2.º CICLO

Effects of historical land-use and land-cover change on the potential distribution of eastern Brazil mountaintop endemic species

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Todas as correções determinadas pelo júri, e só essas, foram efetuadas.

O Presidente do Júri,

Porto, ______/______/_________

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ABSTRACT

Land-use and land-cover (LULC) change, directly associated with habitat loss and degradation, is pointed to as the leading global driver of biodiversity decline. Terrestrial hotspots of endemic species often have faced greater habitat loss and degradation pressures than other terrestrial regions. This is the case of Espinhaço Range (ER), located in eastern Brazil and partially designated as UNESCO's Biosphere Reserve in 2005, owing to its ecological, geomorphological and cultural relevance. Additionally, endemic mountaintop species are expected to experience more intense habitat loss and degradation impacts since they are less tolerant to rapid environmental changes.

In this study – by integrating historical LULC data and ecological niche modelling – we aimed to unravel the LULC change dynamics that occurred in ER between 1985 and 2019 and assess its impacts on the potential distribution of 22 amphibian, 9 bird and 6 reptile endemic species. In addition, we sought to evaluate whether the Espinhaço Range Biosphere Reserve (RBSE) designation (in year 2005, Phase 1) caused any noticeable effect on both LULC change patterns and the potential distribution ranges of target species.

Our results show that the ER underwent intense LULC change dynamics in the last 35 years, with farming and commercial forestry expansion as the primary drivers of native vegetation loss. The institution of the RBSE in 2005 did not change the increasing trajectories of forest plantations, urban-mining and farming areas, although it may have positively influenced a recent secondary forest recovery trend. Concerning the species' potential distributions, niche models predicted that around one-third of the target species underwent potential habitat loss between 1985 and 2019 within the ER. Considering the evaluated groups, birds presented the most consistent and highest trend of habitat loss (up to -28.6% relative variation). Furthermore, the results show that species potential richness tends to aggregate in two hotspots located in the southern and northern parts of ER and suggest that potential biodiversity loss likely results from the combined effects of multiple LULC change processes. Substantial challenges lie ahead to ensure the long-term conservation of Espinhaço endemic fauna. The present study highlights the need for conservation strategies focused on biodiversity hotspots through adequate land-use planning and management. Additionally, it supports the expansion of RBSE towards the northern part of ER to improve the connectivity between the hotspots.

Keywords: LULC change, Biodiversity decline, Espinhaço Range, Espinhaço Range Biosphere Reserve, Endemic mountaintop species, Habitat loss, Ecological Niche Modelling, Hotspot.

RESUMO

Alterações do uso e cobertura do solo (UCS), associadas com a perda e degradação de habitats, são apontadas como a principal causa global de declínio da biodiversidade. *Hotspots* terrestres de espécies endêmicas têm usualmente enfrentado maiores pressões de perda e degradação de habitats que outras regiões. É o caso da Serra do Espinhaço (SE), localizada no leste do Brasil e parcialmente designada como Reserva da Biosfera da UNESCO em 2005 devido a sua relevância ecológica, geomorfológica e cultural. Além disso, espera-se que espécies endêmicas de topos de montanhas sofram impactos mais intensos da perda e degradação de habitats, uma vez que são menos tolerantes a rápidas mudanças ambientais.

No presente estudo, integrando dados históricos de UCS e modelação de nicho ecológico, pretendemos desvendar as dinâmicas de alteração do UCS ocorridas na SE entre 1985–2019; e avaliar seus impactos sobre a potencial distribuição de 22 anfíbios, 9 aves e 6 répteis endêmicos da SE. Adicionamente, buscamos avaliar se a designação da Reserva da Biosfera da Serra do Espinhaço (RBSE) em 2005 (Fase 1) causou algum efeito perceptível nos padrões de alteração do UCS e na potencial área de distribuição das espécies-alvo.

Os resultados mostram que a SE passou por uma intensa dinâmica de alteração do UCS nos últimos 35 anos, sendo a expansão da agropecuária e silvicultura os vetores primários da perda de vegetação nativa. A instituição da RBSE em 2005 não alterou as trajetórias de aumento das plantações florestais, áreas urbanas, de mineração e agropecuárias, embora possa ter influenciado positivamente na recente tendência de recuperação de florestas secundárias. Os modelos de nicho ecológico previram que cerca de um terço das espécies-alvo sofreram perda potencial de habitat entre 1985 e 2019 na SE. Entre os grupos avaliados, as aves apresentaram tendências de perda de habitat mais consistentes e de maior magnitude (variação relativa de até -28,6%). Ademais, os resultados mostram que a riqueza potencial de espécies tende a se agregar em dois *hotspots* (partes sul e norte da SE) e sugerem que a perda potencial de biodiversidade resulta dos efeitos combinados de múltiplos processos de mudança do UCS. Há pela frente desafios substanciais a fim de garantir a conservação a longo prazo da fauna endêmica da SE. O presente estudo destaca a necessidade de estratégias de conservação focadas nos *hotspots* de biodiversidade através do planejamento e manejo adequado do uso da terra. Adicionalmente, apoia a expansão à norte da RBSE de forma a melhorar a conectividade entre os *hotspots*.

Palavras-chave: Alterações no UCS, Declínio da biodiversidade, Serra do Espinhaço, Reserva da Biosfera da Serra do Espinhaço, Espécies endêmicas dos topos de montanhas, Perda de habitat, Modelação de Nicho Ecológico, *Hotspot*.

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ABBREVIATIONS

ANM	National Mining Agency (*Agência Nacional de Mineração*)
ANN	Artificial Neural Networks
AUC	Area Under the Curve
BAM	Biotic-Abiotic-Movement
COBRAMAB	Brazilian Commission for the Man and the Biosphere Programme (*Comissão Brasileira para o Programa Homem e a Biosfera*)
COPAM	State Council for Environmental Policy (*Conselho Estadual de Política Ambiental*)
CTA	Classification Tree Analysis
DD	Data Deficient
DNI	Direct Normal Irradiation
EN	Endangered
ENM	Ecological Niche Model
ER	Espinhaço Range
ESM	Ensemble of Small Models
EVI2	Enhanced Vegetation Index 2
FAO	Food and Agriculture Organization
FDA	Flexible Discriminant Analysis
FMI	International Monetary Fund (*Fundo Monetário Internacional*)
GAM	Generalized Additive Models
GBM	Generalized Boosted Models
GLM	Generalized Linear Models
IABS	Brazilian Institute of Development and Sustainability (*Instituto Brasileiro de Desenvolvimento e Sustentabilidade*)
ICMBIO	Chico Mendes Institute for Biodiversity Conservation (*Instituto Chico Mendes de Conservação da Biodiversidade*)
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
LC	Least Concern

LULC	Land Use and Land Cover
MAB	Man and the Biosphere
MARS	Multivariate Adaptive Regression Splines
MAXENT	Maximum Entropy
MMA	Ministry of the Environment (Ministério do Meio Ambiente)
NDWI	Normalized Difference Water Index
NT	Near Threatened
QGIS	Quantum Geographic Information System
RBSE	Espinhaço Range Biosphere Reserve (*Reserva da Biosfera da Serra do Espinhaço*)
RFO	Random Forests
ROC	Receiver Operating Characteristic
SE	*Serra do Espinhaço*
SEMA	Environment Department (*Secretaria do Meio Ambiente*)
TSS	True Skill Statistics
TWI	Topographic Wetness Index
UCS	*Uso e Cobertura do Solo*
UNESCO	United Nations Educational, Scientific and Cultural Organization
VU	Vulnerable

1. Introduction

In the past decades, global environmental changes caused by human actions have reached unprecedented rates, leading to a biodiversity crisis scenario. Global assessments show alarming trends, such as the overall decline of 60% in vertebrates' population sizes between 1970 and 2014 estimated by the Living Planet Index, being this trend especially pronounced in the Neotropical region (*i.e.,* Central and South America), where the population decrease reached 89% for the same period (WWF, 2018). As a result, approximately 25% across animal and plant species assessed by the International Union for the Conservation of Nature (IUCN) are currently threatened with extinction, many in a short time (IPBES, 2019).

Land-cover and land-use (LULC) change, directly associated with habitat loss and degradation, is pointed as the leading global driver of biodiversity decline, having its impacts synergistically exacerbated by other factors such as climate change (IPBES, 2019; Mantyka-pringle, Martin, & Rhodes, 2012; Newbold et al., 2015). Indeed, about 75% of Earth's land surface has been significantly altered by anthropic action within the last millennium (IPCC, 2019; IPBES, 2019), while land-use change affected almost a third of the global area only in the last six decades (1960-2019) (Winkler et al., 2021).

Terrestrial hotspots of endemic species, which is the case of Espinhaço Range, often have faced more significant pressures of habitat loss and degradation than other terrestrial regions (IPBES, 2019). Located in eastern Brazil, this ancient neotropical mountain range is the core area of the rupestrian grasslands ecosystem (Fernandes et al., 2018). The synergy among geomorphological isolation, environmental filters derived from edaphoclimatic factors (*e.g.,* nutrient impoverished soils, seasonal distribution of rainfall) and biotic interactions over evolutionary time (Fernandes, 2016a; Silveira et al., 2016; Guedes et al., 2020) shaped what it is currently acknowledged as an important center of biological diversity and endemism for plant (Silveira et al., 2016; Vasconcelos et al., 2020) and animal communities (Chaves et al., 2015; Vasconcelos et al., 2008; Leite et al., 2008; Alves et al., 2008; ICMBio, 2012). For instance, despite covering less than 1% of Brazilian territory, it is estimated that rupestrian grasslands shelter around 17% of Brazil's plant diversity and 46% of the Cerrado's diversity (Fernandes et al., 2018), comprising an endemism rate of around 30% for angiosperm species (Giulietti et al., 1997; Rapini et al., 2008).

Biological survey efforts in the Espinhaço region have intensified in the 1960s, resulting in hundreds of new species described and studied (Fernandes et al., 2018). Nevertheless, many knowledge gaps regarding Espinhaço's biodiversity remain nowadays, with many species having their distribution ranges and conservation status poorly understood or overestimated and even becoming known by science under some significant threat degree (Fernandes et al., 2018; Hoffmann, Vasconcelos & Fernandes, 2020).

Espinhaço Range ecological importance and its biogeographic, geomorphological and cultural heritage significance resulted in the designation of the southern part of Espinhaço Range as a Biosphere Reserve by the United Nations Educational Scientific and Cultural Organization (UNESCO) in 2005 (IABS & RBSE, 2017). However, despite this, Espinhaço Range and the biodiversity that it harbours is threatened by increasing rates of human degradation of natural resources posed by mining, non-natural fires, biological invasions, exotic species afforestation, cattle ranching, among other factors (Fernandes, 2016b). The adverse effects of anthropogenic disturbances on biotic communities within Espinhaço Range are potentially exacerbated by the limited availability of highland habitats, their disjointed spatial distribution (which hinders colonization of new areas) and the lower tolerance of endemic mountaintop species to rapid environmental changes owing to high phylogenetic conservatism and niche specialization (Conceição et al., 2016; Hoffmann, Vasconcelos, & Fernandes, 2020).

Since land-use and land-cover transformation underpin habitat loss and degradation, understanding and quantifying LULC change and its spatiotemporal dynamics are crucial aspects in tackling the global challenge of biodiversity decline, as well as many other societal issues such as climate change, hydric depletion and food security (Winkler et al., 2021).

In this context, remote sensing has consolidated as an essential tool for monitoring key drivers of biodiversity change (*e.g.,* LULC dynamics) owing to its advantageous features in comparison to conventional ground surveys, such as its continuity (*i.e.,* the maintenance of long-term products derived from satellite data), large-scale coverage capacity, affordability, lower labour and time consumption (Prasad, Semwal, & Roy, 2015; Turner et al., 2015). Recently, substantial technological progress was made regarding satellite data acquisition, processing and interpretation (*e.g.,* satellite resolution improvements, increase of cloud computing capacity, developments in data science and machine learning algorithms leading to higher accuracy imagery classification) (Prasad, Semwal, & Roy, 2015; Souza et al., 2020). These, together with the advances concerning the broader accessibility to satellite data, allowed more robust applications of remote sensing for conservation purposes (Turner et al., 2015; Winkler et al., 2021). An example is the MapBiomas Project, a Brazilian initiative created in 2015 to reconstruct annual land-use and land-cover information (Souza et al., 2020; URL: *http://mapbiomas.org*).

One of the increasing applications of remote sensing derived data for conservation purposes is developing Ecological Niche Models (ENMs). In a nutshell, these models estimate species potential distribution based on the correlation between occurrence records and environmental predictor variables (Guisan & Thuiller, 2005; Peterson & Soberón, 2012; Prasad, Semwal, & Roy, 2015; Sillero, 2011). ENMs underwent considerable development in the last decades and became a powerful toolkit extensively applied to deal with a varied set of ecological issues (Guisan & Thuiller, 2005; Peterson & Soberón, 2012). For instance, by assessing how abiotic features underpin and shape species potential distribution patterns, ecological niche models can be used for the evaluation of environmental change effects on the habitat suitability of species considering current and forecasted scenarios, as well as for the appropriate setting of priority areas for conservation in the face of these changes (Guisan & Thuiller, 2005).

In this study, by integrating a land-use and land-cover change assessment based on an extended time-series dataset and ecological niche modelling, we aimed to: (1) unravel the LULC change dynamics that occurred in Espinhaço Range over the last 35 years; (2) evaluate whether the Espinhaço Range Biosphere Reserve designation caused any noticeable effect on land-use patterns; and (3) assess the impacts of historical LULC changes on the potential distribution range of bird, amphibian and reptile species endemic to the Espinhaço region.

We hypothesized that the LULC spatiotemporal changes within Espinhaço Range in the past decades may have caused a contraction of the available habitat for most of the endemic species assessed with differential gains/losses among species and groups. Thus, as specific objectives, this study proposes to identify spatial patterns of habitat contraction or expansion; to evaluate and rank the percentage of change in potential distribution ranges in order to draw attention to more vulnerable species from a conservation perspective; as well as to support the definition of essential areas for conservation through the identification of regions with high suitability and diversity for species. Overall, this knowledge is expected to guide and support decision-making and conservation efforts in the Espinhaço Range.

2. Materials and Methods

The present study encompassed two interconnected analytical components: (i) the land-use and land-cover change assessment and (ii) the ecological niche modelling whose temporal projections were based on the aforementioned land-use/cover change data, both comprising the period from 1985 to 2019. The methodological aspects of each component are respectively detailed in sub-sections 2.2 and 2.3.

2.1. Study Area

The Espinhaço Mountain Range is the largest continuous orogenic belt within Brazilian territory and, after the Andes, South America's second-largest mountain range (Fernandes et al., 2018; Neves et al., 2016; Saadi, 1995). Located in eastern Brasil, it extends over 1,200 km in the South-North direction, stretching from Minas Gerais to Bahia with a topographical and litho-structural division between the denominated Meridional and Septentrional plateaux (Saadi, 1995; Gontijo, 2008) (Figure 1).

The "Espinhaço", which means "large spine", was named by the German geologist, geographer and metallurgist Wilhelm Ludwig von Eschwege, alluding to the topographical appearance of this mountain range (Gontijo, 2008; Neves et al., 2016).

The Espinhaço region is within the warm subtropical climate range and presents diverse microclimates associated with topographic variability, characterized by average altitudes around 1,000 m but reaching peaks up to 2,000 m (Verdi et al., 2015). Mean annual temperature is around 18.5°C, and annual precipitation, on average, ranges from 850 to 1,400 mm, considering the north-south direction (COMIG, 1997 *apud* Gontijo, 2008).

Espinhaço Range serves as a hydrographic divider between the São Francisco River basin and the East Atlantic basin, which comprises rivers flowing directly into the Atlantic Ocean (Saadi, 1995; Verdi et al., 2015). It also delimits a contact zone between three major Brazilian biomes (see Figure 1): Cerrado (by western side), Atlantic Forest (by eastern side), two biodiversity hotspots; and Caatinga (by northern side). However, despite the influence of these vegetation domains, some authors understand the

Espinhaço Range as the basis of their own biome[1] due to its ecogeographic singularity, especially concerning the rupestrian highland grasslands (*i.e., "campos rupestres de altitude"*). These form a unique, biodiverse and evolutionary old phytophysionomy that dominates Espinhaço higher elevation areas (Colli-Silva, Vasconcelos, & Pirani, 2019; Fernandes et al., 2018) (Gontijo, 2008).

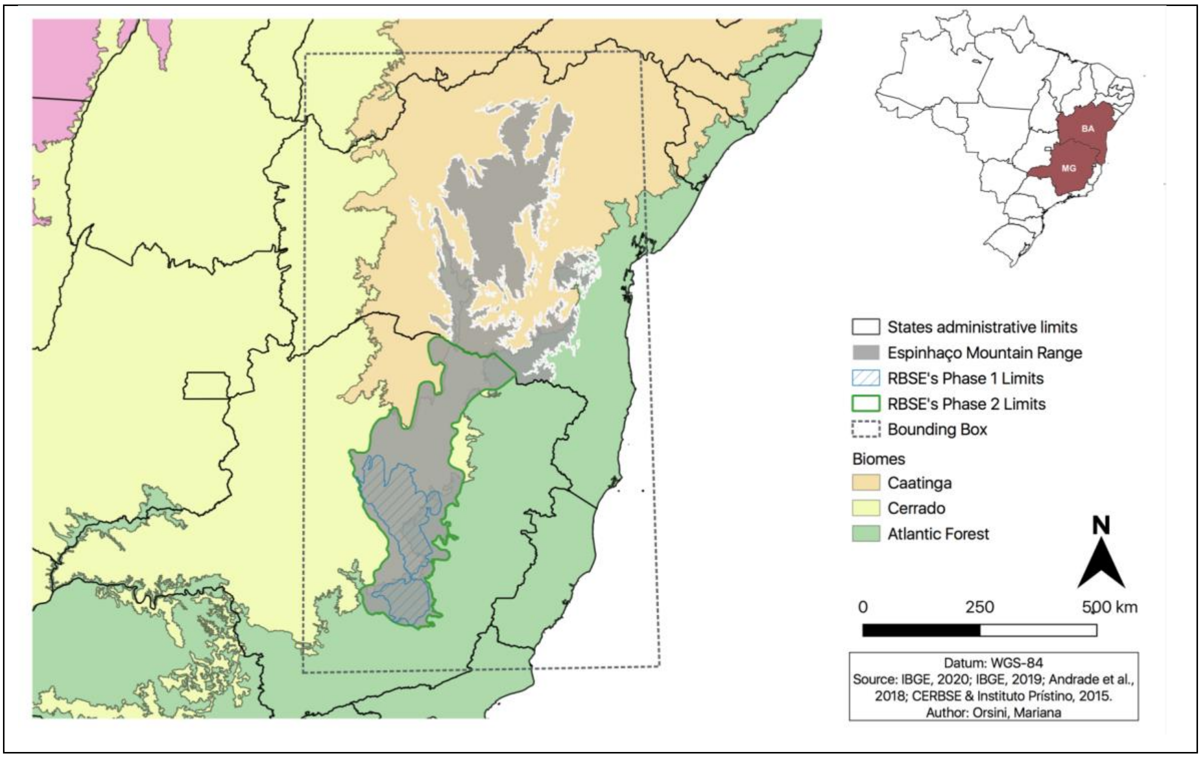

Figure 1: Study Area location.

The geomorphological isolation, in addition to climatic buffering and geological aspects (*e.g.:* acidic, nutrient-impoverished edaphic conditions; Guedes et al., 2020), shaped, over evolutionary history, a considerably heterogeneous landscape and one of the world's highest levels of biodiversity and endemism (Fernandes et al., 2018; Vasconcelos et al., 2020).

Owing to its ecological and geomorphological relevance and its cultural heritage, part of Espinhaço Range was recognized as Biosphere Reserve by the United Nations Educational, Scientific and Cultural Organization (UNESCO) in 2005 (see Figure 2 for RBSE's Phase 1 limits and zoning). By integrating southern Espinhaço to the Man and the Biosphere (MaB) Programme, the primary goals were to promote conservation

[1] Based on Walter's (1986) biome concept, as analyzed by Coutinho (2006): "An area of geographical space, with dimensions up to more than one million square kilometers, represented by a uniform type of environment, identified and classified according to macroclimate, phytophysiognomy, soil and altitude"

strategies, sustainable development and scientific knowledge divulgation (Andrade et al., 2018).

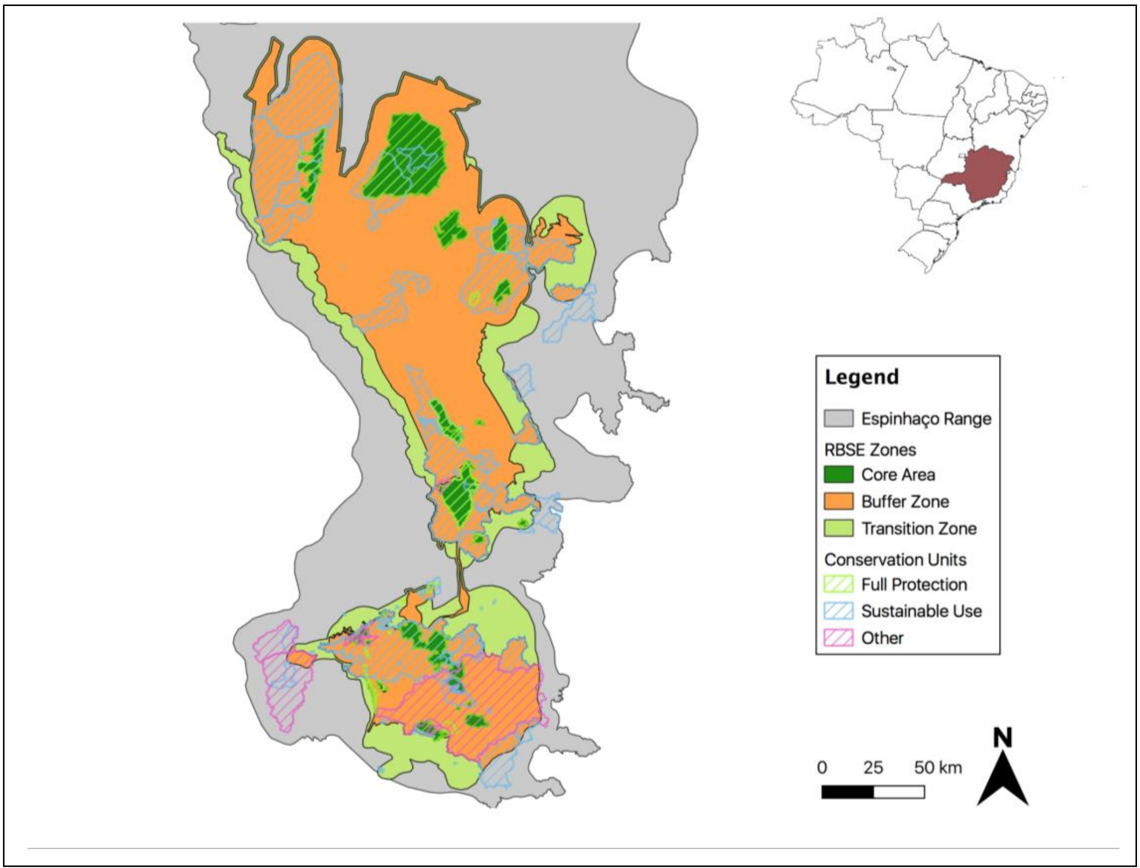

Figure 2: Espinhaço Range Biosphere Reserve (Phase 1) Zoning and Conservation Units.

In the context of the present study, we considered three spatial arrangements regarding the Espinhaço Range as nested study areas (Figure 1):

(i) the Espinhaço Range Biosphere Reserve (RBSE) limits established in 2005 (*i.e.*, Phase 1);
(ii) the totality of Espinhaço Mountain Range extending towards the state of Bahia; and
(iii) a rectangular bounding box surrounding the previous two areas with a 100 km buffer.

For the land-use and land-cover change analysis, we considered both the RBSE and the total Espinhaço Range as study areas. On the other hand, for the ecological niche modelling stage, we considered the bounding box as a study area to increase the environmental variability sampled, the number of available presence records of the species pool and, consequently, the algorithms' learning capacity and model performance. However, tha modelling outputs were subsequently cropped to match the Espinhaço Range and the RBSE extents, with the post-modelling analysis focusing on these areas.

It is important to highlight that the delimitation of Espinhaço Range was based, for the southern portion within Minas Gerais state, on the officially established limits for RBSE's Phase 2 expansion approved in 2019 by UNESCO (UNESCO, 2019). This delimitation was then combined with the manual delimitation of northern boundaries within Bahia state by matching the altitudinal gradients with the southern portion and considering areas above 600 m. The original limits regarding RBSE's Phase 1 were considered for LULC change analysis because its establishment date (2005) was more coherent to the temporal availability of the MapBiomas dataset (from the year 1985 to 2019) since one of the objectives was to assess the potential effects of the Biosphere Reserve creation.

2.2. Land-Use and Land-Cover Assessment

2.2.1. Data collection and processing

The long-term time series of land-use and land-cover was obtained from MapBiomas; a multi-disciplinary network developed for reconstructing annual land-use and land-cover information in Brazil (Souza et al., 2020) (*http://mapbiomas.org*).

MapBiomas Project is based on a pixel to pixel classification of Landsat imagery, through the random forest classification algorithm available in the Google Earth Engine platform, with the supervision of trained specialists. The classification process is separately performed for each Brazilian biome (*i.e.,* Amazon, Atlantic Forest, Caatinga, Cerrado, Pampa and Pantanal) and cross-cutting themes (*i.e.,* the classes that overlap the biomes delimitation, but were individually classified through a specific approach to reduce spectral confusion, namely Pasture, Agriculture, Coastal Zone and Urban infrastructure), resulting in a set of maps which, after applying temporal and spatial filters, are integrated (Souza et al., 2020). For more methodological details, consult Souza et al., 2020.

In the present study, we used MapBiomas Collection 5 (Projeto MapBiomas, 2020), which was released in August 2020, spanning from 1985 to 2019. The 30 m spatial resolution dataset, after downloaded, was processed to match our study area extent, considering the three geographical arrangements mentioned in the previous section.

The MapBiomas classification scheme, compatible with the Food and Agriculture Organization (FAO) classification framework, is a hierarchical system with several organization levels allowing the land-use/cover type to be assigned according to structural aspects of vegetation and human interactions (Souza et al., 2020).

In this context, it is crucial to conceptualize that land-cover refers to the biophysical attributes of the Earth's surface, while land-use is associated with the purpose for and activities by which humans utilize these attributes (Winkler et al., 2021; Hu et al., 2019).

The data processing, performed in QGIS software version 3.6, also included a reclassification of the original land-use and land-cover classes defined by MapBiomas. For analysis simplification purposes, the four original agriculture subclasses and the class comprising mosaic of pasture and agriculture were aggregated into one unique agriculture class. Table 1 describes the land-use/cover classification system used in the present study, showing the adaptations made in the original MapBiomas's classification.

Table 1: The land-use and land-cover classification system.

<table>
<tr><th>Previous Level</th><th>Level</th><th>Original Class</th><th>Reclass</th><th>Code</th><th>Description</th></tr>
<tr><td rowspan="3">1. Forest
1.1 Natural Forest</td><td>1.1.1</td><td>Forest Formation</td><td>Forest Formation</td><td>1</td><td>Vegetation types with a predominance of tree species with high-density continuous canopy, present in several biomes.</td></tr>
<tr><td>1.1.2</td><td>Savanna Formation</td><td>Savanna Formation</td><td>2</td><td>Also present in several biomes, vegetation types with a tree layer varying in density, with semi-continuous canopy and distributed over a continuous shrub-herbaceous layer.</td></tr>
<tr><td>1.1.3</td><td>Mangrove</td><td>Mangrove(ab)</td><td>3</td><td>Dense and evergreen forest formation often flooded by tide and associated with the mangrove coastal ecosystem.</td></tr>
<tr><td>1. Forest</td><td>1.2</td><td>Forest Plantation</td><td>Forest Plantation</td><td>4</td><td>Planted tree species for commercial use.</td></tr>
<tr><td rowspan="5">2. Non Forest Natural Formation</td><td>2.1</td><td>Wetland</td><td>Wetland(abc)</td><td>5</td><td>Floodplain with fluvial and lake influence, subject to periodic or permanent flooding, located along watercourses and in lowlands areas that accumulate water, with herbaceous shrub vegetation and/or arboreal and pioneer formations, and marshes (marine influence).</td></tr>
<tr><td>2.2</td><td>Grassland</td><td>Grassland</td><td>6</td><td>Vegetation type with a predominance of herbaceous stratum, including patches with a well developed shrub-herbaceous stratum.</td></tr>
<tr><td>2.3</td><td>Salt flat</td><td>Salt flat(ab)</td><td>7</td><td>“Apicuns” or salt flats are formations often without tree vegetation, associated to saline and a less flooded area in the mangrove, generally in the transition between this area and the continent.</td></tr>
<tr><td>2.4</td><td>Rocky Outcrop</td><td>Rocky Outcrop</td><td>8</td><td>Naturally exposed rocks in the terrestrial surface, without soil cover, often with partial presence of rupicolous vegetation and high slope.</td></tr>
<tr><td>2.5</td><td>Other Non Forest Natural Formation</td><td>Other Non Forest Natural Formation(a)</td><td>9</td><td>Herbaceous vegetation under fluvial or marine influence.</td></tr>
<tr><td>3. Farming</td><td>3.1</td><td>Pasture</td><td>Pasture</td><td>10</td><td>Pasture areas, mainly planted, related with farming activity. Areas of natural pasture are mainly classified as grassland formation, which may or may not be grazed.</td></tr>
<tr><td rowspan="3">3. Farming
3.2 Agriculture
3.2.1 Temporary Crop</td><td>3.2.1.1</td><td>Soy Bean</td><td rowspan="5">Agriculture</td><td rowspan="5">11</td><td rowspan="5">Areas with both annual and perennial crops. It also included farming areas where it was impossible to distinguish between agriculture and pasture.</td></tr>
<tr><td>3.2.1.2</td><td>Sugar Cane</td></tr>
<tr><td>3.2.1.3</td><td>Other Temporary Crops</td></tr>
<tr><td>3.2 Agriculture</td><td>3.2.2</td><td>Perennial Crop</td></tr>
<tr><td>3. Farming</td><td>3.3</td><td>Mosaic of Agriculture and Pasture</td></tr>
<tr><td rowspan="2">4. Non Vegetated Area</td><td>4.1</td><td>Beach and Dune</td><td>Beach and Dune(ab)</td><td>14</td><td>Sandy areas, with bright white colour, where there is no predominance of vegetation of any kind.</td></tr>
<tr><td>4.2</td><td>Urban Infrastructure</td><td>Urban Infrastructure</td><td>12</td><td>Urban areas with predominance of non-vegetated surfaces, including roads, highways and constructions.</td></tr>
</table>

Previous Level	Level	Original Class	Reclass	Code	Description
	4.3	Mining	Mining	13	Areas related to large mineral extraction, with clear soil exposure due to heavy machinery. Only areas near to Mineral Resources Research Company (CPRM) and AhkBrasilien (AHK) chart references were considered.
	4.4	Other Non Vegetated Areas	Other Non Vegetated Areas	15	Non-vegetated impermeable surface areas (infrastructure, urban areas or mining) not mapped into their classes, and exposed soil (mainly sandy soil) in natural area or off-season crop areas.
5. Water	5.1	River, Lake and Ocean	River, Lake and Ocean	16	Rivers, lakes, dams, reservoir and other water bodies.
	5.2	Aquaculture	Aquaculture	17	Artificial lakes, where aquaculture and/or salt production activities predominate.
6. Non Observed	6	Non Observed	Non Observed	0	Areas blocked by clouds or atmospheric “noise” or with an absence of ground observation.

Legend: LULC classes which are absent at (a) Espinhaço Range Biosphere Reserve; (b) Espinhaço Mountain Range; and (c) Bounding box (Class descriptions adapted from Projeto MapBiomas, 2020 and Souza et al., 2020).

2.2.2. Land-use and land-cover change analysis

The land-use and land-cover (LULC) change analysis encompassed both net change and gross change. As stated by Tomlinson et al. (2018) and Fuchs et al. (2015), net change refers to the difference in the total area of a determined land-use/cover type between two time steps, while gross change comprises the sum of all area gains and losses for this land-use/cover, considering the same period.

The area of each LULC class was calculated yearly from 1985 to 2019. For simplification purposes, area estimation was based on the area of each pixel (30m resolution) and the total amount of pixels for each LULC class, which was measured through QGIS tools.

Net change analysis consisted of the difference between the total area value concerning the final time step and the initial time step per LULC category, with positive values indicating area increase and negative values indicating area decrease. The net change was expressed as area (km^2) and percent (%) units of measurement and, regarding percent change, it denoted both absolute and relative variation (*i.e.,* change in comparison to the total study area and change in comparison to each initial class area, respectively), as detailed below.

- Total net change (%): refers to the difference between the final and initial area of a given land-use/cover in relation to the total area under study. This was assessed according to Equation 1.

$$N_T = \left(\frac{A_F - A_I}{A_T}\right) \times 100 \qquad (1)$$

- Relative net change (%): refers to the difference between the final and initial area of a given land-use/cover type, in relation to the initial area of the class *(i.e.,* indicates the intra-class percent variation). This was assessed according to Equation 2.

$$N_R = \left(\frac{A_F - A_I}{A_I}\right) \times 100 \qquad (2)$$

For E1. 1 and Eq. 2:

- N_T is the percent total net change (%);
- N_R is the percent relative net change (%);
- A_F is the final area of a given land-use/cover type;
- A_I is the initial area of a given land-use/cover type;
- A_T is the total study area.

For the gross LULC change analysis, transition matrices were generated through R software, Terra package. The matrices were established for two spatial extents (the Espinhaço Range Biosphere Reserve and the entire Espinhaço Mountain Range) and two temporal intervals (1985-2005 and 2005-2019).

The transition matrix indicates all the pairwise conversions between all the LULC classes and the amount of each one that remained unchanged during the study period. The vector of row sums represents the gross losses of a given LULC class, while the vector of column sums indicates the gross gains of this class. Hence, the balance between gains and losses of one LULC class results in its total net change, while the sum of gains and losses results in the total gross change, which is mathematically represented by the following equations (Tomlinson et al., 2018):

$$G_U = \left|\sum_{i=1}^{n_U} \beta_{iU} + \sum_{j=1}^{n_U} \beta_{Uj} - \beta_{UU}\right| \qquad (3)$$

$$N_U = \left|\sum_{i=1}^{n_U} \beta_{iU} - \sum_{j=1}^{n_U} \beta_{Uj}\right| \tag{4}$$

For Eq. 3 and Eq. 4:

- G is the total gross change;
- N is the total net change;
- U is the land-use;
- *i* is the row index;
- *j* is the column index;
- n is the area value;
- β denotes the area changing from land-use *i* to land-use *j*;
- β_{iU} is the area removed from U;
- β_{Uj} is the area added to U;
- β_{UU} is the area that remained unchanged;

Finally, Sankey diagrams were generated for graphical visualization of the temporal LULC change dynamics and the LULC conversions within the transition matrices were classified (Table 2) to generate transition maps that illustrate the spatial patterns of land-use/cover change in the study area. In order to improve the detection of patterns, the transition maps with original 30m spatial resolution were aggregated into a 3km pixel and the percentages of the resized pixel occupied by each transition class were calculated.

Table 2: Land-use/cover transitions classification.

ID	Transition class	Description
1	No LULC change	Classes that remained the same, also including the transitions from and into "Non Observed" class
2	Deforestation	Conversion of native forest formations into other LULC classes, except forest plantations
3	Afforestation	Conversion of LULC classes into native forest formation
4	Reforestation	Conversion of native forest formation into exotic forest plantation
5	Forest plantation expansion	Conversion of LULC classes, except native forest (included in Reforestation transition), into exotic forest plantations
6	Native vegetation interchange	Transitions between non forest native vegetation classes (*i.e.,* savanna formation, grasslands and other non forest natural formation)
7	Biomass gain	Transitions characterized by an increase in biomass gradient from rocky outcrops conversion into savannas, grasslands or other non forest natural formation
8	Rocky exposure / Biomass loss	Conversion of LULC classes into rocky outcrops
9	Renaturalization	Transition of anthropic land-uses into native land-covers
10	Farming expansion	Conversion of LULC classes into pasture and agriculture
11	Urban-mining expansion	Conversion of LULC classes into urban areas, mining and other anthropic areas classified as "Other Non Vegetated Areas"
12	Water increase	Conversion of LULC classes into "River Lake and Ocean"
13	Water decrease	Conversion of "River, Lake and Ocean" into other LULC classes

2.3. Ecological Niche Modelling

Ecological Niche Models (ENMs) describe patterns associated with species distribution based on the correlation between occurrence records and environmental factors (Dormann et al., 2012; Guisan & Thuiller, 2005; Kearney, Wintle, & Porter, 2010). These models consider as underlying assumptions the equilibrium state between species and the current environmental conditions, the temporal niche conservatism (*i.e.,* the slow pace of niche changes over evolutionary time; Soberón & Nakamura, 2009) and the independence of species records. ENMs can be used to obtain spatiotemporal predictions (or projections) of species potential distribution represented by habitat suitability maps.

The concept of niche is complex and diverse within ecological literature. Although this is not the scope of the present study, it is important to distinguish some concepts to better understand ecological modelling and its operational basis. Regarding niche definitions, Grinnellian fundamental niche concept comprises the set of environmental conditions (solely abiotic) which allow a positive intrinsic growth rate of species population, disregarding biotic interactions and colonization ability (Grinnell, 1917; Soberón, 2007; Soberón & Nakamura, 2009). Complementary to Grinnell's concept, Jackson and Overpeck (2000) denominated the potential niche as the subset of the fundamental niche which is actually available in a given geographical space and time (Soberón & Nakamura, 2009). On the other hand, the Eltonian niche (Elton, 1927) included the biotic interactions and resource-consumer dynamics, which were later consolidated in Hutchinson's realized niche, defined as a reduced part of the potential niche (and, consequently, the fundamental niche) that species really occupy considering the effects of biotic interactions and dispersal ability (Hutchinson, 1957; Soberón, 2007; Soberón & Nakamura, 2009) (see Figure 3). Thus, Hutchinson (1978) defined niche as a multidimensional space containing both environmental conditions (what he called scenopoetic variables) and resources (what he called bionomic variables, the ones that can be consumed and drive biotic interactions) required for species persistence.

Species distribution and ecological niche are not synonyms, although they are closely related concepts within the spatial dimension where species occur. In summary, species distribution is mainly determined by three interconnected factors which correlate to distinct niche compartments: abiotic (*i.e.,* environmental conditions), biotic (*e.g.,* competition, predation), and movement (*i.e.*, dispersal capacity) elements (Guisan &

Thuiller, 2005; Sóberon & Peterson, 2005), as represented in Biotic-Abiotic-Movement (BAM) diagrams (Peterson, 2011; Figure 3).

A. **B.**

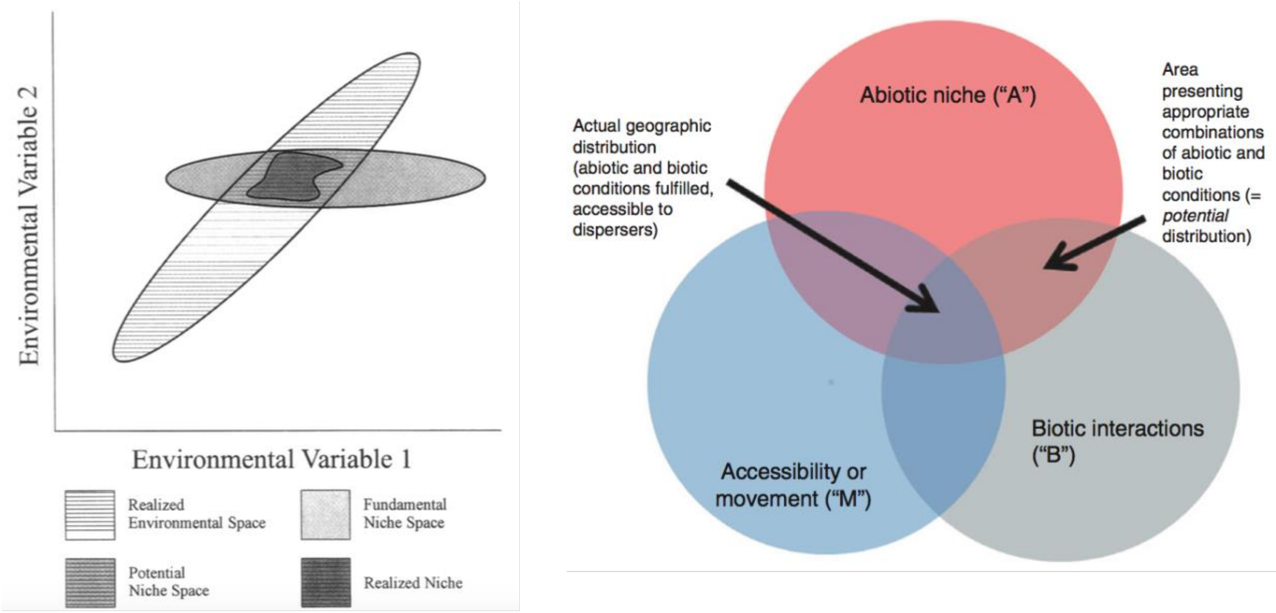

Figure 3: Schematic representation of the fundamental, potential and realized niche of a species in response to two environmental variables (A) (Jackson & Overpeck, 2000); and the BAM Diagram showing the main factors underpinning species distributions (B) (Peterson, 2011).

In this context, it is essential to highlight that Ecological Niche Models operate on the species realized niche (*i.e.,* the spatial dimension where species occur). However, since biotic interactions and dispersal ability are not directly accounted for in the modelling approach, ENMs predict species potential, not actual, distribution (*i.e.,* they provide an intermediate estimation between the realized and the fundamental niches) (Peterson & Soberón, 2012; Figure 3). In other words, this means that areas predicted by ENMs as suitable habitats, despite presenting adequate abiotic conditions for the species occurrence, may include areas outside their actual distribution range due to biotic and dispersal constraints (Peterson & Soberón, 2012; Soberón & Nakamura, 2009). This conceptual elucidation is necessary to ensure that model outputs are translated into proper conclusions from an ecological perspective.

2.3.1. Environmental predictors

Species distributions are driven by a combination of factors operating in distinct spatial scales. While climatic gradients are the main factor shaping species distribution on a macro scale (Thuiller et al., 2004; Soberón, 2007), other aspects such as land-cover affect species occupancy patterns at a finer local scale (Cord et al., 2014; Soberón, 2007; Luoto et al., 2007).

Although still a controversial topic, integrating multi-scale environmental filters to predict species' distribution has become an increasingly studied subject (Fournier et al., 2017; Luoto et al., 2007). In this context, many recent studies support the incorporation of additional environmental variables, especially remote sensing data (*e.g.,* spectral indices, biophysical or ecosystem functioning attributes; Arenas-Castro et al., 2019; Alcaraz-Segura et al., 2017), to bioclimatic ones in order to improve models' performance within regional or local scales (Fournier et al., 2017; Stanton et al., 2012; Wilson et al., 2013; Cord et al., 2014; Cord & Roder, 2011; Thuiller et al., 2004; Pearson et al., 2004; Luoto et al., 2007).

For the development of ENMs in this study, five sets of environmental variables were considered: bioclimatic, solar irradiation, topographic derivatives, land-use/cover variables and spectral indexes derived from satellite data (see summary in Table 3).

The environmental dataset comprised both static and dynamic predictors. The static variables were the ones whose values were considered unchanged over the timeframe under modelling, encompassing the bioclimatic, topographic and solar irradiation factors. On the other hand, the dynamic predictors were those expected to substantially change during the modelling timeframe and whose spatiotemporal variation was taken into account for models training and projection, encompassing variables related to land-use/cover and spectral indexes.

Table 3: Set of environmental variables used for Ecological Niche Modelling.

Category	Environmental factor	Variable Code	Description	Source	Spatial Resolution
Static	Bioclimatic	BIO01	Annual Mean Temperature	CHELSA (version 1.2)	1km
		BIO02	Mean Diurnal Range		
		BIO03	Isothermality		
		BIO04	Temperature Seasonality		
		BIO05	Max Temperature of Warmest Month		
		BIO06	Min Temperature of Coldest Month		
		BIO07	Temperature Annual Range		
		BIO08	Mean Temperature of Wettest Quarter		
		BIO09	Mean Temperature of Driest Quarter		
		BIO10	Mean Temperature of Warmest Quarter		
		BIO11	Mean Temperature of Coldest Quarter		
		BIO12	Annual Precipitation		
		BIO13	Precipitation of Wettest Month		
		BIO14	Precipitation of Driest Month		
		BIO15	Precipitation Seasonality		
		BIO16	Precipitation of Wettest Quarter		
		BIO17	Precipitation of Driest Quarter		
		BIO18	Precipitation of Warmest Quarter		
		BIO19	Precipitation of Coldest Quarter		
	Solar irradiation	DNI	Direct Normal Irradiation	Global Solar Atlas 2.0	250m
	Topographic	DEM	Digital Elevation Model (i.e., height above sea level)	JAXA (version 3.1)	30m
		Slope	Steepness of the surface	Derived from JAXA dataset	
		TWI	Topographic Wetness Index	Derived from JAXA dataset	
Dynamic	Land use and land cover	FORF	Forest Formation	MapBiomas (Collection 5.0)	30m
		SAVF	Savanna Formation		
		MANG	Mangrove		
		FORP	Forest Plantation		
		WETL	Wetland		
		GRAS	Grassland		
		SALF	Salt Flat		
		ROCK	Rocky outcrop		
		ONFN	Other Non Forest Natural Formation		
		PAST	Pasture		
		AGRI	Agriculture		
		URBI	Urban Infrastructure		
		MING	Mining		
		BEDU	Beach and Dune		
		ONVA	Other Non Vegetated Area		
		RILO	River, Lake and Ocean		
		AQUA	Aquaculture		
	Spectral indexes	EVI2	Enhanced Vegetation Index 2	MapBiomas	30m
		NDWI	Normalized Difference Water Index		

The climatic data for current conditions encompasses monthly and annual average temperature and precipitation patterns based on Climatologies for the period between 1979 and 2013 (http://chelsa-climate.org/). It was complemented by solar irradiation data from the Global Solar Atlas 2.0 database. We used the Direct Normal Irradiation (DNI), representing the sum of energy per unit area perpendicularly received from the sun, measured by the long-term yearly average of daily totals between 1999 and 2020 for Brazil (ESMAP, 2019; https://globalsolaratlas.info/).

Topographic variables included direct elevation measures and their derivatives such as slope and the Topographic Wetness Index (TWI), which describes the trend of water flow accumulation in a terrain (Quinn, Beven, & Lamb, 1995).

Regarding the dynamic environmental predictors, this study considered remote sensing-based data comprising both categorical land-use/cover variables and continuous spectral indexes (*i.e.,* the Enhanced Vegetation Index 2, EVI2; and The Normalized Difference Water Index, NDWI).

EVI2 is a two-banded vegetation index which, similarly to EVI, provides optical measures of vegetation canopy greenness comprising multiple aspects such as canopy cover and structure, leaf area and its chlorophyll content (Jiang et al., 2008). Developed for allowing cross-sensor applications and integration to long-term historical vegetation indexes products, the EVI2 was validated, at local and global scales and across various land-cover types, as an accurate substitute of EVI for atmospherically corrected and suitable quality pixels, also keeping its improved sensitivity in high biomass regions (Jiang et al., 2008). NDWI, in turn, provides a measure of the liquid water content in vegetation canopies through its interaction with solar radiation, including some background soil reflectance effects (Gao, 1996). It is therefore used as an environmental moisture indicator.

The use of continuous remote sensing data as model predictors is pointed as advantageous in comparison to the use of categorical land-cover data since it avoids information loss (*i.e.,* continuous remote sensing data preserves variation gradients) and the introduction of additional errors which are inherent to any classification process (Cord et al., 2014). However, the high thematic detail and accuracy levels achieved by MapBiomas Project over time (Souza et al., 2020) encouraged the use of this land-use/cover dataset, which is described in **Section 2.2.1,** as a predictor variable in this study.

Among the aforementioned set of eligible environmental filters, the predictive variables were distinctively selected for each target species through a pre-training process carried out by the Random Forest algorithm (R software). This process was based on three criteria: (i) the importance value of each variable; (ii) avoiding correlation between variables; and (iii) equitable selection among static and dynamic variables (*i.e.,* selection of the same number of variables from each category). In addition, for land-use/cover variables, expert knowledge about the target species' habitat preferences was also considered an additional selection criterion. This process was accomplished by asking experts specialized in each taxonomic group (*i.e.,* reptiles, birds and amphibians) to indicate the main land use/cover categories linked to each species habitat.

The number of selected variables varied according to the number of records available for each target species. For probabilistic purposes, it was considered that a minimum set of ten presence observations is required to support the inclusion of one environmental predictor (*i.e.,* a 1:10 ratio), as stated by Harrell et al. (1996) and Guisan & Zimmermann (2000). The Appendix S1 presents the relation between target species and their respective selection of environmental factors for ENMs development.

2.3.2. Species records

The Espinhaço fauna is characterized by high diversity and endemism levels, including many species occurring within small geographic ranges, patchily distributed across high altitude areas and often with few known occurrence records (Fernandes et al., 2018). Lower tolerance to changes and limited habitat availability within highlands raise a concern about the vulnerability of these species in the face of environmental changes (Hoffmann, Vasconcelos, & Fernandes, 2020).

The present study focused on fauna groups with contrasting dispersal abilities, comprising as target species 22 amphibians, 6 reptiles and 9 birds endemic to Espinhaço Range and/or threatened with extinction within local or global contexts. Since models' accuracy is highly dependent on the number of observations available, although the sensitiveness to small sample sizes differs among statistical modelling techniques (Pearson et al., 2007; Thibaud et al., 2014), we established a 15 records minimum threshold for species selection (*i.e.*, only species presenting a minimum of 15 independent observations at a resolution of 1×1 km were considered). Thus, these procedures resulted in a final database composed by 1742 independent records of 37 target species (Table 4).

Table 4: Target species for Ecological Niche Modelling. Conservation status for Minas Gerais state according to COPAM Normative n° 147/2010; Bahia state according to SEMA Ordinance n° 37/2017; Brazil according to MMA Ordinance n° 444/2014; and global conservation status according to the IUCN Red List of Threatened Species (IUCN, 2021).

Fauna Group	Order	Family	Species	Conservation status				Independent Records
				MG	BA	BR	IUCN	
Bird	Passeriformes	Thraupidae	*Embernagra longicauda*	-	-	-	LC	188
Bird	Passeriformes	Tyrannidae	*Polystictus superciliaris*	-	VU	-	LC	129
Bird	Apodiformes	Trochilidae	*Augastes scutatus*	-	-	-	LC	118
Bird	Passeriformes	Furnariidae	*Asthenes luizae*	-	-	-	NT	100
Bird	Passeriformes	Furnariidae	*Cinclodes espinhacensis*	-	-	EN	-	48
Bird	Apodiformes	Trochilidae	*Campylopterus diamantinensis*	-	-	-	-	29
Bird	Apodiformes	Trochilidae	*Augastes lumachella*	-	EN	EN	NT	28
Bird	Passeriformes	Furnariidae	*Asthenes moreirae*	-	EN	-	LC	20
Bird	Passeriformes	Thamnophilidae	*Formicivora grantsaui*	-	-	EN	EN	15
Amphibian	Anura	Cycloramphidae	*Thoropa megatympanum*	-	-	-	LC	110
Amphibian	Anura	Hylidae	*Scinax machadoi*	-	-	-	LC	97
Amphibian	Anura	Hylidae	*Bokermannohyla alvarengai*	-	-	-	LC	93
Amphibian	Anura	Hylidae	*Bokermannohyla saxicola*	-	-	-	LC	87
Amphibian	Anura	Hylidae	*Scinax curicica*	-	-	-	DD	84
Amphibian	Anura	Hylidae	*Bokermannohyla nanuzae*	-	-	-	LC	46
Amphibian	Anura	Leptodactylidae	*Leptodactylus camaquara*	-	-	-	DD	42
Amphibian	Anura	Hylidae	*Bokermannohyla oxente*	-	-	-	LC	40
Amphibian	Anura	Leiuperidae	*Pseudopaludicola mineira*	-	-	-	DD	39
Amphibian	Anura	Hylidae	*Bokermannohyla martinsi*	-	-	-	LC	36
Amphibian	Anura	Hylidae	*Boana botumirim*	-	-	-	-	31
Amphibian	Anura	Hylodidae	*Crossodactylus trachystomus*	-	-	-	DD	31
Amphibian	Anura	Odontophrynidae	*Odontophrynus juquinha*	-	-	-	-	29
Amphibian	Anura	Hylidae	*Aplastodiscus heterophonicus*	-	-	-	-	25
Amphibian	Anura	Hylodidae	*Hylodes uai*	-	-	-	DD	23
Amphibian	Anura	Leptodactylidae	*Physalaemus orophilus*	-	-	-	-	23
Amphibian	Anura	Hylidae	*Boana cipoensis*	-	-	-	NT	21
Amphibian	Anura	Hylidae	*Phasmahyla jandaia*	-	-	-	LC	21
Amphibian	Anura	Phyllomedusidae	*Pithecopus megacephalus*	-	-	-	DD	19
Amphibian	Anura	Odontophrynidae	*Proceratophrys cururu*	-	-	-	DD	18

Fauna Group	Order	Family	Species	Conservation status				Independent Records
				MG	BA	BR	IUCN	
Amphibian	Anura	Hylidae	*Scinax montivagus*	-	-	-	-	17
Amphibian	Anura	Strabomantidae	*Pristimantis rupicola*	-	-	-	-	15
Reptile	Testudines	Chelidae	*Hydromedusa maximiliani*	VU	-	-	VU	30
Reptile	Squamata	Tropiduridae	*Tropidurus montanus*	-	-	-	LC	23
Reptile	Squamata	Amphisbaenidae	*Amphisbaena acangaoba*	-	-	-	-	18
Reptile	Squamata	Gymnophthalmidae	*Psilops paeminosus*	-	EN	-	VU	17
Reptile	Squamata	Tropiduridae	*Tropidurus erythrocephalus*	-	VU	VU	NT	17
Reptile	Squamata	Tropiduridae	*Eurolophosaurus nanuzae*	-	-	-	LC	15

Species data were obtained through collaboration with Brazilian researchers experts in Espinhaço's selected fauna groups. The database kindly provided consisted of presence-only records with a minimum 1×1 km spatial resolution gathered from primary field surveys, in addition to occurrence data compiled from literature and museum specimens after their identity verification. All available records were verified and temporally and spatially filtered to match the temporal availability of land-use/cover data (*i.e.,* 1985–2019) and decrease spatial autocorrelation by excluding duplicates within the 1×1 km grid.

To obtain the training dataset, we extracted the values of each environmental predictor based on its spatial coordinates. Moreover, for dynamic variables (*i.e.,* spectral indices and land-use/cover), we matched the year of the record with that of each variable. This process maximised the spatiotemporal coherence in model training conditions by avoiding errors due to landscape shifts across time.

Since species' true absence data was unavailable for modelling purposes, the database was complemented by ten pseudo-absences sets randomly generated for the extended study area, being the number of pseudo-absences equal to the number of presences for each set and target species (Barbet-Massin et al., 2012).

2.3.3. Model development: fitting, evaluation and projection

Concerning species distribution modelling, there are many alternative workflows and techniques that vary in the underlying premises, concepts, operational basis, and resulting projections (Araujo & New, 2007; Watling et al., 2015). An example of such workflow is the one we followed, which combines different techniques and explores their resulting range of projections. This process is done by an ensemble approach which can significantly improve the models' forecasts robustness (Araujo & New, 2007).

Therefore, to produce more robust results, the Ecological Niche Models were developed through *biomod2* package within R software (Thuiller et al., 2009). Nine techniques, representing four modelling classes, were combined to produce a consensus model: (1) Generalized Linear Models (GLM); (2) Generalized Additive Models (GAM); (3) Multivariate Adaptive Regression Splines (MARS) as regression-based approaches; (4) Classification Tree Analysis (CTA); (5) Flexible Discriminant Analysis (FDA); (6) Artificial Neural Networks (ANN) as classification/machine-learning approaches; (7) Random Forests (RFO); (8) Generalized Boosted Models (GBM) as bagging and boosting approaches; and (9) Maximum Entropy (MAX) (Thuiller et al., 2009; Guisan, Thuiller & Zimmermann, 2017).

The target species were modelled individually using a random two-fold partition of occurrence dataset, with 75% assigned to model calibration and 25% to model evaluation. Each algorithm was run 10 times using a different set of pseudo-absences each time. The 10% best performing models (according to the True-skill Statistic, TSS values) were selected, and the projections were averaged to generate the final ensemble forecast (Araujo & New, 2007). For evaluation of models' predictive performance (*i.e.*, models 'accuracy), two metrics were considered: the area under the Receiver Operating Characteristic (ROC) curve (AUC; Fielding & Bell, 1997) and the maximum True-Skill Statistic (TSS; Allouche et al., 2006).

Then, the ensemble model calibrated for current conditions – more specifically, the land-use/cover conditions found at each year included in the set of records available for the species (*i.e.,* a multi-annual calibrated model) – was spatially and temporally projected to estimate the past distribution of target species on a biennial basis (*i.e.,* every two years) for the 2018-1985 period. These spatiotemporal projections were performed by replacing the calibration conditions during model training to each year available for the dynamic variables. Habitat suitability spatial projections were obtained for the entire study area (*i.e.,* the bounding box surrounding the Espinhaço Range). These temporal projections allowed us to assess the effects of land-use/cover change in species habitat suitability dynamics and diagnose potential gains or losses in distributions.

Finally, we transformed the continuous probability maps regarding habitat suitability into binary ones (*i.e.,* suitable *vs* unsuitable habitats) using the threshold value which maximized the TSS performance statistic (Liu et al., 2005; Liu et al., 2015). The final outputs were habitat suitability maps for current and past scenarios for each target species. The Figure 4 presents the workflow followed for Ecological Niche Models development, as described above.

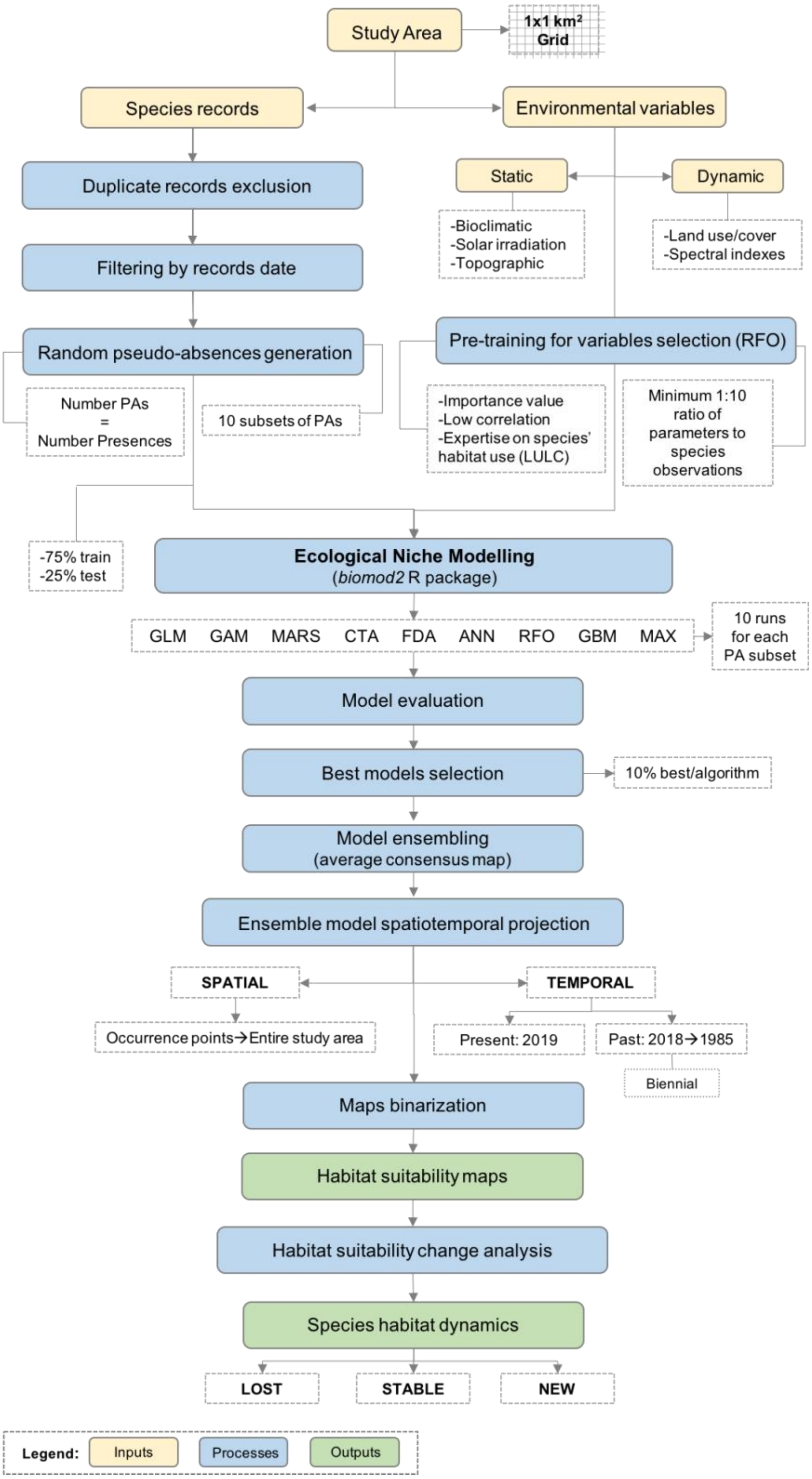

Figure 4: Schematic representation of the methodological steps for model development.

2.3.4. Habitat suitability analysis

Through model projections for current and past conditions, we were able to quantify changes in suitable habitat distribution for target species over the last 35 years within Espinhaço Range and the Espinhaço Range Biosphere Reserve (RBSE, Phase 1 limits). The post-modelling analysis was performed in R and QGIS (version 3.6) and comprised two complementary analytical components: (i) the suitable area trend assessment for each target species; and (ii) the species richness assessment for the study area.

In the first analysis, the temporal variation of the area predicted as suitable for each species was assessed, allowing the evaluation of species potential distribution contraction or expansion trends between 1985 and 2019.

Regarding the second analysis, species richness maps were generated for each fauna group by overlapping the individual species distribution maps for each biennium. Then, we used the nonparametric Theil-Sen estimator to assess the spatiotemporal trends to species richness change (*i.e.,* gains or losses) within the Espinhaço Range between 1985 and 2019. In summary, the Theil-Sen estimator (Sen, 1968) is a robust linear regression method that calculates, for each pixel, the trend curve that best fits the temporal distribution of species richness, estimating the magnitude of temporal change based on the curve's inclination (trends presenting $p<0.05$ were considered statistically significant). Thus, it was possible to assess changes in biodiversity spatial patterns and quantify species turnover in the study area over time.

3. Results

3.1. Land-Use and Land-Cover Assessment

The results of the land-use/cover assessment are presented for two spatial extents (see **Section 2.1** for study area details): the totality of Espinhaço Range *(i.e.,* study area II) and the Espinhaço Range Biosphere Reserve's Phase 1 limits (*i.e.,* study area I).

3.1.1. Espinhaço Range

3.1.1.1. Landscape composition

The Espinhaço Mountain Range (*i.e.,* spatial arrangement II; see **Section 2.1**) is located in a heterogeneous landscape mainly composed of savanna formations (42.02%[2]), pasture (25.33%[2]) and forest formations (14.21%[2]), surrounded, in a lesser extent, by agricultural areas (7.80%[2]), grasslands (5.13%[2]) and forest plantations of commercial species (3.80%[2]) such as *Eucalyptus* spp. and *Pinus* spp. These six land-use/cover classes, together, represented 96,99% and 98,28% of the Espinhaço area in 1985 and 2019, respectively.

Although the general landscape composition was maintained from 1985 to 2019, *i.e.,* the most representative land-use/cover types, in terms of occupied area, remained the same, as seen in Figure 5, many land-use changes occurred during the 35 years analyzed, as detailed in the following sections.

[2] Area value refers to the year 2019 (the last one available in MapBiomas dataset for access date until August 2021)

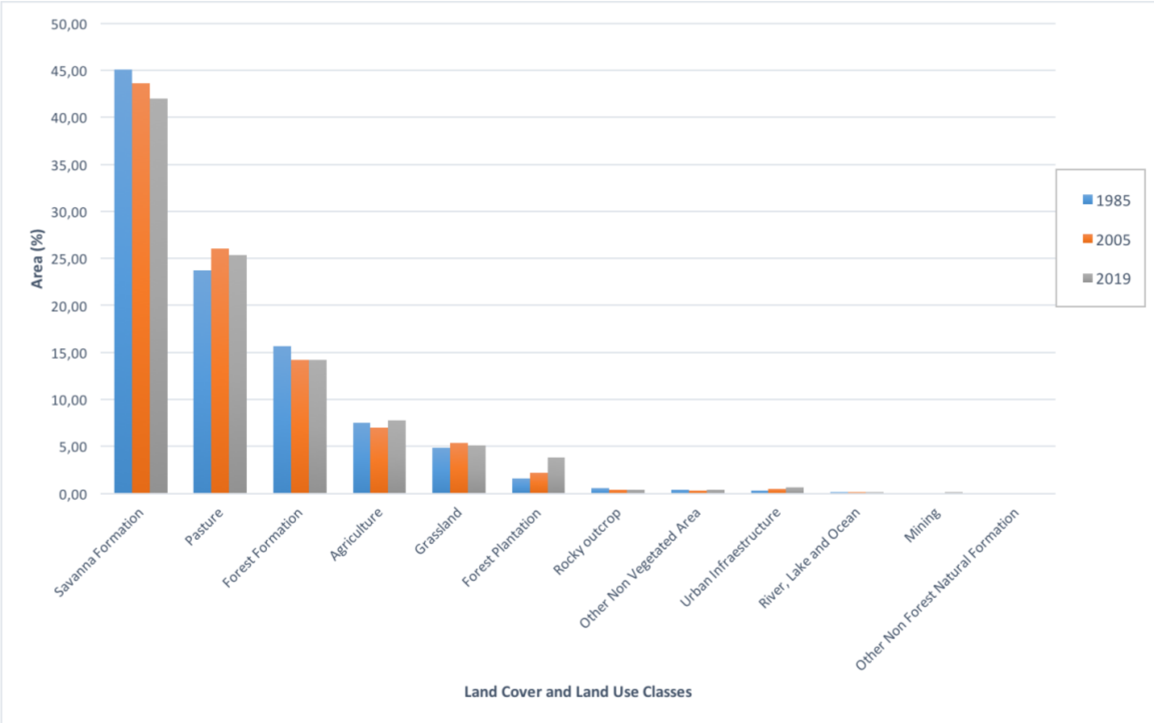

Figure 5: Land-cover and land-use composition in the Espinhaço Mountain Range from 1985 to 2019.

3.1.1.2. LULC temporal and spatial variation trends

From 1985 to 2019, around 223 726 km^2, corresponding to roughly 9.56% of Espinhaço Range total area, has changed. The savanna and forest formations were the land-cover types that suffered the most significant decrease in this period, shrinking, respectively, -6.9% (from 45.1% to 42.0% of Espinhaço's total area, *i.e.,* -3.11% total net change) and -9.4% (from 15.7% to 14.2% of Espinhaço's total area, *i.e.,* -1.47% total net change). In contrast, the classes that underwent the major net increase were the forest plantations (from 1.6% to 3.8% of Espinhaço's total area, *i.e.,* 2,24% total net change) and pasture (from 23.8% to 25.3% of Espinhaço's total area, *i.e.,* 1.56% total net change). The forest plantations, besides presenting a significant total net change, also showed a pronounced relative increase of approximately 144%.

Some land-use/cover classes, despite occupying a small area in proportion to the total Espinhaço Range area (<1%), showed an expressive relative net variation during the analyzed period. Urban infrastructure increased around 150% of its initial area; mining areas, increased by 29.26%, mainly between 2015 and 2016; and the rocky outcrops decreased by -30.39%.

As can be seen in Appendix S2, some land-use and land-cover classes presented a linear change trajectory over time, such as forest plantations and urban infrastructure, which presented an increasing linear trend, while savanna and forest formations presented an opposite decrease trend. The results, however, suggest an attenuation of deforestation rates, with possible ecological succession effects, since 2010.

In contrast, other land-use and land-cover classes presented a more oscillating behaviour, such as the grasslands, rocky outcrops and agriculture; or even bidirectional, like pasture areas, which showed an increasing trend between 1985 – 2004, followed by a reduction tendency until 2018 (see Appendix S2).

The land-use and land-cover net change is presented in Table 5 and Figure 6, whereas Appendix S2 shows in detail the temporal change trajectories presented by each land-use and land-cover class. The spatial distribution of the LULC classes in 1985 and 2019 is shown in Figure 7.

Table 5: Five-yearly summary of land-use and land-cover change in Espinhaço Range from 1985 to 2019.

Class / Year	**Land-Cover / Land-Use Area (%)**								**Total Net Change 1985-2019 (%)**	**Relative Net Change 1985-2019 (%)**
	1985	**1990**	**1995**	**2000**	**2005**	**2010**	**2015**	**2019**		
Savanna Formation	45.13	45.37	44.82	44.24	43.65	43.26	43.04	42.02	-3.11	-6.89
Pasture	23.76	24.16	25.13	25.56	26.02	26.05	25.52	25.33	1.56	6.58
Forest Formation	15.68	15.36	14.64	14.62	14.22	13.83	14.02	14.21	-1.47	-9.39
Agriculture	7.54	7.26	7.27	6.86	7.04	7.17	7.04	7.80	0.27	3.56
Grassland	4.88	5.06	4.92	5.30	5.38	5.21	5.34	5.13	0.24	4.96
Forest Plantation	1.56	1.51	1.87	1.92	2.18	2.92	3.41	3.80	2.24	144.08
Rocky outcrop	0.58	0.41	0.43	0.43	0.42	0.40	0.39	0.41	-0.18	-30.39
Other Non Vegetated Area	0.41	0.35	0.31	0.37	0.34	0.32	0.34	0.38	-0.02	-5.44
Urban Infrastructure	0.27	0.34	0.41	0.47	0.51	0.57	0.65	0.68	0.41	150.55
River, Lake and Ocean	0.12	0.11	0.12	0.14	0.15	0.19	0.16	0.16	0.03	28.50
Mining	0.07	0.08	0.08	0.08	0.08	0.08	0.08	0.10	0.02	29.26
Other Non Forest Natural Formation	0.001	0.001	0.001	0.001	0.001	0.001	0.000	0.001	0.000	-30.58

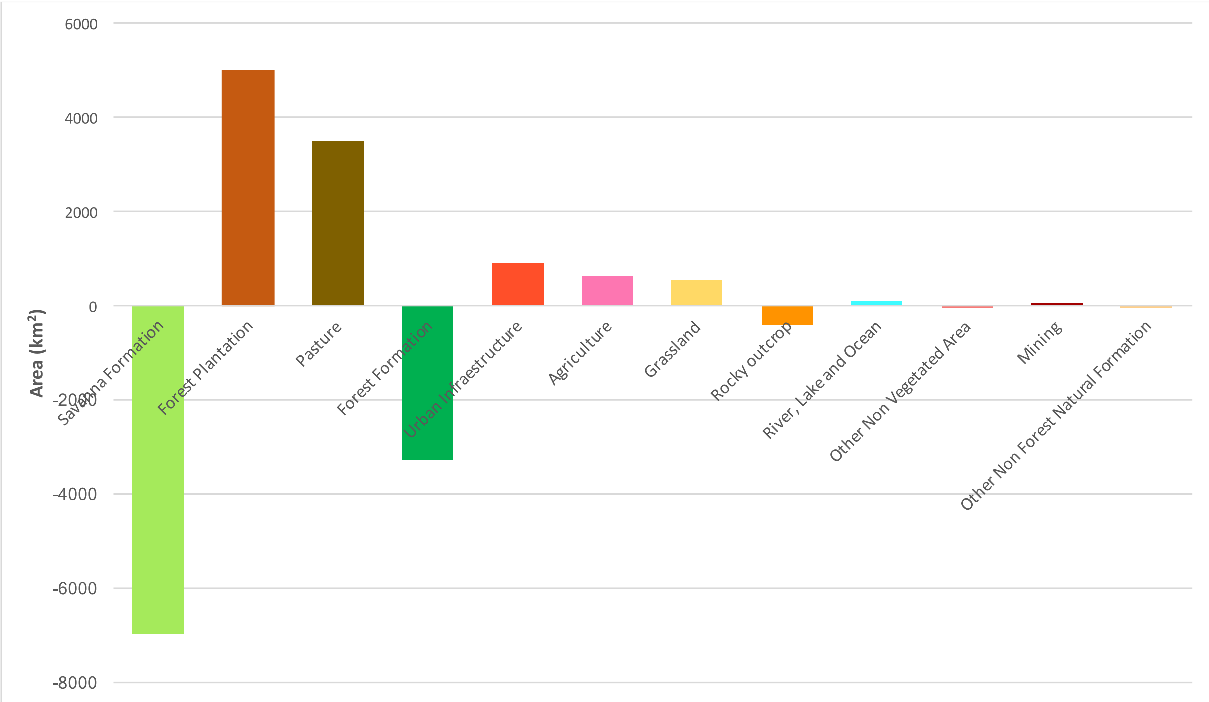

Figure 6: Land-use and land-cover net change in Espinhaço Range during 1985 – 2019. Columns are arranged in descending order of total net change.

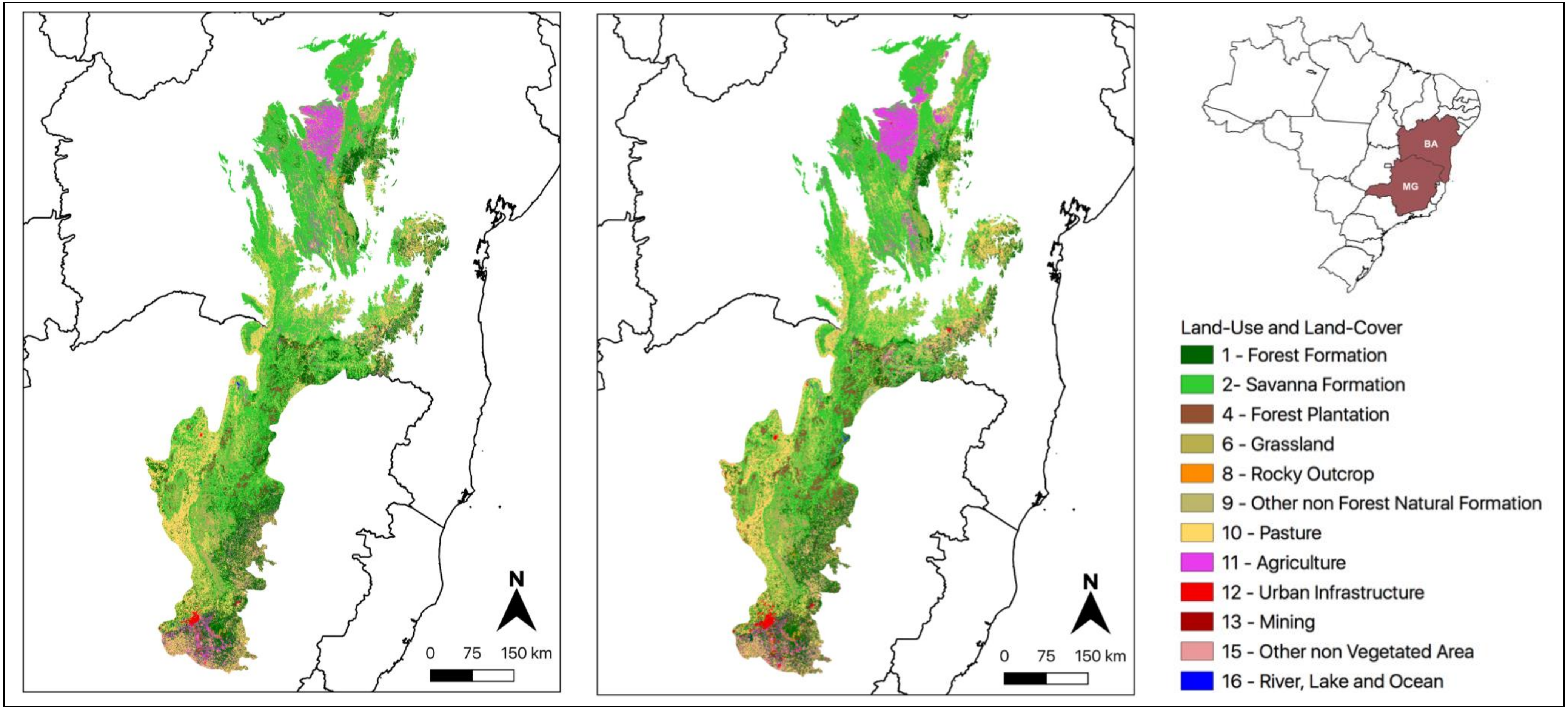

Figure 7: Spatial distribution of land-use and land-cover classes within Espinhaço Range in 1985 (left) and 2019 (right) scenarios.

3.1.1.3. LULC transition analysis

Concerning gross changes, around 25% of total Espinhaço area underwent land-use/cover changes, being the most expressive transitions for the 1985 – 2019 period: farming expansion (42.14%), renaturalization of anthropic areas (20.69%), deforestation (12.49%) and afforestation (10.13%), as presented in Figure 8.

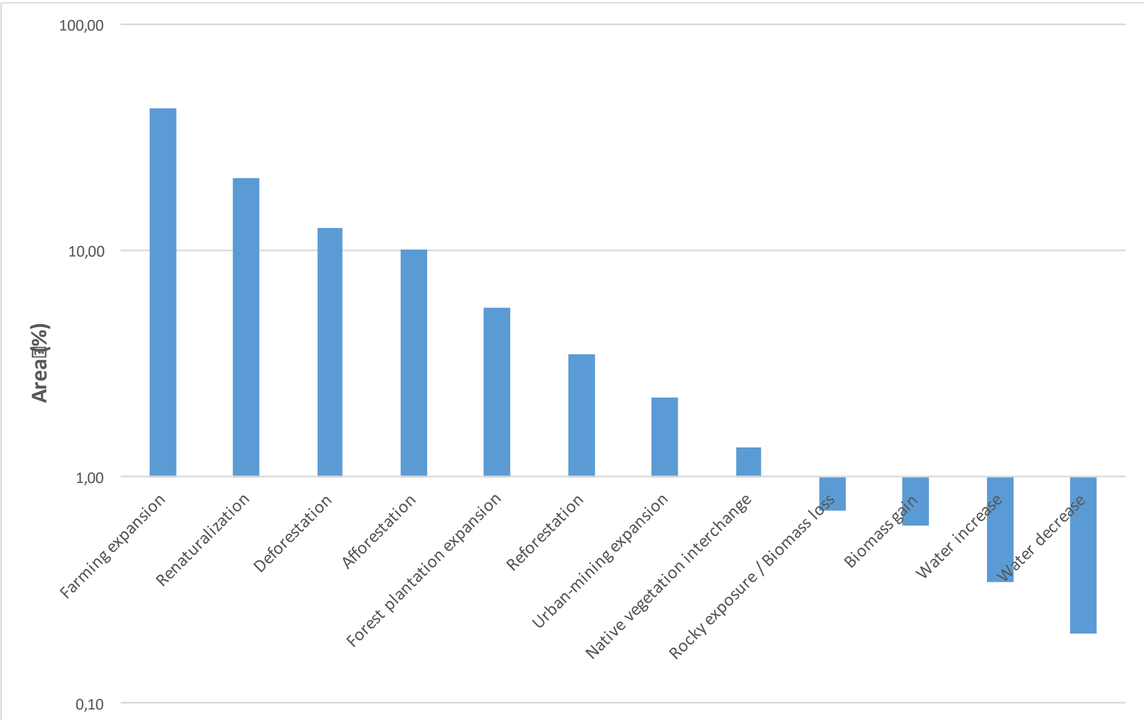

Figure 8: Land-use/cover transitions within the Espinhaço Range during 1985 – 2019.

Besides being among the classes that underwent major net changes, pasture, savanna, and forest formations were also among the land-use types that showed larger area gross change (in km^2, see Figure 9). On the other hand, agriculture, although presented a minor expressive total net change (0.27% of Espinhaço Range's total area), underwent a significant total gross variation, with area gains and losses closely balanced.

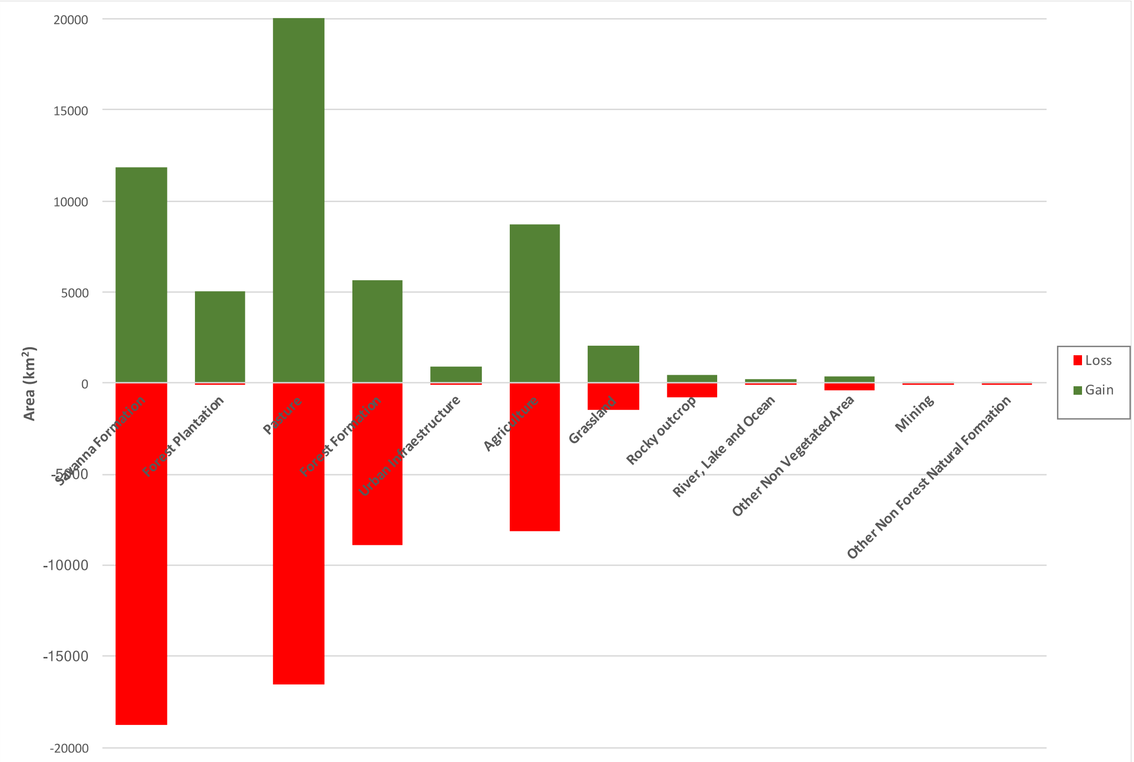

Figure 9: Land-use and land-cover gross change in the Espinhaço Range during the period of 1985 – 2019. Columns are arranged in descending order of total net change.

Concerning the classes which underwent major changes, around 18.6% of savanna formations that existed in 1985 were primarily transformed into other land-use types until 2019, mainly into pastureland (11.1%), agriculture (2.9%), forest plantations (2.0%) and forest formations (1.7%), resulting in a relative net loss of -6.9% for the period of 1985 – 2019.

Exhibiting a similar pattern, 25.4% of forest formations present in Espinhaço landscape in 1985 were converted into pasture areas (11.8%), forest plantations (5.5%), agriculture (3.6%) and savanna formations (3.6%), generating a relative net loss of -9.4% for the period of 1985 – 2019.

Representing a smaller area dimension, the rocky outcrops, which were one of the land-cover classes that suffered major percent gross loss of their original area (-63.2%), were mainly transformed into savannas (17.3%), pasture (16%), agriculture (15.3%) and grasslands (8.7%). Considering the new areas of rocky outcrops gained from 1985 to 2019, this land-cover dynamic resulted in a relative net decrease of around -30%

spatially correlated to the areas of urban-mining expansion (Figure 11). However, it is important to highlight that rare classes (*i.e.,* classes with a small proportion of area) tend to be penalized by the classification algorithm, thus presenting higher classification errors and being under estimated (Souza et al., 2020). Moreover, the rocky outcrops dynamics may be indirectly associated with the changes in the surrounding/rupicolous vegetation, which would possibly affect the rocky surface exposure (*e.g.:* the growing vegetation could cover the rocky surface, reducing its exposure and, consequently, detectability).

These results suggest that the land conversion for farming and logging purposes is a relevant change driver of native vegetation in the study area. Indeed, 19.8% and 7.3% of the total pasture area at Espinhaço Range in 2019 was originated from savanna and forest formations, respectively; while 23.6% and 22.8% of the total area occupied by forest plantations of *Eucaliptus* spp. and *Pinus* spp. in 2019 derived from savannas and forests.

Regarding the anthropogenic land-use classes, the urban areas, which maintained almost its entire initial area (*i.e.,* gross loss near 0), mainly expanded over formerly pasture areas (32.4%), agriculture lands (10.7%), savannas (7.7%) and forest formations (4.9%). On the other hand, forest formations contributed to 16.7% of 2019's total mining areas, followed by agriculture (8.9%) and rocky outcrops (4.0%).

Observing LULC temporal trajectories (Appendix S2) and comparing transitions between 1985 – 2005 and 2005 – 2019 periods, it is noticeable that some LULC classes did not show constant trends over time.

Forest plantations showed accentuated growth trend since 2004, with annual net change rate increasing by 260% from an average of 66.8 km^2/year during 1985 – 2005 to 240.7 km^2/year during 2005 – 2019. In qualitative terms, while the conversion of native forests into forest plantations of commercial species underwent a slight attenuation between 2005 – 2019 (48.5 km^2/year *vs* 50.1 km^2/year in 1985 – 2005); the conversion of pasture and savanna formations into forest plantations underwent, respectively, around a sixfold and threefold increase in the recent context (pasture: 11.7 km^2/year in 1985 – 2005 *vs* 67.5 km^2/year; savanna formation: 36.7 km^2/year in 1985 – 2005 *vs* 110.8 km^2/year in 2005 – 2019).

Pasture areas, in contrast, presented, after a rise period, a decreasing trend since 2004, especially from 2012 to 2018. This change in trajectory was mainly caused by the increased conversion of pasture areas into croplands (by average annual rates of 138.8 km^2/year in 1985 – 2005 *vs* 268.4 km^2/year in 2005 – 2019) and forest plantations (as stated above) in the last 15 years, in comparison to 1985 – 2005. Therefore, pasture shrinkage correlates to agriculture expansion observed for the same period, as well as to the forest formation late recovery (between 2010 and 2018), since deforestation driven by pasture expansion currently underwent a -50% decrease in the average annual rates (185.4 km^2/year in 1985 – 2005 *vs* 96.1 km^2/year in 2005 – 2019).

Table 6 and Figure 10 show the land-use and land-cover change processes in Espinhaço Range from 1985 to 2019, whereas Figure 11 shows the spatial patterns and magnitude (in terms of area) of the land-use/cover transitions occurred within the Espinhaço region. Transition matrices detailing the LULC changes during 1985 – 2005 and 2005 – 2019 are presented in Appendix S3.

Table 6: Transition matrix of LULC change in Espinhaço Range between 1985 – 2019 [unit: (a) %; (b) km^2].

a.

Classes		2019												
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Other Non Forest Natural Formation	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
1985	No class	100.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
	Forest Formation	0.0%	74.6%	3.6%	5.5%	0.1%	0.1%	0.0%	11.8%	3.6%	0.2%	0.1%	0.1%	0.2%
	Savanna Formation	0.0%	1.7%	81.4%	2.0%	0.5%	0.1%	0.0%	11.1%	2.9%	0.1%	0.0%	0.0%	0.1%
	Forest Plantation	0.0%	0.2%	0.1%	99.4%	0.0%	0.0%	0.0%	0.1%	0.1%	0.0%	0.0%	0.0%	0.0%
	Grassland	0.0%	0.1%	2.0%	0.7%	86.3%	0.1%	0.0%	5.8%	3.4%	0.1%	0.0%	1.5%	0.0%
	Rocky outcrop	0.0%	3.2%	17.2%	0.1%	8.7%	36.8%	0.0%	16.0%	15.3%	0.7%	0.7%	1.1%	0.2%
	Other Non Forest Natural Formation	0.0%	28.1%	19.8%	0.0%	0.0%	0.0%	31.6%	13.8%	5.7%	0.0%	0.0%	0.5%	0.5%
	Pasture	0.0%	5.0%	14.3%	1.6%	1.5%	0.0%	0.0%	68.9%	7.4%	0.9%	0.0%	0.2%	0.1%
	Agriculture	0.0%	6.5%	14.2%	1.0%	2.7%	1.1%	0.0%	21.3%	51.7%	1.0%	0.1%	0.2%	0.1%
	Urban Infrastructure	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	100.0%	0.0%	0.0%	0.0%
	Mining	0.0%	0.5%	2.4%	0.1%	0.0%	1.7%	0.0%	1.8%	3.6%	5.6%	82.2%	0.8%	1.3%
	Other Non Vegetated Area	0.0%	1.7%	3.8%	2.0%	10.4%	2.9%	0.0%	18.3%	4.3%	4.5%	0.1%	51.7%	0.2%
	River. Lake and Ocean	0.0%	3.5%	13.4%	0.7%	0.5%	0.9%	0.0%	17.5%	2.4%	0.5%	0.6%	1.2%	58.7%

b.

Classes		2019												
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Other Non Forest Natural Formation	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
1985	No class	502740.05	0.00	0.02	0.00	0.01	0.00	0.00	0.33	0.04	0.00	0.00	0.00	0.00
	Forest Formation	0.01	26173.62	1279.65	1934.09	22.66	34.08	0.37	4155.92	1278.74	73.51	36.05	25.89	63.91
	Savanna Formation	0.02	1761.16	82190.71	2001.97	524.65	133.75	0.45	11249.50	2880.98	116.13	5.79	42.80	56.06
	Forest Plantation	0.00	7.89	4.62	3459.94	0.34	0.00	0.00	2.87	1.84	0.84	0.93	0.09	0.04
	Grassland	0.00	13.10	220.01	75.82	9429.25	11.86	0.00	629.96	373.88	6.86	0.05	161.85	4.98
	Rocky outcrop	0.00	41.94	224.58	0.74	112.84	478.75	0.00	208.54	199.64	9.48	8.63	14.89	2.28
	Other Non Forest Natural Formation	0.00	0.73	0.51	0.00	0.00	0.00	0.82	0.36	0.15	0.00	0.00	0.01	0.01
	Pasture	0.02	2658.05	7624.94	826.32	821.31	24.22	0.11	36604.61	3949.64	490.97	4.75	105.19	50.08
	Agriculture	0.02	1103.62	2390.52	173.56	462.90	192.34	0.04	3590.30	8723.29	161.45	19.11	33.57	8.83
	Urban Infrastructure	0.00	0.05	0.02	0.00	0.00	0.00	0.00	0.02	0.02	604.45	0.15	0.00	0.00
	Mining	0.00	0.82	3.96	0.17	0.02	2.88	0.00	2.93	6.01	9.37	137.03	1.32	2.14
	Other Non Vegetated Area	0.00	15.02	34.85	18.01	94.70	26.24	0.00	166.47	39.06	40.68	1.33	469.11	1.90
	River. Lake and Ocean	0.00	9.58	36.62	1.93	1.43	2.44	0.00	47.76	6.49	1.37	1.61	3.28	160.23

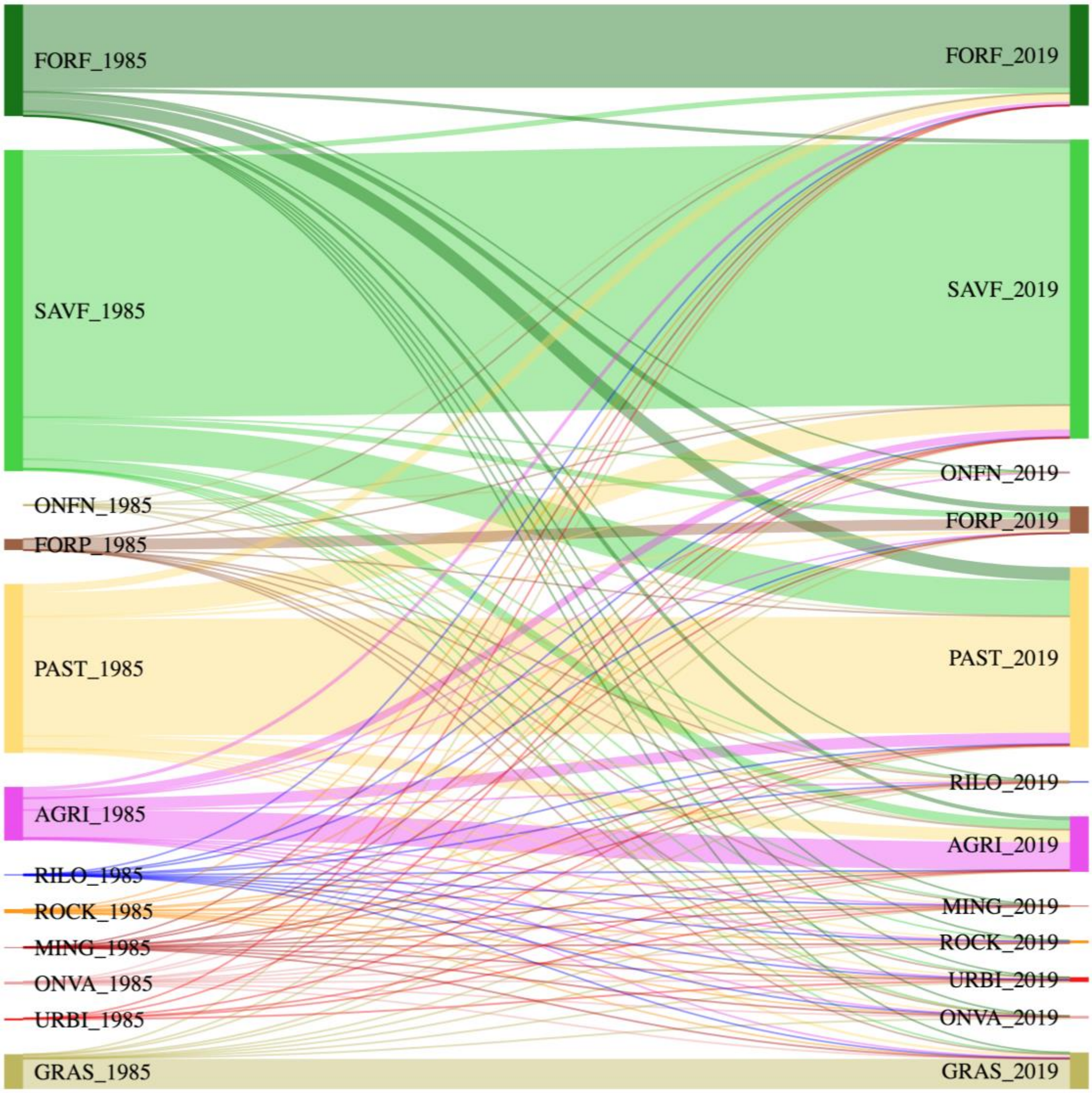

Figure 10: Transition dynamics between land-use and land-cover types in Espinhaço Range during 1985 – 2019 period. Classes are: FORF – Forest Formation; SAVF – Savanna Formation; FORP – Forest Plantation; GRAS – Grassland; ROCK – Rocky Outcrop; ONFN – Other Non-Forest Natural Formation; PAST – Pasture; AGRI – Agriculture; URBI – Urban Infrastructure; MING – Mining; ONVA – Other Non-Vegetated Area; RILO – River, Lake and Ocean.

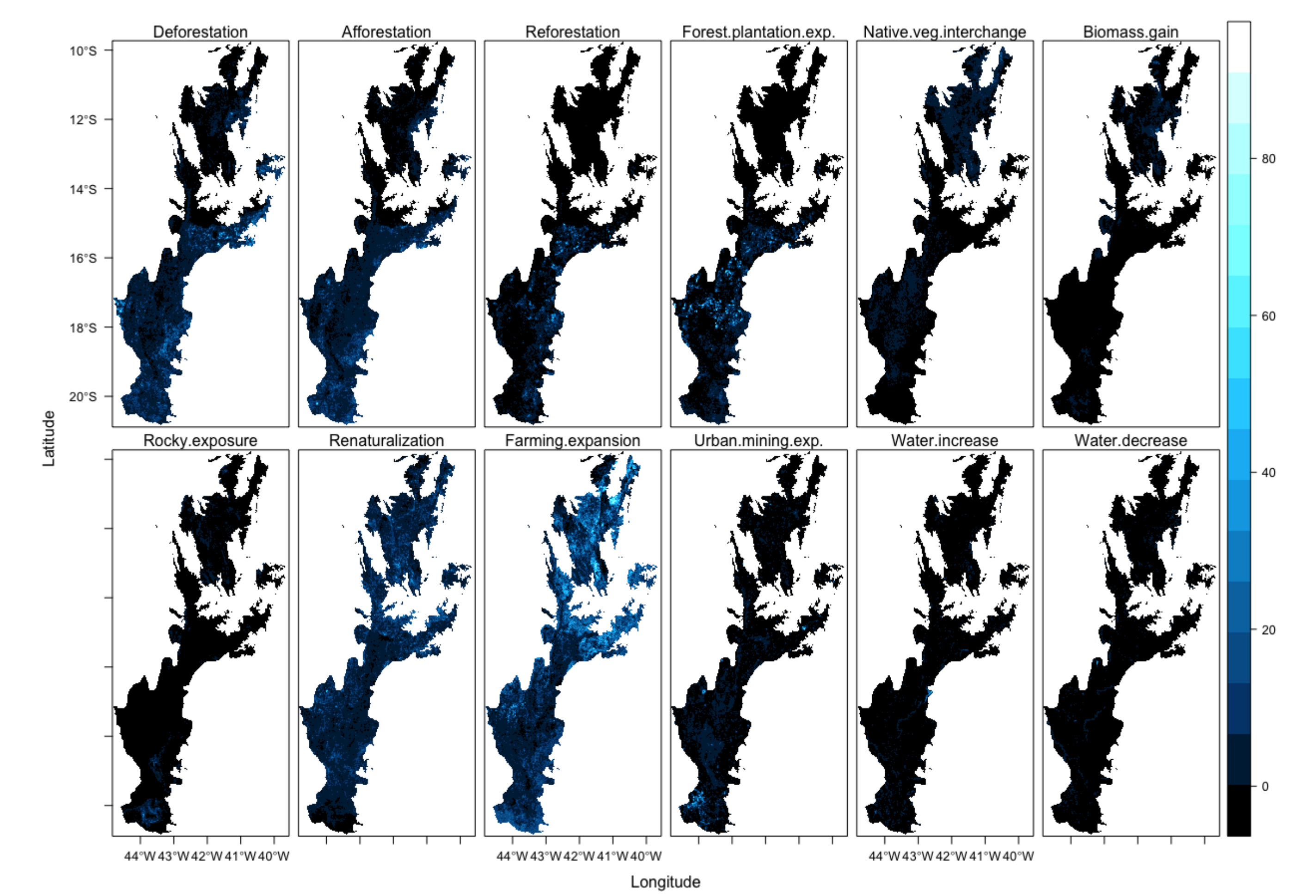

Figure 11: Land-use/cover transitions within the Espinhaço region from 1985 to 2019.

3.1.2. Espinhaço Range Biosphere Reserve

3.1.2.1. Landscape composition

Similarly to the entire Espinhaço Range, the UNESCO's Espinhaço Range Biosphere Reserve (hereafter referred to as RBSE – *Reserva da Biosfera da Serra do Espinhaço*, *i.e.,* spatial arrangement I; see **Section 2.1**) is mainly composed of forest formations (25.12%[3]), savanna formations (23.60%[3]), pasture (22.47%[3]), grasslands (13.28%[3]) and agriculture areas (6.28%[3]). Together, these five land-cover and land-use classes represented 94.87% and 90.75% of the total RBSE area in 1985 and 2019, respectively.

Figure 12 shows the landscape composition in RBSE comparing three distinct years representing the beginning, middle and end of the analyzed time-series: 1985, 2005 and 2019.

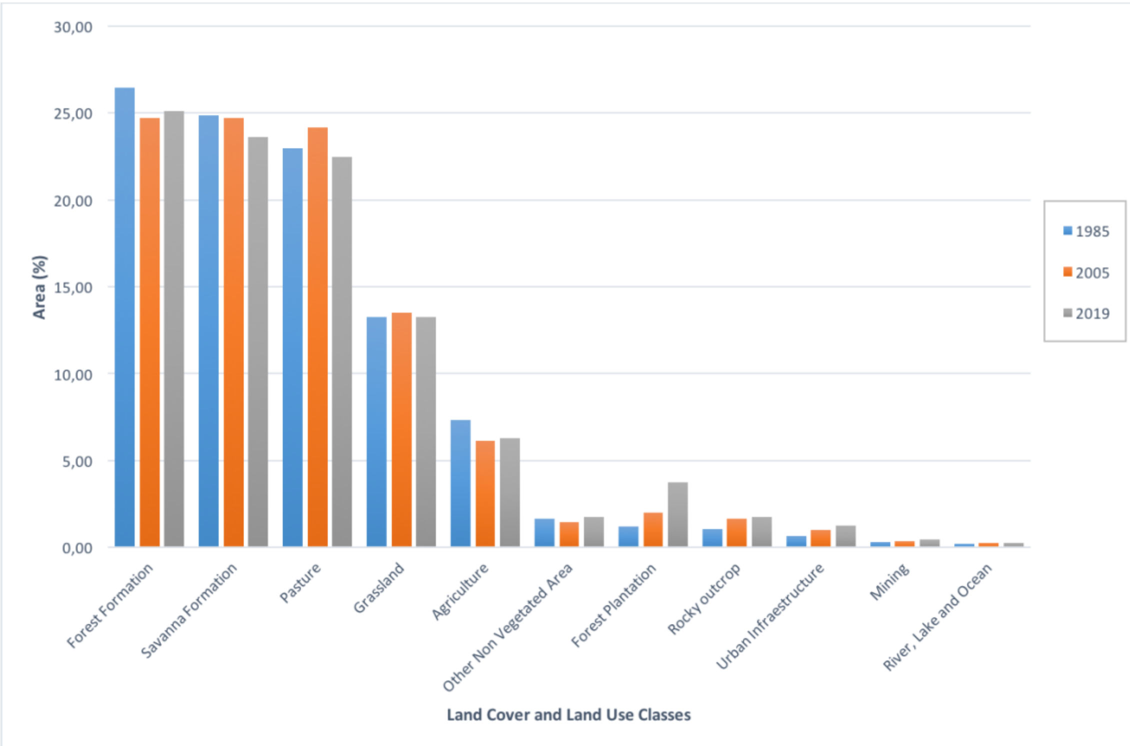

Figure 12: Land-use and land-cover composition in the Espinhaço Range Biosphere Reserve from 1985 to 2019.

[3] Area value refers to 2019 (the last one available in MapBiomas dataset for access date until August 2021)

3.1.2.2. LULC temporal and spatial variation trends

From 1985 to 2019, a total of around 2 815 km^2 (8.32% of Espinhaço Range Biosphere Reserve total area), taking into account all the land-use/cover classes changed.

Among all the land-use and land-cover types, the natural forest and savanna formations were the ones that decreased the most in this period, suffering, respectively, a total net loss of -1.35% and -1.27% relative to RBSE's total area; followed by agricultural areas (-1.04% of the total area). On the other hand, the forest plantations were the land-use that underwent a major increase (2.53% of RBSE's total area), besides being the class which presented the major relative net increase, growing 207% of its initial area during the 35 years analyzed.

Despite occupying a minor extent in relation to the RBSE's total area (<2%), other land-use and land-cover types showed an expressive relative net variation during the analyzed period. It is the case of urban infrastructure, which increased around 94% its initial area (mainly in the southern portion of RBSE, where the state capital Belo Horizonte is located); the rocky outcrops (increased around 64%); and mining areas (increased by 46% and, similarly to the totality of Espinhaço Range, mainly between 2015 and 2016). As stated in previous section, the high variability presented by the rocky outcrops (expected to present a more static behavior), may be related to classification errors associated with this class (Souza et al., 2020) and changes within the surrounding/rupicolous vegetation likely to affect the rocky surface exposure.

As can be seen in Appendix S4, some land-use/cover classes presented a more linear change trajectory over time, such as the urban infrastructure, the forest plantations and the rocky outcrops, which showed a near-constant increasing trend.

In contrast, other classes showed a more irregular change trajectory, such as the pastures (characterized by two notable "cycles" of area increase followed by decrease); the savannas and forest formations, that exhibited a declining trend with fluctuations. However, the natural forests presented a recent (*i.e.,* since 2007) expansion trend, suggesting a potential attenuation of net deforestation rates. The oscillatory behavior is also the case of the grasslands, which presented a small area's total net change, but intense change dynamics over time.

The land-use/cover net change is presented in Table 7 and Figure 13, whereas Appendix S4 shows in detail the temporal change trajectories presented by each land-use and land-cover class. The spatial distribution of the LULC classes in 1985 and 2019 is shown in Figure 14.

Table 7: Five-yearly summary of land-use and land-cover change in Espinhaço Range Biosphere Reserve from 1985 to 2019.

Class / Year	Land-Cover / Land-Use Area (%)								Total Net Change 1985-2019 (%)	Relative Net Change 1985-2019 (%)
	1985	1990	1995	2000	2005	2010	2015	2019		
Forest Formation	26.47	26.03	25.45	26.03	24.71	24.46	25.05	25.12	-1.35	-5.11
Savanna Formation	24.87	25.57	25.01	25.06	24.71	24.62	24.51	23.60	-1.27	-5.10
Pasture	22.97	22.95	24.01	22.90	24.15	23.25	22.13	22.47	-0.50	-2.18
Grassland	13.24	13.60	13.28	13.60	13.53	13.61	13.52	13.28	0.04	0.27
Agriculture	7.32	6.25	6.26	5.90	6.12	6.18	6.14	6.28	-1.04	-14.19
Other Non Vegetated Area	1.65	1.57	1.49	1.59	1.47	1.47	1.47	1.75	0.11	6.65
Forest Plantation	1.22	1.30	1.63	1.73	1.99	2.84	3.46	3.75	2.53	207.00
Rocky outcrop	1.07	1.41	1.45	1.64	1.68	1.86	1.84	1.76	0.69	64.45
Urban Infrastructure	0.65	0.76	0.86	0.96	1.02	1.12	1.24	1.26	0.61	93.78
Mining	0.32	0.33	0.34	0.34	0.35	0.33	0.38	0.46	0.15	46.21
River, Lake and Ocean	0.222	0.230	0.233	0.253	0.263	0.259	0.254	0.259	0.04	16.62

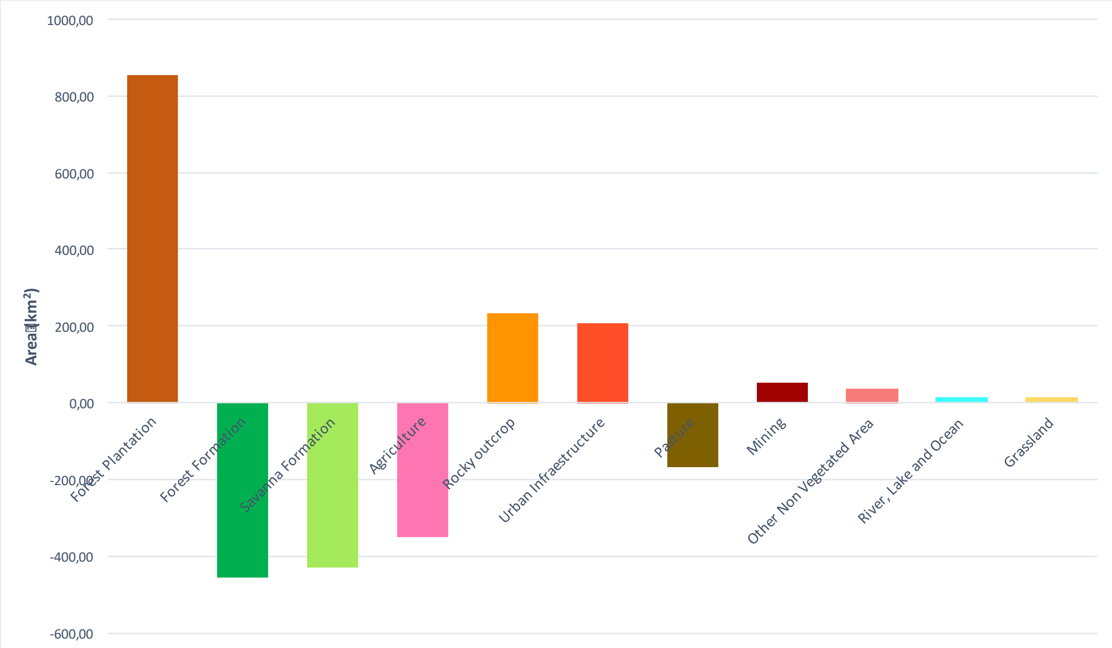

Figure 13: Land-use and land-cover net change in Espinhaço Range Biosphere Reserve during 1985 – 2019. Columns are arranged in descending order of total net change.

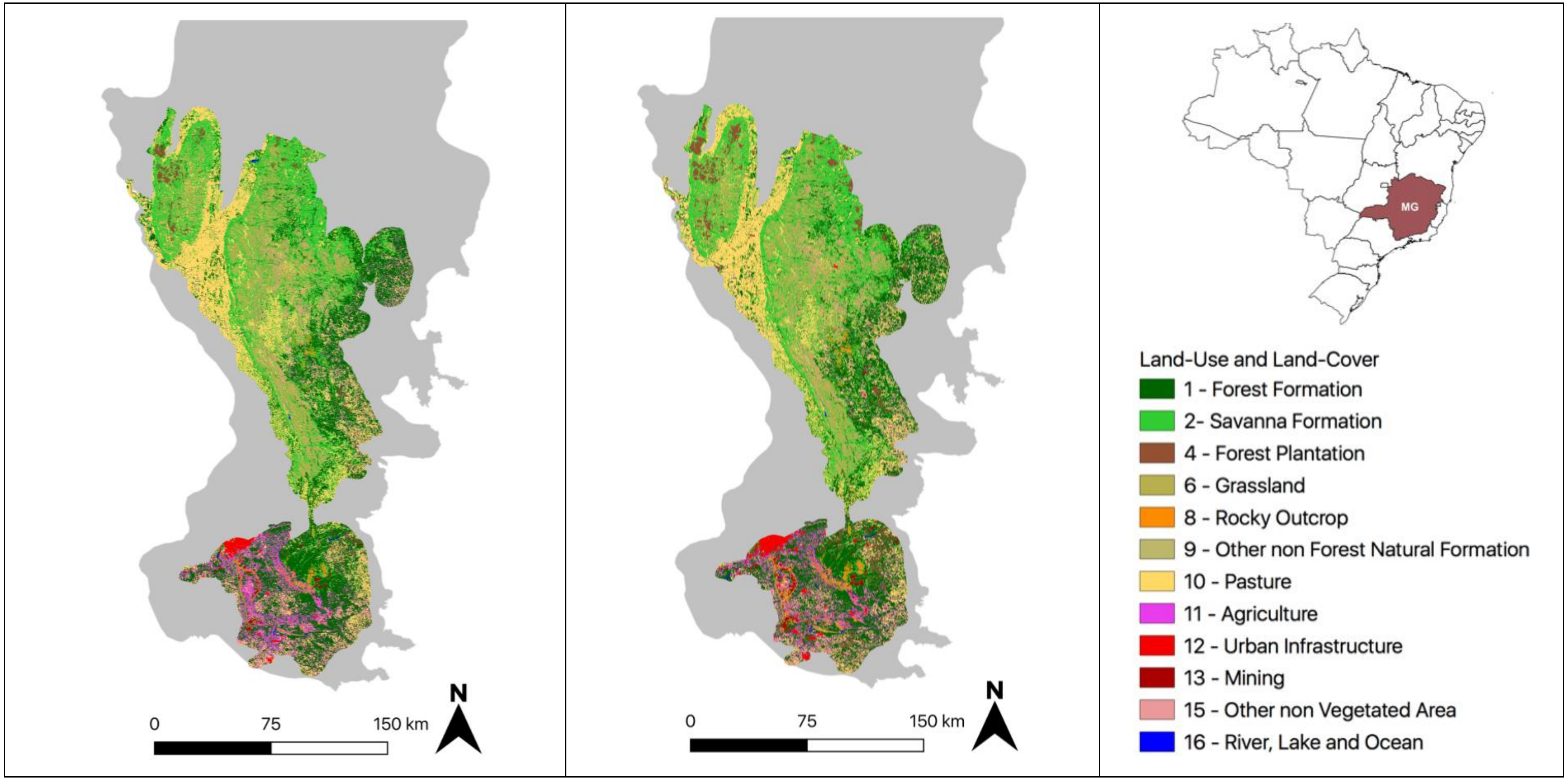

Figure 14: Spatial distribution of land-use and land-cover classes within RBSE in 1985 (left) and 2019 (right) scenarios.

3.1.2.3. LULC transition analysis

Concerning gross changes, around 21% of total RBSE area underwent land-use/cover changes, being the most expressive transitions during the 1985 – 2019 period: farming expansion (23.79%), deforestation (21.15%), afforestation (19.35%) and renaturalization of anthropic areas (13.00%), as presented in Figure 15.

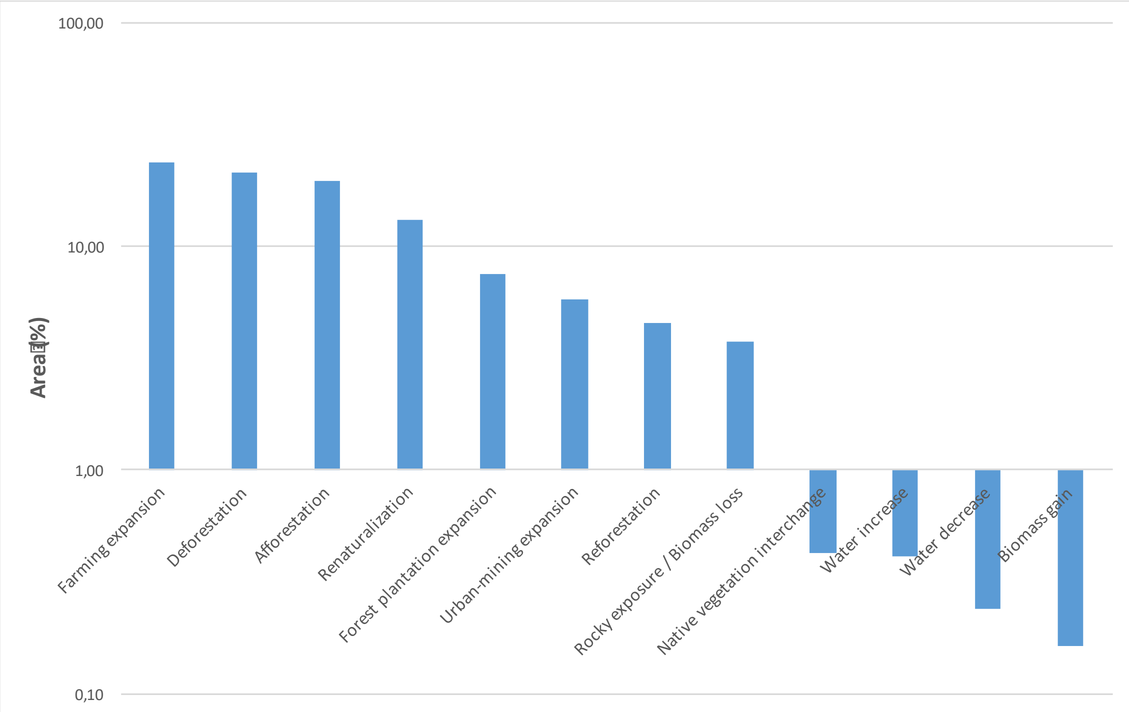

Figure 15: Land-use/cover transitions within the RBSE during 1985 – 2019.

Besides being among the classes that underwent major net changes, the forest formations, agriculture, and savanna formations were among the land-uses that showed more considerable area gross change (in km^2), as seen in Figure 16. Regarding total area values, they were only exceeded by pasture, which was the land-use class that underwent the major gross change from 1985 to 2019, although presented a less significant net change (-0.50% of RBSE's total area).

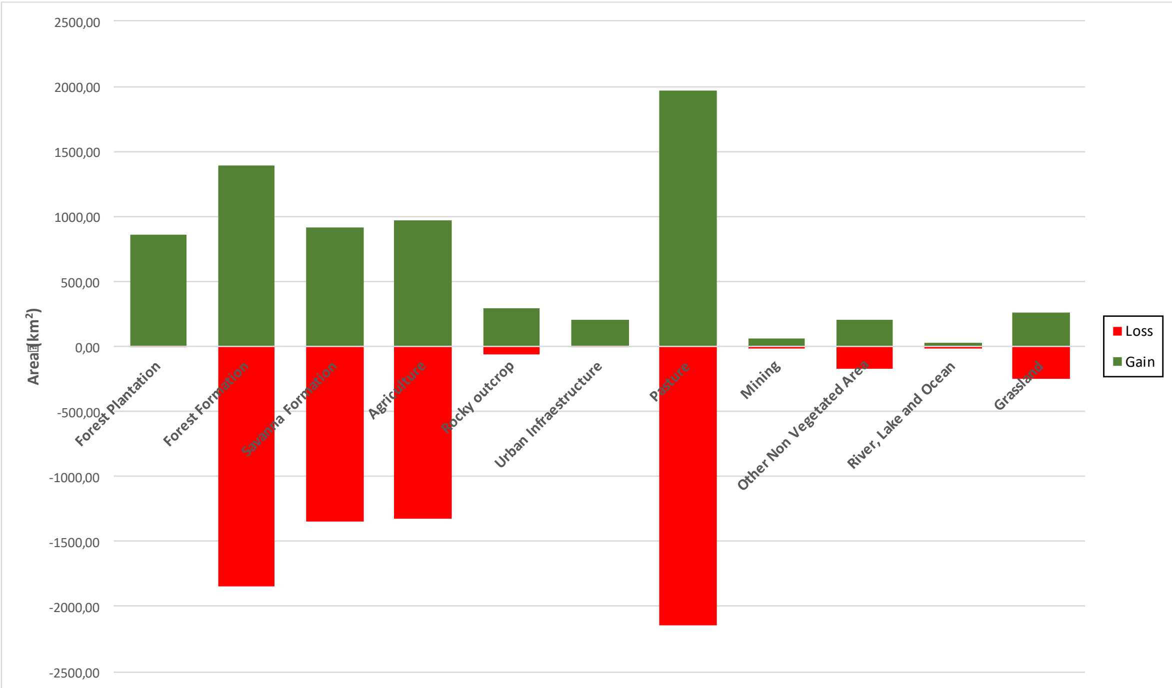

Figure 16: Land-use and land-cover gross change in Espinhaço Range Biosphere Reserve during the period of 1985 – 2019. Columns are arranged in descending order of total net change.

The forest formations, even though showing a relative net decrease of around -5%, underwent a much more expressive gross loss around -21% of their initial area from 1985 to 2019, being transformed mainly into pastureland (9.8%), agriculture (3.9%), forest plantations (3.6%) and savanna formations (2.1%).

However, it is noteworthy that the conversion of forests into pasture underwent a threefold decrease in total area between 2005 and 2019 (264 km^2, corresponding to a loss annual rate of -17.58 km^2/year); in comparison to 1985 – 2005 (822 km^2; corresponding to a loss annual rate of -39.16 km^2/year). The reduction of pasture expansion over forest formations is likely to explain the recent increasing trend observed for native forests in RBSE since 2007, besides contributing to pasture shrinking detected for the same period.

Exhibiting a similar pattern to forests, the savanna formations present at RBSE in 1985 were mostly transformed into pasture (6.3%), forest plantations (3.5%) and forest formations (3.5%), resulting in a gross loss of around -16% and a relative net decrease around -5% for the period of 1985 – 2019.

Similarly to the totality of Espinhaço Range, these results suggest that farming and logging expansion are the main drivers of native vegetation loss in the study region. Indeed, native forest and savanna formations were the primary sources for the increase of forest plantations, accounting, respectively, for 25.4% and 23.4% of this land-use class total area in 2019, as well as they were among the leading sources for pasture expansion between 1985 and 2019.

The forest plantations of commercial species, *i.e.,* the land-use type that presented the major total net change, also expanded over formerly pasture and agriculture areas, classes contributing to, respectively, 11.6% and 5.4% of forest plantations' total area in 2019.

However, the forest plantation expansion was not constant over time, as well as the main sources of new areas. After analyzing the temporal dynamics, there was a threefold increase in the net growth rates of forest plantations from 2005 to 2019 (39.7 km^2/year) compared to the 1985 – 2005 period (12.4 km^2/year). While the conversion rate of native forests into cultivated ones was slightly higher from 2005 to 2019 (10.8 km^2/year *vs* 7.4 km^2/year in 1985 – 2005); the conversion rates of pasturelands and savanna formations into forest plantations underwent a much more expressive increase in the last 15 years: a fivefold increase in pasture conversions (1.8 km^2/year in 1985 – 2005 *vs* 9.8 km^2/year in 2005 – 2019); and a threefold increase in savannas conversion (4.8 km^2/year in 1985 – 2005 vs 14.7 km^2/year in 2005-2019). In fact, instead of forest areas, savanna formations are currently the leading source for new forest plantations.

Regarding overall pasture reduction observed in the last decade (more specifically, between 2007 – 2018), it is a result of both reductions of gross gains (93.2 km^2/year in 1985 – 2005 *vs* 68.0 km^2/year in 2005 – 2019) and an increase of gross loss (-74.2 km^2/year in 1985 – 2005 *vs* -105.8 km^2/year in 2005 – 2019) of this land-use class. The increasing conversion of pasture into agricultural areas in the last 15 years (17.1 km^2/year in 1985 – 2005 *vs* 38.7 km^2/year in 2005 – 2019) played a crucial role in the declining trend observed.

Despite representing a smaller percentage of RBSE's total area, the urban expansion observed between 1985 and 2019 (more expressive between 1985 to 2005) occurred mainly through the transformation of agricultural areas (19.2% of 2019's total urban infrastructure), pasture (12.5%), forest formations (6.0%) and other non-vegetated areas

(4.1%). In addition, forest formations contributed to 19.6% of 2019's total mining areas at RBSE, followed by croplands (10.2%) and rocky outcrops (5.4%).

In summary, comparing the land-use/cover transitions patterns between 1985 – 2005 and 2005 – 2019 within the RBSE territory, there are significant differences, as seen in Appendix S5. After the RBSE creation on 2005, there was a trend to decreasing deforestation (-43.5% relative to annual rate) and increasing afforestation (+16.7% relative to annual rate), resulting in a positive balance likely related to the recovery of forest formations in the recent scenario. On the other hand, during 2005 – 2019, there was also an expressive trend to increasing forest plantations (+252% relative to annual rate), mining and urban areas (+71% relative to annual rate) and farming (+25% relative to annual rate), resulting in a higher overall rate of land-use change in the recent scenario.

Table 8 and Figure 17 show the land-cover and land-use change processes in Espinhaço Range Biosphere Reserve from 1985 to 2019, whereas transition matrices detailing the LULC changes that occurred during 1985 – 2005 and 2005 – 2019 are presented in Appendix S6. Appendix S7 and Appendix S8 show the spatial patterns and magnitude (in terms of area) of the land-use/cover transitions occurred within the RBSE during 1985 – 2005 and 2005 – 2019, respectively.

Table 8: Transition matrix of LULC change in the RBSE between 1985 – 2019 [unit: (a) %; (b) km^2].

a.

Classes		2019											
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
1985	No class	100.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
	Forest Formation	0.0%	79.4%	2.1%	3.6%	0.0%	0.3%	9.8%	3.9%	0.3%	0.3%	0.1%	0.1%
	Savanna Formation	0.0%	3.5%	84.1%	3.5%	0.2%	1.0%	6.3%	1.0%	0.1%	0.0%	0.1%	0.1%
	Forest Plantation	0.0%	1.1%	0.1%	98.4%	0.0%	0.0%	0.1%	0.1%	0.0%	0.2%	0.0%	0.0%
	Grassland	0.0%	0.0%	0.3%	0.5%	94.4%	0.0%	1.7%	0.0%	0.0%	0.0%	2.9%	0.0%
	Rocky outcrop	0.0%	2.5%	3.2%	0.2%	0.0%	83.7%	0.4%	3.0%	2.3%	2.3%	2.4%	0.1%
	Pasture	0.0%	8.1%	7.4%	1.9%	2.2%	0.1%	72.5%	6.6%	0.7%	0.0%	0.4%	0.1%
	Agriculture	0.0%	17.9%	4.2%	2.8%	0.0%	6.2%	17.6%	46.5%	3.3%	0.6%	0.6%	0.2%
	Urban Infrastructure	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	99.9%	0.0%	0.0%	0.0%
	Mining	0.0%	0.2%	0.9%	0.1%	0.0%	2.6%	0.6%	1.9%	5.0%	86.6%	1.0%	1.1%
	Other Non Vegetated Area	0.0%	0.7%	1.1%	0.3%	13.3%	3.1%	6.5%	2.0%	3.1%	0.2%	69.5%	0.2%
	River. Lake and Ocean	0.0%	2.6%	6.7%	2.3%	0.5%	0.0%	8.2%	0.1%	0.7%	0.2%	1.3%	77.3%

b.

Classes		**2019**											
		No class	**Forest Formation**	**Savanna Formation**	**Forest Plantation**	**Grassland**	**Rocky outcrop**	**Pasture**	**Agriculture**	**Urban Infrastructure**	**Mining**	**Other Non Vegetated Area**	**River. Lake and Ocean**
1985	**No class**	44357.57	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Forest Formation	0.00	7110.49	191.33	321.73	2.57	26.87	878.87	353.74	25.56	30.76	6.60	9.18
	Savanna Formation	0.00	295.04	7074.54	296.92	16.39	81.33	533.72	82.60	11.78	4.01	10.49	9.12
	Forest Plantation	0.00	4.64	0.54	406.45	0.13	0.00	0.38	0.31	0.06	0.68	0.00	0.00
	Grassland	0.00	1.11	14.24	23.55	4231.01	0.00	77.45	0.36	2.09	0.00	130.17	0.39
	Rocky outcrop	0.00	9.18	11.61	0.59	0.00	303.77	1.32	10.87	8.41	8.50	8.64	0.21
	Pasture	0.00	628.95	578.05	147.16	167.83	11.18	5633.18	511.60	53.66	3.03	34.58	4.67
	Agriculture	0.00	444.24	104.49	68.53	0.01	154.12	437.02	1151.96	82.17	15.97	14.17	3.82
	Urban Infrastructure	0.00	0.03	0.01	0.00	0.00	0.00	0.00	0.02	220.42	0.10	0.00	0.00
	Mining	0.00	0.25	1.01	0.07	0.00	2.84	0.67	2.05	5.33	93.07	1.10	1.13
	Other Non Vegetated Area	0.00	3.92	6.24	1.81	74.11	17.00	35.93	11.37	17.41	0.92	387.15	0.98
	River. Lake and Ocean	0.00	1.97	5.04	1.69	0.37	0.03	6.13	0.11	0.55	0.18	0.96	58.01

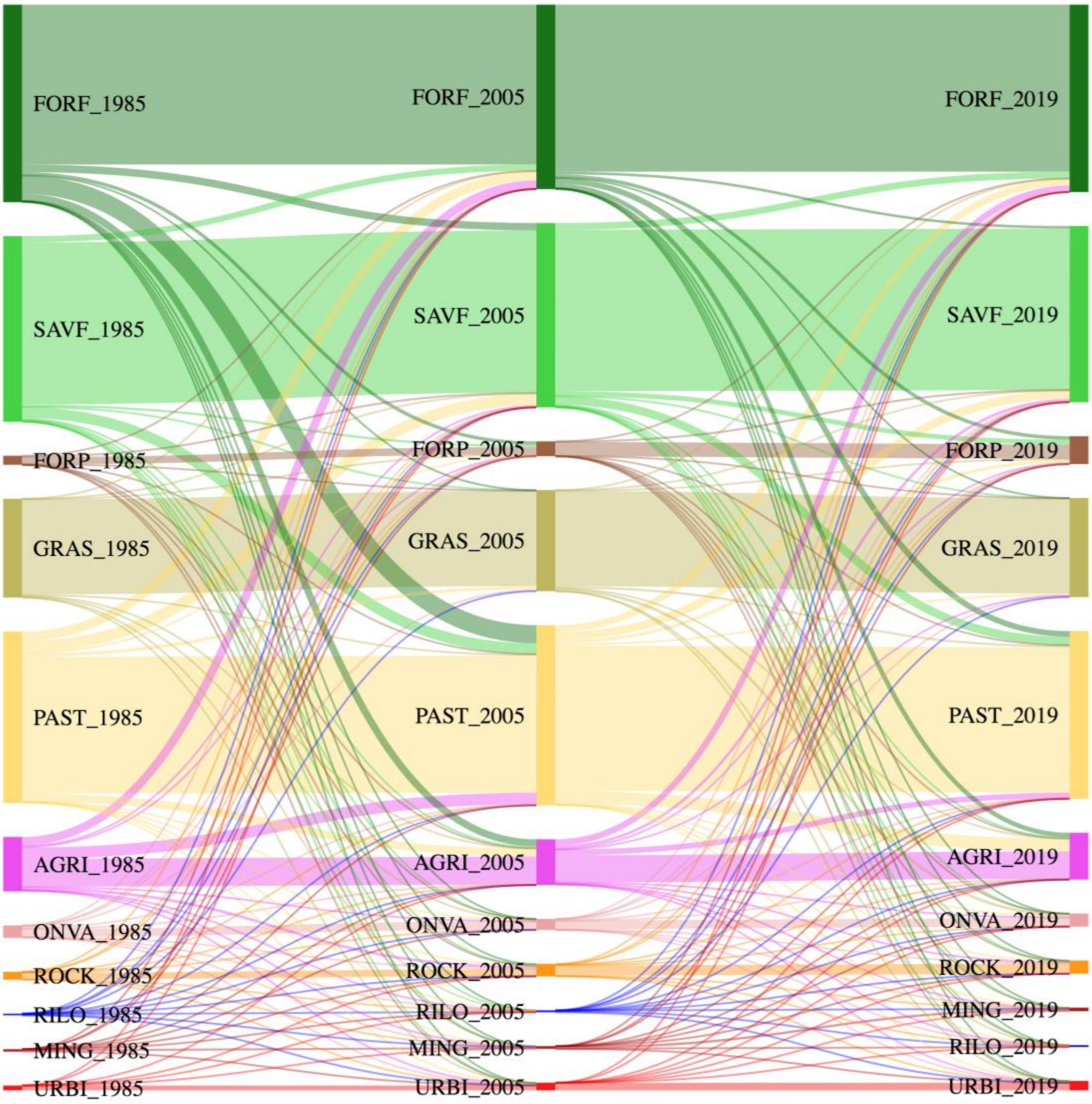

Figure 17: Transition dynamics between land-use and land-cover types in Espinhaço Range Biosphere Reserve during 1985-2005-2019 period. Classes are: FORF – Forest Formation; SAVF – Savanna Formation; FORP – Forest Plantation; GRAS – Grassland; ROCK – Rocky Outcrop; PAST – Pasture; AGRI – Agriculture; URBI – Urban Infrastructure; MING – Mining; ONVA – Other Non-Vegetated Area; RILO – River, Lake and Ocean.

3.2. Ecological Niche Modelling

3.2.1. Models performance

Cross-validation tests indicated that the ensemble models presented overall good performances for all the target species, considering both the test AUC (ranging from 0.982 to 1) and the TSS metrics (ranging from 0.885 to 1) (see Appendix S9). In other words, this means that the ENMs produced reliable predictions for all the modelled species.

3.2.2. Suitable area trend assessment

Regarding the potentially suitable habitat variation between 1985 and 2019, distinct temporal trajectories were observed among the target species. According to the downward or upward trends and their magnitude, the species were classified as presenting (i) stable habitat trend (-3% ≤ suitable area change ≤ 3%); (ii) moderate habitat loss trend (-10% ≤ suitable area change < -3%); (iii) high habitat loss trend (suitable area change < -10%); (iv) moderate habitat gain trend (3% < suitable area change ≤ 10%); or (v) high habitat gain trend (suitable area change > 10%). Figure 18 shows, as examples, two species presenting contrasting temporal trends, while Appendix S10 and Appendix S11 present the potential habitat temporal trajectories for all target species within the Espinhaço Range and the RBSE, respectively.

It is important to highlight that, according to our model spatial projections, seven species do not occur within the RBSE limits (*Bokermannohyla oxente*, *Pristimantis rupicola*, *Scinax montivagus*, *Augastes lumachella*, *Formicivora grantsaui*, *Amphisbaena acangaoba*, and *Tropidurus erythrocephalus*), resulting in a total of 37 modelled species for the totality of Espinhaço Range and 30 modelled species for the RBSE.

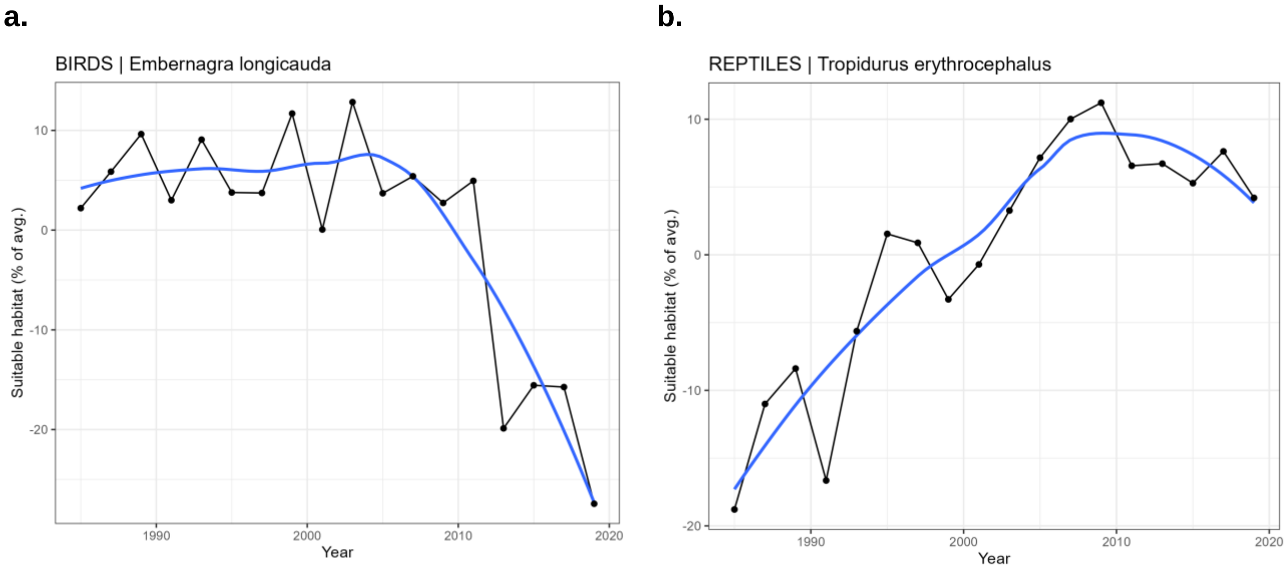

Figure 18: Example of two modelled species presenting contrasting temporal trajectories regarding potential habitat relative change (%) within the Espinhaço Range between 1985 – 2019. (a) depicts habitat loss trend for bird species *Embernagra longicauda* whereas (b) shows habitat gain trend for reptile species *Tropidurus erythrocephalus*. The % variation was calculated in relation to the average of suitable habitat for the entire period.

In general, 54% and 57% of the target species presented stable habitat trends within the totality of Espinhaço Range and the RBSE (Phase 1 limits), respectively. Around 33% of the species within both the Espinhaço and the RBSE presented temporal trends towards habitat loss, being 14% and 10% potentially affected by high levels of habitat loss (*i.e.,* > 10% in relation to the original area) within Espinhaço Range and the RBSE, respectively. The species presenting a higher magnitude of habitat loss trend were the bird species *Embernagra longicauda*, *Polystictus superciliaris*, *Campylopterus diamantinensis*, amphibian species *Boana cipoensis*, and the reptile species *Tropidurus montanus* for the Espinhaço Range; and *Embernagra longicauda, Campylopterus diamantinensis*, and *Tropidurus montanus* for the RBSE.

In turn, around 14% of the species within Espinhaço Range and 10% within the RBSE presented temporal trends towards potential habitat gain. Table 9 summarizes the change trends concerning the potentially suitable habitat of the target species within the Espinhaço and the RBSE over the last 35 years.

Table 9: Summary of species' potential suitable area trend assessment for the 1985 – 2019 period within the Espinhaço Range and the RBSE. "Avg." refers to average; "Dif." refers to difference.

Fauna group	Species	Espinhaço Range					RBSE (Phase 1)				
		Relative variation (% of avg.)			Dif. 2019-1985 (%)	Trend Class	Relative variation (% of avg.)			Dif. 2019-1985 (%)	Trend Class
		1985	2005	2019			1985	2005	2019		
Amphibians	*Aplastodiscus heterophonicus*	4.41	-1.74	2.74	-1.67	Stable	2.37	-1.58	1.44	-0.93	Stable
Amphibians	*Boana botumirim*	2.50	-0.70	2.77	0.26	Stable	2.19	-1.17	0.12	-2.07	Stable
Amphibians	*Boana cipoensis*	-7.83	0.95	-19.45	-11.61	High loss	-7.77	1.55	-17.64	-9.87	Moderate loss
Amphibians	*Bokermannohyla alvarengai*	-1.22	0.16	-8.80	-7.58	Moderate loss	-1.10	-0.12	-6.79	-5.69	Moderate loss
Amphibians	*Bokermannohyla martinsi*	0.22	0.97	2.48	2.26	Stable	-0.56	0.35	1.58	2.14	Stable
Amphibians	*Bokermannohyla nanuzae*	2.21	-3.50	4.68	2.47	Stable	2.07	-1.72	0.89	-1.18	Stable
Amphibians	*Bokermannohyla oxente*	4.75	4.15	6.58	1.83	Stable	-	-	-	-	-
Amphibians	*Bokermannohyla saxicola*	-0.30	-0.70	1.68	1.97	Stable	1.20	-0.94	1.20	-0.01	Stable
Amphibians	*Crossodactylus trachystomus*	-1.21	0.77	-1.79	-0.59	Stable	0.19	0.02	-2.59	-2.77	Stable
Amphibians	*Hylodes uai*	6.11	-2.56	18.49	12.38	High gain	5.65	-2.49	11.38	5.73	Moderate gain
Amphibians	*Leptodactylus camaquara*	-1.43	0.32	1.61	3.04	Stable	-0.20	-0.48	1.16	1.35	Stable
Amphibians	*Odontophrynus juquinha*	-1.73	0.45	-10.51	-8.78	Moderate loss	-2.11	0.62	-7.22	-5.11	Moderate loss
Amphibians	*Phasmahyla jandaia*	0.02	0.03	0.86	0.84	Stable	0.34	-0.14	0.39	0.04	Stable
Amphibians	*Physalaemus orophilus*	11.33	-5.46	6.64	-4.69	Moderate loss	9.41	-3.25	3.41	-5.99	Moderate loss
Amphibians	*Pithecopus megacephalus*	-1.65	0.26	0.74	2.38	Stable	0.19	-1.65	2.10	1.91	Stable
Amphibians	*Pristimantis rupicola*	-0.38	2.37	-1.33	-0.95	Stable	-	-	-	-	-
Amphibians	*Proceratophrys cururu*	-4.70	-0.09	-1.87	2.83	Stable	-4.11	0.30	-1.37	2.74	Stable
Amphibians	*Pseudopaludicola mineira*	-1.92	-0.10	-0.80	1.12	Stable	-1.44	-0.28	-0.29	1.15	Stable
Amphibians	*Scinax curicica*	-0.11	0.19	-1.01	-0.90	Stable	0.30	-0.14	-1.29	-1.59	Stable

Fauna group	Species	Espinhaço Range					RBSE (Phase 1)				
		Relative variation (% of avg.)			Dif. 2019-1985 (%)	Trend Class	Relative variation (% of avg.)			Dif. 2019-1985 (%)	Trend Class
		1985	2005	2019			1985	2005	2019		
Amphibians	*Scinax machadoi*	-0.62	-0.60	-1.90	-1.28	Stable	-0.08	-1.85	-2.15	-2.07	Stable
Amphibians	*Scinax montivagus*	5.49	1.08	-3.28	-8.77	Moderate loss	-	-	-	-	-
Amphibians	*Thoropa megatympanum*	-1.60	0.80	-0.94	0.66	Stable	-1.65	-0.37	-2.35	-0.70	Stable
Birds	*Asthenes luizae*	-0.67	2.22	0.01	0.68	Stable	-1.39	0.69	-1.74	-0.34	Stable
Birds	*Asthenes moreirae*	-2.78	-1.59	-9.13	-6.35	Moderate loss	-6.57	-0.71	-15.99	-9.41	Moderate loss
Birds	*Augastes lumachella*	3.92	0.95	-1.75	-5.66	Moderate loss	-	-	-	-	-
Birds	*Augastes scutatus*	-0.38	0.36	-7.57	-7.20	Moderate loss	-0.36	0.09	-5.68	-5.32	Moderate loss
Birds	*Campylopterus diamantinensis*	-2.01	0.39	-20.04	-18.03	High loss	-1.73	-0.27	-15.13	-13.40	High loss
Birds	*Cinclodes espinhacensis*	-0.72	0.46	-1.75	-1.03	Stable	-0.45	0.26	-1.80	-1.35	Stable
Birds	*Embernagra longicauda*	-1.45	-0.01	-30.03	-28.58	High loss	-2.44	-1.68	-22.05	-19.61	High loss
Birds	*Formicivora grantsaui*	-7.02	9.17	4.18	11.20	High gain	-	-	-	-	-
Birds	*Polystictus superciliaris*	-4.87	6.24	-27.30	-22.43	High loss	-3.11	-0.21	-13.16	-10.05	Moderate loss
Reptiles	*Amphisbaena acangaoba*	0.41	0.66	0.82	0.41	Stable	-	-	-	-	-
Reptiles	*Eurolophosaurus nanuzae*	-0.52	-0.40	24.58	25.10	High gain	2.48	-2.94	12.11	9.62	Moderate gain
Reptiles	*Hydromedusa maximiliani*	3.40	-0.76	2.48	-0.92	Stable	1.52	-0.84	3.13	1.61	Stable
Reptiles	*Psilops paeminosus*	6.41	41.86	13.40	6.99	Moderate gain	-7.34	4.92	-0.70	6.63	Moderate gain
Reptiles	*Tropidurus erythrocephalus*	-20.70	4.65	1.75	22.45	High gain	-	-	-	-	-
Reptiles	*Tropidurus montanus*	-9.33	6.42	-20.94	-11.61	High loss	-5.22	-1.54	-16.31	-11.09	High loss

There were significant trend dissimilarities among the fauna groups assessed (*i.e.,* amphibians, birds and reptiles). In this context, the birds presented the most consistent trend of potential habitat loss for the period analysed. Around 71% of the bird species within the RBSE presented habitat loss trend, as well as 66% of the bird species within the entire Espinhaço Range, from which 33% (*i.e.,* 3 species) were potentially affected by high levels of habitat loss (Table 9). Moreover, the birds comprised the set of species that underwent a higher magnitude of potential habitat losses (namely, in descending order: *Embernagra longicauda*, -28.58%[4] relative variation; *Polystictus superciliaris*, -22.43%[4] relative variation; and *Campylopterus diamantinensis*, -18.03%[4] relative variation) (Table 9). In contrast, one bird species with occurrence restricted to the northern part of the Espinhaço Range presented a moderate trend towards potential habitat gain: *Formicivora grantsaui* (+11.20% relative change).

Among the amphibians, most species (73% and 74% for Espinhaço and the RBSE, respectively) presented a stable trend concerning the potentially suitable areas during the 1985 – 2019 period. Around 20% of the amphibians underwent potential habitat loss within both the Espinhaço Range and the RBSE, being 5% (*i.e.,* one species, namely *Boana cipoensis*, with -11.61% relative variation) potentially affected by high levels of habitat loss considering the totality of Espinhaço limits (Table 9). On the other hand, one species (*Hylodes uai*) presented potential habitat gains, with moderate magnitude within the RBSE (+5.73%) and high magnitude within the totality of Espinhaço Range (+12.38%).

Finally, among the reptiles, 50% of the species presented temporal trends towards potential habitat gain within both the entire Espinhaço Range and the RBSE, being two species potentially affected by high magnitude gains considering the Espinhaço extent (*Eurolophosaurus nanuzae*, with +25.10% relative variation; and *Tropidurus erythrocephalus*, with +22.45% relative variation) (Table 9). In contrast, one reptile species underwent a high magnitude potential habitat loss within both geographical extents analysed (*Tropidurus montanus*, with -11.61% loss for the Espinhaço Range).

In general, the temporal trajectories and potential habitat trends presented by each target species were similar between the spatial extents considered in this study (*i.e.,* the totality of Espinhaço Range and the RBSE Phase 1 limits), with lower magnitude effects

[4] Relative variation values for the totality of Espinhaço Range

(comprising both loss and gain trends) being observed within the RBSE, with exception for *Physalaemus orophilus* and *Asthenes moreirae*. Another interesting pattern observed concerns many inflection points of the species' trend curves around 2005 (*i.e.,* the RBSE's creation year). However, this temporal pattern was also observed for the total Espinhaço extent, suggesting a likely association with land-use/cover changes affecting the entire region rather than local effects of the Biosphere Reserve establishment.

3.2.3. Species richness assessment

The assessed fauna groups (*i.e.,* birds, amphibians and reptiles) presented distinct spatial patterns of species richness over the Espinhaço Range, as shown in Figure 19. While birds and amphibians displayed clear hotspot areas, partially overlapped between these groups and correlated with higher altitude areas (in general, >1,300m), the set of modelled reptiles presented a more spatially scattered species richness over the Espinhaço territory, suggesting these are likely more generalists regarding their environmental requirements.

Based on the spatial patterns of birds and amphibians' potential richness distribution, it is possible to identify two main biodiversity hotspots within Espinhaço Range (see birds and amphibians' richness spatial distributions in Figure 19; for a detailed view of hotspots, see Appendix S12): (i) the southern part of Espinhaço Range (central region of Minas Gerais state); and, (ii) especially for the birds, the Chapada Diamantina region, northern Espinhaço Range (central Bahia state).

The southern hotspot is encompassed by the Phase 1 limits of the Espinhaço Range Biosphere Reserve (established in 2005), yet there are high richness areas for birds and amphibians classified as RBSE's buffering zones and therefore not fully protected. In addition, we highlight the spatial overlap between the southernmost part of this hotspot and the mining region of Iron Quadrangle (Figure 20). On the other hand, the northern hotspot is entirely outside the RBSE's boundaries (even considering its Phase 2 expansion) and, consequently, outside its protection zone. The Appendix S13 presents the potential species richness distribution within the RBSE, Phase 1 limits.

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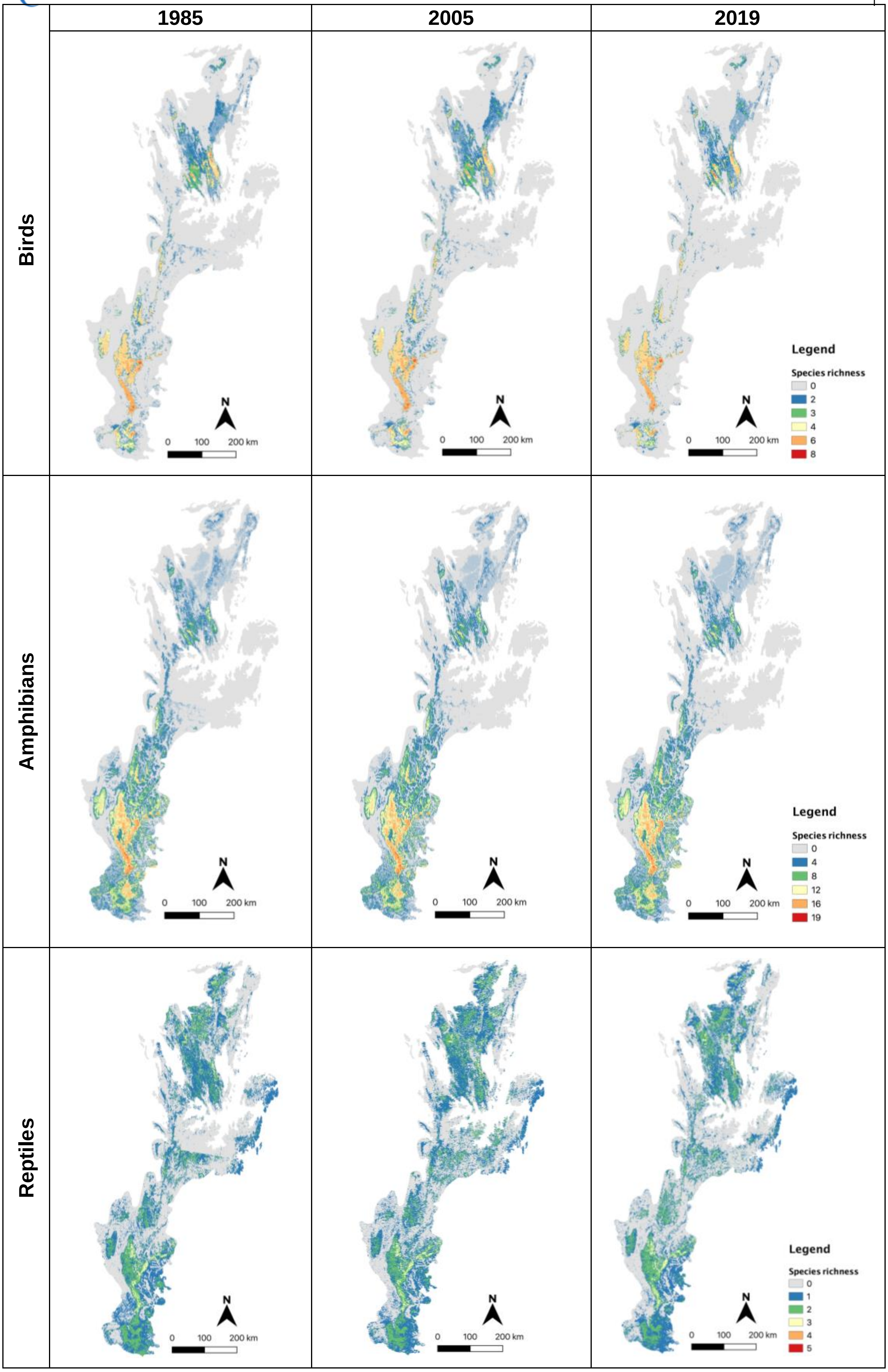

Figure 19: Temporal change of the species richness of birds, amphibians and reptile species for the Espinhaço Range considering 1985, 2005 and 2019.

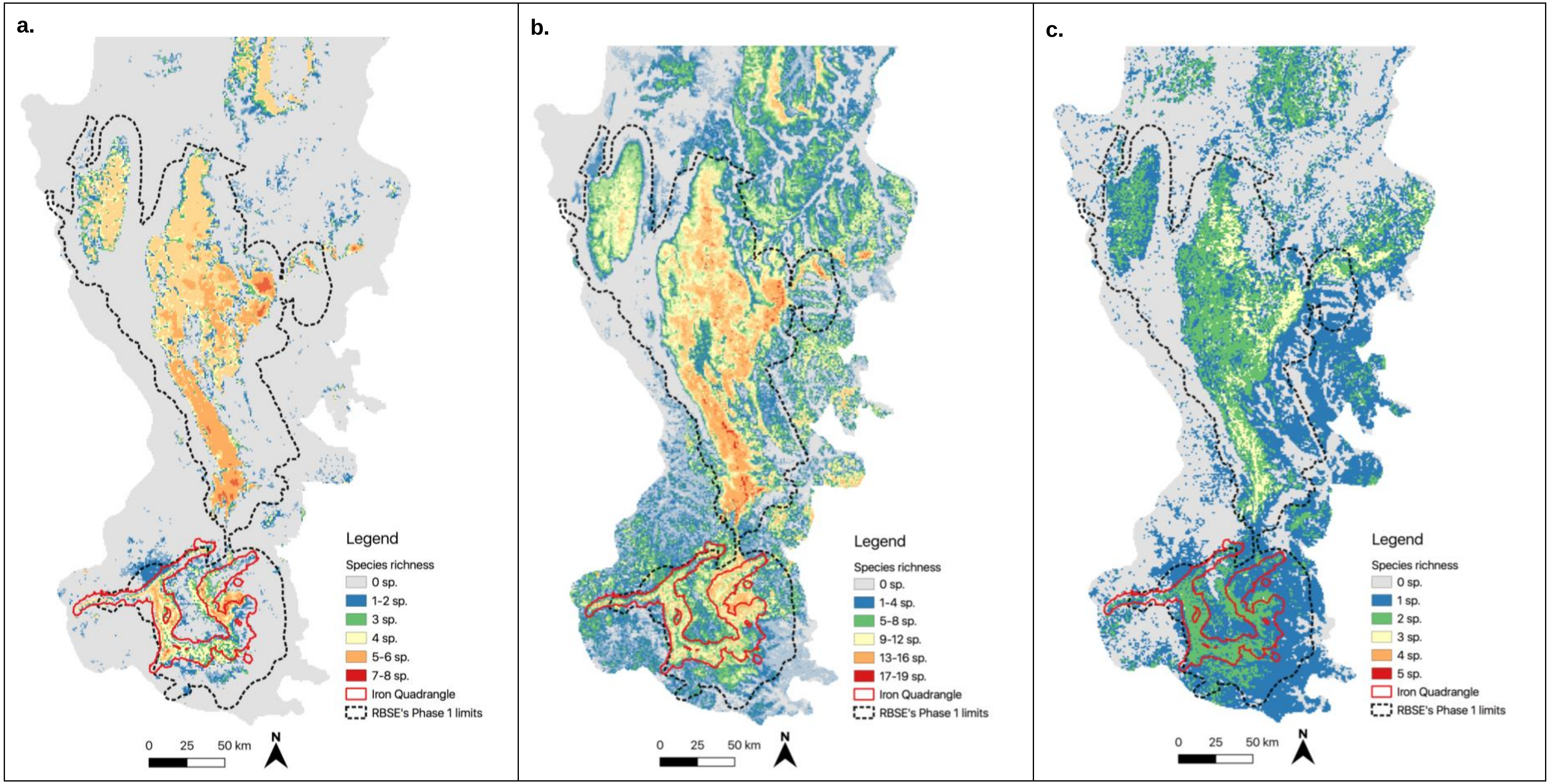

Figure 20: Spatial overlap between the Iron Quadrangle and the southernmost part of species hotspots: (a) birds; (b) amphibians; (c) reptiles. The maps display the current (2019) species richness distribution.

Despite the spatial aggregation of species richness (mainly for birds and amphibians) and the fact that there are species spatially restricted to some parts of the Espinhaço Range (*e.g.,* northern *vs* southern Espinhaço), it is noticeable the connectivity between the areas with higher species diversity across the Espinhaço extent (*i.e.,* the areas with higher species richness are spatially connected through a gradient of lower species richness areas). However, for the birds, the results show a gradual connectivity loss between the potential suitable areas located in the central region of Espinhaço Range from 1985 to 2019 (Figure 19).

Concerning the species richness changing trends within the Espinhaço Range between 1985 and 2019, the results demonstrate potential loss for all the assessed fauna groups, yet with low overall magnitudes (Figure 21). In terms of spatial patterns, the areas presenting higher trends towards species richness loss tend to be on the margins of the higher altitude areas where the biodiversity hotspots are located. This spatial pattern suggests that the mountaintops may consist of a fauna refuge, hindering and buffering the anthropic impacts of land-use/cover changes. In addition, these areas showed a spatial aggregation pattern, forming confined zones with a strong tendency to biodiversity loss within the Espinhaço Range. (Figure 21).

Overall, there was no clear spatial overlap between the areas presenting potential species richness loss and a specific land-use/cover transition within the Espinhaço Range. In general, for all the fauna groups assessed, we observed spatial coincidence among the localities showing higher species richness loss with deforestation, farming expansion, forest plantation development and reforestation (*i.e.,* the conversion of native forests into exotic ones). Notwithstanding, the potential habitat and species richness loss seem to result from multiple and simultaneous land-use/cover transition processes with complex spatiotemporal patterns.

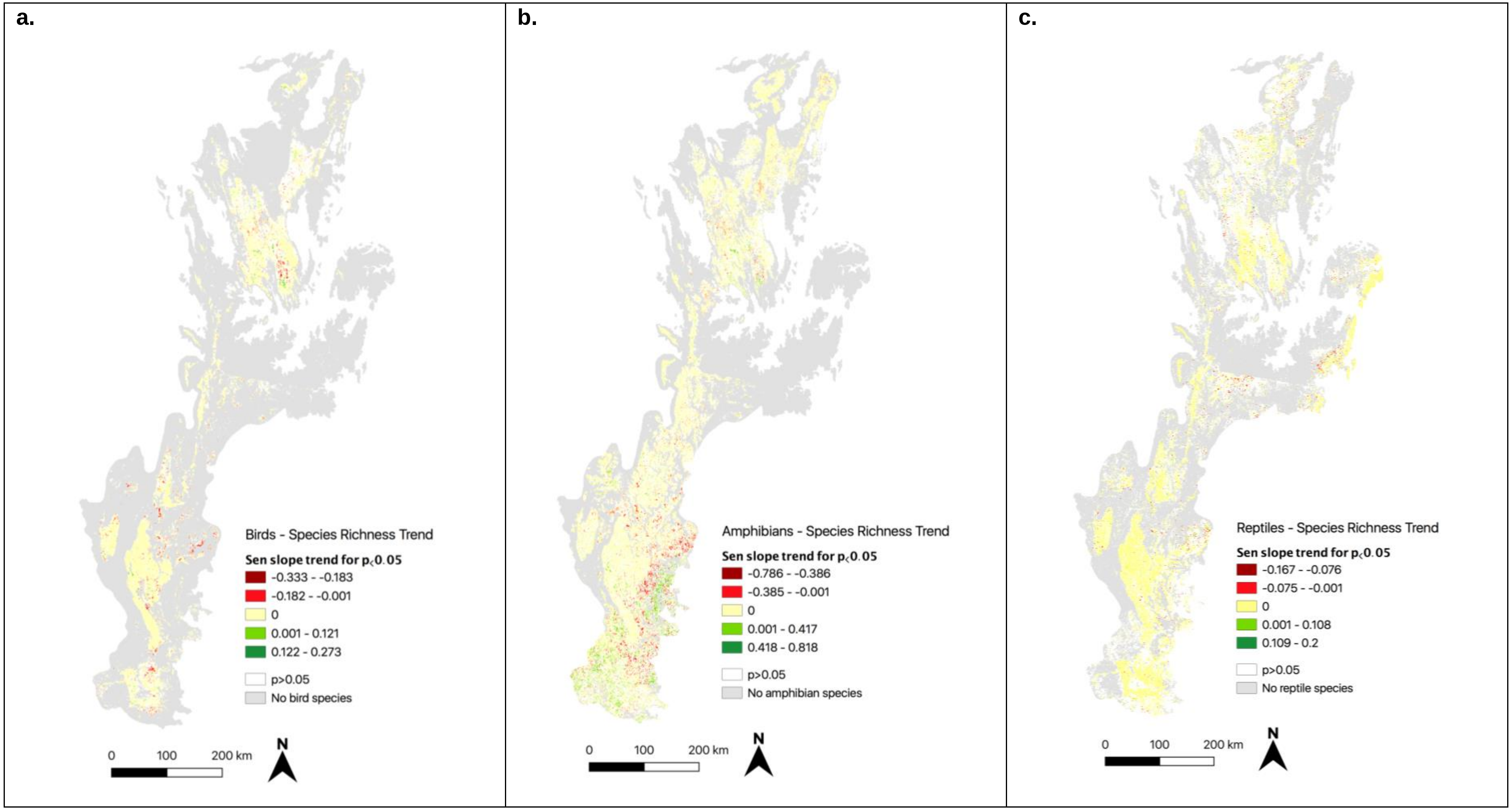

Figure 21: Temporal changes in potential species richness from 1985 to 2019 within the Espinhaço Range for (a) birds, (b) amphibians and (reptiles). Positive Sen's Slope values indicate potential increase of species richness; and negative Sen's Slope values indicate potential decrease of species richness.

4. Discussion and Conclusion

4.1. Land-Use and Land-Cover Assessment

Results summary

- Forest and savanna formations presented decreasing temporal trends from 1985 to 2019 within both the RBSE and the totality of Espinhaço Range.
- In contrast, forest plantations, urban and mining areas presented increasing temporal trends from 1985 to 2019 within both the RBSE and the entire Espinhaço Range.
- Gross changes significantly exceed net changes for some land-use/cover classes (*e.g.,* forests and savannas).
- Farming and logging expansion are the main drivers of native vegetation loss in the study region (Espinhaço Range and RBSE).
- An expressive trend to increasing rates of forest plantations expansion was observed in the recent scenario (*i.e.,* 2005 – 2019 period) both within the RBSE (+220%) and the Espinhaço Range (+260%).
- In contrast, a trend to decreasing pasturelands was observed in the recent scenario (*i.e.,* 2005 – 2019 period) within both RBSE and Espinhaço Range.
- The lower conversion rates of forests into pasturelands during 2005 – 2019 (*ca.* -50%), in comparison with 1985 – 2005 period, is likely related to the recent forest recovery observed for both RBSE and Espinhaço Range.
- Mining impacts on land-use/cover are disproportionate to their direct spatial extent, since mining expansion is usually correlated to the increase of forest plantations and urban areas.
- The institution of the RBSE on 2005 did not change the increasing trajectories of forest plantations, urban-mining and farming areas. However, it may have positively influenced the recent forest recovery trend.
- Although the effects of the RBSE creation on land-use/cover dynamics seem diffuse considering its total area, more expressive outcomes may be locally restricted to RBSE's specific zones.

Discussion

The results show that the Espinhaço Range landscape underwent intense land-use/cover change dynamics in the last 35 years, encompassing extensive anthropic occupation of natural formations.

Land-use change is affecting biodiversity across the planet mainly through promoting habitat loss and fragmentation (Newbold et al., 2015). The reduction of available habitat area and spatial configuration change of remaining habitats have been the main drivers of biodiversity decline in terrestrial ecosystems over the last century (Fletcher et al., 2018; Newbold et al., 2015; Pfeifer et al., 2017), having their adverse effects enhanced by synergistic interaction with other factors like climate change (Mantyka-pringle, Martin, & Rhodes, 2012). Indeed, studies point out the fact that many species currently live in fragmented patches of degraded habitat susceptible to threats from the adjacent anthropogenic matrix (Fletcher et al., 2018; Pfeifer et al., 2017).

In our Study Area, results evidence that, for some LULC classes, gross variation significantly exceeds net variation, such as observed for forest formations, whose gross loss between 1985 and 2019 reached around -25% and -21% of their initial area for Espinhaço Range and RBSE, respectively; and savanna formations, which underwent a gross loss around -17% and -16% for Espinhaço Range and RBSE, respectively.

Even considering partial natural regrowth/recovery offsetting the gross habitat loss, this is a relevant issue from a conservation perspective, since primary and secondary habitats do not perform similar ecological functions (*e.g.,* abiotic conditions and resources provision, carbon stock; Tomlinson et al., 2018) or have similar conservation value. Hence, replacing primary by secondary habitats might frequently result in extinction risk for less tolerant species and impoverished biotic communities (Harris & Pimm, 2004; Gardner et al., 2008; Cava et al., 2017).

Thus, considering that habitat recovery is usually a long-term process (Cava et al., 2017), before focusing on regeneration strategies, the priority conservation goals must protect the remaining primary forests, savannas, and other native vegetation types from degradation pressures.

In this context, the main factors driving native vegetation loss in the Study Area were farming and forest cultivation expansion, considering both RBSE and the entire Espinhaço Range.

Traditional extensive farming is historically practiced in the region, and, recently, it coexists with intensive cattle raising and agriculture (Almada et al., 2016; Neves et al., 2016; Fernandes et al., 2018). The recent pasture retraction observed for the Espinhaço Range and the RBSE (since 2004 and 2007, respectively), which is associated with positive effects from the recovery of forest formations, may reflects a larger scale (at national and global levels) stabilization or decreasing trend drove by herd intensification. This process enabled production to increase, in animal units, on the same pasture area (Blaustein-Rejto, Blomqvist, McNamara, & De Kirby, 2019; Lapola et al., 2013; Parente & Ferreira, 2018). On the other hand, the land-use intensification for cattle ranching also poses socioenvironmental harms, for instance contributing to soil erosion and the consolidation of land ownership inequality that triggers rural-urban migrations and urbanization increase (Lapola et al., 2013). Moreover, in light of land rent theory, studies demonstrate that farming intensification tends to promote expansion rather than contraction of grazing land, despite Brazil is so far experiencing the opposite effect (Kaimowitz & Angelsen, 2008; Lapola et al., 2013).

In contrast, agriculture presented expansion waves in recent years, largely owing to pasture conversion into croplands. The intense fluctuations that marked these land-use dynamics are possibly related to the global economic and food crisis (2007 – 2009) on agricultural production, as hypothesized by Winkler et al. (2021).

The upward trend observed for the forest plantations, in turn, is directly associated with the development of wood-demanding industries in the region, mainly the cellulose industry and steelmaking industry which currently rely on vegetal charcoal (Barbosa et al., 2020; AMS, 2009 *apud* Sonter et al., 2014). Indeed, Minas Gerais, where southern Espinhaço Range is located, possesses the most extensive forest plantation area in Brazil, with forestry promotion programs playing an essential role in this sector development (Basso et al., 2012).

If, on the one hand, the increase of forest plantations for wood supply means a decrease in native wood consumption (Martins et al., 2021) and, consequently, the deforestation aiming wood supply; on the other hand, forest plantation expansion over native forest formations consists in a growing driver of deforestation.

Besides promoting habitat loss and fragmentation (mainly of forest and savannah formations), forest plantations of exotic species (predominantly *Eucalyptus* spp.), often in monoculture arrangements, poses other environmental issues, including hydrological impacts due to excessive water consumption regardless of seasonality (Dye &Versfeld, 2007; Alvarez-Garreton et al., 2019); soil erosion and fertility problems; microclimatic effects; impoverishment of biological communities; and increased biological invasion threat; which may represent long-lasting ecosystem changes (Bull et al., 2006; Dodet & Collet, 2012).

Mining, although presenting a smaller absolute area in the landscape, is worthy of further mention due to its economic and historical relevance to the local context. The discovery of large gold deposits in Espinhaço in the 1700s is related to the Portuguese colonization of this region, triggering a mineral extractive tradition persistent until nowadays (Neves et al., 2016), as noticed by the steady increasing trend observed for mining areas from 1985 to 2019.

Despite the less significant spatial extent, it is important to consider that the indirect effects associated with mining activities may surpass the area directly occupied by mines. Mining expansion is correlated to the rise of forest plantations (for charcoal provision) and to urban development (driven by increased labor opportunities and, in already highly urbanized areas, land competition between mining companies and urban developers) (Sonter et al., 2014). Furthermore, considering that mineral extraction depends on the underlying geology, mining expansion is driven by lithological aspects an, in the Espinhaço context, it tends to concentrate on specific ecosystem (*i.e.,* rupestrian grasslands) which geographically overlap the iron ore deposits associated with rocky outcrops (Fernandes et al., 2018; Sonter et al., 2014). Indeed, Fernandes et al. (2018) estimate that about 18% of rupestrian grasslands are currently under direct and indirect (considering a 5 km buffer) influence of mining and predict that the affected area by this activity may reach 60% of this ecosystem in the following decades, assuming mining expansion (*i.e.,* the authorization of all mines currently under licensing) associated with climate change scenarios. Thus, mining impacts on native land-cover are largely disproportionate to their spatial extent, as Sonter et al. (2014) stated.

The main environmental issues linked to mining, especially open-pit mining usually practiced in this region, comprise the removal of ironstone outcrops and their associated biota, as well as the destruction of springs and drainage systems, streams siltation, toxic waste generation, potential underground and surface water contamination (Neves et al., 2016) among others. It is worth mentioning that the mining sector affects not only biodiversity, but also threatens local traditional communities and their cultural heritage. Moreover, large mining enterprises settled in the region have worsened land use and ownership conflicts, increased real estate speculation and displaced traditional and rural populations (Almada et al., 2016; Neves et al., 2016).

Interestingly, a considerable part (73%) of Espinhaço's total mining area in 2019 is located within the Espinhaço Range Biosphere Reserve, despite its smaller spatial extent in comparison to the entire Espinhaço range. Indeed, the RBSE presented a slightly higher total net change (49.69 km^2) than the entire Espinhaço Range (48.77 km^2) during 1985 – 2019 period, although the absolute area occupied by mining is obviously larger in the totality of Espinhaço (~215km^2) that in the RBSE (~157km^2). This expressive mining area within RBSE is likely associated with the Iron Quadrangle (*i.e., Quadrilátero Ferrífero*), one of the world's most crucial mineral regions[5]. However, it also sheds light on the complex dilemmas regarding the economic exploitation in this protected area. According to IABS & RBSE (2017) , since the Biosphere Reserve recognition in 2005 until 2015, 168 large enterprises from several sectors, with mining leading the predicted investments (MMA & COBRAMAB, 2015), signed intention protocols with Minas Gerais government aiming their implementation and operation within RBSE territory.

Concerning the mining expansion peak observed between 2015 – 2016, this contrasts with the Brazilian steel industry crisis with successive production drops since 2012, derived from the global economic recession (ANM, 2019). According to FMI (2018 *apud* ANM, 2019), 2015 consisted of the worse period since 2009 for the mineral and energy commodities market. In this context, the abrupt increase in mining areas located within Espinhaço Range may reflect, in a global scale, the 2016 initial recovery of metal consumption in the international market caused by general economic growth and the investments in infrastructure and construction sectors made by China, the leading

[5] The Iron Quadrangle consists in a small region of Minas Gerais (4% of total state area) that concentrates the main Brazilian iron ore reserve and it is accountable for the major production of this mineral (Sonter et al., 2014; ANM, 2019)

consumer of Brazilian iron ore (Banco Mundial, 2016 *apud* ANM, 2019); and, in local scale, changes in environmental policies regarding the licensing of mining projects.

This considerable expansion is alarming, especially after the unprecedented socio-environmental disaster caused by Fundão Tailings Dam rupture in November 2015 at Minas Gerais (Carmo et al., 2017). Although the stopping of activities in Samarco Mineração S/A plants following the disaster affected the national iron ore pellets production[6] (39.9% decrease in relation to 2015 production; ANM, 2019), this event seems not to have slowed the pace of mineral enterprises implementation. In contrast, this is likely related to a weakening trend of environmental policies in Minas Gerais state, such as represented by the approval of Law n. 21.972/2016, the State Decree n. 47.042/2016 and the Normative Deliberation n. 217/2017, which focuses on simplifying and accelerating environmental licensing processes (Carmo et al., 2017; Milanez, Magno, & Pinto, 2019).

Finally, there were some noticeable dissimilarities when comparing the entire Espinhaço Range and the Espinhaço Range Biosphere Reserve regarding the land-use and land-cover change trends. The Espinhaço Range lost almost twice the original forest area than the RBSE (-9% *vs* -5%), and presented a higher relative net loss of savanna formations (-7% *vs* -5%). Furthermore, while the Espinhaço Range presented an overall increasing trend for pastures (6.6%) and agriculture areas (3.6%) between 1985 and 2019, the RBSE showed an opposite decrease trend for the same period (-2.2% and -14.2%, respectively). The relative net increase of urban areas was more expressive within the Espinhaço Range (151%) than in the RBSE (94%), whereas the relative net growth of mining and forest plantations was higher in the RBSE (46% and 207%, respectively), in comparison to the Espinhaço Range (29% and 144%, respectively).

As a potential effect of granting protection status to part of the Espinhaço Range, one might expect some mid to long-term changes regarding land-use patterns and trends such as reducing native vegetation loss. However, after analyzing the previous and post 2005 scenarios (*i.e.,* the scenarios before and after the Espinhaço Range Biosphere Reserve creation), the main land-use/cover change trends observed were similar between the RBSE and the totality of Espinhaço Range, such as the recent attenuation of deforestation linked to pastureland reduction, the increased net growth of forest

[6] Minas Gerais is the main iron ore producer state in Brazil, accounting for 63,9% of Brazilian production in 2016 (ANM, 2019)

plantations, and net loss rates of savanna formations. These similarities suggest that the observed trends are likely associated with larger-scale aspects affecting the whole geographical extent of the Espinhaço Range. Thus, the results obtained do not suggest clear evidence of direct effects from the Biosphere Reserve classification on mitigating certain land-use/cover change trajectories meaningful for conservation.

Notwithstanding, it is worthy of mention that RBSE is a vast and heterogeneous area (Phase 1 encompassed 3.210.903 ha; MMA & COBRAMAB, 2015) under a complex management structure that comprises distinct zones (*i.e.,* core, buffer and transition zones) and integrates a network of conservation units with different protection categories (*i.e.*, full protection or sustainable use), created on distinct dates and, consequently, presenting contrasting consolidation status (Figure 2). The mentioned aspects make it difficult to assess the RBSE as a territorial unit. Furthermore, this territory is highly dynamic, and only between 2005 and 2015, 36 new conservation units were established within the RBSE area (MMA & COBRAMAB, 2015). Hence, although the effects of Biosphere Reserve creation seem diffuse considering the total area of RBSE, more expressive effects may be present in specific parts of the landscape (*e.g.,* consolidated full protected areas), consisting in a topic for further assessment.

By elucidating the land-use and land-cover change dynamics in Espinhaço Range over the last 35 years, the present study is expected to identify critical areas of habitat loss and fragmentation and the emergence of potential socio-environmental conflicts.

Understanding the main spatio-temporal land-use/cover change trends is a crucial step in improving territory management and public decision-making, especially within the Espinhaço Range Biosphere Reserve, and for promoting a more effective allocation of conservation resources and efforts.

Thus, we hope that this study, a pioneer in the Espinhaço context, raises awareness about the threats to this highly relevant biogeographic locality, underpinning the debate on the role of environmental policies in making socio-economic development compatible with biodiversity conservation, besides encouraging further investigation initiatives.

As future perspectives, we highlight the importance of assessing land-use/cover change dynamics over the Espinhaço Range, integrating forthcoming advances regarding remote sensing or methodological classification improvements achieved by MapBiomas Project.

4.2. Ecological Niche Modelling

Results summary

- According to the models' projections, around one third of target species underwent potential habitat loss between 1985 and 2019 within the Espinhaço Range.
- The birds were the fauna group that presented the most consistent and highest magnitude trend of potential habitat loss for the period analysed.
- In general, the target species presented similar temporal trajectories and trends regarding potential habitat change between the RBSE and the totality of Espinhaço Range, but with lower magnitude effects observed within the RBSE (Phase 1 limits).
- There was a pattern of inflection points on the species' suitable area trend curves around 2005 (*i.e.,* the RBSE's creation year), within both the RBSE and the totality of Espinhaço Range.
- Species potential richness is unevenly distributed throughout the study area, especially for birds and amphibians, tending to aggregate in two hotspots: (i) in the southern part of Espinhaço Range; and (ii) in the Chapada Diamantina region, within northern Espinhaço.
- Only the southern hotspot is encompassed by the RBSE, while the northern hotspot is entirely outside the RBSE's protection boundaries, even considering its Phase 2 expansion.
- Concerning the southern hotspot, there is a spatial overlap between its southernmost part and the mining region of the Iron Quadrangle.
- All the assessed fauna groups presented potential loss of species richness within the Espinhaço Range during 1985 – 2019, yet with low overall magnitudes.
- There was a spatial aggregation of areas presenting a strong tendency to biodiversity loss within the Espinhaço Range.
- There was no clear spatial overlap between areas with a high trend of species richness loss and a specific land-use/cover transition within the Espinhaço Range, suggesting multiple/synergistic transitions are likely into play.

Discussion

The results show that a considerable part of the target species (32% *i.e.,* 12 out of 37 species) underwent potential habitat loss during 1985 – 2019 within the Espinhaço Range, being this trend significantly higher for the birds, group for which potential habitat loss reached 66% of species (*i.e.,* 6 out of 9 species) and a maximum of -28.6% of the average suitable area estimated (for *Embernagra longicauda*).

It is essential to point out that these results refer to the potential habitat losses that occurred in the last 35 years in the face of land-use/cover changes and, therefore, the species may be actually experiencing more drastic reductions of their original distribution ranges considering cumulative impacts previous to 1985 and additional synergistic factors (*e.g.,* climate change). Indeed, recent studies suggest that some species occurring within Espinhaço Range does not have their distribution ranges and conservation status correctly estimated (Hoffmann et al., 2020), such as stated for *Asthenes luizae* (Freitas et al., 2019) and *Pithecopus megacephalus* (Ramos et al., 2018). Furthermore, predictions considering future climate change scenarios are alarming for eastern Brazil mountaintop endemic birds, suggesting gradual habitat contractions ranging from 44.5% to 100% until both 2050 and 2070 (Hoffmann et al., 2020). Since the target species of the present study are endemic, local habitat loss means a risk of global extinction.

Concerning the relatively stable trends observed for amphibian and reptile species, it is worth recalling that Ecological Niche Models predict species potential distribution based on abiotic environmental conditions and, by doing so, these may forecast suitable habitats areas outside the species actual distribution range owing to biotic and dispersal constraints (*i.e.,* not all the predicted suitability range is actually available or accessible for species). This ENM limitation is an especially relevant issue in the context of spatially discontinuous ecosystems, which is the case of the rupestrian grasslands whose patchy distribution and altitudinal isolation as sky-island archipelagos (Chaves et al., 2015) pose obstacles to species migration/colonization processes. In addition, this effect is of particular importance for low dispersal organisms, such as most amphibian and reptile species, for which ENMs tend to over-predict distribution ranges (Leite, 2012).

Regarding the potential habitat gains, the reptiles were the fauna group presenting most expressive positive trends. In addition to the potential *vs.* real distribution modelling issue previously discussed, this result may be related to a more generalist tendency in terms of habitat requirements presented by the modelled reptile species, as the species richness patterns of this fauna group suggest. In the case of the forest-living amphibian *Hylodes uai* (+12.38% relative change within Espinhaço Range), we hypothesize that the potential habitat increase may be related to recent forest recovery, since there is a temporal coincidence between an increase in forest formations and the inflection point towards habitat gain on this species' suitable area trajectory (see Appendix S10).

Considering the species assemblage, the results reveal that the species richness is unevenly distributed throughout the study area. In contrast, there are clear hotspot areas, especially for birds and amphibians, namely: the southern part of Espinhaço Range (comprising Serra do Cipó, the Iron Quadrangle and Diamantina plateau); and the Chapada Diamantina region within northern Espinhaço, with both hotspots corroborated by Leite (2012) and Hoffman et al. (2020).

These high species richness hotspots overlap with higher altitude areas that, probably due to geoclimatic barriers associated with the landscape topographical discontinuities (Leite, 2012; Chaves et al., 2015), seem to play a refuge role for the fauna. Topography is a key factor that influences land-use patterns and, since steep terrains usually pose an accessibility limitation (Freitas, Hawbaker & Metzger, 2010), it may be expected that the altitudinal gradient also buffers anthropogenic disturbances such as the impacts on biotic communities related to land-cover changes. Indeed, based on our model projections, these elevated hotspots' core area was less affected by potential species richness change during the analysed period (*i.e.,* 1985 – 2019).

However, yet these hotspots consist in stable areas for species populations under current climatic conditions, they may not keep this way in the future. Hoffmann et al. (2020) predicted upwards altitudinal shifts (up to 664 m and 744 m until 2050 and 2070, respectively) for bird species endemic to southern Brazilian mountaintops due to future climate changes. In the context of Espinhaço Range, high levels of altitudinal displacement mean potential distribution contraction owing to the limited availability of higher elevation areas. On the other hand, based on a visual comparison, the birds and amphibians' richness hotspots seem to spatially correlate with the stability areas predicted for the rupestrian grasslands, which is the primary habitat for many modelled species in this study, under future climate change scenarios (2050 and 2070; Fernandes et al., 2018).

Besides the aforementioned altitudinal shifts, latitudinal displacements towards areas with milder temperature are also expected in response to climate change (Parmesan & Yohe, 2003). Indeed, Hoffmann et al. (2020) reported a trend for southwest latitudinal shifts (*i.e.,* to higher latitudes) concerning the center of the distribution area of eastern Brazilian mountaintop endemic birds under climate change scenarios. This observation emphasizes the importance of conservation efforts focused on the southern portion of Espinhaço Range, which is likely to keep stable and suitable habitat areas for endemic mountain species in the future, especially considering that this mountain range does not extend continuously across a wide latitudinal gradient.

Putting aside the uncertainties regarding future predictions and focusing on the current scenario, the strong spatial overlap between the southernmost part of the species richness hotspot and the Iron Quadrangle (*i.e.,* one of the Brazilian and world's most crucial mineral regions; Figure 20) is utmost alarming from a biodiversity conservation perspective.

In this context, Pena et al. (2017) have demonstrated that around 36% and 29% of the median potential distribution of endemic anuran and bird species are currently directly or indirectly affected by mining activities within the Espinhaço region. Especially for the amphibians, species richest areas (*i.e.,* presenting suitable environmental conditions for several species) are even highly affected: 67% of the potentially suitable area for 16 endemic amphibians are currently impacted by mining activities (Pena et al., 2017).

Concerning the adverse effects of mining on highly biodiverse areas within the Espinhaço Range, the current trend towards Brazilian environmental policy weakening may contribute to the expansion of this activity, going against the conservation goals needed. It is the case of the Legislative Proposal 2159/2021 that aims to accelerate and facilitate the licensing of enterprises able to cause environmental degradation, for instance through promoting companies self-regulation, costs reduction and minimizing public participation (Bronz, Zhouri & Castro, 2020). Hence, from a pragmatic perspective, we highlight the importance of do not dissociate the conservation discussions from the governmental policy agenda.

Although a spatial correlation between areas of higher species richness loss and mining expansion was not observed in the present study, this topic needs further investigation. In general, land-use/cover changes displayed complex spatiotemporal patterns, and mining presented a sparse and spatially restricted distribution within the Espinhaço region, hampering the identification of clear overlaps. Notwithstanding, other anthropic change drivers coexist with mining in the Espinhaço landscape, being even more spatially significant. Indeed, we observed spatial coincidence among the zones showing a strong tendency to biodiversity loss and other land-use/cover transitions, namely deforestation/reforestantion, farming and forest plantations expansion. Thus, the results suggest that the potential habitat and biodiversity loss are likely resulted from the combined effects of multiple and interconnected land-use/cover change processes rather than from a specific LULC change.

Overall, results show attenuated trends of potentially suitable habitat change (including both potential losses and gains) within the RBSE (Phase 1 limits) compared to the totality of Espinhaço Range. In addition to the occurrence of many inflection points on the species' trend curves around 2005 (*i.e.,* the RBSE's creation year), this observation could suggest potential effects of the Biosphere Reserve classification on biodiversity conservation at Espinhaço region. However, the fact that these trajectories around 2005 were also observed for the total Espinhaço extent suggests a likely association with land-use/cover changes affecting the entire region rather than local effects of the Biosphere Reserve establishment. Thus, there are likely other factors behind the lower magnitude in potential habitat change observed for the endemic species between the southern and northern portions of Espinhaço Range.

Nevertheless, the existence of an endemic biodiversity hotspot within the northern part of Espinhaço Range supports the expansion of the RBSE's limits, as a future Phase 3 (Andrade et al., 2018), to Bahia state aiming to broaden its protection boundaries and conservation goals to this highly diverse region.

Technical constraints regarding the modelling of species with a low number of occurrence records presented severe limitations to study range shifts of other endemic species. However, considering that many species endemic to the Espinhaço Range have their distribution ranges partially overlapped, we expect that promoting conservation actions focused on the modelled species may also benefit other rare co-occurring species (*i.e.,* the umbrella-species concept, which has been proven an effective strategy for conservation planning; Branton & Richardson, 2011).

In this context, the Ensembles of Small Models (ESM) represent a promising approach to overcome the challenge of small sample sizes and consist in a future perspective for modelling the potential distribution of rare species with few occurrence records while avoiding model overfitting (Breiner et al., 2018). In parallel, we highlight the importance of improving the survey efforts within the Espinhaço Range in order to fill knowledge gaps regarding their endemic biodiversity and enhance the existing occurrence datasets. Model projections obtained in this study can guide these efforts by supporting enhanced sampling schemes for surveying and monitoring (Guisan et al., 2006; Carvalho et al., 2016).

Substantial challenges still lie ahead if the aim is to ensure the long-term conservation of the endemic species within Espinhaço Range. Based on the present assessment, we advise that conservation strategies prioritize the protection of the biodiversity hotspots through adequate land-use planning to reduce pressures on ecosystems and species. By identifying areas with substantial losses in species diversity, model projections can also guide localized restoration efforts and the implementation of land-use constraints focusing on conservation. This study results also suggest that an expansion of RBSE towards the north of Espinhaço Range, improving the connectivity between multiple hotspots (*i.e.,* Phase 3 RBSE expansion) as well as improving management and monitoring, will strongly benefit this protected area.

5. Concluding remarks

Through the integration of land-use/cover change assessment and Ecological Niche Modelling, this study presented a quali-quantitative description of the land-use change dynamics that occurred in the Espinhaço Range in the last 35 years, as well as their effects on the potential distribution of endemic bird, amphibian and reptile species.

As previously detailed, the Espinhaço Range underwent an intense land-use change since 1985, with farming and logging expansion representing the main drivers of native vegetation loss in the region. As a result, around one third of the endemic target species, especially the birds, presented trends towards potential habitat contraction between 1985 and 2019, according to our models' projections.

The present study contributes to the identification of (i) crucial areas susceptible to habitat loss and fragmentation associated with land-use/cover change processes; (ii) areas where socio-environmental conflicts are likely to occur; and (iii) species highly threatened by their habitat degradation. As such this thesis outputs support strengthening territorial management and conservation strategies focused on the endemic fauna in the context of the Espinhaço Range.

We advise that conservation strategies must prioritize the protection of the biodiversity hotspots identified through adequate land-use planning to reduce pressures on ecosystems and species populations. In this context, we draw attention to the conflict (conservation *vs.* economic development) that arises from the spatial overlap between the southern hotspot and the mining exploration region of the Iron Quadrangle. Considering the importance of this hotspot as a likely stable habitat zone for endemic species under future climate changes, we highlight the need of this conflict being properly addressed from a governmental perspective. Besides, our results support the expansion of the Espinhaço Range Biosphere Reserve's limits towards the north part of Espinhaço Range (*i.e.,* Phase 3 RBSE expansion) in order to include the Chapada Diamantina hotspot and therefore improve the connectivity between hotspots. Increasing landscape connectivity is ever more critical in face of climate change allowing species movements to track suitable areas.

Additionally, we propose that the establishment and/or expansion of the Biosphere Reserves should be associated with other actions at a local scale (*e.g.:* creation of Conservation Units, strengthening of environmental policies, among others), allowing to improve their effectiveness in terms of environment protection and achievement of their goals.

Furthermore, we hope that this study, unprecedented for the Espinhaço Range, shed light on this highly biodiverse region, often underestimated in comparison to other spatially extensive mountain ranges like the Andes, raising awareness about its biogeographic and ecological importance and the urgent need of investigative and conservation efforts aiming to protect the biodiversity that it harbors.

Finally, it is worthy of mention the fact that the calibration of a multi-annual model (*i.e.,* model calibration considering the dynamic environmental conditions of all the years comprised by the available species records) consisted in an innovative methodological aspect of the present study. This approach, which allows capturing the temporal variability associated with the predictive variables of the models and, consequently, a more accurate association between occurrence records and environmental conditions, can be explored in other studies based on ecological modelling techniques.

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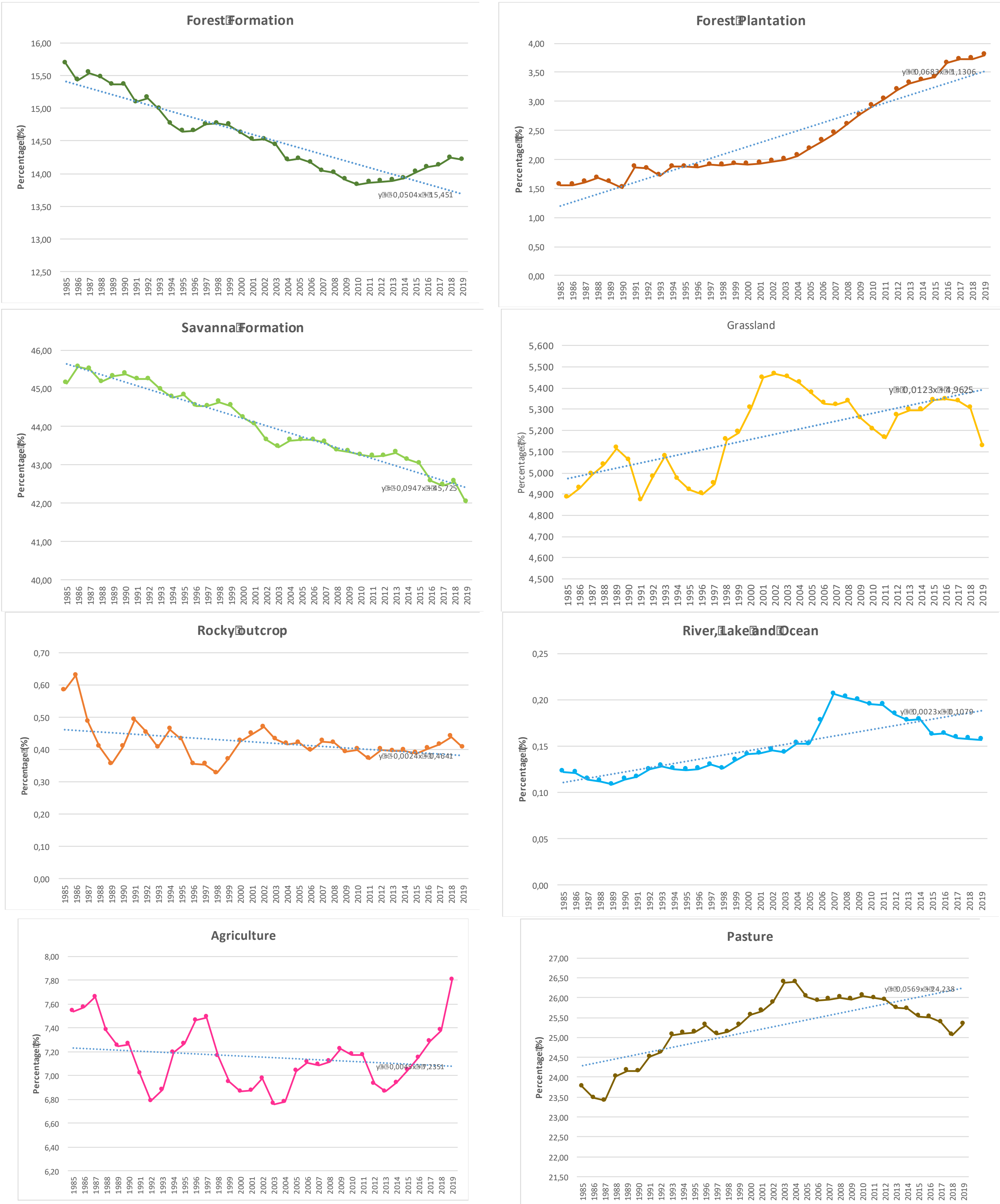
APPENDICES

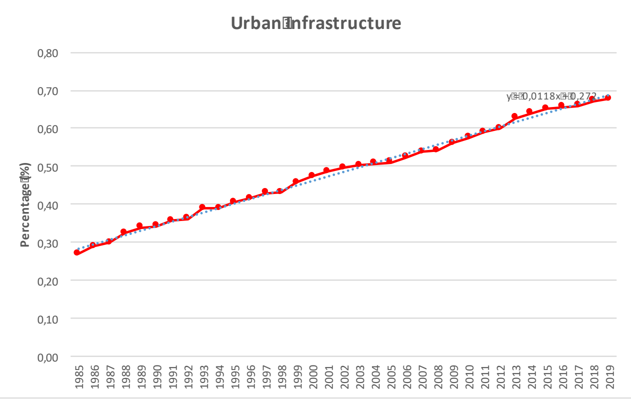
Appendix S1: Environmental variables selected for each target species. “Avg” refers to average; “Med” refers to median; and “Std” refers to standard deviation (see Table 3 for description of variables codes).

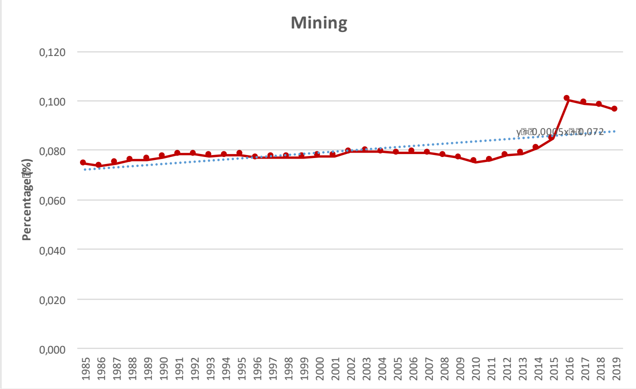
Fauna Group	Species	Nº records	Unique records (1x1 km2)	Nº variables	Selected variables	
					Static	Dynamic
Amphibians	*Aplastodiscus heterophonicus*	39	25	4	BIO_18, BIO_13	FORF, PAST
Amphibians	*Boana botumirim*	151	31	7	BIO_18, BIO_06, BIO_16	GRAS, ONVA, PAST, SAVF
Amphibians	*Boana cipoensis*	137	21	5	BIO_08, BIO_16	GRAS, NDWImed_avg, SAVF
Amphibians	*Bokermannohyla alvarengai*	303	93	11	Elev_Avg, BIO_05, BIO_16, BIO_15, BIO_07	GRAS, PAST, EVI2med_avg, SAVF, NDWImed_std, ROCK
Amphibians	*Bokermannohyla martinsi*	124	36	10	BIO_06, BIO_04, BIO_12, BIO_07	ROCK, PAST, AGRI, NDWImed_std, SAVF, FORF
Amphibians	*Bokermannohyla nanuzae*	225	46	9	BIO_06, BIO_18, BIO_13, DNI_avg	SAVF, PAST, GRAS, AGRI, FORF
Amphibians	*Bokermannohyla oxente*	223	40	9	BIO_03, Elev_Avg, BIO_04, BIO_05	PAST, SAVF, GRAS, EVI2med_std, FORF
Amphibians	*Bokermannohyla saxicola*	519	87	11	BIO_06, BIO_18, BIO_13, BIO_07, BIO_02	GRAS, PAST, SAVF, AGRI, NDWImed_std, FORF
Amphibians	*Crossodactylus trachystomus*	147	31	8	BIO_08, BIO_18, BIO_16	PAST, NDWImed_std, SAVF, FORF, GRAS
Amphibians	*Hylodes uai*	74	23	4	BIO_06, BIO_18	FORF, EVI2med_std
Amphibians	*Leptodactylus camaquara*	135	42	8	Elev_Avg, BIO_05, BIO_16, BIO_04	GRAS, PAST, AGRI, SAVF
Amphibians	*Odontophrynus juquinha*	94	29	5	Elev_Avg, BIO_18	NDWImed_avg, GRAS, SAVF
Amphibians	*Phasmahyla jandaia*	82	21	4	BIO_06, BIO_12	FORF, PAST
Amphibians	*Physalaemus orophilus*	84	23	4	BIO_18, BIO_06	FORF, SAVF
Amphibians	*Pithecopus megacephalus*	108	19	5	BIO_06, BIO_18	GRAS, PAST, SAVF
Amphibians	*Pristimantis rupicola*	46	15	6	BIO_03, Elev_Avg	PAST, SAVF, GRAS, ROCK
Amphibians	*Proceratophrys cururu*	48	18	4	BIO_06, BIO_16	GRAS, AGRI
Amphibians	*Pseudopaludicola mineira*	408	39	8	BIO_06, BIO_04, BIO_13, BIO_07	AGRI, GRAS, PAST, SAVF
Amphibians	*Scinax curicica*	375	84	11	BIO_06, BIO_18, BIO_16, BIO_02, BIO_07	PAST, NDWImed_std, EVI2med_avg, SAVF, FORF, GRAS
Amphibians	*Scinax machadoi*	502	97	11	BIO_06, BIO_18, BIO_16, Elev_Std, BIO_02	PAST, SAVF, NDWImed_std, EVI2med_avg, FORF, GRAS

Fauna Group	Species	Nº records	Unique records (1x1 km2)	Nº variables	Selected variables	
					Static	Dynamic
Amphibians	*Scinax montivagus*	103	17	5	BIO_03, Elev_Avg	PAST, SAVF, GRAS
Amphibians	*Thoropa megatympanum*	579	110	11	Elev_Avg, BIO_18, BIO_13, BIO_15, BIO_07	GRAS, PAST, SAVF, NDWImed_std, AGRI, ROCK
Birds	*Asthenes luizae*	138	100	11	Elev_Avg, BIO_05, BIO_13, BIO_04, TWI_avg	GRAS, AGRI, EVI2med_avg, PAST, SAVF, ROCK
Birds	*Asthenes moreirae*	126	20	5	BIO_05, Elev_Avg	ROCK, EVI2med_avg, SAVF
Birds	*Augastes lumachella*	112	28	7	BIO_03, Elev_Avg, BIO_07	PAST, SAVF, GRAS, ROCK
Birds	*Augastes scutatus*	432	118	11	Elev_Avg, BIO_18, BIO_13, Elev_Std, BIO_07	PAST, GRAS, EVI2med_avg, SAVF, EVI2med_std, ROCK
Birds	*Campylopterus diamantinensis*	56	29	8	Elev_Avg, BIO_13, BIO_18	EVI2med_avg, PAST, ONVA, SAVF, ROCK
Birds	*Cinclodes espinhacensis*	66	48	11	BIO_05, Elev_Avg, BIO_13, BIO_04	GRAS, AGRI, ONVA, NDWImed_avg, SAVF, ROCK, PAST
Birds	*Embernagra longicauda*	308	188	11	Elev_Avg, BIO_05, BIO_13, Elev_Std, BIO_07	PAST, EVI2med_avg, SAVF, FORF, GRAS, ROCK
Birds	*Formicivora grantsaui*	26	15	6	BIO_03, BIO_10	GRAS, PAST, SAVF, ROCK
Birds	*Polystictus superciliaris*	222	129	11	Elev_Avg, BIO_05, BIO_13, Elev_Std, BIO_03	PAST, EVI2med_avg, SAVF, NDWImed_std, FORF, ROCK
Reptiles	*Amphisbaena acangaoba*	31	18	4	BIO_03, Elev_Avg	SAVF, PAST
Reptiles	*Eurolophosaurus nanuzae*	145	15	5	Elev_Avg, BIO_18	SAVF, EVI2med_std, ROCK
Reptiles	*Hydromedusa maximiliani*	49	30	7	BIO_05, BIO_12, DNI_avg	FORF, EVI2med_std, PAST, RILO
Reptiles	*Psilops paeminosus*	72	17	4	BIO_12, BIO_15	NDWImed_std, GRAS
Reptiles	*Tropidurus erythrocephalus*	138	17	6	BIO_03, BIO_16	AGRI, SAVF, ROCK, GRAS
Reptiles	*Tropidurus montanus*	96	23	5	Elev_Avg, BIO_05	NDWImed_avg, NDWImed_std, ROCK

Appendix S2: Temporal trajectories of main land-use and land-cover classes in Espinhaço Range between 1985 and 2019.




Urban Infrastructure
Percentage (%)
y = 0,0118x + 0,272
0,80
0,70
0,60
0,50
0,40
0,30
0,20
0,10
0,00
1985
2019


Mining
Percentage (%)
y = 0,0005x + 0,072
0,120
0,100
0,080
0,060
0,040
0,020
0,000
1985
2019

Appendix S3: Transition matrix of LULC change in Espinhaço Range during 1985-2005 (I) and 2005-2019 (II) [unit: (a) %; (b) km²].

I.a

	Classes	2005												
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Other Non Forest Natural Formation	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
1985	No class	100.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
	Forest Formation	0.0%	77.4%	5.0%	3.0%	0.1%	0.1%	0.0%	11.1%	3.0%	0.1%	0.1%	0.1%	0.1%
	Savanna Formation	0.0%	1.8%	84.5%	0.8%	0.5%	0.1%	0.0%	9.8%	2.4%	0.1%	0.0%	0.0%	0.0%
	Forest Plantation	0.0%	3.0%	12.9%	77.6%	1.0%	0.0%	0.0%	4.8%	0.6%	0.0%	0.0%	0.1%	0.0%
	Grassland	0.0%	0.1%	1.9%	0.3%	90.1%	0.1%	0.0%	4.2%	2.4%	0.0%	0.0%	0.9%	0.0%
	Rocky outcrop	0.0%	1.9%	16.6%	0.0%	10.9%	41.2%	0.0%	14.6%	11.6%	0.3%	0.3%	2.3%	0.2%
	Other Non Forest Natural Formation	0.0%	24.1%	18.1%	0.0%	0.0%	0.0%	34.8%	18.4%	3.7%	0.0%	0.0%	0.5%	0.4%
	Pasture	0.0%	3.3%	13.8%	0.5%	1.7%	0.0%	0.0%	74.5%	5.5%	0.6%	0.0%	0.1%	0.1%
	Agriculture	0.0%	5.7%	13.4%	0.4%	3.0%	1.1%	0.0%	22.5%	53.0%	0.6%	0.1%	0.3%	0.1%
	Urban Infrastructure	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	100.0%	0.0%	0.0%	0.0%
	Mining	0.0%	0.4%	1.8%	0.0%	0.0%	4.3%	0.0%	1.2%	1.7%	3.3%	82.7%	3.8%	0.9%
	Other Non Vegetated Area	0.0%	1.1%	4.1%	1.2%	11.3%	2.5%	0.0%	20.1%	3.3%	3.8%	0.1%	52.3%	0.3%
	River. Lake and Ocean	0.0%	2.4%	10.1%	0.0%	0.4%	0.6%	0.0%	5.2%	1.0%	0.4%	0.3%	0.4%	79.2%

I.b

	Classes	2005												
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Other Non Forest Natural Formation	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
1985	No class	502740.15	0.00	0.02	0.00	0.01	0.00	0.00	0.23	0.04	0.00	0.00	0.00	0.00
	Forest Formation	0.01	27162.27	1767.14	1052.21	37.14	35.66	0.32	3893.71	1035.10	33.43	17.90	21.84	21.77
	Savanna Formation	0.04	1800.11	85362.77	769.74	457.14	125.27	0.60	9913.66	2399.57	57.90	3.06	25.86	48.25
	Forest Plantation	0.00	105.46	447.68	2700.11	34.79	0.07	0.00	167.93	19.24	0.06	0.86	3.05	0.15
	Grassland	0.00	11.03	202.23	35.23	9843.36	10.08	0.00	464.27	260.43	3.57	0.00	93.70	3.71
	Rocky outcrop	0.01	24.95	216.48	0.05	142.07	536.88	0.00	189.81	150.78	3.56	4.19	30.46	3.10
	Other Non Forest Natural Formation	0.00	0.62	0.47	0.00	0.00	0.00	0.90	0.48	0.09	0.00	0.00	0.01	0.01
	Pasture	0.02	1744.76	7332.60	244.83	903.08	17.74	0.03	39610.40	2914.02	293.21	2.03	63.81	33.67
	Agriculture	0.02	954.93	2264.91	68.88	508.05	182.73	0.03	3785.26	8928.85	104.93	9.60	42.74	8.61
	Urban Infrastructure	0.00	0.04	0.05	0.00	0.00	0.01	0.00	0.04	0.03	604.47	0.06	0.01	0.01
	Mining	0.00	0.63	2.96	0.02	0.02	7.10	0.00	1.98	2.87	5.43	137.85	6.37	1.45
	Other Non Vegetated Area	0.00	10.05	37.06	11.13	102.38	22.52	0.00	182.56	29.53	34.31	0.51	474.42	2.90
	River. Lake and Ocean	0.00	6.60	27.52	0.13	1.05	1.77	0.00	14.23	2.64	1.06	0.69	1.00	216.06

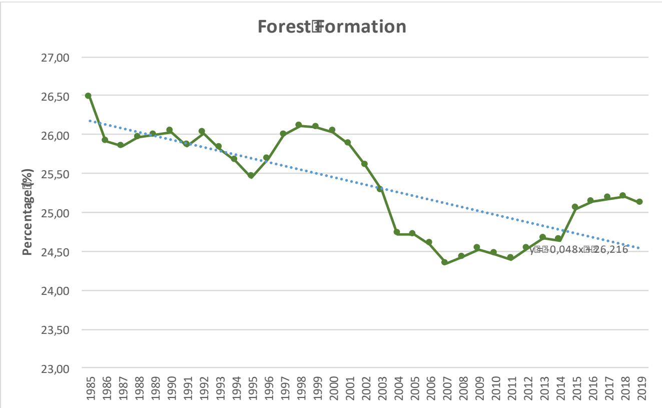
II.a

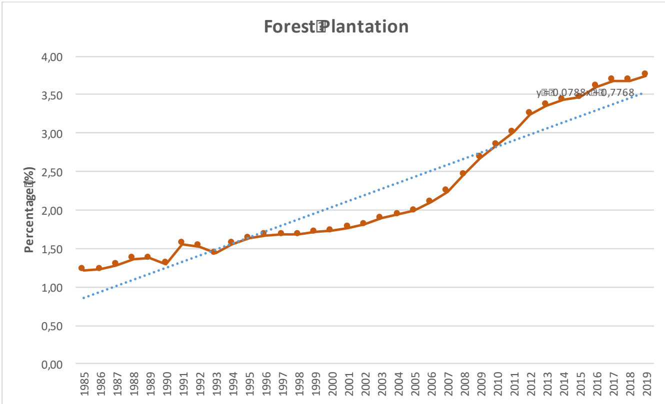
	Classes	2019												
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Other Non Forest Natural Formation	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
2005	No class	100.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
	Forest Formation	0.0%	87.8%	2.9%	2.3%	0.0%	0.0%	0.0%	4.5%	2.2%	0.1%	0.1%	0.0%	0.1%
	Savanna Formation	0.0%	1.6%	87.4%	1.7%	0.3%	0.1%	0.0%	7.0%	1.7%	0.0%	0.0%	0.0%	0.1%
	Forest Plantation	0.0%	0.4%	0.3%	98.9%	0.0%	0.0%	0.0%	0.1%	0.2%	0.0%	0.0%	0.0%	0.0%
	Grassland	0.0%	0.1%	4.0%	0.7%	84.2%	0.2%	0.0%	5.9%	3.6%	0.0%	0.0%	1.4%	0.0%
	Rocky outcrop	0.0%	2.0%	10.0%	0.2%	5.5%	67.4%	0.0%	3.1%	7.6%	0.8%	1.4%	1.8%	0.1%
	Other Non Forest Natural Formation	0.0%	10.2%	31.1%	0.0%	0.0%	0.0%	51.2%	3.3%	3.1%	0.0%	0.0%	0.0%	1.0%
	Pasture	0.0%	2.4%	9.3%	1.7%	1.0%	0.0%	0.0%	78.1%	6.9%	0.4%	0.0%	0.2%	0.1%
	Agriculture	0.0%	5.3%	10.7%	1.0%	2.3%	0.5%	0.0%	13.0%	66.6%	0.4%	0.1%	0.1%	0.0%
	Urban Infrastructure	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	99.6%	0.3%	0.0%	0.0%
	Mining	0.0%	0.2%	1.2%	0.2%	0.0%	0.4%	0.0%	1.5%	3.0%	4.2%	87.5%	0.8%	1.0%
	Other Non Vegetated Area	0.0%	0.7%	1.5%	1.4%	10.8%	3.9%	0.0%	8.9%	4.6%	1.5%	0.6%	66.0%	0.2%
	River. Lake and Ocean	0.0%	2.7%	12.2%	0.6%	0.8%	0.8%	0.0%	12.8%	3.2%	0.2%	0.6%	1.3%	64.8%

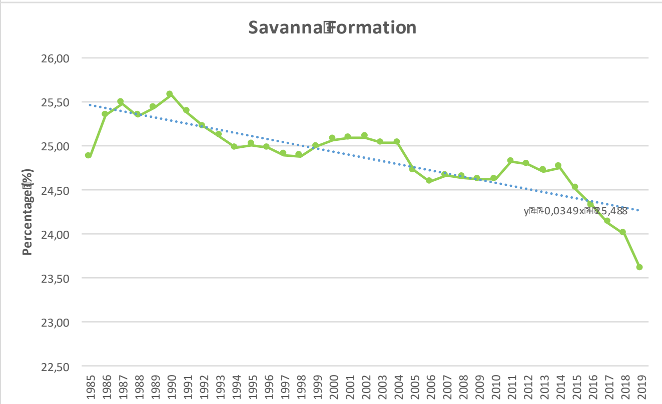
II.b

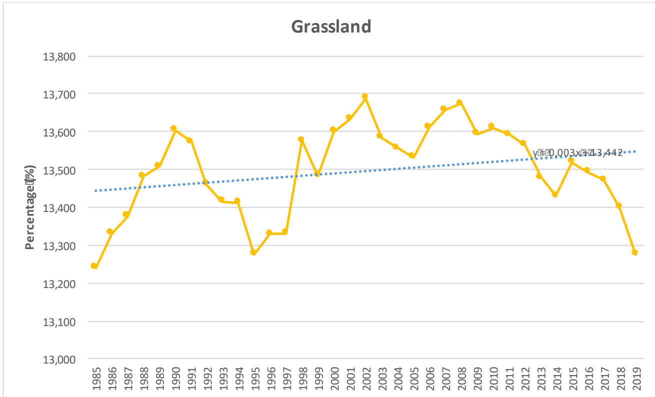
	Classes	2019												
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Other Non Forest Natural Formation	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
2005	No class	502740.02	0.00	0.02	0.00	0.01	0.00	0.00	0.17	0.02	0.00	0.00	0.00	0.00
	Forest Formation	0.00	27926.79	928.94	726.96	5.09	14.26	0.19	1441.23	691.26	20.64	18.87	14.12	33.12
	Savanna Formation	0.03	1575.66	85354.53	1662.49	276.67	101.42	0.43	6866.04	1687.78	43.19	5.95	31.97	55.72
	Forest Plantation	0.00	21.55	13.31	4830.08	0.56	0.04	0.00	6.17	8.99	1.13	0.26	0.16	0.08
	Grassland	0.01	9.56	476.31	87.93	10123.00	20.87	0.00	707.17	435.84	3.85	0.06	162.81	1.70
	Rocky outcrop	0.00	19.12	94.01	1.88	51.48	633.51	0.00	28.89	71.23	7.88	13.26	17.31	1.24
	Other Non Forest Natural Formation	0.00	0.19	0.58	0.00	0.00	0.00	0.96	0.06	0.06	0.00	0.00	0.00	0.02
	Pasture	0.05	1386.72	5404.71	1012.70	572.10	26.57	0.11	45445.55	4025.37	211.32	3.63	103.97	31.76
	Agriculture	0.01	830.96	1683.57	157.93	355.95	76.41	0.10	2050.26	10487.49	69.88	8.97	18.27	3.36
	Urban Infrastructure	0.00	0.27	0.17	0.00	0.00	0.04	0.00	0.22	0.37	1137.71	3.09	0.04	0.03
	Mining	0.00	0.34	2.19	0.27	0.00	0.77	0.00	2.63	5.24	7.47	154.69	1.33	1.81
	Other Non Vegetated Area	0.00	5.32	11.09	10.31	82.60	29.86	0.00	67.74	35.38	11.40	4.46	503.74	1.39
	River. Lake and Ocean	0.00	9.10	41.57	1.98	2.65	2.80	0.01	43.46	10.76	0.65	2.20	4.28	220.24

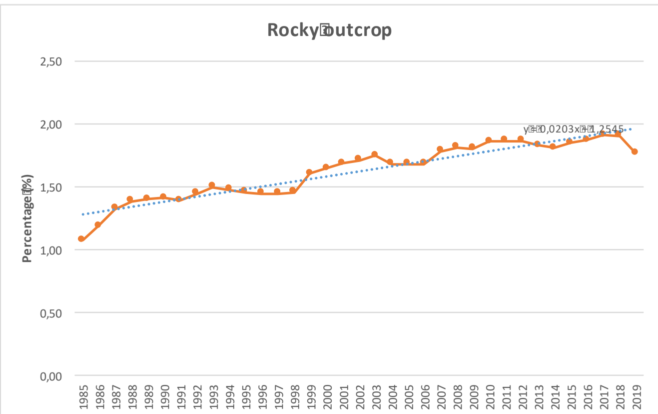
Appendix S4: Temporal trajectories of main land-use and land-cover classes in Espinhaço Range Biosphere Reserve between 1985 and 2019.

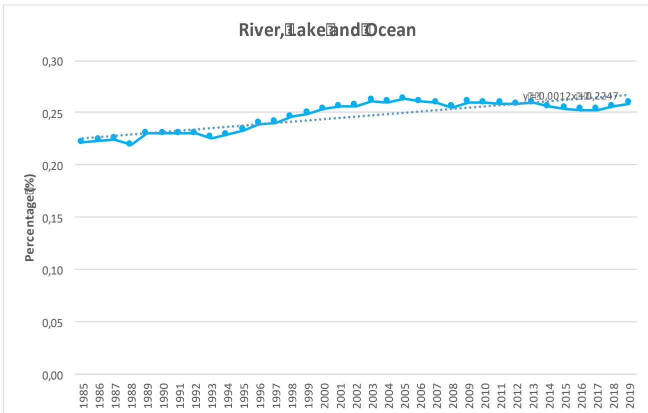


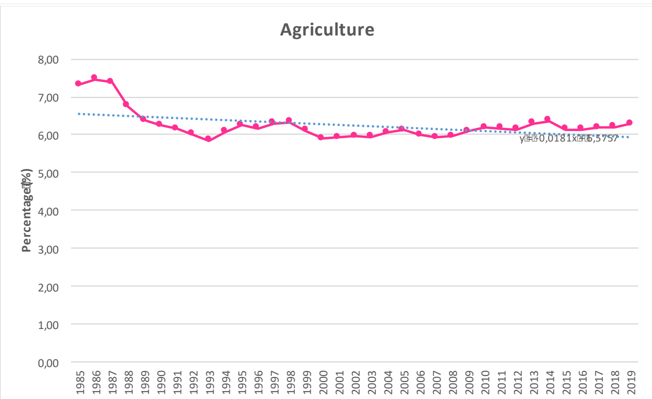


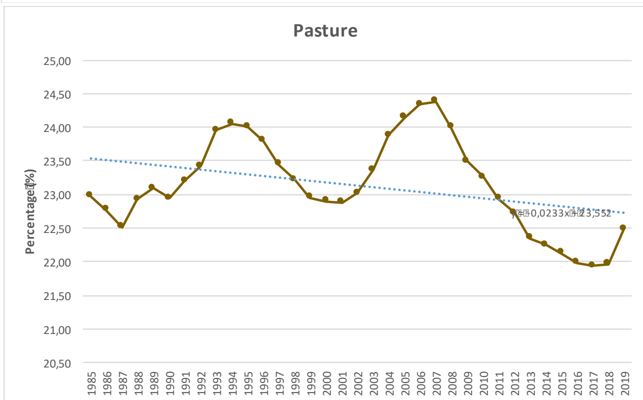


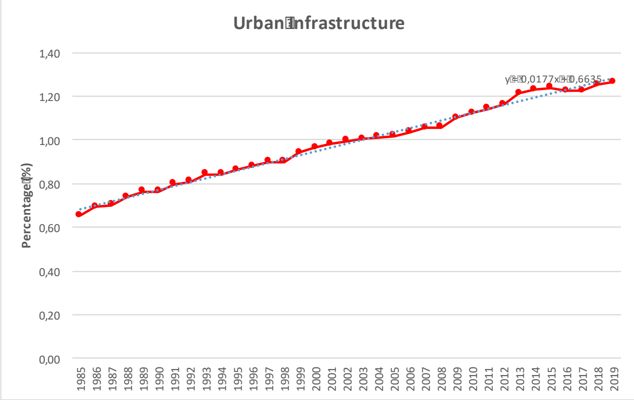


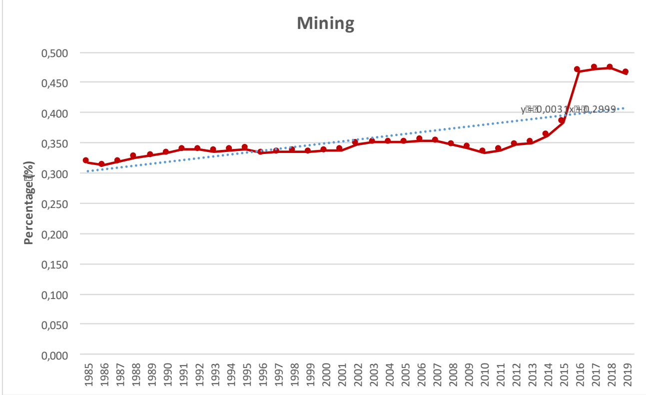






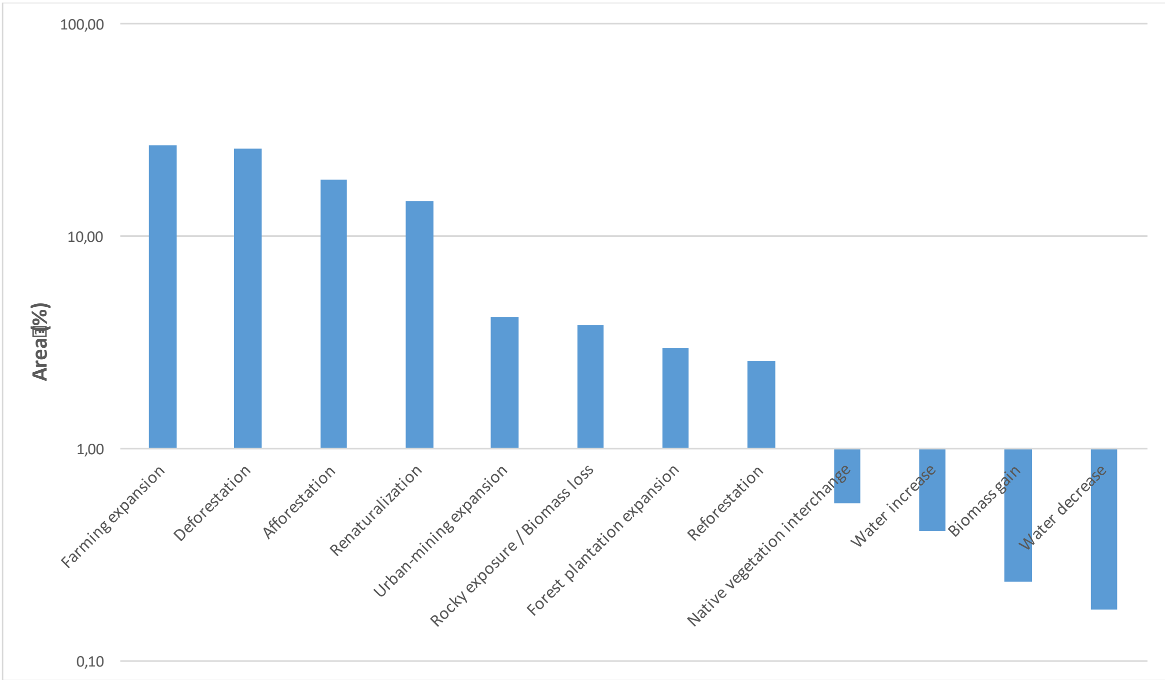



Urban Infrastructure
y = 0,0177x + 0,6635
Percentage (%)
1,40
1,20
1,00
0,80
0,60
0,40
0,20
0,00
1985 1986 1987 1988 1989 1990 1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001 2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016 2017 2018 2019

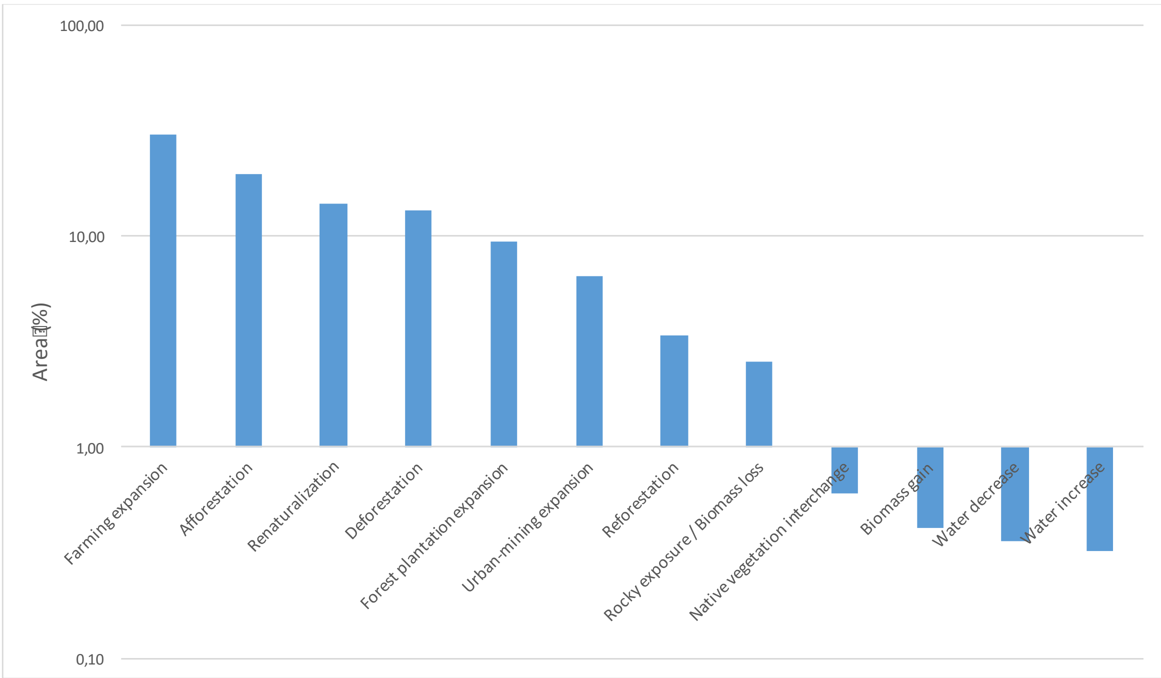

Mining
y = 0,0031x + 0,2999
Percentage (%)
0,500
0,450
0,400
0,350
0,300
0,250
0,200
0,150
0,100
0,050
0,000
1985 1986 1987 1988 1989 1990 1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001 2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016 2017 2018 2019

Appendix S5: Land-use/cover transitions within the RBSE during 1985 – 2005 (a) and 2005 – 2019 (b).

a.



b.



Appendix S6: Transition matrix of LULC change in Espinhaço Range Biosphere Reserve during 1985-2005 (I) and 2005-2019 (II) [unit: (a) %; (b) km^2].

I.a

	Classes	2005											
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
1985	No class	100.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
	Forest Formation	0.0%	80.9%	3.8%	1.7%	0.0%	0.3%	9.2%	3.6%	0.1%	0.2%	0.1%	0.1%
	Savanna Formation	0.0%	3.4%	87.3%	1.2%	0.2%	0.8%	5.8%	1.0%	0.1%	0.0%	0.1%	0.1%
	Forest Plantation	0.0%	4.4%	8.2%	82.0%	1.4%	0.0%	2.5%	1.3%	0.0%	0.1%	0.0%	0.0%
	Grassland	0.0%	0.0%	0.3%	0.2%	96.3%	0.0%	1.4%	0.0%	0.0%	0.0%	1.8%	0.0%
	Rocky outcrop	0.0%	1.4%	3.9%	0.0%	0.0%	87.7%	0.2%	3.2%	0.8%	1.1%	1.6%	0.0%
	Pasture	0.0%	5.4%	6.8%	0.5%	1.9%	0.1%	79.9%	4.6%	0.4%	0.0%	0.2%	0.1%
	Agriculture	0.0%	15.4%	2.6%	1.2%	0.0%	5.2%	21.2%	51.4%	2.1%	0.3%	0.4%	0.1%
	Urban Infrastructure	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	99.9%	0.0%	0.0%	0.0%
	Mining	0.0%	0.3%	0.9%	0.0%	0.0%	6.2%	0.6%	1.5%	3.0%	81.5%	4.9%	1.0%
	Other Non Vegetated Area	0.0%	0.5%	1.4%	0.1%	15.0%	3.3%	8.0%	1.9%	2.6%	0.1%	66.9%	0.3%
	River. Lake and Ocean	0.0%	1.2%	8.3%	0.1%	0.1%	0.0%	2.6%	0.1%	0.6%	0.3%	0.5%	86.2%

I.b

	Classes	2005											
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
1985	No class	44357.57	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Forest Formation	0.00	7249.03	339.58	155.47	3.94	22.55	822.35	325.51	11.36	15.40	4.97	7.52
	Savanna Formation	0.00	282.99	7351.29	101.68	20.32	66.81	489.63	84.56	5.55	1.46	5.38	6.26
	Forest Plantation	0.00	18.10	33.84	338.85	5.82	0.04	10.41	5.43	0.02	0.59	0.10	0.00
	Grassland	0.00	1.04	12.52	8.43	4314.52	0.00	61.95	0.02	1.25	0.00	80.15	0.48
	Rocky outcrop	0.00	5.02	14.15	0.01	0.00	318.52	0.70	11.67	3.03	4.14	5.72	0.15
	Pasture	0.00	419.26	532.49	38.19	151.43	5.77	6214.39	359.96	32.66	1.28	14.24	4.22
	Agriculture	0.00	382.40	63.66	30.11	0.00	129.07	525.10	1273.20	52.22	7.75	9.96	3.00
	Urban Infrastructure	0.00	0.03	0.02	0.00	0.00	0.01	0.00	0.02	220.45	0.05	0.00	0.01
	Mining	0.00	0.30	0.97	0.02	0.00	6.67	0.66	1.65	3.27	87.68	5.27	1.02
	Other Non Vegetated Area	0.00	2.63	7.67	0.69	83.36	18.23	44.48	10.56	14.34	0.32	372.76	1.80
	River. Lake and Ocean	0.00	0.93	6.23	0.06	0.07	0.03	1.97	0.10	0.44	0.20	0.36	64.66

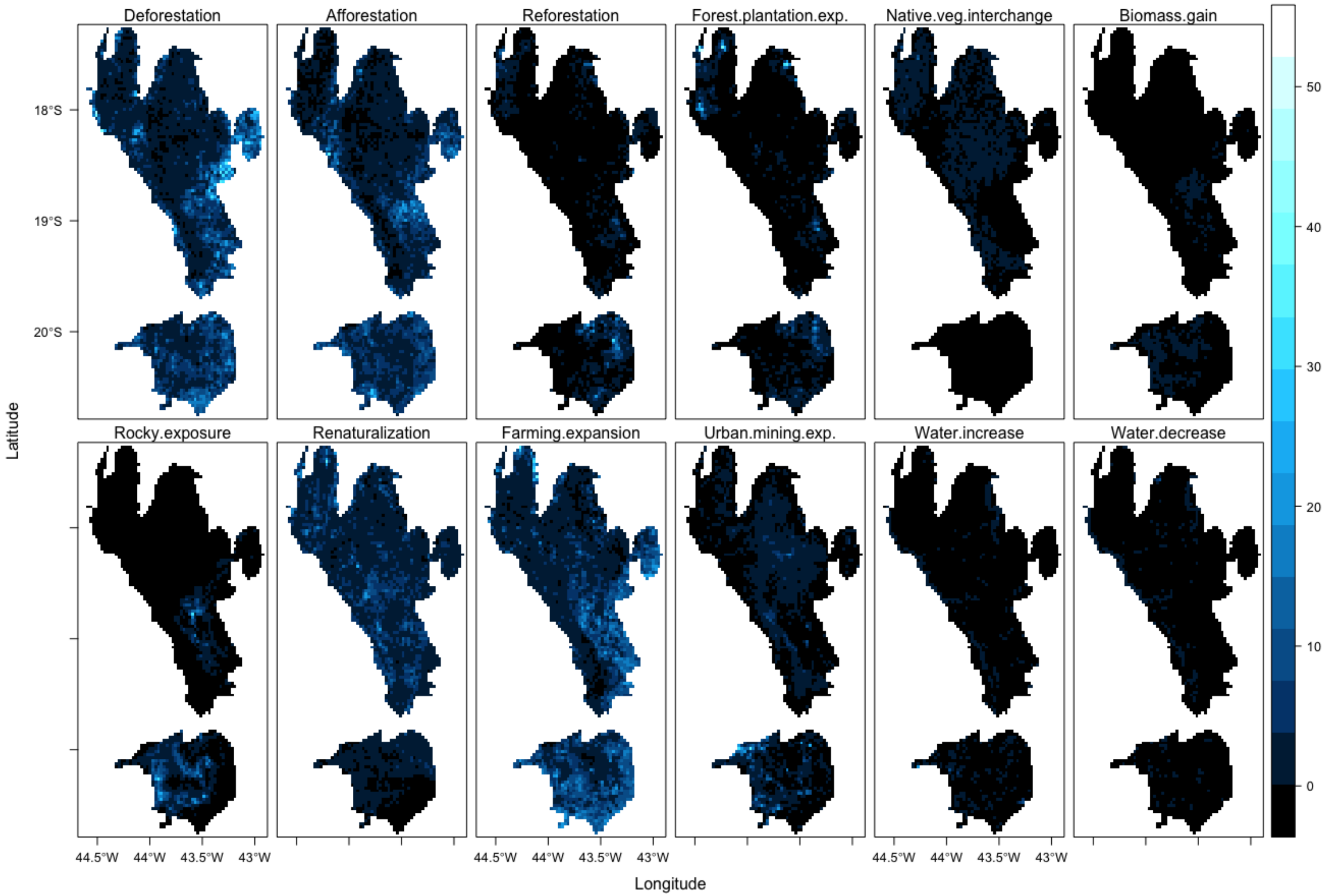
II.a

Classes		2019											
		No class	**Forest Formation**	**Savanna Formation**	**Forest Plantation**	**Grassland**	**Rocky outcrop**	**Pasture**	**Agriculture**	**Urban Infrastructure**	**Mining**	**Other Non Vegetated Area**	**River. Lake and Ocean**
2005	**No class**	100.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
	Forest Formation	0.0%	90.5%	1.4%	1.9%	0.0%	0.1%	3.2%	2.5%	0.1%	0.2%	0.1%	0.0%
	Savanna Formation	0.0%	3.7%	87.5%	2.6%	0.2%	0.5%	4.5%	0.7%	0.1%	0.0%	0.1%	0.1%
	Forest Plantation	0.0%	1.9%	0.3%	97.0%	0.0%	0.0%	0.2%	0.5%	0.0%	0.0%	0.0%	0.0%
	Grassland	0.0%	0.0%	0.3%	0.6%	94.2%	0.0%	2.1%	0.0%	0.0%	0.0%	2.8%	0.0%
	Rocky outcrop	0.0%	1.8%	3.5%	0.2%	0.0%	82.5%	0.8%	5.0%	1.3%	2.2%	2.6%	0.1%
	Pasture	0.0%	3.8%	4.6%	1.8%	1.2%	0.1%	80.6%	7.1%	0.3%	0.0%	0.5%	0.0%
	Agriculture	0.0%	13.9%	6.0%	2.6%	0.0%	3.0%	12.4%	59.8%	1.5%	0.3%	0.4%	0.1%
	Urban Infrastructure	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	99.3%	0.6%	0.0%	0.0%
	Mining	0.0%	0.2%	0.5%	0.1%	0.0%	0.6%	0.4%	1.2%	3.8%	91.3%	0.9%	0.9%
	Other Non Vegetated Area	0.0%	0.3%	0.5%	0.3%	13.1%	1.9%	2.5%	1.2%	1.0%	0.8%	78.4%	0.1%
	River. Lake and Ocean	0.0%	1.8%	5.0%	1.9%	0.5%	0.6%	5.7%	0.8%	0.4%	0.8%	1.8%	80.8%

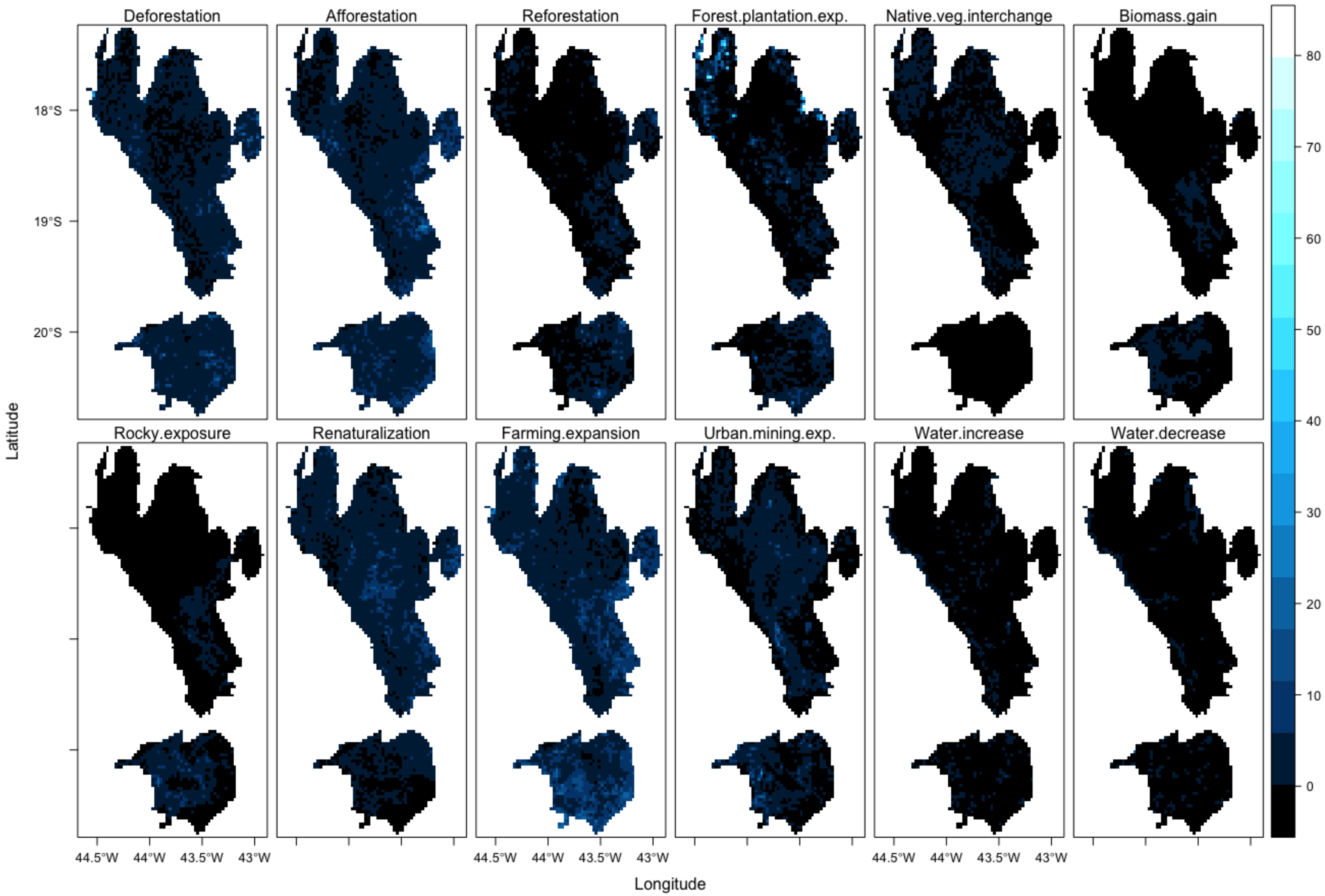
II.b

Classes		2019											
		No class	Forest Formation	Savanna Formation	Forest Plantation	Grassland	Rocky outcrop	Pasture	Agriculture	Urban Infrastructure	Mining	Other Non Vegetated Area	River. Lake and Ocean
2005	No class	44357.57	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Forest Formation	0.00	7571.04	117.75	162.25	0.48	6.89	263.68	206.52	9.24	15.82	4.65	3.39
	Savanna Formation	0.00	305.89	7319.95	221.08	13.36	40.10	378.67	57.80	6.60	3.94	7.77	7.26
	Forest Plantation	0.00	12.48	2.17	653.41	0.19	0.04	1.48	3.15	0.25	0.23	0.06	0.04
	Grassland	0.00	0.68	15.21	26.32	4313.97	0.00	96.09	0.29	0.66	0.00	126.04	0.18
	Rocky outcrop	0.00	10.09	19.66	1.21	0.00	468.38	4.63	28.55	7.56	12.68	14.63	0.31
	Pasture	0.00	308.68	379.68	147.12	98.54	9.56	6584.26	580.84	20.78	2.16	38.48	1.54
	Agriculture	0.00	287.61	125.06	54.00	0.02	61.18	257.96	1239.71	30.60	7.14	8.29	1.12
	Urban Infrastructure	0.00	0.11	0.04	0.00	0.00	0.03	0.02	0.21	342.04	2.11	0.02	0.02
	Mining	0.00	0.23	0.56	0.12	0.00	0.75	0.46	1.45	4.57	108.57	1.13	1.02
	Other Non Vegetated Area	0.00	1.40	2.55	1.34	65.40	9.63	12.37	5.74	4.81	3.88	391.21	0.59
	River. Lake and Ocean	0.00	1.58	4.45	1.65	0.46	0.56	5.05	0.71	0.35	0.68	1.58	72.05

Appendix S7: Land-use/cover transitions within the RBSE from 1985 to 2005.



Appendix S8: Land-use/cover transitions within the RBSE from 2005 to 2019.

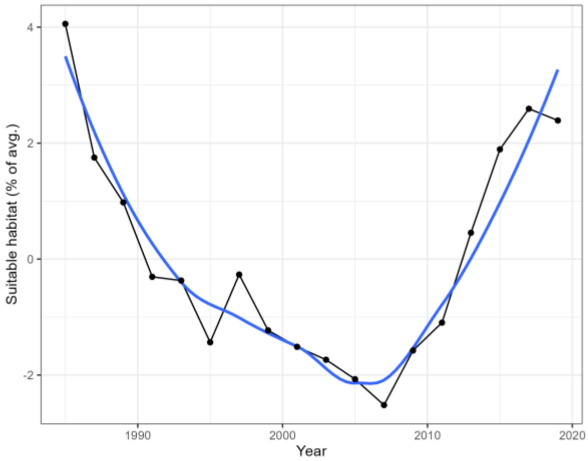


Appendix S9: Model performance statistics for the final Ensemble Models. The results were averaged across the test sets, pseudo-absence sets and model runs. Two evaluation measures were calculated: AUC (Area Under the Curve – [0, 1]), and TSS (True Skill Statistic – [-1, 1]). Column "Threshold" indicates the value used to partition ensemble probability maps into suitable/unsuitable areas. Sensitivity refers to the true positive rate and Specificity refers to the true negative rate.

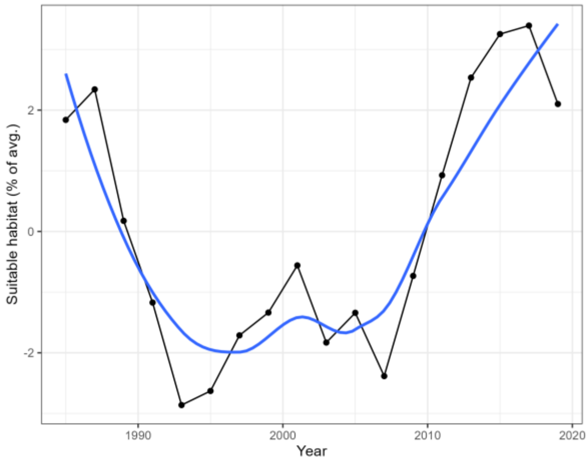
Fauna group	Species	AUC	TSS	Threshold	Sensitivity	Specificity
Amphibians	*Aplastodiscus heterophonicus*	0.996	0.962	678	100	96.250
Amphibians	*Boana botumirim*	0.998	0.984	842	100	98.438
Amphibians	*Boana cipoensis*	1	1	933	100	100
Amphibians	*Bokermannohyla alvarengai*	0.996	0.945	601	100	94.524
Amphibians	*Bokermannohyla martinsi*	1	0.997	858	100	99.667
Amphibians	*Bokermannohyla nanuzae*	0.998	0.977	856	100	97.727
Amphibians	*Bokermannohyla oxente*	0.999	0.973	710	100	97.250
Amphibians	*Bokermannohyla saxicola*	0.994	0.951	542	100	95.114
Amphibians	*Crossodactylus trachystomus*	0.982	0.885	904	90.323	98.214
Amphibians	*Hylodes uai*	0.998	0.975	843	100	97.500
Amphibians	*Leptodactylus camaquara*	0.995	0.938	660	97.619	96.154
Amphibians	*Odontophrynus juquinha*	0.998	0.983	938	100	98.333
Amphibians	*Phasmahyla jandaia*	0.994	0.938	660	100	93.750
Amphibians	*Physalaemus orophilus*	0.995	0.985	890	100	98.500
Amphibians	*Pithecopus megacephalus*	0.997	0.965	667	100	96.500
Amphibians	*Pristimantis rupicola*	1	1	901.5	100	100
Amphibians	*Proceratophrys cururu*	0.990	0.919	659	100	91.875
Amphibians	*Pseudopaludicola mineira*	0.999	0.994	949	100	99.375
Amphibians	*Scinax curicica*	0.995	0.941	611	98.81	95.263
Amphibians	*Scinax machadoi*	0.991	0.929	687	97.938	95
Amphibians	*Scinax montivagus*	1	1	928	100	100
Amphibians	*Thoropa megatympanum*	0.995	0.943	513	99.091	95.175
Birds	*Asthenes luizae*	1	0.988	616	100	98.800
Birds	*Asthenes moreirae*	1	1	626	100	100
Birds	*Augastes lumachella*	0.998	0.977	906	100	97.667
Birds	*Augastes scutatus*	0.999	0.977	766	99.153	98.500
Birds	*Campylopterus diamantinensis*	0.999	0.993	882	100	99.286
Birds	*Cinclodes espinhacensis*	1	0.998	859	100	99.778
Birds	*Embernagra longicauda*	0.993	0.925	816	95.213	97.253
Birds	*Formicivora grantsaui*	1	1	820	100	100
Birds	*Polystictus superciliaris*	0.998	0.965	420	100	96.508
Reptiles	*Amphisbaena acangaoba*	1	1	937	100	100
Reptiles	*Eurolophosaurus nanuzae*	0.995	0.962	783	100	96.250
Reptiles	*Hydromedusa maximiliani*	0.985	0.900	643	100	90
Reptiles	*Psilops paeminosus*	0.992	0.970	810	100	97
Reptiles	*Tropidurus erythrocephalus*	1	0.994	832	100	99.375
Reptiles	*Tropidurus montanus*	0.981	0.893	434	100	89.259

Appendix S10: Temporal trajectories regarding species' potential habitat relative change (%) within the Espinhaço Range between 1985 – 2019.

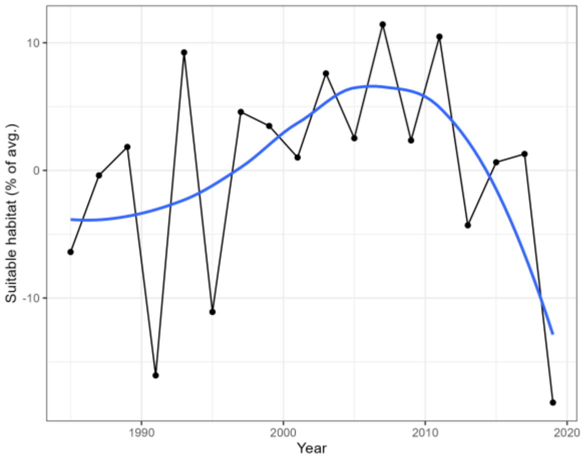
AMPHIBIANS | Aplastodiscus heterophonicus



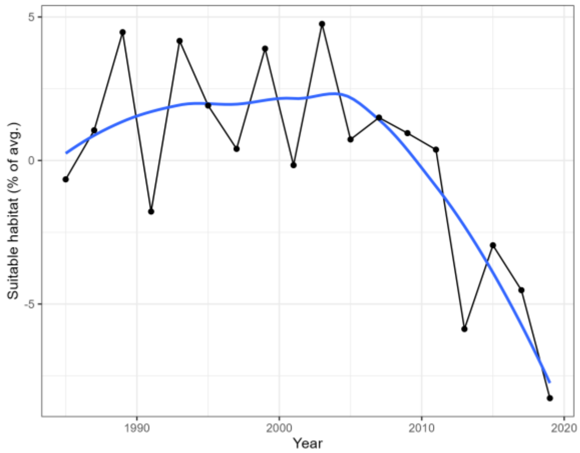
AMPHIBIANS | Boana botumirim



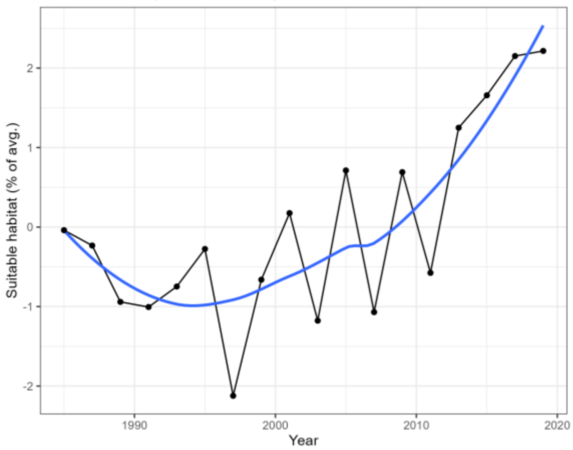
AMPHIBIANS | Boana cipoensis



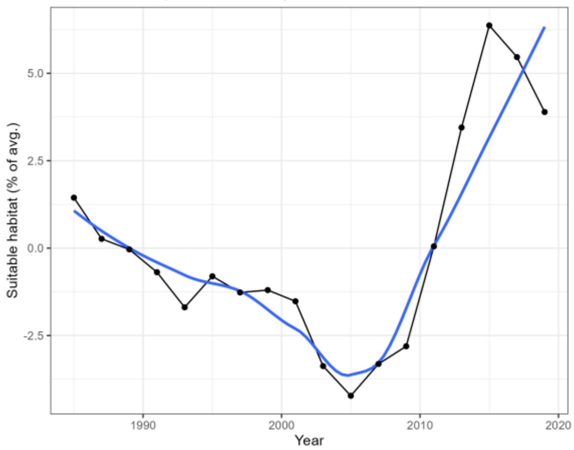
AMPHIBIANS | Bokermannohyla alvarengai



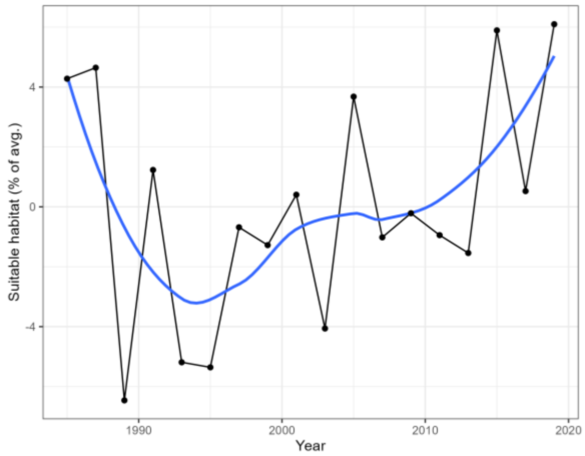
AMPHIBIANS | Bokermannohyla martinsi



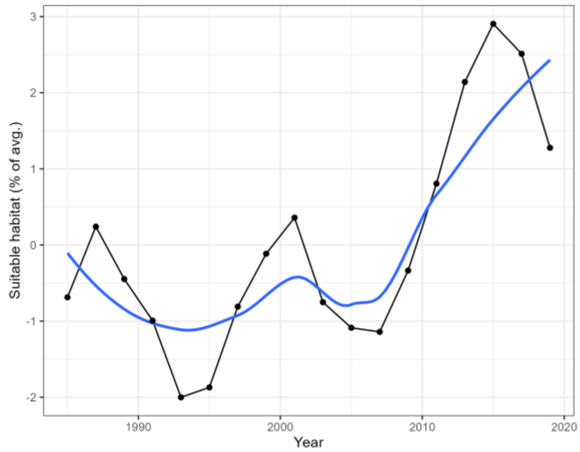
AMPHIBIANS | Bokermannohyla nanuzae



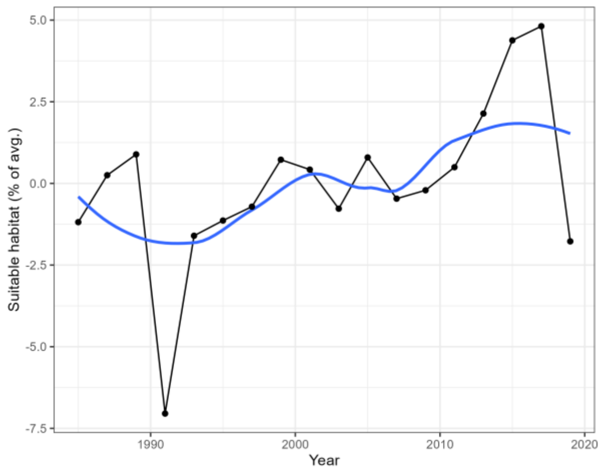
AMPHIBIANS | Bokermannohyla oxente



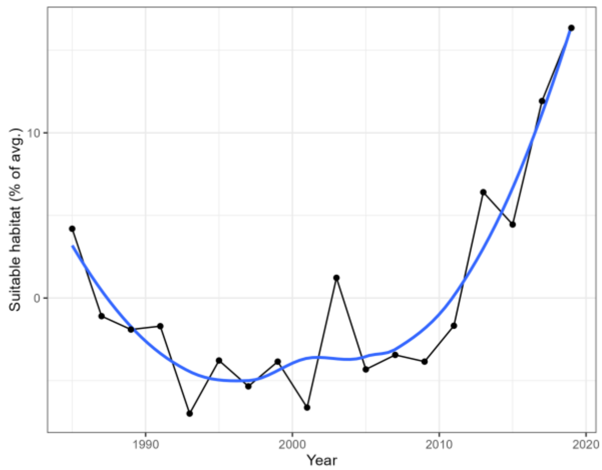
AMPHIBIANS | Bokermannohyla saxicola



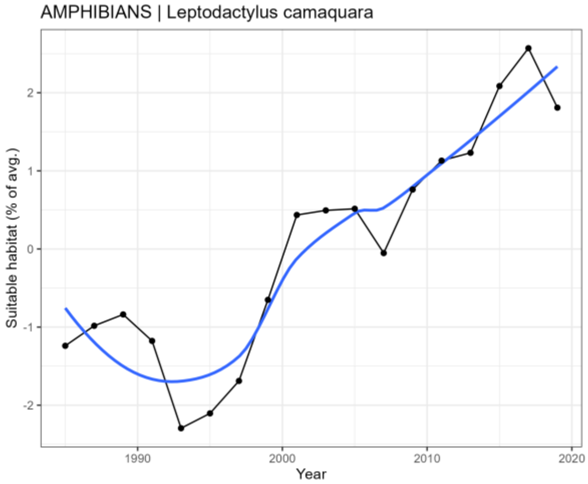
AMPHIBIANS | Crossodactylus trachystomus



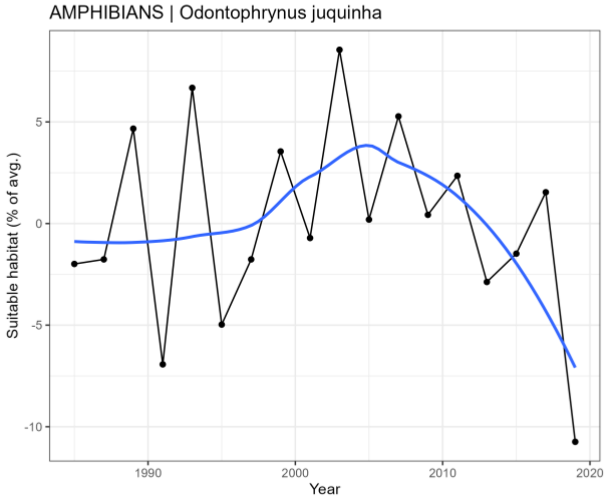
AMPHIBIANS | Hylodes uai

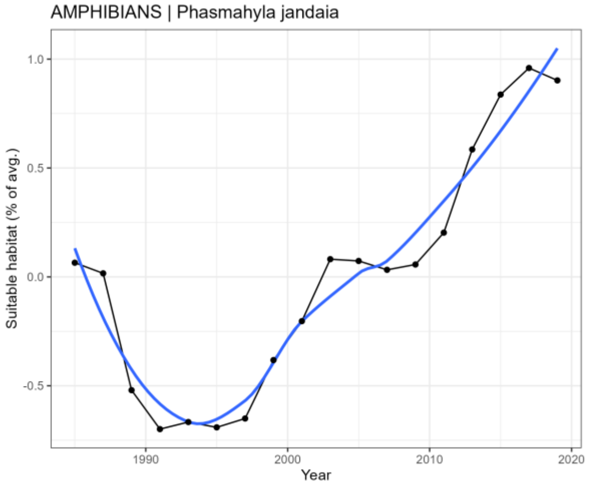


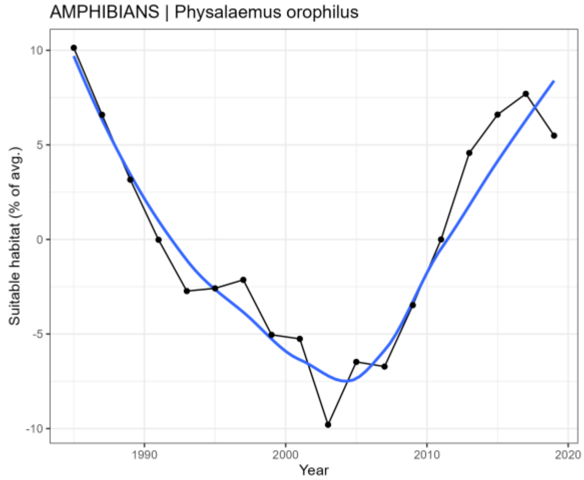
AMPHIBIANS | Leptodactylus camaquara

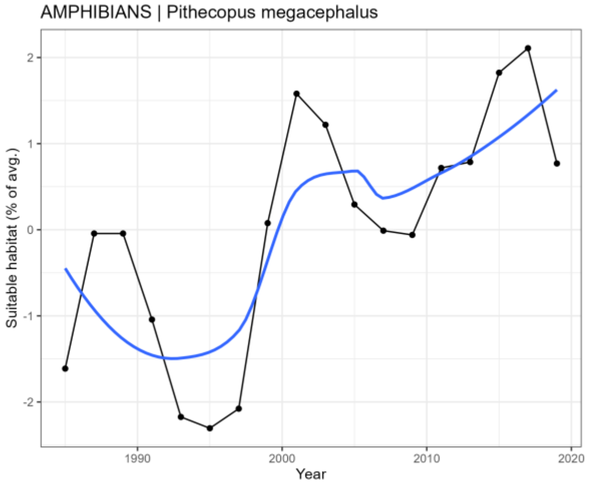


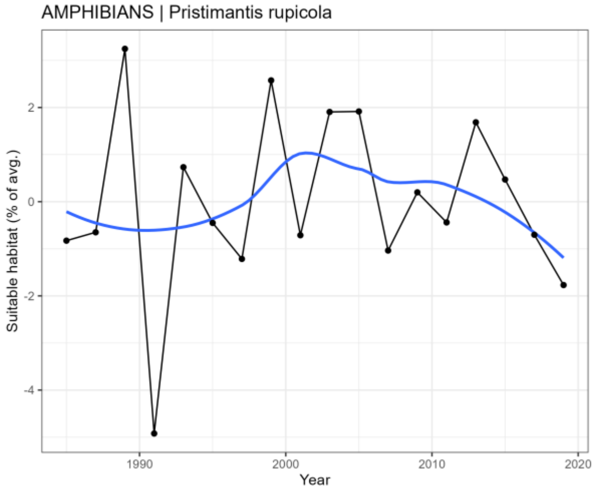
AMPHIBIANS | Odontophrynus juquinha

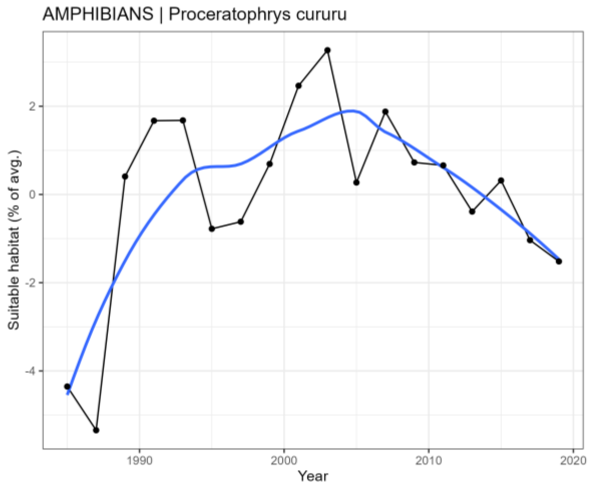


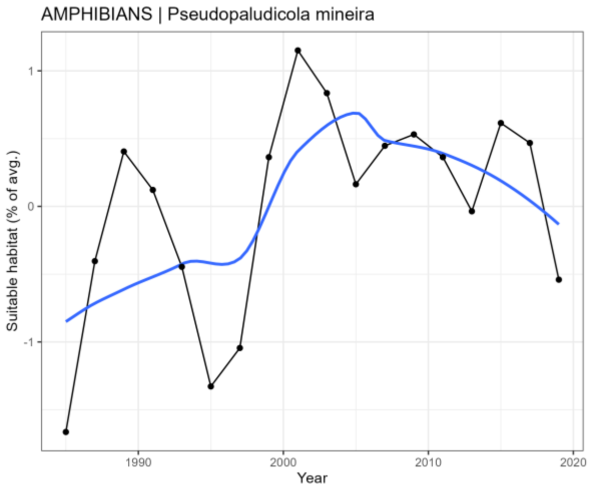

AMPHIBIANS | Phasmahyla jandaia
Suitable habitat (% of avg.)
Year


AMPHIBIANS | Physalaemus orophilus
Suitable habitat (% of avg.)
Year

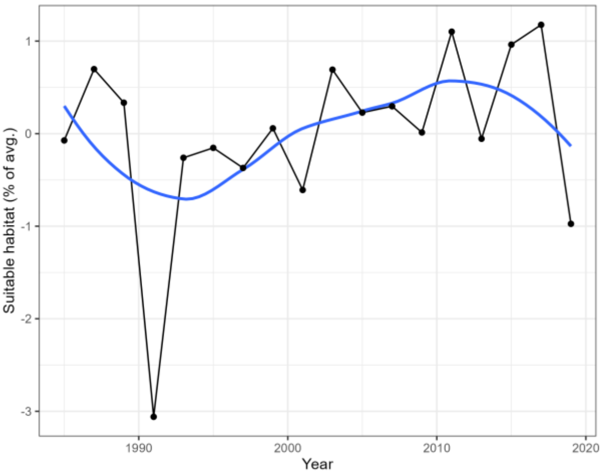

AMPHIBIANS | Pithecopus megacephalus
Suitable habitat (% of avg.)
Year


AMPHIBIANS | Pristimantis rupicola
Suitable habitat (% of avg.)
Year

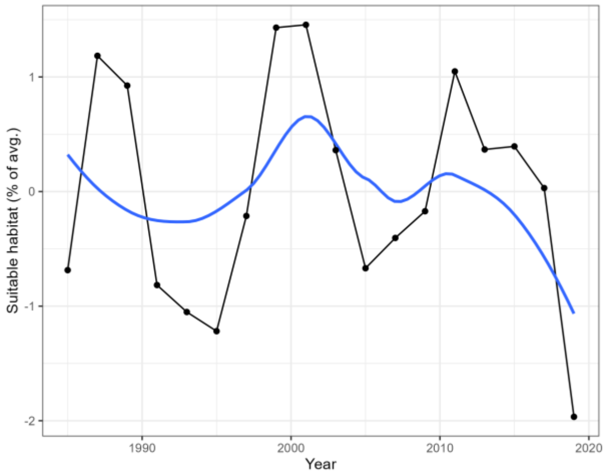

AMPHIBIANS | Proceratophrys cururu
Suitable habitat (% of avg.)
Year


AMPHIBIANS | Pseudopaludicola mineira
Suitable habitat (% of avg.)
Year

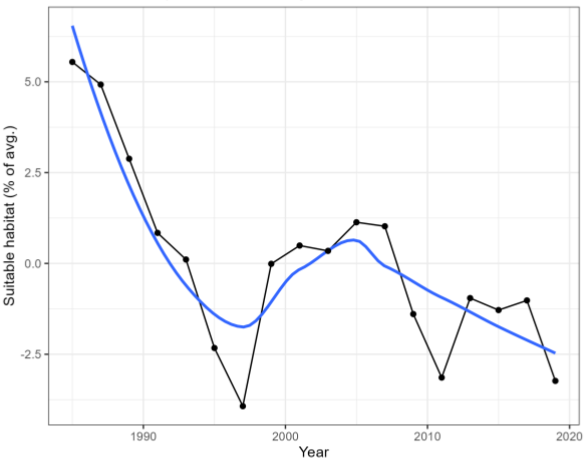
AMPHIBIANS | Scinax curicica



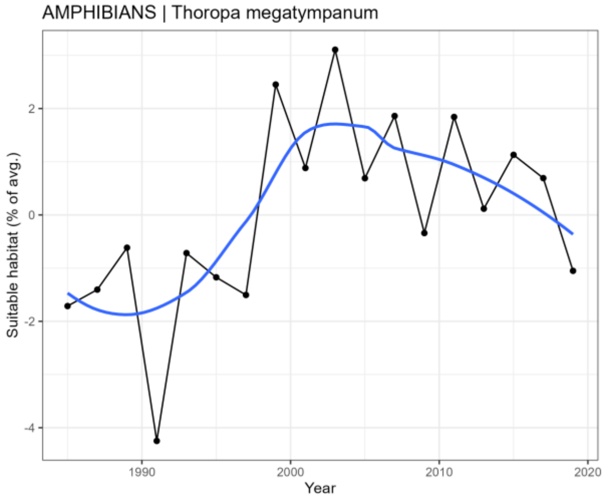
AMPHIBIANS | Scinax machadoi



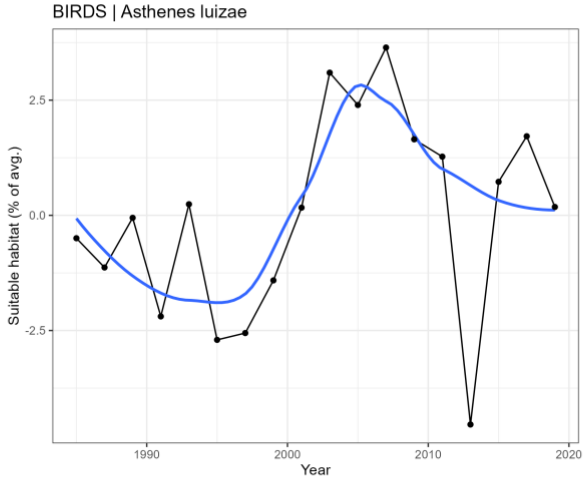
AMPHIBIANS | Scinax montivagus



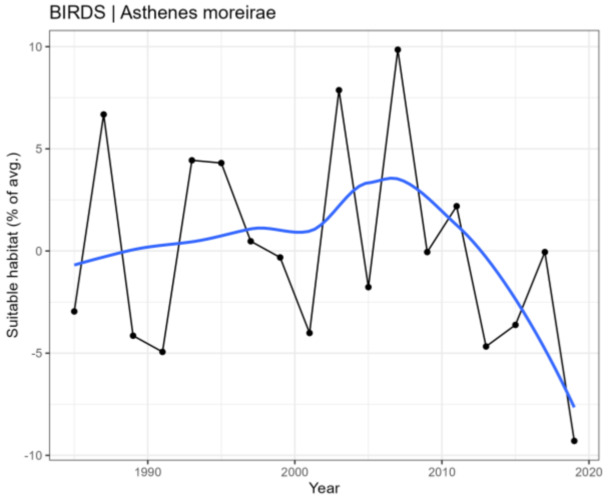
AMPHIBIANS | Thoropa megatympanum



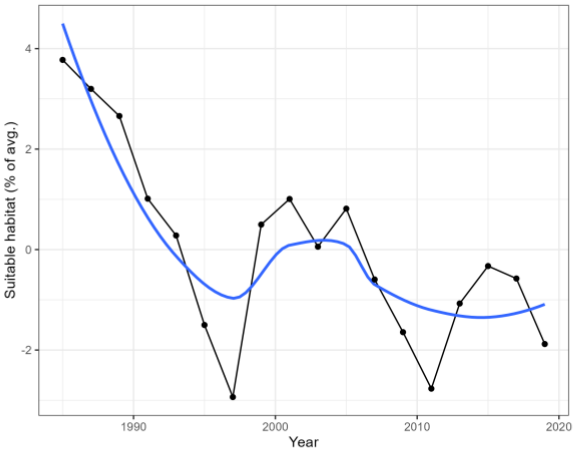
BIRDS | Asthenes luizae



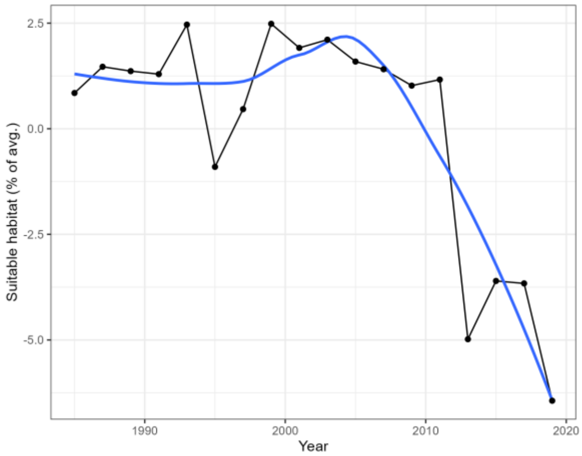
BIRDS | Asthenes moreirae



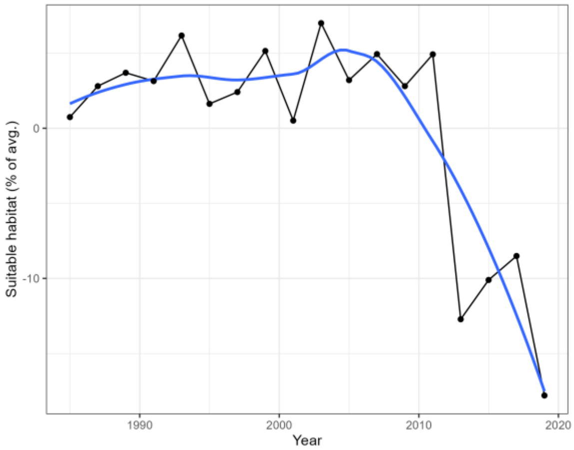
BIRDS | Augastes lumachella



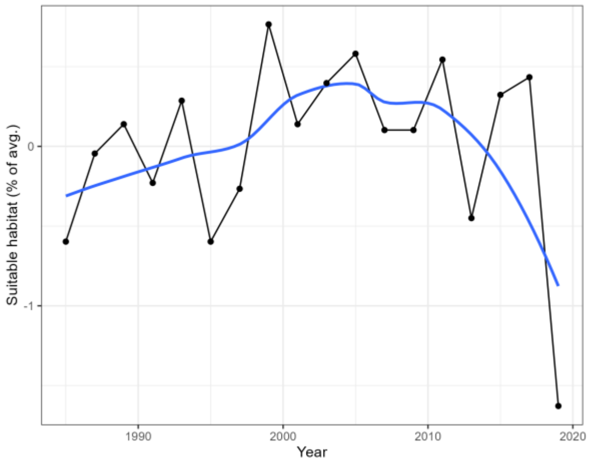
BIRDS | Augastes scutatus



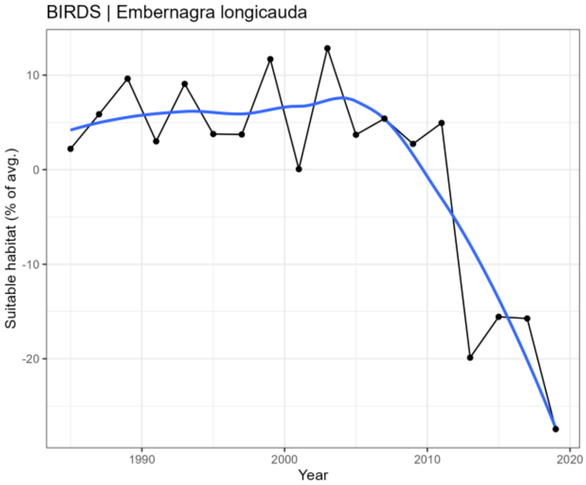
BIRDS | Campylopterus diamantinensis



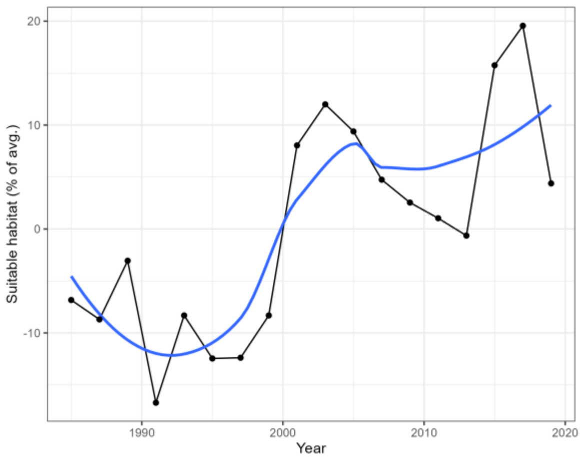
BIRDS | Cinclodes espinhacensis



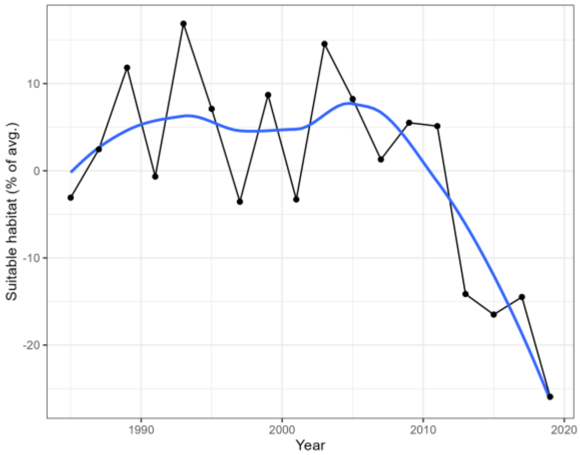
BIRDS | Embernagra longicauda



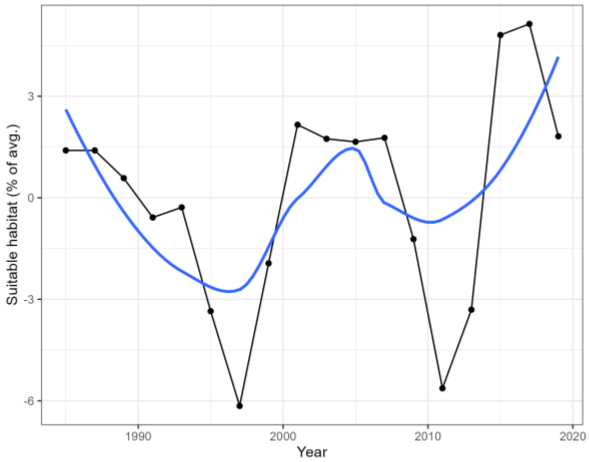
BIRDS | Formicivora grantsaui



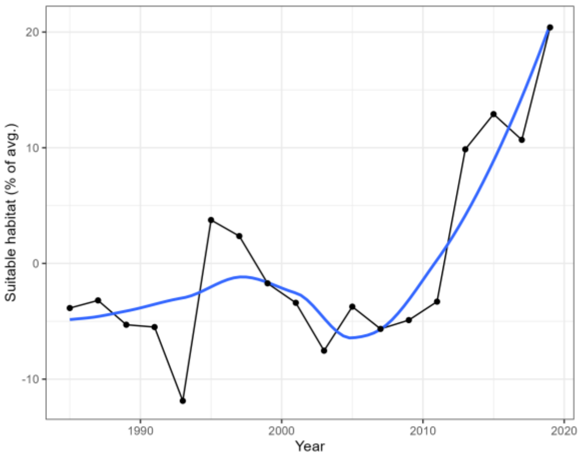
BIRDS | Polystictus superciliaris



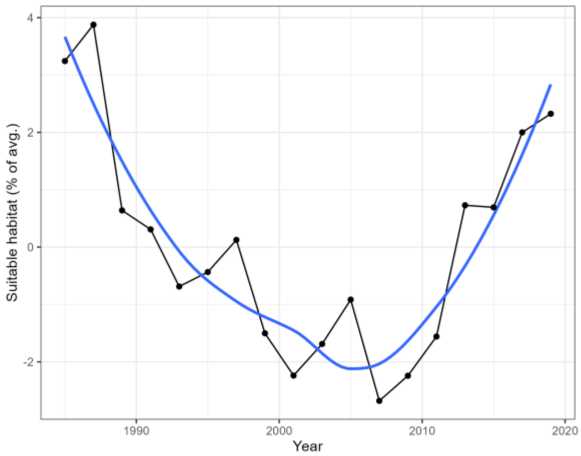
REPTILES | Amphisbaena acangaoba



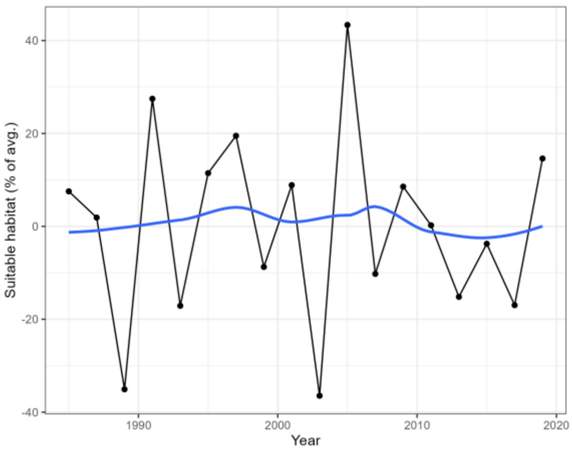
REPTILES | Eurolophosaurus nanuzae



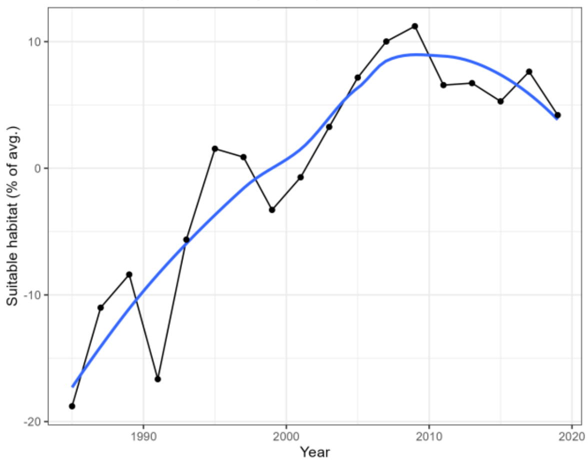
REPTILES | Hydromedusa maximiliani



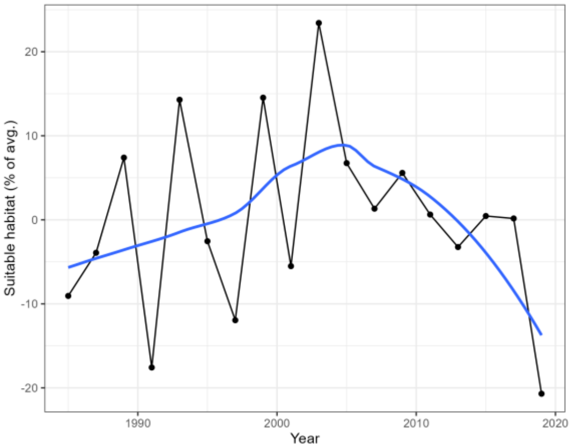
REPTILES | Psilops paeminosus



REPTILES | Tropidurus erythrocephalus

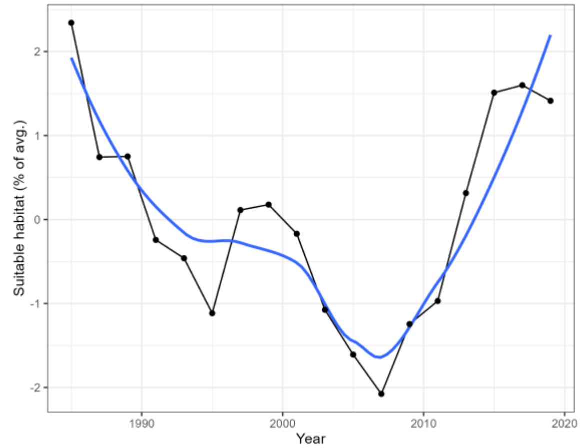


REPTILES | Tropidurus montanus

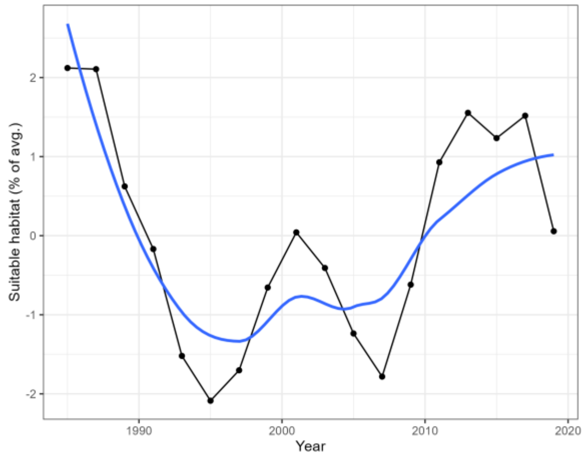


Appendix S11: Temporal trajectories regarding species' potential habitat relative change (%) within the RBSE (Phase 1 limits) between 1985 – 2019.

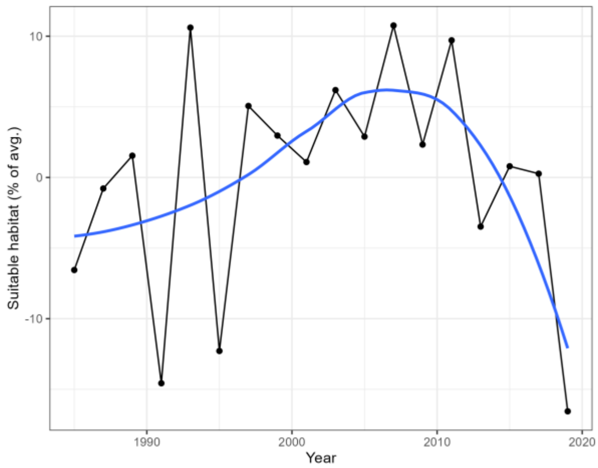
AMPHIBIANS | Aplastodiscus heterophonicus



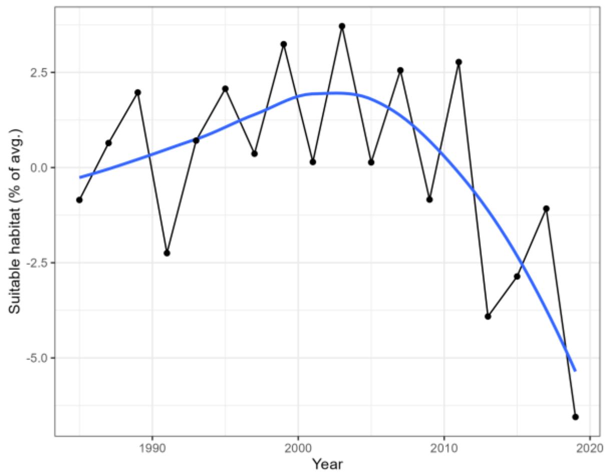
AMPHIBIANS | Boana botumirim



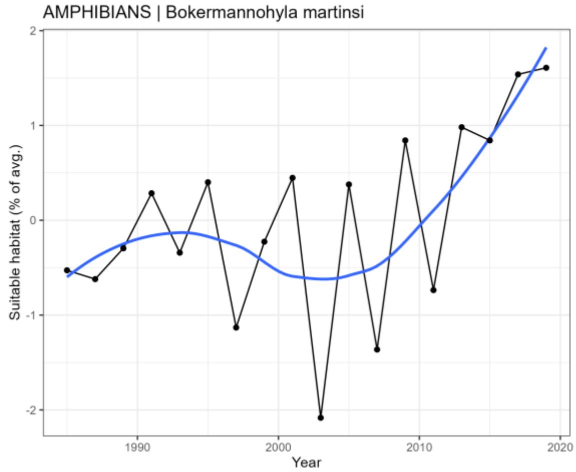
AMPHIBIANS | Boana cipoensis



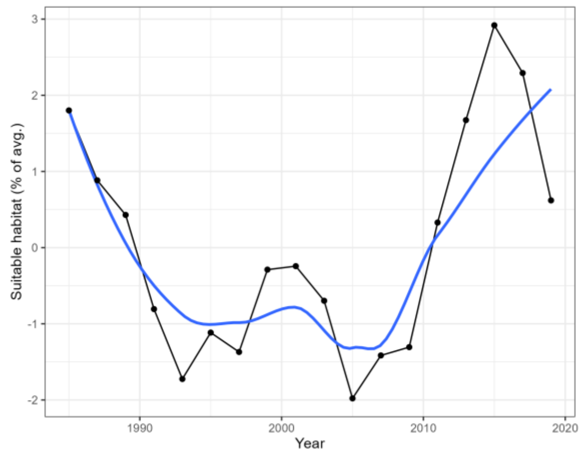
AMPHIBIANS | Bokermannohyla alvarengai



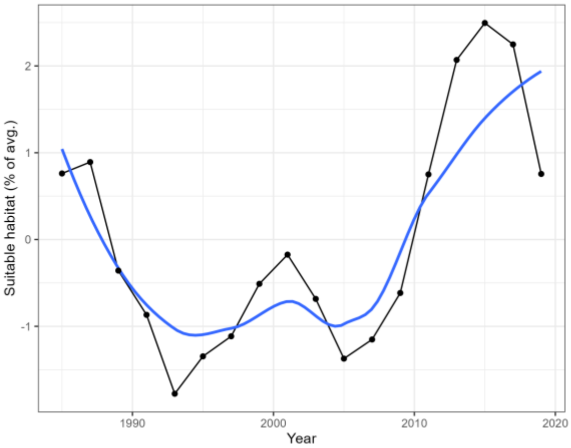
AMPHIBIANS | Bokermannohyla martinsi



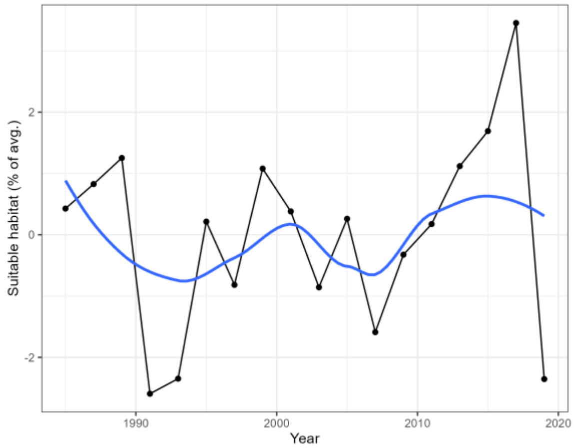
AMPHIBIANS | Bokermannohyla nanuzae



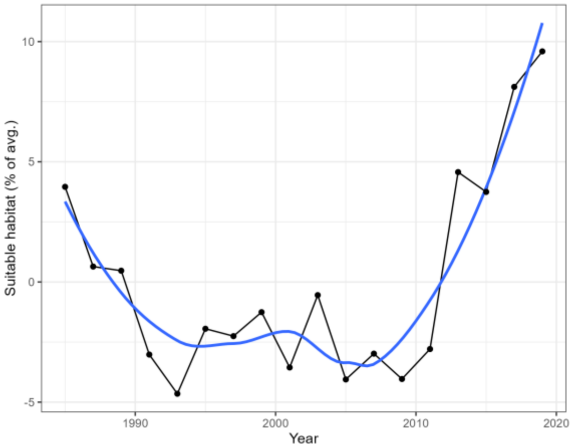
AMPHIBIANS | Bokermannohyla saxicola



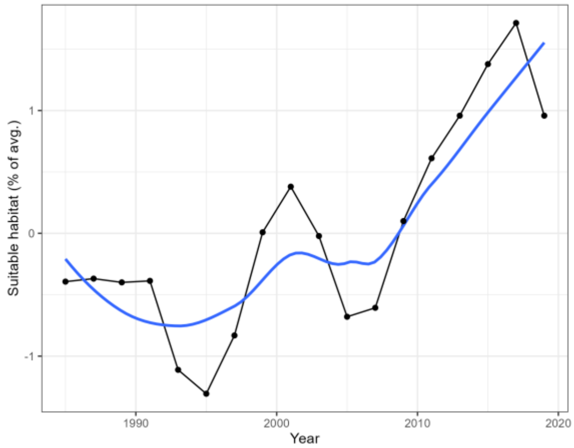
AMPHIBIANS | Crossodactylus trachystomus



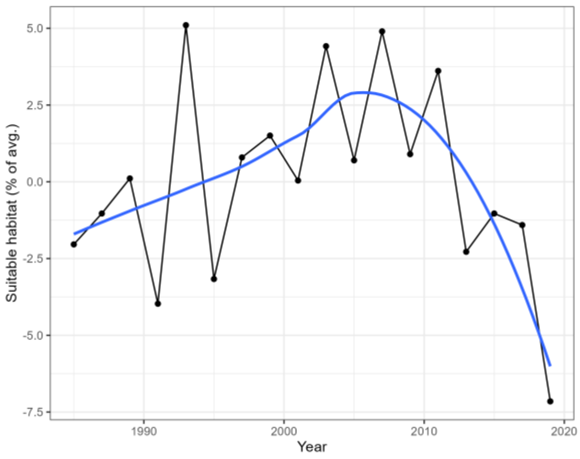
AMPHIBIANS | Hylodes uai



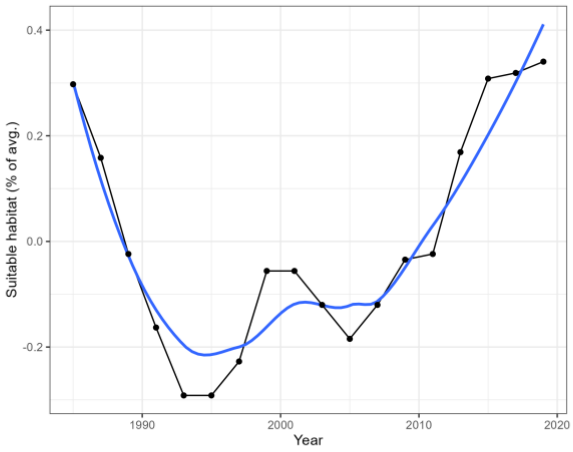
AMPHIBIANS | Leptodactylus camaquara



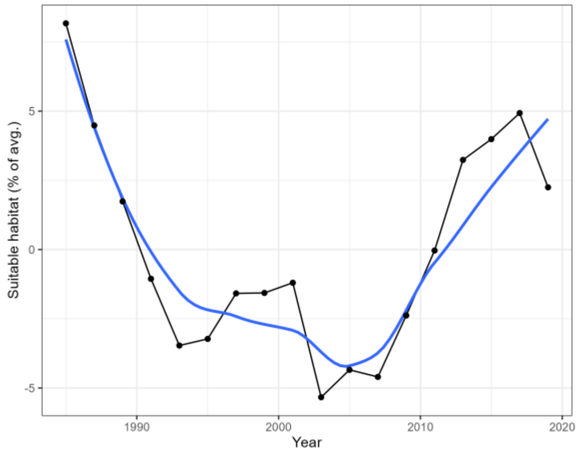
AMPHIBIANS | Odontophrynus juquinha



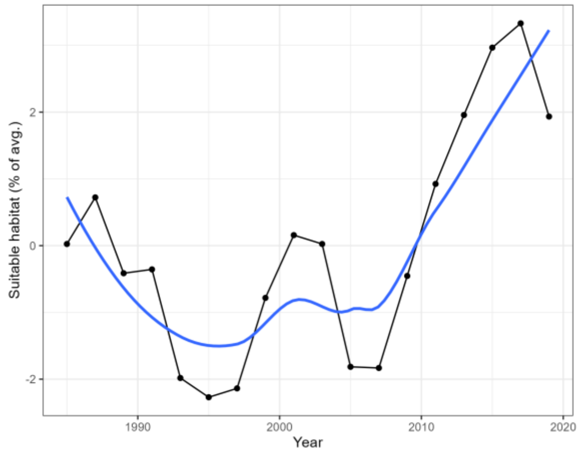
AMPHIBIANS | Phasmahyla jandaia



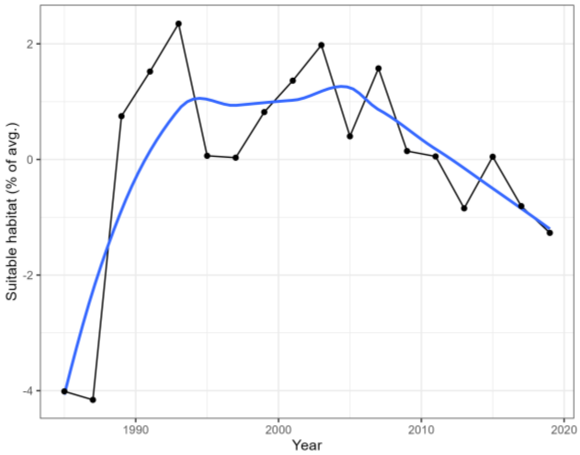
AMPHIBIANS | Physalaemus orophilus



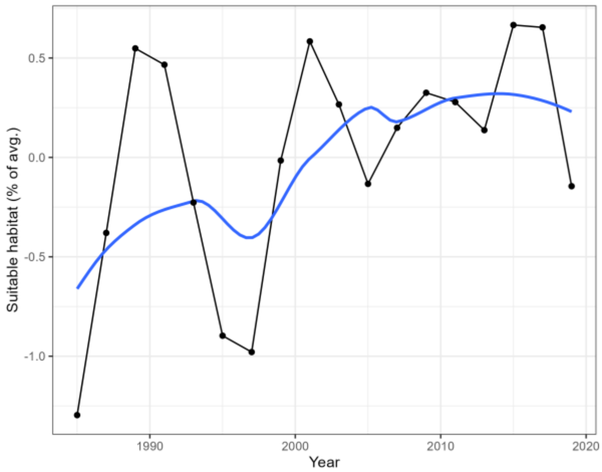
AMPHIBIANS | Pithecopus megacephalus



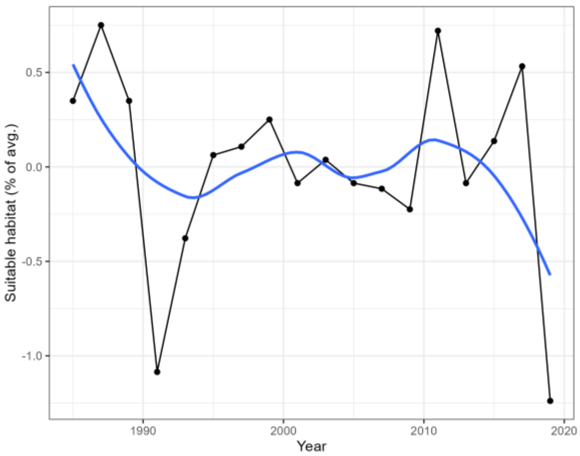
AMPHIBIANS | Proceratophrys cururu



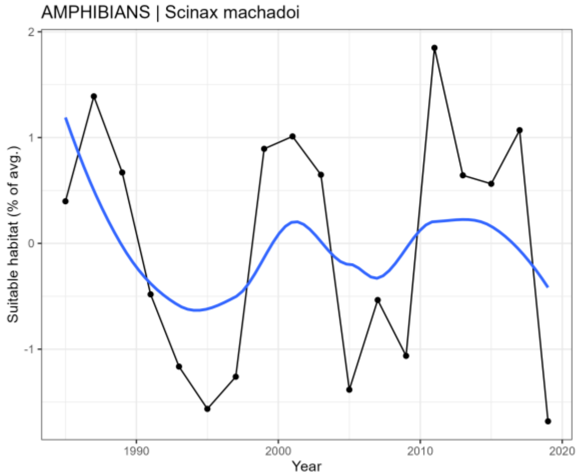
AMPHIBIANS | Pseudopaludicola mineira



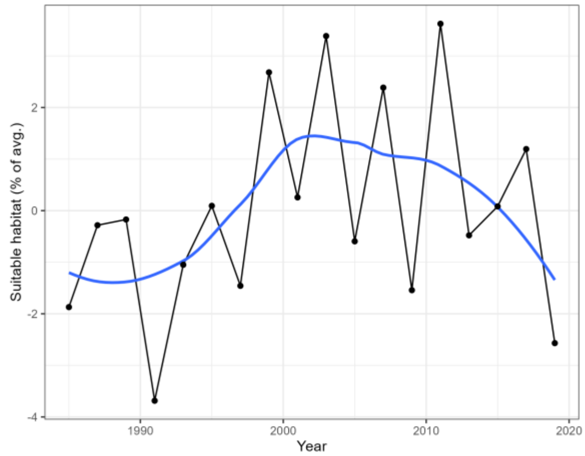
AMPHIBIANS | Scinax curicica



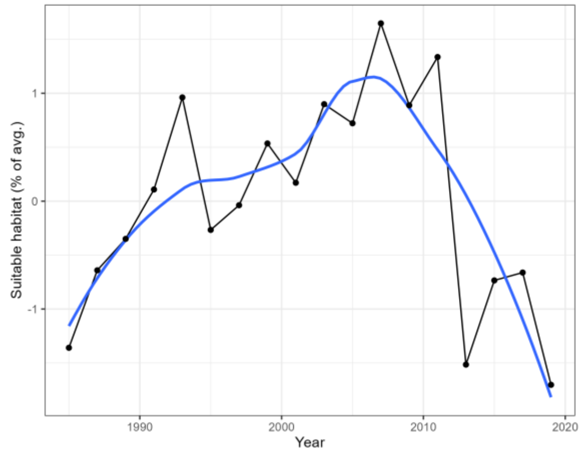
AMPHIBIANS | Scinax machadoi



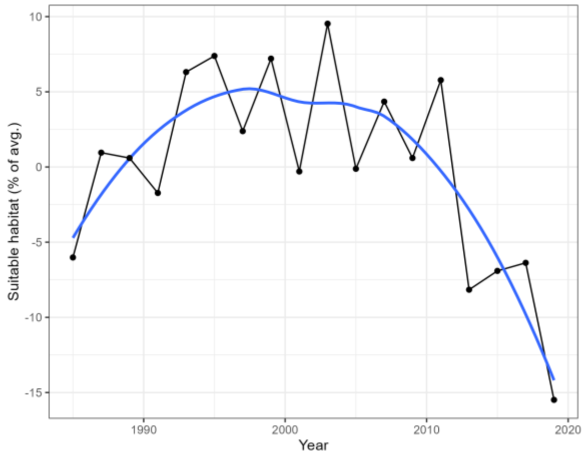
AMPHIBIANS | Thoropa megatympanum



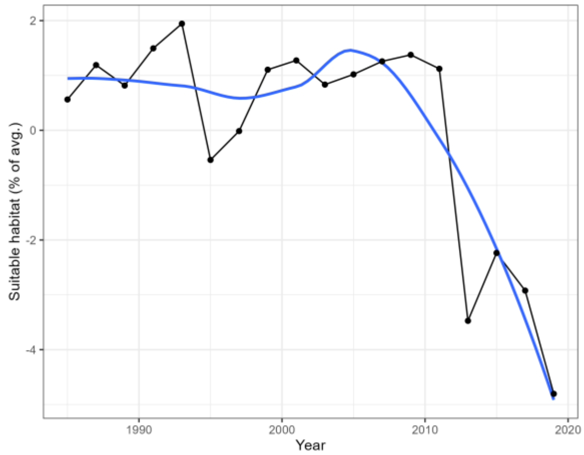
BIRDS | Asthenes luizae



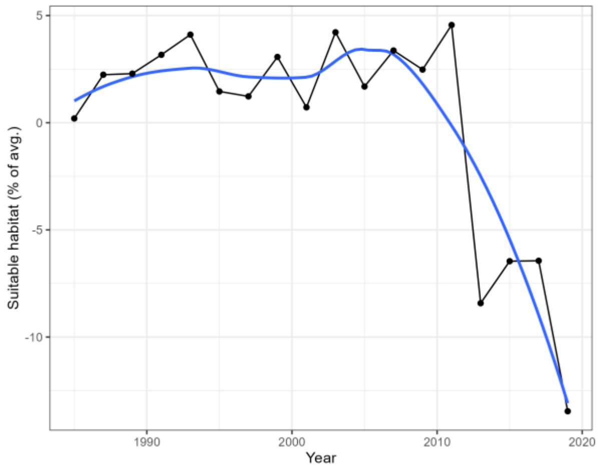
BIRDS | Asthenes moreirae



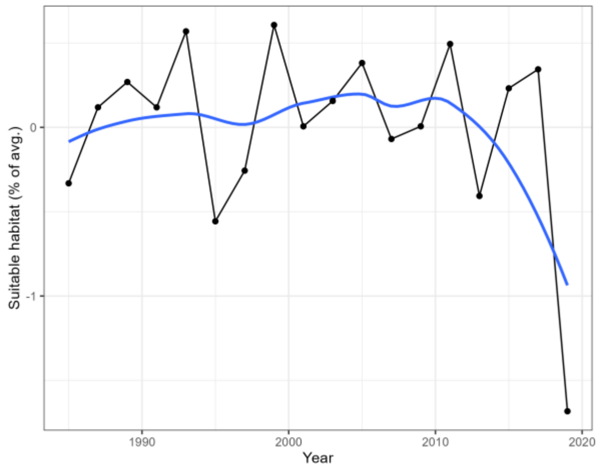
BIRDS | Augastes scutatus



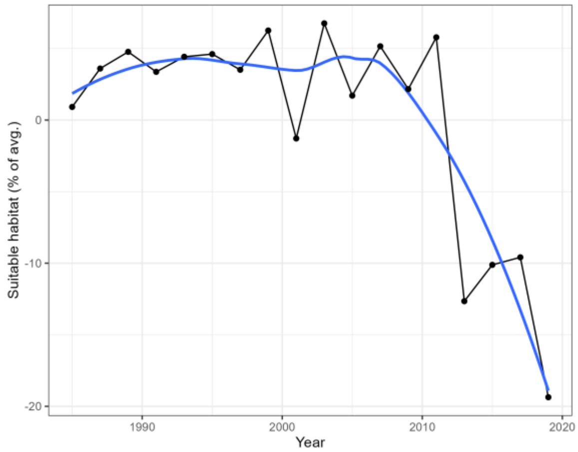
BIRDS | Campylopterus diamantinensis



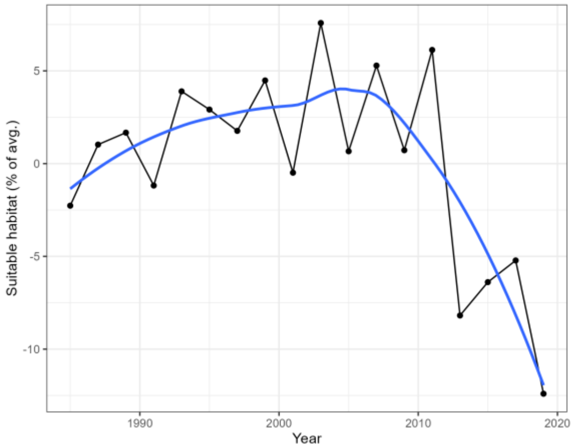
BIRDS | Cinclodes espinhacensis



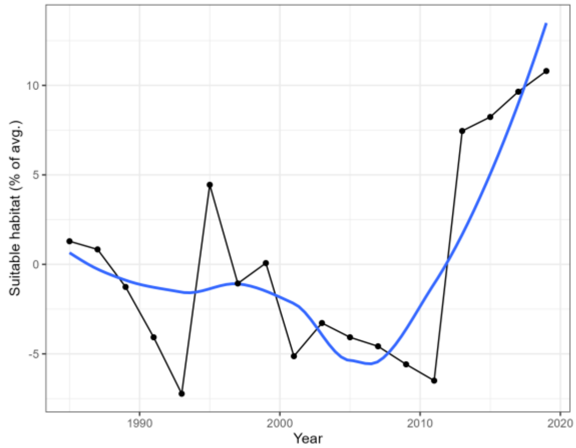
BIRDS | Embernagra longicauda



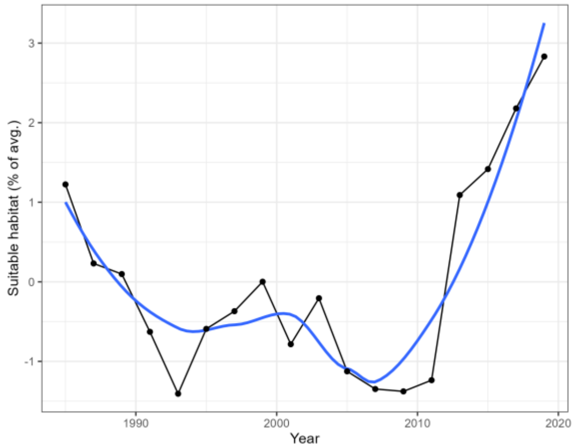
BIRDS | Polystictus superciliaris



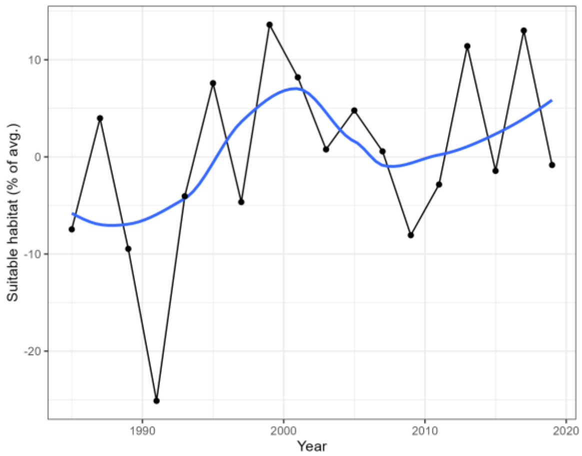
REPTILES | Eurolophosaurus nanuzae



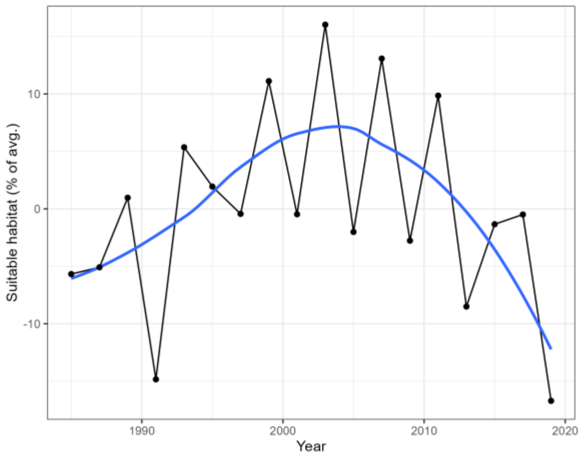
REPTILES | Hydromedusa maximiliani



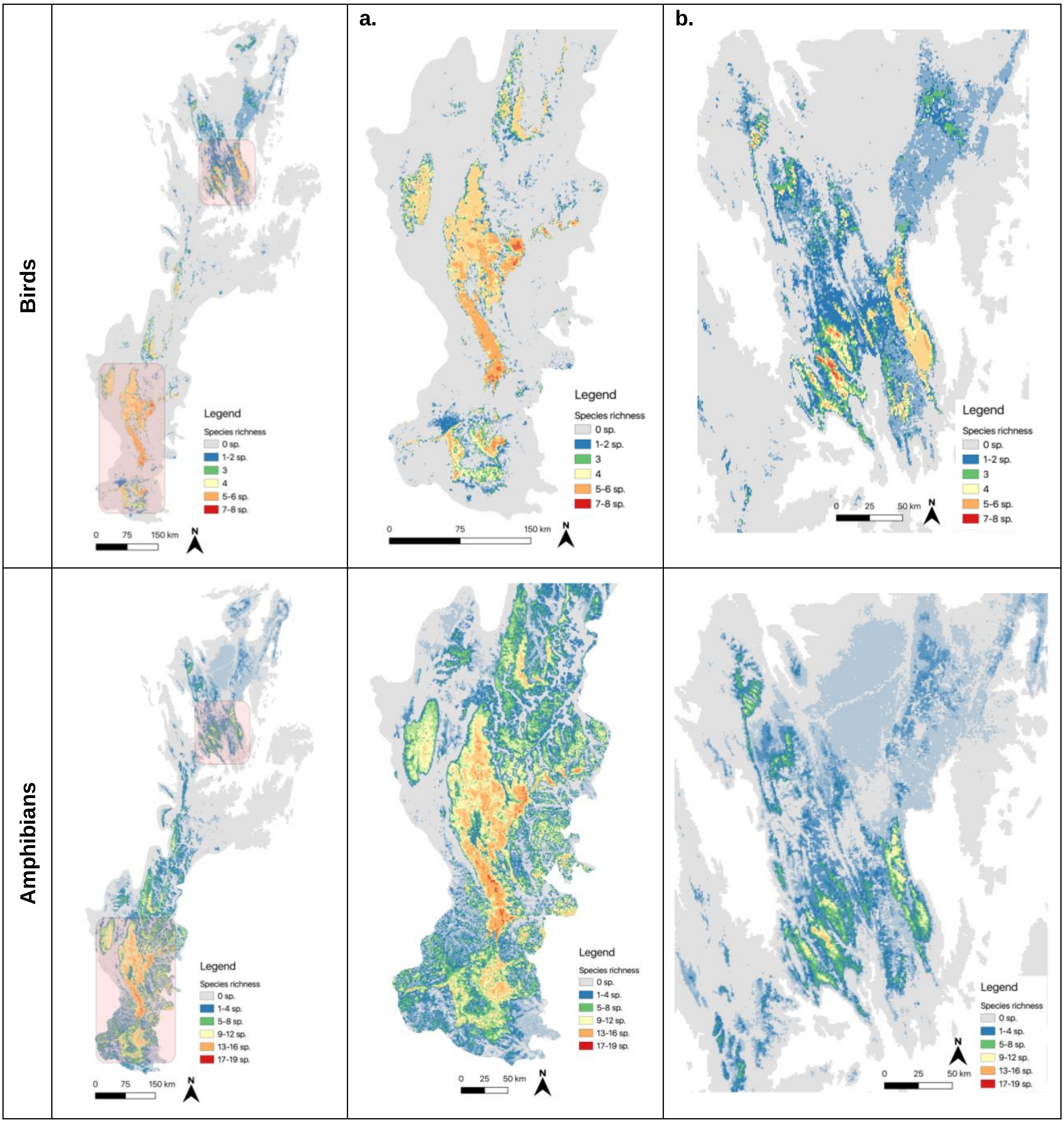
REPTILES | Psilops paeminosus



REPTILES | Tropidurus montanus



Appendix S12: Biodiversity hotspots for birds and amphibians within the Espinhaço Range. (a) displays a zoom in the southern hotspot; whereas (b) displays a zoom in the northern hotspot. The maps present species richness for current (2019) conditions.



Appendix S13: Temporal progression of the species richness for birds, amphibians and reptiles over the RBSE (Phase 1 limits) considering the years of 1985, 2005 and 2019.