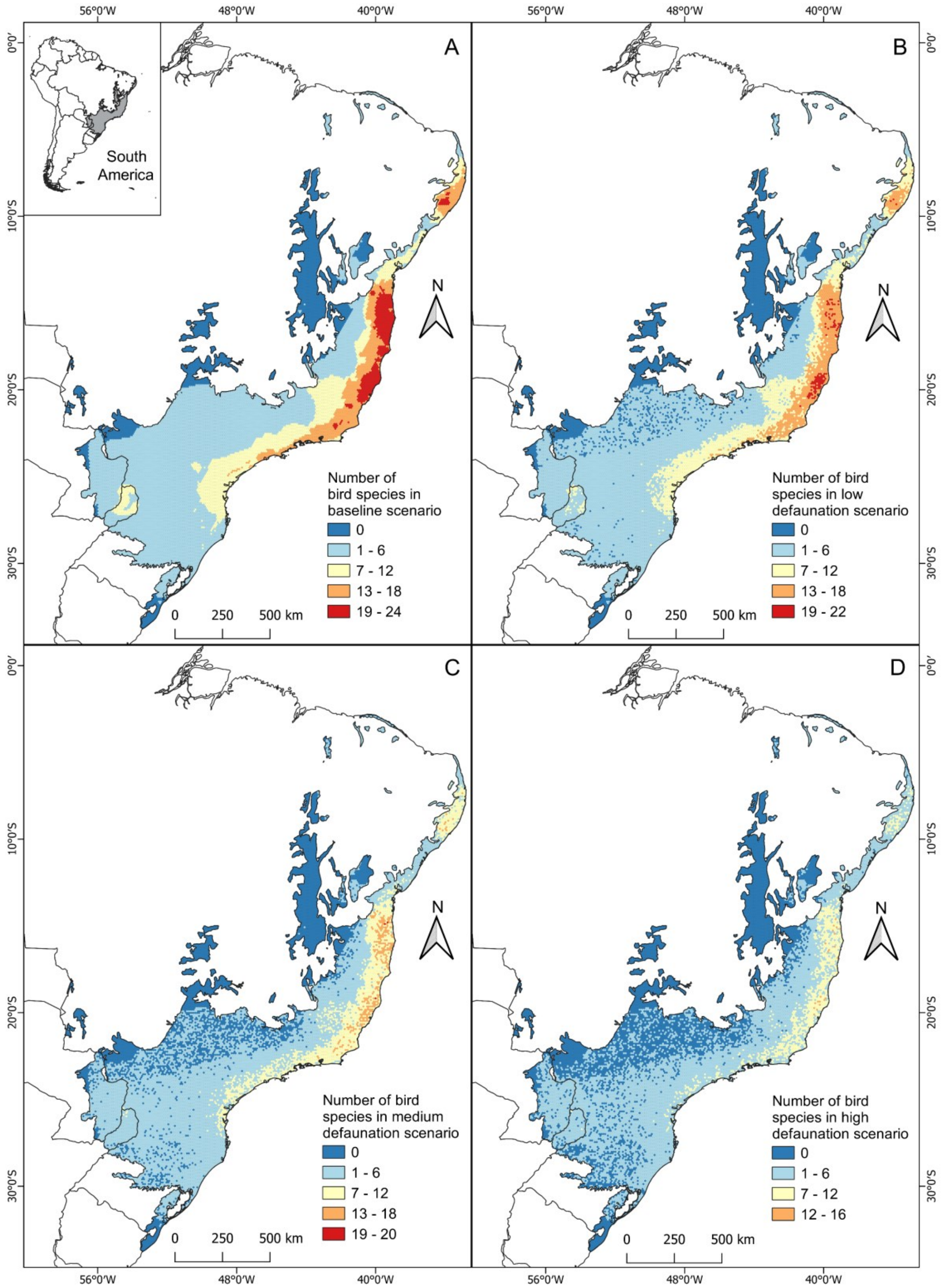

Figure 2. Distribution of 65 endemic and threatened Atlantic Forest bird species based on the matrix of presence and absence of the bird species (obtained from range maps) within each hexagon (size = 10,000 ha). The maps show the number of bird species in (A) baseline, (B) low defaunation (rate of 17.1%), (C) medium defaunation (rate of 42%), and (D) high defaunation (rate of 60.4%) scenarios.

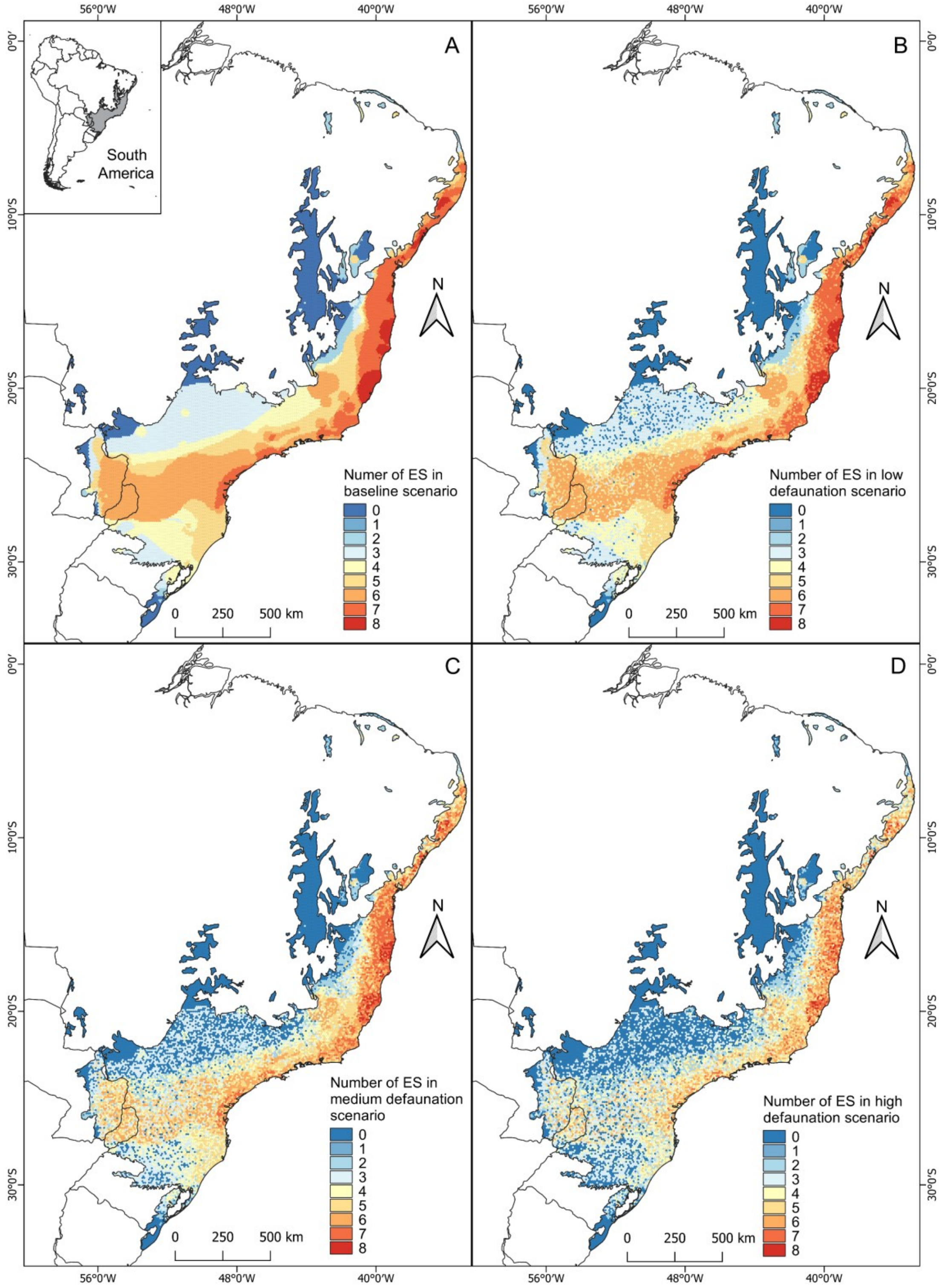

Figure 3. Distribution of ecosystem services (ES) provided by endemic and threatened Atlantic Forest bird species across the realm based on the presence-absence matrix of species per hexagon multiplied by the ES-by-species matrix. The maps show the number of ES in (A) baseline, (B) low defaunation (rate of 17.1%), (C) medium defaunation (rate of 42%), and (D) high defaunation (rate of 60.4%) scenarios.

defaunation scenarios (low: ~27%; medium: ~57%; high: 73%). Other main ES affected were: (i) seed dispersal (low: 24%; medium: 52%; high: ~69%); (ii) seed predation (low: ~21%; medium: 46%; high: ~63%); (iii) pollination (low: ~19%; medium: ~45%; high: ~67%); and (iv) birdwatching (low: 17%; medium: 42%; high: 60%; Table 2).

The network metrics revealed reductions in the interactions between endemic and threatened Atlantic Forest species and their putative ecosystem services per Atlantic Forest ecoregion from baseline to defaunation scenarios (Figure 4). The most prominent reductions were in Ecosystem services and Ecoregion average degrees. The ecosystem services average degree decreased from 31.4 in the baseline scenario to 25.9 in the low defaunation scenario, 18.2 in the medium defaunation scenario, and 12.2 in the high defaunation scenario. The ecoregion average degree decreased from 13.2 in the baseline scenario to 10.9 in the low defaunation scenario, 7.7 in the medium defaunation scenario, and 5.2 in the high defaunation scenario. Nestedness decreased from 83.8 to 81.7, 80.8, and 81.7 in low, medium, and high defaunation scenarios, respectively. Modularity was 0.1 in the baseline scenario and remained the same in the low defaunation scenario, decreasing to 0.09 and 0.08 in medium and high defaunation scenarios, respectively. Lastly, connectance remained the same (0.78) in the baseline, low, and medium defaunation scenarios, and decreased to 0.77 in the high defaunation scenario.

Discussion

Defaunation constitutes a significant disruption to biodiversity, exerting profound effects on ecosystem functioning, and consequently, causing negative impacts on the provisioning of essential ecosystem services (Dirzo et al. 2014). In this study, we explored the impact of varying rates of endemic and threatened bird defaunation on ecosystem services within the Atlantic Forest realm. Our findings demonstrate that the ecological network structure

Table 2. Cumulative ecosystem services (ES) in the hexagons in the baseline and defaunation scenarios (low, medium, and high defaunation) and the respective percentages of ES loss.

ES	Baseline	Low defaunation	Medium defaunation	High defaunation
Biological control	2,199	1,947 (-11.5%)	1,469 (-33.2%)	1,011 (-54.0%)
Birdwatching	74,495	61,729 (-17.1%)	43,231 (-42.0%)	29,490 (-60.4%)
Ecosystem engineering	5,562	4,924 (-11.5%)	3,703 (-33.4%)	2,643 (-52.5%)
Food	25,844	18,971 (-26.6%)	11,131 (-56.9%)	6,967 (-73.0%)
Pest control	27,750	23,932 (-13.8%)	17,754 (-36.0%)	12,517 (-54.9%)
Pollination	2,904	2,365 (-18.6%)	1,606 (-44.7%)	973 (-66.5%)
Seed dispersal	29,021	22,047 (-24.0%)	13,865 (-52.2%)	9,012 (-68.9%)
Seed predation	26,125	20,782 (-20.5%)	13,994 (-46.4%)	9,758 (-62.6%)

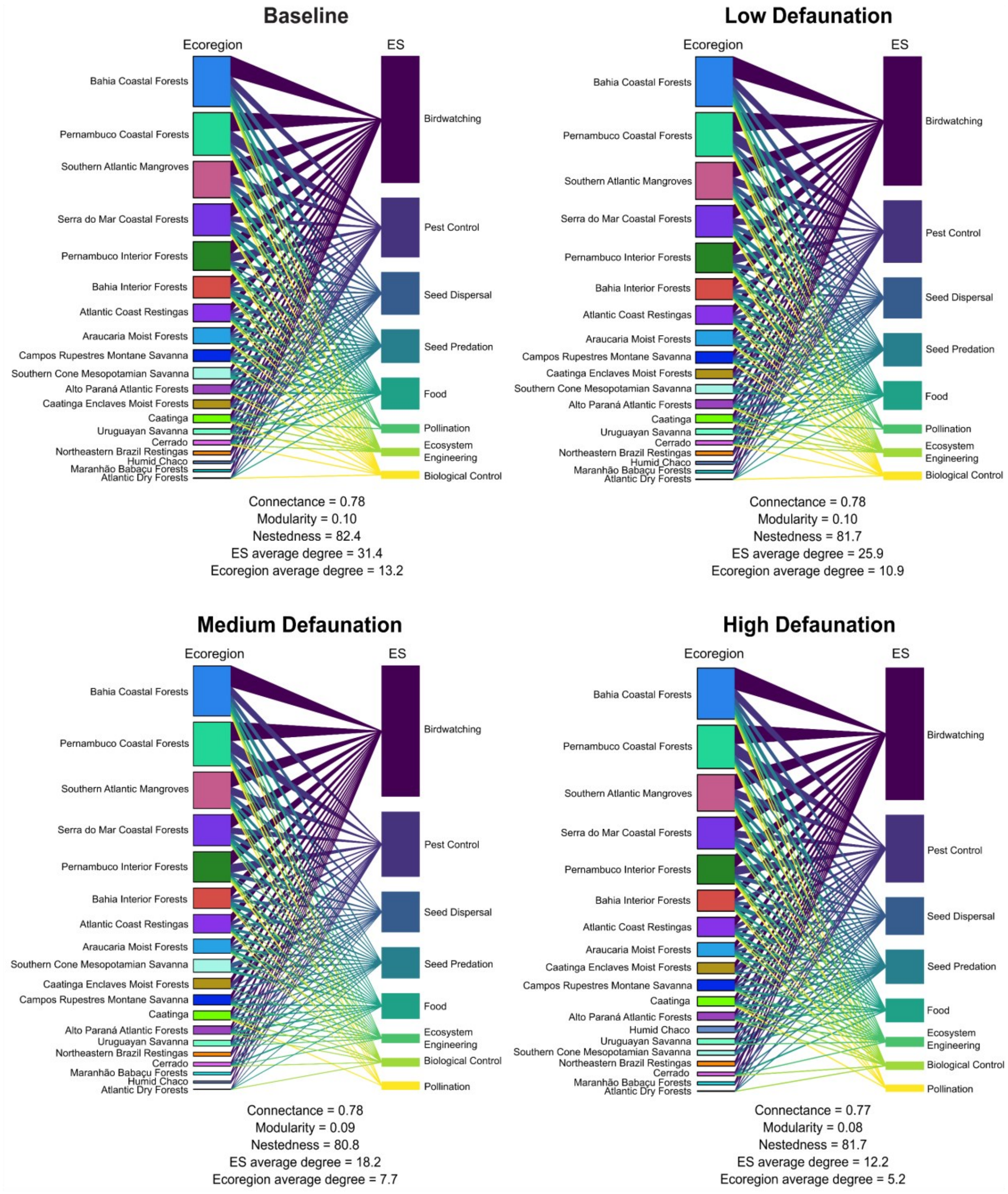

Figure 4. Ecological bipartite networks showing the interactions between endemic and threatened Atlantic Forest species and their putative ecosystem services (ES) per Atlantic Forest ecoregion in the baseline, low defaunation (17.1%), medium defaunation (42%) and high defaunation (60.4%) scenarios. High connectance, modularity, and nestedness indicate greater network robustness, while decreases in ES average degree and Ecoregion average degree indicate the average loss of each ES and ES per ecoregion, respectively.

(connections between bird species and ecosystem services by ecoregion) remained relatively stable with no critical changes even in a high defaunation scenario. However, we observed that both medium and high defaunation scenarios led to a substantial depletion in ecosystem services, raising concerns about potential alterations in ecosystem functioning and human well-being.

Tropical regions have experienced the highest vertebrate extinction rates in the last decades (Dirzo et al. 2014, Ceballos et al. 2017). In tropical forests worldwide, forest conversion was responsible for a decrease in bird species abundance by over 53% in the last 40 decades, severely impacting threatened bird species (Osuri et al. 2020). For the same period, overhunting in tropical regions was responsible for a decrease in bird abundances by 58% (reaching 76% in some areas), decreasing populations and species interactions (Benítez-López et al. 2017). In the Atlantic Forest, predictions showed that deforestation can cause a loss of 30–50% of endemic bird species in the near future (Brooks et al. 1999). Medium to high levels of defaunation in the Atlantic Forest were also estimated for mammals (Canale et al. 2012, Bogoni et al. 2018, Pereira et al. 2021). Hence, the defaunation rates found in this study both in the medium and high defaunation scenarios (~40–60%) were consistent with previous studies. Following these findings and considering the presence of several extinction drivers (e.g., habitat loss, climate change, hunting, invasive species competition, pollution, and diseases), it is likely that species defaunation will align with these scenarios (i.e., medium to high) rather than the low defaunation scenario (17%).

As expected, our findings revealed a substantial decline in ecosystem services provided by endemic and threatened birds in both medium and high defaunation scenarios. In these scenarios, all ecosystem services exhibited a decline by over 30%. Conversely, the low defaunation scenario demonstrated comparatively lower rates, with decreases below 30% in all ecosystem services. Among the ecosystem services, food provision showed the most significant depletion. The most important food resources for humans among birds are the large Neotropical birds such as those from the Cracidae family (Michel et al. 2020). Indeed, in this study, two of the bird species analyzed, *Aburria jacutinga* and *Crax blumenbachii*, have been found to be particularly vulnerable to the combination of intense hunting pressure and habitat loss (Brooks 2006). In addition, some birds used as sources of bushmeat by traditional communities are also threatened by illegal trade (*Amazona vinacea*, *Cotinga maculata*, *Paraclaravis geoffroyi*, *Pionus reichenowi*, and *Xipholena atropurpurea*; IUCN 2022).

Seed dispersal, seed predation, and pollination were also identified as the ecosystem services with the most significant degradation in both medium and high defaunation scenarios. Seed dispersal by frugivorous and granivorous birds plays a crucial role in ecosystem functioning in tropical forests, influencing the recruitment of surviving individuals and plant communities' regeneration (Wunderle 1997, Şekercioğlu 2006, de la Peña-Domene et al. 2014, González-Castro et al. 2019). But seed dispersal seems to be under a high risk of disruption (Michel et al. 2020). Plants that produce large-sized fruits or seeds rely solely on larger frugivores for dispersal (Kitamura et al. 2002). Large-bodied vertebrates are more likely to be extinct than smaller vertebrates (Galetti & Dirzo 2013, Emer et al. 2019, Michel et al. 2020), making plants with large-sized fruits or seeds particularly vulnerable to extinction if their dispersers face population decline or loss (Kitamura et al. 2002). The loss of large frugivorous birds is also associated with microevolutionary changes, leading to reductions in seed size, which facilitates the dispersal by small-sized birds (Galetti et al. 2013, Carvalho et al. 2016). Although smaller vertebrates may exhibit some resilience, persistent disturbances during the Anthropocene can cause the defaunation of small birds, which can have adverse effects on their interactions with plants that produce small and medium-sized fruits and seeds (Cordeiro & Howe 2003).

Seed predation is also an important process in forest regeneration and plant demography (Janzen 1971, Hulme 1998). Considering terrestrial granivorous birds (Columbidae and Tinamidae families), seed predation seems to be more important in fragmented areas because there is a decrease in mammalian seed predator's density and an increase in terrestrial granivorous birds' density (Pizo & Vieira 2004). However, in our study, seven out of 11 species are from the Psittacidae family. Psittacids are considered important pre-dispersal seed predators (controlling plant recruitment) and show different intensities of seed predation between fragmented (high intensity) and continuous forests (low intensity; Francisco et al. 2002). Considering that the psittacids in this study inhabit forests, defaunating these birds can potentially interfere with forest dynamics within their geographical distribution.

Pollination is an indispensable ecosystem service in tropical regions, where plants are highly dependent on animal pollinators: estimates show that 94% of plants in tropical communities are pollinated by animals (Ollerton et al. 2011). In the Atlantic Forest of southeastern Brazil, hermit hummingbirds play a significant role in plant pollination, contributing to a substantial percentage of the process (Buzato et al. 2000). Hermit hummingbirds (represented here by *Glaucis dohrnii* and *Phaethornis margarettae*), a separate group from the majority of hummingbird species, have long bills specialized in tubular flowers, are non-territorials, and show traplining behavior, i.e., travel between clumps of flowers in a regular route and specific sequence trave (Stiles 1975, Hinkelmann & Schuchmann 1997). These characteristics imply that hermits do not stay for long periods in one patch and spend much time searching for food, allowing them to disperse pollen over long distances, making them important pollinators by preventing plant inbreeding (Linhart 2017). Moreover, defaunating bird pollinators reduces pollination, fruit size and consequent seed number per fruit, and plant density, negatively affecting plant dynamics (Anderson et al. 2011).

Our findings also showed important decreases in birdwatching in both medium and high defaunation scenarios. Birdwatching is a cultural service that stimulates the conservation of natural areas, contributes to ornithological knowledge, provides human well-being, and brings socioeconomic benefits (Şekercioğlu 2002, Michel et al. 2020). Examples of economic activities stimulated by birdwatching encompass expenditures on equipment, books and field guides, travel (which mobilizes an entire tourism network including accommodations, attractions, tour guides, tourism agencies, and local commerce), as well as engagement in events associated with this activity (Kronenberg 2014). In addition, many birdwatchers expend a considerable amount of money and time to see rare bird species (Booth et al. 2011, Brock et al. 2020). Considering that the Neotropical region concentrates a great number of endemic and rare species, the defaunation of these birds can negatively impact these economic activities. The main threats to birdwatching seem to be the combination of deforestation and climate change, posing a significant threat to rare, iconic, and charismatic bird species, which tend to inhabit wetter and more forested areas (Echeverri et al. 2021). Consequently, the defaunation of these birds leads to a decline in culturally important avian populations, negatively impacting the provision of valuable cultural ecosystem services (Echeverri et al. 2021).

The losses in ES were confirmed in the network metrics. The decrease of ES average degree and ecoregion average degree from baseline to defaunation scenarios demonstrate a degradation in ecosystem services provided by birds, with higher intensity in regional defaunation. However, our estimates also revealed high nestedness and connectance, and low modularity in all scenarios, which could indicate that the network is more robust and cohesive and, consequently, more persistent and resilient to disturbances (Bascompte et al. 2003, Thébault & Fontaine 2010). This resilience can be explained by a functional redundancy within the network (Biggs et al. 2020), which must be provided by the coarse hexagon scale used in the analysis. It means that, due to the coarse scale, there was a substantial functional redundancy (i.e., different species providing the same ES) to buffer the ES loss that can occur in finer scales (Bogoni et al. 2020).

Our study highlights the concern about losing ecosystem services by bird defaunation and its impacts on ecosystem functioning and human well-being. The estimates suggest that the network structure connecting these bird species and their associated ecosystem services within the Atlantic Forest ecoregions is complex and potentially resilient. Additionally, at a coarse scale, the loss of certain species does not implicate in reducing some ES provision, as other species might perform similar ecosystem services due to functional redundancy. However, we cannot ignore that our results indicate that there may be substantial losses in the ecosystem services within the Atlantic Forest provided by endemic and threatened birds, increasing the potential losses of key processes for ecosystem functioning. Given the potential association of functional redundancy with the coarse scale, we emphasize the necessity for further studies to investigate how the Atlantic Forest birds' defaunation impacts the provision of ecosystem services on finer scales. Endemic and threatened Atlantic Forest birds are facing an imminent risk of extinction due to many threats caused by human activities. The Atlantic Forest is extensively deforested and highly fragmented due to agricultural development and urbanization (da Ponte et al. 2017, Rezende et al. 2018), reflecting in high rates of habitat loss and extinction risk by all bird species analyzed in this study (IUCN 2022). These implications extend beyond the forest ecosystem and directly impact human well-being, as birds play crucial roles in providing food resources and cultural services. Our findings further underscore the well-established discussions in the literature concerning the Atlantic Forest, its threatened bird species, and the provision of ecosystem services, emphasizing the imperative for creating new protected areas, restoring degraded landscapes, implementing strong policies for deforestation control, and mitigating human impacts on biodiversity (Bogoni et al. 2018, Vale et al. 2018a, Pereira et al. 2019, Develey 2021). By doing so, we can safeguard the endemic and threatened Atlantic Forest bird populations and the ecosystem services they provide, ultimately promoting a healthier and more sustainable environment for all living beings that depend on this unique and biodiverse ecosystem.

Supporting information for chapter 3

Table S1. Status and map source of 65 endemic and threatened Atlantic Forest bird species. Map source: the species' range maps were obtained from the IUCN platform (IUCN 2022) or were built using a minimum convex polygon (MCP) enveloping the species' presence records (gathered from online and literature data sources).

Species	Status	Map source
Acrobatornis fonsecai	VU	MPC
Amazona pretrei	VU	IUCN
Amazona rhodocorytha	VU	MPC
Amazona vinacea	EN	IUCN
Antilophia bokermanni	CR	MPC
Anumara forbesi	VU	IUCN
Automolus lammi	EN	IUCN
Biatas nigropectus	VU	IUCN
Buteogallus lacernulatus	VU	IUCN
Calyptura cristata	CR	IUCN
Celeus galeatus	VU	IUCN
Celeus tinnunculus	VU	IUCN
Cotinga maculata	CR	IUCN
Crax blumenbachii	EN	IUCN
Dendrocincla taunayi	EN	MPC
Dysithamnus plumbeus	VU	IUCN
Eleoscytalopus psychopompus	EN	IUCN
Formicivora erythronotos	EN	IUCN
Formicivora littoralis	EN	MPC

Table S1. (continued)

Species	**Status**	**Map source**
Formicivora paludicola	CR	IUCN
Glaucidium mooreorum	CR	IUCN
Glaucis dohrnii	VU	IUCN
Hemitriccus furcatus	VU	MPC
Hemitriccus kaempferi	VU	IUCN
Hemitriccus mirandae	VU	IUCN
Herpsilochmus pileatus	VU	IUCN
Iodopleura pipra	EN	IUCN
Leptodon forbesi	EN	IUCN
Lipaugus conditus	VU	IUCN
Merulaxis stresemanni	CR	IUCN
Myrmoderus ruficauda	EN	IUCN
Myrmotherula minor	VU	IUCN
Myrmotherula snowi	CR	IUCN
Myrmotherula urosticta	VU	IUCN
Nemosia rourei	CR	IUCN
Onychorhynchus swainsoni	VU	IUCN
Paraclaravis geoffroyi	CR	IUCN
*Phaethornis margarettae**	EN	IUCN
Phylloscartes beckeri	EN	IUCN
Phylloscartes ceciliae	CR	IUCN
*Piculus polyzonus**	EN	IUCN
Pionus reichenowi	VU	IUCN
Pipile jacutinga	EN	IUCN
Platyrinchus leucoryphus	VU	IUCN

Table S1. (continued)

Species	**Status**	**Map source**
Pyriglena atra	EN	IUCN
Pyriglena pernambucensis	VU	MPC
Pyrrhura cruentata	VU	IUCN
Pyrrhura griseipectus	EN	IUCN
Pyrrhura leucotis	VU	IUCN
Rhopornis ardesiacus	EN	IUCN
Sclerurus cearensis	VU	IUCN
Scytalopus diamantinensis	EN	IUCN
Scytalopus gonzagai	EN	IUCN
Scytalopus iraiensis	EN	IUCN
Sporophila falcirostris	VU	IUCN
Sporophila frontalis	VU	IUCN
Stilpnia peruviana	VU	IUCN
Synallaxis infuscata	EN	IUCN
Tangara fastuosa	VU	IUCN
Terenura sicki	CR	IUCN
Thalurania watertonii	EN	IUCN
Thripophaga macroura	VU	IUCN
Touit surdus	VU	IUCN
Xipholena atropurpurea	VU	IUCN
Xiphorhynchus atlanticus	VU	IUCN

**Phaethornis margarettae* and *Piculus polyzonus* are considered by IUCN as *Phaethornis malaris* and *Piculus chrysochloros* subspecies, respectively. In this study, we considered as different species and split up the IUCN range maps, separating the Atlantic Forest distribution from the Amazonian distribution.

Table S2. List of bird occurrence records downloaded from Global Biodiversity Information Facility (GBIF) to create distribution maps for species without range maps available at IUCN platform.

Species	**GBIF source**
Acrobatornis fonsecai	https://doi.org/10.15468/dl.hhpuxa
Amazona rhodocorytha	https://doi.org/10.15468/dl.4b963y
Antilophia bokermanni	https://doi.org/10.15468/dl.bhbt2t
Dendrocincla taunayi	https://doi.org/10.15468/dl.dxkhzk
Formicivora littoralis	https://doi.org/10.15468/dl.5gabu3
Hemitriccus furcatus	https://doi.org/10.15468/dl.dxkhzk
Pyriglena pernambucensis	https://doi.org/10.15468/dl.cwkkcm

General conclusion

In this thesis, I investigated the effects of future climate and land-use changes on threatened Neotropical bird species. I also evaluated, under different scenarios, how endemic and threatened bird species defaunation can affect the network structure and the provision of essential ecosystem services in the Atlantic Forest, a biodiversity hotspot in South America and one of the most endangered realms in the world.

The separate and combined climate and forest cover variables in Chapter 1 enable us to identify the risks suffered by a wide-ranging species (*Crax fasciolata*) under different future environmental policy scenarios. Using the applied approach in this chapter, I could identify that forest cover decrease seems to be the strongest driver of environmental suitability loss for *Crax fasciolata*. Many areas where the species occurs will maintain high climate suitability in the future but might not have suitable habitats if no stronger efforts in reforestation and deforestation control are taken into account. Moreover, as *Crax fasciolata* occupies different domains, which have different public conservation policies, understanding the influence of these extinction drivers across these diverse domains enables the development of more effective conservation strategies to be adopted according to the reality of each domain.

I also showed how combining climate and forest cover variables changes environmental suitability from current to future scenarios for endemic and threatened Atlantic Forest birds. The EcoLand approach used in Chapter 2 proved to be a valuable tool for two steps: (i) understanding the species responses to environmental changes; and (ii) identifying the priority areas for both conservation and restoration efforts in current and future scenarios. It is very clear that tropical birds will be negatively impacted for land-use and climate changes and will lose suitable areas in the near future (Şekercioğlu et al. 2008, 2012). The situation worsens for endemic and threatened bird species, which have limited distributions and, in most cases, specialized habitats, exacerbating their vulnerability (Jetz et al. 2007, Borges et al. 2019, Manes et al. 2021). Especially concerning endemic and threatened Atlantic Forest birds, my research revealed that more than 70% of the key conservation zones are outside the extant Protected Areas. This emphasizes the need to establish new protected areas in regions with high suitability, ensuring the preservation of a high-quality environment for the conservation of these species. In addition, restoring the regions with lower forest cover percentage enhances the quality of the forest and the connectivity of highly fragmented areas, which is crucial to increase bird species abundance and facilitate dispersal (Uezu & Metzger 2016, Borges & Loyola 2020, de la Sancha et al. 2021, Diniz et al. 2022).

Beyond the biodiversity loss, we demonstrated in Chapter 3 that key ecosystem services can be lost with medium to high rates of defaunation of endemic and threatened birds assemblages in the Atlantic Forest. These losses raise concerns about how ecosystem processes and human activities and well-being will be affected, and which measures should be taken by decision-makers to minimize the negative impacts. This research is a first step to identify which areas are under high risk to lose the ecosystem services provided by endemic and threatened Atlantic Forest birds, helping in the development of public policies associated with the main bird extinction drivers in tropical areas, particularly habitat loss. The Atlantic Forest is still being degraded due to increase of croplands, pasturelands, urbanization, and mining (MapBiomas 2023). Native forest cover remains stable in this domain due to the trade-off between older and younger native forest cover (Rosa et al. 2021). While increases in younger native forest cover is important to maintain biodiversity, older native forest cover is continuously reduced by these anthropic land-uses, which negatively impacts biodiversity since many organisms cannot recolonize secondary forests, also impacting the provision of key ecosystem services (Rosa et al. 2021). The other bird extinction drivers such as climate change, overexploitation and invasive species should also be considered by decision-makers.

It is crucial to ensure the compliance of environmental public policies to reduce greenhouse gas emissions (such as encouraging the replacement of fossil fuels with renewable energy sources), control deforestation, guarantee sustainable agricultural practices, control invasive species, and punish poachers.

The results presented here are important contributions to develop conservation actions to minimize the human impacts on threatened Neotropical birds and the ecosystem services they provide. Habitat loss and climate change are among the main challenges faced by the planet in short-, medium-, and long-terms. I reinforce the urgent need for reforestation, deforestation control, reduction in biodiversity exploitation, and actions/policies to mitigate climate change. I suggest that further studies should be carried out on other bird species to enhance the comprehension of how the bird extinction drivers can impact their future distribution, as well as the ecosystem services associated with them. Furthermore, studies on finer scales can bring additional information to these predictions and help local governments to develop better conservation efforts.

References

Aiello-Lammens ME, Boria RA, Radosavljevic A, Vilela B & Anderson RP. 2015. spThin: An R package for spatial thinning of species occurrence records for use in ecological niche models. Ecography, 38: 541–545.

Allouche O, Tsoar A & Kadmon R. 2006. Assessing the accuracy of species distribution models: Prevalence, kappa and the true skill statistic (TSS). Journal of Applied Ecology, 43: 1223–1232.

Almeida AC, Tourinho L, Ramalho Q, Jenkins CN & Almeida-Gomes M. 2022. Regional variation in responses of a vulnerable bird species to land-use and climate change. Biotropica, 54: 1190–1204.

Almeida-Neto M, Guimarães P, Guimarães JR PR, Loyola RD & Ulrich W. 2008. A consistent metric for nestedness analysis in ecological systems: reconciling concept and measurement. Oikos, 117: 1227–1239.

Anderson SH, Kelly D, Ladley JJ, Molloy S & Terry J. 2011. Cascading effects of bird functional extinction reduce pollination and plant density. Science, 331: 1068–1071.

Andrén H. 1994. Effects of Habitat Fragmentation on Birds and Mammals in Landscapes with Different Proportions of Suitable Habitat: A Review. Oikos, 71: 355–366.

Antonelli A. 2022. The rise and fall of neotropical biodiversity. Botanical Journal of the Linnean Society, 199: 8–25.

Araújo MB & New M. 2007. Ensemble forecasting of species distributions. Trends in Ecology and Evolution, 22: 42–47.

Araújo MB, Whittaker RJ, Ladle RJ & Erhard M. 2005. Reducing uncertainty in projections of extinction risk from climate change. Global Ecology and Biogeography, 14: 529–538.

Araújo MB, Alagador D, Cabeza M, Nogués-Bravo D & Thuiller W. 2011. Climate change threatens European conservation areas. Ecology Letters, 14: 484–492.

Araújo MB, Ferri-Yáñez F, Bozinovic F, Marquet PA, Valladares, F & Chown, SL. 2013. Heat freezes niche evolution. Ecology Letters, 16: 1206–1219.

Argentina. 2007. Ley 26.331. Presupuestos Mínimos de Protección Ambiental de los Bosques Nativos. Establécense los presupuestos mínimos de protección ambiental para el enriquecimiento, la restauración, conservación, aprovechamiento y manejo sostenible de los bosques nativos. Congreso de la Nación Argentina, Argentina.

Asner GP, Loarie SR & Heyder U. 2010. Combined effects of climate and land-use change on the future of humid tropical forests. Conservation Letters, 3: 395–403.

Assine ML, Merino ER, Pupim FN, Warren LV, Guerreiro RL. & McGlue MM. 2015. Geology and geomorphology of the pantanal basin. In: Dynamics of the Pantanal Wetland in South America (Eds. I Bergier & ML Assine), vol. 37, pp. 23–50. Springer, Cham.

Assis LFFG, Ferreira KR, Vinhas L, Maurano L, Almeida C, Carvalho A, Rodrigues J, Maciel A & Camargo C. 2019. TerraBrasilis: A Spatial Data Analytics Infrastructure for Large-Scale Thematic Mapping. ISPRS International Journal of Geo-Information, 8: 513.

Barbet-Massin M, Jiguet F, Albert CH & Thuiller W. 2012. Selecting pseudo-absences for species distribution models: How, where and how many? Methods in Ecology and Evolution, 3: 327–338.

Barve N, Barve V, Jiménez-Valverde A, Lira-Noriega A, Maher SP, Peterson AT, Soberón J, Villalobos F. 2011. The crucial role of the accessible area in ecological niche modeling and species distribution modeling. Ecological Modelling, 222: 1810–1819.

Bascompte J, Jordano P, Melián CJ & Olesen JM. 2003. The nested assembly of plant–animal mutualistic networks. Proceedings of the National Academy of Sciences, 100: 9383–

9387.

Bellard C, Bertelsmeier C, Leadley P, Thuiller W & Courchamp F. 2012. Impacts of climate change on the future of biodiversity. Ecology Letters, 15: 365–377.

Bellard C, Leclerc C & Courchamp F. 2015. Combined impacts of global changes on biodiversity across the USA. Scientific Reports, 5: 11828.

Bello C, Galetti M, Pizo MA, Magnago LFS, Rocha MF, Lima RAF, Peres CA, Ovaskainen O & Jordano P. 2015. Defaunation affects carbon storage in tropical forests. Science Advances, 1: e1501105.

Benítez-López A, Alkemade R, Schipper AM, Ingram DJ, Verweij PA, Eikelboom JAJ & Huijbregts AJ. 2017. The impact of hunting on tropical mammal and bird populations. Science, 356: 180–183.

Berg MP, Toby Kiers E, Driessen G, van der Heijden M, Kooi BW, Kuenen F, Liefting M, Verhoef HA & Ellers, J. 2010. Adapt or disperse: Understanding species persistence in a changing world. Global Change Biology, 16: 587–598.

Bernardo CSS, Cresswell B, Lloyd H, Azeredo R & Simpson J. 2011. Selection of radio transmitter and attachment method for post-release monitoring of captive-bred reintroduced red-billed curassow *Crax blumenbachii*, Brazil. European Journal of Wildlife Research, 57: 689–694.

Biggs CR, Yeager LA, Bolser DG, Bonsell C, Dichiera AM, Hou Z, Keyser SR, Khursigara AJ, Lu K, Muth AF, Negrete B & Erisman BE. 2020. Does functional redundancy affect ecological stability and resilience? A review and meta-analysis. Ecosphere, 11: e03184.

BirdLife International. 2016. *Crax fasciolata*. IUCN Red List Threat. Species 2016, e.T45092100A95141387.

BirdLife International. 2023. State of the world's birds: 2022 annual update. Available at: http://datazone.birdlife.org/2022-annual-update [Accessed August 1, 2023].

Boccaletti S, Latora V, Moreno Y, Chavez M & Hwang DU. 2006. Complex networks: structure and dynamics. Physics Report, 424: 175–308.

Bogoni JA, Pires JSR, Graipel ME, Peroni N & Peres CA. 2018. Wish you were here: how defaunated is the atlantic forest biome of its medium- to large-bodied mammal fauna? PLoS ONE, 13: e0204515.

Bogoni JA, Peres CA & Ferraz KMPMB. 2020. Effects of mammal defaunation on natural ecosystem services and human well being throughout the entire Neotropical realm. Ecosystem Services, 45: 101173.

Bonfim FCG, Cordeiro PHC, Peres CA, Canale GR & Bernardo CSS. 2018. Combining modeling tools to identify conservation priority areas: A case study of the last large-bodied avian frugivore in the Atlantic Forest. Global Ecology and Conservation, 15: e00426.

Booth JE, Gaston KJ, Evans KL & Armsworth PR. 2011. The value of species rarity in biodiversity recreation: A birdwatching example. Biological Conservation, 144: 2728–2732.

Borges FJA & Loyola R. 2020. Climate and land-use change refugia for Brazilian Cerrado birds. Perspectives in Ecology and Conservation, 18: 109–115.

Borges FJA, Ribeiro BR, Lopes LE & Loyola R. 2019. Bird vulnerability to climate and land use changes in the Brazilian Cerrado. Biological Conservation, 236: 347–355.

Bornschein MR, Maurício GN, Belmonte-Lopes R, Mata H. & Bonatto SL. 2007. Diamantina Tapaculo, a new Scytalopus endemic to the Chapada Diamantina, northeastern Brazil (Passeriformes: Rhinocryptidae). Revista Brasileira de Ornitologia, 15: 151–174.

Brasil. 2000. Lei nº 9.985, de 18 de julho de 2000. Regulamenta o art. 225, § 1o, incisos I, II, III e VII da Constituição Federal, institui o Sistema Nacional de Unidades de conservação da Natureza e dá outras providências. Available at:

http://www.planalto.gov.br/ccivil_03/leis/l9985.htm

Brasil. 2006. Lei nº 11.428, de 22 de dezembro de 2006. Dispõe sobre a utilização e proteção da vegetação nativa do Bioma Mata Atlântica, e dá outras providências. Available at: http://www.planalto.gov.br/ccivil_03/_ato2004-2006/2006/lei/l11428.htm

Brasil. 2012. Lei nº 12.651, de 25 de maio de 2012. Dispõe sobre a proteção da vegetação nativa; altera as Leis nºs 6.938, de 31 de agosto de 1981, 9.393, de 19 de dezembro de 1996, e 11.428, de 22 de dezembro de 2006; revoga as Leis nºs 4.771, de 15 de setembro de 196. Available at: http://www.planalto.gov.br/ccivil_03/_ato2011-2014/2012/lei/l12651.htm

Brazil. 2017. Planaveg: Brazilian National Plan for Native Vegetation Recovery. Brazil's Ministry of Environment, Ministry of Agriculture, Livestock and Food Supply, Ministry of Education. Ministério do Meio Ambiente, Brasília, 823 pp.

Brock M, Fraser I, Law C, Mitchell S & Roberts DL. 2020. An economic analysis of twitching behaviour and species rarity, Journal of Environmental Economics and Policy, 1–20.

Brodie J, Post E & Laurance WF. 2012. Climate change and tropical biodiversity: A new focus. Trends in Ecology and Evolution, 27: 145–150.

Brooks DM. 2006. Conserving Cracids: the most Threatened Family of Birds in the Americas. Miscellaneous Publications of The Houston Museum of Natural Science, Houston, 169 pp.

Brooks T, Tobias J & Balmford A. 1999. Deforestation and bird extinctions in the Atlantic forest. Animal Conservation 2: 211–222.

Buzato S, Sazima M & Sazima I. 2000. Hummingbird-pollinated floras at three atlantic forest sites. Biotropica, 32: 824–841.

Câmara G, Soterroni A, Ramos F, Carvalho A, Andrade P, Souza RC, Mosnier A, Mant R, Buurman M, Pena M, Havlik P, Pirker J, Kraxner F, Obersteiner M, Kapos V, Affonso A, Espíndola G & Bocqueho G. 2015. Modelling land use changes in Brazil 2000–2050: a report by the REDD-PAC project. INPE, IPEA, IIASA, UNEP-WCMC, São José dos Campos, Brasília, Laxenburg, Cambridge, 93 pp.

Canale GR, Peres CA, Guidorizzi CE, Gatto CAF & Kierulff MCM. 2012. Pervasive defaunation of forest remnants in a tropical biodiversity hotspot. PLoS ONE, 7: 41671.

Cannon AJ. 2020. Reductions in daily continental-scale atmospheric circulation biases between generations of global climate models: CMIP5 to CMIP6. Environmental Research Letters, 15: 064006.

Carrara E, Arroyo-Rodríguez V, Vega-Rivera JH, Schondube JE, de Freitas SM & Fahrig L. 2015. Impact of landscape composition and configuration on forest specialist and generalist bird species in the fragmented lacandona rainforest, mexico. Biological Conservation, 184: 117–126.

Carvalho CS, Galetti M, Colevatti RG & Jordano P. 2016. Defaunation leads to microevolutionary changes in a tropical palm. Scientific Reports, 6: 31957.

Ceballos G, Ehrlich PR & Dirzo R. 2017. Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. Proceedings of the National Academy of Sciences of the United States of America, 114: e6089–e6096.

Cockle KL, Martin K & Drever MC. 2010. Supply of tree-holes limits nest density of cavity-nesting birds in primary and logged subtropical Atlantic forest. Biological Conservation, 143: 2851–2857.

Cordeiro NJ & Howe HF. 2003. Forest fragmentation severs mutualism between seed dispersers and an endemic African tree. Proceedings of the National Academy of Sciences, 100: 14052–14056.

Coriolano IR. 2015. Hábitos alimentares do periquito cara-suja *Pyrrhura griseipectus*

Salvadori, 1900 (Aves: Psittacidae). Monograph. Universidade Federal do Ceará, Fortaleza. 29 pp.

Corlett RT. 2016. The Impacts of Droughts in Tropical Forests. Trends in Plant Science, 21: 584–593.

Crain CM, Halpern BS, Beck MW, Kappel CV. 2009. Understanding and managing human threats to the coastal marine environment. Annals of the New York Academy of Sciences, 1162: 39–62.

Crouzeilles R, Vale MM, Cerqueira R & Grelle CEV. 2013. Increasing strict protection through protected areas on Brazilian private lands. Environmental Conservation, 40: 209–210.

Crouzeilles R, Prevedello JA, Figueiredo MSL, Lorini ML & Grelle CEV. 2014. The effects of the number, size and isolation of patches along a gradient of native vegetation cover: how can we increment habitat availability? Landscape Ecology, 29: 479–489.

Crouzeilles R, Santiami E, Rosa M, Pugliese L, Brancalion PHS, Rodrigues RR, Metzger JP, Calmon M, Scaramuzza CAM, Matsumoto MH, Padovezi A, Benini RM, Chaves RB, Metzker T, Fernandes RB, Scarano FR, Schmitt J, Lui G, Christ P, Vieira RM, Senta MMD, Malaguti GA, Strassburg BBN & Pinto S. 2019. There is hope for achieving ambitious Atlantic Forest restoration commitments. Perspectives in Ecology and Conservation, 17: 80–83.

da Fonseca GAB, Rylands A, Paglia A & Mittermeier RA. 2004. Atlantic Forest. In: Hotspots revisited: earth’s biologically richest and most endangered terrestrial ecosystems (Eds. RA Mittermeier, PR Gil, M Hoffmann, J Pilgrim, T Brooks, CG Mittermeier, J Lamoreux & GAB. da Fonseca), pp. 84–88. CEMEX, Mexico City.

da Ponte E, Roch M, Leinenkugel P, Dech S & Kuenzer C. 2017. Paraguay’s Atlantic Forest cover loss – Satellite-based change detection and fragmentation analysis between 2003 and 2013. Applied Geography, 79: 37–49.

Dehling DM. 2018. The structure of ecological networks. In: Ecological Networks in the Tropics: an integrative overview of species interactions from some of the most species-rich habitats on Earth (Eds. W Dáttilo & V Rico-Gray), pp. 29–42. Springer, Cham.

de la Cruz M. 2008. Metodos para analizar datos puntuales. In: Introduccion al Analisis Espacial de Datos en Ecologia y Ciencias Ambientales: Metodos y Aplicaciones (Eds. FT Maestre, A Escudero & A Bonet), pp. 76–127. Asociacion Espanola de Ecologia Terrestre, Universidad Rey Juan Carlos y Caja de Ahorros del Mediterraneo, Madrid.

de La Peña-Domene M, Martínez-Garza C, Palmas-Pérez S, Rivas-Alonso E & Howe HF. 2014. Roles of birds and bats in early tropical-forest restoration. PLoS ONE, 9: e104656.

de la Sancha NU, Boyle SA & McIntyre NE. 2021. Identifying structural connectivity priorities in eastern Paraguay’s fragmented Atlantic Forest. Scientific Reports 11: 16129.

del Hoyo J. 1994. Cracidae (Chachalacas, Guans and Curassows). In: Handbook of the birds of the world (Eds. J del Hoyo, A Elliott & J Sargatal), pp. 322–476. Lynx Edicions, Barcelona.

Della Rocca F & Milanesi P. 2020. Combining climate, land use change and dispersal to predict the distribution of endangered species with limited vagility. Journal of Biogeography, 47: 1427–1438.

Develey PF. 2021. Bird conservation in Brazil: challenges and practical solutions for a key megadiverse country. Perspectives in Ecology and Conservation, 19: 171–178.

Díaz S, Settele J, Brondízio ES, Ngo HT, Agard J, Arneth A, Balvanera P, Brauman KA, Butchart SHM, Chan KMA, Garibaldi LA, Ichii K, Liu J, Subramanian SM, Midgley GF, Miloslavich P, Molnár Z, Obura D, Pfaff A, Polasky S, Purvis A, Razzaque J, Reyers B, Chowdhury RR, Shin Y-J, Visseren-Hamakers I, Willis KJ & Zayas CN. 2019. Pervasive human-driven decline of life on Earth points to the need for transformative change. Science, 366: eaax3100.

Di Marco M, Buchanan GM, Szantoi Z, Holmgren M, Marasini GG, Gross D, Tranquilli S, Boitani L & Rondinini C. 2014. Drivers of extinction risk in African mammals: the interplay of distribution state, human pressure, conservation response and species biology. Philosophical Transactions of the Royal Society B: Biological Sciences, 369: 20130198.

Diniz MF, Dallmeier F, Gregory T, Martinez V, Saldivar-Bellassai S, Benitez-Stanley MA & Sánchez-Cuervo AM. 2022. Balancing multi-species connectivity and socio-economic factors to connect protected areas in the Paraguayan Atlantic Forest. Landscape and Urban Planning, 222: 104400.

Dirzo R, Young HS, Galetti M, Ceballos G, Isaac NJB & Collen B. 2014. Defaunation in the Anthropocene. Science, 345: 401–406.

Dormann CF, Gruber B & Fruend J. 2008. Introducing the bipartite package: analysing ecological networks. R news, 8: 8–11.

Dunne JA, Williams RJ & Martinez ND. 2002. Network structure and biodiversity loss in food webs: robustness increases with connectance. Ecology Letters, 5: 558–567.

eBird Basic Dataset. 2022. Version: EBD_relMay-2022. Cornell Lab of Ornithology, Ithaca, New York.

Echeverri A, Karp DS, Frishkoff LO, Krishnan J, Naidoo R, Zhao J, Zook J & Chan KMA. 2021. Avian cultural services peak in tropical wet forests. Conservation Letters, 14: e12763.

Emer C, Galetti M, Pizo MA, Jordano P & Verdú M. 2019. Defaunation precipitates the extinction of evolutionarily distinct interactions in the anthropocene. Science Advances, 5: 6699–6718.

Estes JA, Terborgh J, Brashares JS, Power ME, Berger J, Bond WJ, Carpenter SR, Essington

TE, Holt RD, Jackson JBC, Marquis RJ, Oksanen L, Oksanen T, Paine RT, Pikitch EK, Ripple WJ, Sandin SA, Scheffer M, Schoener TW, Shurin JB, Sinclair ARE, Soulé ME, Virtanen R & Wardle DA. 2011. Trophic downgrading of planet Earth. Science, 333: 301–306.

Esquivel A, Zarza R, Tiffer-Sotomayor R, Díaz A, Pérez D & Velázquez M. 2019. Conservation status and challenges of the Atlantic Forest birds of Paraguay. Diversity, 11.

Evans TL & Costa M. 2013. Landcover classification of the Lower Nhecolândia subregion of the Brazilian Pantanal Wetlands using ALOS/PALSAR, RADARSAT-2 and ENVISAT/ASAR imagery. Remote Sensing of Environment, 128: 118–137.

Ewers RM & Banks-Leite C. 2013. Fragmentation Impairs the Microclimate Buffering Effect of Tropical Forests. PLoS ONE, 8: e58093.

Fahrig L. 1997. Relative effects of habitat loss and fragmentation on population extinction. The Journal of Wildlife Management, 61: 603–610.

Faria D, Delabie JHC & Dias MH. 2021. The Hileia Baiana: An Assessment of Natural and Historical Aspects of the Land Use and Degradation of the Central Corridor of the Brazilian Atlantic Forest. In: The Atlantic Forest (Eds. MCM Marques & CEV Grelle), pp. 63–90. Springer, Cham.

Felipe-Lucia MR, Soliveres S, Penone C, Fischer M, Ammer C, Boch S, Boeddinghaus RS, Bonkowski M, Buscot F, Fiore-Donno AM, Frank K, Goldmann K, Gossner MM, Hölzel N, Jochum M, Kandeler E, Klaus VH, Kleinebecker T, Leimer S, Manning P, Oelmann Y, Saiz H, Schall P, Schloter M, Schöning I, Schrumpf M, Solly EF, Stempfhuber B, Weisser WW, Wilcke W, Wubet T & Allan E. 2015. Land use intensification alters ecosystem multifunctionality via loss of biodiversity and changes to functional composition. Ecology Letters, 18: 834–843.

Fendrich AN, Barretto A, de Faria VG, de Bastiani F, Tenneson K, Guedes Pinto LF & Sparovek G. 2020. Disclosing contrasting scenarios for future land cover in Brazil: Results from a high-resolution spatiotemporal model. Science of The Total Environment, 742: 140477.

Ferrante L & Fearnside PM. 2019. Brazil's new president and 'ruralists' threaten Amazonia's environment, traditional peoples and the global climate. Environmental Conservation, 46: 261–263.

Ferro VG, Lemes P, Melo AS & Loyola R. 2014. The reduced effectiveness of protected areas under climate change threatens Atlantic Forest tiger moths. PLoS One, 9: e107792.

Fick SE & Hijmans RJ. 2017. WorldClim 2: new 1-km spatial resolution climate surfaces for global land areas. International Journal of Climatology, 37: 4302–4315.

Fleytas, MC. 2007. Cambios en el paisaje: evolución de la cobertura vegetal en la Región Oriental del Paraguay. In: Biodiversidad del Paraguay: una aproximación a sus realidades (Eds. DA Salas-Dueñas & JF Facetti), pp. 77–87. Fundación Moisés Bertoni, Asunción.

Fonseca CR, Ganade G, Baldissera R, Becker CG, Boelter CR, Brescovit AD, Campos LM, Fleck T, Fonseca VS, Hartz SM, Joner F, Käffer MI, Leal-Zanchet AM, Marcelli MP, Mesquita AS, Mondin CA, Paz CP, Petry MV, Piovensan FN, Putzke J, Stranz A, Vergara M & Vieira EM. 2009. Towards an ecologically-sustainable forestry in the Atlantic Forest. Biological Conservation, 142: 1209–1219.

Francisco MR, Oliveira Lunardi V & Galetti M. 2002. Massive seed predation of *Pseudobombax grandiflorum* (Bombacaceae) by parakeets *Brotogeris versicolurus* (Psittacidae) in a forest fragment in Brazil. Biotropica, 34: 613–615.

Friedl M & Sulla-Menashe D. 2015. MCD12Q1 MODIS/Terra+Aqua Land Cover Type Yearly L3 Global 500m SIN Grid. Boston University and MODAPS SIPS - NASA.

Gaiotti MG, Mascarenhas W & Macedo RH. 2017. The critically endangered and endemic araripe manakin (*Antilophia bokermanni*): dietary assessment for conservation purposes. Wilson Journal of Ornithology, 129: 783–791.

Galetti M & Dirzo R. 2013. Ecological and evolutionary consequences of living in a defaunated world. Biological Conservation, 163: 1–6.

Galetti M & Rodrigues M. 1992. Comparative seed predation on pods by parrots in Brazil. Biotropica, 24: 222–224.

Galetti M, Guevara R, Côrtes MC, Fadini R, Von Matter S, Leite AB, Labecca F, Ribeiro T, Carvalho CS, Collevatti RG, Pires MM, Guimarães PR, Brancalion PH, Ribeiro MC & Jordano P. 2013. Functional extinction of birds drives rapid evolutionary changes in seed size. Science, 340: 1086–1090.

Giorgi AP, Rovzar C, Davis KS, Fuller T, Buermann W, Saatchi S, Smith TB, Silveira LF & Gillespie TW. 2014. Spatial conservation planning framework for assessing conservation opportunities in the Atlantic Forest of Brazil. Applied Geography, 53: 369–376.

Giraudo AR, Matteucci SD, Alonso J, Herrera J & Abramson RR. 2008. Comparing bird assemblages in large and small fragments of the Atlantic Forest hotspots. Biodiversity Conservation, 17: 1251–1265.

Gonçalves F, Sales LP, Galetti M & Pires MM. 2021. Combined impacts of climate and land use change and the future restructuring of Neotropical bat biodiversity. Perspectives in Ecology and Conservation, 19: 454–463.

González-Castro A, Yang S & Carlo TA. 2019. How does avian seed dispersal shape the structure of early successional tropical forests? Functinal Ecology, 33: 229–238.

Gouveia SF, Souza-Alves JP, Rattis L, Dobrovolski R, Jerusalinsky L, Beltrão-Mendes R & Ferrari SF. 2016. Climate and land use changes will degrade the configuration of the landscape for titi monkeys in eastern Brazil. Global Change Biology, 22: 2003–2012.

Grantham HS, Duncan A, Evans TD, Jones KR, Beyer HL, Schuster R, Walston J, Ray JC, Robinson JG, Callow M, Clements T, Costa HM, DeGemmis A, Elsen PR, Ervin J, Franco P, Goldman E, Goetz S, Hansen A, Hofsvang E, Jantz P, Jupiter S, Kang A, Langhammer P, Laurance WF, Lieberman S, Linkie M, Malhi Y, Maxwell S, Mendez M, Mittermeier R, Murray NJ, Possingham H, Radachowsky J, Saatchi S, Samper C, Silverman J, Shapiro A, Strassburg B, Stevens T, Stokes E, Taylor R, Tear T, Tizard R, Venter O, Visconti P, Wang S, Watson JEM. 2020. Anthropogenic modification of forests means only 40% of remaining forests have high ecosystem integrity. Nature Communications, 11: 5978.

Guerra A, Roque, FO, Garcia LC, Ochoa-Quintero JM, Oliveira PTS, Guariento RD & Rosa IMD. 2020. Drivers and projections of vegetation loss in the Pantanal and surrounding ecosystems. Land Use Policy, 91: 104388.

Hannah L, Midgley G, Andelman S, Araújo M, Martinez-Meyer E, Pearson R & Williams P. 2007. Protected area needs in a changing climate. Frontiers in Ecology and the Environment, 5:131–138.

Hansen MC, Potapov PV, Moore R, Hancher M, Turubanova SA, Tyukavina A, Thau D, Stehman SV, Goetz SJ, Loveland TR, Kommareddy A, Egorov A, Chini L, Justice CO & Townshend JRG. 2013. High-Resolution Global Maps of 21st-Century Forest Cover Change. Science, 342: 850–853.

Harrison RD, Tan S, Plotkin JB, Slik F, Detto M, Brenes T, Itoh A & Davies SJ. 2013. Consequences of defaunation for a tropical tree community. Ecology Letters, 16: 687–694.

Hasui É, Metzger JP, Pimentel RG, Silveira LF, Bovo AAA, Martensen AC, Uezu A, Regolin AL, Oliveira AAB, Gatto CAFR, Duca C, Andretti CB, Banks-Leite C, Luz D, Mariz D, Alexandrino ER, Barros FM, Martello F, Pereira IMS, Silva JN, Ferraz KMPMB, Naka

LN, Anjos L, Efe MA, Pizo MA, Pichorim M, Gonçalves MSS, Cordeiro PHC, Dias RA, Muylaert RL, Rodrigues RC, Costa TVV, Cavarzere V, Tonetti VR, Silva WR, Jenkins CN, Galetti M & Ribeiro MC. 2017. ATLANTIC BIRDS: a data set of bird species from the Brazilian Atlantic Forest. Ecology, 99: 497–497.

Hausfather Z. 2019. CMIP6: the next generation of climate models explained. CarbonBrief. Available at: https://www.carbonbrief.org/cmip6-the-next-generation-of-climate-models-explained.

Hijmans RJ, Phillips S, Leathwick J & Elith J. 2017. R package dismo: species distribution modeling. R package version 1.1-4.

Hijmans R. 2023. raster: Geographic Data Analysis and Modeling. R package version 3.6-20.

Hinkelmann C & Schuchmann KL. 1997. Phylogeny of the hermit hummingbirds (Trochilidae: Phaethornithinae). Studies on Neotropical Fauna and Environment, 32: 142–163.

Hirzel AH, Hausser J, Chessel D & Perrin N. 2002. Ecological-niche factor analysis: how to compute habitat-suitability maps without absence data? Ecology, 83: 2027–2036.

Hoang NT & Kanemoto K. 2021. Mapping the deforestation footprint of nations reveals growing threat to tropical forests. Nature Ecology and Evolution, 5: 845–853.

Hof, Araújo, MB, Jetz W & Rahbek C. 2011. Additive threats from pathogens, climate and land-use change for global amphibian diversity. Nature, 480: 516–519.

Hoffmann D, de Vasconcelos MF & Martins RP. 2015. Como as mudanças climáticas podem afetar a área de distribuição e o status de conservação de uma ave endêmica das montanhas do leste do Brazil: O caso do papa-moscas-de-costas-cinzentas, *Polystictus superciliaris* (Aves, Tyrannidae). Biota Neotropica, 15: e20130075.

Hole DG, Huntley B, Arinaitwe J, Butchart SHM, Collingham YC, Fishpool LDC, Pain DJ & Willis SG. 2011. Toward a management framework for networks of protected areas in the face of climate change. Conservation Biology, 25: 305–315.

Hosner PA, Braun EL & Kimball RT. 2016. Rapid and recent diversification of curassows, guans, and chachalacas (Galliformes: Cracidae) out of Mesoamerica: Phylogeny inferred from mitochondrial, intron, and ultraconserved element sequences. Molecular Phylogenetics and Evolution, 102: 320–330.

Hulme PE. 1998. Post-dispersal seed predation: consequences for plant demography and evolution. Perspectives in Plant Ecology, Evolution and Systematics, 1: 32–46.

Hurtt GC, Chini LP, Frolking S, Betts RA, Feddema J, Fischer G, Fisk JP, Hibbard K, Houghton RA, Janetos A, Jones CD, Kindermann G, Kinoshita T, Klein Goldewijk K, Riahi K, Shevliakova E, Smith S, Stehfest E, Thomson A, Thornton P, van Vuuren DP & Wang YP. 2011. Harmonization of land-use scenarios for the period 1500–2100: 600 years of global gridded annual land-use transitions, wood harvest, and resulting secondary lands. Climatic Change, 109: 117–161.

Hutyra LR, Munger JW, Nobre CA, Saleska SR, Vieira SA & Wofsy SC. 2005. Climatic variability and vegetation vulnerability in Amazônia. Geophysical Research Letters, 32: L24712.

iNaturalist. 2021. iNaturalist: A community for naturalists. Available at: https://www.inaturalist.org/ [Accessed 26 November, 2021].

Işik K. 2011. Rare and endemic species: why are they prone to extinction? Turkish Journal of Botany, 35: 411–417.

IUCN. 2012. Guidelines for application of IUCN Red List criteria at regional and national levels: version 4.0. IUCN, Gland, Switzerland and Cambridge, 32 pp.

IUCN. 2022. The IUCN Red List of Threatened Species. Version 2022-2. Available at: https://www.iucnredlist.org [Accessed July 16, 2023].

Janzen DH. 1971. Seed predation by animals. Annual Review of Ecology and Systematics, 2:

465–492.

Jaureguiberry P, Titeux N, Wiemers M, Bowler DE, Coscieme L, Golden AS, Guerra CA, Jacob U, Takahashi Y, Settele J, Díaz S, Molnár Z & Purvis A. 2022. The direct drivers of recent global anthropogenic biodiversity loss. Science Advances, 8: 9982.

Jenkins CN, Alves MAS, Uezu A & Vale MM. 2015. Patterns of vertebrate diversity and protection in Brazil. PLoS One, 10: e0145064.

Jetz W, Wilcove DS & Dobson AP. 2007. Projected impacts of climate and land-use change on the global diversity of birds. PLoS Biology, 5: 1211–1219.

Johnson CN, Balmford A, Brook BW, Buettel JC, Galetti M, Guangchun L & Wilmshurst JM. 2017. Biodiversity losses and conservation responses in the anthropocene. Science, 356: 270–275.

Joly CA, Metzger JP & Tabarelli M. 2014. Experiences from the Brazilian Atlantic Forest : ecological findings and conservation initiatives. New Phytologist, 204: 459–473.

Junk WJ, Da Cunha CN, Wantzen KM, Petermann P, Strüssmann C, Marques MI & Adis J. 2006. Biodiversity and its conservation in the Pantanal of Mato Grosso, Brazil. Aquatic Sciences, 68: 278–309.

Kaplan JO, Krumhardt KM & Zimmermann NE. 2012. The effects of land use and climate change on the carbon cycle of Europe over the past 500 years. Global Change Biology, 18: 902–914.

Karatzoglou A, Smola A, Hornik K & Zeileis A. 2004. kernlab – An S4 Package for Kernel Methods in R. Journal of Statistical Software, 11: 1–20.

Keyes AA, McLaughlin JP, Barner AK & Dee LE. 2021. An ecological network approach to predict ecosystem service vulnerability to species losses. Nature Communications, 12: 1586.

Khaliq I, Hof C, Prinzinger R, Böhning-Gaese K & Pfenninger M. 2014. Global variation in thermal tolerances and vulnerability of endotherms to climate change. Proceedings of the Royal Society B: Biological Sciences, 281: 20141097.

Kilpp JC, Prestes NP, Dal Pizzol GE & Martinez J. 2015. Dieta alimentar de *Amazona vinacea* no sul e sudeste de Santa Catarina, Brasil. Atualidades Ornitológicas, 183: 9–13.

Kitamura S, Yumoto T, Poonswad P, Chuailua P, Plongmai K, Maruhashi T & Noma N. 2002. Interactions between fleshy fruits and frugivores in a tropical seasonal forest in thailand. Oecologia, 133: 559–572.

Koo KA & Park SU. 2022. The effect of interplays among climate change, land-use change, and dispersal capacity on plant redistribution. Ecological Indicators, 142: 109192.

Kronenberg, J. 2014. Environmental impacts of the use of ecosystem services: case study of birdwatching. Environmental Management, 54: 617–630.

Kurten EL. 2013. Cascading effects of contemporaneous defaunation on tropical forest communities. Biological Conservation, 163: 22–32.

Lafferty KD. 2009. The ecology of climate change and infectious diseases. Concepts & Synthesis, 90: 888–900.

Lambin EF, Geist HJ & Lepers E. 2003. Dynamics of land-use and land-cover change in tropical regions. Annual Review of Environment and Resources, 28: 205–241.

Lees AC & Peres CA. 2009. Gap-crossing movements predict species occupancy in Amazonian forest fragments. Oikos, 118: 280–290.

Lewis SL, Edwards DP & Galbraith D. 2015. Increasing human dominance of tropical forests. Science, 349: 827–832.

Liaw A & Wiener M. 2002. Classification and Regression by randomForest. R News, 2: 18-22.

Linhart YB. 2017. Ecological and behavioral determinants of pollen dispersal in hummingbird-pollinated *Heliconia*. The American Naturalist, 107: 511–523.

Lira PK, Portela RCQ & Tambosi LR. 2021. Land-cover changes and an uncertain future: will the Brazilian Atlantic Forest lose the chance to become a Hopespot? In: The Atlantic Forest (Eds. MCM Marques & CEV Grelle), pp. 233–251. Springer, Cham.

Liu VJ & Lyons JA. 2012. Notes on the feeding activity and diet of the Blue-fronted Parrotlet (*Touit dilectissima*) in north-west Ecuador. Boletín SAO, 21: evnc4_2012.

Loiselle BA, Graham CH, Goerck JM. & Ribeiro MC. 2010. Assessing the impact of deforestation and climate change on the range size and environmental niche of bird species in the Atlantic forests, Brazil. Journal of Biogeography, 37: 1288–1301.

Lowen JC, Clay RP, Brooks TM, Esquivel EZ, Bartrina L, Barnes R, Butchart SHM & Etcheverry NI. 1995. Bird conservation in the Paraguayan Atlantic Forest. Cotinga, 4: 58–64.

Loyola RD, Lemes P, Nabout JC, Trindade-Filho J, Sagnori MD, Dobrovolski R & Diniz-Filho JAF. 2013. A straightforward conceptual approach for evaluating spatial conservation priorities under climate change. Biodiversity and Conservation, 22:483–495.

Manes S, Costello MJ, Beckett H, Debnath A, Devenish-Nelson E, Grey KA, Jenkins R, Khan TM, Kiessling W, Krause C, Maharaj SS, Midgley GF, Price J, Talukdar G & Vale, MM. 2021. Endemism increases species' climate change risk in areas of global biodiversity importance. Biological Conservation, 257: 109070.

Mantyka-Pringle CS, Visconti P, Di Marco M, Martin TG, Rondinini C & Rhodes JR. 2015. Climate change modifies risk of global biodiversity loss due to land-cover change. Biological Conservation, 187: 103–111.

MapBiomas. 2022. Projeto MapBiomas – Coleção 7.1 da Série Anual de Mapas de Uso e Cobertura da Terra do Brasil. Available at: https://mapbiomas.org/ [Accessed July 17, 2023].

MapBiomas. 2023. Relatório Anual do Desmatamento no Brasil 2022. MapBiomas, São Paulo, 125 pp.

Marengo JA, Oliveira GS & Alves LM. 2015. Climate Change Scenarios in the Pantanal. In: Dynamics of the Pantanal Wetland in South America (Eds. I Bergier & ML Assine), pp. 227–238. Springer, Cham.

Marengo J, Alves L & Torres R. 2016. Regional climate change scenarios in the Brazilian Pantanal watershed. Climate Research, 68: 201–213.

Marini MÂ, Barbet-Massin M, Martinez J, Prestes NP & Jiguet F. 2010. Applying ecological niche modelling to plan conservation actions for the Red-spectacled Amazon (*Amazona pretrei*). Biological Conservation, 143: 102–112.

Marques MCM & Grelle CEV. 2021. The Atlantic Forest: History, Biodiversity, Threats and Opportunities of the Mega-diverse Forest. Springer, Cham, 517 pp.

Marques MCM, Trindade W, Bohn A & Grelle CEV. 2021. The Atlantic Forest: an introduction to the megadiverse forest of South America. In: The Atlantic Forest (Eds. MCM Marques & CEV Grelle), pp. 3–23. Springer, Cham.

May RM. 1973. Stability and complexity in model ecosystems. Princeton University Press, New Jersey, 235 pp.

McCallum H. 2012. Disease and the dynamics of extinction. Philosophical Transactions of the Royal Society B: Biological Sciences, 367: 2828–2839.

Michel NL, Whelan CJ & Verutes GM. 2020. Ecosystem services provided by Neotropical birds. Condor, 122: 1–21.

Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-being: Synthesis. Island Press, Washington DC, 137 pp.

Ministério do Meio Ambiente. 2022. PORTARIA MMA No 148, DE 7 DE JUNHO DE 2022. Altera os Anexos da Portaria no 443, de 17 de dezembro de 2014, da Portaria no 444, de

17 de dezembro de 2014, e da Portaria no 445, de 17 de dezembro de 2014, referentes à atualização da Lista Nacional de Espécies Ameaçadas de Extinção. Diário Oficial da União, Brasília.

Ministério do Meio Ambiente. 2023. Plataforma oficial de dados do Sistema Nacional de Unidades de Conservação da Natureza. Available at: https://cnuc.mma.gov.br/ [Accessed June 16, 2023].

Mittermeier RA, Robles-Gil P, Hoffman M, Pilgrim J, Brooks T, Mittermeier CG, Lamoreux J & Fonseca GAB. 2005. Hotspots revisited: Earth’s biologically richest and most endangered terrestrial ecoregions. CEMEX, Mexico City, 392 pp.

Mohebalian PM, Lopez LN, Tischner AB & Aguilar FX. 2022. Deforestation in South America’s tri-national Paraná Atlantic Forest: Trends and associational factors. Forest Policy and Economics, 137: 102697.

Moreira-Lima L. 2013. Aves da Mata Atlântica: riqueza, composição, status, endemismos e conservação. Dissertation. Instituto de Biociências da Universidade de São Paulo, São Paulo, 513 pp.

Morris RJ. 2010. Anthropogenic impacts on tropical forest biodiversity: a network structure and ecosystem functioning perspective. Philosophical Transactions of the Royal Society B: Biological Sciences, 365: 3709–3718.

Mota FMM, Heming NM, Morante-Filho JC & Talora DC. 2022. Climate change is expected to restructure forest frugivorous bird communities in a biodiversity hot-point within the Atlantic Forest. Diversity and Distributions, 28: 2886–2897.

Muylaert RL, Vancine MH, Bernardo R, Oshima JEF, Sobral-Souza T, Tonetti VR, Niebuhr BB & Ribeiro MC. 2018. A note on the territorial limits of the Atlantic forest. Oecologia Australis, 22: 302–311.

Newbold T. 2018. Future effects of climate and land-use change on terrestrial vertebrate community diversity under different scenarios. Proceedings of the Royal Society B: Biological Sciences, 285: 20180792.

Newbold T, Scharlemann JPW, Butchart SHM, Şekercioğlu ÇH, Alkemade R, Booth H & Purves DW. 2013. Ecological traits affect the response of tropical forest bird species to land-use intensity. Proceedings of the Royal Society B: Biological Sciences, 280: 20122131.

Nunes AP, de Vasconcelos MF, Hoffmann D, Souza LN, Gomes CRG, Epifânio AD, Godoi MN, Tizianel FAT, Straube FC, Silva PA, Posso SR, Laps RR, de Faria SP & Tomas WM. 2018. Aves da borda oeste do Pantanal, Mato Grosso do Sul, Brasil. Atualidades Ornitológicas, 206: 47–69.

O'Connor F. 2019. MOHC UKESM1.0-LL model output prepared for CMIP6 AerChemMIP. Earth System Grid Federation.

Oliveira-Silva AE, Piratelli AJ, Zurell D & da Silva FR. 2022. Vegetation cover restricts habitat suitability predictions of endemic Brazilian Atlantic Forest birds. Perspectives in Ecology and Conservation, 20: 1–8.

Ollerton J, Winfree R & Tarrant S. 2011. How many flowering plants are pollinated by animals? Oikos, 120: 321–326.

Olson DM, Dinerstein E, Wikramanayake ED, Burgess ND, Powell GVN, Underwood EC, D'Amico JA, Itoua I, Strand HE, Morrison JC, Loucks CJ, Allnutt TF, Ricketts TH, Kura Y, Lamoreux JF, Wettengel WW, Hedao P & Kassem KR. 2011. Terrestrial ecoregions of the world: a new map of life on earth: a new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. Bioscience, 51: 933–938.

O'Neill BC, Tebaldi C, Van Vuuren DP, Eyring V, Friedlingstein P, Hurtt G, Knutti R, Kriegler E, Lamarque JF, Lowe J, Meehl GA, Moss R, Riahi K & Sanderson BM. 2016.

The Scenario Model Intercomparison Project (ScenarioMIP) for CMIP6. Geoscientific Model Developmet, 9: 3461–3482.

Osuri AM, Mendiratta U, Naniwadekar R, Varma V & Naeem S. 2020. Hunting and forest modification have distinct defaunation impacts on tropical mammals and birds. Frontiers in Forests and Global Change, 2: 87.

Pacheco AA, Neves ACO & Fernandes GW. 2018. Uneven conservation efforts compromise Brazil to meet the Target 11 of Convention on Biological Diversity. Perspectives in Ecology and Conservation, 16: 43–48.

Pacto pela Restauração da Mata Atlântica. 2023. Sobre o Pacto. Available at: https://www.iucnredlist.org [Accessed October 23, 2023].

Palmeirim AF, Santos-Filho M & Peres CA. 2020. Marked decline in forest-dependent small mammals following habitat loss and fragmentation in an Amazonian deforestation frontier. PLoS ONE, 15: e0230209.

Paraguay. 2001. Resolución nº 200/01. Por la cual se asignan y reglamentan las categorias de manejo; la zonificación y los usos y actividades. Secretaria del Ambiente, Asunción.

Paraguay. 2004. Ley no. 2524 de prohibición en la región Oriental de las actividades de transformación y conversión de superficies con cobertura de bosques. Republica del Paraguay, Asunción.

Pardini R, Bueno AA, Gardner TA, Prado PI & Metzger JP. 2010. Beyond the fragmentation threshold hypothesis: regime shifts in biodiversity across fragmented landscapes. PLoS ONE, 5: e13666.

Pascual M & Dunne JA. 2006. Ecological networks: linking structure to dynamics in food webs. Oxford University Press, Oxford, 386 pp.

Pereira RAS. 2006. Interações antagonísticas de figueiras e psitacídeos. Natureza on line, 4: 25.

Pereira HM, Navarro LM, Martins IS. 2012. Global Biodiversity Change: The Bad, the Good, and the Unknown. Annual Review of Environment and Resources, 37: 25–50.

Pereira GA, Araújo HFP & Azevedo-Júnior SM. 2016. Distribuição e conservação de três importantes grupos de aves na Floresta Atlântica do Nordeste do Brazil. Brazilian Journal of Biology, 76: 1004–1020.

Pereira GA, Dantas SM, Periquito MC, Coelho G, Harrop R, Souza JL, Tenório A, Victor L-S & Raia Y. 2019. Threats to, and conservation of, birds in Mata de Aldeia, north-east Atlantic Forest, Brazil. Cotinga, 41: 12–21.

Pereira AD, Bogoni JA, Bazilio S & Orsi ML. 2021. Mammalian defaunation across the devonian kniferidges and meridional plateaus of the brazilian atlantic forest. Biodiversity and Conservation, 30: 4005–4022.

Peterson AT, Sánchez-Cordero V, Soberón J, Bartley J, Buddemeier RW & Navarro-Sigüenza AG. 2001. Effects of global climate change on geographic distributions of Mexican Cracidae. Ecological Modelling, 144: 21–30.

Peterson AT, Soberón J, Pearson RG, Anderson RP, Martínez-Meyer E, Nakamura M. & Araújo MB. 2011. Ecological Niches and Geographic Distributions. Princeton University Press, New Jersey, 314 pp.

Phillips SJ & Dudík M. 2008. Modeling of species distributions with Maxent: New extensions and a comprehensive evaluation. Ecography, 31: 161–175.

Pimm SL & Raven P. 2000. Extinction by numbers. Nature, 403: 843–845.

Pinto Leite CM, Mariano-Neto E & Rocha PLDb. 2018. Biodiversity thresholds in invertebrate communities: The responses of dung beetle subgroups to forest loss. PLoS ONE, 13: e0201368.

Pinto SR, Melo F, Tabarelli M, Padovesi A, Mesquita CA, de Mattos Scaramuzza CA, Castro P, Carrascosa H, Calmon M, Rodrigues R, César R. G. & Brancalion PHS. 2014.

Governing and delivering a biome-wide restoration initiative: The case of Atlantic Forest Restoration Pact in Brazil. Forests, 5: 2212–2229.

Pivatto MAC, Manço DDG, Straube FC, Urben-filho A & Milano M. 2006. Aves do Planalto da Bodoquena, Estado do Mato Grosso do Sul (Brasil). Atualidades Ornitológicas, 129.

Pizo MA & Vieira EM. 2004. Granivorous birds as potentially important post-dispersal seed predators in a Brazilian forest fragment. Biotropica, 36: 417–423.

Prado LC, Dias TC, Araújo LWL, Silveira LF & Francisco MR. 2022. Population density estimates and key microhabitat parameters for two endangered tropical forest understory insectivorous passerines from the pernambuco endemism center. Avian Conservation and Ecology, 17: 22.

Prieto-Torres DA, Sánchez-González LA, Ortiz-Ramírez MF, Ramírez-Albores JE, García-Trejo EA & Navarro-Sigüenza AG. 2021. Climate warming affects spatio-temporal biodiversity patterns of a highly vulnerable neotropical avifauna. Climatic Change, 165: 57.

Püttker T, Bueno AA, de Barros CS, Sommer S & Pardini R. 2011. Immigration rates in fragmented landscapes – empirical evidence for the importance of habitat amount for species persistence. PLoS ONE, 6: e27963.

QGIS Development Team. 2021. QGIS Geographic Information System. Version 3.16.3. QGIS Association. Available at: http://qgis.osgeo.org

QGIS Development Team. 2022. QGIS Geographic Information System. Version 3.22. QGIS Association. Available at: http://qgis.osgeo.org

R Core Team. 2020. R: A Language and Environment for Statistical Computing, version 4.0.2. R Foundation for Statistical Computing, Vienna, Austria. Available at: https://www.r-project.org/

R Core Team. 2023. R: A Language and Environment for Statistical Computing, version

4.3.0. R Foundation for Statistical Computing, Vienna, Austria. Available at: https://www.r-project.org/

Ramalho Q, Tourinho L, Almeida-Gomes M, Vale MM & Prevedello JA. 2021. Reforestation can compensate negative effects of climate change on amphibians. Biological Conservation, 260: 109187.

Ramos RG, Scarabello M, Soterroni AC, Andrade PR, Simões R, Ruivo HM, Kraxner F & Ramos FM. 2023. Current policies are insufficient to protect or restore Brazil’s cost-effective conservation priority zones. Environmental Research Letters, 18: 065006.

Reside AE, VanDerWal J & Kutt AS. 2012. Projected changes in distributions of Australian tropical savanna birds under climate change using three dispersal scenarios. Ecology and Evolution, 2: 705–718.

Riahi K, van Vuuren DP, Kriegler E, Edmonds J, O'Neill BC, Fujimori S, Bauer N, Calvin K, Dellink R, Fricko O, Lutz W, Popp A, Cuaresma JC, KC S, Leimbach M, Jiang L, Kram T, Rao S, Emmerling J, Ebi K, Hasegawa T, Havlik P, Humpenöder F, Da Silva LA, Smith S, Stehfest E, Bosetti V, Eom J, Gernaat D, Masui T, Rogelj J, Strefler J, Drouet L, Krey V, Luderer G, Harmsen M, Takahashi K, Baumstark L, Doelman JC, Kainuma M, Klimont Z, Marangoni G, Lotze-Campen H, Obersteiner M, Tabeau A & Tavoni M. 2017. The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview. Global Environmental Change, 42: 153–168.

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Strassburg BBN, Tabarelli M, Fonseca GA & Mittermeier RA. 2018. From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspectives in Ecology and Conservation, 16: 208–214.

Rezende GC, Sobral-Souza T & Culot L. 2020. Integrating climate and landscape models to prioritize areas and conservation strategies for an endangered arboreal primate. American Journal of Primatology, 82: e23202.

Ribeiro MC, Metzger JP, Martensen AC, Ponzoni FJ & Hirota MM. 2009. The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biological Conservation, 142: 1141–1153.

Ridgely R. 2010. Mutum-de-penacho *Crax fasciolata*. In: Aves do Brasil: Pantanal & Cerrado (Eds. JA Gwynne, RS Ridgely, T Guy & M Argel), p. 36. Editora Horizonte, São Paulo.

Rios E, McGowan, PJK, Collar NJ, Benchimol M, Canale GR, Olmos F, Santos-Filho M & Bernardo CSS. 2021. Which is worse for the red-billed curassow: habitat loss or hunting pressure? Oryx, 55: 412–420.

Rodrigues RR, Henrique P, Brancalion S & Isernhagen I. 2009. Pacto pela restauração da Mata Atlântica: referencial dos conceitos e ações de restauração florestal. LERF/ESALQ: Instituto BioAtlântica, São Paulo, 256 pp.

Roesch L, Vieira F, Pereira VA, Schunemann A, Teixeira I, Senna A & Stefenon V. 2009. The Brazilian Pampa: a fragile biome. Diversity, 1: 182–198.

Rosa MR, Brancalion PHS, Crouzeilles R, Tambosi LR, Piffer PR, Lenti FEB, Hirota M, Santiami E & Metzger, JP. 2021. Hidden destruction of older forests threatens Brazil's Atlantic Forest and challenges restoration programs. Science Advances, 7: eabc4547.

Ruuskanen S, Hsu BY & Nord A. 2021. Endocrinology of thermoregulation in birds in a changing climate. Molecular and Cellular Endocrinology, 519: 111088.

Sáenz-Jiménez F, Rojas-Soto O, Pérez-Torres JAIRO, Martínez-Meyer E & Sheppard JK. 2020. Effects of climate change and human influence in the distribution and range overlap between two widely distributed avian scavengers. Bird Conservation International, 31: 77–95.

Sala OE, Chapin FS, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, Huber-Sanwald E, Huenneke LF, Jackson RB, Kinzig A, Leemans R, Lodge DM, Mooney HA, Oesterheld M, Pof NLR, Sykes MT, Walker BH, Walker M & Wall DH. 2000. Global biodiversity scenarios for the year 2100. Science, 287: 1770–1774.

Salazar LF & Nobre CA. 2010. Climate change and thresholds of biome shifts in Amazonia. Geophysical Research Letters, 37: L17706.

Salazar LF, Nobre CA & Oyama MD. 2007. Climate change consequences on the biome distribution in tropical South America. Geophysical Research Letters, 34: L09708.

Sales LP, Galetti M & Pires MM. 2020. Climate and land-use change will lead to a faunal "savannization" on tropical rainforests. Global Change Biology, 26: 7036–7044.

Sansevero JBB, Garbin ML, Sánchez-Tapia A, Valladares F & Scarano FR. 2020. Fire drives abandoned pastures to a savanna-like state in the Brazilian Atlantic Forest. Perspectives in Ecology and Conservation, 18: 31–36.

Santos JC, Leal IR, Almeida-Cortez JS, Fernandes GW & Tabarelli M. 2011. Caatinga: the scientific negligence experienced by a dry tropical forest. Tropical Conservation Science, 4: 276–286.

Santos JP, Sobral-Souza T, Brown KS, Vancine MH, Ribeiro MC & Freitas AVL. 2020. Effects of landscape modification on species richness patterns of fruit-feeding butterflies in Brazilian Atlantic Forest. Diversity and Distributions, 26: 196–208.

Sayer JA & Whitmore TC. 1991. Tropical Moist Forests: Destruction and species extinction. Biological Conservation, 55: 199–213.

Scarano FR & Ceotto P. 2015. Brazilian Atlantic forest: impact, vulnerability, and adaptation to climate change. Biodiversity and Conservation, 24: 2319–2331.

Schloss CA, Nuñez TA & Lawler JJ. 2012. Dispersal will limit ability of mammals to track climate change in the Western Hemisphere. Proceedings of the National Academy of Sciences of the United States of America, 109: 8606–8611.

Şekercioğlu ÇH. 2002. Impacts of birdwatching on human and avian communities. Environmental Conservation, 29: 282–289.

Şekercioğlu ÇH. 2006. Increasing awareness of avian ecological function. Trends in Ecology and Evolution, 21: 464–471.

Şekercioğlu ÇH, Schneider SH, Fay JP & Loarie SR. 2008. Climate change, elevational range shifts, and bird extinctions. Conservation Biology, 22: 140–150.

Şekercioğlu ÇH, Primack RB & Wormworth J. 2012. The effects of climate change on tropical birds. Biological Conservation, 148: 1–18.

SiBBr. 2021. Sistema da Informação sobre a Biodiversidade Brasileira (SiBBr). Available at: https://sibbr.gov.br/ [Accessed November 26, 2021].

Sillero N, Arenas-Castro S, Enriquez-Urzelai U, Vale CG, Sousa-Guedes D, Martínez-Freiría F, Real R & Barbosa AM. 2021. Want to model a species niche? A step-by-step guideline on correlative ecological niche modelling. Ecological Modelling, 456: 109671.

Silva JF, Fariñas MR, Felfili JM & Klink CA. 2006. Spatial heterogeneity, land use and conservation in the cerrado region of Brazil. Journal of Biogeography, 33: 536–548.

Silva MPP, Kamino LHY & Pôrto KC. 2014. Is the current network system of protected areas in the Atlantic Forest effective in conserving key species of bryophytes? Tropical Conservation Science, 7: 61–74.

Silva JMC & Barbosa LCF. 2017. Impact of human activities on the Caatinga. In: Caatinga: The Largest Tropical Dry Forest Region in South America (Eds. JMC Silva, IR Leal & M Tabarelli), pp. 359–368. Springer, Cham.

Simberloff DS. 1984. Mass extinction and the destruction of moist tropical forests. Zhurnal Obshchei Biologii, 45: 767–778.

Sistema Federal de Áreas Protegidas (SiFAP). 2023. Áreas Protegidas. Ministério de Ambiente y Desarrollo Sostenible de Argentina, Administración de Parques Nacionales de Argentina, Consejo Federal de Medio Ambiente de Argentina. Available at: https://sifap.gob.ar/areas-protegidas [Accessed November 08, 2023].

Sobral-Souza T, Santos JP, Maldaner ME, Lima-Ribeiro MS & Ribeiro MC. 2021. EcoLand: A multiscale niche modelling framework to improve predictions on biodiversity and conservation. Perspectives in Ecology and Conservation, 19: 362–368.

Song XP, Hansen MC, Stehman SV, Potapov PV, Tyukavina A, Vermote EF & Townshend JR. 2018. Global land change from 1982 to 2016. Nature, 560: 639–643.

Soterroni AC, Mosnier A, Carvalho AXY, Câmara G, Obersteiner M, Andrade PR, Souza RC, Brock R, Pirker J, Kraxner F, Havlík P, Kapos V, Jzu Ermgassen EKH, Valin H & Ramos FM. 2018. Future environmental and agricultural impacts of Brazil's Forest Code. Environmental Research Letters, 13: 074021.

Souza TV, Lorini ML, Alves MAS, Cordeiro P & Vale MM. 2011. Redistribution of threatened and endemic atlantic forest birds under climate change. Natureza e Conservação, 9: 214–218.

Souza EB & Manzi AO. 2014. Mudanças Ambientais de Curto e Longo prazo: Projeções, Reversibilidade e Atribuição. In: Base científica das mudanças climáticas. Volume 1- Primeiro Relatório de Avaliação Nacional (Eds. M Ambrizzi & T Araujo), pp. 320–351. Painel Brasileiro de Mudanças Climáticas, Rio de Janeiro.

speciesLink. 2021. speciesLink: Sistema de Informação Distribuído para Coleções Biológicas. Available at: https://specieslink.net/ [Accessed November 26, 2021].

Stiles FG. 1975. Ecology, flowering phenology, and hummingbird pollination of some Costa Rican *Heliconia* species. Ecology 56: 285–301.

Stockwell DRB & Peterson AT. 2002. Effects of sample size on accuracy of species distribution models. Ecological Modelling, 148: 1–13.

Stouffer DB & Bascompte J. 2011. Compartmentalization increases food-web persistence. Proceedings of the National Academy of Sciences, 108: 3648–3652.

Strassburg BBN, Barros FSM, Crouzeilles R, Iribarrem A, dos Santos JS, Silva D, Sansevero

JBB, Alves-Pinto HN, Feltran-Barbieri R & Latawiec AE. 2016. The role of natural regeneration to ecosystem services provision and habitat availability: a case study in the Brazilian Atlantic Forest. Biotropica, 48: 890–899.

Strassburg BBN, Iribarrem A, Beyer HL, Cordeiro CL, Crouzeilles R, Jakovac CC, Braga Junqueira A, Lacerda E, Latawiec AE, Balmford A, Brooks TM, Butchart SHM, Chazdon RL, Erb KH, Brancalion P, Buchanan G, Cooper D, Díaz S, Donald PF, Kapos V, Leclère D, Miles L, Obersteiner M, Plutzar C, Scaramuzza CAM, Scarano FR, Visconti P. 2020. Global priority areas for ecosystem restoration. Nature, 586: 724–729.

Szabo JK, Khwaja N, Garnett ST & Butchart SHM. 2012. Global patterns and drivers of avian extinctions at the species and subspecies level. PLoS ONE, 7: e47080.

Takemura, T. 2019. MIROC MIROC6 model output prepared for CMIP6 AerChemMIP. Earth System Grid Federation.

Tambosi LR, Martensen AC, Ribeiro MC & Metzger JP. 2014. A framework to optimize biodiversity restoration efforts based on habitat amount and landscape connectivity. Restoration Ecology, 22: 169–177.

Tella JL, Dénes FV, Zulian V, Prestes NP, Martínez J, Blanco G & Hiraldo F. 2016. Endangered plant-parrot mutualisms: seed tolerance to predation makes parrots pervasive dispersers of the Parana pine. Scientific Reports, 6: 31709.

Tewksbury JJ, Huey RB & Deutsch CA. 2008. Ecology: Putting the heat on tropical animals. Science, 320: 1296–1297.

Thébault E. 2013. Identifying compartments in presence-absence matrices and bipartite networks: Insights into modularity measures. Journal of Biogeography, 40: 759–768.

Thébault E & Fontaine C. 2010. Stability of ecological communities and the architecture of mutualistic and trophic networks. Science, 329: 853–856.

Thompson PL, Rayfield B & Gonzalez A. 2017. Loss of habitat and connectivity erodes species diversity, ecosystem functioning, and stability in metacommunity networks. Ecography, 40: 98–108.

Thornton DH, Branch LC & Sunquist ME. 2012. Response of large galliforms and tinamous (Cracidae, Phasianidae, Tinamidae) to habitat loss and fragmentation in northern Guatemala. Oryx, 46: 567–576.

Thuiller W, Georges D, Engler R & Breiner F. 2020. biomod2: Ensemble Platform for Species Distribution Modeling. Available at: https://cran.r-project.org/package=biomod2

Tobias JA, Şekercioğlu ÇH & Vargas FH. 2013. Bird conservation in tropical ecosystems: challenges and opportunities. In: Key Topics in Conservation Biology 2, vol. 2 (Eds. DW Macdonald & KJ Willis), pp. 258–276. John Wiley & Sons.

Tobias JA et al. 2022. Avonet: morphological, ecological and geographical data for all birds. Ecology Letters, 25: 581–597.

Torezan-Silingardi HM, Silberbauer-Gottsberger I & Gottsberger G. 2021. Pollination ecology: natural history, perspectives and future directions. In: Plant-animal interactions: source of biodiversity (Eds. K Del-claro & HM Torezan-Silingardi), pp. 119–174. Ppringer, Cham.

Tourinho LJ, Prevedello A, Carvalho BM, Rocha DSB & Vale MM. 2022. Macroscale climate change predictions have little influence on landscape-scale habitat suitability. Perspectives in Ecology and Conservation, 20: 29–37.

Trancoso R. 2021. Changing Amazon deforestation patterns: urgent need to restore command and control policies and market interventions. Environmental Research Letters, 16: 041004.

Travis JMJ. 2003. Climate change and habitat destruction: A deadly anthropogenic cocktail. Proceedings of the Royal Society B: Biological Sciences, 270: 467–473.

Uezu A & Metzger JP. 2011. Vanishing bird species in the Atlantic Forest: relative importance of landscape configuration, forest structure and species characteristics. Biodiversity Conservation, 20: 3627–3643.

Uezu A & Metzger JP. 2016. Time-lag in responses of birds to Atlantic Forest fragmentation: restoration opportunity and urgency. PLoS ONE, 11: e0147909.

UNEP-WCMC & IUCN. 2023. Protected Planet: The World Database on Protected Areas (WDPA) and World Database on Other Effective Area-based Conservation Measures (WD-OECM). Available at: www.protectedplanet.net [Accessed June 16, 2023].

Vale MM, Souza TV, Alves MAS & Crouzeilles R. 2018a. Planning protected areas network that are relevant today and under future climate change is possible: The case of Atlantic Forest endemic birds. PeerJ, 6: e4689.

Vale MM, Tourinho L, Lorini ML, Rajão H & Figueiredo MSL. 2018b. Endemic birds of the Atlantic Forest: traits, conservation status, and patterns of biodiversity. Journal of Field Ornithology, 89: 193–206.

Vale MM, Lima-Ribeiro MS & Rocha TC. 2021. Global land-use and land-cover data: historical, current and future scenarios. Biodiersity Informatics, 16: 28–38.

van der Putten WH, de Ruiter PC, Martijn Bezemer T, Harvey JA, Wassen M & Wolters V. 2004. Trophic interactions in a changing world. Basic and Applied Ecology, 6: 487–494.

Varela S, Anderson RP, García-Valdés R & Fernández-González F. 2014. Environmental filters reduce the effects of sampling bias and improve predictions of ecological niche models. Ecography, 37: 1084–1091.

Vasconcelos MF & D’Angelo Neto S. 2007. Padrões de distribuição e conservação da avifauna na região central da Cadeia do Espinhaço e áreas adjacentes, Minas Gerais, Brasil. Cotinga, 28: 27–44.

Vasconcelos MF, Hoffmann D, Araújo MC & Vasconcelos PN. 2015. Bird-termite interactions in Brazil: a review with perspectives for future studies. Biota Neotropica, 15:

e20140035.

Venter O, Sanderson EW, Magrach A, Allan JR, Beher J, Jones KR, Possingham HP, Laurance WF, Wood P, Fekete BM, Levy MA & Watson JEM. 2016. Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. Nature Communications, 7: 12558.

VertNet. 2016. VertNet: Distributed databases with backbone. Version 2016-09-29. Available at: http://www.vertnet.org/ [Accessed November 11, 2021].

Wei T & Simko V. 2017. R package "corrplot": Visualization of a Correlation Matrix (Version 0.84). Available at: https://github.com/taiyun/corrplot

WikiAves. 2023. WikiAves, a enciclopédia das aves do Brasil. Available at: http://www.wikiaves.com.br/ [Accessed July 22, 2023].

Williams BA, Venter O, Allan JR, Atkinson SC, Rehbein JA, Ward M, Di Marco M, Grantham HS, Ervin J, Goetz SJ, Hansen AJ, Jantz P, Pillay R, Rodríguez-Buriticá S, Supples C, Virnig ALS & Watson JEM. 2020. Change in terrestrial human footprint drives continued loss of intact ecosystems. One Earth, 3: 371–382.

Williams DR, Rondinini C & Tilman D. 2022. Global protected areas seem insufficient to safeguard half of the world's mammals from human-induced extinction. Proceedings of the National Academy of Sciences, 119: 2200118119.

Wilman H, Belmaker J, Simpson J, de la Rosa C, Rivadeneira MM & Jetz W. 2014. Eltontraits 1.0: species-level foraging attributes of the world's birds and mammals. Ecology, 95: 2027.

Whelan CJ, Wenny DG & Marquis RJ. 2008. Ecosystem services provided by birds. Annals of the New York Academy of Sciences, 1134: 25–60.

WorldClim. 2020. Global climate and weather data. Available at: https://www.worldclim.org/data/index.html

Wunderle JM. 1997. The role of animal seed dispersal in accelerating native forest regeneration on degraded tropical lands. Forestry Ecology and Management, 99: 223–235.

York R, Rosa EA & Dietz T. 2003. Footprints on the Earth: The Environmental Consequences of Modernity. American Sociological Review, 68: 279–300.

Young CEF & de Castro BS. 2021. Financing Conservation in the Brazilian Atlantic Forest. In: The Atlantic Forest (Eds. MCM Marques & CEV Grelle), pp. 451–468. Springer, Cham.

Young HS, McCauley DJ, Galetti M & Dirzo R. 2016. Patterns, causes, and consequences of anthropocene defaunation. Annual Review of Ecology, Evolution and Systematics, 47: 333–358.

Zeppetello LRV, Parsons LA, Spector JT, Naylor RL, Battisti DS, Masuda YJ & Wolff NH. 2020. Large scale tropical deforestation drives extreme warming. Environmental Research Letters, 15: 084012.

Zwiener VP, Padial AA, Marques MCM, Faleiro FV, Loyola R & Peterson AT. 2017. Planning for conservation and restoration under climate and land use change in the Brazilian Atlantic Forest. Diversity and Distributions, 23: 955–966.