Marcia C. M. Marques
Carlos E. V. Grelle *Editors*

# The Atlantic Forest

## History, Biodiversity, Threats and Opportunities of the Mega-diverse Forest

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*To all the environmentalists who defended*
*the Atlantic Forest*

# Foreword

For the past 30 years, the Atlantic Forest has been considered a global priority for biodiversity conservation, one of the top five Biodiversity Hotspots on our planet. Situated mostly within Brazil (but also in Argentina and Paraguay), which is one of the world's top two Megadiversity Countries (Mittermeier et al. 1997), the Atlantic Forest is truly a special part of the world, with exceptional levels of species diversity and endemism.

The numbers speak for themselves, more than 20,000 species of which at least 6,000 are endemic—a number higher than most countries. There are 2,645 non-fish vertebrates, including 384 mammal species, with at least 109 endemics, 1,025 birds with 215 endemics, 517 reptiles with 126 endemics, and 719 amphibians, of which a striking 504 are endemic (Figueiredo et al., this volume). Indeed, if the Atlantic Forest were a country, it would rank as a Megadiversity Country in its own right.

Diversity in certain groups of organisms is especially impressive. Tree diversity in certain parts of the region is among the very highest on Earth, with some early surveys stunning the scientific world, such as 454 arboreal species being surveyed in Bahia in a single hectare of forest (Thomas et al. 1998). This is a number that is matched only by a few sites on the eastern slope of the Tropical Andes, another Biodiversity Hotspot. In this volume, Zwiener et al. greatly expand on the magnificent tree diversity in the Atlantic Forest. Bromeliads and orchids are also exceptionally diverse. And, of course, our favorite group of animals, the non-human primates, are also extremely rich here, with 6 genera and 24 species, of which 21 are endemic, making this region one of the world's top primate conservation priorities.

Diversity and endemism per unit area are also exceptional in the Atlantic Forest, especially when we take into account how many species are packed into the relatively very small area that remains. Even without taking into account the widespread habitat loss that has taken place in the Atlantic Forest compared to Amazonia, its diversity in many groups of organisms is close to that of Amazonia, which is approximately five times larger.

As with the other hotspots, the Atlantic Forest has also suffered from widespread habitat loss over the centuries. It is where Brazil was first colonized and is today a major population center, not just for Brazil but for the entire continent, with São

Paulo being one of the two biggest cities in the Western Hemisphere. Going back to the 1980s, estimates of the remaining natural vegetation cover were as low as 7-10% (Fonseca 1985).

We usually refer to the Atlantic Forest as the Brazilian Atlantic Forest, but it is important to note that it also extends into the moist forests of eastern Paraguay and Misiones Province in northern Argentina. That said, more than 90% of the region is found within the borders of Brazil, making its conservation very largely a Brazilian concern.

Although the global importance of the Atlantic Forest is now recognized by everyone in the conservation world, that was not always the case. If one goes back just 50 years, Brazil and its vast extent of tropical rain forest wasn't even considered a high priority in our community. It was not until the early 1970s that Amazonia began to be recognized as of global significance, especially through the leadership of Dr. Thomas E. Lovejoy, then of the World Wildlife Fund – USA, as international attention focused on the impacts of the TransAmazonian Highway, popularized by Goodland and Irwin of the World Bank in 1975. The Atlantic Forest at this time was still nothing more than an afterthought, if it was discussed at all. One of the very first papers in an international journal that called attention to the plight of the Atlantic Forest was published by one of us (Fonseca 1985), as recognized by Marques et al. (this volume).

The origins of worldwide interest in the Atlantic Forest can be traced back to 1970, and it came about because of one little monkey, the Golden Lion Tamarin (*Leontopithecus rosalia*). Following a couple of short articles by Clyde Hill, Curator of Mammals at the San Diego Zoo, and John Perry, Assistant Director of the National Zoo in Washington, D.C., a number of people in the conservation community started to become concerned about the declining situation of this species and its habitat in the state of Rio de Janeiro. In July 1971, one of us (Mittermeier) travelled to Brazil and met with Prof. Adelmar F. Coimbra-Filho, the pioneer of Brazilian primatology, and together with him started to publish on this species in international journals and also on the two other then-recognized species of lion tamarin, the Golden-headed Lion Tamarin (*Leontopithecus chrysomelas*) and the Black Lion Tamarin (*Leontopithecus chrysopygus*), which Coimbra had rediscovered in 1970 in the states of Bahia and São Paulo, respectively, after neither had been seen for almost 70 years. There followed a series of international conferences on these animals, the first at the National Zoo in Washington, D.C., in February 1972, with many others to follow.

At the end of the decade, Mittermeier began a decade-long survey of primates in protected areas of the Atlantic Forest, together with Coimbra, Prof. Célio Valle of the Universidade Federal de Minas Gerais (UFMG), and Almirante Ibsen de Gusmão Câmara of the Fundação Brasileira para a Conservação da Natureza (FBCN) in Rio de Janeiro, and with the active participation of one of us (Fonseca), then a graduate student working on primates – all of it funded by the World Wildlife Fund – USA. Early on, this program focused heavily on the two endemic genera, *Leontopithecus* and the muriquis (*Brachyteles*), turning them both into flagship species comparable in value to the giant panda of China, the gorillas of Africa, and the orangutans of Southeast Asia. What is more, this research program recognized the importance of the Atlantic Forest region as a whole, with the result that Mittermeier introduced it as a priority region in the World Wildlife Fund's Global Campaign for Primates and Tropical Forests, which began in 1982.

There followed a series of scientific and popular publications, led by the two of us but in collaboration with many other partners, in which we highlighted repeatedly the importance of this region (Coimbra-Filho and Mittermeier 1973a, b; Coimbra-Filho et al. 1975; Mittermeier and Coimbra-Filho 1977; Mittermeier et al. 1982; Fonseca 1985; Tabarelli et al. 2005).

In 1983, at a meeting of senior staff from WWF–US to set priorities for its Latin America Program, the Atlantic Forest along with the Tropical Andes were recognized as the two highest priorities in the region. This meeting was in many ways a precursor to the Forest Hotspots analysis, first published by Dr. Norman Myers 1988, in which he recognized the Atlantic Forest as one of the top 10 on Earth. Following input from a number of us, he published an update in 1990, recognizing 18 of what were then called Biodiversity Hotspots. In July 1989, Mittermeier took the Hotspots concept to the fledgling Conservation International and made it the central focus of the organization for the next 20 years. Fonseca joined Conservation International in 1990 as Executive Director of its Brazil Program, with the Atlantic Forest as the top priority. In parallel, Fonseca led the establishment of the first graduate program in Ecology, Conservation and Wildlife Management at the UFMG focusing especially on the Atlantic Forest. One of the editors of this volume, C. E. V. Grelle, received his Master's degree from that program.

Thanks to the John D. and Catharine T. MacArthur Foundation, which also chose the hotspots as its central organizing principle for its new conservation program, Conservation International was able to secure a major grant for its work in the Atlantic Forest, along with grants to several young in-country conservation organizations (e.g., SOS Mata Atlântica in São Paulo, the Fundação Biodiversitas, Belo Horizonte, and the Sociedade de Pesquisa em Vida Selvagem (SPVS), in Curitiba, among many others).

Conservation International also began to publish a series of books, with support from the Mexican cement company CEMEX, highlighting a variety of different conservation issues. The first of these was entitled *Megadiversity* (Mittermeier et al. 1997), which recognized Brazil as one of the two top countries on Earth for its biodiversity, and this was followed by a number of others, two of them, *Hotspots* (Mittermeier et al. 1999) and *Hotspots Revisited* (Fonseca et al. 2004), prominently highlighting the Atlantic Forest. These also formed the basis of two major Conservation International fund-raising campaigns, "The Campaign to Save the Hotspots," which began in the mid-1990s, and "A Future for Life," which started with the new millennium. Together they raised approximately $2 billion.

All of these activities were instrumental in increasing international recognition of the Atlantic Forest as a truly major global conservation priority, and this continues to the present day through a very wide range activities far too numerous to cover in this brief foreword.

What is especially gratifying to those of us who can trace our Atlantic Forest roots going back 40-50 years is the incredible growth of capacity in this region, including both the scientific research capacity and the conservation capacity to create and manage protected areas, at federal, state, and municipal levels, and even in terms of private protected areas (RPPNs). There is no doubt that the Atlantic Forest can be considered a conservation success story among the world's Biodiversity Hotspots, as exemplified by the recent publication, *The Atlantic Forest: Hotspot to Hopespot*, that documented an increase in forest cover to 28% percent, or 32 million hectares, of native vegetation cover (Rezende et al. 2018). The authors estimate that just the implementation of the existing legislative framework could boost the extent of forest cover to 35%.

To be sure, the region still faces many conservation challenges and we must always be cognizant of the old adage that "there are no final victories in conservation." Nonetheless, we are confident that the large and growing cadre of Brazilian researchers and conservationists will succeed in their efforts. Indeed, there is no better indication of this than the truly amazing and highly competent group of young authors that have contributed to this historic and brilliant volume.

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## References

Coimbra-Filho AF, Mittermeier RA (1973a) New data on the taxonomy of the Brazilian marmosets of the genus *Callithrix* Erxleben, 1777. Folia Primatol 20:241–264

Coimbra-Filho AF, Mittermeier RA (1973b) Distribution and ecology of the genus *Leontopithecus* lesson, 1840 in Brazil. Primates 14(1):47–66

Coimbra-Filho AF, Magnanini A, Mittermeier RA (1975) Vanishing gold – last chance for Brazil's lion tamarins. Animal Kingdom 15(6):20–27

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

Fonseca GAB (1985) The vanishing Brazilian Atlantic Forest. Biol Conserv 34:17–34

Goodland RJA, Irwin HS (1975) Amazon jungle: green hell to Red Desert? An ecological discussion of the environmental impact of the highway construction program in the Amazon Basin. Elsevier, Amsterdam

Mittermeier RA, Coimbra-Filho AF (1977) Conservation of the Brazilian lion tamarins (*Leontopithecus rosalia*). In: H. S. H. Prince Rainier of Monaco, Bourne G (eds) Primate conservation. Academic Press, New York, pp 59–94

Mittermeier RA, Coimbra-Filho AF, Constable ID, Rylands AB, Valle C (1982) Conservation of primates in the Atlantic Forest region of eastern Brazil. Int Zoo Yearbook 22:2–17

Mittermeier RA, Robles-Gil P, Mittermeier CG (eds) (1997) Megadiversity. Earth's biologically wealthiest nations. CEMEX/Agrupación Sierra Madre, Mexico

Mittermeier RA, Fonseca GAB, Rylands AB, Mittermeier CG (1999) Atlantic forest. In: Mittermeier RA, Myers N, Robles Gil P, Mittermeier CG (eds) Hotspots: the Earth's biologically richest and most endangered terrestrial ecoregions. CEMEX/Agrupación Sierra Madre, Mexico, pp 136–147

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Strassburg BBN, Tabarelli M, Fonseca GAB, Mittermeier RA (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:208–214

Tabarelli M, Pinto LP, Silva JMC, Hirota M, Bedê L (2005) Challenges and opportunities for biodiversity conservation in the Brazilian Atlantic Forest. Conserv Biol 19(3):695–700

Thomas WW, Carvalho AMV, Amorim AM, Garrison J, Arbelaéz AL (1998) Plant endemism in two forests in southern Bahia, Brazil. Biodivers Conserv 7:311–322

# Preface

## Time to Rethink the Atlantic Forest

The object of curiosity and admiration of scientists and conservationists, the Brazilian Atlantic Forest is scientifically explored in this book. Over the 23 chapters written by 149 authors from 65 different institutions, the Atlantic Forest goes through a scanner again, updating some works already published in the last 25 years (Dean 1995; Morellato and Haddad 2000; Galindo-Leal and Câmara 2003; Metzger 2009; Eisenlohr et al. 2015). This re-reading is necessary, since both knowledge of the different dimensions of diversity and threats to biodiversity and ecosystem services remain current. Fortunately, in recent years, new tools have also been developed to map and manage the biome, opening up new opportunities for a future based on the conservation and sustainable use of nature.

The history of knowledge of the Atlantic Forest goes back to the end of the eighteenth century, with expeditions by European naturalists, and goes beyond the other centuries with a gradual structuring of research institutions in Brazil (Marques et al. 2000). Over time, a variation was also noticed in the ways of understanding the limits and sectors of the Atlantic Forest as well as the research topics, which include the most descriptive basic sciences (biogeography and systematics), ecology (populations and communities), and the most applied sciences (conservation and biodiversity) (Marques et al. 2021).

The Atlantic Forest has a long history of human interaction and transformation: the early human interactions in the Pleistocene, the actions of more modern native populations of farmers, the arrival of Portuguese colonizers and their large plantation systems, the urbanization and industrialization in mid-twentieth century, and the Anthropocene, from the twentieth century (Solórzano et al. 2021). Altogether, these activities dramatically affected the landscape of the Atlantic Forest, resulting in the highly threatened system. Considering the great environmental, biological, and occupational differences across regions of the Atlantic Forest, some particularities were marked. For example, in the northern Atlantic Forest, the vegetation was mainly converted into sugarcane fields, remaining only 13% of the original area located mainly within particular properties (Lins-e-Silva et al. 2021). In the central Atlantic Forest, specifically the northern portion of Espírito Santo and southern Bahia states (the Hileia Baiana), 500 years of colonization and successive economic cycles (brazilwood, sugarcane, cassava flour, timber, cocoa) significantly reduced and degraded the native vegetation. However, these forests are still significant reservoirs of the regional biota that inhabits the few remaining forests and the complex mosaic of different land uses that comprise the dominant human-modified landscapes (Faria et al. 2021). In the southern Atlantic Forest (from south Doce river), characterized by tropical dense, seasonal, and mixed forests, the degradation was marked by strong land cover changes, but still some of the largest fragments along the Serra do Mar mountain remained (Carlucci et al. 2021).

One of the most striking features of the Atlantic Forest is its megadiversity, already documented in previous studies. The updating of these data in this book shows that the diversity of some groups of organisms may be even greater than that already reported, while for other groups, a large knowledge gap still exists. For terrestrial ecosystems, the analysis of data from inventories and herbarium revealed a greater richness of tree species in the central area of the biome, but data for most species is still scarce or not yet ready for use because of bias in data collection (Zwiener et al. 2021). Similarly, the richness of epiphytes is also biased by the collection, and a greater diversity and endemism is found in the south and southeast regions of Brazil (Ramos et al. 2021). For eusocial insects (ants, bees, wasps, and termites), the compilation made by Feitosa et al. (2021) found the impressive number of 1,401 species distributed in 189 genera occurring in the Atlantic Forest; the authors point to the many gaps for specific groups and regions. Finally, the compilation of Tetrapoda occurring in the Atlantic Forest identified 2,645 species, being 719 species of amphibians, 517 species of reptiles, 1,025 species of birds, and 384 species of mammals (Figueiredo et al. 2021). This impressive fauna represents 2.8% of world's Tetrapoda species, and considering that 157 new Tetrapoda species were described in the Atlantic Forest in the last decade, mostly from poorly sampled regions or environments, the importance of the region for global biodiversity can be still higher (Figueiredo et al. 2021). For freshwater ecosystems, understood as coastal and interior freshwaters distributed in 22 hydrographic basins along the Atlantic Forest, the diversity of fishes (and other vertebrates), macroinvertebrates, zooplankton, macrophytes, and microalgae is also astonishing (Padial et al. 2021). The contrast between the demand for ecosystem services (especially drinking water and hydroelectric energy) and the susceptibility of aquatic organisms and trophic chains to environmental changes underscores the importance of increased ecological studies in these ecosystems.

Associated with this diversity in fauna and flora, a diversity of interactions between plant-pollinator, from generalist to specialist systems, represent part of the complexity of the ecosystems that compose the biome (Varassin et al. 2021).

Although it is one of the most important regions in the world in terms of biodiversity and ecosystem services, the Atlantic Forest has historically been the target of numerous threats. As almost all regions in the world, the ongoing anthropogenic climate change is becoming one of the major threats to Atlantic Forest biodiversity. Vale et al. 2021 shows that, using the predictions of eight different climate models, it is possible to project a mean temperature increase of 4.8°C to 5.6°C under a business-as-usual scenario (RCP 8.5) whereas precipitation is expected decrease in the northern and increase in southern portion of the Atlantic Forest. In fact, there a good number of studies projecting high risk of biodiversity loss in terrestrial ecosystems, but a significant knowledge gap in altitudinal, freshwater, and coastal environments (Valle et al. 2021). At a regional (biome) scale, the land-cover changes along the last 520 years led the Atlantic Forest to the current situation of low forest cover mostly distributed in small and isolated fragments composed by forests of varying ages and degradation states (Lira et al. 2021). The introduction of non-native species, that led to biotic homogenization and species losses, showed a growing pattern in spatiotemporal records and richness in the last years. This pattern was specially marked by the increased number of non-native species of plants and fishes and the increased abundance of non-native insects, that directly alter community structure and ecosystem services, and indirectly affects public health and agriculture (Vitule et al. 2021). Also, the fauna of Atlantic Forest has been strongly affected and reduced, driven mainly by habitat loss, fragmentation and degradation in terrestrial populations. In general, apex predators, other carnivores, large-bodied mammals, large herbivores (Galetti et al. 2021) and pollinators (Varassin et al. 2021) were among the most defaunated functional groups, negatively affecting key ecosystem services.

In order to reverse the effects of these threats on the biodiversity of the Atlantic Forest, it is necessary to take advantage of the windows of opportunities that are now presented. One of these opportunities is centered on the concept of ecosystem services. Pires et al. (2021), argue that Atlantic Forest provides ecosystem services for over 60% of the Brazilian population and that promoting dialogue between multiple sectors that depend on these services could leverage a sustainability agenda. Creating alternatives to finance the conservation of the Atlantic Forest is also a possibility explored by Young & Castro (2021), who list several financial instruments that could be targeted for this purpose. Actions that include encouraging sustainable productive activities, such as agroforestry systems and large-scale ecological restoration are other alternatives. According to Tubenchlak et al. (2021), there are numerous initiatives already underway for agroforestry systems in the Atlantic forest, with great potential for restoring socioecological interactions. Similarly, de Siqueira et al. (2021) presents several ways to stimulate social engagement through ecological restoration, enabling conditions to organizing people in a common view project that significantly contribute to the success of the national restoration agenda. The engagement of people is also important for ecotourism (Viveiros-de-Castro et al. 2021), for conservation actions involving governments, organized civil society and universities (Grelle et al. 2021), and for the advancement of interdisciplinary knowledge about biodiversity, as presented by Bergallo et al. (2021). Altogether, the threats, opportunities, as well as the weaknesses and strengths of the Atlantic Forest are addressed by Grelle, Rajão & Marques (2021), which discuss the possible future of the biome.

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# Part I
# Historical and Biogeographical Aspects

# Chapter 1
# The Atlantic Forest: An Introduction to the Megadiverse Forest of South America

**Marcia C. M. Marques, Weverton Trindade, Amabily Bohn, and Carlos E. V. Grelle**

**Abstract** The Atlantic Forest, the second largest forest in South America and one of the most biodiverse biomes in the world, is also one of the most threatened and important for conservation. In this chapter, we introduce the Atlantic Forest focusing on describing the evolution of knowledge, the geographical limits, and the current proposals of sectorization in ecological units. The knowledge of the Atlantic Forest can be explained by three successive phases: (1) the science of naturalists (the late eighteenth century to the late nineteenth century), where the flora and fauna were described by European travelers; (2) the rise of science in Brazil, characterized by the organization of Atlantic Forest biodiversity in collections (1890–1985); and (3) the contemporary era (1985–2020), characterized by the publication of 8226 studies focused on 4 main topics – biogeography and systematics, conservation and biodiversity, plant-animal interaction, and populations and community. The understanding of the distribution limits of the Atlantic Forest biome (11 different proposals), as well as sectorization (4 different proposals), has been the subject of several studies and legislations, which are presented and discussed. Additionally, we present terminologies usually used to designate the Atlantic Forest as a whole, as well as its sectors, to facilitate understanding in future studies. We conclude that understanding the Atlantic Forest remains a long and endless exercise, given its complexity, increased knowledge, and continuous threats.

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## 1.1 Introduction

Since Brazilian territory was discovered by Europeans, the Atlantic Forest has been the object of admiration and curiosity. This exuberant forest, with fauna and flora of impressive diversity, inhabited by human beings never before seen by the colonizers, was gradually being explored by the five senses of curious members of the Portuguese court charged with describing the New World. The same enchantment that motivated rapporteurs and artists to record the diversity of plants and animals persisted for generations of naturalists and scientific researchers, who for over 500 years have sought to describe, know, and explain the exuberance of this South American megadiverse forest.

The Atlantic Forest is the second largest tropical forest in South America. With an original total area of 1.6 million ha (according to the integrative limits of Muylaert et al. 2018; see below), it was previously distributed mainly in the Brazilian territory (93% of biome total area), but also entering the borders with Paraguay (5.3%) and Argentina (1.7%). Several palynological and phylogeographic evidence show that there were connections between the Atlantic Forest and the Amazon Forest in the Quaternary, 33,000 to 25,000 years BP (Bigarella et al. 1975; Costa 2003). During the last glaciation, a xeric and shrubby vegetation developed in central Brazil, separating the two forests that are still connected by riparian forests (Prado and Gibbs 1993). Therefore, the Atlantic Forest can be understood as a landscape of the geological dynamics of tropical forests in South America, whose identity and specificity have been defined especially in the last 20,000 years.

The Atlantic Forest is one of the most biodiverse and unique regions on Earth (Myers et al. 2000), which is a result of a complex evolutionary history. The mechanisms of flora and fauna diversification have been explained by several, non-exclusive theories along the time: (1) the Pleistocene refuges, which are isolated forest sites formed during the glaciations where organisms could have diverged and originated new lineages (Ab'Saber 1979) and, consequently, resulted in centers of endemism (Carnaval and Moritz 2008); (2) the neotectonic hypothesis, in which the uplift of mountain ranges at about 5.6 Ma, especially the Serra do Mar in the southeastern region, would have modified the climate (Simpson 1979) and created new conditions and landscapes for the local diversification; (3) the riverine barriers, especially the São Francisco, Jequitinhonha, Doce, and Paranapanema systems, that would have caused important differentiation of lineages and species (see examples in Dantas et al. 2011); and (4) the ecological gradient hypothesis, where the gradual transition from humid forests, in the core area of Atlantic Forest, toward those drier biomes surrounding the biome (e.g., Cerrado and Caatinga), would have created differential selective regimes leading to a divergence of organisms between regions (for instance, Cabanne et al. 2011). Most of the hypotheses (except no. 4, explained by parapatric speciation) are based on reductions of gene flow among populations by geographical isolation, which promoted divergence and allopatric speciation, resulting in a cumulative biological diversity (Dantas et al. 2011).

Although these hypotheses are usually used to explain diversification in large scales, the current knowledge about lineage diversification suggests that some differences across taxa do exist (Peres et al. 2020). In general, higher species richness is found on the topographically complex coasts of the states of São Paulo and Rio de Janeiro, despite other regions may have favorable conditions for differentiating niches and accumulating species. Some specific locations are important for the turnover of species, especially the Rio Doce river (Espírito Santo state), the border between São Paulo and Rio de Janeiro, and the state of Bahia, resulting in five main centers of endemism: Pernambuco, coastal Bahia, central Bahia, Serra do Mar, and Paraná/Araucaria (Peres et al. 2020). Altogether, these regions contribute to making the Atlantic Forest a megadiverse biome.

In this chapter, we introduce the Atlantic Forest focusing on describing the evolution of the knowledge about its biodiversity, its geographical limits, and the current proposals of sectorization into smaller ecological units. As opposed to establishing rigid classifications and delimitations, our goal is to show different views of a complex and unique biome that can be useful for future studies.

## 1.2 The Paths of Scientific Knowledge

The discovery and knowledge of the Atlantic Forest are intrinsically related to Brazil's colonization history. This is because it is located in the easternmost portion of the continent, the lands first occupied by European colonists. It is not uncommon bibliometric studies and systematic reviews to point out the Atlantic Forest as the most studied biome in Brazil. This status has been achieved by a long way of scientific exploration and the institutionalization of science in the country. We propose that knowledge of the biodiversity of the Atlantic Forest can be characterized by different phases, as described below: (1) the science of naturalists; (2) the rising of science in Brazil; and (3) the contemporary period.

### *1.2.1 The Science of Naturalists (Late Eighteenth Century to Late Nineteenth Century)*

Almost 300 years after the discovery of Brazil and the Portuguese's indifference for the knowledge of the indigenous peoples who lived there, the colonizers finally began to look at the Atlantic Forest with scientific interest. According to Dean (1995), this look of curiosity arises after a good part of the forests have already been cut down and the demand for new natural products by the European market instigates the crown to a search for economic diversification and efficiency. One of the first investigations was stimulated by Marquês de Pombal, who helped to train generations of naturalists, Portuguese and Brazilian, at the University of Coimbra.

The investigations carried out in the late eighteenth and early nineteenth centuries were financially supported by more visionary crown managers, which were aligned to the scientific advances in France and England. Two important collections and research centers were installed in Rio de Janeiro – the Botanical Garden in 1808 and the Royal Museum (later the National Museum) in 1818 – which were then responsible to search for useful native species and to promote agricultural diversification. With the arrival of the royal family to Brazil, mainly by the encouragement of the Archduchess D. Leopoldina, a group of brilliant European naturalists were invited to Brazil. These included Karl Friedrich Philipp von Martius, Johann Emanuel Pohl, Johann Baptist von Spix, and August Glaziou, the most important naturalists to describe Brazilian flora and fauna. The transfer of the royal family to the colony also favored the arrival of innumerable young naturalists, who were sent by museums, botanical gardens, and European scientific societies to collect in Brazil. It is worth mentioning the French botanist Auguste de Saint-Hilaire, who in 1816 landed in Brazil and, for 6 years, collected 15,000 species of plants and animals that would result in the publication of 14 volumes when he returned to France (Dean 1995). Charles Darwin, while passing with H.M.S. Beagle through Rio de Janeiro in 1832, would have recorded in his logbook the admiration for the grandeur of the Atlantic Forest (Dean 1995). Despite being one of the most exciting periods of discovery about the flora and fauna of the Atlantic Forest, as well as about the first times of Brazil's formation, most of the specimens and data collected by these naturalists were taken to European institutions, and just a few specimens were left to the still-scarce Brazilian infrastructure of museums and researchers.

Accompanying the activities of collecting and describing nature carried out by naturalists, several artistic missions from Europe promoted the encounter between art (the sensitivity) and science (the reason) of the Atlantic Forest. The Atlantic Forest was then portrayed and known through the brushes of artists such as Jean-Baptiste Debret, Nicolas-Antoine Taunay, and Johann Moritz Rugendas, among others (Belluzzo 1996).

### *1.2.2 The Rising of Science in Brazil: The Organization of Atlantic Forest Biodiversity (1890–1985)*

After the emergence of the Brazilian scientific institutions, in the nineteenth century during the reign of D. Pedro II, the sciences started to figure among the government policies, with the state being its main supporter. With the establishment of the Republic (1889), the provinces had the opportunity to develop their science independently of federal power, which resulted in the proliferation of natural history museums, such as the Museum of Zoology of the University of São Paulo (launched in 1895) and, later, the São Paulo Botanical Gardens (1928), the Mello Leitão Biology Museum (1949, Espírito Santo), and the Zoobotanic Foundation (1972, Rio Grande do Sul). The Brazilian Research Council (CNPq) was created in 1951, and the organization of scientists in societies occurred in the 1940s and 1970s, with the creation of the Brazilian Society for the Advancement of Science (1948), the Botanical Society of Brazil (1950), and the Brazilian Society of Zoology (1978). Some of the main Brazilian teaching and research centers were established in the Atlantic Forest region, such as the University of São Paulo (1934), the Federal University of Rio de Janeiro (1920), the Federal University of Pernambuco (1946), the Federal University of Minas Gerais (1927), the Federal University of Paraná (1912), and the Pontifical Catholic Universities of Rio de Janeiro (1940) and São Paulo (1946). These universities created their own collections of fauna and flora or incorporated pre-existing museums, in addition to being the training site for the first generations of scientists in postgraduate courses on natural sciences. In the 1970s and 1980s, other important universities and research centers were consolidated, such as the State University of São Paulo (1976), the State University of Campinas (1962), the Cocoa Research Centre (1957), and the Brazilian Agricultural Research Corporation (1972). The latter two were responsible for the development of researches related to the Brazilian agricultural production and, thus, with direct impacts on the economic development and knowledge of the areas within the Atlantic Forest.

### *1.2.3 The Contemporary Era (1985–2019): A Bibliometric Analysis*

With the facility of current search tools, contemporary Atlantic Forest research can be described through bibliometric research. We performed a bibliometric analysis to describe the evolution of the knowledge about the Atlantic Forest by searching documents on the Web of Science using the topics “Atlantic Forest” and “Mata Atlântica,” from 1985 to December 2019 (see Supplementary Material). A total of 8288 documents have been published in the period; the first article “The Vanishing Brazilian Atlantic Forest” was published in the journal *Biological Conservation* in 1985 (note that obviously other studies were published before this year, but they are not cataloged in Web of Science, a contemporary tool). In this article, Da Fonseca (1985) examined the Atlantic Forest deforestation in the state of Minas Gerais and claimed to urgent efforts to protect the remaining forest remnants, which were few, small, isolated, and unprotected. Until the late 1980s, other three articles were published, but a clear interest on the Atlantic Forest emerged only in the 1990s, especially in 1993, when more than ten articles were published in a single year (Fig. 1.1).

#### **The Rise and Establishment of Conservation Science (1985–2015)**

Analyzing the keywords used by authors from 1992 to 2019, Atlantic Forest research can be divided into four clusters (Fig. 1.2): conservation and biodiversity (green), biogeography and systematics (red), plant-animal ecology (blue), and ecology of population and communities (yellow).

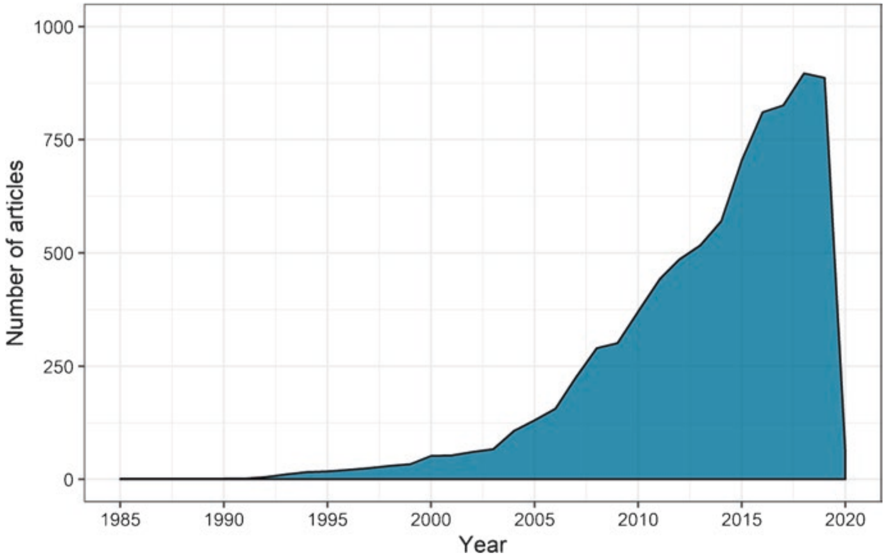

**Fig. 1.1** Number of articles published from 1985 to December 2019 on the topic "Atlantic Forest" (*n* = 8288). Data downloaded from Web of Science and analyzed using Bibliometrix R package (Aria and Cuccurullo 2017)

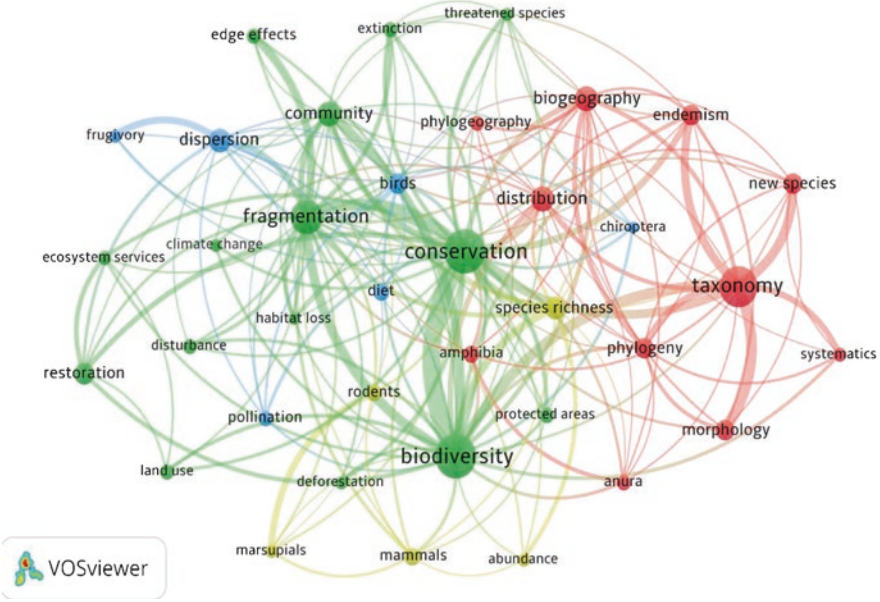

**Fig. 1.2** Bibliometric network based on keywords mentioned in articles on the Atlantic Forest. Size of keyword labels and circles are determined by number of occurrences. Keywords are connected by number of co-occurrence. Network map created in VOSviewer 1.6.14

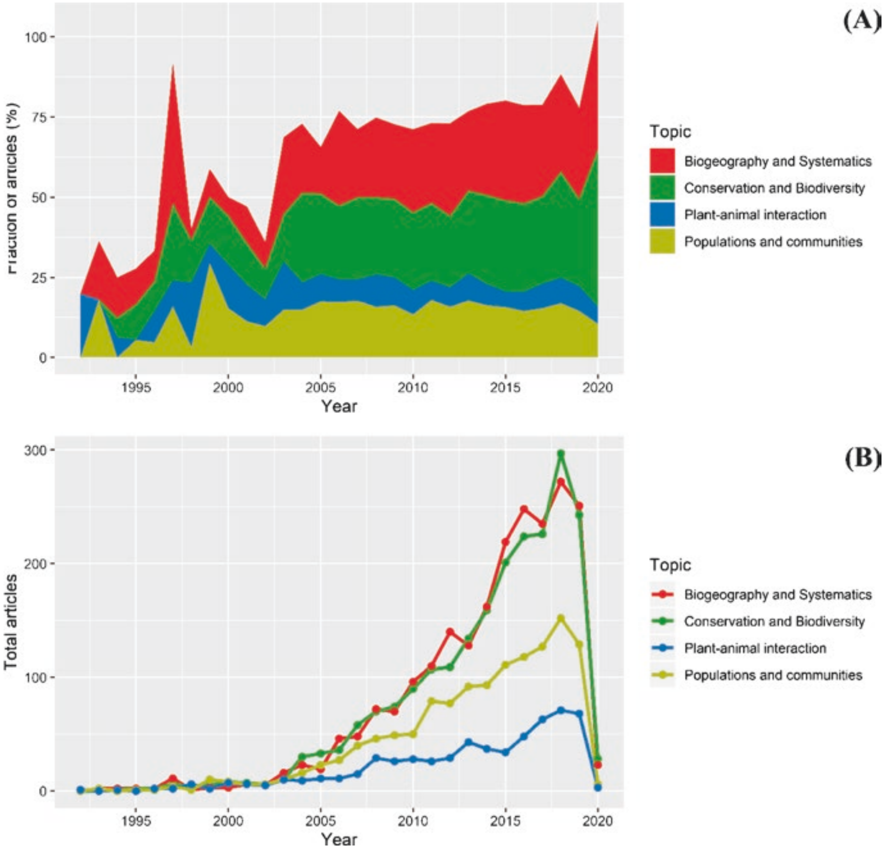

**Fig. 1.3** Number (**a**) and proportion of total articles (**b**), by topic, according to words from title and keywords. Articles were selected and clustered according to keywords and clusters from bibliometric network map (Fig. 1.2)

In the early 1990s, there was a predominance of descriptive studies on plant-animal interaction and ecology of population and communities comprising mammals (rodents, primates, bats, and marsupials) and birds (Fig. 1.3a, b). Since then, especially after Rio 92, concerns in the environment and biodiversity conservation have grown in Brazil. The main factors that contributed to this growth were (1) the promulgation of the Brazilian Constitution in 1988, which placed the environment as a citizen's right and state responsibility, and (2) the rupture of the 30-year military dictatorship regime in 1989, where the number of species and ecosystem losses was still unknown.

In the early 2000s, the information about the Atlantic Forest, still fragmented, begins to be compiled in special numbers of scientific journals and books. In the early 2000, Morellato and Haddad (2000) edited a special issue of the journal *Biotropica*, with 14 articles that dealt mainly on vertebrate distribution, plant-animal interactions, and ecology and plant distribution. It is worth mentioning the work of

Oliveira-Filho and Fontes (2000), who proposed a comprehensive definition of the Atlantic Forest, including humid and semideciduous forests as a great continuum. Also, in this year, the Atlantic Forest has been confirmed as a hotspot for global conservation actions (Myers et al. 2000). In 2003 (and in 2004, the Portuguese version), the Conservation International published a book entirely dedicated to the Atlantic Forest as part of a series dedicated to the world's conservation hotspots (Galindo-Leão and Câmara 2003). In its 31 chapters, the book reported the history of the threats in the different territories and showed the conservation status, with examples focused mainly on vertebrates in the 3 countries where the Atlantic Forest occurs (Brazil, Paraguay, and Argentina).

In parallel, in the early 2000s, the number of articles on biogeography and systematics also increased considerably (Fig. 1.3a, b). This exponential growth is likely due to the beginning of the "big data era" on biodiversity, which has as starting point the establishment of the Global Biodiversity Information Facility (GBIF), in 2001. The massive availability of biodiversity data, together with the rapid emergence of new techniques and tools to analyze such information, provides an invaluable resource to document biodiversity and its distribution through time and space (Maldonado et al. 2015; Smith and Blagoderov 2012). Thanks to that, both conservation and biogeography became consistent research fields in Atlantic Forest, which explains, in part, them being the main research topics currently studied (Fig. 1.3b). Conservation has been the most used keyword (765 occurrences), and it is linked to all 38 keywords on the map (Fig. 1.2). This finding suggests that conservation is the main focus of the research conducted on the Atlantic Forest.

In 2009, a special issue of the *Biological Conservation* journal was also dedicated to the Atlantic Forest (Metzger and Sodhi 2009). In this issue, the 11 articles suggest the high degree of fragmentation of Atlantic Forest and its effects on different groups of organisms and regions of Brazil. In the estimates of that time, only 11% of the biome original area remained (Ribeiro et al. 2009), which motivated several later studies. In 2015, a special issue of the *Biodiversity and Conservation* journal explored, in 14 articles, the flora and vegetation of the Atlantic Forest (Eisenlohr et al. 2015). One of the most impressive results shows that in 70 years of studies in the Atlantic Forest, only 0.01% of the total Atlantic Forest was actually surveyed. This demonstrates how limited our knowledge is on the Atlantic Forest (Lima et al. 2015).

This period definitively marked the fruitful "marriage" between science and the environmental movement in Brazil, which brought quality and greater argumentative capacity to the implementation of legal regulation of several Brazilian environmental laws, including the Atlantic Forest Law (Law 11428/2006; see below).

## Biodiversity Collections and Big Data

The accumulation of scientific work over time has generated a huge collection of data. Recently, some data repository initiatives have been published for the biodiversity of organisms, such as bats (Muylaert et al. 2017), small mammals (Figueiredo et al. 2017), amphibians (Vancine et al. 2018), butterflies (dos Santos et al. 2018), mammals (Souza et al. 2019), primates (Culot et al. 2019), epiphytic plants (Ramos et al. 2019), and trees (Oliveira-Filho 2017; Bergamin et al. 2015; Zwiener et al. 2017, 2019). Databases of attributes of tree plants (Rodrigues et al. 2018) and wood densities (Oliveira et al. 2019), mammals (Gonçalves et al. 2018), and birds (Rodrigues et al. 2019) also proliferated. In this period, all topics considered here reached their peak of publications (Fig. 1.3b), especially conservation and biogeography. Altogether, these data collections have been important for the increase of studies using geospatial tools, allowing a broader understanding of diversity patterns and influencing public policies.

## 1.3 The Limits of the Atlantic Forest

Defining a geographical space with ecological, biological, and environmental significance is one of the most complex tasks for a nature scientist. While there is a lot of controversy regarding the use of terms (see Box 1.1), it is known that the physiognomy of vegetation is one of the most important characteristics to define a biome. The understanding that the Atlantic Forest is a distinct ecological unit within Brazilian territory dates back to the middle of the nineteenth century with the work of von Martius, who contributed greatly to the knowledge of Brazilian flora and vegetation. In his seminal proposal for the floristic division of Brazil, in 1858, Martius named Dryades, the Greek deity, to refer to the forests of the Atlantic coast, located in the southeast and part of the northeast of Brazil (IBGE 2012).

Several other proposals for phytogeographic divisions were suggested (Fig. 1.4). Hueck (1972) identified the Brazil coastal vegetation as an extension of Amazon rainforest (Fig. 1.4b). Ab'Saber (1977) identified "landscape units" based on relief, drainage, climate, soils, and vegetation patterns and called them "morphoclimatic domains." According to this proposal, the Atlantic Forest is considered the hilly areas of *Mares de Morro* (Seas of Hills), with origin in the Tertiary (Fig. 1.4c). Rizzini's proposal of 1966 and 1979 deserves to be highlighted. For him, the Atlantic Forest is closely related to the *Restingas* and the *Pinheiral* (Araucaria Forest) complexes (Fig. 1.4d). The Atlantic Forest, considered by Myers et al. (2000) as a biodiversity hotspot, had an extension similar to the morphoclimatic domain designated by Ab'Saber (Fig. 1.4e). Currently, the Atlantic Forest is considered by the Critical Ecosystem Partnership Fund as a biodiversity hotspot (Fig. 1.4f) and has a similar dimension to "WWF ecoregions" proposed by Olson et al. (2001) (Fig. 1.4g). Silva and Casteleti (2003) also considered a broader extension of Atlantic Forest using areas of endemism of birds, butterflies, and primates (Fig. 1.4h).

From the 1960s onward, a national project (Radambrasil project) led by phytogeographers and geographers proposed an official classification for Brazilian vegetation. A first version was published in 1991 launched by IBGE and updated in 2004 (IBGE 2004; Fig. 1.4i). In the IBGE's classification, the Atlantic Forest does not constitute an isolated unit. Instead, it is composed of a complex of various vegeta-

**Box 1.1 Terms Usually Applied to Refer to the Entire or Parts of the Atlantic Forest and Used in This Book**

Atlantic Forest Complex: The mosaic of vegetation types or vegetation physiognomies present in the Atlantic Forest as a whole. The Atlantic Forest complex would include both vegetation formations that are typically forestry and the shrub and grassland, the aquatic ecosystems, and the ecotonal areas in all their extension.

Atlantic Forest Domain: From a phytogeographic point of view, it refers to a specific area characterized by the presence of endemic species (IBGE 2012). The domain of the Atlantic Forest refers to a region characterized by its specific flora.

Biome: This is, certainly, one of the most controversial terms in ecology. Walter (1986) used this term to mention 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 the macroclimate, phytophysiognomy (formation), and also the ground and altitude (main elements that characterize the different continental environments). According to the Brazilian Vegetation Classification, a biome can be defined as a set of life (plant and animal) constituted by the grouping of contiguous and identifiable vegetation types on a regional scale, with similar geoclimatic conditions and a shared history of changes, which results in its own biological diversity (IBGE 2012). In this book, the term was used by authors to refer to one of these definitions.

Ecoregion: Relatively large units of land or water containing distinct assemblages of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land-use change (Olson et al. 2001). The ecoregions have been used by the World Wildlife Fund (WWF) as a tool to establish important areas for conservation.

Ecosystem: The complex of living organisms, their physical environment, and all their interrelationships in a particular unit of space.

Endemism Centers: The Atlantic Forest's areas of endemism served as the basis for a conservation proposal that accounted three biodiversity centers (Conservation International Brazil et al. 2000; Silva and Casteleti 2003; Da Fonseca et al. 2004; Ayres et al. 2005; Tabarelli et al. 2005, 2010).

Phytophysiognomy: The aspect of vegetation, defined by its structure, based on characteristics such as the predominant life form (or habitus), plant architecture, density of vegetation, seasonality, and deciduousness.

Vegetation Formation: Term used to designate a defined vegetation type, a set of forms of higher-order plant life, which makes up a homogeneous physiognomy, despite its complex structure (IBGE 2012). The sub-formation is used as a subdivision of formation – or part of it – and is differentiating itself by presenting specific facies that alter the physiognomy of formation (IBGE 2012).

Vegetation Types: The same as vegetation formation.

tion types, from forest to non-forest physiognomies, more or less close to the Atlantic coast, and differentiated in terms of terrain, age, location, climate, and flora origin (tropical or temperate) (IBGE 2012).

During the 1990s, environmentalists and scientists organized conservation actions that culminated in the recognition of the Atlantic Forest as a member of the World Biosphere Reserve Network, in 1992 – a title granted by UNESCO. This created a demand for internal legal instruments in Brazil, to guarantee the maintenance of this title by UNESCO. In 1993, a federal decree (Decree 750) defined the IBGE map as the official limit of Atlantic Forest. The following 13 years were marked by the tentative of transform the protection of the Atlantic Forest in law. On December 22, 2006, it was finally approved the project proposed by Deputy Fabio Feldman, becoming the first law specifically created for the Atlantic Forest biome (Law 11.428). It was then promulgated by the Minister of Environment Marina Silva and President Luiz Inácio Lula da Silva. This law had strong impacts on the understanding of what the Atlantic Forest is, as well as on disciplining the use of natural resources within the limits of the biome. Subsequently, other legal instruments complemented details of the law (e.g., CONAMA Resolution 388/2007). Although the effectiveness of this law has sometimes been questioned (Varjabedian 2010), the Atlantic Forest remains the only Brazilian biome with a specific law for its protection.

In Article 2 of the Atlantic Forest Law, the limits of the biome are defined. In this definition, the Atlantic Forest includes several of its local variants, such as the Atlantic Dense Forest, Mixed Forest (Araucaria Forest), Open Forest, Seasonal Semideciduous Forest, and Seasonal Deciduous Forest. These are all part of the Atlantic Forest biome, as well as the associated ecosystems, named as mangroves, *restingas* (scrubs), altitude grasslands, and the inland swamps and mountain forest (Brejos de altitude) in the northeast region. With this more comprehensive delimitation, the Atlantic Forest is distributed in 17 Brazilian states, with a total of 3401 municipalities and housing about 70% of Brazilian population.

After the enactment of the law, a decree (Decree 6.660/2008) established the map of the law enforcement area (Fig. 1.4j), as well as the regulating devices for determining the use and protection of the Atlantic Forest. The application of the law would apply to remnants of native vegetation in the primary stage and in the initial, medium, and advanced secondary stages of regeneration.

Given the differences between the limits established by different studies and maps, Muylaert et al. (2018) compared the similarities and differences of four previously proposed and widely used limits (Atlantic Forest Law, WWF limits, Silva and Casteleti 2003, and MMA-IBGE) and proposed two new limits: the "consensual limit" of the Atlantic Forest (Fig. 1.4k), which consists of the intersection of four other limits, and the "integrating limit" (Fig. 1.4l), which refers to the union of previously defined limits. Together, all these proposals for the limits of the Atlantic Forest (Fig. 1.4) are still used, and some of them are mentioned in the chapters of this book.

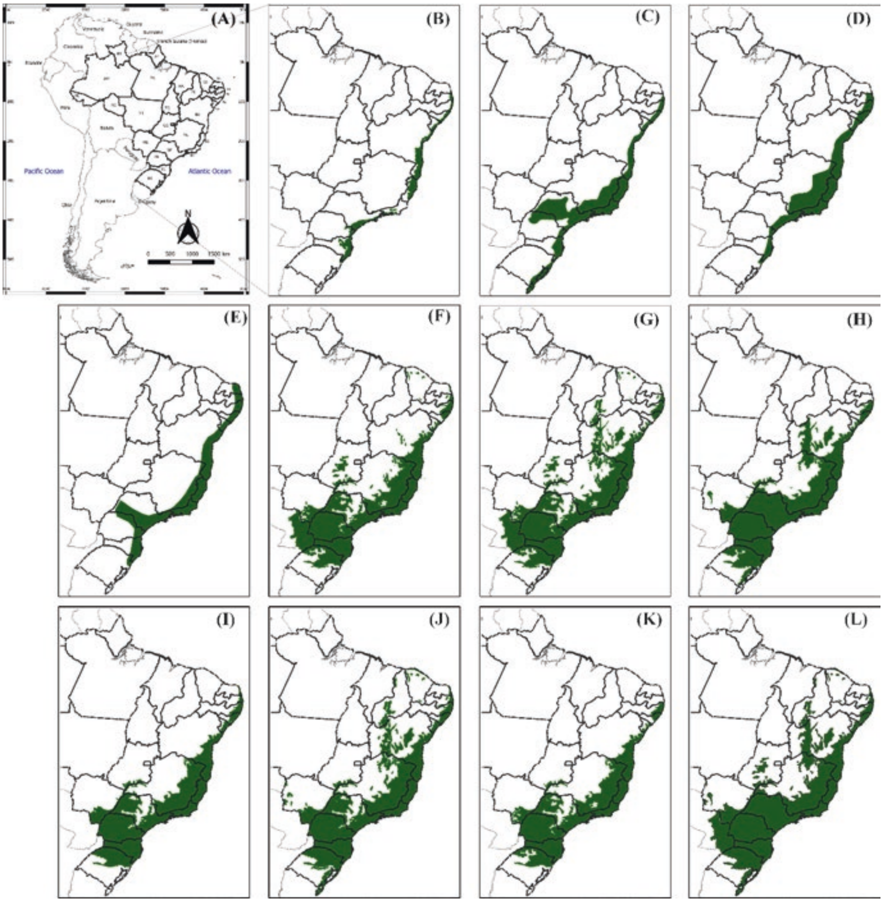

**Fig. 1.4** Different proposals for Atlantic Forest limits: (**a**) South American map overview and Brazilian states limits; (**b**) Coastal Tropical forests identified by Hueck (1972), shapefile digitalized by Hasenack et al. (2017); (**c**) *Mares de Morro* proposed by Ab'Saber (1977), shapefile digitalized by IBGE (2019); (**d**) Atlantic Forest proposed by Rizzini (1979); (**e**) Atlantic Forest identified as a biodiversity hotspot by Myers et al. (2000) and (**f**) updated by Hoffman et al. (2016); (**g**) Atlantic Forests proposed by Olson et al. (2001), WWF ecoregions; (**h**) Atlantic Forest considered by Ribeiro et al. (2009), similar to the one proposed by Silva and Casteleti (2003); (**i**) Atlantic Forest considered by IBGE (2004); (**j**) Atlantic Forest Law application map, digitalized by IBGE (2008); (**k**) consensual and (**l**) integrative limits proposed by Muylaert et al. (2018)

## 1.4 The Atlantic Forest Sectorization

Along its length, which comprises 31° in latitude and 22.9° in longitude (according to the limits of Atlantic Forest Law), the Atlantic Forest presents a great diversity of physiognomies and ecosystems. The altitudes vary from the sea level to 2891 m in elevation, in Serra do Caparaó, between Minas Gerais and Espírito Santo states. The relief includes mountains, plateaus, plains, boards, and depressions (IBGE 2008). The climate, according to the Köppen-Geiger climate classification (Peel et al. 2007), includes the types Af, Am, Aw, BSh, Cfa, Cfb, Cwa, and Cwb. The soils are extremely varied and include types of eutrophic and dystrophic soils, on flat terrain and elevations, floodplains, *restingas*, and mangroves, frosted and thiomorphic soils, humic soils, and rock outcrops (Resende et al. 2002). In addition, 8 of the 12 Brazilian hydrographic regions are located in the Atlantic Forest: South Atlantic, Uruguay, Paraná, Southeast Atlantic, East Atlantic, São Francisco, East Northeast Atlantic, and Parnaíba (CNRH 2003, but see Padial et al. 2021).

With all this variation in space and environment throughout the Atlantic Forest, there is a strong structuring of biodiversity, which is known for various groups of organisms (e.g., Oliveira-Filho and Fontes 2000; Marcilio-Silva et al. 2017; and Zwiener et al. 2019 for plants; Da Silva et al. 2004; Carnaval et al. 2014; Loyola et al. 2014; and De La Sancha et al. 2020, for animals). Based on the combination of different drivers on the biota, and using different criteria and methods, several proposals for sectorization of the Atlantic Forest have historically followed (Fig. 1.5).

Based on global and regional distribution of plants and animals, Olson et al. (2001) subdivided the terrestrial ecosystems into 14 biomes, 8 biogeographic realms, and 867 ecoregions. In the Atlantic Forest, 11 ecoregions were recognized (Fig. 1.5a), which are based mainly on the Brazilian vegetation map from IBGE (1993). Silva and Casteleti (2003) used data on bird, butterfly, and primate distributions to propose the division of Atlantic Forest into eight biogeographical sub-regions (Fig. 1.5b), five as centers of endemism (Bahia, Brejos Nordestinos, Pernambuco, Diamantina, and Serra do Mar) and three as transition zones (São Francisco, Araucaria Forests, and Interior Forests). Ribeiro et al. (2011) refined the biogeographical sub-regions proposed by Silva and Casteleti (2003), using climatic and elevation data, identifying 55 internal divisions. The map of application of the Atlantic Forest Law (IBGE 2008) divided Atlantic Forest into ten vegetation types (Fig. 1.5c). These vegetation types were based on Brazilian vegetation map from IBGE (2004), which divided the vegetation of Brazil according to the dominant plant life forms. More recently, Cantidio and Souza (2019) identified 21 spatially cohesive occurrence ecoregions (Fig. 1.5d) using a data set on the distribution of 4378 shrub and tree species across 711 localities.

In the proposition of sectorizations above (Fig. 1.5), the authors apply terms to identify each sector based on some broader international system (e.g., IBGE 2008; Olson et al. 2001), adopting some regional terminologies (e.g., Silva and Casteleti 2003), or still do not propose any specific term (Cantidio and Souza 2019). In addition, at the infra-sector scale, there are a multitude of terms used to characterize types of vegetation or ecosystems in the Atlantic Forest. Given its wide uses and constant references, including in the chapters of this book, the main terms and their correspondents are presented in Table 1.1.

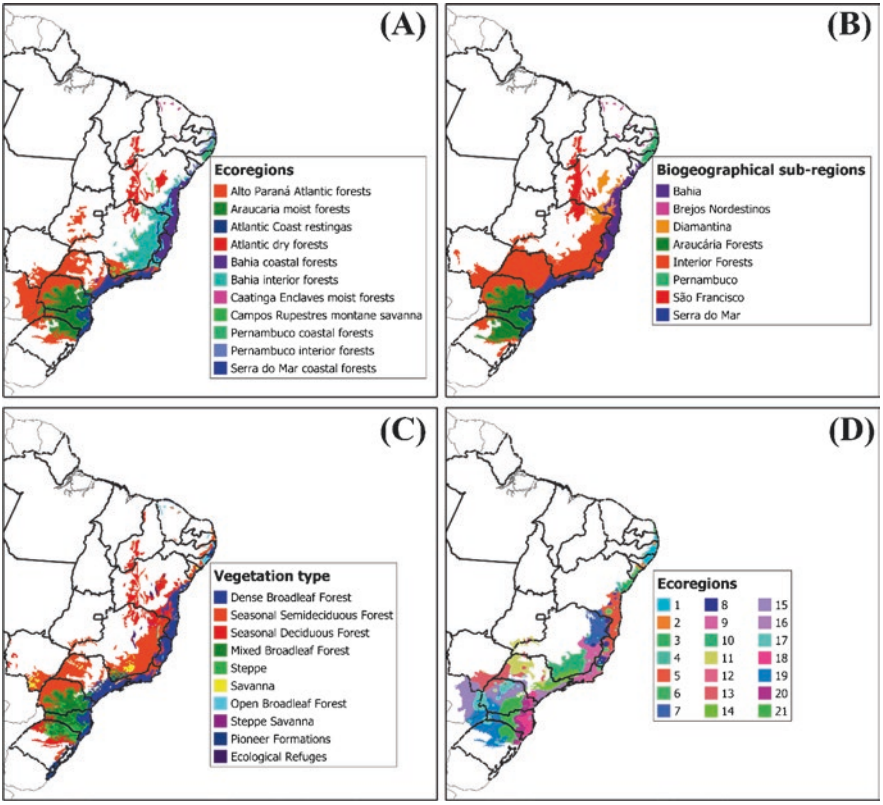

**Fig. 1.5** Different proposals of Atlantic Forest sectorization: (**a**) Atlantic Forest ecoregions defined by Olson et al. (2001), “WWF ecoregions,” (**b**) biogeographical sub-regions proposed by Silva and Casteleti (2003), (**c**) Atlantic Forest vegetation types defined by IBGE (2008) (see correspondent Portuguese names in Table 1.1), (**d**) ecoregions based on woody plant occurrence proposed by Cantidio and Souza (2019)

The limits between the sectors of the Atlantic Forest are defined by specific geographic and geological conditions, which change environmental conditions and biota either gradually or abruptly. Examples of these variations can be seen in the representative vegetation profiles of three regions of the Atlantic Forest (Fig. 1.6): the Northeastern Atlantic Forest, the Central Corridor, and the Southern Atlantic Forest. These regions, which were treated in detail in Chaps. 3 (Lins-e-Silva et al. 2021), 4 (Faria et al. 2021), and 5 (Carlucci et al. 2021), illustrate the diversity of physiognomies and landscapes of the Brazilian Atlantic Forest.

**Table 1.1** Terms and its correspondents used to design different vegetation types in Atlantic Forest (AF) domain and applied in present book

| English terms and correspondents | Portuguese terms and correspondents | General characteristics |
|---|---|---|
| Atlantic Forest; Atlantic Forest *latu sensu* | Floresta Atlântica, Floresta Atlântica *latu sensu*, Mata Atlântica | Used to refer to the vegetation or biome more generically |
| Atlantic Dense Forest, Atlantic Rain Forest, Coastal Dense Forest, Atlantic Pluvial Forest | Floresta Ombrófila Densa Atlântica[a], Floresta Pluvial Atlântica | Wet (or rainy) forest, characterized by precipitation well distributed throughout the year and dense canopy |
| Atlantic Open Forest | Floresta Ombrófila Aberta[a] | Moist to wet forest, characterized by precipitation distributed throughout the year and opened canopy |
| Semidecidual Seasonal Forest, Interior Semidecidual Seasonal Forest | Floresta Estacional Semidecidual[a], Floresta Tropical Subcaducifólia, Floresta Estacional Mesófila Semidecídua, Floresta Latifoliada Tropical | Forest characterized by seasonality (alternation of rainy and dry periods), and deciduous trees (20–50% of individuals) lose their leaves in winter |
| Tabuleiro Forest | Floresta de Tabuleiro, Tabuleiros Costeiros, Floresta Estacional Semidecidual[a] | Forests occurring in lowland area or coastal board, of origin Tertiary, with its species distributed along a climatic gradient in states of Rio de Janeiro to Pernambuco |
| Seasonal Decidual Forest, Seasonal Dry Forest | Floresta Estacional Decidual[a]; Floresta Decídua; Floresta Seca | Forest characterized by strong seasonality (alternation of rainy and long dry periods), and deciduous trees (>50% of individuals) lose their leaves in winter |
| Araucaria Forest, Araucaria Mixed Forest, Mixed Forest | Floresta Ombrófila Mista[a]; Floresta com Araucária, Mata de Araucária, Pinheiral | Wet forest, occurring in cold climates of southern Brazil, dominated by ancient genera from temperate regions (*Drimys*, *Araucaria, Podocarpus*) |
| *Restinga*, coastal scrub, coastal thicket, coastal plain forest, dunes | Vegetação Pioneira com Influência Marinha[a]; Restinga; vegetação (ou floresta) de restinga, vegetação de dunas | Herbaceous, shrub, or tree vegetation that grows on coastal sandy plains formed during the Holocene. All are also referred to as Atlantic Forest-associated ecosystems |
| Salt marsh | Vegetação Pioneira com influência Flúvio-Marinha[a]; marisma, apicum | Herbaceous vegetation flooded by seawater and freshwater. Also referred to as Atlantic Forest-associated ecosystems |
| Mangrove | Vegetação Pioneira com Influência Flúvio-Marinha[a], manguezal | Young tropical vegetation occurring in lowlands in estuary regions and in marginal areas of river deltas, lagoons, bays, and river mouths, affected by tides and not exposed to sea waves. Also referred to as Atlantic Forest-associated ecosystems |

(continued)

**Table 1.1** (continued)

| English terms and correspondents | Portuguese terms and correspondents | General characteristics |
|---|---|---|
| Grasslands; altitude grasslands | Estepes[a], Savanas-Estépicas[a]; Campos; Campos de Altitude | Relatively extensive plateaus, composed of mosaics of grass clumps, sparse herbs, shrubs, and small trees, limited by low temperatures, in mountain ridges over 900 m (e.g., in Serra da Bocaina, Serra da Mantiqueira, Serra do Caparaó) |
| Rupestrian fields, *campo rupestre* | Estepes[a], Savanas-Estépicas[a]; campo rupestre | Graminoid and diverse vegetation, limited by the depth of the substrate, in mountain ridges over 900 m (e.g., in Cadeia do Espinhaço, Chapada Diamantina) |
| *Brejos Nordestinos,* altitude swamps | Brejos nordestinos, brejos de altitude, florestas serranas | Enclaves of dense forests in the semi-arid northeast (in the Caatinga domain), in regions of high altitude and humidity (due to the exposure of wet masses from the coast), forming true vegetation islands |

[a] Term used in the official Brazilian Vegetation Classification (IBGE 2012)

## 1.5 Conclusion

In this chapter, we introduced the Atlantic Forest in terms of the evolutionary history, evolution of scientific knowledge, and subdivision along its distribution. We show that the current identity of the Atlantic Forest biome is the result of (1) an intrinsic interest of biologists who saw this as an ideal case to understand how evolution processes; (2) a continuous exercise in understanding the drivers that determine their unique identity, as well as the variations between their sectors; and (3) the notion that this is a place of intrinsic biological, ecological, and cultural values, of relevance to humanity. This identity is, therefore, complex, which challenges us to immerse ourselves both in deepening the knowledge of its parts (a physicalist view of science, sensu Mayr 2008) and in the interpretation of the functioning of the whole (an organicist and mechanist scientific view). The chapters in this book will enable a trip to these two worlds.

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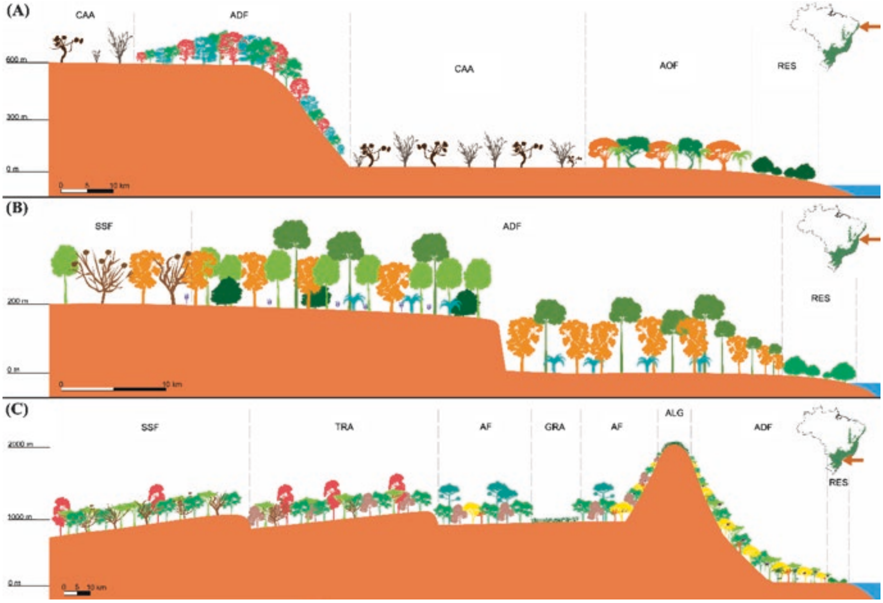

**Fig. 1.6** Profiles of three Atlantic Forest phytophysiognomies in three different regions in Brazil: (**a**) *the Northeastern Atlantic Forest* in Pernambuco state – the *Restinga* is a narrow band followed by Atlantic Open Forest; after a strip of *Caatinga* vegetation, the Dense Forest occurs in the slopes of Serra da Borborema, characterizing the *Brejos Nordestinos* (based on Pôrto et al. 2004 and Lima 2007). (**b**) *The Central Corridor* – Hileia Baiana: in South Bahia, the *Restinga* is followed by Atlantic Dense Forest in the coastal plain; in the interior, the Barreiras Formation establishes a climatic and edaphic transition where the Seasonal Semideciduous Forest (Tabuleiro Forest) occurs (based on Jardim 2003). (**c**) *The Southern Atlantic Forest*: in Paraná state, the coastal plain is narrow, and the *Restinga* is followed by Atlantic Dense Forest in the lowlands and in the Serra do Mar mountains; at higher elevations, outcrops and altitude grasslands occur. In the interior, the first plateau is covered by Araucaria Forest and grasslands, the second by a transitional vegetation, and the third plateau by Seasonal Semideciduous Forest (based on Roderjan 1994, 2001). AF Araucaria Forest, ADF Atlantic Dense Forest, AOF Atlantic Open Forest, ALG altitude grasslands, CAA *Caatinga*, GRA grasslands, SSF Seasonal Semideciduous Forest, RES *Restinga*, TRA transitional vegetation between SSF and AF. The left/vertical scale refers to the altitude; the horizontal scale refers to the ocean distance. All scales are estimates

## References

Ab'Saber NA (1977) Os domínios morfoclimáticos na América do Sul. Geomorfologia 52:1–22

Aria M, Cuccurullo C (2017) Bibliometrix: an R-tool for comprehensive science mapping analysis. J Informet 11(4):959–975

Ayres JM, Da Fonseca GAB, Rylands AB et al (2005) Os corredores ecológicos das florestas tropicais do Brasil. Sociedade Civil Mamirauá, Belém

Belluzzo AM (1996) A propósito d'O Brasil dos viajantes. Revista USP 30:8–19

Bergamin RS, Duarte LDS, Marcilio-Silva V et al (2015) Compilation of woody species occurring in the Brazilian Atlantic Forest complex. Front Biogeogr 7(2):69–72

Bigarella JJ, Andrade-Lima D, Riehs PJ (1975) Considerações a respeito das mudanças paleoambientais na distribuição de algumas espécies vegetais e animais no Brasil. An Acad Bras Cienc 47:411–464

Brown KS Jr, Ab'Saber AN (1979) Ice-age forest refuges and evolution in the neotropics: correlation of paleoclimatological, geomorphological and pedological data with modern biological endemism. Paléo 5:1–30

Cabanne GS, D'Horta FM, Meyer D et al (2011) Evolution of Dendrocolaptes platyrostris (Aves: Furnariidae) between the South American open vegetation corridor and the Atlantic forest. Biol J Linn Soc 103(4):801–820

Cantidio LS, Souza AF (2019) Aridity, soil and biome stability influence plant ecoregions in the Atlantic Forest, a biodiversity hotspot in South America. Ecography 42(11):1887–1898

Carlucci MB, Marcílio-Silva V, Torezan JM (2021) The Southern Atlantic Forest: Use, Degradation, and Perspectives for Conservation. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Carnaval AC, Moritz C (2008) Historical climate modeling predicts patterns of current biodiversity in the Brazilian Atlantic forest. J Biogeogr 35(7):1187–1201

Carnaval AC, Waltari E, Rodrigues MT et al (2014) Prediction of phylogeographic endemism in an environmentally complex biome. Proc R Soc B 281:20141461

CNRH (2003) Divisão hidrográfica nacional. Conselho Nacional de Recursos Hídricos, Brasília

Conservation International do Brasil, Fundação SOS Mata Atlântica, Fundação Biodiversitas et al (2000) Avaliação e ações prioritárias para a conservação da biodiversidade da Mata Atlântica e Campos Sulinos. Secretaria de Biodiversidade e Florestas. Ministério do Meio Ambiente, Brasília

Costa LP (2003) The historical bridge between the Amazon and the Atlantic Forest of Brazil: a study of molecular phylogeography with small mammals. J Biogeogr 30(1):71–86

Culot L, Pereira LA, Agostini I et al (2019) Atlantic-primates: a dataset of communities and occurrences of primates in the Atlantic forests of South America. Ecology 100(1):e02525. https://doi.org/10.1002/ecy.2525

Da Fonseca, GAB (1985) The vanishing Brazilian Atlantic forest. Biological Conservation 34(1): 17–34.

Da Fonseca GAB, Alger K, Pinto LP, Araújo M, Cavalcanti R (2004) Corredores de biodiversidade: o Corredor Central da Mata Atlântica. In: Arruda MB, Sá LFSN (org.) Corredores ecológicos: uma abordagem integradora de ecossistemas no Brasil. Ibama, Brasília, pp 47–65

Da Silva JMC, De Souza MC, Castelletti CHM (2004) Areas of endemism for passerine birds in the Atlantic forest, South America. Glob Ecol Biogeogr 13(1):85–92

Dantas GPM, Cabanne GS, Santos FR (2011) How past vicariant events can explain the Atlantic Forest biodiversity? In: Grillo O, Venora G (eds) Ecosystems Biodiversity. IntechOpen, pp 429–442

De La Sancha NU, Maestri R, Bovendorp RS, Higgis CL (2020) Disentangling drivers of small mammal diversity in a highly fragmented forest system. Biotropica 52:182–195

De Lima RAF, Mori DP, Pitta G et al (2015) How much do we know about the endangered Atlantic Forest? Reviewing nearly 70 years of information on tree community surveys. Biodiversity and Conservation 24:2135–2148

Dean WB (1995) With broadax and firebrand: the destruction of the Brazilian Atlantic Forest. University of California Press, California

Eisenlohr PV, Oliveira-Filho AT, Prado J et al (eds) (2015) Special issue: Brazilian Atlantic Forests. Biodivers Conserv 24(9):2129–2331

Faria D, Delabie JHC, Dias MH (2021) The Hileia Baiana – an assessment of natural and historical aspects of the land use and degradation of the central corridor of the Brazilian Atlantic Forest. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Figueiredo MSL, Barros CS, Delciellos AC et al (2017) Abundance of small mammals in the Atlantic Forest (ASMAF): a data set for analyzing tropical community patterns. Ecology 98(11):2981

Galindo-Leão C, Câmara IG (eds) (2003) The Atlantic Forest of South America: biodiversity status, threats and outlook. Island Press, Washington

GBIF (2016) Science review. Global Biodiversity Information Facility. https://www.gbif.org/pt/science-review. Accessed 26 Mar 2020

Gonçalves F, Bovendorp RS, Beca G et al (2018) Atlantic mammal traits: a data set of morphological traits of mammals in the Atlantic forest of South America. Ecology 99(2):498

Hasenack H, Silva JS, Weber EJ et al (2017) A digital version of Hueck's vegetation map of South America: 50 years after the release of his book on the sub-continent's forests. Geografia y Sistemas de Informacion Geografica 9(9):11–15

Hoffman M, Koenig K, Bunting G et al (2016) Biodiversity Hotspots (version 2016.1). https://zenodo.org/record/3261807#.Xm6JAUHQjIV. Accessed 15 Mar 2020

Hueck K (1972) As florestas da América do Sul: ecologia, composição e importância econômica. Polígono, São Paulo

IBGE (1993) Mapa de Vegetação do Brasil. Instituto Brasileiro de Geografia e Estatistica, Rio de Janeiro

IBGE (2004) Mapa de Vegetação do Brasil. Instituto Brasileiro de Geografia e Estatistica, Rio de Janeiro

IBGE (2008) Mapa da área de aplicação da Lei nº 11.428 de 2006. Instituto Brasileiro de Geografia e Estatistica, Rio de Janeiro. https://www.mma.gov.br/estruturas/sbf_chm_rbbio/_arquivos/mapa_de_aplicao_da_lei_11428_mata_atlantica.pdf. Accessed 16 Mar 2020

IBGE (2012) Manual Técnico da Vegetação Brasileira. Instituto Brasileiro de Geografia e Estatistica, Rio de Janeiro

IBGE (2019) Atlas Nacional Digital do Brasil. Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro. https://www.ibge.gov.br/apps/atlas_nacional. Accessed 16 Mar 2020

Jardim JG (2003) Uma caracterização parcial da vegetação na região sul da Bahia, Brasil. In: Prado PI, Landau EC, Moura RT et al (eds) Corredor de Biodiversidade da Mata Atlântica do Sul da Bahia. Instituto de Estudos Sócio-Ambientais do Sul da Bahia and Conservation International do Brasil, Ilhéus

Lins-e-Silva ACB, Ferreira PSM, Rodal MJN (2021) The North-eastern Atlantic Forest: biogeographical, historical and current aspects in the Sugarcane Zone. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Lima DA (2007) Estudos fitogeográficos de Pernambuco. Anais da Academia Pernambucana de Ciência Agronômica 4:243–274

Loyola RD, Lemes P, Brum FT et al (2014) Clade-specific consequences of climate change to amphibians in Atlantic Forest protected areas. Ecography 37:65–72

Maldonado C, Molina CI, Zizka A et al (2015) Estimating species diversity and distribution in the era of big data: to what extent can we trust public databases? Glob Ecol Biogeogr 24(8):973–984

Marcilio-Silva V, Zwiener VP, Marques MCM (2017) Metacommunity structure, additive partitioning and environmental drivers of woody plants diversity in the Brazilian Atlantic Forest. Divers Distrib 23:1110–1123

Mayr E (2008) Isto é biologia: A ciência do mundo vivo. Companhia das Letras, São Paulo

Metzger JP, Sodhi N (2009) Conservation issues in the Brazilian Atlantic Forest. Biol Conserv 142(6):1137–1252

Morellato LPC, Haddad CFB (2000) Introduction: the Brazilian Atlantic Forest. Biotropica 32(4):786–792

Muylaert RDL, Stevens RD, Esberard CEL et al (2017) Atlantic bats: a data set of bat communities from the Atlantic forests of South America. Ecology 98(12):3227

Muylaert RL, Vancine MH, Bernardo R et al (2018) A note on the territorial limits of the Atlantic forest. Oecol Aust 22:302–311

Myers N, Mittermeier RA, Mittermeier CG et al (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Oliveira LZ, Uller HF, Klitzke AR et al (2019) Towards the fulfillment of a knowledge gap: wood densities for species of the subtropical Atlantic Forest. Data 4(3) https://doi.org/10.3390/data4030104

Oliveira-Filho AT (2017) NeoTropTree, Flora arbórea da região Neotropical: Um banco de dados envolvendo biogeografia, diversidade e conservação. http://www.neotroptree.info. Accessed 26 Mar 2020

Oliveira-Filho AT, Fontes MAL (2000) Patterns of floristic differentiation among Atlantic forests in southeastern Brazil and the influence of climate. Biotropica 32(4b):793–810

Olson DM, Dinerstein E, Wikramanayake E et al (2001) Terrestrial ecoregions of the world: a new map of life on earth. Bioscience 51(11):933–938

Padial A, Costa APL, Bonecker CC, Nogueira DG, Roque FO, Message HJ, Diniz LP, Cavalcante LL, Vallejos MAV, Ota RR, Ludwig TAV (2021). Freshwater Studies in Atlantic Forest: general overview and prospects. In: Marques MCM & Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer

Peel MC, Finlayson BL, Mcmahon TA (2007) Updated world map of the Köppen-Geiger climate classification. Hydrol Earth Syst Sci 11(5):1633–1644

Peres EA, Pinto-da-Rocha R, Lohmann G et al (2020) Patterns of species and lineage diversity in the Atlantic Rainforest of Brazil. In: Rull V, Carnaval AC (eds) Neotropical diversification: patterns and processes. Springer International Publishing, Cham

Pôrto KC, Cabral JJP, Taberelli M (2004) Brejos de Altitude em Pernambuco e Paraíba: História Natural, Ecologia e Conservação. Ministério do Meio Ambiente, Brasília

Prado DE, Gibbs PE (1993) Patterns of species distributions in the dry seasonal forests of South America. Ann Mo Bot Gard 80:902–927

Ramos FN, Mortara SR, Monalisa-Francisco N et al (2019) Atlantic epiphytes: a data set of vascular and non-vascular epiphyte plants and lichens from the Atlantic Forest. Ecology 100(2) https://doi.org/100.10.1002/ecy.2541

Resende M, Lani JL, de Rezende SB (2002) Pedossistemas da Mata Atlântica: considerações pertinentes sobre a sustentabilidade. Rev Árvore 26(3):261–269

Ribeiro MC, Metzger JP, Martensen AC et al (2009) The Brazilian Atlantic Forest: how much is left, and how is the remaining forest distributed? Implications for conservation. Biol Conserv 142(6):1141–1153

Ribeiro MC, Martensen AC, Metzger JP et al (2011) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In: Zachos F, Habel J (eds) Biodiversity hotspots. Springer, Berlin, p 405

Rizzini CT (1979) Tratado de fitogeografia do Brasil. v.2. Aspectos ecológicos. Hucitec/Edusp, São Paulo

Roderjan CV (1994) O gradiente da floresta ombrófila densa no morro Anhangava, Quatro Barras, PR - aspectos climáticos, pedológicos e fitossociológicos. Thesis, Federal University of Paraná

Roderjan CV (2001) As unidades fitogeográficas do estado do Paraná, Brasil. https://www.researchgate.net/publication/285892213_As_unidades_fitogeograficas_do_Estado_do_Parana. Accessed 26 Feb 2020

Rodrigues AV, Bones FLV, Schneiders A et al (2018) Plant trait dataset for tree-like growth forms species of the subtropical Atlantic Rain Forest in Brazil. Data 3(2):16. https://doi.org/10.5281/zenodo.1241023

Rodrigues RC, Hasui E, Assis JC et al (2019) Atlantic bird traits: a data set of bird morphological traits from the Atlantic forests of South America. Ecology 100(6):e02647. https://doi.org/10.1002/ecy.2647

dos Santos JP, Freitas AVL, Brown KS Jr et al (2018) Atlantic butterflies: a data set of fruit-feeding butterfly communities from the Atlantic forests. Ecology 99(12):2875

Silva JMC, Casteleti CHM (2003) Status of the biodiversity of the Atlantic Forest of Brazil. In: Galindo-Leal C, Câmara IG (eds) The Atlantic forest of South America: biodiversity status, threats, and outlook. CABS and Island Press, Washington, pp 43–59

Simpson BB (1979) Quaternary biogeography of the high montane regions of South America. In: Duellman WE (ed) The South American herpetofauna: its origin, evolution, and dispersal. University of Kansas, Lawrence, pp 157–188

Smith VS, Blagoderov V (2012) Bringing collections out of the dark. ZooKeys 209:1–6

Souza Y, Goncalves F, Lautenschlager L et al (2019) Atlantic mammals: a data set of assemblages of medium- and large-sized mammals of the Atlantic Forest of South America. Ecology 100(10):e02785. https://doi.org/10.1002/ecy.2785

Tabarelli M, Pinto LP, Silva JMC et al (2005) Challenges and opportunities for biodiversity conservation in the Brazilian Atlantic Forest. Conserv Biol 19(3):695–700

Tabarelli M, Aguiar AV, Ribeiro MC et al (2010) Prospects for biodiversity conservation in the Atlantic Forest: lessons from aging human-modified landscapes. Biol Conserv 143:2328–2340

Vancine MH, Duarte KS, de Souza YS et al (2018) Atlantic amphibians: a data set of amphibian communities from the Atlantic forests of South America. Ecology 99(7):1692

Varjabedian R (2010) Lei da Mata Atlântica: retrocesso ambiental. Estudos Avançados 24(68):147–160. https://doi.org/10.1590/S0103-40142010000100013

Walter H (1986) Vegetação e Zonas Climáticas. EPU, São Paulo

Zwiener VP, Padial AA, Marques MCM (2019) The mechanisms explaining tree species richness and composition are convergent in a megadiverse hotspot. Biodivers Conserv https://doi.org/10.1007/s10531-019-01910-9

Zwiener VP, Padial AA, Marques MCM et al (2017) Planning for conservation and restoration under climate and land use change in the Brazilian Atlantic Forest. Divers Distrib 23(8):955–966

# Chapter 2
# The Atlantic Forest Ecological History: From Pre-colonial Times to the Anthropocene

**Alexandro Solórzano, Lucas Santa Cruz de Assis Brasil, and Rogério Ribeiro de Oliveira**

**Abstract** The Atlantic Forest has a long history of human interaction and transformation, since as early as the Late Pleistocene. Throughout the centuries, native populations coevolved with their environment, at the same time that always exerted some degree of pressure on the system. The early human interactions may have contributed to the extinction of megafauna, coupled with climate becoming warmer and humid. In a second moment, more modern native populations developed slash-and-burn agriculture, leaving a distinct mark in the landscape, altering forest structure and composition. Up to this point, human interaction, although leaving a footprint, left intact a large portion of forest cover. With the implementation of the Portuguese colonial enterprise, in less than 500 years, much of the Atlantic Forest was transformed into a non-forest matrix. This dramatic spatial reorganization was mainly driven by large plantation systems, particularly sugarcane and coffee, intermixed with the introduction of large grazing mammals, converting forest into pastures using exotic African grasses. With urbanization and industrialization, energy demand leads to the transformation of large forest tracts into charcoal, up until the mid-twentieth century. The Anthropocene accelerated deforestation processes with growing urbanization and industrialization until the late twentieth century with the birth of modern conservation movements that have helped diminish deforestation rates.

**Keywords** Environmental history · Social-ecological systems · Landscape transformation · Colonial enterprise

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## 2.1 Introduction

In order to understand the Atlantic Forest current ecological dynamics and conservation status, we need to historically contextualize how humans have been a part of this biome, exerting different degrees of pressure and resource use. The Atlantic Forest has a long history of human occupation and landscape transformation, with different cultures having occupied the Brazilian Atlantic Coast, each one with a very specific social-ecological legacy. Inhabited by Amerindian populations since at least 8000 years ago, the Atlantic Forest received the first Europeans landing in the late fifteenth century and was both the stage and actor of the first civilizational and biological clash that shaped Brazil's modern history (Cabral 2014). The deforestation that occurred during the colonial and post-colonial period resulted in the current extremely fragmented landscape, but that still harbours biological megadiversity with high rates of endemism, which is why the biome is considered one of the main hotspots for global biodiversity conservation (Myers et al. 2000).

The disciplines of Geography, History and Ecology are important to understand how different elements that compose the landscape are interconnected in the flow of space-time (Solórzano et al. 2009). Therefore, historical ecology "traces the ongoing dialectical relations between human acts and acts of nature, made manifest in the landscape" (Crumley 1994, p. 14). Landscapes retain the physical evidence of the different cultures that imprinted their cognitive models, decision-making, environmental perceptions and lifestyles. In this chapter, we will use the idea of landscape transformation in order to understand how in different historical periods humans interacted and fundamentally modified various aspects of their environment generating novel dynamics and sometimes even novel ecosystems (*sensu* Hobbs et al. 2006).

In this sense, it is important to understand that almost all environments on Earth have been affected, to some degree, by human activities (Balée 2006). This stands true, especially when considering species and landscape domestication since the early Holocene and the current concept of the Anthropocene, a proposal of human epoch based on evidences of anthropogenic changes in the atmosphere, lithosphere and biosphere that dramatically intensified in the twentieth century (Steffen et al. 2007). Therefore, different societies impacted landscapes in different ways and intensities, depending on socioeconomic, political and cultural factors, where a wide variety of human interactions with the landscape, in different historical and ecological contexts, can be studied as a total (integrative) phenomenon (Balée 2006).

In this chapter, we describe a chronology of human cultures interacting with the landscape of the Atlantic Forest. The modern pre-Columbian Atlantic Forest was comprised of an extensive evergreen moist forest (also known as Dense Ombrophilous Forest) covering all of the Brazilian Atlantic Coast. Together with this moist forest, there were other versions of Atlantic Forest that were seasonally dry, in more inland portions and in the northeast, and mixed moist subtropical forest with *Araucaria angustifolia*, in the southern portion. Scarano (2002) describes six distinct versions of Atlantic Forest, some of them non-forest formations such as "the open scrub vegetation of the sandy coastal plains, the swamp forests and the vegetation growing on rocky outcrops at high altitude".

The organization of this chapter follows six different moments, or social-ecological interactions, from pre-colonial times to contemporary urban society, that represent clashes between different human societies and the Atlantic Forest (Fig. 2.1). This is translated and imprinted in the landscape as different levels, or intensities, of human-forest co-existence and ecological transformation. The first moment or clash represents human arrival about 18,000 years BP in the Atlantic Coast and their role in the probable megafauna extinction of the Late Pleistocene. The second clash represents a second moment of forest biomass transformation into human food. Here we explain how native populations in Mid- to Late Holocene developed slash-and-burn agriculture, transforming Atlantic Forest structure and composition and domesticating the landscape. The third clash shows a sharp transition from pre-Columbian societies to the Portuguese colonial enterprise and the implementation of large social-ecological systems (particularly based on sugarcane and coffee plantations) and their disastrous consequence on the Atlantic Forest. The fourth clash/moment delves into the entire process of the introduction of large grazing mammals (cows, oxen, mules, horses, sheep, etc.) throughout the entire colonial period and the transformation of forest into pastures using exotic African grasses. The fifth clash explains how the growing metropolises demand for energy, starting in the eighteenth century, leads to the transformation of large forest tracts into charcoal, lasting until the mid-twentieth century when the Brazilian energy matrix transitioned into petroleum. Finally, the sixth moment ends with concluding remarks of

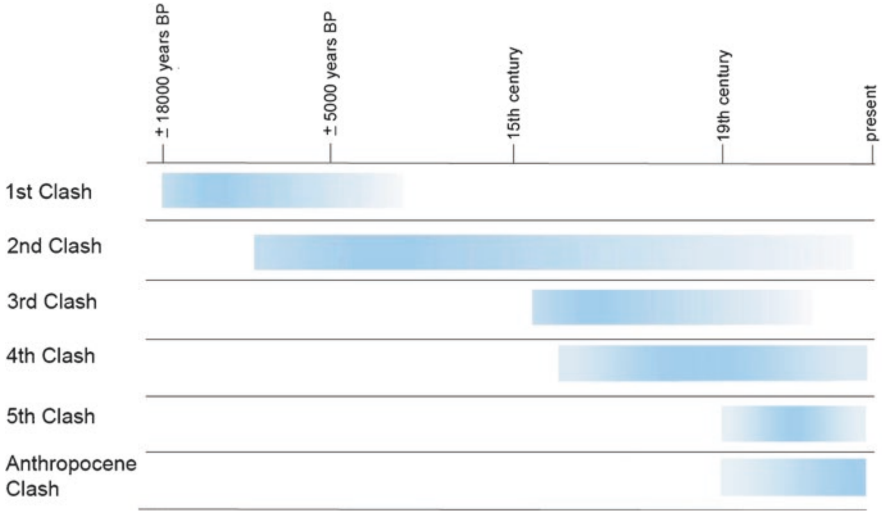

**Fig. 2.1** Timeline representing the different historical moments of the clashes between humans and the Atlantic Forest. Some clashes are overlapped in time, and the gradients of light to dark blue represent roughly degrees of intensity of the activities through history

the Atlantic Forest in the context of the Anthropocene, highlighted by the intensification of urbanization and industrialization and the birth of the conservation movement.

## 2.2 First Clash: Humans' Arrival in the Atlantic Forest and Consequences in the Landscape

It is generally considered that the most pronounced human intervention in the territory of the Atlantic Forest began with the arrival of the European colonizer to the continent. However, this fact is not entirely true. The use of specific techniques, particularly fire, enabled pre-Colombian populations a potential to alter the environment.

The arrival of humans on the American continent by the Bering Strait is an indisputable fact (Bourgeon et al. 2017), although there are other equally robust hypotheses about distinct origins (Lavallée 2000). It is also likely that the first settlers did not cross the strait just once. Migrations may have occurred both ways along this route several times over millennia. Analysis of the oldest genomes suggests that there was an early split within Beringian populations, giving rise to the northern and southern lineages (Moreno-Mayar et al. 2018). There are still other hypotheses that show the non-indigenous origin of humans in the Americas.

In terms of vegetation, the Atlantic Forest was very different from when the first Europeans arrived. We must first understand that the Atlantic Forest inhabited by native populations 1000 years BP is significantly different from what was found by the first human inhabitants of the coast in the Late Pleistocene (around 18,000 years BP). Climatic conditions of the Late Pleistocene were dramatically different, with a colder and drier climate related to the end of the last glaciation period. The Atlantic vegetation was completely different, with moist evergreen forests only occurring in certain lowland and south facing hills of the southeast. Most of the vegetation was either dry forests intermixed with open vegetation formations (a mosaic of open fields with sparse woody vegetation) or Araucaria forests in the south and mountainous areas of the southeast, also intermixed with grassland formations (Jeske-Pierusshka and Ledru 2016). This type of mixed and open landscape was ideal for the large mammals of the Pleistocene ecological necessities.

In a historical perspective, the practice of hunting by populations over a broad spectrum of time may have conditioned the Atlantic Forest diversity in many of its attributes such as composition, structure and functionality. The extinction of large mammals in the Pleistocene presents a great interest due to the possible direct involvement of humans in its process.

According to Cione et al. (2009), the most important and spectacular aspect of the natural history of mammals occurred in the transition from the Pleistocene to the Holocene, when 100% of the megamammals and 80% of the large mammals of South America were extinct. Megamammals are those with body mass over 1000 kg, like *Glyptodon* (a giant armadillo), ground sloth (*Megatherium americanum* and *Eremotherium laurillardi*) and the long llama (*Macrauchenia patachonica*). The arrival of humans in South America was the only new ecological event that took place for thousands of years. The overkill hypothesis is based initially on the coincidence between the geographical expansion of *Homo sapiens* and the extinction of megafauna. In principle, extinction restricted to large animals can be attributed not only to selective hunting but also to the demographic characteristics of the species (Diniz-Filho 2002) or to climatic changes as an accessory factor for extinctions (Cione et al. 2009). Guthrie (2006) links climate change, human colonization and extinction of large mammals. Both climate and humans may have affected the extinction dynamics of South American megafauna in the Late Pleistocene. These results corroborate other recent studies, which conclude that the synergy between humans and climate is considered the most viable cause to explain megafauna extinctions during the Late Quaternary and not one cause or another alone (Lima-Ribeiro 2013).

Although the research by Doughty et al. (2013) has occurred in the Amazon region, its results allow us to understand the broad spectrum that the fauna extinction can have in ecosystems. According to these authors, the Amazonian megafauna extinction decreased the lateral flow of phosphorus from the flooded areas of the Amazon towards the non-flooded *terra firme* forests. This caused a strong phosphorus depletion in the soil's distribution. The current limitation of this element in the Amazon basin may be a relic of an ecosystem that evolved without the functional connectivity that it one day presented, caused by megafaunal dislocation. Although the Atlantic Forest biome may have had very different processes, this study highlights the intimate relationship between biodiversity, nutrient flow, human presence and hunting practice.

The hunting dependence for protein may have historically led to severe depletion of game fauna groups. The historical loss of these groups may represent dramatic changes in the ecosystem due to numerous cascade effects (Mathias et al. 2018). These changes occur in several ecosystem key functions, such as fruit dispersal, pollination, nutrient cycling, herbivory and predation, leading to ecological extinctions of numerous groups. The extinction of giant frugivores reduced the chances of higher plant species seed dispersal (Guimarães et al. 2008). Large herbivores contribute not only to changes in seed dispersal processes but also to plant succession, species richness and composition and even species evolution (Marquis 2010). In this sense, seed dispersal networks significantly changed with the megafauna extinction. Most likely medium and small mammals became key components for the structure of the seed dispersal network, affecting both animal and plant assemblages and contributing to the shaping of modern ecological communities (Pires et al. 2014).

It is likely that not only there was a significant change in population size and distribution of large fruit-bearing trees but also a reconfiguration of landscape structure as well as composition. Therefore, dense forest cover in the Atlantic domain can be related to both Late Pleistocene and Holocene climate becoming warmer and humid, leading to forest expansion over open vegetation areas, and to the extinction of megafauna and large fauna from human overhunting coupled with climate change. This would also be a contributing factor for vegetation densification, in the absence of large mammals migrating and dislocating through the vegetation (Marquis 2010).

## 2.3 Second Clash: Transforming Forest Biomass into Food: Agriculture and Landscape Domestication

The territory of the Atlantic Forest has been inhabited by many social groups, generally referred to as indigenous or traditional populations. Pre-Columbian populations of different ethnic groups occupied the Atlantic Forest domain dating back to 10,000 years BP (Neves et al. 2004). Broad estimations of population size in Brazilian lowland areas, including the Amazon and Atlantic Forest, range from eight to ten million people before European arrival (Denevan 1992). If this statement stands, we could estimate a population of indigenous groups between three and four million people in the Atlantic Forest domain. Most of these cultural groups have a point of convergence in agriculture (Mertz et al. 2009). Due to the ecological characteristics of tropical forests, and in particular the Atlantic Forest, the knowledge produced by these populations undergoes a selection of practices that often result in a convergence between cultural processes that are quite distant from each other, in space and time (Toledo and Barrera-Bassols 2008). Much of the Atlantic rainforest territory was used in the past by some form of migratory, i.e. itinerant, agriculture, which came to be known as slash-and-burn agriculture, due to its use of fire. The predominant ethnic group that occupied most of SE and NE Atlantic Forest was of the Tupi group dating back to 5000 years BP (Barreto and Drummond 2016).

Secular or millenarian populations acting on the same ecosystem end up promoting the consolidation of a set of knowledge regarding its management, regardless of its cultural origin (Adams and Murrieta 2008). The best example is the *coivara* (slash-and-burn agriculture), practised with minimal differences in almost all Brazilian territory and at different times. The success of this method is due in large part to the cultural techniques used to face the issue of soil infertility in areas where it is practised. Therefore, fire can be considered as a fundamental tool for this type of agriculture. It is a cheap and adequate tool for the purposes of forest regeneration (Raison 1980), once certain steps (like multi-cultivation or fallow use) of this type of agriculture are followed. The essence of nomadic agriculture is the opening of a forest tract, its drying and subsequent burning.

Fertilization induced by the forest ashes allows the use of the soil by a certain amount of time. After a period of about 3 years, the productivity declines, and the crop is then abandoned to fallow, where a secondary forest colonizes the area. Whitmore (1990) states that subsistence agriculture allows a maximum of 10 to 20 inhabitants/km$^2$, since at any one time only 10% of the area may be under cultivation, due to the need for fallow land.

The vegetation established in the cultivated areas after abandonment to fallow depends not only on the availability of propagules but is also selected by the management characteristics that are employed (Bahra et al. 2014). In the same way, stump regrowth capacity, fire resistance and the dominance of certain species are fundamental for forest recovery. Thus, through several ways, this type of forest management allows for natural succession to occur and consequently forest biomass resilience through colonizing species.

The energy balance of this system is largely favourable when compared to techniques that depend on industrialized agricultural products (Bayliss-Smith 1982; Altieri 1987; Adams 2000) as these are generally unavailable to poor rural populations. However, this indigenous system is totally dependent on the maintenance of soil fertility by alternating between the cultivation period and a period in which the land is left fallow (Boserup 1972). In terms of the impact of shifting cultivation on the ecosystem, Ewel (1976) pointed out that the restoration of fertility that occurs during fallowing is due in large part to the recycling of organic material and nutrients into the upper layers of the soil through leaf litter production and subsequent decomposition.

Thus, in relation to the composition of forest ecosystems, it is considered that the historical processes of occupation of the Atlantic Forest territory have severely altered the biome's current diversity pattern. Abandoned areas previously subject to the traditional subsistence farming practice generally show a reduction in the species number of tree or shrub and a predominance of pioneer and secondary species (Chazdon 2003).

The relatively low floristic richness is, therefore, a significant characteristic of *coivara* agriculture. According to Sastre (1982), the very slow evolution of secondary formations with a low floristic diversity characterizes the so-called anthropic or anthropogenic climax. Accepting this definition, the establishment of an anthropic climax seems to be the main mark in the structure and composition of the vegetation that the agricultural activity of these traditional populations left on the forested landscape and that should remain for a long time after the end of human intervention on the Atlantic Forest biome (Oliveira 2008). Thus, these managed lands that led to anthropogenic successional trajectories can be understood as an important part of cultural landscapes in the Atlantic Forest. The floristic and structural aspects found in these areas reflect the probable selective pressure exerted by successive periods of fallow and cultivation to which they were submitted for a long time. The reduction of woody species diversity seems to be the main result of this historical process of human subsistence.

The forest landscape "discovered" by early European colonizers was far from being pristine (Denevan 1992, 2011). Spatial analysis of pre-Columbian landscape modification has shown that 60–80% of the forest physiognomies (i.e. dense evergreen and seasonally dry) where secondary forests in some level of recovery from dense human occupation (native villages), slash-and-burn agriculture (in different ages of abandonment or under period of fallow), hunting pressure, gathering and management of native flora (species and landscape domestication) (Denevan 1992; Oliveira and Solórzano 2014; Barreto and Drummond 2016). The Pristine Myth theory also indicates that possibly forest regeneration was greater following mass genocide of native population inhabiting the Atlantic Coast, making human presence less visible in the landscape in 1750 than it was in 1492 (Denevan 1992).

## 2.4 Third Clash: The Colonial Enterprise and the Implementation of Large Social-Ecological Systems

### *2.4.1 A Substantial Shift: From Extractivism to Monoculture Farming*

In the first decades after Brazil's discovery, the extractive activity of *pau-brasil* (*Paubrasilia echinata* (Lam.) Gagnon, H.C. Lima & G.P. Lewis) was the main source of wealth that the Portuguese Crown found in its new lands. In this first encounter between the Atlantic Forest and colonial settlers, the native population played a key role in extracting this valuable resource. The indigenous people, who detained a vast knowledge about these forests, were employed by the Portuguese to remove trunks of this species (Prado Jr 2011; Dias 2016), which produced a red dye (used to dye clothes), that became very popular in the European market. The colony's initial spatial configuration was the establishment of commercial depots, small punctual occupations along the coastline and no major changes in the forest, except for the selective cutting of timber and firewood in a radius around them (Dean 1996; Barreto and Drummond 2016). In any case, the territorial cost of the initial colonial enterprise was low, with *pau-brasil* logging affecting the composition and structure of these forests, however, without exerting great modifications in its functionality. This small-scale activity, both spatially and temporally, would soon lose importance, with settlers turning their attention to another type of enterprise: the great plantation. This activity presented its own set of characteristics that would redefine the course of the Atlantic Forest transformation process with social-ecological consequences reaching our present time.

### *2.4.2 The Sugarcane and the Forest: The Plantation System Begins*

First, we must stress that sugarcane and coffee were species brought from abroad, and they are not native. Along with these cultivars, many other plant species were also brought, especially fruit and ornamental species. Even when not all of them were cultivated in massive plantation systems, they also played a part in the transformation of the landscape (Oliveira and Engemann 2011). Sugarcane plantations were established in the sixteenth century in the captaincy of São Vicente, current coast of São Paulo, and afterwards expanded between the captaincies of the northeast of Pernambuco and Bahia, rapidly becoming a profitable monoculture that would transform much of the Atlantic Forest. The Northeastern Atlantic Forest is constricted by the Serra da Borborema orographic barrier, providing a distinct biodiversity when compared to the rest of the Atlantic Forest (Barreto and Drummond 2016). Although its occurrence does not extend as many kilometres inland as in the rest of the Atlantic Forest, the colonial enterprise of sugar production was not concerned and consumed it without further considerations, motivated by the sugarcane cultivation expansion, increasing sugar production (Prado Jr. 2011; Moreira 2014).

Sugarcane did not develop well on hillside or higher altitudes, demanding well-drained soils of lowland and coastal areas (Ross 2006). This factor, together with the greater proximity of ports to the metropolis when compared to São Vicente, explains the sugarcane diffusion during the seventeenth century along the Brazilian coast and its success in the northeast (Moreira 2014; Barreto and Drummond 2016).

This agricultural format, that became known as the plantation system, would also be adopted by the French, Dutch and English in their colonies (Moreira 2014). The large commercial crop cultivation was structured in three elements: (i) huge tracts of land, (ii) large numbers of slaves and (iii) specialized cultivation of one main cultivar species (Prado Jr. 2011). In the Portuguese colony, sugarcane inaugurates the natural resources appropriation system of the Atlantic Forest. Hence, forest cover and biodiversity paid the price for the establishment of immense monocultural farms.

The colonizer appropriated indigenous knowledge, replaced them and sometimes even erased them (Oliveira and Winiwarter 2010). An emblematic example is the *coivara*, an indigenous slash-and-burn agricultural system. The colonizers appropriated and applied the *coivara* in large areas as a way to clean and fertilize the land quickly and inexpensively for large-scale monoculture plantations (Dean 1996; Marquese 2008). This method adoption was pervasive, generating an agricultural system that can be classified as "pyromaniac" (Cabral 2014). However, the colonial enterprise extends the scale of action once used and does not reserve fallow areas. This poor use of technique produced landscapes dominated by mosaics of agricultural lands, fields in natural regeneration and patches of secondary vegetation interspersed with areas of higher elevation in well-preserved conditions. As the sugarcane crop expanded, new forested areas were required for new plantations.

It is important to note that sugarcane monoculture not only required a large piece of land initially, but in the process of sugarcane conversion into sugar, adjacent forests were needed to supply the mill furnaces with necessary fuel. Engemann (2005) estimated a territorial cost varying between 4 and 22 ha of Atlantic Forest per crop to supply timber and firewood (for the construction of crates to transport the product or as charcoal to feed the fire of the mills). So a new adjacent forest portion was requisitioned with every new year, in a process that did not end with the rise of coffee as the new lucrative monoculture yield – it only lost a little of its breath.

### *2.4.3 The Coffee Plantation System and the Retraction of the Atlantic Forest*

The stories of immense wealth made with coffee (*Coffea arabica* L.) and its consequent destruction of Brazilian forests throughout the nineteenth century are already well known and commented by many authors (Dean 1996; Secreto 2000; Pádua 2002; Marquese 2008; Oliveira and Ruiz 2018). It was in the extensive process of forest transformation into coffee plantations that we identify a great landscape transition. This landscape transition generated changes in social-ecological systems and, consequently, in the resilience of these landscapes.

With the increase of coffee value as an international product, coffee culture became extremely profitable at the beginning of the nineteenth century. In the farms around the city of Rio de Janeiro, immense properties and slave labour-based productions burst into scene (Drummond 1988). However, with the mentality that only virgin forest soils would be suitable for new grain crops, associated with the general belief of Brazil presenting endless lands for agriculture (Padua 2002), coffee plantations presented great mobility and expansion capacity in the forested landscape (Secreto 2000).

In Brazil, animal husbandry was separated from crops, with solid waste from livestock farming not being used (Linhares 1996), generating low soil fertility in a relatively short space of time. This was also one of the factors that led to an intensive agricultural frontier opening (Secreto 2000). Coffee barons and landowners were soon on their way to new sites in order to maximize their cultivation. Thus they climbed the Serra do Mar in the early nineteenth century, using the Paraíba do Sul Valley, an extensive forest corridor, as an axis of dispersion and frontier opening. This expansion occurred both to the west, towards the plateau of São Paulo, and to the east, towards the lowlands of Campos dos Goytacazes, and also in the northern direction reaching the *Zona da Mata* in Minas Gerais (Dean 1996; Moreira 2014). In this process, coffee plantations left behind exhausted lands (Ab'Saber and Bernardes 1958), spreading new plantations in areas still endowed with a large forest cover (Dean 1996; Secreto 2000). In other words, the coffee crop advanced into previously unexplored territories, notably controlled by indigenous groups (Holanda 2017). The arrival of landowners with large estates brought with it a wide network of trade, crafts, villages and small towns to the detriment of the Atlantic Forest. The coffee advance consisted in a land use conversion process that took place from the beginning of the nineteenth century onwards, developing into three main directions and promoting intense changes in forested landscape structural, functional and compositional attributes. This left a deep mark in a process that quickly transformed the forested landscape of colonial Brazil that reverberates until today, with the Atlantic Forest becoming an extremely fragmented and threatened biome.

The extension of land use in Brazil for coffee plantations differs from those found in other grain-producing colonies. For example, in the Paraiba Valley, spacing between coffee shrubs was three times greater than in the Island of Hispaniola (Marquese 2008), consequently consuming more land to allocate the same amount of coffee plants when compared to the Antillean competitors. The land abundance in Brazil, as opposed to the scarcity of new lands on the Caribbean island, would be the main reason for apparent land waste and low productive capacity. Still, the central justification for such a planting architecture – the coffee shrubs arranged in vertical lines "downhill" and open spacing between the ranks – was for the benefit of the foreman that would have better control of the working slaves, being more visible in an open plantation landscape (Padua 2002; Marquese 2008).

In terms of environmental constraints, coffee crop expansion was limited by climatic conditions related to the increase of altitude and annual rainfall and decrease of seasonality (Ab'Saber and Bernardes 1958). Thus, the most suitable lands for coffee plantations would be those located between 300 and 600 metres, in the interior portions of Southeastern Atlantic Forest domain, not being profitable to invest in other localities (Fraga 1963). According to these authors, these unsuitable lands for coffee would have remained largely as forests managed by small farmers, being used for traditional agriculture (*coivara* system), charcoal production and wood extraction (Ab'Saber and Bernardes 1958). The main effects of the Atlantic Forest transformation were the appearance of two environmental results: the deflagration of erosive processes (loss of A horizon, loss of macro and micronutrients) with the consequent alteration of the hydrological cycle and loss of biodiversity. These two processes, taken together, bring an immediate and constant cost to society, represented by structural and functional degradation that keeps ecosystems in operation. The direct result is the decrease and eventual loss of water resources, ecosystem functions and ecosystem services, which affect both the biome and society itself.

The environmental transformations caused by the coffee plantations at that time were so big that even regional climate was estimated to be impacted. The loss of forest cover leads to intensive erosive processes and fundamentally alters hydrological balance, leading to a drier climate, especially in the winter, with summer torrential rains that promote even more erosion (Dantas and Coelho-Netto 2019). The rainfall regime in the occupied and used areas of Atlantic Forest shifted making it similar to the savannah rainfall regime, that is, with a longer dry season. This may have extended the area of a semi-humid climate typical of the Cerrado, the Brazilian Savannah biome, to regions with a humid climate (Dantas and Coelho Netto 1996). At the same time that regional climate changed, after years of forests being converted to agricultural lands and pastures, the landscape physiognomy itself also resembled that of tropical savannahs: grasslands with varying degrees of woody vegetation and with scattered forest fragments in the landscape. This landscape physiognomic shift has led to increasing occurrence of typical savannah animal and plant species in the Paraíba Valley and other Atlantic Forest areas. Species such as the maned wolf (*Chrysocyon brachyurus* Illiger) and wolf's fruit (*Solanum lycocarpum* A.St.-Hil.), typical of the Cerrado, expanded their distribution, reaching even coastal areas (Bereta et al. 2017; Xavier et al. 2017).

The consequence of the coffee enterprise clash with the Atlantic Forest led to one of the most intense and rapid episodes of forest devastation in human history. In the space of less than a century, coffee plantations were responsible for the felling and burning of a substantial portion of the Southeastern Atlantic Forest, leaving a legacy of degraded lands, with their resilience often compromised, even hindering future possibilities of forest natural regeneration, once the majority of those lands were converted into low-performing extensive pastures lasting until present days (Brasil et al. 2018). However, in areas where agricultural and pasture lands were abandoned due to economic decay and rural exodus, natural forest regeneration has been observed, being an important factor of forest cover recovery in the last decades (Rezende et al. 2015). The sites that still maintain sizeable forest fragments are in the steepest terrain, hill tops and ridges, which prevented the development of any type of intense economic activities (Tabarelli et al. 2010). The Serra da Bocaina Park, one of the largest Atlantic Forest protected areas with 9270 ha (ICMBio 2012), is precisely the expression of this historical process in which difficult access lands were spared from human activity.

## 2.5 Fourth Clash: The Introduction of Large Ungulate Mammals and the Transformation of the Forest into Pastures with African Grasses

### *2.5.1 What Came in the First Caravels? The First Introduction of Novel Fauna*

Although cattle breeding was not considered an economic cycle such as sugarcane or coffee, the introduction and breeding of exotic animals in the colony provided changes in the Atlantic Forest ecological relations, especially regarding seed dispersal. In 1532, bovine cattle was introduced: a generalist Portuguese variety acclimatized to the native grasses. Since the first incursions in Atlantic Forest soil, the colonists brought pigs, chickens, goats and sheep to supply the demand for meat (Dean 1996).

Landscape changes start from the beginning of Brazilian colonization, with these species introductions, in a parallel process to what happened in North America (Crosby 2011). It was not only the Portuguese who came to the territory that later was called Brazil but many of the animal and plant species that were part of their culture, as well as their landscape management techniques, and the very conception of nature that the settlers envisioned was inseparably linked to the entire colonial process. Although the integration between farming and livestock was not exercised, with manure not being incorporated in crops, raising animals often had a designated place in the landscape and in properties, for their proper development. Pastures, hen houses and stockyards were frequent structures in the landscape and also within monoculture farms (Linhares 1996).

After all, in addition to food security that came from cattle breeding, they were also responsible for almost all the driving force in these colonial ventures, bearing the production loads in carts, spinning the grain mill and transporting provisions to the farm warehouses. Its dual function – food and mechanics – guaranteed its permanence in the landscape (Brasil 2018). Thus, their importance in the Portuguese colony was not only due to cultural traits or food preferences but also for the colonial monoculture operational logistics (Moreira 2014).

### *2.5.2 Exotic Grass Demand: The Nutritional Issue*

After emphasizing the importance of these animals in the colonial landscape, it is necessary to refer to a nutritional problem that the first shepherds/herdsmen faced: native Brazilian forests did not provide enough food for the development of these grazing mammals. Horse breeding, due to their nutritional requirement, was one that had the most difficulty with native grasses, precisely because these grasses and the Eurasian breed of horses did not coevolve (Dean 1996). High tannin concentrations and sclerophyllous characteristics may constitute feeding barriers to non-specialist herbivores (Oliveira and Solorzano 2014).

Therefore, grass species that were more grazing resistant, with higher nutritional values, became a demand among herbivorous mammal breeders. The exotic grass arrival was mainly due to this demand from tropical livestock. Tropical America, especially South America, had savannahs with native grass dominance, but which did not support the grazing of ungulate mammals. Thus, several $C_4$ metabolic grasses from Africa were brought to support livestock activity on the South American continent (D'Antonio and Vitousek 1992).

The colonial landscape became flooded with exotic grasses from the African continent. The exotic grass introduction history in Brazilian soil is one of uncertainty, especially regarding its intentionality. The most widespread and romanticized version portrays African grass seeds brought by chance in the beds used in slave ships. Guinea grass (*Megathyrsus maximus* Jacq.) and molasses grass (*Melinis minutiflora* P. Beauv) were two exotic species heavily used to form pastures, each of them being adapted and used under different conditions (slope feature and angle, selection by type of grazer, grass management practices, etc.). Early ranchers benefited from the rapid growth rates of these exotic grasses, their high resistance and distinct ability to adapt to local conditions, promoting grazing expansion towards the hinterlands of Brazil during the sixteenth, seventeenth and eighteenth centuries, especially after the sugar cycle decline (Silva et al. 2012).

Pasture areas often assumed two positions in the colonial enterprise: (a) at the settlement frontier and agricultural frontier due to the rusticity of the activity, preparing the landscape for further occupation and guaranteeing new territories for other future activities (Linhares 1996), and (b) in the infertile abandoned lands and weary from excessive planting (Dean 1996), spreading the resistant African grasses and providing food for the farm grazing animals (sugarcane or coffee properties).

One of the observed consequences of using agriculturally eroded soils with exotic pastures is the massive expression of this land use currently in Paraíba do Sul River Valley, which, as we have seen, harboured the largest coffee plantations in the Americas during the nineteenth century. As coffee moved to the São Paulo western lands towards the end of the nineteenth century, the old coffee plantations gradually were converted into pastures, with cattle ranching being the most practised economic activity in the region until present day. This is a pattern that can be observed in most of the Atlantic Forest, not only in the states of Rio de Janeiro, São Paulo and Minas Gerais, since replacing abandoned farm lands with pastures was a very common practice in the past and still is today (Prado Jr. 2011; Cabral 2014; Holanda 2017). Currently the main landscape characteristics of this region is a vast cover of exotic pastures, with small-edge effected forest fragments embedded in an almost impermeable matrix. The livestock permanence in these areas where forest cover has been removed is one of the main constraints for natural regeneration, due to the constant trampling and seedling herbivory, coupled with intense soil erosion and long distance from any source of propagules from remnant forests.

## 2.6 Fifth Clash: The Demand for Energy for the Growing Urbanization and the Transformation of Forest into Charcoal

The great deforestation of the Atlantic Forest was an extensive process and had much to do with the peculiar social system that developed with the initial colonization. For example, cattle raising, together with mining and farming, may have eliminated most of the forest in a 300–400 km arc inland from Rio de Janeiro and Santos (Williams 2006). Besides these three historical factors, the need of energy provided by forest biomass also had great importance in the transformation of the Atlantic Forest landscape. Historically firewood has always accompanied human trajectory as a first necessity energy source. Its transformation into charcoal via muffled combustion (through charcoal kilns) allows an increase of its caloric power with a reduction in mass. This makes it an energy source that can be transported over long distances. The calorific value of charcoal per unit of weight is 3.1 times higher than that of firewood (Genovese et al. 2006). Unlike petroleum, charcoal can be produced locally and is an energy source which production cost is almost exclusively composed by the manual labour invested in it (Olson 1991).

In the nineteenth-century urban society, charcoal had a fundamental role as an energy source, being an important part of iron manufacture. Another important aspect of this energy source was its domestic consumption in household ovens and stoves. Possibly during the nineteenth century, a more intense process of charcoal production began on the slopes of the south and southeastern coastal mountains, called Serra do Mar. This was due to the combination of the following factors: (a) demand for the specific energy of charcoal, (b) availability of the forest resource, (c) ease of production and low cost and (d) the unemployed human contingent after the abolition of slavery (1888) and also of emancipated men before it.

Although firewood has been consumed for thousands of years in the Atlantic Forest, from the eighteenth century on this consumption increased substantially. The use of charcoal as an energy matrix represented an intense demand for firewood, thus increasing pressure on forests. For example, in the mid-nineteenth century, an intense process of charcoal production began on the slopes of the Pedra Branca Massif, located in the West Zone of the city of Rio de Janeiro. Charcoal was the city's energy matrix (Oliveira et al. 2011). As to the volume demanded at that time for the metabolism of the city, it suffices to remember that in order to produce a ton of iron, it was necessary to melt and reduce iron ore from 2.8 to 3.8 tons of charcoal (Paradis-Grenouillet 2013). Another aspect of great demand was civil construction, particularly stonemasonry activities producing cobblestone, portals, facades and sidewalks (Oliveira et al. 2011). In all the work and construction that rocks were used as a construction material, there was a need to sharpen iron forged tools such as chisels. They were taken to the forge to be trimmed to the red on the anvil. These forges, fuelled by charcoal, increased dramatically in number in order to account for the exponential growth of urban areas at the turn of the twentieth century. Thus, the largest cities in the Atlantic Forest domain (such as Rio de Janeiro and São Paulo) had charcoal as its main energy matrix, which production demanded the use of vast areas of forest. On the other hand, the use of charcoal represented a diffuse source of energy from the landscape, due to the fact that it could be manufactured practically from any forest that provided the necessary biomass.

The remnants of historic charcoal production sites today are recognized in the middle of the forest landscape by the plateaus excavated by the charcoal producers on the slopes and by vestiges of dark earth on these plateaus, with fragments of charcoal left in the soil (Rodrigues et al. 2018). These plateaus were built along the slopes of the hilly and mountainous forested areas. The charcoal kilns were constructed with nearby available firewood, producing a 3-metre-high cone. It is estimated that the charcoal producers selected trees by their size and characteristic, performing a selective logging, sparing large trees (being too difficult to cut into manageable pieces for firewood) and trees that produced latex (which could spoil a batch of charcoal by affecting combustion speed and heat in the kiln) (Sales et al. 2014). The wood was covered with clay to allow for a muffled combustion, which is the main step to transform wood into charcoal. Its production and distribution gave rise to a network of paths and roads that crossed the mountains towards the metropolitan areas. Charcoal was partly transported by the charcoal makers by the use of mule troops. Given the great need of this energy source, the region of Pedra Branca Massif became a true production centre of charcoal. So far, 1145 old charcoal kilns have been found in this massif, according to an inventory still in progress, by Oliveira and Fraga (2016). The Pedra Branca Massif consisted only of one of the sources of charcoal for the city of Rio de Janeiro. This research has been expanded to several forest formations in the southeast, and information is now available on the occurrence of past charcoal production in 26 forest remnants, most of which are situated on slopes in part of the southeastern region of the country. Thus, it is very likely that many forest remnants of the Brazilian southeast region have been used in the past as an energy production site (Oliveira and Scheel-Ybert, 2018).

This intense use of the forests of the mountainous slopes of the Serra do Mar and Mantiqueira shows three important points regarding the concealment of human work in the landscape. The first concerns the volume of charcoal produced. Taking into account only the 1145 charcoal kilns found in the Pedra Branca Massif, Sales et al. (2014) estimated that they produced a total of 11,300 tons of charcoal, equivalent to a volume of the order of 61,700 $m^3$.

The second point concerns the export of nutrients from the forest system. When the forest is cleared and converted into charcoal, the nutrients present in the forest biomass are exported out of the forest system and incorporated in urban and industrial metabolism. Oliveira and Fraga (2016) estimated that the export of nutrients in the Pedra Branca Massif in the studied areas was about 887.5, 157.5 and 1304.8 kg/ha, respectively, for N, P and K. In other words, the functional recovery of the ecosystem occurs in less than 16 years. Although this estimate may simplify complex processes and local realities that may change these values, it provides evidence that the forest system rebounds structurally after firewood is removed. This structural resilience of the forest is in large part a consequence of the forest management technique employed by the charcoal producers of selective logging and artisanal production of charcoal.

The third point is about the invisibility of the charcoal workers in the landscape concerning their own social condition. In the last decade of the nineteenth century, slavery abolition represented a reorganization of society as an immense contingent of illiterate, unprepared and deprived people entered the formal job market. In a society that still maintained a strong slavery ideology, even after abolition, manual labour was a degrading task, but the impoverished could not do without this type of work. In this context, becoming a charcoal maker was relatively simple: an axe and a lighter were enough. The work of the poor African descendants is almost invisible in notarial documents. The landscape thus becomes a significant repository of these stories concerning the almost invisible work of the charcoal producers in the forest hinterlands.

## 2.7 Concluding Remarks: The Anthropocene Clash and Current Social-Ecological Configuration

Although the debate is still open on the exact beginning of the Age of Humans – Anthropocene – it is widely accepted that this novel human epoch can be divided into two stages: the Industrial Era (ca. 1800–1945) and the Great Acceleration (ca. 1945–present time) (Steffen et al. 2007). In Brazil, the Anthropocene manifested itself peripherally, that is, triggered by the agricultural and extractive demands of European and North American industrial centres (Cabral and Bustamante 2016). The most complete example of this is the annihilation of the Atlantic Forest in the Paraíba Valley between the mid-nineteenth century and the beginning of the twentieth century by the expansion of coffee cultivation. In the 1930s, the Brazilian Anthropocene began to accelerate, due to the industrialization of the national economy. The growing urbanization was at the forefront of this process, pressuring the Atlantic Forest with demands linked mainly to energy and construction materials (Dean 1996). In the early 1960s, Brazilian agriculture finally began to incorporate modern equipment and inputs, which could have prevented further deforestation through increased productivity on consolidated farmland – but the Atlantic Forest was already practically decimated (Cabral and Bustamante 2016). Few regions of reasonably flat and arable land of the biome remained highly forested, with less than 10% of original cover surviving towards the late twentieth century, mostly concentrated on mountainous areas.

Currently, the Atlantic Forest can be understood as an anthropogenic biome (Ellis and Rammankutty 2008), composed of forest ecosystems and a mosaic of silvo-agro-pastoral systems and urban areas, which interact in different degrees both by the constant transformation of human activity and aging of these heterogeneous landscapes and by the degree of permeability of the non-forest matrix allowing the gene flow of different taxonomic groups (Tabarelli et al. 2010; Oliveira and Solórzano 2014). Thus, the structure and composition of ecological communities as well as the functioning of ecosystems are being affected by natural and anthropogenic processes and may lead to changes never before verified, producing novel or emerging ecosystems (Hobbs et al. 2006).

Concentrating about 70% of Brazilian population and 80% of GDP, the Atlantic Forest today is much more a region of predominantly agriculture, pasturelands, cities, complex road systems, industrial parks and factories than forests per se. So what we have today is a result of thousands of years of interaction between human societies and the forest. Hence, the Atlantic Forest that has survived brings evidence – in its composition, structure and functionality – of the dialectic presence of humans (Oliveira 2008). The closer the forest remnants are to dense human occupation, i.e. urban areas, the more pronounced is the human footprint. Throughout the entire colonial process, numerous exotic plant species were introduced (for food, timber, ornamental purposes) in cities and farmlands. Today many of these species are dominant components of what has been called novel ecosystems, which have composition and relative abundance not previously seen in a given biome (Hobbs et al. 2006).

In the 1980s and 1990s, the great counter-movements of nature conservation emerged in Brazil, expanding a scientific debate that would define the contours and limits of the Atlantic Forest biome. From then on, the academy's bridge with society initiated a legislative body that would envision the protection of Brazil's most threatened biome and one of the hottest hotspots. The Atlantic Forest Law from 2006, which launched a zero deforestation policy for the biome, became the cornerstone for the biome's conservation, yet actions to combat widespread illegal hunting and invasive species must also be enforced and incentivized (Rezende et al. 2018).

## References

Ab'saber AN, Bernardes N (1958) O Vale do Paraíba, Serra da Mantiqueira e arredores de São Paulo. CNG, Rio de Janeiro

Adams C (2000) As populações caiçaras e o mito do bom selvagem: a necessidade de uma nova abordagem interdisciplinar. Revista de Antropologia 43(1):145–182

Adams C, Murrieta RSS (2008) Agricultura de corte e queima e florestas tropicais em um mundo em mudança. Boletim do Museu Paraense Emílio Goeldi 3:1

Altieri MA (1987) Agroecology: the scientific basis of alternative agriculture. Westview Press, Boulder, 227 p

Bahra E, Zaragocinb DC, Makeschina F (2014) Soil nutrient stock dynamics and land-use management of annuals, perennials and pastures after slash-and-burn in the Southern Ecuadorian Andes. Agric Ecosyst Environ 188:275–288

Balée W (2006) The research program of historical ecology. Annu Rev Anthropol 35:75–98

Barreto C, Drummond JA (2016) Com açúcar e sem afeto: A cana e a devastação da Mata Atlântica nordestina. In: Cabral DC, Bustamante AG (eds) Metamorfoses florestais: culturas, ecologias e as transformações históricas da Mata Atlântica. Prismas, Curitiba, pp 265–285

Bayliss-Smith TP (1982) Taro (Colocasia esculenta) in the New Guinea Highlands. Second Papua New Guinea Food Crops Conference (1980, Goroka, Papua New Guinea), 134–147

Bereta A, Freitas SR, Bueno C (2017) Novas ocorrências de Chrysocyon brachyurus (Carnivora) no estado do Rio de Janeiro indicando a expansão de sua distribuição geográfica. Boletim da Sociedade Brasileira de Mastozoologia 78(5–8)

Boserup E (1972) Evolução agrária e pressão demográfica. Hucitec, São Paulo

Bourgeon L, Burke A, Higham T (2017) Earliest human presence in North America dated to the last glacial maximum: new radiocarbon dates from Bluefish Caves, Canada. PLoS One 12:1–15

Brasil LSCA (2018) Legados socioecológicos do café: transição de paisagem no Vale do Rio Paraíba do Sul. Dissertartion, Pontifícia Universidade Católica do Rio de Janeiro

Brasil LSCA et al (2018) Legados Ecológicos e Transição da Paisagem no Médio Vale do Paraíba do Sul: Do Café às Pastagens. Fronteiras J Social Technol Environ Sci 7:78–99

Cabral DC (2014) Na presença da floresta: Mata Atlântica e história colonial. Garamond, Rio de Janeiro

Cabral DC, Bustamante AG (2016) Introdução. In: Cabral DC, Bustamante AG (eds) Metamorfoses florestais: culturas, ecologias e as transformações históricas da Mata Atlântica. Prismas, Curitiba, pp 17–33

Chazdon RL (2003) Tropical forest recovery: legacies of human impact and natural disturbances. Perspect Plant Ecol Evol Systematics 6:51–71

Cione AL, Tonni EP, Soivelzon L (2009) Did humans cause the late Pleistocene-Early Holocene mammalian extinctions in South America in a context of shrinking open areas? In: Haynes G (ed) American megafaunal extinctions at the end of the Pleistocene. Springer, Reno, pp 125–144

Crosby A (2011) Imperialismo ecológico. Companhia das Letras, São Paulo

Crumley C (1994) Historical ecology: a multidimensional ecological orientation. In: Crumley C (ed) Historical ecology: cultural knowledge and changing landscape. School of American Research Press, Santa Fe

D'Antonio C, Vitousek P (1992) Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annu Rev Ecol Syst 23(1):63–87

Dantas ME, Coelho Netto AL (1996) Resultantes geo-hidroecológicas do ciclo cafeeiro (1780-1880) no médio vale do rio Paraíba do Sul: uma análise quali-quantitativa. Anu Inst Geocienc 19:61–78

Dantas ME, Coelho-Neto AL (2019) A denudação antropogênica da paisagem: processos erosivodeposicionais no médio Vale do Rio Paraíba do Sul. In: Oliveira RR, Lazos-Ruiz AE (eds) Geografia Histórica do café no Vale do Rio Paraíba do Sul. Editora PUC-Rio, Rio de Janeiro, pp 105–124

Dean W (1996) A ferro e fogo: a história e a devastação da mata atlânticabrasileira. Companhia das Letras, São Paulo

Denevan WM (1992) The pristine myth: the landscape of the Americas in 1492. Ann Assoc Am Geographer 82(3):369–385

Denevan WM (2011) The "pristine myth" revisited. Geographical Rev 101(4):576–591

Dias MH (2016) O pau-brasil na Bahia colonial: Zonas de ocorrência, condições de exploração e impactos ambientais. In: Cabral DC, Bustamante AG (eds) Metamorfoses florestais: culturas, ecologias e as transformações históricas da Mata Atlântica. Prismas, Curitiba, pp 151–175

Diniz-Filho JAF (2002) Modelos ecológicos de extinção da megafauna do Pleistoceno. Xingó 2:53–81

Doughty CE, Christopher E, Wolf A, Malhi Y (2013) The legacy of the Pleistocene megafauna extinctions on nutrient availability in Amazonia. Nat Geosci 6:761–764

Drummond JA (1988) O jardim dentro da máquina: breve história ambiental da Floresta da Tijuca. Revista Estudos Históricos 1(2):276–298

Ellis E, Ramankutty N (2008) Putting people in the map: anthropogenic biomes of the world. Front Ecol Environ 6:439–447

Engemann C (2005) Consumo de recursos florestais e produção de açúcar no período colonial – o caso do Engenho do Camorim, RJ. In: Oliveira R (ed) As marcas do homem na floresta: história ambiental de um trecho urbano de mata atlântica. PUC-Rio, Rio de Janeiro, pp 119–140

Ewel JJ (1976) Litter fall and leaf decomposition in a tropical forest succession in eastern Guatemala. J Ecol 64:293–308

Fraga C (1963) Resenha histórica do café no Brasil. Agricultura em São Paulo 10(1):1–21

Genovese AL, Udaeta MEM, Galvão LCR (2006) Aspectos energéticos da biomassa como recurso no Brasil e no mundo. In: Proceedings of the VI Encontro de Energia no Meio Rural, Unicamp, Campinas, 6-8 June 2006

Guthrie RD (2006) New carbon dates link climatic change with human colonization and Pleistocene extinctions. Nature 441:207–209

Hobbs RJ et al (2006) Novel ecosystems: theoretical and management aspects of the new ecological old order. Glob Ecol Biogeogr 15:1–7

Holanda S (2017) Caminhos e Fronteiras. Companhia das Letras, São Paulo

Instituto Chico Mendes de Biodiversidade (2012) Plano de Manejo do Parque Nacional da Serra da Bocaina. ICMBio, Brasilia

Jeske-Pierusshka V, Ledru MP (2016) A Mata Atlântica no final do Quaternário: Dinâmicas climatobotânicas e antropogênicas desde o Último Máximo Glacial. In: Cabral DC, Bustamante AG (eds) Metamorfoses florestais: Culturas, ecologias e as transformações históricas da Mata Atlântica. Prismas, Curitiba, pp 37–53

Lavallée D (2000) The first South Americans. University of Utah, Salt Lake City

Lima-Ribeiro M (2013) Mudanças climáticas, colonização humana e a extinção da megafauna na América do Sul. Thesis, Universidade Federal de Goiás

Linhares M (1996) Pecuária, alimentos e sistemas agrários no Brasil (séculos XVII e XVIII). Arquivos do Centro Cultural Calouste Gulbenkian, Le Portugal et l'Europe Atlantique, le Brésil et l'Amérique Latine. Mélanges offerts à Fréderic Mauro 34

Marquese R (2008) Diáspora africana, escravidão e a paisagem da cafeicultura no Vale do Paraíba oitocentista. Almanack braziliense 7:138–152

Marquis RJ (2010) The role of herbivores in terrestrial trophic cascades. In: Terborgh J, Estes JA (eds) Trophic cascades: predators, prey, and the changing dynamics of nature. Island Press, London, pp 109–124

Mathias M et al (2018) Pleistocene megafaunal extinctions and the functional loss of long-distance seed-dispersal services. Ecography 41(1):153–193

Mertz O, Padoch C, Fox F, Cramb RA, Leisz J (2009) Swidden change in Southeast Asia: understanding causes and consequences. Hum Ecol 37(3):259–264

Moreira R (2014) A formação espacial brasileira: contribuição crítica aos fundamentos espaciais da geografia do Brasil. Consequencia, Rio de Janeiro

Moreno-Mayar J et al (2018) Early human dispersals within the Americas. Science 362(6419). https://doi.org/10.1126/science.aav2621

Myers N et al (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Neves WA et al (2004) Early Holocene human skeletal remains from Cerca Grande, Lagoa Santa, Central Brazil, and the origins of the first Americans. World Archaelogy 36(4):479–501

Oliveira RR (2008) When the shifting agriculture is gone: functionality of Atlantic Coastal Forest in abandoned farming sites. Boletim do Museu Paraense Emílio Goeldi 3:213–226

Oliveira RR, Engemann C (2011) História da paisagem e paisagens sem história: a presença humana na Floresta Atlântica do Sudeste Brasileiro. Esboços Revista do Programa de Pós-Graduação em História da UFSC 18(25):9–31

Oliveira RR, Fraga JS (2016) Fluxos de energia, matéria e trabalho na construção da paisagem do Rio de Janeiro do século XIX. In: Franco JLF, Silva SD, Drummond JA, Tavares GG (eds) História ambiental: territórios, fronteiras e biodiversidade. Garamond, Goiania, pp 35–54

Oliveira RR, Ruiz AEL (2018) Geografia Histórica do café no Vale do Rio Paraíba do Sul, 1st edn. Editora PUC-Rio, Rio de Janeiro

Oliveira RR, Sheel-Ybert R (2018) A hidden landscape: the paleo-territory of charcoal makers in Rio de Janeiro, Brazil. In: Paradis-Grenouillet S, Burri S, Rouad R (eds) Charbonnage, charbonniers, charbonnières: confluence de regads autour dún artisanat méconnu. Presses Universitaires de Provence, Limoges, pp 33–38

Oliveira RR, Solórzano A (2014) Três hipóteses ligadas à dimensão humana da biodiversidade da Mata Atlântica. Fronteiras J Social Technol Environ Sci 3:80–95

Oliveira RR, Winiwarter V (2010) Toiling in paradise: knowledge acquisition in the context of colonial agriculture in Brazil's Atlantic Forest. Environment and History 16(4):483–508

Oliveira RR, Fraga JS, Berck DE (2011) Uma floresta de vestígios: metabolismo social e a atividade de carvoeiros nos séculos XIX e XX no Rio de Janeiro, RJ. INTERthesis 8:286–315

Olson SD (1991) Firewood and charcoal in classical athens. Hesperia 60(3):411–420

Pádua JA (2002) Um sopro de destruição: pensamento político e crítica ambiental no Brasil escravista, 1786–1888. Zahar, Rio de Janeiro

Paradis-Grenouillet S (2013) Les forêts metallurgiques: analyses dendroecologiques et approches géohistoriques. Thesis, Université de Limoges

Pires M et al (2014) Reconstructing past ecological networks: the reconfiguration of seed-dispersal interactions after megafaunal extinction. Oecologia 175:1247–1256

Prado C Jr (2011) Formação do Brasil contemporâneo. Companhia das Letras, São Paulo

Raison RJ (1980) Possible forest site deterioration associated with slash-burning. Search 11(3):68–72

Rezende CL et al (2015) Atlantic Forest spontaneous regeneration at landscape scale. Biodivers Conserv 24:2255–2272

Rezende CL et al (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:208–214

Rodrigues AF et al (2018) Humic acid composition and soil fertility of soils near an ancient charcoal kiln: are they similar to Terra Preta de Índios soils? J Soils Sediments 1:1–8

Ross J (2006) Ecogeografia do Brasil: subsídios para planejamento ambiental. Oficina de Textos, São Paulo

Sales GPS, Solorzano A, Patzlafff RG, Oliveira RR (2014) Resultantes ecológicas, práticas culturais e provisão de lenha para a fabricação de carvão nos séculos XIX e XX no Rio de Janeiro. Pesquisas Botânica 65:389

Sastre C (1982) Notion de climax en régions néotropicales. Compte rendu des ceances de la Societé de Biogeographie 58(3):117–123

Scarano FR (2002) Structure, function and floristic relationships of plant communities in stressful habitats marginal to the Brazilian Atlantic rain forest. Ann Bot 90:517–524

Secreto M (2000) Dominando la floresta tropical: desbravamentos para el café paulista (Brasil siglo XIX). Theomai 1:1–16

Silva M, Boaventura V, Fiorovanti M (2012) História do povoamento bovino no Brasil Central. Revista UFG 13(13):34–41

Solórzano A, Oliveira RR, Guedes-Bruni RR (2009) Geografia, história e ecologia: criando pontes para a interpretação da paisagem. Ambiente e Sociedade 12(1):49–66

Steffen W, Crutzen PJ, McNeill JR (2007) The Anthropocene: are humans now overwhelming the great forces of nature? Ambio 36(8):614–621

Tabarelli M et al (2010) Prospects for biodiversity conservation in the Atlantic Forest: lessons from aging human-modified landscapes. Biol Conserv 143(10):2328–2340

Toledo VM, Barrera-Bassols N (2008) La memoria biocultural: la importância ecológica de las sabidurías tradicionales. Icaria editorial, Barcelona

Williams M (2006) Deforesting the earth: from prehistory to global crisis: an abridgment. University of Chicago, Chicago

Xavier MS et al (2017) Noteworthy coastal records of the maned wolf, Chrysocyon brachyurus (Illiger, 1815), in Southeastern Brazil. Boletim Sociedade Brasileira de Mastozoologia 78:9–13

# Chapter 3
# The North-Eastern Atlantic Forest: Biogeographical, Historical, and Current Aspects in the Sugarcane Zone

**Ana Carolina B. Lins-e-Silva, Paulo Sérgio M. Ferreira, and Maria Jesus N. Rodal**

**Abstract** The Northern Atlantic Forest (NAF) covers a narrow strip along the Brazilian coast, from 3° to 19° S, over lowlands and plateaus. *Mangroves*, *dunes*, *restingas*, and forests occur from low to high altitudes, along an East-to-West climatic gradient. Initially covering 23 million hectares, NAF was reduced to 13%, much converted into sugarcane fields. Within NAF, in the North-eastern sugarcane zone (NESZ), land conversion happened in waves, the last in the middle 1970s, resulting in a predominant pattern of small-sized fragments in a sugarcane matrix. Acknowledged for its high endemism for plants, butterflies, and birds, high species numbers occur at a regional level and low numbers at the patch level, with a signal of taxonomic homogenization, simplification, and species loss; edge effects trigger retrogressive succession, while chronic disturbances like harvesting and hunting deplete plant and animal species. Despite all threats, some areas maintain high biodiversity and provide essential ecosystem services. Since most remnants are located within sugarcane properties, forest conservation relies on private-owned lands and their legal compliance to meet a vegetation debt that approaches 280,000 hectares and accomplish an ongoing initiative to restore approximately one million hectares. This is the key to a good prognosis for the NESZ Atlantic Forest.

**Keywords** Biogeography · Biodiversity · Conservation · Anthropogenic drivers

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## 3.1 The Natural Settings: Biogeographical Aspects of the Northern Atlantic Forest

### *3.1.1 Biogeographical Overview*

The Atlantic Forest is acknowledged as the most ancient forest formation in Brazilian territory, and its assembly is likely to have happened before the separation between South America and Africa (Rizzini 1997). Along its original extension, the Atlantic Forest extends approximately from 3° to 33° S and 29° to 58° W, considering the "integrative limit," a sensu *lato* delimitation proposed by Muylaert et al. (2018). Within this range, there are expressive variations in altitude (0 ~ 3000 m), climate, and topography, which determine both local and regional biodiversity distributions (Oliveira-Filho and Fontes 2000; Thomas 2008; Marques et al. 2011). Taking into account all this complexity, it is well recognized that the Atlantic Forest, on a broad scale, does not occur as a homogenous body but is a mosaic of regions with distinct features and dynamics.

A brief search in the literature allows us to identify the individualization of at least two large blocks, the Northern and the Southern Atlantic Forest, divided by the *Rio Doce* Valley, around the latitude of 19° S (Fig. 3.1a) (Bates et al. 1998; Costa et al. 2000). This division is marked not only by geographical features but also by a sharp floristic turnover (Fiaschi and Pirani 2009). The topographical barrier of the *Rio Doce*, however, does not seem to be effective enough, in ecological terms, to create this separation between all taxonomic groups on these two sectors (Saiter et al. 2016). Apparently, there is a coincidence between the location of the *Rio Doce* and other ecological factors that account for species turnover, such as physiological restrictions related to precipitation and temperature gradients (Thomé et al. 2014; Saiter et al. 2015; Saiter et al. 2016), biogeographical hybrid inheritance (Crisci et al. 1991; San Martín and Ronquist 2004), and different dynamics of stability during the climatic changes of the Last Quaternary glaciations (Carnaval and Moritz 2008; Carnaval et al. 2009; Carnaval et al. 2014).

As a consequence, the floristic composition of the Northern Atlantic Forest is predominantly tropical, with Laurasian elements, such as the families Fabaceae and Euphorbiaceae, among others. The Southern portion, on the other hand, is predominantly temperate, with remnants of the Gondwana separation, such as Myrtaceae, Winteraceae, and arborescent ferns, as examples (Anderson et al. 1999; Pennington and Dick 2004; Fiaschi and Pirani 2009; Eisenlohr and Oliveira-Filho 2015). Additionally, climatic changes during the Late Pleistocene and Holocene (last 120 thousand years) (Cox et al. 2016) left distinct impacts on different regions of the Atlantic Forest, with periods during which the forest remained restricted to refuges, while open formations expanded under dryer and colder climates (Carnaval and Moritz 2008; Carnaval et al. 2009; Carnaval et al. 2014). During Quaternary climatic changes, the Northern area was much less climatically stable than the Southern region (Carnaval et al. 2014). This was evidenced by Costa et al. (2017), who used climate modeling to predict shifts in South American ecosystems, including the Atlantic Forest, over the last 21,000 years (Fig. 3.1b).

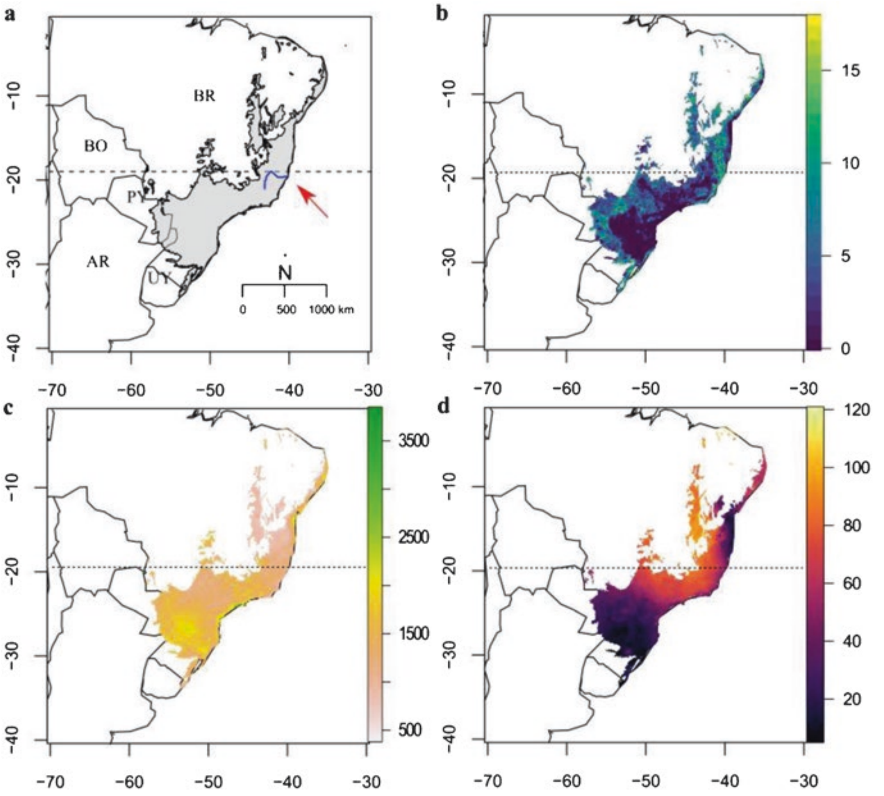

**Fig. 3.1** (**a**) Atlantic Forest sensu *lato*. (From Muylaert et al. 2018) with hypothetical division between North and South (*dotted line*) and *Rio Doce* (*red arrow*), (**b**) biome instability in the last 21,000 (Edited from Costa et al. 2017), (**c**) annual precipitation in mm. (Fick and Hijmans 2017), (**d**) rainfall seasonality in mm. (Fick and Hijmans 2017)

In the present, the most striking environmental factor that influences the variation on vegetation composition and physiognomy in the Northern Atlantic Forest is the precipitation gradient (Oliveira-Filho et al. 2015; Eisenlohr and Oliveira-Filho 2015). Covering a latitudinal range from ~3° to 19° S, on the extremes, from North to South, annual precipitation average in Rio Grande do Norte is around half of the annual total recorded in Bahia, on the Southern limit of the region (Thomas and Barbosa 2008).

### *3.1.2 Vegetation Physiognomies, Climate, and Topography*

The theoretical bases for vegetation classification indicate that evolutionary pressures selected plant strategies. Such strategies are defined by morphofunctional traits (habit, phenology, and height, among others) that occur on individuals, which belong to species. Consequently, it is understood that the recognition of distinct vegetation types depends on the combined occurrence of species with similar strategies within the same physiognomic-floristic dominium.

Distinct vegetation types in the Northern Atlantic Forest primarily occur over two large geomorphologic units, lowlands and coastal plateaus, situated on geological formations of Quaternary and Neogene origin, respectively, and reworked surfaces, with lithologically diverse origins (Silva et al. 1993). According to IBGE (2012), these coastal areas are unstable surfaces covered by a vegetation of first occupation with edaphic influence, lying on terrains influenced by marine sediments, with the formation of marine sand deposits on dunes and restingas. Vegetation types occurring on such sandy terrains borrow their names from the geological background and are also called *dunes* and *restingas*. On the river mouths, over mixed sediments from riverine and marine origin, the so-called mangrove or *mangal* is occupied by typical vegetation, the *mangrove* forest. In the lowlands and coastal plateaus, precipitation and temperature vary along with the latitudinal gradient (Muylaert et al. 2018).

The lowland occupies terrains varying from 0 to 30 m above sea level, covered by *mangroves*, *dunes*, and *restingas*, whose physiognomy adjusts from open shrubby vegetation, with different degrees of aggregation, to forest stands. The coastal plateau relief varies from plain to wavy, reaching slightly over 100 m of altitude, with sediment origin, with a variable degree of carving, which mixes narrow, deep, and steep gorges, open and flat valleys, and broad meadows (Silva et al. 1993). In these areas, low-fertility, sandy, and deep soils sustain forests, according to Rizzini (1997).

Reworked surfaces, another prominent relief feature in the Northern sector, are characterized by a group of dissected relief, with deep valleys and geologically complex plateaus, resulting from uplift and erosion of the pre-Cambrian basement (Silva et al. 1993). A geological feature is notable: the locally called *mar de morros* (sea of hills), a relief formed by dissected hills (Ab'Saber 2003), which precede the Borborema Plateau, along the coast of Pernambuco and Alagoas, and toward the eastern slopes of the Diamantina Plateau, in Bahia (Thomas 2008). Soils on this relief unit vary according to the topography: deep and well-drained latosols on the flat tops, podzols of median depth and good drainage on the steep slopes and bottom of narrow valleys, and also meadow gleysols, organic and water-saturated (Silva et al. 1993). Thomas and Barbosa (2008) recognize the occurrence of rainforest and seasonal forests on the reworked surfaces. It is essential to point out that forests are wetter when closer to the coast and get dryer toward the inner continent, where a heterogeneous mixture of forest types can occur, depending on the altitude and hillside orientation (Thomas and Barbosa 2008).

Different vegetation types in the Northern Atlantic Forest are a result of abiotic factors, such as climatic variation (rainfall, temperature, wind, and other variables) and physiographic features (hillside exposition, topography, continentality effect, and soils, among others), as well as biogeographic histories. As a general pattern, precipitation decreases along a South-to-North gradient, but another essential East-to-West gradient exists, where precipitation diverges from over 2000 mm year$^{-1}$ on the lowlands close to the Atlantic coast down to 1000 mm year$^{-1}$ to the West.

Moreover, there is a significant seasonality, with the number of dry months (with precipitation lower than 100 mm) varying from zero to seven, along the gradient (Fig. 3.1c, d).

The Atlantic Forest is not only influenced by natural factors but also by anthropogenic drivers that shaped the existing forest remnants. In the Northern Atlantic Forest, a rapid land use conversion soon after the country's colonization in the sixteenth century was mostly induced by the same natural features that account for the forest existence: a smooth relief, a humid tropical climate and proximity to the coast, and the primary access of colonizers and explorers to the country. As a consequence, three contrasting features characterize the Northern Atlantic Forest in our time: while it maintains a high biodiversity, related to a high degree of endemism, forests are also subjected to the highest fragmentation process in any forest ecosystems, due to anthropic pressure, and is much more neglected in research and conservation initiatives when compared to the Southern region (Carnaval and Moritz 2008; Thomas 2008; Carnaval et al. 2009).

## 3.2 The Anthropogenic Drivers of Change: Forest Loss and the Definition of a North-Eastern Sugarcane Zone

Initially, the Northern Atlantic Forest covered around 28% (ca. 23 million hectares) of six states as a narrow strip along the Brazilian coast, here listed from North to South, with percentages of original coverage per state: Rio Grande do Norte (6.65%), Paraíba (10.62%), Pernambuco (17.22%), Alagoas (54.89%), Sergipe (46.53%), and Bahia (31.85%) (Fig. 3.2) (data from Fundação SOS Mata Atlântica

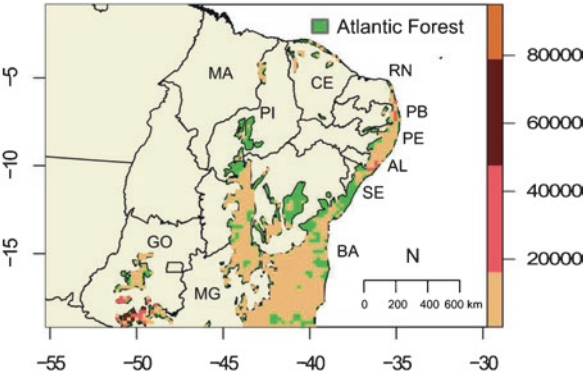

**Fig. 3.2** The North-eastern sugarcane zone within the Atlantic Forest (in green, from Muylaert et al. 2018). Sugarcane plantations are located in the states of Rio Grande do Norte (RN), Paraíba (PB), Pernambuco (PE), Alagoas (AL), and Sergipe (SE). Dark lines mark the state borders. On the left, the orange-to-brown scale marks areas planted with sugarcane, in hectares, per municipality. (Source: https://www.ibge.gov.br/apps/dinamica_agropecuaria/)

2018). Along five centuries, since country colonization, the Northern Atlantic Forest was converted mainly into two land uses, anthropic areas and built areas, according to the classification provided by Rezende et al. (2018). Following the pattern for the entire biome, anthropic areas, which include agriculture, pasture, mining, degraded areas, and all other artificial non-built surfaces, were by far the primary conversion in this Atlantic Forest region and cover nowadays 55% of the lands (data available from Rezende et al. 2018, http://geo.fbds.org.br/). As a result of this intense land conversion, only around 13% of the original forest remains, which correspond to 3.6% of the territory of the six states in contrast with the formerly existent 28% (Fundação SOS Mata Atlântica 2018).

Among anthropic uses, there was a conversion of substantial forested areas into sugarcane (a perennial grass of the family Poaceae, *Saccharum officinarum* L.) fields. This key cultivated crop in hot-humid North-eastern Brazil was the first to happen and is the most extensive agricultural practice up to now in all states, except for Bahia. Sugarcane was introduced by the Portuguese colonizers around 1530 (Cabral D de 2014) and is recognized as one of the bases over which the Brazilian territory was built. Nowadays, sugarcane is the third major cultivated crop in Brazil in planted area and covers 10.2 million hectares (Produção Agrícola Municipal/PAM 2017), of which 5.1 million hectares is in the Atlantic Forest biome (IBGE 2017). Sugarcane fields, however, do not form a continuous area, but is noticeably divided into two main zones, the North-eastern and the Middle-Southern, the latter predominantly concentrated in the state of São Paulo and also covering parts of the neighboring states (IBGE 2017), in which around 80% of the Brazilian sugarcane production is found (Goes et al. 2011). In the North-eastern zone, sugarcane plantations do not correspond to the boundaries of the Northern Atlantic Forest (Fig. 3.2), but is comprised between a North border in the state of Rio Grande do Norte (latitude 5°40′ S) and a South limit in the state of Sergipe (10°49′ S) (de Andrade 1994), excluding the Atlantic Forest region in the state of Bahia. It is essential to highlight that the introduction of sugarcane in North-eastern Brazil occupied the coastal territory down to the state of Bahia, to a region called *Recôncavo* (around the latitude of 12° S), where the activity was prosperous, due to the abundance of appropriate soils and rivers (Freyre 1937). However, sugarcane production declined in Bahia during the sixteenth century, recovered in the following two centuries, but decreased by the end of the nineteenth century (Pires 2009). Sugarcane fields correspond today to less than 3% of the cultivated crops in Bahia (PAM 2017).

At present, the North-eastern sugarcane zone occupies parts of 186 municipalities originally covered by the Atlantic Forest in 5 states (IBGE 2017). Within this sugarcane zone, percentages of Atlantic Forest converted into anthropic non-built, and urban areas reach 81% and 2%, respectively (data available from Rezende et al. 2018), and only 9 to 10% of the original forest remains (Fundação SOS Mata Atlântica 2018). When we look at data on forest original cover in the Atlantic Forest with a focus on the North-eastern sugarcane zone and contrast with recent data on sugarcane planted areas (PAM 2017), it is notable that sugarcane fields cover alone an average of 20% of those previously forested lands (about one million out of five million hectares). However, sugarcane coverage varies among states, replacing from

4% of the original Atlantic Forest in Sergipe up to 20% of the original forest lands in the other states in the sugarcane sector. Numbers fluctuate not only spatially but also temporally, with a decreasing tendency during the last three decades.

The spatial trend in the North-eastern zone contrasts with the national trend for sugarcane: looking at the entire country, the planted area increased steadily from one million to six million hectares from 1955 to 2006 (Goldemberg et al. 2008) and from 6 to 10.5 million, from 2006 to 2017 (IBGE 2017). More than half of this expansion happened in the Cerrado, a savannah-like biome in central Brazil (Lapola et al. 2014), whereas 1.4 million hectares were recently converted for sugarcane plantations in the Atlantic Forest (IBGE 2017). Nevertheless, there are no records of recent land conversion in the North-eastern zone, from which the latest report (PAM 2017) gives a cultivated area of around 800,000 hectares in the municipalities with records of operative cultivation against 1.3 million hectares recorded in 1990. It is important to understand this new tendency, as it apparently creates an opportunity for forest natural regeneration and restoration initiatives.

Before thinking about future opportunities, it is crucial to consider what we know regarding the pattern of past land conversion from native forest to agriculture in the North-eastern zone and its present-day configuration. G. Freyre, a sociologist, reported in 1937 that, during the first centuries after colonization (sixteenth to eighteenth centuries), the first wave of deforestation took place, when sugarcane was planted on the plain river valleys along the coast. Later, in a second wave of forest replacement by agriculture along the nineteenth century, slopes with clay soils in the states of Pernambuco and Alagoas were converted, and forests were left only at the top of the hills and in deep valleys (de Andrade 1994; Freyre 1937). Freyre (1937) listed some available estimates for remaining forest cover at that time: around the 1920s, Philipp von Luetzelburg, a German scientist that lived and studied plants and forests in the state of Pernambuco, estimated a remnant forest cover of 14% for the state, against quoted original 34% – which means approximately a loss of 60% of the original forests; in the 1930s, João de Vasconcelos-Sobrinho, an agronomist and ecologist, estimated 10% of the state covered by remaining coastal forests, including mature and young patches, meaning a forest loss of around 70%. These numbers from 1937 reveal that other episodes of deforestation were still to happen in the twentieth century. In effect, today it is estimated that the original forest once covered 17% of the state area but remains in only 2% of Pernambuco, which means a loss of nearly 90% of the original cover (data calculated from inventories by Fundação SOS Mata Atlântica 2018 and Rezende et al. 2018).

The third episode of intense deforestation occurred in the middle of the twentieth century when flat plateaus of sedimentary origin with well-drained and less fertile soils were occupied (Ferreira et al. 2015). By this time, low-fertile soils could be corrected with agronomic techniques (de Andrade 1994), and sugarcane productivity could be improved through genetic programs. Such programs have made it possible to obtain new varieties with enhanced productivity and resistance to pests, diseases, and climate restrictions (Goes et al. 2011).

The fourth and last wave of conversion began in 1975, due to the establishment of the Brazilian Alcohol Program (*Proálcool*) which aimed at reducing oil imports by producing ethanol from sugarcane (Goldemberg et al. 2008). Deforestation is considered to be the worst threat posed by the program. Although the general claim is that sugarcane expansion should only occur over pasture lands, there are pieces of evidence of a massive land conversion of natural forests into sugarcane plantations (for a detailed example, see Trindade et al. 2008). It appears that sugarcane expansion was limited only by the available logistics for crop management, for instance, relief constraints for planting and harvesting (see Silva et al. 2007 for a general example of the effect of relief on forest conversion in South-eastern Brazil).

Those repeated episodes of land conversion led to highly fragmented and reduced forest landscapes in the Northern Atlantic Forest zone. The resulting pattern of fragmentation is well documented at the large scale. Ribeiro et al. (2009) analyzed the spatial distribution of forest remnants and calculated the total Atlantic Forest cover in Brazil as 11.4% to 16%. Also looking at the entire biome, the Fundação SOS Mata Atlântica (2018) calculated 15%, while Rezende et al. (2018) gave a higher percentage of forest coverage, reaching 28% of native formations. Looking at the North-eastern sugarcane zone, the cover percentage obtained for the biogeographical sub-region (BSR) Pernambuco proposed by Silva and Casteleti (2005) is a good approximation, as it covers 3.8 million hectares, but excludes the state of Sergipe. Ribeiro et al. (2009) calculated 12.1% for the BSR Pernambuco, which includes an impactful low cover of only 0.6% for *restingas* and *mangroves*. From other data, when analyzing forest cover in North-eastern states, excluding Sergipe in order to keep approximately the same area of the BSR Pernambuco, we came to the following percentages: 9.17% (Fundação SOS Mata Atlântica 2018) and 16% (Rezende et al. 2018). Therefore, forest cover calculated in this zone is always lower than in the entire biome. Another fact calls our attention: as a result of enhanced automated detection and higher resolution of satellite imagery, Rezende et al. (2018) reported an amount of mapped native vegetation for the biome that is twice the size of previous numbers. Despite that, forest cover within the North-eastern sugarcane zone remained low. This clearly shows that low percentages of forest cover are not a consequence of improved mapping techniques, but a consistent scenario of forest reduction in this portion of the North-eastern Atlantic Forest.

Beyond the low forest cover, other important and alarming landscape features within the North-eastern sugarcane zone are size and number of forest remnants. This critical situation is so evident that it can be pictured and described regardless of the scale of the study, from particular landscapes to the entire biome. In a pioneer study within this region of severe fragmentation, Ranta et al. (1998) looked at a particular 267,400 hectare landscape in the South of Pernambuco (latitude 8°36' S) and recorded 1839 forest fragments summing together 62,300 hectares or 23% of the area. On the results, two characteristics called attention due to the positive aspect: 23% of coverage was per se a high percentage (much greater than the average for the region), and isolation was low since fragments were located close to each other. However, the study described a scenario with negative aspects regarding size and edge effects: approximately 48% of the fragments were smaller than 10 hectares, with an average fragment size of only 34 hectares. The largest mapped fragment was 1539 hectares and the smallest one 0.06 hectare, with a high proportion of edge habitats when applying any edge distances greater than 100 m. Araujo-Silva

(2015) evaluated a fraction of the same landscape (11,600 ha) and found a similar pattern, with somewhat high forest cover (33.46%), a low mean distance between fragments (93.35 m), but an average fragment size of only 17.2 hectares. Further South, in the state of Alagoas (latitude 9°01 S), a 66,700 ha landscape mainly composed by sugarcane monoculture was described by Santos et al. (2008) as retaining approximately 9000 ha (13%) of forest cover divided into 109 forest fragments, ranging in size from 1.67 ha to 3500 ha. The 3500 ha patch was referred to as the largest and best-preserved forest patch in the region, with core areas of undisturbed forest interior.

In the North of Pernambuco (latitude 7°44' S), Trindade et al. (2008) documented what we earlier called the fourth wave or *Próalcool*-induced deforestation, when studying a 28,000 ha landscape to evaluate not only spatial but also temporal changes along 30 years (from 1975 to 2005). Similarly to the studies reported above, they described a severe fragmentation characterized by relatively high forest coverage but composed by small fragments immersed in a sugarcane matrix. The authors also recorded a decrease in the percentage of forest cover from 45.6% to 24% and a decline in the average patch size from 83 to 61 hectares in this period. As a result of deforestation, isolation (which seemed to be less crucial in Ranta et al. 1998 study) is more than doubled with time, defining a temporal pattern of "forest shrinking." Thus, the largest fragment in this landscape in 2005 did not exceed 500 hectares. Further North, in the state of Paraíba (latitude around 7° S), Lima (2016) described a 95,700 ha landscape, of which 20% is composed by Atlantic Forest fragments with various ages all surrounded by or abandoned after sugarcane cultivation.

The actual scenario of fragmentation within the sugarcane zone has orientated the classification of forest fragments in class sizes, as proposed by Trindade et al. (2008), in small (<30 ha), medium (30–200 ha), or large (>200 ha) in Northern latitudes (around 7° S), or by Mendes Pontes et al. (2016) in Southern latitudes (8–10 °S), in which fragments ranged from very small (≤10 ha) to large (>1000 ha, the largest being 3478.3 ha). In fact, Ribeiro et al. (2009) pointed out the inexistence of a single fragment larger than 10,000 ha in the entire BSR Pernambuco and reported low functional connectivity for the area, in which only 1% was protected in nature reserves. Hence, from local to national scales, the present situation is very critical and poses an important question: what are the consequences for biodiversity maintenance under this picture where small fragments prevail within a matrix of predominantly sugarcane fields?

## 3.3 Anthropogenic Disturbances and Biodiversity: Consequences of Five Centuries of Land Conversion and the Present-Day Setting

In a fragmented landscape, there are many expected consequences for biodiversity, related to permanent forest removal, and chronic disturbances, produced by constant management practices in the matrix and anthropic use of forest resources. The first and straightforward consequence of forest loss is area reduction (Fahrig 2003).

Considering that the predominant landscape in this region is characterized by small-sized fragments accounting for a reasonably high forest cover and low isolation, would it be possible that forest cover helped to moderate the area effect? If so, total forest coverage would create a condition that kept biodiversity at high levels on the landscape scale. In effect, Pinto et al. (2010) examined the microclimatic conditions of forest patches in a matrix of sugarcane and found that microclimate, even in the smallest forest patches, is shaped by landscape configuration, so that forest cover minimizes heat and humidity exchanges between forest and matrix habitats. Therefore, this "forest-mediated buffering" might help to maintain relatively high levels of biodiversity, but a high complementarity among small-sized fragments is also expected.

For sessile groups, such as plants, a large number of endemic taxa and high richness are reported for the region. A search in the Brazilian Plant List (Flora do Brasil 2019), using the following filters, "group: angiosperms," "endemism: only endemic to Brazil," "state," "origin: native," and "phytogeographic domain: Atlantic Forest," reveals that the number of species in this forest block varies from 3354 in the state of Bahia to 244 in Rio Grande do Norte. Species richness is often recorded high at the landscape level, while lower richness occurs at the patch level. A systematic collection of angiosperm plant species from only nine 12 to 388 ha fragments within a 280 ha sugarcane matrix recorded 826 species in 112 families of angiosperms (Melo et al. 2011). This inventory included a great variety of life forms, such as herbs, shrubs, trees, herbaceous and woody vines, epiphytes, parasites, hemiparasites, and myco-heterotroph forms. Lists of only tree species, by far the most studied life form, reviewed from 32 forest fragments in the state of Pernambuco, recorded a total richness of 364 species, 35% out of which were rare, and only 54–58 species per fragment (alpha diversity), along with a high complementarity (beta diversity) (Araujo-Silva et al. 2014). A survey of herbs, carried out in 10 small forest fragments plus the interior areas of a 3500 ha control site, found 134 species in total but recorded that species richness was reduced by half in small fragments, whereas 68% of species were exclusive to interior forest habitat (Lima et al. 2015). Examining the entire BSR Pernambuco, Lôbo et al. (2011) added a historical perspective and evaluated a dataset with 4671 records of 650 native tree species, for 2 periods, before and after 1980. As this turning point marks the fourth wave of deforestation and the last peak of fragmentation in this region, the authors found that local floras are becoming more similar to each other with time, with a signal of floristic convergence and high susceptibility to taxonomic homogenization.

The situation for animal species seems more worrying and confirms the tendency for homogenization and species loss, although this region has been acknowledged for its endemism for butterflies and birds (Silva and Casteleti 2005). For birds, the BSR Pernambuco holds 434 bird species, 15 of which are currently listed as threatened and 65% are strongly associated with forest habitats (Pereira et al. 2014). The first three endemic birds in Brazil to be declared extinct come from this area, and one species is extinct in the wild (Hasui et al. 2017). In local landscapes, a bird survey from 5 forest patches within sugarcane fields registered 184 species and revealed that 30 species previously cited for the area, in the 1940s, were missing, all of them described as more sensitive to the decreasing of forest areas and habitat interior (Farias et al. 2007). For mammals, the study of 21 forest fragments in the BSR Pernambuco resulted in a list of 21 medium- and large-sized mammal species out of 38 species previously occurring in the study area (Mendes Pontes et al. 2016). As no fragment may host the entire lasting mammal community, which is nowadays highly simplified, authors forecasted that mammalian fauna in this region will comprise only four species in this twenty-first century.

Another main consequence of forest fragmentation is the edge effect, which alters the structure and functioning of a belt of forest near the boundaries (Fahrig 2003). Depending on the edge width, small fragments may be entirely affected by the different conditions from the margins. There is no consensus about the edge width that should be adopted in North-eastern sugarcane-dominated landscapes. A width of 100 m for edge-dominated habitats was adopted by Oliveira et al. (2008), whereas Silva et al. (2008) examining the effects on plant assemblages along 100 m concluded that forest borders strongly influence the first 40–60 m. Moreover, Guerra et al. (2013) showed that topographic positions of edges should also be taken into account, as more significant plant dynamics were recorded on steeper borders. Regardless of the adopted distance across edges, the effect is always severe for biodiversity, by removing shade-tolerant species, decreasing plant diversity, enhancing the density of fine stems, and homogenizing assemblages, in a process that initiates a degeneration process or a retrogressive succession, as warned by Santos et al. (2008).

Being considered a principal cause of environmental degradation in developing countries, the frequent removal of small fractions of forest biomass constitutes a chronic form of disturbance (Singh 1998). Along with area and edge effects, such impacts must be measured and understood. Aiming to identify how much pressure the harvesting of wood products put on Atlantic forest ecosystems, Medeiros et al. (2011) measured the volume of all wood products in residences near the forests and found 86 tree species used by residents, mainly for fuel, which accounted for 92% of total annual wood consumption. Monthly income was inversely related to the static wood volume and the rate of wood consumption, revealing a pressure posed by the economic status of people living in these landscapes (Medeiros et al. 2012). Hunting is also a chronic disturbance that has depleted or extirpated bird species, some of them more susceptible, especially large-bodied species and ground frugivores (Silveira et al. 2003) and mammal species, frequently hunted with feral dogs (Mendes Pontes et al. 2016).

Despite all threats, some Atlantic Forest areas in this region preserve their significance in terms of habitat quality and therefore hold expressive biodiversity. For example, a single area in the *Usina* Serra Grande (in Alagoas, latitude around 9° S) keeps 16 species of threatened birds in 3500 ha (Silveira et al. 2003); the Ecological Station of Murici, covering 6116 ha (also in Alagoas, latitude around 9°15' S), has 14 threatened bird species and is listed as an Important Bird Area (IBA) by BirdLife International (2013); and even a single 385 ha fragment (in Pernambuco, at 8° S) is a habitat for 288 butterfly species (Melo et al. 2019).

Forests in the sugarcane zone are also responsible for the maintenance of essential ecosystem services, such as water protection and carbon storage. Based on data from forest interior, edges, and fragments in *Usina* Serra Grande, Dantas de Paula et al. (2011) estimated that forest interiors retain an average carbon pool of 202.8 ton C.ha$^{-1}$; however, this stock can drop to half along edges and small fragments. Regarding water supply, it is known that the Atlantic Forest as a whole provides water for more than 125 million Brazilians (Joly et al. 2014). Although there is still a lack of data quantifying the service in the North-eastern zone, a study carried out in one protected area, the Dois Irmãos State Park, illustrates the role of the forest in maintaining the water quality. The effective costs for water treatment in reservoirs without bordering forests are five times greater than in the Prata reservoir, which is surrounded by mature forests in the Park (Alcântara et al. 2012). From the largest fragment of Atlantic Forest in North-eastern Brazil, a 7000 ha area under the jurisdiction of the Brazilian Army, a survey with stakeholders revealed the value and predominance of water-related services provided by the forest, including availability, regulation, and quality (Guimarães et al. 2017).

## 3.4 A Challenging Future for Atlantic Forest Conservation in the North-Eastern Sugarcane Zone

In a region where anthropic uses converted more than 80% of the lands, most forest remnants are located within rural properties, especially in those where sugarcane is cultivated, the locally called *usinas* (sugar mills). A large amount of data on biodiversity presented in this chapter come from the following *usinas*, in partnerships between landowners and universities or research centers: *Usinas* São José, Serra Grande, Trapiche, and Miriri. Others come from studies in protected areas, such as parks and ecological stations, and also from military-owned forests (Guimarães et al. 2017). Therefore, forest conservation must be based on a system integrating protected areas created by the government and proportions of rural properties where native vegetation must be maintained, according to the Brazilian Native Vegetation Protection Law (NVPL).

Looking at the future, there are the legal demands to be complied with in order to meet the NVPL requirements (Brancalion et al. 2014). In accordance with the law, marginal strips along all water bodies (Areas of Permanent Preservation) must be covered by native vegetation, which creates a legal vegetation debt that reaches approximately 773,000 hectares for the Northern Atlantic Forest as a whole or nearly 276,000 hectares in the Northern sugarcane zone (data calculated from Rezende et al. 2018). There are also ongoing initiatives for restoration, under the Atlantic Forest Restoration Pact (AFRP), joining together stakeholders from non-governmental organizations, governmental agencies, private companies, and research institutions (Melo et al. 2013). In the sugarcane zone, from Rio Grande do Norte to Sergipe, the pact plans to restore approximately one million hectares, totaling three million hectares if the state of Bahia is included, which accounts alone for two million hectares of the planned restoration in the Northern Atlantic Forest (Calmon et al. 2011). The already mentioned decrease in sugarcane planted area in the region, of around 500,000 hectares or 40% since 1990 (PAM 2017), indicates that there are lands available for restoration. Indeed, during the last two decades, sugarcane producers have actively participated in restoration projects, although sometimes with poor outcomes (Costa et al. 2016). There is, hence, an urgent appeal for the adoption of efficient restoration techniques, for framing restoration as a sustainable economic activity (Melo et al. 2013), keeping in mind that the primary beneficiary is agriculture, especially when targeting restoration of riparian sites (Brancalion et al. 2014).

The effect of cultural practices or activities in the matrices of forest landscapes must be investigated. In sugarcane fields in the North-eastern zone, pre-harvest burning is still a common practice, used to facilitate the manual harvest (Coelho et al. 2011; Goes et al. 2011). To what extent such practice and others (chemical fertilization, herbicides, and insecticides, for instance) threaten the biodiversity in forest remnants is still not accurately assessed. Moreover, there are fragments immersed in heavily urbanized matrices, which can cause chronic disturbances of high intensity and define stronger edge effects, as revealed by Guerra et al. (2017), when studying a gradient from urban to rural forests in Pernambuco.

As a synthesis, in this region, forest biodiversity and ecosystem functioning are severely affected by area and edge effects, topography, and chronic disturbances. Time is a variable that should also be considered, given that such assemblages might still be under accommodation since the last fragmentation episode, at around 40 years ago, and a time-delayed extinction or extinction debt is still expected (Tilman et al. 1994). However, time has also revealed that forests are highly resilient and regenerate in landscapes with a relevant forest cover, following a progressive and convergent pattern (Nascimento et al. 2014). Implications for conservation derived from the actual setting of the Atlantic Forest in the sugarcane zone are that, for the maintenance of high biodiversity, every single fragment counts. Integration between productive and conservation sectors in joint initiatives must guarantee the maintenance of existing forests, allow natural regeneration, and perform active restoration, in order to ensure a good prognosis for the Atlantic Forest scenario in the sugarcane zone.

## References

Ab’Saber AN (2003) Os domínios de natureza no Brasil: potencialidades paisagísticas. Ateliê Editorial, São Paulo

Alcântara F, Freitas J, Pinto SRR, Menezes CE (2012) Pagamentos por serviços ambientais no estado de Pernambuco: linha de base e caracterização do serviço ambiental do Parque Estadual de Dois Irmãos. Editora Universitária, Recife

Anderson JM, Anderson HM, Archangelsky S, Bamford M, Chandra S, Dettmann HR, Mcloughlin S, Rosler O (1999) Patterns of Gondwana plant colonization and diversification. J Afr Earth Sci 28:145–167 de Andrade MC (1994) Modernização e pobreza: a expansão da agroindústria canavieira e seu impacto ecológico e social. Unesp, São Paulo

Araujo-Silva MF (2015) influência do relevo na fragmentação e estrutura da vegetação na floresta atlântica, Sub-região Pernambuco. Dissertation. UFRPE, Recife

Araujo-Silva MF, Aguiar MMB, Silva MCG, Feliciano ALP, Lins-e-Silva ACB (2014) Complementaridade ecológica: a diversidade alfa e beta na Floresta Atlântica pernambucana. https://doi.org/10.12702/VIII. SimposFloresta. 2014.238–650-1

Bates JM, Hackett SJ, Cracraft J (1998) Area-relationship in the Neotropical lowlands: an hypothesis based on raw distributions of passerine birds. J Biogeogr 25:783–793

BirdLife International (2013) More than 12,000 important bird and biodiversity areas have been identified on land and at sea. Available via http://www.birdlife.org. Accessed 10 Apr 2019

Brancalion PHS, Garcia LC, Loyola R, Rodrigues RR, Pillar VD, Lewinsohn TM (2014) A critical analysis of the native vegetation protection law of Brazil (2012): updates and ongoing initiatives. Nat Conserv 14S:1–15

Cabral D de C (2014) Na presença da floresta: Mata Atlântica e história colonial. Garamond, Rio de Janeiro

Calmon M, Brancalion PHS, Paes A, Aronson J, Castro P, da Silva SC, Rodrigues RR (2011) Emerging threats and opportunities for large-scale ecological restoration in the Atlantic Forest of Brazil. Restor Ecol 19(2):154–158

Carnaval AC, Hickerson MJ, Haddad CBF, Rodrigues MT, Moritz C (2009) Stability predicts genetic diversity in the Brazilian Atlantic Forest hotspot. Science 323:785–189

Carnaval AC, Moritz C (2008) Historical climate modelling predicts patterns of current biodiversity in the Brazilian Atlantic forest. J Biogeogr 35:1187–1201

Carnaval AC, Waltari E, Rodrigues MT, Rosauer D, VanDerWal J, Damasceno R, Prates I, Strangas M, Spanos Z, Rivera D, Pie MR, Firkowski CR, Bornschein MR, Ribeiro LF, Moritz C (2014) Prediction of phylogeographic endemism in an environmentally complex biome. Proc Royal Soc Lond 281:20141461

Coelho ST, La Rovere EL, Guardabassi P, Grisoli R (2011) Bioenergy Brazil: biofuels environmental zoning in Brazil. GBIO-IEE-USP/CENTROCLIMA-COPPE-UFRJ. Available via http://www.iee.usp.br/gbio/

Costa AS, Malhado ACM, Bragagnolo C, Correia RA, Ladle RJ (2016) Ecological outcomes of Atlantic Forest restoration initiatives by sugarcane producers. Land Use Policy 52:345–352

Costa GC, Ledru M-P, Shepard DB, Hampe A, Werneck FP, Martinez PA, Costa LP, Leite YLR, Fonseca GAB, Fonseca MT (2000) Biogeography of south American forest mammals: endemism and diversity in the Atlantic Forest. Biotropica 32:872–881

Costa GC, Ledru M-P, Shepard DB, Hampe A, Werneck FP, Martinez PA, Moritz C, Mazzochini GG, Carnaval C (2017) Biome stability in South America over the last 30 kyr: inferences from long-term vegetation dynamics and habitat modelling. Glob Ecol Biogeogr 27:285–297

Cox CB, Moore PD, Ladle RJ (2016) Biogeography: an ecological and evolutionary approach. Wiley-Blackwell, London

Crisci JV, Cigliano MM, Morrone JJ, Roig-Juñent S (1991) Historical biogeography of Southern South America. Syst Zool 40:152–171

Dantas de Paula M, Alves Costa CP, Tabarelli M (2011) Carbon storage in a fragmented landscape of Atlantic forest: the role played by edge-affected habitats and emergent trees. Trop Conserv Sci 4(3):349–358

Eisenlohr PV, Oliveira-filho AT (2015) Revisiting patterns of tree species composition and their driving forces in the Atlantic forests of southeastern Brazil. Biotropica 47:689–701

Fahrig L (2003) Effects of habitat fragmentation on biodiversity. Annu Rev Ecol Evol Syst 34:487–515

Farias GB, Alves AGC, Lins-e-Silva ACB (2007) Riqueza de aves em cinco fragmentos de Floresta Atlântica na Zona da Mata Norte de Pernambuco, Brasil. Biotemas 20(4):111–122

Ferreira MP, Alves DS, Shimabukuro YE (2015) Forest dynamics and land-use transitions in the Brazilian Atlantic Forest: the case of sugarcane expansion. Reg Environ Chang 15(2):365–377

Fiaschi P, Pirani JR (2009) Review of plant biogeographic studies in Brazil. Plant Syst Evol 47:477–496

Fick SE, Hijmans RJ (2017) WorldClim 2: new 1-km spatial resolution climate surfaces for global land areas. Int J Climatol 37(12):4302–4315. https://doi.org/10.1002/joc.5086

Flora do Brasil 2020 under construction. Jardim Botânico do Rio de Janeiro. Available via http://floradobrasil.jbrj.gov.br/. Accessed 12 May 2019

Freyre G (1937) Nordeste: aspectos da influência da cana sobre a vida e a paisagem do nordeste do Brasil. Global, São Paulo, 255 p

Fundação SOS Mata Atlântica/Instituto Nacional de Pesquisas Espaciais (2018) Atlas dos remanescentes florestais da Mata Atlântica, Período 2016–2017: Relatório Técnico. Available via http://mapas.sosma.org.br/

Goes T, Marra R, Araújo M, Alves E, Souza MO (2011) Sugarcane in Brazil: current technologic stage and perspectives. Revista de Política Agrícola1, pp 52–65

Goldemberg J, Coelho ST, Guardabassi P (2008) The sustainability of ethanol production from sugarcane. Energ Policy 36:2086–2097

Guerra TNF, Araújo EL, Sampaio EVSB, Ferraz EMN (2017) Urban or rural areas: which types of surrounding land use induce stronger edge effects on the functional traits of tropical forests plants? Appl Veg Sci 20(4):538–548

Guerra TNF, Rodal MJN, Lins-e-Silva ACB, Alves M, Menezes Silva MA, Mendes PGA (2013) Influence of edge and topography on the vegetation in an Atlantic Forest remnant in northeastern Brazil. J For Res 18:200–208

Guimarães H, Braga R, Mascarenhas A, Barros Ramos T (2017) Indicators of ecosystem services in a military Atlantic Forest area, Pernambuco – Brazil. Ecol Indic 80:247–257

Hasui E, Metzger JP, Pimentel RG, Silveira LF, Bovo AADA, Martensen AC, Uezu A, Regolin AEL, De Oliveira AAB, Gatto CAFR, Duca C, Andretti CB, Banks-Leite C, Luz D, Mariz D, Alexandrino ER, De Barros FM, Martello F, Pereira IMDS, Da Silva JN, Ferraz KMPMDB, Naka LN, Dos Anjos L, Efe MA, Pizo MA, Pichorim M, Gonçalves MSS, Cordeiro PHC, Dias RA, Muylaert RDL, Rodrigues RC, Da Costa TVV, Cavarzere V, Tonetti VR, Silva WR, Jenkins CN, Galetti M, Ribeiro MC (2017) Atlantic birds: a data set of bird species from the Brazilian Atlantic Forest. Ecology 99(2):497

IGBE (2012) Manual Técnico da Vegetação Brasileira. IBGE, Rio de Janeiro, 2. ed. 275 p

IBGE (2017) A Geografia da Cana-de-açúcar: dinâmica territorial da produção agropecuária. Instituto Brasileiro de Geografia e Estatística. Available via www.ibge.gov.br/apps/dinamica_agropecuaria/

Joly CA, Metzger JP, Tabarelli M (2014) Experiences from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. New Phytol 204(3):459–473

Lapola DM, Martinelli LA, Peres CA, Ometto JPHB, Ferreira ME, Nobre CA, Aguiar APD, Bustamante MMC, Cardoso MF, Costa MH, Joly CA, Leite CC, Moutinho P, Sampaio G, Strassburg BBN, Vieira ICG (2014) Pervasive transition of the Brazilian land-use system. Nat Clim Chang 4:27–35

Lima PB (2016) Herbáceas da floresta Atlântica nordestina: regeneração natural em uma cronossequência de abandono agrícola e potencial invasor. Thesis, UFRPE, Botânica

Lima PB, Lima LF, Santos BA, Tabarelli M, Zickel CS (2015) Altered herb assemblages in fragments of the Brazilian Atlantic forest. Biol Conserv 191:588–595

Lôbo D, Leão T, Melo FPL, Santos AMM, Tabarelli M (2011) Forest fragmentation drives Atlantic forest of northeastern Brazil to biotic homogenization. Divers Distrib 17:287–296

Marques MCM, Swaine MD, Liebsch D (2011) Diversity distribution and floristic differentiation of the coastal lowland vegetation: implications for the conservation of the Brazilian Atlantic Forest. Biodivers Conserv 20:153–168

Medeiros PM, Almeida ALS, Silva TC, Albuquerque UP (2011) Pressure indicators of wood resource use in an Atlantic Forest Area, Northeastern Brazil. Environ Manag 47:410–424

Medeiros PM, Silva TC, Almeida ALS, Albuquerque UP (2012) Socio-economic predictors of domestic wood use in an Atlantic forest area (north-east Brazil): a tool for directing conservation efforts. Int J Sust Dev World 19(2):189–195

Melo A, Amorim B, García-González J, Souza JAN, Pessoa EM, Mendonça E, Chagas M, Alves-Araújo A, Alves M (2011) Updated floristic inventory of the angiosperms of the Usina São José, Igarassu, Pernambuco, Brazil. Rev Nordestina Biol 20(2):3–26

Melo DHA, Duarte M, Mielke OHH, Robbins RK, Freitas AVL (2019) Butterflies (Lepidoptera: Papilionoidea) of an urban park in northeastern Brazil. Biota Neotropica 19(1):e20180614

Melo FPL, Pinto SRR, Brancalion PHS, Castro PS, Rodrigues RR, Aronson J, Tabarelli M (2013) Priority setting for scaling-up tropical forest restoration projects: early lessons from the Atlantic Forest Restoration Pact. Environ Sci Pol 33:395–404

Mendes Pontes AR, Beltrão ACM, Normande IC, Malta AJR, APd SJ, Santos AMM (2016) Mass extinction and the disappearance of unknown mammal species: scenario and perspectives of a Biodiversity Hotspot's Hotspot. PLoS One 11(5):e0150887

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

Nascimento LM, Sampaio EVSB, Rodal MJN, Lins-e-Silva ACB (2014) Secondary succession in a fragmented Atlantic Forest landscape: evidence of structural and diversity convergence along a chronosequence. J Forest Res-JPN 19(6):501–513

Oliveira MA, Santos AMM, Tabarelli M (2008) Profound impoverishment of the large-tree stand in a hyper-fragmented landscape of the Atlantic forest. Forest Ecol Manag 256:1910–1917

Oliveira-Filho AT, Budke CJ, Jarenkow JA, Eisenlohr PV, Neves DRM (2015) Delving into the variations in tree species composition and richness across South American subtropical Atlantic and Pampean forests. J Plant Ecol 8:242–260

Oliveira-Filho AT, Fontes MAL (2000) Patterns of floristic differentiation among Atlantic forests in southeastern Brazil and the influence of climate. Biotropica 32:793–810

Pennington RT, Dick CW (2004) The role of immigrants in the assembly of the South American rainforest tree flora. Philos Trans Royal Soc B359:1611–1622

Pereira GC, Dantas SM, Silveira LF, Roda SA, Albano C, Sonntag FA, Leal S, Periquito MC, Malacco GB, Lees AC (2014) Status of the globally threatened forest birds of northeast Brazil. Pap Avulsos Zool 54(14):177–194

Pinto SRR, Mendes G, Santos AMM, Dantas M, Tabarelli M, Melo F (2010) Landscape attributes drive complex spatial microclimate configuration of Brazilian Atlantic forest fragments. Trop Conserv Sci 3(4):399–402

Pires FTG (2009) Engenhos de açúcar no Recôncavo baiano. R IHGB a.170(442):233–248

Produção Agrícola Municipal (PAM) 1975–2015 (2017) Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro. Available via www.ibge.gov.br/estatisticas/economicas/agricultura-e-pecuaria/9117-producao-agricola-municipal-culturas-temporarias-e-permanentes.html

Ranta P, Blom TOM, Niemelä J, Joensuu E, Siitonen M (1998) The fragmented Atlantic rain forest of Brazil: size, shape and distribution of forest fragments. Biodivers Conserv 7(3):385–403

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Strassburg BBN, Tabarelli M, Fonseca GA, Mittermeier RA (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:208–214

Ribeiro MC, Metzger JP, Martensen AC, Ponzoni FJ, Hirota MM (2009) The Brazilian Atlantic Forest: how much is left, and how is the remaining forest distributed? Implications for conservation. Biol Conserv 142(6):1141–1153

Rizzini CT (1997) Tratado de Fitogeografia do Brasil: aspectos ecológicos, sociológicos e florísticos. Âmbito Cultural Edições Ltda, Rio de Janeiro

Saiter FZ, Brown JL, Thomas WW, Oliveira-Filho AT, Carnaval AC (2016) Environmental correlates of floristic regions and plant turnover in the Atlantic Forest hotspot. J Biogeogr 43:2322–2331

Saiter FZ, Eisenlohr PV, França GS, Stehmann JR, Thomas WW, Oliveira-Filho AT (2015) Floristic units and their predictors unveiled in part of the Atlantic Forest hotspot: implications for conservation planning. An Acad Bras Ciênc 87:2031–2046

San Martín I, Ronquist F (2004) Southern hemisphere biogeography inferred by event-based models: plant versus animal patterns. Syst Biol 53:216–243

Santos BA, Peres CA, Oliveira MA, Grillo A, Alves-Costa CP, Tabarelli M (2008) Drastic erosion in functional attributes of tree assemblages in Atlantic forest fragments of northeastern Brazil. Biol Conserv 141:249–260

Silva FBR, Riché GR, Tonneau JP, Souza Neto NC, Brito LTL, Correia RC, Cavalcanti AC, Silva FHBB, Silva AB, Araújo Filho JC, Leite AP (1993) Zoneamento agroecológico do Nordeste: diagnóstico do quadro natural e agrossocioeconômico. EMBRAPA – CPATSA/CNPS, Petrolina

Silva HCH, Lins-e-Silva ACB, Gomes JS, Rodal MJN (2008) The effect of internal and external edges on vegetation physiognomy and structure in a remnant of Atlantic Lowland Rainforest in Brazil. Biorem Biodiv Bioavail 2(1):47–55

Silva JMCD, Casteleti CHM (2005) Status of the biodiversity of the Atlantic Forest of Brazil. In: Galindo-Leal C, Câmara IG (eds) The Atlantic Forest of South America: biodiversity status, threats, and outlook. Island Press, Washington, DC, pp 43–59

Silva WG, Metzger JP, Simões S, Simonetti C (2007) Relief influence on the spatial distribution of the Atlantic Forest cover on the Ibiúna Plateau, SP. Braz J Biol 67(3):403–411

Silveira LF, Olmos F, Long AJ (2003) Birds in Atlantic Forest fragments in north-east Brazil. Cotinga 20:32–46

Singh SP (1998) Chronic disturbance, a principal cause of environmental degradation in developing countries. Environ Conserv 25(1):1–2

Thomas WW (2008) Introduction and acknowledgment. In: Thomas WW (ed) The Atlantic Coastal Forest of Northeastern Brazil. The New York Botanical Garden Press, New York, pp 1–5

Thomas WW, Barbosa MRV (2008) Natural vegetation types in the Atlantic Coastal Forest in Northeastern Brazil. In: Thomas WW (ed) The Atlantic Coastal Forest of Northeastern Brazil. The New York Botanical Garden Press, New York, pp 6–20

Thomé MTC, Zamudio KR, Haddad C, Alexandrino J (2014) Barriers, rather than refugia, underlie the origin of diversity in toads endemic to the Brazilian Atlantic Forest. Mol Ecol 23:6152–6164

Tilman D, May RM, Lehman CL, Nowak MA (1994) Habitat destruction and the extinction debt. Nature 371:65–66

Trindade MB, Lins-e-Silva ACB, Silva HP, Figueira SB, Schessl M (2008) Fragmentation of the Atlantic rainforest in the Northern coastal region of Pernambuco, Brazil: recent changes and implications for conservation. Biorem Biodiv Bioavail 2(1):5–13

# Chapter 4

# The Hileia Baiana: An Assessment of Natural and Historical Aspects of the Land Use and Degradation of the Central Corridor of the Brazilian Atlantic Forest

**Deborah Faria, Jacques Hubert Charles Delabie, and Marcelo Henrique Dias**

**Abstract** The exuberant, pristine forest originally occupying an area stretching along today's southern Bahia to the northern portion of Espírito Santo states was coined as the Hileia Baiana and represents a unique and highly diverse biota that, in addition to high levels of endemics, suffered the influence of past connections to the Amazonian biome. Herein we made an effort to present the key social, economic, and historical aspects that ultimately determined the current land use of this region. From the first colonial territorial division that encompasses two hereditary captaincies, Ilhéus in the north and Porto Seguro in the south, the occupation process was fomented by the exploration of the brazilwood and the sugarcane plantation. Unlike the northeastern territories encompassing the Recôncavo and Zona da Mata, where sugar mills multiplied between the sixteenth and seventeenth centuries, in Ilhéus and Porto Seguro, the production declined. In this territory, the production of cassava flour, the food basis of the colonial population, was developed, being more vigorous in the captaincy of Ilhéus due to the greater proximity of the consumer markets of Bahia and Pernambuco. At the same time, the presence of large stocks of valuable timber encouraged the exploitation of such strategic resource to the metropolitan naval industry. In the nineteenth century, the favorable foreign market promoted a new sugarcane cycle and the introduction of new export crops, coffee, and cacao. In Ilhéus, cacao plantations expanded in the second half of the nineteenth century, making the old headquarters of the captaincy the nucleus of a producing area that, in the twentieth century, also extended to territories of the former captaincy of Porto Seguro. Along the nineteenth and the twentieth centuries, Ilhéus – in the south of what is now the state of Bahia – consolidated itself as a major cacao-producing region, while the occupation and the strengthening of the economy of the two regions earlier encompassing the Porto Seguro captaincy (extreme south of Bahia and north of Espírito Santo) were only solidified during the twentieth century.

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Five hundred years of colonization and successive economic cycles significantly reduced and degraded the native vegetation, but these forests are still significant reservoirs of the regional biota that inhabits the few remaining forests – legally protected or not – and the complex mosaic of different land uses that comprise the dominant human-modified landscapes. What is left still faces several threats regarding deforestation, fragmentation, and chronic disturbances that, if not adequately curbed, will probably jeopardize the long-term conservation of such important natural assets.

**Keywords** Hileia Baiana · Atlantic Forest · Cacao plantation · Logging · Biodiversity · Ilhéus · Porto Seguro · Espírito Santo

> The mass of vegetation accumulates around the visitor with an impenetrable strength to the rays of the sun, pouring with force from their own entrails, multiplying in ever new vaulted compartments, closing finally over our heads in a shaded and compact cover, woven by vines interlaced and supported by climbing plants; the eye is no more able to recognize where the plants begin or finish; the place from where the roots sprout fall hidden in a mass of plants of different species that branch out in front of us... (von Habsburg, 1867)

## 4.1 The Natural Features of the Hileia Baiana

Although rarely used in scientific literature in the last decades, the expression “Hileia Baiana” was applied by Andrade-Lima (1966) to describe the resemblance of the lowland forests located in the south of Bahia and north of Espírito Santo states to the exuberant Amazonian forests or “Hileia Amazônica,” a term coined by the German naturalist Alexander von Humboldt (Magalhães and Maio 2007). It corresponds to a rather well-defined region of the coastal Atlantic Forest of northeastern Brazil, agreeing to a region of what Carnaval and Moritz (2008) and Martins (2011) called the “Bahia refuge,” a stretch of the Atlantic Forest remarkably stable during the Pleistocene and Holocene periods. It corresponds too to the “Bahia endemism area,” one of the five main endemism areas of the Brazilian Atlantic Forest described by da Silva and Casteleti (2003).

The regional geomorphology is composed of Quaternary sands deposed along the littoral, the coastal “tabuleiros” on the Barreiras Formation of Oligocene sediments and inland, and an alternation of hills and valleys with several mountains of medium altitude (up to 1000 m) formed between 300 and 200 million years ago and roughly parallel to the littoral (Barbosa et al. 2003). The climate is wet tropical of the Af climatic region in Köppen’s classification (1936), with average monthly temperatures ranging between 20 and 25 °C, locally characterized by a strong gradient of rainfalls from around 1600 to 2000 mm along the coast to less than half of that inland. The main vegetation formations, from east to west, occupy a narrow zone up to 50 km wide within the coastal forest belt and vary in composition depending on elevation, soils, and drainage (Thomas et al. 1998). According to Thomas and

Barbosa (2008), natural formation encompasses a complex mosaic of physiognomies including (1) the "open restinga," comprising open, savanna-like formation growing on sandy soils where cashew (*Anacardium occidentale* L.) is one of the main characteristic trees; (2) the "restinga forest," which is a shrubby woodland of small trees with a closed canopy where one of the most characteristic species in the Hileia Baiana is the piassava palm (*Attalea funifera* Mart.); (3) the "coastal tabuleiros," physiognomies that shelter two types of vegetation living on moister clay soils: the open tabuleiros, sometimes called "campos nativos," which form open clearings within tabuleiro forest, and the tabuleiro forest itself; (5) the "tropical moist forest," comprising the main vegetal formation of the Hileia Baiana, with the "tropical lowland moist forest" or "mata higrófila sul-baiana," which grows at low elevations (until 100 m); (6) the "submontane tropical moist forest," a formation occurring up to 600 m elevation; and (7) the "submontane and montane deciduous seasonal forests" known as "mata de cipó" or liana forest in southwestern Bahia, which are usually distributed in areas with 800–1000 mm of orographic-driven rainfalls per year well distributed in dry and rainy seasons.

The coastal zone was submitted to alternative sea-level changes during the Quaternary (Suguio et al. 2005), according to the climatic changes. Dominguez et al. (1982) reconstituted the detailed history of alluvial and sand deposition according to the Quaternary sea regressions and transgressions along the coast of several regions of Brazil, responsible for the formation of the estuary of the Jequitinhonha River and many portions of the coast of Bahia. The climatic changes and a range of biotic and geographic evidences originated the Theory of Refuges of Quaternary (Haffer 1969) responsible for a fragmentation of the forest by open dry formations such the savannas (Vanzolini and Williams 1981; Martins 2011), accounting itself for the increase in the species richness of the Atlantic Forest by a succession of vicarious processes. Evidence shows the occurrence of two active forest corridors, which are discontinuous in time between the Amazon Forest and the Atlantic Forest since Oligocene but more intensive during Pleistocene (Batalha-Filho et al. 2013; Ledo and Colli 2017).

The Hileia Baiana shelters one of the five centers of endemism along the Atlantic Forest biome (Silva and Casteleti 2003), harboring one of the most diverse areas for plants and animals in the world. For plant species, the region represents one of the richest spots (Martini et al. 2007). Systematic, long-term inventories from a single reserve in Espírito Santo report that the area contains ~17% of the angiosperm species estimated for the entire biome (Peixoto et al. 2008). In a single forest hectare, Thomaz and Monteiro (1997) detected 443 tree species in Espírito Santo, while Thomas et al. (1998) reported 454 in a forest fragment in Bahia, which was near to the forests in which Martini et al. (2007) surveyed a striking number of 144 species in plots of only 0.1 ha. Endemism rates are between 18.9% and 28.1% of local flora (Thomas et al. 1998; Amorim et al. 2008), and Euphorbiaceae, Lauraceae, Sapotaceae, and Myrtaceae are especially the most diverse plant families, including many endemic species (Amorim et al. 2008). A recent revision encompassing the north of Espírito Santo and most of the coastal forests in Bahia state highlighted the uniqueness of this flora, identifying 547 endemic tree species from 69 families, with

7 endemic genera (Ostroski et al. 2018). Populations of a range of trees are especially interesting for regional history, landscape, and conservation, such as the brazilwood [*Paubrasilia echinata* (Lam.) Gagnon, H.C. Lima & G.P. Lewis], the jacarandá-da-Bahia or Brazilian rosewood [*Dalbergia nigra* (Vell.) Allemão ex. Benth.], the juçara palm [*Euterpe edulis* (Mart.)], and the jequitibá (*Cariniana* spp.), species until today representing valuable economic resources like food, timber, and other uses (Dean 1995; Mattos-Silva et al. 2008).

Similar levels of speciose and endemism can be reported for the fauna, including vertebrates and invertebrates. In the "Hileia Baiana," 59 bat species are known from inventories carried out in the south of Bahia state (Faria et al. 2006b) and 12 primate species (Pinto 1994). For birds, Laps (2006) reported 333 species, including 17 species threatened, in Una Biological Reserve and its surrounding areas (Bahia), while Srbek-Araujo et al. (2014) registered 391 species – 44% of the bird species occurring within the Atlantic Forest – from ~30 years of inventories in a single protected area in Espírito Santo state. Sixty-one species of snakes were found in the single cacao-producing region, with 19 of them endemic of the Atlantic Forest (Argolo 2004). There is only partial data about the invertebrate diversity of the Hileia Baiana, particularly on the cacao-producing region, but all the available information suggests that the fauna is unique and rich, at least for the major components of animal biomass such as springtails, spiders, mites, termites, and ants but also for onychophores, isopods, or gastropods (Delabie et al. 2017). For example, at least 250 species of spiders are known for the Hileia Baiana (Delabie et al. 2017), as much as 429 ant species are known for the whole strata of the vegetation only for the single municipality of Ilhéus with 1.712 $km^2$ (Delabie et al. 1998) and 391 ant species living only on the ground were sampled in 11 landscapes of south of Bahia (Santos et al. 2017). In this region, the several types of cacao agroforests are responsible for the conservation of a large proportion of native species (Delabie et al. 2007; Cassano et al. 2014), among them many endemic ones, including some threatened by extinction. It is also the case near Porto Seguro where the monotypic endemic ant genus *Diaphoromyrma* survives in a couple of forest fragments completely surrounded by an ocean of eucalyptus plantations (Fernández et al. 2009).

## 4.2 Forests and Territory: Land-Use Changes along the Colonial Period

When the Portuguese arrived in Brazil, on April 22, 1500, the region encompassing the Hileia Baiana was inhabited by Amerindians from different ethnic groups. The brief meeting of the old and new world civilizations was described with details in a famous letter written by the clerk Pero Vaz de Caminha to King D. Manuel, a moment that sealed the destiny of the territory in general and its natural resources in particular. From the beginning, it was clear that the lavish and pristine coastal forests harbored a rich stock of valuable woods, the most precious being the brazilwood, a tree prized as a natural red die. Indeed, its exploitation yielded a profitable trade to the crown due to its high quality and local abundance. Since the beginning of the sixteenth century and for a long time, brazilwood harvesting was the main economic activity carried out by the Portuguese in Brazil. This huge richness has also attracted the attention of other nations eager to have their share of the profitable resources on the new, unprotected territory, forcing the once reluctant Portugal to take possession of the land.

Nevertheless, the continental scale of the new territory severely limited the capacity of the crown to occupy the land. The strategy adopted by the King D. João III was to divide the land into large territories, starting from the coast until the western limit of the territory, the so-called Hereditary Captaincies. Each captaincy was granted to a donee or a private recipient representing well-provided entrepreneurs that could explore the territory and its resources in exchange for setting the infrastructure, rules, and share of the profits with the crown. Between 1534 and 1566, the crown distributed 14 Hereditary Captaincies, 2 of which generally comprising the north and south limits of the Hileia Baiana, the captaincies of São Jorge dos Ilhéus and Porto Seguro, respectively (Fig. 4.1).

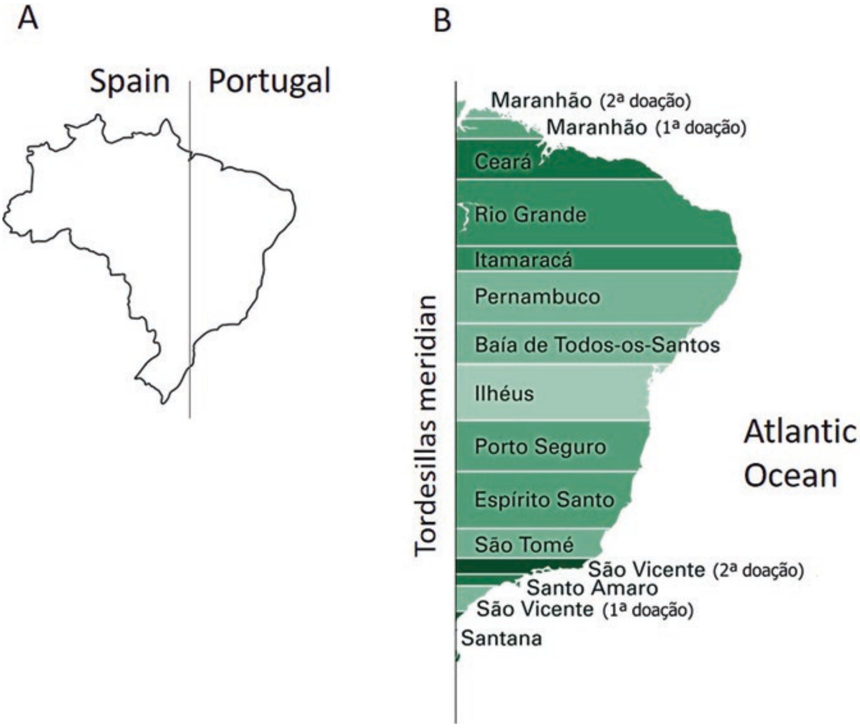

**Fig. 4.1** Representation of the (**a**) territorial division of Tordesillas treaty, the imaginary line (black bar) separating western lands under Spain dominion while the east side comprised the Portuguese America, (**b**) in which each hereditary captaincy of the XVI century is depicted by different shades of green

As depicted in Fig. 4.2, which makes reference of main localities cited in this chapter, the territory corresponding to both captaincies extended, roughly, from the current municipality of Valença in the north, which divided the captaincy of Bahia with that of Ilhéus, until the mouth of the Doce River, which delimited the captaincies of Porto Seguro and Espírito Santo. The Jequitinhonha River, in turn, circumscribed the boundaries between the two. From the second half of the nineteenth century, the territories would be integrated with that of Bahia, forming a single captaincy, which also includes the north of the Doce River. The territories of Ilhéus and Porto Seguro would yet remain as a circumscription of the homonymous counties created in the 1760s, which lasted until the first half of the nineteenth century, when both became cities.

The beneficiaries of the first captaincies were families intimately connected with conquest and overseas business, well aware of the economic potential of the lands they received from the King. For instance, Jorge de Figueiredo Corrêa, the donee of the Ilhéus captaincy, was not a conqueror but rather an active participant in the enterprises in charge of reporting every commercial transaction overseas. This certainly assured him privileged information for choosing his fair share of the Brazilian coast (Coelho Filho 2000). His allotment was replete on valuable woods, including large stocks of brazilwood. The territory was well served by a hydrographic network and natural harbors, making the region convenient for the flourishing enterprise of sugar mills. On the other hand, the donee of the neighbor Porto Seguro captaincy, Pero de Campos Tourinho, came to Brazil with his family and, for more than a decade, traded brazilwood, before engaging in the promotion of sugar production, which was also favored by the geography of his allotment (Souza 1939).

In addition to the rich forest, fertile soil, and a convenient hydrographic system, the coast comprising both captaincies harbored indigenous populations of the Tupi ethnicity. From the bay of Camamu in the north to the Doce River in the south most tip of this land, the Tupiniquim ethnic group predominated, forming an area densely occupied by villages. They became the main allies of the Portuguese settlers, but such a relationship could not guarantee the hegemony of colonizers in the deal with the natives. Indeed, the French traffickers rivaled with the Portuguese and even coopted Lusitanian authorities and settlers for their interests in the acquisition of brazilwood. Thus, the exploitation of such precious dye, which guided the first actions of contact and colonization in the territories of the captaincies of Ilhéus and Porto Seguro, eventually became a source of conflict (Dias 2016).

For the indigenous populations occupying the coast, the alliances with Europeans guaranteed, at first, their strengthening against the traditional internal enemies. However, throughout the sixteenth century, the primitive barter system was decisive to the dismantling of the Tupi villages, despite preserving the communitarian character of those societies. With the intensification of resource exploitation in the period of the captaincies, competition for indigenous labor put more and more tools and weapons into the hands of natives, eventually saturating the junk market. In addition to broadening its bargaining power, trade has changed the economy of the native populations by facilitating such tasks as clearing the forest, hunting, and producing canoes. Relieved at these activities, they were able to invest more time

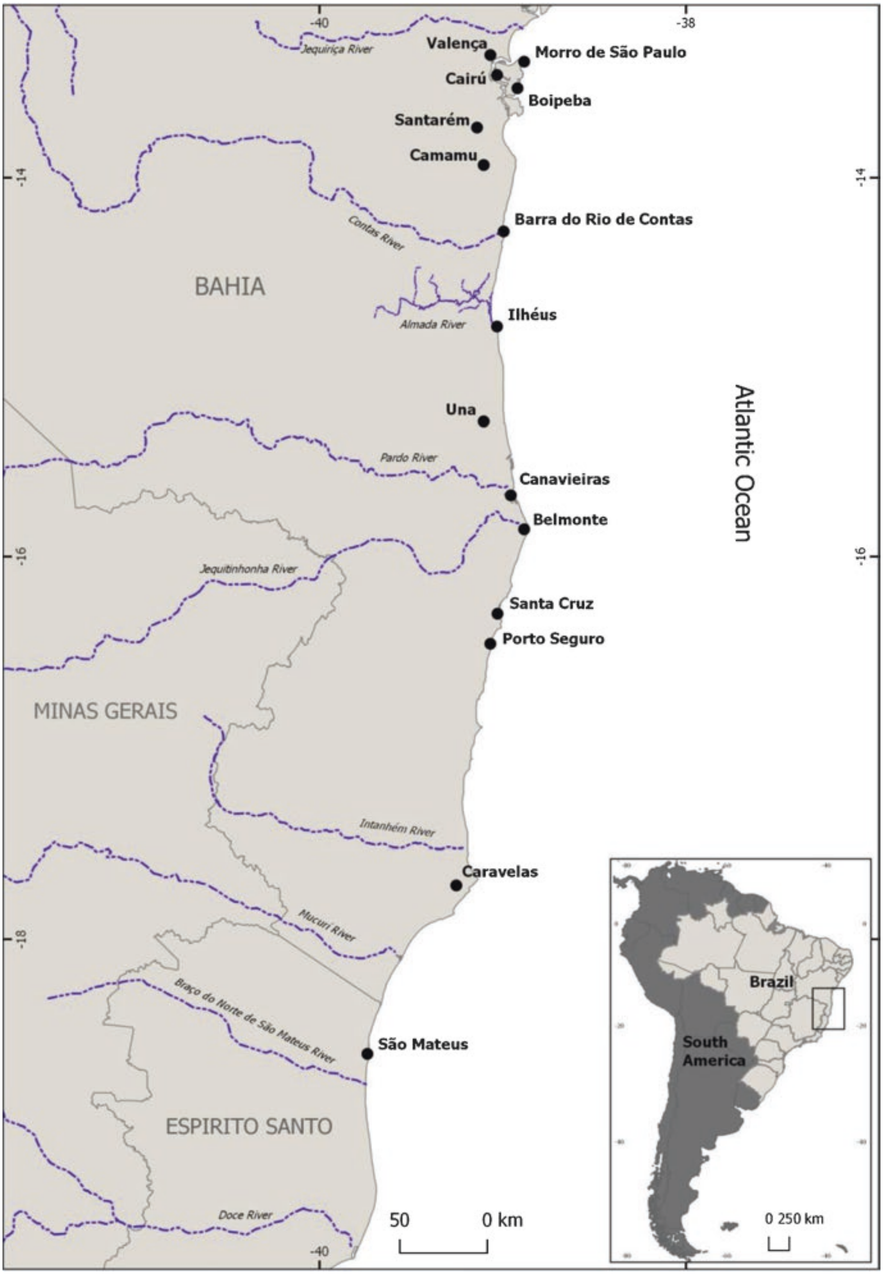

**Fig. 4.2** Depiction of the north-south limits of the Hileia Baiana, highlighting the main rivers and the original name (until the nineteenth century) of the main localities cited in the text

focusing on intertribal wars, fueling the slave-rescue market, while de-structuring villages (Schwartz 1988).

The establishment of the first sugar mills in the 1540s would intensify the interdependence between Portuguese and natives. The situation of the latter worsened by the epidemics of smallpox and measles that, in the 1560s, swept the coast from which today lies the states of Espírito Santo and Bahia. The Jesuits, the Catholic order with the official monopoly upon the indigenous labor, reported the demise of nearly two-thirds of their village Amerindians during these outbreaks (Campos 2006). More and more, the colonists began to practice the "jumps", that is, incursions to the inner lands (sertões) for pursuing new slaves. Thus, the barter system was compromised by the disarrangement of the indigenous communities and by the low population. The emptiness left by the decreasing of the Tupi population attracted hordes of other indigenous, fierce tribes from the interior, identified as Aimorés. The attacks directed at the Portuguese settlements seriously compromised the existence of the incipient urban centers and the functioning of the sugar mills.

The Portuguese Crown concentrated its investments in the defense and administration of the capital city of Salvador and its surroundings (herein Recôncavo), attracting the people and the assets capable of setting up sugar mills. At the same time, Ilhéus and Porto Seguro donees, dependent on private investment, were unable to cope with the obstacles represented by the lack of labor, by the incursions of the Amerindians, and by the conflicts between residents and natives. In the final years of the sixteenth century, the number of mills drastically reduced in these last captaincies. In Porto Seguro, which had five sugar mills in 1570, only one remained in 1612 (Carrara 2007). The danger of the Aimoré besetment in the forests neighboring the Portuguese settlements also paralyzed the brazilwood business. Peace was only observed between 1602 and 1603, yet another difficulty, this time in the external sphere, further prevented the development of colonization in those villages: the war imposed by the Dutch corsairs. From the first decade of the seventeenth century, foreign fleets significantly limited the transit of Portuguese vessels along the coast of Bahia. Such high risk hindered sugar production from crossing the Atlantic Ocean, resulting in many mills quitting production. For instance, in the village of São Jorge dos Ilhéus, of the eight existing mills in the 1570s, only half were operating in 1629 (Dias 2011). From 1649, Portugal imposed the so-called fleet system for the transatlantic trade, which requires that all ships leaving the country should first gather on the main ports (e.g., Salvador) and then sail in convoys. Such rule determined the closing of the last mills in that village, except Santana mill which belonged to the Jesuits. The survival of this unit was only possible by the Jesuit strategy of keeping in parallel to the manufacture of sugar a food production, such as cassava flour, rice, beans, and fish, destined for the domestic market. Santana mill, as defined by the priest Felipe Franco who managed the business between 1656 and 1674, was an exception as a production unit of sugarcane plantation and farm.[1]

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[1] Franco F (1659) Carta do Pe. Felipe Franco para o Pe. Reitor, Ilhéus, 29 de setembro de 1659. Arquivo Nacional da Torre do Tombo (Lisboa), Fundo Cartório dos Jesuítas, Maço 68, n. 306.

The historiography of the last 40 years has shown that self-sustaining sugar mills in colonial Brazil are real exceptions (Linhares and Silva 1981; Barickman 2003; Amantino 2018). The mercantile production of food began as soon as the first colonial settlements were established. With the process of concentration of sugar mills in the northern captaincy of Bahia and the increasing substitution of indigenous labor for that of the enslaved Africans, the production of food in that zone diminished, while the demand increased. On the coast of Ilhéus, a captaincy increasingly isolated from the export economy, settlers of modest conditions were established, unable to claim land in the dynamic sugar zone and thus participate directly in that business. As regular or irregular tenants, or simply landowners with no land titles, they began to produce cassava flour, the main staple food. Such situation contributed to consolidating the colonial settlement on the banks of the navigable rivers that orbited not only on the village of Ilhéus but also in other urban nuclei founded between the end of the sixteenth and the beginning of the seventeenth century (i.e., Cairu, Boipeba Island, and Camamu). The geography of these territories articulated a network of rivers, bays, and natural harbors, promoting a very convenient connection with Salvador and its surroundings.

During the Dutch domination in the northern captaincies, the militias hired to defend the country were concentrated in Salvador, extending significantly the number of mouths to be fed. For this reason, the colonial administration adopted a policy of concentrating the cassava flour production in the Ilhéus captaincy. As unbelievable as it might sound, a series of top-down, draconian measures were imposed on the captaincy. This included preventing the production of export crops, such as sugarcane and tobacco, under penalty of imprisonment of the offenders; regulating the amount of flour that each village should send directly to the central government in Salvador, according to its corresponding productive capacity; appointing local agents and commissioners of the capital to carry out the trade; and establishing a maximum price per bushel (~36.3 liters) to be paid to producers, always below the potential market value. These measures comprised the so-called "conchavo das farinhas"[2] or "collusion of flour", which was in force until 1738. Consequently, the so-called lower villages consolidated their agrarian profile of the internal supply zone for the central government (Dias 2011).

The long period in which such measures lasted, far beyond the end of the conflicts with the Dutch, points to the structural dependence established between the dynamic zone of sugar production – and the overseas trade – and the area designated for food production. Interestingly, a regional market was established that allowed the participation of small producers in the slave system. For most farmers, however, the advantage of this ever-increasing demand for the flour they produced did not

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[2] Another possible English translation for the Portuguese term "conchavo" would be "incumbency" as, in this case, it refers to certain obligation to be fulfilled by some agreed commitment. After agreeing among themselves, the authorities from the administrative capital of Salvador legally forced flour producers from the villages of Cairu, Camamu, and Boipeba to produce and sell fixed quantities of flour, under fixed prices, to Salvador. Only after meeting this demand could producers sell to other markets.

reinforce the expansion of their productive forces, given the limitation represented by the pricing established under the "collusion of flour." Most farmers were limited to producing more due to a small number of slaves, on average two or three per unit.[3] Some peasants were even more restricted, counting only with the family labor. Under these meager conditions, flour producers were also unable to expand their cultivating area, as labor scarcity prevented new clearings in the forest. These producers were left with the early-growth forests (*capoeiras*), thus submitted to a true regime of rotation of fragments of forest, in different levels of regeneration.

On average, productive units in Ilhéus village occupied 220 m of riverfront, and in most cases, property depths were not even measured given that farmers did not have the capacity to exploit the forest beyond the *capoeiras* along the river.[4] Therefore, farms were usually elongated, with lands beyond 3 km from the river usually untouched by axes and fire. This pattern was repeated in other places of the Ilhéus captaincy. Under these conditions, the overall increase in production was only possible with the incorporation of new producers and the expansion of the plantations along the margins of navigable rivers of the coastal zone, as was the case of the Contas River, the northern limit of the captaincy. At the end of the eighteenth century, its banks were already occupied at a distance of 42 km (seven leagues) from the mouth of Contas River, and the village of Barra do Rio de Contas (today the Itacaré city) was the second largest producer of flour in the Ilhéus captaincy, behind only of Camamu village.[5]

The expansion of subsistence agriculture, required to prevent supplying crises in the capital and the Recôncavo, nevertheless implied the destruction of the timber resources so valuable to the royal fleet. Ilhéus authorities, like the squire Francisco Nunes da Costa and Baltasar da Silva Lisboa, were emphatic in denouncing this situation. The latter was appointed "Judge Conservator" of the Ilhéus forests, one of the first positions specifically created by the Brazilian bureaucracy to deal with conservation of natural resources (Padua 2002). He was given the task of implementing a conservation plan that included the withdrawal of farmers from their lands, from which they had no legitimate title. The flour producers could plant only in the *capoeiras*. The opening of new clearings in the forest was conditional on a procedure to save the valuable, hard-wooded timbers. Carpenters were employed to make the inspection and mark the trees that should be cut before burning. The reaction of peasants and local potentate farmers, some of them very influential in the high sphere of colonial rule, reversed the conservation plan as the Judge Lisboa lacked the political strength required to continue his actions (Morton 1978).

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[3] Relação da planta da mandioca da Vila de Cairu, 1786. Biblioteca Nacional do Rio de Janeiro, Seção de Manuscritos. Ms. 512 (34, doc. 31).

[4] TOMBO das terras pertencentes à S.M. desde o rio Aqui até Canavieiras; Auto ou tombo feito em uma e outra margem do rio Taípe... das terras de Mapendipe da Comarca de Ilhéus (1798–1799). Biblioteca Nacional do Rio de Janeiro, Seção de Manuscritos. Ms. 512 (53), doc. 42; I, 31, 21, 35.

[5] Lisboa BS (1808). Memória acerca da abertura de uma estrada pela costa desde a V. de Valença até o Rio Doce apresentada ao Príncipe Regente por Baltazar da Silva Lisboa em 1808. Biblioteca Nacional do Rio de Janeiro, Seção de Manuscritos. Ms. 512 (58, doc.52).

The best performing zone was located in the northern territories of the captaincy of Ilhéus, which in the course of the eighteenth century combined the traditional flour production with timber exploitation. From the beginning of the previous century, logging for naval and civil construction was carried out as an economic activity in the lands exploited by the Jesuits. However, in the second decade of the eighteenth century, the crown began its participation in the timber business by appointing local administrators to carry out the tasks of cutting, transport, and trade. Timber was destined for the royal shipyards of Lisbon, but a significant part of the logs was used in smaller shipyards, spread by the ports that served to the hinterland of flour, between the captaincy of Ilhéus and Bahia (Miller 2000; Dias 2011).

The presence of forests with large reserves of hard-wooded timber, suited for the construction sector such as the vinhático (*Plathymenia foliolosa* Benth.), canarywood (*Centrolobium* spp.), sucupira (e.g., *Bowdichia virgilioides* Kunth), and angelim [e.g., *Andira anthelmia* (Vell.) Benth., *A. fraxinifolia* Benth.], and the presence of rivers facilitating the flow of wooden pieces were determining factors in the choice of the sites to be explored. In 1725, the logging units were initially composed by two cutting mills, or simply “cuts”: one at the site called Maricoabo (currently in the municipality of Valença) and another at Taperoá. Later, as the forests closest to the best ports were overexploited, other cuts appeared on the banks of the Mapendipe, Una, and Jequié rivers. In the navigable limit of the rivers, the port-yard was established, from where the wood already benefited was transported in small boats, until the port of the bar. From this point forward, the conveyance was made by larger boats, as in the case of plows and sums (Dias 2011). It is also worth mentioning that the timber sector in Ilhéus was not limited to the extraction and primary processing of the woods. It involved a more sophisticated industry supplying products of higher aggregate values such as finished goods for shipbuilding (Dias 2010).

Following the trails opened by chariot convoys used in the inland transport of the woods, new squatters were established, with their fields and slaves. At the end of the eighteenth century, communications between vicinal ports, cutting areas, and incipient settlements were well established. From a zone in which flour production was traditionally carried out with vitality, logging activity expanded significantly. Despite the conflicts between the authorities responsible for the preservation of the wood resources and the flour producers who had to burn down the forest for farming (Morton 1978), soon a network of small urban nuclei formed the so-called Royal Road (see in Fig. 4.2).

In the Porto Seguro captaincy, which was further away from the economic center represented by Salvador, the domestic supply of food and the logging activity remained, however, on a much smaller scale than in the Ilhéus captaincy, then a main supplier of goods to the capital. The exploitation of brazilwood remained the economic activity that most mobilized capital and labor during the seventeenth and eighteenth centuries. Only in the last decades of the 1970s, due to the effect of the administrative, economic, and urban reforms carried out in Portugal, Porto Seguro region experienced the economic growth fueled by both agriculture and logging. After the crown changed the capital from Salvador to Rio de Janeiro (1763), Porto Seguro suddenly became closer to the consuming center. The increased demand for subsistence products and services by the new capital triggered an unprecedented demographic growth. For instance, the village of Caravelas, located south of Porto Seguro, went from 360 inhabitants in 1764 to 2300 in 1817. For the same reason, the town of São Mateus (today north of the state of Espírito Santo) jumped from 345 to 3000 inhabitants in the same period.

As expected, the migration of the political and economic center slowed the occupation rate in the territories of the north of Porto Seguro and in the southern territories of the Ilhéus captaincy. In the last years of the eighteenth century, the forests bordering the delta of Pardo and Jequitinhonha rivers still appeared as immense reserves of what was considered the best brazilwood stocks in the Portuguese dominions in America, as well as other species of wood with a wide use in naval and civil construction, such as the tapinhoã (*Mezilaurus navalium* (Allemão) Taub. ex Mez, Lauraceae) and the Brazilian rosewood (*Dalbergia nigra* (Vell.) Fr.All. ex Benth, Fabaceae). This forested region served as a refuge for the indigenous migrant populations from the backlands, such as the Camacã and Pataxó Amerindian groups. However, in the last years of that century, the advance of colonization put these groups in contact with small farmers and loggers. Many of these Amerindians, especially the Camacãs, were employed in the timber industry, inhabiting the village of Belmonte and other small settlements in the Pardo River estuary, where the village of Canavieiras began to flourish. The result of these contacts was very similar to that observed a little more than two centuries earlier with the Tupi Amerindians: their near extinction by the contamination of biological agents against which they had no immunity. This seems to have been the fate of many Camacãs who made alliances with the Portuguese to protect themselves from their Botocudo and Pataxó enemies. In Mogiquiçaba, where they had been settled around 1760, they were gone when German Prince Maximilian de Wied-Neuwied visited the region in 1815 (Wied 1989: 212–227). Instead of indigenous people, there were 18 Africans employed in crafting ropes from the piassava palm tree (*Attalea funifera* Mart., Arecaceae). In Belmonte, in turn, only a few Amerindians joined the 600 inhabitants who produced flour, corn, sugarcane spirits (cachaça), and hardwood. Others, "fleeing from the mosquitos," had migrated to the village of Canavieiras to work in sawmills in 1799. In 1815, however, the scientist Prince did not discriminate them among the white and brown population.

Again, the decimation of the main "human energy converters" of that forest economy prevented the intensification of the ax's work. Employing more valuable assets in these unsafe forests, such as African slaves, was too risky. In addition, these alien captives did not have the knowledge and skills of the natives to select, cut, chop, and transport the timber. In view of the low technical and demographic level employed in both selective extractivism and slash-and-burn agriculture, the forest had the opportunity to regenerate itself, especially those species that, after being cut, sprout from the stumps, such as brazilwood. Indeed, until the first half of the nineteenth centuries, deforestation probably did not exceed 8% of the entire biome (Cabral 2014). This resilience prevented the extinction of even this most commercially endangered tree species, whose populations, although weakened, were able to remain viable throughout the colonial period and even in the following century. In a more general context, it can be said that the Atlantic Forest of southern Bahia obstinately resisted the attacks of that rustic extractive economy and domestic supply. The greatest and most evident losses from this period were in human lives and cultures, not in forests.

Regarding the demographic expansion, the counties of Ilhéus and Porto Seguro had the lowest population in Bahia at the end of the eighteenth century. Even in areas where the colonial economy expanded more vigorously, the urban network was limited to no more than 20 km bordering the coast. In addition, this was only in a few strategic points where private and clerical agents and authorities tried to conquest and expand territorial domination against the resistant groups of Amerindians who insisted on remaining in the forest. The impulse for this new venture came from the foreign market, which had extended the demand for Brazilian sugar after the destabilization of the Antillean production. In the territories of the old captaincies of Ilhéus and Porto Seguro, there were many forested lands available, and the settlement remained rarefied, unlike the old sugar zone of the Recôncavo, on the surroundings of the Bahia captaincy.

In the year 1808, a pivotal event occurred, marking the destiny of the entire country. Fleeing from the Napoleonic invasion, the Portuguese court moved from Lisbon to Rio de Janeiro. An immediate consequence was the opening of the ports, making the demand for the lands of southern Bahia to increase. There was also an impulse of occupation and territorial domination from the hinterland, where, at that time, considerable cattle production was already taking place. It was necessary, therefore, to descend the troops to the coast, in demand of the ports and the markets constituted there. Indigenous people were placed in strategic locations with the purpose of establishing contacts with the still autonomous groups and attracting them to the new settlements.

## 4.3 Three Forest Regions and Their Distinctive History and Land-Use Dynamics

From the last decade of the colonial period, the territory along the Hileia Baiana remained relatively low populated, which somehow explained the maintenance of massive forest stands. Indeed, in the nineteenth century, no more than 30,000 people inhabited the former captaincies of Porto Seguro and Ilhéus, most of them concentrated in a few coastal villages. The Amerindians occupied more remote stands (Cabral 2014). However, the development of the agricultural frontiers defined differences in the land-use dynamic and development within the territory. Herein we will describe key aspects and differences of the land use of three distinct regions, today comprising the southern Bahia and the northern tip of Espírito Santo states. Within Bahia state, the dynamic of the forest was different in two stretches, one between the Contas and Jequitinhonha rivers and the second extending from this north limit to the Mucuri River on the south. These regions are coined here as the southern and extreme south of Bahia and north of Doce River, respectively.

### *4.3.1 The Forests of Southern Bahia*

In addition to the new sugar mills, other export crops contributed to the expansion of the agricultural frontier in the territory of the former Ilhéus captaincy in the early years of the nineteenth century, with emphasis on coffee and cacao (Mahony 1996). As the first was gradually abandoned, most of the territory of the Hileia Baiana experienced an economic stagnation for most of the nineteenth century, particularly the region covering the province of Porto Seguro and north of Doce River. By contrast, the region encompassing the Ilhéus province was marked by a single cash crop, the cacao tree (*Theobroma cacao* L., Malvaceae). Although the birthplace of the species is still a matter of controversy, it probably has its origin in the upper Amazon region and then spread to Central and other parts of South America (Motamayor et al. 2008). It is also debatable the date and place of the introduction of cacao cocoa in Bahia. The most popular version accounts that the crop was first grown in 1746 in what today is the Canavieiras municipality after the botanist Louis Frédéric Warneaux sent the first seeds from the state of Pará, where the crop was cultivated since 1679 (Andrade 2003). Nevertheless, the earliest official report comes from the naturalist Manuel Ferreira da Câmara who experimentally planted cacao in the 1780s on his farm in Camamu village (Câmara 1789). Despite such dispute, during the next two and a half centuries, its cultivation contributed to forging the regional culture (Rocha 2008).

The first plantations started along with the fertile and more humid soils near the rivers around Ilhéus. From the Almada and Cachoeira River basins, the cacao crop spread inland and to the north. At first, cocoa was grown on the small-scale family unit, *following the classic slash-and-burn* agricultural practice (Ruf and Schroth 2004; Piasentin and Saito 2014). According to Bondar (1939), after cleaning and burning the native vegetation, seeds were planted together with annual food crops such as cassava and corn, which provided shade for seedlings. After the harvesting of the staple food, cacao shrubs were eventually shaded by the spontaneous growth of pioneer tree species for 7–10 years, after which the shade was removed and cacao was kept at the full sun (Piasentin and Saito 2014). The crop was labor intensive, requiring many hands not only to clean the forest and establish the planting but requiring a constant workforce for the digging, pruning, and thinning for controlling the shade. Usually, such labor was conducted by diarist Amerindians (mainly Camacã ethnic group) and by African slaves.

Due to favorable climatic conditions and being free from its native pests, from 1810 to 1910, cacao production experienced a striking expansion in the region. In the middle of the nineteenth century, when sugar production was in crisis due to increased foreign competition and lower prices, cacao was already the main product exported from Ilhéus. In 1867, more than 31,000 arrobas of cacao (~455 tons) were exported at a price/ton of ~2.5 times higher than the 10,000 arrobas (~150 tons) of sugar (Ribeiro 2017).

At the end of the nineteenth century, the banks of the Almada River owned some of the most important cacao farms in Bahia, and the plantations expanded toward the present municipalities of Uruçuca (formerly Água Preta) and Itajuípe (formerly Pirangi). Gradually, cacao also spread along the western border in the territories of the present municipalities of Itabuna (formerly Tabocas) and Buerarema (formerly Macuco). To the south of Ilhéus, a new agricultural frontier also driven by cacao began to advance in the second decade of the nineteenth century in the territories of the current municipalities of Una and Canavieiras (Dias and Araujo 2016; Ribeiro 2017).

The rising prices of cacao beans on the international market in the second half of the nineteenth century attracted both the capital and migrants to work on the farms. The plantation was further boosted by an influx of capital from the decaying sugar sector (Lamberti 2017). Regarding labor force, two important migratory waves also promoted the economic and demographic expansion in the region. The first one came from overseas and brought Europeans in the condition of new owners, but also in that of poor settlers, to be installed in agricultural colonies. In addition to the Europeans, the region received migrants from the Middle East, mainly Arabs, Lebanese, and Syrians. The second migratory wave came from the interior of the northeast, especially from Sergipe and northern Bahia, where many refugees from the droughts migrated, trying their luck on the new agricultural front (Lyra 2007). In parallel with this migratory pressure, external demand for cacao increased. As a result, southern Bahia would become, in the last decades of the nineteenth century, the main cacao-producing area in Brazil.

The incomplete census of 1872 registered a total population of 10,692 inhabitants within the Ilhéus district. To this population, the following year would add almost 2,000 immigrants from the north of Europe, introduced in the agricultural colonies of Una and Comandatuba villages. Already in 1881, the year of the elevation of Ilhéus from village to the category of city, only this municipality had a population of approximately 10,000 inhabitants, of which 977 were slaves. Of this total, only 1042 free and 65 slaves lived within the urban nucleus. From the following decade, there will be an even faster growth of the internal migration to the south of Bahia, boosting the expansion of cacao by interior territories and in the coastal zones, as much of the north as of the south. The "cacao region" also presented the highest relative growth of the population of Bahia in the period 1890–1920, with rates exceeding 100%, as was the case of municipalities such as Ilhéus (725.9%), Itacaré (402.7%), Maraú (403.8%), Belmonte (240.3%), and Canavieiras (344.6%) (Silva et al. 1989). In that context, this region within the Atlantic Forest was beginning a new cycle of deforestation and transformation, irreversible and of great proportions.

It was only at the beginning of the twentieth century that farmers started to grow cocoa as agroforests. From that moment until 1976, the vast majority of the plantations occurred in a traditional and local mode of cultivation regionally known as "cabruca." The word is probably short for "cabrocagem," meaning the drilling of the forest. Indeed, as cacao (*Theobroma cacao*) is an understory species, farmers removed the entire original understory layer to plant cacao shrubs, but a few canopy trees were left for shade. At first glance, these shaded plantations resemble the original forest, only thinned and with a homogenous understory of cocoa trees. Cabrucas soon replaced vast lands of native forests, but compared with other land uses such as coffee or sugar, their presence assured the maintenance of a green tapestry of forest cover. Despite the significant changes compared with the original forests, this farming mode, often enriched by exotic shading trees, still keeps a structurally complex, multi-strata environment that is known to harbor many species of plants and animals (Cassano et al. 2009, 2014).

From 1911 to 1929, this crop became not only the most important agricultural product in Bahia (Virgens-Filho et al. 1993), but in some periods, production represented 20% of public incomes in the country (May and Rocha 1996). As a commodity, prices fluctuate internationally according to its supply and demand. In the case of cacao, the market is highly controlled by a few industries, so demand, not supply, largely determines the price. Because of the high volatility of cocoa's international market prices, aggravated by variations of exchange rates, farmers were consistently kept vulnerable to cyclical crisis, becoming chronically in debt. Uncertainties on climatic conditions, pests, and other endogenous features are also important factors regulating production and, in many cases, aggravated debts (see Caldas and Perz 2013). For instance, after the stock market *crash* of *1929,* cacao prices in 1933 were only 30% of those from 1927 (Rocha 2008). Traditionally, farmers used to compensate for the low prices not by an intensification and technological input but rather by expanding the crop area (Costa and Soares 2016).

In the 1950s, the situation of the cacao sector was meager, with production continuously dropping and leading to a serious stagnation on the regional economy. As a response, in 1957, the federal government created the Executive Commission of the Cacao Plantation Crop (CEPLAC), a federal agency aimed to organize the sector by providing planning, financial, and technological assistance. In 1962, CEPLAC carried out a widespread policy to modernize plantations under the framework of the green revolution (Johns 1999). Among other things, the strategy included subsidized loans conditioned to the adoption of a technological package that included the combined reduction of shade levels with an increase in the use of agrochemical, specifically fertilizers, and pesticides (Johns 1999). Under the technical supervision of CEPLAC, in 1964, the cacao farming expanded over the native vegetation, mostly from areas occupied by old-growth or secondary forests (Piasentin and Saito 2014). Under this system, known as "derruba total" (full removal), cacao shrubs were planted together with the exotic *Erythrina* trees, nitrogen-fixing species that also provide partial shade. In addition, from 1967 to 1986, CEPLAC launched a plan to renew the traditional plantations, through a massive thinning of old cabrucas. Surveys in 61 properties within the core region of cocoa plantation reported the presence of 171 different tree species, an average of 76 trees/ha (Alvim and Pereira 1965). The target was to eliminate 50–60% of the shade trees from the traditional plantations, thus reducing the average density of shade to 25 individuals trees/ha or less than a third of the regional average (Alvim and Pereira 1965). Johns (1999) estimated that the full implementation of such policy would have led to the removal of 25.8 million canopy trees, yet the author presented compiling evidence and reasons for the low compliance of the landowners to this shade clearance. Yet, until the 1960s, there were relatively more native forests than cocoa plantations.

The rehabilitation measures succeeded, leading to an increase in regional production from 88,000 tons in the early 1960s to nearly 400,000 tons in 1987 (Santos Filho et al. 2008). It was achieved by a significant rise in *productivity* levels – 220 kg/ha in 1962 to 740 kg/ha in 1980 (Rocha 2008) – with cacao farming spreading through nearly 600,000 ha in Bahia. The economic success of such a two-pronged strategy encompassing intensification and full removal came at a cost: driving a massive forest loss (Alger and Caldas 1994). During the 1970s, cacao achieved peak prices of U$ 4000/ton, suddenly stimulating the entering of new producers from Africa and Asia. The increment of the areas devoted to cacao farming led to successive records of harvesting, ultimately dropping the prices to less than U$ 800/ton (Martins 2007).

In 1989, the region was hard hit by a pest outbreak, after the criminal introduction of the fungus known as witch's broom (Pereira et al. 1990). This introduction was probably motivated by political reasons, not yet satisfactorily explained, and was later described as "agroterrorism" (Caldas and Perz 2013), greatly contributing to aggravate the social crisis throughout the region. The fungus (*Moniliophthora perniciosa* (Stahel) Aime & Phillips-Mora, *Marasmiaceae*) provoked the crash of the regional production, reduced to 1/4, or 96,000 tons annually (Pereira et al. 1990). The disease was a game-changer, although several factors have also contributed to the unprecedented crisis faced by the sector in Bahia (Alger and Caldas 1994). For instance, as international prices crashed down to less than half of the production costs, the shortage of subsidies and credit maintained the low level of technological input of the farms. Bankrupt landowners abandoned their properties, dismissing nearly 250,000 workers that fled to urban zones. Between 1980 and 2000, the rural population of southern Bahia dropped from 45% to 24% (Caldas and Perz 2013), and the lack of economic alternatives put the region in an unprecedented economic stagnation.

During this harsh time, logging not farming became the leading driver for deforestation. Wood exploitation has always been a pervasive activity well illustrated by the brazilwood cycle and the constant extraction of other high-quality timber. In the cacao region, logging activity increased during the periods in which the cacao farming expanded. However, in the 1970s, the opening of a state road (BA 001) and a main federal highway (BR 101) encouraged the arrival of a new cycle of wood exploration. After exhausting the regional stock of woods from the forests of northern Espírito Santo and the extreme south of Bahia (see below), many lumber companies eagerly settled in the cacao region, taking advantage of the timber stocked within the cabrucas. Decapitalized by the successive crisis, cacao farmers sold large quantities of trees, after which some properties were converted to pastures. According to Mesquita (1997), in 1971, forests comprised 11,000 km$^2$ of the region, but 10 years and 230 sawmills later, the forest was reduced to 20%. In 1996, a group of specialists concluded that the logging industry in the region was not sustainable, and despite some legal victories, the activity somehow continued (Mesquita 1997). It would take an extra decade for the publication of the Federal Decree 6660 (21 November 2008) finally regulating the full, legal protection of the entire biome.

In addition to native forests, a relevant part of the regional biota is reported within the cabrucas (Faria et al. 2006a, 2007; Cassano et al. 2009). The forested nature of this crop translates in a biodiversity-friendly matrix that allows movement for many species throughout the landscape (Faria et al. 2007; Cassano et al. 2009), mitigating the negative effects of forest loss and fragmentation (Pardini et al. 2009). But similar to what happened during the 1960s, there is a growing effort to increase local productivity by decreasing shade levels in cabrucas. This intensification is currently supported by a state law, which defines that a minimum of 40 native trees/ha must be kept in each cabruca (Portaria No. 10.225/2015, Bahia state), a value much lower than the average of 197 trees/ha found in those traditional cabrucas (see Sambuichi et al. 2012 or Schroth et al. 2015). The structural simplification of plantations is expected to negatively affect biodiversity and ecosystem services in this region. The removal of shade trees will not only reduce local tree diversity, but it is estimated that if this thinning policy is extended for all the existing cabrucas, it will lead to an estimated regional releasing of 21,000 tons of carbon, an amount equivalent of 2/3 of all carbon stored in region's native forests (Schroth et al. 2015). Although the identification of synergies and trade-offs between management intensification and the conservation role of traditional cabrucas is still under debate (Schroth et al. 2016), current efforts to improve the regional viability of cacao include other strategies. Among them, we highlight the introduction of more productive, genetically modified varieties, the niche market for organic crops, and the creation of new business models like the so-called "tree-to-bar" or "bean-to-bar", in which factories within local farms or in the region, respectively, allow the production of chocolate, the final product with a higher aggregated value (Assad 2017).

### 4.3.2 The Forests on the Extreme South of Bahia

Limited north and south by two massive forested areas, the region between the Jequitinhonha and Mucuri rivers remained apart from the economic growth experienced by the cacao region during the nineteenth century. With few and distant villages concentrated along the coastal zone, connected by precarious roads and by the sea, the region continued low populated and economically isolated from the consumer market. The arrival of a railroad linking the mining areas of Minas Gerais state to the coast, in 1882, encouraged a migratory flux from the hinterland, increasing the economic activities in the area (Giffoni 2006). The region also experienced the introduction of cacao, though it was irrelevant for bringing similar economic prosperity experienced in southern Bahia (Cerqueira-Neto 2013).

A significant occupation of the territory and its economic development started only in the 1950s, with the arrival of the BR 5 highway connecting the area with the north of Espírito Santo. Logging was the first business opening the devastation of large areas of forests. For instance, the arrival of a company called BRALANDA (Brasil Holanda SA) in 1952, with new technology capable of extracting and processing a large stock of timber, marked the setting of a new industry based on wood exploitation that led to an unprecedented rate of regional deforestation. As woods were extracted, the degraded forest was cleared and burnt, and the agricultural frontier expanded largely with coffee and cattle ranching. The latter economic activity was a key driver consolidating the regional development, opening vast areas of pastures at the expanse of the Atlantic Forests (Amorim and Oliveira 2007).

This development was accelerated when the old BR 5 was enlarged to form the BR 101, a national highway that completely integrated this territory to most developed states and consuming centers (Amorim and Oliveira 2007). This important logistical infrastructure boosted the logging business at the point that, at the end of the 1970s, the city of Eunápolis was considered the "Meca" of the logging business in Bahia (Mesquita 1997). The pace of deforestation was astonishing, as illustrated in Fig. 4.3. As pointed by Cerqueira-Neto (2013), despite some efforts, the region was never economically integrated on the cacao-producing region led by southern Bahia, but rather "The history of the region showed that it was the loggers from Espírito Santo and cattle ranchers from Minas Gerais states the main actors who have modified and landscape of the far South, both in rural and in urban areas."

In the 1980s, large reforestation programs using exotic trees such as *Pinus* were implemented first devoted to supplying wood for the steel industries of Minas Gerais and Espírito Santo states, and later came the *Eucalyptus* to supply the raw material for pulp mills. Today several national and multinational pulp companies are installed in the region and are supplied by an area larger than 450,000 ha of *Eucalyptus* plantations. The few forest remnants are only testimonies of the once exuberant formation of the original Hileia Baiana, today immersed in landscapes dominated by pastures, agriculture, and forest plantations. The current regional economy heavily relies on the pulp industry but also on diversification of agricultural production and new activities such as tourism, with the city of Porto Seguro representing an important destiny.

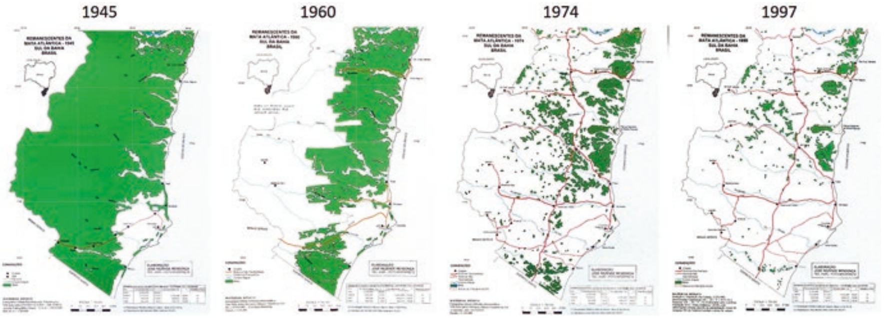

**Fig. 4.3** Time series showing the progressive loss of native forests (green) in the extreme south region of the state of Bahia, Brazil, from 1945 to 1997 (Reproduced from Mendonça et al. 1993)

### 4.3.3 The Forests on the North of Doce River

With the ending of the monarchy period by the Proclamation of the Republic in 1889, the forests from the north of Doce River that once belonged to the Porto Seguro captaincy were incorporated to the Espírito Santo state, originally comprising a captaincy itself. Until the first half of the nineteenth century, sugarcane was still the main economic activity in that state, a region with a few populated villages scattered along the coast, a stretch hardly exceeding 20 km (Saint-Hilaire 1974). However, it was the coffee production, starting in 1847, that allowed a broader and significant occupation of this territory, particularly its inner parts. Indeed, the population jumped from 49,092 people in 1856 to 82,137 inhabitants in less than 30 years (Macedo and Magalhaes 2011). As expected, this rapid growth triggered radical changes in the social, demographic, and economic structure of this territory. Moreover, most of these transformations were limited to the south, central, and inner parts of the state, as the expansion of coffee plantations was a direct result of the extension of the agriculture frontier from São Paulo and Minas Gerais states (Batista 2017). The vast, unexploited land of this new frontier allowed farmers to set larger properties, perfect to follow the production model of plantations, where large properties are devoted to produce and export a single crop, labored by slaves. Meanwhile, the presence of the Doce River imposed a natural barrier to the inflow of human occupation to the north. A map from 1878 still designated this northern region as a “Little-known land inhabited by indigenous people” (Egler 1962). Despite some controversy regarding the actual role played by natives in curbing new settlements, the existence of endemic diseases has also contributed to explaining why the region, that accounted for 47% of the state territory, concentrated less than 12% of the population in 1872 (Macedo and Magalhaes 2011). For these reasons, the original forest remained largely intact until the first decade of the twentieth century. As pointed by Egler (1962), the Doce River marked the physical segregation between an occupied territory in the south and an unknown, uninhabited block of forests in the north. In other words, the river was a major barrier to the occupation of a vast part of the territory, and the forests were left relatively intact.

The situation started to change in 1906 when a railway finally connected these northern lands to the central, more developed portion of the territory. This event suddenly brought more people to the region and, at the same time, the first sawmills that explore the forests for railroad ties, while coffee plantations significantly expanded that in 1920 the region became the leading producer on the state (Macedo and Magalhaes 2011). However, a significant increase in the influx of people and goods occurred only after the construction of the first bridge over the Doce River, in 1928. A wave of migrants from the central and southern parts pushed the forest further north, replacing the native areas by coffee plantations. Most properties were small, established as family units that have to be self-sufficient, not necessarily highly productive. Cacao was also another important cash crop dating back to the 1880s, but only after 1917, farming spread significantly over the native forests, forming a cacao-producing zone around the Linhares municipality (Santos-Lima et al. 2018).

Despite such attempts to settle and develop the region, the forests from the northern tip of Espírito Santo remained relatively unexplored until 1950 when compared with other regions along the Atlantic Forest. This dynamic, however, changed fast and dramatically. The progressive urbanization and industrialization of the country rapidly increased the national demand for wood. Suddenly, the large remaining forest tracts, that once precluded development, now provided new opportunities for the economic expansion of the region. In the 1960s, the region had 1700 working sawmills, supplying raw material for a range of sectors such as construction, furniture, and steel industry (Dean 1995). Forests were rapidly lost as they were first logged, cleaned, and then burned for agriculture and, increasingly, for pastures. For instance, between 1950 and 1975, cattle farming increased the unprecedented proportion of 681% (Bergamim 2016). Together, these activities severely reduced the once pristine forests, leaving behind a vast range of the lowland forests logged, degraded, and cleared.

Deforestation accelerated in 1971 after the arrival of the BR 101. By the end of the twentieth century, most of the native forests vanished, and the wood became rarer and more expensive to exploit due to the higher costs to assess the ever-distant stock areas. Between 1967 and 1986, a large government initiative promoted the occupation of large areas of forestry, mainly the monocultures of *Eucalyptus* and *Pinus*, rapidly transforming the region into a major pole of wood and paper industry (Siqueira et al. 2004). In fact, due to the right climatic conditions and implanted infrastructure, the region achieved the world's highest productivity level of *Eucalyptus*, with the state in general, and this region in particular, leading the world rank of cellulose exports. In addition to agriculture, in which coffee is a major source of income, pastures and extensive areas are devoted to forestry. Although the mining industry is more concentrated upstream of the Doce River Valley, in November 2015, an area of 650 km along the valley received 35 million $m^3$ of mining rejects after a dam, located in the municipality of Mariana, collapsed (Hatje et al. 2017). The "Mariana disaster" killed 19 people and led to unprecedented destruction and contamination of aquatic and terrestrial ecosystems (Magris et al. 2019).

## 4.4 The Challenge of Conserving What Remains

The Hileia Baiana suffered an intensive process of human occupation, an ultimate result of the dynamism of the historical, social, and ecological systems described here. Land use was, and still is, chiefly driven by the establishment of commodities and, thus, highly influenced by the boom and bust of such unpredictable economic cycles.

What is left from the native forests is still under severe pressure including not only the effect of habitat loss and its fragmentation but also chronic disturbances. According to the 2016 annual report of the NGO SOS Mata Atlântica, the Atlantic Forest remaining in the entire states of Espírito Santo and Bahia comprises 2,005,710 ha and 483,172 ha, respectively (SOS Mata Atlântica and INPE 2017). Although this figure represents a significant reduction from its original cover, by far, the region encompasses the largest extent of the northern portion of this biome. It is also impressive that inventories still confirm that the region, despite such aggressions, harbors high levels of biodiversity, part of which is endemic or remains unknown. It is not uncommon to observe that tree inventories rarely identify more than 80% of the samples, quite often having a high number of new species still waiting to be described and named.

During the 1980s and 1990s, Brazil made a significant effort to increase the representation of protected areas, some along the Hileia Baiana (Mittermeier et al. 2005). Despite their importance, these spaces are limited in size, and a considerable part of the extant biodiversity exists in modified areas. To increase the connectivity among the isolated fragments (protected or unprotected remnants) and the surrounding mosaic of different land uses, these areas are immersed in the current "Central Corridor of the Atlantic Forest" (CCAF), an instrument prioritizing public policies of conservation enforced by the federal government (Fig. 4.4). The polygon of the CCAF comprises a land of ~112,135 km$^2$, of which 32.9% is covered by native forests under different successional stages and disturbance levels (http://mapbiomas.org/), with only 2.5% legally protected by 45 protected areas (municipal, state, and federal protected units under IUCN categories Ia, II, and III).

Biodiversity within the extant fragments and the surrounding mosaic land uses that characterize the anthropogenic landscapes is constantly exposed to many threats. Habitat remnants immersed in more deforested landscapes experience significant changes including the shrinkage of forest structure (Rocha-Santos et al. 2016), selective extinctions (Morante-Filho et al. 2015; Benchimol et al. 2017a), and alteration of ecological processes (Menezes et al. 2016; Benchimol et al. 2017b; Rocha-Santos et al. 2017; Morante-Filho et al. 2018) which are key to maintain the functionality of the system. Fire, logging, and hunting are also important vectors of chronic disturbance that further cause biodiversity erosion not only in this region but also for the entire biome (Tabarelli et al. 2010). Except for some large, well-protected conservation units in Espírito Santo and extreme southern Bahia, medium- and large-sized mammals are rare or absent from most fragments within this corridor (Canale et al. 2012), with serious consequences for the functioning and regeneration of these forests (Culot et al. 2017). There is an urgent need to reverse this scenario and ensure the maintenance of biodiversity and the ecological integrity of remaining systems that are also facing the likely and unprecedented effects of climate change.

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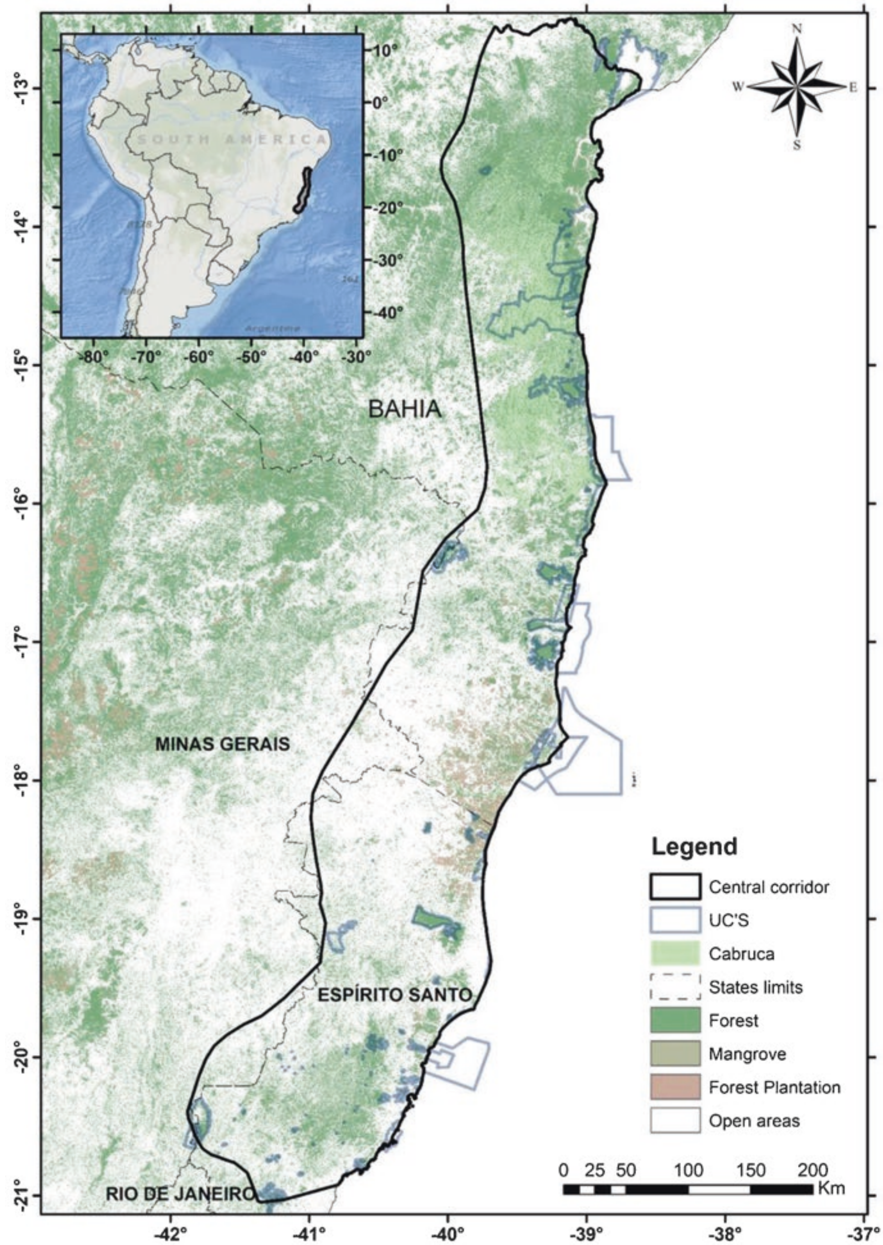

**Fig. 4.4** Limits of the Central Corridor of the Atlantic Forest, depicting the current location of forest remnants and other main land uses, and the contour of main conservation units (UCs). (Source: MapBiomas http://mapbiomas.org/)

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## References

Alger K, Caldas M (1994) The declining of the cocoa economy in the Atlantic forest of southern Bahia, Brazil: conservation attitudes of cocoa planters. Environmentalist 14(2):107–119

Alvim PT, Pereira CP (1965) Sombra e espaçamento nas plantações de cacau no Estado da Bahia. In: Relatório anual 1964. Centro de Pesquisas do Cacau, Comissão Executiva do Plano da Lavoura Cacaueira, Itabuna, Brasil, pp 18–19

Amantino MS (2018) A Companhia de Jesus na cidade do Rio de Janeiro. O caso do Engenho Velho, século XVIII. Paço Editorial, Jundiaí

Amorim RR, Oliveira RC (2007) Degradação ambiental e novas territorialidades no extremo sul da Bahia. Caminhos de Geografia 8(22):18–37

Amorim AM, Thomas WW, Carvalho AMV, Jardim JG (2008) Floristics of the Una biological reserve, Bahia, Brazil. In: Thomas WW (ed) The Atlantic coastal Forest of northeastern Brazil. Memoirs of The New York Botanical Garden, New York, pp 67–146

Andrade MP (2003) Ilhéus: Passado e Presente, 2rd edn. Editus, Ilhéus

Andrade-Lima D (1966) Vegetação. In: Instituto Brasileiro de Geografia e Estatística & Conselho Nacional de Geografia, ed. Atlas Nacional do Brasil, Rio de Janeiro, 11p

Argolo AJS (2004) As Serpentes dos Cacauais do Sudeste da Bahia. Editus, Ilhéus, 250p

Assad L (2017) Cacau no Brasil: uma oportunidade que (ainda) não se concretizou. Ciência e Cultura 69(2):11–13

Barbosa JSF, Correa-Gomes LC, Marinho MM, da Silva FCA (2003) Geologia do segmento sul do Orógeno Itabuna-Salvador-Curaçá. Rev Bras Geociências 33(1-Suplemento):33–47

Barickman BJ (2003) Um contraponto baiano: açúcar, fumo, mandioca e escravidão no Recôncavo, 1780–1860. Civilização Brasileira, Rio de Janeiro

Batalha-Filho H, Fjeldså J, Fabre PH, Miyaki CY (2013) Connections between the Atlantic and the Amazonian forest avifaunas represent distinct historical events. J Ornithol 154:41–50

Batista MR (2017) O processo de transição da mão de obra escrava para a livre e a imigração no Espírito Santo do século XIX. In: VI Congresso Internacional UFES Paris/ Est, Vitória, pp 912–930

Benchimol M, Mariano E, Faria D, Rocha-Santos L, Pessoa MS, Gomes F, Talora D, Cazetta E (2017a) Translating plant community responses to habitat loss into conservation practices: Forest cover matters. Biol Conserv 209:499–507

Benchimol M, Talora DC, Mariano E, Oliveira TL, Leal A, Mielke MS, Faria D (2017b) Losing our palms: the influence of landscape-scale deforestation on Arecaceae diversity in the Atlantic forest. Forest Ecol Mang 304:314–322

Bergamim MC (2016) A pequena propriedade rural no Espírito Santo: constituição e crise de uma agricultura familiar. In: Congresso da Sociedade Brasileira de Economia, Administração e Sociologia Rural, 44, 2006. Sober, Fortaleza/Anais/Brasília, DF

Bondar G (1939) A Cultura de Cacau na Bahia. Livraria Duas Américas, Salvador, Bahia

Cabral DC (2014) Na Presença da Floresta: Mata Atlântica e História Colonial. Garamond/ FAPERJ, Rio de Janeiro

Caldas M, Perz S (2013) Agro-terrorism? The causes and consequences of the appearance of witch's broom disease in cocoa plantations of southern Bahia, Brazil. Geoforum 47:147–157

Câmara MF (1789) Ensaio de descrição física e econômica da Comarca de São Jorge dos Ilhéus. Memórias Econômicas da Academia de Sciencias de Lisboa 1:304–350

Campos JS (2006) Crônicas da Capitania de São Jorge dos Ilhéus, 3rd edn. Editus, Ilhéus

Canale GR, Peres CA, Guidorizzi CE, Gatto CAF, Kierulff MCM (2012) Pervasive defaunation of forest remnants in a tropical biodiversity hotspot. PLoS One 7:e41671

Carnaval AC, Moritz C (2008) Historical climate modelling predicts patters of current biodiversity in the Brazilian Atlantic Forest. J Biogeogr 35:1187–1201

Carrara AA (2007) Fiscalidade e estrutura agrária: Ilhéus, Porto Seguro e Espírito Santo, séculos XVI-XVIII. In: Dias MH, Carrara AA (eds) Um lugar na história: a capitania e comarca de Ilhéus antes do cacau. Editus, Ilhéus, pp 15–46

Cassano CR, Schroth G, Faria D, Delabie JHC, Bede L (2009) Landscape and farm scale management to enhance biodiversity conservation in the cocoa producing region of southern Bahia, Brazil. Biodivers Conserv 18:577–603

Cassano CR, Schroth G, Faria D, Delabie JHC, Bede L, Oliveira LC, Mariano-Neto E (2014) Desafios e recomendações para a conservação da biodiversidade na região cacaueira do sul da Bahia. CEPLAC/CEPEC Boletim Técnico 205. 54p

Cerqueira-Neto SPG (2013) Construção geográfica do extremo sul da Bahia. Rev Geogr 30(1):243–263

Coelho Filho LW (2000) A capitania de São Jorge e a década do açúcar (1541–1550). Vila Velha, Salvador

Costa FM, Soares NS (2016) Cacau riqueza de pobres (1ª edição). Ilhéus-Ba Editus 286p

Culot L, Bello C, Batista JLF, do Couto HTZ, Galetti M (2017) Synergistic effects of seed disperser and predator loss on recruitment success and long-term consequences for carbon stocks in tropical rainforests. Sci Rep 7:7662

Dean W (1995) With broadax and firebrand - the destruction of the Brazilian Atlantic Forest. University of California Press, Berkeley, 482 p

Delabie JHC, Andrade A, Silva APS, Brescovit AD, Jahyny BJB, Jared C, Mariano CSF, Castaño-Meneses G, Carvalho JTL, Lopes JMS, Boccardo L, Antoniazzi MM, Santos PP, Bouzan RS, Calvanese V, Reis YT (2017) Fauna fossória e da serapilheira associada ao cultivo do cacaueiro no Sudeste da Bahia. In: Souza JO Jr (ed) Cacau: Cultivo. Pesquisa e Inovação, Ilhéus, Editus

Delabie JHC, Jahyny B, Nascimento IC, Mariano CSF, Lacau S, Campiolo S, Philpott SM, Leponce M (2007) Contribution of cocoa plantations to the conservation of native ants (Insecta: Hymenoptera: Formicidae) with a special emphasis on the Atlantic Forest fauna of southern Bahia, Brazil. Biodivers Conserv 16:2359–2384. https://doi.org/10.1007/s10531-007-9190-6

Delabie JHC, Mariano CSF, Nascimento IC (1998) As formigas do Município de Ilhéus (Insecta: Hymenoptera: Formicidae). Especiaria 1(2):133–152

Dias MH (2010) A floresta mercantil: exploração madeireira na capitania de Ilhéus no século XVIII. Rev Bras Hist 30:193–214

Dias MH (2011) Farinha, madeiras e cabotagem: a capitania de Ilhéus no antigo sistema colonial. Editus, Ilhéus

Dias MH (2016) O pau-brasil na Bahia colonial: zonas de ocorrência, condições de exploração e impactos ambientais. In: Cabral DC, Bustamante AG (eds) Metamorfoses florestais: culturas, ecologias e as transformações históricas da Mata Atlântica. Prismas, Curitiba, pp 151–175

Dias MH, Araujo GS (2016) Ocupação territorial em uma fronteira indígena do sul da Bahia: Una e Olivença em meados do século XIX. Territórios & Fronteiras 9:215–135

Dominguez JML, Martin L, Bittencourt ACSP (1982) Evolução paleogeográfica do delta do rio Jequitinhonha durante o quaternário: influência das variações do nível do mar. In: Suguio K, Meis MR, Mousinho DE, Tessler MG (eds) Atas do Quarto Simpósio do Quaternário no Brasil. CTCQ-SBG, Rio de Janeiro, pp 69–92

Egler WA (1962) A zona pioneira ao norte do Rio Doce. Rev Bras Geogr 167:147–180

Faria D, Laps RR, Baumgarten J, Cetra M (2006a) Bat and bird assemblages from forests and shade cacao plantations in two contrasting landscapes in the Atlantic Forest of southern Bahia, Brazil. Biodivers Conserv 15:587–612

Faria D, Paciencia MLB, Dixo M, Laps RR, Baumgarten J (2007) Ferns, frogs, lizards, birds and bats in forest fragments and shade cacao plantations in two contrasting landscapes in the Atlantic forest, Brazil. Biodivers Conserv 16:2335–2357

Faria D, Soares-Santos B, Sampaio E (2006b) Bats from the Atlantic rainforest of southern Bahia, Brazil. Biota Neotrop 6(2):1–13

Fernández F, Delabie JHC, Nascimento IC (2009) *Diaphoromyrma*, a new myrmicine ant genus (Hymenoptera: Formicidae) from north eastern Brazil. Zootaxa 2204:55–62

Giffoni JMS (2006) Trilhos Arrancados: História da Estrada de Ferro Bahia e Minas (1878–1966). PhD Thesis, Universidade Federal de Minas Gerais, Belo Horizonte

Haffer J (1969) Speciation in Amazonian forest birds. Science 165:131–137

Hatje V, Pedreira RMA, De Rezende CE, Schettini CAF, De Souza GC, Marin DC, Hackspacher PC (2017) The environmental impacts of one of the largest tailing 748 dam failures worldwide. Sci Rep 7:1–13

Johns ND (1999) Conservation in Brazil's chocolate forest: the unlikely persistence of the traditional cocoa agroecosystem. Environ Manag 23:31–47

Köppen W (1936) Das Geographisches System der Klimate. In: Köppen W, Geiger W (eds) Handbuch der Klimatologie (Kapitel 3 V.1). Teil. C. Ebr. Bornträger, Berlin

Lamberti F (2017) Economic history of cocoa in Southern Bahia its role on economy, society and culture. Master Dissertation - Escola de Administração de Empresas de São Paulo, Fundação. 80p

Laps RR (2006) Efeito da fragmentação e alteração do hábitat na avifauna da região da Reserva Biológica de Una, Bahia, PhD Thesis, Universidade Estadual de Campinas, São Paulo

Ledo RMD, Colli GR (2017) The historical connections between the Amazon and the Atlantic Forest revisited. J Biogeogr 44:2551–2563. https://doi.org/10.1111/jbi.13049

Linhares MY, Silva FCT (1981) História da agricultura brasileira. Brasiliense, São Paulo

Lyra HB (2007) A 'crise' de mão-de-obra e a criação de colônias agrícolas na Bahia: 1850–1889. In: Dias MH, Carrara AA (eds) Um lugar na história: a capitania e comarca de Ilhéus antes do cacau. Editus, Ilhéus, pp 233–305

Macedo FC, Magalhaes DF (2011) Formação econômica do Espírito Santo: do isolamento econômico à inserção aos mercados nacional e internacional. Rev Hist Reg 16(1):61–99

Magalhães RCS, Maio MC (2007) Desenvolvimento, ciência e política: o debate sobre a criação do Instituto Internacional da Hiléia Amazônica. História, Ciências, Saúde – Manguinhos, Rio de Janeiro 14:169–189

Magris RA, Marta-Almeida M, Monteiro JÁ, Ban NC (2019) A modelling approach to assess the impact of land mining on marine biodiversity: assessment in coastal catchments experiencing catastrophic events (SW Brazil). Sci Total Environ 659:828–840

Mahony MA (1996) The world cacao made: society, politics and history in Southern Bahia, Brazil, 1822–1919. PhD dissertation, Yale University, Yale

Martini AMZ, Fiaschi P, Amorim AMA, Paixão JL (2007) A hot-point within a hot-spot: a high diversity site in Brazil's Atlantic Forest. Biodivers Conserv 16:3111–3128

Martins FM (2011) Historical biogeography of the Brazilian Atlantic Forest and the Carnaval–Moritz model of Pleistocene refugia: what do phylogeographical studies tell us? Biol J Linn Soc 104 (3:499–509

Martins PTA (2007) Os reflexos da crise da lavoura cacaueira nos ecossistemas de manguezal do município de Ilhéus, Bahia. Geografia (Londrina) 16(1):39–49

Mattos-Silva LA, Jardim JG, Thomas WW, Santos TS (2008) Common names of vascular plants of the Atlantic coastal forest region of southern Bahia, Brazil. In: Thomas WW (ed) The Atlantic coastal Forest of northeastern Brazil. The New York Botanical Garden Press, Bronx, New York, pp 245–318

May PH, Rocha RB (1996) O sistema agrossilvicultural do cacau-cabruca. In: Lopes IV, Bastos Filho GS, Biller D, Bale M (eds) Gestão ambiental no Brasil: experiência e sucesso. Editora FGV, Rio de Janeiro, pp 35–62

Mendonça JR, de Carvalho AM, Mattos Silva LA, Thomas WW (1993) 45 Anos de Desmatamento no Sul da Bahia, Remanescentes da Mata Atlântica – 1945, 1960, 1974, 1990. Projeto Mata Atlântica Nordeste, CEPEC, Ilhéus

Menezes I, Cazetta E, Morante-Filho JC, Faria D (2016) Forest cover and bird diversity: drivers of fruit consumption in forest interiors in the Atlantic Forest of southern Bahia, Brazil. Trop Conserv Sci 9:549–562

Mesquita CAB (1997) Exploração madeireira no sul da Bahia: situação atual e perspectivas. In: Revista dos Mestrandos em Direito Econômico da Universidade Federal da Bahia. Ed. Especial de Direito Ambiental, pp 121–135

Miller S (2000) Fruitless tress: Portuguese conservation and Brazil's colonial timber. Stanford University Press, Stanford

Mittermeier RA, Fonseca GAB, Rylands AB, Brandon K (2005) Uma breve história da conservação da biodiversidade no Brasil. Megadiversidade 1(1):4–21

Morante-Filho JC, Faria D, Mariano-Neto E, Rhodes J (2015) Birds in anthropogenic landscapes: the responses of ecological groups to forest loss in the Brazilian Atlantic Forest. PLoS One 10:e0128923

Morante-Filho JC, Arroyo-Rodrigues V, Pessoa MS, Cazetta E, Faria D (2018) Direct and cascading effects of landscape structure on tropical forest and non-forest frugivorous birds. Ecol Appl 28:2024–2032

Morton FWO (1978) The royal timber in late colonial Bahia. Hispanic Am Hist Rev 58:41–61

Motamayor JC, Lachenaud P, Da Silva E, Mota JW, Loor R, Kuhn DN, Brown JS, Schnell RJ (2008) Geographic and genetic population differentiation of the Amazonian chocolate tree (*Theobroma cacao* L). PLoS One 3(10):e3311

Ostroski P, Saiter FZ, Amorim AM, Fiaschi P (2018) Endemic angiosperms in Bahia Coastal Forests, Brazil: an update using a newly delimited area. Biota Neotrop. 18(4):e20180544. https://doi.org/10.1590/1676-0611-BN-2018-0544

Pardini R, Faria D, Accacio GM, Laps RR, Mariano-Neto E, Paciencia MLB, Dixo M, Baumgarten J (2009) The challenge of maintaining Atlantic forest biodiversity: a multi-taxa conservation assessment of specialist and generalist species in an agro-forestry mosaic in southern Bahia. Biol Conserv 142:1178–1190

Padua JA (2002) Um sopro de destruição: pensamento político e critica ambiental no Brasil escravista (1786–1888). Jorge Zahar, Rio de Janeiro, 318p

Peixoto AL, Silva IM, Pereira OJ, Simonelli M, Jesus RM, Rolim SG (2008) Tabuleiro forests north of the Rio Doce: their representation in the Vale do Rio Doce natural reserve, Espírito Santo, Brazil. Mem New York Bot Gard 100:319–350

Pereira JL, Ram A, Figueiredo JM, Almeida LCC (1990) First occurrence of witches' broom disease in the principal cocoa growing region of Brazil. Trop Agr 67:188–189

Piasentin FB, Saito CH (2014) Os diferentes métodos de cultivo de cacau no sudeste da Bahia, Brasil: Aspectos históricos e percepções. Boletim Do Museu Paraense Emilio Goeldi: Ciências Humanas Belém 9(1):61–78. https://doi.org/10.1590/S1981-81222014000100005

Pinto LPS (1994) Distribuição geográfica, população e estado de conservação de mico-leão-da-cara-dourada, *Leontopithecus chrysomelas* (Callithrichidae, Primates). Master thesis, Universidade Federal de Minas gerais, Belo Horizonte

Ribeiro ALR (2017) *In Memoriam*: urbanismo, literatura e morte. Editus, Ilhéus

Rocha LB (2008) A região cacaueira da Bahia – dos coronéis à vassoura-de-bruxa: saga, percepção, representação. Editus, Ilhéus, 255p

Rocha-Santos L, Benchimol M, Mayfield MM, Faria D, Pessoa MS, Talora D, Mariano E, Cazetta E (2017) Functional decay in tree community within tropical fragmented landscapes: effects of landscape-scale forest cover. PLoS ONE (Electronic) 12:e0175545, 2017

Rocha-Santos L, Pessoa MS, Cassano CR, Talora D, Orihuela RL, Mariano E, Morante-Filho JC, Faria D, Cazetta E (2016) The shrinkage of a forest: landscape-scale deforestation leading to overall changes in local forest structure. Biol Conserv 196:1–9

Ruf F, Schroth G (2004) Chocolate forests and monocultures: a historical review of cocoa growing and its conflicting role in tropical deforestation and forest conservation. In: Schroth G, Fonseca GAB, Harvey CA, Gascon C, Vasconcelos H, Izac AMN (eds) Agroforestry and biodiversity conservation in tropical landscapes. Island Press, Washington

Saint-Hilaire A (1974) Viagem ao Espírito Santo e Rio Doce. Belo Horizonte Itatiaia; São Paulo: EDUSP; Niteroi: EDUFF

Sambuichi RHR, Vidal DB, Piasentin FB, Jardim JG, Viana TG, Menezes AA, Mello DLN, Ahnert D, Baligar VC (2012) Cabruca agroforests in southern Bahia, Brazil: tree component, management practices and tree species conservation. Biodivers Conserv 21:1055–1077

Santos Filho LP, Pires JL, Ruiz MAM, Valle RR, Afonso JMC, Marrocos PCL, Freire ES (2008) Produção de cacau e a vassoura-de-bruxa na Bahia. Agrotrópica 20:73–82

Santos RJ, Koch EBA, Leite CMP, Porto TJ, Delabie JHC (2017) An assessment of leaf-litter and epigaeic ants (Hymenoptera: Formicidae) living in different landscapes of the Atlantic Forest biome in the state of Bahia, Brazil. J Insect Biodivers 5(19):1–19

Santos-Lima S, Souza CAS, Patrocínio NGRB, Silva RA, Santos RSG, Gramacho KP (2018) Favorabilidade, distribuição e prevalência da vassoura-de-bruxa do cacaueiro no estado do Espirito Santo, Brasil. Agrotrópica 30(1):5–14

Schroth G, Bede LC, Paiva AO, Cassano CR, Amorim AM, Faria D, Mariano-Neto E, Martini AMZ, Sambuichi RHR, Lôbo RN (2015) Contribution of agroforests to landscape carbon storage. Mitig Adapt Strat Gl 20:1175–1190

Schroth G, Jeusset A, Gomes AS, Florence CT, Coelho NAP, Faria D, Läderach P (2016) Climate friendliness of cocoa agroforests is compatible with productivity increase. Mitig Adapt Strat Gl 21:67–80

Schwartz S (1988) Segredos internos. Companhia das Letras, São Paulo

Silva JMC, Casteleti CH (2003) Estado da biodiversidade da Mata Atlântica brasileira. In: Galindo-Leal C, Gusmão-Câmara I (eds) Mata Atlântica, Biodiversidade, Ameaças e Perspectivas. Fundação SOS Mata Atlântica, Conservação Internacional, Belo Horizonte, pp 43–59, 471p

Silva SCB, Leão SO, Silva BCN (1989) Urbanização e metropolização no estado da Bahia: evolução e dinâmica. UFBA Press, Salvador

Siqueira JDP, Lisboa RS, Ferreira AM, Souza MFR, Araújo E, Lisbão Júnior L, Siqueira MM (2004) Estudo ambiental para os programas de fomento florestal da Aracruz Celulose S.A. e extensão florestal do Governo do Estado do Espírito Santo. Floresta, Edição Especial, pp 3–67

SOS Mata Atlântica, INPE. (2017). Atlas dos remanescentes florestais da Mata Atlântica – Período de 2015–2016. Fundação SOS Mata Atlântica and Instituto Nacional de Pesquisas Espaciais 63p. http://sosma.org.br/projeto/atlas-da-mata-atlantica/dados-mais-recentes/

Souza JBO (1939) Pau-brasil na história nacional. Companhia Editora Nacional, Rio de Janeiro

Srbek-Araujo AC, Simon JE, Magnago GR, Pacheco JF, Fonseca PSM, Whitney BM, Silveira LF (2014) A avifauna da Reserva Natural Vale, Linhares, Espírito Santo, Brasil. Ciencia & Ambiente 49:169–191

Suguio K, Angulo RJ, Carvalho AM, Corrêa ICS, Tomazelli LJ, Willwock JÁ, Vital H (2005) Paleoníveis do mar e paleolinhas de costa. In: Souza CRG, Suguio K, Oliveira MAS, Oliveira PE (eds) Quaternário do Brasil. Ribeirão Preto, Holos, pp 114–129

Tabarelli M, Aguiar AV, Ribeiro MC, Metzger JP, Peres CA (2010) Prospects for biodiversity conservation in the Atlantic Forest: lessons from aging human-modified landscapes. Biol Conserv 143(10):2328–2340

Thomas WW, Barbosa MRV (2008) Natural vegetation types in the Brazilian Atlantic coastal forest north of the Rio Doce. Mem N Y Bot Gard 100:6–20

Thomaz LD, Monteiro R (1997) Composição florística da mata Atlântica de encosta da Estação Biológica de Santa Lucia, município de Santa Tereza, ES. Boletim do Museu de Biologia Mello Leitão 7:3–48

Thomas WM, Carvalho AMV, Amorim AMA, Garrison J, Arbeláez AL (1998) Plant endemism in two forests in southern Bahia, Brazil. Biodivers Conserv 7:311–322

Vanzolini PE, Williams EE (1981) The vanishing refuge: a mechanism for ecogeographic speciation. Papéis Avulsos de Zoologia 34:251–255

Virgens-Filho AC, Santos AHC, Figueroa GES, Isensee HCF, Silva IM, Abreu JM, Melo JRV, Cruz lP (1993) A CEPLAC e a crise da lavoura cacaueira. CEPLAC, Ilhéus, 30p

Wied, MPv (Maximiliano, príncipe de Wied-Neuwied) (1989) Viagem ao Brasil. Itatiaia/USP Press, Belo Horizonte

# Chapter 5
# The Southern Atlantic Forest: Use, Degradation, and Perspectives for Conservation

**Marcos Bergmann Carlucci, Vinícius Marcilio-Silva, and José Marcelo Torezan**

**Abstract** The southern Atlantic Forest comprises tropical dense, seasonal, and mixed forests, occurring south of the Doce River in S and SE Brazil, NE Argentina, and SE Paraguay. These forests harbor high biodiversity but have been severely degraded along the centuries following the arrival of Europeans to South America. We revisit the history of use and degradation of the southern Atlantic Forest, identifying commonalities and idiosyncrasies among its forest types. We also discuss the role of protected areas, sustainable use of the forest, and restoration to indicate trends and challenges for the conservation of the remaining and future forests.

**Keywords** *Araucaria* mixed forest · Atlantic dense forest · Hotspot · Rainforest · Seasonal forest

## 5.1 Introduction

The Atlantic Forest in southern and southeastern Brazil is represented by three major forest types, namely, Atlantic dense forest, seasonal forest, and *Araucaria* mixed forest (Fig. 5.1). The biogeographic history of the Atlantic Forest indicates floristic affinities with the Amazon rainforests and Caatinga xeric shrublands and thorny, short forests (Oliveira-Filho et al. 2005; Mori et al. 1981), as well as with the Andes (SanMartín and Ronquist 2004). The Doce River constitutes a natural barrier

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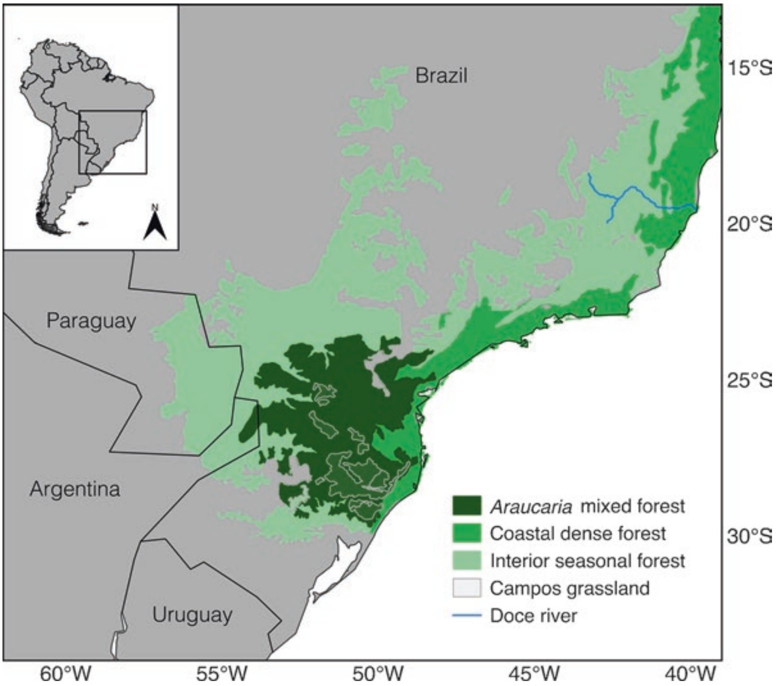

**Fig. 5.1** The distribution of the main forest types of the Atlantic Forest, which spread mostly throughout S and SE Brazil, also reaching NE Argentina and SE Paraguay. The Doce River, which divides the North and South block of the Atlantic Forest, is depicted. (Forest types according to ecoregions adapted from Olson et al. (2001) and Campos grassland distribution according to IBGE (1992))

for most of the Atlantic Forest biota since the Pleistocene (Carnaval and Moritz, 2008), naturally dividing the Atlantic Forest into North and South blocks. In this chapter, we will focus on the South block of the Atlantic Forest, ranging from 19° S in the southern margin of the Doce River, in the state of Espírito Santo, SE Brazil, to 32° S in the state of Rio Grande do Sul, S Brazil.

Atlantic dense forests are distributed along the coast, mainly in the Serra do Mar range, in warm, humid, non-seasonal climates. Few tree species are deciduous, and usually less than 20% of tree individuals lose their leaves in the winter, from June to August (IBGE 2012). Lianas are relatively scarce and epiphytes abundant. Seasonal forests occur in the less humid hinterlands, in moderately seasonal climates. Contrasting with Atlantic dense forests of southern Brazil, which are known for being influenced by the orographic rains at the Serra do Mar range, and the *Araucaria* mixed forests of southern and alto-montane portions of inland, where lower temperatures and high rainfall are observed, seasonal forests cover large inland areas with a drier climate. Deciduousness varies greatly, from about 20% to over 50% of tree individuals, following regional patterns of rainfall distribution and soil water storage capacity. In some places, epiphytes are scarce, while lianas are abundant. The *Araucaria* mixed forests occur in large tracts in hinterland plateaus and river valleys and in smaller patches associated with dense forests, seasonal forests, and subtropical grasslands.

The history of degradation of the three main forest types of the southern Atlantic Forest holds several similarities, but, similar to the North block, the dense forests were impacted earlier. The degradation of the Atlantic dense forests started with the arrival of Portuguese colonizers in the state of Bahia in 1500 (Dean 1997). Five centuries of deforestation led to only 11.3% of remnants (mostly secondary, young forests) of the original distribution of the Atlantic Forest (Ribeiro et al. 2009). A more recent evaluation with automated fragment detection and high-resolution imagery (5 m) found 26% of Atlantic Forest cover, most of small secondary vegetation islands, disconnected from large fragments (Rezende et al. 2018). Broad-scale deforestation is usually explained by unregulated, often illegal and ambitious profit pursuit. However, from the colonial period endeavor of occupation to the tax-based impulse to forest conversion by the twentieth-century governments, deforestation in southern Brazil has received state support.

Major economic cycles of natural resource exploitation of the southern Atlantic Forest started in the sixteenth century (reviewed by Rambo 1956; Maack 1968; Dean 1997; Soares and Medri 2002; Castella and Britez 2004; Baer 2008; Joly et al. 2014). Sugarcane has been an important crop in many areas of São Paulo since the sixteenth century. Extractivism of the mate (*Ilex paraguariensis* A. St.-Hil.) leaves represented a major economic cycle for southern Brazil since the sixteenth century. Coffee plantation was important in the northern region of Paraná and most of São Paulo state since the early nineteenth century. A shift from coffee plantation to cattle ranching in the Southeast due to soil degradation occurred during the twentieth century. Brazilian pine [*Araucaria angustifolia* (Bertol.) Kuntze] logging represented the main economic cycle for the South during most of the twentieth century. Soybean has occupied large areas of land in the west portion of the South region since the 1970s. *Eucalyptus* tree plantations in the Southeast and *Pinus* tree plantations in the South were established in former areas of cattle ranching (including native grasslands) during the late twentieth century and the early twenty-first century. Infrastructure expansion (urbanization, roads, gas and oil pipelines, water reservoirs, hydroelectric reservoirs) has occurred mostly during the last half century, expanding anthropic impacts toward native ecosystems.

During this entire period, southern Atlantic Forests were subject to selective logging, potentially leading to tree population declines, and defaunation, where large-bodied vertebrates are often extirpated from forest fragments due to over-poaching (Joly et al. 2014). Once degraded, landscapes have been subject to further land use shifts, which has usually led to additional degradation and biodiversity declines. Here we revisit the history of use and degradation of the major Atlantic Forest types south of the Doce River valley, which comprehends the *Araucaria* mixed forest, the seasonal forests, and the dense forests in southern and southeastern Brazil (Fig. 5.1).

## 5.2 Use and Degradation of the Three Major Atlantic Forests South of the Doce River

### 5.2.1 Atlantic Dense Forest

#### Characteristics

The Atlantic dense forest is an evergreen tropical forest structured in layers, presenting a high-canopy arboreal stratum, formed by trees up to 40-m tall, and an understory with high diversity and density of herbs and shrubs, distributed along the Brazilian Atlantic coastal plains and mountains. The South block of dense forests covers a wide range of elevations, from coastal lowlands (0–50 m a.s.l.), to slopes and peaks of the Serra do Mar mountain range (50–2,200 m a.s.l.), and latitudes, from 19 °S to 32 °S. Southern Atlantic dense forests have their west distribution in inland Brazil, reaching the *Araucaria* mixed forest in the South and the seasonal forest in the Southwest and Northwest.

Atlantic dense forest plant composition is mainly driven by climate (Oliveira-Filho and Fontes 2000; Marcilio-Silva et al. 2017). In general, climate is warm and wet in lowlands and cooler and wetter in the slopes (IBGE 1992; Oliveira-Filho and Fontes 2000). The average total annual rainfall is 1,600 mm, varying slightly among seasons (average of 500 mm in the summer and 340 mm in the winter). The average temperature ranges from 25 °C in lowlands to 16 °C in higher altitudes (Marcilio-Silva et al. 2017). The Atlantic dense forests can be subdivided into five formations, according to topography, latitude, and elevation (IBGE 2012): *alluvial formation*, not conditioned to topography and with repetitive environments, inside the alluvial terraces of river basins; *lowland formation*, which occurs in Tertiary or Quaternary sedimentary terrain areas such as flat terraces, plains, and depressions not susceptible to flooding, between 5 and 50 m a.s.l.; *submontane formation*, which occurs on the slopes of hills, between 50 and 500 m a.s.l.; *montane formation*, which occurs at the top of plateaus and mountain ranges, between 500 and 1,500 m a.s.l.; and *alto-montane formation*, which occurs above the boundaries of montane formations.

The southern Atlantic dense forests are highly fragmented in general. Patches of (almost) undisturbed forest are located mainly on steep slopes (Ribeiro et al. 2009), while fragments of secondary forests usually occur on abandoned sites that were used for slash-and-burn agriculture during the last two centuries (Liebsch et al. 2008). Atlantic dense forests encompass extensive urban areas such as the metropolitan areas of São Paulo and Rio de Janeiro, the two largest Brazilian cities, and their original region is inhabited by more than 50 million people. Only 12% (~160,000 km$^2$) from the original cover still remains, and less than half of these are in protected areas (SOS Mata Atlântica and INPE 2017). The Atlantic Forest was the first forested area to be colonized by the European settlers in the sixteenth century (Dean 1997). The history of this colonization and Brazil's development help understand how this ancient and biodiverse forest came to be so endangered in present days.

## History of Degradation and Present Socioeconomic Drivers

The Atlantic dense forest has been intensively reduced along centuries of exploitation. Dense forests were widely exploited in an unsustainable way, which was driven by the mindset of a colonial government pursuing fast economic development (Dean 1997). From the European colonizer standpoint, the perception of tropical forests was bizarre in nature and constituted a constraint to the progress. The exploitation and replacement of natural forests by exotic species, commercially viable and capable of generating a more homogeneous landscape, similar to the Old Continent, have been the modus operandi during Brazil's development (Dean 1997).

Soon after the first Portuguese ships reached the Brazilian coast in 1500, the *pau-brasil* tree (*Paubrasilia echinata* (Lam.) Gagnon, H.C. Lima & G.P. Lewis), which stands for reddish wood and gave the name to the country, was a good extracted from the Atlantic Forest and traded with Europe. This species is typical from dry coastal cactus scrubs, also occurring in tall restingas (Gagnon et al. 2016) and dense forests (Lima 2020). *Pau-brasil* wood, which provides an intense red dye, was widely logged throughout the Northeast and Southeast coast during the sixteenth century (Arruda and Piletti 1996). At the same time, the colony started the extraction of precious metals, boosting international trade through maritime exports, and, consequently, the degradation of the forest (Prado Junior 2006). Areas of Atlantic dense forest were removed for creating the first European villages and, soon after, croplands for sustaining the villages (Dean 1997).

As reviewed by Baer (2008), the sugarcane has been first introduced in 1533 in the Northeast. Then, only after the development of the colony during the late sixteenth century, this crop gained importance as the main driver of the colony's economy, and, consequently, of deforestation. In order to prepare the land for crop cultivation, the dense forests were increasingly subject to slash-and-burn by European colonizers (Dean 1997). The sugarcane cultivation directly increased the population density, not only by developing the colony but also by introducing enslaved Africans brought as the workforce to Brazilian farms. In the second half of the seventeenth century, Brazilian sugarcane production faced a decline due to marginal access to the international market and the rise of new foreigner producers (Baer 2008).

Around 1730, the coffee has been introduced into Brazil. After the Brazilian independence, in 1822, coffee production was consolidated in the Southeast region, especially in São Paulo state, where large areas of the Atlantic dense forest were converted into coffee plantations (Silva 1986). In the nineteenth century, coffee exportation had an enormous economic impact on Brazil's development, and massive areas of dense forests have been converted into croplands and urban areas (Baer 2008; Silva 1986). In the eighteenth and nineteenth centuries, the intense conversion of denses forests into coffee plantation, even over hills, has led the capital of the colony, Rio de Janeiro, to a huge crisis of water supply. This crisis encouraged a pioneer restoration action, which occurred in the Tijuca Forest in 1861 as a measure to recover the natural springs of the city (Dean 1997).

With the end of slavery in the nineteenth century, the demand for labor in coffee production, especially in the west of São Paulo state, hastened immigration movements, especially of Europeans. The arrival of immigrants promoted the development of new villages and increased the populations of the main cities of the Atlantic dense forest in the Southeast (Arruda 1981; Silva 1986). The good financial condition provided by the coffee cycle allowed the country to industrialize, helping the development of the Brazilian society but also bringing new threats to the remaining forests. The coffee cycle faced its end in the global economic crisis of 1929.

Between the 1950s and 1960s, a 4650-km-long road (BR-101), linking the Northeast and South regions of Brazil, was built. BR-101 crosses 12 states near the coast of the Atlantic Ocean and is established mostly over the Atlantic dense forest. Together with social and economic benefits, the road promoted deforestation, not only per se but also by encouraging human occupation in its surroundings along most of its course. A rare example of a stretch of the BR-101 covered by continuous Atlantic dense forest is the region between São Paulo and Paraná coasts, where this road was not built. This region currently encompasses eight protected areas, which totalize >190,000 ha of protected ecosystems (SOS Mata Atlântica and INPE 2017).

Brazil's agricultural production diversified between the 1960s and the 1990s after the creation of Embrapa, the Brazilian Agriculture and Livestock Research Company, in 1973. Soon after, an expansion of the agricultural frontiers occurred in the country, with an emphasis on monocultures like soybean, cotton, and bean, on a semi-industrial scale. There was a great advance in the Brazilian agriculture between 1960 and 1990 – the number of exported agricultural products increased from 4 to 19, and the representation of processed agricultural products increased from 16% to 80% of the total exportation (Baer 2008). Most of this production came from areas that originally were covered by the other Atlantic Forest types, as present in the other sections of this chapter. Part of the production was, and still is, exported through ports situated along the Atlantic dense forest, promoting the economy in the region as well as the urbanization degree. Nowadays, the main urban, industrial, petroleum, and port poles of Brazil, accounting for approximately 80% of the national GDP (https://cidades.ibge.gov.br/), are established in the Atlantic dense forest region.

The deforestation and fragmentation rates of the Atlantic dense forest have decreased but never stopped (SOS Mata Atlântica and INPE 2017). Two out of the ten municipalities with higher deforestation rates in a period of 30 years (1985 to 2015) in Brazil are located in the Atlantic dense forest region, more specifically in the Rio de Janeiro state. These two municipalities deforested together a total of 25,617 ha of dense forests (SOS Mata Atlântica and INPE 2017), which is a larger area than the nearby protected area (Desengano State Park) and most of the national parks protecting Atlantic dense forests. Today, common threats to the dense forests are agriculture and livestock, disordered urban sprawl, predatory exploitation of plants (timber, ornamental), illegal trade of wild animals, fragmentation of protected areas, industrialization, predatory fishing, disordered tourism, and pollution.

## Conservation Efforts

The creation of major protected areas has been crucial for the conservation of currently remaining dense forests. The first Brazilian national park, Itatiaia, was created in 1937, today encompassing 28,000 ha of Atlantic dense forest on the border between Rio de Janeiro and Minas Gerais states. Nowadays, the remaining Atlantic dense forest is mainly distributed in protected areas or in slopes hard to access. The largest continuum of Atlantic dense forest protects more than 800,000 ha of forests distributed along São Paulo, Rio de Janeiro, and Paraná states and is part of the Atlantic Forest Biosphere Reserve created by UNESCO in 1994. Some of the main protected areas of dense forests south of the Doce River are the following: Serra do Mar State Park (332,000 ha), in São Paulo; Serra da Bocaina National Park (104,000 ha), in the border between Rio de Janeiro and São Paulo; Três Picos State Park (65,000 ha), in Rio de Janeiro; Serra do Itajaí National Park (57,000 ha), in Santa Catarina; Alto Ribeira State Park (35,000 ha), in São Paulo; Superagui National Park (34,000 ha), in Paraná; Caparaó National Park (31,800 ha), in the border between Minas Gerais and Espírito Santo; and Itatiaia National Park (28,000 ha), Desengano State Park (21,000 ha), and Serra dos Órgãos National Park (20,000 ha), in Rio de Janeiro.

Governments and civil society have been working for the conservation of this world biodiversity hotspot. There are an increasing number of nongovernmental organizations (NGOs) dealing not only with biodiversity conservation but also with the socioeconomic issues indirectly involved in deforestation. The SOS Mata Atlântica (https://www.sosma.org.br) is a good example toward the conservation of the Atlantic Forest, providing annual scientific reports on deforestation, promoting environmental education, and stimulating the exercise of socio-environmental citizenship. SOS Mata Atlântica is part of the "Network of NGOs of the Atlantic Forest" (http://rma.org.br), which brings together 300 civil society organizations acting to protect the Atlantic Forest.

## Perspectives and Challenges

Millions of people depend on the ecosystem services provided by the Atlantic dense forest, because native ecosystems supply water, food, and medicinal products; control floods, landslides, and erosion; and regulate the climate. Besides the public and private protected areas, there are different initiatives to protect and restore the dense forest. For example, there is an ongoing initiative that brings together researchers, NGOs, and civil society in an effort to restore the forest and its ecosystem services called "Pact for the Restoration of the Atlantic Forest" (https://www.pactomataatlantica.org.br/), which has already restored 86,332 ha of forest (see De Siqueira et al. 2021, Chap. 18).

Banks-Leite et al. (2014) assessed the ecological benefits and economic costs of paying landowners to set aside private land for restoration in the Atlantic Forest. They found that an annual investment equivalent to 6.5% of what Brazil spends on agricultural subsidies would recover species composition and ecosystem services across farmlands to levels found inside protected areas. Therefore, a shift in the way the money is invested can change the outcome not just for biodiversity but also for human well-being.

Ecotourism, or environmentally friendly tourism, is another possibility to overcome the degradation of dense forests. For instance, an initiative funded by the Inter-American Development Bank (IDB; www.iadb.org/) in São Paulo has improved economic opportunities for local populations and increased public awareness about biodiversity conservation by increasing the numbers of visits to six state parks and providing training to local residents. Ecotourism has a high potential to provide sustenance for families surrounding dense forest areas while protecting the forest.

Agroecology practices may enable to keep economic development together with forest protection, because of its lower impact in land management (Wezel et al. 2009). Analyzing family farm sustainable production in southern Brazil, Fernandes and Woodhouse (2008) show evidence of ecological and social advantages of ecological farms, but the cost of alternative marketing arrangements for agroecology still produces a financial disadvantage. To cope with these initial disadvantages, the Brazilian government, through the Secretariat of Family Agriculture and Cooperatives (http://www.mda.gov.br/), has promoted some programs to the establishment of family farms with organic and agroecology production. The Safra Plan for Family Agriculture 2017/2020 brought low annual interest rates for the financing of organic and agroecological products in the "National Program for Strengthening Family Agriculture". The "National School Feeding Program" requires the purchase of at least 30% of the food served in public schools from family farms, which is a way to guarantee fresh and quality food in school meals while generating sustainable income for family farmers. The promotion of agroecology practices at all levels, from production to markets, could move Brazil toward more sustainable agriculture.

Brazil has a high potential to attain sustainability in using biodiversity (Marcilio-Silva and Marques 2017). Emphasis on more efficient management of existing croplands can sustain biodiversity while increasing ecosystem service provision, contributing to human well-being (Strassburg et al. 2014; Alves-Pinto et al. 2017; Marcilio-Silva et al. 2018). This potential to increase productivity in a sustainable way can be the key factor to save what remains of the Atlantic dense forest.

### *5.2.2* Araucaria *Mixed Forest*

#### Characteristics

The *Araucaria* mixed forest occurs in highlands under a climate characterized by rainfalls well distributed along the year (annual average of ca. 1,500 mm), monthly temperature averages typically below 23 °C, and common frosts during the coldest months (Rambo 1956; Maack 1968). It comprises two types of formation, namely, montane and alluvial; the montane formation occurs typically between 400 and 1000 m a.s.l., while the alluvial formation is associated with rivers and streams embedded in the montane formation (Scheer and Blum 2011). In its southern limit of distribution, *Araucaria* mixed forests occur under 400 m a.s.l. (Carlucci et al. 2011a, 2013; Wrege et al. 2017). *Araucaria* mixed forest tree communities are characterized by the dominance of *Araucaria angustifolia* (Bertol.) Kuntze, popularly known as the Brazilian pine (or Paraná pine), and high species richness of Myrtaceae, Lauraceae, and Fabaceae families (Scheer and Blum 2011).

Originally, the *Araucaria* mixed forest was spread throughout the highlands of S Brazil – in Paraná, Santa Catarina, and Rio Grande do Sul states – also occurring as forest patches in the states of São Paulo, Rio de Janeiro, and Minas Gerais (Hueck 1953). Its northernmost limit has been historically suggested as small remnants in the Caparaó National Park (Ruschi 1950), in Espírito Santo, but patches of this forest type apparently vanished during the twentieth century in the region, so that its actual northern limit has been considered the State Park of Serra do Papagaio, in Minas Gerais (Kersten et al. 2015). Its southernmost limit has been recognized as several mixed forest patches with *A. angustifolia* and *Podocarpus lambertii* Klotzsch ex Endl. scattered in the hills of the Pampa grasslands in Rio Grande do Sul (Carlucci et al. 2011a, 2013; Wrege et al. 2017). Its eastern limits are the escarpments of the southern Brazilian plateau (*Planalto Sul-Brasileiro*), while its westernmost distribution currently is in the Misiones Province, in Argentina (Hueck 1953). Historical record of a native *A. angustifolia* small (possibly vanishing) population has been reported for the Department of Alto Paraná, in eastern Paraguay (Thomas 2013), specifically in the National Reserve of Kuri'y, an area of 2000 ha created in 1973 to protect the species (Paraguay's Decree No. 30956, 1973).

The *Araucaria* mixed forest is not as rich in tree species as the Atlantic dense forests but presents high phylogenetic diversity comparatively (Duarte et al. 2014). The *Araucaria* mixed forest harbors seed plant lineages that originated during the Early Triassic (~250 Ma), such as *Araucaria* (Kershaw and Wagstaff 2001), and during the Early Cretaceous (~125–110 Ma), such as several Magnoliidae families (Feild and Arens 2007). Characteristic groups of the *Araucaria* mixed forests, such as magnoliids (e.g., Winteraceae, Canellaceae), families from other angiosperm clades (e.g., Proteaceae, Cunoniaceae), conifers (Araucariaceae and Podocarpaceae), and arborescent ferns (Dicksoniaceae and Cyatheaceae), are considered "survivors" of the Gondwanan moist forests of the Cretaceous (Kooyman et al. 2014). The presence of these so diverse and evolutionarily distantly related groups of vascular plants represents a living evolutionary heritage in the *Araucaria* mixed forests of southern Brazil. It is important to note that, despite the phylogenetic composition of *Araucaria* mixed forests differs from other types of Atlantic Forests, the species-level floristic composition is highly influenced by surrounding forest types (Duarte et al. 2014).

Evidences from palaeo-pollen research indicate that the *Araucaria* mixed forest has migrated southward in response to climatic changes of the last millennia (Behling and Pillar 2007). Several millennia ago, a drier and colder climate predominated, so that grasslands used to form almost the totality of landscapes in southern Brazil. A warmer and wetter climate in the last three millennia has promoted the expansion of *Araucaria* mixed forests over grasslands in southern Brazil (Behling and Pillar 2007). *Araucaria* mixed forests advance over the grasslands when not limited by fire or domestic cattle grazing and trampling (Oliveira and Pillar 2004). Analyses of satellite images and carbon isotopes indicated that fire and grazing retard forest expansion over grassland, which appears to occur slowly when these disturbances are present (Oliveira and Pillar 2004; Silva and Anand 2011). The seed disperser fauna plays a fundamental role in the expansion process, carrying seeds of pioneer tree species that facilitate the arrival and establishment of late successional species in the grassland (Duarte et al. 2006, 2007). The expansion of the *Araucaria* mixed forest occurs through edge dynamics (Oliveira and Pillar 2004; Carlucci et al. 2011b) or through nucleation, in which nurse plants such as the Brazilian pine or nurse objects as rocks promote the formation of forest patches on the grassland (Duarte et al. 2006, 2007; Carlucci et al. 2011c; Marcilio-Silva et al. 2015, Sühs et al. 2018).

The major devastation imposed to the *Araucaria* mixed forest has led several species to a process of extinction. Regarding plants, extinction threat may be related to utility, such as species providing timber or essential oils. This is the case of the Brazilian pine (*Araucaria angustifolia*), endangered (EN) at the national and critically endangered at the international level (Carlucci et al. 2013; Thomas 2013), which provides timber and an edible seed (*pinhão*) widely consumed by southern Brazilian people. Another example is the imbuia tree [*Ocotea porosa* (Nees & Mart.) Barroso], endangered nationally and vulnerable internationally due to excessive wood harvesting (Quinet et al. 2013; Varty and Guadagnin 1998).

## History of Degradation and Present Socioeconomic Drivers

The recognition of the *Araucaria* mixed forest for its wealth of woods dates back to the eighteenth century. However, the region kept practically unexploited through the late eighteenth century because the Serra do Mar mountain range was a major natural barrier separating the Paranaguá port from the highlands where the *Araucaria* mixed forests occur. The first steam powered sawmill of the region was established in 1872 near the city of Curitiba, focusing on the timber of the Brazilian pine, imbuia, and cedro-rosa (*Cedrela fissilis* Vell.) (Carneiro 2014). This sawmill was owned by the Paranaense Forestry Company, which was created in 1871 to start a massive exploitation of the *Araucaria* mixed forest (Companhia Florestal Paranaense 1872). In order to present the araucaria timber to the international market, the company owners remounted and exposed a 33-m-tall Brazilian pine in the international fair of Vienna, Austria, in 1873 (Carneiro 2014). This exposal possibly rendered the fame for the future exportation of Brazilian pine timber to Europe in the twentieth century, despite the Paranaense company per se has not succeeded in its ambition.

The massive devastation of the *Araucaria* mixed forest actually started after the inauguration of the Graciosa road linking Curitiba to the coastal city of Antonina in

1873 and the railway linking the city of Curitiba to the Paranaguá port in the Atlantic Ocean coast in 1885 (Castella and Britez 2004). Initially, the Brazilian pine timber had the Scots pine (or Riga pine; *Pinus sylvestris* L.) as a major competing product that was imported from Europe. The Brazilian pine timber became the main wood after the beginning of the First World War, which precluded the importation of the Scots pine to supply the Brazilian and Argentinean markets (Castella and Britez 2004). Several sawmills then settled in southern Brazil, going south and westward (Castella and Britez 2004) as railways expanded over the countryside. In 1906, there were 108 sawmills in the state of Paraná (Koch and Corrêa 2002).

During the first half of the twentieth century, the major sawmill in Latin America, the "Southern Brazil Lumber and Colonization Company," was responsible for exploring and logging the forests occurring up to 15 km at each side of the railway linking the states of São Paulo and Rio Grande do Sul (Koch and Corrêa 2002). The company was established in Três Barras, in the border between Paraná and Santa Catarina. The Brazilian pine wood was the leading timber exploited by the Lumber Company (as was broadly known), followed by the wood of imbuia and other less noble woods, accounting for 85%, 10%, and 5%, respectively, of the whole sawmill's production (Nogueira 1920). The main fate of this wood was Latin America markets, headed by the cities of São Paulo, Buenos Aires, and Rio de Janeiro (Nogueira 1920). After an expedition through the states of Paraná and Santa Catarina in 1928, the botanist Frederico Carlos Hoehne warned about the ongoing devastation of the *Araucaria* mixed forest (Hoehne 1930). From the beginning of the colonization process of the Paraná state to 1930, almost half of this forest type (34,200 out of 73,780 km$^2$) had been logged in the state of Paraná (Maack 1968).

After 1930, with the development of trucks, the deforestation went increasingly afar from railways, further penetrating in southern Brazil countryside (Koch and Corrêa 2002). During the Second World War, the exploitation of the Brazilian pine timber was boosted again (Castella and Britez 2004). The wood of the Brazilian pine was indicated by specialists as the ideal wood to help rebuild Europe (Gill 1947) as the war destroyed entire cities. Until 1940, in states like Paraná and Rio Grande do Sul, deforested areas usually gave place to croplands, mostly corn, bean, and wheat (Rambo 1956; Maack 1968). In 1941, the Brazilian government created the National Pine Institute (*Instituto Nacional do Pinho*), which had as one of its objectives to restore *Araucaria* mixed forests exploited for timber (Gill 1947). In 1948, there were 2843 sawmills registered in southern Brazil (Koch and Corrêa 2002). From 1915 to 1960, Brazil exported 18.5 billion m$^3$ of wood, almost the totality of which originated from the *Araucaria* mixed forest (Koch and Corrêa 2002).

In 1963, a major forest fire burned 9650 km$^2$ of the *Araucaria* mixed forest in Paraná state, further accentuating the devastation (Maack 1968). In 1965, 57,848 out of 73,780 km$^2$, or 78.4% of the original distribution of the forest type, had been devastated in Paraná (Maack 1968). In the same year, the first Brazilian Forest Code was published, ruling that the clear-cutting of *Araucaria* mixed forests was prohibited, unless a management plan was specifically written to explore a given area (Law No. 4771, 1965). By the end of the 1970s, the thousands of existing sawmills in southern Brazil experienced the lack of wood supply and started to end their operation (Koch and Corrêa 2002). The coffee cycle itself, which started in the 1940s and ended after a major frost event in 1975 in Paraná (Une 1982), also contributed to reduce the *Araucaria* mixed forests (Koch and Corrêa 2002). Unfortunately, the existence of federal and state laws protecting the *Araucaria* mixed forest has been often disrespected. Several foci of deforestation have still been found in the next decades (Koch and Corrêa 2002).

Analyses of satellite images have been useful to update the situation of native ecosystems worldwide. In southern Brazil, only 12.6% of the original distribution of the *Araucaria* mixed forest remained forested by 2005 (Ribeiro et al. 2009), mostly occupied by secondary forests. Assuming an area of original distribution of the *Araucaria* mixed forest in Paraná of 82,958 km$^2$, Castella and Britez (2004) analyzed satellite images and showed that there were no more original, pristine forests in the state by 1998; rather, only 0.8% of old-growth (661 km$^2$) and 29.4% of secondary *Araucaria* mixed forests under varying degrees of regeneration (24,404 km$^2$) remained by then. In Santa Catarina, satellite imagery analyses indicated that 24.4% (13,741 km$^2$) of the original distribution of the *Araucaria* mixed forest remained by 2008 in the state (Vibrans et al. 2012). In Rio Grande do Sul state, analyses showed that only 12% of the original area covered by *Araucaria* mixed forest (3592 out of 29,875 km$^2$) remained by 2002, while 85.6% of the area were converted to agricultural uses (Cordeiro and Hasenack 2009). Most of the remnant forests in this state should be secondary too. All these percentages may be lower today as deforestation kept occurring in some regions (SOS Mata Atlântica and INPE 2017).

From 1900, in the beginning of the devastation of the *Araucaria* mixed forest, to 2000, the population of southern Brazil increased from 1.8 million to 25 million people (IBGE 2007). The major demand for food and services, besides urbanization and infrastructure expansion per se, led to the conversion of most *Araucaria* mixed forests into croplands. Currently the economy of southern Brazilian states is mostly based in services (62–66%), industry (25–33%), and farming (4–9%) (IBGE 2014). Urban areas are widespread in the region, but higher human population density is found in the following top 10 largest cities and surroundings (IBGE Resolution No. 2, 28 August 2018): Curitiba-PR (>1.92 million inhabitants), Caxias do Sul-RS (>504,000), Ponta Grossa-PR (>348,000), Cascavel-PR (>324,000), São José dos Pinhais-PR (>317,000), Colombo-PR (>241,000), Chapecó-SC (>217,000), Passo Fundo-RS (>202,000), Guarapuava-PR (>180,000), and Lages-SC (>158,000).

## Conservation Efforts

Pioneer environmentalists and scientists such as Romário Martins, Frederico Carlos Hoehne, Reinhard Maack, and Balduíno Rambo, have made several warnings on the risks of overexploitation and attempts to protect *Araucaria* mixed forests since the early twentieth century (Hoehne 1930; Rambo 1956; Maack 1968; Carneiro 2014). The warnings included risks of soil degradation and extinction of species or of the whole forest type, while attempts of protection included the creation of major protected areas. In this sense, it is worth to mention that the Aparados da Serra National Park (13,000 ha), protecting canyons, *Araucaria* mixed forests, and native grasslands in the border between Rio Grande do Sul and Santa Catarina, was created in 1959 after Balduíno Rambo's plea. Unfortunately, most of the efforts of these pioneers were ignored, and laws kept encouraging massive deforestation (Koch and Corrêa 2002).

Establishing and monitoring protected areas is still the best way to protect large representative areas of native ecosystems, as large tracts of old-growth native forests remain not formally protected, but are still subject to unsustainable use. In the 2000s, a few protected areas were created to conserve some of the last remnants representative of the forest type (Carneiro 2014). Two national parks (Campos Gerais and Araucárias), two biological reserves (Araucárias and Perobas), one ecological station (Mata Preta), and one wildlife refuge (Campos de Palmas) were created between 2005 and 2006 in the states of Paraná and Santa Catarina, totalizing 809.2 km$^2$ (ICMBio 2019). Some priority areas in the center-south region of Paraná and west of Santa Catarina were deforested as landowners knew about the project for creation of protected areas (MMA 2005). In 2009, a major operation was held in Paraná and found an illegal network of deforestation in those areas (Carneiro 2014). Continuous exploitation of the last remnants of old-growth *Araucaria* mixed forests located in center-south Paraná has occurred since then notwithstanding (IBAMA 2018).

Analyses of satellite imagery indicated that only 3.1% of the remaining *Araucaria* mixed forests were protected by 2007 (Ribeiro et al. 2009). This scenario has likely not changed ever since, because no major protected areas were created in the region after 2005/2006. Several conservation priority areas in the *Araucaria* mixed forest still lacks legal protection (MMA 2007a) or at least an assurance that they will be protected by landowners. For instance, a study conducted by the Ministry of the Environment suggested the creation of a major protected area (Wildlife Refuge of Corredor do Rio Pelotas, with 2,730 km$^2$ of *Araucaria* mixed forest-grassland mosaics) between Santa Catarina and Rio Grande do Sul (MMA 2007b). Although this region has been threatened by damming for hydroelectric power stations (Mähler-Junior and Larocca 2009) and the study for creation of a protected area is ready for more than a decade, the region is not formally protected yet.

Another major problem is that most of protected areas already created were not fully implemented, so that many landowners have not been paid for their lands and, in most cases, still live inside strictly protected areas like national parks (Araujo 2016). As a consequence, several protected areas have often been subject to attempts of downgrading, downsizing, and degazettement (sensu Mascia and Pailler 2011). A legal alternative to solve this problem is the "compensation for legal reserve" (Law of Protection of the Native Vegetation, Brazil's Federal Law No. 12.651, 2012), in which landowners in deficit with areas of native vegetation inside their properties (20% of each property within the Atlantic Forest domain) may pay for areas within protected areas of the same biome, thereby enabling the government to buy the lands of the protected area from prior landowners with the money of compensation.

### Perspectives and Challenges

The major reduction suffered by the *Araucaria* mixed forest imposes big challenges to conserve their species and ecosystems in the long run. Considering the vast area deforested and that many landscapes within the original distribution lack forest remnants (Tambosi et al. 2014), the restoration of *Araucaria* mixed forests becomes urgent. A study suggests that, despite the major populational reduction suffered, the Brazilian pine possesses sufficient genetic diversity among their populations for conservation and restoration (Stefenon et al. 2008). Climate changes impose additional challenges to species and ecosystem conservation in the modern world (Ackerly et al. 2010). In this sense, it is important to evaluate whether the geographical distribution of native species projected to future climate scenarios will coincide or not with existing protected areas or areas to be restored. In synthesis, the conservation of *Araucaria* mixed forest urges that (1) new protected areas are created in the last large remnants representative of the formation, (2) the existing protected areas are implemented and properly monitored, and (3) degraded lands within the original distribution are restored according to specific legislation and current scientific knowledge.

## *5.2.3 Atlantic Seasonal Forest*

### Characteristics

Atlantic seasonal forests covered large mid-elevation inland areas, where a combination of high temperatures and marked rainfall seasonality is associated with some degree of deciduousness in forest trees. Such deciduousness varies according to local combinations of rainfall and soil. In the Brazilian vegetation map, the semideciduous seasonal forests are described as a vegetation in which 20 to 50% of the tree individuals lose their leaves in the dry or cool Southern Hemisphere winter conditions (IBGE 2012). Small remnants of deciduous seasonal forests, where loss of leaves occurs in over 50% of the trees, occur scattered along all major Brazilian regions, associated with rock outcrops and shallow often limestone soils. In spite of such simple classification, several regional forms of Atlantic semideciduous seasonal forests have been recognized, based on edaphic, climatic, and biogeographical features.

Silva and Casteleti (2003) proposed a classification of the Atlantic Forest into eight biogeographical sub-regions, based on bird, butterfly, and primate data. In this classification, the forests with the term “Interior” (meaning inland) are comprised entirely of Atlantic seasonal forests. There are Atlantic seasonal forests also in other sub-regions, for instance, in the northeastern São Francisco basin.

The Interior forests used to be the largest Atlantic Forest (49% of its total area). There were large blocks of Interior forests in W and N Paraná, W São Paulo, E Mato Grosso do Sul, and E Minas Gerais states. In spite of such a huge representation in original Atlantic Forest vegetation, Interior forests, and therefore Atlantic seasonal forests, are the most threatened Atlantic Forest vegetation type in southern and southeastern Brazil. Only 7% of Interior Atlantic seasonal forests remained by 2005 (against 11.3% of the Atlantic Forest as a whole; Ribeiro et al. 2009), and only 6.8% of these remnants are protected (0.8% of the original cover).

Ribeiro et al. (2011) refined Silva and Casteleti (2003) mapping, subdividing the 8 biogeographical units in 55 sectors, by incorporating bioclimatic and elevation data. For the Silva and Casteleti's Interior seasonal forests, 25 sectors were recognized, reflecting the large area and the heterogeneity in soil and regional climates in which the Atlantic seasonal forests are placed.

Scheer and Blum (2011) reviewed a number of tree inventories, restricted to Paraná state, including the three major types of the southern Atlantic Forest, and suggested that the Atlantic seasonal forests have more tree species (282, in 11.8 ha in 10 sites) than *Araucaria* mixed forests (220, in 8.3 ha in 19 sites) and less tree species than the Atlantic dense forests along the coast (469, 9.1 ha in 29 sites). The authors, however, did not account for different sample sizes and tree size sampling criteria (usually reported in stem diameter), so these numbers could change after standardizing sample size and sampling criteria. Oliveira-Filho and Fontes (2000), using a large dataset from several southern Brazilian states and Paraguay, also pointed to higher richness in Atlantic dense forests than in Atlantic seasonal forests. They suggested that the Atlantic seasonal forest flora is a subset of drought-resistant species from Atlantic dense forests but also holds some floristic influence from Cerrado. While Atlantic dense forests can present consistently high average local species richness and alpha diversity, Atlantic seasonal forests have a rich flora, which is possibly explained by beta diversity patterns across the Atlantic seasonal forest distribution. Such high beta diversity can be seen in the datasets used by Scheer and Blum (2011) and Oliveira-Filho and Fontes (2000) and might be explained by rainfall gradients, soil heterogeneity, and influence of other vegetation types (e.g., Cerrado). Atlantic seasonal forests also show floristic differentiation along a latitudinal gradient, associated with temperature.

## History of Degradation and Present Socioeconomic Drivers

The economic drivers of deforestation of Atlantic seasonal forests are diverse, and took place at different moments of eastern Brazilian development (Dean 1997), but converge nowadays in posing further threats to forest biodiversity by bringing together landscape alteration and disruptive human activities. On one hand, Minas Gerais and São Paulo states share (at least) a century-old history of forest conversion, mainly in coffee plantations and pasture lands. Currently, however, sugarcane plantations in São Paulo and iron mining in Minas Gerais appear to be the strongest drivers of change. On the other hand, Atlantic seasonal forests in Paraná, Santa Catarina, and Rio Grande do Sul states were subject to exploitation later. For instance, Paraná state had its full Atlantic seasonal forest cover until early twentieth century. A fast, strong deforestation process occurred in a few decades after the

1930s, firstly to replace the decadent coffee production from São Paulo and Minas Gerais and then to give place to the “soybean system,” which was consolidated in the 1970s. This system consists of a soybean-maize-wheat rotation culture very intensive in fertilizers and pesticides. Indeed, there were replacements of coffee plantations in (usually small) family farms for soybean cultivation in large estates, which led to another wave of conversion of forests on arable land into croplands. This more recent cycle of deforestation reduced the forest cover to as low as 2% in some landscapes of N Paraná (IPARDES 1993).

## Conservation Efforts

Fragmentation and reduction of Atlantic seasonal forests were widespread throughout states and regions, leaving no large tracts such as the dense forests of the Serra do Mar. For instance, Vicente et al. (2009) showed that Atlantic seasonal forests in Paraná state were reduced from 37.7% to 8.2% of the state area and that only 3.3% of the remnant vegetation is formally protected. Few protected areas in the Atlantic seasonal forest region are larger than a few thousand hectares. The most remarkable protected areas harboring seasonal forests are the national parks of Iguaçu in Brazil and its Argentinean neighbor Iguazú (Table 5.1), which form a major block of continuous forest.

In some places, Atlantic seasonal forest remnants comprise less than 1% of the original cover of the “county” (set of neighbor municipalities), as in the fertile lands of N Paraná region (IPARDES 1993), where most of the remnant forest patches are smaller than 10 ha (Torezan 2004). This pattern of high habitat loss (usually over

**Table 5.1** Largest seasonal Atlantic forest protected areas in Brazil and neighbor countries of Argentina and Paraguay

| Protected area | Location | Area (thousands ha) |
|---|---|---|
| Iguaçu NP | Paraná, Brazil | 185.3 |
| Ilha Grande NP | Paraná/ Mato Grosso do Sul, Brazil | 78.9 |
| Iguazú NP | Argentina | 67.6 |
| Rio Doce SP | Minas Gerais, Brazil | 37 |
| Morro do Diabo SP | São Paulo, Brazil | 33.8 |
| Turvo SP | Rio Grande do Sul, Brazil | 17.5 |
| Caaguazú NP | Paraguay | 16 |
| Itabó NP | Paraguay | 15.2 |
| Limoy NP | Paraguay | 14.8 |
| Cerro Corá NP | Paraguay | 12 |
| Aguapeí SP | São Paulo, Brazil | 9 |
| Perobas BR | Paraná, Brazil | 8.7 |
| Ybycuí NP | Paraguay | 3.8 |
| Ñacunday NP | Paraguay | 1.7 |

*BR* biological reserve, *NP* national park, *SP* state park

90%) and strong fragmentation is the rule in the economically dynamic, highly populated inlands of Minas Gerais, São Paulo, and Paraná states (Ribeiro et al. 2009).

### Perspectives and Challenges

The fate of forest remnants amidst the sugarcane and soybean plantations challenges our commitment to conserve Atlantic seasonal forest ecosystems. Landscape connectivity is deeply impaired, and edge effects are increased by the strong contrast between monocultures and the forest, while the effects of huge amounts of fertilizers and pesticides are still out of the bounds of ecological science. In the policy front, while Serra do Mar dense forests lie in a relatively undisputed place devoted to conservation, most of the Atlantic seasonal forest range is also the home of large-scale agribusiness, where each square meter of arable land is contested, which turns conservation of seasonal Atlantic forests a grueling task.

## 5.3 Conclusions

The history of use and degradation of southern Atlantic Forest types holds some similarities but also several idiosyncrasies. Among the similarities, the mindset of colonizers toward the exploitation of natural resources has driven unsustainable deforestation along centuries. Among the idiosyncrasies, we observe that the Atlantic dense forests were the first to be severely impacted by European settlers occupying the coast of Brazil. However, because of a predominance of steep relief, dense forests constitute the largest remnants nowadays in the southern Atlantic Forest. The *Araucaria* mixed forests were severely degraded mostly since the First World War and mainly due to the value of its woods, especially the Brazilian pine. The Interior seasonal forests also have a one-century history of large-scale conversion in states like Minas Gerais and São Paulo due to agriculture but a more recent degradation in the west of the South region of Brazil (last 50 years), after the expansion of agriculture frontiers on fertile soils.

Because of the history of use and degradation of the Atlantic Forest, the existing forest remnants, especially still existing old-growth forests outside protected areas, urge conservation actions. Increasing formal protection in public and private areas, as well as promoting policies for the sustainable use of the Atlantic Forest, is important to conserve not only biodiversity but also ecosystem service provisioning. Most of the Brazilian population lives within the Atlantic Forest boundaries and thus directly depends on basic ecosystem services provided by the forest, such as water supply, climate regulation, and prevention of natural hazards. Agriculture and livestock production can also take advantage of a sustainable management, with compliance to the environmental law and efforts to diminish impacts on the remaining forest. The compliance to current environmental law will also demand large-scale restoration inside private lands in the next years. Importantly, while several landscapes may rely on natural regeneration because they still harbor a sufficient area of forests, active restoration should take place in most of the severely degraded landscapes (i.e., lacking propagule source areas).

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## References

Ackerly DD, Loarie SR, Cornwell WK et al (2010) The geography of climate change: implications for conservation biogeography. Divers Distrib 16:476–487

Alves-Pinto HN, Latawiec AE, Strassburg BB et al (2017) Reconciling rural development and ecological restoration: strategies and policy recommendations for the Brazilian Atlantic Forest. Land Use Policy 60:419–426

Araujo SCDB (ed) (2016) Anais do 1° Seminário de Pesquisas do Parque Nacional dos Campos Gerais e da Reserva Biológica das Araucárias. ICMBio, Ponta Grossa

Arruda BKG (1981) Padrões alimentares da população brasileira. Instituto Nacional da Alimentação e Nutrição, Brasília

Arruda JJDA, Piletti N (1996) Toda a história. Ática, São Paulo

Baer W (2008) A economia brasileira, 6th edn. Nobel, São Paulo, p 416

Banks-Leite C, Pardini R, Tambosi LR et al (2014) Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot. Science 345:1041–1045

Behling H, Pillar VD (2007) Late Quaternary vegetation, biodiversity and fire dynamics on the southern Brazilian highland and their implication for conservation and management of modern *Araucaria* forest and grassland ecosystems. Philos T Roy Soc B 362:243–251

Carlucci MB, Jarenkow JA, Duarte LDS, Pillar VDP (2011a) Conservação da Floresta com Araucária no extremo sul do Brasil. Nat Conservacao 9:111–114

Carlucci MB, Teixeira FZ, Brum FT, Duarte LDS (2011b) Edge expansion of Araucaria forest over southern Brazilian grasslands relies on nurse plant effect. Community Ecol 12:196–201

Carlucci MB, Duarte LDS, Pillar VD (2011c) Nurse rocks influence forest expansion over native grassland in southern Brazil. J Veg Sci 22:111–119

Carlucci MB, Prieto PV, Hering RLO et al (2013) Araucariaceae. In: Martinelli G, Moraes MA (eds) Livro Vermelho da Flora do Brasil. Jardim Botânico do Rio de Janeiro, Rio de Janeiro, pp 185–186

Carnaval AC, Moritz C (2008) Historical climate modelling predicts patterns of current biodiversity in the Brazilian Atlantic forest. J Biogeogr 35:1187–1201

Carneiro JAS (2014) Carta ao futuro de Araucarilândia. In: Hoehne FC (ed) Araucarilândia – Observações geraes e contribuições ao estudo da flora e phytophysionomia do Brasil. Companhia Melhoramentos de São Paulo, São Paulo

Castella PR, Britez RM (eds) (2004) A Floresta com Araucária no Paraná: conservação e diagnóstico dos remanescentes florestais. FUPEF/PROBIO. 233 p. Ministério do Meio Ambiente, Brasília

Companhia Florestal Paranaense (1872) Companhia Florestal Paranaense. Companhia Florestal Paranaense, Rio de Janeiro

Cordeiro JLP, Hasenack H (2009) Cobertura vegetal atual do Rio Grande do Sul. In: Pillar VD, Müller SC, Castilhos ZMS, Jacques AVA (eds) Campos Sulinos: Conservação e uso sustentável da biodiversidade, 1st edn. Ministério do Meio Ambiente, Brasília, pp 285–299

De Siqueira LP, Tedesco AM, Rodrigues RR, Chaves RB, Albuquerque NCB, Corrêa FF, Santiami EL, Tambosi LR, Guimarães TMG, Brancalion PHS (2021) Engaging people for large-scale forest restoration: governance lessons from the Atlantic Forest of Brazil. In: M. C. M. Marques,

C. E. V. Grelle (eds.), Atlantic Forest: biodiversity, threats and solutions of the megadiverse forest. Springer, Switzerland

Dean W (1997) With broadax and firebrand: the destruction of the Brazilian Atlantic Forest. University of California Press, Berkeley

Duarte LDS, Machado RE, Hartz SM, Pillar VD (2006) What saplings can tell us about forest expansion over natural grasslands. J Veg Sci 17:799–808

Duarte LDS, Carlucci MB, Hartz SM, Pillar VD (2007) Plant dispersal strategies and the colonization of Araucaria forest patches in a grassland-forest mosaic. J Veg Sci 18:847–858

Duarte LDS, Bergamin RS, Marcilio-Silva V et al (2014) Phylobetadiversity among forest types in the Brazilian Atlantic Forest Complex. PLoS One 9:e105043

Feild TS, Arens NC (2007) The ecophysiology of early angiosperms: early angiosperm ecophysiology. Plant Cell Environ 30:291–309

Fernandes LADO, Woodhouse PJ (2008) Family farm sustainability in southern Brazil: an application of agri-environmental indicators. Ecol Econ 66:243–257

Gagnon E, Bruneau A, Hughes CE et al (2016) A new generic system for the pantropical Caesalpinia group (Leguminosae). PhytoKeys 71:1–160

Gill T (1947) Parana Pine – a source of wood for reconstruction in Europe. Unasylva 1(2)

Hoehne FC (1930) Araucarilandia. Observações geraes e contribuições ao estudo da flora e phytophysionomia do Brasil. Companhia Melhoramentos de São Paulo, São Paulo

Hueck K (1953) Distribuição e habitat natural do pinheiro-do-Paraná (*Araucaria angustifolia*). Bol Fac Filos Cienc Letr USP Bot 10:709–723

IBAMA (2018) Ibama embarga 10 empresas por exploração ilegal de araucárias no PR. Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis. http://www.ibama.gov.br/noticias/436-2018/1432-ibama-embarga-10-empresas-por-exploracao-ilegal-de-araucarias-no-pr. Accessed on 07 May 2019

IBGE (1992) Manual técnico da vegetação brasileira. Série Manuais Técnicos em Geociências, n.1. IBGE, Rio de Janeiro

IBGE (2007) Estatísticas do Século XX. Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro

IBGE (2012) Manual Técnico da Vegetação Brasileira, 2nd ed, Manuais Técnicos em Geociências núm 1. Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro

IBGE (2014) Contas Regionais do Brasil – 2012. Instituto Brasileiro de Geografia e Estatística, Rio de Janeiro

ICMBio (2019) Unidades de Conservação – Mata Atlântica. Instituto Chico Mendes de Conservação da Biodiversidade. http://www.icmbio.gov.br/portal/unidadesdeconservacao/biomas-brasileiros/mata-atlantica/unidades-de-conservacao-mata-atlantica. Accessed on 07 May 2019

IPARDES (1993) Cobertura florestal e consumo de madeira, lenha e carvão nas microrregiões de Londrina, Maringá e Paranavaí: subsídio para uma política florestal no Estado do Paraná. Instituto Paranaense de Desenvolvimento Econômico e Social, Curitiba

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

Kershaw P, Wagstaff B (2001) The southern conifer family Araucariaceae: history, status, and value for paleoenvironmental reconstruction. Annu Rev Ecol Syst 32:397–414

Kersten RA, Borgo M, Galvão F (2015) Floresta Ombrófila Mista: aspectos fitogeográficos, ecológicos e métodos de estudo. In: Eisenlohr PV, Felfili JM, JAA M-N (eds) Fitossociologia no Brasil: métodos e estudos de caso, 1st edn. UFV, Viçosa, pp 156–182

Koch Z, Correa MC (2002) Araucária: a floresta do Brasil meridional. Olhar Brasileiro, Curitiba

Kooyman RM, Wilf P, Barreda VD et al (2014) Paleo-Antarctic rainforest into the modern Old World tropics: the rich past and threatened future of the 'southern wet forest survivors'. Am J Bot 101:2121–2135

Liebsch D, Marques MCM, Goldenberg R (2008) How long does the Atlantic Rain Forest take to recover after a disturbance? Changes in species composition and ecological features during secondary succession. Biol Conserv 141:1717–1725

Lima HC. *Paubrasilia.* In Flora do Brasil 2020 under construction. Jardim Botânico do Rio de Janeiro. Available at: http://floradobrasil.jbrj.gov.br/reflora/floradobrasil/FB602728. Accessed on 01 Nov 2019

Maack R (1968) Geografia física do Estado do Paraná. Banco de Desenvolvimento do Paraná, Curitiba

Mähler-Junior JKF, Larocca JF (2009) Fitofisionomias, desmatamento e fragmentação da Floresta com Araucária. In: Fonseca CR, Souza AF, Leal-Zanchet AM et al (eds) Floresta com Araucária – ecologia, conservação e desenvolvimento sustentável. Holos, Ribeirão Preto

Marcilio-Silva V, Cavalin PO, Varassin IG et al (2015) Nurse abundance determines plant facilitation networks of subtropical forest-grassland ecotone: abundance determines plant facilitation network. Austral Ecol 40:898–908

Marcilio-Silva V, Zwiener VP, Marques MCM (2017) Metacommunity structure, additive partitioning and environmental drivers of woody plants diversity in the Brazilian Atlantic Forest. Diversity Distrib 23:1110–1119

Marcilio-Silva V, Marques MCM (2017) New paradigms for Atlantic Forest agriculture and conservation. Biodiversity 18:201–205

Marcilio-Silva V, Marques MC, Cavender-Bares J (2018) Land use trade-offs between tree biodiversity and crop production in the Atlantic Forest. Conserv Biol 32:1074–1084

Mascia MB, Pailler S (2011) Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. Conserv Lett 4:9–20

MMA (2005) Devastação avança em áreas destinadas à preservação da araucária. Ministério do Meio Ambiente. http://www.mma.gov.br/informma/item/2602-devastacao-avanca-em-areas-destinadas-a-preservacao-da-araucaria.html. Accessed on 07 May 2019

MMA (2007a) Áreas prioritárias para conservação, uso sustentável e repartição de benefícios da biodiversidade brasileira – atualização – Portaria MMA n° 9, de 23 de janeiro de 2007. Ministério do Meio Ambiente, Brasília

MMA (2007b) Proposta de unidade de conservação de proteção integral, na categoria de Refúgio de Vida Silvestre, formando corredor ecológico, no Rio Pelotas e nos Campos de Cima da Serra, sul do Brasil – Relatório Técnico. Ministério do Meio Ambiente, Brasília

Mori SA, Boom BM, Prance GT (1981) Distribution patterns and conservation of eastern Brazilian coastal forest tree species. Brittonia 33:233–245

Nogueira J (1920) Do Rio Iguassú e ao Guayra. Editora Carioca, Rio de Janeiro

Oliveira JM, Pillar VD (2004) Vegetation dynamics on mosaics of Campos and *Araucaria* forest between 1974 and 1999 in Southern Brazil. Community Ecol 5:197–202

Oliveira-Filho A, Tameirão Neto E, Carvalho W et al (2005) Análise floristica do compartimento arbóreo de áreas de Floresta Atlântica sensu lato na região das Bacias do Leste (Bahia, Minas Gerais, Espírito Santo e Rio de Janeiro). Rodriguésia 56:185–235

Oliveira-Filho AT, Fontes MAL (2000) Patterns of floristic differentiation among Atlantic Forests in Southeastern Brazil and the influence of climate. Biotropica 32:793–810

Olson DM, Dinerstein E, Wikramanayake ED et al (2001) Terrestrial ecoregions of the world: a new map of life on Earth. Bioscience 51:933–938

Prado Junior C (2006) História Econômica do Brasil. Brasiliense, São Paulo

Quinet A, Kutschenko DC, Barros FSM et al (2013) Lauraceae. In: Martinelli G, Moraes MA (eds) Livro vermelho da flora do Brasil. Instituto de Pesquisas Jardim Botânico do Rio de Janeiro, Rio de Janeiro, p 1100

Rambo B (1956) A fisionomia do Rio Grande do Sul, 2nd edn. Selbach, Porto Alegre

Rezende CL, Scarano FR, Assad ED et al (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:208–214

Ribeiro MC, Metzger JP, Martensen AC et al (2009) The Brazilian Atlantic Forest: how much is left, and how is the remaining forest distributed? Implications for conservation. Biol Conserv 142:1141–1153

Ribeiro MC, Martensen AC, Metzger JP et al (2011) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In: Zachos F, Habel J (eds) Biodiversity hotspots. Springer, Berlin

Ruschi A (1950) Fitogeografia do Estado do Espírito Santo – Considerações gerais sobre a distribuição da flora no Estado do Espírito Santo. Bol. Mus. Biol. Mello Leitão Bot 1:1–384

SanMartín I, Ronquist F (2004) Southern Hemisphere biogeography inferred by event-based models: plant versus animal patterns. Syst Biol 53:216–243

Scheer MB, Blum CT (2011) Arboreal diversity of the Atlantic Forest of southern Brazil: from the beach ridges to the Paraná River. In: Grillo O (ed) The dynamical processes of biodiversity – case studies of evolution and spatial distribution. InTechOpen, London

Silva JMC, Casteleti CH (2003) Status of the biodiversity of the Atlantic forest of Brazil. In: Galindo-Leal C, Câmara IG (eds) The Atlantic Forest of South America: biodiversity status, threats, and outlook. Island Press, Washington-DC, pp 43–59

Silva LCR, Anand M (2011) Mechanisms of *Araucaria* (Atlantic) forest expansion into southern Brazilian grasslands. Ecosystems 14:1354–1371

Silva S (1986) Expansão cafeeira e origens da indústria no Brasil. Alfa-Omega, São Paulo

Soares FS, Medri ME (2002) Alguns aspectos da colonização da bacia do rio Tibagi. In: Medri ME, Pimenta JA, Shibatta OA (eds) Bianchini E. A bacia do rio Tibagi, Londrina, pp 69–79

SOS Mata Atlântica, INPE (2017) Atlas dos remanescentes florestais da Mata Atlântica, período de 2015 a 2016. SOS Mata Atlântica/Instituto Nacional de Pesquisas Espaciais, São Paulo/São José dos Campos

Stefenon VM, Gailing O, Finkeldey R (2008) Genetic structure of plantations and the conservation of genetic resources of Brazilian pine (*Araucaria angustifolia*). Forest Ecol Manag 255:2718–2725

Strassburg BB, Latawiec AE, Barioni LG et al (2014) When enough should be enough: improving the use of current agricultural lands could meet production demands and spare natural habitats in Brazil. Global Environ Chang 28:84–97

Sühs RB, Giehl ELH, Peroni N (2018) Interaction of land management and araucaria trees in the maintenance of landscape diversity in the highlands of southern Brazil. PLoS One 13:e0206805

Tambosi LR, Martensen AC, Ribeiro MC, Metzger JP (2014) A framework to optimize biodiversity restoration efforts based on habitat amount and landscape connectivity: optimizing restoration based on landscape resilience. Restor Ecol 22:169–177

Thomas P (2013) *Araucaria angustifolia*. The IUCN Red List of Threatened Species 2013: e.T32975A2829141. https://doi.org/10.2305/IUCN.UK.2013-1.RLTS.T32975A2829141.en. Accessed on 04 Mar 2019

Torezan JMD (2004) Estrutura de paisagens fragmentadas e prioridades para conservação da biodiversidade [Structure of fragmented landscapes and priorities for biodiversity conservation]. In: Disperati AA, Santos JR (eds) Aplicações de geotecnologias na engenharia florestal. Fupef, Curitiba, pp 138–149

Une MY (1982) An analysis of the effects of frosts on the principal coffee areas of Brazil. GeoJournal 6:129–140

Varty N, Guadagnin DL (1998) Ocotea porosa. The IUCN Red List of Threatened Species 1998: e.T32978A9739985. https://doi.org/10.2305/IUCN.UK.1998.RLTS.T32978A9739985.en. Accessed on 04 Mar 2019

Vibrans AC, McRoberts RE, Lingner DV et al (2012) Diversidade e conservação dos remanescentes florestais. In: Vibrans AC, Sevegnani L, Gasper AL, Lingner DV (eds) Inventário Florístico Florestal de Santa Catarina, vol 1. Edifurb, Blumenau, pp 65–76

Vicente RF, Vanzela ALL, Torezan JMD (2009) Effectiveness of Paraná State conservation units system, South Brazil. Nat Conservacao 7:166–180

Wezel A, Bellon S, Doré T et al (2009) Agroecology as a science, a movement and a practice: a review. Agron Sustain Dev 29:503–515

Wrege MS, Fritzsons E, Soares MTS et al (2017) Distribuição natural e habitat da araucária frente às mudanças climáticas globais. Pesquisa Florestal Brasileira 37:331–346

# Part II
# Biodiversity

# Chapter 6
# Tree Diversity in the Brazilian Atlantic Forest: Biases and General Patterns Using Different Sources of Information

**Victor P. Zwiener, Renato A. F. de Lima, Andrea Sánchez-Tapia, Diogo S. B. Rocha, and Marcia C. M. Marques**

**Abstract** The Atlantic Forest is one of the most important biodiversity hotspots, yet only a few studies have attempted to summarize tree diversity patterns across its full extent. With the increasing availability of primary biodiversity data, such a synthesis is potentially feasible; however, a critical assessment of the available information is needed to understand the limitations of data and increase knowledge on broad-scale biodiversity patterns. Here we (i) explore the potential limitations and biases of both herbarium and inventory data, (ii) provide a synthesis of diversity patterns, and (iii) present a spatial prioritization, based on complementary scenarios, for sampling tree species across the domain. We show that despite the large amounts of herbarium data, the number of unique localities per species is rather small (median, 53) and data for most species is still scarce or not yet ready for use. The spatial patterns of both herbarium records and inventory data are influenced by the presence of protected areas, proportion of forest cover, distance to graduate programs, and variables representing ease of access. Species richness presented peaks in the mid-portion of the domain. Such a pattern is related to spatial and historical constraints, environmental variation, and influence from other phytogeographical domains.

**Keywords** Herbarium records · Forest inventory · Data leakage · Sampling bias · Biodiversity synthesis · Spatial sampling priorities

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## 6.1 Introduction

The Atlantic Forest is one of the most species-rich regions in the world for many organisms, including trees. For instance, nearly 7000 species of trees and shrubs occur in this biodiversity hotspot, of which ca. 50% are endemic to the domain (Zappi et al. 2015). The composition and distribution of Atlantic Forest trees have been studied for centuries now. The first records of tree species date back to the eighteenth and nineteenth centuries, registered during expeditions carried in Brazil by European naturalists. At first, the information of Atlantic Forest tree diversity was restricted to herbarium records and taxonomic descriptions, but in the middle of the twentieth century, biodiversity information also became available from different sources with the publication of the first quantitative forest inventories in the Atlantic Forest (Davis 1945; Veloso 1945, 1946; Cain et al. 1956; Veloso and Klein 1957, 1968). Since then, such information has become increasingly available in both herbarium and forest inventory data.

Despite the long history of botanical and ecological studies in the Atlantic Forest, there are surprisingly few syntheses of tree diversity patterns. Examples focus mostly in assessing patterns and environmental correlates of tree species composition (Oliveira-Filho and Fontes 2000; Eisenlohr and Oliveira-Filho 2015; Rezende et al. 2015; Neves et al. 2017; Marcilio-Silva et al. 2017; Zwiener et al. 2020) but rarely focus on other measures of diversity (i.e., species richness, Zwiener et al. 2020) or do not span the entire domain (Oliveira-Filho et al. 2013; Duarte et al. 2014; Cerqueira and Martins 2015). Recent evaluations spanning all the Atlantic Forest domain were based only on herbarium georeferenced data from a restricted set of species (Werneck et al. 2011) or on distribution models projected to different scenarios (Murray-Smith et al. 2009; Zwiener et al. 2017, 2018). Such studies are fundamental to a better understanding of local and regional processes that generate and maintain tree diversity in the Atlantic Forest, but a comprehensive synthesis based on multiple sources of information is still lacking, constraining our knowledge and conservation actions in one of the most important biodiversity hotspots.

The reason why we still lack synthesis of broad-scale diversity patterns for trees in the Atlantic Forest is unknown but unlikely related to a lack of primary biodiversity information. Today, primary biodiversity data, defined as information that places a specific taxon at a given time and location (Sousa-Baena et al. 2013), is available in large quantities in both herbarium (Sousa-Baena et al. 2013; Peterson et al. 2018) and forest inventory data (Lima et al. 2015). Herbarium records have long been organized and maintained by the systematic community, and most databases are increasingly becoming digitalized and openly available (Zappi et al. 2015; GBIF 2016; Silva et al. 2017; CRIA 2019; REFLORA 2019). However, the initiatives to compile data from forest inventories are more recent (Oliveira-Filho and Ratter 1994; Bergamin et al. 2015; Oliveira-Filho 2017). In the Atlantic Forest, most forest inventory data still remains scattered in hundreds of studies, many of which are not published in scientific journals or available in data repositories, making it difficult to synthesize biodiversity patterns (Lima et al. 2015). Given the astonishing number of species and the overwhelming task of compiling, carefully checking, and cleaning datasets from different sources of information, it may be hypothesized as a reason for the scarcity of broad-scale studies in the Atlantic Forest.

Furthermore, the available data is not always ready for use. The difference between the full set of primary data and data that are available and usable for science and policy applications is often called "data leakage" (Peterson et al. 2018), and it is mainly driven by a lack or inaccuracy of sampling, data digitalization, species identification, georeferencing, and open availability (Peterson et al. 2018). Additionally, many factors, such as detectability and proximity to access routes, may lead to spatial variation in the presence and intensity of sampling, a phenomenon known as sampling bias (Sheth et al. 2008; Oliveira et al. 2016). Sampling bias may directly affect broad-scale estimates of species richness, endemism, and beta-diversity (Yang et al. 2013). Therefore, a critical assessment of available biodiversity information is needed to understand potential shortfalls of knowledge, sampling bias, sources of data leakage, and, ultimately, broad-scale biodiversity patterns.

In this chapter, we first explore the biases of both herbarium and inventory data, and then we attempt to synthesize the current patterns of tree diversity and richness for the entire Atlantic Forest, based on multiple sources of biodiversity information. We also assess the amount of data leakage for herbarium records and potential effects of spatially explicit factors in generating sampling bias in primary biodiversity data. Finally, we provide a pioneer attempt to identify priority areas for sampling under different scenarios, in order to guide future efforts toward the collection of primary biodiversity data in the Atlantic Forest.

## 6.2 Primary Biodiversity Data

We describe knowledge gaps and patterns of tree diversity based on information available in herbarium collections and forest inventories. We consider trees as free-standing woody plants with an adult height greater than 4 meters and/or diameter at breast height (DBH) higher or equal to 5 cm, including non-woody species with tree-like growth forms such as some cacti, palms, and ferns.

### *6.2.1 Herbarium Data*

To summarize primary biodiversity information available in herbaria and natural history museums, we compiled occurrence data of tree species that were registered in forest inventories and checklists across the Brazilian Atlantic Forest domain. We only considered information that was published in scientific journals or that was available in master and PhD dissertations (Lima et al. 2015; Zwiener et al. 2017; Zwiener et al. 2020), and we consider our compilation of herbarium data a representative subset of all forest inventory datasets. The complete checklist was searched for spelling errors, incompatible homonyms, varieties, and unambiguous synonyms in the following electronic databases: “Flora do Brasil 2020” (floradobrasil.jbrj.gov.br), Tropicos (tropicos.org), and The Plant List (theplantlist.org).

The result was a list of 2906 accepted species from 110 botanical families. For each species, we applied a protocol, in the following sequence, to obtain and assess data quality: (1) download georeferenced data from speciesLink (http://splink.cria.org.br), an electronic database of Brazilian primary biodiversity data; (2) quantify and remove geographically duplicated information; (3) quantify and remove data lacking longitude or latitude; (4) quantify and remove data lacking coordinate precision information (decimals) or coordinate uncertainty, as this information is crucial to many applications (e.g., ecological niche modeling) and prevents misinterpretation of coarse-resolution coordinates; (5) plot data on reference maps of Brazilian states and phytogeographical domains, and compare the geographic location of data with species checklist at domain- and state-level information, available at “Flora do Brasil 2020” (floradobrasil.jbrj.gov.br); and (6) quantify and remove records located on the ocean and away from the state or domain of confirmed occurrence.

We obtained a total of 674,750 georeferenced occurrence records, of which 354,851 (52.6%) had duplicated latitude-longitude information; 8 (<0.01%) and 6 (<0.01%) of the unique georeferenced records lacked latitude or longitude, respectively; 5691 (0.8%) of the unique and complete georeferenced records did not present coordinate precision (decimals) or uncertainty information; and 7499 (1.1%) records were located on the ocean or were considered outliers. After the data-cleaning protocol, 1938 species (67%) presented less than 100 unique georeferenced records, 967 species (33%) had less than 30 records, and 373 species (13%) had less than 10 records (Fig. 6.1). Note that the lower percentages are included in the quantification of larger ones (e.g., the 967 species with less than 30 records are included in the 1938 species with less than 100 records).

The final dataset had 217,116 records from 2906 species, located within the boundaries of the Atlantic Forest based on a 20 × 20 km grid overlaid across the domain extent (Fig. 6.2a). Our evaluation illustrates that despite large amounts of digital primary biodiversity data for tree species in the Atlantic Forest, the number of unique localities per species is rather small (median, 53, range, 1–1571) and data for most species is scarce or not yet ready for use in biodiversity assessments and applications (Sousa-Baena et al. 2013; Peterson et al. 2018; Cornwell et al. 2019). However, the high number of duplicates regards only to geographic coordinate information and may not reflect true duplicates, in the sense of specimens that have been deposited in different herbaria. Such data-cleaning approach may be appropriate to reduce spatial autocorrelation of records and characterize the environment where species occur (Boria et al. 2014); however, it may introduce bias in cases where two specimens have been collected on the same locality but contain different coordinates.

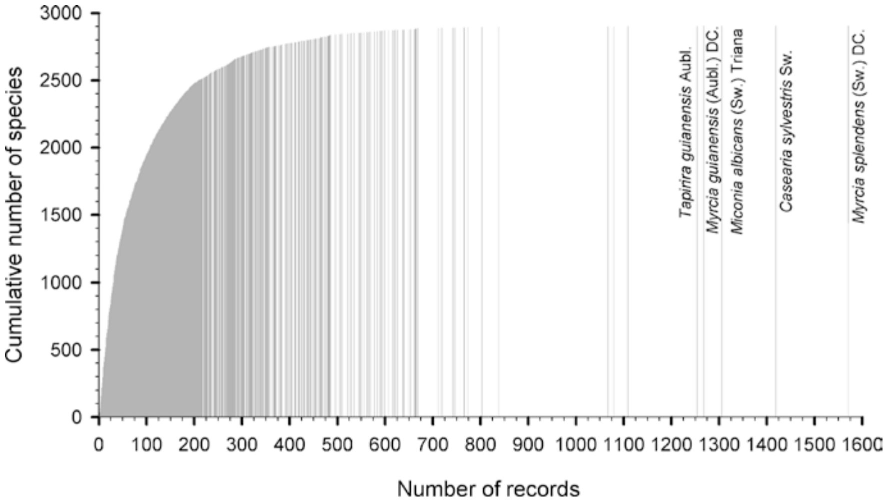

**Fig. 6.1** Frequency of unique georeferenced records and the cumulative number of tree species in the Brazilian Atlantic Forest. We highlight the five species with the largest number of records

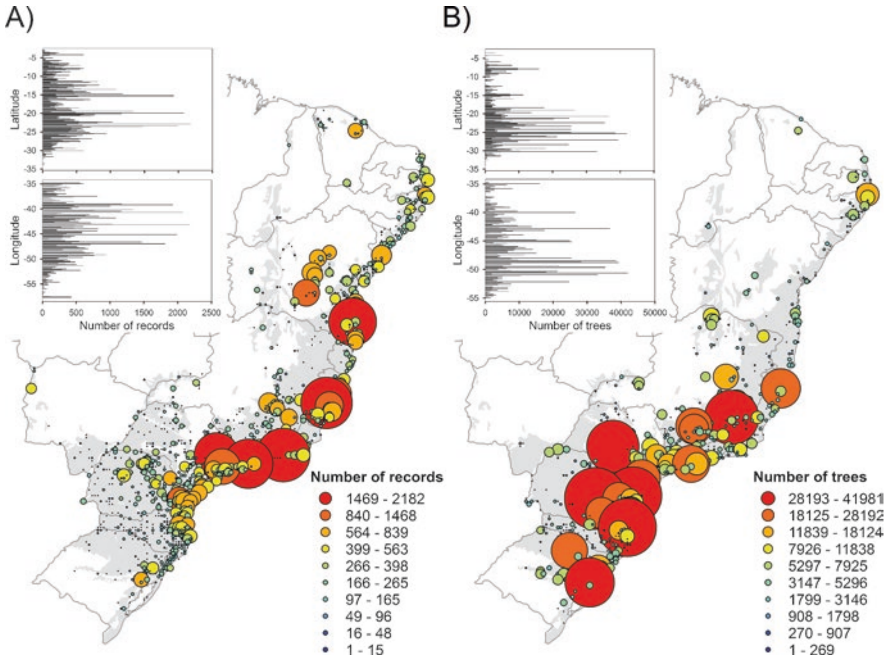

**Fig. 6.2** Total number of unique georeferenced herbarium records (**a**) and living trees in forest surveys (**b**). Values were obtained by extracting the sum in each grid cell of a 20 × 20 km grid overlaid across the Brazilian Atlantic Forest

It is important to notice that our study only used herbarium data from speciesLink; however, there are other available sources of occurrence records, such as the Global Biodiversity Information Facility (GBIF) and Herbarium Collection of the Rio de Janeiro Botanical Garden (RB), that could complement datasets. Furthermore, records classified as outliers that contain inconsistent coordinates may have the original herbarium deposit file scrutinized in order to retrieve information of the approximate location of sampling, thus increasing the number of useful records per species.

### 6.2.2 *Forest Inventory Data*

We compiled inventories of Atlantic Forests from the Neotropical Tree Community database (v. 4.0 – http://labtrop.ib.usp.br/doku.php?id=projetos:treeco:start). We considered all Atlantic Forest formations available in the database and all successional stages, including early secondary forests. We made no restrictions related to sampling methods, effort, or type of results presented. We included inventories of the dominant and regeneration strata of the forest, although the completeness of our list of studies is biased toward the upper stratum of the forest. In the case of studies presenting the same data or different censuses at the same site, we considered only the most recent study/census or the one published in peer-reviewed journals. Therefore, inventory data were obtained from a total of 1162 studies accessible to us, which contained 2645 surveys, and 2.18 million trees were located within the limits of the domain (Fig. 6.2b). These surveys ranged from 0.01 to 26 ha (average 0.68 ha), and the main inclusion criteria were DBH ≥5 cm (46%), ≥10 cm (26%), and ≥3 cm (8%). For each of these inventories, we extracted the number of living trees and the number of species in the sample. We also retrieved the geographical coordinates of each inventory and verified their precision. Whenever needed, the coordinates provided by the authors were corrected, based on maps or site description provided in the study.

## 6.3 Quantifying and Explaining Sampling Bias in Tree Diversity Data

One important step to bridge the existing knowledge gaps of primary biodiversity data is to understand why some parts of the Atlantic Forest have more information than others. In other words, we asked the following question: what are the drivers of the spatial occurrence and amount of tree diversity data? To answer this question, we first calculated the number of herbarium records and the number of species records from inventories in 20 km grid cells. For the same cells, we obtained six spatially explicit independent variables (Table 6.1), which we hypothesized to cor-

**Table 6.1** Spatially explicit variables used in the analyses and how they were obtained from sources. These were calculated for each cell of a 20 × 20 km grid across the Brazilian Atlantic Forest

| Data | Data type and calculation | Source |
|---|---|---|
| Remaining forest cover for the year 2016 | The shapefile with forest remnants was rasterized by calculating the *cover* in each pixel (continuous data) | Fundação SOS Mata Atlântica (2017) (https://www.sosma.org.br) |
| Area covered by protected areas | The conservation unit shapefile was rasterized by calculating the *cover* in each pixel (continuous data) | Ministério do Meio Ambiente (http://mapas.mma.gov.br/i3geo/datadownload.htm)<br>World Database on Protected Areas (WDPA) 1.0 (https://www.unep-wcmc.org/resources-and-data/wdpa) |
| The closest institution with graduate programs in Botany or Ecology | The list of graduate programs was assigned coordinates for the municipalities and rasterized into binary data. The minimum geographic distance to any pixel with the presence of an institution was calculated (continuous data) | Plataforma Sucupira (2019) (https://sucupira.capes.gov.br/sucupira/) |
| The closest federal or state road | The shapefile for roads was transformed into a binary raster, and the geographic distance to the nearest pixel with a road was calculated (continuous data) | Instituto Brasileiro de Geografia e Estatística (https://portaldemapas.ibge.gov.br) |
| Density of cities | The shapefile with the centroids for capitals and municipalities was rasterized by counting the number of centroids per pixel (discrete, count data) | Instituto Brasileiro de Geografia e Estatística (https://portaldemapas.ibge.gov.br) |
| Total human population | The official GDP data for 2016 was rasterized to the standard grid and summed up for each pixel (continuous data) | Instituto Brasileiro de Geografia e Estatística (https://portaldemapas.ibge.gov.br) |

relate with primary data availability and represent proxies of remaining forest, human presence, and ease of access (Sousa-Baena et al. 2013; Oliveira et al. 2016).

Both response variables had an excess of zeros; therefore, we described the variation in herbarium and inventory data using a hurdle model (Zuur et al. 2009), which first fits the presence and absence of the data (using a binomial model) and then models the counts at those sites with non-zeros (using the negative binomial distribution). For this specific analysis, we removed the data from the Santa Catarina state forest inventory (Vibrans et al. 2015), which was the only study conducted using systematic sampling, following a grid across the entire state, meaning that data will not be related to the independent variables selected here. Prior to analysis, all independent variables were transformed using a Box-Cox procedure and standardized (i.e., (observed − mean)/standard deviation), to make their estimated effects comparable. Comparison between the null and hurdle models was based on the Akaike information criterion (AIC). For each independent variable, we present the estimated parameter, its 95% confidence interval, and the test statistics. This analysis was performed in R (R Core Team 2018) using package *pscl* (Zeileis et al. 2008).

We found that 61% and 17% of the 20 km grid cells had at least one record for herbarium and inventory data, respectively. Overall, 46% of the cells had only herbarium data, and 2% had only inventory data. Consequently, both sources of data had significantly aggregated patterns, with inventory data being more aggregated than herbarium data (Morisita's coefficient of dispersion, 17.8 and 10.1, respectively) and with a higher concentration of records in the southern part of the Atlantic Forest (Fig. 6.2). These results reveal that more than one-fifth of the cells (21%) that still have remaining forest fragments have no information at all regarding their tree diversity and that our current knowledge on the rest of the cells is highly aggregated in space. If we consider ten herbarium or inventory records as a minimum to characterize tree diversity of a given cell, then only 38% of the Atlantic Forest tree diversity can be properly characterized.

The coverage of herbarium data was more than three times the coverage of inventory data. However, since each inventory generally provides records for many tree species at once, the number of records provided by inventory data (total, 2,302,423; median, 986; range, 29–41,981) was more representative relative to herbarium data (total, 217,116; median, 15; range, 1–2182). Considering only the grid cells with both herbarium and inventory data (15% of the cells), we found a positive correlation between the sums of living trees in inventory data and number of herbarium records; however, the predictive power of this relationship was low (adjusted $R^2 = 12\%$; $F = 120.99$; $p < 0.001$). There are two possible explanations for this result, which are not self-excluding: (i) the collection of herbarium and inventory data are not being carried in the same areas and (ii) the plant specimens collected in forest inventories are not always deposited in herbaria. In practice, the weak relationship between the two sources of biodiversity data makes them complementary to each other. This means that the combined use of both sources of data would provide a better characterization of the tree species composition and diversity in the Atlantic Forest.

The results of the models aiming to explain the current availability of biodiversity data were qualitatively similar for both herbarium and inventory data. The presence and the amount of data increase with forest cover, presence of protected areas, population size, and density of cities, while data decrease with distance from graduate programs and access roads (Fig. 6.3). These trends were stronger for herbarium than inventory data, particularly regarding the density of cities. For herbarium data, there was a shift in the relative importance of the distance to roads and to graduate programs between the presence/absence and count parts of the model (Fig. 6.3). While the proximity of roads is more important to predict the existence of a herbarium record, it becomes less important to predict the total number of records. This could suggest that the proximity to graduate programs is more important to define the amount of knowledge on tree diversity. For inventory data, the variables related to ease of access (i.e., road and grad school distances) were more important

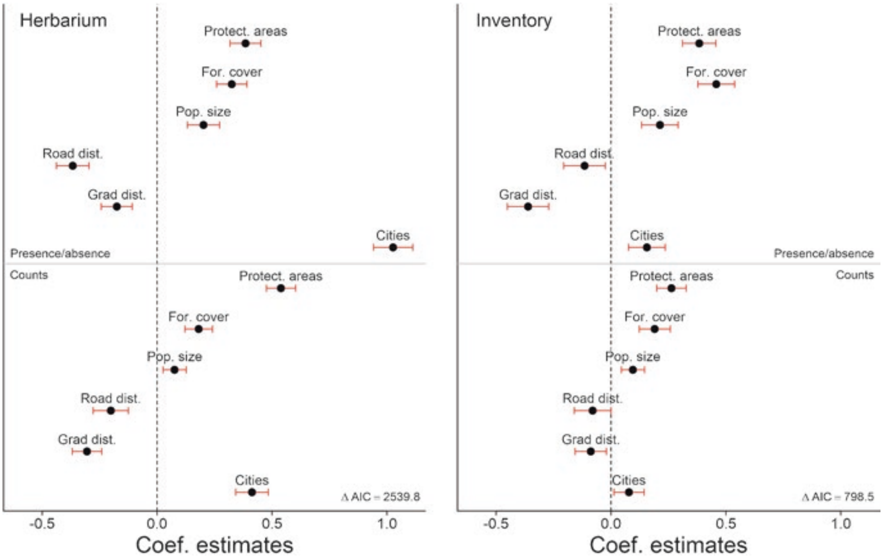

**Fig. 6.3** Influence of selected independent variables on the presence and amount of herbarium (left) and inventory data (right) for trees in the Brazilian Atlantic Forest. Analyses were performed using a hurdle model, and each point represents the estimated effect of each independent variable, along with the 95% confidence interval (red brackets). The vertical dashed line in each panel separates the negative (left) and positive (right) influences, while the horizontal gray line separates the parts of the hurdle model regarding the presence/absence (top) from the counts/amount of data (bottom). We also present the ΔAIC value between the hurdle model containing all independent variables and the model without them. Legend: Protect. Areas, area covered by protected areas; For. cover, 2016 remaining forest cover; Pop. size, total human population; Road dist., distance to the closest federal or state road; Grad dist, distance to the closest institution with graduate programs in Botany or Ecology; Cities, density of cities

for the presence/absence part of the model, meaning that although these independent variables can predict the existence of inventories, they predict less their total effort.

## 6.4 Insights on Patterns of Tree Diversity

Attempts to describe, explain, and predict diversity patterns across the Atlantic Forest have been mostly based on local scales. Most broad-scale studies have assessed patterns of species composition and potential environmental drivers (Oliveira-Filho and Fontes 2000; Eisenlohr and Oliveira-Filho 2015; Rezende et al. 2015; Neves et al. 2017; Marcilio-Silva et al. 2017), but only a few have looked at species richness patterns (Oliveira-Filho et al. 2013; Cerqueira and Martins 2015; Zwiener et al. 2020). In the herbarium dataset, we found that Myrtaceae (358; 12.3%), Fabaceae (344; 11.8%), Rubiaceae (191; 6.6%), Melastomataceae (158;

5.4%), and Lauraceae (144; 5%) were the five most species-rich families, respectively. These results were closely followed by the forest inventory data, in which Myrtaceae (422; 13.5%), Fabaceae (387; 12.4%), Rubiaceae (188; 6%), Lauraceae (163; 5.3%), and Melastomataceae (139; 4.5%) were the most specious families.

Species were not evenly distributed across the Atlantic Forest, with some regions having a higher concentration than others (Fig. 6.4). Forests near the ocean presented the greatest number of species according to both herbarium and inventory data, especially from southern Bahia to Rio de Janeiro states, whereas interior forests were less species-rich. However, some patterns differed among the datasets: in the herbarium data, sites with comparatively higher species richness in the south could potentially reflect higher sampling intensity (see Fig. 6.2a), and also the total species value reinforces that data from herbarium collections and forest inventories represent different and complementary aspects of biodiversity (Bottin et al. 2019). For instance, from the herbarium data, we found that the maximum number of species per cell was more than twofold the maximum value of inventories. Considering that the number of species for the herbarium data was obtained by summing species occurrences in each cell and, for the inventory data, values were obtained by averaging observed richness from studies within cells, the two datasets are fundamentally

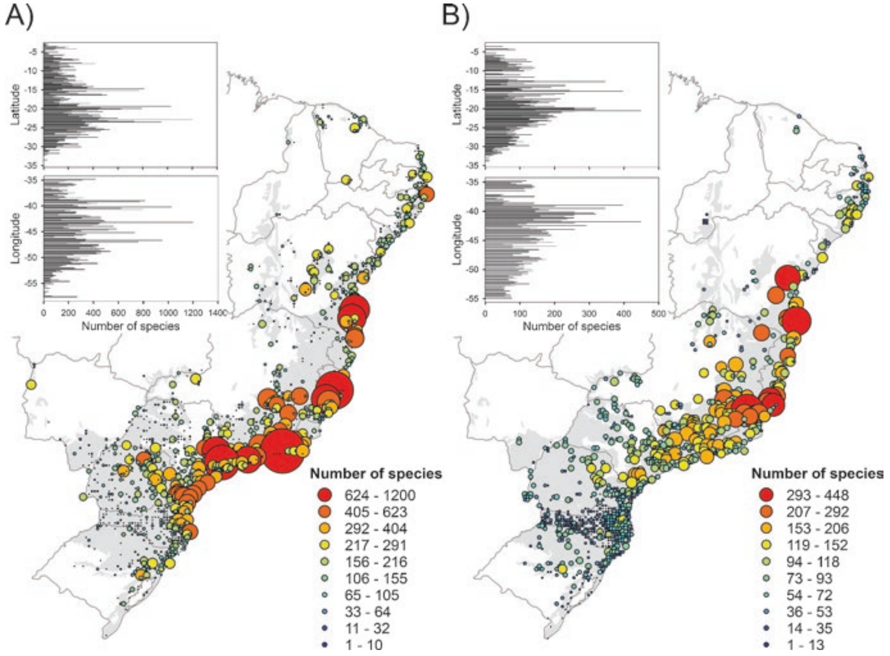

**Fig. 6.4** Number of species from unique georeferenced herbarium records (**a**) and forest inventories (**b**) overlaid on 20 × 20 km grid across the Brazilian Atlantic Forest. Values for herbarium data were obtained by extracting the sum of species occurrences in each grid cell, whereas values for forest inventories consist of an average of the number of species obtained from studies in each grid cell

different, with the herbarium dataset possibly more associated with a regional species pool than with local communities (Bottin et al. 2019). Further studies aiming at comparing species richness from these two primary biodiversity data sources should control for sampling bias and spatial scales that influence overall values.

Contrary to the classical richness-latitude gradient, species richness is lower and spatially correlated in the geographic extremes and peaks in the mid-portion of the domain (Fig. 6.4). Such broad-scale patterns, also observed for longitude, have long been described and discussed in the literature (Mori et al. 1981; Fiaschi and Pirani 2009). However, recent empirical evidence demonstrates the complexity of non-mutually exclusive mechanisms, such as environmental gradients, and spatial and historical constraints that structure geographic patterns of tree biodiversity, for example, species coherence to environmental gradients, nestedness and turnover in ecological communities, limits of physiological plasticity, relative contribution of marginal habitats, speciation, and dispersal processes (Neves et al. 2017; Marcilio-Silva et al. 2017; Sánchez-Tapia et al. 2018; Rezende et al. 2018; Zwiener et al. 2018; Cantidio and Souza 2019; Zwiener et al. 2020).

In this sense, patterns associated with environmental gradients have been the main focus of studies that explore broad-scale tree diversity across the Atlantic Forest (e.g., Oliveira-Filho and Fontes 2000; Eisenlohr and Oliveira-Filho 2015; Rezende et al. 2015; Neves et al. 2017; Marcilio-Silva et al. 2017; Zwiener et al. 2020). The results have highlighted climatic variables related to productivity (i.e. evapotranspiration) annual means, minimum and maximum variation and seasonality of temperature and precipitation. Comparatively, fewer studies have assessed the influence of soil variables and other biologically meaningful correlates (e.g., fire and aridity) at multiple scales and extents (Marques et al. 2011; Eisenlohr and Oliveira-Filho 2015; Neves et al. 2017; Cantidio and Souza 2019; Zwiener et al. 2020).

Latitude, longitude, distance from the ocean, altitude, and other topographic variables have also been described as important environmental predictors of tree diversity (Oliveira-Filho and Fontes 2000; Marques et al. 2011; Eisenlohr and Oliveira-Filho 2015; Rezende et al. 2015; Marcilio-Silva et al. 2017; Neves et al. 2017). These variables certainly contribute to the current understanding of biodiversity patterns across the domain and are even used as surrogates for plant distributions (Eisenlohr and Oliveira-Filho 2015). However, they lack a clear and direct biological mechanism and likely represent indirect effects that are correlated to underlying spatially structured environmental gradients (Austin 2002). For example, altitude covaries with temperature, which is known to affect organisms; however, the rate of covariation – and by extension its indirect effect associated with temperature – is dependent on latitude. As such, indirect variables have a restricted role in representing mechanisms that affect patterns across scales and geographical extents.

Along with current environmental variation, historical and spatial constraints such as Quaternary climate fluctuations (Costa et al. 2018), dispersal barriers (e.g., Doce and São Francisco rivers, Carnaval and Moritz 2008), recent geological movements (e.g., Ribeira Iguape Valley; Maia et al. 2017), phylogenetic resemblance

(Duarte et al. 2014), and geographical space partitioning (Sánchez-Tapia et al. 2018) also determine biogeographical patterns of tree diversity in the Atlantic Forest. For instance, the most species-rich sites coincide with specious regions for other taxa and Pleistocene forest refugia (Carnaval and Moritz 2008; Oliveira et al. 2017; Costa et al. 2018). Moreover, the Atlantic Forest shows floristic similarities with the Amazon Forest and Caatinga in the north (Oliveira-Filho and Ratter 1994) and is influenced by the flora of the Andes and elements of the ancient southern Gondwana in the south (SanMartín and Ronquist 2004; Duarte et al. 2014). Sites in the mid-portion that present the highest species richness may suffer floristic influence by both biogeographical regions (Oliveira-Filho et al. 2013; Zwiener et al. 2020).

## 6.5 Spatial Priorities for Future Sampling of Primary Biodiversity Data

There are different ways to define priorities for the sampling of tree primary biodiversity data, but they all depend basically on the current distribution of already available data and on remaining forests. In a simplistic scenario, high priority would be given to sites with high forest cover but low or null sampling intensity. However, as shown in this chapter, the spatial patterns of sampling are correlated with the presence of protected areas and variables representing ease of access. This influence is likely related to the scarcity of resources and infrastructure that would be necessary to sample more remote areas. Therefore, a more realistic prioritization should consider such factors (features) under different scenarios.

In order to do this, we used the program Zonation v4.0 to generate maps of spatial priorities for sampling, using the basic core-area removal rule (Moilanen et al. 2014). By adding multiple features and changing their relative weights, we generated prioritizations under three scenarios: (A) simplistic, which considers only forest cover (positive weight) and intensity of previous sampling (negative weight); (B) low-cost opportunistic, which considers the previous features plus distance to graduate programs and roads (negative features), protected areas (positive), density of cities (positive), and total human population (positive); and (C) high-cost exploratory, which considers the same previous features but with inverted weights of distance to graduate programs and roads, protected areas, density of cities, and total human population in order to favor remote forested unsampled sites. Weights for features in scenarios "B" and "C" were established proportionally to the sum of coefficients of the herbarium and inventory count hurdle model (Fig. 6.3). For all scenarios, the sum of weights was equal to zero, thus generating a balanced solution between positive and negative features.

Sites with high priority were distributed across the entire Atlantic Forest but with some regions having a higher concentration (Fig. 6.5). For all scenarios, the southwestern region (west and central of Paraná and Mato Grosso do Sul states) and

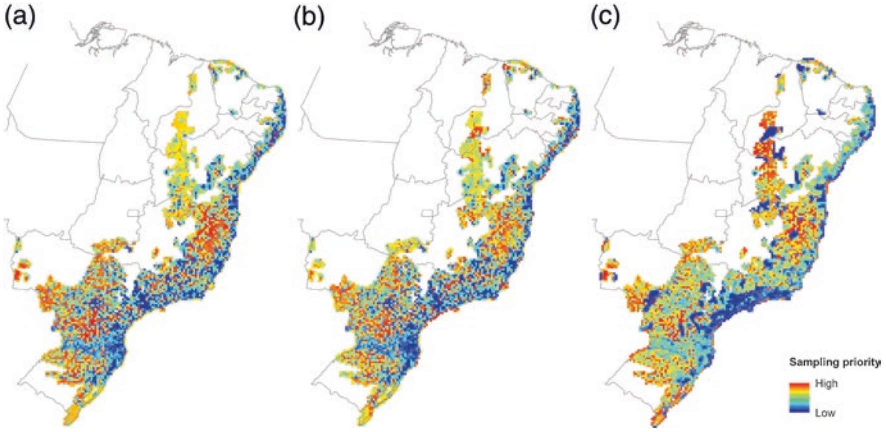

**Fig. 6.5** Spatial priorities for sampling trees in the Brazilian Atlantic Forest with positive and negative features weighted under different scenarios. Features and weights of scenario (**a**), forest cover (+1), density of sampling (−1); scenario (**b**), forest cover (0.27), density of cities (0.25), human population (0.09), protected areas (0.40), density of sampling (−0.66), distance to graduate program (−0.20), distance to roads (−0.14); scenario (**c**), forest cover (0.66), density of cities (−0.25), human population (−0.09), protected areas (−0.40), density of sampling (−0.27), distance to graduate program (0.20), distance to roads (0.14)

interior forests of the mid part of the domain (northeast of Minas Gerais) had a high concentration of priority sites. Interior forests of Goiás in central Brazil and deciduous forests of southern Atlantic Forest were also highlighted in the prioritization exercise. Other regions also presented high priority depending on the scenario under consideration (e.g., Piauí in scenario C).

Our exercise demonstrates an approach to prioritize areas for sampling the remaining Atlantic Forest vegetation under different scenarios that weigh accessibility (scenario "B") and exploration of more remote undersampled areas (scenario "C"). Scenario "B" represents a low-cost alternative that may also replicate the sampling bias of currently available primary biodiversity data, whereas scenario "C" is likely to better complement the current knowledge, yet with higher field expedition costs.

All scenarios show that sites along the coastlines of southeastern and southern Brazil have been sampled extensively, which thus are not considered a priority in comparison to other sites in the Atlantic Forest. This is true in particular for trees and regions such as the state of Santa Catarina, which has been systematically sampled across its full extent (Vibrans et al. 2015). Such an initiative greatly contributed to the floristic knowledge in southern Brazil and highlights the importance of planned field efforts to support herbarium collections and forest inventory data in this biodiversity hotspot. Furthermore, sampling high-priority sites in more remote areas (i.e., Piauí) may contribute to understand the floristic link between different forests and define better the limits of the domain.

## 6.6 Concluding Remarks: Where to Go from Here?

Overcoming knowledge shortfalls such as species misidentification or lack of a formal description and availability of precise information on geographical distributions (i.e., the Linnean and Wallacean shortfalls) represents a current challenge in biodiversity assessments (Hortal et al. 2015). Our approach was not intended to assess knowledge shortfalls in detail; rather, we present a general picture of the limitations in herbarium and forest inventory datasets that are commonly used in macroecological studies and biodiversity synthesis. Based on the results and information provided here, we can conclude that:

1. Despite an apparently large amount of data from both herbarium collections and forest inventories, we still urge for high-quality georeferenced biodiversity data. Efforts should be directed at identifying species and the geographic location where they were collected and provide representative information on the number of individuals, species composition, and community structure, particularly in more remote and undersampled sites.
2. Data from herbarium collection and forest inventories are fundamentally different but provide important and complementary information on the distribution of tree biodiversity in the Atlantic Forest. Future studies should include as many sources of primary biodiversity data, be aware and control for sampling bias, and consider the effects of spatial scale on observed patterns.
3. Hypotheses to explain species richness patterns in the domain remain understudied. Efforts that focus on assessing clear biological hypotheses regarding niche-based, stochastic, and historical processes, along with anthropogenic factors, would greatly contribute to understand and predict patterns of tree biodiversity in the Atlantic Forest.
4. Future prioritization for the sampling of primary biodiversity data should also consider species individually and the representation of environmental variation across the domain, given that such information is crucial to characterize ecological niches, explore correlations with biodiversity patterns, and generate a comprehensive synthesis.

## References

Austin MP (2002) Spatial prediction of species distribution: an interface between ecological theory and statistical modelling. Ecol Model 157:101–118

Bergamin RS, Duarte LDS, Marcilio-Silva V, Seger GDDS, Liebsch D, Marques MCM (2015) Compilation of woody species occurring in the Brazilian Atlantic Forest complex. Front Biogeogr 7:69–72

Boria RA, Olson LE, Goodman SM, Anderson RA (2014) Spatial filtering to reduce sampling bias can improve the performance of ecological niche models. Ecol Model 275:73–77

Bottin M, Peyre G, Vargas C, Raz L, Richardson JE, Sanches A (2019) Phytosociological data and herbarium collections show congruent large scale patterns but differ in their local descriptions of community composition. J Veg Sci. https://doi.org/10.1111/jvs.12825

Cain SA, Castro GMO, Pires JM, Silva NT (1956) Application of some phytosociological techniques to Brazilian rain forest. Am J Bot 43:911–941

Cantidio LS, Souza AF (2019) Aridity, soil and biome stability influence plant ecoregions in the Atlantic Forest, a biodiversity hotspot in South America. Ecography 42:1887–1898

Carnaval AC, Moritz C (2008) Historical climate modelling predicts patterns of current biodiversity in the Brazilian Atlantic Forest. J Biogeogr 35:1187–1201

Cerqueira RM, Martins FR (2015) Does climate influence spatial variation of tree species alpha diversity in the Brazilian Atlantic rainforest? Acta Sci Biol Sci 37:185–196

Cornwell WK, Pearse WD, Dalrymple RL, Zanne AE (2019) What we (don't) know about global plant diversity. Ecography 42:1819–1831

Costa GC, Hampe A, Ledru MP et al (2018) Biome stability in South America over the last 30 kyr: inferences from long-term vegetation dynamics and habitat modelling. Glob Ecol Biogeogr 27:285–297

CRIA - Centro de Referência em Informação Ambiental (2019) SpeciesLink – Electronic Database. http://splink.cria.org.br/. Assessed 13 May 2019

Davis DE (1945) The annual cycle of plants, mosquitoes, birds, and mammals in two Brazilian forests. Ecol Monogr 15:243–295

GBIF (2016) Science review 2016. Global Biodiversity Information Facility, Copenhagen

Duarte SD, Bergamin RS, Marcilio-Silva V, Seger GDS, Marques MCM (2014) Phylobetadiversity among Forest Types in the Brazilian Atlantic Forest Complex. PLoS One 9:e105043

Eisenlohr PV, Oliveira-Filho AT (2015) Revisiting patterns of tree species composition and their driving forces in the Atlantic Forests of southeastern Brazil. Biotropica 46:689–701

Fiaschi P, Pirani JR (2009) Review of plant biogeographic studies in Brazil. J Syst Evol 47:477–496

Fundação SOS Mata Atlântica (2017) Atlas dos remanescentes florestais da Mata Atlântica – Período. Fundação SOS Mata Atlântica, INPE, São Paulo, pp 2015–2016. (Relatório técnico).

Hortal J, Bello F, Diniz-Filho JAF, Lewinsohn TM, Lobo JM, Ladle R (2015) Seven shortfalls that beset large-scale knowledge of biodiversity. Annu Rev Ecol Evol Syst 46:523–549

Lima RAF, Mori DP, Pita G et al (2015) How much do we know about the endangered Atlantic Forest? Reviewing nearly 70 years of information on tree community surveys. Biodivers Conserv 24:2135–2148

Maia FR, Zwiener VP, Morokawa R, Silva-Pereira V, Goldenberg R (2017) Phylogeography and ecological niche modelling uncover the evolutionary history of Tibouchina hatschbachii (Melastomataceae), a taxon restricted to the subtropical grasslands of South America. Bot J Linn Soc 183:616–632

Marcilio-Silva V, Zwiener VP, Marques MCM (2017) Metacommunity structure, additive partitioning and environmental drivers of woody plants diversity in the Brazilian Atlantic Forest. Divers Distrib 23:1110–1119

Marques MCM, Swaine MD, Liebsch D (2011) Diversity distribution and floristic differentiation of the coastal lowland vegetation: implications for the conservation of the Brazilian Atlantic Forest. Biodivers Conserv 20:153–168

Moilanen A, Pouzols FM, Meller L, Veach V, Arponen A, Leppänen J, Kujala H (2014) Zonation –Spatial conservation planning methods and software. Version 4. User Manual. University of Helsinki, Finland

Mori SA, Boom BA, Prance GT (1981) Distribution patterns and conservation of eastern Brazilian coastal forest tree species. Brittonia 33:233–245

Murray-Smith C, Brummitt NA, Oliveira-Filho AT, Bachman S, Moat J, Lughadha EMN, Lucas EJ (2009) Plant diversity hotspots in the Atlantic coastal forests of Brazil. Conserv Biol 23:151–163

Neves DM, Dexter KG, Pennington RT et al (2017) Dissecting a biodiversity hotspot: the importance of environmentally marginal habitats in the Atlantic Forest domain of South America. Divers Distrib 23:8989–8909

Oliveira U, Paglia AP, Brescovit AD et al (2016) The strong influence of collection bias on biodiversity knowledge shortfalls of Brazilian terrestrial biodiversity. Divers Distrib 22:1232–1244

Oliveira U, Brescovit AD, Santos AJ (2017) Sampling effort and species richness assessment: a case study on Brazilian spiders. Biodivers Conserv 26:1481–1493

Oliveira-Filho AT (2017). NeoTropTree, Flora arbórea da Região Neotropical: Um banco de dados envolvendo biogeografia, diversidade e conservação. Universidade Federal de Minas Gerais. Retrieved from http://www.neotroptree.info

Oliveira-Filho AT, Budke JC, Jarenkow JA, Eisenlohr PV, Neves DM (2013) Delving into the variations in tree species composition and richness across South American subtropical Atlantic and Pampean forests. J Plant Ecol 8:242–260

Oliveira-Filho AT, Fontes MAL (2000) Patterns of floristic differentiation among Atlantic Forests in southeastern Brazil, and the influence of climate. Biotropica 32:793–810

Oliveira-Filho AT, Ratter JA (1994) Database: Woody flora of 106 forest areas of eastern tropical South America. Royal Botanic Garden, Edinburgh

Peterson AT, Asase A, Canhos DAL, de Souza S, Wieczorek J (2018) Data leakage and loss in biodiversity informatics. Biodivers Data J 6:e26826

Plataforma Sucupira (2019) Cursos recomendados e reconhecidos. Área de Ensino. https://www.sucupira.capes.gov.br/sucupira/. Assessed 12 Jan 2019

R Core Team (2018) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria

REFLORA – Plantas do Brasil: Resgate Histórico e Herbário Virtual para o Conhecimento e Conservação da Flora Brasileira (2019). http://reflora.jbrj.gov.br/reflora/PrincipalUC/PrincipalUC.do. Assessed 14 Nov 2019

Rezende VL, Miranda PLS, Meyer L et al (2015) Tree species composition and richness along altitudinal gradients as a tool for conservation decisions: the case of Atlantic semideciduous forest. Biodivers Conserv 24:2149–2163

Rezende VL, Bueno ML, Eisenlohr PV, Oliveira-Filho AT (2018) Patterns of tree species variation across southern South America are shaped by environmental factors and historical processes. Perspect Plant Ecol Evol Syst 34:10–16

Sánchez-Tapia A, Garbin ML, Siqueira MF, Guidoni-Martins KG, Scarano FR, Carrijo TT (2018) Environmental and geographical space partitioning between core and peripheral Myrsine species (Primulaceae) of the Brazilian Atlantic Forest. Bot J Linn Soc 187:633–652

SanMartín I, Ronquist F (2004) Southern Hemisphere biogeography inferred by event-based models: plant versus animal patterns. Syst Biol 53:216–243

Sheth SN, Lohmann LG, Consiglio T, Jiménez I (2008) Effects of detectability on estimates of geographic range size in Bignonieae. Conserv Biol 22:200–211

Silva LAE, Fraga CN, Almeida TMH et al (2017) Jabot – Sistema de Gerenciamento de Coleções Botânicas: a experiência de uma década de desenvolvimento e avanços. Rodriguésia 68:391–410

Sousa-Baena MS, Garcia LC, Peterson AT (2013) Completeness of digital accessible knowledge of the plants of Brazil and priorities for survey and inventory. Divers Distrib 20:369–381

Veloso HP (1945) As comunidades e estações botânicas de Teresópolis, Estado do Rio de Janeiro (com um ensaio de chave dendrológica). Bol Mus Nac 3:1–95

Veloso HP (1946) A vegetação do município de Ilhéus, Estado da Bahia, I—Estudo sinecológico das áreas de pesquisas sobre a febre amarela silvestre realizado pelo SEPFA. Mem Inst Oswaldo Cruz 44:13–103

Veloso HP, Klein RM (1957) As comunidades e associações vegetais da mata pluvial do sul do Brasil I. As comunidades do município de Brusque, Estado de Santa Catarina. Sellowia 8:81–235

Veloso HP, Klein RM (1968) As comunidades e associações vegetais da mata pluvial do sul do Brasil VI. Agrupamentos arbóreos dos contra-fortes da Serra Geral situados ao sul da costa catarinense e ao norte da costa sul-riograndense. Sellowia 20:127–180

Vibrans AC, Gasper AL, Müller JJ, Mantovani A, Aguilar MD, Marschalek R (2015) Inventário florístico florestal de Santa Catarina. FURB, Blumenau

Yang W, Ma K, Kreft H (2013) Geographical sampling bias in a large distributional database and its effects on species richness-environment models. J Biogeogr 40:1415–1426

Werneck MDS, Sobral MEG, Rocha CTV, Landau EC, Stehmann JR (2011) Distribution and endemism of angiosperms in the Atlantic Forest. Nat Conservacao 9:188–193

Zappi C, Filardi FLR, Leitman P et al (2015) Growing knowledge: an overview of Seed Plant diversity in Brazil. Rodrigésia 66:1085–1113

Zeileis A, Kleiber C, Jackman S (2008) Regression models for count data in R. J Stat Softw 27(8). http://www.jstatsoft.org/v27/i08/

Zuur AF, Ieno EN, Walker NJ, Saveliev AA, Smith GM (2009) Mixed effects models and extensions in ecology with R. Springer, New York

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

Zwiener VP, Lira-Noriega A, Grady CJ, Padial AA, Vitule JRS (2018) Climate change as a driver of biotic homogenization of woody plants in the Atlantic Forest. Glob Ecol Biogeogr 27:298–309

Zwiener VP, Padial AA, Marques MCM (2020) The mechanisms explaining tree species richness and composition are convergent in a megadiverse hotspot. Biodivers Conserv 29:799–815

# Chapter 7
# Vascular Epiphytes of the Atlantic Forest: Diversity and Community Ecology

**Flavio Nunes Ramos, Sara Ribeiro Mortara, and João Pedro Costa Elias**

**Abstract** Epiphytes are plants that germinate and root non-parasitically on other plants, without contact with the soil. They are hyper-diverse and comprise approximately 9% of all vascular plant species. We discuss here vascular epiphytes growing in the Atlantic Forest, focusing on (i) their diversity, the number of epiphyte species, and the main groups and families; (ii) their biogeography and the influence of biotic and abiotic factors on species distributions; (iii) their relationships with phorophyte trees, considering mainly trunk and crown influences on epiphyte communities; and (iv) conservation, discussing epiphyte conservation statuses and reintroductions.

**Keywords** Biogeography · Conservation · Endemism · Epiphytes · Phorophytes · Richness

## 7.1 Introduction

Epiphytes are plants that germinate and root non-parasitically on other plants, without contact with the soil (Madison 1977; Bennet 1986; Zotz 2016), obtaining most of their nutrients from atmospheric sources (Benzing 1983). The classification of plants into epiphytic life forms has been controversial, and many attempts have been made to define them (Batke et al. 2016; Benzing 1987, 1990). The currently most useful life form classification for mechanically dependent plants (Kelly 1985) divides species into six categories: holo-epiphytes, primary

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hemi-epiphytes, nomadic vines (includes secondary hemi-epiphytes), climbers (lianas and vines), stranglers, and hemi-parasites (i.e., mistletoes) (Zotz 2013). Zotz (2013, 2016) only consider holo- and primary hemi-epiphytes as epiphytes (including stranglers).

Epiphytes are hyper-diverse, comprising approximately 9% of all vascular plant species (Cascante-Marín and Nivia-Ruíz 2013; Ibisch et al. 1996; Kress 1986; Zotz 2013). They provide ecological services related to hydrology and nutrient cycling (Jarvis 2000; Stanton et al. 2014; Bruijnzeel et al. 2011) and contribute to diversity through their interactions with other biota (Benzing 1983; Yanoviak et al. 2007). A great number of invertebrate and vertebrate organisms use epiphytes as habitat or as food resources (Angelini and Silliman 2014; Lasky and Keitt 2012; Scheffers et al. 2014). Epiphytes are one of the first life forms to become extinct due to overexploitation and deforestation, the loss of large trees (their support), and changes in microclimatic conditions (Sodhi et al. 2008). Because of their arboreal lifestyles and sensitivity to environmental stress, epiphytes can be important bioindicators (Kull and Hutchings 2006; Li et al. 2015).

The South American Atlantic Forest is considered one of the most endangered ecosystems in the world (Myers et al. 2000; Mittermeier et al. 2004) and one of the main floristic centers in the Neotropics (Gentry 1982; Stehmann et al. 2009). Most of that forest (~ 95%) occurs in eastern Brazil, but it also extends into Argentina, Paraguay, and Uruguay (from 3° S to 30° S, Ribeiro et al. 2009, Stehmann et al. 2009). The biome shows wide variations in rainfall and elevation (from sea level up to >2800 m) and comprises a vast heterogeneity of habitats (Oliveira-Filho and Fontes 2000), including several vegetation types, partly reflecting environmental gradients (Oliveira-Filho et al. 2005). While forest physiognomies in the continental interior largely receive rainfall concentrated in only a single season (~1000 mm/year), coastal forests can receive as much as 4000 mm distributed relatively evenly throughout the year (Câmara 2003). The Atlantic Forest originally covered approximately 150 million hectares but now occupies only 12% of its original area, with 80% of its fragments being <50 ha, with a mean distance between them of 1.4 km (Ribeiro et al. 2009). The Atlantic Forest presents very high richness of the epiphyte species, representing more than 15% of the total vascular plant richness (2000 species, Ramos et al. 2019), being most (78%) of them endemic from this biome (Freitas et al. 2016).

We present an overview of vascular epiphyte studies in the Atlantic Forest in this chapter and discuss conservation implications and gaps in our knowledge concerning epiphyte diversity.

## 7.2 Vascular Epiphyte Studies

Most epiphyte assemblage studies undertaken in the Atlantic Forest have been floristic inventories that documented the most common representatives of local floras (Ramos et al. 2019). The first epiphyte inventory in Paraná State, Brazil, was made in 1950 (Hertel 1950). The most common sampling method adopted in epiphyte assemblage inventories in the Atlantic Forest has been the walk-through methodology, as proposed by Filgueiras et al. (1994). Those surveys follow a transect in a selected forest area, without quantifications of the number of individuals of each species, which makes new epiphyte inventories necessary as new epiphyte species are still being found. Lana et al. (2018), for example, studying only the Orchidaceae assemblage in a conservation area in Minas Gerais State, recently found four species that had not yet been reported in any Atlantic Forest flora. Quantitative studies are fundamental to describing patterns and understanding community ecology processes, such as those related to assemblage changes in response to environmental disturbances (Ricklefs 2007). There have been very few quantitative studies (11 studies or 12% of the data), however, of epiphyte communities in the Atlantic Forest (Ramos et al. 2019) or even in the world (Mendieta-Leiva and Zotz 2015). The quantification methodology most commonly adopted in epiphyte inventories in the Atlantic Forest involves counting the number of individuals or stands (sense Sanford 1968) (99%) (Ramos et al. 2019) – but there have been only rare studies examining them by coverage (1%) and only one quantifying biomass (Petean et al. 2018). There have also been collection biases in epiphyte assemblage inventories in the Atlantic Forest, as those inventories have been concentrated in southern and southeastern Brazil, mainly near the coast (Ramos et al. 2019). Studies examining epiphyte-tree relationships have likewise been very scarce, and few workers have presented any information about phorophytes (e.g., Chaves et al. 2016; Francisco et al. 2018).

## 7.3 Vascular Epiphyte Diversity

Floristic surveys in the Atlantic Forest have largely focused on tree species, although there has been increasing interest in epiphyte diversity in the Atlantic Forest [e.g., Kersten (2010), Leitman et al. (2015), Menini Neto et al. (2016), Freitas et al. (2016), and Ramos et al. (2019)]. The first estimates of Atlantic Forest vascular epiphyte richness were made by Kersten (2010), who cited 1074 species; current estimated epiphyte richness is approximately 2000 species. Our consultations of the official list of Brazilian flora (Brazilian Flora 2020 under construction) identified 2227 epiphyte species in the Atlantic Forest. Freitas et al. (2016) reported 2256 species of vascular epiphytes, while Ramos et al. (2019) reported 2080 species of vascular and avascular epiphytes. Although the species list of Freitas et al. (2016) was incorporated in the database published by Ramos et al. (2019), the latter only reported records of species with validated geographic coordinates – which may have contributed to the differences in the number of species records between Freitas et al. (2016) and Ramos et al. (2019).

### 7.3.1 Number of Vascular Epiphyte Species

We present here a brief description of the number behind epiphyte diversity in the Atlantic Forest. Almost 80% of the vascular epiphytes found in the Atlantic Forest are endemic (Freitas et al. 2016; Brazil Flora G 2020). That degree of epiphyte species endemism is very high when compared both to (i) other groups in the Atlantic Forest, such as total vascular plants (57%), trees (54–60%), or amphibians (60%) (Mittermeier et al. 2004, Brazilian Flora 2020 under construction), and (ii) epiphytes in other forests from Central America or north of South America (0 from 37%, Kreft 2004, Cascante-Marin and Nivia-Ruíz 2013). Species not endemic to the Atlantic Forest have disjunct distributions between the Atlantic Forest and the Amazon or Cerrado biomes (~15%), but only a few species (~5%) occur in more than two phytogeographic domains (Freitas et al. 2016, Brazilian Flora 2020 under construction). Angiosperms have a higher proportion of Atlantic Forest endemic species than ferns (Fig. 7.1). The richest families of vascular epiphytes are, in descending order, Orchidaceae, Bromeliaceae, Piperaceae, Arecaceae, Cactaceae, and Gesneriaceae (among angiosperms) and Polypodiaceae, Dryopteridaceae, Hymenophyllaceae, and Aspleniaceae (among ferns). The richest families of angiosperm epiphytes show high levels of endemism, especially Bromeliaceae and Gesneriaceae (91% and 89% endemic species, respectively).

Fontoura et al. (2012) and Menini Neto et al. (2016) reported that the southeastern and southern regions of the Atlantic Forest were the most species-rich. According to Ramos et al. (2019), the southern and southeastern regions of Brazil and southern region of Bahia State (Fig. 7.2) contain the richest known areas of epiphyte species diversity – which also coincide with areas that have experienced the highest

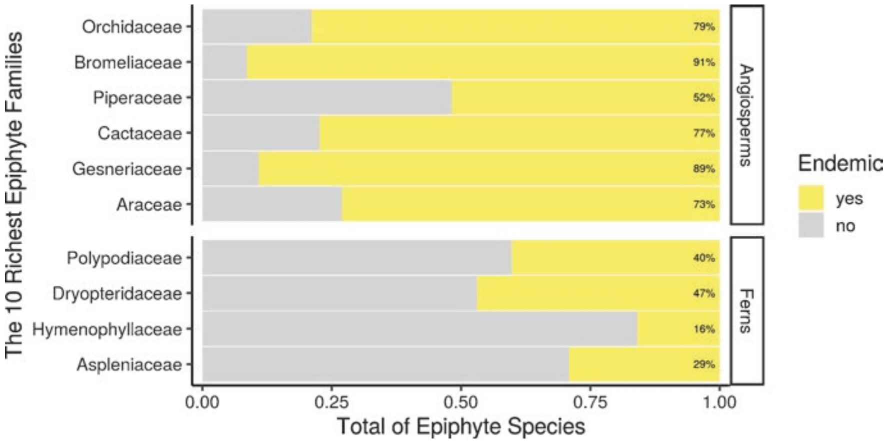

**Fig. 7.1** Proportions of endemic species among the ten richest families of Atlantic Forest epiphytes. The percentage numbers indicate the percentage of endemic species within each family. The data in this figure was derived from the Brazil Flora G (2020)

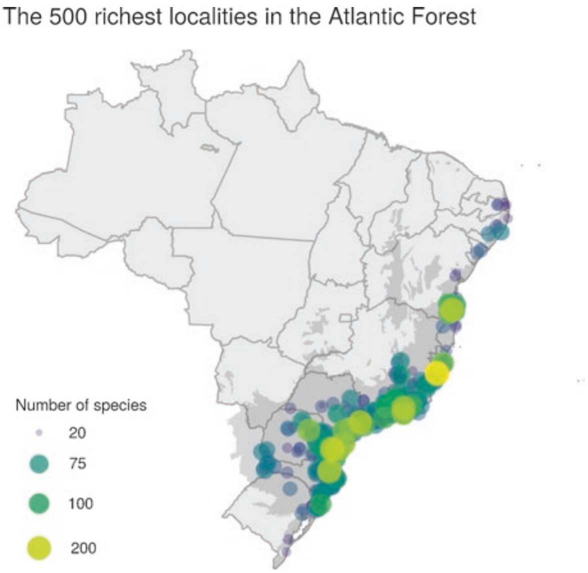

**Fig. 7.2** Number of species recorded in each locality in the Atlantic Forest (Atlantic Forest domain in dark gray). Regions with the highest number of species are (1) southern Bahia State and (2) southeastern and (3) southern Brazil. (Data from Ramos et al. (2019))

sampling efforts (Kersten 2010; Ramos et al. 2019), being close to important research centers in large cities (with the exception of southern Bahia State).

The areas holding the greatest epiphyte richness are southeastern and southern Brazil, especially near central-southern Rio de Janeiro and Espírito Santo States, along the Serra do Mar mountain range (Fontoura et al. 2012; Menini Neto et al. 2016). Similarly, studies with angiosperms (Werneck et al. 2011) and some animal groups (Ferro and Melo 2011; Vasconcelos et al. 2014) reported that same pattern and also included southern Bahia State as one of the richest areas (Menini Neto et al. 2016). Many other organisms occurring in the Atlantic Forest share similar diversity patterns, such as butterflies (Brown Jr. and Freitas 2000), moths (Ferro and Melo 2011), termites (Cancello et al. 2014), and harvestmen spiders (Pinto-da-Rocha and Silva 2005).

Most Atlantic Forest epiphyte records are from Ombrophilous Forest sites (60%) (both Mixed and Dense), with fewer records from Semideciduous Forests (25%) (Ramos et al. 2019). Leitman et al. (2015) identified 38 epiphytic species indicators of Atlantic Forest vegetation types, most of them bromeliads and orchids. Seasonal Semideciduous Forest was the only vegetation type where indicator species were not identified. Epiphyte species occurring in Seasonal Semideciduous Forest represent a subset of the species occurring in Ombrophilous Forests (more humid and richer forest), which can tolerate lower precipitation rates, lower air humidity, and greater seasonality (Leitman et al. 2015). Epiphyte assemblages from Seasonal Semideciduous Forests representing subsets of Ombrophilous Forest species adapted to seasonal conditions were also reported in a local study of epiphyte assemblages (Forzza et al. 2014) and in tree assemblage studies (Oliveira-Filho and Fontes 2000). Additionally, adaptations of epiphyte species to lower temperatures could explain the high number of indicator species found in Mixed Ombrophilous Forests (Leitman et al. 2015).

## 7.4 Biogeography of Vascular Epiphytes

There have been few studies of epiphyte biogeography in the world, but epiphyte distributions have been observed to be influenced by both climatic conditions and vicariance events (Kessler 2001, 2002a; Kreft et al. 2004; Kessous et al. 2019; Moreno-Chacón and Saldaña 2019). Epiphyte species show wider distributions as compared to terrestrial or rupicolous species, due to their greater capacities of diaspore dispersal (Kessler 2002a). Epiphyte species, however, appear to be more susceptible to water stress and low temperatures (especially frost) as compared to terrestrial plants (Gentry and Dodson 1987; Benzing 1990; Kessler 2002b; Kreft et al. 2004), although more research is still needed in that area (Zotz 2016).

Epiphyte richness has been found to be greater in areas with high precipitation rates and low seasonality, as those plants live without soil and are highly dependent on moisture derived directly from rainfall or water vapor (Kreft et al. 2004). Precipitation is therefore an important climatic factor influencing epiphyte diversity (Gentry and Dodson 1987; Zotz 2016). Additionally, although at smaller scales, the presence of humidity derived from clouds, rivers, or streams (even temporary ones) can influence epiphyte richness (Gentry and Dodson 1987; Zotz 2016; Furtado and Menini Neto 2018). Another limiting climatic factor for epiphyte distribution is temperature (Gentry and Dodson 1987; Krömer et al. 2005; Kessler et al. 2011; Hsu et al. 2014). Low temperatures, especially frost events, appear to be the most important climatic condition limiting the latitudinal and altitudinal distributions of epiphyte species (Mayo et al. 1997; Nervo et al. 2019). Altitude can also influence vascular epiphyte richness. Several studies have shown a peak of epiphyte richness at intermediate altitudes between 1000 and 1500 m a.s.l. (Gentry and Dodson 1987; Hietz and Hietz-Seifert 1995; Küper et al. 2004; Krömer et al. 2005; Cardelus et al. 2006; Krömer et al. 2008; Hsu et al. 2014; Sanger and Kirkpatrick 2015; Ding et al. 2016).

Epiphyte richness and composition along latitudinal and altitudinal gradients in the Atlantic Forest are influenced not only by climatic conditions but also by historic events, such as river formation and Pleistocene refuges (Fontoura et al. 2012; Leitman et al. 2015; Menini Neto et al. 2016; Nervo et al. 2016; Furtado and Menini Neto 2018; Nervo et al. 2019), as is presented below in detail.

### *7.4.1 Latitude*

Latitudinal distributions of epiphyte species in the Atlantic Forest are influenced by climatic factors, mainly precipitation (from ocean to inland) and temperature (Fontoura et al. 2012; Leitman et al. 2015; Menini Neto et al. 2016). Areas with high precipitation show high epiphyte richness, such as the “Serra do Mar,” a mountain chain in southeastern Brazil that receives orographic rains on its highest slopes, providing both horizontal and vertical precipitations (Almeida and Carneiro 1998) – and thus sustaining considerable epiphyte diversity. Areas with low precipitation and (mainly) high seasonality, on the other hand, show lower epiphyte richness, such as Seasonal Semideciduous Forests, an inland forest physiognomy (Menini Neto et al. 2016). Temperatures also affect epiphyte richness in some areas of the Atlantic Forest, especially minimum and mean annual temperatures in low latitudinal areas (Fontoura et al. 2012). Epiphyte assemblages in the Atlantic Forest are also influenced by geographical distances (Leitman et al. 2015), with epiphyte communities being very similar between neighboring areas, even those with differing environmental conditions – suggesting that seed dispersal is an important limitation (Leitman et al. 2015).

### *7.4.2 Elevation*

There is a strong influence of altitude on the general pattern of epiphyte distribution in the Atlantic Forest (Leitman et al. 2015; Nervo et al. 2016; Furtado and Menini Neto 2018; Nervo et al. 2019), although that influence decreases with increasing latitude (Leitman et al. 2015). Fontoura et al. (2012) suggested that elevational influences were stronger at regional scales. Because most of the higher-altitude areas in the Atlantic Forest mountains are equivalent to mid-elevation peaks (between 1000 and 1500 m a.s.l.), however, some studies investigating the influence of elevation on epiphyte assemblage diversity did not show sharp reductions of species richness at the highest reaches of those gradients (Nervo et al. 2016, Furtado and Menini Neto 2018, Nervo et al. 2019), as opposed to reductions in taller mountains, such as the Andes (Krömer et al. 2005).

### *7.4.3 Past Events*

Plant (Prance 1982; Oliveira-Filho and Fontes 2000; Ledru et al. 2007; Prata et al. 2018) and animal (Pellegrino et al. 2005; Cabanne et al. 2007; Thome et al. 2010; DaSilva and Pinto-da-Rocha 2010; DaSilva et al. 2015) species distributions in the Atlantic Forest are not homogenous but rather have two main sections that are divided by the Doce River in northeastern Espírito Santo State. The northern block comprises the northeastern region of the Atlantic Forest, while the southern block comprises the southeastern and southern forest regions – with epiphyte distribution following the same pattern (Fontoura et al. 2012; Menini Neto et al. 2016). That disjunction between the two blocks has been attributed to (i) their geomorphologic differentiation (DaSilva and Pinto-da-Rocha 2010), (ii) oceanic invasions of the Doce River region during Pleistocene interglacial periods (Suguio et al. 2005), and

(iii) the creation of forest refuges to the north of the São Francisco River during the Pleistocene era (Carnaval and Moritz 2008).

## 7.5 Relationships Between Epiphytes and Their Phorophytes

The relationships between epiphytes and their support plants (phorophytes) are consensually defined as commensal relationships. Epiphytes show non-random distribution patterns, which are influenced by phorophyte traits and climatic conditions. In general, the relationships between epiphytes and phorophytes in the Atlantic Forest show only low levels of specialization (Francisco et al. 2018; Francisco et al. 2019; Zotarelli et al. 2019), similar to other epiphyte/phorophyte patterns in tropical forests (e.g., Sáyago et al. 2013) – suggesting stronger associations with specific phorophyte features and not the phorophyte species themselves. Traits associated with specific phorophyte species, however, can explain variations of epiphyte diversity in specific cases [e.g., the bark-shedding species *Piptadenia gonoacantha* (Fabaceae.) harbors fewer epiphyte individuals in the Atlantic Forest as compared to other phorophytes] (Dislich and Mantovani 2016).

In terms of other phorophyte traits, larger tropical forest trees support the most epiphyte species, and just a few large trees may satisfactorily describe local epiphyte species diversity (Zotz and Bader 2011). The largest trees in the Atlantic Forest support a considerable fraction of epiphyte diversity (Schmitt and Windisch 2010; Dislich and Mantovani 2016), and a single large tree was found to bear 46% of the total local epiphyte richness (Francisco et al. 2018). Epiphyte diversity on Atlantic Forest phorophytes is not equally distributed along the tree surface, with the intermediate height of the tree harboring the highest epiphyte diversity (Kersten et al. 2009). The trunk tends to support more epiphyte richness, but with lower abundance (individuals/biomass), than the crown (Francisco et al. 2018; Petean et al. 2018); that pattern is not consistent, however, in other forest types (e.g., the Amazon Forest; Pos and Sleegers 2010). Even though some epiphytes may show preferences for specific phorophyte zones (e.g., Flores-Palacios and García-Franco 2005), that does not necessarily result in considerable differences in species compositions between Atlantic Forest zones (Machado et al. 2016).

Phorophyte traits influencing epiphyte distributions generally reflect the combined effect of several tree features, as opposed to a single isolated trait. Larger trees have more habitat available for epiphytes and are thus able to support more epiphyte coverage, while trees with more habitat heterogeneity (e.g., rugose bark) show the richest epiphyte species compositions (Batista and Santos 2016). Phorophyte habitat availability and heterogeneity can therefore better explain epiphyte distribution patterns when analyzed together. Small trees, for example, usually harbor fewer epiphytes but become excellent hosts to atmospheric bromeliads if they are also deciduous (habitat heterogeneity proxy) (Chaves et al. 2016). The decisive role of large trees in Atlantic Forest epiphyte diversity is well-established, but we know very little about the traits associated with habitat heterogeneity and even less about the effects of combined tree features. In comparisons of ecosystems analogous to the Atlantic Forest (i.e., tropical forests), bark (e.g., peeling bark, Boelter et al. 2014) and canopy characteristics (e.g., canopy structure, Fayle et al. 2006) were found to be associated with epiphyte assemblage in different ways, but have not yet been extensively examined in the Atlantic Forest.

It is possible that the unique distribution patterns of Atlantic Forest epiphytes are due to the high proportion of endemic epiphyte species found there. As such, a vast gap in our knowledge of Atlantic Forest epiphyte/phorophyte relationships exists in terms of (I) the combined effects of two or more traits and (II) traits associated with phorophyte habitat heterogeneity.

## 7.6 Conservation

Anthropic disturbances reducing forest cover represent the main threat to epiphyte species (Barberena et al. 2018; Cardoso et al. 2018), followed by the absence of mutualistic organisms (pollinators, seed dispersers, mycorrhiza) and collection pressure (Barros 2007). According to the CNCFlora red list of endangered species (Martinelli and Moraes 2013), only 8% (171) of the Brazilian epiphyte species (Brazil Flora G 2020) listed are considered endangered [including the categories of critically endangered (33 species), endangered (82), or vulnerable (56)]; on the other hand, only 11% of the epiphyte species are considered as of least concern. Given that the majority (~80%) of Brazilian epiphyte species are endemic to the Atlantic Forest (Freitas et al. 2016) and a considerable number are represented by less than ten records from the Atlantic Forest (Ramos et al. 2019), considerably more work will be needed to determine their conservation statuses. Our knowledge of epiphyte diversity is therefore still incipient, and data on epiphyte species distributions and local abundances are still extremely scarce. Fully 19% of all epiphyte species are represented by only a single collection, and 59% have less than ten records (Ramos et al. 2019). The combination of high endemism levels, few records, and limited data on species abundances and conservation raises concern about how little we actually know about epiphyte ecology in the Atlantic Forest.

Deforestation (due to farming and cattle raising) represents one of the greatest impacts on epiphyte assemblages. Studying the effects of forest fragmentation and land use on this plant group will be important for determining effective conservation and management actions. The few studies already undertaken to examine the impacts of anthropic land use on epiphyte assemblages in the Neotropics have shown that areas having more intense land use [especially plantations, such as shade coffee, teak, and pine (Moorhead et al. 2010, Einzmann and Zotz 2016, Alzate-Q et al. 2019), and pasture land (Köster et al. 2009, Larrea and Werner 2010, Werner et al. 2011)] demonstrate decreased richness and alterations in epiphyte assemblage composition, mainly due to altered microclimatic conditions (Werner 2011). Studies of land use or edge effects in the Atlantic Forest have shown similar patterns. Only two studies comparing the effects of land use on epiphyte assemblages in the

Atlantic Forest have been published that examined differences in epiphyte assemblages in intact forest versus shaded cocoa plantations (Pardini et al. 2009; Fernandes et al. 2015). Those studies showed that although epiphyte richness was lower in shaded cocoa plantations and their species composition differed from the forest, that type of agroforestry could still preserve important number of species. It is important, however, to highlight the fact that the richness and abundances of generalist species were higher than those of specialist/forest species in shaded cocoa plantations. There have been six studies focusing on edge effects on epiphyte assemblages in the Atlantic Forest (Bataghin et al. 2008; Bernardi and Budke 2010; Bataghin et al. 2012; Bianchi and Kersten 2014; Dias-Terceiro et al. 2015; Silva et al. 2017), and they demonstrated that the forest fragments showed impoverished epiphyte assemblages that become even poorer near forest edges. Not only did richness decrease, but there were also compositional changes, and solar radiation levels appeared to be correlated with epiphyte abundances and distribution within the fragments (Bataghin et al. 2008, Bernardi and Budke 2010, Bataghin et al. 2012, Bianchi and Kersten 2014, Dias-Terceiro et al. 2015, Silva et al. 2017).

Few local studies addressing epiphyte conservation in the Atlantic Forest have been conducted in past decades. Some of them evaluated the conservation status of species in a specific area (Cardoso et al. 2016, 2018; Barberena et al. 2018) and economic impacts on epiphyte conservation (Santos et al. 2005), while others evaluated the growth and survival of reintroduced epiphyte species in remnant or restored forest fragments (Jasper et al. 2005; Endres-Júnior et al. 2015, 2018; Duarte and Gandolfi 2013, 2017).

Epiphyte restoration or reintroduction attempts can be difficult because of (a) a lack of knowledge of a given species' conservation status and its population dynamics; (b) their complex propagation and cultivation requirements; (c) a limited knowledge of their interactions with biotic and abiotic factors in forest remnants; and (d) problems related to the maintenance of genetic diversity. The few reintroduction projects with epiphyte species conducted in Atlantic Forests have achieved high (>50%) survival rates with both adult and young individuals (Jasper et al. 2005; Dorneles and Trevelin 2011; Duarte and Gandolfi 2013, 2017; Endres-Júnior et al. 2015, 2018). Reintroduction success was affected mainly by solar radiation levels (Endres-Júnior et al. 2015, 2018), the age or biomass of the epiphyte individuals (Duarte and Gandolfi 2013; Endres-Júnior et al. 2015, 2018), the season when planted (Duarte and Gandolfi 2017), and the substrate used to attach the individual to the tree (Jasper et al. 2005; Duarte and Gandolfi 2017). Those studies did not find any effects of phorophyte tree traits on epiphyte reintroduction success. Given that epiphyte reintroduction success was species dependent, detailed studies of the biology of epiphyte species will be necessary (Jasper et al. 2005), as both abiotic and biotic conditions could affect epiphyte reintroduction in Atlantic Forests. Solar radiation, for example, positively affected *Catasetum fimbriatum* (Orchidaceae) (Dorneles and Trevelin 2011; Endres-Júnior et al. 2015, 2018) (Duarte and Gandolfi 2017) but negatively affected *Rhipsalis floccosa* (Cactaceae) (Duarte and Gandolfi 2017) epiphyte reintroductions, while herbivory negatively affected the survival of *Cattleya intermedia* (Orchidaceae) (Dorneles and Trevelin 2011, Endres-Júnior et al. 2015, 2018). Additionally, many *Tillandsia* (Bromeliaceae) species showed higher transplantation survival rates when associated with other Orchidaceae, Cactaceae, or bryophyte species (by maintaining humidity) (Jasper et al. 2005).

## 7.7 Concluding Remarks

We reviewed in this chapter a number of studies that examined vascular epiphyte diversity, ecology, biogeography, and conservation in the Atlantic Forest and identified some significant gaps in our knowledge. Although there have been in situ conservation and restoration efforts using Atlantic Forest epiphytes, their conservation is still incipient. Epiphyte inventories undertaken in the Atlantic Forest have a longer history than studies of their ecology and conservation, but they are still not sufficient to fully support their management or unify political efforts that could guarantee their long-term protection. The combination of high endemism levels, few records, and only limited data concerning species' abundances or conservation statuses raises concern about how little we actually know about epiphyte distribution in the Atlantic Forest.

## References

Almeida FFM, Carneiro CDR (1998) Origem e evolução da Serra do Mar. Revista Brasileira de Geociências 28:135–150

Alzate-Q NF, García-Franco JG, Flores-Palacios A, Krömer T, Laborde J (2019) Influence of land use types on the composition and diversity of orchids and their phorophytes in Cloud Forest fragments. Flora. https://doi.org/10.1016/j.flora.2019.151463

Angelini C, Silliman BR (2014) Secondary foundation species as drivers of trophic and functional diversity: evidence from a tree-epiphyte system. Ecology 95:185–196

Barberena FFVA, Baumgratz JF, Barros F (2018) Ecological data for an orchid diversity hotspot show that the subtribe Laeliinae may be endangered in the Brazilian Atlantic Forest. Nord J Bot 36:e01728

Barros F (2007) A família Orchidaceae na Flora do Estado de São Paulo e suas espécies ameaçadas. In: Mamede MCH et al. (orgs), Livro vermelho das espécies vegetais ameaçadas do estado de São Paulo. Instituto de Botânica, São Paulo, p 47–52

Bataghin FA, Fiori A, Toppa RH (2008) Efeito de borda sobre epífitos vasculares em Floresta Ombrófila Mista, Rio Grande do Sul, Brasil. O Mundo da Saude 32:329–338

Bataghin FA, Pires JSR, Barros F (2012) Epifitismo vascular em sítios de borda e interior em Floresta Estacional Semidecidual no Sudeste do Brasil. Hoehnea 39(2):235–245

Batista WVS, Santos ND (2016) Can regional and local filters explain epiphytic bryophyte distributions in the Atlantic forest of southeastern Brazil? Acta Botanica Brasilica 30(3):462–472

Batke S, Cascante-Marín A, Kelly DL (2016) Epiphytes in Honduras: a geographical analysis of the vascular epiphyte flora and its floristic affinities to other Central American countries. J Trop Ecol 58:1–27

Bennet BC (1986) Patchiness, diversity, and abundance relationships of vascular epiphytes. Selbyana 9:70–75

Benzing DH (1983) Vascular epiphytes: a survey with special reference to their interactions with other organisms. In: Sutton SL, Whitmore TC, Chadwick AC (eds) Tropical rain forests: ecology and management. Blackwell Scientific Publications, Oxford, pp 11–24

Benzing DH (1987) Vascular epiphytism: taxonomic participation and adaptive diversity. Ann Mo Bot Gard 74:183–204

Benzing DH (1990) Vascular epiphytes. General biology and related biota. Cambridge University Press, Cambridge

Bernardi S, Budke J (2010) Estrutura da sinúsia epifítica e efeito de borda em uma área de transição entre Floresta Estacional Semidecídua e Floresta Ombrófila Mista. Floresta 40(1):81–92

Bianchi JS, Kersten RA (2014) Edge effect on vascular epiphytes in a subtropical Atlantic Forest. Acta Botanica Brasilica 28(1):120–126

Boelter CR, Dambros CS, Nascimento HEM, Zartman CE (2014) A tangled web in tropical treetops: effects of edaphic variation, neighbourhood phorophyte composition and bark characteristics on epiphytes in a central Amazonian forest. J Veg Sci 25(4):1090–1099

Brazil Flora G (2020) Brazilian Flora 2020 project – Projeto Flora do Brasil 2020. v393.222. Instituto de Pesquisas Jardim Botanico do Rio de Janeiro. Dataset/Checklist. https://doi.org/10.15468/1mtkaw

Brown KS Jr, Freitas AVL (2000) Atlantic forest butterflies: indicators for landscape conservation. Biotropica 32:934–956

Bruijnzeel LA, Mulligan M, Scatena FN (2011) Hydrometeorology of tropical montane cloud forests: emerging patterns. Hydrol Process 25:465–498

Cabanne GS, Santos FR, Miyaki C (2007) Phylogeography of *Xiphorhynchus fuscus* (Passeriformes, Dendrocolaptidae): vicariance and recent demographic expansion in southern Atlantic forest. Biol J Linn Soc 91:73–84

Câmara IG (2003) Brief history of conservation in the Atlantic Forest. In: Galindo-Leal C, Câmara IG (eds) The Atlantic Forest of South America: biodiversity status, trends, and outlook. Center for Applied Biodiversity Science and Island Press, Washington, DC, pp 31–42

Cancello EM, Silva RR, Vasconcellos A, Reis YT, Oliveira LM (2014) Latitudinal variation in termite species richness and abundance along the Brazilian Atlantic Forest Hotspot. Biotropica 46(4):441–450

Cardelus CL, Colwell RK, Watkins JE (2006) Vascular epiphyte distribution patterns: explaining the mid-elevation richness peak. J Ecol 94(1):144–156

Cardoso JC, Silva JAT, Vendrame WA (2016) Impacts of deforestation on some orchids of São Paulo State, Brazil. Natureza & Conservação 14:28–32

Cardoso WC, Calvente A, Dutra VF, Sakuragui CM (2018) Anthropic pressure on the diversity of Cactaceae in a region of Atlantic Forest in eastern Brazil. Haseltonia 24(1):28–35

Carnaval AC, Moritz C (2008) Historical climate modeling predicts patterns of current biodiversity in the Brazilian Atlantic forest. J Biogeogr 35:1187–1120

Cascante-Marín A, Nivia-Ruíz A (2013) Neotropical flowering epiphyte diversity: local composition and geographic affinities. Biodivers Conserv 22:113–125

Chaves CJN, Dyonisio JC, Rossatto DR (2016) Host trait combinations drive abundance and canopy distribution of atmospheric bromeliad assemblages. AoB PLANTS 8:1–13

DaSilva MB, Pinto-da-Rocha R (2010) Systematic review and cladistic analysis of the Hernandariinae (Opiliones: Gonyleptidae). Fortschr Zool 27:577–642

DaSilva MB, Pinto-da-Rocha R, DeSouza AM (2015) A protocol for the delimitation of areas of endemism and the historical regionalization of the Brazilian Atlantic rain forest using harvestmen distribution data. Cladistics 31:692–705

Dias-Terceiro RG, Peixoto GM, Gomes VS, Menezes MC, Neco EC, Pessoa TSA, Fabricante JR, Albuquerque MB (2015) Edge effect on vascular epiphytic composition in a fragment of Atlantic Forest in northeastern Brazil. Acta Botanica Brasilica 29(2):270–273

Ding Y, Liu G, Zang R, Zhang J, Lu X, Huang J (2016) Distribution of vascular epiphytes along a tropical elevational gradient: disentangling abiotic and biotic determinants. Sci Rep 6. https://doi.org/10.1038/srep19706

Dislich R, Mantovani W (2016) Vascular epiphyte assemblages in a Brazilian Atlantic Forest fragment: investigating the effect of host tree features. Plant Eco 217(1):1–12

Dorneles LT, Trevelin V (2011) Aclimatização e reintrodução de *Cattleya intermedia* Graham ex Hook (Orchidaceae) obtidas por propagação in vitro. Iheringia – Série Botânica 66:167–174

Duarte MM, Gandolfi S (2013) Enriquecimento de florestas em processo de restauração: aspectos de epífitas e forófitos que podem ser considerados. Hoehnea 40(3):507–514

Duarte MM, Gandolfi S (2017) Diversifying growth forms in tropical forest restoration: enrichment with vascular epiphytes. For Ecol Manag 401:89–98

Einzmann HJR, Zotz G (2016) How diverse are epiphyte assemblages in plantations and secondary forests in tropical lowlands? Tropical Conserv Sci 9(2):629–647

Endres-Júnior D, Sasamori MH, Silveira T, Schmitt JL, Droste A (2015) Reintrodução de *Cattleya intermedia* Graham (Orchidaceae) em borda e interior de um fragmento de Floresta Estacional Semidecidual no sul do Brasil. Revista Brasileira de Biociências 13(1):33–40

Endres-Júnior D, Sasamori MH, Schmitt JL, Droste A (2018) Survival and development of reintroduced *Cattleya intermedia* plants related to abiotic factors and herbivory at the edge and in the interior of a forest fragment in South Brazil. Acta Bot Bras 32(4):555–556

Fayle TM, Chung AYC, Dumbrell AJ, Eggleton P, Foster WA (2006) The effect of Rain Forest canopy architecture on the distribution of epiphytic ferns (*Asplenium* spp.) in Sabah, Malaysia. Biotropica 41(6):676–681

Fernandes V, Assis Bomfim J, Fontoura T, Cazetta E (2015) Richness and abundance of *Aechmea* and *Hohenbergia* (Bromeliaceae) in forest fragments and shade cocoa plantations in two contrasting landscapes in Southern Bahia, Brazil. Tropical Conserv Sci 8:58–75

Ferro VG, Melo AS (2011) Diversity of tiger moths in a Neotropical hotspot: determinants of species composition and identification of biogeographic units. J Insect Conserv 15:643–651

Filgueiras TS, Nogueira PE, Brochado AL, Guala GF (1994) Caminhamento: um método expedito para levantamentos florísticos qualitativos. Caderno de Geociências 12:39–43

Flores-Palacios A, García-Franco JG (2005) The relationship between tree size and epiphyte species richness: testing four different hypotheses. J Biogeogr 33(2):323–330

Fontoura T, Scudeller VV, Costa AF (2012) Floristics and environmental factors determining the geographic distribution of epiphytic bromeliads in the Brazilian Atlantic rain Forest. Flora 207(9):662–672

Forzza RC, Pifano DS, Oliveira Filho AT, Meireles LD, Faria PL, Salimena FRG, Mynssen CM, Prado J (2014) Flora vascular da reserva biológica da represa do grama, descoberto, Minas Gerais e Suas relações florísticas com florestas ombrófilas e semidecíduas do Sudeste Brasileiro. Rodriguésia 65:275–292

Francisco TM, Couto DR, Evans DM, Garbin ML, Ruiz-Miranda CR (2018) Structure and robustness of an epiphyte–phorophyte commensalistic network in a neotropical inselberg. Austral Ecol 43:903–914

Francisco TM, Couto DR, Garbin ML, Muylaert RL, Ruiz-Miranda CR (2019) Low modularity and specialization in a commensalistic epiphyte–phorophyte network in a tropical cloud forest. Biotropica 51(4):509–518

Freitas L, Salino A, Menini Neto L, Almeida T, Mortara S, Stehmann J, Amorim AM, Guimaraes E, Nadruz Coelho MA, Zanin A, Forzza R (2016) A comprehensive checklist of vascular epiphytes of the Atlantic Forest reveals outstanding endemic rates. PhytoKeys 58:65–79

Furtado SG, Menini Neto L (2018) Elevational and phytophysiognomic gradients influence the epiphytic community in a cloud forest of the Atlantic phytogeographic domain. Plant Ecol 219:677–690

Gentry AH (1982) Neotropical floristic diversity: phytogeographical connections between central and South America, Pleistocene climatic fluctuations, or an accident of the Andean orogeny? Ann Mo Bot Gard 69:557–593

Gentry A, Dodson CH (1987) Diversity and biogeography of neotropical vascular epiphytes. Ann Mo Bot Gard 74:205–233

Hertel RJG (1950) Contribuição à ecologia de flora epifítica da serra do mar (vertente oeste) do Paraná. Arquivos do Museu do Paranaense 8:3–63

Hietz P, Hietz-Seifert U (1995) Composition and ecology of vascular epiphyte communities along an altitudinal gradient in Central Veracruz, Mexico. J Veg Sci 6(4):487–498

Hsu RC, Wolf JH, Tamis WL (2014) Regional and elevational patterns in vascular epiphyte richness on an East Asian Island. Biotropica 46:549–555

Ibisch PL, Boegner A, Nieder J, Barthlott W (1996) How diverse are Neotropical epiphytes? An analysis based on the “Catalogue of the flowering plants and gymnosperms of Peru”. Ecotropica 2:13–28

Jarvis MT (2000) Measuring and modelling the impact of land-use change in tropical hillsides: the role of cloud interception to epiphytes. Adv Environ Monitor Model 1:118–148

Jasper A, Freitas EM, Musskopf EL, Bruxel J (2005) Metodologia de salvamento de Bromeliaceae, Cactaceae e Orchidaceae na Pequena Central Hidrelétrica (PCH) Salto Forqueta – São José do Herval/Putinga-RS-Brasil. Pesquisas, Botânica 56:265–284

Kelly D (1985) Epiphytes and climbers of a Jamaican Rain Forest: vertical distribution, life forms and life histories. J Biogeogr 12(3):223–241

Kersten RA (2010) Epífitas vasculares – Histórico, participação taxonômica e aspectos relevantes, com ênfase na Mata Atlântica. Hoehnea 37(1):9–38

Kersten RA, Kuniyoshi YS, Roderjan CV (2009) Epífitas vasculares em duas formações ribeirinhas adjacentes na bacia do rio Iguaçu – Terceiro Planalto Paranaense. Iheringia - Série Botânica 64:33–43

Kessler M (2001) Pteridophyte species richness in Andean forests in Bolivia. Biodivers Conserv 10:1473–1495

Kessler M (2002a) Environmental patterns and ecological correlates of range size among bromeliad communities of Andean forests in Bolivia. Bot Rev 68:100–127

Kessler M (2002b) Environmental patterns and ecological correlates of range size among bromeliad communities of Andean forests in Bolivia. Bot Rev 68:100–127

Kessler M, Kluge J, Hemp A, Ohlemüller R (2011) A global comparative analysis of elevational species richness patterns of ferns. Glob Ecol Biogeogr 20:868–880

Köster N, Friedrich K, Nieder J, Barthlott W (2009) Conservation of epiphyte diversity in an Andean landscape transformed by human land use. Conserv Biol 23(4):911–919

Kreft H, Köster N, Küper W, Nieder J, Barthlott W (2004) Diversity and biogeography of vascular epiphytes in Western Amazonia, Yasuní, Ecuador. J Biogeogr 31:1463–1476

Kress WJ (1986) The systematic distribution of vascular epiphytes: an update. Selbyana 9:2–22

Kessous IM, Neves B, Couto DR, Paixão-Souza B, Pederneiras LC, Moura RL, Barfuss MHJ, Salgueiro F, Costa AF (2019) HistoEerical biogeography of a Brazilian lineage of Tillandsioideae (subtribe Vrieseinae, Bromeliaceae): the Paranaean Sea hypothesized as the main vicariant event. Bot J Linn Soc, boz038

Krömer T, Kessler M, Gradstein R, Acebey A (2005) Diversity patterns of vascular epiphytes along an elevational gradient in the Andes. J Biogeogr 32:1799–1809

Krömer T, Jimenez I, Kessler M (2008) Diversity and vertical distribution patterns of vascular epiphytes in the Cordillera Mosetenes, Cochabamba, Bolivia. Revista Boliviana de Ecología y Conservacíon Ambiental 23:27–38

Kull T, Hutchings MJ (2006) A comparative analysis of decline in the distribution ranges of orchid species in Estonia and the United Kingdom. Biol Conserv 129:31–39

Küper W, Kreft H, Nieder J, Köster N, Barthlott W (2004) Large-scale diversity patterns of vascular epiphytes in Neotropical montane rain forests. J Biogeogr 31(9):1477–1487

Lana LG, Escobar TF, Godinho EMM, Peluzio LE (2018) Survey of Orchidaceae species of Serra das Cabeças, in Parque Estadual da Serra do Brigadeiro, Araponga-MG, Brazil. Hortic Bras 36:156–166

Larrea ML, Werner FA (2010) Response of vascular epiphyte diversity to different land-use intensities in a neotropical montane wet forest. For Ecol Manag 260(11):1950–1955

Lasky JR, Keitt TH (2012) The effect of spatial structure of pasture tree cover on avian frugivores in eastern Amazonia. Biotropica 44:489–497

Ledru MP, Salatino MLF, Ceccantini GT, Salatino A, Pinheiro F, Pintaud JC (2007) Regional assessment of the impact of climatic change on the distribution of a tropical conifer in the lowlands South America. Diver Distrib 13(6):761–771

Leitman P, Amorim AM, Sansevero JB, Forzza RC (2015) Epiphyte patterns in the Atlantic Forest. Bot J Linn Soc 179:587–601

Li P, Pemberton R, Zheng G (2015) Foliar trichome-aided formaldehyde uptake in the epiphytic *Tillandsia velutina* and its response to formaldehyde pollution. Chemosphere 119:662–667. https://doi.org/10.1016/j.chemosphere.2014.07.079

Machado LS, Gonzatti F, Windisch PG (2016) Epiphytic ferns in swamp forest remnants of the coastal plain of Southern Brazil: latitudinal effects on the plant community. Acta Botanica Brasilica 30(4):644–657

Madison M (1977) Vascular epiphytes: their systematic occurrence and salient features. Selbyana 2:1–13

Martinelli G, Moraes MA (2013) Livro vermelho da flora do Brasil. Andrea Jakobsson Estúdio Editorial, Jardim Botânico do Rio de Janeiro, Rio de Janeiro. 1100p

Mayo SJ, Bogner J, Boyce P (1997) The genera of Araceae. Royal Botanic Gardens, Kew, London. 370p

Mendieta-Leiva G, Zotz G (2015) A conceptual framework for the analysis of vascular epiphyte assemblages. Perspect Plant Ecol Syst 17(6):510–521

Menini Neto L, Furtado SG, Zappi DC, Oliveira-Filho AT, Forzza RC (2016) Biogeography of epiphytic Angiosperms in the Brazilian Atlantic forest, a world biodiversity hotspot. Rev Bras Bot 39:261–273

Mittermeier RA, Gil PR, Hoffman M, Pilgrim J, Brooks T, Mittermeier CG, Lamoreux J, Fonseca GAB (2004) Hotspots revisited: earth's biologically richest and most endangered terrestrial ecoregions. CEMEX & Agrupacion Sierra Madre, Cidade do México. 392p

Moorhead LC, Philpott SM, Bichier P (2010) Epiphyte biodiversity in the coffee agricultural matrix: canopy stratification and distance from forest fragments. Conserv Biol 24(3):737–746

Moreno-Chacón M, Saldaña A (2019) α, β and γ-diversity of vascular epiphytes along the climatic gradient of continental Chile. N Z J Bot 57(1):18–31

Myers N, Mittermeier RA, Mittermeier CG, Fonseca GA, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Nervo MH, da Silva Coelho FV, Windisch PG, Overbeck GE (2016) Fern and lycophyte communities at contrasting altitudes in Brazil's subtropical Atlantic Rain Forest. Folia Geobot 51:305

Nervo MH, Andrade BO, Tornquist CG, Mazurana M, Windisch PG, Overbeck GE (2019) Distinct responses of terrestrial and epiphytic ferns and lycophytes along an elevational gradient in Southern Brazil. J Veg Sci 30:55–64

Oliveira-Filho AT, Fontes MAL (2000) Patterns of floristic differentiation among Atlantic Forests in Southeastern Brazil and the influence of climate. Biotropica 32:793–810

Oliveira-Filho AT, Neto ET, Carvalho WAC, Werneck M, Brina AE, Vidal CV, Rezende SC, Pereira JAA (2005) Análise florística do compartimento arbóreo de áreas de floresta atlântica sensu lato na região das bacias do leste. Rodriguésia 56:185–235

Pardini R, Faria D, Accacio GM, Laps RR, Mariano-Neto E, Paciencia MLB, Dixo M, Baumgarten J (2009) The challenge of maintaining Atlantic Forest biodiversity: a multi-taxa conservation assessment of specialist and generalist species in an agroforestry mosaic in southern Bahia. Biol Conserv 142(6):1178–1190

Pellegrino KCM, Rodrigues MT, Waite AN, Mrando M, Yonenaga-Yassuda Y, Sites JW Jr (2005) Phylogeography and species limits in the *Gymnodactylus darwinii* complex (Gekkonidae, Squamata): genetic structure coincides with river systems in the Brazilian Atlantic Forest. Biol J Linn Soc 85:13–26

Petean MP, Marcon AK, Liebsch D, Franklin G, Kersten RA (2018) Vascular epiphyte biomass in a South Brazilian fragment of Atlantic Forest. An Acad Bras Cienc 90(4):3717–3728

Pinto-da-Rocha R, Silva MB (2005) Faunistic similarity and historic biogeography of the harvestmen of Southern and Southeastern Atlantic Rain Forest of Brazil. J Arachnol 33:290–299

Pos ET, Sleegers ADM (2010) Vertical distribution and ecology of vascular epiphytes in a lowland tropical rain forest of Brazil. Boletim do Museu Paraense Emílio Goeldi 5(3):335–344

Prance GT (1982) Biological diversification in the tropics. Columbia Univ. Press, New York

Prata B, Magalhães E, Teixeira AP, Joly CA, Assis MA (2018) The role of climate on floristic composition in a latitudinal gradient in the Brazilian Atlantic Forest. Plant Eco Evo 151(3):303–313

Ramos FN, Mortara SR, Monalisa-Francisco N, Elias JPC, Neto LM, Freitas L, Kersten RA, Amorim AM, Matos FB, Nunes-Freitas AF et al (2019) ATLANTIC EPIPHYTES: a data set of vascular and non-vascular epiphyte plants and lichens from the Atlantic Forest. Ecology 100(2):e02541

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

Ricklefs RE (2007) History and diversity: explorations at the intersection of ecology and evolution. Am Nat 170:56–70

Sanford W (1968) Distribution of epiphytic orchids in semi-deciduous tropical forest in southern Nigeria. J Ecol 56:697–705

Sanger JC, Kirkpatrick JB (2015) Moss and vascular epiphyte distributions over host tree and elevation gradients in Australian subtropical rainforest. Aust J Bot 63(8):696–704

Santos AJ, Bittencourt AM, Nogueira AS (2005) Aspectos econômicos da cadeia produtiva das bromélias na região metropolitana de curitiba e litoral paranaense. Floresta 35(3):409–417

Sáyago R, Lopezaraiza-Mikel M, Quesada M, Álvarez-Añorve MY, Cascante-Marín A, Bastida JM (2013) Evaluating factors that predict the structure of a commensalistic epiphyte-phorophyte network. Proceed Royal Soc B:280. https://doi.org/10.1098/rspb.2012.2821

Scheffers BR, Phillips BL, Shoo LP (2014) Asplenium bird's nest ferns in rainforest canopies are climate-contingent refuges for frogs. Global Ecol Conserv 2:37–46

Schmitt JL, Windisch PG (2010) Biodiversity and spatial distribution of epiphytic ferns on *Alsophila setosa* Kaulf. (Cyatheaceae) caudices in Rio Grande do Sul, Brazil. Braz J Biol 70(3):521–528

Silva VL, Mallmann IT, Cunha S, Schmitt JL (2017) Impacto do efeito de borda sobre a comunidade de samambaias epifíticas em floresta com araucária. Revista Brasileira de Ciências Ambientais 45:19–32

Sodhi NS, Koh LP, Peh KSH, Tan HTW, Chazdon RL, Corlett RT, Lee TM (2008) Correlates of extinction proneness in tropical angiosperms. Diver Distrib 14:1–10

Stanton DE, Huallpa Chávez J, Villegas L, Villasante F, Armesto J, Hedin LO, Horn H (2014) Epiphytes improve host plant water use by microenvironment modification. Funct Ecol 28:1274–1283

Stehmann JR, Forzza RC, Salino A, Sobral M, Costa DP, Kamino LHY (2009) Plantas da Floresta Atlântica. Jardim Botânico do Rio de Janeiro, Rio de Janeiro, 505 p

Suguio K, Angulo RJ, Carvalho AM, Corrêa ICS, Tomazelli LJ, Willwock JA, Vital H (2005) Paleoníveis do mar e paleolinhas de costa. In: Souza CRG, Suguio K, Oliveira AMS, Oliveira PE (eds) Quaternário do Brasil. Holos Editora, Ribeirão Preto, pp 114–129

Thome MTC, Zamudio KR, Giovanelli JGR, Haddad CFB, Baldissera FA Jr, Alexandrino JMB (2010) Phylogeography of the endemic toads and post-Pliocene persistence of the Brazilian Atlantic forest. Mol Phylogenet Evol 55:1018–1031

Vasconcelos TS, Prado VHM, Silva FR, Haddad CFB (2014) Biogeographic distribution patterns and their correlates in the diverse frog fauna of the Atlantic forest hotspot. PLoS One 9:e104130

Werneck FP, Costa GC, Colli GR, Prado DE, Sites JW Jr (2011) Revisiting the historical distribution of seasonally dry tropical forests: new insights based on palaeodistribution modeling and palynological evidence. Glob Ecol Biogeogr 20:272–288

Werner FA, Homeier J, Oesker M, Boy J (2011) Epiphytic biomass of a tropical montane forest varies with topography. J Trop Ecol 28:23–31

Werner FA (2011) Reduced growth and survival of vascular epiphytes on isolated remnant trees in a recent tropical montane forest clear-cut. Basic Appl Ecol 12:172–181. https://doi.org/10.1016/j.baae.2010.11.002

Yanoviak S, Nadkarni NM, Solano J (2007) Arthropod assemblages in epiphyte mats of Costa Rican cloud forests. Biotropica 36:202–210

Zotarelli HG, Molina JM, Ribeiro JE, Sofia SH (2019) A commensal network of epiphytic orchids and host trees in an Atlantic Forest remnant: a case study revealing the important role of large trees in the network structure. Austral Ecol 44:114–125

Zotz G (2013) The systematic distribution of vascular epiphytes-a critical update. Bot J Linn Soc 171:453–481

Zotz G (2016) Plants on plants – the biology of vascular epiphytes. Springer, Cham

Zotz G, Bader MY (2011) Sampling vascular epiphyte diversity – Species richness and community structure. Ecotropica 17:103–112

# Chapter 8
# Social Insects of the Atlantic Forest

**Rodrigo Machado Feitosa, Maria Santina de Castro Morini, Aline Cristina Martins, Taís Mattoso de Andrade Ribeiro, Fernando Barbosa Noll, Eduardo Fernando dos Santos, Eliana Marques Cancello, and Joice Paulo Constantini**

**Abstract** Despite the outstanding diversity and ecological relevance of social insects in most terrestrial ecosystems, current knowledge of these insects in the Atlantic Forest is still somewhat dispersed in literature. In this chapter, we offer an overview covering the origin, evolution, diversity, functional composition, and distribution of all the eusocial species of ants, bees, wasps (Hymenoptera), and termites (Blattaria, Isoptera) known to occur in the Atlantic Forest. We compiled a database consisting of 1401 species distributed in 189 genera of the two insect orders assessed here. A total of 1250 species of social hymenopterans and 151 species of termites were here recorded for the Atlantic Forest. Additionally, we update the information regarding the state of knowledge, diversity gaps, and prospects for the eusocial insects of the Atlantic Forest. Considering the impressive richness presented in this compilation and the crucial role of social insects in the main ecological processes on Atlantic rainforest landscape, it became urgent to target those organisms in conservation actions and research. A thoughtful study on current, past, and future species distribution of social insects in the Atlantic Forest could indicate

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priority areas for conservation and endangered species in different scales, including in face of climate change.

**Keywords** Ants · Bees · Wasps · Termites · Eusociality · Checklist

## 8.1 Introduction

Eusociality can be defined by cooperative brood care, overlapping generations within a colony of adult individuals, and division of reproductive labor (Wilson and Hölldobler 2005b). This organizational level of animal sociality is a major evolutionary innovation involving adaptations in natural history, morphology, and behavior (Cardinal and Danforth 2011). While the origin of this behavior is still extensively debated (e.g., Johnstone et al. 2012; Legendre and Condamine 2018), eusociality is a widespread syndrome that has independently appeared in several lineages of insects, at least three times in crustaceans, and twice in mammals (Bourke 2011).

Considering that reproductive potential is a basic premise of natural selection, the evolution and persistence of sterile individuals consist in a challenging biological paradox. In fact, even Darwin (1859), in his theory of natural selection, depicted the eusocial insects as a "special difficulty, which at first appeared to [him] insuperable, and actually fatal to the whole theory." There have been several not mutually exclusive hypotheses proposed for the evolution of "worker behavior." Wilson and Hölldobler (2005a) trace the origins of eusociality through a route that starts with solitary organisms acquiring benefits to group behavior, eventually leading to a "point of no return," wherein certain individuals no longer have the physical ability to reproduce and only gain evolutionary fitness indirectly.

In general, colonies of eusocial insects present marked differences between their castes, with queens and fertile males taking the roles of sole reproducers while soldiers, if present, defend the nest and workers forage and maintain resources for the colony (Wilson 1971). The insect order Hymenoptera (ants, bees, and wasps) is the largest and most conspicuous animal group with eusocial species. The social behavior has arisen multiple times within the order, but most hymenopterans are solitary. Eusociality in Hymenoptera is largely attributed to the haplodiploid sex determination system (whereby females arise from fertilized diploid eggs and males arise from unfertilized haploid eggs). This system favors the altruistic behavior, since the relatedness between full-sibling sisters is greater than between a mother and her offspring in monandrous colonies. In this scenario, it would be more advantageous for a haplodiploid female to raise sisters rather than invest in its own offspring (Hamilton 1964a).

The haplodiploid sex determination system, however, does not explain the evolution of the social behavior in all the eusocial insect lineages. Termites (Blattaria, Isoptera) are highly evolved diplodiploid social cockroaches. The origin of eusociality in termites may be related to the fact that these organisms have been ancestrally living within their food, i.e., rotting wood (Thorne 1997). Also, termites present a complex mutualism with cellulose-digesting protozoans and bacteria, in which young individuals acquire these symbionts via anal trophallaxis. The primitively xylophagous habit and the physical dependence of adults to obtain their intestinal symbionts may have favored a longer permanence of brood within the shelter and, consequently, originated the colonial habit in termites (Nalepa 2010). Nevertheless, it is an open issue for termites.

Although important as models for the study of origin and evolution of eusociality, hymenopterans and termites have also been extensively studied regarding their key ecological roles in most terrestrial ecosystems (Richter 2000; Lach et al. 2010; Ollerton et al. 2011; Ahmad et al. 2018). Among the Hymenoptera, bees have been long known as the most important pollinator agents in nature (Bailes et al. 2015), while ants and wasps play an essential role on the regulation of herbivorous insect populations, being also role models for the study of insect-plant interactions (New 2018). Termites are among the most important "soil engineers" in tropical and subtropical environments, with a fundamental impact on soil biophysicochemical processes in forests and grasslands (DeSouza and Cancello 2010).

As one of the most diverse rain forests in the world (Mittermeier et al. 2004), the Atlantic Forest holds a significative number of social insect species (Brandão et al. 2000). However, our knowledge about this prominent portion of the fauna is somewhat fragmented since most comprehensive studies deal with more inclusive categories, as families or functional groups, in local or regional scales, and particular ecosystems (e.g., Gonçalves and Brandão 2008; Cancello et al. 2014; Santos et al. 2014, 2016; Silva and Brandão 2014; Ribeiro et al. 2019).

In this chapter, we provide an overview on the origin, evolution, diversity, ecology, and biogeography of ants, eusocial bees and wasps, and termites. We also update the information regarding the state of knowledge, diversity gaps, and prospects for the eusocial insects of the Atlantic Forest. Finally, we present an unprecedented comprehensive list of the eusocial species of insects known to occur in this biome, based on records contained in scientific papers, online repositories, entomological collections, field endeavors, and unpublished surveys (Supplementary Table 8.1). Hopefully, the information presented here may support the creation of measures for species conservation and recovery plans of the Atlantic Forest, also representing the basis for future research on social insects in this biome.

## 8.2 Ants

### *8.2.1 Origin and Evolution*

With their origin estimated in about 145 million years (Moreau and Bell 2013), ants are the most diverse group of social insects both in species richness and ecological roles (Hölldobler and Wilson 2008). Contrary to the related groups of hymenopterans (bees and wasps), all ant species are truly eusocial and belong to a single insect family, Formicidae. Until the beginning of the twenty-first century, the knowledge about the phylogeny and evolution of ants was marked by the lack of consensus on the delimitation and internal relationships of the main ant lineages. This situation was mainly caused by the disagreement between the phylogenetic proposals of studies that had morphological data as their main source of evidence, not considering the potential for widespread anatomical convergence in different ant lineages, especially in the worker caste (Brown 1954; Baroni-Urbani et al. 1992; Grimaldi et al. 1997).

Thanks to recent initiatives to reevaluate the morphological evidence in ant evolution (Bolton 2003; Keller 2011), a series of phylogenetic studies employing molecular data (Brady et al. 2006; Moreau et al. 2006; Rabeling et al. 2008; Ward et al. 2010, 2015; Schmidt 2013; Branstetter et al. 2017a, b), and the reinterpretation of fossil lineages based on recent discoveries (Dlussky et al. 2004; Engel and Grimaldi 2005; Perrichot et al. 2008, 2016; Barden and Grimaldi 2016), our knowledge of ant phylogeny has considerably improved. In this scenario, the outlines of ant evolutionary history are becoming gradually more apparent.

Moreau et al. (2006), based on a lineage-through-time plot derived from a molecular phylogeny, found evidence for a significant increase in diversification rate of ants about 100 Mya, which corresponds to the rise of angiosperm dominance. However, Pie and Tschá (2009) argued that ant and flowering plants diversification are not related, since the pattern found by Moreau et al. (2006) could be an artifact of incomplete taxon sampling. The same authors (Pie and Tschá 2009) showed that closely related genera have diversities that are more similar to one another than one would expect by chance, suggesting that the capacity for diversification may be itself a biological trait that evolved during the radiation of ants, independently of the rise of angiosperms.

Regarding the precise habitats and conditions favoring the rise of the most successful groups of social organisms on Earth, Lucky et al. (2013) tested the hypothesis that ants arose in the leaf litter ("Dynastic Succession Hypothesis" (Wilson and Hölldobler 2005b)), as well as the alternative "Out of the Ground" hypothesis that ants evolved in the soil and then, secondarily, colonized the leaf-litter and the arboreal strata. By reconstructing the habitat transitions of crown-group ants through time, focusing on where they nest and forage (in the canopy, litter, or soil), and based on ancestral character reconstructions, Lucky et al. (2013) showed that, in contrast to the arguments that ants evolved in tropical leaf litter, the soil is supported as the ancestral stratum of all ants.

### *8.2.2 Diversity and Biogeography*

With more than 13,500 species described in about 330 genera (Bolton 2019), ants are one of the most ubiquitous and widespread groups of animals on Earth. Some estimates suggest that ant total diversity in the globe could exceed 25,000 species

(Wilson 2003), and although this represents less than 1% of all insect species known so far (May 1988), ants may represent at least 15% of total terrestrial animal biomass, including vertebrates (Fittkau and Klinge 1973).

While it is common to emphasize how impressive is the diversity and ecological role of ants in most terrestrial biomes, not all ants share the same level of ecological dominance. Wilson (1976) explored the species diversity, geographical range, variety of adaptations, and local abundance of ants, concluding that three taxa – *Camponotus*, *Pheidole*, and *Crematogaster* – are the most prevalent ant genera worldwide, being the most conspicuous elements of the ant fauna in basically all biogeographic regions. Identifying the underlying causes of such high diversity, however, has been elusive. Wilson (1976) suggests that these three hyperdiverse genera have "conquered the World" not because they share distinctive morphological or behavioral traits, but because they are sufficiently different from one another to allow for their coexistence.

Currently, there are 17 extant subfamilies of ants, but four of these – Dolichoderinae, Formicinae, Myrmicinae, and Ponerinae – account together for about 90% of all known species (Bolton 2019). Despite the studies focusing on the consequences of highly diverse ant clades, their causes such as variation in diversification rates are still superficially understood. Another poorly explored issue involves the relictual ant lineages, with a relictual lineage being defined as a clade showing disproportionately low species richness (often including only one or two species) when compared with other, closely related diverse lineages (e.g., the monotypic ant subfamilies Aneuretinae, Paraponerinae, and Martialinae). Using both analytical and simulation results to assess evolutionary scenarios that could lead to current levels of ant diversity, Pie and Feitosa (2016) found that, despite widespread in Formicidae, such relictual lineages are highly unlikely given constant rate models of speciation and extinction. They suggest that relictual taxa experienced differentially low extinction rates in relation to other contemporaneous lineages by adopting alternative ecologies or colonizing specific environmental strata or regions.

In some cases, the breakup of the continents satisfactorily explains current distributions of ant lineages (e.g., Ward et al. 2010, 2015), with the world's tropics harboring the highest diversity of ant species and biomass (Moreau and Bell 2013). Also, a latitudinal gradient has been observed in Formicidae, with diversity decreasing from the equator to the poles (Guénard et al. 2012). Coupling molecular phylogenetic data with an extensive fossil record and through biogeographic range reconstructions, Moreau and Bell (2013) found that the Neotropics, in particular, were considerably important in the early and continued evolutionary history and biogeography of ants. More specifically, the Neotropical region acted as a museum (where the major ant lineages appeared) and a cradle for continued ant diversification (as suggested by the current high species richness of the region).

### 8.2.3 Ecological Importance and Functional Composition

With a total abundance estimated at more than 110 quadrillion (110,000,000,000,000,000) individuals (Hölldobler and Wilson 1994), ant numerical dominance is visibly reflected in their ecological importance. The main key ecological processes mediated by ants in most terrestrial ecosystems include physically and chemically modifying soil, regulating herbivore populations by predation, primarily or secondarily dispersing seeds, protecting plants against herbivores in obligatory or facultative associations, and engaging in mutualistic interactions with a myriad of other organisms (Lach et al. 2010). Ants are also a model of many mimetic assemblages among different insects and even other arthropods (Hölldobler and Wilson 2008). Along termites and earthworms, the effects of the underground activities of ants on soil and edaphic organisms gave them the title of "ecosystem engineers" (Folgarait 1998). In addition, because of their sensitivity to a range of disturbances, ants are commonly used as bioindicators in land management (Andersen and Majer 2004; Ribas et al. 2012).

Most ants are omnivores, combining predation, scavenging animal carcasses, and consumption of plant-derived resources to a variable extent. Specialized hunters, granivores, primary consumers of plant diets, and even fungivores are also found among ant species (Stradling 1978). However, maybe the most remarkable food habit among ants can be observed in the fungus-farming species (Myrmicinae: Attini: Attina). Attine ants are endemic to the New World and obligately depend on the cultivation of fungus gardens for food. This dependence is so complete that, upon leaving the maternal nest, a young queen must carry within her mouth a fragment of fungus that serves as the starting culture for her new garden (Ihering 1898).

Ant agriculture achieves its evolutionary apex in the leaf-cutting ants of the genera *Acromyrmex* and *Atta*, considered the dominant herbivores of the New World tropics, with colony populations that can exceed five million (5,000,000) individuals (Hölldobler and Wilson 1990). Different from more primitive fungus-farming ants that cultivate their fungus gardens on organic detritus, leaf-cutting ants have acquired the ability to cut and process fresh vegetation (leaves, flowers, and grasses) to serve as the nutritional substrate for their fungal cultivars (Schultz and Brady 2008). Because of their foraging and nesting strategies, leaf-cutter ants have a disproportionately large influence on ecosystem processes as soil physical and chemical characteristics and plant community composition (Swanson et al. 2019).

### 8.2.4 Representativeness and State of Knowledge of Ant Fauna from the Atlantic Forest

The first ant species described and currently registered to the Brazilian Atlantic Forest are present in the initial work of the zoological taxonomy, the *Systema Naturae* (1758) by Carolus Linnaeus (1707–1778), father of modern taxonomy. In this work, among other ant species, Linnaeus described *Atta sexdens* (leaf-cutter ant), *Cephalotes atratus* (turtle ant), and *Odontomachus haematodus* (trap-jaw ant) are widely distributed and easily found in practically all Brazilian territory. Specifically, in the Atlantic Forest, the earliest ant records correspond to specimens collected by the German naturalists Hermann von Lüderwaldt and Hermann von Ihering, at the beginning of the twentieth century (Klingenberg and Brandão 2005).

Currently, some of the most prolific myrmecological research groups in South America are established along the Brazilian coastal strip, within the domains of the Atlantic Forest. Among these, we highlight the two major ant collections in Latin America, the Museum of Zoology of the University of São Paulo (MZUSP), in São Paulo, and the myrmecological collection of the Executive Planning Commission of Cocoa Farming (CEPLAC) in Bahia. Both collections together are the most representative not only for the Atlantic Forest but also to the Neotropical region as a whole, in number of type specimens and ant species, as well as for their geographic coverage.

However, it is basically impossible to consider our knowledge on the Atlantic Forest ants without mentioning the monumental effort by the Biota-FAPESP project, coordinated by Dr. Carlos R. Brandão at MZUSP. In this project, researchers surveyed 26 regularly spaced Atlantic rainforest sites from the southern portion of the biome in the state of Santa Catarina to the northern limit of the Atlantic Forest in the state of Paraíba, Northeast Brazil. Along each site 50 1-m$^2$ leaf-litter samples were collected, and mini-Winkler apparatuses were used to extract the ant fauna. From this massive effort, authors obtained 530 ant species with more than 18,000 records and 1300 1-m$^2$ samples of leaf litter in the 26 areas covered by the Atlantic Forest (Silva and Brandão 2014). Interestingly, a single ant species has proven to be, perhaps, one of the most common organisms of the rich soil macrofauna of the Atlantic Forest. The ant *Strumigenys denticulata* Mayr, 1887, (Fig. 8.1) was found in 2/3 of the leaf-litter/m$^2$ samples of the biome (Silva and Brandão 2010).

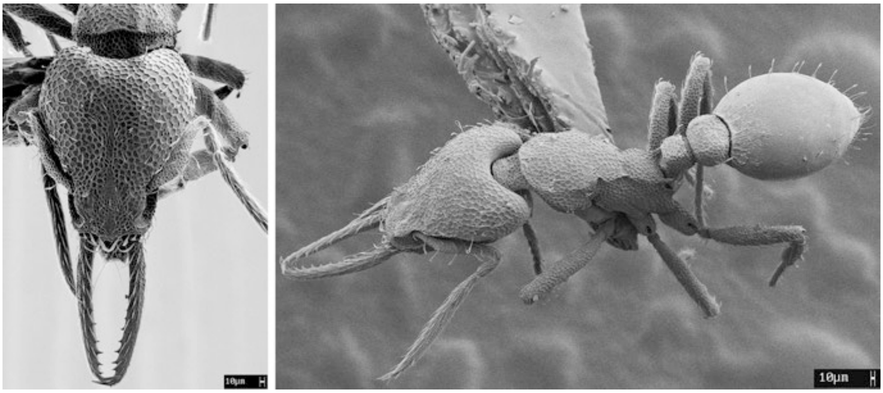

**Fig. 8.1** Worker of *Strumigenys denticulata* (frontal and lateral views), the most frequent and widely distributed leaf-litter ant species in the Atlantic Forest (Silva and Brandão 2010). Scanning electron micrography by Lara M. Guimarães (Museum of Zoology of the University of São Paulo)

Contrary to the expected and most common pattern along latitudinal gradients, the Atlantic Forest leaf-litter ant communities show an inverse pattern in richness, with richer communities in higher than in lower latitudes. It may be due to the fact that an inverse latitudinal gradient in primary productivity and environmental heterogeneity across the Atlantic Forest may affect morphological diversity and species richness, enhancing species coexistence mechanisms and producing the inverse pattern observed (Silva and Brandão 2010, 2014).

In this chapter, we compile the basic data record on the Atlantic Forest ant fauna contained in scientific papers and unpublished monographs. Since not all the identifications could be verified, the occurrences were used only when deemed credible (taxa already recorded from Brazil). Each entry in the table is backed by at least one published reference or data source (the most recent reference including the taxa). The species names' validity and authority were verified in the AntCat platform (Bolton 2019). Species described as morphospecies and subspecies and specimens identified only to a level higher than species were not included in the checklist. As a result, in Supplementary Table 8.1, we list 977 species in 94 genera and ten ant subfamilies known to occur in the Atlantic Forest. The records presented here are though incipient and ephemerous considering that a massive compilation including online repositories, myrmecological collections, field endeavors, and unpublished surveys is being currently prepared and must considerably surpass the diversity recorded here to the ants of the Atlantic Forest (Rogério R. da Silva et al. – Atlantic Ants data paper, in prep.).

### *8.2.5 Knowledge Gaps and Prospects*

As seen, most of our knowledge on ant diversity in the Atlantic Forest is restricted to the leaf-litter stratum (Silva and Brandão 2014). Despite leaf-litter species in tropical forests may represent nearly 70% of the local ant diversity (Kaspari 1996), much still remains to be known about the processes that led to the impressive ant diversity in this stratum. With up to 30 ant species co-occurring in one square meter (Silva and Brandão 2010), the Atlantic Forest offers a unique scenario for future investigation in this field. In addition, only a few studies carried out in the Atlantic Forest have addressed the two "frontiers of knowledge" in myrmecology, the canopy and subterranean ant communities (e.g., Silva and Silvestre 2004; Schmidt and Solar 2010; Da Rocha et al. 2015).

Studies on endemism rates and biogeography of Atlantic Forest ants are also incipient, and the extent to which endemism within the Atlantic Forest ant fauna is associated with particular environments or subregions is yet uncertain (but see Ströher et al. 2019). However, at least three ant genera are exclusively found within the Atlantic Forest domain: *Anillidris* (Dolichoderinae) and *Phalacromyrmex* (Myrmicinae), known to the submontane forests of the southern portion of the biome, and *Diaphoromyrma* (Myrmicinae), which only occurs in lowland forests of the Bahia state, Northeast Brazil. The fact that at least three genera are restricted to the Atlantic Forest suggests that the number of ants endemic to the biome may be considerably high at the specific level. Ants also seem to be an interesting model to assessing the ancient biotic connections between the Amazon Forest and the Atlantic Forest (Ledo and Colli 2017), since many phylogenetic-related taxa are uniquely registered in both biomes (e.g., the myrmicine genera *Cryptomyrmex* and *Oxyepoecus* and the doryline *Sphinctomyrmex*).

The true size of the Atlantic Forest ant fauna is without a doubt substantially higher than the 977 species that we have listed here. Some important ecosystems of the biome have not been intensively sampled for ants, as the highland grasslands and *Araucaria* forests of Southern Brazil and the lowland forests at northeast region. In addition, we still do not have efficient techniques to sample the canopy and subterranean ant assemblages. Nevertheless, the sampling effort that has already been conducted in these environments reveals high rates of species turnover between localities. It is also noteworthy that all the 13 ant species currently included in the Brazilian red list of the endangered fauna are exclusively found in the Atlantic Forest biome (Instituto Chico Mendes de Conservação da Biodiversidade 2018). As a global hot spot of biodiversity historically threatened by extensive anthropogenic disturbance, the preservation of the Brazilian Atlantic Forest remnants is a high priority for maintaining ant biodiversity into the future.

## 8.3 Eusocial Bees

### *8.3.1 Origin and Evolution*

The bees – one of the most charismatic group of insects – comprehend seven families in the superfamily Apoidea (Michener 2007), among which five occur in the Atlantic Forest. They present an expressive array of nesting places and behavior, sociality degrees, floral resource utilization, and morphology (Michener 2007). Every bee species depends on angiosperm flowers, for growing and developing during larval stage and to obtain energy during adult life (Wcislo and Cane 1996). Therefore, they maintain a close association with plants, especially eudicots, since their synchronous origin 125 Mya in the early Cretaceous (Cardinal and Danforth 2013).

The intimate association with flowers differentiates bees from their closest relatives, the apoid wasps, which are mostly carnivore insects (Branstetter et al. 2017a, b). They evolved a series of morphological, behavioral, and physiological adaptations to locate, collect, and feed on floral pollen, nectar, or oil (Thorp 1979). However, the pollinivory per se was not the main driver of bee diversification (Murray et al. 2018). Instead, other factors such as the sociality found in several groups may have driven the enormous bee diversity when compared to their wasp relatives.

Although the most well-known bees – honey bees, bumble bees and stingless bees – are social insects, most bee species are solitary. They obviously do not form colonies, but solitary behavior mainly implies that each female bee is responsible for constructing her own nest and providing it with food, usually dying before the maturation of her offspring (Michener 2007). Many forms of sociality, i.e., intraspecific interactions related to brood care, evolved in different bee lineages (Danforth et al. 2013). On the other hand, eusociality is only found in the corbiculate bees in the family Apidae and is an ancient behavior traced back to Cretaceous (Cardinal and Danforth 2011). Michener (1979b) defines eusocial bees as those who live in colonies consisting of closely related females from two generations (mothers and daughters) of somewhat recognizable castes, i.e., egg layers (or *queens*) and workers. Meliponini (stingless bees) and *Apis* (honey bees) are often called "highly" or "advanced" eusocial. However, the multiple origins of eusocial behavior do not obey an evolutionary sequence, and "fixed-caste" eusocial is a more precise terminology (Almeida and Porto 2014). In a fixed-caste eusocial bee, the queen is not able to survive alone and starts her own nest as do the totipotent gynes of other corbiculates (Michener 1979b).

### *8.3.2 Diversity and Biogeography*

Unlike many insect groups, most of the 20,000 described species of bees prefer subtropical or temperate, xeric regions of the world, especially Mediterranean basin, Californian region, and Central Chile (Michener 1979a). Presumably, bees originated in xeric areas derived from Gondwana (South America or Africa) (Michener 1979a), although a modern treatment of bee biogeography based on phylogenetic comparative methods is lacking. The misbalanced number of species between xeric and humid areas could be partly explained by the ground-nesting behavior presented by most species. Not surprisingly, to escape from the threats of nesting in humid soil, bees occurring in tropical areas use waterproofing products (e.g., floral oils or resins) or nest inside wood (holes in trunks, bark, twigs) or construct aerial nests like many eusocial bees (Michener 2007).

In the Neotropical region, 5000 described species of bees occur (and possibly much more new species), classified in five families: Andrenidae, Apidae, Colletidae, Halictidae, and Megachilidae (Melo 2007). They are distributed from tropical rainforests to deserts, but not reaching extreme environments in high latitudes or altitudes. Apidae, Halictidae, and Megachilidae occur throughout the Neotropics, although the first is much more diverse in tropical areas; Andrenidae and Colletidae are restricted to or much more diverse to subtropical or temperate areas, especially arid and semiarid environments (Melo 2007).

### *8.3.3 Ecological Importance and Functional Composition*

Bees exert a major role in pollinating crop and native flowers, and for this reason, wild bees are one of the most important targets in ecosystem service conservation and food security (Potts et al. 2010; Bailes et al. 2015). In tropical rainforests, the pollination by bees exceeds any other animal pollination (Bawa 1990; Ollerton et al. 2011). Crescent literature demonstrates the ongoing decline of bee fauna, consequently threatening pollination services, due to intense habitat loss, climate change, pesticides, and alien and pathogen species (Vanbergen et al. 2013; Klein et al. 2017; Potts et al. 2010).

Functionally, bees can be considered herbivore insects, since all species depend on floral sources. However, some bees, called cleptoparasites, do not collect food sources for their own offspring but lay eggs on other bees' nests (Wcislo and Cane 1996) instead. Notwithstanding, both female and male cleptoparasitic species still forage on flowers for nectar for their own maintenance.

### *8.3.4 Representativeness and State of Knowledge of Eusocial Bee Fauna from the Atlantic Forest*

In tropical humid areas, like the Atlantic rainforest, eusocial bees of the tribe Meliponini (Apidae family) dominate in both abundance and species number (Gonçalves and Brandão 2008; Gonçalves et al. 2012). Other eusocial species include those of *Bombus*, a Holarctic group with only seven Brazilian species, and the introduced Africanized honey bee *Apis mellifera*. In the present chapter, we will consider only the fixed-caste eusocial species (see definition above), which in Brazil is represented by the Meliponini tribe.

Meliponini or stingless bees are the most diverse lineage of corbiculate bees (about 450 species, 200 of which occurring in Brazil (Pedro 2014)). Approximately 60% of bee species caught by different methods in the Atlantic Forest are stingless bees, but this number decreases in southern latitudes (Gonçalves and Brandão 2008; Gonçalves et al. 2012). Due to their large colonies and resource requirements, they prefer mass-flowering plants that offer a large amount of pollen, preferably the dominant families Asteraceae, Myrtaceae, and Melastomataceae, of which they are the most important pollinators (Wilms et al. 1996). Usually, mass-flowering plants are high trees, thus stingless bee foraging pattern follows a stratification gradient that favors the canopy (Ramalho 2004). Therefore, it is often challenging to collect stingless bees in the Atlantic Forest, thus requiring specific methods, such as nets attached to long poles and tree climbing.

To compile the species list presented in this chapter (Supplementary Table 8.1), Meliponini bee fauna from Atlantic Forest domain was retrieved from relevant entomological collections deposited in GBIF (https://www.gbif.org/) and SpeciesLink (http: //www.splink.org.br/), totalizing approximately 27,000 records. For downloading, cleaning, and constructing of distribution maps, a series of R (R Core Team 2016) packages were used, especially SpeciesGeoCoder (Töpel et al. 2017), following the scripts from Alexander Zizka (https://github.com/azizka/). Additional records from the literature were compiled (Melo and Costa 2004). All species from museum collections and literature were checked on the Moure's Bee Catalogue (Camargo and Pedro 2013) for their validity, spelling, and author names.

A compiled list of Meliponini bee fauna from the Atlantic Forest is presented on Supplementary Table 8.1. Atlantic stingless bee fauna is composed of 21 genera (*Camargoia*, *Cephalotrigona*, *Friesella*, *Frieseomelitta*, *Geotrigona*, *Lestrimelitta*, *Leurotrigona*, *Melipona*, *Mourella*, *Nannotrigona*, *Oxytrigona*, *Paratrigona*, *Partamona*, *Plebeia*, *Scaptotrigona*, *Scaura*, *Schwarziana*, *Tetragona*, *Tetragonisca*, *Trigona*, and *Trigonisca*) and 75 valid species, among which three are endangered of extinction.

### *8.3.5 Knowledge Gaps and Prospects*

Large regions of Atlantic Forest domain are still underexplored (see map on Fig. 8.2a), and most are punctual sampling using specific collecting methods (see Gonçalves and Brandão 2008; Gonçalves et al. 2012), thus possibly underestimating the species diversity. A combination of methods that explores high canopies and suitable nesting sites (e.g., trunk holes) would be ideal to an exhaustive knowledge of the Atlantic Forest's Meliponini bees. However, this exhaustive approach has never been applied on a single area. In addition, most of the 75 species found are unknown regarding their ecological aspects, nesting sites and behavior, floral preferences, or natural enemies. Therefore, a conclusion about their real situation under the severe threat faced by the Atlantic Forest is hampered by this knowledge gap.

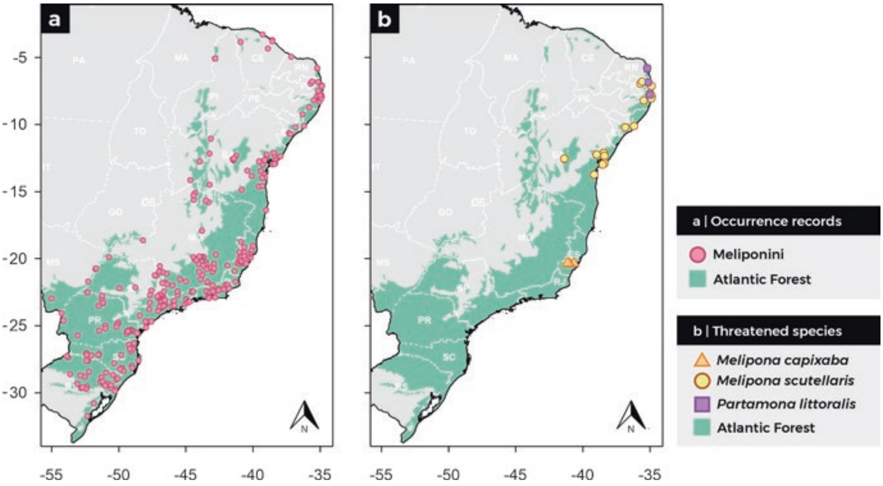

**Fig. 8.2** Distribution maps for stingless bee species (Meliponini) occurring in Atlantic Forest domain (sensu Lei da Mata Atlântica). (**a**) Occurrence records for all Meliponini species (pink circles). (**b**) Occurrence records for three threatened species from the Atlantic Forest: *Melipona capixaba* (orange triangles), *Melipona scutellaris* (yellow circles), and *Partamona littoralis* (purple squares). Letters indicate the abbreviations for the Brazilian states

In spite of the widespread occurrence and abundance of stingless bees in the Atlantic Forest, some species have very restricted distribution, increasing their vulnerability (e.g., *Schwarziana bocainensis* and *Melipona capixaba*). Three stingless bee species from the Atlantic Forest were recently included in the Brazilian red list of threatened fauna: *Melipona* (*Michmelia*) *capixaba* Moure and Camargo, 1994; *Melipona* (*Michmelia*) *scutellaris* Latreille, 1811; and *Partamona littoralis* Pedro and Camargo, 2003 (ICMBio 2018). Allied to increasing habitat loss, those and other stingless bee species suffer with intense exploitation by collectors on native nests, often destroying them to collect honey or the entire nest to keep in private collections. The predatory action of collectors must be differentiated from responsible beekeeping.

Considering the crucial role of stingless bees to the pollination of the most important vegetal elements on Atlantic rainforest landscape, it became urgent to target those bees in conservation actions and research. A profound study on current, past, and future species distribution in the Atlantic Forest could indicate more endangered species in different scales, including in face of climate change.

## 8.4 Eusocial Wasps

### *8.4.1 Origin and Evolution*

Eusocial wasps are represented by species of two families of Hymenoptera, Crabronidae, and Vespidae, and the latter presents the great diversity of social behavior reported for wasps. The origin of the eusocial behavior in Vespidae is yet somewhat controversial, with two concurrent hypotheses: (1) a single origin, with Stenogastrinae as the sister group of Vespinae+Polistinae (Pickett and Carpenter 2010; Piekarski et al. 2014), and (2) dual origin, with Stenogastrinae as sister group of all other subfamilies of Vespidae (Hines et al. 2007; Piekarsky et al. 2018). These two hypotheses are correlated to the data type used in the phylogenetic analyses, in order that the dual origin is obtained with molecular data alone, while the monophyly is obtained when phenotypic (morphology and behavior) characters are employed, even together with molecular data (Hines et al. 2007; Piekarsky et al. 2014). On the other hand, the phylogenetic relationships within Polistinae are more consensual, and the hypothesis obtained from molecular data corroborates that one proposed from total evidence, in which phenotypic characters are included (Pickett and Carpenter 2010; Piekarsky et al. 2014, 2018).

Irrespective of the controversies in terms of the phylogenetic relationships, it is clear that independent foundation is ancient and swarm foundation is derived from all phylogenetic hypotheses for paper wasps. In all cases, independent founders present small colonies which live less than one year, and swarm founders (a homoplastic condition that occurs several times for social wasps) produce long-living large colonies. Also, in this evolution of social traits, it is possible to note that queens dominate and centralize decisions in independent founders, while in swarm founders, decisions are decentralized and workers take responsibility for several decisions (Noll 2013).

### 8.4.2 Diversity and Biogeography

The eusociality in Crabronidae is typical of a single genus, *Microstigmus* Ducke, which comprises 27 Neotropical species (Amarante 2002), while Vespidae includes 996 eusocial species of three different subfamilies: Polistinae, Stenogastrinae, and Vespinae. Vespinae embraces 69 highly eusocial species and shows a wide distribution, occurring in the Australian, Nearctic (a few invasive species reached the northern Neotropical), Oriental, and Palearctic regions (Carpenter and Kojima 1997; Pickett and Carpenter 2010). Stenogastrinae, on the other hand, is restricted to the Oriental region and includes 58 species primitively eusocial (Carpenter and Kojima 1996; Pickett and Carpenter 2010). Polistinae, the most diverse group of social wasps in number of species as well as in social behaviors, which vary from primitive to highly eusocial, shows cosmopolitan distribution, with greater diversity in the Neotropical region (Jeanne 1991; Carpenter 1996; Carpenter and Kojima 1997; Pickett and Carpenter 2010).

The Polistinae are commonly known as paper wasps and are taxonomically classified in four tribes: Epiponini (19 Neotropical genera), Mischocyttarini (one genus, *Mischocyttarus* Saussure, with occurrence in the Neotropical region and reaching the southern Nearctic and British Columbia), Polistini (one cosmopolitan genus, *Polistes* Latreille), and Ropalidiini (four genera occurring in the Afrotropical, Oriental, and Australian regions). The sociality of Polistinae can be divided in two behavioral groups: (1) independent founding (ancient condition), in which reproductive females (queens) initiate a colony singly or with other few reproductive females but without workers (nonreproductive individuals), and (2) swarm founding, in which a new colony is founded by one or more reproductive individuals accompanied by workers (Gadagkar 1990; Noll 2013). The first group includes *Polistes*, *Mischocyttarus*, and three genera of Ropalidiini, *Belonogaster* Saussure, *Parapolybia* Saussure, and *Ropalidia* Guérin-Méneville (partially). The other behavioral group, swarm founding, has all Epiponini and two genera of Ropalidiini, *Polybioides* Buysson, and *Ropalidia* Guérin-Méneville (some species) (Jeanne 1991; Gadagkar 2001; Noll 2013).

Independent-founding wasps build small nests, and queen's dominance is established by physical aggressive behaviors, while the swarming wasps' nests are usually bigger and architecturally more complex than the independent wasps' nests (Jeanne 1991; Wenzel 1998; Gadagkar 2001; Noll 2013). Also, differently from the independent-founding wasps, the reproductive dominance in Epiponini is established during the immature phase, with the development of an individual that could bear physiological and/or morphological differences for the other nest mates (Noll et al. 2004). Epiponini swarms are modulated by pheromones (Jeanne 1980, 1981, 1991), which can be released in the air (Hunt et al. 1995; Howard et al. 2002; Mateus 2011) or left on leaves, or other substrates present between the old and new nest locations (Jeanne 1981; Mateus 2011).

Species of *Microstigmus* show primitive eusocial organization (Gadagkar 2001), with no morphological distinction between reproductive and nonreproductive individuals (Matthews 1968, 1991; Lucas et al. 2011), but present some reproductive division of labor (Ross and Matthews 1989). Moreover, some species perform mass provisioning, while other ones show progressive provisioning after laying an egg in an empty cell (Matthews 1991).

### *8.4.3 Ecological Importance and Functional Composition*

The knowledge on the ecological aspects of the species of *Microstigmus* is quite incipient, and it is restricted to feeding habit and to the activity period, defining the *Microstigmus* as diurnal predatory wasps of Collembola, leafhoppers (Insecta: Hemiptera, Cicadellidae) or trips (Insecta: Thysanoptera) (Gadagkar 2001; Asís 2003). Paper wasps are generalist predators of arthropods, like spiders and other insects, and some species can act as necrophagous of invertebrates and vertebrates to obtain the protein necessary to feed their offspring (Evans and West-Eberhard 1970; Richter 2000; Oliveira et al. 2010). However, a facultative prey preference has been reported for social wasps (Richter 2000), since foragers usually return to sites of previous successful hunting and may catch repeatedly the same prey (Takagi et al. 1980; Richter 1990). Moreover, some preference for generalist herbivores has also been reported for social wasps (Richter 2000).

Considered one of the most important prey sources for several species of social wasps (Hunt et al. 1987; Richter 1990, 2000), caterpillars are usually specialist herbivorous insects and feed on species of one or a few genera or of a single subfamily or family of plants (Bernays 1988; Bernays and Graham 1988). Caterpillars with narrow diet present more effective chemical defenses against predators than generalist species due to casual absorption (Jones et al. 1989) or the physiological adaptations that enable the specialized herbivores to accumulate the phytochemicals consumed over time (Dyer 1995). This specialization may be established by the selective pressure exerted by predators on the generalist herbivores (Bernays 1988; Bernays and Graham 1988; Bernays and Cornelius 1989; Hay et al. 1989).

During the adult stage, social wasps require only carbohydrates, which are obtained from flowers (nectar), aphids (honeydew), sugary exudates, or fruits (Evans and West-Eberhard 1970; Hunt et al. 1987; Letourneau and Choe 1987; Oliveira et al. 2010). As a consequence of the flower visiting, social wasps can act as pollinators (Vieira and Shepherd 1999; Brodmann et al. 2008; Burger et al. 2017), so that their flower-visiting networks show high generalization, with wasp preferring the most abundant plants and presenting great niche overlapping (Mello et al. 2011). Mello et al. (2011) detected some modularity in the flower visiting, which was established by hub wasps connected to several plant species with fewer connections. One of this hub species is *Brachygastra lecheguana* (Latreille), and its modularity is probably determined by the fact that it stores nectar in addition to animal food source for their brood. However, more studies are necessary to confirm such proposition. Based on the high generalization of the paper wasps to visit flowers, the pollination role, and their regular predatory action on the population herbivores, Mello et al. (2011) suggest that the importance of social wasps as mutualists for the maintenance of plant populations is higher than it has been previously assumed.

Besides the diet demand, paper wasps also forage for water and plant material or mud to build their nests (Wenzel 1998; Richter 2000). Species of *Microstigmus* also use plant material, embedding them in a silk matrix (Matthews and Starr 1984; Matthews 1991). Different from the flower-visiting networks, there is not any study about the structure of the social paper wasps' nest material-collecting networks. Another important ecological aspect is that most social paper wasps are diurnal, but species of *Apoica* Lepeletier are nocturnal (Hunt et al. 1995), indicating that species of this genus show different function in the ecosystems.

### 8.4.4 Representativeness and State of Knowledge of Eusocial Wasp Fauna from the Atlantic Forest

The fauna of social wasps, in the Atlantic Forest, is represented by a total of 198 species, 188 Polistinae and 10 *Microstigmus* (Crabronidae). Such species number is equivalent to 75% and 10% of all mammals and terrestrial vertebrates, respectively, known for the Atlantic Forest, considering the numbers presented by Ribeiro et al. (2011). The richest group of Polistinae, in the Atlantic Forest, is Mischocyttarini, with 85 species of *Mischocyttarus*, followed by Epiponini, with 72 species and 16 genera, and Polistini, with 31 species of *Polistes* (Fig. 8.3).

In total, 65 species of eusocial wasps are endemic to the Atlantic Forest. *Mischocyttarus* is the taxon with the highest endemism, with 46 species (almost 55% of the known species of the genus), followed by *Microstigmus*, with ten endemic species, Epiponini, with seven species, and *Polistes*, with five species. However, the high endemism status of *Mischocyttarus* may change by new revisionary studies, since many endemic species were reported only by the original descriptive papers. Silveira (2019) synonymized eight names, so that six of them had endemic status for the Atlantic Forest. The endemic status of the species of *Microstigmus* is also determined by records restricted to the type of locality but differently from *Mischocyttarus*; the taxonomy of *Microstigmus* is well resolved although few studies on the diversity of Crabronidae have been performed in the domain of the Atlantic Forest. One of the endemic species of Epiponini, *Agelaia vicina* (Saussure), builds huge nests, with a colossal number of individuals (over one million) (Zucchi et al. 1995), and was pointed out as a keystone species, since it can provoke great predation impact on the arthropod population and influence

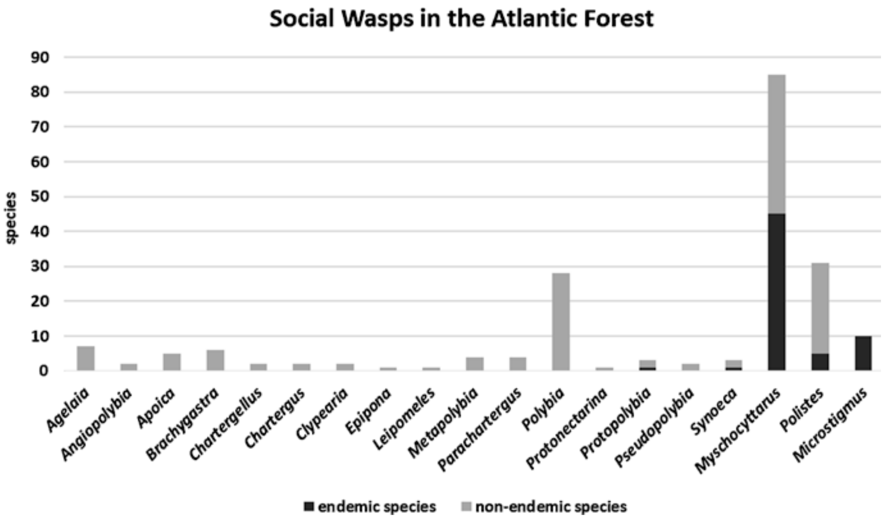

**Fig. 8.3** Number of endemic and non-endemic wasp species reported by genus for the Atlantic Forest

substantially the scavenging dead vertebrates and invertebrates (Oliveira et al. 2010).

On the other hand, many species of Polistinae are widespread in the Neotropical region (Silva and Noll 2015), so that some of them, like *Angiopolybia pallens*, *Agelaia angulata*, and *Polistes carnifex*, show discontinuous distribution and occur in Amazonian and Atlantic forests. These two biomes have historical relationships, which were established as a consequence of the forest expansion and retraction (Prance 1987; Por 1992; Costa 2003). Silva and Noll (2015), based on the distribution of the species of *Brachygastra* Perty, pointed out that the Atlantic Forest is historically related to the southeastern Amazon. The events of distension and retraction were modulated by historical climatic variation (Pennington et al. 2004; Carnaval and Moritz 2008), resulting in different areas of endemism (Silva et al. 2004; Werneck et al. 2011; Hoffmeister and Ferrari 2016; Garrafoni et al. 2017) and contributing to the establishment of the current biodiversity patterns of the Atlantic Forest (Araújo et al. 2008; Carnaval and Moritz 2008; Sandel et al. 2011). The Atlantic Forest's areas of endemism served as the basis for a conservation proposal that accounted three biodiversity centers (Conservation International Brazil et al. 2000; Silva and Casteleti 2003; Fonseca et al. 2003; Ayres et al. 2005; Tabarelli et al. 2005, 2010). However, for the social wasps, only two biodiversity centers have been recognized, once nonecological or molecular distinctions have been found between the northern and central centers (Carvalho 2014; Santos et al. 2016).

Even though the Atlantic Forest is one the most diverse and threatened biome in the world, very little is known about the diversity patterns of social wasps and the processes related to the establishment of such patterns. Ribeiro et al. (2019) described the variation in paper wasp richness, indicating that it decreases from 630 meters above the sea level as altitude increases. However, the lower altitudes were not included in the study; that is unfortunate because Atlantic Forest lowlands have received more anthropic impact than higher altitudes (Ribeiro et al. 2011). Since predatory wasps are sensible to forest fragmentation (Santos et al. 2014), this lack of data for lowlands prevents the recognition of a complete variation in species richness in this biome.

### 8.4.5 Knowledge Gaps and Prospects

Most studies on social wasps from the Atlantic Forest consist of local inventories, and few studies have treated wider question, like regional diversity patterns, historical biogeography, anthropogenic impacts on the fauna of social wasps, or intrinsic question related to the ecology, biology, and evolution of social wasps.

One of the oldest ecological questions related to the paper wasps is the relationship between the colony size and latitudinal gradients, so that colony size increases with latitude (Jeanne 1991). Jeanne (1991) highlights that the local predator-parasite pressure could have a major effect on colony size over much of the species ranges, so that larger nests could be an adaptation to the high predation pressure imposed by vertebrates and low pressure by army ants, which is considered the major predatory force that influences the colony size in wet habitats at the equatorial region (Kaspari and O'Donnell 2003). The Atlantic Forest presents suitable conditions to carry out studies that evoke problems like that, since it stretches over 27° of latitude and includes great habitat heterogeneity (Ribeiro et al. 2011). Moreover, the same relationship could be investigated for altitudinal variations, once the Atlantic Forest also covers a great elevational variation along its latitudinal distribution (Ribeiro et al. 2011). Some other aspects, like seasonal synchrony and nesting cycle, could change along latitudinal gradients and should be also investigated (Jeanne 1991).

## 8.5 Termites (Blattaria, Isoptera)

### 8.5.1 Origin and Evolution

Termites, or "social cockroaches" as Wilson (1971) called them, are currently classified in the order Blattaria and infraorder Isoptera (Krishna et al. 2013). The group is considered monophyletic in the most recent and more comprehensive articles (Inward et al. 2007; Legendre et al. 2008, 2015; Bourguignon et al. 2015, 2016b). Termites, along with other cockroaches and mantises, are members of Dictyoptera, also a monophyletic clade of Polyneoptera, all hemimetabolous insects.

Termites live in colonies with reproductive (queen and king) and sterile individuals (soldiers and workers or helpers) organized in castes: workers, responsible for almost every task in the colony; soldiers, responsible for defense; and the royal pair, both formerly alates that lost their wings after the nuptial flight. Termites have very different caste systems than Hymenoptera, with true soldiers, i.e., individuals with conspicuously different morphology from workers and with remarkable adaptations for defense; true workers; or pseudergates (= false workers, a totipotent juvenile, which can become soldier or reproductive or can undergo "stationary molts" or "regressive molts," meanwhile it functions as a worker). Their caste system is very complex, considering the different taxonomic groups, which is beyond the scope of this chapter (for a more comprehensive explanation, see Korb and Hartfelder 2008; Matsuura 2010; Roisin and Korb 2010).

Legendre et al. (2015) rooted a phylogenetic analysis (~800 taxa, 10 kbp) suggesting that crown Dictyoptera date to the Late Carboniferous (~300 Mya) and that the most recent common ancestor of the clade (cockroaches + termites) dates to the Permian (~275 Mya) and stem termites to the Early Jurassic (~195 Mya) and a crown diversification in the Late Jurassic (~150 Mya). Bourguignon et al. (2015) gave a more recent date (136–170 Mya), while other authors proposed an older origin (Davis et al. 2009; Ware et al. 2010). Following Legendre et al. (op cit), their dating estimates suggested that termites could have been the first extant insect lineage to evolve eusociality. To address the fossil record currently known, see Krishna et al. (2013), Engel (2016), and Zhao et al. (2019).

Termites have been challenging the principal theory proposed to explain Hymenoptera eusociality (Hamilton 1964a, b), because these insects are haplodiploid, with a genetic difference between females (diploid) and males (haploid) and a consequent relatedness asymmetry among castes (i.e., relatedness among sisters is higher than daughter-mother) as the explanation for altruism and so the most important driver for eusociality. Termites are diplodiploid insects without the genetic difference between male and female; nevertheless, some proposals were made as an analogy with the same mechanism as in Hymenopteran (see a comprehensive revision in Thorne 1997). Nevertheless, it is important to emphasize that haplodiploidy is not a determinant path toward eusociality, as some researchers have been discussing, for instance, Bourke and Franks (1995), among others. For a comparison of eusocial evolution in termites and hymenopterans, see Howard and Thorne (2010). Another approach has been developed by Nalepa in many articles, where she proposed discussions on the origin of termite eusociality related to the altricial development in wood-feeding cockroaches, mainly Cryptocercidae, the sister group of termites (Nalepa 2010). Korb and Heinze (2004), although recognizing the fundamental role of relatedness in the evolution of eusociality, consider it would be less important for its maintenance and stated that "Insect societies can thus be regarded as a level of selection with novelties that provide benefits beyond the scope of a solitary life." Korb (2008), going deeper in the same approach, discussed the ecological drives that underline the evolutionary transitions in termite eusociality and compared with those in cooperative breeding vertebrates, providing potential explanations of why eusociality is so rare in vertebrates, despite the similar ecological pressures on both groups. Bourguignon et al. (2016a), hypothesizing that the true worker phenotype (wingless) originated as a dispersal strategy for fertile wingless individuals before eusociality, developed very different ideas from all the above-mentioned authors.

### 8.5.2 Diversity and Biogeography

Krishna et al. (2013) considered nine extant families: Mastotermitidae, Archotermopsidae, Hodotermitidae, Stolotermitidae, Kalotermitidae, Stylotermitidae, Serritermidae (exclusively Neotropical), Rhinotermitidae, and Termitidae. Five occur in the Neotropical region: the last three plus Stolotermitidae and Kalotermitidae. Termites have been separated into the traditional and paraphyletic "lower" termites and the monophyletic "higher" termites, represented only by Termitidae, while all other families are included in the former (lower), which all feed on wood and harbor flagellated protists as intestinal symbionts, responsible for the digestion of cellulose. Termitidae comprises about 75% of the extant species and is the most diverse in all criteria, with broader distribution, and comprehends most diverse lifestyles, that is, defense strategies, including nest construction, reproductive strategies, and feeding habits, in a gradient of humification from hard wood (*xylophages*) to humus (humivores, soil feeders, or humus feeders), including wood in different breakdown stages, grass, and litter, being classified in different ways (Donovan et al. 2001; Bourguignon et al. 2011). All species of this family have lost the flagellated protist symbionts, which were replaced by a vastly diverse microbial community of prokaryotes as bacteria (see Bignell (2010), Ohkuma and Brune (2010), Brune and Ohkuma (2010)). Termitidae includes the subfamilies Sphaerotermitinae, Macrotermitinae, Foraminitermitinae (all absent from the Neotropics), Apicotermitinae, Nasutitermitinae, Termitinae, and Syntermitinae (exclusively Neotropical). Worldwide, Isoptera includes about 3100 species, with several hundred genera in the tropics (≤23.5° N and S), while only a few species reach latitudes ≥40° north or south. Termites are absent from high altitudes, and their diversity sharply declines above 800 m; nevertheless, Scheffrahn (2015) recorded *Rugitermes laticollis* (Kalotermitidae) at 2800 m (and its type locality is La Paz, Bolivia, at 3700 m). Termites are distributed unequally among continents; the primary cause is historical, as Termitidae likely originated about 50 Mya in Africa and many dispersal events have occurred to produce the current worldwide distribution (Bourguignon et al. 2016b). Consequently, the wood feeders occur worldwide, sometimes in a relict Gondwanan distribution (e.g., Stolotermitidae), while the soil feeders are more abundant in Africa, followed by South America and Southeast Asia but almost absent in Australia. The fungus-growing termites (Macrotermitinae) occur only in Africa and Southeast Asia. For a comprehensive view of termite evolution, diversity, phylogeny, and the fossil record, see Krishna et al. (2013, vol 1: p 147–170).

### *8.5.3 Ecological Importance and Functional Composition*

Abe (1987) grouped the termites in three life types, which Shellman-Reeve (1997) slightly modified (in parentheses): "one-piece type" (single-site nesters) that nests in wood and consumes that wood (Stolotermitidae, Archotermopsidae, Kalotermitidae); "intermediate type" (multiple-site nesters) that nests in wood, feeds on it, and also constructs galleries to other wood pieces (Mastotermitidae, Stylotermitidae, and most Rhinotermitidae); and "separate type" (central-site nesters) that nests in diverse sites (on living or dead trees, in soil, on ground surface, etc.) and constructs galleries to access different sources of dead plant material, i.e., from wood to humus (mainly Termitidae). Shellman-Reeve (1997) added the "inquiline-site nesters," for example, *Serritermes serrifer* (Serritermitidae) which lives inside nests of *Cornitermes* spp. (Termitidae, Syntermitinae) and seems to feed on the small pieces of wood inside the host nest's wall (Barbosa and Constantino 2017). The lifestyles of many species are unknown, such as many of the Apicotermitinae, which is a soldierless group in the Neotropics and is mostly found dispersed in the soil or inside nests constructed by other species.

Called "soil ecosystem engineers" (Jones et al. 1994), termites are central to tropical and subtropical ecosystems as primary consumers and detritivores and have a remarkable diversity of significant roles in soil biophysicochemical processes (DeSouza and Cancello 2010), mainly through their constructions (nests above and below ground, galleries) and feeding activities (foraging tunnels and their sheetings plus feeding on soil, influencing chemical, physical, and hydraulic soil proprieties). Termite biomass comprises about 45–65% of overall soil macrofaunal biomass at some sites and is comparable to the ungulates and megaherbivores in Africa (Loveridge and Moe 2004). Jouquet et al. (2016) concisely reviewed this matter, emphasizing that termite roles in tropical soils are still neglected compared to earthworms in temperate regions. The immense biomass of termites and their ecological success result mainly from two characteristics: eusociality and the capacity to digest cellulose through intestinal microsymbionts. Some references from the vast literature on termite importance in tropical soils are mentioned here, as a guide (Lee and Wood 1971; Black and Okwakol 1997; Holt and Lepage 2000; Schaefer 2001; DeSouza and Cancello 2010; Ackerman et al. 2007; Pardeshi and Prusty 2010).

### *8.5.4 Representativeness and State of Knowledge of Termite Fauna from the Atlantic Forest*

The primary source of data in Supplementary Table 8.1 is the termite collection of the Museum of Zoology of the University of São Paulo (MZUSP). Supplementary Table 8.1 lists 151 species of the three families (Kalotermitidae, Rhinotermitidae, and Termitidae) that occur in the Atlantic Forest.

The Atlantic Forest definition adopted (IBGE) encompasses many different ecosystems, from mangroves, restinga forests, and highland grasslands to ombrophilous dense forests, which makes it difficult or impossible to discuss a single "termitofauna" of this biome. Cancello et al. (2014) surveyed termite species through the ombrophilous dense forest at 15 regularly spaced sites from 7° S to 27° S, using a standardized sampling protocol. Briefly, the total observed species richness and abundance were negatively related to latitude, which was explained mainly by differences in temperature, rainfall, and a proxy for energy (potential evapotranspiration). Of the 87 morphospecies found, an estimated 50% were new (Oliveira et al. 2015; Oliveira and Constantino 2016). Epigean and arboricolous termite nests (Fig. 8.4a) were absent in latitudes higher than 21° S, and only the small epigean nests of *Anoplotermes pacificus* were found in the southern ombrophilous dense forest. In the Neotropics, the majority of epigean nests are constructed by species of Syntermitinae, and many species of Termitinae are inquilines inside them. So, the low representation of Syntermitinae, mainly in the south (>21° latitude), may explain the low representation of the soil-feeding Termitinae. As the Apicotermitinae is poorly known, Constantini (2018) carried out her doctoral research on that material, along with other samples housed in the collection (Schlemmermeyer 2000;

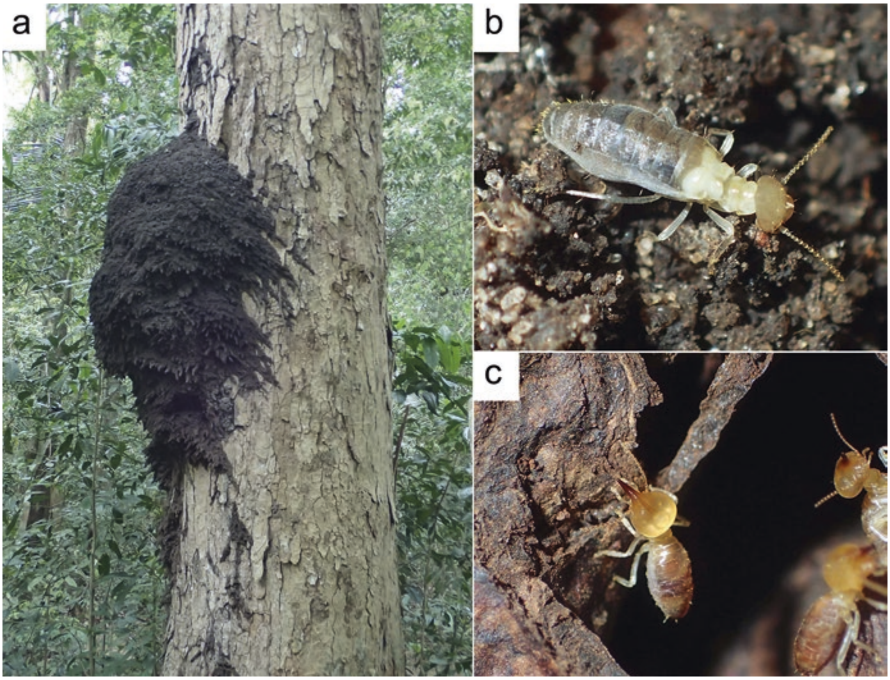

**Fig. 8.4** Examples of termites occurring in the Atlantic Forest. (**a**) *Labiotermes labralis*, arboricolous nest, occurring in the northeastern Atlantic Forest. (**b**) *Dissimulitermes invisibilis*, worker. (**c**) *Nasutitermes aquilinus*, soldier, common species in Atlantic domain, outside the ombrophilous dense forest

Reis and Cancello 2007), and found 23 species of which 20 were new (Constantini et al. 2020) (Fig. 8.4b). Reis and Cancello (2007) compared taxocenoses from different forest types (ombrophilous dense forest, mesophyll forests, and seasonal dry tropical forests) in southern Bahia. Their results showed different composition, richness, and abundance of termites among these environments, with the rain forest being the richest and the liana forest the poorest in number of species. *Microcerotermes* (Termitinae) (Supplementary Table 8.1) is represented only by morphospecies in the Atlantic Forest material from the MZUSP collection; the taxonomy of this genus is confused, although its species are frequent and abundant in the biome.

### 8.5.5 Knowledge Gaps and Prospects

The MZUSP collection houses 6445 samples from the Atlantic Forest, of which 3000 are identified only to genus level. Considering that collections are the working material for taxonomists and systematics, these numbers give an idea of the amount of material from the Atlantic domain yet to be studied. On the other hand, many of these samples have not yet been examined by the curators and perhaps are neither difficult to identify nor represent new taxa. Except for the ombrophilous dense forest, all other ecosystems lack termite fauna surveys; perhaps the least known is highland grasslands. In a taxonomic approach, some brief considerations follow. Kalotermitidae: this family is underrepresented in every biome; nevertheless, Schlemmermeyer (2000) has found many occurrences in the well-preserved Boracéia Biological Station (BBS – MZUSP), in standing dead trees alone, which means that with appropriate collecting techniques, it would be possible to enlarge collections and understand its actual role in the forest. Rhinotermitidae: the species have potential pest status and occur in all formations; except for the exotic pest species *Coptotermes gestroi* (Ferraz and Cancello 2001, 2004), the life histories of the other species are unstudied. Termitidae: Apicotermitinae: in spite of the study by Constantini (2018), many samples (about 300) from other formations of the Atlantic Forest remain to be identified and are a priority for taxonomic studies. Nasutitermitinae is the most frequent and abundant group found in many surveys, and *Nasutitermes* (see Fig. 8.4c) urgently requires taxonomic revision (nearly 950 samples have not been determined to species level), along with other groups, such as the smallest nasutes ((*Subulitermes*, *Araujotermes*, and *Atlantitermes*, now under revision by Carolina Cuezzo (Cuezzo and Cancello 2018)). Syntermitinae is the best known, as many genera have been revised and a recent phylogenetic hypothesis was proposed by our group (Rocha et al. 2017); this family is least represented, at least in the ombrophilous dense forest. Termitinae: there are about 600 identified only to genus; *Dentispicotermes* is under revision in the MZUSP Termite Laboratory (by Isabel Mosch), and *Microcerotermes*, as mentioned above, is in need of revision, as is *Termes*.

In conclusion, as the Atlantic domain is one of the most threatened biomes in Brazil, all studies in all its ecosystems have high priority.

## 8.6 Concluding Remarks

In this compilation, we listed 1401 species distributed in 189 genera and seven families of the two insect orders assessed here. A total of 1250 species of social hymenopterans and 151 species of termites were recorded in the Atlantic Forest. Ants were the most speciose group with 977 species, followed by wasps (198), termites (151), and bees (75). As a widely known biodiversity hot spot historically threatened by anthropogenic disturbance, the preservation of the Atlantic Forest remnants is a high priority for maintaining its biodiversity into the future. In this sense, considering the impressive richness presented in this compilation and the crucial role of social insects in the main ecological processes on Atlantic rainforest landscapes, it is essential to target those organisms in conservation actions and research. Comprehensive studies on current, past, and future species distribution of social insects in the Atlantic Forest could indicate priority areas for conservation and endangered species in different scales, including in face of climate change.

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## References

Abe T (1987) Evolution of life types in termites. In: Kawano S, Connell JH, Hidaka T (eds) Evolution and coadaptation in biotic communities. University of Tokyo Press, Tokyo

Ackerman IL, Teixeira WG, Riha SJ et al (2007) The impact of mound-building termites on surface soil properties in a secondary forest of Central Amazonia. Appl Soil Ecol 37(3):267–276

Ahmad SK, Dawah HA, Khan MA (2018) Ecology of termites. In: Khan MA, Ahmad W (eds) Termites and sustainable management. Sustainability in plant and crop protection. Springer, Cham

Almeida EAB, Porto DS (2014) Investigating eusociality in bees while trusting the uncertainty. Sociobiology 61(4):355–368

Amarante STP (2002) A synonymic catalog of the neotropical Crabronidae and Sphecidae (Hymenoptera: Apoidea). Arq Zool S Paulo 37:1–139

Araújo MB, Nogués-Bravo D, Diniz-Filho JAF, Haywood AM, Valdes PJ, Rahbek C (2008) Quaternary climate changes explain diversity among reptiles and amphibians. Ecography 31:8–15

Andersen AN, Majer JN (2004) Ants show the way down under: invertebrates as bioindicators in land management. Front Ecol Environ 2:291–298

Asís JD (2003) Nesting biology of *Microstigmus thripoctenus* Richards, with a study on nest recognition (Hymenoptera: Crabronidae). J Insect Behav 16(1):49–65

Ayres JM, Fonseca GAB, Rylands AB et al (2005) Os corredores ecológicos das florestas tropicais do Brasil. Sociedade Civil Mamirauá, Belém. https: //www.conservation.org/global/brasil/publicacoes/Pages/livros.aspx. Accessed on 04 Mar 2019

Bailes EJ, Ollerton J, Pattrick JG, Glover BJ (2015) How can an understanding of plant-pollinator interactions contribute to global food security? Current Opinion Plant Biol 26:72–79

Barbosa JRC, Constantino R (2017) Polymorphism in the neotropical termite *Serritermes serrifer*. Entomol Exp Appl 163(1):43–50

Barden P, Grimaldi DA (2016) Adaptive radiation in socially advanced stem-group ants from the Cretaceous. Curr Biol 26:515–521

Baroni-Urbani C, Bolton B, Ward PS (1992) The internal phylogeny of ants (Hymenoptera: Formicidae). Syst Entomol 17:301–329

Bawa KS (1990) Plant-pollinator interactions in tropical rainforests. Ann Rev Ecol Syst 21:399–422

Bernays EA (1988) Host specificity in phytophagous insects: selection pressure from generalist predators. Entomol Exp Appl 49:131–140

Bernays EA, Cornelius ML (1989) Generalist caterpillar prey are more palatable than specialist for the generalist predator *Iridomyrmex humilis*. Oecologia 79:427–430

Bernays E, Graham M (1988) On the evolution of host specificity in phytophagous arthropods. Ecology 69:886–892

Bignell DE (2010) Termites as soil engineers and soil processors. In: Helmut K, Varma A (eds) Intestinal microorganisms of termites and other invertebrates. Springer, Berlin Heidelberg, New York, pp 183–220

Black HIJ, Okwakol MJN (1997) Agricultural intensification, soil biodiversity and agroecosystem function in the tropics: the role of termites. Appl Soil Ecol 6(1):37–53

Bolton B (2003) Synopsis and classification of Formicidae. Mem Am Entomol Inst 71:1–370

Bolton B (2019) An online catalog of the ants of the world. Available from http://antcat.org. Accessed on 15/05/2019

Bourguignon T, Šobotník J, Lepoint G et al (2011) Feeding ecology and phylogenetic structure of a complex neotropical termite assemblage, revealed by nitrogen stable isotope ratios. Ecol Entomol 36(2):261–269

Bourguignon T, Lo N, Cameron SL et al (2015) The evolutionary history of termites as inferred from 66 mitochondrial genomes. Mol Biol Evol 32(2):406–421

Bourguignon T, Chisholm RA, Evans TA (2016a) The termite worker phenotype evolved as a dispersal strategy for fertile wingless individuals before eusociality. Am Nat 187(3):372–387

Bourguignon T, Lo N, Šobotník J et al (2016b) Mitochondrial phylogenomics resolves the global spread of higher termites, ecosystem engineers of the tropics. Mol Biol Evol 34(3):589–597

Bourke AFG, Franks NR (1995) Social evolution in the ants. Princeton University Press, Princeton

Bourke A (2011) Principles of social evolution. Oxford University Press, Oxford

Brady SG, Schultz TR, Fisher BL, Ward PS (2006) Evaluating alternative hypotheses for the early evolution and diversification of ants. Proc Natl Acad Sci U S A 13(48):18172–18177

Brandão CRF, Cancello EM, Yamamoto CI, Scott-Santos CP (2000) Invertebrados terrestres. In: Avaliação do Estado de conhecimento da biodiversidade brasileira. Ministério do Meio Ambiente, Brasília

Branstetter MG, Danforth BN, Pitts JP, Faircloth BC, Ward PS, Buffington ML et al (2017a) Phylogenomic insights into the evolution of stinging wasps and the origins of ants and bees. Curr Biol 27(7):1019–1025

Branstetter MG, Longino JT, Ward PS, Faircloth BC (2017b) Enriching the ant tree of life: enhanced UCE bait set for genome-scale phylogenetics of ants and other Hymenoptera. Methods Ecol Evol 8:768–776

Brodmann J, Twele R, Francke W et al (2008) Orchids mimic green-leaf volatiles to attract prey-hunting wasps for pollination. Curr Biol 18:740–744

Brown WL Jr (1954) Remarks on the internal phylogeny and subfamily classification of the family Formicidae. Insect Soc 1:21–31

Brune A, Ohkuma M (2010) Role of the termite gut microbiota in symbiotic digestion. In: Bignell DE, Roisin Y, Lo N (eds) Biology of termites: a modern synthesis. Springer Dordrecht Heidelberg, New York, pp 439–475

Burger H, Jürgens A, Ayasse M et al (2017) Floral signals and filters in a wasp- and a bee-pollinated *Gomphocarpus* species (Apocynaceae: Asclepiadeae). Flora 232:83–91

Cancello EM, Silva RR, Vasconcellos A et al (2014) Latitudinal variation in termite species richness and abundance along the Brazilian Atlantic Forest hotspot. Biotropica 46(4):441–450

Cardinal S, Danforth BN (2011) The antiquity and evolutionary history of social behavior in bees. PLoS One 6(6):e21086

Cardinal S, Danforth BN (2013) Bees diversified in the age of eudicots. Proc R Soc B 280:20122686

Camargo JMF, Pedro SRM (2013) Meliponini Lepeletier, 1836. In Melo GAR, Moure MJS, Urban D (Eds) Catalogue of bees (Hymenoptera, Apoidea) in the neotropical region - online version. Retrieved from http://www.moure.cria.org.br/catalogue

Carnaval AC, Moritz C (2008) Historical climate modelling predicts patterns of current biodiversity in the Brazilian Atlantic Forest. J Biogeogr 35:1187–1201

Carpenter JM (1996) Distributional checklist of species of the genus *Polistes* (Hymenoptera: Vespidae; Polistinae, Polistini). Am Mus Novit 3188:1–39

Carpenter JM, Kojima J (1996) Checklist of the species in the subfamily Stenogastrinae (Hymenoptera: Vespidae). J New York Entomol Soc 104:21–36

Carpenter JM, Kojima J (1997) Checklist of the species in the subfamily Vespinae (Insecta: Hymenoptera: Vespidae). Nat Hist Bull Ibaraki Univ 1:51–92

Carvalho AF (2014) Sobre a biogeografia dos polistíneos neotropicais: uma abordagem ecológica e evolutiva. Dissertation, Universidade Federal de São Carlos

Conservation International-Brazil, Fundação SOS Mata Atlântica, Fundação Biodiversitas, Instituto de Pesquisas Ecológicas, Secretaria do Meio Ambiente do Estado de São Paulo, Semad/Instituto Estadual de Florestas-MG (2000) Avaliação e ações prioritárias para a conservação da biodiversidade da Mata Atlântica e Campos Sulinos. Brasília, MMA/SBF. https://www.conservation.org/global/brasil/publicacoes/Pages/livros.aspx. Accessed 04 Mar 2019

Constantini JP (2018) Estudo taxonômico dos Apicotermitinae da Mata Atlântica. PhD thesis, University of São Paulo. Available via DIALOG. http://www.teses.usp.br/teses/disponiveis/38/38131/tde-30102018-172307/. Accessed 14 Mar 2019

Constantini J, Janei V, Costa-Leonardo AM, Cancello, EM (2020) Dissimulitermes, a new soldierless termite genus (Blattaria: Isoptera: Termitidae: Apicotermitinae) from the Neotropics and the histology of its dehiscent organ. Insect Syst Evol https://doi.org/10.1163/1876312X-bja10014

R Core Team (2016) R: a language and environment for statistical computing, Vienna, Austria

Costa LP (2003) The historical bridge between the Amazon and the Atlantic Forest of Brazil: a study of molecular phylogeography with small mammals. J Biogeogr 30:71–86

Cuezzo C, Cancello E (2018) Phylogenetic Reconstruction of the Neotropical nasutes genera (Isoptera, Termitidae), based on molecular evidence and worker gut characters. In: Abstracts of the 18th Congress of International Union for the Study of Social Insects (IUSSI), Guarujá, Brazil, 5–10 August 2018. Available via DIALOG. https://www.iussi2018.com/node/1219. Accessed 14 Mar 2019

Da Rocha W, Ribeiro SP, Neves FS, Fernandes GW, Leponce M, Delabie JHC (2015) How does bromeliad distribution structure the arboreal ant assemblage (Hymenoptera: Formicidae) on a single tree in a Brazilian Atlantic Forest agroecosystem? Myrmecol News 21:83–92

Danforth BN, Cardinal S, Praz C, Almeida EAB, Michez D (2013) The impact of molecular data on our understanding of bee phylogeny and evolution. Annu Rev Entomol 58(1):57–78

Darwin C (1859) On the origin of species by means of natural selection. John Murray, London

Davis RB, Baldauf SL, Mayhew PJ (2009) Eusociality and the success of the termites: insights from a supertree of dictyopteran families. J Evol Biol 22:1750–1761

DeSouza O, Cancello EM (2010) Termites and ecosystem function. In: Encyclopedia of Life Support Systems (EOLSS). Eolss Publishers, Oxford. Available via DIALOG. http://www.eolss.net/sample-chapters/c20/E6-142-CS-12.pdf. Accessed 14 Mar 2019

Dlussky GM, Brothers DJ, Rasnitsyn AP (2004) The first Late Cretaceous ants (Hymenoptera: Formicidae) from southern Africa, with comments on the origin of the Myrmicinae. Insect Syst Evol 35:1–13

Donovan SE, Eggleton P, Bignell DE (2001) Gut content analysis and a new feeding group classification of termites. Ecol Entomol 26:356–366

Dyer LA (1995) Tasty generalists and nasty specialists? Antipredator mechanisms in tropical lepidopteran larvae. Ecology 176:1483–1496

Engel MS, Grimaldi DA (2005) Primitive new ants in Cretaceous amber from Myanmar, New Jersey, and Canada (Hymenoptera: Formicidae). Am Mus Novit 3485:1–23

Engel MS, Barden P, Riccio ML, Grimaldi DA (2016) Morphologically specialized termite castes and advanced sociality in the Early Cretaceous. Curr Biol 26:522–530

Evans HE, West-Eberhard MJ (1970) The Wasps. The University of Michigan Press, USA, Ann Arbor

Ferraz M, Cancello EM (2001) Swarming behavior of the economically most important termite, *Coptotermes havilandi* (Isoptera: Rhinotermitidae), in Southeastern Brazil. Sociobiology 38:683–694

Ferraz MV, Cancello EM (2004) Strategies on the developmental biology of incipient colonies of *Coptotermes gestroi* (Isoptera: Rhinotermitidae) in different substrates. Sociobiology 44(1):109–122

Fittkau EJ, Klinge H (1973) On biomass and trophic structure of the central Amazonian rain forest ecosystem. Biotropica 5:2–14

Folgarait PJ (1998) Ant biodiversity and its relationship to ecosystem functioning: a review. Biodivers Conserv 7:1221–1244

Fonseca GAB, Alger K, Pinto LP, Araújo M, Cavalcanti R (2003) Corredores de biodiversidade: o Corredor Central da Mata Atlântica. In: Prado PI, Landau EC, Moura RT, Pinto LPS, Fonseca GAB, Alger K (org) Corredor de Biodiversidade da Mata Atlântica do Sul da Bahia. IESB / CI / CABS / UFMG / UNICAMP Corredores de biodiversidade: o Corredor Central da Mata Atlântica, pp 1–11. http://programas.inema.ba.gov.br/sigbiota/iesb/Documentos/Corredores%20de%20Biodiversidade.pdf. Accessed 04 Mar 2019

Gadagkar R (1990) Origin and evolution of eusociality: a perspective from studying primitively eusocial wasps. J Genet 69:113–125

Gadagkar R (2001) The social biology of *Ropalidia marginata*: toward understand the evolution of Eusociality. Harvard University Press, Cambridge

Garrafoni ARS, Moura FR, Lourenço AP (2017) Areas of endemism in the Atlantic Forest: quantitative biogeography insights from orchids bees (Apidae: Euglossini). Apidologie 48:513–522

Gonçalves RB, Brandão CRF (2008) Diversidade de abelhas (Hymenoptera, Apidae) ao longo de um gradiente latitudinal na Mata Atlântica. Biota Neotrop 8(4):51–61

Gonçalves RB, Santos EF, Scott-Santos CF (2012) Bees (Hymenoptera: Apoidea: Apidae s.l.) captured with Malaise and pan traps along an altitudinal gradient in the Parque Estadual da Serra do Mar, Ubatuba, São Paulo, Brazil. Check List 8(1):53–56

Grimaldi D, Agosti D, Carpenter JM (1997) New and rediscovered primitive ants (Hymenoptera: Formicidae) in Cretaceous amber from New Jersey, and their phylogenetic relationships. Am Mus Novit 3208:1–43

Guénard B, Weiser MD, Dunn RR (2012) Global models of ant diversity suggest regions where new discoveries are most likely are under disproportionate deforestation threat. Proc Natl Acad Sci U S A 109:7368–7373

Hamilton WD (1964a) The genetical evolution of social behaviour. I. J Theor Biol 7:1–16

Hamilton WD (1964b) The genetical evolution of social behaviour. II. J Theor Biol 7:17–52

Hay ME, Pawlik JR, Duffy JE et al (1989) Seaweed-herbivore-predator interactions: host-plant specialization reduces predation on small herbivores. Oecologia 81:418–427

Hines HM, Hunt JH, O'Connor TK et al (2007) Multigene phylogeny reveals eusociality evolved twice in vespid wasps. PNAS 104:3295–3299

Hoffmeister CH, Ferrari A (2016) Areas of endemism of arthropods in the Atlantic Forest (Brazil): an approach based on a metaconsensus criterion using endemicity analysis. Biol J Linn Soc 119:126–144

Hölldobler B, Wilson EO (1990) The ants. The Belknap Press of Harvard University Press, Cambridge, MA

Hölldobler B, Wilson EO (1994) Journey to the ants: a story of scientific exploration. The Belknap Press of Harvard University Press, Cambridge, MA

Hölldobler B, Wilson EO (2008) The Superorganism: the beauty, elegance, and strangeness of insect societies. W.W. Norton & Company, New York

Holt JA, Lepage M (2000) Termites and soil properties. In: Abe T, Bignell DE, Higashi M (eds) Termites: evolution, sociality, symbioses, ecology, 1st edn. Kluwer Academic Publishers, Dordrecht, pp 389–407

Howard KJ, Thorne BL (2010) Eusocial evolution in termites and Hymenoptera. In: Bignell DE, Roisin Y, Lo N (eds) Biology of termites: a modern synthesis. Springer Dordrecht Heidelberg, New York, pp 97–132

Howard KJ, Smith AR, O'Donnell S, Jeanne RL (2002) Novel method of swarm emigration by epiponine wasp, *Apoica pallens* (Hymenoptera: Vespidae). Ethol Ecol Evol 14:365–371

Hunt JH, Jeanne RL, Baker I, Grogan DE (1987) Nutrients dynamics of swarm-founding social wasp species, *Polybia occidentalis* (Hymenoptera: Vespidae). Ethology 75:291–305

Hunt JH, Jeanne RL, Keeping MG (1995) Observations on *Apoica pallens*, a nocturnal neotropical social wasp (Hymenoptera: Vespidae, Polistinae, Epiponini). Insect Soc 42:223–236

Ihering RV (1898) The founding of new colonies and fungus gardens in *Atta sexdens* (translated from German). Zool Anz 21:238–245

Instituto Chico Mendes de Conservação da Biodiversidade (2018) Livro Vermelho da Fauna Brasileira Ameaçada de Extinção: Volume VII - Invertebrados. In: Instituto Chico Mendes de Conservação da Biodiversidade (org) Livro Vermelho da Fauna Brasileira Ameaçada de Extinção. ICMBio, Brasília

Inward DJG, Vogler AP, Eggleton P (2007) A comprehensive phylogenetic analysis of termites (Isoptera) illuminates key aspects of their evolutionary biology. Mol Phylogenet Evol 44:953–967

Jeanne RL (1980) Evolution of social behaviour in the Vespidae. Annu Rev Entomol 25:371–396

Jeanne RL (1981) Chemical communication during swarm emigration in the social wasp *Polybia sericea* (Olivier). An Behav 29:102–113

Jeanne RL (1991) The swarm-founding Polistinae. In: Ross K, Matthews RW (eds) The Social Biology of Wasps. Comstock Publishing Associates, Ithaca, pp 191–231

Johnstone RA, Cant MA, Field J (2012) Sex-biased dispersal, haplodiploidy and the evolution of helping in social insects. Proc Biol Sci 279:787–793

Jones CG, Whitman DW, Compton SJ et al (1989) reduction in diet breadth results in sequestration of plant chemicals and increases efficacy of chemical defense in a generalist grasshopper. J Chem Ecol 15:1811–1822

Jones CG, Lawton JH, Shachak M (1994) Organisms as Ecosystem Engineers. Oikos 69:373–386

Jouquet P, Bottinelli N, Shanbhag RR et al (2016) Termites: the neglected soil engineers of tropical soils. Soil Sci 181:157–165

Kaspari M (1996) Litter ant patchiness at the 1-m$^2$ scale: disturbance dynamics in three Neotropical forests. Oecologia 107:265–273

Kaspari M, O'Donnell S (2003) High rates of army ant raids in the Neotropics and implications for ant colony and community structure. Evol Ecol Res 5:933–939

Keller RA (2011) A phylogenetic analysis of ant morphology (Hymenoptera: Formicidae) with special reference to the poneromorph subfamilies. B Am Mus Nat Hist 355:1–90

Klein S, Cabirol A, Devaud J, Barron AB, Lihoreau M (2017) Why bees are so vulnerable to environmental stressors? Trends in Ecol Evol 32(4):268–278

Klingenberg C, Brandão CRF (2005) The type specimens of fungus growing ants, Attini (Hymenoptera, Formicidae, Myrmicinae) deposited in the Museu de Zoologia da Universidade de São Paulo, Brazil. Pap. Avulsos de Zool 45(4):41–50

Korb J (2008) The ecology of social evolution in termites. In: Korb J, Heinze J (eds) Ecology of Social Evolution. Springer, Berlin Heidelberg, Berlin, Heidelberg, pp 151–174

Korb J, Hartfelder K (2008) Life history and development - a framework for understanding developmental plasticity in lower termites. Biol Rev 83:295–313

Korb J, Heinze J (2004) Multilevel selection and social evolution of insect societies. Naturwissenschaften 91(6):291–304

Krishna K, Grimaldi DA, Krishna V, Engel MS (2013) Treatise on the Isoptera of the world. Bull Am Museum Nat Hist 1:1–2704

Lach L, Parr CL, Abbott KL (2010) Ant ecology. Oxford University Press, Oxford

Ledo RMD, Colli GR (2017) The historical connections between the Amazon and the Atlantic Forest revisited. J Biogeogr 44:2551–2563

Lee KE, Wood TG (1971) Termites and soils. Academic Press of London, London

Legendre F, Condamine LF (2018) When Darwin's special difficulty promotes diversification in insects. Syst Biol 67(5):873–887

Legendre F, Whiting MF, Bordereau C et al (2008) The phylogeny of termites (Dictyoptera: Isoptera) based on mitochondrial and nuclear markers: Implications for the evolution of the worker and pseudergate castes, and foraging behaviors. Mol Phylogenet Evol 48:615–627

Legendre F, Nel A, Svenson GJ et al (2015) Phylogeny of Dictyoptera: dating the origin of cockroaches, praying mantises and termites with molecular data and controlled fossil evidence. PLoS One 10:e0130127

Letourneau DK, Choe JC (1987) Homopteran attendance by wasps and ants: the stochastic nature of interactions. Psyche 94(1–2):81–91

Linnaeus C (1758) Systema naturae per regna tria natura, secundum classes, ordines, genera, species, cum characteribus, differentiis synonymis, locis, vol 1, 10th ed. (revised) edn. Laurentii Salvii, Holmiae [Stockholm]. 824 pp

Loveridge JP, Moe SR (2004) Termitaria as browsing hotspots for African megaherbivores in miombo woodland. J Trop Ecol 20:337–343

Lucas ER, Martins RP, Zanette LRS et al (2011) Social and genetic structure in colonies of social wasp *Microstigmus nigrophtalmus*. Insect Soc 58:107–114

Lucky A, Trautwein MD, Guénard BS, Weiser MD, Dunn RR (2013) Tracing the rise of ants - out of the ground. PLoS One 8(12):e84012. https://doi.org/10.1371/journal.pone.0084012

Mateus S (2011) Observations on forced colony emigration in *Parachartergus fraternus* (Hymenoptera: Vespidae: Epiponini): new nest site marked with sprayed venom. Psyche 2011:1–8

Matsuura K (2010) Sexual and asexual reproduction in termites. In: Bignell DE, Roisin Y, Lo N (eds) Biology of termites: a modern synthesis. Springer Dordrecht Heidelberg, New York, pp 255–277

Matthews RW (1968) *Microstigmus* comes: Sociality in a Sphecid Wasp. Science 160:787–788

Matthews RW (1991) Evolution of Social Behavior in Sphecidae wasps. In: Ross KG, Matthews RW (eds) The Social Biology of Wasps. Comstock Publishing Associates, Ithaca, pp 570–602

Matthews RW, Starr CK (1984) Microstigmus comes wasps have a method of nest construction unique among social insects. Biotropica 16:55–58

May RGE (1988) How many species are there on Earth? Science 241:1441–1449

Mello MAR, Santos GMM, Mecho MR, Hermes MG (2011) high generalization in flower-visiting networks of social wasps. Acta Oecol 37(1):37–42

Melo GAR, Costa MA (2004) A new stingless bee species of the genus *Scaura* (Hymenoptera, Apidae) from the Brazilian Atlantic Forest, with notes on *S. latitarsis* (Friese). Zootaxa 544:1–10

Melo GAR (2007) Introductory remarks. In Melo GAR, Moure MJS, Urban D (Eds) Catalogue of bees (Hymenoptera, Apoidea) in the neotropical region. Sociedade Brasileira de Entomologia, Curitiba

Michener CD (1979a) Biogeography of the Bees. Ann Miss Bot Garden 66(3):277–347

Michener CD (1979b) The social behavior of bees: a comparative study. Cambridge University Press

Michener CD (2007) The bees of the world, 2nd edn. The John Hopkins University Press, Baltimore

Mittermeier RA, Gil PR, Hoffmann M, Pilgrim J, Brooks T, Mittermeier CG, Lamoreux J, Da Fonseca GAB (2004) Hotspots revisited: Earth's biologically richest and most endangered terrestrial ecoregions. CEMEX/Agrupación Sierra Madre, Mexico City

Moreau CS, Bell CD (2013) Testing the museum versus cradle biological diversity hypothesis: Phylogeny, diversification, and ancestral biogeographic range evolution of the ants. Evolution 67(8):2240–2257

Moreau CS, Bell CD, Vila R, Archibald B, Pierce NE (2006) Phylogeny of the ants: diversification in the age of angiosperms. Science 312:100–104

Murray EA, Bossert S, Danforth BN, Murray EA (2018) Pollinivory and the diversification dynamics of bees. Biol Lett. https://doi.org/10.1098/rsbl.2018.0530

Nalepa CA (2010) Altricial development in wood-feeding cockroaches: the key antecedent of termite eusociality. In: Bignell DE, Roisin Y, Lo N (eds) Biology of termites: a modern synthesis. Springer Dordrecht Heidelberg, New York, pp 69–95

New TR (2018) Mutualisms and Insect Conservation. Springer, Cham

Noll FB (2013) "Marimbondos": a review on the neotropical swarm-founding polistines. Sociobiology 60:347–354

Noll FB, Wenzel JW, Zucchi R (2004) Evolution of caste in Neotropical swarm-founding wasps (Hymenoptera: Vespidae; Epiponini). Am Mus Novit 3467:1–24

Ohkuma M, Brune A (2010) Diversity, structure, and evolution of the termite gut microbial community. In: Bignell DE, Roisin Y, Lo N (eds) Biology of termites: a modern synthesis. Springer Dordrecht Heidelberg, New York, pp 413–438

Oliveira DE, Constantino R (2016) A taxonomic revision of the neotropical termite genus *Diversitermes* (Isoptera: Termitidae: Nasutitermitinae). Zootaxa 4158:221

Oliveira OAL, Noll FB, Wenzel JW (2010) Foraging behavior and colony cycle of *Agelaia vicina* (Hymenoptera: Vespidae; Epiponini). J Hymenopt Res 19:4–11

Oliveira DE, Rocha MM, Cancello EM (2015) *Muelleritermes*: a new termite genus with two species from the Brazilian Atlantic Forest (Isoptera: Termitidae: Nasutitermitinae). Zootaxa 4012:258

Ollerton J, Winfree R, Tarrant S (2011) How many flowering plants are pollinated by animals? Oikos 120(3):321–326

Pardeshi M, Prusty BAK (2010) Termites as ecosystem engineers and potentials for soil restoration. Curr Sci 99(1):11

Pedro SRM (2014) The stingless bee fauna in Brazil (Hymenoptera: Apidae). Sociobiology 61(4):348–354

Pennington RT, Lavin M, Prado DE et al (2004) Historical climate change and speciation: neotropical seasonally dry forest plants show patterns of both Tertiary and Quaternary diversification. Philos T Roy Soc B 359:515–537

Perrichot V, Lacau S, Néraudeau D, Nel A (2008) Fossil evidence for the early ant evolution. Naturwissenschaften 95:85–90

Perrichot V, Wang B, Engel MS (2016) Extreme morphogenesis and ecological specialization among cretaceous basal ants. Curr Biol 26:1–5

Pickett KM, Carpenter JM (2010) Simultaneous analysis and the origin of eusociality in the Vespidae (Insecta: Hymenoptera). Arthropod Syst Phylo 68:3–33

Pie MR, Feitosa RM (2016) Relictual ant lineages (Hymenoptera: Formicidae) and their evolutionary implications. Myrmecol News 22:55–58

Pie MR, Tschá MK (2009) The macroevolutionary dynamics of ant diversification. Evolution 63:3023–3030

Piekarski PK, Longair R, Rogers S (2014) Monophyly of eusocial wasps (Hymenoptera: Vespidae): molecules and morphology tell opposing histories. JURA 4:11–14

Piekarski PK, Carpenter JM, Lemmon AR et al (2018) Phylogenetic evidence overturns current conceptions of social evolution in wasps (Vespidae). Mol Biol Evol 35:2097–2109

Por FD (1992) Sooretama: the Atlantic rain forest of Brazil. SPB Academic Publishing, The Hague

Potts SG, Biesmeijer JC, Kremen C, Neumann P, Schweiger O, Kunin WE (2010) Global pollinator declines: trends, impacts and drivers. Trends Ecol Evol 25(6):345–353

Prance GT (1987) Biogeography of Neotropical plants. In: Whitmore TC, Prance GT (eds) Biogeography and quaternary history in Tropical America. Oxford Science Publications, Oxford, pp 46–65

Rabeling C, Brown JM, Verhaagh M (2008) Newly discovered sister lineage sheds light on early ant evolution. Proc Natl Acad Sci U S A 105(39):14913–14917

Ramalho M (2004) Stingless bees and mass flowering trees in the canopy of Atlantic Forest: a tight relationship. Acta Bot Brasil 18(1):37–47

Reis YT, Cancello EM (2007) Riqueza de cupins (Insecta, Isoptera) em áreas de Mata Atlântica primária e secundária do sudeste da Bahia. Iheringia, Série Zool 97:229–234

Ribas CR, Campos RBF, Schmidt FA, Solar RRC (2012) Ants as indicators in Brazil: a review with suggestions to improve the use of ants in environmental monitoring programs. Psyche 2012:636749. https://doi.org/10.1155/2012/636749

Ribeiro MC, Martensen AC, Metzger JP et al (2011) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In: Zachos FE, Habel JC (eds) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. Biodiversity hotspots. Springer, Berlin, pp 405–434

Ribeiro DG, Silvestre R, Garcete-Barret BR (2019) Diversity of wasps (Hymenoptera: Aculeata: Vespidae) along an altitudinal gradient of Atlantic Forest in Itatiaia National Park, Brazil. Rev Bras Entomol 63(1):22–29

Richter MR (1990) Hunting wasp interactions: influence of prey size, arrival order, and wasp species. Ecology 71(3):1018–1030

Richter MR (2000) Social wasp (Hymenoptera: Vespidae) foraging behaviour. Annu Rev Entomol 45:121–150

Rocha MM, Castro ACMC, Cuezzo C et al (2017) Phylogenetic reconstruction of Syntermitinae (Isoptera, Termitidae) based on morphological and molecular data. PLoS One 12(3):e0174366

Roisin Y, Korb J (2010) Social organization and the status of workers in termites. In: Bignell DE, Roisin Y, Lo N (eds) Biology of termites: a modern synthesis. Springer Dordrecht Heidelberg, New York, pp 133–164

Ross KG, Matthews RW (1989) New evidence for eusociality in the sphecid wasp *Microstigmus comes*. Anim Behav 38:613–619

Sandel B, Arge L, Dalsgaard B et al (2011) The influence of late Quaternary climate-change velocity on species endemism. Science 334:660–664

Santos EF, Noll FB, Brandão CRF (2014) Functional and Taxonomic diversity of stinging wasps in Brazilian Atlantic Rainforest Areas. Neotrop Entomol 43:97–105

Santos EF, Noll FB, Brandão CRF (2016) Structural organization of social paper wasp (Hymenoptera: Polistinae) assemblage along a latitudinal gradient in the Atlantic Rainforest: correlating fauna partitioning to biodiversity centers. J Insect Conserv 20:597–609

Schaefer CER (2001) Brazilian latosols and their B horizon microstructure as long-term biotic constructs. Soil Res 39:909

Scheffrahn RH (2015) Global elevational, latitudinal, and climatic limits for termites and the redescription of *Rugitermes laticollis* Snyder (Isoptera: Kalotermitidae) from the Andean Highlands. Sociobiology 62:426

Schlemmermeyer T (2000) Levantamento da termitofauna da estação biológica de Boracéia, Salesópolis, SP. PhD Thesis, University of São Paulo

Schmidt C (2013) Molecular phylogenetics of ponerine ants (Hymenoptera: Formicidae: Ponerinae). Zootaxa 3647:201–250

Schmidt FA, Solar RRC (2010) Hypogaeic pitfall traps: methodological advances and remarks to improve the sampling of a hidden ant fauna. Insect Soc 57:261–266

Schultz TR, Brady SG (2008) Major evolutionary transitions in ant agriculture. Proc Natl Acad Sci U S A 105:5435–5440

Shellman-Reeve JS (1997) The spectrum of eusociality in termites. In: Choe JC, Crespi BJ (eds) The Evolution of Social Behaviour in Insects and Arachnids, 1st edn. Cambridge University Press, Cambridge, pp 52–92

Silva RR, Brandão CRF (2010) Morphological patterns and community organization in leaf-litter ant assemblages. Ecol Monogr 80:107–124

Silva RR, Brandão CRF (2014) Ecosystem-Wide morphological structure of leaf-litter ant communities along a tropical latitudinal gradient. PLoS One 9(3):e93049. https://doi.org/10.1371/journal.pone.0093049

Silva JMC, Casteleti CHM (2003) Status of the biodiversity of the Atlantic Forest of Brazil. In: Galindo-Leal C, Câmara IG (eds) The Atlantic Forest of South America: biodiversity status, threats, and outlook. Island Press, Washington, DC, pp 43–59

Silva M, Noll FB (2015) Biogeography of the social wasp genus *Brachygastra* (Hymenoptera: Vespidae: Polistinae). J Biogeogr 42:833–842

Silva RR, Silvestre R (2004) Riqueza da fauna de formigas (Hymenoptera: Formicidae) que habita as camadas superficiais do solo em Seara, Santa Catarina. Pap Avulsos Zool 44(1):1–11

Silva JMC, Sousa MC, Castelletti CHM (2004) Areas of endemism for passerine birds in the Atlantic Forest. Glob Ecol Biogeogr 13:85–92

Stradling DJ (1978) Food and feeding habits of ants. In: Brian MV (ed) Production ecology of ants and termites. Cambridge University Press, Cambridge, pp 81–106

Ströher PR, Meyer ALS, Zarza E, Tsai W, McCormack JE, Pie M (2019) Phylogeography of ants from the Brazilian Atlantic Forest. Org Divers Evol 19(3):435–445

Swanson AC, Schwendenmann L, Allen MF et al (2019) Welcome to the *Atta* world: a framework for understanding the effects of leaf-cutter ants on ecosystem functions. Funct Ecol:1–14. https://doi.org/10.1111/1365-2435.13319

Tabarelli M, Pinto LP, Silva JMC et al (2005) Challenges and opportunities for Biodiversity Conservation in the Brazilian Atlantic Forest. Conserv Biol 19:695–700

Tabarelli M, Aguiar AV, Ribeiro MC, Metzger JP, Peres CA (2010) Prospects for biodiversity conservation in the Atlantic Forest: lessons from aging human-modified landscapes. Biol Conserv 143:2328–2340

Takagi M, Hirose Y, Yamasaki M (1980) Prey-location learning in *Polistes jadwigae* Dalla Torre (Hymenoptera, Vespidae). Kontyû 48:53–58

Thorne BL (1997) Evolution of eusociality in termites. Annu Rev Ecol Syst 28:27–54

Thorp RW (1979) Structural, behavioral, and physiological adaptations of bees (Apoidea) for collecting pollen. Ann Miss Bot Gard 66:788–812

Töpel M, Zizka A, Calió MF, Scharn R, Silvestro D, Antonelli A (2017) SpeciesGeoCoder: fast categorization of species occurrences for analyses of biodiversity, biogeography, ecology, and evolution. Syst Biol 66(2):145–151

Vanbergen AJ, The Insect Pollinator Initiative (2013) Threats to an ecosystem service: pressures on pollinators. Front Ecol Envirn 11(5):251–259

Vieira MF, Shepherd GJ (1999) Pollinators of *Oxypetalum* (Asclepiadaceae) in southeastern Brazil. Rev Brasil Biol 59:693–704

Ward PS, Brady SG, Fisher BL, Schultz TR (2010) Phylogeny and biogeography of Dolichoderine ants: effects of data partitioning and relict taxa on historical inference. Syst Biol 59(3):342–362

Ward PS, Brady SG, Fisher BL, Schultz TR (2015) The evolution of myrmicine ants: phylogeny and biogeography of a hyperdiverse ant clade (Hymenoptera: Formicidae). Syst Entomol 40:61–81

Ware JL, Grimaldi DA, Engel MS (2010) The effects of fossil placement and calibration on divergence times and rates: an example from the termites (Insecta: Isoptera). Arthropod Struct Dev 39:204–219

Wcislo WT, Cane JH (1996) Floral resource utilization by solitary bees (Hymenoptera: Apoidea) and exploitation of their stored foods by natural enemies. Ann Rev Entomo l41:257–286

Wenzel JW (1998) A generic key to the nests of Hornets, Yellows, and Paper Wasps Worldwide (Vespidae: Vespinae, Polistinae)

Werneck MS, Sobral MEG, Rocha CTV et al (2011) Distribution and endemism of angiosperms in the Atlantic Forest. Nat Conservação 9:188–193

Wilms W, Imperatriz-Fonseca VL, Engels W (1996) Resource partitioning between highly eusocial bees and possible impact of the introduced africanized honey bee on native stingless bees in the Brazilian Atlantic Rainforest. Stud Neot Fauna Envir 31:137–151

Wilson EO (1971) The insect societies. The Belknap Press of the Harvard University Press, Cambridge, Cambridge (MA)

Wilson EO (1976) Which are the most prevalent ant genera? Studia Entomologica 19:187–200

Wilson EO (2003) *Pheidole* in the New World: a dominant, hyperdiverse ant genus. The Belknap Press of the Harvard University Press, Cambridge, MA

Wilson EO, Hölldobler B (2005a) The rise of the ants: a phylogenetic and ecological explanation. Proc Natl Acad Sci U S A 102:7411–7414

Wilson EO, Hölldobler B (2005b) Eusociality: origin and consequences. Proc Natl Acad Sci U S A 102:13367–13371

Zhao Z, Leggleton P, Chuyin X, Gao T, Shih C, Ren D (2019) The oldest known mastotermitids (Blattodea: Termitoidae) and phylogeny of basal termites. Syst Entomol 44:612–623

Zucchi R, Sakagami SF, Noll FB et al (1995) *Agelaia vicina*, a swarm founding polistine with the largest colony size among wasps and bees (Hymenoptera: Vespidae). J New York Entomol Soc 103:129–137

# Chapter 9
# Tetrapod Diversity in the Atlantic Forest: Maps and Gaps

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**Abstract** The Atlantic Forest is a heterogeneous and complex vegetation mosaic caused by variety of climatic, geomorphological, and edaphic conditions. It has long been known that the Atlantic Forest has one of the most diversified biotas on the planet, presenting high levels of endemism. Here, we update the knowledge regarding terrestrial vertebrates occurring in the Atlantic Forest, focusing on endemic species and presenting its main spatial patterns of diversity. We also analyzed the main knowledge gaps associated with these species. We identified 2,645 species of Tetrapoda in the Atlantic Forest, being 719 species of amphibians, 517 species of reptiles, 1,025 species of birds, and 384 species of mammals. The uniqueness of its fauna is impressive even in a global scale, as 2.8% of the world's Tetrapoda species occurs only in the Atlantic Forest. For reptiles, this percentage is 1.3%, while for both birds and mammals, it hovers around 1.9%, but for amphibians, it reaches an impressive 6.6%. Spatially, most groups exhibit their highest species

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richness at the core of the Atlantic Forest, and this pattern becomes more evident when only endemic species are considered. Even with all its impressive diversity, 157 new Tetrapoda species were described in the Atlantic Forest in the last decade, mostly from poorly sampled regions or environments. An increase of sampling effort on these regions might increase the number of species on this biome, which already is one of the most diverse in the world.

**Keywords** Endemism · Linnean shortfall · Richness · Spatial patterns · Wallacean shortfall

## 9.1 Overview

The Atlantic Forest is among the largest blocks of tropical forests of the world and the second largest rainforest in South America (Sobral-Souza and Lima-Ribeiro 2017), once covering more than 1.3 million km$^2$ (Paglia and Pinto 2010). It stretches from northeastern Brazil to northern Argentina and eastern Paraguay, comprising a variety of climatic, geomorphological, and edaphic conditions that result from this wide latitudinal, longitudinal, and altitudinal range (Ribeiro et al. 2011). There is a longitudinal gradient strongly correlated with precipitation, running inland from the coast, that results in a gradient between evergreen and semideciduous forests. Simultaneously, there are latitudinal and altitudinal gradients associated with both temperature and rainfall, resulting in a variety of forest formations, such as the subtropical *Araucaria* moist forest, cloud forest, and montane forest (Oliveira-Filho and Fontes 2000; Eisenlohr and Oliveira-Filho 2015). The Atlantic Forest is a complex vegetation mosaic composed of many distinct phytophysiognomies. Although much of the Atlantic Forest was originally covered by forests, other types of vegetation are also found within the ecoregion, such as the *campos rupestres*, *campos de altitude*, mangroves, and sand coastal plains or *restingas* (Scarano 2002).

During its evolutionary history, the Atlantic Forest was repeatedly connected to the Amazon (Vivo and Carmignotto 2004; Sobral-Souza et al. 2015; Ledo and Colli 2017), with eventual dispersal and biotic interchange between these ecoregions (Costa 2003; Lynch Alfaro et al. 2012; Buckner et al. 2015). These connections were followed by periods of isolation with subsequent allopatric speciation (Silva et al. 2004). The Atlantic Forest also experienced other major geological events and environmental changes, such as the establishment of major river systems, the uplift of the Serra do Mar (Safford 1999), and sea level changes (Leite et al. 2016), which influenced biological diversification (Costa 2003; Pellegrino et al. 2005; Sobral-Souza and Lima-Ribeiro 2017; Figueiredo and Grelle 2018). Its complex evolutionary history, coupled with the high environmental heterogeneity, is the main factor associated with the high levels of diversity and endemism of the Atlantic Forest (Sobral-Souza and Lima-Ribeiro 2017).

The Atlantic Forest has one of the most diversified biotas on the planet, comprising 1–8% of the world's biodiversity (Silva and Casteleti 2003) and more than 5% of the world's vertebrates (Paglia and Pinto 2010). A previous assessment of the Tetrapoda diversity in the Atlantic Forest includes 990 species of birds, 370 amphibians, 295 mammals, and 200 reptiles, in a total of 1,855 species of terrestrial vertebrates (Paglia and Pinto 2010), while another assessment includes 861 species of birds, 625 amphibians, 321 mammals, and approximately 300 reptiles, in an approximate total of 2,107 species of Tetrapoda (Monteiro-Filho and Conte 2017). Its biota is also very distinct, with endemism rates ranging from 25% in birds to 78% in amphibians (Monteiro-Filho and Conte 2017), and 2% of all vertebrates are endemic to the Atlantic Forest (Myers et al. 2000).

New information is now available to better describe Atlantic Forest vertebrate diversity, as new species were described and databases became available. Herein, we summarize and update the knowledge regarding terrestrial vertebrates occurring in the Atlantic Forest, focusing on endemic species and describing spatial patterns of diversity. We examine and discuss the main knowledge gaps associated with these species.

## 9.2 Endemism

To obtain a list of occurring and endemic species of terrestrial vertebrates, we used the integrative limit of the Atlantic Forest following Muylaert et al. (2018). This limit is comprehensive and inclusive, because it includes Atlantic Forest remnants occurring today in ecotone or enclave condition, as is the case of the *Brejos de Altitude* humid forest enclaves, in the Caatinga ecoregion. These remnants are important as a register of the diversity of plants and animals that once were part of the Atlantic Forest and are currently distributed in other biomes, serving as an evidence of some patterns of historical biogeography.

First, we performed a spatial overlay between the integrative limit and the specialist-drawn geographic distribution maps of reptiles (Meiri et al. 2017), birds (BirdLife International and NatureServe 2011), amphibians, and mammals (IUCN 2014), which were drawn by specialists and are available in digital format. All species whose distribution intersected with the limit of the Atlantic Forest were considered as occurring in this ecoregion. We gave preference to maps which represented the historical geographic distribution of the species. In addition to the digital maps, we also conducted a non-systematic search of the literature for newly described species. From those sources, we formed our list of species occurring in the ecoregion. To identify the endemic species, we followed a criteria applied by Vale et al. (2018) and defined as endemics those species that have at least 80% of their distribution within the limit of the Atlantic Forest.

We projected the digital distribution maps over an equal-area hexagonal grid (~500 km$^2$ cells) that encompassed the entire Atlantic Forest and obtained a list of all species present in each cell. Based on this list, we estimated total and endemic species richness for each cell of the hexagonal grid. We also estimated the restricted-range richness, which is richness estimative weighted by the range size of each species. It was defined as the sum of the inverse of the geographic range size of all endemic species present in a cell, as species with restricted geographic ranges contributed more to this index than those with large geographic ranges.

As species with distinct ecological requirements may respond to the environment in different ways, we chose to use a deconstructive approach of diversity (Marquet et al. 2004) to better represent the spatial patterns of diversity in the Atlantic Forest. That is, we first provide summaries for the major groups of terrestrial vertebrates (amphibians, reptiles, birds, mammals) and then proceed to progressively more restricted taxonomic subgroups (e.g., anurans, caecilians, turtles, lizards, snakes, amphisbaenids, crocodilians, nonpasserine birds, passerine birds, terrestrial mammals, and bats).

The subgroup anurans include all species of the order Anura, while the caecilians include all species of the order Gymnophiona. The subgroup turtles include all species of the order Testudines, while the crocodilians include all species of the order Crocodylia, and the amphisbaenids include all species of the family Amphisbaenidae. The subgroup lizards include all species of the families Dactyloidae, Diploglossidae, Gekkonidae, Gymnophthalmidae, Hoplocercidae, Iguanidae, Leiosauridae, Liolaemidae, Phyllodactylidae, Polychrotidae, Scincidae, Sphaerodactylidae, Teiidae, and Tropiduridae. The subgroup snakes include all species of the families Aniliidae, Anomalepididae, Boidae, Colubridae, Elapidae, Leptotyphlopidae, Tropidophiidae, Typhlopidae, and Viperidae. The subgroup passerine birds include all species of the order Passeriformes, while the subgroup nonpasserine birds include the species of all the other orders of the class Aves. The subgroup bats include all species of the order Chiroptera, while the subgroup terrestrial mammals include the species of all the other orders of the class Mammalia.

## 9.3 Tetrapod Diversity

We identified 2,645 species of Tetrapoda in the Atlantic Forest, comprising 719 species of amphibians, 517 reptiles, 1,025 birds, and 384 mammals. Thus, the Atlantic Forest comprises one of the most diverse ecoregions on the planet, with 9.5% of all amphibians, 5.1% of reptiles, 9.2% of birds, 6.7% of mammals, and 7.7% of all Tetrapoda species of the world (Table 9.1). With recent studies and our definition of Atlantic Forest residents, the number of species of birds increased somewhat, while the number of species of mammals, amphibians, and, especially, reptiles increased much more as compared to previous studies (Paglia and Pinto 2010).

The tetrapod fauna uniqueness in the Atlantic Forest justifies its designation as an important biodiversity hotspot (Myers et al. 2000). One in three species of Tetrapoda (36%) in the Atlantic Forest is endemic. Even so, endemism is uneven among taxonomic groups, reaching 70% for amphibians, while the other three groups range from 20% to 30% (Table 9.1). This uniqueness extends to the global

**Table 9.1** Species diversity and endemism of the four major groups of tetrapods in the Atlantic Forest. The first three columns are total numbers of species. The next two columns are the proportions of the global total found in the Atlantic Forest. The final column is the percentage of Atlantic Forest species that are endemic. The sources used for the numbers of described species in the world are Jetz and Pyron (2018) for amphibians, Roll et al. (2017) for reptiles, BirdLife International (2018) for birds, and Faurby and Svenning (2015) for mammals

| | Number of species | | | Relative number of species (percentage of total) | | |
|---|---|---|---|---|---|---|
| | Atlantic Forest | | | World | | Atlantic Forest Endemic |
| Taxonomic group | Total | Endemic | World | Total | Endemic | |
| Amphibian | 719 | 504 | ≈7,600 | 9.5 | 6.6 | 70.1 |
| Reptile | 517 | 126 | 10,064 | 5.1 | 1.3 | 24.4 |
| Bird | 1,025 | 215 | 11,126 | 9.2 | 1.9 | 21.0 |
| Mammal | 384 | 109 | 5747 | 6.7 | 1.9 | 28.4 |
| Tetrapoda | 2,645 | 954 | 34,537 | 7.7 | 2.8 | 36.0 |

scale, because for every forty species of tetrapod in the world, one (2.7%) is found only in the Atlantic Forest, compared to 2% as estimated by Myers et al. (2000). Amphibians are very unique, with 6.6% of the global total of species being endemic to the Atlantic Forest. Endemic reptiles comprise 1.3%, while endemic birds and mammals are 1.9% of the global total of species (Table 9.1). A complete list of species which occur at the Atlantic Forest is available in the Supplementary Material.

As observed in these major groups, endemism is variable among subgroups. A high level of endemism is associated to some subgroups, where more than 70% of the species of caecilians and anurans (Amphibia) and half (50%) of the species of amphisbaenids are endemic to the Atlantic Forest. There are subgroups with moderate levels of endemism, where 38% of terrestrial mammals, 27% of passerine birds, and ~20% of snakes and lizards are endemic. The turtles, nonpasserine birds, and bats have fewer endemics (<15%), with no crocodilians being endemic (Table 9.2).

### 9.3.1 *Spatial Patterns of Diversity*

In describing distributions, we excluded caecilians and crocodilians because they are represented by too few species (ten, four, respectively) to provide meaningful maps. Because anurans comprise more than 98% of all amphibians, we simply joined all amphibians in one map. Spatial variation in species richness follows a well-known geographic pattern, with the greatest number of species at the center of the ecoregion: in the mountainous region of the Serra do Mar, decreasing toward the extremes (Fig. 9.1). This distribution pattern is clearly discernable in amphibians, birds, mammals, and most of their subgroups, including nonpasserines, passerines, and terrestrial mammals. Bats were the exception with a clear gradient of increasing species richness toward the equator. This pattern of increased species richness

**Table 9.2** Diversity and endemism of the Atlantic Forest's species of Tetrapoda, presented by subgroups of species that share taxonomic affiliation and/or ecological function

| | Number of species | Number of endemics | Percentage of endemics |
|---|---|---|---|
| Amphibia | | | |
| Caecilians | 12 | 9 | 75.0 |
| Anurans | 707 | 495 | 70.0 |
| Reptilia | | | |
| Snakes | 278 | 64 | 23.0 |
| Lizards | 174 | 39 | 22.4 |
| Turtles | 21 | 3 | 14.3 |
| Crocodilians | 4 | 0 | 0.0 |
| Amphisbaenids | 40 | 20 | 50.0 |
| Aves | | | |
| Nonpasserines | 449 | 59 | 13.1 |
| Passerines | 576 | 156 | 27.1 |
| Mammalia | | | |
| Terrestrial mammals | 262 | 101 | 38.5 |
| Bats | 122 | 8 | 6.6 |

toward the core of the Atlantic Forest is well-known (Costa et al. 2000; Campos et al. 2017; Vale et al. 2018) and is usually associated with the topographic variability in the Serra do Mar. Due to this variable topographic relief, mountainous regions present strong environmental gradients and spatial heterogeneity over short geographic distances (Janzen 1967; Ruggiero and Hawkins 2008) that together favor greater speciation rates (Fine 2015) and species turnover, resulting in greater overall diversity (Melo et al. 2009).

Reptiles, in turn, have contrasting patterns, with greater species richness in the semideciduous forests of the states of São Paulo and Minas Gerais (Fig. 9.1), in a contact zone with the Cerrado ecoregion. Contrary of other Tetrapoda, reptile species richness was not congruent among subgroups. Species richness of lizards was greatest in the dry forests around the Caatinga, while amphisbaenid diversity was greatest in the westernmost portions of the Atlantic Forest, and turtles showed a pattern similar to that of all tetrapods.

When considered only endemic species, different distribution patterns by taxonomic group all disappeared. While the exact center of richness varied somewhat, most endemic groups were more species-rich in the core areas of the Atlantic Forest, in the Serra do Mar region (Fig. 9.2). Two subgroups diverged somewhat from this pattern, with amphisbaenids and bats (both of which with few endemics, 19 and 9, respectively) that had more than one peak of species richness (Fig. 9.2). Restricted-range species richness distributions tended to be similar to those of endemic species, with most groups being most diverse in the Serra do Mar region, with some notable exceptions in the northeast (Fig. 9.3).

Larger versions of the individual maps combined in Figs. 9.1, 9.2, 9.3 and 9.4 are available in the Supplementary Materials.

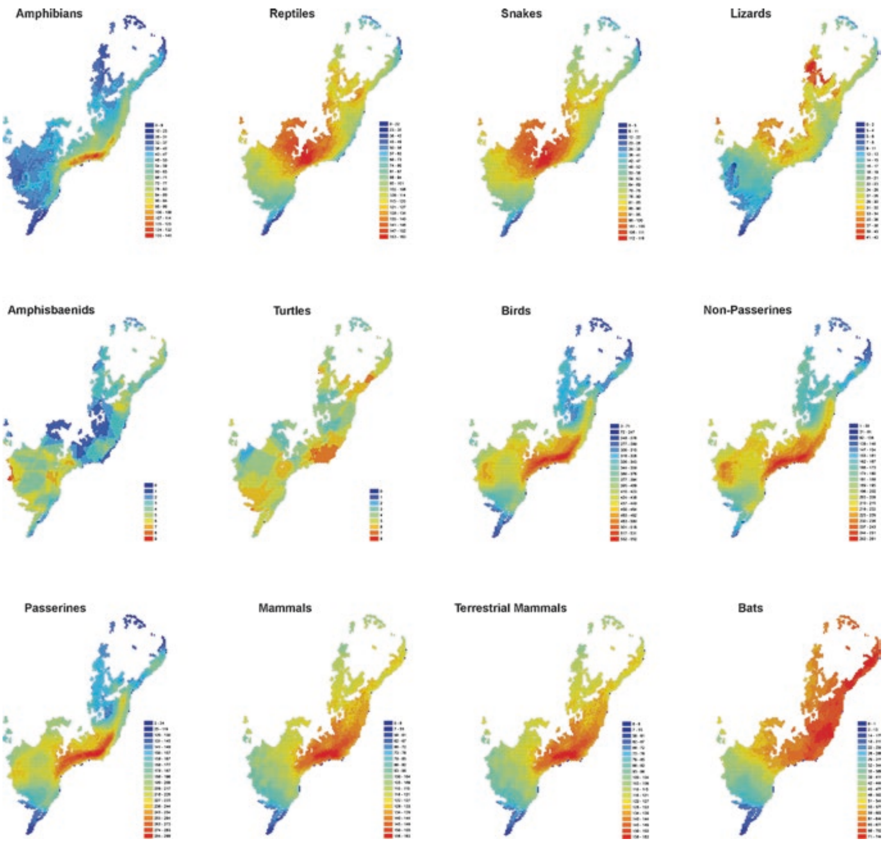

**Fig. 9.1** Distribution of observed species richness for all the groups and subgroups of terrestrial vertebrates in the Atlantic Forest, mapped to a 500 km$^2$ resolution equal-area (Albers) projection grid. The gradient runs from dark red, which indicates the high-richness areas, to dark blue which indicates the low-richness areas

## 9.4 Knowledge Quality

Any analysis of large-scale patterns of diversity must consider the extent of our knowledge regarding biodiversity itself. The unevenness in sampling effort and the lack of adequate taxonomy result in high spatial variation in the quality and reliability of the data available. This compromises our ability to find and interpret biodiversity patterns, as we work with limited and often biased information (Riddle et al. 2011). These gaps in knowledge, also known as shortfalls, are related to limited information regarding species' taxonomy, ecology, evolution, and/or biogeography (Hortal et al. 2015). There are seven identified shortfalls, two of them have preponderant effects on large-scale patterns of diversity. We discuss these two shortfalls below.

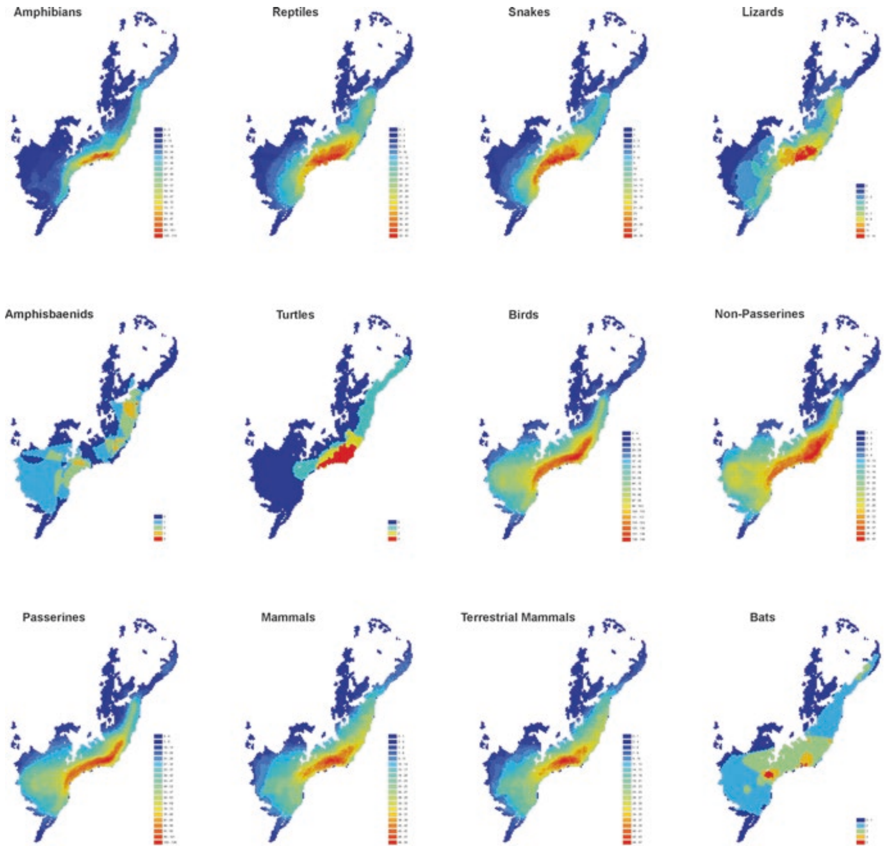

**Fig. 9.2** Distribution of observed endemic species richness for all the groups and subgroups of terrestrial vertebrates in the Atlantic Forest, mapped to a 500 $km^2$ resolution equal-area (Albers) projection grid. The gradient runs from dark red, which indicates the high-richness areas, to dark blue which indicates the low-richness areas

### 9.4.1 *Linnean Shortfall*

This is the first shortfall and refers to the discrepancy between the number of described (and named) species and the estimated number of species that actually exist (Raven and Wilson 1992). The magnitude of the Linnean shortfall is unknown for many reasons, the main one of which is that we cannot rely on an accurate estimate of unknown species. In practice, the number of formally described species changes constantly as a result of new descriptions and taxonomic revisions. There are two categories of unknown species: those yet to be sampled and those sampled but yet to be described. Both categories influence our knowledge of the tetrapods of the Atlantic Forest.

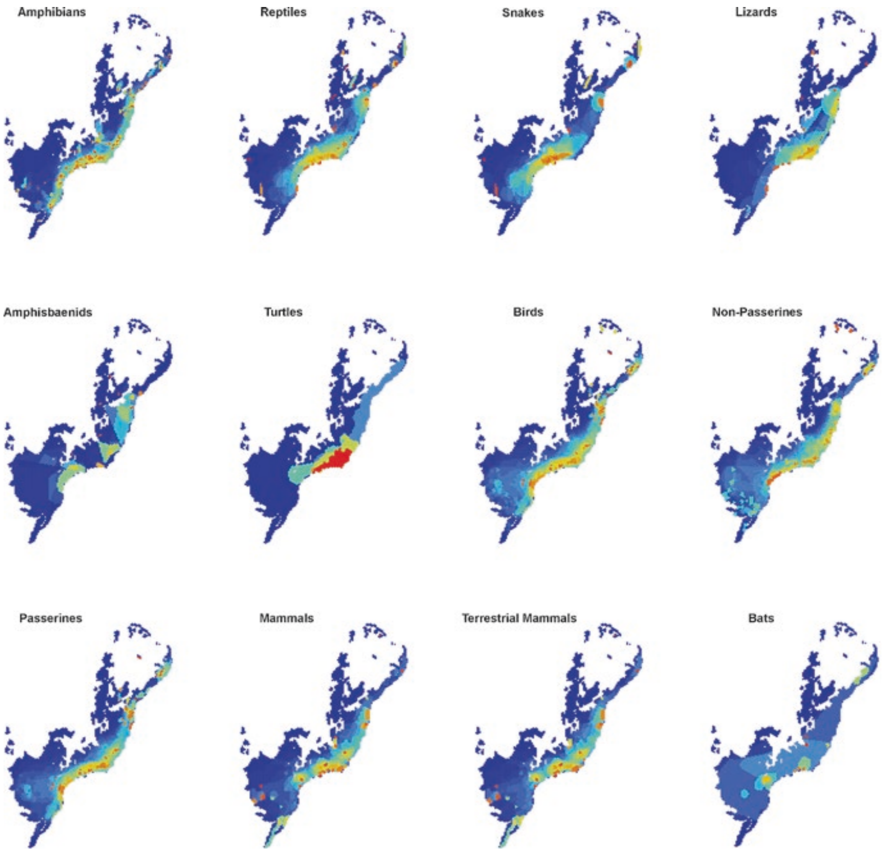

**Fig. 9.3** Distribution of observed restricted-range richness for all the groups and subgroups of terrestrial vertebrates in the Atlantic Forest, mapped to a 500 km$^2$ resolution equal-area (Albers) projection grid. The gradient runs from dark red, which indicates the areas with high values of restricted-range richness, to dark blue which indicates the areas with low values of restricted-range richness

Species yet to be sampled include those inhabiting unsurveyed or hard-to-sample locations or habitat types, such as rainforest canopies or mountain tops (Hortal et al. 2015). Because sampling effort has been concentrated in the center-south of the Atlantic Forest (Figueiredo et al. 2017; Lima et al. 2017; Muylaert et al. 2017; Hasui et al. 2018; Vancine et al. 2018), it is not surprising that many newly described species were based on newly collected individuals from the northeast. For example, the lizard *Placosoma limaverdorum* (Borges-Nojosa et al. 2016), the prehensile-tailed porcupines *Coendou baturitensis* and *C. speratus*, the agouti *Dasyprocta iacki*, and the passerine *Scytalopus gonzagai* (Feijó and Langguth 2013; Pontes et al. 2013; Maurício et al. 2014) were all found in enclaves of the Atlantic Forest surrounded by the Caatinga. Other species have recently been discovered in other poorly sampled environments, such as montane or cloud forests (Whitney et al.

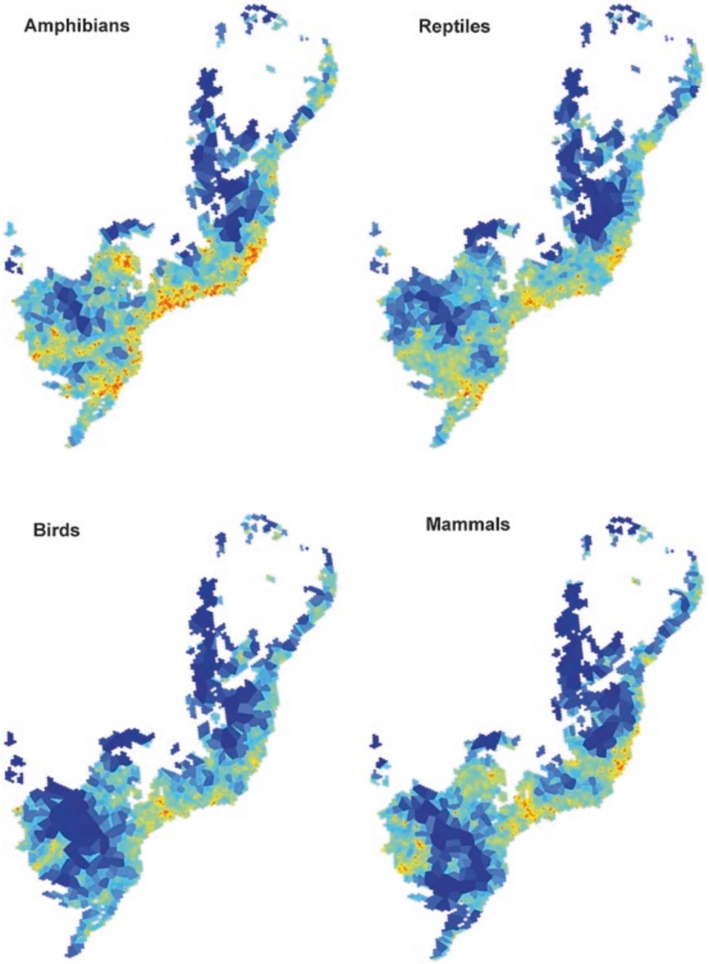

**Fig. 9.4** Thiessen polygon networks to represent sampling density. Maps derive from data from the Global Biodiversity Information Facility (GBIF) and from Brazil's SpeciesLink site. The gradient runs from dark red, which indicates the well-sampled areas, to dark blue which indicates the most poorly sampled areas

2010; Brusquetti et al. 2013; Costa et al. 2015; Ribeiro et al. 2015; Bornschein et al. 2016), wetlands (Buzzetti et al. 2013; Quintela et al. 2014, 2017), coastal sandy plains (Tavares et al. 2011; Cardozo et al. 2018), and coastal islands (Barbo et al. 2012, 2016).

Not all new species, however, are from poorly sampled regions. For example, the type localities for the vesper mouse *Calomys cerqueirai* (Bonvicino et al. 2010) and the bat *Dryadonycteris capixaba* (Nogueira et al. 2012) are located in the Rio Doce basin, and the short-tailed opossum *Monodelphis pinocchio* was first identified in the Morro Grande Forest Reserve (Pavan 2015). Two species of the ground-dwelling frogs of the genus *Ischnocnema* were described from individuals collected in the Serra dos Órgãos National Park and Augusto Ruschi Biological Reserve (Taucce et al. 2018). All of these places are in typical and reasonably well-known and studied places within the Atlantic Forest in southeastern Brazil.

The second category of unknown species is more common for small or cryptic species (Riddle et al. 2011), and all are from taxonomic reviews using museum specimens or molecular phylogenies. For instance, two species of snakes of the genus *Atractus* (Passos et al. 2010) and three cryptic species of frogs of the genus *Chiasmocleis* (Forlani et al. 2017) were described based on taxonomic reviews of these genera. The bat species *Lonchophylla peracchii* was split from *Lonchophylla bokermanni* (Dias et al. 2013), while six bird species that were former Atlantic Forest endemic subspecies were recently elevated to the species level (Vale et al. 2018). The rodent species *Delomys altimontanus* and *Rhipidomys itoan* were recently supported by phylogeography and genetics, in addition to traditional morphological analyses (Costa et al. 2011; Gonçalves and Oliveira 2014). Thus, with increased information, new species arise by discovery as well as new by improved analysis.

Lastly, five additional species fit somewhat in both categories, because they were described based on collected specimens that, with additional field work, were recognized as being new species. The bat *Myotis izecksohni* and the dwarf boas *Tropidophis grapiuna* and *T. preciosus* were described after a taxonomic review of *Myotis nigricans* and *Tropidophis paucisquamis* based on recent capture efforts (Moratelli et al. 2011; Curcio et al. 2012). The passerine bird *Cichlocolaptes mazarbarnetti* was split (and included in a distinct genus) from *Philydor novaesi* due to the differences in their behaviors and vocalizations examined during recent field work (Mazar Barnett and Buzzetti 2014). The rat *Drymoreomys albimaculatus*, a very distinctive new genus and species of the tribe Oryzomyini, was described from museum specimens after that inventories on the Atlantic Forest in the state of São Paulo sampled several new specimens (Percequillo et al. 2011).

The Linnean shortfall for Neotropical species is always decreasing over time thanks to the work of taxonomists (Stevens et al. 2020). New species will continue to be described in many places in the world that are already studied, as well as in relatively well-known taxonomic groups. During the last decade, 157 new species of tetrapods were described in the Atlantic Forest, including ten birds (four nonpasserines and six passerines), 14 mammals (three bats and 12 terrestrial mammals), 14 reptiles (one amphisbaenid, six lizards, and seven snakes), and 118 amphibians. The number of endemic species and the evidence of the Linnean shortfall might increase considerably if future taxonomic reviews find that other Atlantic Forest endemic subspecies should be elevated to species level (Vale et al. 2018) or that new species should be described, as expected by the example of the discovery of three yet undescribed new species of the rodent *Phyllomys* (Loss and Leite 2011).

### *9.4.2 Wallacean Shortfall*

The second shortfall refers to the gap of knowledge associated with the geographic distribution of species (Lomolino 2004). The Wallacean shortfall is due to the temporal and spatial variation in surveying efforts (Hortal et al. 2015) that tend to be biased toward more accessible areas (Dennis and Thomas 2000; Vale and Jenkins 2012) and with a greater number of researchers (Rodrigues et al. 2010). These biases may influence perceived and described spatial patterns of species richness (e.g., Nelson et al. 1990; Reddy and Liliana 2003; Bini et al. 2006; Tobler et al. 2007; Werneck et al. 2011).

The Atlantic Forest is the Brazilian ecoregion with the greatest number of biological studies (Jenkins et al. 2015), and access to those data has increased dramatically as online databases become available. We investigated spatial bias in biological surveys of terrestrial vertebrates in the Atlantic Forest, testing whether biological surveys are spatially biased toward access points, such as cities and roads, or high-richness areas.

We gathered occurrence records for all birds ($N = 1661$), mammals ($N = 2502$), reptiles ($N = 3547$), and amphibians ($N = 3883$) within the Atlantic Forest from Global Biodiversity Information Facility (http://www.gbif.org) and SpeciesLink (http://www.splink.org.br). We assumed that if there is a record, there was a biological survey. We compiled records from material sample, literature, and preserved specimen only (excluding mere "observations") and eliminated duplicates and poorly defined geographic coordinates. Using a geographic information system, we generated the same number of randomly distributed points as biological survey sites for each taxon. We then calculated the distance between inventory sites and random points to the nearest access point (roads and cities), using data derived from OpenStreetMap.org (downloaded from MapCruzin, https://mapcruzin.com/free-south-america-arcgis-maps-shapefiles.htm). We also extracted species richness values at each inventory site and random point from richness maps in raster format at a spatial resolution of 10 × 10 km (in an equal area projection – South America Albers Equal Area Conic) derived from Biodiversity Mapping (Jenkins et al. 2013) for birds, mammals, and amphibians, and from Meiri et al. (2017) for reptiles. We compared data from inventory sites and random points using Cohen's $d$, a measure of effect size defined as the difference between two means divided by a standard deviation for the data (Cohen 1988). Effect size is small when $d \approx 0.2$, medium when $d \approx 0.5$, and large when $d \approx 0.8$ (Cohen 1988). We created a network of Thiessen polygons for each taxon to represent the spatial distribution of sampling effort. In the network, each locality point generates one polygon; the larger the polygon, the lower the survey density (Jenkins et al. 2015).

Biological surveys in the Atlantic Forest tended to be biased toward roads for reptiles, moderately biased toward access points for birds and amphibians, and almost unbiased for mammals (Table 9.3). For amphibians, the analysis suggested the existence of a moderately high survey bias toward high-richness areas, while for birds and mammals, this survey bias is moderate and is moderately low for reptiles

**Table 9.3** Cohen's *d* of the distance to the nearest access point in the distribution of biological surveys and species richness at biological survey sites. The effect was considered small when $d \approx 0.2$, while $d \approx 0.5$ represents a medium effect size, and $d \approx 0.8$ represents a large effect size

| |  |  |  |  |
|---|---|---|---|---|
| Distance to nearest city | 0.5 | 0.5 | 0.6 | 0.2 |
| Distance to nearest road | 0.6 | 0.7 | 0.6 | 0.4 |
| Species richness | 0.7 | 0.3 | 0.6 | 0.5 |

(Table 9.3). Biological surveys tend to be concentrated along the Atlantic coast, mostly in the states of Rio de Janeiro, São Paulo, and Espírito Santo, for all four tetrapod groups (Fig. 9.4). Conversely, large gaps in sampling effort were identified in the interior forests of the northeast and in the meridional plateau in the south, roughly corresponding to four of the bioregions identified by Silva and Casteleti (2003): *Araucaria forests*, *Brejos Nordestinos*, *Diamantina*, and *São Francisco*.

Wallacean shortfall is being reduced through new approaches. For example, the use of increasingly common collaborative databases of species occurrences is providing better information of species occurrence and abundance (e.g., Figueiredo et al. 2017; Lima et al. 2017; Muylaert et al. 2017; Hasui et al. 2018; Vancine et al. 2018). Likewise, citizen science programs result in large volumes of data gathered (Silvertown 2009; Follett and Strezov 2015), often filling sample gaps on a large spatial scale. Even with these initiatives, some species are still hard to find and study, and so due to missing occurrence data, often poorly understood or very rare species suddenly have large range expansions (Passamani et al. 2011; Cerboncini et al. 2014). For example, the bat *Lasiurus ebenus* was described from a single individual on the Ilha do Cardoso State Park, São Paulo (Fazzolari-Correa 1994), and its taxonomic status was questioned. Recently, Cláudio et al. (2018) reported the second record of this species that was collected in Carlos Botelho State Park, also in the state of São Paulo, more than 20 years after its description and 90 km away. This is the most extreme case of Wallacean shortfall for an endemic bat of the Atlantic Forest but is likely to be similar to that for other endemic bats. For example, *Dryadonycteris capixaba* is known from six localities (Rocha et al. 2014), *Myotis izecksohni* is known from seven (Dias et al. 2015), and *Eptesicus taddei* is known from 12 (Bernard et al. 2013). The situation is similar for many species of endemic birds of the Atlantic Forest, and birds are often the best-known group in any region. Using data from the often-used citizen science database WikiAves (www.wikiaves.com.br), 14 species are known from fewer than ten localities (Santos 2019). Eight of those species (*Acrobatornis fonsecai*, *Cichlocolaptes mazarbarnetti*, *Eleoscytalopus psychopompus*, *Glaucis dohrnii*, *Leptodon forbesi*, *Merulaxis stresemanni*, *Philydor novaesi*, and *Scytalopus gonzagai*) are endemic to the forests of the northeast, which reinforces the importance of increasing sampling effort in this region.

Another way to avoid the Wallacean shortfall is using species distribution modeling (SDM). This approach uses occurrence data of a species and environmental variables (such as temperature, precipitation) to describe the distribution (Peterson and Soberón 2012), and it is especially useful to predict distributions of rare or endangered species (Weber et al. 2010; Teixeira et al. 2014). For example, *Lonchophylla peracchii* is known from about 20 localities (Teixeira et al. 2013), and a distribution model for this species indicated where undiscovered populations might occur and guided subsequent sampling expeditions (Teixeira et al. 2014). Indeed, a new population of *L. peracchi* was found (Teixeira et al. 2013). SDM, when developed properly, may provide useful and realistic estimates of distributions of rare and endemic species or, at the very least, indicate future study sites to attempt to fill the gaps (Weber et al. 2010; Teixeira et al. 2014).

## 9.5 Concluding Remarks

The Atlantic Forest is one of the most diverse ecoregions on Earth, encompassing 8% of all species of tetrapods in the world, ~3% of which are found nowhere else. The Atlantic Forest biota is noteworthy in having an impressive diversity of amphibians (>700 species), with almost 10% of the global total number of amphibian species. It is astonishing that 20% of all Atlantic Forest amphibian species were only described during the past 10 years.

Species richness is greatest at the core of the Atlantic Forest, especially with respect to endemic species. This central, coastal portion of the Atlantic Forest shows a high spatial heterogeneity in topographic conditions, ranging from the sea level to the mountain tops of the Serra do Mar. Apparently, there is a strong, positive association between topography and tetrapod species richness in the Atlantic Forest. Only three groups do not follow this general pattern of richness distribution. Snakes and lizards, due to their ecologic characteristics, showed a different pattern where species richness reaches the greatest values at the contact zones with the dryer savannas, the Cerrado and Caatinga. Bats also follow a distinct, clear pattern of increasing species richness toward the equator.

In addition to the extreme spatial heterogeneity in topography, the core of the Atlantic Forest has also been more extensively studied than other regions in South America and concentrates a disproportionate amount of biological surveys. We found the largest sampling gaps in the interior forests of the northeast and south, where several new species were recently described. Thus, in the future, the Atlantic Forest, already among the most diverse in the world, will soon be found to have even greater species diversity.

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## References

Barbo FE, Gasparini JL, Almeida AP et al (2016) Another new and threatened species of lancehead genus *Bothrops* (Serpentes, Viperidae) from Ilha dos Franceses, Southeastern Brazil. Zootaxa 4097:511–529. https://doi.org/10.11646/zootaxa.4097.4.4

Barbo FE, Grazziotin FG, Sazima I et al (2012) A new and threatened insular species of lancehead from Southeastern Brazil. Herpetologica 68:418–429

Bernard E, do Nascimento JL, de Aguiar LMS (2013) Flagging a species as threatened: the case of *Eptesicus taddeii*, an endemic bat from the Brazilian Atlantic Forest. Biota Neotrop 13:314–318

Bini LM, Diniz-Filho JAF, Rangel TFLVB et al (2006) Challenging Wallacean and Linnean shortfalls: knowledge gradients and conservation planning in a biodiversity hotspot. Divers Distrib 12:475–482. https://doi.org/10.1111/j.1366-9516.2006.00286.x

BirdLife International (2018) HBW and BirdLife Taxonomic Checklist. Version 3. http://datazone.birdlife.org/species/taxonomy

BirdLife International, NatureServe (2011) Bird species distribution maps of the world. BirdLife International & NatureServe, Cambridge & Arlington

Bonvicino CR, Oliveira JA, Gentile R (2010) A new species of *Calomys* (Rodentia: Sigmodontinae) from eastern Brazil. Zootaxa 2336:19–25

Borges-Nojosa DM, Caramaschi U, Rodrigues MT (2016) A new species of lizard *Placosoma* Tschudi, 1847 (Squamata: Gymnophthalmidae) from the relictual forest mountains of the state of Ceará, Brazil. Zootaxa 4169:160–170. https://doi.org/10.11646/zootaxa.4169.1.8

Bornschein MR, Ribeiro LF, Blackburn DC et al (2016) A new species of *Brachycephalus* (Anura: Brachycephalidae) from Santa Catarina, southern Brazil. PeerJ 4:e2629. https://doi.org/10.7717/peerj.2629

Brusquetti F, Thomé MTC, Canedo C et al (2013) A new species of *Ischnocnema parva* species series (Anura, Brachycephalidae) from northern state of Rio de Janeiro, Brazil. Herpetologica 69:175–185. https://doi.org/10.1655/herpetologica-d-12-00050

Buckner JC, Lynch Alfaro JW, Rylands AB, Alfaro ME (2015) Biogeography of the marmosets and tamarins (Callitrichidae). Mol Phylogenet Evol 82:413–425. https://doi.org/10.1016/j.ympev.2014.04.031

Buzzetti DRC, Belmonte-Lopes R, Reinert BL et al (2013) A new species of *Formicivora* Swainson, 1824 (Thamnophilidae) from the state of São Paulo, Brazil. Rev Bras Ornitol 21:269–291

Campos FS, Lourenço-de-Moraes R, Llorente GA, Solé M (2017) Cost-effective conservation of amphibian ecology and evolution. Sci Adv 3:1–10. https://doi.org/10.1126/sciadv.1602929

Cardozo DE, Baldo D, Pupin N et al (2018) A new species of *Pseudopaludicola* (Anura, Leiuperinae) from Espírito Santo, Brazil. PeerJ 6:e4766. https://doi.org/10.7717/peerj.4766

Cerboncini RAS, Zanata TB, da Cunha WL et al (2014) Distribution extension of *Abrawayaomys ruschii* Cunha and Cruz, 1979 (Rodentia: Cricetidae) with the first records in the state of Paraná, southern Brazil. Check List 10:660–662. https://doi.org/10.15560/10.3.660

Cláudio VC, Barbosa GP, Novaes RLM et al (2018) Second record of *Lasiurus ebenus* (Chiroptera, Vespertilionidae), with comments on its taxonomic status. Zootaxa 4403:513–522. https://doi.org/10.11646/zootaxa.4403.3.5

Cohen J (1988) Statistical power analysis for the behavioral sciences, 2nd edn. Lawrence Erlbaum, Hillsdale de Costa BMA, Geise L, Pereira LG, Costa LP (2011) Phylogeography of *Rhipidomys* (Rodentia: Cricetidae: Sigmodontinae) and description of two new species from southeastern Brazil. J Mammal 92:945–962. https://doi.org/10.1644/10-MAMM-A-249.1

Costa HC, Resende FC, Teixeira-Jr M et al (2015) A new *Amphisbaena* (Squamata: Amphisbaenidae) from southern Espinhaco Range, Southeastern Brazil. An Acad Bras Cienc 87:891–901. https://doi.org/10.1590/0001-3765201520140088

Costa LP (2003) The historical bridge between the Amazon and the Atlantic Forest of Brazil: a study of molecular phylogeography with small mammals. J Biogeogr 30:71–86

Costa LP, Leite YLR, Fonseca GAB, Fonseca MT (2000) Biogeography of South American forest mammals: endemism and diversity in the Atlantic Forest. Biotropica 32:872–881

Curcio FF, Sales Nunes PM, Argolo AJS et al (2012) Taxonomy of the South American dwarf boas of the genus *Tropidophis* Bibron, 1840, with the description of two new species from the Atlantic Forest (Serpentes: Tropidophiidae). Herpetol Monogr 26:80–121. https://doi.org/10.1655/herpmonographs-d-10-00008.1

Dennis RLH, Thomas CD (2000) Bias in butterfly distribution maps: the influence of hot spots and recorder's home range. J Insect Conserv 4:73–77. https://doi.org/10.1023/A:1009690919835

Dias D, Carvalho WD, Teixeira TSM et al (2015) First record of *Myotis izecksohni* (Chiroptera, Vespertilionidae) for the Atlantic Forest of Minas Gerais, southeastern Brazil. Mastozool Neotrop 22:149–153. https://doi.org/10.1146/annurev.biochem.67.1.509

Dias D, Esbérard CEL, Moratelli R (2013) A new species of *Lonchophylla* (Chiroptera, Phyllostomidae) from the Atlantic Forest of southeastern Brazil, with comments on *L. bokermanni*. Zootaxa 3722:347–360. https://doi.org/10.11646/zootaxa.3722.3.4

Eisenlohr PV, de Oliveira-Filho AT (2015) Revisiting patterns of tree species composition and their driving forces in the Atlantic Forests of Southeastern Brazil. Biotropica 47:689–701. https://doi.org/10.1111/btp.12254

Faurby S, Svenning JC (2015) A species-level phylogeny of all extant and late quaternary extinct mammals using a novel heuristic-hierarchical Bayesian approach. Mol Phylogenet Evol 84:14–26. https://doi.org/10.1016/j.ympev.2014.11.001

Fazzolari-Correa S (1994) *Lasiurus ebenus*, a new vespertilionid bat from southeastern brasil. Mammalia 58:119–124. https://doi.org/10.1515/mamm.1994.58.1.119

Feijó A, Langguth A (2013) Mamíferos de médio e grande porte do Nordeste do Brasil: distribuição e taxonomia, com descrição de novas espécies. Rev Nord Biol 22:3–225. https://doi.org/10.1017/CBO9781107415324.004

Figueiredo MSL, Barros CS, Delciellos AC et al (2017) Abundance of small mammals in the Atlantic Forest (ASMAF): a data set for analyzing tropical community patterns. Ecology 98:2981. https://doi.org/10.1002/ecy.2005

Figueiredo MSL, Grelle CEV (2018) Phylogenetic diversity as a key to understand mechanisms of New World marsupials diversification (Didelphimorphia: Didelphidae). Oecologia Aust 22:168–178

Fine PVA (2015) Ecological and evolutionary drivers of geographic variation in species diversity. Annu Rev Ecol Evol Syst 46:369–392. https://doi.org/10.1146/annurev-ecolsys-112414-054102

Follett R, Strezov V (2015) An analysis of citizen science based research: usage and publication patterns. PLoS One 10:e0143687. https://doi.org/10.1371/journal.pone.0143687

Forlani MC, Tonini JFR, Cruz CAG et al (2017) Molecular and morphological data reveal three new cryptic species of *Chiasmocleis* (Mehely 1904) (Anura, Microhylidae) endemic to the Atlantic Forest, Brazil. PeerJ 5:e3005. https://doi.org/10.7717/peerj.3005

Gonçalves PR, Oliveira JA (2014) An integrative appraisal of the diversification in the Atlantic Forest genus *Delomys* (Rodentia: Cricetidae: Sigmodontinae) with the description of a new species. Zootaxa 3760:1–38. https://doi.org/10.11646/zootaxa.3760.1.1

Hasui É, Metzger JP, Pimentel RG et al (2018) ATLANTIC BIRDS: a data set of bird species from the Brazilian Atlantic Forest. Ecology 99:497. https://doi.org/10.1002/ecy.2119

Hortal J, de Bello F, Diniz-Filho JAF et al (2015) Seven shortfalls that beset large-scale knowledge of biodiversity. Annu Rev Ecol Evol Syst 46:523–549. https://doi.org/10.1146/annurev-ecolsys-112414-054400

IUCN (2014) The IUCN Red List of Threatened Species. Version 2014.1. http://www.iucnredlist.org

Janzen DH (1967) Why mountain passes are higher in the tropics. Am Nat 101:233–249

Jenkins CN, Alves MAS, Uezu A, Vale MM (2015) Patterns of vertebrate diversity and protection in Brazil. PLoS One 10:e0145064. https://doi.org/10.1371/journal.pone.0145064

Jenkins CN, Pimm SL, Joppa LN (2013) Global patterns of terrestrial vertebrate diversity and conservation. Proc Natl Acad Sci 110:E2602–E2610. https://doi.org/10.1073/pnas.1302251110

Jetz W, Pyron RA (2018) The interplay of past diversification and evolutionary isolation with present imperilment across the amphibian tree of life. Nat Ecol Evol 2:850–858. https://doi.org/10.1038/s41559-018-0515-5

Ledo RMD, Colli GR (2017) The historical connections between the Amazon and the Atlantic Forest revisited. J Biogeogr 44:2551–2563. https://doi.org/10.1111/jbi.13049

Leite YLR, Costa LP, Loss AC et al (2016) Neotropical forest expansion during the last glacial period challenges refuge hypothesis. Proc Natl Acad Sci 113:1008–1013. https://doi.org/10.1073/pnas.1513062113

Lima F, Beca G, Muylaert RL et al (2017) ATLANTIC-CAMTRAPS: a dataset of medium and large terrestrial mammal communities in the Atlantic Forest of South America. Ecology 98:2979. https://doi.org/10.1002/ecy.1998

Lomolino MV (2004) Conservation biogeography. In: Lomolino MV, Heaney LR (eds) Frontiers of biogeography: new directions in the geography of nature. Sinauer Associates, Sunderland, pp 293–296

Loss AC, Leite YLR (2011) Evolutionary diversification of *Phyllomys* (Rodentia: Echimyidae) in the Brazilian Atlantic Forest. J Mammal 92:1352–1366. https://doi.org/10.1644/10-MAMM-A-320.1

Lynch Alfaro JW, Boubli JP, Olson LE et al (2012) Explosive Pleistocene range expansion leads to widespread Amazonian sympatry between robust and gracile capuchin monkeys. J Biogeogr 39:272–288. https://doi.org/10.1111/j.1365-2699.2011.02609.x

Marquet PA, Fernández M, Navarrete SA, Valdovinos C (2004) Diversity emerging: toward a deconstruction of biodiversity patterns. In: Lomolino MV, Heaney LR (eds) Frontiers of biogeography: new directions in the geography of nature. Sinauer Associates, Sunderland, pp 191–209

Maurício GN, Belmonte-Lopes R, Pacheco JF et al (2014) Taxonomy of "Mouse-colored Tapaculos" (II): an endangered new species from the montane Atlantic Forest of southern Bahia, Brazil (Passeriformes: Rhinocryptidae: *Scytalopus*). Auk 132:951–952. https://doi.org/10.1642/auk-15-123.1

Mazar Barnett J, Buzzetti DRC (2014) A new species of *Cichlocolaptes* Reichenbach 1853 trace of the fading bird life of northeastern Brazil. Rev Bras Ornitol 22:75–94

Meiri S, Roll U, Grenyer R et al (2017) Data from: the global distribution of tetrapods reveals a need for targeted reptile conservation. Dryad Digital Repository

Melo AS, Rangel TFLVB, Diniz-Filho JAF (2009) Environmental drivers of beta-diversity patterns in new-world birds and mammals. Ecography 32:226–236. https://doi.org/10.1111/j.1600-0587.2008.05502.x

Monteiro-Filho ELA, Conte CE (2017) Revisões em Zoologia: Mata Atlântica, 1st edn. Editora UFPR, Curitiba

Moratelli R, Peracchi AL, Dias D, Oliveira JA (2011) Geographic variation in South American populations of *Myotis nigricans* (Schinz, 1821) (Chiroptera, Vespertilionidae), with the description of two new species. Mamm Biol 76:592–607. https://doi.org/10.1016/j.mambio.2011.01.003

Muylaert RDL, Stevens RD, Esbérard CEL et al (2017) ATLANTIC BATS: a data set of bat communities from the Atlantic Forests of South America. Ecology 98:3227. https://doi.org/10.1002/ecy.2007

Muylaert RL, Vancine MH, Bernardo R et al (2018) Uma nota sobre os limites territoriais da Mata Atlântica. Oecologia Aust 22:302–311. https://doi.org/10.4257/oeco.2018.2203.09

Myers N, Mittermeier RA, Mittermeier CG et al (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858. https://doi.org/10.1038/35002501

Nelson BW, Ferreira CAC, Silva MF, Kawasaki ML (1990) Endemism centres, refugia and botanical collection density in Brazilian Amazonia. Nature 345:714–716

Nogueira MR, Lima IP, Peracchi AL, Simmons NB (2012) New genus and species of nectar-feeding bat from the Atlantic Forest of southeastern Brazil (Chiroptera: Phyllostomidae: Glossophaginae). Am Museum Novit 3747:1–30. https://doi.org/10.1206/3747.2

Oliveira-Filho AT, Fontes MAL (2000) Patterns of floristic differentiation among Atlantic Forests in southeastern Brazil and the influence of climate. Biotropica 32:793–810. https://doi.org/10.1111/j.1744-7429.2000.tb00619.x

Paglia AP, Pinto LP (2010) Biodiversidade da Mata Atlântica. In: Marone E, Riet D, Melo T (eds) Brasil Atlântico: um país com a raiz na mata. Instituto BioAtlântica, Rio de Janeiro, pp 102–129

Passamani M, Cerboncini RAS, Oliveira JE (2011) Distribution extension of *Phaenomys ferrugineus* (Thomas, 1894), and new data on *Abrawayaomys ruschii* Cunha and Cruz, 1979 and *Rhagomys rufescens* (Thomas, 1886), three rare species of rodents (rodentia: Cricetidae) in Minas Gerais, Bra. Check List 7:827–831. https://doi.org/10.15560/7.6.827

Passos P, Fernandes R, Bérnils RS, Moura-Leite JC (2010) Taxonomic revision of the Brazilian Atlantic Forest *Atractus* (Reptilia: Serpentes: Dipsadidae). Magnolia Press, Auckland

Pavan S (2015) A new species of *Monodelphis* (Didelphimorphia: Didelphidae) from the Brazilian Atlantic Forest. Am Museum Novit 3832:1–15. https://doi.org/10.1206/3832.1

Pellegrino KCM, Rodrigues MT, Waite AN et al (2005) Phylogeography and species limits in the *Gymnodactylus darwinii* complex (Gekkonidae, Squamata): genetic structure coincides with river systems in the Brazilian Atlantic Forest. Biol J Linn Soc 85:13–26

Percequillo AR, Weksler M, Costa LP (2011) A new genus and species of rodent from the brazilian Atlantic Forest (Rodentia: Cricetidae: Sigmodontinae: Oryzomyini), with comments on Oryzomyine biogeography. Zool J Linnean Soc 161:357–390. https://doi.org/10.1111/j.1096-3642.2010.00643.x

Peterson AT, Soberón J (2012) Species distribution modeling and ecological niche modeling: getting the concepts right. Nat Conserv 10:102–107. https://doi.org/10.4322/natcon.2012.019

Pontes ARM, Gadelha JR, Melo ERA et al (2013) A new species of porcupine, genus *Coendou* (Rodentia: Erethizontidae) from the Atlantic Forest of northeastern Brazil. Zootaxa 3646:300

Quintela FM, Bertuol F, González EM et al (2017) A new species of *Deltamys* Thomas, 1917 (Rodentia: Cricetidae) endemic to the southern Brazilian Araucaria Forest and notes on the expanded phylogeographic scenario of *D. kempi*. Zootaxa 4294:71–92. https://doi.org/10.11646/zootaxa.4294.1.3

Quintela FM, Gonçalves GL, Althoff SL et al (2014) A new species of swamp rat of the genus *Scapteromys* Waterhouse, 1837 (Rodentia: Sigmodontinae) endemic to *Araucaria angustifolia* Forest in Southern Brazil. Zootaxa 3811:207–225. https://doi.org/10.11646/zootaxa.3811.2.3

Raven PH, Wilson EO (1992) A fifty-year plan for biodiversity surveys. Science 258:1099–1100

Reddy S, Liliana MD (2003) Geographical sampling bias and its implications for conservation priorities in Africa. J Biogeogr 30:1719–1727

Ribeiro LF, Bornschein MR, Belmonte-Lopes R et al (2015) Seven new microendemic species of *Brachycephalus* (Anura: Brachycephalidae) from southern Brazil. PeerJ 3:e1011. https://doi.org/10.7717/peerj.1011

Ribeiro MC, Martensen AC, Metzger JP et al (2011) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In: Zachos FE, Habel JC (eds) Biodiversity hotspots: distribution and protection of conservation priority areas. Springer-Verlag, Heidelberg, pp 405–434

Riddle BR, Ladle RJ, Lourie SA, Whittaker RJ (2011) Basic biogeography: estimating biodiversity and mapping nature. In: Ladle RJ, Whittaker RJ (eds) Conservation biogeography. Blackwell Publishing, West Sussex, pp 47–92

Rocha PA, Feijó A, Dias D et al (2014) Major extension of the known range of the Capixaba nectar-feeding bat *Dryadonycteris capixaba* (Chiroptera, Phyllostomidae). Is this rare species widely distributed in eastern Brazil? Mastozoología Neotrop 21:361–366

Rodrigues ASL, Gray CL, Crowter BJ et al (2010) A global assessment of amphibian taxonomic effort and expertise. Bioscience 60:798–806. https://doi.org/10.1525/bio.2010.60.10.6

Roll U, Feldman A, Novosolov M et al (2017) The global distribution of tetrapods reveals a need for targeted reptile conservation. Nat Ecol Evol 1:1677–1682. https://doi.org/10.1038/s41559-017-0332-2

Ruggiero A, Hawkins BA (2008) Why do mountains support so many species of birds? Ecography 31:306–315. https://doi.org/10.1111/j.2008.0906-7590.05333.x

Safford HD (1999) Brazilian Páramos I. An introduction to the physical environment and vegetation of the *campos de altitude*. J Biogeogr 26:693–712

Santos ÉBG (2019) A contribuição biogeográfica de bases participativas para o estudo de lacunas espaciais em aves endêmicas da Mata Atlântica. Programa de Pós-Graduação em Biodiversidade Neotropical, Universidade Federal do Estado do Rio de Janeiro

Scarano FR (2002) Structure, function and floristic relationships of plant communities in stressful habitats marginal to the Brazilian Atlantic rainforest. Ann Bot 90:517–524. https://doi.org/10.1093/aob/mcf189

Silva JMC, Casteleti CHM (2003) Status of the biodiversity of the Atlantic Forest of Brazil. In: Galindo-Leal C, de Câmara IG (eds) The Atlantic Forest of South America: biodiversity status, trends, and outlook. Center for Applied Biodiversity Science e Island Press, Washington, D.C., pp 43–59

Silva JMC, Sousa MC, Castelletti CHM (2004) Areas of endemism for passerine birds in the Atlantic Forest, South America. Glob Ecol Biogeogr 13:85–92. https://doi.org/10.1111/j.1466-882X.2004.00077.x

Silvertown J (2009) A new dawn for citizen science Jonathan. Trends Ecol Evol 24:467–471. https://doi.org/10.1016/j.chemosphere.2018.03.203

Sobral-Souza T, Lima-Ribeiro MS (2017) De volta ao passado: revisitando a história biogeográfica das florestas neotropicais úmidas. Oecologia Aust 21:93–107. https://doi.org/10.4257/oeco.2017.2102.01

Sobral-Souza T, Lima-Ribeiro MS, Solferini VN (2015) Biogeography of Neotropical rainforests: past connections between Amazon and Atlantic Forest detected by ecological niche modeling. Evol Ecol 29:643–655. https://doi.org/10.1007/s10682-015-9780-9

Stevens RD, Weber MM, Villalobos F (2020) Contemporary biogeography. In: Fleming T, Dávalos LM, Mello MAR (eds) Phyllostomid bats: a unique mammalian radiation. University of Chicago Press, Chicago, pp 391–410

Taucce PPG, Canedo C, Parreiras JS et al (2018) Molecular phylogeny of *Ischnocnema* (Anura: Brachycephalidae) with the redefinition of its series and the description of two new species. Mol Phylogenet Evol 128:123–146. https://doi.org/10.1016/j.ympev.2018.06.042

Tavares WC, Pessôa LM, Gonçalves PR (2011) New species of *Cerradomys* from coastal sandy plains of southeastern Brazil (Cricetidae: Sigmodontinae). J Mammal 92:645–658. https://doi.org/10.1644/10-mamm-096.1

Teixeira TSM, Rosa DTC, Dias D, Vale MM (2013) First record of *Lonchophylla bokermanni* Sazima, Vizotto & Taddei, 1978 (Chiroptera, Phyllostomidae) in São Paulo State, Southeastern Brazil. Oecologia Aust 17:1–11

Teixeira TSM, Weber MM, Dias D et al (2014) Combining environmental suitability and habitat connectivity to map rare or data deficient species in the tropics. J Nat Conserv 22:384–390. https://doi.org/10.1016/j.jnc.2014.04.001

Tobler M, Honorio E, Janovec J, Reynel C (2007) Implications of collection patterns of botanical specimens on their usefulness for conservation planning: an example of two neotropical plant families (Moraceae and Myristicaceae) in Peru. Biodivers Conserv 16:659–677. https://doi.org/10.1007/s10531-005-3373-9

Vale MM, Jenkins CN (2012) Across-taxa incongruence in patterns of collecting bias. J Biogeogr 39:1744–1748. https://doi.org/10.1111/j.1365-2699.2012.02750.x

Vale MM, Tourinho L, Lorini ML et al (2018) Endemic birds of the Atlantic Forest: traits, conservation status, and patterns of biodiversity. J F Ornithol 89:193–206. https://doi.org/10.1111/jofo.12256

Vancine MH, da S Duarte K, de Souza YS et al (2018) ATLANTIC AMPHIBIANS: a data set of amphibian communities from the Atlantic Forests of South America. Ecology 99:1692. https://doi.org/10.1002/ecy.2392

Vivo M, Carmignotto AP (2004) Holocene vegetation change and the mammal faunas of South America and Africa. J Biogeogr 31:943–957. https://doi.org/10.1111/j.1365-2699.2004.01068.x

Weber MM, Terribile LC, Caceres NC (2010) Potential geographic distribution of *Myotis ruber* (Chiroptera, Vespertilionidae), a threatened Neotropical bat species. Mammalia 74:333–338. https://doi.org/10.1515/mamm.2010.037

Werneck MS, Sobral MEG, Rocha CTV et al (2011) Distribution and endemism of angiosperms in the Atlantic Forest. Nat Conserv 9:188–193. https://doi.org/10.4322/natcon.2011.024

Whitney BM, de Vasconcelos MF, Silveira LF, Pacheco JF (2010) *Scytalopus petrophilus* (Rock Tapaculo): a new species from Minas Gerais, Brazil. Rev Bras Ornitol 18:73–88

# Chapter 10
# Freshwater Studies in the Atlantic Forest: General Overview and Prospects

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**Abstract** In this chapter, we reviewed the main studies on freshwater ecosystems of the Atlantic Forest, Brazil. We have firstly provided a list of the hydrographic basins that cross the Atlantic Forest and propose a classification of Atlantic Forest freshwaters based on a proposed wetland classification for Brazil. Anthropogenic disturbances and freshwater changes were discussed. We then made a panorama of studies on the main aquatic communities in the Atlantic Forest, ranging from vertebrates to microalgae. It is clear that basic research is still needed for many biological groups, although recent studies on aquatic biota deserve attention considering their theoretical advances in ecology. Common to studies on all biological groups are the concerns of anthropogenic actions, ranging from habitat alteration due to impoundments to water pollution, over- and illegal exploitation of vertebrates, and massive biological invasions. For many groups, such concerns are reflected in extinction risks of endemic species and biotic homogenization. We present prospects for future studies and urge for greater attention to preserve integrity of freshwaters in the Atlantic Forest.

**Keywords** Aquatic communities · Biodiversity · Anthropogenic disturbances · Dams · Endemic species

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## 10.1 Introduction

Freshwater ecosystems are known to have, disproportionally, a rich diversity of species. With less than 1% of the world's surface (Gleick 1998), the impact of the services that freshwater ecosystems provide is overwhelming. Somewhat paradoxically, a substantial amount of freshwater is used for human survival, and freshwater ecosystems are highly impacted by human activities (Dodds et al. 2013). Threats are diverse, such as flow alteration, organic pollution, species extinctions and invasions (Dudgeon et al. 2006), biotic homogenization (Rahel 2002), thermal alterations, global climate change, increases in ultraviolet radiation (Dodds et al. 2013), and (more recently) microplastic contamination (Zeng 2018). In Brazilian freshwater ecosystems, the scenario is not different, and such threats are strengthened due to the fact that Brazil is one of the countries with the richest freshwater biodiversity in the world (Collen et al. 2014) but with intense dam constructions (Agostinho et al. 2005). Not surprisingly, conservation of Brazilian freshwater biodiversity has been an issue for intense debate (Azevedo-Santos et al. 2019). Aquatic ecosystems in the Atlantic Forest are highly diverse, with high endemism rates and also high proportion of species with data deficiency (see Fig. 1; Collen et al. 2014). It is clear that a review on the actual status of the scientific knowledge on freshwater biodiversity in the Atlantic Forest is extremely necessary.

According to Ribeiro et al. (2011), there are 2650 sub-watersheds in the Atlantic Forest, and only few of them are entirely protected by conservation units (Azevedo-Santos et al. 2019). Due to a major mountain chains that divide watersheds in the Atlantic Forest ("Serra do Mar"; see Leal and Câmara 2003), several watersheds are isolated in the east part of this biome (see watersheds from 1 to 15 and 17, in Fig. 10.1). Probably, a combination of isolation and abundance of freshwater explains the high endemism (Leal and Câmara 2003). Relatedly, the natural barrier promoted by worldwide known "Iguaçu falls" promotes an isolation of fauna and explains an astonishing rate of ~70% of endemism in its native fish fauna (Baumgartner et al. 2012).

In this chapter, we made a panorama of studies in freshwater ecosystems in the Atlantic Forest. We described the main freshwater ecosystems and their overall features, conservation status, and threats. We have also reviewed the overall scenario of studies on the main aquatic communities, from vertebrates to microalgae. We hope our review informs scientists for research prospectus and also provides a support for managers and decision-makers on the best strategies to protect freshwater in the major hydrographic basins of the Atlantic Forest biome.

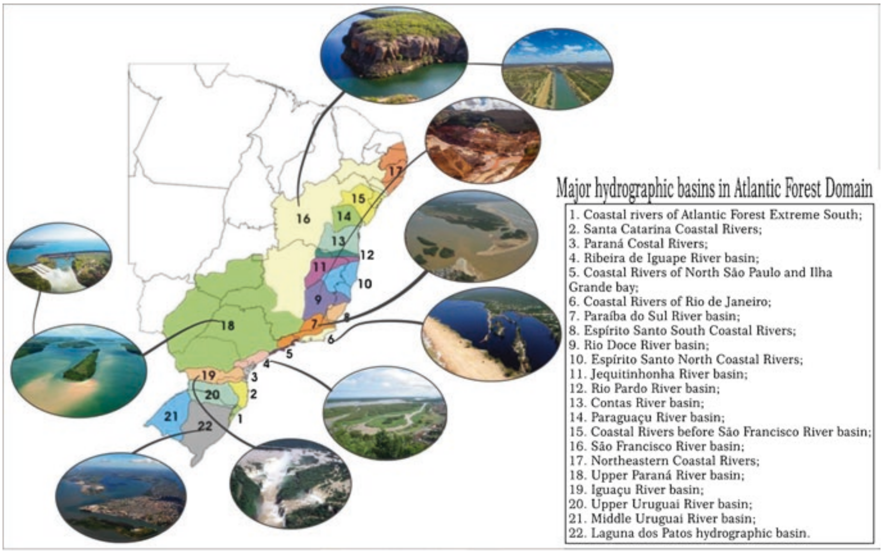

**Fig. 10.1** Major hydrographic basins in the Atlantic Forest domain modified from "Reserva da Biosfera da Mata Atlântica" (available at <http://www.rbma.org.br>), highlighting the diversity of habitats, classified according to Junk et al. (2014), and also major anthropogenic impacts: hydropower reservoirs in Paraná River basin, transposition channel in São Francisco River basin, and tailing dam collapse in Rio Doce River basin (Photos: all freely available at the Internet)

## 10.2 A Classification of Freshwater Ecosystems and the Freshwater Ecosystems in the Atlantic Forest

Freshwater ecosystems comprise many kinds of habitats, and a classification of variety of ecosystems may be a quixotic task. Even so, a tentative classification is useful for managers and decision-makers in conservation or restoration efforts and may support better predictions considering freshwater's main features. A clear classification of freshwater may also help society in understanding and preventing the relatively common events of catastrophic floods and droughts in areas of high human occupancy in the Atlantic Forest, which may become a central concern due to global climate changes. The fact that nearly 20% of Brazilian territory is covered by wetlands is simply astonishing (Junk et al. 2014).

Maybe the most comprehensive attempt to define freshwaters is the one provided by Junk et al. (2014). They hierarchically classified freshwaters based on the features of the land area of freshwater influence, i.e., "wetland". Here, we followed their approach and propose correspondent "freshwater ecosystems" for Atlantic Forest biome. We modified their classification by considering only features of the aquatic ecosystem itself, not the wetland area. The coarser classification proposed by Junk et al. (2014) is "System" and is formed by three units: "Coastal freshwaters", "Inland freshwaters", and "Artificial freshwaters". We agree that this is, so far, also the best attempt to coarsely classify freshwater ecosystems in the Atlantic Forest. Differentiating “Coastal freshwaters” and “Inland freshwaters” makes sense considering either the direct influence of oceans in coastal freshwater due to tides or saline intrusions or atmospheric depositions of dissolved or particulate substances and/or propagules from the ocean, which do not occur with “Inland freshwaters”. Particularly for Atlantic Forest biome, such classification defines entirely numerous hydrographic basins (see Fig. 10.1) given that the major “Serra do Mar” mountain chain divides several basins with or without ocean influence: e.g., several small isolated coastal basins are observed in Costal Atlantic forest. More uncommon in Atlantic forest, “Inland freshwaters” may have a dynamic and continuous transition to “Costal freshwaters” in large and important basins, such as “São Francisco River basin” (see Fig. 10.1).

The fact that artificial (i.e., man-made) freshwaters can be located near or far from the coast was not discussed by Junk et al. (2014). Either way, such ecosystems are different from natural wetlands given they are constantly managed and projected with a clear goal: to provide ecosystem services for human activities. Then, we agree that the best approach is to keep “Artificial freshwaters” as a separate system unit.

Following the hierarchical classification of Junk et al. (2014), we also classified freshwater ecosystems in “subsystems” based on their hydrological main features and then into “classes”, “subclasses” and “macrohabitats”. Some macrohabitats particular to other Brazilian biomes listed in Junk et al. (2014) that were excluded from our classification are not represented, such as palm forest wetlands only from Cerrado biome, wetlands due to riverine habitats only observed in the Amazon basin, the major floodplain in Pantanal biome, and temporary riverine systems in semiarid Northeast Caatinga biome. On the other hand, some units are very representative, such as the numerous “Riparian freshwaters along small rivers (1–5th order)” observed all along the “Serra do Mar” mountain chain, major floodplains (for instance, associated with Paraná River basin), and coastal lagoons. We also consider that some freshwater ecosystems deserve attention for their high importance in ecosystem services and biodiversity conservation. This is the case of phytotelmata (Richardson 1999) in Atlantic Forest domain, where epiphyte bromeliads are highly abundant and diverse, which in turn contribute to high diversity of species in their tanks (Freitas et al. 2016).

Considering the abovementioned facts, the proposed classification of Atlantic Forest freshwaters is summarized in Fig. 10.2. Similarities are clear with Fig. 5 in Junk et al. (2014), and as said, phytotelmata were proposed as a new freshwater “class”. We also proposed that the class “River-floodplains with seasonal precipitation in Atlantic forest” should be included in freshwaters with low-amplitude fluctuating water level. We did so given that tributary rivers with seasonal precipitation in the Atlantic Forest do have low-amplitude floods but with long duration, compared to large floodplain rivers. Most tributaries of São Francisco River, Paraná River, Iguaçu River basin, and Uruguay River will fall into this class.

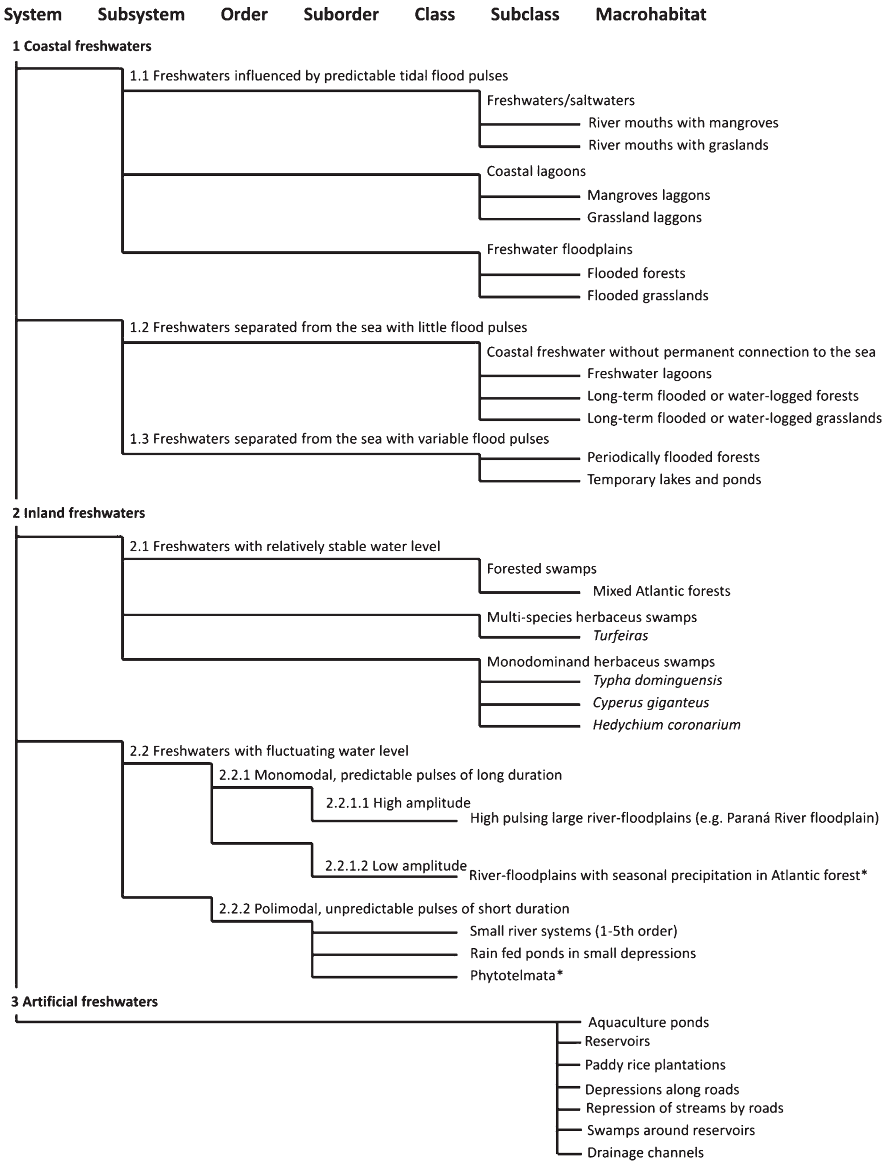

**Fig. 10.2** A proposed classification of Atlantic Forest freshwaters based on a previous classification of Brazilian wetlands (see Fig. 5 in Junk et al. (2014)) and on a non-systematic literature review. * indicates freshwater water bodies not cited as part of wetlands in Junk et al.'s (2014) classification

### 10.2.1 The Novel Ecosystems in the Atlantic Forest and Anthropogenic Threats

Novel ecosystems are defined as those in which biotic and/or abiotic characteristics are altered by humans (Chapin and Starfield 1997). Apart from the fact that nearly all natural freshwater ecosystems in the Atlantic Forest are likely altered by human activities (Agostinho et al. 2005), the number of artificial freshwater ecosystems is disproportionally high. This can be explained by the fact that the highest human occupation in Brazil coincides with Atlantic Forest domain (Ribeiro et al. 2011). The high expansion of hydroelectric dams in South and Southeast Brazil during the last four decades caused major impacts in biodiversity and ecosystem functioning in the Atlantic Forest's major river basins (Agostinho et al. 2005). On the other hand, aquaculture production has also grown in the past decades, aiming not only to the food market but also to the aquarium and ornamental production (Lima-Junior et al. 2018). Among the major threats of these activities are the unsustainable management and the lack of public strategies and supervision in their implementation (Daga et al. 2015).

Among all the Brazilian area covered by some impoundment, more than half of it is located in the Atlantic Forest territory, mostly in the Upper Paraná River and São Francisco River basins (Agostinho et al. 2016). There are more than 200 impoundments in the Atlantic Forest hydrographic basins (Fig. 10.1) including hydropower reservoirs, reservoirs for water supply, and impoundments for flood control (https://www.ana.gov.br). The direct impact of a reservoir is the modification of the river regime flow and the flooding of the surroundings. Ecological alterations are huge, ranging from direct suppression of habitats to the homogenization of the species composition and the introduction of non-native species (Daga et al. 2015). Relatedly, change in regime flow has been one of the causes of biodiversity loss in freshwater environments (Cavalcante et al. 2017). In São Francisco River, the flow has drastically decreased due to the combination of reservoir construction and the major transposition channel constructed for water supply in Brazil's arid northeast region (see Fig. 10.1). As a consequence, there is high saline water intrusion in the largest inland-coastal ecotone freshwater ecosystem in Brazil (Cavalcante et al. 2017). Such intrusion has profound effects in fish assemblage and, as a consequence, to artisanal fisheries (Barbosa and Soares 2009). Finally, water diversion is a new threat for São Francisco River basin. In order to mitigate the decrease in water flow, Brazilian government is about to approve the diversion of water from Amazon River basin to São Francisco River basin. Diversion will likely have profound impacts in aquatic communities and may represent another major invasion event of biota from distinct drainage basins (Daga et al. 2020).

It is also important to notice that after a long policy of construction of large dams, Atlantic Forest freshwater is now threated by small hydropower dams installed in tributary rivers with less influence of the large dams (Arvor et al. 2018).

Although policy makers and dam builders claim that small hydropower dams have few impacts and represent a “clean energy” production, impacts in biota and freshwater ecosystems are severe mainly due to the intense proliferation of dams (Couto and Olden 2018). Numerous small hydropower dams may be even more damaging to aquatic biota than few large dams, given the intense discontinuity that they promote in organism dispersal, including migratory fish (Couto and Olden 2018).

Aquaculture pounds are other novel freshwater ecosystems that deserve attention. The lack of security in fish production leads to accidental escapes of non-native species, besides the emission of large amounts of contaminants (Lima-Junior et al. 2018), leading to the increased pollution and eutrophication of freshwaters (Lima-Junior et al. 2018). However, a new concern is now associated with expansion of aquaculture: the fish culture in reservoirs. At a first sight, it can be seen as an opportunity: aquaculture in an already altered ecosystem, such as reservoirs. However, Brazil does not have any public law that regulates the expansion of aquaculture in reservoirs. It is now clear that such activities facilitate the invasion of non-native species (Lima-Junior et al. 2018). Such “meltdown” of freshwater alterations may be a major impact of aquaculture expansion in reservoirs and should be the focus of intense studies in the near future.

It should also be noticed that the continuous and still under-measured impacts of mine-tailing dam collapse in Atlantic Forest watersheds. We were all witnesses of two recent and unbelievable environmental, social, and economic disasters involving tailing dam collapse in two major watersheds of the Atlantic Forest. The majority of Rio Doce drainage basin (see Fig. 10.1) was completely vanished by the 2015 failure in “Fundão” tailing dam (Hatje et al. 2017). Mud killed 19 people, altered environmental features, and killed biota over more than 650 km until the end of the river in the Atlantic Ocean. Ecological effects are still being discussed and analyzed, but it is clear that effects will last for decades, both in freshwaters and in the ocean shore (Hatje et al. 2017). After such clearly preventable disaster, another tailing dam collapse occurred in 2019, killing more than 300 people and destroying one of the major tributary of Paraopeba River. Ecological effects are still unknown but are incommensurable (Cionek et al. 2019). Lessons not learned and the continuous fear of new tailing dam collapse represent a major threat for freshwater ecosystems in the Atlantic Forest.

It is clear that regularization of public policies and sustainable practices is required for most artificial freshwater systems (Lima-Junior et al. 2018). There is an urgent need to protect and restore the aquatic freshwaters of the Atlantic Forest. Future prospects should be based on the implementation of sustainable programs for artificial wetlands and public strategies that allied scientific studies with socioeconomic needs, having as major objective the conservation of the aquatic freshwaters (Azevedo-Santos et al. 2019).

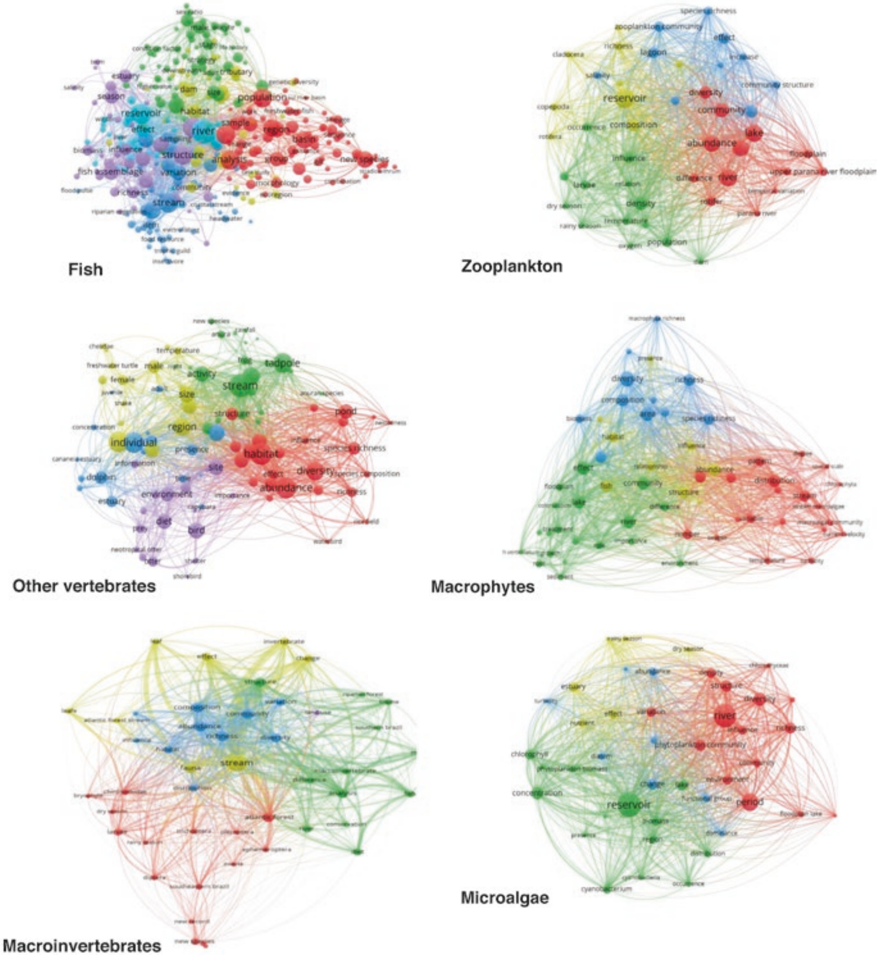

**Fig. 10.3** *(part 1)*. Most common words in studies in the Atlantic Forest for the following groups: *Fish* – 10,281 links extracted from 790 references. Light blue cluster indicates that rivers and reservoirs were the most studied ecosystems. Descriptive studies on new species, group and population evaluations, lineages, and morphology are grouped in the red cluster. Studies on habits such as diets are grouped in blue cluster. Purple cluster highlights studies describing temporal and spatial variation in fish assemblages. The green cluster indicates studies on descriptors of species and populations, like sex ratios, sizes, and strategies. *Amphibians, reptiles, birds, and mammals* – 3737 links extracted from the 714 references. Colors separated mainly biological groups. Birds, otters, and capybaras (purple cluster) are mainly linked with studies describing diet, region, habitat, population, and abundance. Amphibians (mainly tadpoles – green) congregate studies describing stream, biological activities, temperature, variation, population, richness, diversity, abundance, and new species. Studies in mammals (blue cluster) included mainly dolphins and are likely describing sex, regions, estuaries (mainly "Cananéia"), abundance, diversity, and diet. Yellow-green cluster connected turtles in studies describing size, sex, and regions. Red cluster did not link a particular group and represents studies describing habitats, abundance, and diversity. *Macroinvertebrates* – 878 links extracted from 137 references. Groups showed the role of macroinvertebrates on ecological processes (yellow cluster), the relationship between macroinvertebrates and their habitats

## 10.3 A Panorama of Studies in the Main Aquatic Communities of the Atlantic Forest

Here, we also carried out a panorama of studies on main aquatic communities considering that, together, we are experts in most aquatic communities listed below: fish, semiaquatic vertebrates, aquatic macroinvertebrates, zooplankton, phytoplankton, and aquatic macrophytes. To support our knowledge, we carried out a standardized search in the Web of Science Core Collection to capture the literature on the abovementioned aquatic communities in the Atlantic Forest, and then, we used the VOSviewer 1.6.9 software to analyze the main research topics based on keyword co-occurrence patterns (Van Eck and Waltman 2010). For that, all searches included the terms freshwater* or wetland* or river* or stream* and (Brazil OR Brasil) not amazon* or amazonia*. We then screened published papers and excluded studies in other biomes than the Atlantic Forest. Search for each group included the group name and variations. We generated figures in which circle sizes are proportional with the occurrence of each keyword. Co-occurrence links indicate the keywords that appear together in articles, and the thickness of the line is proportional to the number of co-occurrences. Colors correspond to the clustering performed with the VOSviewer software used to identify the four main research topics presented in the title, abstract, and keywords of the article (Fig. 10.3).

### *10.3.1 Freshwater Fish: The Most Emblematic Aquatic Vertebrate Scrutinized and Threatened*

Ichthyofaunistic surveys are the most common approach in studies to identify populations trends, new records, and potential threats and discover possible new species to science. Surveys have already been conducted in most Atlantic Forest drainage

**Fig. 10.3** (continued) (green cluster), taxonomic studies (red cluster), studies on spatial and temporal distribution (blue cluster), and effect of land use and other anthropogenic on macroinvertebrates (purple cluster)
Most common words in studies in the Atlantic Forest for the following groups: *Zooplankton* – 899 links extracted from 290 references. Zooplankton groups were not separated in clusters. Even though clusters were formed with species, in all colors, there are words that relate to studies that focus in describing community composition, abundance, richness, and variation. *Aquatic macrophytes* – 1478 links extracted from 152 references. Clusters separate studies on reporting community descriptors, such as composition and diversity (blue), from studies indicating the likely causes and consequences of macrophyte distribution (red) effects on different ecosystems (green) and role of macrophytes in structuring habitats (yellow green). *Microalgae* – 866 links extracted from 429 references. Clusters separate studies on reservoirs and lakes, which are mainly linked to biomass and cyanobacteria (green), from studies in river systems, and link Chlorophyceae with words such as richness and diversity (blue). Studies on seasonal variation and effects of abiotic variables are in one cluster (yellow green) different from studies on diatoms, dominance, and functional groups (blue)

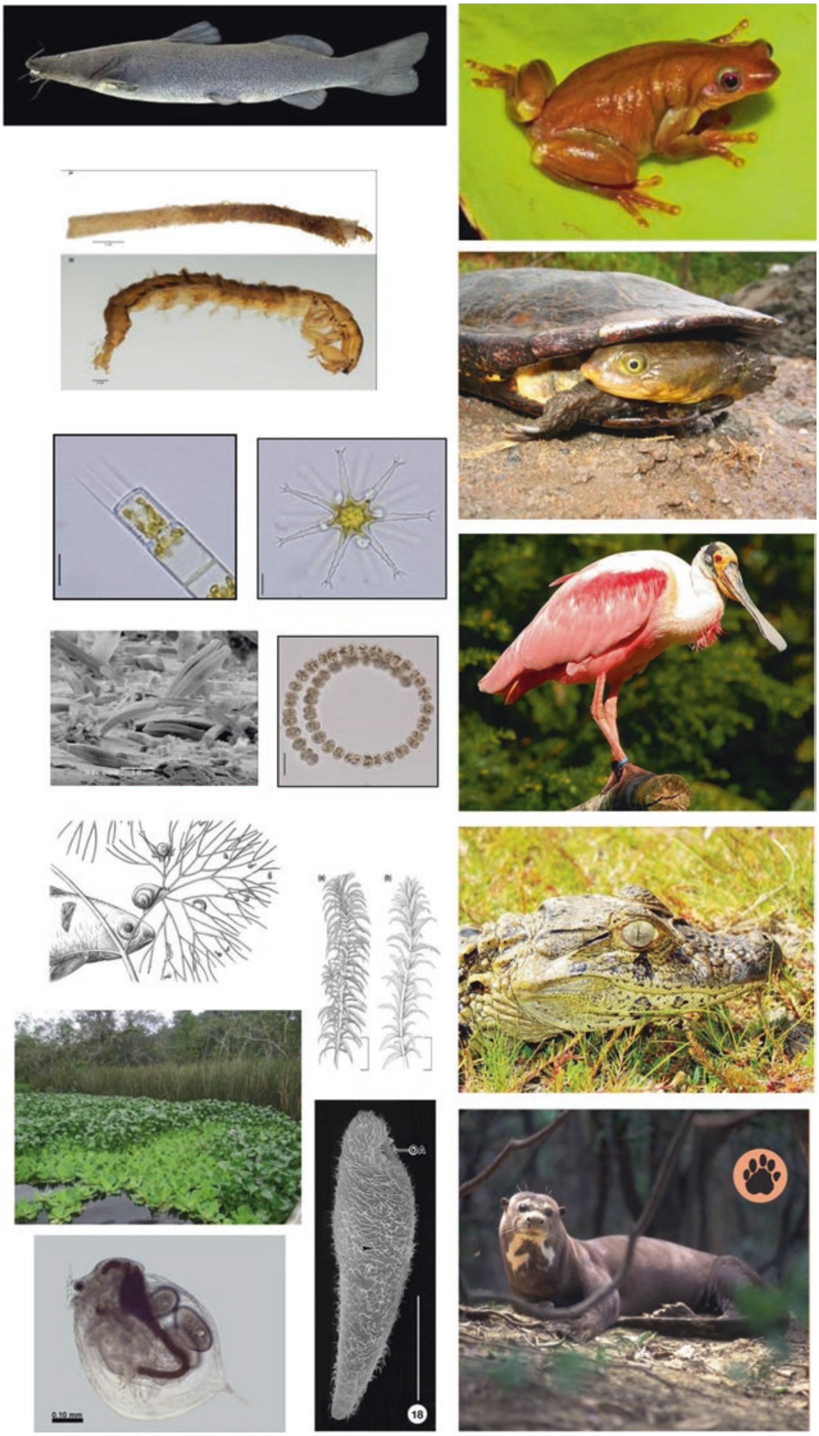

**Fig. 10.4** Some emblematic species at the Atlantic Forest. In clockwise starting from upper left: the endemic and threated large catfish of Iguaçu River basin *Steindachneridion melanodermatum* (source and photograph: Baumgartner et al. (2012)); *Hydromedusa maximiliani*, an endemic turtle to the Atlantic Forest with aquatic habits (source: <http://www.icmbio.gov.br>); *Xenohyla truncata*,

basins, from large to small ones, and large reviews include the Northeastern Atlantic Forest (Camelier and Zanatta 2014), Upper Paraná River floodplain (Graça and Pavanelli 2007; Ota et al. 2018), lower Iguaçu River basin (Baumgartner et al. 2012), and São Francisco River basin (Alves and Pompeu 2010). The most remarkable pattern is the high level of endemism in many drainage basins. Albert and Reis (2011) found 109 endemic species in the Northeastern Atlantic Forest (total of 180), 40 in the Paraiba do Sul (total of 97), 46 in the Fluminense (total of 110), 35 in the Ribeira de Iguape (total of 110), and 36 in the Southeastern Atlantic Forest (total of 97) ecoregions. The Iguaçu River is also a drainage basin with high rate of endemism (~70%; see Baumgartner et al. (2012) and example in Fig. 10.4). Such pattern was likely a result of a set of ecological aspects that are associated with climate changes along time, provided speciation. In freshwater fishes of the Brazilian coastal drainages, the speciation processes (by both vicariance and dispersion) have been associated with the past fluctuations at sea level and events of river capture, isolating or uniting watercourses (Menezes et al. 2007).

Relatedly, Price (2005) made some general considerations about viviparous species of the Atlantic Forest, and evolutionary trends related both to molecular systematics and to biogeography have been recently performed (Camelier et al. 2018). Even so, there is a lack of studies evaluating the loss of genetic diversity, as well as how population structure is affected by deforestation, particularly those with severe restrictions on population size (Menezes et al. 2007). Both levels of endemism and lack of knowledge on genetic diversity confer certain "fragility" to disturbances for the Atlantic Forest fish fauna. In watersheds mostly impacted by dam constructions, severe alterations in fish fauna are indeed well recorded in large and small drainage basins (Agostinho et al. 2016). For instance, a massive invasion event after the disruption of a major barrier for fish dispersal in the Paraná River (the "Sete Quedas" falls) caused intense community homogenization (Vitule et al. 2012). Biotic homogenization was also evidenced in several scales due to dam construction in three drainage basins of Paraná State, affecting also endemic species of Iguaçu River

**Fig. 10.4** (continued) a tree frog endemic to the Atlantic Forest (source: <http://www.icmbio.gov.br>); *Platalea ajaja*, a common species of Pelecaniformes in the Atlantic biome (source: <http://mataatlantica.macae.ufrj.br/node/39>); *Caiman latirostris*, the most representative, although not endemic, alligator of the Atlantic Forest (source: <http://www.icmbio.gov.br>); *Pteronura brasiliensis*, an endangered species of otter in the Atlantic Forest (Source: Milich and Bérnils (2004)); *Lambornella trichoglossa*, a ciliate species that supposedly occur only in Brazilian bromeliad tanks (source: Foissner (2003)); *Ceriodaphnia cornuta*, a typical cladoceran from Atlantic Forest ponds (source: L.P. Diniz); and a macrophyte bed in the Guaraguaçu River showing the coexistence and organization of species (source: A.A. Padial). Two schemes showing how macrophytes affect colonization at multiple scales (sensu Thomaz and Cunha (2010)), left, and illustrating similarities and differences between a native (*Egeria najas*, left) and non-native (*Hydrilla verticillata*, right) submerged species (Mormul et al. 2010) – right. A sequence of four planktonic microalgae photos from Brazilian reservoirs (all from T.A.V. Ludwig): an electron microscopic photo from a set of periphytic diatoms used as bioindicators, left below; *Dolichospermum* sp. (Cyanobacteria), right below; *Aulacoseira* sp. (diatom), left above; and *Staurastrum* sp. (desmid). Finally, *Plectromacronema solaris*, a new species of Trichoptera recorded in Jequitinhonha River basin (see location in Fig. 10.1) (Source: Paprocki and Moreira-Silva (2018)) basin (Daga et al. 2015). At the same time, several fish species endangered in the Atlantic Forest inhabit streams that are facing continuous destruction (Menezes et al. 2007).

Numerous studies have already been developed in rivers, streams, and lagoons aiming to understand, mainly, the relationships between fish and environmental variables in local and/or regional scale. For instance, there are studies on community organization and variability in the Upper Paraná River floodplain (e.g., Fernandes et al. 2009), coastal lagoons (Fortes et al. 2014), coastal blackwater rivers (Esteves et al. 2019), and small mountain streams in "Serra do Mar" mountain chain (Gerhard et al. 2004). More recently, metacommunity studies evidenced high beta diversity (Almeida and Cetra 2016). A proposed pattern is that fish communities are more dissimilar when closer to the headwaters than to confluence of streams (Almeida and Cetra 2016). This pattern is maintained by the degree of connectivity between streams and micro-basins, as well as the ability of each species to disperse and feed through these environments (Mazzoni et al. 2018).

It is also interesting to highlight the interactions between aquatic and terrestrial ecosystems through fish ecology. The diets of many species are composed by terrestrial insects (Abilhoa et al. 2010). In this sense, the destruction of riparian vegetation, which is widespread in the Atlantic Forest, may reduce food and organic matter intake that supplies fishes. Indeed, feeding patterns between the preserved and the impacted environments differ, and as a consequence, land degradation can cause severe taxonomic and functional changes in the Atlantic Forest (Lobón-Cerviá et al. 2016; Camilo et al. 2018). Even though terrestrial and aquatic ecosystems can be intimately related, most studies agree that Atlantic Forest conservation units are not efficient for aquatic systems, which should be protected with specific environmental management. Conservation unit is indeed a generic term, including the various types of protection areas provided by law, but in aquatic ecosystems, micro-basins could be considered as the finest-scale conservation unit (Azevedo-Santos et al. 2019).

Given the severity of threats, it is necessary to intensify the integrative efforts to understand functioning of fish community in Atlantic Forest freshwaters. Future studies more likely will address role of functional traits in ecosystem functioning, the genetic structure of population, systematics, and biogeography. Integrative studies will allow comprehension of the historical evolution, and more detailed data may be used to predictions in future climate change scenarios. Also, preservation of the Atlantic Forest needs to extrapolate the boundaries of conservation units and include local communities of artisanal fishers (e.g., the agreements of fishing and community management in the Amazon and the south region of Brazil; see Castello et al. (2009)). A recent challenge to the environmental managers is to overcome the refusal of government decision-makers to adopt the results of scientific researches on preservation decisions. An effective scientific disclosure practice that urges the society to pressure the politicians to adopt science for the preservation of the Atlantic Forest is urgent.

### *10.3.2 Other Aquatic and Semiaquatic Vertebrates: Few Studies, Many Threats, and Some Prospects*

#### Amphibians

The Atlantic Forest has great richness and endemism of anurans in Brazil (~81% of the registered species), which is the country with the highest diversity of amphibians in the world (Rossa-Feres et al. 2017). Although amphibians are one of the most diverse groups of vertebrates in tropical forest, there is a gap in the knowledge about its ecology and ecosystem services (Córtes-Gomes et al. 2015). Not surprisingly, popular studies on amphibians in freshwater are those related to the ecosystem services provided by this group. Amphibians are essentially carnivores, and their habits integrate terrestrial and aquatic environments. However, pollination and seed dispersion can also involve amphibians. For instance, the endemic species*Xenohyla truncate* (Fig. 10.4), which occurs in the south of Rio de Janeiro state, near the sandbanks, feeds on fruits of *Anthurium harrisii* and *Erythroxylum ovalifolium* and eliminates viable seeds (Córtes-Gomes et al. 2015). Other important service, mainly in urban centers, is the consumption of insect's larvae by tadpoles, acting in the control of many diseases (Whiles et al. 2006).

The high diversity of microhabitats and conditions, besides the high altitudinal variability, is likely a driver of amphibian diversity in the Atlantic Forest (Vancine et al. 2018). One example is their occupancy in phytotelmata. Only in 2015, four new species of anurans were described in the Atlantic Forest having an extremely dependent relation with Bromeliaceae species (Ferreira et al. 2015). Also, altitudinal variability promotes geographical barriers and speciation islands, which is the likely explanation for microendemic species in hill tops or rocky outcrops (Alves et al. 2006).

Known as a sedentary group, the major threat for amphibians is habitat fragmentation. It has been extensively discussed in this book that the Atlantic Forests has only a small portion of its original distribution, and almost all the remnant fragments are near human-occupied areas. More than that, changes in hydroperiod and wetland size may represent major impacts (Baber et al. 2004), as well as the spread of infectious fungus that seems to be particularly important in the Atlantic Forest (Ruggeri et al. 2015). Putting information in a scenario of already mentioned alterations in aquatic system, amphibians are a group with great extinction threat. Indeed, amphibians have more than 32% of the species in the IUCN list, and 43% of the species are in population decline (Becker et al. 2007; Stuart et al. 2004).

#### Reptiles

Among all reptiles, turtles and crocodiles have a major interaction with aquatic environments. For example, of the 13 turtle species described in the Atlantic biome, only two has terrestrial habits (see an example of an endemic turtle with aquatic habitats in Fig. 10.4). For crocodiles, two native species can be found, the *Caiman latirostris* (the most representative considering its abundance in the Atlantic Forest, Fig. 10.4) and *Paleosuchus palpebrosus*, and one introduced species: *Caiman yacare* (native from Pantanal biome). Besides that, there are several species of snakes that can be found in inland waters (Rodríguez 2005).

The most relevant papers with freshwater turtles and crocodiles are related to phylogeographic patterns (Gaffney et al. 2011). There are also some studies describing reproduction and feeding behaviors, species checklists, and reviews on conservation status (e.g., Bujes 2011; Marques et al. 2013). It is clear how little we know about the aquatic reptiles and how much baseline research (including systematics and diversity; Rodríguez 2005). At the same time, this is a critical scenario, once three of the described species of turtles and the two species of alligators founded in the Atlantic Forest are in the IUCN list of endangered species, most of them with the status vulnerable (https://www.iucn.org).

Threats concerning freshwater turtles and crocodiles include human hunting for food resources (both its meat and eggs), as well as the use of crocodile's skin by fashion industry (Alves et al. 2012). The species *Caiman latirostris*, for example, has a worrying conservation status, mainly because its distribution area is highly fragmented and most of the remnants are located near urban areas, where conflicts can occur (Alves et al. 2012).

Degradation of habitats is a likely cause for population declines, and as for fish, dam constructions may represent the main impact, given the environmental changes and fragmentation. Indeed, it has been suggested that the combination of physical and chemical changes, along with alterations in the regime flow and inundation of habitats, affects the distribution of aquatic reptiles (Rodríguez 2005). Pollution and eutrophication of rivers impact turtle and crocodile populations, primordially because sex determination is dependent on environmental features, including physical and chemical aspects (Rodríguez 2005). Also, nidification and spawning of freshwater turtles are extremely dependent on climatic seasons and on the regularity of dry and wet seasons, so one may also expect impacts due to climate changes (Souza et al. 2002).

## Birds

Several birds can be distinguished as "aquatic birds" by taking into account the ecological dependence of the group to aquatic ecosystems. Some examples are species in orders Podicipediformes, Pelecaniformes (such as *Platalea ajaja,* Fig. 10.4), Ciconiiformes, and Charadriiformes. The main interactions with birds and water bodies are for foraging, reproduction, or refuge (Vieira 2017). Most studies in aquatic birds in Brazil can be classified in five research categories, relating to the following areas: distribution, behavior, biodiversity, community structure, and other topics. Most common were studies that performed faunal surveys in Atlantic Forest sites (e.g., Just et al. 2018). Somewhat related to this topic, some studies assessed aspects of community ecology, such as seasonal dynamics (e.g., Koury 2019) or the influence of environmental variables on assemblage composition. There were also studies describing documented records of selected species in new localities or expanding the known distribution of rare and threatened birds (e.g., Camacho and Accordi 2016). Behavioral studies mainly described breeding or feeding ecology (Noguchi 2015; Gheler-Costa et al. 2018). Aquatic birds were also related to tourism potential, wetland conservation, ethnozoology, migratory reports, and broad reviews related to avian influenza.

Studies reflect a pivotal topic of interest of the ornithological community in Brazil both in natural and in anthropogenic sites. Brazilian ornithology is still related to "natural history", but it is clear that it is currently shifting toward other subjects, particularly into taxonomy and systematics (Alves et al. 2008). Future ecological research could tackle a plethora of different subjects, such as to obtain a deeper understanding of diversity patterns in time and space, including investigations on functional and phylogenetic diversities; to further understand the influences of environmental variables in aquatic bird community assembly processes, especially in human-altered landscapes; and to include an ecosystem functioning approach into these studies, considering that waterbirds are particularly important agents of wetland connectivity on a landscape and regional scales.

## Mammals

Freshwaters of Atlantic biome also have a substantial colonization of semiaquatic and aquatic mammals, which are important to nutrient cycling between the aquatic and terrestrial habitats, to integrate different water bodies and to structure trophic webs (Bakker et al. 2016). In most freshwaters of the Atlantic Forest, the main aquatic mammals considered abundant are those from Mustelidae family: *Pteronura brasiliensis* Zimmermann, 1780 ("Ariranha", Fig. 10.4), and *Lontra longicaudis* Olfers, 1818 (Brazilian otter). They are carnivorous semiaquatic species that live mostly in marginal areas between river and lakes, with some records in estuaries (Rodrigues 2013). Their diet is basically fish and crustaceans, but they can feed on come fruits acting as seed dispersers (Quadros and Monteiro-Filho 2001).

Most researches focus on conservation areas, feeding habits, impacts in fisheries, or veterinary assistance (Rheingantz et al. 2017). In the Atlantic forest, there are fewer studies than in Amazonia or Pantanal, where mammal abundance is very high. The small populations are found mainly in remnants nearby urban areas or in the main preserved area of the Atlantic Forest between the "Serra do Mar" mountain chain and the shore (Ribeiro 2006). As a consequence, studies on conservation/restoration efforts of Mustelidae are urgent: *L. longicaudis* is Vulnerable (VU) in IUCN red list, mainly due to habitat degradation (see Ekkos (2017)). Conservation status of *P. brasiliensis* is even more concerning, Critically Endangered according to IUCN red list, and efforts are being rather inefficient (ICMBIO 2015). Main threats for this species are conflicts with fishermen, changes in the river flow, pollution, and bioaccumulation (Rheingantz et al. 2017). Another important aquatic mammal inhabiting most Atlantic Forest freshwaters is the "capybara" (*Hydrochoerus* *hydrochaeris*). Most studies investigate their responses to anthropogenic pressures, given the high ability of this species to live near or inside urban and rural zones. They are usually (but not legally) hunted for meat consumption, pointed out as a dangerous activity given heavy metal contamination in its body can be high (Machado et al. 2018).

Sea mammals can invade Atlantic Forest freshwaters mainly for food consumption, like the cetaceans or sirenians. One example of critically endangered is the manatee *Trichechus manatus manatus*, an herbivorous aquatic mammal of the northeast Brazil that can temporally invade freshwaters in the coast (Balensiefer et al. 2017). Dolphins can also visit freshwaters and lagoons in many estuaries (Bakker et al. 2016). Species mainly recorded in Atlantic Forest coastal areas are *Sotalia guianensis*, *Pontoporia blainvillei* (the most threatened dolphin according to IUCN red list), *Tursiops truncatus*, and *Delphinus capensis*.

The most concern for future studies on aquatic mammals in the Atlantic Forest should be the conservation status of species. Like other aquatic vertebrates, there is still a need for basic ecological studies before any conservation strategy (Pacini and Harper 2008). Surely, preserving the pristine, and restoring impacted, is a general recommendation for aquatic, riparian, and coastal ecosystems. In the case of mammals, given their large distribution and response to anthropogenic alterations, they can serve as "flagship species" in conservation efforts.

### *10.3.3 Aquatic Macroinvertebrates: Astonishing and Threaten Diversity, Still Underestimated*

Freshwater macroinvertebrates have a central role in many functions, processes, and services in freshwaters. Among all freshwater animals, insects are the dominant group (60.4% of total biodiversity), followed by vertebrates (14.5%), crustaceans (10%), arachnids (5%), mollusks (4%), and annelids (1.4%) (Balian et al. 2008). In tropical forests, besides streams, rivers, and ponds, macroinvertebrates also occur in any system that accumulates water, such as phytotelmata (Kitching 2000). Despite the recent advances on synthetizing the knowledge on freshwater invertebrate diversity in South America, aquatic macroinvertebrate diversity from Brazilian tropical forests remains poorly known.

Freshwater macroinvertebrates from the Atlantic Forest have been the focus of research since the eighteenth and nineteenth centuries: Fritz Müller described an amazing larvae of Psychodidae (Diptera) from streams in Santa Catarina, just a few years before Charles Darwin published *On the Origin of Species* (Müller 1895). In the last few years, we witness a great expansion of research on macroinvertebrates in the Atlantic Forest, with new experts, research groups in different institutions, and strong international collaboration. Even so, big challenges still remain in terms of taxonomic studies, and it is not surprising that we have enormous gaps of knowledge in terms of distribution, abundance, evolutionary patterns, abiotic tolerances of species, species traits, and biotic interactions (see Hortal et al. (2015)).

Over the past decades, the topics covered by the studies on macroinvertebrates have been diversified. Studies that document the relationships of anthropogenic drivers of environmental change with macroinvertebrate started in the 1970s in the Atlantic Forest, but it is only in the 1990s that these studies and the use of macroinvertebrates as bioindicators of freshwater condition prospered. Efficient ways to measure and monitor freshwater ecosystems using macroinvertebrates as bioindicators exist in different regions of the Atlantic Forest, supporting ecosystem management and conservation (Hepp et al. 2017). Even so, understanding the large-scale drivers of variation in macroinvertebrates and untangling relationships between biological indicators and large-scale predictor variables are still a challenge. Studies on the effects of climate change or large-scale patterns in Atlantic Forest macroinvertebrates started to appear in the literature in recent years (e.g., Silva et al. 2018), so this topic is a frontier almost unexplored. It is also important to highlight that most studies on macroinvertebrates in the region were based on short time series (less than 2 years), so long-term studies on macroinvertebrate dynamics are still a dream.

The most effective conservation strategy for conserving biodiversity – including freshwater macroinvertebrates – is the creation and maintenance of protected areas for aquatic systems (Azevedo-Santos et al. 2019). To date, we have little information if the net of protected areas in the Atlantic Forest is effective to conserve freshwater macroinvertebrates. We have a clear demand for mapping areas of biological relevance as a first step toward an effective conservation planning that includes freshwater macroinvertebrates as a target group. Ongoing declines of species indicate that conservation strategies must also include restoration and sustainable solutions addressing the social, cultural, economic, and ecological interdependencies.

### *10.3.4 Zooplankton: Panorama of Studies in the "Bridge Community" Between Producers and Secondary Consumers*

Zooplankton is a biological group that represents the main link between primary producers (mainly microalgae) and consumers, being central in energy transfer and nutrient cycling (Melão et al. 2005). The short life cycle and high reproductive ability made them as a proxy group to environmental changes, as well as surrogates of trophic level and degradation and conservation status of freshwaters. Considering the search described in Fig. 10.3, most studies on zooplankton in Atlantic Forest freshwaters were carried out in South and Southeast Brazil. Such pattern coincided with main centers of aquatic researches regarding the zooplankton experts (Silva and Perbiche-Neves 2017).

Studies range from records of new species and population status to inventories of up to 541 taxa in ecosystems monitored in long-term ecological projects. Most studies evaluate zooplankton in lakes, reservoirs, and floodplains. This is not a surprise given the numerous reservoirs in Brazil (Tundisi and Matsumura-Tundisi 2003). In such ecosystems, a common goal was to evaluate the impact of dam constructions in structure and dynamics of zooplankton assemblages. On the other hand, some ecosystems seem to have high number of species, but studies on community variation are poor. Indeed, there is a big gap considering studies on streams and small rivers, which are abundant freshwater ecosystems in the Atlantic Forest. Another example is phytotelmata, where community dynamics may be highly dependent on physical and chemical properties, promoting endemism (Lopez et al. 2009). Indeed, new species have been described in bromeliad tanks (e.g., Foissner 2003). Given the high abundance and diversity of bromeliads in the Atlantic Forest, studies on zooplankton in phytotelmata should be encouraged.

Zooplankton inventories were the main goals of studies, as well as studies describing community features, such as abundance and diversity indexes. Recent efforts have been also devoted to understand trophic interactions between zooplankton and phytoplankton, causes for community variation among space and time and the role of zooplankton secondary production in ecosystems. There are still few studies on functional patterns in communities, metacommunity structuring, interactions with fish, genetic structure and diversity, dormant stages, and exotic species.

A better understanding of genetic diversity and taxonomy can also help one to identify biological invasions, which probably occur but are apparently neglected in zooplankton (Simões et al. 2009). Also, a better development of functional features for zooplankton species is needed to better link the role of zooplankton to ecosystem functioning such as production, biomass, and nutrient cycling. The most used functional classification only considers the main biological groups of zooplankton – ciliates, amoebae, rotifers, cladocerans, and copepods. However, some functional traits indicate specific responses to how species interact with others and with the environment, such as the variety of buccal structures in rotifers related to their feeding habits. We believe that such gaps may be circumvented with improvement of experts.

The increase in studies on interactive webs is also a future prospect of zooplankton studies. Indeed, reconciling aquatic trophic webs with environmental heterogeneity can make better predictions of freshwater ecosystem functioning, as well as to understand the consequences of anthropogenic disturbances. Simões et al. (2015), for instance, indicated that the negative effect of reservoir constructions in zooplankton is likely the cause for changes in ecosystem functioning by disrupting aquatic trophic webs. Challenges for freshwater conservation will increase as anthropogenic impacts accelerate in the Atlantic Forest.

### *10.3.5 Aquatic Macrophytes: Massive Primary Production, Structuring Role, and Biological Invasions*

In terms of biomass, aquatic macrophytes are the main primary producers in most aquatic freshwaters (Cook 1990). It is not a surprise that first studies in the Atlantic Forest evaluated the role of macrophytes in ecosystem functioning (e.g., Esteves and Camargo 1986), which is indeed reflected in the word analyses presented here in term such as “biomass”, “abundance”, “area”, “effect”, “influence”, and “growth” (Fig. 10.3). Along with studies reporting and describing primary production, studies in decomposition were pioneer (e.g., Camargo et al. 1983). Therefore, it is clear that macrophytes have a key role in nutrient cycling and energy transfer (Thomaz and Bini 2003), and in most freshwater ecosystems of Atlantic Forest, this is the case.

It has been long suggested that most biomass produced by macrophytes does not support trophic webs due to herbivory. Instead, macrophytes are the main basis of detritivory. Although detritivory is indeed high, macrophytes can also be direct food source for many invertebrates (Newman 1991). However, role of macrophytes in ecosystem functioning is probably not directly related to its biomass consumption, but to their structuring role in freshwaters. Serving as shelter for preys, feeding areas for predators, and substrate for attached algae, the multiple-scale effects of macrophytes on ecosystem functioning are overwhelming and explain high freshwater diversity (Thomaz and Cunha 2010; see also Fig. 10.4).

Colonization and consequent effects of macrophytes were reported mainly in floodplains, associated lakes, artificial reservoirs, coastal lagoons, and swamps (Thomaz and Bini 2003). Due to abiotic filters, macrophyte colonization is indeed diminished in small streams with poor solar light and in river courses with poor margin development. Indeed, main abiotic conditions that promote macrophyte colonization are the low depth and slope, high solar light, low fetch, nutrient input, and developed soil and margins (Lacoul and Freedman 2006). In this sense, it is also worth mentioning that the likely causes for macrophyte community organization, reporting mainly the correlation between macrophytes and abiotic features, were the focus of numerous publications on this community in Atlantic Forest freshwaters, including artificial reservoirs. More recently, likely causes for macrophyte communities were investigated considering a metacommunity approach, disentangling mechanisms associated with environmental filtering to mechanisms associated with dispersal, for instance (Trindade et al. 2018).

In this sense, human alterations in freshwaters have direct impacts in development of macrophyte beds in both natural and artificial freshwaters. Indeed, massive development of macrophytes is a common environmental problem in Atlantic Forest freshwaters such as reservoirs (Thomaz et al. 1999) and also in floodplains with human alterations. This is the case of the Upper Paraná River floodplain: due to the increase in transparency and flood control caused by upstream and downstream reservoirs, intense colonization of submerged macrophytes occurred, including the exotic and highly invasive *Hydrilla verticillata* (Sousa 2011).

As a consequence, the last pattern listed here considering studies on macrophytes in the Atlantic Forest is the understanding of invasion processes. Invasion of macrophytes has been intensively studied considering their effects on native biota (Mormul et al. 2010; Michelan et al. 2010), the likely causes for invasibility (Thomaz et al. 2015), including several experiments testing which factors can explain colonization of exotic species in Atlantic Forest freshwaters (Ribas et al. 2017) and indicating the negative effect of invasive macrophytes causing other invasions and ecosystem changes (Michelan et al. 2014). Macrophyte massive development, causes and consequences of invasion processes, and community organization in the Anthropocene seem to be the main research prospectus for studies on this important aquatic community in Atlantic Forest freshwaters.

### *10.3.6 Microalgae: The Most Known Primary Producers and Indicators of Water Quality*

Microalgae are fundamental primary producer in several aquatic ecosystems, being the principal source of organic carbon deposition and playing an important role as the base of many aquatic food webs and nutrient cycling. They comprise a polyphyletic group of organisms, including several algae lineages and Cyanobacteria that are phylogenetically distantly related. Microalgae are largely photoautotrophic organisms and characterized by a vast range of different species and diversity of form, function, and indicator (Reynolds 2006; Stevenson 1996). In general, two strata of microalgae can be distinguished: phytoplankton comprise microalgae that live suspended in the water column "liable to passive movement by wind and current" (Reynolds 2006) and phycoperiphyton (periphyton hereafter) that "live on or in association with substrata" (Stevenson 1996). Studies in Atlantic Forest freshwaters are abundant given the proximity to the main research groups. As a consequence, among the *c*. 4747 described species of microalgae in Brazil, Atlantic Forest freshwaters has *c*. 1545 species, being the most diverse biome (Menezes et al. 2015). Even so, it is clear that this is an underestimation, given that new microalgae taxa have been continuously described, mainly in diatoms and cyanobacteria (e.g., Tremarin et al. 2013). Regarding the ecosystems, the most common studies were carried out in reservoirs, for similar reasons explained by the other biological groups described in this chapter. However, it is also worth mentioning the high number of studies in streams and river mainly considering the periphyton (Bicudo et al. 1995).

Studies in microalgae became popular mainly due to the fact that they are frequently used as bioindicators of water quality and respond quickly to environmental alterations. In this case, diatoms are the microalgae group mostly used. Indeed, several researches have been devoted to understand the eutrophication process in aquatic ecosystems, using microalgae as the main focus (mainly diatoms; see Lobo et al. (2019)).

Diatoms are also used to reconstruct ecological history of ecosystems, in the field known as "paleolimnology". Due to their resistant siliceous frustule, diatoms can be recorded in sediment reconstructing thousands of years in community composition (e.g., Ruwer et al. 2018), including evidences that anthropogenic disturbances due to reservoir dams are key to understand eutrophication and biotic homogenization (Wengrat et al. 2018).

Relatedly, numerous studies have been devoted to understand microalgae responses to environmental alterations in reservoirs, including metacommunity approach (Wojciechowski et al. 2017), eutrophication, and cyanobacterial blooms that have consequences to water quality for human consumption (Bittencourt-Oliveira et al. 2014). Future prospects on microalgae organization include responses of the functional traits of species to ecosystem variation (Bovo-Scomparin et al. 2013) and what can help the understanding of how microalgae affects and is affected by ecosystem functioning. As for other groups, particular ecosystems seem to be associated with particular species composition. This is the case of phytotelmata, in which new and peculiar species have been continuously been described (Ramos et al. 2018). Other gaps include the understanding of phylogeny and genetic variation of microalgae, a known challenge that researchers must address.

## 10.4 Prospects for Freshwaters in the Atlantic Forest

It became clear that anthropogenic disturbances are pervasive in Atlantic Forest freshwaters. The main concerns are the extinctions of endemic fauna and flora and consequent biotic homogenization. It seems that there is no magic bullet for preserving the astonishing freshwater biodiversity of the Atlantic Forest. The environmental agenda of Brazil needs to urgently move toward Aichi Biodiversity Targets (https://www.cbd.int/sp/targets/), for the sake of safety of ecosystem services and prevention of irreversible damages related to extinctions of endemic species. Restoration and conservation of freshwaters may benefit from the classification proposed here and from the information of the key aquatic communities. Efforts must be toward reducing fragmentation and pollution and supporting not only maintenance of pristine but also restoring degraded areas. Initiatives on financial support for ecosystem services are urgent and necessary. In this sense, we reinforce that any plan to preserve freshwaters may consider the intervention on the entire micro-basin (sensu Azevedo-Santos et al. 2019), ensuring the full restoration/protection/conservation of ecosystem.

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## References

Abilhoa V, Vitule JRS, Bornatowski H (2010) Feeding ecology of *Rivulus luelingi* (Aplocheiloidei: Rivulidae) in a Coastal Atlantic Rainforest stream, southern Brazil. Neotrop Ichthyol 8(4):813–818

Agostinho AA, Thomaz SM, Gomes LC (2005) Conservation of the biodiversity of Brazil inland waters. Conserv Biol 19:646–652

Agostinho AA, Gomes LC, Santos NCL, Ortega JC, Pelicice FM (2016) Fish assemblages in Neotropical reservoirs: colonization patterns, impacts and management. Fish Res 173(1):26–36

Albert JS, Reis R (eds) (2011) Historical biogeography of Neotropical freshwater fishes. Univ of California Press, London

Almeida RSA, Cetra M (2016) Longitudinal gradient effects on the stream fish metacommunity. Nat Conservação 14:112–119

Alves CBM, Pompeu OS (2010) Peixes do Rio das Velhas: passado e presente. Argvmentvm, Belo Horizonte

Alves AC, Ribeiro LF, Haddad CF et al (2006) Two new species of Brachycephalus (Anura: Brachycephalidae) from the Atlantic forest in Paraná state, southern Brazil. Herpetologica 62(2):221–233

Alves MAS, Silva JMC, Costa ES (2008) Brazilian ornithology: history and current trends. Ornitol Neotrop 19(Supplement):391–399

Alves RRN, Vieira KS, Santana GG et al (2012) A review on human attitudes towards reptiles in Brazil. Environ Monit Assess 184:6877–6901

Arvor D, Daher FR, Briand D et al (2018) Monitoring thirty years of small water reservoirs proliferation in the southern Brazilian Amazon with Landsat time series. ISPRS J Photogramm Remote Sens 145:225–237

Azevedo-Santos VM, Frederico RG, Fagundes CK et al (2019) Protected areas: a focus on Brazilian freshwater biodiversity. Divers Distrib 25(3):442–448

Baber MJ, Fleishman E, Babbitt KJ et al (2004) The relationship between wetland hydroperiod and nestedness patterns in assemblages of larval amphibians and predatory macroinvertebrates. Oikos 107:16–27

Bakker ES, Pagès JF, Arthur R et al (2016) Assessing the role of large herbivores in the structuring and functioning of freshwater and marine angiosperm ecosystems. Ecography 39(2):162–179

Balensiefer DC, Attademo FLN, Sousa GP et al (2017) Three Decades of Antillean Manatee (*Trichechus manatus manatus*) Stranding Along the Brazilian Coast. Trop Conserv Scien 10:1–9

Balian EV, Segers H, Lévéque C et al (2008) The freshwater animal diversity assessment: an overview of the results. Hydrobiologia 595:627–637

Barbosa JM, Soares EC (2009) Perfil da ictiofauna da bacia do São Francisco: estudo preliminar. Rev Bras Engenh Pesca 4(1):155–172

Baumgartner G, Pavanelli CS, Baumgartner D et al (2012) Peixes do baixo rio Iguaçu. EDUEM, Maringá

Becker CG, Fonseca CR, Haddad CFB et al (2007) Habitat split and the global decline of amphibians. Science 318(5857):1775–1777

Bicudo DC, Necchi OJR, Chamixaes CB (1995) Periphyton studies in Brazil: present status and perspectives. In: Tundisi JB, Bicudo CEM, Matsumura-Tundisi T (eds) Limnology in Brazil. Academia Brasileira de Ciências e Sociedade Brasileira de Limnologia, Rio de Janeiro, pp 37–42

Bittencourt-Oliveira MC, Piccin-Santos V, Moura AN et al (2014) Cyanobacteria, microcystins and cylindrospermopsin in public drinking supply reservoirs of Brazil. An Acad Bras Ciênc 86:297–310

Bovo-Scomparin VM, Train S, Rodrigues LC (2013) Influence of reservoirs on phytoplankton dispersion and functional traits: a case study in the Upper Paraná River, Brazil. Hydrobiologia 702:115–127

Bujes CS (2011) Os Testudines continentais do Rio Grande do Sul, Brasil: taxonomia, história natural e conservação. Iheringia Sér Zool 100:413–424

Camacho I, Accordi M (2016) Confirmação da sora, Porzana carolina, em território brasileiro e contribuições para a conservação das áreas úmidas da Área de Proteção Ambiental de Maricá (RJ) para espécies migratórias neárticas. Atualidades Ornitológicas 191:60–66

Camargo AFM, Ishii IH, Esteves FA (1983) Liberação de compostos orgânicos e inorgânicos para a coluna d'água durante o processo de decomposição de duas espécies de macrófitas aquáticas tropicais. Anais do Seminário Regional de Ecologia 3(1983):87–99

Camelier P, Zanatta AM (2014) Biogeography of freshwater fishes from the Northeastern Mata Atlântica freshwater ecoregion: distribution, endemism, and area relationships. Neotrop Ichthyol 12(4):683–698

Camelier P, Menezes NA, Costa-Silva G et al (2018) Molecular and morphological data of the freshwater fish Glandulocauda melanopleura (Characiformes: Characidae) provide evidences of river captures and local differentiation in the Brazilian Atlantic Forest. PLoS One 13(3):1–25

Camilo GS, Terra BF, Araújo FG (2018) Using the relationship between taxonomic and functional diversity to assess functional redundancy in streams of an altered tropical watershed. Environ Biol Fish 101:1395–1405

Castello L, Viana JP, Watkins G et al (2009) Lessons from integrating fishers of Arapaima in small-scale fisheries management at the Mamirauá Reserve, Amazon. Environ Manag 43(2):197–209

Cavalcante G, Miranda LBD, Medeiros PRP (2017) Circulation and salt balance in the São Francisco river Estuary (NE/Brazil). RBRH 22

Chapin FS, Starfield AM (1997) Time lags and novel ecosystems in response to transient climatic change in arctic Alaska. Clim Chang 35(4):449–461

Cionek VM, Alves GHZ, Tófoli RM et al (2019) Brazil in the mud again: lessons not learned from Mariana dam collapse. Biodivers Conserv 28(7):1935–1938

Collen B, Whitton F, Dyer EE et al (2014) Global patterns of freshwater species diversity, threat and endemism. Glob Ecol Biogeogr 23(1):40–51

Cook CDK (1990) Aquatic plant book. SPB Academic, The Hague, 228 p

Cortés-Gomez AM, Ruiz-Agudelo CA, Valencia-Aguilar A et al (2015) Ecological functions of neotropical amphibians and reptiles: a review. Univers Scient 20(2):229–245

Couto TB, Olden JD (2018) Global proliferation of small hydropower plants–science and policy. Front Ecol Environ 16(2):91–100

Daga VS, Skóra F, Padial AA et al (2015) Homogenization dynamics of the fish assemblages in Neotropical reservoirs: comparing the roles of introduced species and their vectors. Hydrobiologia 746(1):327–347

Daga VS, Azevedo-Santos VM, Pelicice FM et al (2020) Water diversion in Brazil threatens biodiversity. Ambio 49(1):165–172

Dodds WK, Perkin JS, Gerken JE (2013) Human impact on freshwater ecosystem services: a global perspective. Environ Sci Technol 47(16):9061–9068

Dudgeon D, Arthington AH, Gessner MO et al (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. Biol Rev 81:163–182

Ekkos (2017) Activities report: otter project. http://www.ekkobrasil.org.br. Acessed 08 May 2019

Esteves FA, Camargo AFM (1986) Sobre o papel das macrófitas aquáticas na estocagem e ciclagem de nutrientes. Acta Limnol Bras 1:273–298

Esteves KE, Silva MHC, Nanini-Costa MH et al (2019) Organization of fish assemblages in Blackwater Atlantic Forest streams. Neotrop Ichthyol 17(1):1–12

Fernandes R, Agostinho AA, Ferreira EA et al (2009) Effects of the hydrological regime on the ichthyofauna of riverine environments of the Upper Paraná River floodplain. Braz J Biol 69(2):669–680

Ferreira RB, Faivovich J, Beard KH et al (2015) The First Bromeligenous Species of Dendropsophus (Anura: Hylidae) from Brazil's Atlantic Forest. PLoS One 10(12):1–21

Foissner W (2003) Morphology and ontogenesis of Lambornella trichoglossa nov. spec., a new tetrahymenid ciliate (Protozoa, Ciliophora) from Brasilian tank bromeliads (Bromeliaceae). Eur J Protistol 39(1):63–82

Fortes WL, Almeida-Silva PH, Prestrelo L et al (2014) Patterns of fish and crustacean community structure in a coastal lagoon system, Rio de Janeiro, Brazil. Mar Biol Res 10(2):111–122

Freitas L, Salino A, Neto LM et al (2016) A comprehensive checklist of vascular epiphytes of the Atlantic Forest reveals outstanding endemic rates. PhytoKeys 58:65

Gaffney AES, Meylan PA, Wood RC et al (2011) Evolution of the side-necked turtles: the family Podocnemididae. Bul Am Mus Nat Hist 350:1–237

Gerhard P, Moraes R, Molander S (2004) Stream fish communities and their associations to habitat variables in a rainforest reserve in southeastern Brazil. Environ Biol Fish 71:321–340

Gheler-Costa C, Comin FH, Gilli LC et al (2018) Foraging behavior of Brazilian cormorant, Nannopterum brasilianus (Suliformes: Phalacrocoracidae). Fortschr Zool 35:1–4

Gleick PH (1998) The human right to water. Water Policy 1(5):487–503

Graça WJ, Pavanelli CS (2007) Peixes da planície de inundação do alto rio Paraná e áreas adjacentes. EDUEM, Maringá, 241p

Hatje V, Pedreira RM, de Rezende CE et al (2017) The environmental impacts of one of the largest tailing dam failures worldwide. Sci Rep 7:10706

Hepp LU, Pratas JA, Graça MA (2017) Arsenic in stream waters is bioaccumulated but neither biomagnified through food webs nor biodispersed to land. Ecotoxicol Environ Saf 139:132–138

Hortal J, de Bello F, Diniz-Filho JAF et al (2015) Seven shortfalls that beset large-scale knowledge of biodiversity. Annu Rev Ecol Evol Syst 46:523–549

ICMBIO (2015) Monitoria Final: Plano De Ação Nacional Para Conservação Da Ariranha. Planos De Ação Nacionais De Conservação De Espécies Ameaçadas De Extinção – PAN. Available in: http://www.icmbio.gov.br/portal/images/stories/docs-plano-de-acao/pan-ariranha/painel_gestao_final_ariranha.pdf. Acessed in May 08, 2019

Junk WJ, Piedade MTF, Lourival R et al (2014) Brazilian wetlands: their definition, delineation, and classification for research, sustainable management, and protection. Aquat Conserv Mar Freshw Ecosyst 24(1):5–22

Just JPG, Rosoni JRR, Romagna RS et al (2018) Bird diversity and conservation in the southern coast of Santa Catarina state, Brazil. Pap Avul Zool 58

Kitching RL (2000) Food webs and container habitats: the natural history and ecology of phytotelmata. Cambridge Univ. Press, Cambridge, USA, 431pp

Koury HA (2019) Dinâmica, sazonalidade e estrutura da comunidade de aves em uma área úmida. Dissertação de Mestrado. Universidade Estadual Paulista “Júlio de Mesquita Filho”

Lacoul P, Freedman B (2006) Environmental influences on aquatic plants in freshwater ecosystems. Environ Rev 14:89–136

Leal CG, Câmara IG (eds) (2003) The Atlantic Forest of South America: biodiversity status, threats, and outlook. Island Press, Washington DC

Lima-Junior DP, Magalhães ALB, Pelicice FM et al (2018) Aquaculture expansion in Brazilian freshwaters against the Aichi biodiversity targets. Ambio 47(4):427–440

Lobo EA, Freitas NW, Salinas VH (2019) Diatoms as bioindicators: ecological aspects of the algae response to eutrophication in Latin America. Mex J Biotechnol 4:1–24

Lobón-Cerviá J, Mazzoni R, Rezende CF (2016) Effects of riparian forest removal on the trophic dynamics of a Neotropical stream fish assemblage. J Fish Biol 89:50–64

Lopez LCS, Alves RRN, Rios RI (2009) Micro-environmental factors and the endemism of bromeliad aquatic fauna. Hydrobiologia 625:151–156

Machado KS, Froehner S, Rizzi J et al (2018) Tracking capybara (Hydrochoerus hydrochaeris) feces contribution method in aquatic environments using sterols. Environ Toxicol Chem 37:353–361

Marques TS, Lara NRF, Bassetti LAB et al (2013) Population structure of Mesoclemmys vanderhaegei (Testudines, Chelidae) in a silvicultural system in southeastern Brazil. Herpetol Notes 6:179–182

Mazzoni R, Pinto MP, Iglesias-Rios R et al (2018) Fish movement in an Atlantic Forest stream. Neotrop Ichthyol 16:e170065

Melão MGG, Rocha O, Roche KF (2005) Produtividade, biomassa, flutuações populacionais e interações biológicas da comunidade planctônica e suas implicações na transferência de energia na cadeia alimentar de um reservatório raso e oligotrófico. In: Roche KF, Rocha O (eds) Ecologia Trófica de Peixes com ênfase na planctivoria em ambientes lênticos de água doce no Brasil. Rima, São Carlos, pp 25–80

Menezes NA, Weitzman SH, Oyakawa OT et al (2007) Peixes de água doce da Mata Atlântica: lista preliminar das espécies e comentários sobre conservação de peixes de água doce neotropicais. Museu de Zoologia, Universidade de São Paulo, São Paulo

Menezes M, Bicudo CEM, Moura CWN et al (2015) Update of the Brazilian floristic list of algae and cyanobacteria. Rodriguésia 66:1047–1062

Michelan TS, Thomaz SM, Mormul RP et al (2010) Effects of an exotic invasive macrophyte (tropical signalgrass) on native plant community composition, species richness and functional diversity. Freshw Biol 55:1315–1326

Michelan TS, Silveira MJ, Petsch DK et al (2014) The invasive aquatic macrophyte Hydrilla verticillata facilitates the establishment of the invasive mussel Limnoperna fortunei in Neotropical reservoirs. J Limnol 73(3):598–602

Mormul RP, Thomaz SM, Higuti J et al (2010) Ostracod (Crustacea) colonization of a native and a non-native macrophyte species of Hydrocharitaceae in the Upper Paraná floodplain (Brazil): an experimental evaluation. Hydrobiologia 644:185–193

Müller F (1895) XVIII. Contribution towards the history of a new form of larvæ of Psychodidæ (Diptera), from Brazil. Trans Royal Entomol Soc London 43:479–482

Newman RM (1991) Herbivory and detritivory on freshwater macrophytes by invertebrates: a review. J North Amer Benthol Soc 10:89–114

Noguchi RG (2015) Reprodução de ardeídeos em uma área periurbana no município de Cananéia, sul do estado de São Paulo. Dissertação de Mestrado. Universidade Federal do Paraná, 58p

Ota RR, Deprá GC, Graça WJ et al (2018) Peixes da planície de inundação do alto rio Paraná e áreas adjacentes: revised. Neotrop Ichthyol 16:e170094

Pacini N, Harper DM (2008) Aquatic, semi-aquatic and riparian vertebrates. In tropical stream ecology. In: Dudgeon D (ed) Tropical stream ecology. Academic Press, pp 147–197

Price DJ (2005) The genetic considerations in the conservation of freshwater viviparous fish. In: Uribe MC, Grier JH (eds) Viviparous fishes. New life international, Homestead

Quadros J, Monteiro-Filho EL (2001) Diet of the neotropical otter, Lontra longicaudis, in an Atlantic forest area, Santa Catarina State, southern Brazil. Stud Neotrop Fauna Environ 36(1):15–21

Rahel FJ (2002) Homogenization of freshwater faunas. Annu Rev Ecol Syst 33(1):291–315

Ramos GJP, Bicudo CEM, Moura CWN (2018) Some new, rare and interesting desmids from bromeliad phytotelmata in Brazil. Phytotaxa 346:059–077

Reynolds C (2006) Ecology of phytoplankton. Cambridge University Press, Cambridge

Rheingantz ML, Santiago-Plata VM, Trinca CS (2017) The Neotropical otter Lontra longicaudis: a comprehensive update on the current knowledge and conservation status of this semiaquatic carnivore. Mammal Rev 47:291–305

Ribas LGDS, Cunha ER, Vitule JRS et al (2017) Biotic resistance by snails and fish to an exotic invasive aquatic plant. Freshw Biol 62:1266–1275

Ribeiro AC (2006) Tectonic history and the biogeography of the freshwater fishes from the coastal drainages of eastern Brazil: an example of faunal evolution associated with a divergent continental margin. Neotrop Ichthyol 4:225–246

Ribeiro MC, Martensen AC, Metzger JP et al (2011) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In: Zachos F, Habel J (eds) Biodiversity hotspots. Springer, Berlin, Heidelberg

Richardson BA (1999) The bromeliad microcosm and the assessment of faunal diversity in a Neotropical forest. Biotropica 31:321–336

Rodrigues LA (2013) Avaliação do risco de extinção da lontra neotropical Lontra longicaudis (Olfers, 1818) no Brasil. Biodiversidade Bras 3:216–227

Rodríguez M (2005) The conservation of Brazilian reptiles: challenges for a Megadiverse Country. Conserv Biol 19:659–664

Rossa-Feres DDC, Garey MV, Caramaschi U et al (2017) Anfíbios da Mata Atlântica: lista de espécies, histórico dos estudos, biologia e conservação. Revisões em Zoologia, Mata Atlântica, pp 237–314

Ruggeri J, Longo AV, Gaiarsa MP et al (2015) Seasonal variation in population abundance and Chytrid infection in stream-dwelling frogs of the Brazilian Atlantic Forest. PLoS One 10(7):e0130554

Ruwer DT, Bernardes MC, Rodrigues L (2018) Diatom responses to environmental changes in the Upper Paraná River floodplain (Brazil) during the last ~1000 years. J Paleolimnol 60:543–551

Silva WM, Perbiche-Neves G (2017) Trends in freshwater microcrustaceans studies in Brazil between 1990 and 2014. Braz J Biol 77:527–534

Silva DP, Dias AC, Lecci LS et al (2018) Potential effects of future climate changes on Brazilian cool-adapted stoneflies (Insecta: Plecoptera). Neotrop Entomol 48:57–70

Simões NR, Robertson BA, Lansac-Tôha FA et al (2009) Exotic species of zooplankton in the Upper Paraná River floodplain, Daphnia lumholtzi Sars, 1885 (Crustacea: Branchiopoda). Braz J Biol 69:551–558

Simões NR, Nunes AH, Dias JD et al (2015) Impact of reservoirs on zooplankton diversity and implications for the conservation of natural aquatic environments. Hydrobiologia 758:3–17

Sousa WTZ (2011) Hydrilla verticillata (Hydrocharitaceae), a recent invader threatening Brazil's freshwater environments: a review of the extent of the problem. Hydrobiologia 669:1

Souza FL, Cunha AF, Oliveira MA et al (2002) Estimating dispersal and gene flow in the neotropical freshwater turtle Hydromedusa maximiliani (Chelidae) by combining ecological and genetic methods. Genet Mol Biol 25:151–155

Stevenson RJ (1996) An introduction to algal ecology in freshwater benthic habitats. In: Stevenson RJ, Bothwell ML, Lowe RL (eds) Algal ecology freshwater benthic ecosystems. Academic Press, New York

Stuart SN, Chanson JS, Cox NA et al (2004) Status and trends of amphibian declines and extinctions worldwide. Science 306:1783–1786

Thomaz SM, Bini LM (2003) Ecologia de Macrófitas Aquáticas. EDUEM, Maringá

Thomaz SM, Cunha ERD (2010) The role of macrophytes in habitat structuring in aquatic ecosystems: methods of measurement, causes and consequences on animal assemblages' composition and biodiversity. Acta Limnol Bras 22:218–236

Thomaz SM, Bini LM, Souza MCD et al (1999) Aquatic macrophytes of Itaipu Reservoir, Brazil: survey of species and ecological considerations. Braz Arch Biol Technol 42:1

Thomaz SM, Mormul RP, Michelan TS (2015) Propagule pressure, invasibility of freshwater ecosystems by macrophytes and their ecological impacts: a review of tropical freshwater ecosystems. Hydrobiologia 746:39–59

Tremarin PI, Freire EG, Ludwig TA (2013) Ultrastructure and distribution of Urosolenia obesa sp. nov. (Rhizosoleniaceae, Diatomeae) in reservoirs from southern Brazil. Phytotaxa 125:1–9

Trindade CRT, Landeiro VL, Schneck F (2018) Macrophyte functional groups elucidate the relative role of environmental and spatial factors on species richness and assemblage structure. Hydrobiologia 823:217–230

Tundisi JG, Matsumura-Tundisi T (2003) Integration of research and management in optimizing multiple uses of reservoirs: the experience in South America and Brazilian case studies. Hydrobiologia 500:231–242

Van Eck NJ, Waltman L (2010) Software survey: VOSviewer, a computer program for bibliometric mapping. Scientometrics 84:523–538

Vancine MH, Duarte KDS, de Souza YS et al (2018) Atlantic amphibians: a data set of amphibian communities from the Atlantic Forests of South America. Ecology 99:1692–1692

Vieira BP (2017) Conceitos utilizados no Brasil para aves aquáticas. Atualidades Ornitológicas 196:41–48

Vitule JRS, Skóra F, Abilhoa V (2012) Homogenization of freshwater fish faunas after the elimination of a natural barrier by a dam in Neotropics. Divers Distrib 18(2):111–120

Wengrat S, Padial AA, Jeppesen E et al (2018) Paleolimnological records reveal biotic homogenization driven by eutrophication in tropical reservoirs. J Paleolimnol 60:299

Whiles MR, Lips KR, Pringle CM et al (2006) The effects of amphibian population declines on the structure and function of Neotropical stream ecosystems. Front Ecol Environ 4:27–34

Wojciechowski J, Heino J, Bini LM et al (2017) Temporal variation in phytoplankton beta diversity patterns and metacommunity structures across subtropical reservoirs. Freshw Biol 62:751–766

Zeng EY (ed) (2018) Microplastic contamination in aquatic environments: an emerging matter of environmental urgency. Elsevier

# Part III
# Threats

# Chapter 11
# Land-Cover Changes and an Uncertain Future: Will the Brazilian Atlantic Forest Lose the Chance to Become a *Hopespot*?

**Paula Koeler Lira, Rita de Cássia Quitete Portela, and Leandro Reverberi Tambosi**

**Abstract** Land-cover changes led the Brazilian Atlantic Forest to the current situation of low forest cover mostly distributed in small and isolated fragments composed by forests of varying ages and degradation states. This worrying situation has huge consequences for biodiversity conservation and for the provision and maintenance of ecosystem services. Despite that, until recently, we had reasons to believe that there was a great opportunity of turning the Atlantic Forest *hotspot* into a *hopespot*. Unfortunately, however, the actual Brazilian government is dismantling the country's environmental policies and, thus, is making this turning point in the Brazilian Atlantic Forest into an obscure and appalling scenario.

**Keywords** Anthropic expansion · Biodiversity threats · Habitat fragmentation · Habitat loss · Environmental policies · Forest degradation

## 11.1 Historical Changes

Even before the arrival of the Portuguese in Brazil in 1500, the Brazilian Atlantic Forest was already subject to some level of anthropogenic disturbance though on a smaller scale. The indigenous people that dominated the Brazilian Atlantic Forest region for approximately 1000 years before the arrival of Portuguese settlers prac-

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tised nomadic slash-and-burn agriculture (Dean 1996). This agricultural method locally known as *coivara* consisted in opening a clearing in the forest, waiting until the driest months of the year so that the forest remains would be dry and then burning it. This process transformed forest into ashes that would briefly fertilize the agricultural cultivation. After some harvests, the cultivated area was abandoned and the forest was let to recover (Dean 1996). Considering the site-specific and sporadic nature of this cultivation system, it might not have significantly impacted the Brazilian Atlantic Forest (Pinto et al. 2014), but some forest that we consider today as pristine might have actually been through several cycles of cutting, drying, burning and regeneration (Dean 1996).

After the Portuguese arrival, the Brazilian indigenous population suffered a huge demographic crash (Pádua 2004). In the first hundred years of contact with the colonizers, their population was reduced by 90–95% mostly as a result of epidemics propagation (Dean 1996; Pádua 2004). Upon arrival, Portuguese colonizers overexploited brazilwood trees (*Paubrasilia echinata*) as a source of red dye for cloth, taking down approximately two million trees and impacting nearly 600,000 ha of forest in the first century of occupation (Dean 1996). Besides the vast areas of deforestation, this over-exploitation is also responsible for the fact that now, the brazilwood tree is threatened to extinction (Martinelli and Moraes 2013). Although the over-exploitation of brazilwood tree ceased in the mid-1800 due to the production of synthetic dye, the extraction of wood for making bows for violin gained notability making the brazilwood trees known as the tree of music and contributing to the ongoing population decline (Martinelli and Moraes 2013; Martinelli et al. 2018).

Simultaneous with the brazilwood exploitation, the Portuguese crown provided land concessions and many incentives to encourage people to rapidly occupy the Brazilian territory and consolidate and expand the sugarcane in large monocultures properties. The production of sugar, especially in the northeast of Brazil, led to severe deforestation since forests were cleared for the cultivation and to provide wood to be used as fuel in sugar mills. The sugarcane economic cycle started on 1530s and lasted until the 1750s. Although often overlooked, another important cause of deforestation in the Brazilian northeast during the early three centuries of colonization was the extensive cattle raising. This activity implied in cutting and burning vast areas of the north-eastern Brazilian Atlantic Forest to be turned into pastures (Câmara 2003). This economic cycle, known as the leather cycle (Coimbra-Filho and Câmara 1996), had devastating impacts on forests especially on forest formations near water bodies (Câmara 2003).

At the eighteenth century but also in the beginning of the nineteenth century, the mining cycle, especially for gold, had also a great impact on the Brazilian Atlantic Forest. The techniques for extracting gold demanded lumber, firewood and the establishment of new towns, especially in the Minas Gerais state (Scarano 2014). This economic cycle was not very long but left some important legacies. In addition to the transfer of the Brazilian economic centre to the south-east region (in 1763, the capital moved from Salvador in the north-east to Rio de Janeiro in the south-east), prospecting activities, metal production and even a metallurgical kick-start were crucial to the beginning of Brazilian industrialization.

However, it was the coffee economic cycle that contributed the most to the urbanization and industrial development of Brazil. From the mid-eighteenth century to the beginning of the twentieth century, coffee plantations occupied a great proportion of the south-east region which was once covered by the Atlantic Forest. In the state of São Paulo, for example, the Atlantic Forest cover was reduced from 80% to approximately 8% between 1854 and 1973 largely due to coffee plantation expansion to sustain exports to the United States and Europe (Victor et al. 2005).

The cocoa cycle, from the end of the nineteenth century to the middle of the twentieth century, was not as meaningful for the economy as the previous ones, but also produced an impact on the Brazilian Atlantic Forest, especially in the south of Bahia state. Cocoa shrubs (*Theobroma cacao*) were mainly planted under thinned native forests, a system regionally known as *cabruca*, but shaded cocoa plantations were also established by felling all native trees to plant cocoa seedlings under exotic species (Faria et al. 2007; Cassano et al. 2009).

In the south of Brazil, the Atlantic Forest was devastated for the Brazilian pine *Araucaria angustifolia* exploitation (araucaria logging cycle), especially between the First World War and the 1970s (Fonseca et al. 2009), for use in construction and furniture and for cellulose. In the 1970s, when the araucaria trees were gone, the lands were quickly converted into agricultural cultivation or forest plantation with exotic species, such as *Eucalyptus* and *Pinus* spp.

In the 1970s, sugarcane plantations had again a devastating effect on the Brazilian Atlantic Forest (Câmara 2003). The rise in the price of petroleum sparked the interest on ethanol as a substitute to petroleum derivatives. As a consequence, large tracts of the Brazilian Atlantic Forest, especially on the north-east and on the interior of São Paulo state, were again converted to sugarcane plantations this time for ethanol production. Additionally, during the last 50 years, the construction of hundreds of hydroelectric dams has also contributed to the Brazilian Atlantic Forest devastation. Some of the largest dams of the world are in the Brazilian Atlantic Forest region. The Itaipu Dam, in the Parana River, flooded at least 1350 Km$^2$ of the Atlantic Forest and the Porto Primavera Dam, also in the Parana River, more than 2200 Km$^2$.

The historical anthropogenic disturbance on the Brazilian Atlantic Forest, briefly described here, is a result of the long history of human interaction with the biome which is described in detail by Solórzano et al. 2021 (Chap. 2), Lins-e-Silva et al. 2021 (Chap. 3), Faria et al. 2021 (Chap. 4) and Carlucci et al. 2021 (Chap. 5).

## 11.2 Recent Changes and the Current Situation

Since 1985, the native Brazilian Atlantic Forest cover has been monitored by the NGO SOS Mata Atlântica and the Brazilian Space Agency (INPE) in order to produce deforestation reports, initially every 5 years and, more recently, every year (SOS Mata Atlântica and INPE 2018). Although these reports were extremely relevant to help control deforestation in the Brazilian Atlantic Forest, they have not allowed the detailed monitoring of land-use and land-cover changes including the

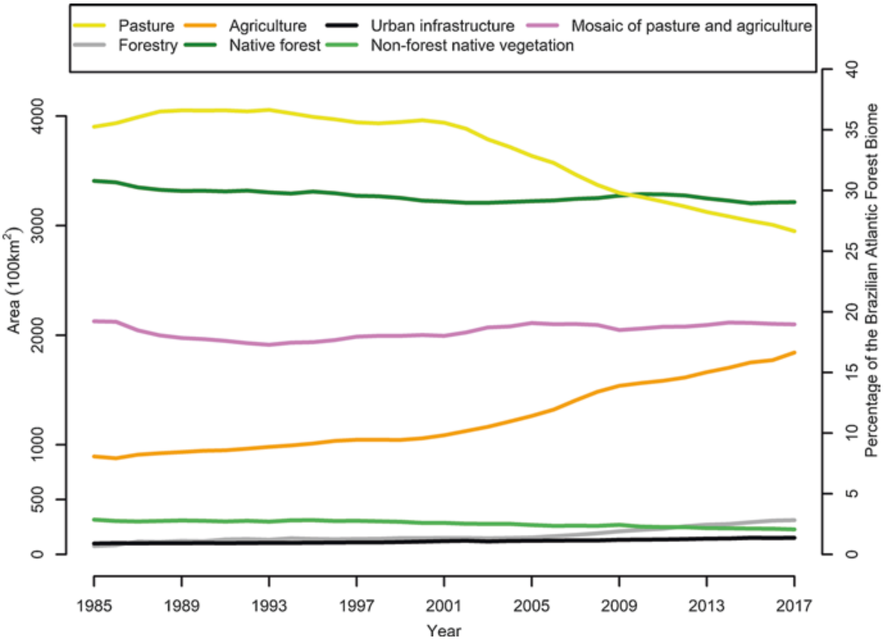

**Fig. 11.1** Area and percentage of the Brazilian Atlantic Forest biome covered by different land-use and land-cover classes from 1985 to 2017 (Data obtained from Project MapBiomas v 3.0)

recovery of native vegetation. Recently, with the improvement on the capability of processing large images dataset, the Project MapBiomas (http://mapbiomas.org/) generated a collection of annual land-use and land-cover maps from 1985 to 2017 for the entire Brazilian Atlantic Forest (Project MapBiomas v 3.0). The Project MapBiomas is a multi-institutional initiative that was born in 2015 with the goal to produce annual land-use and land-cover maps with a consistent methodology to allow monitoring the changes in the entire Brazilian territory.

For the first time since the Brazilian Atlantic Forest occupation, there is consistent data for analysing land-use and land-cover change for the entire biome (Project MapBiomas v 3.0) which unveiled interesting results. In 1985, the main land-use and land-cover types in the Brazilian Atlantic Forest region were pastures (35.3% of the biome), followed by native forest (30.8%), mosaic of pastures and agriculture (19.2%), areas exclusively occupied by agriculture (8.1%), forestry (i.e. forest plantation with exotic species, 0.7%) and urban areas (0.9%; Fig. 11.1). Although the main land-use and land-cover types were the same through the last decades, there was an important change in the extent of pasture and agricultural areas. Pastures were mainly replaced by agricultural areas resulting in a reduction of almost 25% in pasture areas and a double-fold increase of agricultural areas from 1985 to 2017 (Fig. 11.1).

Today, pasture and agriculture are the two predominant land-use types in the Brazilian Atlantic Forest region and particularly cover a large part of the interior forest biogeographic subregion (Ribeiro et al. 2011). The current outsized pasture lands have a huge impact in the Brazilian Atlantic Forest conservation. In order to access water, cattle cause large impacts in riparian forests and riverine systems which reduce their potential to increase landscape connectivity (Ribeiro et al. 2011). Moreover, a commonly used management technique in pastures is using fire to stimulate the regrowth of exotic grasses and kill tree seedlings and bushes, and this frequently ends up burning forest fragments. These frequent burning events in combination with cattle grazing inside fragments result in the degradation of nearby forests (Ribeiro et al. 2011). Despite the large extent of pastures distributed in both small and large properties, in Brazil, the focus of agricultural production expansion is based on large monocultures properties since the early economic cycles although small properties are essential for food production for local consumption (Paulino 2014). The recent replacement of pastures by agriculture occurred mostly due to the expansion of citrus and soybeans plantations but also due to large expansions of sugarcane plantations for biofuel production in the beginning of the 1980s and in the end of the 1990s (Ferraz et al. 2014). Those expansions displaced pastures to lower productivity and non-mechanizable areas and also resulted in the expansion of pastures in other Brazilian biomes such as Cerrado, Pantanal and Amazon (Sá et al. 2013; Strassburg et al. 2014; Meyfroidt et al. 2014). The presence of pastures in hilly areas, the inadequate cattle management and the abandonment of these pastures result in a great number of pastures with low productivity (Strassburg et al. 2014) and millions of hectares of degraded land (Pacto 2011).

Traditional sugarcane harvesting, conducted since the beginning of European settlement, was based on burning the crop which causes air pollution and can accidentally burn the surrounding forest fragments (Durigan et al. 2007). These management practices had important negative impact on forest fragments quality and also on human health, since the pollution caused by burning the sugarcane increased the number of respiratory disease in densely populated areas (Cançado et al. 2006). With the improvement of mechanization associated with environmental law enforcement and environmental certifications, the burning of sugarcane was drastically reduced in the southern states in the middle of the 2000s. The mechanization of sugarcane harvesting caused a reduction in the connectivity between forest fragments since isolated trees – which considerably increase the connectivity between forest fragments acting as stepping stones (e.g. Boscolo et al. 2008) – have been cut down to facilitate mechanical harvest (Ribeiro et al. 2011).

Although occupying a smaller portion of the biome, the urban areas and forestry also presented an increase in their extent of almost 50% and 200%, respectively (Fig. 11.1). The expansion of urban areas mainly occurred close to large cities and metropolitan areas due to real estate market for second household for leisure in the countryside. The urban areas expanded over pastures, agriculture and native forests (e.g. Teixeira et al. 2009). The expansion of forestry also occurred over pasture areas but also over abandoned citrus plantations and in sandy and hilly sites (Brockerhoff et al. 2013), initially in the southern and south-eastern regions and more recently in the central and northern regions of the Atlantic Forest (ABRAF 2013). Forestry may function as a higher-quality matrix when good ecological management practices are employed (Fonseca et al. 2009). Unfortunately, forest plantations are more commonly managed focusing on high productivity with the most common practices being large plantations of a few clones, short-cycle rotation, understorey cleaning, intense chemical use and large areas cropped at the same time that, in most cases, results in a low-quality matrix which is detrimental to the Atlantic Forest conservation (Ribeiro et al. 2011).

The recent expansion of agriculture and forestry activities was based on intense mechanization and is also focused on international commodities market which, in most cases, requires environmental certification and law compliance. As a consequence, some low-productivity agricultural and forestry areas such as those located in high slopes and riparian zones were allowed to naturally regenerate and, in some cases, were actively restored (Rodrigues et al. 2009; Ferraz et al. 2014; Molin et al. 2017). Today, the existing native forest covers 28% of the biome, as it was first reported by the high-resolution mapping done by Rezende et al. (2018) for the year 2013. Although there is a net loss of forest cover from 1985 to 2017, in 2002, there was a change from decreasing to expanding forest area (Fig. 11.1). Some authors suggest that the Brazilian Atlantic Forest may be experiencing a forest transition process due to high economic development, agricultural adjustment to most productive areas, reduction of rural population and increase in environmental concern (Baptista and Rudel 2006; Igari and Tambosi 2016; Calaboni et al. 2018). Although this scenario seems optimistic at first, the availability of Project MapBiomas data allowed, for the first time, a spatially explicit analysis of the forest cover change for the entire biome and unveiled a critical scenario. Despite the forest cover increase, forest recovery has occurred concomitantly with deforestation across the biome (Project MapBiomas v 3.0) resulting in the replacement of old-growth by young regenerated forests as it was previously detected at small spatial scales (Teixeira et al. 2009; Lira et al. 2012; Ferraz et al. 2014). Consequently, almost 20% of the current forest cover is less than 30 years old (Project MapBiomas v 3.0), and as already anticipated, the age of regeneration plays an important role in defining biodiversity in human-modified landscapes (Lira 2011). This forest cover rejuvenation is a recurrent pattern in tropical forests, and recent studies have shown that regenerated forests usually do not persist in tropical regions (Reid et al. 2018). This situation is worrisome because the provision of ecosystem services, such as the accumulation of carbon, greatly differs between early and late successional forests. Shimamoto et al. (2014), studying lowland and montane Atlantic Forest in Southern Brazil, found that second-growth forests (41–60 years old) accumulate more than twofold carbon than immature forest (21–40 years old) and much more than tenfold carbon than young forests (7–20 years old). So young regenerated forests that are not allowed to age have a reduced potential for conserving plant species richness (Rozendaal et al. 2019) and for mitigating carbon emission (Chazdon et al. 2016) which will result in positive emissions in the Brazilian Atlantic Forest despite the forest cover gain. Additionally, some other ecosystem services provided by forests (e.g. soil stabilization and water provision) are also reduced if the forest is young secondary (Tabarelli et al. 2010).

All these intense land-use and land-cover dynamics occurred mainly in private properties as the protection of native forest cover is higher inside protected areas. However, only 30% of the existing Brazilian Atlantic Forest vegetation is located inside protected areas being 21% in sustainable use protected areas (Rezende et al. 2018) and only 9% in strictly protected areas (IUCN Categories I–IV) (Ribeiro et al. 2009; Rezende et al. 2018) which ensures a higher level of protection. Although protected areas are cornerstone of efforts to conserve biodiversity, Laurance et al. (2012) revealed that about half of all tropical protected areas are experiencing an erosion of biodiversity because of environmental changes, such as habitat disruption, hunting and forest-product exploitation, both inside and immediately outside protected areas. Indeed, Xavier da Silva et al. (2018) found that not all areas of Iguaçú National Park, a large (185,200 ha) strictly protected area in the Brazilian Atlantic Forest, are suitable for mammal species due to the negative effects associated with the park boundary (i.e. edge effects and associated impacts), presence of tourism infrastructure and hunting pressure. Likewise, Portela et al. (2010) suggest that the absence of top predators and the isolation of Poço das Antas Biological Reserve (3500 ha) are probably the main reasons why the palm *Euterpe edulis*, a key resource species for the frugivorous community, is declining due to a top-down effect caused by a superabundant predator, the capuchin monkey *Sapajus nigritus*. These kinds of negative effects should be even stronger on small protected areas, which are the majority in the Brazilian Atlantic Forest (Ribeiro et al. 2009). To make the situation worse, the protected areas in the Brazilian Atlantic Forest are not only frequently small but also not connected to other protected areas (Ribeiro et al. 2009). This means that the effectiveness of protected areas in conserving biodiversity also depends on the ability of their land-use and land-cover surroundings to increase habitat availability and/or habitat connectivity. The implementation of strategies to increase habitat availability and facilitate biological flow such as the long-distance trail known as *Caminho da Mata Atlântica* (Viveiro de Castro et al. 2021 Chap. 19) in the Brazilian Atlantic Forest may boost an increase in the connectivity among the protected areas.

In the current context of low forest cover composed by a mosaic of forests with varying ages of regeneration, evaluating the configuration of the Brazilian Atlantic Forest is essential to understand the real threats to the biome and thus to be able to efficiently manage forest fragments in order to maximize biodiversity conservation. Information about the spatial distribution of the forest fragments was only available for small spatial scales (e.g. Jorge and Garcia 1997; Ranta et al. 1998) until 2009 when Ribeiro and collaborators have analyzed, for the first time, the forest configuration for the entire Brazilian Atlantic Forest by using a map produced by SOS Mata Atlântica and INPE (2008) for the year 2005. Results revealed a situation which is expected to compromise biodiversity conservation on the Brazilian Atlantic Forest *hotspot*: (1) More than 80% of the forest fragments are smaller than 50 ha, and only 0.03% of the total forest fragments are larger than 10,000 ha; (2) at least 73% of the existing forest is located less than 250 m from any forest edge, almost half is less than 100 m apart from any edge and only 7.7% is located farther than 1000 m into the forest from any edge; and (3) the mean distance between forest fragments is around 1440 m (but it varies widely), and the importance of small fragments (<50 ha) in reducing isolation is enormous. Additionally, most of the biome is below the minimum amount of forest required to preserve biodiversity integrity (Banks-Leite et al. 2014) with only 5% of its extent with enough forest cover to maintain high levels of biodiversity (Tambosi et al. 2014).

The spatial configuration of the Brazilian Atlantic Forest reported by Ribeiro et al. (2009, 2011) is very worrisome. Fragment size is considered to be positively related with population size, and consequently, larger fragments have better prospects of sustaining species over the long term (Brooks et al. 1999). Moreover, only large forest fragments are capable of preserving species with large area requirements, and this is why Peres (2005) urges for megareserves in the Amazon, for example. Additionally, since most of the existing Atlantic Forest is very close to forest edges, a huge part of the existing forest is under the process known as edge effects, i.e. forests near the edge suffer changes in abiotic conditions and, consequently, on species and ecological process (Murcia 1995; Laurance et al. 2002). Many forest-dependent species have been reported to be negatively affected by edge effects worldwide (Ewers and Didham 2006; Pfeifer et al. 2017; Ries et al. 2004), and for the Brazilian Atlantic Forest, this is no different. For example, edge effects were reported as one of the most important determinants of avifauna (Banks-Leite et al. 2010) and tree community structure (Oliveira et al. 2004) at Atlantic Forest landscapes in São Paulo and Alagoas states, respectively. Finally, it is well known that isolation and the connectivity level of fragments are key components for maintaining species in Atlantic Forest fragmented landscapes (e.g. Boscolo and Metzger 2011); well-connected fragments can sustain a higher number of species and also a large number of individuals/population sizes because (re)colonization and rescue effect are often occurring in those fragments (Pardini et al. 2010).

The types of land-use in the Atlantic Forest region have an extremely important influence on the biodiversity conservation within Atlantic Forest fragments. The biodiversity conservation capability of forest fragments, especially the small and isolated ones, is dependent on the land-use type (i.e. matrix) surrounding it. When matrix permeability is high (i.e. when matrix vegetation structure is similar with forest), edge effects are less intense, and structural connectivity is high, possibly allowing the biological flow to be larger. Boesing et al. (2018a) found that matrix type was a key factor influencing avian cross-habitat spillover; species movement into high-permeable matrices was facilitated while low-permeable matrices acted as a barrier to spillover. As a consequence of matrix influence on edge effects intensity and landscape connectivity, biodiversity extinction thresholds are also modulated by matrix types. Boesing et al. (2018b) found that higher-quality matrices postponed the abrupt loss of birds expected to occur with habitat loss and the associated decline in habitat connectivity. However, even higher-quality matrices cannot postpone diversity loss when the amount of habitat is below a critical threshold of 20% (Boesing et al. 2018b). Therefore, the land-use types have a great importance in biodiversity conservation within Atlantic Forest fragments.

The negative influences of forest cover reduction, forest fragmentation and isolation and land-use change can have synergistic effects that might trigger profound local modifications on forest structure and composition, i.e. forest degradation, by conducting forest fragments towards early successional stages and biotic homogenization (Santos et al. 2008; Lôbo et al. 2011). As old-growth Atlantic Forest remnants are mostly within fragmented landscapes, edge-induced microclimatic conditions may trigger a retrogressive succession known as 'secondarization' (Santos et al. 2008; Tabarelli et al. 2012). Thereby, shade-tolerant species typical of old-growth forests (e.g. large-seeded, emergent and hardwood tree species) are replaced by a set of persistent, fast-growing pioneer species (Oliveira et al. 2004) leading to a low biomass due to their smaller size and low wood density (Lôbo et al. 2011, Fig. 11.2). Lôbo et al. (2011) found that the tree flora across the north-eastern region of the Brazilian Atlantic Forest became 20–40% more similar to each other post-1980 (i.e. after the expansion of sugarcane plantations in the region) being largely dominated by a set of native pioneer species. This secondarization process has negative impacts on forest functioning such as lowering the ability of hosting medium- and large-sized mammals (Canale et al. 2012) and on the provision of ecosystem services such as carbon accumulation since degraded and defaunated forest accumulates less carbon than pristine forest (Bello et al. 2015; Rocha-Santos et al. 2016). Moreover, the proliferation of some 'winner' native pioneer species and the parallel decline of old-growth or disturbance-sensitive species (i.e. 'loser' species) resemble the classical biotic homogenization driven by the replacement of native biota by exotic invading species (Lôbo et al. 2011; Vitule et al. 2021 Chap. 13). This biotic homogenization can undermine the natural immunity of the Atlantic Forest against exotic species invasion as low diversity and functionally similar communities are more susceptible to exotic invasions (Tilman 1997). All these changes that lead to forest degradation are stronger on small and isolated forest fragments (Putz et al. 2011) which are the majority in the Brazilian Atlantic Forest (Ribeiro et al. 2009).

Changes in forest cover and configuration are frequently associated with illegal exploitative human activities, such as selective logging, harvest of non-timber forest products (e.g. fruits) and hunting that can also contribute to forest degradation. The over-exploitation by selective logging and harvest of non-timber forest products reduces harvested populations sizes but can also have a negative effect in the community as a whole leading to forest degradation. For example, as the palm *E. edulis* has a huge impact in attracting the seed disperser fauna (Galetti et al. 1999), the over-exploitation of its palm heart, which leads to individual's death, has influenced the seed rain composition by decreasing the density of animal-dispersed seeds and by increasing the seed density of light-dependent plant species (Muler et al. 2014). Another change observed in the forest after the palm heart over-exploitation was the change in the light regime of the understorey (Muler et al. 2014). All those changes may affect the forest regeneration process and potentially change the whole plant community and, consequently, the structure of the forest.

The most profound effect of over-exploitative hunting is defaunation which involves not only global species extinctions but also declines in animal local abundance and geographic range contraction leading to local populations loss (Dirzo et al. 2014; Young et al. 2016; Ceballos et al. 2017). In human-altered landscapes,

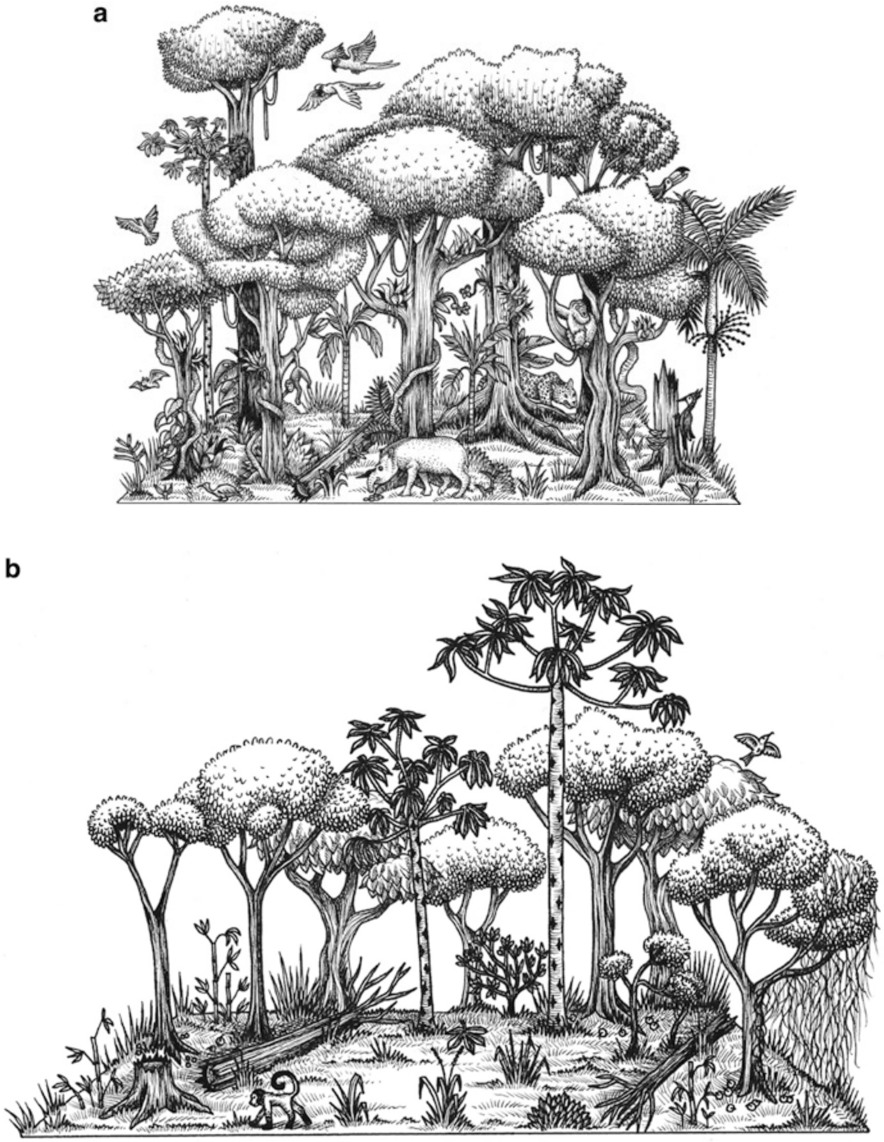

**Fig. 11.2** Preserved (a) and degraded (b) Brazilian Atlantic Forest. The forest depicted in (a) is much more diverse and structurally and functionally complex than in (b). Shade-tolerant plant species, typical of old-growth forests (e.g. large-seeded, emergent and hardwood tree species), are present in (a) but are missing from (b) which is mostly composed of fast-growing pioneer species and, thus, has low accumulated biomass. Medium- and large-sized animals are present in (a), and thus, seedling recruitment of large-seeded species is observed. As (b) is defaunated, a collapse in its natural regeneration is observed as fruits are not dispersed and accumulate in the forest floor. Artwork by Luís Gustavo Barretto Rodrigues

defaunation tends to occur at higher rates because of the easy accessibility by hunters (Galetti and Dirzo 2013). As most part of the existing Brazilian Atlantic Forest is within human-altered landscapes, defaunation has been detected in great part of the fragments (e.g. Galetti et al. 2006, 2013; Canale et al. 2012; Galetti et al. 2021 Chap. 14).

A salient pattern of defaunation is that medium- and large-bodied animals tend to be more vulnerable to hunting than small-bodied animals (e.g. Dirzo et al. 2014; Peres 2000) leading to a pattern of animal community downsizing (Young et al. 2016). Galetti et al. (2006) observed a collapse of the seedling recruitment and, consequently, of the population regeneration of the palm *Astrocaryum aculeatissimum*, which has large fruits and, thus, relies on medium- and large-sized vertebrates for seed dispersal. Another example of the impact of the animal community downsizing in defaunated Atlantic Forest areas was the rapid evolutionary seed size reduction observed for *E. edulis* which has led to the establishment of smaller seedlings that are less vigorous (Galetti et al. 2013). Therefore, the effects of defaunation can be especially pervasive for large-seeded plants that rely on a few medium- and large-sized frugivore species for seed dispersal, such as the two palm species mentioned above. Thus, defaunated forest, i.e. empty forests (Redford 1992), will, in long term, also lose large-seeded plants by demographic collapses, and consequently, the composition, structure and the carbon storage of the forest will change (Bello et al. 2015). All those changes in composition, richness and ecosystems function have led some Atlantic Forest areas towards a much simpler structurally and functionally state (Fig. 11.2).

## 11.3 The Uncertain Future of the Brazilian Atlantic Forest

Historical and recent changes led the Brazilian Atlantic Forest to the current situation of low forest cover mostly distributed in small and isolated fragments composed by forests of varying ages and degradation states which have, as aforementioned, huge consequences for biodiversity conservation and the provision and maintenance of ecosystem services. However, despite this worrying situation, until recently, we had a number of reasons to believe that there was a great opportunity of turning the Atlantic Forest *hotspot* into a *hopespot*, i.e. a history of loss, fragmentation and degradation turned into a sustainable future whereby society's vulnerability to climate change is reduced by protecting and restoring nature and improving human life standards (Scarano and Ceotto 2015; Rezende et al. 2018). Unfortunately, however, the actual Brazilian government is dismantling the country's environmental policies and turning this *hopespot* perspective into an obscure and appalling scenario.

The first indicator that could shed light in the future of the Atlantic Forest is the current proportion of Brazilian Atlantic Forest inside protected areas (30%; Rezende et al. 2018) which surpasses the 17% of terrestrial and inland water that must be conserved according to the AICHI Target 11 aimed at increasing and improving protected areas (www.cbd.int/sp). However, when considering only strictly protected areas, the percentage of Atlantic Forest cover protected is only 9%. Moreover, a gap analysis by Rodrigues et al. (2004) demonstrated the inadequacy of such conservation targets as they are blind to the fact that biodiversity is not evenly distributed across countries and/or biomes. In the Brazilian Atlantic Forest, at least the strictly protected areas are unevenly distributed across the eight subregions (Ribeiro et al. 2009) and between forest and non-forests habitats (Ribeiro et al. 2011). The AICHI Target 11 also states that the protected area should include areas of particular importance for biodiversity and ecosystem services and should be ecologically representative, effectively and equitably managed and well-connected which still might not be the case for the protected areas network of the Brazilian Atlantic Forest.

Although protected areas are conservation cornerstones, most of what remains of the Brazilian Atlantic Forest vegetation is located outside protected areas in private rural properties (Rezende et al. 2018). Thus, most part of the Brazilian Atlantic Forest is protected by measures stipulated by the Brazilian law, especially the Atlantic Forest Law (AFL; Law No. 11.428, 22 December 2006) and the Brazilian Native Vegetation Protection Law (NVPL; Law No. 12.727, 17 October 2012). Therefore, the future of the Brazilian Atlantic Forest and consequently the development of agriculture and human well-being and safety within the biome largely depend on the AFL and NVPL enforcement.

The AFL protects pristine and secondary fragments of the Atlantic Forest in initial, intermediate and advanced stages of succession, and deforestation of these fragments in the Brazilian Atlantic Forest is only allowed in cases of public interest. However, the AFL also states that an area abandoned for a maximum period of 10 years can be considered as fallow management to recover soil properties. As a consequence, landowners are allowed to clear the regenerated vegetation in fallow areas, reducing the persistence of regenerated forests. Despite the legal protection of the AFL, the rates of deforestation in the biome still occur in areas older than 10 years and go way beyond any possible inference of public interest. The NVPL defines the area of native vegetation that must be maintained under protection or restricted use in a given rural property (Areas of Permanent Protection and Legal Reserves; APP and RL in Portuguese), and it also defines the situations in which landholders are required to recover natural vegetation on their land (Brancalion et al. 2016). The NVPL revoked and replaced the Forest Code from 1965 (Law No. 4.771, 15 September 1965) under a huge criticism of the Brazilian scientific community (e.g. Metzger et al. 2010). The main environmental setbacks of the NVPL highlighted by the scientific community were (i) the regularization of properties that were non-compliant with the preceding legislation by the concession of amnesty of all fines and deforestation previous to 22 July 2008, (ii) the removal of protection of some environmentally fragile areas and (iii) the reduction and relaxation of the obligation of restoring native vegetation (Brancalion et al. 2016). Those changes were definitely not good news for the environmental conservation in Brazil as the total protected area on hills and mountaintops was reduced by 87% and the potential restoration area was reduced to 58% compared to the precedent legislation (Soares-Filho et al. 2014).

Even with all these changes, the rural properties in the Brazilian Atlantic Forest region are still far from being in compliance with NVPL. Rezende et al. (2018) have mapped the hydrology and the riparian APP, i.e. marginal strips along all water bodies that must be covered by native vegetation according to the NVPL, and found 7.2 Mha of degraded riparian areas of which at least 5.2 Mha must be restored before 2038 by landowners for legislation compliance. Restoring this legal debt could increase native vegetation cover in the Brazilian Atlantic Forest up to 35% which is above the critical biodiversity threshold established for different taxonomic groups and ecosystem functions and services (Banks-Leite et al. 2014; Boesing et al. 2018a). The compliance with the NVPL is key for the conservation of what is left of the Brazilian Atlantic Forest flora and fauna, and it is also essential to ensure the provision of ecosystem services such as water for agriculture and human consumption and climate regulation.

Fortunately, a great advance of NVPL was the establishment of innovative programmes of control and incentive to facilitate and promote compliance with the law (Brancalion et al. 2016). One advance of NVPL was the establishment of the mandatory Environmental Rural Registry (CAR, in Portuguese). The CAR is a self-declaratory system for registry of rural properties and of their status regarding the environmental NVPL demands. Properties that are not listed in the CAR will not have access to low-interest rates agricultural loans and will not have the amnesty for illegal deforestation before 2008. Once registered in the CAR database, non-compliant rural properties may enlist in the Environmental Compliance Program (PRA) to comply with the law. After signing with PRA, the environmental liability of rural properties can be settled through the Project for Recovery of Degraded and Altered Land (PRADA) where the landholder commits to maintain and recover native vegetation in APPs and/or RLs or to compensate their RL deficit by purchasing either area with native vegetation or Environmental Reserve Quotas (CRA).

The NVPL also established the possibility of using an economic mechanism known as payment for ecosystem services (PES, in English) to stimulate the conservation and recovery of native vegetation in Brazil, but comprehensive and explicit regulation protocols to guide PES application are missing in NVPL (Brancalion et al. 2016). Before the approval of the NVPL, this mechanism has been employed in several parts of the Brazilian Atlantic Forest by either governmental or non-governmental organizations, but nowadays, PES schemes are mainly conducted by government agencies. The PES schemes involved not only restoration of native vegetation but also avoiding degradation and preventing soil erosion and water contamination (Guedes and Seehusen 2011). Some of these PES schemes were extremely successful, such as the programme called *Conservador das Águas* at Extrema municipality (Minas Gerais state) which almost doubled the forest cover in the programme's watershed (Richards et al. 2015). These mechanisms seem to be resulting in the increase of forest cover inside private properties; however, they are usually in pilot projects situation, and their high implementation and transaction costs do not allow increasing scale to achieve all the restoration goals (Ruggiero et al. 2019). Banks-Leite et al. (2014) suggested a conservation scheme for the entire Brazilian Atlantic Forest that involves PES as a mechanism to increase land-owner participation on setting aside private land for conservation. This set-aside programme would maximize the provision of habitat to biodiversity and of ecological functions across rural areas while minimizing the costs to society (an annual investment equivalent to 6.5% of what Brazil spends on agricultural subsidies).

Another reason to believe in the opportunity of turning the Atlantic Forest *hotspot* into a *hopespot* is that the Brazilian commitment to restore approximately 12 million hectares by 2030 (National Plan for Native Vegetation Restoration; PLANAVEG, in Portuguese) will be partly implemented in the Brazilian Atlantic Forest. As initial efforts, recent studies have estimated the potential of natural regeneration (MMA 2017) and also the costs of restoration for the Brazilian biomes (Benini and Adeodato 2017). A great part of the Brazilian Atlantic Forest is considered of low potential of natural regeneration due to the low forest cover, high agricultural intensification and long time since deforestation (MMA 2017), probably demanding active restoration efforts. Despite the low potential for natural regeneration, there was approximately 700 thousand hectares of forest recovery between 2011 and 2015 in the Atlantic Forest (Crouzeilles et al. 2019). So, there is great potential of forest recovery, demanding only law enforcement and deforestation avoidance, including land clearance in considered fallow areas according to the AFL. In order to foster the restoration in the Brazilian Atlantic Forest, a multi-sectorial initiative engaging more than 200 institutions, the Atlantic Forest Restoration Pact, was launched in 2009 (De Siqueira et al. 2021 Chap. 18) and contributed with data and knowledge to the PLANAVEG creation and also to the studies for its implementation.

However, all the positive aspects created during the last decades to promote biodiversity conservation and restoration in the Brazilian Atlantic Forest are at risk due to economic and political instability. Although Brazil has long been a frontrunner in environmental diplomacy and climate change policy (e.g. the international conventions on biological diversity and climate change were born during the Earth Summit in Rio de Janeiro in 1992), the actual Brazilian government is aiming to reduce the environmental protection by preventing new indigenous and protected areas demarcation, reducing the existing protected areas (i.e. revocation of areas that have already been designated), allowing mining activities and water resources exploitation within indigenous and protected areas, legalizing wild animals hunting, weakening existing regulations for the use and sale of pesticides and eliminating or considerably reducing the environmental licences restrictions for new infrastructure projects and other economic activities despite the recent two largest Brazilian environmental disasters caused by mining dam collapses in Mariana and Brumadinho (Minas Gerais state) (Abessa et al. 2019). As if it was not enough, the actual Brazilian government is also openly hostile to the fight against climate change as the new president insinuated to pull Brazil out of the Paris Agreement.

The Brazilian government has been severely criticized at home and abroad and risks losses on all fronts (Abessa et al. 2019). In May 2019, the seven former environment ministers have presented a live pronunciation and signed a letter warning that Brazil's new administration is systematically trying to destroy Brazil's environmental protection policies. Additionally, more than 600 European scientists signed a letter urging the European Union (Brazil's second largest trading partner) to make trade negotiations with Brazil only if it meets the commitments to reduce deforestation and indigenous conflicts (Kehoe et al. 2019). More recently, an essay, with 1230 signatories representing the Brazilian science community as well as indigenous and traditional community members, has called on international trading partners, state and municipal governments, members of parliament and concerned citizens to pressure the Brazilian government to reverse its destructive agenda (Levis et al. 2020).

If the actual Brazilian government succeeds in dismantling environmental policies, Brazil will lose the opportunity of turning the Atlantic Forest *hotspot* into a *hopespot* as glimpsed by Scarano and Ceotto (2015) and Rezende et al. (2018). Moreover, an important part of agricultural commodities production comes from the Atlantic Forest biome such as sugarcane plantations for biofuel, forestry for pulp and paper, citrus plantations and part of the beef and soy production. These products may face international boycotts from environmentally concerned consumer markets which will have an impact on the Brazilian economy and unknown consequences for the country's inhabitants.

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## References

Abessa D, Famá A, Buruaem L (2019) The systematic dismantling of Brazilian environmental laws risks losses on all fronts. Nat Ecol Evol 3:510–511

ABRAF - Associação Brasileira de Produtores de Florestas Plantadas (2013) Anuário estatístico ABRAF 2013 ano base 2012. ABRAF, Brasília

Banks-Leite C, Ewers RM, Metzger JP (2010) Edge effects as the principal cause of area effects on birds in fragmented secondary forest. Oikos 119:918–926

Banks-Leite C, Pardini R, Tambosi LR, Pearse W, Bueno A et al (2014) Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot. Science 345:1041–1045

Baptista SR, Rudel TK (2006) A re-emerging Atlantic Forest? Urbanization, industrialization and the forest transition in Santa Catarina, southern Brazil. Environ Conserv 33:195–202

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

Benini RM, Adeodato S (eds) (2017) Forest restoration economy. The Nature Conservancy, São Paulo

Boesing AL, Nichols E, Metzger JP (2018a) Land use type, forest cover and forest edges modulate avian cross-habitat spillover. J Appl Ecol 55:1252–1264

Boesing AL, Nichols E, Metzger JP (2018b) Biodiversity extinction thresholds are modulated by matrix type. Ecography 41:1520–1533

Boscolo D, Metzger JP (2011) Isolation determines patterns of species presence in highly fragmented landscapes. Ecography 34:1–12

Boscolo D, Candia-Gallardo C, Awade M, Metzger JP (2008) Importance of interhabitat gaps and stepping-stones for lesser woodcreepers (Xiphorhynchus fuscus) in the Atlantic Forest, Brazil. Biotropica 40:273–276

Brancalion PHS, Garcia LC, Loyola R, Rodrigues RR, Pillar VD, Lewinson TM (2016) A critical analysis of the Native Vegetation Protection Law of Brazil (2012): updates and ongoing initiatives. Nat Conservacao 14:1–15

Brockerhoff EG, Jactel H, Parrota JA, Ferraz SFB (2013) Role of eucalypt and other planted forests in biodiversity conservation and the provision of biodiversity-related ecosystem services. For Ecol Manage 301:43–50

Brooks TM, Pimm SL, Oyugi JO (1999) Time lag between deforestation and bird extinction in tropical forests fragments. Conserv Biol 13:1140–1150

Calaboni A, Tambosi LR, Igari AT, Farinaci JS, Metzger JP, Uriarte M (2018) The forest transition in São Paulo, Brazil: historical pattern and potential drivers. Ecol Soc 23:7

Câmara IG (2003) Breve história da conservação da Mata Atlântica. In: Galindo-Leal C, Câmara IG (eds) Mata Atlântica: biodiversidade, ameaças, perspectivas. Fundação SOS Mata Atlântica and Conservação Internacional, Belo Horizonte

Canale GR, Peres CA, Guidorizzi CE, Gatto CAF, Kierulff MCM (2012) Pervasive defaunation of forest remnants in a tropical biodiversity hotspot. PLoS One 7:e41671

Cançado JED, Saldiva PHN, Pereira LAA, Lara LBLS, Artaxo P, Martinelli LA, Arbex MA, Zanobetti A, Braga ALF (2006) The impact of sugarcane-burning emissions on the respiratory system of children and the elderly. Environ Health Perspect 114:725–729

Cassano CR, Schroth G, Faria D, Delabie JHC, Bede L (2009) Landscape and farm scale management to enhance biodiversity conservation in the cocoa producing region of southern Bahia, Brazil. Biodivers Conserv 18:577–603

Ceballos G, Ehrlich PR, Dirzo R (2017) Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. PNAS 114:E6089–E6096

Chazdon RL, Broadbent EN, Rozendaal DMA, Bongers F, Zambrano AMA et al (2016) Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. Sci Adv 2(5):e1501639

Coimbra-Filho A, Câmara IG (1996) Os limites originais do bioma Mata Atlântica na região nordeste do Brasil. Fundação Brasileira para Conservação da Natureza (FBCN), Rio de Janeiro

Crouzeiles R, Santiami E, Rosa M, Pugliese L, Brancalion PHS et al (2019) There is hope for achieving ambitious Atlantic Forest restoration commitments. Perspect Ecol Conserv 17:80–83

Dean WJ (1996) A ferro e fogo: a história e a devastação da Mata Atlântica Brasileira. Companhia das Letras, São Paulo

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

Durigan G, Siqueira MF, Franco GADC (2007) Threats to the Cerrado remnants of the state of São Paulo, Brazil. Sci Agricola 64:355–363

Ewers RM, Didham RK (2006) Confounding factors in the detection of species responses to habitat fragmentation. Biol Rev 81:117–142

Faria D, Barradas MLP, Dixo M, Laps RR, Baumgarten J (2007) Ferns, frogs, lizards, birds and bats in forest fragments and shade cacao plantations in two contrasting landscapes in the Atlantic forest, Brazil. Biodivers Conserv 16:2335–2357

Ferraz SFB, Ferraz KMPMB, Cassiano CC, Brancalion PHS, da Luz DTA, Azevedo TN, Tambosi LR, Metzger JP (2014) How good are tropical forest patches for ecosystem services provisioning? Landscape Ecol 29:187–200

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

Galetti M, Dirzo R (2013) Ecological and evolutionary consequences of living in a defaunated world. Biol Conserv 163:1–6

Galetti M, Zipparro VB, Morellato PC (1999) Fruiting phenology and frugivory of Euterpe edulis in a lowland Atlantic Forest of Brazil. Ecotropica 5:115–112

Galetti M, Conatti CI, Pires AS, Guimarães JRPR, Jordano P (2006) Seed survival and dispersal of an endemic Atlantic forest palm: the combined effects of defaunation and forest fragmentation. Bot J Linn Soc 151:141–149

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

Guedes FB, Seehusen SE (eds) (2011) Pagamentos por serviços ambientais na Mata Atlântica: lições aprendidas e desafios. Ministério do Meio Ambiente, Brasília

Igari AT, Tambosi LR (2016) Agribusiness and socioeconomic drivers of land cover change in Brazil. In: Gheler-Costa C, Lyra-Jorge MC, Verdade LM (eds) Biodiversity in agricultural landscapes in southeastern Brazil, first edn. De Gruyter Open, Berlin, pp 28–42

Jorge LAB, Garcia GJ (1997) A study of habitat fragmentation in Southeastern Brazil using remote sensing and geographic information systems (GIS). For Ecol Manage 98:35–47

Kehoe L, Reis T, Virah-Sawmy M, Balmford A, Kuemmerle T and 604 signatories (2019) Make UK trade with Brazil sustainable. Science 364: 341

Laurance WF, Lovejoy TE, Vasconcelos HL, Bruna EM, Didham RK, Stouffer PC, Gascon C, Bierregaard RO, Laurance SG, Sampaio E (2002) Ecosystem decay of Amazonian forest fragments: a 22-year investigation. Conserv Biol 16:605–618

Laurance WF, Useche CD, Rendeiro J, Kalka M, Bradshaw CJA et al (2012) Averting biodiversity collapse in tropical forest protected areas. Nature 489:290–293

Levis C, Flores BM, Mazzochini GG, Manhães AP, Campos-Silva JV, Amorim PB, Peroni N, Hirota M, Clement CR and 1230 signatories (2020) Help restore Brazil's governance of globally important ecosystem services. Nat Ecol Evol 4(2):172–173

Lira PK (2011) Effects of landscape change history on birds and small mammals in the Atlantic Forest. PhD Thesis, Universidade de São Paulo

Lira PK, Tambosi LR, Ewers RM, Metzger JP (2012) Land-use and land-cover change in Atlantic Forest landscapes. For Ecol Manage 278:80–89

Lôbo D, Leão T, Melo FPL, Santos AMM, Tabarelli M (2011) Forest fragmentation drives Atlantic forest of northeastern Brazil to biotic homogenization. Divers Distrib 17:287–296

Martinelli G, Moraes MA (2013) Livro vermelho da flora do Brasil. Instituto de Pesquisas Jardim Botânico do Rio de Janeiro, Rio de Janeiro

Martinelli G, Martins E, Moraes M, Loyola R, Amaro R (2018) Livro Vermelho da Flora Endêmica do Estado do Rio de Janeiro. Jardim Botânico do Rio de Janeiro, Secretaria de Estado do Ambiente (SEA), Rio de Janeiro

Metzger JP, Lewinson TM, Joly CA, Verdade LM, Martinelli LA, Rodrigues RR (2010) Brazilian law: full speed in reverse? Science 329:276–277

Meyfroidt P, Carlson KM, Fagan ME, Gutiérrez-Vélez VH, Macedo MN et al (2014) Multiple pathways of commodity crop expansion in tropical forest landscapes. Environ Res Lett 9:074012

MMA (2017) Potencial de regeneração da vegetação natural no Brasil. Ministério do Meio Ambiente, Brasília

Molin PG, Gergel SE, Soares-Filho BS, Ferraz SFB (2017) Spatial determinants of Atlantic Forest loss and recovery in Brazil. Landscape Ecol 32:857–870

Muler AE, Rother DC, Brancalion PS, Naves RP, Rodrigues RR, Pizo MA (2014) Can overharvesting of a non-timber-forest-product change the regeneration dynamics of a tropical rainforest? The case study of *Euterpe edulis*. For Ecol Manage 324:117–125

Murcia C (1995) Edge effects in fragmented forests: implications for conservation. Trends Ecol Evol 10:58–62

Oliveira MA, Grillo AS, Tabarelli M (2004) Forest edge in the Brazilian Atlantic forest: drastic changes in tree species assemblages. Oryx 38:389–394

Pacto pela restauração da Mata Atlântica (2011) Mapa de áreas potenciais para restauração florestal. https://docs.wixstatic.com/ugd/5da841_c4c4a0c5c32f44298a2808ef2df520f1.pdf. Accessed 10 May 2019

Pádua JA (2004) A ocupação do território brasileiro e conservação dos recursos naturais. In: Milano M et al (eds) Unidades de conservação: atualidades e tendências. Fundação O Boticário, Curitiba

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

Paulino ET (2014) The agricultural, environmental and socio-political repercussions of Brazil's land governance system. Land Use Policy 36:134–144

Peres CA (2000) Effects of subsistence hunting on vertebrate community structure in amazonian forests. Conserv Biol 14:240–253

Peres CA (2005) Why we need megareserves in Amazonia. Conserv Biol 19:728–733

Pfeifer M, Lefebvre V, Peres CA, Banks-Leite C, Wearn OR, Marsh CJ, Butchart SHM, Arroyo-Rodríguez V, Barlow J, Cerezo A, Cisneros L, D'Cruze N, Faria D, Hadley A, Harris SM, Klingbeil BT, Kormann U, Lens L, Medina-Rangel GF, Morante-Filho JC, Olivier P, Peters SL, Pidgeon A, Ribeiro DB, Scherber C, Schneider-Maunoury L, Struebig M, Urbina-Cardona N, Watling JI, Willig MR, Wood EM, Ewers RM (2017) Creation of forest edges has a global impact on forest vertebrates. Nature 551:187–191

Pinto SR, Melo F, Tabarelli M, Padovesi A, Mesquita CA, Scaramuzza CAM, Castro P, Carrascosa H, Calmon M, Rodrigues R, César RG, Brancalion PHS (2014) Governing and delivering a biome-wide restoration initiative: the case of the Atlantic Forest Restoration Pact in Brazil. Forests 5:2212–2229

Portela RCQ, Bruna EM, Santos FAM (2010) Are protected areas really protecting populations? A test with an Atlantic rain forest palm. Trop Conserv Sci 3:361–372

Project MapBiomas - Collection v 3.0 of Brazilian Land Cover & Use Map Series, accessed on 10 May 2019 through the link: http://mapbiomas.org

Pütz S, Groenevelda J, Alves LF, Metzger JP, Hutha A (2011) Fragmentation drives tropical forest fragments to early successional states: a modelling study for Brazilian Atlantic forests. Ecol Model 222:1986–1997

Ranta P, Blom T, Niemelä J, Joensuu E, Siitonen M (1998) The fragmented Atlantic rain forest of Brazil: size, shape and distribution of forest fragments. Biodivers Conserv 5:385–403

Redford K (1992) The empty forest. Bioscience 42:412–422

Reid JL, Fagan ME, Lucas J, Slaughter J, Zahawi RA (2018) The ephemerality of secondary forests in southern Costa Rica. Conserv Lett 12:e12607

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Strassburg BBN, Tabarelli M, Fonseca GA, Mittermeier RA (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:208–214

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

Ribeiro MC, Martensen AC, Metzger JP, Tabarelli M, Scarano F, Fortin MJ (2011) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In: Zachos FE, Habel JC (eds) Biodiversity hotspots. Springer, Berlin, Heidelberg, pp 405–434

Richards RC, Rerolle J, Aronson J, Pereira PH, Gonçalves H, Brancalion PHS (2015) Governing a pioneer program on payment for watershed services: stakeholder involvement, legal frameworks and early lessons from the Atlantic forest of Brazil. Ecosyst Serv 16:23–32

Ries L, Fletcher RJ, Battin J, Sisk TD (2004) Ecological responses to habitat edges: mechanisms, models and variability explained. Annu Rev Ecol Evol Syst 35:491–522

Rocha-Santos L, Pessoa MS, Cassano CR, Talora DC, Orihuela RLL, Mariano-Neto E, Morante-Filho JC, Faria D, Cazetta E (2016) The shrinkage of a forest: landscape-scale deforestation leading to overall changes in local forest structure. Biol Conserv 196:1–9

Rodrigues ASL, Andelman SJ, Bakarr MI, Boitani L, Brooks TM et al (2004) Effectiveness of the global protected area network in representing species diversity. Nature 428:640–643

Rodrigues RR, Lima RAF, Gandolfi S, Nave AG (2009) On the restoration of high diversity forests: 30 years of experiences in the Brazilian Atlantic Forest. Biol Conserv 142:1242–1251

Rozendaal DMA, Bongers F, Aide TM, Alvarez-Dávila E, Ascarrunz N et al (2019) Biodiversity recovery of Neotropical secondary forests. Sci Adv 5:eaau3114

Ruggiero PGC, Metzger JP, Tambosi LR, Nichols E (2019) Payment for ecosystem services programs in the Brazilian Atlantic Forest: effective but not enough. Land Use Policy 82:283–291

Sá SA, Palmer C, di Falco S (2013) Dynamics of indirect land-use change: empirical evidence from Brazil. J Environ Econ Manag 65:377–393

Santos BA, Peres CA, Oliveira MA, Grilloc A, Alves-Costa CP, Tabarelli M (2008) Drastic erosion in functional attributes of tree assemblages in Atlantic forest fragments of northeastern Brazil. Biol Conserv 141:249–260

Scarano F (2014) Mata Atlântica: uma história do futuro. Edições de Janeiro, Rio de Janeiro

Scarano FR, Ceotto P (2015) Brazilian Atlantic forest: impact, vulnerability, and adaptation to climate change. Biodivers Conserv 24:2319–2331

Shimamoto CY, Botosso PC, Marques MCM (2014) How much carbon is sequestered during the restoration of tropical forests? Estimates from tree species in the Brazilian Atlantic forest. For Ecol Manage 329:1–9

Soares-Filho B, Rajão R, Macedo M, Carneiro A, Costa W, Coe M, Rodrigues H, Alencar A (2014) Cracking Brazil's Forest Code. Science 344:363–364

SOS Mata Atlântica, Instituto Nacional de Pesquisas Espaciais (2008) Atlas dos remanescentes florestais da Mata Atlântica: período 2000–2005. SOS Mata Atlântica and INPE, São Paulo

SOS Mata Atlântica, Instituto Nacional de Pesquisas Espaciais (2018) Atlas dos remanescentes florestais da Mata Atlântica: período 2016–2017. SOS Mata Atlântica and INPE, São Paulo

Strassburg BBN, Latawiec AE, Barioni LG, Nobre CA, da Silva VP, Valentim JF, Vianna M, Assad ED (2014) When enough should be enough: improving the use of current agricultural lands could meet production demands and spare natural habitats in Brazil. Glob Environ Chang 28:84–97

Tabarelli M, Aguiar AV, Ribeiro MC, Metzger JP, Peres CA (2010) Prospects for biodiversity conservation in the Atlantic Forest: lessons from aging human-modified landscapes. Biol Conserv 143:2328–2340

Tabarelli M, Peres CA, Melo FPL (2012) The 'few winners and many losers' paradigm revisited: emerging prospects for tropical forest biodiversity. Biol Conserv 155:136–140

Tambosi LR, Martensen AC, Ribeiro MC, Metzger JP (2014) A Framework to optimize biodiversity restoration efforts based on habitat amount and landscape connectivity. Restor Ecol 22:169–177

Teixeira AMG, Soares-Filho BS, Freitas SR, Metzger JPM (2009) Modeling landscape dynamics in an Atlantic Rainforest region: implications for conservation. For Ecol Manage 257:1219–1230

Tilman D (1997) Community invasibility, recruitment limitation, and grassland biodiversity. Ecology 781:81–92

Victor MAM, Cavalli AG, Guillaumon JR, Filho RS (2005) Cem anos de devastação - Revisitada 30 anos depois. Ministério do Meio Ambiente, Brasília

Xavier da Silva M, Paviolo A, Tambosi LR, Pardini R (2018) Effectiveness of Protected Areas for biodiversity conservation: mammal occupancy patterns in the Iguaçu National Park, Brazil. J Nat Conserv 41:51–62

Young H, McCauley DJ, Galetti M, Dirzo R (2016) Patterns, causes, and consequences of anthropocene defaunation. Annu Rev Ecol Evol Syst 47:333–358

# Chapter 12
# Climate Change and Biodiversity in the Atlantic Forest: Best Climatic Models, Predicted Changes and Impacts, and Adaptation Options

**Mariana M. Vale, Paola A. Arias, Geusep Ortega, Manoel Cardoso, Beatriz F. A. Oliveira, Rafael Loyola, and Fabio R. Scarano**

## 12.1 Current and Future Climate

The Atlantic Forest, with its large latitudinal and altitudinal range, is under different climatic regimes. The current spatial distribution of the Brazilian Atlantic Forest can be linked to several meteorological processes currently at play in the region. These processes have important influences on the observed temperature and rainfall, which in turn drive the environmental conditions needed for the occurrence of the Atlantic Forest vegetation (Salazar et al. 2007; Carnaval et al. 2009; Colombo and Joly 2010). One of these processes is the occurrence of cold fronts (Cavalcanti and Kousky 2009), which are characterized by relatively colder and denser air masses moving from the polar region interacting with moist and hot air, causing a substantial drop in temperature and an increase in precipitation. Cold fronts are most common in the central and southern portions of the Atlantic Forest (latitudes <15° S) (Cavalcanti and Kousky 2009), where their impacts are most relevant. Another process linked to the range of precipitation and temperature observed in a large portion of the Atlantic Forest is a large-scale atmospheric circulation pattern known as the South Atlantic Convergence Zone. This meteorological system is characterized

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by an elongated northwest-southeast region, from the Amazon to southeastern Brazil, where convergent winds, clouds, and substantial precipitation are observed during the summer (Carvalho and Jones 2009). In the portions of the Atlantic Forest located in the northeast of Brazil, the South Atlantic portion of the Intertropical Convergence Zone plays an essential role on the precipitation (Melo et al. 2009). This system is also characterized by the convergence of surface air, clouds, and precipitation and happens typically during March and April.

An additional, essential feature that interacts with the atmospheric patterns discussed above is topography, which in the Atlantic Forest is particularly relevant in the Serra do Mar mountain range (see Carlucci et al. 2021 Chap. 5). Mountain ridges may lift air masses, enhancing cloud and rain formation. Also, as the temperature usually decreases with altitude in the troposphere, locations at sea level or mountaintops will usually present different species.

As we see, atmosphere, ocean, and topography have a great influence on climate and vegetation cover. Indeed, these are essential components of the general circulation models (GCMs) used both for meteorological and climate change predictions. The GCMs used to project future changes in climate due to ongoing climate change incorporate both the natural and anthropogenic dynamics in the main components of the Earth system, usually the atmosphere and oceans, but also the cryosphere and land use/land cover, among others. They are developed by dozens of research institutions worldwide, using standard basic protocols established by the Coupled Model Intercomparison Project and adopted by the Intergovernmental Panel on Climate Change (IPCC) (Taylor et al. 2012). The projections are made under different scenarios of future greenhouse gas concentrations in the atmosphere, called representative concentration pathways (RCPs). There are four such scenarios, ranging from the most optimistic RCP 2.6, where emissions are reduced by about 90% in 2100 compared with the present and a projected average global increase of 1.5 °C by the end of the century, to the most pessimistic RCP 8.5, where greenhouse gas emissions continue mostly untapped, and an average global increase of 4.0 °C is projected by the end of the century (Van Vuuren et al. 2011; Knutti and Sedláček 2012; IPCC 2013).

The Brazilian National Institute for Space Research (INPE) has developed a GCM, the Brazilian Earth System Model (BESM), with the objective of assembling the scientific expertise capable of developing and maintaining a state-of-the-art Earth system model and the aim of participating in the Coupled Model Intercomparison Project Phase 6 (Veiga et al. 2019). On top of global climate models, which have a global extent, there are regional climate models (RCMs), which cover a specific region of the globe, such as a country or a continent, and typically have a higher spatial resolution and a better performance within the region of interest. RCMs need to be nested within a GCM that provides the input data for the external geographic boundary of the RCM. The Brazilian National Institute for Space Research has also developed an RCM for South America, the CPTEC Eta model, with versions nested within the HadGem (UK), MIROC (Japan), and BESM (Brazil) (Chou et al. 2014).

Projected changes in climate can differ widely among GCMs, and different GCMs are known to perform better in specific regions of the globe (e.g., Cai et al. 2009; Yin et al. 2013). Therefore, studies that aim at projecting the future impacts of ongoing climate change on biodiversity, such as species distribution models, should use GCMs that show a good performance in the region of study. This information, however, is not readily available for most regions, especially in the Tropics, and definitely not for the Atlantic Forest. To fill this gap, we provide here an evaluation of the performance of different GCMs over the Atlantic Forest.

We evaluated the performance of 48 GCMs from CMIP5 Phase 5 (used in the last IPCC Assessment Report; Taylor et al. 2012) using Taylor Diagrams (Taylor 2001). Simulations are available at https://esgfnode.llnl.gov/search/esgf-llnl/. The Taylor diagram provides a graphical framework that allows a suite of variables from a variety of models to be compared to reference data. We compared the modeled (GCM) historical data (1850–2005) with the observed historical data (1979–2005) derived from TRMM (Tropical Rainfall Measuring Mission) and ERA-Interim for precipitation and air surface temperature, respectively (Dee et al. 2011; Huffman et al. 2014). Taylor diagrams quantify the spatial similarity of each GCM with respect to observations in terms of the spatial correlation coefficient, the root-mean-square error (RMSE), and the ratio of their variances (Taylor 2001).

We worked under the assumption that, if the models realistically simulate the present climate, they will be able to provide more confident projections of future states. Therefore, after identifying the set of models with the best simulation of seasonal patterns for precipitation and air surface temperature according to the Taylor diagrams, we analyzed their projections for the twenty-first century under the RCP 8.5 scenario. We evaluated the projected change by the end of the twenty-first century (2071–2100) using the 1971–2000 period as the baseline.

The results point to eight GCMs as the best models for the Atlantic Forest (Fig. 12.1, Table 12.1). The data for these GCMs, downscaled and calibrated (bias-corrected), is freely available for download in standard GIS format in the WorldClim Global Climate data portal (https://www.worldclim.org/CMIP5v1). The projected change under the RCP 8.5 scenario showed, on average, the regional increase in temperature between ca. 4.8 °C and 5.6 °C, while precipitation did not show a clear future trend (Fig. 12.2). The lack of trend for precipitation for the Atlantic Forest as a whole is likely because it lies in a region that shows different projections, with an expected decrease in precipitation in its northern portion but an increase in its southern and southeastern portion (Fig. 12.3). This effect is why the Atlantic Forest is often separated into two portions (north and south) in studies related to climate change (e.g., PBMC 2014).

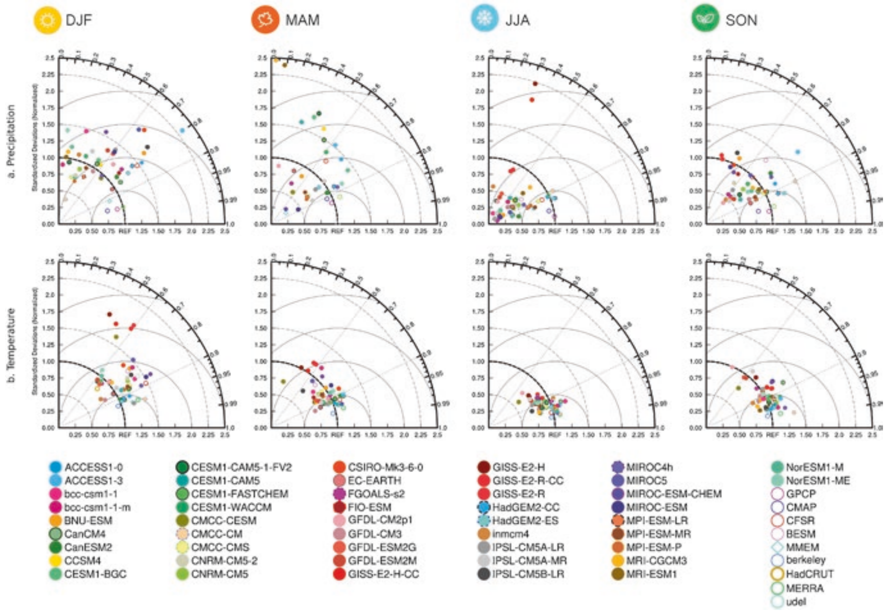


**Fig. 12.1** Taylor diagrams for seasonal simulations of precipitation (top) and air surface temperature (bottom). GCMs are shown by full circles, while observational datasets and BESM (Brazilian Earth System Model) are shown with open circles. For each data point, three statistics are plotted: the Pearson correlation coefficient is shown in the azimuthal angle (dashed straight lines), the root-mean error in GCM is proportional to the distance from the point on the x-axis identified as "REF" (bold black dashed line contours), and the ratio of variance of GCM is proportional to the radial distance from the origin (black solid line contours). The distance between each data point and "REF" is a measure of how realistically each GCM reproduces the observational datasets (see Taylor 2001 for further details). DJF, December, January, February; MAM, March, April, May; JJA, June, July, August; SON, September, October, November

**Table 12.1** Subset of best global climate models for the Atlantic Forest according to Taylor diagrams. Model types: Atmosphere-Ocean General Circulation Models (AOGCMs) and Earth System Models (ESM; includes land use/land cover and the biosphere)

| Model name | Institution | Type |
|---|---|---|
| ACCESS1.0 | CSIRO and Bureau of Meteorology (BOM), Australia | AOGCM |
| CanCM4 | Canadian Centre for Climate Modelling and Analysis, Canada | AOGCM |
| CanESM2 | Canadian Centre for Climate Modelling and Analysis, Canada | ESM |
| CMCC-CM | Centro euro-Mediterraneo per I CambiamentiClimatici, Italy | AOGCM |
| HadGEM2-CC | Met Office Hadley Centre, United Kingdom | ESM |
| HadGEM2-ES | Met Office Hadley Centre, United Kingdom | ESM |
| MIROC4h | Japan Agency for Marine-Earth Science and Technology, atmosphere and ocean research institute, and national institute for environmental studies, Japan | ESM |
| MPI-ESM-LR | Max Planck Institute for Meteorology, Germany | ESM |

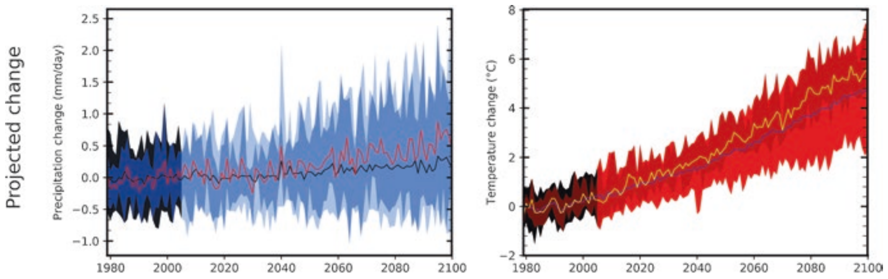
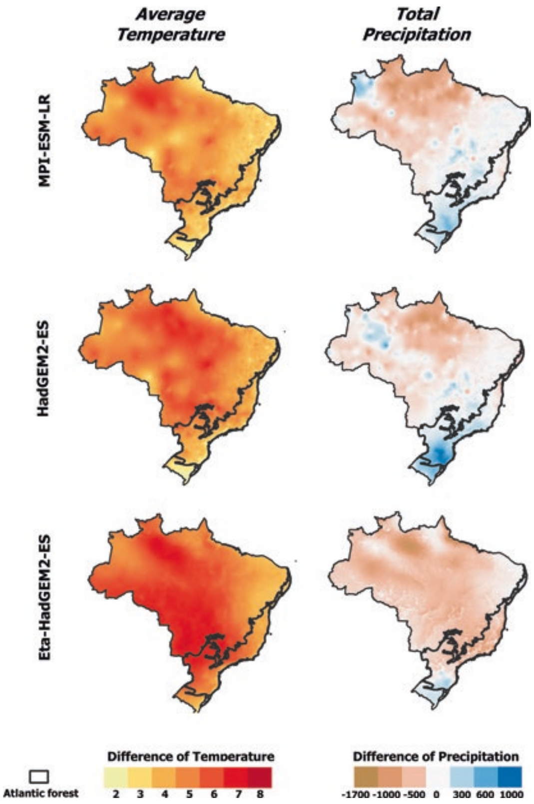


**Fig. 12.2** Projected change of precipitation (left) and air surface temperature (right) over the Atlantic Forest by the end of the twenty-first century. Changes were calculated as the difference between the mean RCP 8.5 projection by the end of the century (2071–2100) and the mean historical simulation (1971–2000), using only the best global climate models for the Atlantic Forest according to the Taylor diagrams

**Fig. 12.3** Projected change in mean temperature (left) and total precipitation (right) in Brazil. Change was calculated for two global climate models with good performance over the Atlantic Forest (MPI-ESM-LR and HadGEM2-ES) using data from the WorldClim Global Climate database and a South America regional climate model (ETA-HadGEM ES)



## 12.2 Impacts of Climate Change on Biodiversity

No studies to date have shown observed impacts of ongoing climate on biodiversity in the Atlantic Forest, but there is a growing number of studies that project future impacts. A global-scale study that combined vulnerability associated with future climate change hazard, future suitability to the invasion by invasive alien species, and current land use changes placed the Atlantic forest among the top three most vulnerable biodiversity hotspots in the world (Bellard et al. 2014).

The bulk of the studies on project impacts of climate change the Atlantic First biodiversity rely on species distribution models under future climatic conditions, which are increasingly being combined with land use change. The Atlantic Forest, together with Cerrado, is possibly the hotspot with the highest number of such studies in South America (Bustamante et al. 2019). There is a clear taxonomic bias in studies toward terrestrial vertebrates, especially not only towards birds and amphibians (e.g., Marini et al. 2010; Souza et al. 2011; Loyola et al. 2014; Lemes and Loyola 2013; Lemes et al. 2014; Hoffmann et al. 2015; Vasconcelos and Nascimento 2016; Vale et al. 2018; Vasconcelos et al. 2018) but also mammals (e.g., Meyer et al. 2014; Gouveia et al. 2016; Lima et al. 2019) and reptiles (e.g., Lourenço-de-Moraes et al. 2019), and also some studies on invertebrates, especially insects (Ferro et al. 2014; Gianinni et al. 2012, 2015; Beltramino et al. 2015; Faleiro et al. 2018; Françoso et al. 2019) and plants (Colombo and Joly 2010; Cupertino-Eisenlohr et al. 2017). The studies typically predict a reduction of the distribution or climatic suitability in the future for the vast majority of the species and expansion for few. An exception is Zwiener et al. (2017a), who predicted a general increase in local richness of woody plants, but mainly for the generalist and disturbance-tolerant species, and a decrease in beta diversity and biotic homogenization at large scales. Few studies consider biological interactions under climate change (e.g., see Vasconcelos et al. 2017 for mutualism and Braz et al. 2019 for competition) and invasive species (e.g., Nori et al. 2011; Assunção et al. 2018). Many studies also predict a southward range shifts (e.g., Colombo and Joly 2010; Ferro et al. 2014; Lemes et al. 2014; Beltramino et al. 2015; Hoffmann et al. 2015; Vale et al. 2018; Silva et al. 2019), which might be a compensation for increased temperatures, and is congruent with a projected southward expansion of the Atlantic Forest vegetation (Salazar et al. 2007). The result is a predicted reduction in species richness and an increase in turnover, in general (which might be clade-specific for amphibians at least; Loyola et al. 2014). Based on these studies, we can assert with high confidence (*sensu* Mastrandrea et al. 2010) that there is a high risk of biodiversity loss in the Atlantic Forest, including species extinction, due to climate change.

A number of studies predict a reduced effectiveness of the network of protected areas in the Atlantic Forest under climate change (e.g., Meyer et al. 2014; Lemes et al. 2014; Ferro et al. 2014; Beltramino et al. 2015; Giannini et al. 2015; Lourenço-de-Moraes et al. 2019; Silva et al. 2019). Systematic conservation planning that takes climate change into account, however, can minimize future loss of species in protected areas throughout meaningful guidance for protected areas network expansion (Vale et al. 2018; Vasconcelos and Prado 2019; Lemes and Loyola 2013). Protection of forest remnants alone will not suffice, however, and well-planned forest restoration is a necessary complementary action to safeguard the Atlantic Forest's biodiversity under climate change (Giannini et al. 2015; Zwiener et al. 2017b).

Despite a large number of studies projecting climate change impacts on the Atlantic Forest's biodiversity, there are substantial taxonomic and methodological bias, which generate significant knowledge gaps, particularly on altitudinal, freshwater, and coastal environments. Given the complex topography of the Atlantic Forest, the lack of observational studies and scarcity of predictive studies (see Hoffmann et al. 2015) on climate change impacts on high-altitude environments and mountain species is surprising. Mountain species and environments are well known for their high vulnerability to climate change both worldwide (La Sorter and Jetz 2010; Öztürk et al. 2016) and in Brazil (Scarano et al. 2016; Fernandes et al. 2018, but see Esser et al. 2019). Several studies have observed range shifts and reduction in mountains. These studies typically replicate altitudinal gradient studies at the community level carried decades ago, revealing upward range shifts and contraction (e.g., Forero-Medina et al. 2011), and could be carried out in the Atlantic Forest. The lack of studies on observed or predicted climate change impacts on Atlantic Forest freshwater ecosystems is also worrisome, given their high diversity and vulnerability (Collen et al. 2013; Roland et al. 2012; but see Esser et al. 2019). Finally, the Atlantic Forest has many associated coastal ecosystems, such as restingas and mangroves, which are also vulnerable to climate change, especially sea-level rise, but there is blatant lack of studies on the topic (Godoy and Lacerda 2015; Oliveira et al. 2016; Copertino et al. 2010). The review of Godoy and Lacerda (2015), for example, reveals that, taking into consideration climate change alone, mangroves in most areas will display a positive response. However, mangroves in southeastern Brazil, which are in constrained coastlines, will most probably not survive (Godoy and Lacerda 2015).

## 12.3 Adaptation Strategies

Climate change and deforestation are the main causes of biodiversity loss in terrestrial ecosystems in the present and the near future. In addition to contributing individually to biological degradation, the interaction between these factors induces negative feedbacks on ecosystem resilience and contributes synergistically to biological degradation at species, genetic, and/or habitat level. However, reversing current and estimated trends of climate change effects on biodiversity is a socio-ecological problem.

We need to perceive the Atlantic forest as an inherently human-nature coupled system, rather than social and natural systems separately. Within the domain of the Atlantic Forest, we find both some of Brazil's largest urban centers (such as Rio de Janeiro and São Paulo) and more than half of the land dedicated to horticulture and food production (Joly et al. 2014). Forest is no longer the norm in the landscape; it is mostly a collection of small vegetation remnants surrounded by a matrix of urban and agricultural ecosystems (Rezende et al. 2018a).

Given this situation, using ecosystems to promote societal adaptation to climate change is particularly appropriate for the Atlantic Forest (Scarano and Ceotto 2015). Ecosystem-based adaptation to climate change (EbA) is defined by the Convention on Biological Diversity (CBD 2009) as "the use of biodiversity and ecosystem services as part of an overall adaptation strategy to help people to adapt to the adverse effects of climate change." One can then expect that a successful EbA program could improve livelihoods across the Atlantic Forest by implementing actions related to ecosystem conservation and restoration (Scarano 2017, 2019).

Although current political and economic instability are obvious hurdles (Loyola 2014; Dobrovolski et al. 2018), recent optimism with EbA applied to the Atlantic Forest has to do with several factors: (1) Favorable legal background that makes mandatory restoration and conservation within private properties to pay for environmental debt; (2) Favorable legal background, in the shape of payment for ecosystem services (PES) legislation in many federal states covered by the Atlantic Forest, to fund restoration; (3) Existence of successful case studies related to PES in several states covered by the Atlantic Forest states; (4) the presence of influential civil society organizations acting in issues related to climate change, conservation, and restoration, such as the Atlantic Forest Pact; (5) Presence of strong academic institutions; and (6) Existence of thriving on-the-ground experiences in project implementation (Scarano and Ceotto 2015; Brancalion et al. 2016; Scarano 2017).

The favorable scenario is such that it has led to a discussion on the possibility of the Atlantic forest gradually change its status from "shrinking biodiversity hotspot" (Ribeiro et al. 2011) to "future climate hope spot" (Scarano and Ceotto 2015; Rezende et al. 2018a). For instance, many municipalities with high legal vegetation debt also have high poverty and/or low human development index, such as those in the northern portion of the state of Rio de Janeiro (Rezende et al. 2018b) or those in the Rio Doce valley, in the state of Minas Gerais (Pires et al. 2017). In such cases, economic incentives must apply in order to foster local restoration-based economies. The injection of resources through mechanisms like PES, for example, could strengthen the economic chain of restoration in degraded municipalities – from the production and commercialization of inputs to the implementation of restoration in the field – stimulating job generation and boosting the local economy while restoring the vegetation. The state of Espírito Santo, for instance, has legislation that ensures the redirection of 3% of oil revenues – so-called "royalties" – to fund restoration (Sossai et al. 2016). If applied in the state of Rio de Janeiro, for example, a similar program would have an annual budget of around USD 40 million, based on 3% of 2016 royalties collected by the state government, not considering the amounts collected by the municipalities (Rezende et al. 2018b). This figure covers the annual costs of planting 39% of the environmental debt in private rural properties of the state, considering 20 years (Rezende et al. 2018b). In the case of the Rio Doce valley, funds from compensation and fines owing to a major spill of mining tailings could also cover large areas with forest restoration (Pires et al. 2017).

Beyond restoration, forest conservation is also a critical component to safeguard biodiversity and the ecosystem services it provides and to foster economy. Protected areas contribute to climate change mitigation. By mitigating the emission of $CO_2$ and other greenhouse gases resulting from the degradation of natural ecosystems, protected areas help to prevent the increase in the concentration of these gases in the atmosphere. These areas also play a crucial role in protecting strategic resources for the development of the country. For example, Young and Medeiros (2018) estimated that ecosystem services delivered from protected areas generate economic contributions that significantly exceed the amount that has been allocated by public administrations to the maintenance of the Protected Areas System in Brazil. They also found that 80% of the country's hydroelectricity comes from generating sources that have at least one tributary downstream from a protected area; 9% of the water for human consumption is directly captured in protected areas, 26% is taken in sources downstream of them, and 4% of the water used in agriculture and irrigation is taken from sources in or downstream of protected areas. Finally, the authors argue that public visitation in Brazil's 67 national parks has the potential to generate between R$ 1.6 and 1.8 billion per year, considering the estimated flows of tourists projected for the country. Protected Areas in the Atlantic forest have enormous potential in all these fronts, and expanding its network in the region represents a crucial joint objective to provide synergy between climate change mitigation and adaptation (Locatelli et al. 2015, see below).

It has been argued that climate change adaptation (Agrawal and Lemos 2015) and EbA in particular (Pant et al. 2015; Scarano 2017; Kasecker et al. 2018) can often be an essential step in the transition from a conventional to a sustainable development paradigm. Moreover, sustainable development can both be the cause and consequence of mitigation and adaptation to climate change, but only rarely, the links between these processes are examined in an integrated fashion (see Agrawal and Lemos 2015; Scarano 2017).

The conservation and restoration of natural ecosystems, and in particular forests, are prone to bring together mitigation, adaptation, and sustainable development (e.g., Locatelli et al. 2011; Thornton and Comberti 2017; Strassburg et al. 2019). Trade-offs have also been reported, for instance, between carbon sequestration and biodiversity values, local livelihoods, and tenure security (Ingalls and Dwyer 2016). Nevertheless, careful planning for restoration in the Atlantic forest can optimize costs, biodiversity conservation, and carbon mitigation, which altogether might result in climate change adaptation (Crouzeilles et al. 2015; Zweiner et al. 2017b; Strassburg et al. 2019).

Locatelli et al. (2015) described three processes whereby mitigation and adaptation synergy may take place. The first process is "joint outcome," i.e., activities that are undertaken without climatic objectives that deliver joint adaptation and mitigation outcomes. For instance, in the Atlantic Forest, and Brazil as a whole, indigenous lands are designed for human and land rights and cultural preservation. These areas also play an essential role in protecting threatened species (Ribeiro et al. 2018). However, the 1.2 million hectares of indigenous lands in the region (Pinheiro et al. 2014) are also important for carbon mitigation and climate change adaptation (see Walker et al. 2014; Nogueira et al. 2018). The second process is called "unintended side-effects," whereby activities aimed at one climate objective, either mitigation or adaptation, can deliver outcomes for the other objective. For example, actions that target disaster risk reduction and climate change based on ecosystems may often have a mitigation effect of carbon stock or sequestration. This is the case of conservation or restoration of mangroves and sand dunes to avoid coastal erosion (Scarano 2002, 2009) or of hillside forests to avoid landslides (Brancalion et al. 2016; Renaud et al. 2016). Finally, the third process is "joint objectives" and refers to the association between adaptation and mitigation objectives in a climate-related activity. Activities such as ecosystem restoration, payment for ecosystem services, and climate-smart agriculture, among others, are increasingly designed to achieve both goals, often resulting in sustainable development (see Harvey et al. 2014; Kasecker et al. 2018). The potential for synergy between mitigation and adaptation has been estimated based on the presence or absence of four enabling conditions for integration: policies and strategies, programs and projects, institutional arrangements, and financial mechanisms (Duguma et al. 2014). This potential is measured by a score, and these authors found that Mexico (with a score of 8) has the most enabling conditions for synergy between mitigation and adaptation in Latin America, followed by Brazil (7), Ecuador (5), and Chile (4). In all these countries, ecosystems are a key piece in the potential for synergy, and in Brazil, owing to its capacity and available infrastructure, the Atlantic Forest has the highest potential. This is the reason why the Atlantic Forest is increasingly perceived as a biodiversity hotspot that can upgrade to the status of a "climate hope spot" (Scarano and Ceotto 2015; Rezende et al. 2018a, b): a biome that becomes an example that the path of degradation and extinction can be transformed to one of prosperity for humans and nature alike.

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## References

Agrawal A, Lemos MC (2015) Adaptive development. Nat Clim Chang 5:186–187

Assunção ACR, Alexandrino RV, Caiafa AN, Oliveira G (2018) The invasion of *Artocarpus heterophyllus*, jackfruit, in protected areas under climate change and across scales: from Atlantic Forest to a natural heritage private reserve. Biol Invasions 21:481–492

Bellard C, Leclerc C, Leroy B, Bakkenes M, Veloz S, Thuiller W, Courchamp F (2014) Vulnerability of biodiversity hotspots to global change. Glob Ecol Biogeogr 23:1376–1386

Beltramino AA, Vogler RE, Gregoric DEG, Rumi A (2015) Impact of climate change on the distribution of a giant land snail from South America: predicting future trends for setting conservation priorities on native malacofauna. Clim Chang 131:621–633

Brancalion PHS, Garcia LC, Loyola R, Rodrigues RR, Pillar VD, Lewinsohn TM (2016) A critical analysis of the native vegetation protection law of Brazil (2012): updates and ongoing initiatives. Nat Conserv 14:1–15

Braz AG, Lorini ML, Vale MM (2019) Climate change is likely to affect the distribution but not parapatry of the Brazilian marmoset monkeys (Callithrix spp.). Divers Distrib 25:536–550

Bustamante MMC, Metzger JP, Scariot A, Bager A, Turra A, Barbieri A, Neves A, Boesing AL, Agostinho AA, Marques AC, Dias B, Grelle CEV, Caixeta D, Sawyer D, Scarano FR, Sousa FDR, Fernandes GW, Queiroz H, Miranda HS, Schongart J, Quintao JMB, Martinelli LA, Gomes LC, da Cunha MC, Piedade MTF, Sato MN, Vale MM, Aquino MFS, Vogt N, May P, Fearnside P, Prado RB, Rodrigues RR, Thomaz SM, Pivello VR, Imperatriz-Fonseca VL, Farjalla VF (2019) Capitulo 3: Tendencias e impactos dos vetores de degradação e restauração da biodiversidade e dos serviços ecossistemicos. In: Joly CA, Scarano FR, Seixas CS, Metzger JP, Ometto JP, Bustamante MMC, Padgurschi MCG, Pires APF, Castro PFD, Gadda T, Toledo P (eds) 1° Diagnostico Brasileiro de Biodiversidade e Serviços Ecossistêmicos. Editora Cubo, São Carlos

Cai X, Wang D, Zhu T, Ringler C (2009) Assessing the regional variability of GCM simulations. Geophys Res Lett 36:L02706

Carnaval AC, Hickerson MJ, Haddad CFB, Rodrigues MT, Moritz C (2009) Stability predicts genetic diversity in the Brazilian Atlantic Forest hotspot. Science 323:785–789

Carvalho LMV, Jones C (2009) Zona de Convergência do Atlântico Sul. In: Cavalcanti IFA, Ferreira NJ, Silva MJ, Dias MAF (eds) Tempo e clima no Brasil. Oficina de Textos, São Paulo

Cavalcanti IFA, Kousky VE (2009) Frentes Frias sobre o Brasil. In: Cavalcanti IFA, Ferreira NJ, Silva MJ, Dias MAF (eds) Tempo e clima no Brasil. Oficina de Textos, São Paulo

CBD (Convention of Biological Diversity) (2009) Connecting Biodiversity and Climate Change Mitigation and Adaptation: Report of the Second Ad HocTechnical Expert Group on Biodiversity and Climate Change. CBD Technical Series 41, Montreal

Chou SC, Lyra A, Mourão C, Dereczynski C, Pilotto I, Gomes J, Bustamante J, Tavares P, Silva A, Rodrigues D, Campos D, Chagas D, Sueiro G, Siqueira G, Nobre P, Marengo J (2014) Evaluation of the eta simulations nested in three global climate models. Am J Clim Change 3:438–454

Collen B, Whitton F, Dyer EE, Baillie JEM, Cumberlidge N, Darwall WRT, Pollock C, Richman NI, Soulsby A, Böhm M (2013) Global patterns of freshwater species diversity, threat and endemism. Glob Ecol Biogeogr 23:40–51

Colombo AF, Joly CA (2010) Brazilian Atlantic Forest lato sensu: the most ancient Brazilian forest, and a biodiversity hotspot, is highly threatened by climate change. Braz J Biol 70:697–708

Copertino MS, Garcia AM, Muelbert JH, Garcia CAE (2010) Introduction to the special issue on climate change and Brazilian coastal zone. Pan-Am J Aquat Sc 5:1–8

Crouzeilles R, Beyer HL, Mills M, Grelle CEV, Possingham HP (2015) Incorporating habitat availability into systematic planning for restoration: a species-specific approach for Atlantic Forest mammals. Divers Distrib 21:1027–1037

Cupertino-Eisenlohr MA, Vinícius-Silva R, Meireles LD, Eisenlohr PV, Meira-Neto JAA, Santos-Gonçalves AP (2017) Stability or break down under climate change? A key group of woody bamboos will find suitable areas in its richness center. Biodivers Conserv 26:1845–1861

Dee DP, Uppala S, Simmons A, Berrisford P, Poli P, Kobayashi S, Andrae U, Balmaseda M, Balsamo G, Bauer DP, Bechtold P, Beljaars ACM, Berg L, Bidlot J, Bormann N, Delsol C, Dragani R, Fuentes M, Geer AJ, Haimberger L, Healy SB, Hersbach H, Hólm EV, Isaksen L, Kållberg P, Köhler M, Matricardi M, McNally AP, Monge-Sanz BM, Morcrette JJ, Park BK, Peubey C, Rosnay P, Tavolato C, Thèpaut JN, Vitart F (2011) The ERA-interim reanalysis: configuration and performance of the data assimilation system. Q J Royal Meteorol Soc 137:553–597

Dobrovolski R, Loyola R, Rattis L, Gouveia SF, Cardoso D, Santos-Silva R, Gonçalves-Souza D, Bini LM, Diniz-Filho JAF (2018) Science and democracy must orientate Brazil's path to sustainability. Perspect Ecol Conserv 16:121–124

Duguma LA, Wambugu SW, Minang PA, van Noordwijk M (2014) A systematic analysis of enabling conditions for synergy between climate change mitigation and adaptation measures in developing countries. Environ Sci Pol 42:138–148

Esser LF, Neves DM, Jarenkow JA (2019) Habitat-specific impacts of climate change in the Mata Atlântica biodiversity hotspot. Divers Distrib 25:1846–1856

Faleiro FV, Nemésio A, Loyola R (2018) Climate change likely to reduce orchid bee abundance even in climatic suitable sites. Glob Change Biol 24:2272–2283

Fernandes GW, Barbosa NPU, Alberton B, Barbieri A, Dirzo R, Goulart F, Guerra TJ, Morellato LPC, Solar RRC (2018) The deadly route to collapse and the uncertain fate of Brazilian rupestrian grasslands. Biodivers Conserv 27:2587–2603

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

Forero-Medina G, Terborgh J, Socolar SJ, Pimm SL (2011) Elevational ranges of birds on a tropical montane gradient lag behind warming temperatures. PLoS One 6:e28535

Françoso E, Zuntini AR, Arias MC (2019) Combining phylogeography and future climate change for conservation of *Bombus morio* and *B. pauloensis* (Hymenoptera: Apidae). J. Insect Conserv 23:63–73

Giannini TC, Acosta AL, Garófalo CA, Saraiva AM, Alves dos Santos I, Imperatriz-Fonseca VL (2012) Pollination services at risk: bee habitats will decrease owing to climate change in Brazil. Ecol Model 244:127–131

Giannini TC, Tambosi LR, Acosta AL, Jaffé R, Saraiva AM, Imperatriz-Fonseca VL, Metzger JP (2015) Safeguarding ecosystem services: a methodological framework to buffer the joint effect of habitat configuration and climate change. PLoS One 10:e0129225

Godoy MDP, Lacerda LD (2015) Mangroves response to climate change: a review of recent findings on mangrove extension and distribution. An Acad Bras Cienc 87:651–667

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

Harvey CA, Chacon M, Donatti CI, Garen E, Hannah L, Andrade A, Bede L, Brown D, Calle A, Chara J, Clement C, Gray E, Hoang MH, Minang P, Rodríguez AM, Seeberg-Elverfeldt C, Semroc B, Shames S, Smukler S, Somarriba E, Torquebiau E, Etten J, Wollenberg E (2014) Climate-smart landscapes: opportunities and challenges for integrating adaptation and mitigation in tropical agriculture. Conserv Lett 7:77–90

Hoffmann D, Vasconcelos MF, Martins RP (2015) How climate change can affect the distribution range and conservation status of an endemic bird from the highlands of eastern Brazil: the case of the gray-backed tachuri, *Polystictus superciliaris* (Aves, Tyrannidae). Biota Neotropica 15:20130075

Huffman GJ, Adler RF, Bolvin DT, Nelkin EJ (2014) The TRMM multi-satellite precipitation analysis (TMPA). In: Gebremichael M, Hossain F (eds) Satellite rainfall applications for surface hydrology. Springer, Dordrecht/Heindelberg/London/New York, pp 3–22

Ingalls ML, Dwyer MB (2016) Missing the forest for the trees? Navigating the trade-offs between mitigation and adaptation under REDD. Clim Change 136:353–366

IPCC (2013) Summary for Policymakers. In: Stocker TF, Qin D, Plattner GK, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM (eds.) Climate change 2013: the physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge

Joly C, Metzger JP, Tabarelli M (2014) Experiences from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. New Phytol 204:459–473

Kasecker TP, Ramos-Neto MB, Silva JMC, Scarano FR (2018) Ecosystem-based adaptation to climate change: defining hotspot municipalities for policy design and implementation in Brazil. Mitig Adapt Strat Gl 23:981–993

Knutti R, Sedláček J (2012) Robustness and uncertainties in the new CMIP5 climate model projections. Nat Clim Chang 3:369–373

La Sorte FA, Jetz W (2010) Projected range contractions of montane biodiversity under global warming. Proc R Soc B 277:3401–3410

Lemes P, Loyola RD (2013) Accommodating species climate-forced dispersal and uncertainties in spatial conservation planning. PLoS One 8:e54323

Lemes P, Melo AS, Loyola RD (2014) Climate change threatens protected areas of the Atlantic Forest. Biodivers Conserv 23:357

Lima AA, Ribeiro MC, Grelle CEV, Pinto MP (2019) Impacts of climate changes on spatio-temporal diversity patterns of Atlantic Forest primates. Perspect Ecol Conserv 17:50–56

Locatelli B, EvansV WA, Andrade A, Vignola R (2011) Forest and climate change in Latin America: linking adaptation and mitigation. Forests 2:431–450

Locatelli B, Pavageau C, Pramova E, Di Gregorio M (2015) Integrating climate change mitigation and adaptation in agriculture and forestry: opportunities and trade-offs. WIREs Climate Change 6:585–598

Lourenço-de-Moraes R, Lansac-Toha FM, Schwind LTF, Arrieira RL, Rosa RR, Terribile LC, Lemes P, Rangel TF, Diniz-Filho JAF, Bastos RP, Bailly D (2019) Climate change will decrease the range size of snake species under negligible protection in the Brazilian Atlantic Forest hotspot. Sci Rep 9:1–14

Loyola R (2014) Brazil cannot risk its environmental leadership. Divers Distrib 20:1365–1367

Loyola RD, Lemes P, Brum FT, Provete DB, Duarte LDS (2014) Clade-specific consequences of climate change to amphibians in Atlantic Forest protected areas. Ecography 37:65–72

Marini MA, Barbet-Massin M, Martinez J, Prestes NP, Jiguet F (2010) Applying ecological niche modelling to plan conservation actions for the red-spectacled amazon (*Amazona pretrei*). Biol Conserv 143:102–112

Mastrandrea MD, Field CB, Stocker TF, Edenhofer O, Ebi KL, Frame DJ, Held H, Kriegler E, Mach KJ, Matschoss PR, Plattner G-K, Yohe GW, Zwiers FW (2010) Guidance note for lead authors of the IPCC fifth assessment report on consistent treatment of uncertainties. Intergovernmental Panel on Climate Change (IPCC). Available at https://wg1.ipcc.ch

Melo ABC, Cavalcanti IFA, Souza PP (2009) Zona de Convergência Intertropical do Atlântico. In: Cavalcanti IFA, Ferreira NJ, Silva MJ, Dias MAF (eds) Tempo e clima no Brasil. Oficina de Textos, São Paulo

Meyer ALS, Pie MR, Passos FC (2014) Assessing the exposure of lion Tamarins (*Leontopithecus* spp.) to future climate change. Am J Primatol 76:551–562

Nogueira EM, Yanai AM, Vasconcelos SS, Graça PMLA, Fearnside PM (2018) Carbon stocks and losses to deforestation in protected areas in Brazilian Amazonia. Reg Environ Chang 18:261–270

Nori J, Urbina-Cardona JN, Loyola RD, Lescano JN, Leynaud GC (2011) Climate change and American bullfrog invasion: what could we expect in South America? PLoS One 6:e25718

Oliveira IS, Rödder D, Capinha C, Ahmadzadeh F, Oliveira AKC, Toledo LF (2016) Assessing future habitat availability for coastal lowland anurans in the Brazilian Atlantic rainforest. Stud Neotrop Fauna Environ 51:45–55

Öztürk M, Hakeem KR, Faridah-Hanum I, Efe R (eds) (2016) Climate change impacts on high-altitude ecosystems. Springer, Cham

Pant LP, Adhikari B, Bhattarai KK (2015) Adaptive transition for transformations to sustainability in developing countries. Curr Opin Env Sust 14:206–212

PBMC (2014) Mudanças Ambientais de Curto e Longo Prazo: projeções e atribuição. In: Ambrizzi T, Araujo M (eds) Base científica das mudanças climáticas. Contribuição do Grupo de Trabalho 1 do Painel Brasileiro de Mudanças Climáticas ao Primeiro Relatório da Avaliação Nacional sobre Mudanças Climáticas. COPPE, Rio de Janeiro, pp 320–346

Pinheiro RS, Mellinger L, Scarano FR (2014) Os povos da Mata Atlântica. In: Scarano FR (ed) Mata Atlântica: uma história do futuro. Edições de Janeiro, Rio de Janeiro

Pires APF, Rezende CL, Assad ED, Loyola R, Scarano FR (2017) Forest restoration can increase the Rio Doce watershed resilience. Perspect Ecol Conserv 15:187–193

Renaud FG, Nehren U, Sudmeier-Rieux K, Estrella M (2016) Developments and opportunities for ecosystem-based disaster risk reduction and climate change adaptation. In: Renaud FG, Sudmeier-Rieux K, Estrella M, Nehren U (eds) Ecosystem-based disaster risk reduction and adaptation in practice. Springer, Cham, pp 1–20

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Strassburg BBN, Tabarelli M, Fonseca GA, Mittermeier RA (2018a) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:208–214

Rezende CL, Fraga JS, Sessa JC, de Souza GVP, Assad ED, Scarano FR (2018b) Land use policy as a driver for climate change adaptation: a case in the domain of the Brazilian Atlantic forest. Land Use Policy 72:563–569

Ribeiro MC, Martensen AC, Metzger JP, Tabarelli M, Scarano FR, Fortin MJ (2011) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In: Zachos FE, Habel JC (eds) Biodiversity hotspots. Springer, Heidelberg, pp 405–434

Ribeiro BR, Martins E, Martinelli G, Loyola R (2018) The effectiveness of protected areas and indigenous lands in representing threatened plant species in Brazil. Rodriguésia 69:1539–1546

Roland FA, Huszar VLM, Farjalla VF, Enrich-Prast A, Amado AM, Ometto JPHB (2012) Climate change in Brazil: perspective on the biogeochemistry of inland waters. Braz J Biol 72:709–722

Salazar LF, Nobre CA, Oyama MD (2007) Climate change consequences on the biome distribution in tropical South America. Geophys Res Lett 34:L09708

Scarano FR (2002) Structure, function and floristic relationships of plant communities in stressful habitats marginal to the Brazilian Atlantic rain forest. Ann Bot 90:517–524

Scarano FR (2009) Plant communities at the periphery of the Atlantic rain forest: rare-species bias and its risks for conservation. Biol Conserv 142:1201–1208

Scarano FR (2017) Ecosystem-based adaptation to climate change: concept, scalability and a role for conservation science. Perspect. Ecol. Conserv. 15:65–73

Scarano FR (2019) Biodiversity sector: risks of temperature increase to biodiversity and ecosystems. In: Nobre CA, Marengo JA, Soares WR (eds) Climate change risks in Brazil. Springer Nature, Cham, pp 131–141

Scarano FR, Ceotto P (2015) Brazilian Atlantic forest: impact, vulnerability and adaptation to climate change. Biodivers Conserv 24:2319–2331

Scarano FR, Ceotto P, Martinelli G (2016) Climate change and “campos de altitude”: forecasts, knowledge and action gaps in Brazil. Oecologia Australis 20:1–6

Silva DP, Dias AC, Lecci LS, Simião-Ferreira J (2019) Potential effects of future climate changes on Brazilian cool-adapted stoneflies (Insecta: Plecoptera). Neotrop Entomol 48:57–70

Sossai MF, Platais GH, Machado JA Jr, Novelli FZ, Beloti T (2016) Forest landscape restoration in the state of Espírito Santo – Brazil: from pilot projects to state policy for large-scale restoration. In: Moraes MA (ed) Forest landscape restoration in Brazil. IUCN, Brasília, pp 205–229

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

Strassburg BBN, Beyer HL, Crouzeilles R, Iribarrem A, Barros F, Siqueira MF, Sanchez-Tapia A, Balmford A, Sansevero JBB, Brancalion PHS, Broadbent EN, Chazdon RL, Oliveira Filho A, Gardner T, Gordon A, Latawiec AE, Loyola R, Metzger JP, Mills M, Possingham HP, Rodrigues RR, Scaramuzza CA, Scarano FR, Tambosi L, Uriarte M (2019) Strategic approaches to restoring ecosystems can triple conservation gains and halve costs. Nat Ecol Evol 3:62–70

Taylor KE (2001) Summarizing multiple aspects of model performance in a single diagram. J Geophys Res Atmos 106:7183–7192

Taylor KE, Stouffer RJ, Meehl GAA (2012) Summary of the CMIP5 experiment design. Bull Am Meteorol Soc 93:485–498

Thornton TF, Comberti C (2017) Synergies and trade-offs between adaptation, mitigation and development. Clim Chang 140:5–18

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

Van Vuuren DP, Edmonds J, Kainuma M, Riahi K, Thomson A, Hibbard K, Hurtt GC, Kram T, Krey V, Lamarque J, Masui T, Meinshausen M, Nakicenovic N, Smith SJ, Rose SK (2011) The representative concentration pathways: an overview. Clim Chang 109:5–31

Vasconcelos TS, Nascimento BTM (2016) Potential climate driven impacts on the distribution of generalist treefrogs in South America. Herpetologica 72:23–31

Vasconcelos TS, Prado VHM (2019) Climate change and opposing spatial conservation priorities for anuran protection in the Brazilian hotspots. J Nat Conserv 49:118–124

Vasconcelos TS, Antonelli CP, Napoli MF (2017) Mutualism influences species distribution predictions for a bromeliad-breeding anuran under climate change. Austral Ecol 42:869–877

Vasconcelos TS, Nascimento BTM, Prado VHM (2018) Expected impacts of climate change threaten the anuran diversity in the Brazilian hotspots. Ecol Evol 8:7894–7906

Veiga SF, Nobre P, Giarolla E, Capistrano V, Baptista M, Marquez AL, Figueroa SN, Bonatti JP, Kubota P, Nobre CA (2019) The Brazilian earth system model ocean–atmosphere (BESM-OA) version 2.5: evaluation of its CMIP5 historical simulation. Geosci Model Dev 12:1613–1642

Walker W, Baccini A, Schwartzman S, Ríos S, Oliveira-Miranda MA, Augusto C, Ruiz MR, Arrasco CS, Ricardo B, Smith R, Meyer C, Jintiach JC, Campos EV (2014) Forest carbon in Amazonia: the unrecognized contribution of indigenous territories and protected natural areas. Carbon Manag 5-6:479–485

Yin L, Rong F, Shevliakova E, Dickinson RE (2013) How well can CMIP5 simulate precipitation and its controlling processes over tropical South America? Clim Dyn 41:3127–3143

Young CEF, Medeiros R (eds) (2018) Quanto vale o verde: a importância econômica das unidades de conservação brasileiras. Conservação Internacional, Rio de Janeiro

Zwiener VP, Lira-Noriega A, Grady CJ, Padial AA, Vitule JRS (2017a) Climate change as a driver of biotic homogenization of woody plants in the Atlantic Forest. Glob Ecol Biogeogr 27:298–309

Zwiener VP, Padial AA, Marques MCM, Falero FV, Loyola RD, Peterson AT (2017b) Planning for conservation and restoration under climate and land use change in the Brazilian Atlantic Forest. Divers Distrib 23:955–966

# Chapter 13
# Non-native Species Introductions, Invasions, and Biotic Homogenization in the Atlantic Forest

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**Abstract** Anthropogenic introduction of non-native species has occurred since the first European colonization of South America in the 1500s, with the Atlantic Forest being the most heavily affected biome in the continent. Biological invasions, together with other anthropogenic pressures occurring over the subsequent 500 years, led to many biological changes such as biotic homogenization. In this chapter, we discuss patterns of non-native species introductions, highlight invasions or population explosions of problematic native species, and explore the phenomenon of biotic homogenization in the Atlantic Forest, Brazil. We explore examples related to the effect of recent introductions of non-native species, highlighting the loss of native biodiversity (e.g., rare, specialist, and endemic) and the proliferation of human-mediated non-native species of economic importance (e.g., genrealist, common and widely distributed). We also discuss the role of society and policymakers in developing policies of public interest. Finally, we discuss how raising

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awareness of the negative effects of invasive non-native species will contribute to inform management policies and provoke more in-depth research, resulting in greater protection and sound management strategy for the Atlantic Forest.

**Keywords** Human-mediated species introductions · Alien species · Hot spot of biodiversity · Biotic changes

## 13.1 Introduction

In South America, the Atlantic Forest (hereafter AF) was the first Brazilian biome to experience invasions by non-native species (hereafter NNS – see Box 13.1 for all definitions used here), with severe consequences including death and disease of both human and animal indigenous populations. Human-mediated introduction of invasive non-native species (hereafter INNS – see Box 13.1) has a long history in the AF, in both terrestrial and aquatic ecosystems.

The AF harbors ~70% of Brazil's population (Metzger 2009) in some of the largest cities of South America. In addition, AF hosts many urban and built ecosystems (including cities, plantations, and reservoirs) with large numbers of NNS (e.g., Bezerra et al. 2019). Substantial human population density and other anthropogenic pressures linked to biological invasions mean that the AF biome is at high risk from the spread and establishment of NNS and so requires substantial research and management.

Where possible, the introduction of NNS should be always avoided, and the adoption of the precautionary principle for species introductions has been recommended by the Convention on Biological Diversity and the Rio Declaration on

**Box 13.1 Concepts and Definitions**

We recognize that terms and definitions relating to invasion biology are still intensely debated. Often, it is difficult to clearly define the natural range of a species and therefore whether it is native to a certain area. Many cryptic species, for example, create this difficulty (see Jarić et al. (2019) for an extended discussion on cryptic species and biological invasions). Furthermore, there is uncertainty in species definitions throughout the invasion process including at what stage a species is described as introduced, invasive, or established and whether their negative or positive impacts are considered (see Essl et al. (2018) for more details). Developing robust, replicable, and consistent definitions of non-native, exotic, or alien species that serve scientific and policy purposes is of critical importance.

We use the following definitions for the present chapter: (i) non-native species – NNS: species introduced by humans (intentionally or accidentally) into areas in which they do not naturally occur, it may also be referred to as exotic, alien, or nonindigenous species. A NNS is considered to be established once a viable breeding populations is formed – non-native individuals may be introduced but fail to survive and reproduce so may not become established, (ii) invasive non-native species – INNS: species introduced by humans (intentionally or accidentally) into areas in which they do not naturally occur and that have any recognizable negative impact, (iii) negative impacts: significant detrimental modification of a natural property or process, (iv) colonization pressure: a measure of the number of NNS arriving in an area, (v) propagule pressure: a measure of the number of individuals (or proxies such as number of records) of a NNS arriving in an area, and (vi) biotic homogenization (BH): an increase in the similarity among communities through time. For further detail on terminology, see Lockwood et al. (2009), Richardson (2011), and Simberloff and Rejmánek (2011). In several analyses in this chapter, we have chosen to adopt a more conservative definition, assuming all species as NNS. As explained above, we recognize the limitations of the term and the fact that many NNS may be INNS. This is largely due to the scale of analysis we have chosen to use and the fact that the NNS vs INNS distinction is complex.

Environment and Development (UN 1992). In the vast and diverse biome of the AF, the precautionary principle seems justified as the most cost-effective measure. There are few examples of successful eradication of INNS and no examples for the AF. Furthermore, once NNS are introduced into a new environment, mitigation measures are required to control the spread of the species. If control is not possible, INNS still need to be monitored in the long term to avoid negative impacts. The cost of such approaches globally is extensive. The USA spends US$120,000 million per year trying to control the spread of 50,000 INNS (Pimentel et al. 2005). Applying a similar level of control to mitigate the potential events of invasion by the ~7000 occurence records of NNS in the AF would cost US$17,000 million per year.

In addition to mitigation, prevention of introductions through awareness-raising programs can be successful, particularly in urban areas. Raising awareness may also increase management efficiency of species introduced for commercial purposes (e.g., farming or aquaculture) and highlight the need for effective biosecurity to prevent accidental escapes from captivity. For example, some INNS of fish (e.g., carp and tilapia) are so widespread in the AF that many people consider them "native," despite all the information available about their invasive potential and impacts associated with their introduction (Bezerra et al. 2019). The same is true for other organisms, such as ornamental plants and domestic pets (e.g., Humair et al. 2014; Patoka et al. 2018). The AF comprises a large and speciose biome that has experienced extensive and long-term detrimental effects as a result of the introduction of varied INNS (Myers et al. 2000). Such detrimental impacts of INNS have been shown to lead to reduced productivity and human well-being (Bacher et al. 2017), which disproportionately affect areas of high diversity (Early et al. 2016). In the AF, Brazil's most populated biome, NNS have the potential to create substantial detrimental ecological and socioeconomic impact. Despite the recognition of these impacts, there is a paucity of data regarding mitigation or control effectiveness, and more research is required to inform policy regarding NNS.

We organize this chapter by describing trends and gaps about introductions and patterns of NNS in the AF with quantitative analysis. We then focus on topics relevant to the authors' expertise highlighting invasions by discussing specific examples from the AF and population blooms of some problematic native species, referred to as native invaders. After which, we explore biotic homogenization (hereafter BH) in AF, which is largely related to invasions, but such ecological phenomenon is a very complex process and, at least in theory, can be independent of non-native introductions or invasions. We conclude with a discussion about conservation and management of biological invasions into the AF, including public policies.

## 13.2 Non-native Species Trends and Patterns

To investigate patterns of NNS (see Box 13.1 for differentiating NNS from INNS) occurrence and reporting, we performed quantitative analyses of NNS records from the AF. We used NNS data from the "I3N Brasil" database (I3N Brasil 2019) for the AF to identify spatiotemporal patterns related to colonization and propagule pressure of NNS (see Box 13.1 for definitions). We retrieved 395 NNS and 6959 occurrence records in the AF from the database (for detailed inclusion criteria, see I3N Brasil (2019), and for definitions, see Box 13.1).

In the AF, most NNS records were of plant and fish species, accounting for 178 and 105 species, respectively, followed by crustaceans, mammals, and molluscs with 22, 20, and 12 species, respectively. The number of occurrences (i.e., number

**Box 13.2 Invasive Mosquitoes**

Public health problems related to INNS are largely neglected (Mazza et al. 2014) but of great importance in a biome as densely populated as the AF. Insects, and especially mosquitoes, act as vectors for human diseases (Juliano and Lounibos 2005). Emerging infectious diseases such as dengue, chikungunya, and Zika have been transmitted by *Aedes aegypti* and *A. albopictus*. These mosquitoes are native to Africa and Asia and have been responsible for the accelerated geographical spread of these viruses in urban and suburban areas within the AF. The first outbreak of the Zika virus in the Americas (AF region of Northeast Brazil) occurred in 2015 (Campos et al. 2015).

Understanding the role of biological invasions and new emerging ecological interactions in these disease vector species is likely to be of critical importance in managing the public health issues. All three diseases are caused by the association of INNS of mosquitoes and viruses. It is also known that positive interactions among INNS may cause synergistic impacts to the natural communities and eventually lead to a process called “invasional meltdown” (Simberloff and Von Holle 1999). There are many examples where species from distinct localities when together engage in facilitative interactions in ways that increase their likelihood of establishment and spread (Braga et al. 2018). Genetic alterations of chikungunya allowed it to replicate efficiently in a new vector, *A. albopictus*, facilitating the spread of the virus worldwide (Enserink 2014). The interaction of the viruses and the mosquitoes (*A. aegypti* and *A. albopictus*) appears to be a case of invasional meltdown. The viruses’ outbreaks are associated with the introduction and spread of the mosquitos as it only occurred after the genetic mutation mentioned. It fits the invasional meltdown hypothesis as their interaction can be synergistic, that is, the impact of the virus and the mosquito together is much higher than summed impact of their presence alone.

of times NNS have been recorded for any given location at any given time) demonstrates that a few species constitute the vast majority of individual records. Insects had nine species with 1195 records, birds had nine species with 375 records, and mammals had 24 species with 404 records in the AF. The remaining 16 species, not on the previous cited groups, are from different taxonomic groups. Insects are a clear example of major bias in terms of taxon recording (also previously shown by Pyšek et al. (2008) and Frehse et al. (2016)), with *Aedes* spp. (747 records) and *Apis mellifera* (406 records) accounting for the majority of records (see Box 13.2 for more information on invasion by *Aedes* and human health problems related to this bias). Although NNS occurrence records in AF are high (6959), two species (*Aedes* spp. and *A. mellifera*), out of 395, account for 16.6% of records. Both species are abundant in populated areas and therefore are more easily recorded. For other taxonomic groups, most recorded species are also easily seen in urban areas. That is the case for plants (*Tecoma stans* was recorded 318 times) and birds (*Passer domesticus* was recorded 211 times). These biases (toward few species and urban areas) in data recording illustrate the paucity of data available for the vast majority of species. Pristine or sensitive areas such as protected areas may experience even more dramatic recording biases (see Sect. 13.2.1 for further discussion of protected areas).

Temporal trends of NNS records exhibited an exponential increase in all taxonomic groups over the last ca. 70 years (Fig. 13.1). There was a rapid increase of records in the early 2000s for all major groups (i.e., plants, fish, birds, mammals, reptiles, and insects). Overall, in the AF, the accumulation of NNS did not plateau and still seems to be increasing, which is in line with global temporal trends (Seebens et al. 2017). However, for AF, the major increase in NNS records occurred later than the global pattern (Seebens et al. 2017) and may reflect delayed awareness or recording of NNS. Invasion biology research only emerged in Brazil in the last 20 years (Frehse et al. 2016). As there is generally a time lag of several years between field observations and publication, we do not yet have data for the most recent 4 years.

As a general pattern, occurrence records are not expected to plateau as no effort for NNS management and control has been effectively implemented in the AF, and with time, more researchers are expected to study NNS. However, new NNS species records increase at different rates for each taxon (Fig. 13.1). For the analyzed dataset, new NNS of plants and fish are recorded at a higher rate than birds and mammals. At first, this pattern can arise due to the number of described species globally for each group. More speciose groups (plants and fish) are more probable to be introduced than less speciose groups (birds and mammals) simply by chance. Additionally, plants and fish are frequently introduced unintentionally, whereas mammals and birds are less expected to be introduced accidentally (e.g., hitchhikers), and this difference can lead to a much higher number of plants and fish being constantly introduced. Reptile introductions are expected to follow the same pattern as birds and mammals due to the lower relative number of species described for the group. However, one or few recent introductions of NNS may cause a dramatic increase in occurrence records as it is proportionally more significative with the low number of NNS. These results from the I3N Brasil database show that fish and plants should be prioritized in actions focused on prevention as their number is increasing more rapidly in the AF.

We analyzed the number of records in each of the AF-registered cities as a proxy for propagule pressure in the AF (Fig. 13.2). Analyzing information on NNS richness as a proxy for colonization pressure for each federal unit, we demonstrate that more NNS occur near densely populated areas (Fig. 13.2). This pattern varies according to taxonomic group; for example, there are more NNS of plants near Curitiba and more NNS of vertebrates near Rio de Janeiro. Indeed, NNS are not occurring only in densely populated areas, but these may increase propagule and colonization pressure to natural adjacent areas.

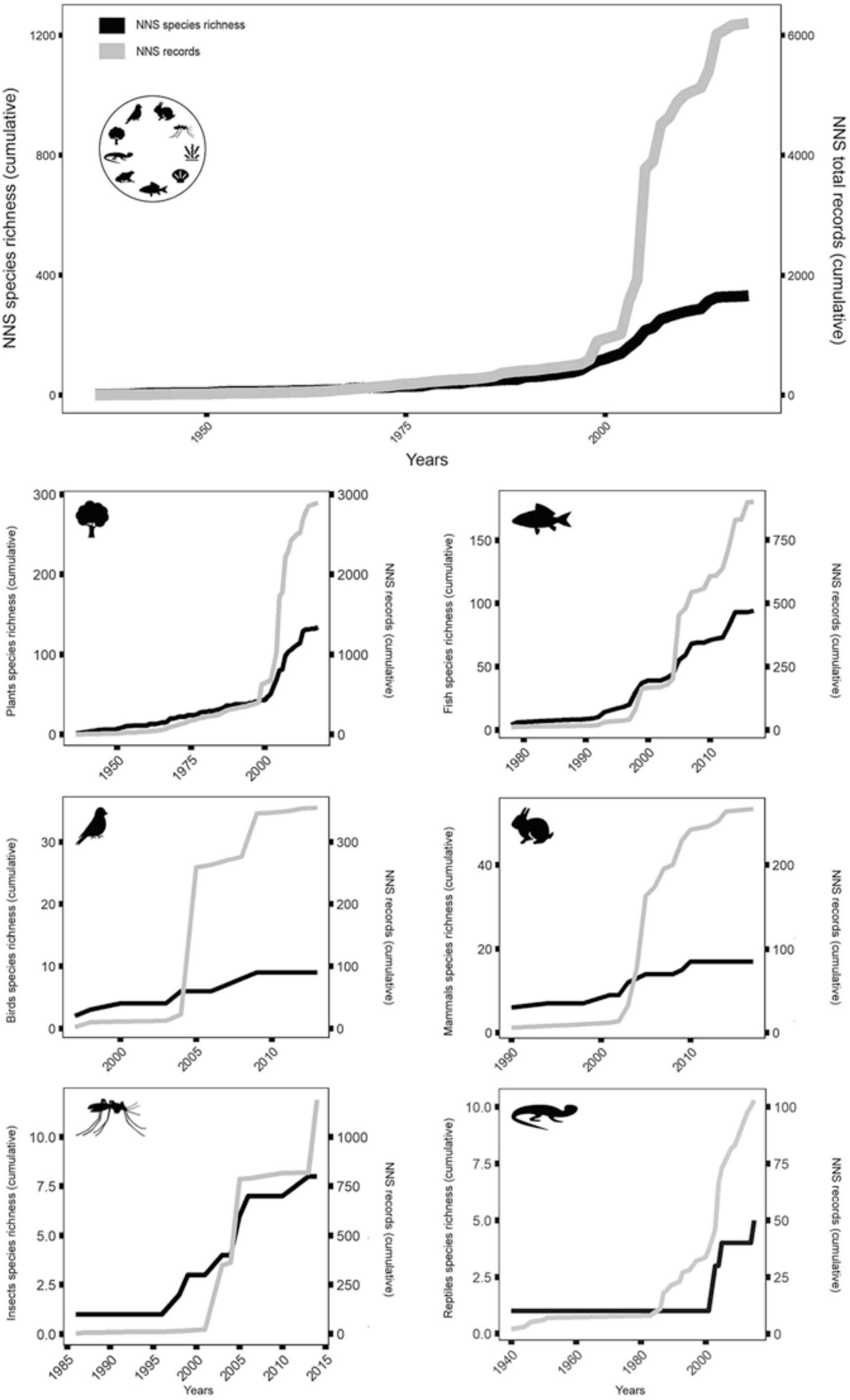

**Fig. 13.1** Cumulative curves of first recorded occurrence (black lines), i.e., NNS richness and colonization trends/propagule pressure (gray lines) of NNS in AF. Total first recorded occurrence and colonization trends along the time (1930–2018) are shown in the top chart; after that, we presented only the most representative groups with complete information (e.g., location and year). Time period varies for each group according with the first record for the group (Data source I3N Brasil database – excluding records without year)

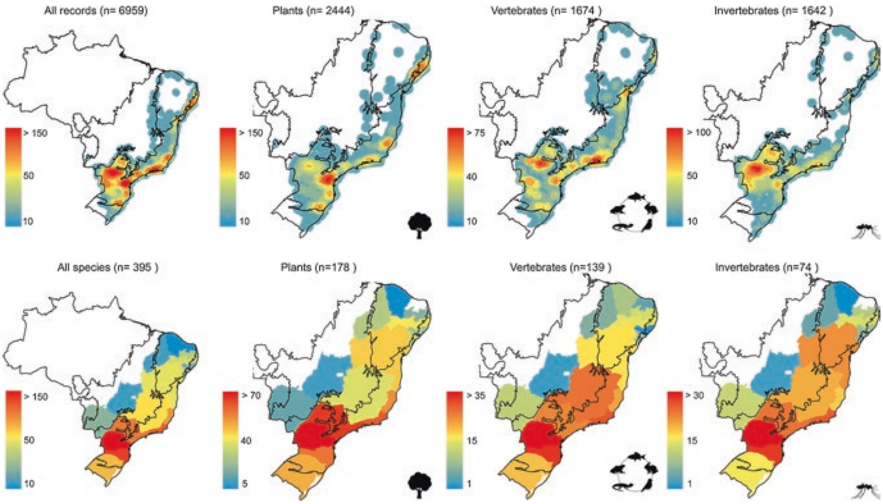

**Fig. 13.2** Geographical patterns of total and main group occurrence records of NNS in AF (top figures) and total and main group NNS richness for each Brazilian state within AF (below figures)

South and Southeastern Brazil have experienced a substantial number of invasions. Sampling bias may impact the number of reported invasions; Frehse et al. (2016) found far fewer studies in the state of Santa Catarina than in São Paulo and Rio de Janeiro. The data from the I3N Brasil database was mostly compiled by researchers based in Santa Catarina. Further work to integrate state and institutional NNS lists from across Brazil would substantially improve the robustness of the data and usefulness of the database.

### *13.2.1 NNS in Protected Areas: An Important Knowledge Gap for the AF*

Protected areas that exclude human activities constitute 2% of the AF territory with an average size of ~10,000 ha (Scarano and Ceotto 2015). Brazilian protected areas are designated to protect important biodiversity; however, the lack of management strategy for NNS means that their impact in protected areas is largely unknown (Ziller and Dechoum 2014), and NNS are thought to be underestimated (Sampaio and Schmidt 2014). A recent assessment of NNS also highlighted the lack of management plans in biodiversity protected areas (Guimarães and Schmidt 2017).

Rosa et al. (2017) found that feral dogs were the main INNS predators threatening wildlife conservation in protected areas, mainly in AF. *Canis familiaris* (domestic dog) are predators of native species (Paschoal et al. 2012, 2016, 2018; Silva et al. 2018) and act as a vector of pathogens to other carnivorous species (Srbek-Araujo and Chiarello 2008; Lessa et al. 2016). The AF is currently the biome with highest level of domestic dog invasion (Lessa et al. 2016; Rosa et al. 2017).

Examples of INNS such as *Callithrix jacchus* (marmoset monkey) (Silva et al. 2018) and *Lepus europaeus* (European hare) (Faria et al. 2016) negatively affect native species through competition and predation. The introduction of domestic mammals could be related to the expansion of urban and rural areas near PAs, increasing NNS occurrence and establishment (Paschoal et al. 2012; Faria et al. 2016; Lessa et al. 2016).

The poor management and low awareness of the risks and negative impacts of INNS hamper ongoing efforts to conserve biodiversity and negatively affect the ecosystem services provided by the AF (Travassos et al. 2018; Azevedo-Santos et al. 2019).

### *13.2.2 Hot Spots of NNS in AF Estuary Ecosystems*

Despite the high diversity of NNS recorded in estuaries (Ruiz et al. 1997), occurrence is likely underestimated due to the lack of available integrated multi-taxa studies. Estuaries are highly connected with other aquatic ecosystems, increasing the dispersal of NNS to adjacent environments. Human modification of aquatic ecosystems (e.g., breakwater, ports, dredging) alters the natural eco-hydrology of the system (e.g., introducing or removing waterfalls, rapids, or tide regime), which may favor invading species over the native biota. Estuaries experience multiple stressors that may combine to increase the likelihood of colonization and establishment of NNS: high propagule pressure supported by the constant movement of ships, several types of artificial structures and available habitat (e.g., piers, sunken ships), aquaculture and farming of NNS, and changes in water quality through human usage (de Castro et al. 2017). Additionally, estuaries also act as “saline bridges” for freshwater INNS that can survive and disperse through saline waters in corridors between riverine and adjacent freshwater systems (Gutierre et al. 2013).

Estuaries are particularly relevant as hot spots of NNS occurrence in the AF due to high levels of human activities (e.g., shipping, tourism, aquaculture), which introduced and spread NNS through ballast water and hull fouling (Souza and Silva 2004; Molnar et al. 2008). In recent years, many and diverse aquatic NNS, including algae, anthozoans, bivalve molluscs, polychaetes, copepods, cladocerans, decapods, isopods, barnacles, entoprocts, ascidians, and fish, have been unintentionally introduced along the Brazilian coast (which is mostly within the AF) (I3N Brasil 2019; Occhi et al. *in prep.*). Estuaries associated with shipping, oil platforms, and other artificial structures have been particularly affected (Ferreira et al. 2008). However, most records are single time points, lacking long-term monitoring programs; thus, we can expect an increase in NNS novel occurrence records and negative impacts. In the AF, despite the existence of a considerable amount of evidence regarding the problems caused by anthropogenic interventions and NNS, there are few prevention or control strategies in place.

## 13.3 Selected Examples of INNS and Native Invaders of the AF

We highlight a few examples of problematic INNS in the AF, acknowledging that there are many other species that are also of major concern. However, we chose species according to the authors' expertise and based on the level of scientific data available. We consider that the selected examples are relevant to comprehend broader processes underlying biological invasions (including native invaders) and their impacts in the AF, noting some general characteristics and scenarios that may increase invasion risk.

### *13.3.1 Fish*

Competition for resources is one of the main ways in which INNS impact native species. Such ecological competition can locally reduce the functional diversity of freshwater fish species, through reduction in the abundance of native species (Vitule et al. 2012; Daga et al. 2015; Bezerra et al. 2019).

The Nile tilapia (*Oreochromis niloticus* Linnaeus, 1758) is a cichlid native to Africa and the Middle East and is one of the most widespread freshwater fish species. It occurs in at least 97 countries and has been introduced in 75 of them (Froese and Pauly 2019). The species is well-known for its high invasive potential and ecological impacts (Canonico et al. 2005). The Nile tilapia is omnivorous, and negative impacts of its introduction outside its natural range include – but are not restricted to – predation of eggs, larvae, and small fish of other species; competition for food, space, and spawning places; alteration of water quality and phytoplankton community; and, in Africa, hybridization with native species where conspecific species occur (Canonico et al. 2005). Tilapia species have also been reported to accelerate the BH or "benthification" processes in altered ecosystems (Bezerra et al. 2019).

Introduced piscivorous fish can also have a large impact on the natural food web; for example, the silver croaker *Plagioscion squamosissimus*, from the Amazonian region, has been introduced in at least three basins in AF: Doce River, Paraíba do Sul River in Southeastern Brazil, and Contas River, Bahia (Queiroz-Sousa et al. 2018). The introduction in the Doce River was likely due to aquaculture activities (Barros et al. 2012). In other Neotropical basins, this species has replaced the native *Pimelodus maculatus* in Tietê River basin and preys intensively on young *Hypophthalmus edentatus* (Queiroz-Sousa et al. 2018). In the Contas River, *Plagioscion squamosissimus* probably takes advantage of the high number of available prey species, demonstrating opportunistic feeding behavior (Santos et al. 2013). The colonization and spread over wide areas in reservoirs result from the reproductive strategy of *P. squamosissimus*, the production of small, pelagic, and buoyant eggs, spawned in several batches (Agostinho et al. 1999). In the Paraíba do

Sul River basin, this species is now established as one of the five most frequent and abundant during both wet and dry seasons (Santos et al. 2010).

### 13.3.2 Bullfrogs and Amphibian Diseases

The AF is home to an astonishing diversity of amphibians. There are about 700 amphibian species known to occur in this biome, of which 70% to 80%, depending on the reference, are endemic, corresponding to about 10% of global anuran diversity (Haddad et al. 2013; Figueiredo et al. 2021). However, dozens of species are under threat and some have been declared extinct (Scheele et al. 2019). Therefore, identifying and understanding the causes of amphibian declines are urgently required. Besides anthropogenic habitat change, native amphibians are threatened by INNS, mainly the North American bullfrog (*Lithobates catesbeianus*) and the chytrid fungus (*Batrachochytrium dendrobatidis*).

In Brazil, ranaculture began in the 1930s, but its popularization increased around the 1970s (Ferreira et al. 2002), matching the most intense amphibian declines in the AF (Carvalho et al. 2017). Due to a lack of proper biosecurity, bullfrogs were introduced into wild environments, and nowadays, the species is spread over a large extent of the biome (Both et al. 2011). While it is not clear whether the spread of the bullfrog has directly caused the decline of native amphibians or only had a contributory effect, invasive bullfrogs may jeopardize native anuran acoustic communication (Forti et al. 2017; Medeiros et al. 2017), prey on native amphibian species (Toledo et al. 2007), and compete for resources (Kiesecker et al. 2001; Boone et al. 2004). Furthermore, bullfrogs might serve as chytrid reservoirs (Daszak et al. 2004) and an international vector of chytrid fungus (Schloegel et al. 2012; O'Hanlon et al. 2018).

Chytrid fungus has been linked to over 500 amphibian population declines and at least 90 species extinctions in recent times (Scheele et al. 2019). In the AF, the fungus can be endemic or invasive, as different strains have been found (Schloegel et al. 2012), and some had been present in the region for a long time (Rodriguez et al. 2014). One plausible hypothesis is that native AF anurans declined when they were infected with a novel chytrid strain, as witnessed in Central America (Lips et al. 2008). Different chytrid lineages vary in degree of infectiousness, but the strain related to global amphibian declines likely originated in East Asia and spread across other continents in the last 100 years (O'Hanlon et al. 2018).

### 13.3.3 Birds

Birds are a target of illegal traders and frequently escape or are released from captivity. This is the main vector of introduction of bird species worldwide (Carrete and Tella 2008). A single assessment has identified INNS of bird in Brazil (Fontoura

2013; but see Klemann-Junior et al. 2017); however, the impact of INNS birds in the AF is not well-documented. One reason for this is the strong linkage of the INNS distribution with habitat conversion in the AF. This can mask the potential impacts of INNS of bird, leading to the conclusion that the decline of native species comes only from habitat modification. The most frequently recorded INNS in the AF are associated with deforested areas (I3N Brasil 2019). For example, the house sparrow (*Passer domesticus*) and the rock dove (*Columba livia*) are strongly associated with urban areas and cattle egret (*Bubulcus ibis*) with pastures (Sick 2001). The cattle egret – native to Africa – is not considered an INNS because its arrival in Brazil is not viewed as mediated by humans (Sick 2001); however, this is a questionable inference (for a more detailed discussion, see Crosby (1972)). The arrival of the cattle egret in the American continent dates to the beginning of the last century. Initially, there was no signal that the species substantially affected native species; nonetheless, few studies or data focused on verifying impacts. However, its predatory behavior along with high-density populations means it has large potential to impact the trophic chain.

Evans et al. (2016) categorized NNS of bird based on levels of impact; for instance, the species *P. domesticus*, *C. livia*, and *B. ibis* were classified as moderate impact (which can cause decline in local native populations), but there are no existing studies reporting impacts of these species in the AF. Other examples include the common waxbill *Estrilda astrild* that was brought to Brazil by the pet industry, and several feral populations are now established throughout South America (Sick 2001; Klemann-Junior et al. 2017) and the turquoise-fronted parrot (*Amazona aestiva*), one of the most trafficked birds in Brazil (Vanstreels et al. 2010), which is spreading through the AF (Klemann-Junior et al. 2017).

### *13.3.4 Wild Boar*

In South America, the first records of wild boar (*Sus scrofa* L.) occurred in the early twentieth century with an introduction for commercial purpose in Argentina and Uruguay and subsequent escapes into areas of Chile and Brazil (Jaksic 1998; Deberdt and Scherer 2007). Currently, the wild boar occurs in almost all Brazilian regions (Pedrosa et al. 2015) and features in the potential for invasion extremely high in AF, Pantanal, and Cerrado and substantial risk of expansion to the Amazon region, as a result of the species being able to easily adapt to varying climatic and environmental conditions (Sales et al. 2017). Although there is concern about the potential harm it may cause, few studies quantify its real negative impacts.

In the transition area among AF and Pampa biomes, wild boar removed the vegetal cover in coastal areas, resulting in chemical and biological changes in the ground (Quintela et al. 2010). In the AF, Hegel and Marini (2013) quantified the negative impact of wild boar on different types of vegetation at the Aracuri-Esmeralda Ecological Station, showing that in about 20 days, the wild boar destroyed 56.2 kg of dry biomass of native vegetation, 94% of it belonging to the most conserved forests. In an analysis of presence and absence in south AF areas, Hegel (2017) showed that besides having a great preference in feeding on native forest (82%), the wild boar also showed great preference for *Pinus* spp. plantation (100%), corn (77%), and oat crops (71%). However, analyzing the occupation patterns by modeling of wild boar in the same areas, the models of Hegel et al. (2019b) showed that boars occurred in all types of vegetation in the landscape, corroborating the adaptability to the different environmental conditions indicated by Sales et al. (2017). Morais et al. (2019) showed a higher frequency of occurrence of wild boar in the AF at lower elevations, in forest areas more humid and warmer, and away from human disturbances.

Negative effect on native mammals' species by competition (probably for food and territory) and predation has been reported in AF (Hegel et al. 2019a). The co-occupancy models among the authors showed that in the presence of wild boar, the occurrence and richness of other native mammals are reduced. In a stomach content analysis with wild boar from AF, Campos Sulinos and Pantanal, Cervo (2017) recorded wild plants, fruits and seeds, native invertebrates and vertebrates, and sheep as wild boar diet items. The wild boar is also a vector of numerous diseases such as influenza, rabies, tuberculosis, or leptospirosis, which affect other animal species and humans (Rosell et al. 2001; De Oliveira et al. 2018).

The estimates of wild boar population densities in the AF and other Brazilian biomes are still little known (estimated between 0.22 and 22.3 individuals/km$^2$ (Oliveira 2012; Puertas 2015)), but it is known that the population is growing. Wild boar population in AF control occurs through predation of piglets and juveniles by pumas (*Puma concolor*) and possibly jaguars (*Panthera onca*) (Hegel and Marini 2018). Therefore, low rates of predation of wild boar have led countries such as Brazil to authorize its hunting by accredited controllers (IBAMA 2005, 2013) as a management strategy to reduce the population size (PAN 2017). Boars hunting combined with other control methods is recommended in AF, especially for protected areas (Rosa et al. 2018), but requires planning and monitoring by enforcement agencies to prevent native animals from being killed or injured by traps, as is already recorded (Carvalho et al. 2019).

### *13.3.5 When a Native Organism Becomes Invasive: Problems and Questions*

Under certain conditions, some native organisms can proliferate intensively and unexpectedly in the community, overwhelming other species in both abundance and in biomass, in a process similar to the invasion by exotic species, which is known as super-dominance (Pivello et al. 2018). However, the nature of the harm caused by any native organism and the reason why some are problematic lead to complex and idiosyncratic questions that can be answered only by persistent studies (Simberloff and Vitule 2014). However, the difficulty is to distinguish natural phenomena from invasion processes.

Species interactions and ecosystem processes of native populations can help us to understand demographic patterns of the invasion process. For example, after disturbance, pulses of population expansion of pioneer native species may occur but tend to return to initial conditions, following natural ecological succession. In this case, peaks of native population explosion are predicted. But in some abnormal context, a super-dominance of a native species may last for long periods and impact the whole community – food webs, pollination, seed dispersion, nutrient cycling, productivity, and other ecological processes – being able to drive other native species to local extinction. The ecosystem can change into a new alternative state, where the resistance of the super-dominant species may prevent natural regeneration or succession (Simberloff 2011; Simberloff et al. 2012).

Most studies associate the super-dominance phenomena with anthropogenic activities. In the Atlantic Forest, deforestation and habitat fragmentation, fire, pollution, the building of roads and dams, drainage disruption, land uses in the surroundings of the remnant fragments, and the introduction of exotic species as a result of these activities are usually the causes for breaking the natural equilibrium and causing the phenomenon of super-dominance of some native species (Pivello et al. 2018). Examples of super-dominant native plant species in the AF are mostly woody bamboos (genera *Chusquea*, *Merostachys*, *Aulonemia*, *Guadua*), lianas (genera *Mikania*, *Piptocarpha*, *Paullinia*), and ferns (*Gleichenia pectinata*). Among animals, there are also potential examples for birds (e.g., *Zenaida auriculata* – Klemann-Junior et al. 2017) and amphibians (e.g., *Phyllodytes luteolus* – Forti et al. 2017).

In the future, we expect land use intensification, increased pollution, and climate change, which will promote an increase in the invasiveness of both novel NNS and super-dominance of natives in the AF. In this sense, it is essential to identify not only the situations of super-dominance but also how such stressors interact to cause explosive population growth of a given native. From the moment these species are identified as super-dominants, important issues such as their management and long-term studies to evaluate the community behavior come to be considered, hence the importance of addressing this issue in discussions about invasive species.

Below, we explore the topic of biotic homogenization (BH) which is a process that has involved non-native introductions and biological invasions but that can be a much more complex and context-dependent process. At least in theory, BH can occur independently of non-native introductions, for example, through extinctions of native species or even by expansions of super-dominant native species in large scales.

## 13.4 Biotic Homogenization in AF

Human activities are dramatically threatening the Earth's biota, causing ecosystem modification and disruptions, species extirpations, and extinctions and increasingly favoring a deterministic group of overdominant native and INNS. This biotic upheaval has led to the simplification of ecological communities, caused primarily by a range expansion of few species, which are gradually dominating the community or replacing endemic native organisms, in a process named biotic homogenization, which is defined as an increase in the similarity among communities through time (McKinney and Lockwood 1999) or a decrease in beta diversity along time, and occurs at any level of organization, including genetic, taxonomic, phylogenetic, and functional attributes (Olden et al. 2004; Winter et al. 2009). At least in theory, BH can occur through the extinction or extirpation of native species or expansion of some overdominant native species. However, current research suggests that the majority of BH impacts result from introductions and INNS, in particular human-associated ones.

Although BH has become one of the most prominent and pressing topics in conservation biology (Olden et al. 2010), few studies have elucidated the complex interactions of the multiple ecological and environmental determinants of this process (Rahel 2000; Hermoso et al. 2012; Brice et al. 2017). In addition, several components of the BH process remain unclear, especially the dynamics at multiple spatial and temporal scales (Olden et al. 2018). To date, around 150 articles have quantified BH around the world, showing a global trend of increased taxonomic similarity across ecosystems and taxonomic groups (Baiser et al. 2012; Magurran et al. 2015; Toussaint et al. 2016). In contrast, little is known about the factors influencing BH in the AF, which has been investigated in seven published articles (Table 13.1), most focusing on freshwater and terrestrial habitats, mainly evaluating plants, invertebrates (zooplankton and ants), and vertebrates (freshwater fish). Most of the articles quantified taxonomic homogenization in the AF, mainly investigating large temporal (>10 years) and moderate spatial (<1000 km$^2$) scales. The majority of the articles were based on simple metrics, such as Jaccard and Sørensen indexes. The main determinant of BH was the introduction/extinction of species rather than habitat modification.

Additionally, some articles investigated the patterns of changes in community composition along the gradients of habitat fragmentation (Filgueiras et al. 2016; Thier and Wesenberg 2016) or human-modified landscapes (Siqueira et al. 2015; Vallejos et al. 2016). Although these articles did not quantify BH intrinsically, mainly due to not considering the temporal scale in the analyses, they provide indirect evidence to estimates of BH for plants and birds in AF (see Box 13.3 for more information). Also, a smaller number of articles assessed the effects of INNS in homogenizing AF assemblages, such as the tropical signal grass on native macrophytes in reservoirs and lakes (Michelan et al. 2010) and the bullfrog on native anuran communities in ponds (Both and Melo 2015). Moreover, some models provide a novel approach to investigate the drivers of future community changes such as BH (Zwiener et al. 2018).

**Table 13.1** Review of the published articles that quantified the BH in the AF

| Habitat | Taxonomic group | Type of BH | Temporal scale | Spatial grain | Spatial extent | Determinants of BH | Reference |
|---|---|---|---|---|---|---|---|
| Freshwater | Plants | Taxonomic | Large | Small | Provincial | Introduction/modification | Ceschin et al. (2018) |
| | Vertebrates | | | | | | |
| Freshwater | Invertebrates | Taxonomic | Small | Moderate | Local | Introduction/extinction | Lopes et al. (2017) |
| Freshwater | Vertebrates | Taxonomic | Small | Small | Provincial | Introduction/extinction | Daga et al. (2015) |
| | | | | | Regional | | |
| | | | | | Regional | | |
| | | | | | Regional | | |
| Freshwater | Vertebrates | Taxonomic | Large | Moderate | Regional | Introduction/extinction/ modification | Petesse and Petrere Jr (2012) |
| | | | Large | | | | |
| | | | Large | | | | |
| Freshwater | Vertebrates | Taxonomic | Large | Large | Provincial | Introduction/extinction/ modification | Vitule et al. (2012) |
| Terrestrial | Plants | Taxonomic | Large | Moderate | Provincial | Introduction/extinction | Lôbo et al. (2011) |
| Terrestrial | Invertebrates | Taxonomic | Large | Moderate | Provincial | Modification | Martello et al. (2018) |
| | | Functional | | | | | |

Spatial scale = small ($\leq$10 km$^2$), moderate (11–999 km$^2$), large (>1000 km$^2$)
Spatial extent = local (the smallest sample unit), regional (considered more than two locals), provincial (considered more than two regions)
[a] Temporal scale = small ($\leq$10 years), large (>10 years)

**Box 13.3 Birds and Biotic Homogenization (BH)**

Birds in the AF have to date seldom been assessed with a focus on INNS, but several studies have considered the impact of human activities on bird populations. Landscape changes have been found to exert a homogenizing effect on bird assemblages, particularly due to urbanization (Vallejos et al. 2016). Larger native vegetation patches have been shown to reduce the impacts of urbanization on birds (Enedino et al. 2018), suggesting that edge effects are important. In the southern portion of the AF, a recent review has identified several species that are recent colonizers and "native invaders," whose geographical distribution seems to be expanding along landscape changes caused by human activities, mainly through deforestation (Vallejos et al. 2016; Klemann-Junior et al. 2017). In the northern part of the AF, birds suffer a beta diversity loss in fragmented rural landscapes (Faria et al. 2007; Morante-Filho et al. 2015). These shifts are only evident at some spatial scales, and the homogenizing effects may be hard to identify on a coarser scale (Morante-Filho et al. 2016).

## 13.5 Future Steps and New Approaches to Biological Invasions in the AF

After describing trends, patterns, and particularities of biological invasions and BH into the AF, we suggest in this topic recent approaches, tools, and concerns toward better management and understand of the complex factors that involve the invasion process and conservation of AF.

### *13.5.1 INVASIVESNET, Citizen Science, Apps, and e-DNA*

We consider government incentives and partnerships between research groups acting in AF, and established international groups will ensure a rapid knowledge growth in the next decades. In this regard, some major recent projects such INVASIVESNET could integrate networks and increase interactions (e.g., Lucy et al. 2016). This new association will facilitate greater understanding and improved management of INNS and biological invasions globally, by developing a sustainable network for effective knowledge exchange. In turn, this will improve topics such as coordination, cooperation, and information exchange among scientists, management, the community of practice, and the public. In tandem with increased networking and collaboration, the rapid development of new technologies is accelerating the detection and monitoring of INNS across the globe.

Citizen science initiatives have the benefit of covering large areas that might not be possible from traditional survey techniques while also helping to raise awareness of the impacts and management of INNS in localized areas among a wider audience than immediate stakeholders. Increasing uptake in smartphone technology across the globe has led to a substantial number of projects employing smartphone applications for the recording and reporting of INNS. The use of apps is particularly fitting where there already exists a target citizen audience that is invested in recording species distributions (e.g., natural historians, bird watchers, or anglers), and previous studies have suggested that a key factor in the success of such citizen science projects is the engagement of large numbers of recorders and sufficient training or feedback to ensure the submitted records are accurate and consistent (Jepson and Ladle 2015). Once again, the need for collaboration and effective communication is clear – between scientists, ecologists, habitat managers, and citizen scientists. Future developments are investigating the use of machine learning for automatic species identification from photographs without the need for expert identification (e.g., Kress et al. 2018).

Further technological developments relevant to the detection of INNS include the advance in sequencing technologies that allow massive parallel sequencing for the metabarcoding of environmental samples. The advent of environmental DNA (eDNA) sequencing protocols has heralded a new paradigm in INNS detection. The burgeoning field is based on the sequencing of free DNA found in the environment (e.g., from mucus, shed skin cells, or pollen), without the need to necessarily identify or even see the target species, and effectively negates the need for destructive sampling or tissue collection. Such a technique may be particularly relevant to early invasion stages when INNS are still at low population densities and less likely to be recorded in standard surveys. To our knowledge, eDNA techniques have yet to be applied to the AF biome for INNS detection; however, they have been extensively applied elsewhere, including both freshwater and marine aquatic environments for a range of taxonomic groups (e.g., Mahon and Jerde 2016; Clusa et al. 2017; Fernandez et al. 2018) and have even been used to detect the presence of terrestrial mammals (invasive wild boars) by sampling water from wallowing sites (Williams et al. 2018).

### *13.5.2 Provenance Trials, Forestry Plantations, and Conifer Invasions in the AF*

In terrestrial environments, INNS of pines from the family Pinaceae and Cupressaceae can invade *Araucaria* forests, grasslands, salt marshes, rocky outcrops, and other degraded environments (e.g., early stages of secondary succession) (Zenni and Ziller 2011; Zenni 2014). Several of these conifer species have been introduced as ornamental species, but most were introduced as part of forestry provenance trials during the 1950s, 1960s, and 1970s (Zenni and Simberloff 2013). Those trials were designed to find the seed sources from all species of interest that exhibited the best growth and survivorship in the new environment (Zenni et al.

2014a, b). Currently, species like *Pinus taeda*, *Pinus elliottii*, and *Cupressus lusitanica* are found naturalized or invading many sites in the AF biome. These pine invasions have had substantial negative impacts on the native biota, including BH (Bourscheid and Reis 2010; Falleiros et al. 2011; Bechara et al. 2013). Previous work has shown that the considerable invasive success of pines is a result of a positive genotype-by-environment interaction, where the genotypes selected for plantations in each region had the greatest chances of success (Zenni et al. 2017). The pine populations can evolve and rapidly adapt to the invaded range (Zenni et al. 2014a; Zenni and Hoban 2015). Like foresters of the past, we could use genetic and, more recently, genomic technologies to identify economically relevant tree species or seed sources for timber and resin production. Provenance trials may be maintained as seed and genetic banks, and genetic technologies (e.g., transgenes and gene editing) could be applied to develop sterile tree lineages. Suppression of floral genes in forestry trees has been proposed in the past and is deemed possible and viable (Strauss et al. 1995; Fritsche et al. 2018).

### *13.5.3 Inclusion of NNS in Museums and Scientific Collections*

Museums and scientific collections (MSCs) have a unique importance in creation and dissemination of the knowledge of Earth's biodiversity, as well as to changes in the abundance and distributions of organisms. However, MSCs may show skewed data, and this bias could be substantial for NNS.

The focus of MSCs is native and rare species, so the inclusion of NNS is frequently ignored. Most MSCs do not include good records of occurrence or abundance and distribution of NNS; therefore, appropriate historical record, storage, and maintenance of NNS collections in MSCs could be as important as information regarding native species. This may skew conclusions on conservation status (see Reis et al. (2016), Vitule et al. (2016), Bezerra et al. (2019)), by not considering NNS properly in the database. For example, simple information about records of occurrence might help to elucidate a myriad of unsuccessful NNS that can be as important as successful ones to elucidate patterns and processes of NNS into the AF and might be a key component of several research topics for invasion science (Zenni and Nuñes 2013; Frehse et al. 2018).

### *13.5.4 Conservation, Management, and Decision-Making*

Besides the importance of scientific efforts in the study of invasion ecology and BH, management and prevention still depend on policymakers. Historically, decisions related to NNS (including INNS) in Brazil did not focus on prevention and tended to favor new introductions (see Azevedo-Santos et al. (2017) and references therein). In general, decrees and laws that allow new introductions in Brazil are justified by the economic development (e.g., commodities, aquaculture, and deforestation). However, the use of NNS to improve economic development is highly contradictory. Recent cuts in science funding have serious implications for science and consequently in the study of NNS. Developmental policies in large urban centers of the AF are related to increases in NNS introductions, both in terms of propagule and colonization pressure, which consequently makes their future control difficult and expensive (Angelo 2016). With the goal of conserving the biodiversity and ecosystem services, scientists and decision-makers must work collaboratively, combining ideas and future directions to define strategies. One important first step is to become aware of the threat that NNS, invaders, and super-dominant organisms poses to natural ecosystems, to ecosystem services, and to the future generations.

## 13.6 Conclusion

We showed that there is a growing pattern in spatiotemporal records and richness of NNS in the AF. Additionally, growth trends (records and richness) and spatial distribution can vary according to taxonomic groups and regions. Plants and fish in the more developed regions of the AF had markedly more NNS richness. Insects had high number of occurrence records (~1000) although with lower species richness (~10), but the majority of species are related to public health and agricultural issues. Growth in NNS numbers and records can directly and indirectly alter community structure and ecosystem services at the local, regional, or biome scales.

The effects of INNS remain underexplored in the AF compared to other environmental problems. Many papers and reviews discuss the strong fragmentation, deforestation, hunting, and related conservation issues in the AF. However, the negative impacts of NNS, INNS, super-dominant organisms, and BH processes require special attention. In this sense, we highlighted some examples of impacts for different taxonomic groups and biological scales (e.g., individuals, populations, and ecosystems).

At last, we raised information on new techniques and approaches which can help to improve the knowledge to help, prevent, and manage NNS and INNS. We hope to provide useful information to scientists and decision-makers involved with conservation, specially related to NNS, in the AF biome.

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## References

Agostinho AA, Miranda LE, Bini LM et al (1999) Patterns of colonization in neotropical reservoirs, and prognoses on aging. In: Tundisi JG, Straskraba M (eds) Theoretical reservoir ecology and its applications. Backhuys Publishers, Leiden, pp 227–265

Angelo C (2016) Brazil's scientists battle to escape 20-year funding freeze. Nature News 539:7630

Azevedo-Santos VM, Fearnside PM, Oliveira CS et al (2017) Removing the abyss between conservation science and policy decisions in Brazil. Biodivers Conserv 26:175–1752

Azevedo-Santos VM, Frederico RG, Fagundes CK et al (2019) Protected areas: a focus on Brazilian freshwater biodiversity. Divers Distrib 25:442–448

Bacher S, Blackburn TM, Essl F et al (2017) Socio-economic impact classification of alien taxa (SEICAT). Methods Ecol Evol 9:159–168

Baiser B, Olden JD, Record S et al (2012) Pattern and process of biotic homogenization in the New Pangaea. Proc R Soc B 279:4772–4777

Barros LC, Santos U, Zanuncio JC, Dergam JA (2012) Plagioscion squamosissimus (Sciaenidae) and Parachromis managuensis (Cichlidae): a threat to native fishes of the Doce River in Minas Gerais, Brazil. PloS One 7:e39138

Bechara FC, Reis A, Trentin BE (2013) Invasão biológica de *Pinus elliottii* var. *elliottii* no Parque Estadual do Rio Vermelho, Florianópolis, SC. Floresta 44:63–72

Bezerra LAV, Freitas MO, Daga VS et al (2019) A network meta-analysis of threats to South American fish biodiversity. Fish Fish 20(4):620–639. https://doi.org/10.1111/faf.12365

Boone MD, Little EE, Semlitsch RD (2004) Overwintered bullfrog tadpoles negatively affect salamanders and anurans in native amphibian communities. Copeia 2004:683–690

Both C, Melo AS (2015) Diversity of anuran communities facing bullfrog invasion in Atlantic Forest ponds. Biol Invasions 17:1137–1147

Both C, Lingnau R, Santos-Jr A et al (2011) Widespread occurrence of the American bullfrog, *Lithobates catesbeianus* (Shaw, 1802) (Anura: Ranidae), in Brazil. S Am J Herpetol 6:127–134

Bourscheid K, Reis A (2010) Dinâmica da invasão de *Pinus elliottii* Engelm. em restinga sob processo de restauração ambiental no Parque Florestal do Rio Vermelho, Florianópolis, SC. Biotemas 23:23–30

Braga RR, Gómez-Aparicio L, Heger T et al (2018) Structuring evidence for invasional meltdown: broad support but with biases and gaps. Biol Invasions 20:923–936

Brice M-H, Pellerin S, Poulin M (2017) Does urbanization lead to taxonomic and functional homogenisation in riparian forests? Divers Distrib 23:828–840

Campos GS, Bandeira AC, Sardi SI (2015) Zika Virus Outbreak, Bahia, Brazil. Emerg Infect Dis 21:1885–1886

Canonico GC, Arthington A, McCrary JK, Thieme ML (2005) The effects of introduced tilapias on native biodiversity. Aquat Conserv 15:463–483

Carrete M, Tella JL (2008) Wild-bird trade and exotic invasions: a new link of conservation concern? Front Ecol Environ 6:207–211

Carvalho T, Becker CG, Toledo LF (2017) Historical amphibian declines and extinctions in Brazil linked to chytridiomycosis. Proc R Soc B 284:20162254

Carvalho WD, Mustin K, Paulino JS, Adania CH, Rosalino LM (2019) Recreational hunting and the use of non-selective traps for population control of feral pigs in Brazil. Biodivers Conserv 28:3045–3050

Cervo IB (2017) Dieta de *Sus scrofa* e suas implicações na agropecuária e na biodiversidade no Brasil. MSc Dissertation, Universidade Federal do Rio Grande do Sul, Brazil. [In Portuguese]

Ceschin F, Bini LM, Padial AA (2018) Correlates of fish and aquatic macrophyte beta diversity in the Upper Paraná River floodplain. Hydrobiologia 805:377–389

Clusa L, Miralles L, Basanta A, Escot C, García-Vázquez E (2017) eDNA for detection of five highly invasive molluscs. A case study in urban rivers from the Iberian Peninsula. PLoS One 12:e0188126

Crosby GT (1972) Spread of the cattle egret in the Western Hemisphere. Bird-Banding 43:205

Daga VS, Skóra F, Padial AA et al (2015) Homogenization dynamics of the fish assemblages in Neotropical reservoirs: comparing the roles of introduced species and their vectors. Hydrobiologia 746:327–347

Daszak P, Strieby A, Cunningham AA et al (2004) Experimental evidence that the bullfrog (*Rana catesbeiana*) is a potential carrier of chytridiomycosis, an emerging fungal disease of amphibians. Herpetol J 14:201–207

De Castro MCT, Fileman TW, Hall-Spencer JM (2017) Invasive species in Northern and Southwestern Atlantic Ocean: a review. Mar Poll Bull 116:41–47

De Oliveira SV, Vargas A, Rocha SM et al (2018) The nature of attacks by wild boar (*Sus scrofa*) and wild boar/domestic pig hybrids ('javaporcos') and the conduct of anti-rabies care in Brazil. Int Am J Med Health 1:e201801001

Deberdt AJ, Scherer SB (2007) O javali asselvajado: ocorrência e manejo da espécie no Brasil. Nat Conserv 5:31–44

Early R, Bradley BA, Dukes JS et al (2016) Global threats from invasive alien species in the twenty-first century and national response capacities. Nat Commun 7:12485

Enedino TR, Loures-Ribeiro A, Santos BA (2018) Protecting biodiversity in urbanizing regions: the role of urban reserves for the conservation of Brazilian Atlantic Forest birds. Perspect Ecol Conserv 16:17–23

Enserink M (2014) Crippling virus set to conquer Western Hemisphere. Science 344:678–679

Essl F, Bacher S, Genovesi P et al (2018) Which taxa are alien? Criteria, applications, and uncertainties. Bioscience 68:496–509

Evans T, Kumschick S, Blackburn TM (2016) Application of the environmental impact classification for alien taxa (EICAT) to a global assessment of alien bird impacts. Divers Distrib 22:919–931

Falleiros RM, Zenni RD, Ziller SR (2011) Invasão e manejo de Pinus taeda em campos de altitude do Parque Estadual do Pico Paraná, Paraná, Brasil. Floresta 41:123–134

Faria D, Paciencia MLB, Dixo M et al (2007) Ferns, frogs, lizards, birds and bats in forest fragments and shade cacao plantations in two contrasting landscapes in the Atlantic forest, Brazil. Biodivers Conserv 8:2335–2357

Faria GMM, Rosa CA, Corrêa GLC et al (2016) Geographic distribution of the European hare (*Lepus europaeus*) in Brazil and new records of occurrence for the Cerrado and Atlantic Forest biomes. Mammalia 80:497–505

Fernandez S, Sandin MM, Beaulieu PG et al (2018) Environmental DNA for freshwater fish monitoring: insights for conservation within a protected area. PeerJ 6:e4486

Ferreira CM, Pimenta AGC, Paiva-Neto JS (2002) Introdução à ranicultura. B Inst Pesca 33:1–15

Ferreira CEL, Junqueira AOR, Villac MC, Lopes RM (2008) Marine bioinvasions in the Brazilian coast: brief report on history of events, vectors, ecology, impacts and management of non-indigenous species. In: Rilov G, Crooks JA (eds) Biological invasions in marine ecosystems. Springer, Berlin, pp 459–478

Figueiredo MSL, Weber MM, Brasileiro CA et al (2021) Tetrapod diversity in the Atlantic Forest: maps and gaps. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Filgueiras BK, Tabarelli M, Leal IR et al (2016) Spatial replacement of dung beetles in edge-affected habitats: biotic homogenization or divergence in fragmented tropical forest landscapes? Divers Distrib 22:400–409

Fontoura PM (2013) Non-native bird species in Brazil. Neotrop Biol Conserv 3:165–175

Forti LR, Becker CG, Tacioli L et al (2017) Perspectives on invasive amphibians in Brazil. PLoS One 12:e0184703

Frehse FA, Braga RR, Nocera GA, Vitule JRS (2016) Non-native species and invasion biology in a megadiverse country: scientometric analysis and ecological interactions in Brazil. Biol Invasions 18:3713–3725

Frehse FA, Andrade PDB, Vitule JRS (2018) Absence of the invasive golden mussel in a reservoir near Curitiba, Brazil: a possible case of invasion failure. Neotrop Biol Conserv 13:86–89

Fritsche S, Klocko AL, Boron A et al (2018) Strategies for engineering reproductive sterility in plantation forests. Front Plant Sci 9:1671

Froese R, Pauly D (2019) FishBase. World wide web electronic publication. http://www.fishbase.org. Accessed 10 April 2019

Guimarães TCS, Schmidt IB (2017) A systematization of information on Brazilian federal protected areas with management actions for animal invasive alien species. Perspect Ecol Conserv 15:136–140

Gutierre SMM, Vitule JRS, Freire CA, Prodocimo V (2013) Physiological tools to predict invasiveness and spread via estuarine bridges: tolerance of Brazilian native and worldwide introduced freshwater fishes to increased salinity. Marine Fresh Res 65:425–436

Haddad CFB, Toledo LF, Prado CPA, Loebmann D, Gasparini JL, Sazima I (2013) Guide to the amphibians of the Atlantic Forest: diversity and biology, 1st edn. Anolis Books, São Paulo, 550p

Hegel CGZ (2017) Padrões de ocupação do javali (*Sus scrofa* L.) na Mata Atlântica sul-brasileira. MSc Dissertation, Universidade Regional Integrada do Alto Uruguai e das Missões at Erechim

Hegel CGZ, Marini MÂ (2013) Impact of the wild boar, *Sus scrofa*, on a fragment of Brazilian Atlantic Forest. Neotrop Biol Conserv 8:17–24

Hegel CGZ, Marini MÂ (2018) Large felids as predators of wild boars in the Atlantic Forest: reconciling Verdade et al. and Rosa et al. Anim Conserv 21:363–364

Hegel CGZ, Santos LR, Marinho JR, Marini MÂ (2019a) Is the wild pig the real “big bad wolf”? Negative effects of wild pig on Atlantic Forest mammals. Biol Invasions 21:3561–3574

Hegel CGZ, Santos LRD, Pichorim M, Marini MÂ (2019b) Wild pig (*Sus scrofa* L.) occupancy patterns in the Brazilian Atlantic forest. Biota Neotrop 19:e20180719

Hermoso V, Clavero M, Kennard MJ (2012) Determinants of fine-scale homogenization and differentiation of native freshwater fish faunas in a Mediterranean Basin: implications for conservation. Divers Distrib 18:236–247

Humair F, Kueffer C, Siegrist M (2014) Are non-native plants perceived to be more risky? Factors influencing horticulturists’ risk perceptions of ornamental plant species. PLoS One 9:e102121

I3N Brasil (2019) Base de dados nacional de espécies exóticas invasoras. Instituto Hórus de Desenvolvimento e Conservação Ambiental, Florianópolis. http://i3n.institutohorus.org.br. Accessed 04 Apr 2019

IBAMA (2005) Instrução Normativa Ibama N° 71 de 04 de agosto de 2005 do Instituto Brasileiro do Meio Ambiente e dos Recursos Renováveis (IBAMA). Ministério do Meio Ambiente (MMA), Brasília

IBAMA (2013) Instrução Normativa Ibama N° 03 de 31 de janeiro de 2013 do Instituto Brasileiro do Meio Ambiente e dos Recursos Renováveis (IBAMA). Ministério do MeioAmbiente (MMA), Brasília

Jaksic FM (1998) Vertebrate invaders and their ecological impacts in Chile. Biodivers Conserv 7:1427–1445

Jarić I, Heger T, Monzon FC et al (2019) Crypticity in biological invasions. Trends Ecol Evol 34:291–302

Jepson P, Ladle RJ (2015) Nature apps: waiting for the revolution. Ambio 44:827

Juliano SA, Lounibos LP (2005) Ecology of invasive mosquitoes: effects on resident species and on human health. Ecol Lett 8:558–574

Kiesecker JM, Blaustein AR, Miller CL (2001) Potential mechanisms underlying the displacement of native red-legged frogs by introduced bullfrogs. Ecology 82:1964–1970

Klemann-Junior L, Vallejos MAV, Scherer-Neto P, Vitule JRS (2017) Traditional scientific data vs. uncoordinated citizen science effort: a review of the current status and comparison of data on avifauna in Southern Brazil. PLoS One 12:e0188819

Kress WJ, Garcia-Robledo C, Soares JVB et al (2018) Citizen science and climate change: mapping the range expansions of native and exotic plants with the mobile app Leafsnap. Bioscience 68:348–358

Lessa I, Guimarães TCS, Bergallo HG et al (2016) Domestic dogs in protected areas: a threat to Brazilian mammals? Nat Conserv 14:46–56

Lips KR, Diffendorfer J, Mendelson JR, Sears MW (2008) Riding the wave: reconciling the roles of disease and climate change in amphibian declines. PLoS Biol 6:e72

Lôbo D, Leão T, Melo FPL et al (2011) Forest fragmentation drives Atlantic forest of northeastern Brazil to biotic homogenization. Divers Distrib 17:287–296

Lockwood JL, Hoopes MF, Marchetti MP (2009) Invasion ecology. Blackwell, Oxford

Lopes VG, Branco CWC, Kozlowsky-Suzuki B et al (2017) Predicting temporal variation in zooplankton beta diversity is challenging. PLoS One 12:e0187499

Lucy FE, Roy H, Simpson A et al (2016) INVASIVESNET towards an International Association for Open Knowledge on invasive alien species. Manage Biol Invasions 7:131–139

Magurran AE, Dornelas M, Moyes F et al (2015) Rapid biotic homogenization of marine fish assemblages. Nat Commun 6:8405

Mahon AR, Jerde CL (2016) Using environmental DNA for invasive species surveillance and monitoring. In: Bourlat S (ed) Marine genomics. Methods in molecular biology. Humana Press, New York

Martello F, Bello F, Morini MSC et al (2018) Homogenization and impoverishment of taxonomic and functional diversity of ants in Eucalyptus plantations. Sci Rep 8:3266

Mazza G, Tricarico E, Genovesi P, Gherardi F (2014) Biological invaders are threats to human health: an overview. Ethol Ecol Evol 26:112–129

McKinney ML, Lockwood JL (1999) Biotic homogenization: a few winners replacing many losers in the next mass extinction. Trends Ecol Evol 14:450–453

Medeiros CI, Both C, Grant T, Hartz SM (2017) Invasion of the acoustic niche: variable responses by native species to invasive American bullfrog calls. Biol Invasions 19:675–690

Metzger JP (2009) Conservation issues in the Brazilian Atlantic Forest. Biol Conserv 142:1138–1140

Michelan TS, Thomaz SM, Mormul RP, Carvalho P (2010) Effects of an exotic invasive macrophyte (tropical signalgrass) on native plant community composition, species richness and functional diversity. Freshwater Biol 55:1315–1326

Molnar JL, Gamboa RL, Revenga C, Spalding MD (2008) Assessing the global threat of invasive species to marine biodiversity. Front Ecol Environ 6:485–492

Morais TA, Rosa CA, Azevedo CS, Viana-Junior AB, Santos P, Passamani M (2019) Factors affecting space use by wild boars (*Sus scrofa*) in high-elevation tropical forests. Can J Zool 97:971–978

Morante-Filho JC, Faria D, Mariano-Neto E et al (2015) Birds in anthropogenic landscapes: the responses of ecological groups to forest loss in the Brazilian Atlantic Forest. PLoS One 6:e0128923

Morante-Filho JC, Arroyo-Rodríguez V, Faria D (2016) Patterns and predictors of β-diversity in the fragmented Brazilian Atlantic forest: a multiscale analysis of forest specialist and generalist birds. J Anim Ecol 1:240–250

Myers N, Mittermeier RA, Mittermeier CG et al (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

O'Hanlon SJ, Rieux A, Farrer RA et al (2018) Recent Asian origin of chytrid fungi causing global amphibian declines. Science 360:621–627

Olden JD, Poff NL, Douglas MR et al (2004) Ecological and evolutionary consequences of biotic homogenization. Trends Ecol Evol 19:18–24

Olden JD, Kennard MJ, Leprieur F et al (2010) Conservation biogeography of freshwater fishes: recent progress and future challenges. Divers Distrib 16:496–513

Olden JD, Comte L, Giam X (2018) The Homogocene: research prospectus for the study of biotic homogenization. NeoBiota 37:23–36

Oliveira CHS (2012) Ecologia e manejo de javali (*Sus scrofa* L.) na América do Sul. Dissertation, Universidade Federal do Rio de Janeiro, Rio de Janeiro, Brazil. [In Portuguese]

PAN - Plano Nacional de Prevenção, Controle e Monitoramento do Javali (2017) Ministério do Meio Ambiente e Ministério da Agricultura, Pecuária e Abastecimento. Brasília, Brasil

Paschoal AMO, Massara RL, Santos JL, Chiarello AG (2012) Is the domestic dog becoming an abundant species in the Atlantic forest? A study case in southeastern Brazil. Mammalia 76:67–76

Paschoal AMO, Massara RL, Bailey LL et al (2016) Use of Atlantic Forest protected areas by free-ranging dogs: estimating abundance and persistence of use. Ecosphere 7:e01480

Paschoal AMO, Massara RL, Bailey LL et al (2018) Anthropogenic disturbances drive domestic dog use of Atlantic Forest protected areas. Trop Conserv Sci 11:1–14

Patoka J, Magalhães ALB, Kouba A et al (2018) Invasive aquatic pets: failed policies increase risks of harmful invasions. Biodivers Conserv 27:3037

Pedrosa F, Salerno R, Padilha FVB, Galetti M (2015) Current distribution of invasive feral pigs in Brazil: economic impacts and ecological uncertainty. Nat Conserv 13:84–87

Petesse ML, Petrere M Jr (2012) Tendency towards homogenisation in fish assemblages in the cascade reservoir system of the Tietê river basin, Brazil. Ecol Eng 48:109–116

Pimentel D, Zuniga R, Morrison D (2005) Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecol Econ 52:273–288

Pivello VR, Vieira MV, Grombone-Guaratini MT, Matos DMS (2018) Thinking about super-dominant populations of native species – examples from Brazil. Perspect Ecol Conser 16:74–82

Puertas FHG (2015) A invasão do javali na Serra da Mantiqueira: aspectos populacionais, uso do habitat e sua relação com o homem. Thesis, University Federal de Lavras, Lavras, Minas Gerais, Brazil. [In Portuguese]

Pyšek P, Richardson DM, Pergl J et al (2008) Geographical and taxonomic biases in invasion ecology. Trends Ecol Evol 23:237–244

Queiroz-Sousa J, Brambilla EM, Garcia-Ayala JR et al (2018) Biology, ecology and biogeography of the South American silver croaker, an important Neotropical fish species in South America. Rev Fish Biol Fisher 28:693–714

Quintela FM, Santos MB, Oliveira SV, Costa RC, Christoff AU (2010) Wild boars and feral pigs (Suidae, *Sus scrofa*) in the Restinga of Rio Grande, RS, Brazil: ecosystems of occurrence and preliminary data on environmental impacts. Neotrop Biol Conserv 5:172–178

Rahel FJ (2000) Homogenization of fish faunas across the United States. Science 288:854–856

Reis RE, Albert JS, Dario FD et al (2016) Fish biodiversity and conservation in South America. J Fish Biol 89:12–47

Richardson DM (2011) Fifty years of invasion ecology: the legacy of Charles Elton. Wiley-Blackwell, Oxford

Rodriguez D, Becker CG, Pupin NC et al (2014) Long-term endemism of two highly divergent lineages of the amphibian-killing fungus in the Atlantic Forest of Brazil. Mol Ecol 23:774–787

Rosa CA, Curi NHA, Puertas F, Passamani M (2017) Alien terrestrial mammals in Brazil: current status and management. Biol Invasions 19:2101–2123

Rosa CAD, Wallau MO, Pedrosa F (2018) Hunting as the main technique used to control wild pigs in Brazil. Wildl Soc Bull 42:111–118

Rosell C, Fernández-Llario P, Herrero J (2001) El jabalí (*Sus scrofa* Linnaeus, 1758). Galemys 13:1–25

Ruiz GM, Carlton JT, Grosholz ED, Hines AH (1997) Global invasions of marine and estuarine habitats by non-indigenous species: mechanisms, extent, and consequences. Am Zool 37:621–632

Sales LP, Ribeiro BR, Hayward MW et al (2017) Niche conservatism and the invasive potential of the wild boar. J Anim Ecol 86:1214–1223

Sampaio AB, Schmidt IB (2014) Espécies exóticas invasoras em unidades de conservação federais do Brasil. Biodivers Bras 2:32–49

Santos ABI, de Freitas TB, Araújo FG (2010) Influence of the river flow on the structure of fish assemblage along the longitudinal gradient from river to reservoir. Fortschr Zool 27:732–740

Santos ABI, Albieri RJ, Araújo FG (2013) Seasonal response of fish assemblages to habitat fragmentation caused by an impoundment in a Neotropical river. Environ Biol Fish 96:1377–1387

Scarano FR, Ceotto P (2015) Brazilian Atlantic forest: impact, vulnerability, and adaptation to climate change. Biodivers Conserv 24:2319–2331

Scheele BC, Pasmans F, Berger L et al (2019) Amphibian fungal panzootic causes catastrophic and ongoing loss of biodiversity. Science 363:1459–1463

Schloegel LM, Toledo LF, Longcore JE et al (2012) Novel, panzootic, and hybrid genotypes of amphibian chytridiomycosis associated with the bullfrog trade. Mol Ecol 21:5162–5177

Seebens H, Blackburn TM, Dyer EE et al (2017) No saturation in the accumulation of alien species worldwide. Nat Commun 8:14435

Sick H (2001) Ornitologia brasileira: uma introdução. Nova Fronteira, Brasília

Silva MAF, Verona CE, Conde M, Pires AS (2018) Frugivory and potential seed dispersal by the exotic-invasive marmoset *Callithrix jacchus* (Primates, Callitrichidae) in an urban Atlantic Forest, Rio de Janeiro, Brazil. Mammalia 82:343–349

Simberloff D (2011) Native invaders. In: Simberloff D, Rejmanek M (eds) Encyclopedia of biologica invasions. University of California Press, California, pp 472–475

Simberloff D, Rejmánek M (2011) Encyclopedia of biological invasions. University of California Press, Los Angeles

Simberloff D, Vitule JRS (2014) A call for an end to calls for the end of invasion biology. Oikos 123:408–413

Simberloff D, Von Holle B (1999) Positive interactions of nonindigenous species invasional meltdown. Biol Invasions 1:21–32

Simberloff D, Souza L, Nunez MA et al (2012) The natives are restless, but not often and mostly when disturbed. Ecology 93:598–607

Siqueira T, Lacerda CG-LT, Saito VS (2015) How does landscape modification induce biological homogenization in tropical stream metacommunities? Biotropica 47:509–516

Souza RSCL, Silva JV (2004) Água de Lastro e Bioinvasão. Interciência, Rio de Janeiro

Srbek-Araujo AC, Chiarello AG (2008) Domestic dogs in Atlantic forest preserves of South-Eastern Brazil: a camera-trapping study on patterns of entrance and site occupancy rates. Braz J Biol 68:771–779

Strauss SH, Rottmann WH, Brunner AM, Sheppard LA (1995) Genetic engineering of reproductive sterility in forest trees. Mol Breed 1:5–26

Thier O, Wesenberg J (2016) Floristic composition and edge-induced homogenization in tree communities in the fragmented Atlantic rainforest of Rio de Janeiro, Brazil. Trop Conserv Sci 9:852–876

Toledo LF, Silva RR, Haddad CFB (2007) Anurans as prey: an exploratory analysis and size relationships between predators and their prey. J Zool 271:170–177

Toussaint A, Beauchard O, Oberdorff T et al (2016) Worldwide freshwater fish homogenization is driven by a few widespread non-native species. Biol Invasions 18:1295–1304

Travassos L, Carvalho ID, Pires AS et al (2018) Living and lost mammals of Rio de Janeiro's largest biological reserve: an updated species list of Tinguá. Biota Neotrop 18:e20170453

UN United Nations (1992) A/CONF.151/26 (Vol. I) - Rio Declaration. 12 Aug 1992. Report on the United Nations Conference on Environment and Development

Vallejos MAV, Padial AA, Vitule JRS (2016) Human-induced landscape changes homogenize Atlantic Forest bird assemblages through nested species loss. PLoS One 11:e0147058

Vanstreels RET, Teixeira RHF, Camargo LC et al (2010) Impacts of animal traffic on the Brazilian Amazon parrots (*Amazona* species) collection of the Quinzinho de Barros Municipal Zoological Park, Brazil, 1986-2007. Zoo Biol 29:600–614

Vitule JRS, Skóra F, Abilhoa V (2012) Homogenization of freshwater fish faunas after the elimination of a natural barrier by a dam in Neotropics. Divers Distrib 18:111–120

Vitule JRS, Costa APL, Frehse FA et al (2016) Comments on 'Fish biodiversity and conservation in South America by Reis et al (2016)'. J Fish Biol 90:1182–1190

Williams KE, Huyvaert KP, Vercauteren KC et al (2018) Detection and persistence of environmental DNA from an invasive, terrestrial mammal. Ecol Evol 8:688–695

Winter M, Schweiger O, Klotz S et al (2009) Plant extinctions and introductions lead to phylogenetic and taxonomic homogenization of the European flora. Proc Natl Acad Sci U S A 106:21721–21725

Zenni RD (2014) Analysis of introduction history of invasive plants in Brazil reveals patterns of association between biogeographical origin and reason for introduction. Austral Ecol 39:401–407

Zenni RD, Hoban SM (2015) Loci under selection during multiple range expansions of an invasive plant are mostly population specific, but patterns are associated with climate. Mol Ecol 13:3360–3371

Zenni RD, Nuñes MA (2013) The elephant in the room: the role of failed invasions in understanding invasion biology. Oikos 122:801–815

Zenni RD, Simberloff D (2013) Number of source populations as a potential driver of pine invasions in Brazil. Biol Invasions 15:1623–1639

Zenni RD, Ziller SR (2011) An overview of invasive plants in Brazil. Brazilian J Bot 34:431–446

Zenni RD, Bailey JK, Simberloff D (2014a) Rapid evolution and range expansion of an invasive plant are driven by provenance -environment interactions. Ecol Lett 17:727–735

Zenni RD, Lamy J-B, Lamarque LJ, Porté AJ (2014b) Adaptive evolution and phenotypic plasticity during naturalization and spread of invasive species: implications for tree invasion biology. Biol Invasions 16:635–644

Zenni RD, Dickie IA, Wingfield MJ et al (2017) Evolutionary dynamics of tree invasions: complementing the unified framework for biological invasions. AoB Plants. https://doi.org/10.1093/aobpla/plw085

Ziller SR, Dechoum MS (2014) Plantas e vertebrados exóticos invasores em unidades de conservação no Brasil. Biodivers Bras 2:4–31

Zwiener VP, Lira-Noriega A, Grady CJ et al (2018) Climate change as a driver of biotic homogenization of woody plants in the Atlantic Forest. Glob Ecol Biogeogr 27:298–309

# Chapter 14
# Causes and Consequences of Large-Scale Defaunation in the Atlantic Forest

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**Abstract** The Atlantic Forest of South America hosts one of the world's most diverse and threatened tropical forest biota. After five centuries of European human expansion, most Atlantic Forest landscapes are archipelagos of small forest fragments surrounded by open-habitat matrices. In this chapter, we describe the causes and consequences of large-scale defaunation in the Atlantic Forest of South America. We identify and quantify the magnitude of the main anthropogenic drivers of defaunation and stimulate a debate on how to revert the loss of fauna to restore biodiversity and ecosystem functions and services. The magnitude of the impact of defaunation in the Atlantic Forest is hard to estimate, but we can predict that, at large scale, habitat loss, fragmentation, and degradation are the most common threats to terrestrial populations. Other threats vary in importance according to the

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taxonomic group. In general, apex predators, other carnivores, large-bodied mammals, and large herbivores were among the most defaunated functional groups and the loss of these animals has also a strong impact on the ecosystem services. Given the extent of the consequences of defaunation in the Atlantic Forest, mitigation strategies are imperative. Habitat restoration would clearly be effective in building space for defaunation mitigation but reversing the pervasive defaunation that occurred in the Atlantic Forest is by no means a straightforward task. Nonetheless, it will be fundamental to assure the persistence of the biodiversity in the Atlantic Forest remnants.

**Keywords** Animal conservation · Defaunation cascades · Animal overexploitation · Terrestrial fauna · Defaunation drivers · Tropical forest

## 14.1 Introduction

Tropical forests hold the highest diversity of animals on Earth, but human-driven disturbance has led many populations to decline and species at imminent risk of extinction (Dirzo et al. 2014; Ceballos et al. 2015). Hunting for subsistence has been considered a major driver for the extinction of vertebrates since the Pleistocene (Ripple et al. 2014; Young et al. 2016) and even today it is a major cause of decline in the populations of large-bodied vertebrates (Ripple et al. 2014). Humans have now affected most of the Earth's land area and just 15% of the natural areas are protected for nature (Sloan et al. 2014). Today we have lost more than a third of all the forest world cover (Hansen et al. 2013). In addition to hunting and deforestation, other human-driven disturbances have been incorporated to threaten the persistence of biodiversity in the rainforests: new emerging diseases, the widespread of invasive species, expansion of roadway network, and traffic that increase wildlife-vehicle collisions and the alterations in climatic condition.

The Atlantic Forest of South America is vanishing rapidly. It used to stretch from latitudes 3° to 30° S and to occupy an area of approximately 150 million ha (Ribeiro et al. 2009). It is estimated that the Atlantic Forest holds 861 species of birds (213

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endemic) (Hasui et al. 2018; Rodrigues et al. 2019), more than 300 reptiles (95 endemic), 512 species of amphibians (260 endemic) (Vancine et al. 2018), and 321 species of mammals (89 endemic) (Bovendorp et al. 2017; Lima et al. 2017; Muylaert et al. 2017; Gonçalves et al. 2018a, b; Culot et al. 2019; Souza et al. 2019, see also Figueiredo et al. 2021 Chap. 9). This biome has suffered intense deforestation, fragmentation, and defaunation in the twentieth century (Dean 1997) and currently it is restricted to less than 16 million ha, or 12% of its original distribution (Ribeiro et al. 2009). As a result, the Atlantic Forest has the highest number of endangered species compared to other Brazilian biomes, having 593 vertebrate species threatened by extinction, and six already extinct in the wild (www.iucnredlist.org; ICMBio 2018).

We defined defaunation as the global, local, or functional extinction of animal population or species (Dirzo et al. 2014). For instance, one endemic amphibian (*Phrynomedusa fimbriata*), four endemic birds (*Cichlocolaptes mazarbarnetti*, *Philydor novaesi*, *Glaucidium mooreorum*, and *Mitu mitu*) and one endemic mammal from Fernando de Noronha Island (*Noronhomys vespucci*) are now globally extinct. The Giant Otter (*Pteronura brasiliensis*) is locally extinct in the Atlantic Forest, but still occur in the Amazon and Pantanal, and the jaguar (*Panthera onca*) is functionally extinct in the Atlantic Forest, where the species is so rare that the top predator function is lost from most of the biome (Galetti et al. 2013a).

In this chapter, we identify and quantify the magnitude of the main anthropogenic drivers of defaunation in the Atlantic Forest of South America (Fig. 14.1). We discuss the ecological consequences of defaunation in the Atlantic Forest and stimulate a debate on how to revert the loss of fauna to restore biodiversity and ecosystem functions and services. How many individuals are lost per year due to human-driven causes? And what are the consequences of the defaunation to human-well-being and ecosystem processes? Here, we attempt to answer these important questions according to each of the human-driven disturbances identified as follows.

## 14.2 Infectious Diseases

Wildlife can be negatively impacted due to diseases (Doherty et al. 2017) which are commonly a silent threat for wild animals (Hudson 2002). Emerging diseases may become one of the major defaunation drivers in the Anthropocene. Patterns of epizootia are characterized as a disease that is only occasionally found in a population, but which can spread very rapidly. Emerging infectious diseases (EIDs) of free-living wild animals can be classified into three major groups on the basis of key epizootiological criteria: (1) associated with spill-over from domestic to wildlife populations; (2) related directly to human intervention via host or parasite translocations; and (3) with no overt human or domestic animal involvement (Daszak et al. 2000). These phenomena have biological implications as many wildlife species are reservoirs of pathogens that threaten domestic animal and human health, and

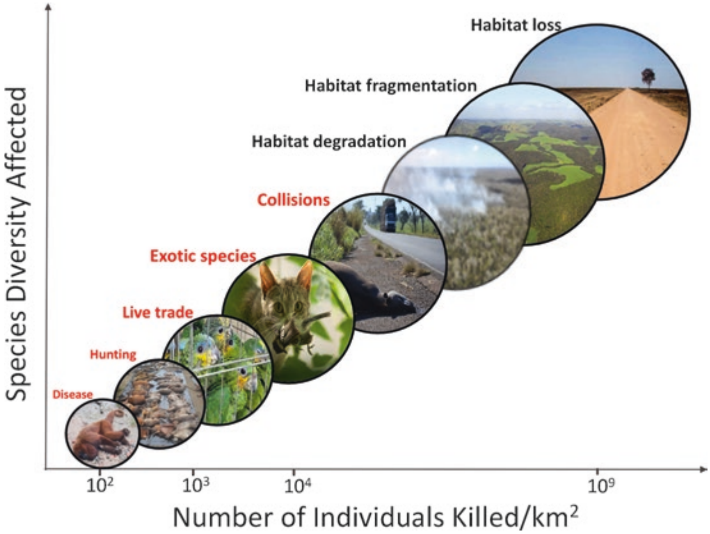

**Fig. 14.1** Conceptual model on the magnitude of the effects of defaunation in both richness and abundance of tetrapods in the Atlantic Forest. The size of the circles indicates the number of species and individuals affected by direct (red colors) or indirect (black colors) drivers. A direct driver influences ecosystem processes and can therefore be identified and measured to differing degrees of accuracy. An indirect driver operates more diffusely, often by altering one or more direct drivers, and its influence is established by understanding its effect on a direct driver. Both indirect and direct drivers often operate synergistically. The magnitude of the impact of defaunation in the Atlantic Forest is hard to estimate, but we can predict that large-scale effects will affect more species and individuals depending on the driver

wildlife EIDs pose a substantial threat to the conservation of global biodiversity (Daszak et al. 2000).

Infections not necessarily cause death but also symptoms that lead to the decline of the population, directly or indirectly due to reducing competition capacity for hunt or escape from the predator, infertility, altering secondary sex ratios and movement patterns, morbidity, and increasing susceptibility to other infectious diseases (Preece et al. 2017). This scenario can be reinforced by the stress due to the impact caused by environmental changes, which consequently reduce the immunity capacity contributing to new infections (Fig. 14.2).

Yellow fever virus (YFV) causes a devastating disease that is resulting in widespread defaunation in the Atlantic Forest; it is a highly pathogenic virus for New World primates (Monath and Vasconcelos 2015). This disease is currently considered an EID, once a gradual expansion of that has been observed toward the southeast of Brazil, an area that was considered YF-free for almost 80 years without vaccine recommendations (Romano et al. 2014). In later 2016, the virus re-emerged in Brazil, initiating the largest epidemic of sylvatic YFV ever recorded in the Country (Cardoso et al. 2019). It was recorded more than 5500 deaths of monkeys during this outbreak (Brasil et al. 2019) (Fig. 14.2), affecting mostly howler

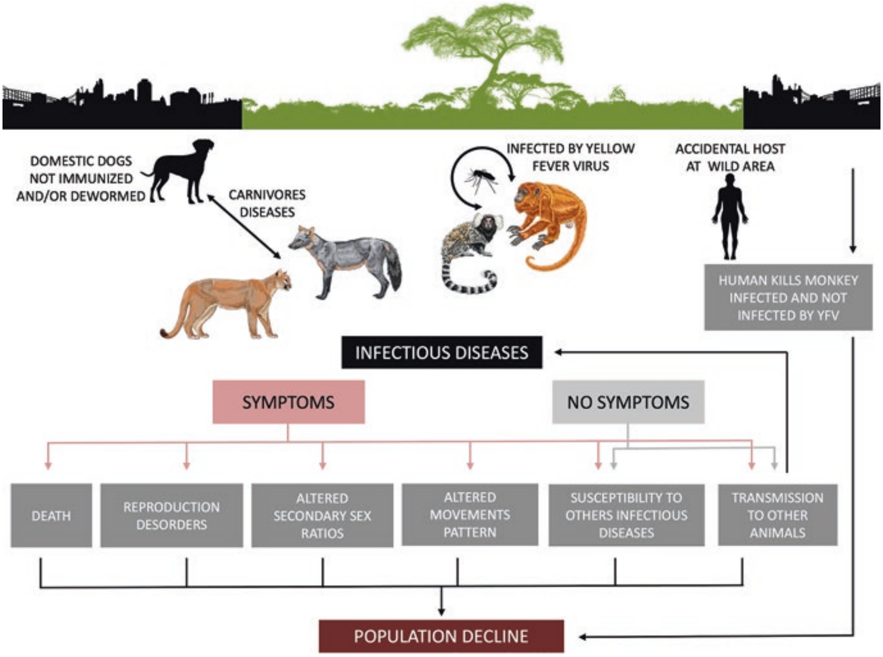

**Fig. 14.2** Two-way flow of diseases between domestic dogs and wild carnivores, and dynamic of yellow fever disease in monkeys, associated with an impact due to the human concept regarding the risk to become infected. Some alterations in wild animals can be caused by these infectious diseases, which result in their population decline and even local extinction

monkeys (*Alouatta* spp.), marmosets (*Callithrix* spp.), titi monkey (*Callicebus* spp.), and the critically threatened muriqui (*Brachyteles* spp.) (Abreu et al. 2019).

Domestic dogs are the most important reservoirs and maintainers of virulent pathogens that affect wild animals (Doherty et al. 2017), specially carnivore species (Courtenay et al. 2001), favoring pathogens transmission across wild, domestic animals and humans (Whiteman et al. 2007). Important diseases transmitted from dogs to wild carnivores are named as “The Big Three” (rabies, distemper, and parvovirosis) due to the strongly negative impact over their populations (Gompper 2014). In Brazil, the risk of transmission of rabies from domestic dogs to wild animals is reduced once they are frequently immunized by Brazilian public policy programs, but the disease is still common in wild animals of Atlantic Forest, such as bats, foxes, and marmosets (Megid et al. 2015; Gonçalves et al. 2020). In case of parvovirosis and distemper virus disease, they are frequent in stray and semi-domiciled domestic dogs (Curi et al. 2016), which are normally not immunized, so their contact with the wild environment can be a threat for wild animals (Martinez et al. 2013). Parvovirosis is a disease with high lethality for very young puppies, and as an aggravating factor the virus can remain for a long time in favorable environments (Megid et al. 2015).

Canine distemper virus (CDV) is considered one of the most important diseases for domestic dogs, with high lethality (Negrão et al. 2006), and is endemic in Brazil (Megid et al. 2015). Dogs are the most important reservoirs; however, some species are susceptible to the virus, such as Canidae (foxes), Mustelidae (otter, ferret, and badger), Procyonidae (coati), Myrmecophagidae (anteaters), and Felidae (wild cats) (Megid et al. 2015). Although outbreaks of CDV disease in wild animals in the Atlantic Forest have not yet occurred, it has been classified as a threat to biodiversity with elements associated with spill-over by domestic dogs (Whiteman et al. 2007). In a large fragment of Atlantic Forest, CDV in wild felids seems to be related with home range use and the close association with domestic dogs living in nearby areas, as 31.5%, 11.3% and 34.6% of the sampled jaguars, pumas, and dogs were seropositive for the disease, respectively (Nava et al. 2008).

## 14.3 Hunting

Hunting for food or retaliation against predators of domestic species continues to be widespread in the Atlantic Forest even within protected areas (Cullen Jr. et al. 2001; Galetti et al. 2017; Sousa and Srbek-Araujo 2017), causing local extinctions or decline of many mammal populations (Canale et al. 2012; Galetti et al. 2009). In the Atlantic Forest, the abundance of medium and large vertebrates is on average 37% to 90% lower in intensely hunted areas compared to low hunted areas (Galetti et al. 2017). Through cascading effects, the abrupt decline or even extinction of populations probably have had pervasive impacts on ecosystems structure and dynamics (Dirzo et al. 2014).

At least one bird (Alagoas curassow, *Pauxi mitu*) and one mammal species (Giant otter, Pteronura brasiliensis) had gone extinct in the wild due primarily to hunting, although both species were threatened by habitat loss and fragmentation of the Atlantic Forest. Other game species (e.g., Aburria jacutinga and Crax blumenbachii) had their distribution reduced in more than 95% (BirdLife International 2019) and species such as the lowland tapir (Tapirus terrestris) and the white-lipped peccary (Tayassu pecari) were locally extinct in most of the Atlantic Forest (Jorge et al. 2013; Ferreguetti et al. 2018).

Hunting is ubiquitous in virtually all fragments in the Atlantic Forest and although it is considered an unregulated activity since 1967, hunting for sport or subsistence never stopped. The recent invasion of wild boar, including feralized domestic pig, and consequently legalization of pig hunting may increase wildlife poaching, due to weak law enforcement. On the other hand, wild pigs may function as a shield to wildlife since it is a preferable target among hunters (Desbiez et al. 2011).

At least 43 bird, 40 mammal, and 14 reptile species are known to be consumed as food in the Atlantic Forest. The impact of hunting on the vertebrate community in the Atlantic Forest differs from other Neotropical sites because while hunting in the Amazon is more evenly distributed among primates, large rodents, and ungulates (Bodmer et al. 1997; Alvard 1998; Peres 2000), in the Atlantic Forest, hunting concentrates heavily on large birds (guans, tinamous), ungulates, large caviomorph rodents, and armadillos (Cullen Jr et al. 2001; Galetti et al. 2017).

The lack of quantitative information on the magnitude of hunting is partly explained by the fact that hunting activities are illegal in Brazil. Therefore, most of the studies on the effects of hunting are based on comparative densities of game species (Canale et al. 2012; Galetti et al. 2017), but not effective measures of harvest from local populations. So far, one study estimated that each hunter harvest 100 kg/year of game meat in a study area in Southeastern Brazil (Nobre 2007).

## 14.4 Animal Trade

The illegal capture and trade of wild animals are certainly main defaunation drivers in the Atlantic Forest since the European invasion to South America. The highly priced red-and-green macaw (Ara chloropterus) was known to occur from the Northeastern Brazilian coast to Rio de Janeiro state and today is extirpated in most of the Atlantic Forest, with a small relict population in Morro do Diabo State Park in São Paulo state and a reintroduced population toward the north in Espírito Santo State.

The illegal capture and trade of wild animals are widespread throughout the Atlantic Forest (Alves et al. 2013) where a total of 31 species are known to be illegally traded. Birds represent the most traded group among all animals in Brazil, and the same pattern is observed in the Atlantic Forest (Renctas 2007). Species belonging to the families Emberizidae (finches) and Psittacidae (parrots and parakeets) stand out among the taxa of wild birds most sold as pets in Brazil, corroborating a trend reported in previous studies (e.g., Gastañaga et al. 2011). There are many reasons for the observed preference for species in these families. The Emberizidae are traded as cage birds because of the popular appeal of their colorful plumage. Moreover, they are extremely resistant, and their small size allows that large numbers of them being kept together in small cages, which facilitates smuggling (Frisch and Frisch 1995; Sick 1997).

The animal trade has recently acquired a new ally – the internet. A survey carried out by Renctas (2007) found 4892 advertisements in Brazilian and international websites promoting the illegal sale and exchange of wild animals from the Brazilian fauna. Most were advertisements for birds and reptiles, but mammals, amphibians, and ornamental fish were also offered (Renctas 2007). The internet illegal market must be viewed with considerable alarm because of the efficiency with which virtual markets allow buyers and sellers to connect with ease and speed that was never possible. These vast potential markets pose new challenges to legislators and enforcement agencies.

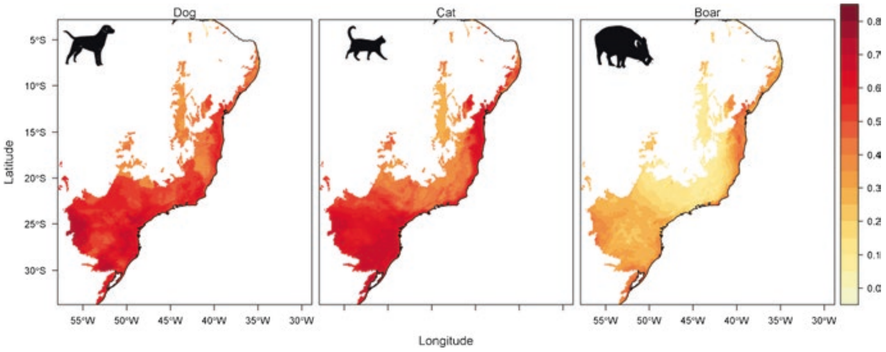

**Fig. 14.3** Potential distribution for the domestic dog, cat, and the wild pig, estimated as a weighted (accuracy-based) ensemble of projections from three modeling methods (generalized linear models – GLM, random forest – RF, bioclimate envelope – BioClim). The domestic cat and dog are spread out most of the Atlantic Forest, while the wild pig is concentrated in the southernmost regions (from L. Sales, unpublished)

## 14.5 Invasive Species

Invasive species can impair native species through competition for space or resources, through direct predation and through spreading diseases that were not previously present in the environment. Humans also transport new diseases from one area of the globe to another. In the Atlantic Forest, three invasive species are known to have widespread distribution and impact on native wildlife: domestic or feral cats, dogs, and the wild pigs Sus scrofa (Fig. 14.3, see also Vitule et al. 2021 Chap. 13).

Although the occurrence of domestic cats and dogs is usually associated with human habitations (Srbek-Araujo and Chiarello 2008; Paschoal et al. 2018), they can reach high densities in forest remnants (Paschoal et al. 2018). This happens mostly because domestic cats and dogs are allowed by their owners to forage freely (Torres and Prado 2010). Forest fragmentation also contributes much to this scenario (Cullen Jr. et al. 2001; Manor and Saltz 2004, see also Vitule et al. 2021 Chap. 13). As a consequence of forest fragmentation, domestic animals have more access to wild areas. Cats have already been pointed out by different studies as the major predator of many birds and mammalian species, causing a decline in many of their populations around the world (Bonnaud et al. 2010). Cats can kill 700 reptiles, 150 birds, and 50 native mammals per square kilometer per year in the USA (Read and Bowen 2001). Dogs are also strong predators, and mammals tend to be their most common victims (Galetti and Sazima 2006).

The third most destructive species for wildlife along with dogs and cats is probably the wild boar, which threatens native fauna by competing for resources, destroying micro-habitats, or depleting prey-base (Ilse and Hellgren 1995; Lowe et al. 2000, see also Vitule et al. 2021 Chap. 13). They also impact natural environments by shifting water quality of streams and accelerating invasion of exotic plants over natural or disturbed areas when dispersing their seeds (Lynes and Campbell 2000; Dovrat et al. 2012; Rosa et al. 2019, see also Vitule et al. 2021 Chap. 13). Wild pigs are hosts of several zoonoses and inflict losses for agriculture by directly destroying crops or transmitting diseases to livestock (Ruiz-Fons 2017; Maciel et al. 2018). In the Atlantic Forest, wild pigs are widespread, likely representing the largest biomass of terrestrial wildlife in many Atlantic Forest fragments (Pedrosa et al. 2015; Rosa et al. 2017; Beca et al. 2017; Rosa et al. 2020, see also Vitule et al. 2021 Chap. 13).

## 14.6 Wildlife Collision (Infrastructure)

Brazil has the fourth biggest road network in the world (DNIT 2019) and the road density is higher in the Southeast region. The effects of roads and traffic on wildlife vary and range from habitat loss (Forman et al. 2003), a reduction in habitat quality in a zone adjacent to the road (e.g., noise, lights, pollution, visual disturbance) (Parris et al. 2009), barrier effect, including interruption of migration and dispersion (Lesbarrères and Fahrig 2012) and direct mortality through collisions with vehicles (Forman and Alexander 1998; Fahrig and Rytwinski 2009). Direct road mortality has the potential to alter the demographic structure of wildlife populations (Steen and Gibbs 2004) and create local population sinks (Nielsen et al. 2006). Such changes may alter the structure and function of communities and ecosystems adjacent to the road (Trombulak and Frissell 2000).

At least 179 species of birds, 92 species of mammals, 71 species of reptiles, and 25 species of amphibians have been recorded roadkilled in different projects on the roads that cross the Atlantic Forest (Grilo et al. 2018). Fifteen species are listed as vulnerable, including the jaguar (Panthera onca), puma (Puma concolor), maned wolf (Chrysocyon brachyurus), hoary fox (Lycalopex vetulus), giant anteater (Myrmecophaga tridactyla), pygmy brocket deer (Mazama nana), maned sloth (Bradypus torquatus), brown howler monkey (*Alouatta guariba*), and small wild cats (*Herpailurus yagouaroundi,* Leopardus colocolo*,* Leopardus geoffroyi*,* Leopardus guttulus*,* and Leopardus wiedii). The northern tiger cat (Leopardus tigrinus), listed as Endangered in Brazil, is another victim of roadkills, as well as other four near-threatened species, the black howler monkey (Alouatta caraya), greater guinea pig (Cavia magna), black-bellied slider (Trachemys dorbigni), and shrike-like cotinga (Laniisoma elegans).

In São Paulo State alone, a survey in 6.500 km of paved roads (18% of total length of paved roads in the State) recorded 37,744 roadkilled individuals, from 32 medium to large-sized mammals (0.6 animals roadkilled/km/year) in 10 years (Abra et al. unpublished). The most roadkilled species are common and generalist mammals such as Capybara (Hydrochoerus hydrochaeris, $n = 12{,}614$; 33.42%), European hare (Lepus europaeus*,* $n = 5406$; 14.32%), crab-eating fox (Cerdocyon thous*,* $n = 4957$; 13.13%), nine-banded armadillo (Dasypus novemcinctus*,* $n = 2375$; 6.29%), porcupine (*Coendou* sp., $n = 2299$; 6.09%), six-banded armadillo (Euphractus sexcinctus*,* $n = 1537$; 4.07%), southern tamandua (Tamandua tetradactyla, $n = 1193$; 3.16%), and raccoon (Procyon cancrivorus, $n = 906$; 2.40%). These species together account for more than 80% of all roadkills in the state of São Paulo.

The frequencies of roadkills vary both temporally and spatially and the extension of these impacts depends on the characteristics of the roads, such as road density, traffic volume, landscape features, proximity with protected areas, the wildlife population and their natural history (Fahrig et al. 1995; Frair et al. 2008; Freitas et al. 2015; Rytwinski and Fahrig 2013). Beyond vehicles, animals also collide with electric power lines and wind turbines, although these collisions are restricted to birds and bats. The information about animal collisions with transmission lines is scarce for the Atlantic Forest, but the available reports show that migratory birds are the main victims of this type of collisions (Marques et al. 2020).

## 14.7 Habitat Loss, Fragmentation, and Degradation

The conversion of natural landscapes to agriculture and cattle fields represents 80% of habitat loss and fragmentation worldwide being the main cause of global biodiversity loss (Laurance 2007). Changes in the landscape matrix can affect the persistence of several species, such as birds, mammals, and others in the forest fragments (Mazerolle and Villard 1999). Population sizes of large and specialist species, for example, tend to decline, recruitment rates and genetic diversity decrease, reduced due to higher inbreeding and extinction rates (see Lira et al. 2021). Meanwhile, generalist and small species tend to be benefited by the habitat loss due to the reduction and/or extinction of predators or competitors, or by the exploitation of new resources from the matrix around (Beca et al. 2017). There is evidence that habitat fragmentation severely affects the composition of local communities in the Atlantic Forest (Beca et al. 2017, see also Lira et al. 2021), creating landscapes with impoverished communities and simplified network interactions (Fahrig 2003).

The Brazilian territory has 1.2 million km$^2$ of pastures and one of the largest cattle herds with about 218 million heads, being considered the largest beef exporter in the world (Vale et al. 2019). Although human-made pasture areas could be considered heterogeneous, associated with deforestation age and type/intensity of management (Dias-Filho and Ferreira 2013), the replacement of rainforest by open farmlands has potentially severe consequences for animal biodiversity and forestry. Only a few forest species can maintain viable populations in tropical livestock systems (Esquivel et al. 2008). Moreover, livestock impacts on wildlife can be direct, through interference competition, or indirect with changes in vegetation structure that influence the availability of natural resources and nesting sites (Gonçalves et al. 2017).

The relationship between habitat and animal loss is complex to estimate, particularly because we do not have enough information on how many animals there were in the area before deforestation. However, based on a few sites where wildlife densities have been estimated, we can calculate that for every km$^2$ of forest destroyed thousands of vertebrates (particularly mammals and birds) and millions of invertebrates are extinct locally if they cannot move to adjacent forests.

For instance, at Cocha Cashu Biological Station, Peru, scientists have estimated a density of 263 understory birds, 288 primates, 127 marsupials, and 33 carnivores in 100 ha (1 km$^2$) lowland Amazon forest (Terborgh et al. 1984). In the Brazilian Atlantic Forest, the density of mammals varied with altitude, rainfall, forest productivity, but hunting pressure and forest size explained most of its diversity and abundance (Galetti et al. 2009). The mean density of terrestrial medium and large-bodied mammals is about 28 individuals per km$^2$ and the mean density of primates is about 90 individuals per km$^2$ (Galetti et al. 2017; Chiarello 2000) where highland forests have lower densities. For small mammals (rodent and marsupials), forest landscapes with a high proportion of forest cover (80–100%) are expected to hold about 1123 individuals per km$^2$ of forest specialist species (76%) that are unable to persist in smaller fragments (Bovendorp et al. 2018). A decline in forest cover favored rodent abundance of a few disturbance-adapted species (Bovendorp et al. 2018) represented by 73% of individuals in landscapes with $\leq$10% forest cover. Birds and mammals only represent a small fraction of the number of individuals and species that 1 km$^2$ of rainforest can sustain. For example, in the Brazilian Amazon, each hectare can contain one billion of invertebrate individuals (Wilson 1987).

## 14.8 Climate Change

Climate change will redistribute the biodiversity as we know it, with negative effects to ecosystem services and human well-being (Pecl et al. 2017, see also Vale et al. 2021). In the Atlantic Forest, where most forest remnants are small (<50 ha) and embedded within human-dominated matrices (Ribeiro et al. 2009), climate-driven migrations will be hampered by the lack of habitat for most terrestrial species, especially those that are canopy-dependent (Gouveia et al. 2016, see also Vale et al. 2021). Projections of climate change effects on the biodiversity of the Atlantic Forest usually indicate range shrinks, upslope and poleward movements (Ferro et al. 2014, see also Vale et al. 2021). Climate-driven faunal movements may force native species outside protected areas (Ferro et al. 2014) and/or drive invasive species inside protected areas (Loyola et al. 2014, see also Vale et al. 2021).

Amphibians are probably the taxa most readily threatened by climate change (Lemes et al. 2014; Loyola et al. 2014). Ecological characteristics such as limited dispersal abilities, water dependency on at least one life stage, and special physiological requirements make amphibians particularly sensitive to future climatic changes (Lawler et al. 2010). At the Atlantic Forest, up to 10% of amphibian species will lose all climatically suitable area by the year 2070. If species are not able to adapt to changing conditions, or migrate to newly suitable habitat, this will translate into physiological stress, with effects on fitness, reproduction, and survival, ultimately leading to species extinctions (Lawler et al. 2010). Unexplained population declines in pristine regions have already been reported on more than 29 Brazilian amphibian species, including the families Leptodactylidae (19), Hylidae (7), Centrolenidae (2), Dendrobatidae (2), and Bufonidae (Eterovick et al. 2005). Species inhabiting higher elevations seem to be at most risk and the frog species Colostethus carioca, C. olfersioides, Crossodactylus dispar, C. gaudichaudii, Cycloramphus boraceiensis, C. duseni, *C eleutherodactylus,* C. granulosus, C. semipalmatus, Hylodes babax, Paratelmatobius lutzii, Thoropa lutzi, and T. petropolitana seem to have undergone population declines in pristine areas due to climate change (Eterovick et al. 2005, see also Vale et al. 2021).

As species relocate in response to climate change, colonization by invasive species may also be enhanced. For example, alterations in climatic patterns may drive the invasive American bullfrog Lithobates catesbeianus into reserves currently established in the Atlantic Forest (Loyola et al. 2014, see also Vale et al. 2021), posing another threat to this already imperiled fauna. Climate change may accelerate disease spread among amphibians in the Atlantic Forest. The fungus *Batrachochytrium dendrobatidis* has been associated with population declines and local extinctions of several amphibian species and is now broadly distributed in the Atlantic Forest (Carnaval et al. 2006; Carvalho et al. 2017). The interaction between climate change per se and the spread of such lethal pathogens may be the cause of widespread amphibian extinctions in South American highland forests.

The spread of vector-borne diseases will likely follow the climate-driven redistribution of parasites. In addition to range expansion, the higher temperatures expected for the twenty-first century are likely to enhance transmission rates of emerging diseases to the wildlife. Climate change may, therefore, interact with other defaunation agents to reduce vertebrate biomass in the Atlantic Forests. Yellow fever has recently decimated hundreds of howler monkeys and was concomitant with extraordinarily high temperatures in South America in the last decade. However, some studies suggest that climate is rarely the main driver of epidemic bursts (Reiter 2001). Human activities and impacts on local habitats may interact with higher temperatures to affect pathogen cycles in complex ways. Habitat degradation, for example, may enhance the vulnerability of howler monkeys to yellow fever (Chapman et al. 2005).

Most projections of climate change effects on the Atlantic Forest biodiversity, however, do not account for landscape permeability across migratory routes. If landscape mosaics are impermeable to the movement of habitat-specialist species, populations will be confined to forest remnants with unsuitable climate conditions (Gouveia et al. 2016). Within such patches, novel climates are likely to exceed the amplitude, extremes, and seasonality characteristics to which such species are adapted (Ribeiro et al. 2016, see also Vale et al. 2021). Primates and other canopy-dependent groups, in addition to other forest-specialist species, may not be able to move as the climate changes (Pecl et al. 2017). The interaction between climate and land use changes is likely to prevent climate-driven migrations, by degrading the landscape configuration for Atlantic Forest vertebrates, such as titi monkeys (Gouveia et al. 2016), lion tamarins (Meyer et al. 2014), and mountain birds. In addition, climate change will also redistribute key food resources (Raghunathan et al. 2015), thus potentially affecting the fauna dependent on them.

Models based on future expectations of greenhouse gas emissions predict large-scale changes, high rates of loss of climatically suitable areas, and reorganized communities of small mammals (rodents and marsupials) in the Atlantic Forest (Bovendorp et al. unpublished).

## 14.9 The Spatial Distribution of Defaunation in the Atlantic Forest

Defaunation does not affect all the groups of animals evenly. Thirty-four percent of the amphibians, 26% of the mammals, and 16% of the birds reported to the Atlantic Forest are under some category of threat (CR, EN, NT, VU, DD; www.iucnredlist.org). Seven mammal, 10 bird, and one frog species are at critical risk of extinction in the Atlantic Forest (Bello et al. 2017; Bovendorp et al. 2017; Culot et al. 2019; Gonçalves et al. 2018a; Hasui et al. 2018; IUCN 2019; Lima et al. 2017; Muylaert et al. 2017; Vancine et al. 2018). In general, apex predators, other carnivores, large-bodied mammals, and large herbivores were among the most defaunated functional groups (Bogoni et al. 2018; Nagy-Reis et al. 2020). Similarly, the defaunation process is not evenly distributed along the Atlantic Forest. According to Bogoni et al. (2018), most Atlantic Forest remnants are classified as medium to high levels of defaunation of mammals (i.e., half of the species were lost). Regions dominated by forest converted into cropland and cattle pastures, timber extraction, and forest edges comprise the most defaunated areas in the Northeast and Southwest parts of the Atlantic Forest (ombrophilous mixed forests and semi-deciduous forests; Fig. 14.4). Historically, these areas have presented the highest human population expansion since the seventeenth century with the highest concentration of sugarcane, coffee, and cacao plantations of colonial and modern Brazil (Ribeiro et al. 2009). In addition to bioclimatic processes related to forest fragmentation, hunting, and species invasion play important roles in the maintenance of these forest fragments. The Northeast and Southwestern Atlantic Forest concentrate the remaining indigenous lands of the Atlantic Forest, where hunting is a common practice (FUNAI 2019). Besides, the inner part of the Atlantic forest has suffered the strongest invasion of feral pigs (Pedrosa et al. 2015). On the other hand, areas that are not under high defaunation process also face threats. Particularly, the concentration of roads and the recent blooms of yellow fever can decimate the remaining animal populations of the center of the Atlantic Forest (Fig. 14.4).

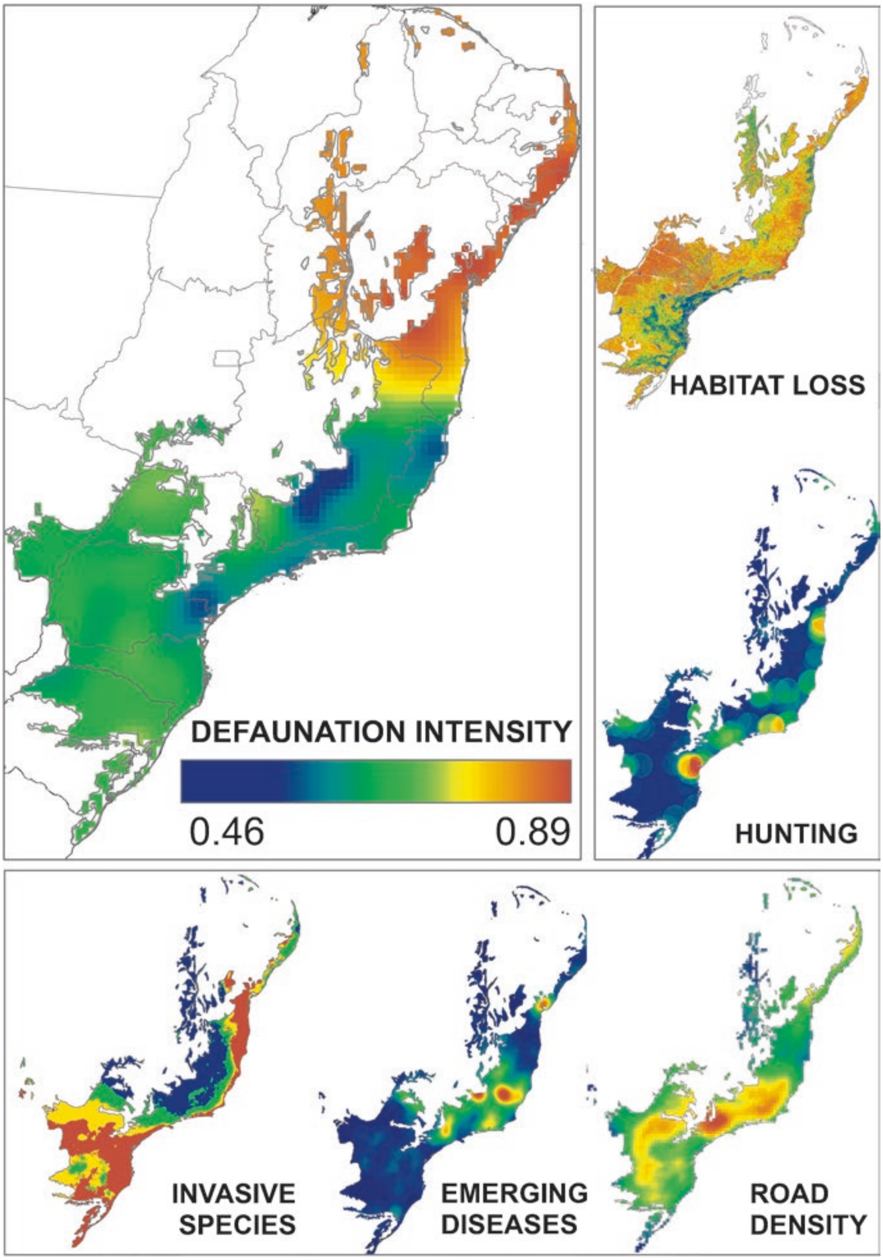

**Fig. 14.4** Defaunation of mammals in the Atlantic Forest and its drivers (according to Bogoni et al. 2018). The index range between 0 (not defaunated) to 1 (highly defaunated). Red colors represent areas with high concentration of each driver while blue colors represent low concentration of each driver

## 14.10 Functional Defaunation, Trophic Cascades and the Loss of the Atlantic Forest Top Predators

Species extinctions and reductions in animal populations promote pervasive trophic cascades that affect ecological processes and functions, disease dissemination, fire dynamics, biogeochemical cycles, and more (Dirzo et al. 2014). An iconic example is the case of Yellowstone National Park (USA), where the extinction of wolves (Canis lupus) triggered a series of trophic cascades through increasing the population of its prey, the elk (Cervus elaphus), which in turn caused significant changes in the habitat through limiting the recruitment of certain plant species and ultimately led to profound changes in riparian ecosystems (Beschta and Ripple 2018). Top-down control by herbivores is widely determinant for plant communities (Jia et al. 2018). Therefore, annihilation of such animals is likely to significantly modify the shape of ecological communities as we see now. Although defaunation is a local process, its effects could scale up globally, for example, through shifts in carbon stocks (Bello et al. 2015). Upscaling effects are likely to be more pronounced in tropical forests.

Most of the Atlantic Forest fragments are under high levels of defaunation, having lost animal populations from every functional group (Bogoni et al. 2018), with likely consequences for ecosystem structure and functioning. Estimates suggest that the largest apex predator (jaguar – Panthera onca), the largest herbivore (tapir – Tapirus terrestris), the largest seed predator (white-lipped peccary – Tayassu pecari), and the largest arboreal seed disperser (muriqui – *Brachyteles* spp.) are missing from 88% of the Atlantic Forest (Jorge et al. 2013). Jaguars are functionally extinct in most of its original distribution (Galetti et al. 2013a; Nagy-Reis et al. 2020) and less than 300 individuals are left in the whole biome, divided into sub-populations of extremely low densities, of which only three are considered viable (Paviolo et al. 2016). Historically, the main driver of jaguar population declines was land conversion and fragmentation, which not only reduced their habitat, but further resulted in population declines of their prey. Currently, retaliation to livestock predation and roadkill are the main threat to the few remaining jaguar individuals that still roam through the Atlantic Forest. According to the trophic cascade model such declines should contribute toward increasing prey population densities, higher consumption of edible plants, and a series of cascading effects that should alter entire ecological communities throughout the whole biome. Critically, empirical evidence for top-down control of herbivorous prey by jaguar is still missing, yet evidence of trophic cascades is well supported by defaunation studies on herbivory, seed dispersal, and indirect interactions at lower trophic levels.

## 14.11 The Loss of Critical Seed Dispersal Functions

Seed dispersal is a critical ecosystem process heavily affected by defaunation. About 89% of all woody plant species in the Atlantic Forest are animal dispersed (Almeida-Neto et al. 2008), with more than 331 vertebrate frugivore species (including birds, mammals, fishes, amphibians, and reptiles) in the biome (Bello et al. 2017). Vertebrate species inhabiting the Atlantic Forest vary in their functional role as seed dispersal agents, and therefore the differential defaunation might affect ecosystem processes in different ways. For example, muriqui monkeys (*Brachyteles* spp.) and tapirs (Tapirus terrestris) are efficient seed dispersers thought to play a complementary role in seed dispersal through their different size-selective and spatial seed deposition patterns. This is also true even within functional guilds, as different primate species also have different seed deposition patterns (Culot et al. 2017). Large terrestrial species contribute disproportionately to long-distance dispersal (Pires et al. 2018) and thus might play a critical role in genetic flow among populations of fruiting trees. Scatter hoarders are effective seed dispersal agents that reduce predation of seeds by insects, and whose dispersal function collapses in fragments below <1000 ha (Galetti et al. 2006).

By taking a closer look at the quality of seed dispersal mediated by Atlantic Forest mammals, evidence shows species may differentially affect seed fate because their feces attract different abundances and richness of dung beetles (Scarabaeidae: Scarabaeinae), which secondarily disperse those seeds (Lugon et al. 2017). Extirpation of large- and medium-sized herbivores in the Atlantic Forest has modified the structure of dung beetle communities and has likely caused co-extinctions (Fig. 14.5; Culot et al. 2013; Genes et al. 2019). Dung beetle abundance and diversity respond to mammal declines because they rely on mammalian droppings for feeding and nesting (Andresen 1999). While manipulating the feces, dung beetles secondarily disperse seeds to microsites that are generally favorable for germination and recruitment (Nichols et al. 2008). As a result, defaunation is likely to additionally affect plant communities through declines in secondary seed dispersal mediated by an impoverished community of dung beetles. Generally, smaller-sized dung beetles dominate in defaunated areas (Fig. 14.5; Culot et al. 2013). Since their size restricts the size of the seeds they can secondari0ly disperse, the dispersal of larger-seeded plant species could be negatively affected.

Fruits from large and dense wood tree species in the Atlantic Forest are generally dispersed by large frugivores, and therefore the extinction of the latter may cause a reduction in up to 5.8% carbon stocked on forests over time (Fig. 14.5; Bello et al. 2015). Furthermore, the functional extinction of large seed dispersers and large seed and seedling predators can have synergic negative effects on the recruitment of large-seeded plants, reducing recruitment success from 30% in an intact community to 7.5% in a strongly defaunated one (Fig. 14.5; Culot et al. 2017), with subsequent effects on carbon storage.

The loss of seed dispersal function carried out by large frugivores has also a strong impact on the phenotypic and genotypic architecture of tree populations. In

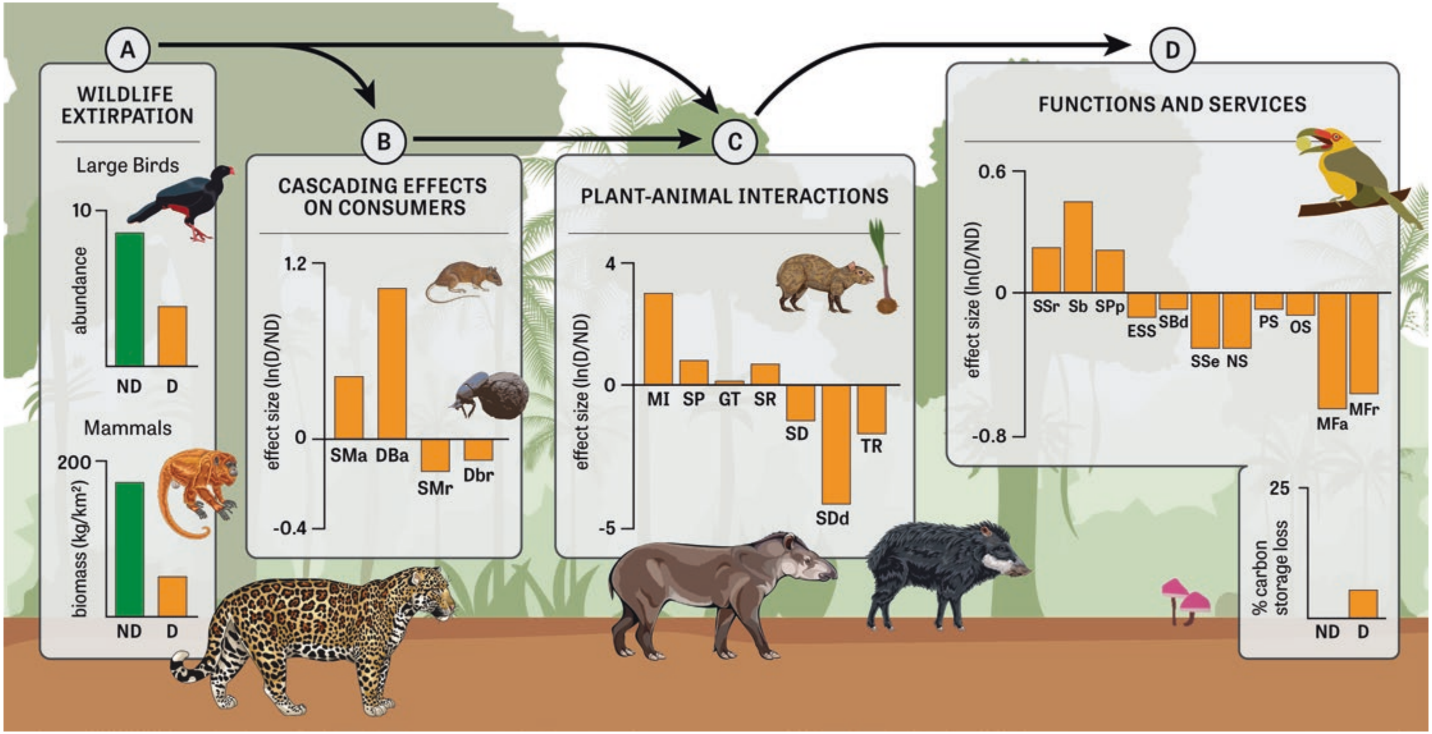

**Fig. 14.5** Evidence of the ecological consequences of defaunation in the Atlantic Forest. Non-defaunated areas (ND) in green and defaunated areas (D) in orange. (a) Mean abundance decline of large frugivorous birds (Galetti et al. 2013b) and arboreal and terrestrial mammal mean biomass loss in defaunated fragments (Galetti et al. 2017). (b) Wildlife extirpation leads to increases on small-mammal (SMa) and dung beetle abundance (DBa), while small-mammal (SMr) and dung beetle richness (DBr) decreases (Galetti et al. 2015a; Culot et al. 2013). (c) Defaunation also affects plant-animal interactions through increasing the number of seed disperser interactions that are missing (MI, Marjakangas et al. 2018), increasing seed predation (SP, Galetti et al. 2015b), increasing germination time (GT), and increasing the number of seedling (SR, Villar et al. 2020). Defaunation decreases seed dispersal (SD, Galetti et al. 2006), maximum seed dispersal distance (SDd) (Culot et al. 2017), and trampling (TR) (Villar et al. 2020). (d) Defaunation and changes in plant-animal interaction patterns alter ecological functions and services, increasing mean seedling species richness (SSr), total seedling biomass (Sb), and total seedling productivity (SPp, Villar et al. 2020), decline in seedling beta diversity (SBd), seedling species evenness (SSe) (Villar et al. 2020), available nitrogen in soil (NS), total phosphorus in soil (PS), and organic matter in soil (OS) (Villar et al. 2020), and reduction in the number of arbuscular mycorrhizal fungal (AMF) spores (MFa) and AMF morphotype richness (MFr) (Paz et al. unpub.) Loss of large bird seed dispersers caused the reduction of palm seed size (ESS, Galetti et al. 2013b). Defaunation of large herbivores also erodes carbon storage (Bello et al. 2015)

forest fragments smaller than 10.000 ha, bird dispersal of large seeds has been lost and only interactions between small generalist birds and small-seeded plant species remain (Emer et al. 2018). Using a key resource tree species as a model system (the juçara palm *Euterpe edulis*), recent studies have shown that the functional extinction of birds drives rapid evolutionary changes in seed size (Fig. 14.5; Galetti et al. 2013b), that also has a genotypic basis (Carvalho et al. 2016). Critically, such studies indicate that selection for smaller seeds on defaunated landscapes occurs on ecological time scales and at a very fast pace, suggesting that the functional extinction of large seed dispersers might have profound and perhaps irreversible consequences on seed traits over evolutionary timescales.

## 14.12 The Ecological Consequences of Losing Large Herbivores

If in one hand seed dispersal is crucial for the maintenance of high plant diversity in the Atlantic Forest, antagonist interactions also play an important role; however, empirical evidence is still scarce. Lower herbivory and seed predation rates due to loss of fauna are expected to have variable impacts on the trophic interactions, depending on the abundance of the animals and regional environmental condition. For example, the dominance of palatable plant species was found to increase with loss of herbivores, reducing the local diversity of plants in a tropical forest (Harrison et al. 2013). As a result of the loss of large herbivores and increase in dominance of certain plant species, both functional and life-form diversity among plant communities can be affected (Bulascoschi-Cagnoni et al. unpublished, Souza et al. unpublished), which might in turn lead to changes in ecosystem functions, such as C storage, decomposition of organic matter, water and nutrient retention (Lavorel and Garnier 2002).

Another effect of defaunation of large mammals in the Atlantic Forest is the competitive release of small mammals (a phenomenon also known as "rodentization", Fig. 14.5; Galetti et al. 2015a), which can trigger subsequent compensatory seed predation by small mammals. For instance, it has been shown that this process leads to a twofold increase in seed predation of a threatened keystone palm species (*Euterpe edulis,* Fig. 14.5; Galetti et al. 2015b), and demographic simulations calibrated with empirical data suggest that this increase in seed predation might have further consequences for recruitment rates in other large-seeded species (Fig. 14.5; Culot et al. 2017). On the other hand, long-term experiments in the Atlantic Forest show that excluding large mammals increases the abundance of palm species and alters the dominance of plant species, and results in an increase in community-level seedling recruitment, primary productivity, and biomass (Villar et al. 2020). Clearly, more evidence is needed to discern in which ecological contexts defaunation leads to increases or decreases in plant recruitment. Nonetheless, the rodentization of the Atlantic Forest due to the loss of large mammals may also have potential consequences for the increase of emerging diseases, such as hantavirus (Galetti et al. 2015a).

Non-trophic effects also result from defaunation processes. Some large herbivore/omnivorous mammals such as the peccaries can be considered as “ecosystem engineers” because of their indirect effects on plants, vertebrates, and soil organisms (Fleming et al. 2014; Coggan et al. 2018). These indirect effects relate with their foraging activities as they disturb the soil surface in search for soil organisms and roots, creating spatial heterogeneity that benefits other vertebrates (e.g., creating optimum ponds for anurans; Beck et al. 2010), soil invertebrates (e.g., termites, Coggan et al. 2018), and soil microbial communities (Eldridge et al. 2016). In addition to decreasing seedling recruitment, primary productivity and biomass, large seed and seedling predators of the Atlantic Forest such as the white-lipped peccary (Tayassu pecari), for instance, can also have substantial effects on seedling communities through trampling and increased soil compaction (Villar et al. 2020). Furthermore, as a result of their top-down control of plant communities and foraging behavior in large herds peccaries can affect soil nutrient cycling (e.g., ammonium- and nitrate-N; Villar et al. 2020, Fig. 14.5) and soil carbon dynamics, which consequently influence plant productivity and alpha- and beta-diversity (Villar et al. 2020, Fig. 14.5) and the diversity of the root-symbionts (e.g., arbuscular mycorrhizal fungi, Paz et al. unpublished, Fig. 14.5). Yet, the evidence at hand still represents the tip of the iceberg, and the longer-term consequences of local extinctions or reductions in the abundance of large wildlife from the Atlantic Forest still remain largely unknown.

## 14.13 How to Revert Defaunation and Its Consequences: A Tool Kit

The Atlantic Forest lost 88% of its original distribution and its biodiversity is rapidly eroding, but paradoxically we have experienced a few global extinctions. Very rare populations have been monitored over large periods (e.g., Golden lion tamarin) and there is still a knowledge gap on most vertebrate species population size (Joly et al. 2014).

Given the extent of the consequences of defaunation in the Atlantic Forest, mitigation strategies are imperative. Habitat restoration would clearly be effective in building space for defaunation mitigation. Despite all the logistical challenges, ambitious programs such as *The Atlantic Forest Restoration Pact* aims to restore 15 million ha by 2050 (Crouzeilles et al. 2019). However, considering that most of the matrix between forest fragments are composed by agricultural lands or cities, restoration will not be suitable to every patch and vertebrates will not be able to re-colonize most forest fragments without being assisted. Reintroduction of the golden lion tamarin (Leontopithecus rosalia) has successfully rescued its population from the brink of extinction (Kierulff et al. 2012), a good example of species restoration project. Although some progress has been made toward understanding the extent to which ecological processes can be restored through rewilding (Genes et al. 2019), and which areas would best benefit from the reintroduction of seed dispersers (Marjakangas et al. 2018), it is still unclear how to scale-up such conservation strategy in practice. Reversing the pervasive defaunation that occurred in the Atlantic Forest is by no means a straightforward task. Nonetheless, it will be fundamental to assure the persistence of the biodiversity in the Atlantic Forest remnants.

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## Literature Cited

Abreu FVS, Delatorre E, Santos AAC, Ferreira-de-Brito A, Castro MG, Ribeiro IP, Furtado ND, Vargas WP, Ribeiro MS, Meneguete P, Bonaldo MC (2019) Combination of surveillance tools reveals that yellow fever virus can remain in the same Atlantic Forest area at least for three transmission seasons. Memorias do Instituto Oswaldo Cruz 114:e190076

Alves RRN, Lima JRF, Araujo FPA (2013) The live bird trade in Brazil and its conservation implications: an overview. Bird Conser Int 23(1):53–65. https://doi.org/10.1017/S095927091200010X

de Almeida Curi NH, Massara RL, de Oliveira Paschoal AM, Soriano-Araújo A, Lobato ZIP, Demétrio GR, Chiarello AG, Passamani M (2016) Prevalence and risk factors for viral exposure in rural dogs around protected areas of the Atlantic forest. BMC Vet Res 12:21. https://doi.org/10.1186/s12917-016-0646-3

Almeida-Neto M, Campassi F, Galetti M, Jordano P, Oliveira-Filho A (2008) Vertebrate dispersal syndromes along the Atlantic forest: broad-scale patterns and macroecological correlates. Glob Ecol Biogeogr 17:503–513. https://doi.org/10.1111/j.1466-8238.2008.00386.x

Alvard MS (1998) Evolutionary ecology and resource conservation. Evol Anthropol Issues News Rev 7(2):62–74

Andresen E (1999) Seed dispersal by monkeys and the fate of dispersed seeds in a Peruvian rain forest. Biotropica 31:145

Beca G, Vancine MH, Carvalho CS, Pedrosa F, Alves RSC, Buscariol D, Peres CA, Ribeiro MC, Galetti M (2017) High mammal species turnover in forest patches immersed in biofuel plantations. Biol Conserv 210:352–359

Beck H, Thebpanya P, Filiaggi M (2010) Do Neotropical peccary species (Tayassuidae) function as ecosystem engineers for anurans? J Trop Ecol 26(4):407–414

Bello C, Galetti M, Pizo MA, Magnago LFS, Rocha MF, Lima RA, Peres CA, Ovaskainen O, Jordano P (2015) Defaunation affects carbon storage in tropical forests. Sci Adv 1:1501105

Bello C, Galetti M, Montan D, Pizo MA, Mariguela TC, Culot L, Bufalo F, Labecca F, Pedrosa F, Constantini R, Emer C, Silva WR, da Silva FR, Ovaskainen O, Jordano P (2017) Atlantic frugivory: a plant–frugivore interaction data set for the Atlantic Forest. Ecology 98:1729. https://doi.org/10.1002/ecy.1818

Beschta RL, Ripple WJ (2018) Can large carnivores change streams via a trophic cascade? Ecohydrology 12:e2048

BirdLife International 2019 IUCN Red List for birds. Downloaded from http://www.birdlife.org on 11/05/2019

Bodmer RE, Eisenberg JF, Redford KH (1997) Hunting and the likelihood of extinction of Amazonian mammals: Caza y Probabilidad de Extinción de Mamiferos Amazónicos. Conserv Biol 11(2):460–466

Bogoni JA, Pires JSR, Graipel ME, Peroni N, Peres CA (2018) Wish you were here: how defaunated is the Atlantic Forest biome of its medium-to large-bodied mammal fauna? PLoS One 13:e0204515

Bonnaud E, Medina FM, Vidal E, Nogales M, Tershy B, Zavaleta E, Donlan CJ, Keitt B, Le Corre M, Horwath SV (2010) The diet of feral cats on islands: a review and a call for more studies. Biol Invasions 13:581–603

Bovendorp RS, Villar N, Abreu-Junior EF, Bello C, Regolin AL, Percequillo AR, Galetti M (2017) Atlantic small-mammal: a dataset of communities of rodents and marsupials of the Atlantic forests of South America. Ecology 98:2226. https://doi.org/10.1002/ecy.1893

Bovendorp RS, Brum FT, McCleery RA, Baiser B, Loyola R, Cianciaruso MV, Galetti M (2018) Defaunation and fragmentation erode small mammal diversity dimensions in tropical forests. Ecography 42:23–35

Brasil MS (2019) Centro de operações de emergências em saúde pública sobre febre amarela. Monitoramento do Período Sazonal da Febre Amarela Brasil 2018/2019

Canale GR, Peres CA, Guidorizzi CE, Gatto CAF, Kierulff MCM (2012) Pervasive defaunation of forest remnants in a tropical biodiversity hotspot. PLoS One 7:e41671. https://doi.org/10.1371/journal.pone.0041671

Cardoso DC, Silva LG, Rabelo RM (2019) Predicting yellow fever through species distribution modeling of virus, vector and monkeys. EcoHealth 16:95–108. https://doi.org/10.1007/s10393-018-1388-4

Carnaval ACOQ, Puschendorf R, Peixoto OL, Verdade VK, Rodrigues MT (2006) Amphibian Chytrid fungus broadly distributed in the Brazilian Atlantic Rain Forest. EcoHealth 3:41–48. https://doi.org/10.1007/s10393-005-0008-2

Carvalho CS, Galetti M, Colevatti RG, Jordano P (2016) Defaunation leads to microevolutionary changes in a tropical palm. Sci Rep 6:31957

Carvalho T, Becker CG, Toledo LF (2017) Historical amphibian declines and extinctions in Brazil linked to chytridiomycosis. Proc R Soc B Biol Sci 284:20162254. https://doi.org/10.1098/rspb.2016.2254

Ceballos G, Ehrlich PR, Barnosky AD, Garcia A, Pringle RM, Palmer TM (2015) Accelerated modern human induced species losses: entering the sixth mass extinction. Sci Adv 1:e1400253. https://doi.org/10.1126/sciadv.1400253

Chapman CA, Gillespie TR, Goldberg TL (2005) Primates and the ecology of their infectious diseases: how will anthropogenic change affect host-parasite interactions? Evol Anthropol Issues News Rev 14:134–144. https://doi.org/10.1002/evan.20068

Chiarello AG (2000) Density and population size of mammals in remnants of Brazilian Atlantic Forest. Conserv Biol 14:1649–1657. https://doi.org/10.1111/j.1523-1739.2000.99071.x

Coggan NV, Hayward MW, Gibb H (2018) A global database and "state of the field" review of research into ecosystem engineering by land animals. J Anim Ecol 87:974–994. https://doi.org/10.1111/1365-2656.12819

Courtenay O, Quinnell RJ, Chalmers WSK (2001) Contact rates between wild and domestic canids: no evidence of parvovirus or canine distemper virus in crab-eating foxes. Vet Microbiol 81:9–19. https://doi.org/10.1016/S0378-1135(01)00326-1

Crouzeilles R, Santiami E, Rosa M, Pugliese L, Brancalion PHS, Rodrigues RR, Metzger JP, Calmon M, de Scaramuzza M, Matsumoto MH, Padovezi A, de Benini RM, Chaves RB, Metzker T, Fernandes RB, Scarano FR, Schmitt J, Lui G, Christ P, Vieira RM, Senta MMD, Malaguti GA, Strassburg BBN, Pinto S (2019) There is hope for achieving ambitious Atlantic Forest restoration commitments. Perspect Ecol Conserv 17:80–83. https://doi.org/10.1016/j.pecon.2019.04.003

Cullen L Jr, Bodmer RE, Valladares-Pádua C (2001) Ecological consequences of hunting in Atlantic forest patches, São Paulo, Brazil. Oryx 35:137–144

Culot L, Bovy E, Vaz-de-Mello FZ, Guevara R, Galetti M (2013) Selective defaunation affects dung beetle communities in continuous Atlantic rainforest. Biol Conserv 163:79–89. https://doi.org/10.1016/j.biocon.2013.04.004

Culot L, Bello C, Batista JLF, do Couto HTZ, Galetti M (2017) Synergistic effects of seed disperser and predator loss on recruitment success and long-term consequences for carbon stocks in tropical rainforests. Sci Rep 7:7662. https://doi.org/10.1038/s41598-017-08222-4

Culot L, Pereira LA, Agostini I, Almeida MAB, Alves RSC, Aximoff I et al (2019) ATLANTIC-PRIMATES: a dataset of communities and occurrences of primates in the Atlantic Forests of South America. Ecology 100:e02525. https://doi.org/10.1002/ecy.2525

Daszak P, Cunningham AA, Hyatt AD (2000) Emerging infectious diseases of wildlife--threats to biodiversity and human health. Science 287:443–449. https://doi.org/10.1126/science.287.5452.443

Dean W (1997) With broadax and firebrand: the destruction of the Brazilian Atlantic Forest. University of California Press, Berkeley

Desbiez ALJ, Keuroghlian A, Piovezan U, Bodmer RE (2011) Invasive species and bushmeat hunting contributing to wildlife conservation: the case of feral pigs in a Neotropical wetland. Oryx 45(1):78–83

Dias-Filho MB, Ferreira JN (2013) As pastagens e o meio ambiente. In: Reis, RA, Bernardes TF, Siqueira, GR (Ed.) Forragicultura: ciência, tecnologia e gestão dos recursos forrageiros. Jaboticabal, p. 93–105

Dirzo R, Young HS, Galetti M, Ceballos G, Isaac NJB, Collen B (2014) Defaunation in the Anthropocene. Science 345:401–406. https://doi.org/10.1126/science.1251817

DNIT – Departamento Nacional de Infraestrutra de Transporte. Available at: http://www.dnit.gov.br/. Accessed in: 18/04/2019

Doherty TS, Dickman CR, Glen AS, Newsome TM, Nimmo DG, Ritchie EG, Vanak AT, Wirsing AJ (2017) The global impacts of domestic dogs on threatened vertebrates. Biol Conserv 210:56–59. https://doi.org/10.1016/j.biocon.2017.04.007

Dovrat G, Perevolotsky A, Neeman G (2012) Wild boars as seed dispersal agents of exotic plants from agricultural lands to conservation areas. J Arid Environ 78:49–54. https://doi.org/10.1016/j.jaridenv.2011.11.011

Eldridge DJ, Delgado-Baquerizo M, Woodhouse JN, Neilan BA (2016) Mammalian engineers drive soil microbial communities and ecosystem functions across a disturbance gradient. J Anim Ecol 85:1636–1646. https://doi.org/10.1111/1365-2656.12574

Emer C, Galetti M, Pizo MA, Guimarães PR Jr, Moraes S, Piratelli A, Jordano P (2018) Seed-dispersal interactions in fragmented landscapes – a metanetwork approach. Ecol Lett 21:484–493. https://doi.org/10.1111/ele.12909

Esquivel MJ, Harvey CA, Finegan B, Casanoves F, Skarpe C (2008) Effects of pasture management on the natural regeneration of neotropical trees. J Appl Ecol 45:371–380. https://doi.org/10.1111/j.1365-2664.2007.01411.x

Eterovick PC, De Queiroz Carnaval ACO, Borges-Nojosa DM, Silvano DL, Segalla MV, Sazima I (2005) Amphibian declines in Brazil: an overview. Biotropica 37:166–179. https://doi.org/10.1111/j.1744-7429.2005.00024.x

Fahrig L (2003) Effects of habitat fragmentation on biodiversity. Annu Rev Ecol Evol Syst 34:487–515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Fahrig L, Rytwinski T (2009) Effects of roads on animal abundance: an empirical review and synthesis. Ecol Soc 14(1):21. http://www.ecologyandsociety.org/vol14/iss1/art21/

Fahrig L, Pedlar JH, Pope SE, Taylor PD, Wegner JF (1995) Effect of road traffic on amphibian density. Biol Conserv 73:177–182. https://doi.org/10.1016/0006-3207(94)00102-V

Ferreguetti AC, Davis CL, Tomas WM, Bergallo HG (2018) Using activity and occupancy to evaluate niche partitioning: the case of two peccary species in the Atlantic Rainforest, Brazil. Hystrix. Ital J Mammal 29:168–174. https://doi.org/10.4404/hystrix-00068-2018

Ferro VG, Lemes P, Melo AS, Loyola R (2014) The reduced effectiveness of protected areas under climate change threatens Atlantic forest tiger moths. PLOS ONE 9(9):e107792. https://doi.org/10.1371/journal.pone.0107792

Fleming PA, Anderson H, Prendergast AS, Bretz MR, Valentine LE, Hardy GES (2014) Is the loss of Australian digging mammals contributing to a deterioration in ecosystem function? Mammal Rev 44:94–108. https://doi.org/10.1111/mam.12014

Forman RTT, Alexander LE (1998) Roads and their major ecological effects. Annu Rev Ecol Syst 29:207–231. https://doi.org/10.1146/annurev.ecolsys.29.1.207

Forman RTT, Sperling D, Bissonette JA, Clevenger AP, Cutshall CD, Dale VH et al (2003) Road ecology: science and solutions. Island Press, Washington DC

Frair JL, Merrill EH, Beyer HL, Morales JM (2008) Thresholds in landscape connectivity and mortality risks in response to growing road networks. J Appl Ecol 45:1504–1513. https://doi.org/10.1111/j.1365-2664.2008.01526.x

Freitas SR, Oliveira AN, Ciocheti G, Vieira MV, Matos DMS (2015) How landscape features influence road-kill of three species of mammals in the Brazilian savanna? Oecologia Australis 18:35–45

Frisch JD, Frisch CD (1995) O jardim dos beija-flores. Ed. Dalgas- Ecoltec, São Paulo

FUNAI (2019) Terras Indígenas. Fundação do Índio- Funai Ministério de Justiça. http://mapas2.funai.gov.br/portal_mapas/pdf/terra_indigena.pdf

Galetti M, Sazima I (2006) Impact of feral dogs in an urban Atlantic forest fragment in southeastern Brazil. Natureza e Conservação 4:146–151

Galetti M, Donatti CI, Pires AS, Guimarães PR Jr, Jordano P (2006) Seed survival and dispersal of an endemic Atlantic forest palm: the combined effects of defaunation and forest fragmentation. Bot J Linn Soc 151:141–149. https://doi.org/10.1111/j.1095-8339.2006.00529.x

Galetti M, Giacomini HC, Bueno RS, Bernardo CSS, Marques RM, Bovendorp RS, Steffler CE, Rubim P, Gobbo SK, Donatti CI, Begotti RA, Meirelles F, de Nobre RA, Chiarello AG, Peres CA (2009) Priority areas for the conservation of Atlantic forest large mammals. Biol Conserv 142:1229–1241. https://doi.org/10.1016/j.biocon.2009.01.023

Galetti M, Eizirik E, Beisiegel B, Ferraz K, Cavalcanti S, Srbek-Araujo AC, Crawshaw P, Paviolo A, Galetti PM, Jorge ML, Marinho-Filho J, Vercillo U, Morato R (2013a) Atlantic Rainforest's jaguars in decline. Science 342:930. https://doi.org/10.1126/science.342.6161.930-a

Galetti M, Guevara R, Côrtes MC, Fadini R, Von Matter S, Leite AB, Labecca F, Ribeiro T, Carvalho CS, Collevatti RG, Pires MM, Guimarães PR, Brancalion PH, Ribeiro MC, Jordano P (2013b) Functional extinction of birds drives rapid evolutionary changes in seed size. Science 340:1086–1090. https://doi.org/10.1126/science.1233774

Galetti M, Guevara R, Neves CL, Rodarte RR, Bovendorp RS, Moreira M, Hopkins JB, Yeakel JD (2015a) Defaunation affects the populations and diets of rodents in Neotropical rainforests. Biol Conserv 190:2–7. https://doi.org/10.1016/j.biocon.2015.04.032

Galetti M, Bovendorp RS, Guevara R (2015b) Defaunation of large mammals leads to an increase in seed predation in the Atlantic forests. Glob Ecol Conserv 3:824–830. https://doi.org/10.1016/j.gecco.2015.04.008

Galetti M, Brocardo CR, Begotti RA, Hortenci L, Rocha-Mendes F, Bernardo CSS, Bueno RS, Nobre R, Bovendorp RS, Marques RM, Meirelles F, Gobbo SK, Beca G, Schmaedecke G, Siqueira T (2017) Defaunation and biomass collapse of mammals in the largest Atlantic forest remnant. Anim Conserv 20:270–281. https://doi.org/10.1111/acv.12311

Gastañaga M, MacLeod R, Hennessey B, Nunez JU, Puse E, Arrascue A, Hoyos J, Chambi WM, Vasquez J, Engblom G (2011) A study of the parrot trade in Peru and the potential importance of internal trade for threatened species. Bird Conserv Int 21(1):76–85

Genes L, Fernandez FAS, Vaz-de-Mello FZ, da Rosa P, Fernandez E, Pires AS (2019) Effects of howler monkey reintroduction on ecological interactions and processes. Conserv Biol 33:88–98. https://doi.org/10.1111/cobi.13188

Gompper ME (2014) Free-ranging dogs and wildlife conservation, First edn. Oxford University Press, Oxford, UK

Gonçalves F, Fischer E, Dirzo R (2017) Forest conversion to cattle ranching differentially affects taxonomic and functional groups of Neotropical bats. Biol Conserv 210(January):343–348. https://doi.org/10.1016/j.biocon.2017.04.021

Gonçalves F, Bovendorp RS, Beca G, Bello C, Costa-pereira R, Renata L et al (2018a) Atlantic mammal traits: a dataset of morphological traits of mammals in the Atlantic Forest of South America. Ecology 99:498

Gonçalves F, Hannibal W, Godoi MN, Martins FI, Oliveira RF, Figueiredo VV, Casella J, Sá ÉFGG (2018b) Non-volant mammals from the Upper Paraná River Basin: a data set from a critical region for conservation in Brazil. Ecology 99:499. https://doi.org/10.1002/ecy.2107

Gonçalves F, Magioli M, Bovendorp R, Ferraz KMPMB, Bulascoschi L, Moreira MZ, Galetti M (2020) Prey choice of introduced species by the common vampire bat on an Atlantic forest land-bridge island. Acta Chiropterol 22(1):167–174. https://doi.org/10.3161/15081109ACC2020.22.1.015

Gouveia SF, Souza-Alves JP, Rattis L, Dobrovolski R, Jerusalinsky L, Beltrão-Mendes R, Ferrari SF (2016) Climate and land use changes will degrade the configuration of the landscape for titi monkeys in eastern Brazil. Glob Chang Biol 22:2003–2012. https://doi.org/10.1111/gcb.13162

Grilo C, Coimbra MR, Cerqueira RC, Barbosa P, Dornas RAP, Gonçalves LO et al (2018) Brazil road-kill: a data set of wildlife terrestrial vertebrate road-kills. Ecology 99:2625. https://doi.org/10.1002/ecy.2464

Hansen MC, Potapov PV, Moore R, Hancher M, Turubanova SA, Tyukavina A, Thau D, Stehman SV, Goetz SJ, Loveland TR, Kommareddy A, Egorov A, Chini L, Justice CO, Townshend JRG (2013) High-resolution global maps of 21st-century forest cover change. Science 342:850–853. https://doi.org/10.1126/science.1244693

Harrison RD, Tan S, Plotkin JB, Slik F, Detto M, Brenes T, Itoh A, Davies SJ (2013) Consequences of defaunation for a tropical tree community. Ecol Lett 16:687–694. https://doi.org/10.1111/ele.12102

Hasui E, Metzger JP, Pimentel RG, Silveira LF, Bovo AADA, Martensen AC et al (2018) Atlantic birds: a data set of bird species from the Brazilian Atlantic Forest. Ecology 99:497. https://doi.org/10.1002/ecy.2119

Hudson PJ (2002) The ecology of wildlife disease, 1st edn. Oxford University Press, Oxford

ICMBio (2018) Livro Vermelho da Fauna Brasileira Ameaçada de Extinção: ed. ICMBio/MMA, Brasília, DF, 492 p

Ilse LM, Hellgren EC (1995) Resource partitioning in sympatric populations of collared peccaries and feral hogs in Southern Texas. J Mammal 76:784–799. https://doi.org/10.2307/1382747

IUCN (2019) IUCN red list of threatened species. Version 2019-1 www.iucnredlist.org

Jia S, Wang X, Yuan Z, Lin F, Ye J, Hao Z, Luskin MS (2018) Global signal of top-down control of terrestrial plant communities by herbivores. Proc Natl Acad Sci 115:6237

Joly CA, Metzger JP, Tabarelli M (2014) Experiences from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. New Phytol 204:459–473. https://doi.org/10.1111/nph.12989

Jorge MLSP, Galetti M, Ribeiro MC, Ferraz KMPMB (2013) Mammal defaunation as surrogate of trophic cascades in a biodiversity hotspot. Biol Conserv 163:49–57. https://doi.org/10.1016/j.biocon.2013.04.018

Kierulff MCM, Ruiz-Miranda CR, de Oliveira PP, Beck BB, Martins A, Dietz JM, Rambaldi DM, Baker AJ (2012) The Golden lion tamarin Leontopithecus rosalia: a conservation success story. Int Zoo Yearbook 46:36–45. https://doi.org/10.1111/j.1748-1090.2012.00170.x

Laurance F (2007) Have we overstated the tropical biodiversity crisis?. Trends Ecol Evol 22 (2):65–70

Lavorel S, Garnier E (2002) Predicting changes in community composition and ecosystem functioning from plant traits: revisiting the Holy Grail. Funct Ecol 16:545–556. https://doi.org/10.1046/j.1365-2435.2002.00664.x

Lawler JJ, Shafer SL, Bancroft BA, Blaustein AR (2010) Projected climate impacts for the amphibians of the Western Hemisphere. Conserv Biol 24:38–50. https://doi.org/10.1111/j.1523-1739.2009.01403.x

Lemes P, Melo AS, Loyola RD (2014) Climate change threatens protected areas of the Atlantic Forest. Biodivers Conserv 23:357–368. https://doi.org/10.1007/s10531-013-0605-2

Lesbarrères D, Fahrig L (2012) Measures to reduce population fragmentation by roads: what has worked and how do we know?. Trends Ecol Evol 27(7):374–380. https://doi.org/10.1016/j.tree.2012.01.015

Lima F, Beca G, Muylaert RL, Jenkins CN, Perilli MLL, Paschoal AMO et al (2017) ATLANTIC-CAMTRAPS: a dataset of medium and large terrestrial mammal communities in the Atlantic Forest of South America. Ecology 98:2979. https://doi.org/10.1002/ecy.1998

Lira PK, Portela RCQ, Tambosi LR (2021) Land-cover changes and an uncertain future: will the Brazilian Atlantic Forest lose the chance to become a hopespot? In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Lowe S, Browne M, Boudjelas S, De Poorter M (2000) 100 of the World's worst invasive alien species a selection from the global invasive species database. Published by the invasive species specialist group (ISSG) a specialist group of the species survival commission (SSC) of the world conservation union (IUCN), 12pp. First published as special lift-out in Aliens 12, December 2000. Updated and reprinted version: November 2004

Loyola RD, Lemes P, Brum FT, Provete DB, Duarte LDS (2014) Clade-specific consequences of climate change to amphibians in Atlantic Forest protected areas. Ecography 37:65–72. https://doi.org/10.1111/j.1600-0587.2013.00396.x

Lugon AP, Boutefeu M, Bovy E, Vaz-de-Mello FZ, Huynen M-C, Galetti M, Culot L (2017) Persistence of the effect of frugivore identity on post-dispersal seed fate: consequences for the assessment of functional redundancy. Biotropica 49:293–302. https://doi.org/10.1111/btp.12418

Lynes BC, Campbell SD (2000) Germination and viability of mesquite (Prosopis pallida) seed following ingestion and excretion by feral pigs (Sus scrofa). Trop Grasslands 34(2):125–128

Maciel ALG, Loiko MR, Bueno TS, Moreira JG, Coppola M, Dalla Costa ER, Schmid KB, Rodrigues RO, Cibulski SP, Bertagnolli AC, Mayer FQ (2018) Tuberculosis in southern Brazilian wild boars (Sus scrofa): first epidemiological findings. Transbound Emerg Dis 65:518–526. https://doi.org/10.1111/tbed.12734

Manor R, Saltz D (2004) The impact of free-roaming dogs on gazelle kid/female ratio in a fragmented area. Biol Conserv 119:231–236

Marjakangas E-L, Genes L, Pires MM, Fernandez FAS, de Lima RAF, de Oliveira AA, Ovaskainen O, Pires AS, Prado PI, Galetti M (2018) Estimating interaction credit for trophic rewilding in tropical forests. Philos Trans R Soc B Biol Sci 373:20170435. https://doi.org/10.1098/rstb.2017.0435

Marques AT, Santos CD, Hanssen F, Muñoz A-R, Onrubia A, Wikelski M et al (2020) Wind turbines cause functional habitat loss for migratory soaring birds. J Anim Ecol 89(1):93–103. https://doi.org/10.1111/1365-2656.12961

Martinez E, Cesario C, Boere V (2013) Domestic dogs in rural area of fragmented Atlantic Forest: potential threats to wild animals. Cienc Rural 43:1998–2003

Mazerolle MJ, Villard M-A (1999) Patch characteristics and landscape context as predictors of species presence and abundance: A review. Ecoscience 6:117–124

Megid J, Ribeiro MG, Paes AC (2015) Doenças Infecciosas Em Animais de Produção e de Companhia, 1st edn. Roca, São Paulo

Meyer ALS, Pie MR, Passos FC (2014) Assessing the exposure of lion tamarins (Leontopithecus spp.) to future climate change. Am J Primatol 76:551–562. https://doi.org/10.1002/ajp.22247

Monath TP, Vasconcelos PFC (2015) Yellow fever. J Clin Virol 64:160–173. https://doi.org/10.1016/j.jcv.2014.08.030

Muylaert RDL, Stevens RD, Esbérard CEL, Mello MAR, Garbino GST, Varzinczak LH et al (2017) ATLANTIC BATS: a data set of bat communities from the Atlantic Forests of South America. Ecology 98:3227. https://doi.org/10.1002/ecy.2007

Nava AFD, Cullen L Jr, Sana DA, Nardi MS, Ramos Filho JD, Lima TF, Abreu KC, Ferreira F (2008) First evidence of canine distemper in Brazilian free-ranging felids. EcoHealth 5:513–518. https://doi.org/10.1007/s10393-008-0207-8

Nagy-Reis MB, Oshima JE, Kanda CZ, Palmeira FBL, Melo FR, Morato RG (2020) Neotropical Carnivores: a data set on carnivore distribution in the Neotropics. Ecology (in press). https://doi.org/doi:10.1002/ecy.3128

Negrão FJ, Wosiacki SH, Alfieri AA, Alfieri AF (2006) Perfil de restrição de um fragmento do gene da hemaglutinina amplificado pela RT-PCR a partir de estirpes vacinais e selvagens do virus da cinomose canina. Arquivo Brasileiro de Medicina Veterinaria e Zootecnia 58:1099–1106

Nichols E, Spector S, Louzada J, Larsen T, Amezquita S, Favila ME (2008) Ecological functions and ecosystem services provided by Scarabaeinae dung beetles. Biol Conserv 141:1461–1474. https://doi.org/10.1016/j.biocon.2008.04.011

Nielsen SE, Stenhouse GB, Boyce MS (2006) A habitat-based framework for grizzly bear conservation in Alberta. Biol Conserv 130:217–229

Nobre RA (2007) Modelos de sustentabilidade de caça de subsistência na Serra do Mar, Mata Atlântica (Master thesis, Universidade de São Paulo)

Parris KM, Lord MV, North JMA (2009) Frogs call at a higher pitch in traffic noise. Ecol Soc 14:25

Paschoal AMO, Massara RL, Bailey LL, Doherty PF Jr, Santos PM, Paglia AP, Hirsch A, Chiarello AG (2018) Anthropogenic disturbances drive domestic dog use of Atlantic Forest protected areas. Trop Conserv Sci 11:1940082918789833. https://doi.org/10.1177/1940082918789833

Paviolo A, De Angelo C, Ferraz KMPMB, Morato RG, Martinez Pardo J, Srbek-Araujo AC et al (2016) A biodiversity hotspot losing its top predator: the challenge of jaguar conservation in the Atlantic Forest of South America. Sci Rep 6:37147

Pecl GT, Araújo MB, Bell JD, Blanchard J, Bonebrake TC, Chen I-C, Clark TD et al (2017) Biodiversity redistribution under climate change: impacts on ecosystems and human well-being. Science 355. https://doi.org/10.1126/science.aai9214

Pedrosa F, Salerno R, Padilha FVB, Galetti M (2015) Current distribution of invasive feral pigs in Brazil: economic impacts and ecological uncertainty. Nat Conservação 13:84–87. https://doi.org/10.1016/j.ncon.2015.04.005

Peres CA (2000) Effects of subsistence hunting on vertebrate community structure in Amazonian forests. Conserv Biol 14:240–253. https://doi.org/10.1046/j.1523-1739.2000.98485.x

Pires MM, Guimarães PR, Galetti M, Jordano P (2018) Pleistocene megafaunal extinctions and the functional loss of long-distance seed-dispersal services. Ecography 41:153–163. https://doi.org/10.1111/ecog.03163

Preece ND, Abell SE, Grogan L, Wayne A, Skerratt LF, van Oosterzee P, Shima AL, Daszak P, Field H, Reiss A, Berger L, Rymer TL, Fisher DO, Lawes MJ, Laurance SG, McCallum H, Esson C, Epstein JH (2017) A guide for ecologists: detecting the role of disease in faunal declines and managing population recovery. Biol Conserv 214:136–146. https://doi.org/10.1016/j.biocon.2017.08.014

Raghunathan N, François L, Huynen M-C, Oliveira LC, Hambuckers A (2015) Modelling the distribution of key tree species used by lion tamarins in the Brazilian Atlantic forest under a scenario of future climate change. Reg Environ Chang 15:683–693. https://doi.org/10.1007/s10113-014-0625-9

Read J, Bowen Z (2001) Population dynamics, diet and aspects of the biology of feral cats and foxes in arid South Australia. Wildl Res 28:195–203

Reiter P (2001) Climate change and mosquito-borne disease. Environ Health Perspect 109:141–161. https://doi.org/10.1289/ehp.01109s1141

Renctas (2007) Vida silveste: o estreito limiar entre preservação e destruição. Diagnóstico do Tráfico de Animais Silvestres na Mata Atlântica. Accesed 02/Sep/2020. http://www.renctas.org.br/wp-content/uploads/2014/06/livro-vida-silvestre.pdf

Ribeiro MC, Metzger JP, Martensen AC, Ponzoni FJ, Hirota MM (2009) The Brazilian Atlantic Forest: how much is left, and how is the remaining forest distributed? Implications for conservation. Biol Conserv 142:1141–1153. https://doi.org/10.1016/j.biocon.2009.02.021

Ribeiro BR, Sales LP, De Marco P Jr, Loyola R (2016) Assessing mammal exposure to climate change in the Brazilian Amazon. PLoS One 11:1–13. https://doi.org/10.1371/journal.pone.0165073

Ripple WJ, Estes JA, Beschta RL, Wilmers CC, Ritchie EG, Hebblewhite M, Berger J, Elmhagen B, Letnic M, Nelson MP, Schmitz OJ, Smith DW, Wallach AD, Wirsing AJ (2014) Status and ecological effects of the world's largest carnivores. Science 343. https://doi.org/10.1126/science.1241484

Rodrigues RC, Hasui É, Assis JC, Pena JCC, Muylaert RL, Tonetti VR et al (2019) ATLANTIC BIRD TRAITS: a data set of bird morphological traits from the Atlantic forests of South America. Ecology 100:e02647. https://doi.org/10.1002/ecy.2647

Romano APM, Costa ZGA, Ramos DG, Andrade MA, de Sa Jayme V, de Almeida MAB, Vettorello KC, Mascheretti M, Flannery B (2014) Yellow fever outbreaks in unvaccinated populations, Brazil, 2008–2009. PLoS Negl Trop Dis 8:1–8. https://doi.org/10.1371/journal.pntd.0002740

Rosa CA, de Almeida Curi NH, Puertas F, Passamani M (2017) Alien terrestrial mammals in Brazil: current status and management. Biol Invasions 19:2101–2123. https://doi.org/10.1007/s10530-017-1423-3

Rosa CA, Passamani M, Pompeu P (2019) Differential effects of exotic Eurasian wild pigs and native peccaries on physical integrity of streams in the Brazilian Atlantic Forest. Manag Biol Invasions 10. In press

Rosa CA, Ribeiro BR, Bejarano V, Puertas FH, Bocchiglieri A, Santos ALB et al. (2020) Neotropical alien mammals: a data set of occurrence and abundance of alien mammals in the Neotropics. Ecology (in press). https://doi.org/10.1002/ecy.3115

Ruiz-Fons FA (2017) Review of the current status of relevant zoonotic pathogens in wild swine (Sus scrofa) populations: changes modulating the risk of transmission to humans. Transbound Emerg Dis 64(1):68–88

Rytwinski T, Fahrig L (2013) Why are some animal populations unaffected or positively affected by roads? Oecologia 173:1143–1156. https://doi.org/10.1007/s00442-013-2684-x

Sick H (1997) Ornitologia brasileira. Editora Nova Fronteira, Rio de Janeiro

Sloan S, Jenkins CN, Joppa LN, Gaveau DLA, Laurance WF (2014) Remaining natural vegetation in the global biodiversity hotspots. Biol Conserv 177:12–24. https://doi.org/10.1016/j.biocon.2014.05.027

Sousa JAC, Srbek-Araujo AC (2017) Are we headed towards the defaunation of the last large Atlantic Forest remnants? Poaching activities in one of the largest remnants of the Tabuleiro forests in southeastern Brazil. Environ Monit Assess 189:129. https://doi.org/10.1007/s10661-017-5854-1

Souza Y, Gonçalves F, Lautenschlager L, Akkawi P, Mendes C, Carvalho MM et al (2019) ATLANTIC MAMMALS: a dataset of assemblages of medium and large-sized mammals of the Atlantic Forest of South America. Ecology (In press)

Srbek-Araujo AC, Chiarello AG (2008) Domestic dogs in Atlantic forest preserves of South-Eastern Brazil: a camera-trapping study on patterns of entrance and site occupancy rates. Braz J Biol 68:771–779

Steen DA, Gibbs JP (2004) Effects of roads on the structure of freshwater turtle populations. Conserv Biol 18:1143–1148

Terborgh JW, Emmons L, Fitzpatrick JW (1984) Annotated checklist of bird and mammal species of Cocha Cashu Biological Station, Manu National Park, Peru. Field Museum of Natural History, Chicago

Torres PC, Prado PI (2010) Domestic dogs in a fragmented landscape in the Brazilian Atlantic Forest: abundance, habitat use and caring by owners. Braz J Biol 70:987–994

Trombulak SC, Frissell CA (2000) Review of ecological effects of roads on terrestrial and aquatic communities. Conserv Biol 14:18–30. https://doi.org/10.1046/j.1523-1739.2000.99084.x

Vale P, Gibbs H, Vale R, Christie M, Florence E, Munger J, Sabaini D (2019) The expansion of intensive beef farming to the Brazilian amazon. Global Environ Chang 57:101922. https://doi.org/https://doi.org/10.1016/j.gloenvcha.2019.05.006

Vale MM, Arias PA, Ortega G et al (2021) Climate change and biodiversity in the Atlantic Forest: best climatic models, predicted changes and impacts, and adaptation options. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Vancine MH, Duarte K, da Souza YS, Giovanelli JGR, Martins-Sobrinho PM, López A et al (2018) ATLANTIC AMPHIBIANS: a data set of amphibian communities from the Atlantic forests of South America. Ecology 99:1692. https://doi.org/10.1002/ecy.2392

Villar N, Siqueira T, Zipparro V, Farah F, Schmaedecke G, Hortenci L, Brocardo CR, Jordano P, Galetti M (2020) The cryptic regulation of diversity by functionally complementary large tropical forest herbivores. J Ecol 108:279–290. https://doi.org/10.1111/1365-2745.13257

Whiteman CW, Matushima ER, Confalonieri UEC, Palha M, Silva A, Monteiro VC (2007) Human and domestic animal populations as a potential threat to wild carnivore conservation in a fragmented landscape from the Eastern Brazilian Amazon. Biol Conserv 138:290–296. https://doi.org/10.1016/j.biocon.2007.04.013

Wilson EO (1987) The little things that run the world* (the importance and conservation of invertebrates). Conserv Biol 1:344–346. https://doi.org/10.1111/j.1523-1739.1987.tb00055.x

Young HS, McCauley DJ, Galetti M, Dirzo R (2016) Patterns, causes, and consequences of anthropocene defaunation. Annu Rev Ecol Evol Syst 47:333–358. https://doi.org/10.1146/annurev-ecolsys-112414-054142

# Chapter 15
# Pollination Systems in the Atlantic Forest: Characterisation, Threats, and Opportunities

**Isabela Galarda Varassin, Kayna Agostini, Marina Wolowski, and Leandro Freitas**

**Abstract** In the Brazilian Atlantic forest, pollination systems encompass several animal groups and vary from very specialised to highly generalised ones. Plant-pollinator interactions are at the origin and maintenance of diversity and affect ecosystems' functioning. Moreover, pollination deficit may impact agricultural systems and the dynamics in natural systems with varying importance according to interaction specialisation. We present here examples of pollination studies in the Atlantic forest, highlighting current stressors of plant-pollinator interactions and opportunities to mitigate them. Habitat loss, climate change, and invasive species are the major threats to pollination interactions. Despite the risk, the opportunities for change (restoration, ecological corridors, and protected areas, and landscape management) can contribute to the maintenance of pollination services in the Atlantic forest.

**Keywords** Biodiversity conservation · Climate change · Ecological restoration · Ecosystem services · Landscape management · Pollination deficit

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## 15.1 Introduction

Pollination is a fascinating process that involves pollen transport either by abiotic (wind and water) or biotic vectors (invertebrate and vertebrate animals), in which the expected result is the formation of fruits and seeds (Agostini et al. 2014). Most pollination events in native or exotic cultivated plants are due to plant-animal interactions (Ollerton et al. 2011; Roubik 2018). Therefore, the interactions between plants and pollinators are involved in the origin of species diversity and the coexistence of species, i.e. the maintenance of diversity, as well as functioning ecosystems (Andresen et al. 2018).

The dependence of plants on pollinators appears to be higher in tropical than in temperate environments. The proportion of animal-pollinated species rises from a mean of 78% in temperate-zone communities to 94% in tropical communities (Ollerton et al. 2011). Most pollinators are insects, such as bees, flies, butterflies, moths, wasps, beetles, and thrips, but there are also vertebrate pollinators, such as birds, bats, non-flying mammals, and lizards (Rech et al. 2014; BPBES/REBIPP 2019). Most plant species are pollinated by bees in the Atlantic forest (Maués et al. 2012), and bees visit ca. 79% of the main cultivated crops in Brazil (BPBES/REBIPP 2019).

Like other hyperdiverse tropical ecosystems (Barlow et al. 2018), the Atlantic forest has a long history of human pressure exemplified by the conversion of natural areas into agricultural and urban areas (Jeske-Pieruschka et al. 2010; Oliveira and Engemann 2011; Nehren et al. 2013), which requires actions to prevent the collapse of their biodiversity and functioning. The original coverage of the Atlantic forest was reduced by ca. 72% (Rezende et al. 2018), causing extinctions and habitat reduction for the remaining species. Effects of anthropogenic changes on species loss are also expressed in ecological interactions (Morris 2010), including the relationships between plants and pollinators (Potts et al. 2010). For the Atlantic forest, the consequences of losing species – threatened and/or rare and/or endemic plants and pollinators – and the interactions they establish may have cascading effects on the functioning of this ecosystem, with a significant impact on the services it provides (i.e. climate regulation, erosion control, pollination). We present here some examples of pollination studies in the Brazilian Atlantic forest, highlighting major threats to plant-pollinator interactions and the opportunities for the maintenance of pollination services in the Atlantic forest.

## 15.2 Pollination Systems in the Atlantic Forest

The knowledge about pollination in Brazilian biomes began to be systematised, according to Maués et al. (2012), only in the 1970s (e.g. Sazima 1972; Sazima and Sazima 1975; Gottsberger 1977). Currently, there is no accurate estimate of the proportion of Atlantic forest species that have been studied regarding the aspects of pollination biology. However, based on a previous meta-analysis (Wolowski et al. 2014) and unpublished surveys for some of the most studied families, e.g. Bromeliaceae and Gesneriaceae (Freitas and Wolowski, unpubl. data), we may roughly estimate that no more than 15% to 20% of the species in the Atlantic forest has some systematic study on pollination.

Studies about pollination systems in the Atlantic forest encompass mostly study cases of single species (Sazima et al. 2003; Agostini et al. 2006) but also works based on pollinator guilds or related plants and at the community level (Agostini et al. 2011; Pinheiro et al. 2008; Brito et al. 2016). Some examples of studies based on guilds are those with stingless bees (Meliponini – Ramalho 2004), hawkmoths (Sphingidae – Avila Jr. 2009; Amorim et al. 2014), hummingbirds (Sazima et al. 1996; Buzato et al. 2000), and bats (Sazima et al. 1999). Taxonomic bias is marked with a large number of studies on reproductive biology of Bromeliaceae (e.g. Wendt et al. 2008; Canela and Sazima 2003, 2005; Cavalcante et al. 2019; Zambon et al. 2019a, b), Melastomataceae (Goldenberg and Varassin 2001; Santos et al. 2012; Brito et al. 2016; Maia et al. 2016), Orchidaceae (e.g. Singer and Sazima 2001; Pansarin and Amaral 2009), and heterostylous species of Rubiaceae (e.g. Castro et al. 2004; Klein et al. 2009). Regarding the habitat, the studies are concentrated in herbs, especially epiphytes, mainly due to studies in Bromeliaceae and Gesneriaceae (e.g. SanMartin-Gajardo and Sazima 2005; Matallana et al. 2010). Data on the pollination biology of tree species are still scarce (e.g. Pires and Freitas 2008; Rocca and Sazima 2008; Danieli-Silva and Varassin 2013). Pollination by vertebrates may represent approximately 15% to 20% of species in tropical forests (Feinsinger 1983) and has attracted more attention from researchers in the Atlantic forest (Sazima et al. 1999; Buzato et al. 2000; Agostini et al. 2006, 2011). Most of these studies were carried out with plants easily accessible from the ground, which can hardly represent ecologically dominant groups of plants.

In a review of published studies from 1981 to 2020 (February) using the database "Web of Science® Core Collection", we have found 524 studies. We used the combination of keywords "pollinat*" and "Atlantic". Based on this database, we selected the studies that (1) report interactions; (2) were done within the Brazilian Atlantic forest domain (i.e. studies conducted within semi-deciduous and rain forests). Our final database was composed of 186 studies, published from 1992 to 2020 (see Appendix 15.1). For those studies, we searched for the following information: groups of pollinators, plant habitat, plant families, kind of Atlantic forest vegetation, the region of the Atlantic forest, and Brazilian political state. We used the abstracts of these studies to build two word clouds, one from 1992 to 2011, before the IPBES initiative, and other from 2012 to 2020 (Fig. 15.1). Among 186 studies, we found three theoretical studies that were used to build the word cloud, but not to account for the distribution of studies on the Atlantic forest. Twelve studies were large surveys, sampling either many plant families, including different habitats or sampling many groups of insects. For those studies, we usually only recorded where the study was done. Five studies reported the impact of interaction on reproductive success, but with no direct record of pollinators. For those, all information but pollinators were reported.

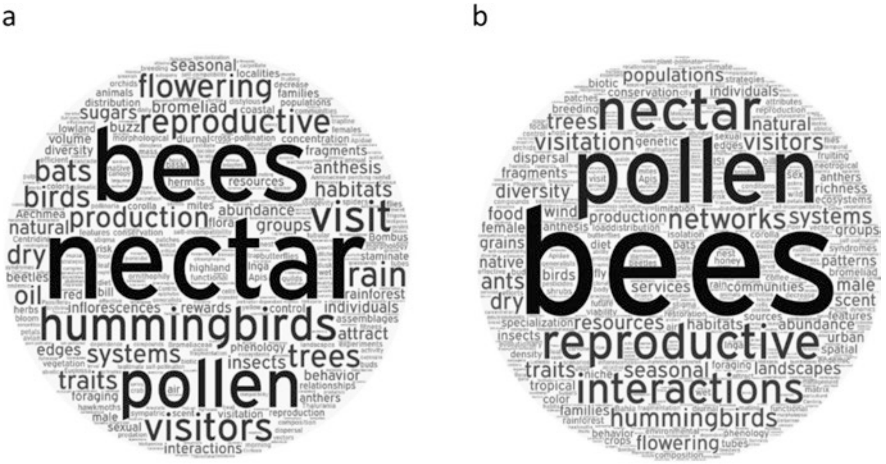

**Fig. 15.1** Temporal changes in the studies done at the Atlantic forest between the early years, 1992–2011, before the IPBES initiative (a) and more recent years, 2012–2020 (b). A comparison word cloud compares the relative frequency that a word was used in the two time periods. The size of the word is proportional to the difference in the relative frequency by which the word was used between the two time periods compared

Our survey from the literature indicates temporal changes in the scope of studies done during a 28-year period (Fig. 15.1). Terms associated with studies done with particular taxonomic groups (bees, hummingbirds, bats, birds, bromeliads), floral biology (nectar, oil, sugar, pollen) occur in both time periods, with higher importance in early year studies, 1992 to 2011 (Fig. 15.1a). In recent year studies, 2012 to 2020, new words such as crops, food, landscape, networks, services, and urban points to a shift towards emerging research topics such as ecosystem services provided by pollination and the effect of anthropogenic changes on plant-pollinator interactions (Fig. 15.1b).

Most studies reported interactions with bees (94), followed by hummingbirds (57), butterflies (24), beetles or flies (13), bats or wasps (11), birds – excluding hummingbirds (7), moths (6), hoverflies (5) hawkmoths or wind (4), ants (3), and primates or bugs/aphids (1). Herbs were the most sampled habit (69), followed by trees (43), shrubs (33), and vines (12). Twenty-five studies had extensive sampling and did not report plant habit. 51 plant families were studied at the Atlantic forest, 27 studies were done with many different plant families and are not included in the 51 families above. Most studies were done in the rain forest (113), followed by semi-deciduous forests (27), restinga coastal vegetation (16), mixed araucaria forests (8), and grasslands (1). Five studies did not report where they were done, and five were done in many different vegetations. Most studies were done on the southeastern regions of the Atlantic forest (106), followed by the northeastern (34) and southern (23) areas. Five studies involved all the regions. Most studies were done in São Paulo state (50), followed by Rio de Janeiro (33), Minas Gerais (17), Pernambuco (16), Bahia and Paraná (12), Santa Catarina (7), Espírito Santo (6), Rio Grande do Sul (4), Alagoas (3), Mato Grosso, Mato Grosso do Sul, Paraíba or Sergipe (1) states.

## 15.3 Generalist and Specialist Pollination Systems in Atlantic Forest

Plant-pollinator interactions vary from obligate specialists (a single animal species pollinates one plant species) to facultative generalists in which flowers are pollinated by a taxonomically broad range of pollinators that also visit the flowers of many other species (Ollerton 2017). Obligate specialisation and extreme generalisation represent two ends of a continuum in resource use (Waser and Ollerton 2006). More generalised systems constitute the majority of flowering plant interactions (Waser et al. 1996, though see Johnson and Steiner 2000). Still, more specialised pollination systems have received more attention, perhaps due to the persuasive influence of the pollination syndrome concept and the examples of intricate coevolution (see Faegri and van der Pijl 1979; Proctor et al. 1996; Waser et al. 1996).

### *15.3.1 Specialised Pollination Systems in Atlantic Forest*

Different specialised pollination systems were studied in the Atlantic forest. Here, we highlight the systems that involve just one functional group of invertebrate or vertebrate pollinators.

A specialised case of pollination by bees called buzz-pollination was extensively studied in some species of Melastomataceae and Solanaceae in the Atlantic forest. In these buzz-pollinated flowers, bees use vibrations (sonications) to extract pollen from the pore of anthers (Bezerra and Machado 2003; Brito and Sazima 2012; Falcão et al. 2016; Malucelli et al. 2018), as for C*haetogastra cerastifolia* (Fig. 15.2a–c). An example is *Tibouchina pulchra,* a common tree species from the Atlantic forest, buzz-pollinated by five bee species *Bombus morio*, *Xylocopa brasilianorum*, *Xylocopa frontalis*, *Epicharis flava*, and *Eufriesea* sp. (Brito and Sazima 2012).

Another remarkable example of invertebrate specialist pollination systems is the orchid bees. *Catasetum cernuum* and *Gongora bufonia* are epiphytic orchids native to the Atlantic forest of southeastern South America, exclusively pollinated by male *Eufriesea violacea* orchid-bees (Nunes et al. 2017). These two orchid species share 12 volatile organic compounds to attract the same euglossine pollinator. The exclusive pollination by a single orchid-bee species together with the similarity of the composition of floral scents of *C. cernuum* and *G. bufonia*, two species from distinct megadiverse lineages of Cymbidioid orchids, strongly suggest that these species have converged to exploit the exclusive niche of pollination by male *E. violacea* (Nunes et al. 2017).

Within vertebrate pollination systems, specialisation is also associated with an intriguing floral morphology. Examples from the genus *Mucuna* (Fabaceae) are *M. japira* and *M. urens* with an explosive pollen release that can be triggered only by pollinators with a large body mass. *Mucuna japira* is pollinated by the

**Fig. 15.2** Specialised and generalised pollination systems in Melastomataceae: a pollen-flower of *Chaetogastra cerastifolia* buzz-pollinated by the bees *Melipona bicolor* (a), *M. marginata* (b) and *Bombus pauloensis* (c), and a nectar-flower of *Miconia hyemalis* pollinated by a fly (Bibionidae, d), a wasp (Vespidae, e) and a hummingbird (*Thalurania glaucopis*, Trochilidae, f) (Photos by Isabela Varassin)

insectivorous bird *Cacicus haemorrhous,* whereas *M. urens* is pollinated by the glossophagine bat *Glossophaga soricina* (Agostini et al. 2006, 2011).

### *15.3.2 Generalised Pollination Systems in Atlantic Forest*

Generalised pollination systems may be highlighted in plants pollinated by two or more functional groups of pollinators. This system occurs in *Miconia hyemalis* (Fig. 15.2d–f), where changes from a classic Melastomataceae buzz-pollinated flower (Fig. 15.2a–c) to a nectar-producing flower with large-pored anthers are associated with generalisation in the pollination system (Brito et al. 2016) that incorporates flies (Fig. 15.2d), wasps (Fig. 15.2e), and hummingbirds (Fig. 15.2f). Other examples are some species of tribe Malveae (Malvaceae) in the Atlantic forest that have floral features overlapping hummingbird and bat pollination systems. *Abutilon ramiflorum*, *Callianthe regnellii,* and *A.* aff. *ramiflorum* present bell to bowl-shaped flowers lasting about 30 h, coloured pinkish to purplish or yellowish, crepuscular anthesis, and very faint cabbage odour. The flowers are visited and pollinated by a species of phyllostomid bat at night, and by several species of hummingbirds during the day (Buzato et al. 1994). Thus, these three species of Malveae share two functional groups of vertebrate pollinators.

Another example of a generalist pollination system is *Inga sessilis* (Fabaceae) that is self-incompatible and pollinated by hummingbirds, hawkmoths, and bats. Diurnal pollinators contribute less than nocturnal ones to fruit production, but the former are the more constant and reliable visitors. These results indicate *I. sessilis* has floral adaptations that encompass both diurnal and nocturnal pollinator requirements, suggesting a complementary and mixed pollination system (Amorim et al. 2013). Three functional groups of invertebrate and vertebrate are the pollinators of *Inga sessilis*.

The last example of a generalist pollination system is *Paypayrola blanchetiana* (Violaceae). Flowers of this species are visited during the night and the day. All visitors forage for nectar. Nocturnal visitors are perching moths (Noctuidae, Geometridae), hawkmoths (Sphingidae), and the crepuscular bee *Megalopta* sp. (Halicticidae). During the day, *P. blanchetiana* is visited by various butterflies (Lycaenidae, Riodinidae, Hesperiidae, Ithomiinae, *Heliconius* spp.), bees (*Euglossa* spp., *Apis mellifera*) and a hummingbird, *Phaetornis ruber* (Trochilidae) (Braun et al. 2012). These Violaceae species are pollinated by five functional groups of invertebrate and vertebrate pollinators.

## 15.4 The Threats to Pollination Interactions: Habitat Loss, Climate Change, and Invasive Species

Landscapes with sharp contrasts predominate in most of the Atlantic forest, where the most significant remnants are composed of fully protected areas, alternated with extensive agricultural or urban areas punctuated with small fragments of vegetation, often in earlier successional stages (Fonseca 1985; Ribeiro and Freitas 2010). Fragmented landscapes have contrasting effects on plant-pollinator interactions in the Atlantic forest. For example, adverse effects on seed production due to a reduction in the visit frequency by pollinators and/or in the number of partners (abundance of plants) were observed in smaller fragments, forest borders or less connected landscapes (Pires et al. 2014; Franceschinelli et al. 2015). However, the effects of fragmentation on the diversity of pollinators in the Atlantic forest can also be neutral or positive, particularly in plants of open-canopy environments, which benefit from increased edges (e.g. Ramos and Santos 2006; Lopes and Buzato 2007; Dunley et al. 2009; Nery et al. 2018, see also Tonhasca et al. 2002). Those results in the Atlantic forest are from case studies with one or a few species, and there is not much data for whole communities, where analysis is noticeably complicated due to the high biodiversity and physiognomic complexity of tropical forests (Freitas et al. 2014).

The Atlantic forest is nowadays composed of a myriad of patches of different sizes and ages distributed on distinct landscapes (Rezende et al. 2018). As a result, the dynamic species turnover is driving many forest fragments towards an early-successional system (Tabarelli et al. 2010) with particular sets of plant reproductive traits (Lopes et al. 2009). For instance, half of the pollination systems differed among disturbed areas (edge) compared to conserved ones, with a decrease in vertebrate-pollinated systems in disturbed areas (Lopes et al. 2009). Plant

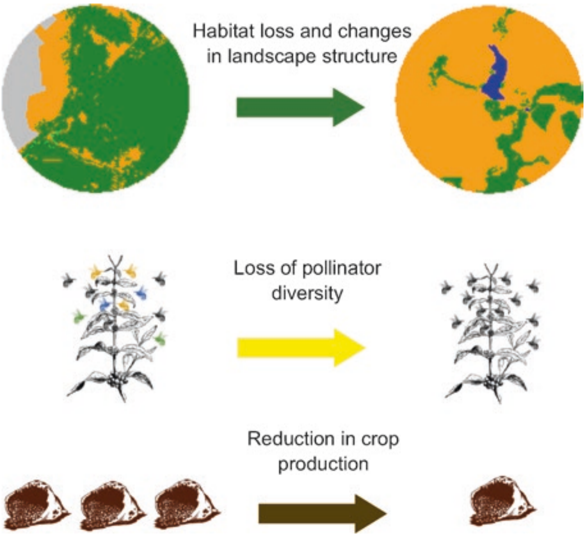

**Fig. 15.3** Habitat loss and changes in landscape structure and its impact on the diversity of pollinators and crop production (Drawings by Israel Schneiberg and Fernando Jeronimo)

reproductive traits in areas restored through natural regeneration changes across time; for instance, early successional systems have a decrease in bee-pollinated, bell-shaped, small and androgynous flowers (Warring et al. 2016). Taken together, these results indicate that forests under disturbance and old successional ones are composed of different sets of plants with consequences to pollinator maintenance and interactions. Moreover, plant reproductive traits that drive community assembly processes in Atlantic forest patches indicate there is an increased role of biotic interactions and limiting similarity across the successional process (Warring et al. 2016). This enhanced role suggests that disturbance may shift the processes driving community assembly to stronger environmental filtering.

Besides that, habitat loss affects pollination interactions through the reduction of pollinator populations (Ramalho et al. 2009; Ferreira et al. 2015). In the fragmented landscapes of the Atlantic forest, long-distance pollen flows within and among populations may increase connectivity among individuals. Still, low plant density in the populations may restrict outcrossing (Gaiotto et al. 2003; Santos et al. 2018). Besides that, for crop plants, habitat loss results in changes in landscape structure. It compromises pollination success (Fig. 15.3), which is the ecosystem services that overflow from natural areas (de Marco and Coelho 2004; Saturni et al. 2016).

There are no historical records on the effects of recent climate change on pollination interactions. However, geographic shifts of bees are expected in the future scenarios at the Atlantic forest (Giannini et al. 2012; Martins et al. 2015). For 62 hummingbird-pollinated plant species, a tendency is expected towards spatial discontinuity, reduction, and spatial displacement due to climate change (Correa-Lima et al. 2019). Besides that, their current flowering patterns are predicted to change, with gains or losses in the length of the flowering season and the possible emergence of resource gaps throughout the year (Correa-Lima et al. 2019). These gaps

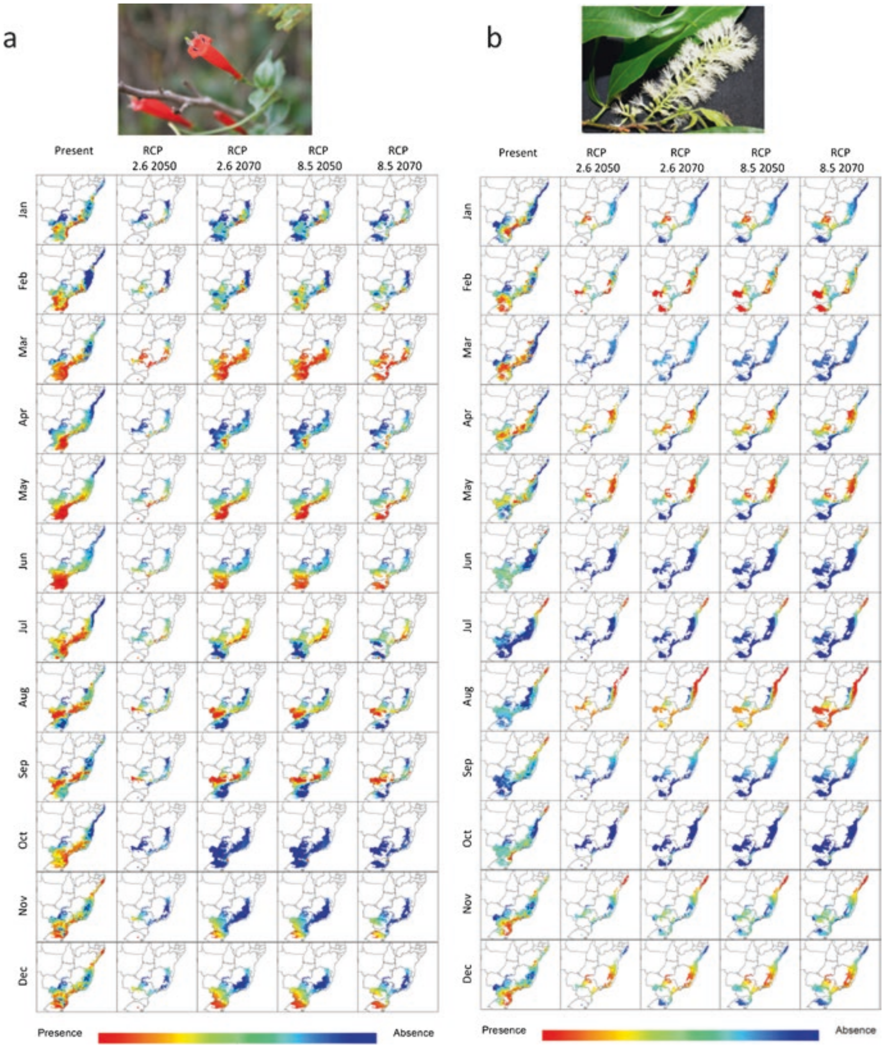

**Fig. 15.4** Predictions of climate suitability for the flowering of *Manettia cordifolia* (a) a species pollinated by only one species of hummingbird, and *Inga marginata* (b), a species pollinated by hummingbirds and insects. Adequacy values of each model represent the probability of presence (in red) and absence (in blue) – (photos by Eduardo Giehl, Maps by Ana Paula Araujo Correa-Lima)

might affect a few pollinator species, as depicted for *Manettia cordifolia* (Fig. 15.4a), which is pollinated by only one species of hummingbird (Pedro Bergamo *pers. comm.*). But these gaps may affect a more significant number of species, e.g. the pollinators of *Inga marginata* (Fig. 15.4b) which is pollinated by many species of hummingbirds and insects (Pedro Bergamo *pers. comm*). Climate changes may impose a threat even for cultivated plants in the Atlantic forest where a potential spatial mismatch between crop plants and their pollinators is predicted (Bezerra et al. 2019). Negative shifts on the probability of pollinator presence are expected to reduce crop production in many municipalities in Brazil, including those in the Atlantic forest (Giannini et al. 2017). Although climate-driven changes are expected, there is strong plasticity in plant-pollinator interactions (Burkle and Alarcón 2011; CaraDonna et al. 2017) due to species turnover or interaction rewiring (Burkle et al. 2016). However, how successful this plasticity will be for the reshaping of plant-pollinator interactions and rescue interactions at the Atlantic forest is unknown.

At the Atlantic forest, the primary threat of an invasive pollinator is *Apis mellifera,* a widespread invasive bee (Schneider et al. 2004). Because *A. mellifera* is a supergeneralist species, floral resources that they use overlaps with most other native pollinators (Giannini et al. 2015b), creating the potential for competition (Carneiro and Martins 2012). Besides that, *A. mellifera* may glue together otherwise disconnected species subsets and increase network asymmetry (Giannini et al. 2015b), thus changing the ecological and evolutionary dynamics within the invaded network. Another bee, *Bombus terrestris*, already spread in Chile and Argentina, is turning Southern Brazil into a highly susceptible invasion area (Acosta et al. 2016), with the same potential effects of *A. mellifera*. Invasive plants at the Atlantic forest share the capacity for generality and interact with more species than native ones (Maruyama et al. 2016). At least for hummingbird-plant interactions, for some hummingbirds, these plants may be a more exclusive floral resource (Maruyama et al. 2016), suggesting the potential displacement of pollinators from native plant resources.

## 15.5 Impacts of Pollinator Decline on Plant Reproduction in Natural and Agricultural Environments

### *15.5.1 Pollination Deficit in Natural and Agricultural Environments*

Pollinators are vital organisms for ecosystem functions and services as they are fundamental to plant reproduction in natural communities (Ashman et al. 2004; Ollerton et al. 2011) and crop productivity (Klein et al. 2007). However, evidence of the pollinator decline has accumulated over the last decades (Potts et al. 2010). This decline can be seen both in natural and agricultural areas as a consequence of several threats to pollinators: mainly loss of habitat and pesticides, but also climate change, pathogens, and invasive species (see Sect. 15.4). Pollinators decline in natural and agricultural areas, and the mortality of bees in crops reinforces this evidence in Brazil. Especially in the Atlantic forest, the historical conversion of natural areas into crops and pastures has led to the reduction and fragmentation of the forest remnants, which now constitute just 28% of the original native vegetation cover (Rezende et al. 2018). Pesticides have a direct impact on pollinator life cycle, causing mortality, reduction of offspring, disorientation, and repellent effect (Freitas and Pinheiro 2010; Pinheiro and Freitas 2010).

Pollinator decline leads to problems in pollination. The pollination deficit of native or agricultural plants brings the inadequate or insufficient transfer of pollen, which limits the quantity and/or quality of fruit production and its economic yield (Vaissière et al. 2011). First, inadequate pollen transfer may be caused by the lack of compatibility between floral morphology and pollinator size. Pollen production and the attractiveness of flowers to pollinators are affected by environmental factors, e.g. soil, temperature, humidity, light, and radiation. Finally, factors linked to the landscape that surround natural communities or plantations may not support pollinator populations, which will also contribute to pollinator decline.

### *15.5.2 Implications of Pollination Deficit for the Maintenance of Natural Plant Populations and Crop Productivity*

Pollination deficit impacts the preservation of natural plant populations with effects on the plant life cycle, community dynamics, and overall ecosystem processes. The majority of the plant species are pollinated by animals (e.g. 94% in tropical communities, Ollerton et al. 2011), while many species are pollen-limited (62%, Ashman et al. 2004). The concept of pollen limitation, which is the reduction of plant reproductive success due to inadequate quantity or quality of pollen (seed quantity or quality) (Ashman et al. 2004), is intrinsically related to pollination deficit. Although pollen limitation may vary across space and time, it is associated with specific plant features as self-incompatibility and pollination specialisation (Knight et al. 2005). Specifically, in the Brazilian Atlantic forest, overall pollen limitation was moderate and was reported for 39% of plants (Wolowski et al. 2014). Within this biome, pollination specialisation (i.e. functional and ecological specialisation sensu (Ollerton et al. 2007) predicted pollen limitation, suggesting that specialist plants are more likely to fail reproductively in response to pollinator decline (Wolowski et al. 2014). However, plant mating system did not explain pollen limitation in the Atlantic forest, i.e. self-compatible and self-incompatibility species did not show different levels of pollen limitation (Wolowski et al. 2014). Indeed, this may be because the majority of plants (84%) are pollinator-dependent (i.e. non-autogamous species), thus plant reproduction in the Atlantic forest is a concern considering the scenario of pollinator deficit.

Pollinator deficit also impacts crop productivity. The direct impact of pollination deficit on crops is the reduction in productivity even for non-pollinator dependent crops. Considering the dependence of each crop on pollinators and the value of crop production, we can infer the economic impact of the pollination deficit for agricultural productivity (see Sect. 15.5.1). For instance, in the Atlantic forest, there are important animal-pollinated crops that depend on pollination at different levels (BPBES/REBIPP 2019). The two largest crop commodities, soybean and coffee, rely on animals to increase their productivity by 25% (Giannini et al. 2015a). Considering the estimated economic value of pollination on both crops for the whole planted areas in Brazil, a pollination deficit is expected to cause an economic loss annually up to US$6.7 billion for soybean and US$1.4 billion for coffee per year (BPBES/REBIPP 2019). Other important crops in the Atlantic forest are apple, bean, cocoa beans, cashew nut, cotton, orange, and tomato, which rely on animals to increase their productivity by 5% to 95% depending on the crop (Giannini et al. 2015a). For those crops, a pollination deficit is expected to cause additional losses up to US$3.0 billion (apple: US$399 million, bean: US$124 million; orange: US$534 million; tomato US$70 million; data from BPBES/REBIPP 2019; cocoa bean: US$533 million, cashew nut: US$13 million, cotton US$827 million, data from Giannini et al. 2015a), considering the whole planted area in Brazil.

## 15.6 Opportunities: Restoration, Landscape Management, and Protected Areas Can Contribute to the Maintenance of Pollination Service

More specialised pollination systems are more prone to pollen limitation (Wolowski et al. 2014, see Sect. 15.5.1) and to the adverse effects of fragmentation at the Atlantic forest (Girão et al. 2007, see Sect. 15.4). Then, actions enhancing its landscape connectivity are crucial to the conservation of plant-pollinator interactions, specially to specialised pollination systems. Besides connectivity, increasing the total forest area is a critical factor for biodiversity recovery and global change mitigation in the Atlantic forest (Newmark et al. 2017), in particular for specific target groups of plants and pollinators (Giannini et al. 2012).

Initiatives for the restoration of the Atlantic forest date back to the nineteenth century, but just recently (from the 1980s) projects have gained in scale and incorporated principles of community and landscape ecology (Rodrigues et al. 2009). While the idea that frugivore attraction should be considered for selection of species is present in the programs since the 1980s, pollination began to be recognised in the last decade. Currently, the initiatives consider the pollination syndromes among the functional attributes for species selection (Martins and Antonini 2016). Another example is the enrichment of young restored forests with herbs, mainly epiphytes, to favour the establishment of hummingbirds and scarab beetles since few ornithophilous and cantharophilous species are arboreal (Domene 2018). There is a growing demand for restoration actions in the Atlantic forest due to the Brazilian Native Vegetation Protection Law, approved in 2012. Restoration of approximately 15 million hectares (http://www.pactomataatlantica.org.br/; about 8% of the original area of the biome) by farmers is expected in the next decades. Farmers may accomplish, in different contexts, restoration within their estates or compensate it outside the farms. Consortia of farmers for implementation of restoration areas in the same locality combine the advantages of reducing costs and conserving larger areas (Strassburg et al. 2019), to allow more complex networks of plant-pollinator interactions than in small fragments (Hagen et al. 2012).

However, restoration of smaller patches within farms supports the provision of local ecosystem services (e.g. soil retention, local climate, and air quality, tourism, and biological control), including pollination (Metzger and Brancalion 2013; Banks-Leite et al. 2014; Strassburg et al. 2019). Several studies have found that farms within more connected landscapes and/or with greater forest coverage in the Atlantic forest harbour a greater diversity of crop pollinators (Garibaldi et al. 2014, 2016). So small fragments patchily distributed in local properties can contribute to the conservation of pollinators and plants associated with them, working as ecological corridors or stepping stones, and thus maintaining species metapopulations in the landscape (Laurance 2004; Townsend and Levey 2005; Kormann et al. 2016). For example, hummingbirds are reported to adapt well to fragmentation (Stouffer and Bierregaard 1995). Still, they are known to use forest corridors rather than an agricultural matrix to move between forest patches (Hadley and Betts 2009). We do not have much data for hummingbirds and fragmentation in the Atlantic forest, but at least some species seem to be sensitive to higher landscape connectivity and the successional stage of vegetation (Marsden et al. 2001; Valle 2018). The increase of functional richness and dispersion in areas under natural regeneration (Warring et al. 2016) indicates that ecological restoration is urgent to conserve interactions that are absent in younger forests. Because young reforested areas are composed of early successional planted trees which flowers only in limited periods (Fragoso and Varanda 2011), the presence of naturally occurring herbs and shrubs may act as bridging plants in times of resource shortage (Ceccon and Varassin 2014). In this sense, pollination as a process is still dependent on naturally occurring species even in reforested areas, which adds to the discussion that active restoration may not result in a faster or more complete recovery than passive restoration as shown for many structural parameters (Crouzeilles et al. 2017; Jones et al. 2018).

Actions aimed at increasing the pollination of crops may generate more favourable environments for native pollinator species (Viana et al. 2012). In addition to forest restoration, those actions encompass agricultural practices with a lower impact on the environment, e.g. organic agriculture, managed agroecosystems, and ecological intensification (Holzschuh et al. 2008, 2010; Nicholls and Altieri 2013; Garibaldi et al. 2016). These three classes of practices cover different contexts of planting area, crop types, and market positioning of farms. They, therefore, are not mutually exclusive as national strategies on the biome scale. Concerning pollinators, more friendly agricultural management can alleviate the harmful effects of the excessive use of pesticides on bees and other non-target insects (Nicholls and Altieri 2013; Cham et al. 2018; Mengoni Goñalons and Farina 2018). For example, bee pollination service is affected by landscape structure at different spatial scales in coffee plantations at Atlantic forest, but native bee diversity was positively impacted by forest cover (Saturni et al. 2016). Another relevant action is the best use of native bees rather than exotic species, notably *Apis mellifera,* for pollination of crops. We do not know how many bee species are in the Atlantic forest, but a latitudinal sampling on forested areas registered 105 species from 50 genera (Gonçalves and

Brandão 2008). This number contrasts with another extensive sampling done in one locality at the Atlantic forest that reports 253 species from 84 genera (Wilms 1995). We can estimate that about 20% may act as crop pollinators in the Atlantic forest, based on the number of known bee species from Brazil (1678 species; Moure et al. 2007) and the number of species registered as crop pollinators (311 species; BPBES/REBIPP 2019). Although *A. mellifera* is the primary crop pollinator in Brazil (as well in the Atlantic forest), there is a potential for increased use of native species, stingless bees (Apidae, Meliponini) in particular (e.g. Slaa et al. 2006).

We still know almost nothing about the natural history of the interactions between flowers and pollinators for most species of the Atlantic forest, in particular for trees and certain groups of insects, such as beetles, wasps, and flies. In addition to these fundamental gaps, we have many open questions about the process of pollination at the community scale and its relation to landscape patterns (e.g. plant dependence on pollinators and dispersal capacity of different pollinator groups). Besides that, the effect of human disturbances on pollination processes is mostly unknown. For example, we do not know which are the threshold values of habitat loss that would lead to drastic pollinator extinctions (see Viana et al. 2012). Joining more sustainable agricultural practices and increasing protected areas and restoration efforts is necessary to guarantee that plant-pollinator interactions in the Atlantic forest may not collapse, keeping ecosystem functioning and evolutionary processes intact while providing essential services to humanity.

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## References

Acosta AL, Giannini TC, Imperatriz-Fonseca VL et al (2016) Worldwide alien invasion: a methodological approach to forecast the potential spread of a highly invasive pollinator. PlosOne 11:e0148295

Agostini K, Sazima M, Sazima I (2006) Bird pollination of explosive flowers while foraging for nectar and caterpillars. Biotropica 38:674–678

Agostini K, Sazima M, Galetto L (2011) Nectar production dynamics and sugar composition in two *Mucuna* species (Leguminosae, Faboideae) with different specialized pollinators. Naturwissenschaften 98:933–942

Agostini K, Lopes AV, Machado IC (2014) Recursos florais. In: Rech AR, Agostini K, Oliveira PE, Machado IC (eds) Biologia da Polinização. Projeto Cultural, Rio de Janeiro, pp 129–150

Amorim FW, Galetto L, Sazima M (2013) Beyond the pollination syndrome: nectar ecology and the role of diurnal and nocturnal pollinators in the reproductive success of *Inga sessilis* (Fabaceae). Plant Biol 15:317–327

Amorim FW, Wyatt GE, Sazima M (2014) Low abundance of long-tongued pollinators leads to pollen limitation in four specialized hawkmoth-pollinated plants in the Atlantic Rain forest, Brazil. Naturwissenschaften 101:893–905

Andresen E, Arroyo-Rodríguez V, Escobar F (2018) Tropical biodiversity: the importance of biotic interactions for its origin, maintenance, function, and conservation. In: Dáttilo W, Rico-Gray V (eds) Ecological networks in the tropics. Springer, Cham, pp 1–13

Ashman TL, Knight TM, Steets JA et al (2004) Pollen limitation of plant reproduction: ecological and evolutionary causes and consequences. Ecology 85:2408–2421

Avila Jr RS (2009) Composição, fenologia reprodutiva e biologia da polinização de espécies esfingófilas no Parque Estadual da Serra do Mar (Núcleo Picinguaba). PhD Thesis, Ecologia, Universidade Estadual de Campinas, Campinas

Banks-Leite C, Pardini R, Tambosi LR et al (2014) Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot. Science 345:1041

Barlow J, França F, Gardner TA et al (2018) The future of hyperdiverse tropical ecosystems. Nature 559:517–526

Bezerra ELS, Machado IC (2003) Floral biology and pollination system of *Solanum stramonifolium* Jacq. (Solanaceae) in an Atlantic Forest remnant in Pernambuco. Acta Bot Bras 17:247–257

Bezerra ADM, Pacheco-Filho AJS, Bomfim IGA et al (2019) Agricultural area losses and pollinator mismatch due to climate changes endanger passion fruit production in the Neotropics. Agric Syst 169:49–57

BPBES/REBIPP (2019) Relatório temático sobre Polinização, Polinizadores e Produção de Alimentos no Brasil. Wolowski M, Agostini K, Rech AR, Varassin IG, Maués M, Freitas L, Carneiro LT, Bueno RO, Consolaro H, Carvalheiro L, Saraiva AM, Silva CI, Padgurschi MCG (Org.). Editora Cubo, São Carlos

Braun M, Dötterl S, Schlindwein C et al (2012) Can nectar be a disadvantage? Contrasting pollination natural histories of two woody Violaceae from the neotropics. Int J Plant Sci 173:161–171

Brito VLG, Sazima M (2012) *Tibouchina pulchra* (Melastomataceae): reproductive biology of a tree species at two sites of an elevational gradient in the Atlantic rainforest in Brazil. Plant Syst Evol 298:1271–1279

Brito VLG, Fendrich TG, Smidt EC et al (2016) Shifts from specialised to generalised pollination systems in Miconieae (Melastomataceae) and their relation with anther morphology and seed number. Plant Biol 18:585–593

Burkle LA, Alarcon R (2011) The future of plant–pollinator diversity: understanding interaction networks across time, space, and global change. Am J Bot 98:528–538

Burkle LA, Myers JA, Belote RT (2016) The beta-diversity of species interactions: untangling the drivers of geographic variation in plant-pollinator diversity and function across scales. Am J Bot 103:118–128

Buzato S, Sazima M, Sazima I (1994) Pollination of three species of *Abutilon* (Malvaceae) intermediate between bat and hummingbird flower syndrome. Flora 189:327–334

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

Canela MB, Sazima M (2003) *Aechmea pectinata*: a hummingbird-dependent bromeliad with inconspicuous flowers from the rainforest in South-Eastern Brazil. Ann Bot 92:731–737

Canela MB, Sazima M (2005) The pollination of *Bromelia antiacantha* (Bromeliaceae) in Southeastern Brazil: ornithophilous versus melittophilous features. Plant Biol 7:411–416

CaraDonna PJ, Petry WK, Brennan RM et al (2017) Interaction rewiring and the rapid turnover of plant–pollinator networks. Ecol Lett 20:385–394

Carneiro LT, Martins CF (2012) Africanized honey bees pollinate and preempt the pollen of *Spondias mombin* (Anacardiaceae) flowers. Apidologie 43:474–486

Castro CC, Oliveira P, Alves MC (2004) Breeding system and floral morphometry of distylous *Psychotria* L. species in the Atlantic rain forest, SE Brazil. Plant Biol 6:755–760

Cavalcante BP, Souza EH, Williams JH, Versieux LM (2019) Reproductive systems and post-pollination barriers between two closely related eu-bromelioids (Bromeliaceae) in the Atlantic Forest of Brazil. Botl J Linn Soc. https://doi.org/10.1093/botlinnean/boz101

Ceccon E, Varassin IG (2014) Plant-pollination interactions in ecosystem restoration. In: Benítez M, Miramontes O, Valiente-Banuet A (eds) Frontiers in ecology, evolution and complexity, 1st edn. CopItArXives, Mexico, pp 50–63

Cham KO, Nocelli RCF, Borges LO et al (2018) Pesticide exposure assessment paradigm for stingless bees. Environ Entomol 48:36–48

Correa-Lima APA, Varassin IG, Barve N et al (2019) Spatio-temporal effects of climate change on the geographic distribution and flowering phenology of hummingbird-pollinated plants. Ann Bot 124:389–398

Crouzeilles R, Ferreira MS, Chazdon RL et al (2017) Ecological restoration success is higher for natural regeneration than for active restoration in tropical forests. Sci Adv 3:e1701345

Danieli-Silva A, Varassin IG (2013) Breeding systems and thrips (Thysanoptera) pollination in an endangered tree (Lauraceae): implications for conservation. Plant Biol 28:31–40

de Marco P, Coelho FM (2004) Services performed by the ecosystem: forest remnants influence agricultural cultures' pollination and production. Biodivers Conserv 13:1245–1255

Domene F (2018) Reintrodução de epífitas vasculares em plantios de restauração florestal PhD. University of São Paulo

Dunley BS, Freitas L, Galetto L (2009) Reproduction of *Byrsonima sericea* (Malpighiaceae) in Restinga fragmented habitats in Southeastern Brazil. Biotropica 41:692–699

Faegri K, van der Pijl L (1979) Principles of pollination ecology. Pergamon Press, Oxford

Falcão BF, Schlindwein C, Stehmann JR (2016) Pollen release mechanisms and androecium structure in *Solanum* (Solanaceae): does anther morphology predict pollination strategy? Flora 224:211–217

Feinsinger P (1983) Coevolution and pollination. In: Futuyma DJ, Slatkin M (eds) Coevolution. Sinauer Associates, Sunderland, pp 282–310

Ferreira PA, Boscolo D, Carvalheiro LG et al (2015) Responses of bees to habitat loss in fragmented landscapes of Brazilian Atlantic Rainforest. Landsc Ecol 30:2067–2078

Fonseca GAB (1985) The vanishing Brazilian Atlantic forest. Biol Conserv 34:17–34

Fragoso FP, Varanda EM (2011) Flower-visiting insects of five tree species in a restored area of semideciduous seasonal forest. Neotrop Entomol 40:431–435

Franceschinelli EV, Carmo RM, Silva Neto CM et al (2015) Reproductive success of *Cabralea canjerana* (Meliaceae) in Atlantic forest fragments, Brazil. Rev Biol Trop 63:515–524

Freitas BM, Pinheiro JN (2010) Efeitos sub-letais dos pesticidas agrícolas e seus impactos no manejo de polinizadores dos agroecossistemas brasileiros. Oecol Aust 14:282–298

Freitas L, Vizentin-Bugoni J, Wolowski M, Souza JMT, Varassin IG (2014) Interações planta-polinizador e a estruturação das comunidades. In: Rech AR, Agostini K, Oliveira PE, Machado IC (eds) Biologia da Polinização. Projeto Cultural, Rio de Janeiro, pp 373–397

Gaiotto FA, Grattapaglia D, Vencovsky R (2003) Genetic structure, mating system, and long-distance gene flow in heart of palm (*Euterpe edulis* Mart.). J Hered 94:399–406

Garibaldi LA, Carvalheiro LG, Leonhardt SD et al (2014) From research to action: enhancing crop yield through wild pollinators. Front Ecol Environ 12:439–447

Garibaldi LA, Carvalheiro LG, Vaissière BE et al (2016) Mutually beneficial pollinator diversity and crop yield outcomes in small and large farms. Science 351:388–391

Giannini TC, Acosta AL, Garófalo CA et al (2012) Pollination services at risk: bee habitats will decrease owing to climate change in Brazil. Ecol Model 244:127–131

Giannini TC, Cordeiro GD, Freitas BM et al (2015a) The dependence of crops for pollinators and the economic value of pollination in Brazil. J Econ Entomol 108:849–857

Giannini TC, Garibaldi LA, Acosta AL et al (2015b) Native and non-native supergeneralist bee species have different effects on plant-bee networks. PLoS One 10:e0137198

Giannini TC, Costa WF, Cordeiro GD et al (2017) Projected climate change threatens pollinators and crop production in Brazil. PLoS One 12:e0182274

Girão LC, Lopes AV, Tabarelli M et al (2007) Changes in tree reproductive traits reduce functional diversity in a fragmented Atlantic Forest landscape. PLoS One 2:e908

Goldenberg R, Varassin IG (2001) Sistemas reprodutivos de espécies de Melastomataceae da Serra do Japi, Jundiaí, São Paulo, Brasil. Rev Bras Bot 24:283–288

Gonçalves RB, Brandão CRF (2008) Diversidade de abelhas (Hymenoptera, Apidae) ao longo de um gradiente latitudinal na Mata Atlantica. Biota Neotropica 8:51–61

Gottsberger G (1977) Some aspects of beetle pollination in the evolution of flowering plants. Plant Syst Evol 1:211–226

Hadley AS, Betts MG (2009) Tropical deforestation alters hummingbird movement patterns. Biol Lett 5:207–210

Hagen M, Kissling WD, Rasmussen C et al (2012) Biodiversity, species interactions and ecological networks in a fragmented world. Adv Ecol Res 46:89–210

Holzschuh A, Steffan-Dewenter I, Tscharntke T (2008) Agricultural landscapes with organic crops support higher pollinator diversity. Oikos 117:354–361

Holzschuh A, Steffan-Dewenter I, Tscharntke T (2010) How do landscape composition and configuration, organic farming and fallow strips affect the diversity of bees, wasps and their parasitoids? J Anim Ecol 79:491–500

Jeske-Pieruschka V, Fidelis A, Bergamin RS et al (2010) Araucaria forest dynamics in relation to fire frequency in southern Brazil based on fossil and modern pollen data. Rev Palaeobot Palynol 160:53–65

Johnson SD, Steiner KE (2000) Generalization versus specialization in plant pollination systems. Trends Ecol Evol 15:140–143

Jones HP, Jones PC, Barbier EB et al (2018) Restoration and repair of Earth's damaged ecosystems. Proc R Soc B 285:20172577

Klein AM, Vaissiere BE, Cane JH et al (2007) Importance of pollinators in changing landscapes for world crops. Proc R Soc B 274:303–313

Klein D, Freitas L, da Cunha M (2009) Self-incompatibility in a distylous species of Rubiaceae: is there a single incompatibility response of the morphs? Sex Plant Reprod 22:121–131

Knight TM, Steets JA, Vamosi JC et al (2005) Pollen limitation of plant reproduction: pattern and process. Annu Rev Ecol Evol Syst 36:467–497

Kormann U, Scherber C, Tscharntke T et al (2016) Corridors restore animal-mediated pollination in fragmented tropical forest landscapes. Proc Biol Sci 283:20152347

Laurance SG (2004) Landscape connectivity and biological corridors. In: Schroth G, Fonseca GAB, Harvey CA et al (eds) Agroforestry and biodiversity conservation in tropical landscapes. Island Press, Washington, pp 50–63

Lopes LE, Buzato S (2007) Variation in pollinator assemblages in a fragmented landscape and its effects on reproductive stages of a self-incompatible treelet, *Psychotria suterella* (Rubiaceae). Oecologia 154:305–314

Lopes AV, Girão LC, Santos BA et al (2009) Long-term erosion of tree reproductive trait diversity in edge-dominated Atlantic forest fragments. Biol Conserv 142:1154–1165

Maia FR, Varassin IG, Goldenberg R (2016) Apomixis does not affect visitation to flowers of Melastomataceae, but pollen sterility does. Plant Biol 18:132–138

Malucelli TS, Maia FR, Varassin IG (2018) Breeding system and pollination of *Pleroma trichopodum* DC. (Melastomataceae): a potential species for the restoration of Atlantic Forest in southern Brazil. Acta Bot Bras 32:402–409

Marsden SJ, Whiffin M, Galetti M (2001) Bird diversity and abundance in forest fragments and *Eucalyptus* plantations around an Atlantic forest reserve, Brazil. Biodivers Conserv 10:737–751

Martins R, Antonini Y (2016) Can pollination syndromes indicate ecological restoration success in tropical forests? Restor Ecol 24:373–380

Martins AC, Silva DP, De Marco P et al (2015) Species conservation under future climate change: the case of *Bombus bellicosus*, a potentially threatened South American bumblebee species. J Insect Conserv 19:33–43

Maruyama PM, Vizentin-Bugoni J, Sonne J et al (2016) The integration of alien plants in mutualistic plant–hummingbird networks across the Americas: the importance of species traits and insularity. Divers Distrib 22:672–681

Matallana G, Godinho M, Guilherme F et al (2010) Breeding systems of Bromeliaceae species: evolution of selfing in the context of sympatric occurrence. Plant Syst Evol 289:57–65

Maués MM, Varassin IG, Freitas L et al (2012) A Importância dos polinizadores nos biomas brasileiros, conhecimento atual e perspectivas futuras para conservação. In: Imperatriz-Fonseca VL, Canhos DAL, Alves DA et al (eds) Polinizadores no Brasil. Edusp, São Paulo, pp 49–66

MengoniGoñalons C, Farina WM (2018) Impaired associative learning after chronic exposure to pesticides in young adult honey bees. J Exp Biol 221:jeb176644

Metzger JPW, Brancalion PHS (2013) Challenges and opportunities in applying a landscape ecology perspective in ecological restoration: a powerful approach to shape neolandscapes. Nat Conservacao 2:103–107

Morris RJ (2010) Anthropogenic impacts on tropical forest biodiversity: a network structure and ecosystem functioning perspective. Philos Trans R 365:3709–3718

Moure JS, Urban D, Melo GAR (org.) (2007) Catalogue of bees (Hymenoptera, Apoidea) in the neotropical region. Sociedade Brasileira de Entomologia, Curitiba

Nehren UDO, Kirchner A, Sattler D et al (2013) Impact of natural climate change and historical land use on landscape development in the Atlantic Forest of Rio de Janeiro, Brazil. An Acad Bras Cienc 85:497–518

Nery LS, Takata JT, Camargo BB, Chaves AM, Ferreira PA, Boscolo D (2018) Bee diversity responses to forest and open areas in heterogeneous Atlantic Forest. Sociobiology 65:686–695

Newmark WD, Jenkins CN, Pimm SL et al (2017) Targeted habitat restoration can reduce extinction rates in fragmented forests. Proc Natl Acad Sci U S A 114:9635–9640

Nicholls C, Altieri M (2013) Plant biodiversity enhances bees and other insect pollinators in agroecosystems. A review. Agron Sustain Dev 33:257–274

Nunes CEP, Gerlachb G, Bandeirac KDO et al (2017) Two orchids, one scent? Floral volatiles of *Catasetum cernuum* and *Gongora bufonia* suggest convergent evolution to a unique pollination niche. Flora 232:207–216

Oliveira RR, Engemann C (2011) História da paisagem e paisagens sem história: a presença humana na Floresta Atlântica do Sudeste Brasileiro. Esboços: histórias em contextos globais 18:9–1

Ollerton J (2017) Pollinator diversity: distribution, ecological function, and conservation. Annu Rev Ecol Evol Syst 48:353–376

Ollerton J, Killick A, Lamborn E et al (2007) Multiple meanings and modes: on the many ways to be a generalist flower. Taxon 56:717–728

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

Pansarin E, Amaral M (2009) Reproductive biology and pollination of Southeastern Brazilian *Stanhopea*Frost ex Hook. (Orchidaceae). Flora 204:238–249

Pinheiro JN, Freitas BM (2010) Efeitos letais dos pesticidas agrícolas sobre polinizadores e perspectivas de manejo para os agroecossistemas brasileiros. Oecol Aust 14:266–281

Pinheiro M, Abrão B, Harter-Marques B et al (2008) Floral resources used by insects in a grassland community in Southern Brazil. Rev Bras Bot 31:469–489

Pires J, Freitas L (2008) Reproductive biology of two tree species of Leguminosae in a montane rain forest in Southeastern Brazil. Flora 203:491–498

Pires JPA, Silva AG, Freitas L (2014) Plant size, flowering synchrony and edge effects: what, how and where they affect the reproductive success of a Neotropical tree species. Austral Ecol 39:328–336

Potts SG, Biesmeijer JC, Kremen C et al (2010) Global pollinator declines: trends, impacts and drivers. Trends Ecol Evol 25:345–353

Proctor M, Yeo P, Lack A (1996) The natural history of pollination. Harper Collins Publishers, London

Ramalho M (2004) Stingless bees and mass flowering trees in the canopy of Atlantic Forest: a tight relationship. Acta Bot Bras 18:37–47

Ramalho AV, Gaglianone MC, de Oliveira ML (2009) Comunidades de abelhas Euglossina (Hymenoptera, Apidae) em fragmentos de Mata Atlântica no sudeste do Brasil. Rev Bras Entomol 53:95–101

Ramos FN, Santos FAM (2006) Floral visitors and pollination of *Psychotria tenuinervis* (Rubiaceae): distance from the anthropogenic and natural edges of an Atlantic Forest fragment. Biotropica 38:383–389

Rech AR, Agostini K, Oliveira PE, Machado IC (eds) (2014) Biologia da polinização. Projeto Cultural, Rio de Janeiro

Rezende CL, Scarano FR, Assad ED et al (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conser 16:208–214

Ribeiro KT, Freitas L (2010) Potential impacts of changes to Brazilian Forest Code in Campos rupestres and Campos de altitude. Biota Neotrop 10:239–246

Rocca M, Sazima M (2008) Ornithophilous canopy species in the Atlantic Rain Forest of Southeastern Brazil. J Field Ornithol 79:130–137

Rodrigues RR, Brancalion PHS, Isernhagen I (2009) Pacto pela restauração da Mata Atlântica: referencial dos conceitos e ações de restauração florestal. Instituto BioAtlântica, São Paulo

Roubik DW (2018) The pollination of cultivated plants: a compendium for practitioners. Food and Agriculture Organization of the United Nations (FAO), Roma

SanMartin-Gajardo I, Sazima M (2005) Chiropterophily in Sinningieae (Gesneriaceae): *Sinningia brasiliensis* and *Paliavana prasinata*are bat-pollinated, but *P. sericiflora*is not. Not yet? Ann Bot 95:1097–1103

Santos APM, Fracasso CM, Santos ML et al (2012) Reproductive biology and species geographical distribution in the Melastomataceae: a survey based on New World taxa. Ann Bot 110:667–679

Santos J, Varassin IG, Muschner VC et al (2018) Estimating seed and pollen dispersal kernels from genetic data demonstrates a high pollen dispersal capacity for an endangered palm species. Am J Bot 105:1–11

Saturni FT, Jaffé R, Metzger JP (2016) Landscape structure influences bee community and coffee pollination at different spatial scales. Agric Ecosyst Environ 235:1–12

Sazima M (1972) Observações sobre polinização em Velloziaceae. Ciênc Cult 24:335

Sazima M, Sazima I (1975) Quiropterofilia em *Lafoensia pacari* St.Hil (Lythraceae), na Serra do Cipó, Minas Gerais. Ciênc Cult 27:406–416

Sazima I, Buzato S, Sazima M (1996) An assemblage of hummingbird-pollinated flowers in a montane forest in Southeastern Brazil. Bot Acta 109:149–160

Sazima M, Sazima I, Buzato S (1999) Bat-pollinated flower assemblages and bat visitors at two Atlantic Forest sites in Brazil. Ann Bot 83:705–712

Sazima M, Buzato S, Sazima I (2003) *Dyssochroma viridiflorum* (Solanaceae): a reproductively bat-dependent epiphyte from the Atlantic Rainforest in Brazil. Ann Bot 92:725–730

Schneider SS, DeGrandi-Hoffman G, Smith DR (2004) The African honey bee, factors contributing to a successful biological invasion. Annu Rev Entomol 49:351–376

Singer R, Sazima M (2001) The pollination mechanism of three sympatric *Prescottia* (Orchidaceae: Prescottinae) species in Southeastern Brazil. Ann Bot 88:999–1005

Slaa EJ, Chaves LAS, Malagodi-Braga KS et al (2006) Stingless bees in applied pollination: practice and perspectives. Apidologie 37:293–315

Stouffer PC, Bierregaard RO Jr (1995) Effects of forest fragmentation on understory hummingbirds in Amazonian Brazil. Conserv Biol 9:1085–1094

Strassburg BBN, Beyer HL, Crouzeilles R et al (2019) Strategic approaches to restoring ecosystems can triple conservation gains and halve costs. Nat Ecol Evol 3:62–70

Tabarelli T, Aguiar AV, Ribeiro MC et al (2010) Prospects for biodiversity conservation in the Atlantic Forest: lessons from aging human-modified landscapes. Biol Conserv 143:2328–2340

Tonhasca A Jr, Blackmer JL, Albuquerque G (2002) Abundance and diversity of Euglossine bees in the fragmented landscape of the Brazilian Atlantic Forest. Biotropica 34:416–422

Townsend PA, Levey DJ (2005) An experimental test of whether habitat corridors affect pollen transfer. Ecology 86:466–475

Vaissière BE, Freitas BM, Gemmill-Herren (2011) Protocol to detect and assess pollination deficits in crops: a handbook for its use. FAO, Roma

Valle BRALG (2018) Diversidade de beija-flores (Trochilidae) em fragmentos florestais na zona de amortecimento e entorno do Parque Nacional do Itatiaia. Professional MSc, Jardim Botânico do Rio de Janeiro, Rio de Janeiro

Viana BF, Boscolo D, Mariano-Neto E et al (2012) How well do we understand landscape effects on pollinators and pollination services? J Pollinat Ecol 7:31–41

Warring B, Cardoso FCG, Marques MM et al (2016) Functional diversity of reproductive traits increases across succession in the Atlantic forest. Rodriguésia 67:321–333

Waser NM, Ollerton J (2006) Plant-pollinator interactions – from specialization to generalization. The University of Chicago Press, Chigaco

Waser NM, Chittka L, Price MV et al (1996) Generalization in pollination systems, and why it matters. Ecology 77:1043–1060

Wendt T, Coser T, Matallana G, Guilherme F (2008) An apparent lack of prezygotic reproductive isolation among 42 sympatric species of Bromeliaceae in Southeastern Brazil. Plant Syst Evol 275:31–41

Wilms W (1995) Die bienenfauna im Küstenregenwald Brasiliens und ihre Beziehungen zu Blütenplanzen: Fallstudie Boracéia, São Paulo. PhD thesis, Fakultät für Biologie der Eberhard-Karls-Universität Tübingen, Tübingen

Wolowski M, Ashman T-L, Freitas L (2014) Meta-analysis of pollen limitation reveals the relevance of pollination generalization in the Atlantic Forest of Brazil. Plos One 9:e89498

Zambon V, Agostini K, Nepi M, Rossi ML, Martinelli AP, Sazima M (2019a) The role of nectar traits and nectary morphoanatomy in the plant-pollinator interaction between *Billbergia distachia* (Bromeliaceae) and the hermit *Phaethornis eurynome* (Trochilidae). Bot J Linn Soc. https://doi.org/10.1093/botlinnean/boz107

Zambon V, Agostini K, Nepi M, Rossi ML, Martinelli AP, Sazima M (2019b) Nectar as manipulator: how nectar traits influence changes in pollinator groups of *Aechmea vanhoutteana*, a bromeliad from the Brazilian Atlantic Forest. Bot J Linn Soc. https://doi.org/10.1093/botlinnean/boz086

# Part IV
# Opportunities

# Chapter 16
# Atlantic Forest: Ecosystem Services Linking People and Biodiversity

**Aliny P. F. Pires, Carolina Y. Shimamoto, Maíra C. G. Padgurschi, Fabio R. Scarano, and Marcia C. M. Marques**

**Abstract** The Atlantic forest biome is a key provider of ecosystem services (ES) delivered worldwide. The region presents a complex history of land use changes based on human demands that have decreased its native vegetation and the potential to integrate biodiversity and human wellbeing. In this chapter, we described the profile of the scientific output on ES for the Atlantic forest; the spatial patterns of ES; projected scenarios, particularly considering the potential for ecosystem restoration; and the profile of public policies and private practices based on ES in the region. We emphasize that the integration of multiple strategies that fill the knowledge gaps, identify local demands and trade-offs, and promote dialogue between multiple sectors will be key to ensure the provisioning of ES in the Atlantic Forest in the coming years.

**Keywords** Ecosystem service · Nature's contributions to people · Knowledge gaps · Forest restoration · Public policy · Private sector · Payment for environmental services (PES)

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## 16.1 Introduction

Can an ecosystem that lost nearly 3/4 of its original vegetation cover continue to supply more than 100 million people with food, water, and maintain a stable climate? This is a central question to the Brazilian Atlantic forest biodiversity hotspot, and a cautious answer would probably be *"yes, so far, but possibly not for long"*. The Atlantic forest domain (hereafter AF) holds more than half of the land dedicated to horticulture and food production in Brazil, contributes to 70% of the country's gross domestic product (GDP), and 2/3 of the industrial economy. On the other hand, symptoms of ES decline are more frequent and include major water crises in big cities and metropolitan areas, such as São Paulo, Rio de Janeiro, and Belo Horizonte; a marked increase in the frequency and intensity of extreme climatic events, such as heavy rains, landslides, heat waves, floodings, and droughts; an impressive increase in the occurrence of biological invasions; and key pollinators for agriculture and native ecosystems are at the brink of extinction (Joly et al. 2014; Vitule et al. 2014; Scarano and Ceotto 2015; Scarano 2019a; Wolowski et al. 2019).

This combination of high supply and demand of key ES and the threats to such services seem to indicate a socioecological system on the boundary of transition. Political, economic, social, and scientific actions at present are likely to define whether this transition is for a more or less sustainable future. Therefore, current opinion on the present and future status of the AF ranges from "shrinking hotspot" (Ribeiro et al. 2011) to "future hotspot" (Scarano and Ceotto 2015; Rezende et al. 2018b). For that, it will be determinant to understand the spatial differences in the AF region.

The biome occupies most of the coastal region of the country and a significant percentage of inland areas. It also comprises urban and rural areas, with different land uses and climates that establish different challenges and perspectives for each portion of the biome. Thus, as much as it is essential to understand local patterns of supply and demand of ES, it is also critical to consider the flow of ES throughout distances beyond local range, from where benefits are delivered to areas where they are used (Torres et al. 2017). It will allow us to distinguish the best way we should address the main issues ensuring human wellbeing, social and economic development, and biodiversity conservation in the biome.

Irrespective of how accurate and spatialized projections can be, the fact is that ES is a key concept, both from theoretical and operational points of view. It is what the Argentinian philosopher Mario Bunge calls a "glue concept", i.e., a concept that binds together distinct approaches and allows for the emergence of a new one (Bunge 2003). In this case, the ES concept offers a glue to bind natural and social sciences and allow for the emergence of a more inter- and transdisciplinary sustainability science (Pires et al. 2018). It also helps to create links and mixes between environmental and socioeconomic policies (Crouzat et al. 2018; Scarano 2019b). Recently, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has proposed a new terminology: "nature's contribution to people" (NCP – Díaz et al. 2018). It is supposed to be more inclusive of the

### Box 16.1: Ecosystem Services and Nature's Contributions to People: Building Bridges

It is estimated over eight million eukaryotic species existing globally. All of them interact with their biotic and abiotic environment establishing a flow of materials and energy in an ecologically dynamic system. From that flow and its effects on soil and atmosphere emerge ecosystem functions, irrespective of whether functions are useful for humans (Braat and de Groot 2012). An example is provided by nutrient cycling, a process mediated by the decomposers and needed for water purification that is, in turn, a function that provides clean water. Therefore, since biodiversity models many natural traits, it is essential for human wellbeing and thus these functions impart to society a variety of benefits.

In millennia of evolution, both biological and culturally, the ability to understand and use such benefits allowed human societies to create innovations that led them to the world's domination. However, especially after the industrial era, these innovations eventually established a gap between man and nature, a situation that began to reverse only in the 1960/1970s with the perception of the finiteness of natural resources (Helliwell 1969). Scholars started to frame ecological concerns in economic terms to demonstrate that the costs to society appear when the functions of an ecosystem are impaired (Ehrlich and Mooney 1983; de Groot 1987; Braat and de Groot 2012). In 1981, such benefits and the role of functioning ecosystems were then summarized by Ehrlich and Ehrlich in the "Ecosystem Services" (ES) concept (Ehrlich and Ehrlich 1981).

Since then, the concept has evolved and in 2005 definitely entered in policy agenda with the Millennium Ecosystem Assessment (MEA). In MEA definition, derived from Daily (1997) and Costanza et al. (1997) approaches, *ES are the benefits people obtain from ecosystems* (MEA 2005). From the former, the term "services" encompasses both the tangible (goods) and the intangible (services) benefits; from the other, the inclusion of natural and human-modified ecosystems as sources of ES (MEA 2005). Originally, such essentiality of natural environments to human life came from the economic worldview. Some argue this utilitarian framing can modify the way humans perceive and relate to nature in a manner that may be counterproductive for conservation purposes (see Gómez-Baggethun et al. 2010 for a historical analysis). In addition, ES would be failing to engage a range of perspectives and dimensions of value (Chan et al. 2012; Díaz et al. 2018).

People have different experiences, truths, and knowledge that may change over time, differ from the scientific lens, and which affect decision making (Daily 2000; Bennett et al. 2015; Pascual et al. 2017; Ishihara 2018). This cultural diversity has a central role in defining links between people and nature and ES concept is lacking to catch such multiple worldviews (Díaz et al. 2018). Alternatively, the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) gives, in its conceptual framework, a broader approach named Nature's Contributions to People (NCP) (Díaz et al. 2015). NCP is *all the positive contributions or benefits, and occasionally negative contributions, losses, or detriments, that people obtain from nature* (Pascual et al. 2017). The concept emphasizes the importance of culture as a *cross-cutting factor shaping the human perception of nature and good quality of life* (Peterson et al. 2018). It innovates connecting present and future through the *maintenance of options*, i.e., the capacity of ecosystems to keep options open (Peterson et al. 2018). It also emphasizes the possibility of *negative contributions from nature* or *disservices* that in most cases do not surplus the positive contributions, but highlights the importance to discuss trade-offs between multiple aspects (Sacchi et al. 2017).

On the other hand, ES has connected ecologists, economists, and social scientists in their efforts to understand the interactions between nature and people (Maes et al. 2018). After each of its milestones (1981 – first conceptualization; 1997 – global quantitative economic estimates; 2005 – policy agenda), the knowledge of many aspects (biological, economic, political, etc.) and the use itself of ES approach increased. Mostly after 2005, many scholars in developing countries, such as Brazil, have been dedicating more attention and studies on ecosystem services and such achievements cannot be overlooked (Peterson et al. 2018; Pires et al. 2018). Furthermore, we live in the Anthropocene era where humans are shaping the Earth both direct, by land use changes, for example, and indirect, by the consequences of these changes (Rockström et al. 2009). In some way, changes are necessary for a good quality of life as agro-ecosystems which are not typically classified as natural and thus replacing "ecosystems" with "nature" may underestimate the importance of socio-ecological processes (Peterson et al. 2018). Occasionally, interactions between people and environment result in the enhancing/coproduction of ecosystem services through the use of advanced technology (Rieb et al. 2017) and in a one-directional flow from "nature" to "people" this cannot be possible (Peterson et al. 2018). But what separates them unites them.

The multiplicity of visions, values, cultures, biodiversity, and ecosystem functions are at the base of both concepts. A broad use of them, in addition to their derivatives (natural capital, ecosystem goods, green infrastructure, nature's benefits, etc.), would be positive to communicate the value of biodiversity to a variety of stakeholders, considering the existence of different approaches and expertise (Peterson et al. 2018; Maes et al. 2018). It is critical to incorporate such multiple views into the sustainability arena to consistently promote biodiversity conservation and human wellbeing (Pires et al. 2020).

variety of, and less represented, values and knowledge systems, such as those of indigenous and traditional people, than ecosystem services (Peterson et al. 2018). However, the controversy is such (Masood 2018) that the NCP concept will be treated here as a synonym of ES, following the proposition of previous studies

(Costanza et al. 2017; de Groot et al. 2018). In Box 16.1, we present an overview of the differences between both concepts.

One evidence of the usability of ES concept is its marked increase in Brazil – and in particular in the AF – both in scientific output and in the diversity of policies and practices based on this concept (Pires et al. 2018). In this chapter, we will explore these multiple aspects and examine the profile of the scientific output on ecosystem services for the Atlantic forest. We also explore the spatial patterns of ES in AF municipalities; the potential for ecosystem restoration; and the profile of public policies and private practices based on ES.

## 16.2 Profile of Scientific Production on Atlantic Forest Ecosystem Services

The scientific knowledge about ES in Brazil has exponentially increased in the last decade, which is particularly outstanding for AF that is the focus of almost 57% of all ES studies in the country (Pires et al. 2018). However, although one can observe an exponential increase in the number of studies related to ecosystem services in Brazil, only 2.4% are linked to human wellbeing. When compared with other developing countries, including South Africa, Mexico, and Chile, Brazil has the worst rate (Pires et al. 2018).

AF concentrates a significant part of the ES delivered in Brazil. The region is responsible for food production, several cities depend on ecotourism based on biodiversity and water resources conservation, and several carbon sequestration programs were implemented in the biome (Brancalion et al. 2014). Nevertheless, changes in land use, such as those due to mining, agriculture, and urbanization, reduced its native vegetation in 28% of its original cover (Rezende et al. 2018b), which represent an increase compared with previous estimates (Ribeiro et al. 2009).

Water scarcity has promoted substantial economic losses and it has compromised the human wellbeing in the whole biome (Nobre et al. 2016). However, studies focused on water provisioning remain scarce (Table 16.1, Fig. 16.1) and investigate mostly the role of native vegetation on supporting (e.g., species diversity) and regulating services (e.g., carbon storage) (Pires et al. 2018). Several biodiversity dimensions are underrepresented in the scientific literature potentially limiting our ability to determine the contribution of the biome in provisioning ES, both considering local and global perspectives. Additionally, supporting and regulating services are hard to be perceived directly by the population, which can hinder social engagement. Other important services such as ecotourism, air, and water quality are also underrepresented in our survey (Table 16.1) but strongly valued by people (Brancalion et al. 2014). In São Paulo, diseases caused by air pollution are considered twice more fatal than traffic accidents (Instituto Saúde e Sustentabilidade 2018).

These biases became even worse when we consider the existence of possible trade-offs between multiple ecosystem services (Kennedy et al. 2016). Few studies

**Table 16.1** Ecosystem services studied in the Atlantic Forest

| Type | Ecosystem services | References |
|---|---|---|
| Provision | Water provisioning | Sovacool (2011); Förster et al. (2015); Richards et al. (2015); Cao et al. (2015); Richards et al. (2017) |
| | Food production | Tscharntke et al. (2011); de Souza et al. (2012); Turetta et al. (2016); Alves-Pinto et al. (2017) |
| Supporting | Species diversity | Gardner et al. (2007); Tabarelli et al. (2012); Brancalion et al. (2013); Banks-Leite et al. (2014); Lacerda (2016); Galetti et al. (2017); Audino et al. (2017); Campos et al. (2017); Begotti et al. (2018); Raymundo et al. (2018) |
| | Habitat-related services | Gomiero et al. (2010); Friess and Webb (2014); Strassburg et al. (2014); Santos and Scotti (2018) |
| Regulating | Water regulation | Soares-Filho et al. (2016); Taffarello et al. (2016); Pires et al. (2017) |
| | Soil erosion | Alarcon et al. (2015); Filoso et al. (2015); Strassburg et al. (2016); Duarte et al. (2016) |
| | Air quality | Filoso et al. (2015) |
| | Carbon storage and sequestration | Petz et al. (2014); Robinson et al. (2015); Magnago et al. (2015); Alarcon et al. (2015); Ferraro et al. (2015); Strassburg et al. (2016); Duarte et al. (2016); Chazdon et al. (2016); d'Albertas et al. (2018) |
| | Pollination | De Marco and Coelho (2004); Aleixo et al. (2014); Giannini et al. (2015); Moreira et al. (2015); Saturni et al. (2016); Hipólito et al. (2018) |
| | Seed dispersal | Sarmento et al. (2014); Bufalo et al. (2016); Oliveira-Silva et al. (2018) |
| | Biotic resistance | Jordani et al. (2015); Rocha-Santos et al. (2016); Kremer and da Rocha (2016); Fonseca et al. (2017); Librán-Embid et al. (2017) |
| Cultural | Cultural and personal value | Adams et al. (2008); de Souza et al. (2016); França et al. (2016); Fernandes et al. (2016); Correia et al. (2016) |
| | Ecotourism | Ruiz-Mallén et al. (2015) |

The table includes some key references retained from the database produced by Pires et al. (2018) for the Atlantic Forest, updated references and complemented with the authors' database

have suggested that forest restoration has a positive effect in improving water quality and quantity (Pires et al. 2017; Sone et al. 2019), but it can also cause negative effects on water yield (Filoso et al. 2017). If we consider that the region is vulnerable to drought (Nobre et al. 2016) and most of the restoration programs are focused on water-related and carbon services (Young and de Bakker 2014), maybe better spatial planning is required to reach the expected results. Although still incipient, prioritization modeling points to restoration solutions that avoid the loss of biodiversity while increasing substantially the carbon sequestration service (Strassburg et al. 2019). There is evidence that biodiversity restoration and carbon sequestration service can be synergistic and part of a common agenda. Therefore, prioritization efforts that integrate multiple services will be determinant to establish

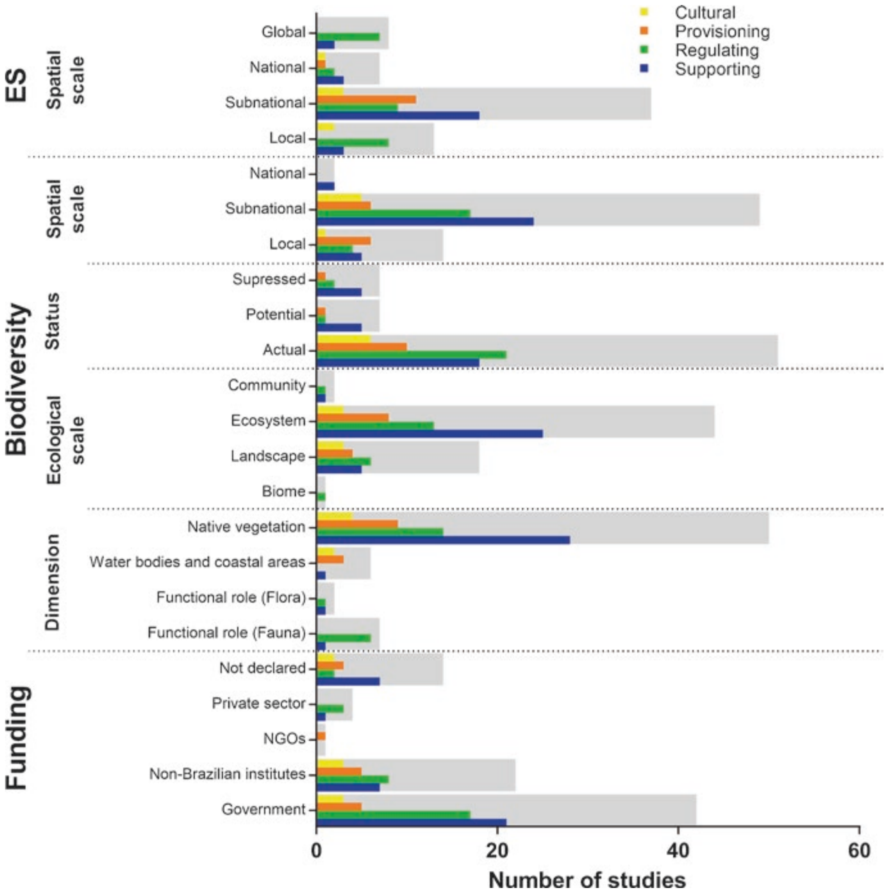

**Fig. 16.1** Scientific knowledge produced in the Atlantic Forest biome. Data were collected following the methods described in Pires et al. (2018) by using the search terms "biodiversity" and "ecosystem services" in the Web of Science database and focused on AF. For all categories, there is a great bias for a specific class. There is a prevalence of government funding in studies focusing on the role of actual native vegetation for supporting and regulating ecosystem services able to reach subnational scales. For detailed methods, categories, and classes used, see Pires et al. (2018)

the best-fitted areas for restoration programs by considering synergies and trade-offs among ES.

Studies considering future land cover, climatic changes, and potential socioeconomic scenarios for ES are key but still scarce. Most studies are dedicated to the role of actual biodiversity on ES, rather than addressing future or past conditions. It is strategic to integrate these multiple temporal perspectives since modeled scenarios would help to verify the efficiency of planning strategies and to evaluate the losses of suppressed biodiversity that could allow designing the best valuation technics (Fig. 16.1).

## 16.3 Spatial Patterns of Ecosystem Services in Atlantic Forest

The mapping of ecosystem services identifies locations where they are concentrated or lacking (Naidoo et al. 2008; Raudsepp-Hearne et al. 2010). Different spatial patterns are found across the AF, which is consistent with the geographical distribution of land uses and protected areas (Fig. 16.2). Regulating services (water balance, carbon stock, and soil productive capacity) show higher occurrence (≥60% of total) in municipalities with protected areas, mainly in the Central and Southern regions of the AF (Santa Catarina, Paraná, São Paulo, and Bahia states). These regions coincide with the largest AF remnants and correspond to the *Serra do Mar* mountain chain with 36.5% (4,169,797 ha) of original vegetation preserved, followed by Bahia with 17.7% (2,162,287 ha) of preserved vegetation (Ribeiro et al. 2009).

The provision services related to crop (sugarcane, soybean, and corn) and cattle productions are distributed mainly in inland areas of Southern AF (São Paulo, Paraná, and Mato Grosso do Sul States). Soybean and corn show higher percentages for provisioning services in municipalities with no protected areas (55% and 62% respectively), while sugarcane and cattle represent more than 50% values in municipalities containing protected areas. Coffee production is higher in the Southern region of AF (Espírito Santo state, by the coast, and in the inland state of Minas Gerais) with 61% in municipalities with no protected areas (Fig. 16.2).

This spatial pattern suggests that socioeconomic and/or ecological patterns are possible drivers for ES in AF (Molin et al. 2017). The provisioning services were more prominent in the Southern region (São Paulo and Paraná states) of the Atlantic Forest. This region shows adequate edapho-climatic and logistic conditions for the development of crop production (Buainain et al. 2014). On the other hand, in the South region, soil productive capacity, a regulation service, is more prominent. This difference can be explained by agricultural modernization, which made use of mechanization and chemical fertilizers to expand production areas and increase fertility in regions with poorer soils, such as in the inland of the Southern AF (Buainain et al. 2014). Yet, the predominance of agriculture in these regions is a reflection of a plan topography.

Protected areas (PAs) are important to the conservation of ecosystem services (Gaston et al. 2008) and Brazil has the largest protected areas system in the world, covering approximately 220 million ha (Bernard et al. 2014). AF has 1225 PAs that are key elements to preserving important forest remnants (BPBES 2019). The distributions of different ecosystem services across AF (Fig. 16.2) demonstrate that regulating services (water balance, carbon stock, and soil productive capacity) are concentrated in these PAs, although they represent less than 10% of AF (Jenkins et al. 2015; BPBES 2019), whereas provision services (coffee, soybean, and corn) are concentrated outside the limits of these areas. This emphasizes the importance of PAs to promote key ecosystem services. For instance, Medeiros et al. (2011) have shown that, in Brazil, more than 80% of the water that feeds into reservoirs for energy generation comes from PAs. It has also been demonstrated that the proximity

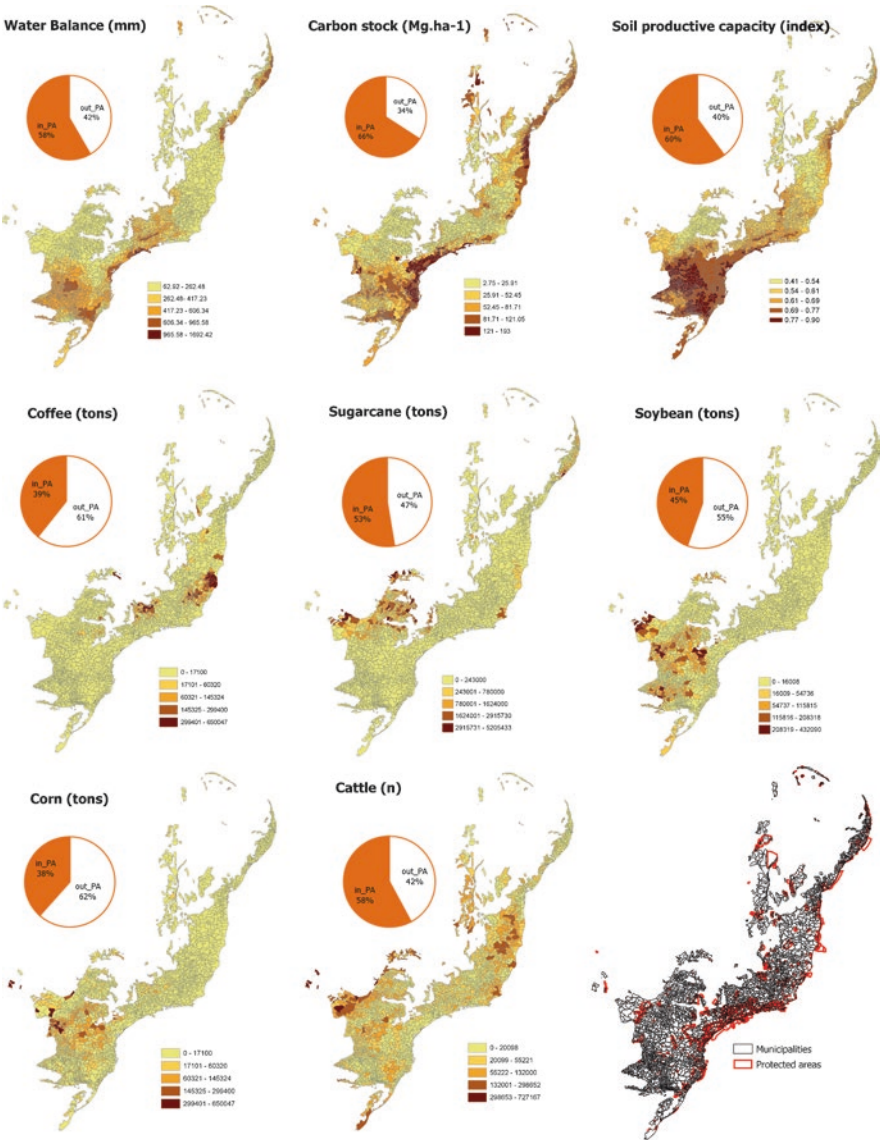

**Fig. 16.2** Spatial distribution of 8 selected ecosystem services in the 3039 municipalities of Atlantic Forest, considering services in protected areas (in_PA; orange) and outside protected areas (out_PA; white). Data from Hijmans et al. (2005), Batistella et al. (2012), Trabucco and Zomer (2010), Ruesch and Gibbs (2008), and Hengl et al. (2014). The last figure (bottom right) shows the limits of the Brazilian Atlantic Forest biome, the municipalities and protected areas (Source: MMA 2015 and SOS Mata Atlântica 2014)

of native vegetation areas can increase crop productivity by ensuring the biological activity and presence of pollinators, such as in coffee plantations (Saturni et al. 2016; Wolowski et al. 2019). Pollinators favor a 30% increase in the yield of this crop (Hipólito et al. 2018).

Ecosystem services are often supplied at long distances. A great part of the food production in the AF supplies demands outside the biome. Conversely, the population living in the domain of this biome also demand services that are delivered by other regions or biomes. These teleconnections (e.g., Torres et al. 2017; Scarano 2019b) should be considered for design and implement conservation and restoration strategies.

## 16.4 The Future of Ecosystem Services Provision in the Atlantic Forest

We can expect great changes in the AF environmental condition in the coming decades. In a business-as-usual scenario, it is projected that (i) the biome will become more urbanized, with increased human densities, especially in areas surrounding PAs (Güneralp and Seto 2013); (ii) pollution will further compromise water sources and air quality causing a set of diseases that will affect human health with great economic effects (Instituto Saúde e Sustentabilidade 2018, Instituto Trata Brasil 2018); (iii) invasive species occurrence will increase, compromising important services; and (iv) climatic changes will increase the occurrence of extreme drought and flood events (Lyra et al. 2017). Thus, strategies that enhance the capacity of the AF to provide ES are key to ensure human wellbeing in the biome and elsewhere (Scarano and Ceotto 2015).

Several studies have shown that ecological restoration provides critical ES (Blangy and Mehta 2006; Rey Benayas et al. 2009; Alexander et al. 2016). Thus, this is one of the most promising practices to ensure ES in highly degraded and fragmented systems (Latawiec et al. 2015). It is no coincidence that audacious global initiatives such as the Bonn Challenge and the Initiative 20×20 have focused on restoring ecosystems globally, especially tropical forests. Restoration actions traditionally focus on the recovery of both structure (including biodiversity) and function of degraded ecosystems (Lamb 2005). However, recently, restoration projects have focused on the recovery of ecosystem services, which has attracted the attention of funders, decision makers, and the general population, given the benefits that services bring for human wellbeing (Bullock et al. 2011).

Ecological restoration improves soil conditions by retaining and delivering nutrients to plants, affecting nutrient and biogeochemical cycles (Amazonas et al. 2011). Additionally, the incorporation of biomass potentially increases carbon sequestration (Chazdon et al. 2016), affecting the global carbon cycle. Specifically, in tropical forests, restoration actions contribute to a significant increase in the levels of ecological indicators of carbon pool, soil attributes, and biodiversity protection, comparatively to disturbed areas (Shimamoto et al. 2018). Specifically, in the AF, restoration can promote an exponential increase in carbon sink service over a 60-year period (Shimamoto et al. 2014).

A set of factors stimulates AF restoration. It includes investments from the private and public sectors, payment for environmental services (PES), and the engagement of several NGOs. The Brazilian Native Vegetation Protection Law (NVPL)[1] is expected to enforce mandatory restoration actions in all biomes and a governance system (Planaveg) was created to optimize the implementation of such restoration action (Brancalion et al. 2016). At the same time, the Pact for the Restoration of the Atlantic Rainforest, a broad and ambitious restoration initiative, has mobilized different actors to ensure effective restoration in the biome, despite still lacks evaluation of its efficiency. Although primarily focused on biological diversity, these and other initiatives also seek the restoration of ecosystem services. This large-scale restoration initiative requires effective spatial planning in order to reduce costs and to involve the population. Practices focused on increasing land productivity will diminish the deforestation pressure in the biome, including land sharing and land sparing, making land available for large restoration projects (Latawiec et al. 2015). Some models have been proposed for the prioritization of the restoration of the AF and indicated the regions where the gain of biodiversity and the reduction of restoration costs (Tambosi et al. 2014; Duarte et al. 2016; Zwiener et al. 2017) and climate change mitigation are optimized (Strassburg et al. 2019).

Although large-scale restoration planning addresses many of the demands for restoration management, prioritization may lose the true dimension of regional and local conditions, resulting in ineffective approaches to ensuring effective restoration in the practice. Thus, large-scale prioritization models need to be better calibrated, considering the multiple functions and services that are best evaluated locally or regionally (Latawiec et al. 2015). On the other hand, local and regional initiatives, if not tied to large-scale planning, fail to provide sensitive ecosystem services at regional scales, such as the water supply service, which is sensitive only at the regional watershed scale (Pires et al. 2017).

## 16.5 Defining Strategies for Ensuring Ecosystem Services in the Atlantic Forest

It is expected that the implementation of the NVPL will promote a significant increase in the AF cover (Brancalion et al. 2016; Pires et al. 2017; Rezende et al. 2018b) since it promotes new mechanisms and policies to support the implementation of the law. Legal gaps in restoration include estimates of more than 720 thousand ha of native vegetation in the Rio Doce watershed (Pires et al. 2017) and more than 412 thousand ha for the state of Rio de Janeiro (Rezende et al. 2018a). Most of

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[1] Law number 12.651, 2012.

the states in the biome have PES programs implemented, mainly focused on restoring areas surrounding spring areas, to ensure water provisioning. In 2005, the "Conservador das Águas" project was implemented in Extrema, Minas Gerais with the support of several institutions (Richards et al. 2015). The Oasis Project started in São Paulo in 2006 and received funds from the Mitsubishi Foundation that expanded the project to other states, including Parana (Young and de Bakker 2014). In the metropolitan region of São Paulo, the project "Produtor de Água" was developed from 2009 to 2014 in the Cantareira, responsible to supply the main water reservoir system in the state. All these initiatives are focused on compensating landowners that accepted to protect forest areas to ensure the watershed protection and they emphasized the potential of such initiatives in providing ES. The efficiency of such PES programs will be guaranteed if the establishment of a payment scheme combines the opportunity cost of land and indices for the quality and quantity of conservation (Young and de Bakker 2014). Studies suggest that the monetary expectation of landowners is 35% higher for restoration than conservation PES programs (Alarcon et al. 2015). To reduce costs with restoration programs, modeling studies considering the potential of natural regeneration in the biome can help to design the best cost-effective planning proposals (Rezende et al. 2015).

These PES programs can promote adaptation to climate change. The potential of using ecosystem-based options to complement the multiple strategies present in the Brazilian National Adaptation Plan will be critical for the maintenance of ES (Scarano 2017; see also Vale et al. 2021, Chap. 12). Therefore, ES accountability studies focusing on the implementation of ecosystem-based adaptation strategies in the biome is a prominent field of research. ES accountability is also key to the private sector, in special when considering the potential to incorporate the contribution of private areas to ES provisioning (Kennedy et al. 2016). The sector can be more than a source of funding for project implementation, but develop new opportunities for biodiversity conservation and economic activities.

There are multiple alternatives to conserve and enhance ES in the AF that include policy, methodological, social, and economic components (Fig. 16.3) and communication between the multiple sectors will be determinant to ensure the full implementation of such strategies (Pires et al. 2018). It is clear that several actions will be required and they should be fitted considering the specific context for project implementation and opportunities to promote multiple benefits, such as adaptation to climate change. Agroforestry is also an alternative to maximize the provision of ecosystem services and it can facilitate restoration processes by incorporating proper methods during its implementation (Turetta et al. 2016).

## 16.6 Conclusion

Despite the severe historical reduction of vegetation cover, the AF remains a key provider of ecosystem services, often related to food, carbon, water, and cultural services. Thus, AF is a global biodiversity hotspot but also a hotspot for several ES delivered worldwide. There is some optimism regarding ongoing initiatives of eco-

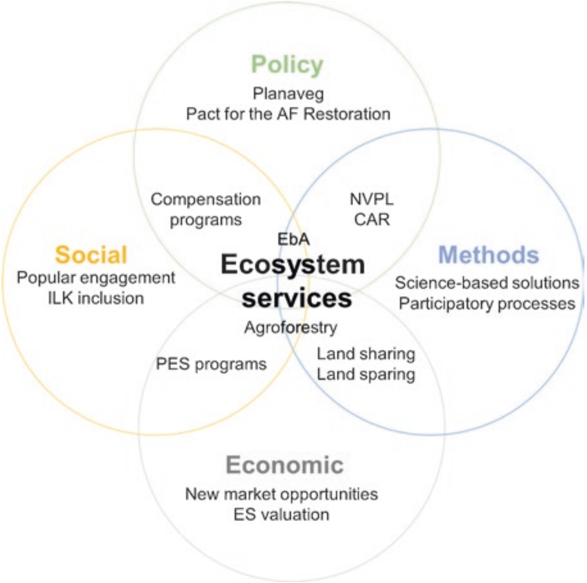

**Fig. 16.3** Tools for ensuring ecosystem services in the Atlantic Forest. The biome has the potential to integrate multiple solutions to keep its ability to provide ecosystem services and human wellbeing by integrating social (popular engagement, indigenous and local knowledge (ILK) inclusion), policy (National Policy for Native Vegetation Restoration (Planaveg), Pact for the AF Restoration), methodological (science-based solutions and participatory processes) and economic strategies (ES valuation and new market opportunities). Several tools are in the interface between these multiple components, including the Native Vegetation Protection Law (NVPL) and the Rural Environmental Register (CAR); ecosystem-based adaptation (EbA); the Payment for Environmental Services (PES) and compensation programs; agroforestry and land sharing and land sparing methods. The key component that will catalyze this integration of all strategies is the communication between these several actors

system restoration taken in parallel to the legislation that fosters conservation of vegetation remnants. For instance, it has been suggested that the AF can actually become a hopespot by implementing policies that promote human wellbeing while protecting nature and fostering climate change adaptation (Scarano and Ceotto 2015; Rezende et al. 2018b). The evidence reviewed here shares the vision that the AF can move out from the biodiversity hotspot status and become a model that demonstrates the feasibility of reaching sustainability through restoration and other practical measures. For that purpose, ES remains as a key operational concept, since it directly links the biodiversity component with people. Nevertheless, given the pressing global changes, time is of the essence for successful local transformations and to upscale solutions to the biome level.

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## References

Adams C, Seroa da Motta R, Ortiz RA, Reid J, Ebersbach Aznar C, de Almeida Sinisgalli PA (2008) The use of contingent valuation for evaluating protected areas in the developing world: economic valuation of Morro do Diabo State Park, Atlantic Rainforest, São Paulo State (Brazil). Ecol Econ 66:359–370

Alarcon GG, Ayanu Y, Fantini AC, Farley J, Filho AS, Koellner T (2015) Weakening the Brazilian legislation for forest conservation has severe impacts for ecosystem services in the Atlantic Southern Forest. Land Use Policy 47:1–11

Aleixo KP, de Faria LB, Groppo M, do Castro MMN, da Silva CI (2014) Spatiotemporal distribution of floral resources in a Brazilian city: implications for the maintenance of pollinators, especially bees. Urban For Urban Green 13:689–696

Alexander S, Aronson J, Whaley O, Lamb D (2016) The relationship between ecological restoration and the ecosystem services concept. Ecol Soc 21:art34

Alves-Pinto HN, Latawiec AE, Strassburg BBN, Barros FSM, Sansevero JBB, Iribarrem A, Crouzeilles R, Lemgruber L, Rangel MC, Silva ACP (2017) Reconciling rural development and ecological restoration: strategies and policy recommendations for the Brazilian Atlantic Forest. Land Use Policy 60:419–426

Amazonas NT, Martinelli LA, Piccolo M d C, Rodrigues RR (2011) Nitrogen dynamics during ecosystem development in tropical forest restoration. For Ecol Manag 262:1551–1557

Audino LD, Murphy SJ, Zambaldi L, Louzada J, Comita LS (2017) Drivers of community assembly in tropical forest restoration sites: role of local environment, landscape, and space. Ecol Appl 27:1731–1745

Banks-Leite C, Pardini R, Tambosi LR, Pearse WD, Bueno AA, Bruscagin RT, Condez TH, Dixo M, Igari AT, Martensen AC, Metzger JP (2014) Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot. Science 345:1041–1045

Batistella M, Bolfe EL, De Victoria DC, De Custodio DO, Da Silva GBS, Drucker DP (2012) SOMABRASIL: Sistema de Observação e Monitoramento da Agricultura no Brasil. Comunicado Técnico 29. Embrapa, Campinas-SP

Begotti RA, dos Pacífico ES, de Ferraz SFB, Galetti M (2018) Landscape context of plantation forests in the conservation of tropical mammals. J Nat Conserv 41:97–105

Bennett EM, Cramer W, Begossi A, Cundill G, Díaz S, Egoh BN, Geijzendorffer IR, Krug CB, Lavorel S, Lazos E, Lebel L, Martín-López B, Meyfroidt P, Mooney HA, Nel JL, Pascual U, Payet K, Harguindeguy NP, Peterson GD, Prieur-Richard AH, Reyers B, Roebeling P, Seppelt R, Solan M, Tschakert P, Tscharntke T, Turner BL, Verburg PH, Viglizzo EF, White PCL, Woodward G (2015) Linking biodiversity, ecosystem services, and human well-being: three challenges for designing research for sustainability. Curr Opin Environ Sustain 14:76–85

Bernard E, Penna LAO, Araújo E (2014) Downgrading, downsizing, Degazettement, and reclassification of protected areas in Brazil. Conserv Biol 28:939–950

Blangy S, Mehta H (2006) Ecotourism and ecological restoration. J Nat Conserv 14:233–236

BPBES (2019) 1° Diagnóstico Brasileiro de Biodiversidade e Serviços Ecossistêmicos. Joly CA; Scarano FR; Seixas CS; Metzger JP; Ometto JP; Bustamante MMC; Padgurschi MCG; Pires APF; Castro PFD; Gadda T; Toledo P (eds.) Editora Cubo, São Carlos pp.178

Braat LC, de Groot R (2012) The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosyst Serv 1:4–15

Brancalion PHS, Melo FPL, Tabarelli M, Rodrigues RR (2013) Biodiversity persistence in highly human-modified tropical landscapes depends on ecological restoration. Trop Conserv Sci 6:705–710

Brancalion PHS, Cardozo IV, Camatta A, Aronson J, Rodrigues RR (2014) Cultural ecosystem services and popular perceptions of the benefits of an ecological restoration project in the Brazilian Atlantic Forest. Restor Ecol 22:65–71

Brancalion PHS, Garcia LC, Loyola R, Rodrigues RR, Pillar VD, Lewinsohn TM (2016) A critical analysis of the native vegetation protection law of Brazil (2012): updates and ongoing initiatives. Natureza e Conservacao 14:1–16

Buainain AM, Alves E, Silveira JM, Navarro Z (2014) O mundo rural no Brasil do século 21: a formação de um novo padrão agrário e agrícola. Embrapa, Brasília

Bufalo FS, Galetti M, Culot L (2016) Seed dispersal by Primates and implications for the conservation of a biodiversity hotspot, the Atlantic Forest of South America. Int J Primatol 37:333–349

Bullock JM, Aronson J, Newton AC, Pywell RF, Rey-Benayas JM (2011) Restoration of ecosystem services and biodiversity: conflicts and opportunities. Trends Ecol Evol 26:541–549

Bunge M (2003) Emergence and convergence: qualitative novelty and the unity of knowledge. University of Toronto Press, Toronto

Campos FS, Lourenço-de-Moraes R, Llorente GA, Solé M (2017) Cost-effective conservation of amphibian ecology and evolution. Sci Adv 3:e1602929

Cao V, Margni M, Favis BD, Deschênes L (2015) Aggregated indicator to assess land use impacts in life cycle assessment (LCA) based on the economic value of ecosystem services. J Clean Prod 94:56–66

Chan KMA, Satterfield T, Goldstein J (2012) Rethinking ecosystem services to better address and navigate cultural values. Ecol Econ 74:8–18

Chazdon RL, Broadbent EN, Rozendaal DMA, Bongers F, Zambrano AMA, Aide TM, Balvanera P, Becknell JM, Boukili V, Brancalion PHS, Craven D, Almeida-Cortez JS, Cabral GAL, de Jong B, Denslow JS, Dent DH, DeWalt SJ, Dupuy JM, Duran SM, Espirito-Santo MM, Fandino MC, Cesar RG, Hall JS, Hernandez-Stefanoni JL, Jakovac CC, Junqueira AB, Kennard D, Letcher SG, Lohbeck M, Martinez-Ramos M, Massoca P, Meave JA, Mesquita R, Mora F, Munoz R, Muscarella R, Nunes YRF, Ochoa-Gaona S, Orihuela-Belmonte E, Pena-Claros M, Perez-Garcia EA, Piotto D, Powers JS, Rodriguez-Velazquez J, Romero-Perez IE, Ruiz J, Saldarriaga JG, Sanchez-Azofeifa A, Schwartz NB, Steininger MK, Swenson NG, Uriarte M, van Breugel M, van der Wal H, Veloso MDM, Vester H, Vieira ICG, Bentos TV, Williamson GB, Poorter L (2016) Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. Sci Adv 2:e1501639

Correia RA, Jepson PR, Malhado ACM, Ladle RJ (2016) Familiarity breeds content: assessing bird species popularity with culturomics. PeerJ 4:e1728

Costanza R, Arge R, De Groot R, Farberk S, Grasso M, Hannon B, Limburg K, Naeem S, Neill RVO, Paruelo J, Raskin RG, Suttonkk P (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–260

Costanza R, de Groot R, Braat L, Kubiszewski I, Fioramonti L, Sutton P, Farber S, Grasso M (2017) Twenty years of ecosystem services: how far have we come and how far do we still need to go? Ecosyst Serv 28:1–16

Crouzat E, Arpin I, Brunet L, Colloff MJ, Turkelboom F, Lavorel S (2018) Researchers must be aware of their roles at the interface of ecosystem services science and policy. Ambio 47:97–105

d'Albertas F, Costa K, Romitelli I, Barbosa JM, Vieira SA, Metzger JP (2018) Lack of evidence of edge age and additive edge effects on carbon stocks in a tropical forest. For Ecol Manag 407:57–65

Daily GC (2000) The value of nature and the nature of value. Science 289:395–396

de Groot RS (1987) Environmental functions as a unifying concept for ecology and economics. Environmentalist 7:105–109

de Groot R, Sukhdev P, Gough M (2018) Biodiversity: sparring makes us strong. Nature 561:309–309

De Marco P, Coelho FM (2004) Services performed by the ecosystem: forest remnants influence agricultural cultures' pollination and production. Biodivers Conserv 13:1245–1255

de Souza HN, de Graaff J, Pulleman MM (2012) Strategies and economics of farming systems with coffee in the Atlantic Rainforest Biome. Agrofor Syst 84:227–242 de Souza SEXF, Vidal E, Chagas G d F, Elgar AT, Brancalion PHS (2016) Ecological outcomes and livelihood benefits of community-managed agroforests and second growth forests in Southeast Brazil. Biotropica 48:868–881

Díaz S, Demissew S, Carabias J, Joly C, Lonsdale M, Ash N, Larigauderie A, Adhikari JR, Arico S, Báldi A, Bartuska A, Baste IA, Bilgin A, Brondizio E, Chan KMA, Figueroa VE, Duraiappah A, Fischer M, Hill R, Koetz T, Leadley P, Lyver P, Mace GM, Martin-Lopez B, Okumura M, Pacheco D, Pascual U, Pérez ES, Reyers B, Roth E, Saito O, Scholes RJ, Sharma N, Tallis H, Thaman R, Watson R, Yahara T, Hamid ZA, Akosim C, Al-Hafedh Y, Allahverdiyev R, Amankwah E, Asah TS, Asfaw Z, Bartus G, Brooks AL, Caillaux J, Dalle G, Darnaedi D, Driver A, Erpul G, Escobar-Eyzaguirre P, Failler P, Fouda AMM, Fu B, Gundimeda H, Hashimoto S, Homer F, Lavorel S, Lichtenstein G, Mala WA, Mandivenyi W, Matczak P, Mbizvo C, Mehrdadi M, Metzger JP, Mikissa JB, Moller H, Mooney HA, Mumby P, Nagendra H, Nesshover C, Oteng-Yeboah AA, Pataki G, Roué M, Rubis J, Schultz M, Smith P, Sumaila R, Takeuchi K, Thomas S, Verma M, Yeo-Chang Y, Zlatanova D (2015) The IPBES conceptual framework – connecting nature and people. Curr Opin Environ Sustain 14:1–16

Díaz S, Pascual U, Stenseke M, Martín-López B, Watson RT, Molnár Z, Hill R, Chan KMA, Baste IA, Brauman KA, Polasky S, Church A, Lonsdale M, Larigauderie A, Leadley PW, van Oudenhoven APE, van der Plaat F, Schröter M, Lavorel S, Aumeeruddy-Thomas Y, Bukvareva E, Davies K, Demissew S, Erpul G, Failler P, Guerra CA, Hewitt CL, Keune H, Lindley S, Shirayama Y (2018) Assessing nature's contributions to people. Science 359:270–272

Duarte GT, Ribeiro MC, Paglia AP (2016) Ecosystem services modeling as a tool for defining priority areas for conservation. PLoS One 11:e0154573

Ehrlich PR, Ehrlich A (1981). Extinction: the causes and consequences of the disappearance of species. Page (R. House, Ed.). New York

Ehrlich PR, Mooney HA (1983) Extinction, substitution, ecosystem services. BioScience 33:248–254

Fernandes GW, Goulart FF, Ranieri BD, Coelho MS, Dales K, Boesche N, Bustamante M, Carvalho FA, Carvalho DC, Dirzo R, Fernandes S, Galetti PM, Millan VEG, Mielke C, Ramirez JL, Neves A, Rogass C, Ribeiro SP, Scariot A, Soares-Filho B (2016) Deep into the mud: ecological and socio-economic impacts of the dam breach in Mariana, Brazil. Natureza & Conservação 14:35–45

Ferraro PJ, Hanauer MM, Miteva DA, Nelson JL, Pattanayak SK, Nolte C, Sims KRE (2015) Estimating the impacts of conservation on ecosystem services and poverty by integrating modeling and evaluation. Proc Natl Acad Sci 112:7420–7425

Filoso S, do Carmo JB, Mardegan SF, Lins SRM, Gomes TF, Martinelli LA (2015) Reassessing the environmental impacts of sugarcane ethanol production in Brazil to help meet sustainability goals. Renew Sust Energ Rev 52:1847–1856

Filoso S, Bezerra MO, Weiss KCB, Palmer MA (2017) Impacts of forest restoration on water yield: a systematic review. PLoS One 12:e0183210

Fonseca MM, Lima E, Lemos F, Venzon M, Janssen A (2017) Non-crop plant to attract and conserve an aphid predator (Coleoptera: Coccinellidae) in tomato. Biol Control 115:129–134

Förster J, Barkmann J, Fricke R, Hotes S, Kleyer M, Kobbe S, Kübler D, Rumbaur C, Siegmund-Schultze M, Seppelt R, Settele J, Spangenberg JH, Tekken V, Václavík T, Wittmer H (2015) Assessing ecosystem services for informing land-use decisions: a problem-oriented approach. Ecol Soc 20:art31

França JUB, Lamano-Ferreira AP d N, Ruiz MS, Quaresma CC, Kniess CT, Ramos HR, Ferreira ML (2016) Ecological knowledge about protected areas in the east zone of São Paulo, SP: implications for sustainability in urban area. HOLOS 3:174

Friess DA, Webb EL (2014) Variability in mangrove change estimates and implications for the assessment of ecosystem service provision. Glob Ecol Biogeogr 23:715–725

Galetti M, Brocardo CR, Begotti RA, Hortenci L, Rocha-Mendes F, Bernardo CSS, Bueno RS, Nobre R, Bovendorp RS, Marques RM, Meirelles F, Gobbo SK, Beca G, Schmaedecke G, Siqueira T (2017) Defaunation and biomass collapse of mammals in the largest Atlantic forest remnant. Anim Conserv 20:270–281

Gardner TA, Hernández MIM, Barlow J, Peres CA (2007) Understanding the biodiversity consequences of habitat change: the value of secondary and plantation forests for neotropical dung beetles. J Appl Ecol 45:883–893

Gaston KJ, Jackson SF, Cantú-Salazar L, Cruz-Piñón G (2008) The ecological performance of protected areas. Annu Rev Ecol Evol Syst 39:93–113

Giannini TC, Tambosi LR, Acosta AL, Jaffé R, Saraiva AM, Imperatriz-Fonseca VL, Metzger JP (2015) Safeguarding ecosystem services: a methodological framework to buffer the joint effect of habitat configuration and climate change. PLoS One 10:e0129225

Gómez-Baggethun E, de Groot R, Lomas PL, Montes C (2010) The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecol Econ 69:1209–1218

Gomiero T, Paoletti MG, Pimentel D (2010) Biofuels: efficiency, ethics, and limits to human appropriation of ecosystem services. J Agric Environ Ethics 23:403–434

Güneralp B, Seto KC (2013) Futures of global urban expansion: uncertainties and implications for biodiversity conservation. Environ Res Lett 8

Helliwell DR (1969) Valuation of wildlife resources. Reg Stud 3:41–47

Hengl T, De Jesus JM, MacMillan RA, Batjes NH, Heuvelink GBM, Ribeiro E, Samuel-Rosa A, Kempen B, Leenaars JGB, Walsh MG, Gonzalez MR (2014) SoilGrids1km – global soil information based on automated mapping. PLoS One 9

Hijmans RJ, Cameron SE, Parra JL, Jones PG, Jarvis A (2005) Very high resolution interpolated climate surfaces for global land areas. Int J Climatol 25:1965–1978

Hipólito J, Boscolo D, Viana BF (2018) Landscape and crop management strategies to conserve pollination services and increase yields in tropical coffee farms. Agric Ecosyst Environ 256:218–225

Instituto Saúde e Sustentabilidade (2018) Um minuto de ar limpo

Instituto Trata Brasil (2018) Benefícios econômicos e sociais da expansão do saneamento no Brasil

Ishihara H (2018) Relational values from a cultural valuation perspective: how can sociology contribute to the evaluation of ecosystem services? Curr Opin Environ Sustain 35:61–68

Jenkins CN, Alves MAS, Uezu A, Vale MM (2015) Patterns of Vertebrate Diversity and Protection in Brazil. PLOS ONE 10(12):e0145064.

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

Jordani MX, Hasui É, da Silva VX (2015) Natural enemies depend on remnant habitat size in agricultural landscapes. J For Res 26:469–477

Kennedy CM, Hawthorne PL, Miteva DA, Baumgarten L, Sochi K, Matsumoto M, Evans JS, Polasky S, Hamel P, Vieira EM, Develey PF, Sekercioglu CH, Davidson AD, Uhlhorn EM, Kiesecker J (2016) Optimizing land use decision-making to sustain Brazilian agricultural profits, biodiversity and ecosystem services. Biol Conserv 204:221–230

Kremer LP, da Rocha RM (2016) The biotic resistance role of fish predation in fouling communities. Biol Invasions 18:3223–3237

Lacerda AEB (2016) Conservation strategies for Araucaria Forests in Southern Brazil: assessing current and alternative approaches. Biotropica 48:537–544

Lamb D (2005) Restoration of degraded tropical forest landscapes. Science 310:1628–1632

Latawiec AE, Strassburg BBN, Brancalion PHS, Rodrigues RR, Gardner T (2015) Creating space for large-scale restoration in tropical agricultural landscapes. Front Ecol Environ 13:211–218

Librán-Embid F, De Coster G, Metzger JP (2017) Effects of bird and bat exclusion on coffee pest control at multiple spatial scales. Landsc Ecol 32:1907–1920

Lyra A, Tavares P, Chou SC, Sueiro G, Dereczynski C, Sondermann M, Silva A, Marengo J, Giarolla A (2017) Climate change projections over three metropolitan regions in Southeast Brazil using the non-hydrostatic Eta regional climate model at 5-km resolution. Theor Appl Climatol:1–20

Maes J, Burkhard B, Geneletti D (2018) Ecosystem services are inclusive and deliver multiple values. A comment on the concept of nature's contributions to people. One Ecosyst 3:e24720

Magnago LFS, Magrach A, Laurance WF, Martins SV, Meira-Neto JAA, Simonelli M, Edwards DP (2015) Would protecting tropical forest fragments provide carbon and biodiversity cobenefits under REDD+? Glob Chang Biol 21:3455–3468

Masood E (2018) The battle for the soul of biodiversity. Nature 560:423–425

MEA (2005) Millennium ecosystem assessment: ecosystems and human well-being, Washington, DC

Medeiros R, Young CEF, Pavese HB, Araújo FFS (2011) Contribuição das Unidades de Conservação Brasileira para a Economia Nacional: Sumário Executivo. UNEP-WCMC. 44 p. Brasília

MMA (2015) Shapefile of conservation units of Brazil

Molin PG, Gergel SE, Silvio BSS (2017) Spatial determinants of Atlantic Forest loss and recovery in Brazil. Landsc Ecol 32:857–870

Moreira EF, Boscolo D, Viana BF (2015) Spatial heterogeneity regulates plant-pollinator networks across multiple landscape scales. PLoS One 10:e0123628

Naidoo R, Balmford A, Costanza R, Fisher B, Green RE, Lehner B, Malcolm TR, Ricketts TH (2008) Global mapping of ecosystem services and conservation priorities. Proc Natl Acad Sci 105:9495–9500

Nobre CA, Marengo JA, Seluchi ME, Cuartas LA, Alves LM (2016) Some characteristics and impacts of the drought and water crisis in Southeastern Brazil during 2014 and 2015. J Water Res Protect 08:252–262

Oliveira-Silva LRB, Campêlo AC, Lima IMS, Araújo ACL, Bezerra BM, Souza-Alves JP (2018) Can a non-native primate be a potential seed disperser? A case study on Saimiri sciureus in Pernambuco state, Brazil. Folia Primatol 89:138–149

Pascual U, Balvanera P, Díaz S, Pataki G, Roth E, Stenseke M, Watson RT, Başak Dessane E, Islar M, Kelemen E, Maris V, Quaas M, Subramanian SM, Wittmer H, Adlan A, Ahn SE, Al-Hafedh YS, Amankwah E, Asah ST, Berry P, Bilgin A, Breslow SJ, Bullock C, Cáceres D, Daly-Hassen H, Figueroa E, Golden CD, Gómez-Baggethun E, González-Jiménez D, Houdet J, Keune H, Kumar R, Ma K, May PH, Mead A, O'Farrell P, Pandit R, Pengue W, Pichis-Madruga R, Popa F, Preston S, Pacheco-Balanza D, Saarikoski H, Strassburg BB, van den Belt M, Verma M, Wickson F, Yagi N (2017) Valuing nature's contributions to people: the IPBES approach. Curr Opin Environ Sustain 26–27:7–16

Peterson GD, Harmáčková ZV, Meacham M, Queiroz C, Jiménez-Aceituno A, Kuiper JJ, Malmborg K, Sitas N, Bennett EM (2018) Welcoming different perspectives in IPBES: "nature's contributions to people" and "ecosystem services". Ecol Soc 23(1):39

Petz K, Alkemade R, Bakkenes M, Schulp CJE, van der Velde M, Leemans R (2014) Mapping and modelling trade-offs and synergies between grazing intensity and ecosystem services in rangelands using global-scale datasets and models. Glob Environ Chang 29:223–234

Pires APF, Rezende CL, Assad ED, Loyola R, Scarano FR (2017) Forest restoration can increase the Rio Doce watershed resilience. Perspect Ecol Conserv 15:187–193

Pires APF, Amaral AG, Padgurschi MCG, Joly CA, Scarano FR (2018) Biodiversity research still falls short of creating links with ecosystem services and human well-being in a global hotspot. Ecosyst Serv 34:68–73

Pires APF, Padgurschi MCG, de Castro PD, Scarano FR, Strassburg B, Joly CA, Watson RT, de Groot R (2020) Ecosystem services or nature's contributions? Reasons behind different interpretations in Latin America. Ecosyst Serv 42:101070

Raudsepp-Hearne C, Peterson GD, Bennett EM (2010) Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc Natl Acad Sci 107:5242–5247

Raymundo D, Prado-Junior J, de Oliveira-Neto NE, Santana LD, do Vale VS, Jacobson TB, de Oliveira PEAM, Carvalho FA (2018) Persistence of Coffea arabica and its relationship with the structure, species diversity and composition of a secondary forest in Brazil. PLoS One 13:e0194032

Rey Benayas JM, Newton AC, Diaz A, Bullock JM (2009) Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. Science 325:1121–1124

Rezende CL, Uezu A, Scarano FR, Araujo DSD (2015) Atlantic Forest spontaneous regeneration at landscape scale. Biodivers Conserv 24:2255–2272

Rezende CL, Fraga JS, Sessa JC, de Souza GVP, Assad ED, Scarano FR, Linhares C, Stingel J, Cabral J, Vinagre G, De Souza P, Delgado E, Rubio F, Rezende CL, Fraga JS, Sessa JC, de Souza GVP, Assad ED, Scarano FR (2018a) Land use policy as a driver for climate change adaptation: a case in the domain of the Brazilian Atlantic forest. Land Use Policy 72:563–569

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Tabarelli M, Fonseca GA, Mittermeier RA (2018b) From hotspot to hopespot: an opportunity for the Brazilian Atlantic. Perspect Ecol Conserv 16:208–214

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

Ribeiro MC, Martensen AC, Metzger JP, Tabarelli M, Scarano FR, Fortin MJ (2011) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. pp. 405–434. In: Zachos FE, Habel JC (eds) Biodiversity hotspots. Springer, Heidelberg

Richards RC, Rerolle J, Aronson J, Pereira PH, Gonçalves H, Brancalion PHS (2015) Governing a pioneer program on payment for watershed services: stakeholder involvement, legal frameworks and early lessons from the Atlantic forest of Brazil. Ecosyst Serv 16:23–32

Richards RC, Kennedy CJ, Lovejoy TE, Brancalion PHS (2017) Considering farmer land use decisions in efforts to ‘scale up’ payments for watershed services. Ecosyst Serv 23:238–247

Rieb JT, Chaplin-Kramer R, Daily GC, Armsworth PR, Böhning-Gaese K, Bonn A, Cumming GS, Eigenbrod F, Grimm V, Jackson BM, Marques A, Pattanayak SK, Pereira HM, Peterson GD, Ricketts TH, Robinson BE, Schröter M, Schulte LA, Seppelt R, Turner MG, Bennett EM (2017) When, where, and how nature matters for ecosystem services: challenges for the next generation of ecosystem service models. Bioscience 67:820–833

Robinson SJB, van den Berg E, Meirelles GS, Ostle N (2015) Factors influencing early secondary succession and ecosystem carbon stocks in Brazilian Atlantic Forest. Biodivers Conserv 24:2273–2291

Rocha-Santos L, Pessoa MS, Cassano CR, Talora DC, Orihuela RLL, Mariano-Neto E, Morante-Filho JC, Faria D, Cazetta E (2016) The shrinkage of a forest: landscape-scale deforestation leading to overall changes in local forest structure. Biol Conserv 196:1–9

Rockström J, Steffen W, Noone K, Persson Å, Chapin FS, Lambin EF, Lenton TM, Scheffer M, Folke C, Schellnhuber HJ, Nykvist B, de Wit CA, Hughes T, van der Leeuw S, Rodhe H, Sörlin S, Snyder PK, Costanza R, Svedin U, Falkenmark M, Karlberg L, Corell RW, Fabry VJ, Hansen J, Walker B, Liverman D, Richardson K, Crutzen P, Foley JA, Revolution I (2009) A safe operating space for humanity. Nature 461:472–475

Ruesch, A, Gibbs, H (2008) IPCC Tier-1 Global Biomass Carbon Map for the Year 2000. Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, http://cdiac.ess-dive.lbl.gov

Ruiz-Mallén I, Schunko C, Corbera E, Rös M, Reyes-García V (2015) Meanings, drivers, and motivations for community-based conservation in Latin America. Ecol Soc 20:art33

Sacchi LV, Powell PA, Gasparri NI, Grau R (2017) Air quality loss in urban centers of the Argentinean Dry Chaco: wind and dust control as two scientifically neglected ecosystem services. Ecosyst Serv 24:234–240

Santos HV, Scotti MR (2018) Riparian reforestation with a single exotic species restores soil aggregation and porosity but not humic substances. Soil Use Manag 34:124–133

Sarmento R, Alves-Costa CP, Ayub A, Mello MAR (2014) Partitioning of seed dispersal services between birds and bats in a fragment of the Brazilian Atlantic Forest. Zoologia (Curitiba) 31:245–255

Saturni FT, Jaffé R, Metzger JP (2016) Landscape structure influences bee community and coffee pollination at different spatial scales. Agric Ecosyst Environ 235:1–12

Scarano FR (2019a) Biodiversity Sector: Risks of Temperature Increase to Biodiversity and Ecosystems. In: Nobre C., Marengo J., Soares W. (eds) Climate Change Risks in Brazil. Springer, Cham

Scarano FR (2019b) The emergence of sustainability. pp. 51–71. In: Wegner L, Lüttge U (eds) Emergence and modularity in life sciences. Springer, Cham

Scarano FR (2017) Ecosystem-based adaptation to climate change: concept, scalability and a role for conservation science. Perspect Ecol Conserv 15:65–73

Scarano FR, Ceotto P (2015) Brazilian Atlantic forest: impact, vulnerability and adaptation to climate change. Biodivers Conserv 24:2319–2331

Shimamoto CY, Botosso PC, Marques MCM (2014) How much carbon is sequestered during the restoration of tropical forests? Estimates from tree species in the Brazilian Atlantic Forest. For Ecol Manag 329:1–9

Shimamoto CY, Padial AA, Da Rosa CM, Marques MCM (2018) Restoration of ecosystem services in tropical forests: a global meta-analysis. PLoS One 13:1–16

Soares-Filho B, Rajão R, Merry F, Rodrigues H, Davis J, Lima L, Macedo M, Coe M, Carneiro A, Santiago L (2016) Brazil's market for trading forest certificates. PLoS One 11:e0152311

Sone JS, Gesualdo GC, Zamboni PAP, Vieira NOM, Mattos TS, Carvalho GA, Rodrigues DBB, Sobrinho TA, Oliveira PTS (2019) Water provisioning improvement through payment for ecosystem services. Sci Total Environ 655:1197–1206

SOSMA (2014) Atlas dos remanescentes florestais da Mata Atlântica período 2013–2014

Sovacool BK (2011) Using ecosystem valuation to protect the Atlantic rainforest: the case of the oasis project. Soc Nat Resour 24:1096–1104

Strassburg BBN, Latawiec AE, Barioni LG, Nobre CA, da Silva VP, Valentim JF, Vianna M, Assad ED (2014) When enough should be enough: improving the use of current agricultural lands could meet production demands and spare natural habitats in Brazil. Glob Environ Chang 28:84–97

Strassburg BBN, Barros FSM, Crouzeilles R, Iribarrem A, dos Santos JS, Silva D, Sansevero JBB, Alves-Pinto HN, Feltran-Barbieri R, Latawiec AE (2016) The role of natural regeneration to ecosystem services provision and habitat availability: a case study in the Brazilian Atlantic Forest. Biotropica 48:890–899

Strassburg BBN, Beyer HL, Crouzeilles R, Iribarrem A, Barros F, de Siqueira MF, Sánchez-Tapia A, Balmford A, Sansevero JBB, Brancalion PHS, Broadbent EN, Chazdon RL, Filho AO, Gardner TA, Gordon A, Latawiec A, Loyola R, Metzger JP, Mills M, Possingham HP, Rodrigues RR, de Scaramuzza CAM, Scarano FR, Tambosi L, Uriarte M (2019) Strategic approaches to restoring ecosystems can triple conservation gains and halve costs. Nat Ecol Evol 3:62–70

Tabarelli M, Aguiar AV, Ribeiro MC, Metzger JP (2012) The conversion of the Atlantic Forest in anthropogenic landscapes: lessons for the conservation of biological diversity of tropical forests. Interciência 37:88–92

Taffarello D, Samprogna Mohor G, do Carmo Calijuri M, Mendiondo EM (2016) Field investigations of the 2013–14 drought through quali-quantitative freshwater monitoring at the headwaters of the Cantareira System, Brazil. Water Int 41:776–800

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

Torres S, Moran E, Silva R (2017) Property rights and the soybean Revolution: shaping how China and Brazil are Telecoupled. Sustainability 9:954

Trabucco A, Zomer RJ (2010) Global Soil Water Balance Geospatial Database

Tscharntke T, Clough Y, Bhagwat SA, Buchori D, Faust H, Hertel D, Hölscher D, Juhrbandt J, Kessler M, Perfecto I, Scherber C, Schroth G, Veldkamp E, Wanger TC (2011) Multifunctional shade-tree management in tropical agroforestry landscapes – a review. J Appl Ecol 48:619–629

Turetta APD, Tonucci R, De Mattos LM, Amaro G, de Balieiro FC, Prado RB, de Souza HA, de Oliveira AP (2016) An approach to assess the potential of agroecosystems in providing environmental services. Pesq Agrop Brasileira 51:1051–1060

Vitule J, Bornatowski H, Freire C, Abilhoa V (2014) Extralimital introductions of Salminus brasiliensis (Cuvier, 1816) (Teleostei, Characidae) for sport fishing purposes: a growing challenge for the conservation of biodiversity in neotropical aquatic ecosystems. BioInvasions Records 3:291–296

Wolowski M, Agostini K, Rech AR, Varassin IG, Maués M, Freitas L, Carneiro LT, de Bueno RO, Consolaro H, Carvalheiro L, Saraiva AM, da Silva CI (2019) Relatório temático sobre polinização, polinizadores e produção de alimentos no Brasil. Editora Cubo, São Carlos

Young CEF, de Bakker LB (2014) Payments for ecosystem services from watershed protection: a methodological assessment of the oasis project in Brazil. Natureza a Conservacao 12:71–78

Zwiener VP, Padial AA, Marques MCM, Faleiro FV, Loyola R, Peterson AT (2017) Planning for conservation and restoration under climate and land use change in the Brazilian Atlantic Forest. Divers Distrib:1–12

# Chapter 17
# Changing the Agriculture Paradigm in the Brazilian Atlantic Forest: The Importance of Agroforestry

**Fernanda Tubenchlak, Carolina Giudice Badari, Guilherme de Freitas Strauch, and Luiz Fernando Duarte de Moraes**

**Abstract** Agricultural expansion has been the main driver of land-use change in the Atlantic Forest. Nowadays, more than 60% of the biome is dedicated to mostly unsustainable agricultural land-uses; however, most people live in urban areas and overall production is low. Here, we discuss the impacts of agriculture in shaping the landscape and its potential to restore socioecological interactions. The focus is on agroforestry, which is broadly understood as the integration of trees in productive systems and landscapes. In the first part of the chapter, we review the concept of agroforestry, benefits, its relation to urgent global issues, its role in Forest Landscape Restoration and achieving the United Nations Sustainable Development Goals. In the second part of the chapter, we present examples in the Atlantic Forest and discuss the opportunities and challenges for upscaling the adoption of agroforestry and overcome the apparent dichotomy between production and conservation.

**Keywords** Agroecology · Eco-agriculture · Regenerative agriculture · Integrated landscape management · Forest landscape restoration

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## 17.1 Introduction

Historically, deforestation and habitat degradation were driven by anthropogenic activities such as logging, intensive agriculture and extensive cattle ranching. The Atlantic Forest was the first agricultural frontier of Brazil, and large areas have been converted for commodity plantations such as sugarcane, coffee, cotton, cocoa and most recently soybeans (Dean 1996; Campos 2011). The production of such crops has occurred in cycles of "climax and crisis" (Young 2006), comprising intensive land uses and natural resources exhaustion that, along with the territorial occupation model, have led to highly fragmented landscapes today (Ribeiro et al. 2011). After more than 500 years of exploitation, 62% of the biome is covered by agricultural land-uses (Mapbiomas 2019).

Nonetheless, farmland contraction has been observed over the last decades, especially in the 1975–1996 period (Barretto et al. 2013). The area dedicated to pastures has decreased from 51 to 40 million hectares in the 1985–2017 period, representing now 36% of the biome, while the size of the cattle herd remained relatively stable (LAPIG 2019). Part of this area has been converted to agriculture, which increased by 106%, occupying 18 million hectares in 2017 (Mapbiomas 2019). Despite the intensification trends, overall production is low, as most of the agricultural land is dedicated to extensive pastures, with low stocking rates compared to carrying capacity and notable degradation levels (Strassburg et al. 2014; LAPIG 2019).

Intensive monocultures represent most of the area dedicated to agriculture; however, small-scale family farming plays a major role in maintaining rural livelihoods and feeding the mostly urban population (Kageyama et al. 2013; IBGE 2017). The simplified agroecosystems that dominate farmlands of the biome deliver limited socioeconomic benefits and represent a matrix with low permeability, which hinders ecological flows and negatively impacts biodiversity and ecosystem services provision (Ricketts 2001; Perfecto and Vandermeer 2010). In turn, the outcomes are soil degradation, environmental contamination and water-related problems (pollution, droughts or floods, depending on the region) which adversely impacts yields and stability of agricultural production.

Nonetheless, from 1985 to 2015, almost 220 thousand hectares of secondary forests have grown following farmland contraction (SOS Mata Atlântica 2017). From an ecological perspective, regrowing forests may enhance the provision of ecosystem services (Chazdon et al. 2009b; Chazdon et al. 2016), but might not fulfil their potential socioecological benefits if their occurrence is coupled with unsustainable agricultural intensification and rural exodus. Nevertheless, agroforestry practices have been contributing to restore forests while improving livelihoods, suggesting that agriculture can be a driver of regeneration instead of a driver of degradation in human-dominated landscapes.

In this chapter, we synthesize the current knowledge on agroforestry and discuss its potential to restore socioecological interactions in the Atlantic Forest. First, we review agroforestry definitions, classifications and benefits. Then we debate the role of agroforestry in agroecological and integrated landscape management approaches, which are increasingly recognized as key strategies in Forest Landscape Restoration (FLR) and for achieving the United Nations Sustainable Development Goals (SDGs). In the second part of the chapter, we focus on agroforestry practices in the Atlantic Forest, with practical examples and recent studies that show their potential as part of the solutions for enhancing biodiversity and human well-being. Finally, we discuss opportunities and challenges for upscaling agroforestry in the biome.

## 17.2 Agroforestry

### *17.2.1 Definitions and Classifications*

The concept of agroforestry has evolved over the last four decades and various definitions have been proposed, from plot to landscape and policy-level concepts. The most cited definition states that agroforestry is "*a collective name for land-use systems and technologies where woody perennials (trees, shrubs, palms, bamboos etc.) are deliberately used on the same land-management units as agricultural crops and/or animals, in some form of spatial arrangement or temporal sequence*" (Nair 1993). As such, it encompasses a wide range of 'systems', 'sub-systems' and 'practices', but its main feature is the integration of trees in productive landscapes. It is important to emphasize that this integration is not new, agroforestry is a relatively recent name given to ancient practices developed by traditional communities all over the world, especially in the tropics (Posey 1985; Nair 1993; van Noordwijk 2019).

Following the above rationale, the Brazilian legislation defines agroforestry as "*land use systems in which woody perennials are managed in association with herbaceous plants, shrubs, trees, agricultural crops, forage species in the same management unit, according to spatial arrangement and temporal, with high species diversity and interactions between these components*" (MMA 2009). This definition adds and highlights the importance of biodiversity and complexity in agroforestry systems. However, agroforestry has also been defined more simply as "*a combination of agriculture and forestry, is land use that combines aspects of both, including the agricultural use of trees*" (Leakey 2012). This definition aims to overcome the segregation and between 'agriculture' and 'forestry', highlighting the necessity to understand and manage multifunctional landscapes as a *continuum* of functions, in order to align regulations and incentives for achieving the SDGs (van Noordwijk 2019).

Following the multiple understandings of what agroforestry is, different classifications have been proposed, based on structural, functional, socioeconomic and ecological characteristics; these different classifications are neither independent nor mutually exclusive (Nair 1985; May and Trovatto 2008). As land-use systems, the definition of boundaries is often uncertain and thus agroforestry can be understood at different scales, from the plot level to the landscape level. At the plot level, agroforestry systems include trees on the same patch of agricultural crops and/or in association with livestock. At broader scales, such as the property or the landscape

**Fig. 17.1** Multiple examples of agroforestry systems. Categorization is based on the nature of their components: arboreal, agricultural and animal. Traditionally this classification includes the upper three categories: agrisilvicultural, agrosilvipastoral and silvipastoral systems. The inclusion of the fourth category is based on the understanding that managed forest systems are part of the space-time *continuum* of agroforestry. (Source: Author's elaboration, based on Nair (1985), illustration by Raissa Theberge)

level, agroforestry can be understood as the integrated management of different 'sub-systems' (i.e. agricultural plots and forested areas). Moreover, as the structure and composition of agroforestry systems can vary over time, the same system can be categorized in different ways depending on the timing of evaluation.

The first categorization that can be made is based on the components of the system, according to their nature: 'agrisilvicultural systems' are composed of arboreal, shrub or palm species combined with short cycle agricultural species; 'silvipastoral systems' consist of a combination of trees, shrubs or palms with forage and animal species; 'agrosilvipastoral systems' combine the two previous ones (Nair 1985); and 'forest systems' are based on the arboreal component (planted or 'natural'), comprising primarily timber and/or non-timber forest products extraction (Fig. 17.1).

This last category includes forestry and forest management as part of the space-time *continuum* of agroforestry systems, understanding the role of humans in planting, maintaining and diversifying forests, which might result in the creation of anthropogenic landscapes (Clement et al. 2015; Levis et al. 2017).

Another proposal to classify agroforestry systems regards their design in time: (i) sequential, when annual crops and the introduction of trees succeed in time, as in shifting agriculture; (ii) simultaneous or concomitant, when there is spatial integration of annual and perennial crops, such as alley cropping of timber trees with annual crops; and (iii) complementary, when using trees in hedgerows or as windbreaks (May and Trovatto 2008; de Moraes et al. 2011).

Among the simultaneous agroforestry systems, two categories can be distinguished: (i) static agroforestry systems and (ii) successional (or dynamic) agroforestry systems. In the first one, management practices do not modify the composition nor the structure of the system, as for example in the systems of shaded coffee (*Coffea* spp), cacao (*Theobroma cacao*) or yerba mate (*Ilex paraguariensis*), where there are basically two strata: the cash crops occupy the lower stratum under a canopy of shade trees (which may represent one or more strata, depending on species composition). In the second category, management practices modify the systems' structure and lead to the formation of complex and multi-stratified systems that mimic the dynamics of ecological succession of native forests, where herbaceous and shrub species are gradually replaced over time, evolving into perennial biodiverse systems. These systems are also known as *regenerative agroforestry systems* (Amador 2003) and r*egenerative and analogous agroforestry systems* (Peneireiro 2003; de Souza and Piña-Rodrigues 2013).

This last denomination relates to the *analogue forestry* concept, which designs agroforestry systems that mimic native forests in terms of structure, functioning and biodiversity (Senanayake and Jack 1998; Dickinson 2014). According to the International Analog Forestry Network, analogous forests consist of an approach to ecological restoration, which uses natural forests as guides to create ecologically balanced and socioeconomically productive landscapes. Most recently, these systems have been referred to as *syntropic agroforestry,* in reference to Ernst Götsch's *Syntropic Farming*, one of the main references in the development and diffusion of successional agroforestry systems in Brazil (May and Trovatto 2008; Miccolis et al. 2016).

### *17.2.2 Agroforestry Benefits*

The scientific literature provides many examples of the multiple socioecological benefits of agroforestry, which are highly variable depending on the type of system evaluated, the geographical location, and the biophysical and socioeconomic contexts. Overall, it is widely recognized that agroforestry systems increase resource use efficiency, contribute to biodiversity conservation and enhance ecosystem services provision (Jose 2009; van Noordwijk 2019). Agroforestry is also recognized as an alternative for improving food security and nutrition, as it increases agrobiodiversity and food production, contributing to poverty alleviation and enhancing human well-being (Leakey 1996; Cacho et al. 2003; Jose 2009; Altieri et al. 2012; de Souza et al. 2016a).

From the biodiversity perspective, the conservation potential will depend on the type of agroforestry, composition and characteristics of species, the intensity of agricultural management, as well as characteristics of the landscape (Jose 2012). In order to be multifunctional (delivering multiple benefits), agroforestry systems require greater numbers of species, since different plant species influence different functions (Hector and Bagchi 2007; de Moraes et al. 2011; Sloan and Sayer 2015). Typically, forest biodiversity declines along a management intensification gradient, broadly reflecting the decline in floristic and structural diversity (Perfecto et al. 2003; Perfecto and Vandermeer 2008; Badari et al. 2020). Consequently, the retention or management of structurally and floristically complex habitats like agroforests and secondary forests can often ensure the persistence of some forest species in managed landscapes (Gardner et al. 2009).

Although some authors suggest that agroecosystems tend to conserve mainly generalist species, complex agroecosystems represent a more permeable matrix that contribute to landscape connectivity (Uezu et al. 2008; Perfecto and Vandermeer 2010; Badari et al. 2020). Multiple studies have demonstrated the importance of agroforestry systems for tropical avifauna (Bakermans et al. 2012) and mammal species, such as carnivores (Ferreira et al. 2018) and primates, including vulnerable and endangered species (Cullen et al. 2005; Oliveira et al. 2009). A review by Estrada et al. (2012) on the use of agroecosystems by primates in the tropics showed that 57 primate taxa used 38 types of agroecosystems as temporary or permanent habitats. Fifty-one per cent of the taxa recorded were classified as least concern in the IUCN Red List, but the rest were classified as endangered (20%), vulnerable (18%), near threatened (9%) or critically endangered (2%). These results show that common species are not the only ones occupying agroecosystems and highlight the role of the agricultural matrix in biodiversity conservation.

Ecosystems services provided by agroforestry systems vary over a range of spatial and temporal scales (Jose 2009). At the local level (i.e. plot and farm), these systems can increase net primary production, enhance pest control and pollination, conserve and regenerate soils (by combining of erosion control and maintenance or enhancement of soil fertility), enhancing productivity and supporting rural livelihoods. At the landscape level, agroforestry systems can improve air and water quality, mitigate extreme events such as floods and droughts, and increase habitat availability and connectivity for native species. At the global level, agroforestry makes an important contribution to climate change action, sequestrating carbon and increasing socioecological resilience (Jose 2009; Altieri and Nicholls 2013; Altieri et al. 2015; Zomer et al. 2016).

### *17.2.3 Agroforestry, Agroecology and Integrated Landscape Management*

Transitioning to sustainability requires increasing forest cover and managing multifunctional landscapes that balance environmental, economic, social and cultural objectives (Aronson and Alexander 2013; Mansourian et al. 2017). Considering that agroecosystems are the dominant ecosystems of the Anthropocene, improving their management is essential to address the urgent challenges of sustainable development (DeClerck et al. 2016). In this sense, the adoption of agroecological practices and integrated landscape management are increasingly recognized as central strategies for balancing biodiversity conservation and socioeconomic development (Sayer et al. 2013; Besseau et al. 2018)

Among the interventions directly linked to the (re)generation of ecological and socioeconomic benefits, agroforestry options are the most cited and studied (Lamb et al. 2005; de Souza et al. 2016a; Schulz and Schröder 2017; Latawiec et al. 2018). The integration of valuable trees (e.g. fodder, fuelwood, or timber) into agricultural systems can diversify and improve income generation while enhancing the provision of ecosystems services (FAO 2016; IUFRO 2017). Remote sensing data show that 43% of all agricultural land globally had at least 10% tree cover in 2010 (Zomer et al. 2016). However, despite their multiple benefits, trees constitute an underutilized resource in modern agriculture, mainly after agricultural modernization of the so-called ‘green revolution’ (Leakey 2019).

Nonetheless, agroforestry is more than just the presence of trees on agricultural areas, it should be “*considered as a dynamic, ecologically based, natural resource management system that, through the integration of trees in farm and rangeland, diversifies and sustain smallholder production for increased social, economic and environmental benefits*” (Leakey 1996). This understanding of agroforestry is aligned with the agroecological approach, which can be defined as the *“application of ecological principles and concepts in the design and management of sustainable agroecosystems”* (Gliessman 2001; Altieri 2002).

Agroecology started as a theoretical and methodological approach to study agricultural activity from an ecological perspective; however, it has developed by incorporating new theoretical and practical dimensions, including social, economic, ethical and political issues. Agroecology embraces the various factors that lie beyond the limits of the actual agricultural production, bringing together different disciplines in a systemic approach to understand agriculture and food systems (Hecht 2002). It also embraces traditional knowledge systems, understanding the ecological rationale that supports ancient agricultural practices (Toledo 1993; Guzmán Casado et al. 2000). More than a scientific discipline (or field of knowledge), the term is currently used to refer to ecological agricultural practices and also a social movement (Wezel and Soldat 2009).

Agroforestry and agroecology constitute key elements in integrated landscape management. ‘Landscape approaches’ have been increasingly adopted by scientists, policymakers and local stakeholders, who recognize the need to think beyond site-level interventions (i.e. protected areas and ecological restoration) and increase the multifunctionality of landscapes in order to conciliate biodiversity conservation, food production and enhance human well-being (Chazdon et al. 2009a; Perfecto and Vandermeer 2010; Sayer et al. 2013; Laestadius et al. 2015). A landscape approach can be defined as "*a long-term collaborative process bringing together diverse stakeholders aiming to achieve a balance between multiple and sometimes conflicting objectives in a landscape or seascape"* (Sayer et al. 2013). In other words, it is an integrated approach that considers more than the spatial dimension, but also the complex socioecological context of a site or system (defined case by case based on a meaningful planning unit) (Stanturf et al. 2014; Mansourian 2017). This approach underlies the international commitments to restore degraded land and is central for achieving the SDGs, as discussed below.

## Forest Landscape Restoration (FLR)

Large-scale restoration goals have been targeted at different levels, from global to local scales, including the Atlantic Forest (Laestadius et al. 2015; Suding et al. 2015; De Siqueira et al. 2021, chap. 18). The recognition that restoration needed to be expanded beyond small-scale plots and contribute to multifunctional landscapes led to the proposal of the concept of FLR, which stands for "*a planned process that aims to regain ecological integrity and enhance human well-being in deforested or degraded landscapes*" (Mansourian et al. 2005, 2017). This integrated vision of FLR expands the discussion of biodiversity and ecological functions and requires addressing economic, cultural and political issues (Latawiec et al. 2015; Mansourian et al. 2017; Reed et al. 2017).

From the original concept of FLR, Mansourian et al. (2017) stress three fundamental principles: a) intentionality (which excludes spontaneous natural regeneration); b) multifunctionality (social and ecological approaches) and c) scale (from local to regional landscape). Meaning more than simply expanding forest cover, FLR will succeed whenever there are targets clearly established, a participatory selection of areas to be restored (to address multiple interests), and the careful establishment of techniques to measure the restoration success at the landscape level (Mansourian et al. 2017; Troya and Kumar 2016). Those are essential conditions to reach a multifunctional landscape by FLR.

Hence, restoration should not be one more competitor for land. Rather than displacing productive activities to other areas, restoration should enhance and diversify agricultural production, contribute to food security and generate economic returns to farmers (Brancalion et al. 2012; Latawiec et al. 2015; Maier et al. 2018). Therefore, agroforestry systems have been recommended as a cost-alternative approach strategy for FLR, complementing other restoration approaches (e.g. sowing and planting of seedlings, assisted natural regeneration) (Vieira et al. 2009; Gliessman and Tittonell 2015; Chazdon and Uriarte 2016; Miccolis et al. 2016).

## Sustainable Development Goals (SDGs)

The benefits of agroforestry discussed above evidence their role in sustainable development strategies. According to FAO (2018), agroecology offers a unique approach to transform agri-food systems in a sustainable and inclusive way and is essential for implementing the 2030 Agenda and achieving the SDGs. The most obvious contributions of agroecology and agroforestry are towards goal 2: "*End hunger, achieve food security and improved nutrition and promote sustainable agriculture*"; more specifically to target 2.4: *"By 2030, ensure sustainable food production systems and implement resilient agricultural practices that increase productivity and production, that help maintain ecosystems, that strengthen capacity for adaptation to climate change, extreme weather, drought, flooding and other disasters and that progressively improve land and soil quality"*. In this sense there is ample evidence that agroecology and agroforestry are effective for fighting poverty (goal 1), as most people in vulnerable situations live in rural contexts.

The transition to agroecological agroecosystems contributes directly to goal 15: *"Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss"*, and to goal 13 *"Take urgent action to combat climate change and its impacts"*. The incorporation of trees on farms and other agroecological management practices (e.g. crop diversification, maintenance of local genetic varieties, organic soil management, animal integration) increase carbon sequestration and agroecosystem resilience, contributing simultaneously to mitigation and adaptation (Altieri and Nicholls 2013; Altieri et al. 2015; Zomer et al. 2016).

Beyond those, recent studies indicate that agroecology and agroforestry can help achieve at least 13 out of the 17 SDGs (Fig. 17.2). A meta-analysis by Farrely (2016) with 50 case studies from 22 African countries demonstrates agroecology's contribution towards achieving 10 SDGs. Similarly, a report by the Agroforestry Network reviews the evidence on the multiple benefits of agroforestry and its role to achieve nine SDGs (Agroforestry Network 2018). FAO (2018) highlights agroecology contributions to 10 SDGs and calls for scaling up and transform food and agricultural systems.

**Fig. 17.2** Agroecology and agroforestry can contribute to achieve at least 13 out of the 17 Sustainable Development Goals (SDGs). (Source: SDGs logos – Sustainable Development Knowledge Platform of United Nations (https://sustainabledevelopment.un.org/sdgs.))

The contribution of agroecology and agroforestry to goal 3 "*Ensure healthy lives and promote well-being for all at all ages*" is most linked to target 3.9: "*Reduce illnesses and death from hazardous chemicals and pollution"* as they reduce the use of chemical fertilizers and pesticides, which have negative effects on the surrounding environment and on human health (van Lexmond et al., 2015). The reduction in agrochemicals also relates to target 6.3 "*By 2030, improve water quality by reducing pollution, eliminating dumping and minimizing release of hazardous chemicals and materials, halving the proportion of untreated wastewater and substantially increasing recycling and safe reuse globally*". Furthermore, agroforestry can contribute to goal 6, specifically to target 6.6: "*By 2020, protect and restore water-related ecosystems, including mountains, forests, wetlands, rivers, aquifers and lakes*".

The contribution to goal 7 "*Ensure access to affordable, reliable, sustainable and modern energy for all*" is since agroforestry can reduce pressure on forests, providing fuelwood and timber for farmers at home. Similarly, contribution to goal 12 "*Ensure sustainable consumption and production patterns*" relates to target 12.2: "*By 2030, achieve the sustainable management and efficient use of natural resources*". Agroecological food systems also include alternative markets that are based on shorter supply chains and traceability, contributing to target 12.3: "*By 2030, halve per capita global food waste at the retail and consumer levels and reduce food losses along production and supply chains, including post-harvest losses*".

Additionally, agroforestry can contribute to goal 8 "*Promote sustained, inclusive and sustainable economic growth, full and productive employment and decent work for all*". Considering that agroforestry is innovative and labour-intensive compared to conventional agricultural practices, its adoption can help generate jobs and incomes, contributing to targets 8.2 and 8.3. In particular, the adoption of agroforestry contributes directly to target 8.9: "*By 2030, devise and implement policies to promote sustainable tourism that creates jobs and promotes local culture and products*". In this sense, agroecological management can help to maintain people in rural areas and even to stimulate migration from cities to those areas, contributing to goal 11 "*Make cities and human settlements inclusive, safe, resilient and sustainable*" (Tubenchlak 2018).

Contributions to goal 4 are based on the fact that agroecology and agroforestry projects can disseminate the "*knowledge and skills needed to promote sustainable development"* (target 4.7). In this context, agroecology can contribute to goals 5 and 10, as it gives priority to the most marginalized sectors of society: rural women, youth, family farmers and indigenous peoples, promoting empowerment and inclusion (targets 5.5 and 10.2).

## 17.3 Agroforestry in the Brazilian Atlantic Forest

Agroforestry practices are not new in the Atlantic Forest: agroforestry homegardens, shaded crops known as '*cabruca*' (with cocoa) and '*faxinais*' (with yerba mate), and '*coivara*' agriculture (shifting agriculture, associated with the use of fire and fallow practices) are part of indigenous heritage, and are used by traditional communities that inhabit the biome, such as '*quilombolas*' and '*caiçaras*' (May and Trovatto 2008). Despite its socioecological importance, these traditional agroforestry systems are increasingly rare as a result of extensive deforestation, agricultural intensification, rural outmigration and restrictive environmental laws (e.g. by shortening the fallow period, law restrictions may avoid the recovery of soil health) (May and Trovatto 2008). A recent meta-analysis shows that agroforestry systems may provide up to 45% and 65% more benefits for biodiversity and ecosystem service levels, respectively, than conventional production systems in the Atlantic Forest; and that biodiverse agroforestry performs better than simple agroforestry systems (Santos et al. 2019).

Due to increasing recognition of the benefits of agroforestry, multiple governmental and non-governmental institutions, including farmers associations and the land reform movement (e.g. the Landless Rural Workers' Movement – MST), have been researching, encouraging and supporting their adoption throughout the biome (Cullen et al. 2005; May and Trovatto 2008). In general, agroforestry systems promoted by those initiatives have been implemented in accordance with the principles of agroecology, enabling the transition from simplified models to biodiverse systems, through participatory and systemic strategies, recognizing the endogenous sociocultural potential and rescuing traditional knowledge (May and Trovatto 2008; Steenbock et al. 2013; Canuto 2017). These efforts include the implementation of demonstrative units, technical training, strengthening rural extension services, securing markets, and resulted in the creation of innovative regulations and cross-sectoral public policies (May and Trovatto 2008; Porro and Miccollis 2011; Sambuichi et al. 2017). In the following pages, we present some examples of agroforestry practices and initiatives.

### *17.3.1 Case Studies*

In the south of Bahia and in some regions of Espírito Santo state, shaded cocoa systems have been managed for more than two and a half centuries (May and Trovatto 2008). The traditional '*cabruca*' system is classified as an authentic 'static' agroforestry system, as its implementation consists of thinning the understory to plant cocoa trees, while preserving the canopy of larger trees. This system maintains some of the characteristics of the original forest, including part of its biodiversity. This allows for the maintenance of fertility and thus a greater longevity of production without the need for external inputs (Ruf and Schroth 2004). However, there was a dramatic reduction in the area covered by these systems between the 1960s and 1990s, as a result of agricultural modernization, declining prices of cacao in international markets and the outbreak of the witches' broom disease, which led many landowners to systematically exploit the native woods and convert their agroforests into pastures (Ruf and Schroth 2004; May and Trovatto 2008).

Environmental groups have engaged in avoiding further loss of these systems due to the recognition of their biodiversity outcomes, such as providing habitat for the endangered golden-headed lion tamarin (*Leontopithecus chrysomelas*), and other forest species (Oliveira et al. 2009; Pardini et al. 2009). As a response to these efforts, farmers are trying to revitalize the remaining area of *cabruca* practicing fertilizer pruning and increasing the diversity of species (May and Trovatto 2008). Similarly, other agroforestry initiatives promoted by environmental NGOs in partnership with rural reform movements in the biome contribute to the conservation of endangered endemic primates: the black lion tamarin (*L. chrysopygus*) in Pontal do Paranapanema, state of São Paulo; and the golden-lion tamarin (*L. rosalia*) in the coastal plain of the state of Rio de Janeiro (Cullen et al. 2005).

The restoration program in Pontal do Paranapamena region, led by the Institute of Ecological Research (IPE), created job opportunities for rural settlers and benefited the black-lion tamarin and other native species (Uezu et al. 2008; Cullen et al. 2005; Valladares-Pádua et al. 2002). Different restoration plantings and agroforestry systems have been promoted over the last 20 years. Among those, several coffee agroforestry systems (a diversified system that combines *Coffea arabica* L. cultivation with annual crops and Atlantic Forest tree species) were implemented, functioning as 'biodiversity islands' or 'stepping stones' that enhance landscape connectivity while providing income for local people (Cullen et al. 2005). A study recently conducted by Badari et al. (2020) shows that the ecological responses of natural regeneration (in terms of quantity and biodiversity of natural regeneration plants) in those systems vary according to the intensity of coffee production, being inversely proportional to the density of coffee plants. Although there may be a tradeoff between the number of coffee plants and the abundance of natural regeneration, the coffee agroforestry systems had a higher overall ecological performance (including the percentage of animal-dispersed regeneration species) compared to contemporary conventional restoration plantings located at the same landscape, proving to be a potentially promising option for forest restoration in this context.

Further south along the Atlantic Coast, another successful agroforestry experience has been developed in Vale do Ribeira region, between São Paulo and Paraná States. The agroforestry farmers' cooperative '*Cooperafloresta*' shows that it is possible to convert soil degradation through food production, providing further evidence that agroforestry is an ecological and socioeconomically viable alternative land use in the biome. After the adoption of biodiverse agroforestry systems, inspired in the work of Ernst Gotsch and adapted with the local knowledge (most of these farmers can be considered as traditional farmers, many are part of '*quilombola*' communities), the families reached a sevenfold increase in their annual revenues, as well as numerous non-monetary benefits that are not easily measurable, such as increased food sovereignty and self-realization (Steenbock et al. 2013).

Part of this success is due to collective initiatives for processing and selling, and we can highlight communal infrastructures, such as the packing hall, the honey processing house and the agroindustry (where banana, guava, fruit-pulps, ice cream and jams are processed). Around 100 families (300 people) participate in the cooperative, and the area occupied by agroforestry systems is approximately 750 hectares, of which 250 are intensely managed agroforests and another 500 are less intensely managed, allowing natural regeneration to predominate. Data collected in these systems show that there was an average annual increment of 6.6 tons of total carbon per hectare in SAFs, aged between 4 and 15 years (Steenbock et al. 2013).

In the state of Rio de Janeiro, various agroforestry experiences have been developing over the last decades, including *neorurals* (term coined in France in the 1970s to describe urban exodus movement for rural areas), promoted by multiple stakeholders, inspired by Ernst Götsch and Cooperafloresta (Tubenchlak 2018). The first initiatives promoting agroecological agroforestry systems started at the end of the 1990s in the municipality of Paraty (Mattos 2011). Also, in 1993 was created the '*Fazendinha Agroecológica*', the experimental unit of EMBRAPA Agrobiologia in partnership with the Federal Rural University of Rio de Janeiro (UFRRJ), which has multiple examples of agroforestry systems and has contributed to agroforestry training and diffusion in the state (Neves et al. 2005; Campello et al. 2006).

Studies in the region of Paraty demonstrate the importance of traditional and novel agroforestry practices for biodiversity and ecosystem functions (Silveira et al. 2007; de Souza et al. 2016b). Garrote (2004) verified the existence of 347 plant species, with an average of 67 species per homegarden in three management zones, demonstrating the great diversity and complexity of these systems, their multiple functions and uses, and their contribution to maintaining the existing broad biodiversity. Also, Mattos (2011) and Strauch (2016) identify the characteristics and existence of traditional 'ways of life' as success factors in the implementation and expansion of the 40 agroforestry systems that existed at the time of their academic research. These agroforestry practices generated innovations, alternative products and income, increased the level of security and food sovereignty of families (and of the territory) and helped to reduce predatory use of local flora and fauna. Furthermore, they contributed to the resistance and territorial autonomy of the traditional populations.

The state of Rio de Janeiro also provides examples of how public policies can promote the adoption of agroforestry systems. The '*Rio Rural*' program directly provided resources to farmers to adopt environmental conservation practices associated with improved productive practices, supporting the implementation of a large number of agroforestry systems. This implementation was supported by several partner institutions that created an agroecological network of in the northwestern region of the state. This integrated action resulted in the creation of three participatory research units, and the implementation of 37 '*Horta-floresta*' (intensely managed successional agrisilvicultural system, based on short-cycle species planted along with fruit trees, eucalyptus, banana and native trees – Miccolis et al. 2016). The '*Horta-floresta*' systems were implemented in a joint effort scheme, with the participation of farmers, technicians and students (Tubenchlak 2018).

The '*Rio Rural*' project has diversified agricultural production systems, which was predominantly based on coffee and dairy cattle farms. With the diversification of production systems, the farmers now access other commercial channels (including the organic market), stimulating short marketing circuits, such as farmers market and institutional markets (national program for school meals – PNAE). Another example in the state is the '*Conexão Mata Atlântica*' project, which aims to recover climate and biodiversity services within the 'Paraíba do Sul' River Basin. This pilot project of Payments for Environmental Services (PES) has started in six municipalities and the results achieved as of 2018 indicate that 1773 hectares of Atlantic Forest were under environmental conservation, 269 hectares were undergoing restoration and 43 hectares were allocated to productive conversion throughout agroforestry systems, benefiting 165 farmers in total.

## 17.4 Looking Forward

As debated in this chapter, one of the strategies towards the sustainability of agroecosystems is to try to get as close as possible to the ecological relations existing in the natural ecosystems of their surroundings, in order to reproduce the essential ecological functions that underpin productivity. In this sense, rural landscapes in the Atlantic Forest should be necessarily forested. The megadiverse biome offers a vast amount of possibilities for designing biodiverse agroforestry systems (Oliveira & Carvalhaes, 2016). Canosa and Moraes (2016) have identified the functional attributes of more than 700 tree species native to the Atlantic Forest to support restoration initiatives. The functional attributes analysed were: (i) biological nitrogen fixation potential; (ii) wildlife attractiveness and (iii) provision of forest products (timber or non-timber); highlighting the potential of agroforestry systems to conserve native biodiversity while providing multiple forest goods and services.

Recent advances in regulatory frameworks, such as the inclusion of agroforestry systems as alternatives to areas set aside for recovery in the Native Vegetation Protection Law, offer opportunities to further stimulate their adoption. However, there are still bureaucratic barriers and legal insecurity in managing those systems, especially for native trees, which hinder its adoption by farmers and favours the use of exotic species (Miccolis et al. 2016; Tubenchlak 2018). As presented before, some environmental legislation can restrict agroforestry practices; while, there is no legislation establishing minimum ecological criteria for the current conventional agriculture, which is increasingly more dependent on the use of pollutant agrochemicals.

The integration between environmental and agricultural policies is thus fundamental to overcome the dichotomy between use and conservation. Multiple governmental programs and public policies, such as PNAE, PAA (Family Agriculture Food Acquisition Program), PRONAF (National Program for the Strengthening of Family Farming) and some PES initiatives, have effectively contributed to increase farmers' incomes and have contributed to agroforestry development in Brazil over recent decades (Miccolis et al. 2011). This policy mix evidences the complexity of effectively promoting sustainable agriculture: efforts need to go beyond the agricultural practices and establish direct links between supply and demand, including public procurement and payment for ecosystem services.

The adoption of agroecological principles and practices in landscape approaches offers opportunities to engage people in restorative processes, maintain rural livelihoods, and also to promote the return to rurality, which can be considered a search for – and results in – a more harmonious relationship between man and nature (Foley et al. 2011; Folke et al. 2011; Raymond et al. 2013). Hence, restoration efforts in the Atlantic Forest should embrace agroecosystems, understanding their role in "*integral ecological restoration*", which is defined as the restoration of human–nature connection (Celentano and Rousseau 2016). The advance of agroforestry systems at greater spatial and temporal scales can contribute to recovering areas through food production, fostering the transition from a "*shrinking hotspot"* into a "*hope spot"* (Ribeiro et al. 2011; Scarano and Ceotto 2015).

## References

Agroforestry Network (2018) Achieving the Global Goals through agroforestry, 28

Altieri MA (2002) Agroecologia: bases científicas para uma agricultura sustentável. Livraria e Editora Agropecuária, Guaíba/RS, 592p

Altieri MA, Nicholls CI (2013) The adaptation and mitigation potential of traditional agriculture in a changing climate. Clim Chang 140(1):33–45. https://doi.org/10.1007/s10584-013-0909-y

Altieri MA, Funes-Monzote FR, Petersen P (2012) Agroecologically efficient agricultural systems for smallholder farmers: contributions to food sovereignty. Agron Sustain Dev 32(1):1–13. https://doi.org/10.1007/s13593-011-0065-6

Altieri MA, Nicholls CI, Henao A, Lana MA (2015) Agroecology and the design of climate change-resilient farming systems. Agron Sustain Dev 35(3):869–890

Amador DB (2003) Restauração de Ecossistemas com Sistemas Agroflorestais. Retrieved from http://saf.cnpgc.embrapa.br/publicacoes/14.pdf

Aronson J, Alexander S (2013) Ecosystem restoration is now a global priority: time to roll up our sleeves. Restor Ecol 21(3):293–296. https://doi.org/10.1111/rec.12011

Badari CG, Bernardini LE, de Almeida DRA, Brancalion PHS, César RG, Gutierrez V, Chazdon RL, Gomes HB, Viani RAG (2020) Ecological outcomes of agroforests and restoration 15 years after planting, Restoration Ecology. https://doi.org/10.1111/rec.13171

Bakermans MH, Rodewald AD, Vitz AC, Rengifo C (2012) Migratory bird use of shade coffee: the role of structural and floristic features. Agrofor Syst 85(1):85–94. https://doi.org/10.1007/s10457-011-9389-0

Barretto AGOP, Berndes G, Sparovek G, Wirsenius S (2013) Agricultural intensification in Brazil and its effects on land-use patterns: an analysis of the 1975–2006 period. Glob Chang Biol 19(6):1804–1815. https://doi.org/10.1111/gcb.12174

Besseau P, Graham S, Christophersen T (2018) Restoring forests and landscapes: the key to a sustainable future. Vienna, Austria. Retrieved from www.unenvironment.org

Brancalion PHS, Viani RAG, Strassburg BBN, Rodrigues RR (2012) Finding the money for tropical forest restoration. Unasylva 63(239):25–34. Retrieved from http://www.lcb.esalq.usp.br/publications/articles/2012/2012uv63n239p25-34.pdf

Cacho O, Marshall GR, Milne M (2003) Smallholder agroforestry projects: potential for carbon sequestration and poverty alleviation. ESA Working Paper 3(6):77

Campello EF, Silva G, Nóbrega P, Vieira A, Franco A, Resende A (2006) Implantação e manejo de SAFs na Mata Atlântica: A Experiência da Embrapa Agrobiologia. In: Gama-Rodrigues, A. C.; Barros, N. F.; Gama-Rodrigues, E. F. Sistemas agroflorestais: Bases científicas para o desenvolvimento sustentável. Campos dos Goytacazes, RJ: Universidade Estadual do Norte Fluminense Darcy Ribeiro

Campos MC (2011) Modernização da agricultura, expansão da soja no Brasil e as transformações socioespaciais no Paraná. Rev Geogr 6(1):161–191

Canosa GA, Moraes LFD (2016) Atributos funcionais de espécies da Mata Atlântica: ferramentas para o planejamento ambiental e econômico. Embrapa. Documentos 305:1–168

Canuto JC (2017) Sistemas Agroflorestais: experiências e reflexões. Brasília, DF

Celentano D, Rousseau G (2016) Integral ecological restoration: restoring the link between human culture and nature. Ecol Restor 34(2):94–97. https://doi.org/10.3368/er.34.2.94

Chazdon RL, Uriarte M (2016) Natural regeneration in the context of large-scale forest and landscape restoration in the tropics. Biotropica 48:709–715. https://doi.org/10.1111/btp.12409

Chazdon RL, Harvey CA, Komar O, Griffith DM, Ferguson BG, Martínez-Ramos M, Morales H, Nigh R, Soto-Pinto L, van Breugel M, Philpott SM (2009a) Beyond reserves: a research agenda for conserving biodiversity in human-modified tropical landscapes. Biotropica 41(2):142–153. https://doi.org/10.1111/j.1744-7429.2008.00471.x

Chazdon RL, Peres CA, Dent D, Sheil D, Lugo AE, Lamb D, Stork NE, Miller SE (2009b) The potential for species conservation in tropical secondary forests. Conserv Biol 23(6):1406–1417. https://doi.org/10.1111/j.1523-1739.2009.01338.x

Chazdon R, Broadbent EN, Rozendaal DMA, Bongers F, Zambrano AMA, Aide TM, Balvanera P, Becknell JM, Boukili V, Brancalion PHS, Craven D, Almeida-Cortez JS, Cabral GAL, de Jong B, Denslow JS, Dent DH, DeWalt SJ, Dupuy JM, Durán SM, Espírito-Santo MM, Fandino MC, César RG, Hall JS, Hernández-Stefanoni JL, Jakovac CC, Junqueira AB, Kennard D, Letcher SG, Lohbeck M, Martínez-Ramos M, Massoca P, Meave JA, Mesquita R, Mora F, Muñoz R, Muscarella R, Nunes YRF, Ochoa-Gaona S, Orihuela-Belmonte S, Peña-Claros M, Pérez-García EA, Piotto D, Powers JS, Rodríguez-Velazquez J, Romero-Pérez IE, Ruíz J, Saldarriaga JG, Sanchez-Azofeifa A, Schwartz NB, Steininger MK, Swenson NG, Uriarte M, van Breugel M, van der Wal H, Veloso MDM, Vester H, Vieira ICG, Bentos TV, Williamson GB, Poorter L (2016) Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. Sci Adv 333(6045):988–993. https://doi.org/10.1126/science.1201609

Clement CH, Junqueria AB, Moreira PA, Lins J, Levis C, Cabral TS, Lima PP (2015) A ecologia histórica e a criação da agrobiodiversidade na Amazônia. In: Santilli J, Bustamante PG, Barbieri RL (orgs), Agrobiodiversidade (Coleção Transição Agroecológica). Brasília, DF, EMBRAPA, 308p

Cullen L, Alger K, Rambaldi DM (2005) Land reform and biodiversity conservation in Brazil in the 1990s: conflict and the articulation of mutual interests. Conserv Biol 19(3):747–755. https://doi.org/10.1111/j.1523-1739.2005.00700.x

de Moraes LFD, da Amâncio COG, de Resende AS (2011) Sistemas agroflorestais para o uso sustentável do solo: considerações agroecológicas e socioeconômicas. Seropédica, RJ

de Souza MC, Piña-Rodrigues FC (2013) Desenvolvimento de espécies arbóreas em sistemas agroflorestais para recuperação de áreas degradadas na Floresta Ombrófila Densa, Paraty, RJ. Revista Árvore 37(1):89–98. Retrieved from http://www.redalyc.org/pdf/488/48825658019.pdf

de Souza SEXF, Vidal E, Chagas, G. de F., Elgar, A. T., & Brancalion, P. H. S. (2016a) Ecological outcomes and livelihood benefits of community-managed agroforests and second growth forests in Southeast Brazil. Biotropica 48(6):868–881. https://doi.org/10.1111/btp.12388

de Souza MCS, Piña-Rodrigues FCM, Casagrande JC, Da Silva SF, Scoriza RN (2016b) Funcionalidade ecológica de sistemas agroflorestais biodiversos: Uso da serapilheira como indicador da recuperação de áreas de preservação permanente. Floresta 46(1):75–82. https://doi.org/10.5380/rf.v46i1.34991

Dean W (1996) A ferro e fogo: a história e a devastação da Mata Atlântica brasileira. Companhia das Letras, São Paulo, Brazil

DeClerck F, Jones S, Attwood S, Bossio D, Girvetz E, Chaplin-Kramer B et al (2016) Agricultural ecosystems and their services: the vanguard of sustainability? Curr Opin Environ Sustain 23:92–99. https://doi.org/10.1016/j.cosust.2016.11.016

Dickinson AK (2014) Analog forestry: creating productive landscapes. In: ETFRN news 56: towards productive landscapes, 103

Estrada A, Raboy BE, Oliveira LC (2012) Agroecosystems and Primate Conservation in the tropics: a review. Am J Primatol 74(8):696–711

Farrely M (2016) Contribuições da Agroecologia para os Objetivos do Desenvolvimento Sustentável. In: Revista Agriculturas: experiências em Agroecologia. LEISA Brasil, AS-PTA/RJ, volume 13, n° 3

Ferreira AS, Peres CA, Bogoni JA, Cassano CR (2018) Use of agroecosystem matrix habitats by mammalian carnivores (Carnivora): a global-scale analysis. Mammal Rev 48(4):312–327. https://doi.org/10.1111/mam.12137

Foley JA, Ramankutty N, Brauman KA, Cassidy ES, Gerber JS, Johnston M, Mueller ND, O'Connell C, Ray DK, West PC, Balzer C, Bennett EM, Carpenter SR, Hill J, Monfreda C, Polasky S, Rockstrom J, Sheehan J, Siebert S, Tilman D, Zaks DPM (2011) Solutions for a cultivated planet. Nature 478(7369):337–342. https://doi.org/10.1038/nature10452

Folke C, Sa Jansson A, Rockström J, Olsson P, Carpenter SR, Chapin FS et al (2011) Reconnecting to the biosphere. Ambio 40:719–738. https://doi.org/10.1007/s13280-011-0184-y

Food and Agricultural Organization of the United Nations – FAO. (2016) Forest landscape restoration for Asia-Pacific forests, 198

Food and Agricultural Organization of the United Nations – FAO. (2018) Scaling up agroecology initiative: transforming food and agricultural systems in support of the SDGs

Gardner TA, Barlow J, Chazdon R, Ewers RM, Harvey CA, Peres CA, Sodhi NS (2009) Prospects for tropical forest biodiversity in a human-modified world. Ecol Lett 12(6):561–582. https://doi.org/10.1111/j.1461-0248.2009.01294.x

Garrote V (2004) Os quintais caiçaras, suas características socioambientais e perspectivas para a comunidade do Saco do Mamanguá, Paraty/RJ. Dissertação de mestrado em Recursos Florestais, apresentada à Escola Superior de Agricultura Luiz de Queiroz/USP, São Paulo, 198p

Gliessman SR (2001) Agroecologia: processos ecológicos em agricultura sustentável. 2ª ed., Porto Alegre/RS, Ed. UFRGS, 655p

Gliessman S, Tittonell P (2015) Agroecology for food security and nutrition. Agroecol Sustain Food Syst 39(2):131–133. https://doi.org/10.1080/21683565.2014.972001

Guzmán Casado G, Sevilla Guzmán E, González de Molina M (2000) Introducción a la Agroecologia como desarrollo rural sostenible. Mundi Prensa, Madrid, 535p

Hecht S (2002) A evolução do Pensamento Agroecológico. In: Altieri, M. A. Agroecologia: bases científicas para uma agricultura sustentável. Livraria e Editora Agropecuária, Guaíba/RS, 592p

Hector A, Bagchi R (2007) Biodiversity and ecosystem multifunctionality. Nature 448(7150):188

IBGE – Instituto Brasileiro de Geografia e Estatística – Censo Agropecuário (2017) Disponível em: https://sidra.ibge.gov.br/pesquisa/censo-agropecuario/censo-agropecuario-2017. Acesso em: 26 mar.2019

IUFRO (2017) Implementing Forest Landscape Restoration: a practitioner's guide. (J. Stanturf, S. Mansourian, & M. Kleine, Eds.). Viena, Austria

Jose S (2009) Agroforestry for ecosystem services and environmental benefits: an overview. Agrofor Syst 76(1):1–10. https://doi.org/10.1007/s10457-009-9229-7

Jose S (2012) Agroforestry for conserving and enhancing biodiversity. Agrofor Syst 85(1):1–8. https://doi.org/10.1007/s10457-012-9517-5

Kageyama AA, Bergamasco SMPP, de Oliveira JTA (2013) Uma tipologia dos estabelecimentos agropecuários do Brasil a partir do censo de 2006. Rev Econ Sociol Rural 51(1):105–122. https://doi.org/10.1590/s0103-20032013000100006

Laestadius L, Buckingham K, Maginnis S, Saint-Laurent C (2015) Before Bonn and beyond: the history and future of forest landscape restoration. Unasylva 245(66):11–17

Lamb D, Erskine PD, Parrotta JA (2005) Restoration of degraded tropical Forest landscapes. Science 310:1628–1632. https://doi.org/10.1126/science.1111773

LAPIG – Laboratório de Processamento de Imagens e Geoprocessamento (2019) Disponível em: http://maps.lapig.iesa.ufg.br/. Acesso em 30 mar. 2019

Latawiec AE, Strassburg BBN, Brancalion PHS, Rodrigues RR, Gardner T (2015) Creating space for large-scale restoration in tropical agricultural landscapes. Front Ecol Environ 13(4):211–218. https://doi.org/10.1890/140052

Latawiec AE, dos Santos JS, Maioli V, Junqueira AB, Crouzeilles R, Jakovac AC, Tubenchlak F, Strassburg BB (2018) 7 Forest landscape restoration and land sparing–sharing. In: Mansourian S, Parrotta J (eds) Forest landscape restoration: integrated approaches to support effective implementation. Routledge, London.

Leakey RRB (1996) Definition of agroforestry revisited. Agrofor Today 8(1):5–7. https://doi.org/10.1016/b978-0-12-805356-0.00001-5

Leakey RRB (2012) Living with the trees of life. Towards the transformation of tropical agriculture. CAB International, Wallingford, UK

Leakey RRB (2019) A holistic approach to sustainable agriculture: trees, science and global society. In Agroforestry for Sustainable Agriculture

Levis C, Costa FRC, Bongers F, Peña-Claros M, Clement CR, Junqueira AB et al (2017) Persistent effects of pre-Columbian plant domestication on Amazonian forest composition. Science 355(6328):925–931. https://doi.org/10.1126/science.aal0157

Maier TF, de Benini RM, Fachini C, de Santana PJA (2018) Financial analysis of enrichment. Model using timer and non-timber produtcs of secondary remnants in the Atlantic Fores. Revista Árvore 42(6):420602. https://doi.org/10.1590/1806-90882018000600002

Mansourian S, Vallauri D, Dudley N (2005) Forest restoration in landscapes: beyond planting trees. Springer, New York

Mansourian S (2017) Understanding the relationship between governance and forest landscape restoration. 14(3):267–278. https://doi.org/10.4103/0972-4923.186830

Mansourian S, Stanturf JA, Derkyi MAA, Engel VL (2017) Forest landscape restoration: increasing the positive impacts of forest restoration or simply the area under tree cover? Restor Ecol 25(2). https://doi.org/10.1111/rec.12489

Mapbiomas. Coleção 3.1 (2019). Disponível em: http://mapbiomas.org. Acesso em: 15 mar. 2019

Mattos C (2011) Expressões agroecológicas a partir de percepções socioambientais da agricultura tradicional e camponesa em Paraty/RJ. Dissertação de mestrado em Ciências Ambientais na Universidade Federal Fluminense

May PH, Trovatto CMM (2008) Manual agroflorestal para a Mata Atlântica. Ministério Do Desenvolvimento Agrário, Brasília, p 196

Miccolis A, Vivan JL, Gonçalves ALR, Méier M, Porr R (2011) Políticas públicas e Sistemas Agroflorestais: lições aprendidas a partir de cinco estudos de caso no Brasil. In Políticas Públicas e Financiamento para o Desenvolvimento Agroflorestal no Brasil (pp. 1–24). Retrieved from http://www.worldagroforestry.org/latinamerica/sites/worldagroforestry.org.latinamerica/files/politicas.pdf

Miccolis A, Mongeli F, Henrique P, Marques R, Luis D, Vieira M, Arco-Verde MF, Hoffmann MR, Rehder T, Pereira B (2016) Restauração ecológica com sistemas agroflorestais: como conciliar conservação com produção. Opções para Cerrado e Caatinga. Brasilia: Instituto Sociedade, População e Natureza – ISPN/Centro Internacional de Pesquisa Agorflorestal – ICRAF,. Retrieved from http://www.cgiar.org/who-we-are/cgiar-fund/fund-donors-2/

MMA – Ministério do Meio Ambiente, 2009. http://www.mma.gov.br/estruturas/pnf/_arquivos/in_mma_05_2009_5.pdf

Nair PKR (1985) Classification of agroforestry systems. Agrofor Syst 3(2):97–128. https://doi.org/10.1007/BF00122638

Nair PK (1993) An introduction to agroforestry. ICRAF (Vol. 73). Kluwer academic publishers. https://doi.org/10.1016/0378-1127(95)90008-X

Neves MCP, Guerra JGM, Carvalho SR, Ribeiro RLD, Almeida DL (2005) Sistema integrado de produção agroecológica ou Fazendinha Agroecológica do Km 47. In: de Aquino AM, de Assis RL (eds) Agroecologia: Princípios e técnicas para uma agricultura orgânica sustentável. Brasília, DF, Embrapa Informação Tecnológica, pp 147–172

Oliveira RE, Carvalhaes MA (2016) Agroforestry as a tool for restoration in Atlantic forest: can we find multi-purpose species? Oecologia Australis 20(4):425–435. https://doi.org/10.4257/oeco.2016.2004.03

Oliveira LC, Hankerson SJ, Dietz JM, Raboy BE (2009) Key tree species for the golden-headed lion tamarin and implications for shade-cocoa management in Southern Bahia, Brazil. Animal Conservation, (April), 1–11. https://doi.org/10.1111/j.1469-1795.2009.00296.x

Pardini R, Faria D, Accacio GM, Laps RR, Mariano-Neto E, Paciencia MLB, Dixo M, Baumgarten J (2009) The challenge of maintaining Atlantic forest biodiversity: A multi-taxa conservation assessment of specialist and generalist species in an agro-forestry mosaic in southern Bahia. Biol Conserv 142(6):1178–1190. https://doi.org/10.1016/j.biocon.2009.02.010

Peneireiro FM (2003) Fundamentos da Agrofloresta Sucessional. II Simpósio sobre Agrofloresta Sucessionais. Embrapa/Petrobrás, Sergipe

Perfecto I, Vandermeer J (2008) Biodiversity conservation in tropical agroecosystems: A new conservation paradigm. New York Acad Sci 1134:173–200. https://doi.org/10.1196/annals.1439.011

Perfecto I, Vandermeer J (2010) The agroecological matrix as alternative to the land-sparing/agriculture intensification model. Proc Natl Acad Sci U S A 107(13):5786–5791. https://doi.org/10.1073/pnas.0905455107

Perfecto I, Mas A, Dietsch T, Vandermeer J (2003) Conservation of biodiversity in coffee agroecosystems: a tri-taxa comparison in southern Mexico. Biodivers Conserv 12:1239–1252. Retrieved from https://link.springer.com/content/pdf/10.1023%2FA%3A1023039921916.pdf

Porro R, Miccollis A (2011) Políticas Públicas e Financiamento para o Desenvolvimento Agroflorestal no Brasil

Posey DA (1985) Indigenous management of tropical forest ecosystems: the case of the Kayapo Indians of the Brazilian Amazon. Agrofor Syst 3:139–158. https://doi.org/10.1007/BF00122640

Raymond CM, Singh GG, Benessaiah K, Bernhardt JR, Levine J, Nelson H, Turner NJ, Norton B, Tam J, Chan KMA (2013) Ecosystem services and beyond: using multiple metaphors to understand human–environment relationships. Bioscience 63(7):536–546. https://doi.org/10.1525/bio.2013.63.7.7

Reed J, van Vianen J, Barlow J, Sunderland T (2017) Have integrated landscape approaches reconciled societal and environmental issues in the tropics? Land Use Policy 63:481–492. https://doi.org/10.1016/j.landusepol.2017.02.021

Ribeiro MC, Martensen AC, Metzger JP, Tabarelli M, Scarano F, Fortin M-J (2011) The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In Biodiversity hotspots (pp. 405–434). https://doi.org/10.1016/S0169-5347(98)01363-9

Ricketts (2001) The matrix matters: effective isolation in fragmented landscapes. Am Nat 158(1):87–99. https://doi.org/10.2307/3078900

Ruf F, Schroth G (2004) Chocolate forests and monocultures: a historical review of cocoa growing and its conflicting role in tropical deforestation and forest conservation (Chapter 6). In: Schroth G et al (eds) Agroforestry and biodiversity conservation in tropical landscapes. Island Press, Washington, DC, 535 p

Sambuichi RHR, de Moura IF, de Mattos LM, de Ávila ML, Spínola PAC, da Silva APM (2017) A política nacional de agroecologia e produção orgânica no Brasil : uma trajetória rural de luta pelo desenvolvimento sustentável. A política nacional de agroecologia e produção orgânica no Brasil: uma trajetória de luta pelo desenvolvimento rural sustentável. Brasília

Santos PZF, Crouzeilles R, Sansevero JBB (2019) Can agroforestry systems enhance biodiversity and ecosystem service provision in agricultural landscapes? A meta-analysis for the Brazilian Atlantic Forest. For Ecol Manag 433(November):140–145. https://doi.org/10.1016/j.foreco.2018.10.064

Sayer J, Sunderland T, Ghazoul J, Pfund J-L, Sheil D, Meijaard E, Venter M, Boedhihartono AK, Day M, Garcia C, van Oostenj C, Buck LE (2013) Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. Proc Natl Acad Sci 110(21):8349–8356. https://doi.org/10.1073/pnas.1210595110

Scarano FR, Ceotto P (2015) Brazilian Atlantic forest: impact, vulnerability, and adaptation to climate change. Biodivers Conserv. https://doi.org/10.1007/s10531-015-0972-y

Schulz JJ, Schröder B (2017) Identifying suitable multifunctional restoration areas for Forest landscape restoration in Central Chile. Ecosphere 8(1). https://doi.org/10.1002/ecs2.1644

Senanayake R, Jack J (1998) Analogue forestry: an introduction. Monash publications in geography no. 49. Monash Print Services, Melbourne

Silveira ND, Pereira MG, Polidoro JC, Tavares SRDL, Mello RB (2007) Aporte de nutrientes e biomassa via serrapilheira em sistemas agroflorestais em Paraty (RJ). Ciencia Florestal 17(2):129–136. https://doi.org/10.5902/198050981944

Sloan S, Sayer JA (2015) Forest Ecology and Management Forest Resources Assessment of 2015 shows positive global trends but forest loss and degradation persist in poor tropical countries. Forest Ecol Manag 352:134–145

SOS Mata Atlântica (2017) Atlas dos remanescentes florestais da Mata Atlântica: período 2015–2016. Relatório técnico. São Paulo: Fundação SOS Mata Atlântica. Disponível em: http://mapas.sosma.org.br/site_media/download/atlas_2015-2016_relatorio_tecnico_2017.pdf

Stanturf JA, Palik BJ, Williams MI, Dumroese RK, Madsen P (2014) Forest restoration paradigms. J Sust Forestr 33(SUP1). https://doi.org/10.1080/10549811.2014.884004

Steenbock W, da Silva LCE, da Silva RO, Rodrigues AS, Perez-Cassarino J, Fonini R (2013) Agrofloresta, Ecologia e Sociedade. Kairós Edições, Curitiba. Retrieved from http://www.tandfonline.com/doi/abs/10.1080/11263508809430593

Strassburg BBN, Latawiec AE, Barioni LG, Nobre CA, da Silva VP, Valentim JF, Vianna M, Assad ED (2014) When enough should be enough: improving the use of current agricultural lands could meet production demands and spare natural habitats in Brazil. Glob Environ Chang 28(1):84–97. https://doi.org/10.1016/j.gloenvcha.2014.06.001

Strauch GFE (2016) Redes sócio técnicas camponesas: inovações agroecológicas, autonomia, e articulação territorial em Paraty, estado do Rio de Janeiro. Tese de doutorado em Agroecologia apresentada ao Programa de Recursos Naturais da Universidade de Córdoba/Espanha, 259 p. Acesso em 13/02/2018, disponível em http://hdl.handle.net/10396/13258

Suding BK, Higgs E, Palmer M, Callicott JB, Anderson CB, Gutrich JJ et al (2015) Committing to ecological restoration. Science 348(6235):638–640

Toledo VM (1993) La racionalidad de la producción campesina. In: Sevilla Guzmán E, González de Molina M (eds) Ecología, campesinado y historia. Las Ediciones de la Piqueta, Madrid, 253 p

Troya V, Kumar C (2016) O que é restauração de paisagens e florestas? In: Moraes, M. A. (Org.). Restauração de florestas e paisagens no Brasil. Brasília: UICN. pág. 24–32

Tubenchlak F (2018) Restauração de Paisagens e Transição (Agro)Florestal: o caso do Estado do Rio de Janeiro, Brasil. Dissertação apresentada ao Programa de Pós-Graduação em Ecologia do Instituto de Biologia da Universidade Federal do Rio de Janeiro

Uezu A, Beyer DD, Metzger JP (2008) Can agroforest woodlots work as stepping stones for birds in the Atlantic forest region? Biodivers Conserv 17(8):1907–1922. https://doi.org/10.1007/s10531-008-9329-0

Valladares-Pádua C, Pádua SM, Cullen Junior L, Ditt HM (2002) Módulos agroflorestais na conservação de fragmentos florestais da Mata Atlântica. Revista Experiências PDA 2:7–33

van Noordwijk M (2019) Sustainable development through trees on farms: agroforestry in its fifth decade. World Agroforestry, Bogor, Indonesia. Retrieved from http://www.worldagroforestry.org/downloads/Publications/PDFS/B19029.pdf

van Lexmond, MB, Bonmatin, J, Goulson, D, Noome AD (2015) Worldwide integrated assessment on systemic pesticides. Environ Sci Pollut Res 22, 1–4. https://doi.org/10.1007/s11356-014-3220-1

Vieira DLM, Holl KD, Peneireiro FM (2009) Agro-successional restoration as a strategy to facilitate tropical forest recovery. Restor Ecol 17(4):451–459. https://doi.org/10.1111/j.1526-100X.2009.00570.x

Wezel A, Soldat V (2009) A quantitative and qualitative historical analysis of the scientific discipline of agroecology. Int J Agric Sustain 7(1):3–18. https://doi.org/10.3763/ijas.2009.0400

Young CEF (2006) Desmatamento e desemprego rural na Mata Atlântica. Floresta e Ambiente 13(2):75–88

Zomer RJ, Neufeldt H, Xu J, Ahrends A, Bossio D, Trabucco A, van Noordwijk M, Wang M (2016) Global tree cover and biomass carbon on agricultural land: the contribution of agroforestry to global and national carbon budgets. Sci Rep 6(April):29987. https://doi.org/10.1038/srep29987

# Chapter 18
# Engaging People for Large-Scale Forest Restoration: Governance Lessons from the Atlantic Forest of Brazil

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**Abstract** Large-scale forest restoration strategies should be designed not only in the light of ecological aspects, but also considering the diverse interests and uses of landscape to expand project strategies and methodologies while also supporting a more effective, long-lasting and inclusive restoration. Here, we discuss the process of social engagement through restoration, incorporating this broader view of projects and evaluating restoration initiatives with different spatial scales and objectives within the Atlantic Forest of Brazil. These projects incorporate initiatives from different networks, platforms and sites distributed across regions with high ecological and socioeconomic heterogeneity. Despite the different restoration approaches used, the creation of a common ground for sharing information, communication,

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vision and expectations, as well the continuous dialogue between multiple stakeholders, is a common key factor among projects in the Atlantic Forest. Thus, we aim to shed light on relevant aspects of these strategies, trying to capture the enabling conditions to organizing people in a common view project and argue that transferring or adapting it more broadly could significantly contribute to the success of the national restoration agenda.

**Keywords** Governance · Stakeholder engagement · Large-scale restoration · Atlantic Forest

## 18.1 Introduction

Ecological restoration can be interpreted as the greatest “mea culpa” declaration of current civilization on environmental themes. Following on the ideas of Aronson et al. (2007), restoration is the recognition of human overexploitation of natural capital with no consideration of sustainability or future generations. In our crowded world, for our own wellbeing and survival, we must act to recover what has been degraded and lost. The cited phrase presents two interesting pieces of information: it confirms our responsibility for environmental and climatic changes, and indicates that we, as a society, have the challenge of seeking solutions to change this reality. With that said, and agreed upon, how can we put into practice the complex task of restoring degraded ecosystems as complex as tropical rainforests? The good news is that many people are willing to take up this challenge. Over the past 40 years, Brazil has played a prominent role in the development of restoration methods and techniques (Bustamante et al. 2019) and is an internationally recognized player and leader. The question is even more critical in the current context of Brazilian national commitments to restore 12 million hectares in the coming years. How to engage multiple stakeholders and organize people to promote regional strategies such as the Initiative 20x20 for Latin American and Caribbean countries, or the Bonn Challenge, expanded in the New York Declaration on Forests at the United Nations Climate Summit to include a total of 350 million hectares restored and recovered around the world by 2030?

There have been significant advances in ecological restoration in Brazil via the application of restoration concepts, especially in experimental projects, both small- and large-scaled, positioning the country as a global reference source in this field (Crouzeilles et al. 2019). A spatial database of the primary restoration projects in the Atlantic Forest, led by the initiative known as the Atlantic Forest Restoration Pact (PACTO), currently includes records of over 36,000 hectares undergoing restoration in the biome (database: https://pactoma.esalq.usp.br/pacto/ – accessed on 10/11/2019).

The way stakeholders interact, negotiate and make decisions about the landscape is what is typically referred to as landscape governance (Kusters et al. 2018; van Oosten et al. 2014). The use of the term governance in association with environmental issues casts light on the challenge faced by humanity, which is experiencing climate change, loss of biodiversity and natural capital and increased complexity, requiring an expansion of understanding and tools to address environmental issues (Ball et al. 2014). The definition of governance one uses should be explicit with regards to the varying ways in which public and private actors from state, market and civil society negotiate and manage public issues (Arts 2014). For a more detailed review about this issue, it is necessary to back in 1968, with the concepts of Garrett Hardin on his classic article "The tragedy of the commons" where he indicates the eminent conflict or collapse due to overuse of shared resources. Alternatively, Elinor Ostrom, the laureate Nobel Prize in economics, suggests an approach in the book "Governing the Commons" (Ostrom 1990) based on large observation on the field, demonstrating how local communities were able to deal with the commons pools, without top-down regulations.

Governance is critical for landscape restoration actions to ensure that the results meet the interests of the inhabitants and users and that, above all, their actions remain sustainable over the years. Regarding this objective, some studies have focused on understanding how the process of organizing stakeholders to reconcile their interests in the landscape occurs. Wilson and Cagalanan (2016) conducted a review of several studies and models of governance at the landscape level and found, among other results, that the expansion of more democratic and inclusive governance mechanisms can ensure the success of restoration initiatives. Pistorius and Freiberg (2014) discuss complementary forms of governance for the implementation of top-down models, such as international agreements, in the specific case of the Aichi Target. Forest Certification and Payment for Ecosystem Services are also referred to as forest governance models (Arts 2014). Van Oosten et al. (2014) suggested that governance at the landscape level may be considered according to three approaches: as a management tool; as a multi-stakeholder decision-making process; or as the creation of a new institutional space, known as "institutional bricolage". Institutional bricolage is a process in which actors consciously and unconsciously reshape or piece together different institutional arrangements. Mansourian et al. (2014) proposed that the factors that influence governance may be described in three pillars that operate at different landscape scales, incorporating stakeholders, processes and structures. Mansourian (2017) conducted a literature review about restoration governance, showing that there are limited guidelines available for practitioners, especially for large-scale restoration initiatives, and then proposed a framework to help fulfil this gap, based on the understanding of stakeholders' connections, contexts and scale.

In a literature review on how large-scale forest restoration affects local livelihoods, Adams et al. (2016) concluded that restoration should be carried out as a form of investment where the financial, environmental and social benefits are obtained simultaneously, attempting to meet the different demands from a variety of stakeholders. When this challenging task is not properly addressed, as observed by Brancalion and Chazdon (2017) for tropical forest and landscape restoration, the negative effects can include inequality in the distribution and access to benefits, generating an imbalance among the stakeholders involved. However, the authors indicate that bottom-up processes tend to bring more benefits to the community, which is reinforced by the studies of Pinto et al. (2014) for the Atlantic Forest biome, and by Holl (2017) for other tropical areas. Brancalion and Chazdon (2017) also warn that the assessment of the impacts of restoration initiatives on women groups and other vulnerable groups are underestimated in the literature.

Therefore, restoration governance is an emerging field, with methodological, practical and conceptual challenges ahead. Different approaches and scales are addressed in the process for organizing people and institutions for forest and landscape restoration. Understanding that governance models are fundamentally important in the scaling-up of restoration programs and in increasing the effectiveness of each restoration project is thus crucial. Here, we evaluated the governance mechanisms that operate in different restoration initiatives across the Atlantic Forest region in Brazil, identifying the main characteristics and dimensions of the governance structure and highlighting strategies to upscale restoration. We explored this overarching question in different restoration projects that served as case studies and helped us to illustrate the importance of engaging people for restoration success.

## 18.2 Methodology

### Case Studies

In this chapter, we described the governance approaches employed by four large-scale restoration projects at the Brazilian Atlantic Forest: (1) the Brazilian Ecological Restoration Network (national level, all biomes); (2) The Atlantic Forest Restoration Pact (national level, Atlantic Forest biome); (3) Arboretum program (Atlantic Forest of northeast Brazil); and IV) Ecological Corridor (Atlantic Forest of southeast Brazil) (Fig. 18.1). These case studies were intentionally selected because they represent different governance models, objectives and structure. The information about those case studies was shared from their leaderships, which are co-author in this chapter.

### Brazilian Society for Ecological Restoration

In 2010, a group of professionals and researchers in the field of ecological restoration, from across Brazil, gathered to discuss the creation of a support network and locus for debate on how to confront the challenges associated with restoration. In the same year, the Brazilian Ecological Restoration Network (Rede Brasileira de Restauração Ecológica – REBRE), an open, horizontal and non-hierarchical network, was established to expand the synergy of restoration initiatives and deepen the dialogue among the various stakeholders involved in the restoration process. The creation of REBRE was formalized through the publication of its Charter of Principles, available at http://www.rebre.org. Its initial operation comprised a list of

**Fig. 18.1** Location of the four case study areas, Brazil. The National Scale case study (in grey), is represented by the Brazilian Society for Ecological – SOBRE; the Biome Scale case study (in green) is the Atlantic Forest Restoration Pact; and the Regional Scale case studies (in red) are, from north to south, respectively, the Arboretum Program and the Ecological Corridor

email addresses for exchanging information, questions and other subjects and interests shared by the various restoration stakeholders in Brazil. Once the dialogue platform was established, it was possible to move beyond the exchange of experiences and advances towards establishing the necessary basis for the collective construction of knowledge and creating enabling conditions for upscaling restoration in Brazil.

In the early years of operation of the REBRE, the desire of the restoration community to establish more structured dialogue channels (especially a Brazilian Conference on Ecological Restoration) was growing, as was the search by restoration experts for public agencies that could publish technical positions, as the network format is limited in this respect. Thus, during the third REBRE workshop in 2014, the participants founded the Brazilian Society for Ecological Restoration

(*Sociedade Brasileira de Restauração Ecologica* – SOBRE), as registered in the *Antonina Charter*. As a not-for-profit civil association, SOBRE is a scientific, cultural and educational association promoting ecological restoration in Brazil, seeking to expand knowledge in the area, support the training of human resources and contribute to decision-making and public policies. Its main characteristic is its governance model, with an evident bottom-up logic, developed from a base with broad national representation, after a process of collective maturity. Since its creation, it has informed policy through technical advice, when demanded by governments, NGOs and others, both for encouraging new policies, such as offsetting mechanisms for restoration and prioritizing criteria, and for avoiding setbacks such as weakening legislation on restoration. As an institution, with more than 400 associates, it is managed by a board of directors, supported by councillors from every region of the country (see www.sobrestauracao.org).

SOBRE is therefore an institution that brings together restoration stakeholders in Brazil while maintaining a commitment to the collective and democratic principles that guide it, since before its creation, based on dialogue and synergy in the pursuit of economies of scale in restoration actions, with quality and benefits for people and the environment.

**The Atlantic Forest Restoration Pact**

The Atlantic Forest Restoration (hereafter Pact) is a multi-stakeholder platform that aims to restore and reforest 15 million hectares in the Atlantic Forest biome by 2050, integrating the efforts of its members and enabling large-scale restoration with socioeconomic benefits. The Pact is presented in several studies as an example of a process built from the bottom up in a collaborative and dynamic way (Calmon et al. 2011; Crouzeilles et al. 2019; Holl 2017). The initiative is based on a governance model that takes advantage of its more than 280 members from different sectors, active in the chain of restoration and reforestation in the 17 states of the Atlantic Forest biome, thus allowing the engagement of a diverse group of experts acting in working groups (WGs) according to the movement's strategic themes.

In constructing the connections necessary to address gaps, the platform brings together different sectors for the consolidation of a strategy based on an innovative approach. Brancalion et al. (2016) referred the main innovations on the Pact's governance to promote large-scale restoration. Thus, a pioneering Geospatial Working Group was established, which was responsible for preparing the first product of this coalition, a map of potential restoration areas in the Atlantic Forest biome. As a result of this mapping initiative, an initial effort was also made to register the restoration initiatives that have occurred in the Atlantic Forest biome since 2009, because the institutions involved and actions being taken were previously unknown, and this was the first basis for sharing actions and a vision. Currently, this registry of projects is consolidated in an online geospatial platform that allows projects to be registered, as well as queries and cross-checking to be performed with other databases, in addition to monitoring the progress made in areas and their level of quality according to a protocol defined for restoration actions.

Based on these initial surveys, the importance of understanding and managing the efforts being made towards achieving the expected results of restoration initiatives was observed, and a monitoring protocol for assessing the success of these actions was established. The Pact then began to gather experts in the field to consolidate knowledge on the science and practice of restoration. Important publications include the Theoretical Framework of Forest Restoration Concepts and the Monitoring Protocol (Viani et al. 2017). The set of maps, geospatial platform, reference documents and monitoring protocol is what is referred to as the Pact "toolbox", which serves as a system of governance, aligning strategies and establishing a common vision shared by the participating members.

In addition to an internal governance structure composed of a National Coordination, Executive Secretariat and Coordination Council, a network of regional units (RUs) was established and are considered operational bodies of the Pact. These RUs include partner institutions, whose legitimacy and local actions are in accordance with the strategies established by the movement, possessing national influence, to the extent that they identify barriers or opportunities, bringing these issues to the movement and the Pact overarching body. However, forest and landscape restoration not only address environmental aspects but also include a socioeconomic component, considering the various stakeholders and interests within a territory. Thus, the Pact understands that it is essential to consider restoration in a more holistic and innovative manner, integrating many different perspectives in its scope of action. Based on this new perspective, the platform initiated a more inclusive approach, with the Gender and Diversity Working Group (PACTO and IUCN 2017).

**Arboretum Programme**

The Arboretum Programme is a public and inter-institutional programme that operates in the Atlantic Forest biome in the territory of Hileia Baiana (north of Espírito Santo and south of Bahia), whose focus is on the conservation and valorization of forest biodiversity. The proposal of the Arboretum was developed by the Brazilian Forest Service and made feasible by the Public Prosecution Office of the state of Bahia through a conduct adjustment agreement (*Termo de Ajustamento de Conduta* – TAC, in Portuguese) signed by pulp companies, by financing the implementation and maintenance costs of the Arboretum Programme. This TAC allowed a seed money for the construction of the physical structure of the program and the maintenance of basic activities for 10 years. The programme is currently managed by a council composed of representatives from research, outreach and normalization institutions in a structural arrangement that is being replicated in other regions of Brazil and is recognized as a model of public policy, referred to as the Centre for Sustainable Forest Development by the Brazilian Forest Service.

The Arboretum program established a seed network that comprises seven communities and more than 40 active seed collectors, which have sold by 2018 more than five tons of seeds from 393 native forest species. Most of these species had never been priced or marketed before. Currently, their forest nursery has approximately 300 native tree species in production. Among the four nurseries established in local communities, the production focus is species with greater market demand, without abandoning the conservationist approach. This seedling network adds value to the community centres, involving more than 30 nursery workers and has an approximate annual production capacity of 500,000 seedlings per year.

The impact of the programme has important significance for forests and the people living nearby them. The seed collectors and nursery workers who work in the programme experience forests in their daily activities, and based on this relationship, they value forests beyond only an economic context. Establishing a relationship between people with forests is undoubtedly one of the best mechanisms for forest protection and possibly even one of the least widespread, especially in the Atlantic Forest. The Arboretum Programme initiated this process through seeds and seedlings. However, many value niches still require exploration, from the perspective of practical and palpable fundamentals for sustainable forests.

**Ecological Corridor**

Remnant fragments of the Atlantic Forest in the São Paulo portion of the Paraíba do Sul River valley constitute a complex mosaic, composed of pastures and urban areas, whose land-use history has led to the region's forest suitability, which is currently characterized as a strategic region for large-scale forest restoration. In addition to the natural suitability of the region, the 2013–2014 water crisis brought special attention to this region due to its significance to the public water supply of the metropolitan region of Rio de Janeiro and the city of São Paulo, which contribute considerably to the national Gross Domestic Product. As a result, among the initiatives proposed for confronting the water crisis, some public policies and efforts from national and international non-governmental organizations have focused on leveraging numerous forest restoration projects in the region. These projects are motivated not only by the water crisis but also by the international commitments made by Brazil and the state of São Paulo to address the international restoration challenges previously mentioned in this chapter. In this context, in 2009, the Paraíba Valley Ecological Corridor was established, an organization created with the mission of reconnecting these isolated fragments by planting an estimated 6000 hectares of native forest species, thus promoting the reconnection of approximately 150,000 hectares in the region. In its first 10 years, the Ecological Corridor and its partners planted 523,000 native seedlings on 430 hectares in the Paraíba Valley region.

In 2016, the Oikos Institute of Agroecology began a preliminary diagnosis of Paraíba Valley's forest restoration chain, which detected that "regionally, the forest restoration segment is very disconnected and local stakeholders are unprepared to face the challenge and the opportunity to promote large-scale forest restoration. This perception is shared by many regional stakeholders – from seed collectors and seedling producers, to companies that plan and execute projects, and organizations that promote and foster forest restoration in Paraíba Valley" (Andrade et al. 2019). Thus, based on this diagnosis, a popular movement initiated by a group known as "Paraíba Valley Restoration Stakeholders", a network established based on the workshops promoted by the Restoration Opportunities Assessment Methodology – ROAM workshops in the region, conducted by World Resources Institute – WRI

Brazil, led to the beginning of conversations and coordination in favour of synergies for the effectiveness of actions in the region.

In this arrangement, the Ecological Corridor realized that it played a fundamental role in promoting the connection between people of the valley, not only to raise community awareness about the importance of having a forest stand and all the benefits it provides but also to promote dialogue and partnerships between different "valley stakeholders" to seek convergence and synergy between forest restoration actions, the mobilization of rural producers and national and international opportunities and public policies. The Ecological Corridor performs forest restoration activities by engaging the population in this context, helping local governments understand the importance of strategic landscape planning, aligning the actions performed in the region to enhance results. The Ecological Corridor is part of a network of approximately 200 stakeholders, including local, national and international organizations, universities, institutions, producers, nurseries, public authorities and companies. Their actions are led primarily by women, who occupy prominent positions in the governance structure of the projects.

## 18.3 Results and Discussion

Based on these studies and the conceptual frameworks of Mansourian et al. (2014), Mansourian and Rambeloarisoa (2005), Sapkota et al. (2018) and Van Oosten et al. (2014), we developed a matrix to consider the different relevant aspects in the case studies. Thus, the pillars of components, structure and stakeholders were considered, in addition to the scale of action, which is fundamental in the case studies presented. These comparisons are presented in Table 18.1.

The case studies employed different approaches to engage society to expand restoration initiatives. The analyses suggest that such models must be associated with the specific conditions of each landscape, according to their components, structure, stakeholders and scale. Contextual factors also should be included in these analyses, as historical and cultural aspects, as well the understanding of political and socioeconomic issues (Mansourian et al. 2014). The initial conditions, collaborative process and leadership are some of the factors strongly relevant for the results of restoration actions (Ansell and Gash 2008). The initial conditions refer to the existence of conflicts or cooperation that determine the degree of trust between stakeholders.

The initial conditions and context of the related studies are rather heterogeneous among the presented cases because they address different scales and regions. However, in general, defining a common goal and sharing perspectives promoted collaboration among the stakeholders. This favourable condition is not as evident in projects such as the Corridors or in PACT, where it is imperative to equalize divergent interests, such as that of businesses, NGOs and government in determining the application of efforts and resources. This becomes even more critical in the case of the Arboretum Programme, where the origin of the project is associated with com-

**Table 18.1** Matrix of comparison among the different governance models, based on strategies and components of governance

| Case study | Components[a] | Structure[b] | Stakeholders | Scale |
|---|---|---|---|---|
| Arboretum | Economical and social | Governmental | Public Prosecutor's office and Forest Service, businesses, local community, traditional communities, fomented groups, NGOs and local and international universities | Regional |
| Corridors | Social | NGOs | State Department of the environment, businesses, universities, national and international NGOs, landowners | Regional |
| PACT | Science, social, political and economical | Coalition or institutional bricolage[c] | National Coordination Council, regional units and working groups, members, | Biome |
| SOBRE | Science and political | Society of researchers, practitioners and policy makers | National Board, regional Councillors, associated members (persons and institutions) | National |

[a]Representing major focus: Science, Social, Economic, Political (Sapkota et al. 2018)
[b]Relevant bodies that help to organize the initiative (Mansourian et al. 2014)
[c]Institutional Bricolage: a creation of institutional space (Van Oosten et al. 2014)

pliance with a Conduct Adjustment Agreement. The subsequent involvement of the pulp and paper companies in the region indicates that the initial obstacles were overcome, and it was possible to advance a propositional agenda for the region. In these cases, the key role played by the initiatives' leaders is clear, as they are responsible for gathering the best strategies and convincing stakeholders of themes relevant to the landscape.

| Case study | Objective | Context |
|---|---|---|
| Arboretum | The arboretum Programme is a public and inter-institutional initiative focusing on biodiversity conservation and restoration, by promoting seed and seedling network | The proposal of the arboretum was developed by the Brazilian forestry service and made feasible by the public prosecutor of the state of Bahia through a conduct adjustment agreement (TAC) signed by the pulp companies, by financing the implementation and maintenance costs of the arboretum Programme. |
| Corridors | The Paraíba Valley ecological corridor is an NGO that was established with the mission of reconnecting isolated fragments planting an estimated 6000 hectares of native forest species to reconnect 150,000 hectares of isolated fragments | Restoration projects were proposed for confronting the water crisis and to increase the vegetation connection by some public policies and efforts from national and international non-governmental organizations |

| Case study | Objective | Context |
|---|---|---|
| PACT | PACTO is a multi-stakeholder platform that aims to restore 15 million hectares in the Atlantic Forest biome by 2050, enabling large-scale restoration, socioeconomic benefits | To construct the bridges necessary to address gaps of landscape restoration, PACTO brings together different sectors for the consolidation of a strategy based on an innovative approach |
| SOBRE | SOBRE aims to promote ecological restoration in Brazil, expanding knowledge and dialogue, supporting the training of human resources and contributing to decision-making and public policies | A group of professionals and researchers in the field of restoration, from across Brazil, gathered to create a support network and locus for debate on how to confront the restoration challenges, ending up establishing an institutional framework to develop science, subsidize policy and share practice experiences on restoration |

Nevertheless, none of these projects would be successful if platforms for dialogue and collaboration were not established in support of decision-making and public policies. The PACT information and monitoring database is an interesting example in this direction. Based on this tool, it is possible to create scenarios and develop action strategies in a shared manner among the different stakeholders. The construction of dialogue in a consistent and constant manner could be made with social media or by more traditional approach, as meetings and field visits. In the case of SOBRE, an online net is very efficient, but national and regional conferences occur to improve the synergy and change experience and knowledge.

According to the literature assessing governance can be made through different ways and perspectives (Arts 2014; Lemos and Agrawal 2006). Within case studies governance models could be led initially by government agencies, as in the case of the Arboretum Project, or by civil society, as in the case of Corridors, and in both cases the arrangement encompasses many stakeholders. In the example of SOBRE, researchers and practitioners met in a non-hierarchical manner to discuss and advance central themes for restoration. The PACT is considered more like a coalition, or as an institutional Bricolage, with more flexible forms of institutionalization, rather than formal planning structure. The number of members, close to 300 institutions on PACT and 400 associates on SOBRE, indicates the recognition of these initiatives as legitimate representatives of the interests of their members.

The objective or motivation for each initiative is fundamental to get people involved. In the Arboretum programme, building a network for seed and seedling promotes economic benefits and link to that initiative, which result in a more participatory process of restoration and conservation in that region. In the case of the Ecological Corridor, the motivation was more related to a gap and lack of connection on the value chain. In this way, to face the water crisis and gain more native vegetation in the region, institutions joined on effort to recover and reconnect forest remnants. The case of SOBRE was driven by a science-practice concern that was efficiently addressed in a first moment by a platform for sharing knowledge and experience, and in a second moment by periodic conferences.

The observations suggest that the participatory or collaborative approach should be considered in all processes to enhance results. The sharing of this purpose in a clear manner can be decisive for the involvement of stakeholders. Thus, a geospatial platform presents itself as a visual tool for sharing goals and records of accomplishments. The creation of references, knowledge generation and scientific achievements in understanding can be fuel for others. Supporting policies is certainly one of the roles of academia in its full exercise of activities. Awareness and outreach through training and workshop is what motivates the public in a broader way, as the reality and possibilities of effective actions are presented. This broader scope and heterogeneity of situations suggest a more holistic approach at the governance issue, as in Complex Systems Science (Filotas et al. 2014), which presupposes the integration of landscape properties. This perspective indicates that, perhaps, the challenge of understanding governance is grounded in the question ‘What can engage people?’ rather than ‘How to organize people?’ This study suggests that factors such as vision and shared cause, and creation of a sense of identity and belonging can promote this understanding, connecting people and generating their engagement on forest landscape restoration initiatives.

## 18.4 Key Messages

- To foster frank and fair dialogue among stakeholders, successive interactions are necessary through open meetings, where conflicts – when they arise – are openly confronted and negotiated.
- Objectives should not be set a priori, but rather should be defined during the process, throughout meetings and based on the participation and interaction of different stakeholders. Thus, scientific questions are considered but relevant social aspects are also included.
- The strategical planning of restoration should not be restricted to academia, but a network of stakeholders, including academia, can help promoting effective restoration programs. This allowed to credit different stakeholders outside academia as co-authors of the scientific results obtained in this partnership.
- Academic researchers need to move between scientific and political interfaces and agree to work with data that is not necessarily novel for science but highly critical for guiding best practices and policies.
- The representation of various sectors and interests, as well as gender balance in the links of the restoration chain, are relevant issues that can provide further insight into how to obtain greater involvement of various stakeholders in the expansion of efforts for large-scale restoration.

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## References

Adams C, Rodrigues ST, Calmon M, Kumar C (2016) Impacts of large-scale forest restoration on socioeconomic status and local livelihoods: what we know and do not know. Biotropica 48(6):731–744

Andrade A, Bargos D, Fialho T, Cristofaro S (2019) Desafios da cadeia da restauração florestal no Vale do Paraíba Paulista. Soc Nat 30(3):257–277

Ansell C, Gash A (2008) Collaborative governance in theory and practice. J Public Adm Res Theory 18(4):543–571

Aronson J, Milton SJ, Blignautames NJ (2007) Restoring natural capital: science, business, and practice. Island Press, Washington, DC, p 401

Arts B (2014) Assessing forest governance from a 'triple G' perspective: government, governance, governmentality*. Forest Policy Econ 49:17–22

Ball AA, Gouzerh A, Brancalion PHS (2014) Multi-scalar governance for restoring the Brazilian Atlantic Forest: a case study on small landholdings in protected areas of sustainable development. Forests 5(4):599–619

Brancalion PHS, Chazdon RL (2017) Beyond hectares: four principles to guide reforestation in the context of tropical forest and landscape restoration. Restor Ecol 25(4):491–496

Brancalion PHS, Pinto SR, Pugliese L, Padovezi A, Rodrigues RR, Calmon M, Carrascosa H, Castro P, Mesquita B (2016) Governance innovations from a multi-stakeholder coalition to implement large-scale forest restoration in Brazil. World Dev Perspect 3:15–17

Bustamante MMC, Silva JS, Scariot A, Sampaio AB, Mascia DL, Garcia E, Sano E, Fernandes GW, Durigan G, Roitman I, Figueiredo I, Rodrigues RR, Pillar VD, de Oliveira AO, Malhado AC, Alencar A, Vendramini A, Padovezi A, Carrascosa H, Freitas J, Siqueira JA, Shimbo J, Generoso LG, Tabarelli M, Biderman R, de Paiva Salomão R, Valle R, Junior B, Nobre C (2019) Ecological restoration as a strategy for mitigating and adapting to climate change: lessons and challenges from Brazil. Mitig Adapt Strateg Glob Chang 24:1249–1270

Calmon M, Brancalion PHS, Paese A, Aronson J, Castro P, da Silva SC, Rodrigues RR (2011) Emerging threats and opportunities for large-scale ecological restoration in the Atlantic Forest of Brazil. Restor Ecol 19(2):154–158

Crouzeilles R, Santiami E, Rosa M, Pugliese L, Brancalion PHS, Rodrigues RR, Metzger JP, Calmon M, Scaramuzza CA d M, Matsumoto MH, Padovezi A, Benini R d M, Chaves RB, Metzker T, Fernandes RB, Scarano FR, Schmitt J, Lui G, Christ P, Vieira RM, Senta MMD, Malaguti GA, Strassburg BBN, Pinto S (2019) There is hope for achieving ambitious Atlantic Forest restoration commitments. Perspect Ecol Conserv 17(2):80–83

Filotas E, Parrott L, Burton PJ, Chazdon RL, Coates KD, Coll L, Haeussler S, Martin K, Nocentini S, Puettmann KJ, Putz FE, Simard SW, Messier C (2014) Viewing forests through the Lens of complex systems science. Ecosphere 5(1):1–23

Holl KD (2017) Restoring tropical forests from the bottom up. Science 355(6324):455–456

Kusters K, Buck L, de Graaf M, Minang P, van Oosten C, Zagt R (2018) Participatory planning, monitoring and evaluation of multi-stakeholder platforms in integrated landscape initiatives. Environ Manag 62(1):170–181

Lemos MC, Agrawal A (2006) Environmental governance. Annu Rev Environ Resour 31(1):297–325

Mansourian S (2017) Governance and Forest landscape restoration: a framework to support decision-making. J Nat Conserv 37:21–30

Mansourian S, Rambeloarisoa G (2005) Case study: madagascar: developing a forest landscape restoration initiative in a landscape in the moist forest. In: Forest restoration in landscapes: beyond planting trees. Springer, New York

Mansourian S, Aquino L, Erdmann TK, Pereira F (2014) A comparison of governance challenges in forest restoration in Paraguay's privately-owned forests and Madagascar's co-managed state forests. Forests 5(4):763–783

Ostrom E (1990) Governing the commons: The evolution of institutions for collective action. New York, University Press.

PACTO & UICN (2017) Coleção Semeando Equidade – perspectivas de gênero na restauração de paisagens florestais. Pacto pela Restauração da Mata Atlântica: Brasil. Disponível em: http://genderandenvironment.org/resource/semeando-equidade-perspectivas-de-genero-na-restauracao-de-paisagens-florestais/

Pinto SR, Melo F, Tabarelli M, Padovesi A, Mesquita CA, Scaramuzza CA d M, Castro P, Carrascosa H, Calmon M, Rodrigues R, César RG, Brancalion PHS (2014) Governing and delivering a biome-wide restoration initiative: the case of Atlantic Forest restoration pact in Brazil. Forests 5(9):2212–2229

Pistorius T, Freiberg H (2014) From target to implementation: perspectives for the international governance of forest landscape restoration. Forests 5(3):482–497

Sapkota RP, Stahl PD, Rijal K (2018) Restoration governance: an integrated approach towards sustainably restoring degraded ecosystems. Environ Dev 27:83–94

van Oosten C, Gunarso P, Koesoetjahjo I, Wiersum F (2014) Governing forest landscape restoration: cases from Indonesia. Forests 5(6):1143–1162

Viani RAG, Holl KD, Padovezi A, Strassburg BBN, Farah FT, Garcia LC, Chaves RB, Rodrigues RR, Brancalion PHS (2017) Protocol for monitoring tropical forest restoration: perspectives from the Atlantic Forest restoration pact in Brazil. Trop Conserv Sci 10:1–8

Wilson SJ, Cagalanan D (2016) Governing restoration: strategies, adaptations and innovations for tomorrow’s forest landscapes. World Dev Perspect 4:11–15

# Chapter 19
# The Atlantic Forest Trail: Reconnecting People, Biodiversity, and Protected Areas

**Ernesto B. Viveiros de Castro, Andre Monnerat Lanna, Anna Carolina Lobo, Felipe Feliciani, Renata B. Bradford, Jorge Luis do Nascimento, and Carlos E. V. Grelle**

**Abstract** The Atlantic Forest Trail (AFT) is a 4000-km trail under implementation in the Southern and Southeastern regions of Brazil, following the Serra do Mar and extending to the Serra Geral mountain ranges. The trail crosses 90 protected areas that include federal, state, and private lands, as well as several other protected areas at different levels of implementation and management. There are almost 3,000,000 ha of protected areas crossed by the trail, with the potential to form a major corridor for biodiversity protection. The AFT's mission is to promote society's engagement in conservation and recovery of the Atlantic Forest through outdoor activities, while promoting inclusive socioeconomic development and adding value to our natural and cultural heritage. The project is structured in four strategic axes: (1) trail implementation and protected areas strengthening; (2) outreach, people engagement and volunteering; (3) tourism supply chain; and (4) biodiversity and ecological corridors. Governance is organized at regional, state, and local levels and follows the premises of collaborative and participatory management, bringing together protected areas agencies, climbing and hiking organizations, tour operators, and NGOs. Still in the implementation phase, there are over 4,000 registered volunteers, 450 km

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of signed trails and partnerships with scientific research agricultural and tourism institutions.

**Keywords** Ecological corridors · Conservation engagement · Ecotourism · Outdoor activities · Conservation strategies

## 19.1 Introduction

Considered the hottest of the hotspots in the world (Myers et al. 2000; Laurance 2009), the Atlantic Forest hosts a strikingly high biological diversity, being its biodiversity richness per area higher than the Amazon (Thomas et al. 1998; Morellato and Haddad 2000). Recognized as a Biosphere Reserve by UNESCO, the biome was, in the past, one of the largest rainforests in the Americas, reaching Paraguay (Cartes and Yanosky 2003; Huang et al. 2007) and Argentina (Giraudo et al. 2003), with an area of 1,480,000 km$^2$. Currently, less than 28% of the biome original area is preserved (Rezende et al. 2018). Brazilian Atlantic Forest is a mosaic of phytophysiognomies remnants such as evergreen forest (most of all in the coast), semideciduous forest, deciduous forest, mixed forest (Araucaria forest), mangroves, restingas (sandbank vegetation), and "campos de altitude" (Brazilian paramos) along an elevation gradient from 0 to 2891 m a.s.l. (Marques et al. 2021, Chap. 1).

Although there are over 700 protected areas in the biome, they only cover 8.4% of its area (Jenkins et al. 2015) and, usually, these areas are isolated and not large enough to conserve viable populations of many species (Brito and Grelle 2004; Brito et al. 2008; Ribeiro et al. 2009). One of the most important Brazilian mountain ranges is the "Serra do Mar". It is, paradoxically, the most populated area and where the largest remnants of the biome occurred (Ribeiro et al. 2009). Its rugged, steep terrain preserved relatively large natural areas. Maintaining or recovering the connection between these preserved areas is one of the main challenges for the Atlantic Forest conservation.

One hundred and fifty million people live in this biome, which concentrates 80% of the country's Gross Domestic Product (GDP). Nowadays, 90% of people living in the Atlantic Forest are located in urban centers (Scarano and Ceotto 2015). The historical process of urbanization alienated people from natural environments, with negative results for the natural environment and society. In fact, WWF-Brazil conducted a survey that found out that 87% of the people in Southeastern Brazil do not experience nature activities (WWF-Brazil, unpublished data). The disconnection with nature causes negative health effects, which led Louv (2005) to coin the metaphoric term "Nature Deficit Disorder." On the other hand, many studies have shown that outdoor experiences generate health benefits, such as improvement in affection and cognition (Bratman et al. 2015), reduction of the symptoms of attention-deficit hyperactivity disorder (Kuo and Taylor 2004), and even the reduction in the prevalence of myopia in children (Rose et al. 2008).

Nature-based tourism is a fast-growing sector within the tourism industry worldwide (Hugo 1999). Although there is some controversy about its sustainability (Wight 1993), this activity is considered an important tool to promote conservation and an alternative to other land uses that would have worse consequences for the local biodiversity (Broadbent et al. 2012). Studies have shown that people who experience natural environments and protected areas are more likely to support conservation efforts (Formica and Uysal 2001). Besides, nature-based tourism and outdoor recreation in protected areas positively affect human well-being (Naidoo et al. 2019) and local economies (Eagles 2002; Driml 2010; Souza et al. 2018). Thus, reconnecting people to natural environments can boost public health and nature conservation.

Brazil has the largest and most diverse natural environment on the planet (WEF 2017), but the potential for nature-based tourism is under-exploited and, hence, not reflected in large tourist flows. Undeveloped, isolated parks and destinations hinder the development of ecotourism and outdoor recreation in Brazil (Viveiros de Castro et al. 2015). Despite these limitations, nature-based tourism is relevant to local economies in many Brazilian regions and has great potential for growth (Souza et al. 2018).

Trails are one of the most basic elements that connect people and natural environments. From primitive trails for experienced recreationists to structured paths for people with disabilities or elders, trails can be planned and managed for different users and uses (Clark and Stankey 1979). Long-distance trails, which traverse large territories and connect different ecosystems and protected areas, have been proposed as an effective conservation tool on a regional scale since the first decades of the twentieth century (Mackaye 1921). The US Appalachian Trail (AT), for instance, protects at least a 300-meter wide corridor (Potere et al. 2007) and mobilizes thousands of volunteers who are stewards of the trail and surrounding areas (Mittlefehldt 2013). The AT is seen both as a biological corridor and an important recreation tool (Moore and Barthlow 1998). Many other long-distance trails have been implemented worldwide in the last hundred years (Tynon et al. 1998; Heffield 2017).

In Brazil, many paths were developed by the Natives and then later used by European colonizers to explore, occupy the territory and transport products and crops (Capistrano de Abreu 1998). After the advent of railroads in the nineteenth century, most of these trails were abandoned. They began to be explored again for recreation in the early twentieth century by a small group of people grouped in mountain clubs, but it was only in the 1980s that hiking began to gain more popularity. However, in spite of the trails' historical importance in the settlement process in Brazil and their revitalization in the past 30 years, their use for recreation, sports, and tourism is still incipient, especially long-distance trails. In fact, the country has neither a tradition in long-distance trails, nor a well-developed outdoor culture, although this reality is changing.

In recent years, several long-distance trails have been proposed in Brazil by hiking organizations, tourism, and protected areas managers, and the Atlantic Forest Trail (AFT) was the first one over 1,000 km, in 2012. The trail crosses the

**Fig. 19.1** The Atlantic Forest Trail and the main National and State Parks crossed by the trail (State initials: RS - Rio Grande do Sul State, SC - Santa Catarina State, PR - Paraná State, SP - São Paulo State, RJ - Rio de Janeiro State)

Serra do Mar mountain range, totaling more than 4,000 km along the south-southeast coast of Brazil (Fig. 19.1). It has been implemented by protected areas agencies, mountain hiking and climbing organizations, tour operators, and NGOs under the umbrella of WWF-Brazil (Lobo et al. 2017). The project has four main goals: trail implementation and consolidation of protected areas; civil society engagement in conservation; promotion of green business, strengthening the ecotourism value chain; and the increase of the connectivity among the remnants of this biome. This chapter presents the AFT as an initiative that contributes to the reconnection of people, biodiversity, and protected areas, promoting and improving the conservation state of the biome.

## 19.2 The Atlantic Forest Trail

Inspired by the Appalachian Trail, which crosses 3,500 km in the Appalachian Mountains along the East Coast of the USA, the AFT follows the Serra do Mar, a coastal mountain range in Southeastern Brazil (Fig. 19.2). This region comprises the largest remnants of the Atlantic Forest. The trail continues farther south of the range, reaching the canyons of Serra Geral, an area of outstanding beauty and great potential for tourism and recreation, especially due to its proximity to major urban centers.

The trail crosses 10 national parks, 28 state parks, and other 52 protected areas that include federal, state, and private lands, totaling almost 3,000,000 ha

**Fig. 19.2** Hiker in the Pico do Paraná State Park (1,822 m a.s.l.), the highest point of Brazilian South Region. (Photo by: Ernesto V. Castro)

(Table 19.1). The AFT also crosses many municipal protected areas, but they are not included in this study because most of them have not been implemented yet, and are known as "paper parks." There are exceptions of municipal protected areas effectively managed, such as some municipal parks in Florianópolis and Rio de Janeiro but inaccuracies in the decrees of creation and unavailability of data make the analysis difficult and we chose not to consider them in this survey.

Along its thousands of kilometers, hikers cross different ecosystems: tropical rainforest, beaches, mangroves, caves, the coastal plain of "restingas" and "campos de altitude", providing a complete sample of the phytophysiognomies of the Atlantic Forest. The trail also provides cultural experiences opportunities, such as the visitation of traditional small-scale fishermen communities in Superagüi National Park, quilombo villages in the Ribeira Valley, historical towns such as Paraty, many of them recognized as World Heritage Sites.

The southern end of the trail is the Aparados da Serra National Park, from where it heads north crossing five states. It passes less than 50 km from the City of São Paulo, the largest metropolis in Latin America, and crosses the entire City of Rio de Janeiro, where hikers can visit world-known tourist attractions like Corcovado (where Christ, the Redeemer statue is located) and Sugar Loaf. At Três Picos State Park, the trail passes by the highest point of Serra do Mar, Pico Maior de Friburgo at 2,316 m (7,598 ft.), to finally reach its northern limit, at Desengano State Park, in the State of Rio de Janeiro.

**Table 19.1** Federal and State protected areas along the Atlantic Forest Trail

| IUCN category | Category | Level | Number of PAs | Total area (ha.) | Areas (State) |
|---|---|---|---|---|---|
| I | Biological Reserve | State | 1 | 7672 | Aguaí (SC) |
| II | Park | Federal | 10 | 330,178 | Aparados da Serra (RS/SC), Serra Geral (RS/SC), São Joaquim (SC), Serra do Itajaí (SC), Saint-Hilaire/Lange (PR), Guaricana (PR), Superagui (PR), Serra da Bocaina (SP/RJ), Tijuca (RJ), Serra dos Órgãos (RJ) |
|  |  | State | 28 | 940,391 | Serra Furada (SC), Serra do Tabuleiro (SC), Rio Vermelho (SC), Acaraí (SC), Boguaçu (PR), Pau Oco (PR), Pico do Marumbi (PR), Graciosa (PR), Serra da Baitaca (PR), Pico do Paraná (PR), Ilha do Mel (PR), Ilha do Cardoso (SP), Lagamar de Cananeia (SP), Rio Turvo (SP), Caverna do Diabo (SP), Alto Ribeira (SP), Intervales (SP), Nascentes do Paranapanema (SP), Carlos Botelho (SP), Jurupará (SP), Serra do Mar (SP), Restinga de Bertioga (SP), Ilhabela (SP), Cunhambebe (RJ), Ilha Grande (RJ), Pedra Branca (RJ), Três Picos (RJ), Desengano (RJ) |
|  | Wildlife Refuge | State | 1 | 4811 | Serra da Estrela (RJ) |
| IV | Private Natural Heritage Reserve | Federal | 11 | 11,187 | Grande Floresta das Araucárias (SC), Morro das Aranhas (SC), Chácara Edith (SC), Porto Franco (SC), Emilio Battistella (SC), Caetezal (SC), Volta Velha (SC), Fazenda Palmital (SC), Perna do Pirata(PR), Pousada Graciosa (PR), Salto Morato (PR) |
|  |  | State | 13 | 18,508 | Morro da Mina (PR), Fazenda Santa Maria (PR), Rio Cachoeira (PR), Águas Belas (PR), Serra do Itaqui (PR), Hércules Florence (SP), Costa Blanca (SP), Sítio Serra Negra (RJ), Bacchus (RJ), Rio Bonito de Lumiar (RJ), Jardim das Delicias (RJ), Sítio Peito de Pomba (RJ), Refúgio do Bugio (RJ) |
| V | Environmental Protection Area | Federal | 7 | 731,065 | Baleia Franca (SC), Anhatomirim (SC), Guaraqueçaba (PR), Cananeia-Iguape-Peruíbe (SP), Cairuçu (RJ), Guapimirim (RJ), Petrópolis (RJ) |
|  |  | State | 10 | 853,289 | Guaratuba (PR), Serra do Mar (PR), Iraí (PR), Planalto do Turvo (SP), Cajati (SP), Quilombos do Médio Ribeira (SP), Serra do Mar (SP), Tamoios (RJ), Bacia dos Frades (RJ), Macaé de Cima (RJ) |

(continued)

**Table 19.1** (continued)

| IUCN category | Category | Level | Number of PAs | Total area (ha.) | Areas (State) |
|---|---|---|---|---|---|
| VI | National Forest | Federal | 1 | 519 | Ibirama (SC) |
| | | State | 1 | 29 | José Zago (RJ) |
| | Extractive Reserve | Federal | 1 | 1178 | Mandira (SP) |
| | | State | 2 | 2787 | Taquari (SP), Ilha do Tumba (SP) |
| | Sustainable Development Reserve | State | 3 | 3153 | Itapanhapima (SP), Lavras (SP), Aventureiro (RJ) |
| | Ecological Reserve | State | 1 | 9960 | Juatinga (RJ) |
| | | Total | 90 | 2,913,549 | |

State initials: *RS* - Rio Grande do Sul, *SC* - Santa Catarina, *PR* - Paraná, *SP* - São Paulo, *RJ* - Rio de Janeiro

The route follows historical trails, such as ancient muleteers trails at the edges of the canyons, historical paths that link the Paraná Plateau to the coastal plain (Itupava, Graciosa, and Conceição), and the Bandeirantes trails used to transport gold in the eighteenth century in São Paulo. Other trails have been used by traditional communities for centuries, such as the Juatinga and Ilha Grande trails. It also follows trails already used for hiking and other recreational and ecotourism activities: trails that surround the Santa Catarina Island, the Paranapiacaba Continuum Trail, the Transcarioca Trail, and the Petrópolis-Teresópolis Crossing. About 80% of the total route is formed by existing trails or small rural roads.

## 19.3 History and Project Structure

Proposed in 2012 during the Brazilian Congress of Protected Areas, the idea has been developed since 2014 in partnership between federal and state protected areas agencies, hiking/climbing organizations, tour operators, NGOs and volunteers, supported by WWF-Brazil. The trail has been implemented in a democratic and participatory way, so many workshops were organized along the trail to discuss the route, establish principles and guidelines, and engage local partners and volunteers. Its development is based on promoting real commitment toward the Atlantic Forest, using the trail and the contact with natural environments to engage the society in the agenda pro-protected areas.

The governance of the project is structured in three levels: a Coordination Committee responsible for advancing the mission and writing guidelines, which encompasses members of the protected areas agencies at federal and state levels, representatives of national NGOs (such as WWF-Brazil), and the Brazilian Mountain Hiking and Climbing Confederation (CBME), among others; State Committees, which articulate the planning and implementation of the project, formed by protected areas managers, mountain hiking and climbing federations and other partners; and Local Groups, responsible for organizing field works and action, mobilizing local partners and volunteers, and implementing the trail on the ground.

The entire governance structure considers the following principles: (1) give an identity to the AFT as a long-distance trail, creating unity in communication over its 4000 km; (2) recognize, value, and maintain the identity, culture and traditions of each place, section, and group; (3) work synergistically to inspire the autonomy of local groups and make them proud of the project; (4) ensure that local communities feel part and benefit from the project; (5) strengthen the relationships between the stakeholders and also between the public and private sectors; (6) empower civil society in the decision-making process, promoting greater engagement with the Atlantic Forest.

Clearly expressing that the project goes beyond trail implementation, the mission of the Atlantic Forest Trail is to “Engage society in the conservation and recovery of the Atlantic Forest through outdoor recreation and landscape connection along its 4,000 km, promoting inclusive socioeconomic development and valorization of both natural and cultural heritage.” To accomplish this mission, the project is divided into four strategic axes that cover its main objectives, as shown below:

## 19.4 Project Strategic Axes

### *19.4.1 Trail Implementation and Protected Areas Strengthening*

The first strategic axis involves the implementation of the trail itself. The success and sustainability of long-distance trails are strongly related to management arrangements. Models involving partnerships between management agencies, volunteers, and tour operators have shown effective results (Stender et al. 2018).

One of the premises adopted to stimulate engagement was the direct participation of the stakeholders involved in all stages of planning and implementation, including the definition of criteria for route selection, management techniques, and the choice of the trail sign symbol. The precise definition of the route has been validated in workshops with potential partners and local stakeholders, who have a greater knowledge of the terrain. A unique main route has been defined as a spinal cord with many ramifications to important attractions or secondary paths that can be accessed on out-and-back trails. Alternative routes for specific profiles of users (cyclists, canoeists, and riders, among others) can also be considered.

The general guidelines for route definition include: prioritization of passage through protected areas, existing trails, tourist circuits, important attractions, and the presence of complementary activities (diving, wildlife observation, boat trips, etc.); prioritization of areas with communities and landowners interested in the project; existence of overnight accommodation and other support infrastructures; strategic locations for strengthening institutional presence and inhibiting irregular activities, such as areas used by poachers; and important areas threatened by projects that jeopardize the conservation of local biodiversity. Large urban areas, dangerous neighborhoods, and areas with conflicts with private landowners, as well as especially sensitive vegetation and soils are to be avoided. The trail implementation needs to be negotiated with land managers and owners, and passage through territories of traditional communities must be conditioned to their previous consent and interest manifested by them.

Management techniques were proposed in order to standardize as much as possible the level of intervention in the trail, considering users' profiles, management goals, and local characteristics. One of the management actions defined was trailblazing, as consistent, standardized signs are a great way to create the trail's identity, promote emotional connection, and generate support. Thus, close attention was given to the trail symbol, which was designed based on recommendations and proposals gathered during a participatory meeting, and then three options were submitted to an online vote (Fig. 19.3). While we encourage the use of the AFT symbol in all instances, we also value local ethics. There is no imposition, and the AFT symbol can be used by itself or combined with local symbols, empowering local initiatives, and recognizing their importance and traditions.

Another goal of this strategic axis is to support land management agencies in improving visitation management in parks, focusing on the implementation and

**Fig. 19.3** The AFT arrow painted in Ilhabela State Park (SP). (Photo by: Ernesto V. Castro)

maintenance of trail systems, trail blazing, overlooks, and interpretive folders for visitors. This stage includes the prioritization of parks and other protected areas to receive investment, itineraries consultation, and capacity building for protected areas management.

### 19.4.2 Outreach, Engagement, and Volunteering

International experiences show that engaging the community and promoting volunteer work is essential to implement and maintain a long-distance trail (Martinez and McMullin 2004). The AFT is associated to a social movement incubated by WWF-Brazil called Borandá (the agglutination of the expression "let's walk" in Portuguese), which promotes community engagement and partnerships for the protection of the Atlantic Forest. The idea behind Borandá is to promote outdoor activities as a means to raise awareness of the importance of nature conservation, and the AFT is at the heart of this movement. By combining the goals and strategies of Borandá and the AFT, we want to inspire the conservation of our natural heritage, the AFT implementation, as well as the appreciation and enjoyment of trails and protected areas (Cohen 1985).

The main channels of communication, mobilization, and engagement are the websites and social media of both initiatives: Atlantic Forest Trail (http://caminhodamataatlantica.org.br/) and Borandá Movement (http://movimentoboranda.org/). The websites present the projects with useful information for those who want to hike the AFT, attracting interests, promoting actions, and registering volunteers. Partner institutions, such as climbing associations and environmental NGOs mobilize and train the volunteers, who take over many activities, and the highlights are trail maintenance and blazing, as well as the long-term volunteers that oversee the entire project. Finally, the large number of people volunteering in climbing and hiking clubs and organized groups that support the implementation and maintenance of long-distance trails indicates that they can play an important role in engaging people for conservation actions.

### 19.4.3 Tourism Supply Chain

Implementation of trails has been encouraged in many places as a vector of tourism-related economic activities generating income for local communities (Bowker et al. 2007; Stender et al. 2018). Besides supplementing income, these activities can reduce local impacts by representing alternatives to more impactful land uses, such as mining. In a study in Costa Rica, Broadbent et al. (2012) concluded that forests in areas dedicated to ecotourism showed signs of recovery as opposed to areas dedicated to other uses, which resulted in economic and educational gains for the local population.

The tourism supply chain strategic axis strengthens local initiatives, generating complementary income for traditional, rural, and local populations through small businesses and services related to tourism. Room and board services, campsites, rental and sale of equipment, guiding and portering are some possibilities that can be offered and strengthened. The tourism supply chain involves several levels and has the potential to generate a cascade effect on local communities. Souza et al. (2018) estimated that each dollar invested in the Brazilian protected areas system produced seven dollars in economic benefits.

The Coordination Committee approved guidelines for partnerships with tour operators, and the Local Groups have the autonomy to register and encourage new local partners. Partners need to have a sustainable approach to their businesses and value diversity, and the AFT partners with businesses and areas that encourage but do not require the hiring of guides, valuing the local knowledge, services, and the visitors' experience as well. The project offers a recommendation stamp, partners are showcased on the website and social media and get preference in capacity-building programs. Partners should offer benefits to the trail or hikers, such as support to volunteers who manage the trail, project promotion, or discounts for hikers, in a win-win relationship that aims to enable long-term trail maintenance.

The project intends to carry out surveys on the activities along the trail in partnership with municipalities and tourism institutions. Based on this diagnosis, business plans will be developed to guide the initiatives considered as priorities, carefully adjusting the expectations to the demand considering short, medium, and long-term perspectives. Moreover, training will be provided to local partners to improve their service provision, tourism infrastructure, and offer integrated itineraries for the various potential visitors. The project values local initiatives strengthening existing trails and touristic circuits. The close cooperation with the tourism sector is an important premise of this strategic axis.

### *19.4.4 Biodiversity and Ecological Corridors*

Due to the north-south orientation, the Serra do Mar and Serra Geral mountain ranges have altitudinal and latitudinal gradients that are strategic for the conservation of Atlantic Forest species. These gradients make the Serra do Mar a hotspot of species of many groups, such as amphibians, birds, and terrestrial mammals (Figueiredo et al. 2021– Chap. 9).

Many authors argue that north-south forest corridors protect a climate gradient that allows the movement of species to more suitable areas, for both seasonal migrations and in response to global climate change (Halpin 1997; Carroll et al. 2018), even indicating trail corridors as possible migration paths that can lead to changes in species distribution (Hunter et al. 1988). Besides this, Serra do Mar has an altitudinal gradient that varies from sea level to 2,891 a.s.l. allowing the vertical migration of species in response to climate change. The conservation relevance of this region is exceptional because of the high concentration of threatened species of amphibians, birds, and plants in the Serra do Mar (Jenkins et al. 2015; Ribeiro et al. 2018).

These coastal mountains show the largest fragments of the Brazilian Atlantic Forest (Ribeiro et al. 2009; Rezende et al. 2018) and also the lowest rates of mammalian defaunation in the biome (Bogoni et al. 2018). The large remnants of forest in the intermediate regions of the Serra do Mar allow high diversity and abundance of species, including large mammals such as jaguars (*Panthera onca*), tapirs (*Tapirus terrestris*), peccaries (*Tayassu pecari*), and southern muriquis (*Brachyteles arachnoides*). On the other hand, habitat loss and fragmentation at its northern and southern terminus prevent the recolonization of locally extinct species. For instance, the endangered southern muriqui is the largest primate in the Americas and has a restricted distribution, being present predominantly on the Serra do Mar (Strier et al. 2017). Most southern muriquis populations are isolated due to forest fragmentation, especially in the State of Rio de Janeiro (Cunha et al. 2009). These characteristics make the Serra do Mar a hotspot within the biodiversity hotspot of the Atlantic Forest.

The main goal of this strategic axis is to establish a multi-partner collaboration network to monitor and conserve biodiversity, protecting remnants and restoring corridors along the AFT. Therefore, the initiative promotes the engagement of the public sector, research institutes, NGOs, local communities, tourists, and volunteers, making sure the entire community has a voice and is empowered.

One of the planned initiatives is to create a mega-transect to collect climate and biodiversity data to monitor the conservation status of the ecosystem. Furthermore, this mega-transect will help to promote awareness about the connectiveness of the socio-ecological system that is the Biodiversity Corridor along the Serra do Mar and Serra Geral. This idea is inspired by other similar positive experiences, especially in North America and Africa (e.g., McKinley et al. 2019). At various levels, we seek to involve local communities and hikers to participate as citizen scientists, which has been effective in both engaging and generating quality data (Cohn 2008), as well as engage research and government institutions to study socio-ecological large-scale research considering the Serra do Mar biodiversity corridor as a whole. Local Groups could gather data from a 4,000 km line of research stations, monitoring animal movements, climate variables and other data defined by the scientists in a scale that would be very difficult to maintain with conventional research projects and techniques, especially in a scenario of low research budget (Escobar 2019).

Future plans involve the development of a citizen science mobile application as a tool for data collection and investigations involving a large-scale landscape, such as the ecology of the movement of large-displacement species, e.g., jaguar and puma (*Puma concolor*). The development of this vision of the landscape will enable broader and more precise responses to issues related to biodiversity conservation. As a result, we expect to achieve not only robust, scaled up monitoring scheme throughout the mega-transect, but also greater effectiveness of management plans, ecosystem recovery, and creation of public and private protected areas along the AFT.

The last step will be sharing knowledge to engage and educate hikers and the general public, as well as involve decision-makers in conserving this biodiversity corridor. A practical result for conservation will be the proposition for the creation of protected areas between existing ones and restoring the connectivity where there is forest fragmentation. Many sections of the Atlantic Forest Trail were already identified as strategic for the restoration of the Atlantic Forest biome when considering priority factors such as biodiversity conservation, $CO_2$ storage, and the financial cost of restoration (Strassburg et al. 2019).

## 19.5 State of the Art of the Atlantic Forest Trail

Regarding its development and considering that it is a long-term project, significant results were obtained in the first years of the AFT implementation. As previously mentioned, organized, active groups are present along the entire trail overseeing its management, trail blazing initiatives, the definition of the route, and relationship with volunteers, among other activities.

In the State of Rio de Janeiro, the Rio de Janeiro Climbing and Mountain Hiking Federation (FEMERJ) coordinates the trail implementation. In association with federal and state land management agencies five workshops on trail management, including trail blazing, were promoted, involving more than 280 volunteers. The AFT was presented in several events, congresses, and symposiums, such as the Climbing Season Opening Festivals. About 80% of the trail already exists as disconnected traditional, local, and regional trails. Trail blazing efforts were made at Bocaina, Tijuca and Serra dos Órgãos National Parks, and Ilha Grande and Desengano State Parks. Currently, about 190 km in the State are signed, always valuing local traditions and ethics, as well as making our symbol compatible with the local trails. In 2019, a pilot experience with schoolteachers and students in the Serra dos Órgãos National Park has produced trail signs interpreting trails in a participatory way that involved more people in conservation actions. The registration of partners of the supply chain has recently begun, and we already have several campsites, hostels, guides, and other service providers accredited.

In the State of São Paulo, several sections were managed and signed, with groups of volunteers participating in capacity-building workshops and directly acting toward the progress of the project. The northern coast of São Paulo has some sections ready, especially the Ilhabela State Park, where a motivated volunteer group signed the entire local section of the AFT. In the southern region of the State, private properties such as Legado das Águas, a private reserve that promotes ecotourism, were engaged, and managed and signed local sections of the trail. There are about 100 km signed in the State.

The state of Paraná shows good evolution with several sections signed, as well as good engagement of volunteers. The state was responsible for bringing new forms of trail signing techniques and also different ways of approaching communities. The results have been positive and many institutions have approached the project, including NGOs that manage Private Reserves. At the moment there are about 90 km signed in this State.

In Santa Catarina, 70 km of the trail is already signed. Besides the management and trail blazing workshops that happened frequently, the State shows some of the most interesting advances so far. The Agricultural Research and Rural Extension Company of Santa Catarina is engaged in the project as a strategic partner that will implement the project in several cities in the State, with a primary focus on community-based tourism.

Considering the whole project, over 450 km of the 4000 km of the proposed trail were managed and signed in recent years, thanks to the dedicated efforts of more than 30 institutions, which promoted about 25 workshops to define the route, train volunteers on trail management and blazing, and effective work in the field.

The Atlantic Forest Trail was highlighted in the media several times, which, together with the work of the field teams, made the enrollment of volunteers significantly positive, with about 4,000 volunteers enrolled on the website. The Social media presence is also worth mentioning, bringing together nearly 20,000 followers on Facebook and Instagram.

In the biodiversity strategic axis, in addition to the formation of a network of researchers to evaluate and monitor the conservation status of the Serra do Mar and discuss how the project can further this agenda, a cooperative network for large-scale monitoring of the Serra do Mar is being developed using tools of citizen science. One initiative is the partnership with the country's main health research institution, the Osvaldo Cruz Foundation, for the use and promotion of an application and database on wild animals and health (Chame et al. 2015). This application gathers information about problems that may arise from the humans-wildlife interactions, such as yellow fever that has severely affected primate populations in the Atlantic Forest in recent years (Strier et al. 2019). Beyond this original goal, the application can be used to register evidence of wildlife and contribute to monitoring biodiversity along the trail and the corridor.

Despite being recent and still under construction, the AFT was recognized as a good practice in the III Seminar of Good Management Practices for Protected Areas promoted by the Institute of Ecological Research (IPÊ) and ICMBio, in 2017. It is part of the World Trails Network, and was presented at the IUCN World Parks Congress (Sidney – Australia/Dec 2014), the IUCN World Congress (Hawaii – USA/Sep 2016), the World Trails Network Conference (Tottori – Japan/Oct 2016), and the Congress of Protected Areas of Latin America and the Caribbean (Peru/Oct 219).

## 19.6 Perspectives

The Atlantic Forest Trail, as any long-distance trail, is a long-term project, and the expected results should consider this perspective. In the coming years, our goal is to improve governance and implement the whole trail, which may require using rural roads or highways temporarily. One of the goals of the engagement axis is to implement a digital platform to publicize, mobilize, and inform the public about the project and to empower the largest number of local groups with management skills. As for the tourism supply chain, in the medium-term, the perspective is to map local and potential businesses that can turn the trail opportunities into income generation, and have them registered and inserted on the project's database. Finally, the next steps for the biodiversity axis are to identify the priority areas for connectivity and articulate local projects for the implementation of ecological corridors, as well as to set up a monitoring network with social participation, using citizen science tools.

In the long term, our goal is to implement the entire trail considering the diversity of opportunities and users, which should guide decisions about trail blazing and level of intervention. We hope to have the trail recognized as an important outdoor recreation opportunity and conservation strategy as well. On another note, the project aims to support visitation planning and management in protected areas, improving the public use structures, promoting different outdoor experiences, offering access to natural areas in a democratic way, and increasing the number and variety of outdoor opportunities. The engagement axis expects to have a significant part of society emotionally connected to the Atlantic Forest and sensitized to the importance of protected areas and nature conservation, as well as to have engaged volunteers working on the path of the Atlantic Forest. In our vision of the future, the supply chain around the trail will strengthen local businesses, improving user experience and generating income for local communities sustainably. Finally, we expect to expand the connectivity among remnants along the trail, with the designation of public and private protected areas and effective corridors for animals and seeds' flow contributing to protect and recover threatened species in the Atlantic Forest.

Bottom-line, we envision the AFT supported by a wide network of users, volunteers, community and tour operators, and recognized as an important heritage to humankind. Moreover, in the long term, the four combined project axes must contribute to improve the conservation of the Atlantic Forest, with remnants and protected areas conserved and connected, as well as develop and promote diverse recreation opportunities.

## References

Bogoni JA, Pires JSR, Graipel ME, Peroni N, Peres CA (2018) Wish you were here: how defaunated is the Atlantic Forest biome of its medium- to large-bodied mammal fauna? PLoS One 13(9):e0204515

Bowker JM, Bergstrom JC, Gill J (2007) Estimating the economic value and impacts of recreational trails: a case study of the Virginia Creeper Rail Trail. Tour Econ 13(2):241–260

Bratman GN, Daily GC, Levy BJ, Gross JJ (2015) The benefits of nature experience: improved affect and cognition. Landsc Urban Plan 138:41–50

Brito D, Grelle CEV (2004) Effectiveness of a reserve network for the conservation of the endemic marsupial Micoureus travassosi in Atlantic Forest remnants in southeastern Brazil. Biodivers Conserv 13:2519–2536

Brito D, Grelle CEV, Boubli JP (2008) Is the Atlantic Forest protected area network efficient in maintaining viable populations of Brachyteles hypoxanthus? Biodivers Conserv 17:3255–3268

Broadbent EN, Almeyda-Zambrano AM, Dirzo R, Durham WH, Driscoll L, Gallagher P, Salters R, Schultz J, Colmenares A (2012) The effect of land use change and ecotourism on biodiversity: a study of Manuel Antonio, Costa Rica, from 1985-2008. Landsc Ecol 27:731–744

Capistrano de Abreu J (1998) Os Caminhos antigos e o povoamento do Brasil, 2nd edn. Editora Universidade de Brasília, Brasília

Carroll C, Parks S, Dobrowski S, Roberts D (2018) Climatic, topographic, and antrhopogenic factors determine connectivity between current and future climate analogs in North America. Glob Chang Biol 24(11):5318–5331

Cartes JL, Yanosky A (2003) Dynamics of biodiversity loss in the Paraguayan Atlantic Forest: an introduction. In: Galindo-Lean C, Câmara IG (eds) The Atlantic Forest of South America: biodiversity status, threats, and outlook. Island Press, Washington, pp 267–269

Chame M, Barbosa HJ, Gadelha L, Augusto DA, Krempser E, Abdalla L (2015) Sistema de Informação em Saúde Silvestre - SISS-Geo. In: Salgado AC, Motta CLR, Santoro FM (org.). Grandes Desafios da computação no Brasil - Relatos do 3o. Seminário. 1 edn. Rio de Janeiro: Instituto Tércio Pacitti de Aplicações e Pesquisas Computacionais - UFRJ. pp 72–87

Clark RN, Stankey GH (1979) The recreation opportunity spectrum: a framework for planning, management, and research. U.S. Department of Agriculture, Forest Service (General Technical Report), Portland

Cohen JL (1985) Strategy or identity: new theoretical paradigms and contemporary social movements. Soc Res 52(4):663–716

Cohn JP (2008) Citizen science: can volunteers do real research? Bioscience 58(3):192–197

Cunha A, Grelle CEV, Boubli JP (2009) Distribution, population size and conservation of the endemic muriquis (Brachyteles spp.) of the Brazilian Atlantic forest. Oryx 43(2):254–257

Driml S (2010) The economic value of tourism to national parks and protected areas in Australia. CRC for Sustainable Tourism, Gold Coast

Eagles PJ (2002) Trends in park tourism: economics, finance and management. J Sustain Tour 10(2):132–153

Escobar H (2019) Brazilian scientists lament ‘freeze’ on research budget. Science 364(6436):111

Formica S, Uysal M (2001) Segmentation of travelers based on environmental attitudes. J Hosp Leis Mark 9(3–4):35–49

Giraudo AR, Povedano H, Belgrano MJ, Krauczuc ER, Pardiñas U, Miquelarena A (2003) Biodiversity status of the interior Atlantic Forest of Argentina. In: Galindo-Leal C, Gusmão-Câmara I (eds) The Atlantic Forest of South America: biodiversity status, threats and outlook. Island Press, Washington DC, pp 160–180

Halpin PN (1997) Global climate change and natural area protection: management responses and research directions. Ecol Appl 7:828–843

Heffield J (2017) Record numbers on the Te araroa trail [online]. Australas Leis Manag 125:64–65

Huang C, Kim S, Alstatt A, Townshend JRG, Davis P, Song K, Tucker CJ, Rodas O, Yanosky A, Clay R, Musinski J (2007) Rapid loss of Paraguay’s Atlantic forest and the status of protected areas — a Landsat assessment. Remote Sens Environ 106:460–466

Hugo ML (1999) A comprehensive approach towards the planning, grading and auditing of hiking trails as ecotourism products. Curr Issue Tour 2(2–3):138–173

Hunter ML Jr, Jacobson GL Jr, Webb T III (1988) Paleoecology and the coarse-filter approach to maintaining biological diversity. Conserv Biol 2:375–385

Jenkins CN, Alves MAS, Uezu A, Vale MM (2015) Patterns of vertebrate diversity and protection in Brazil. PLoS One 10(12):e0145064

Kuo FE, Taylor AF (2004) A potential natural treatment for attention-deficit/hyperactivity disorder: evidence from a national study. Am J Public Health 94(9):1580–1586

Laurance WF (2009) Conserving the hottest of the hotspots. Biol Conserv 142:1137

Lobo AC, Sagi L, Viveiros de Castro EB, Matheus FS, Meireles T (2017) Borandá social movement and the Atlantic Forest trail implementation. World Trails Mag 3:10–23. Available at https://worldtrailsnetwork.org/magazine/

Louv R (2005) Last child in the woods: saving our children from nature-deficit disorder. Algonquin Books, Chapel Hill

MacKaye B (1921) An Appalachian Trail: a project in regional planning. J Am Inst Architects 9:325–330

Martinez TA, McMullin SL (2004) Factors affecting decisions to volunteer in nongovernmental organizations. Environ Behav 36(1):112–126

McKinley PS, Belote TR, Aplet GH (2019) An assessment of ecological values and conservation gaps in protection beyond the corridor of the Appalachian Trail. Conserv Sci Pract 1:e30

Mittlefehldt S (2013) Tangled roots: the Appalachian Trail and American environmental politics. University of Washington Press, Washington, 255pp

Moore R, Barthlow K (1998) The economic impacts and uses of long-distance trails. USDI National Park Service, Washington, DC

Morellato LPC, Haddad CFB (2000) Introduction: the Brazilian Atlantic Forest. Biotropica 32(4b):786–792

Myers N, Mittermeier RA, Mittermeier CG, Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Naidoo R, Gerkey D, Hole D, Pfaff A, Ellis AM, Golden CD, Herrera D, Johnson K, Mulligan M, Ricketts TH, Fisher B (2019) Evaluating the impacts of protected areas on human well-being across the developing world. Sci Adv 5:eaav3006

Potere D, Woodcock C, Schneider A, Ozdogan M, Baccini A (2007) Patterns in Forest clearing along the Appalachian Trail Corridor. Photogramm Eng Remote Sens 73(7):1–9

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Strassburg BBN, Tabarelli M, Fonseca GA, Mittermeier RA (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16(2018):208–214

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

Ribeiro BR, Martins E, Martinelli G, Loyola R (2018) The effectiveness of protected areas and indigenous lands in representing threatened plant species in Brazil. Rodriguesia 69:1539–1546

Rose KA, Morgan IG, Ip J, Kifley A, Huynh S, Smith W et al (2008) Outdoor activity reduces the prevalence of myopia in children. Ophthalmology 115(8):1279–1285

Scarano FR, Ceotto P (2015) Brazilian Atlantic forest: impact, vulnerability, and adaptation to climate change. Biodivers Conserv 24:2319–2331

Souza TVSB, Thapa B, Rodrigues CGO, Imori D (2018) Economic impacts of tourism in protected areas of Brazil. J Sustain Tour 26(1):1–15

Stender K, Sanders D, Dowling R (2018) Sustainable long-distance trail management: international perspectives. Tour Anal 23:365–376

Strassburg BBN et al (2019) Strategic approaches to restoring ecosystems can triple conservation gains and halve costs. Nat Ecol Evol 3(1):62–70

Strier KB et al (2017) Demographic monitoring of wild muriqui populations: criteria for defining priority areas and monitoring intensity. PLoS One 12(12):1–14

Strier KB, Tabacow FP, de Possamai CB et al (2019) Status of the northern muriqui (Brachyteles hypoxanthus) in the time of yellow fever. Primates 60(1):21–28

Thomas WM, Carvalho AMV, Amorim AMA, Garrison J, Arbeláez AL (1998) Plant endemism in two forests in southern Bahia. Braz Biodiver Conserv 7:311–322

Tynon JF, Harding JA, Chave DJ (1998) National recreation trails: an overview. In: Vogelsong HG (ed.) Proceedings of the 1997 Northeastern Recreation Research Symposium; 1997 April 6–9; Bolton Landing, NY Gen Tech Rep. NE-241

Viveiros de Castro EB, Souza TVSB, Thapa B (2015) Determinants of tourism attractiveness in national parks of Brazil. 2015. PARKS. Int J Protect Areas Conserv 21(2):51–62

WEF (2017) The travel & tourism competitiveness report 2017. World Economic Forum, Geneva, Switzerland. Available at https://www.weforum.org/reports/the-travel-tourism-competitiveness-report-2017

Wight P (1993) Ecotourism: ethics or eco-sell? J Travel Res 31:3–9

# Chapter 20
# Conservation Initiatives in the Brazilian Atlantic Forest

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**Abstract** With a wide distribution across eastern South America, the Brazilian Atlantic Forest is a mosaic of lowland and montane vegetation types, such as evergreen forest, semideciduous and deciduous forest, mixed forest (e.g., *Araucaria*), mangroves, and restingas. It has long been recognized as having one of the most diversified biotas on the planet, with high levels of endemism of plants and animals. Due to its location, European colonization and exploration began six centuries ago in the coastal areas, spreading to the interior and increasing over the last 70 years.

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In response to long-standing deforestation, many conservation actions have been planned and performed by federal and state governments, NGOs, and universities. Here, we compiled some of these initiatives, showing the conservationists' goals and multi-institutional actions to save species and ecosystems across the Brazilian Atlantic Forest. Furthermore, we confirmed here that with a dialogue among government, NGOs, and universities, it is possible to design and perform actions to the conservation of the Brazilian Atlantic Forest.

**Keywords** Action plans for conservation · Landscape connectivity · Protected areas · Reintroduction · Spatial analysis of prioritization · Threatened species

## 20.1 Introduction

Two centuries ago, the Prussian explorer and naturalist Carl F. P. Von Martius launched the term dryad to describe the splendid evergreen forest that occurs along the coast of Brazil. The name dryad, in honor of a Greek nympha, represents how von Martius saw the splendid Brazilian Atlantic Forest. Nowadays, after studies of phytogeographers and botanists, such as Helmut Hueck and Henrique P. Veloso, we know that the Brazilian Atlantic Forest is a mosaic of evergreen forest (mostly along the coast), semideciduous forest, deciduous forest, mixed forest (*Araucaria* forest), mangroves, and "restingas." Oliveira-Filho and Fontes (2000) described a gradient of species composition but definitively stated that all these phytophysonomies formed a unique biome known as Brazilian Atlantic Forest. The limits of the Brazilian Atlantic Forest are hard to unravel and depend on the approach and question to be answered (Muilaert et al. 2018).

With around 1,200,000 km$^2$ of extension and dynamic climatic fluctuations during the Quaternary era, the Brazilian Atlantic Forest harbors an impressive number of species – many of them endemic to the Biome – of trees (Zwiener et al. 2021), epiphytes (Ramos et al. 2021), social insects (Feitosa et al. 2021), fishes and aquatic invertebrates (Padial et al. 2021), and tetrapods (Figueiredo et al. 2021). Human exploration of Brazilian Atlantic Forest began in the sixteenth century but increased in the last 50 years (Fonseca 1985; SOS Mata Atlântica and INPE 2017), and this biome is recognized as a world biodiversity hotspot (Myers et al. 2000). Nowadays, this biome remains as one of the most deforested areas in Brazil with the remaining vegetation cover comprising 28% of the original (Rezende et al. 2018) distributed mainly in small and isolated fragments, where long-term survival without direct human intervention is controversial (Fundação SOS Mata Atlântica, INPE and Instituto Socio Ambiental 1998).

Due to this long-standing deforestation and its expected consequences on plants (e.g., Lima et al. 2015) and vertebrates (e.g., Brooks et al. 1999; Grelle et al. 1999, 2005), the Brazilian Atlantic Forest is the place of many conservations initiatives, including the creation of national and state protected areas (categories I to VI of

IUCN), private protected areas (RPPNs in the Portuguese acronym), corridors, and actions plans for protected species (e.g., Galindo-Leal and Câmara 2003; Pinto et al. 2006; Rocha et al. 2006; Joly et al. 2014). Furthermore, some analyses of spatial prioritizations were already performed in the Brazilian Atlantic Forest using a systematic conservation planning concept, although with few taxa (Pinto and Grelle 2009; Loyola et al. 2014) or focusing on restoration ecology (Crouzeilles et al. 2015; Zwiener et al. 2017; Strassburg et al. 2019).

Herein, we compiled some of the pioneering and large-scale initiatives for the conservation of the Brazilian Atlantic Forest, including initiatives and studies of the Brazilian government, NGOs, and universities. The initiatives described in this chapter allow the implementation of conservation strategies to extensive biodiversity threats such as changes in land use and land cover (Lira et al. 2021 – Chap. 11), climate change (Vale et al. 2021 – Chap. 12), and defaunation (Galetti et al. 2021 – Chap. 14). Note that along with this chapter, we have not included a comprehensive list of all conservation initiatives, but a selection done by the first author following criteria of potential application for all the Brazilian Atlantic Forest.

## 20.2 Priority Areas for Biodiversity Conservation in the Brazilian Atlantic Forest

In 2017 and 2018, the Brazilian Ministry of the Environment coordinated the second update process to establish "Priority areas for Conservation, Sustainable Use and Benefit Sharing of Brazilian Biodiversity." The establishment of priority areas for biodiversity conservation is a public policy designed to support decision-making in the planning and implementation of conservation measures. These areas are used to direct protected area siting, licensing, and inspection of activities that may cause significant environmental harm and to foster the sustainable use of the nation's territory. Guidelines for the identification of priority areas and actions were established in Brazil by Decree No. 5092 of May 21, 2004 (http://www.planalto.gov.br/ccivil_03/_Ato2004-2006/2004/Decreto/D5092.htm) within the scope of the Ministry of the Environment's responsibilities.

Like previous efforts, the second update process of priority areas was carried out respecting the limits of biomes in Brazil. For the Brazilian Atlantic Forest biome, the process was carried out with the support of a Brazilian conservation think tank and nongovernmental organization, the Instituto de Pesquisas Ecológicas (IPÊ), which was selected through a public bid to conduct the priority areas process.

Beginning with the first updating of priority areas, done in 2006–2007, the process has been based on the use of the systematic conservation planning methodology (Margules and Pressey 2000). This method seeks to identify and select a set of priority areas for the conservation and sustainable use of different features of biodiversity and conservation targets, such as species, habitats, landscapes, ecosystem services, and ecological processes. This approach selects areas considering not only biological and environmental criteria but also anthropogenic variables that may affect the conservation of biodiversity. The objective is to establish a system of areas that contribute to the achievement of conservation goals while considering socioeconomic constraints.

The process follows some key principles, including (i) representation, wide representation of biodiversity components; (ii) functionality, preservation of conservation objects in the long term while maintaining their viability and ecological integrity; (iii) efficiency, maximum biodiversity protection through a solution that achieves conservation goals with the best possible cost-benefit; (iv) complementarity, consideration of existing protected areas so as to maximize the total number of protected objects when adding new areas; (v) flexibility, achieving conservation goals by various combinations of priority areas; and (vi) irreplaceability, identification of areas indispensable for achieving conservation goals.

### *20.2.1 Participatory Process*

Systematic conservation planning is meant to be a highly participatory process, one that includes the opportunity for feedback, revision, and iteration where needed (Margules and Sarkar 2007). The process consists of six stages (Margules and Pressey 2000):

1. Measure and map biodiversity: an extensive review of existing data locates the most suitable datasets to represent biodiversity, identifying similarities and differences among areas. These data can include the locations of species, species assemblages, habitat types, or other desired conservation targets.
2. Identify conservation goals for the planning region: the overall objectives of the process (representativeness and persistence) are translated into quantitative goals for species, vegetation types, and other important features in the planning region.
3. Review existing conservation areas: analyze how much of the goals set in stage 2 are already achieved by existing conservation areas and so what the gaps are.
4. Select additional reserves: with the information on the gaps in the existing conservation areas, new areas are identified to achieve the set of goals established in stage 2. This identification is usually made using selection algorithms or decision-support software considering constraints such as costs, opportunities, and the land use of the planning area.
5. Implement conservation actions: the set of conservation actions to be implemented in each individual priority area is decided.
6. Management and monitoring of reserves: as in stage 2, the management must be monitored based on goals and targets. This monitoring requires the definition of indicators that reflect the success of the conservation actions and drive adaptive management with a continued review of proposed actions.

The second update of priority areas for the Brazilian Atlantic Forest focused on stages one to five and proceeded in the following five broad steps.

### 20.2.2 Step I: Assessment of the Previous Priority Area Update

The first step's aim was to evaluate the results generated and impacts achieved with the first updating of the priority areas of the Brazilian Atlantic Forest biome, published by MMA Administrative Ruling No. 09/2007. This included a public consultation of 229 representatives from government agencies, educational and research institutions, civil society organizations, and businesses, among others.

The first priority areas revision, done in 2007, indicated a total of 880 priority areas for biodiversity conservation in the Atlantic Forest, distributed over 428,409 km$^2$. This corresponded to 37.9% of the biome, with 30.6% of the areas lacking protection and 7.3% of areas somehow protected by conservation units or indigenous lands. According to the public consultation, the results of this process were adequate or satisfactory for several activities, including directing research, projecting further planning, creating new protected areas, and directing financial resources, although there was a need for improvement.

In the period between 2007 and 2018, 551 new protected areas were created that have their limits partially or totally overlapping the priority areas established in 2007. This corresponds to 9239 km$^2$ of protected areas established in areas identified as priorities. Only 240 protected areas were created outside the priority areas, corresponding to 4460 km$^2$. This is about half the number and extent compared to areas that intersect priority areas, demonstrating a positive effect of this prioritization and an advance in biodiversity conservation in the Brazilian Atlantic Forest over the last 10 years.

### 20.2.3 Step II: Definition of Targets and Goals

The second step was to gather and systematize spatial data on potential conservation targets. This was followed by a consultation of experts during the "Workshop for the Definition of Targets and Goals for Biodiversity Conservation," held in Atibaia, São Paulo, between April 11 and 13, 2018. The workshop was attended by 40 participants, mainly specialists in various taxonomic groups. During the workshop, the specialists chose the best databases to use for the analysis and defined the specific targets and goals for biodiversity conservation.

To be selected, the targets had to represent the overall biodiversity and have adequate spatial data across the Brazilian Atlantic Forest. Targets chosen included birds, reptiles, amphibians, fish, mammals, and datasets related to plants. Additional species information used was the status of endangerment, whether species had restricted distributions or were endemic to the Brazilian Atlantic Forest, the level of evolutionary distinctiveness, the presence of rare functional traits, overall rarity (for plant species), and plants used for extractive purposes. In addition, conservation targets related to speleological heritage, vegetation physiognomy, and ecosystem services were considered as targets.

For all targets, quantitative conservation goals were assigned using a range of criteria to establish how much of the distribution of each target should be conserved. Among the criteria considered were the vulnerability or biological importance of the target, as well as its spatial distribution (broad or restricted). In total, there were more than 2500 conservation targets selected, distributed in these groups: mammals, birds, reptiles and amphibians, fishes, humid environments, caves, plants, and targets related to ecosystem services and vegetation physiognomies. This was more than 10 times the number of targets considered compared to the last update of the priority areas and was a much wider diversity of targets.

### *20.2.4 Step III: Definition of the Cost Layer*

The third step included the compilation of available spatial data on economic activities or biophysical conditions that are potentially harmful to the environment or somehow incompatible with biodiversity conservation. This step was part of the process of constructing what is known as a cost surface, which would enter later analyses jointly with the biodiversity data. The cost surface indicates areas with relative difficulty to implement conservation and sustainable actions. Such information helps in choosing priority areas while considering both the reduction of conflicts with the productive sector and the achievement of the biodiversity conservation goals.

The data organizing process was followed by a consultation with specialists and representatives of several economic sectors operating in the biome. The “Workshop for Cost Analysis for the Conservation of Atlantic Forest Biodiversity” took place in the city of Florianópolis, Santa Catarina, between June 19 and 21, 2018. It was attended by 45 participants, including experts and representatives of economic sectors that develop activities in the biome.

Of the variables that contributed to the definition of the cost surface, standing out in descending order (from higher to lower incompatibility): urbanization, major ports, refineries, potentially polluting industrial activities, paved roads, agriculture, airports, petroleum distribution terminals, aquaculture, reservoirs (hydropower reservoirs and others), unpaved roads, pasture areas, thermoelectric power generation, wind power generation, mining-related dams, hydrocarbon exploration wells, railways, plantation forestry, smaller ports, hydropower units, biodiesel production, production of ethanol, transportation waterways, power transmission lines, gas pipeline, solar energy capture, and production of biogas.

### *20.2.5 Step IV: Definition of the Opportunity Layer*

The fourth step was the creation of an opportunities layer for conservation and sustainable use, including the production of a georeferenced database of the information. The opportunity layer represents the information about activities and uses of the territory that are more compatible with biodiversity conservation and that promote or facilitate the maintenance of areas for conservation. The "Workshop on Opportunities for the Conservation of Atlantic Forest Biodiversity" took place in Porto Seguro, Bahia, between August 27 and 29, 2018, with 59 participants. Among them were a series of professionals and representatives of groups and institutions working with Brazilian Atlantic Forest conservation, sustainable use, and indigenous and traditional peoples.

Among the variables contributing to the definition of the opportunities layer, standing out in descending order (from higher to lower favorability), is highlighted: the presence of state and municipal conservation units – not yet officially included in the National Register of Conservation Units (CNUC), the proportion of natural vegetation remaining, the presence of restricted use areas and permanent preservation areas (APPs in Portuguese) legally defined based on slope and altitude, landscape connectivity index, governmental proposals for the creation of new protected areas, the presence of quilombo communities and indigenous lands, environmentally differentiated settlements, ecological corridors, birdwatching tourism, and long-distance trekking trails, among others.

The cost and opportunity layers were then integrated to produce the final cost surface, which entered as an input in the analyses to define the priority areas. To join these individual layers, the opportunity layer (resulting from the sum of individual opportunity variables) was subtracted from the costs layer (resulting from the sum of individual cost variables). To avoid a higher weight for one layer in detrimental of the layer, the data layers were each rescaled during the process to vary between 1 and 10.

It is worth noting that other costs and opportunities can be imagined for the Brazilian Atlantic Forest. However, the selection of variables depended on the availability of adequate spatial data that met minimum criteria:

1. Coverage throughout the biome.
2. Needed information in metadata and attribute tables.
3. Accuracy of the location of occurrence.
4. Lack of redundancy with other variables.

### 20.2.6 Step V: Definition of Priority Areas and Actions for the Conservation of Brazilian Atlantic Forest Biodiversity

The fifth step, the "Workshop on the Definition of Priority Areas and Actions for the Conservation of Atlantic Forest Biodiversity," was held in Brasília-DF from November 6 to 8, 2018, with 79 participants. They represented 40 institutions linked to educational and research sectors, governmental and nongovernmental agencies, private initiatives, indigenous and traditional peoples, and expert consultants.

Based on the systematic conservation planning approach and the use of the Marxan software, a map of proposed priority areas for biodiversity conservation in the Brazilian Atlantic Forest was presented. With the contribution of workshop participants, it was further refined and adjusted, synthesizing the four previous steps into one final layer. This represented the best set of areas to meet conservation goals, regarding landscape permeability and connection and reduction of conflicts with productive activities. During the workshop, the participants also discussed the best set of actions to recommend for each priority area.

## 20.3 Outcomes of the Participatory Process

In December 2018, MMA Ordinance No. 463 of December 18, 2018, was published, recognizing the new priority areas for the conservation, sustainable use, and sharing of benefits of Brazilian biodiversity (Fig. 20.1). The maps, databases, and associated information were made publicly available for consultation on the MMA website: http://areasprioritarias.mma.gov.br.

Priority areas for the Brazilian Atlantic Forest biome covered a total of 246,893 km$^2$, representing 22.3% of the biome and a reduction compared to 2007, which selected 346,191 km$^2$ (30.6% of the biome), excluding areas already having

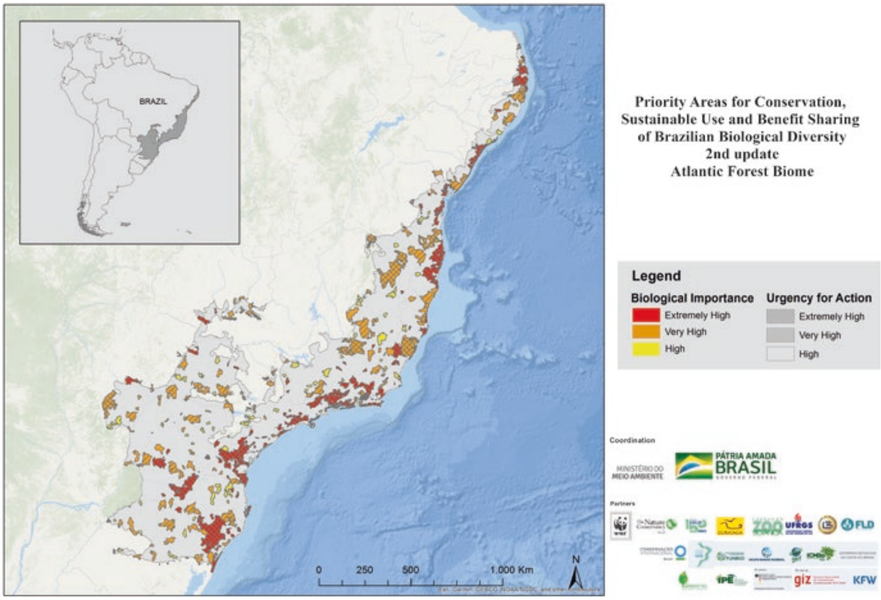

**Fig. 20.1** Priority areas for Conservation of Brazilian Atlantic Forest biodiversity, recognized by MMA Ordinance No. 463, of December 18, 2018

protection. This decrease is a result of an exercise that used more consistent databases and georeferenced information, as well as the use of the Marxan system that allowed the selection of areas that, although smaller in number and area, managed to reach 99.5% of the goals for the more than 2500 targets chosen.

Actions and measures that were most frequently cited as the main conservation action for priority areas were limitation/regulation of degrading activities, recovery of degraded areas, integrated and participatory management of protected areas, establishment of ecological corridors and recognition of territories of traditional peoples and communities, creation of conservation units, creation and strengthening of territorial management, measures targeting the protection of species, and monitoring, inspection, and control of illegal activities (e.g., deforestation, hunting, predatory fishing).

Going forward, it is recommended that the priority areas and recommended actions be periodically reviewed by MMA, within a period not exceeding 5 years. During this period, the MMA will disseminate the instrument to subnational governments such as state and municipal regulators, as well as other public and private bodies and institutions that work with biodiversity conservation. The aim is to guide specific actions that can be effectively implemented, achieving results and not just being recommendations on paper.

Throughout the updating effort, the careful elaboration of spatial databases and detailed records of the priority areas process brought more reliability to the results. Now, it is possible to understand which elements have stood out in the choice of each priority area. In addition to greater transparency, this allows the tool to be incorporated for use in other environmental management actions, such as prioritization of research and licensing of development or environmental activities. The extensive involvement of many segments of society in this search for better conservation strategies was a major contribution. Such broad involvement makes the results more likely to be viable for practical application in one of the most threatened biomes on the planet.

## 20.4 The Brazilian National Centre for Flora Conservation (CNCFlora in the Portuguese Acronym)

The colossal Brazilian plant biodiversity imposes overwhelming difficulties in terms of setting up coordinated efforts regarding the implementation of effective conservation actions. This question becomes even more complex to overcome when it is clear that the amount of newly described species annually in a given country is similar to the number of species that have their extinction risk assessed, and ways to scale up in the detection of species at risk sharply contrast with the ever-increasing rate of natural ecosystems depletion. Although this conundrum remains only partially addressed, two major steps of effective conservation strategies are (i) knowing the identity of *foci* subjects, e.g., name and taxonomic description of all organisms in a given area, and (ii) estimating the extinction risk of the present organisms using the best information available in order to deliver sound conservation actions.

Brazil is a signatory of the Convention on Biological Diversity (CBD) and follows the Global Strategy for Plant Conservation (GSPC) objectives to set integrated conservation actions to guarantee the perpetuation of plant diversity in the country. Acknowledging the need to identify and detect which plants are facing elevated risk of extinction, the GSPC defined 16 essential targets to put forward conservation actions and bring back plant species from the brink of extinction. Among these, Target 1 calls for “a widely accessible working list of known plant species as a step toward a complete world flora,” while Target 2 calls for “an assessment of the conservation status of all known plant species, as far as possible, to guide conservation action.” Nationally, Target 1 is almost complete due to efforts undertaken to prepare the list of species of the Brazilian flora (2008–2015) and the Brazilian flora 2020 (2016–present). Knowledge of plant conservation status is currently available because of the establishment in 2008 of the National Centre for Flora Conservation (CNCFlora).

CNCFlora was created within the infrastructure of the research Institute of Rio de Janeiro’s Botanic Garden (JBRJ in Brazil acronym) that is a federal autarchy subordinated to the Brazilian Ministry of the Environment, with the purpose of conservation of the Brazilian flora. Its main responsibility is to provide, to the Ministry of the Environment and other governmental agencies, technical and scientific information in order to support decisions related to the conservation of the Brazilian flora. Since its creation, CNCFlora supported REFLORA project in its aim to issue a new list of the Brazilian flora – replacing the previous one, *Flora Brasiliensis*, from 1908 (BFG 2018) – mapping out almost 46,000 species in the country, including all terrestrial plants (angiosperms, gymnosperms, ferns, and lycophytes and bryophytes) and algae and fungi species (Flora do Brasil 2020 *in prep*).

Additionally, the center assessed to date the extinction risk of over 6830 terrestrial plant species. Outcomes of these efforts resulted in the update of the National Red List, where 2113 plant species are considered threatened and therefore are subject to specific regulations by the Brazilian government (MMA 2014).

In summary, CNCFlora’s mission is to (i) prevent the extinction of species of the Brazilian flora and coordinate efforts to save those detected as threatened (CR, EN, VU), (ii) meet international commitments assumed by the country in the CBD through the GSPC, and (iii) meet the national goals expressed in the Brazilian Biodiversity Policy. In practice, CNCFlora is responsible for the preparation of red lists and action plans (National Action Plans) of Brazilian plants, as well as the coordination of ex situ conservation efforts and the execution of scientific expeditions in priority areas for plant conservation and research.

Although the national scope of the center’s work, many initiatives focusing on the Brazilian Atlantic Forest were conducted in previous years. Considering the importance of this iconic biome of both regional and global conservation relevance, here, we synthesize the main activities of CNCFlora for the conservation of the Brazilian Atlantic Forest flora, targeting initiatives that the organization directly or indirectly conducted or promoted in order to secure a healthy destiny for unique flora assemblages contained in this biome and its marginal ecosystems.

The incredible plant diversity in the Atlantic Forest is revealed by substantial increase in knowledge verified in recent years, as since 2010, at least 1471 seed plant species have been added to the national list (Martinelli and Moraes 2013). From this rate, it is evident that many areas within the Atlantic forest remain poorly botanized, and some highly diversified taxonomic groups still neglected from robust scientific investigations (Sobral and Stehmann 2009). Many additions to this ever-growing species list are constantly being made, and nowadays, the most up-to-date tally indicates the occurrence of nearly 18,200 terrestrial species occurring in the biome, of which 8720 (48%) represent single-biome endemics (Flora do Brasil 2020 *in prep*). However, regarding the whole flora conservation status detection and actions to save them, we still have a long way to go.

Over the last 8 years, 19% of the flora inhabiting Brazil had its conservation status assessed by CNCFlora. These assessments deal with plants distributed throughout the whole country and across all Brazilian biomes and were assessed in different initiatives: the first Brazil's Flora Red Data book comprised reassessments of 4617 species previously included in regional red lists published by governmental or nonprofitable agencies (Martinelli and Moraes 2013) and is the baseline of the Brazilian Official List of Threatened Species (MMA 2014), from which 1212 (57%) are (not exclusively) Atlantic Forest species.

Martinelli et al. (2018) also assessed the conservation status of 884 plant species endemic to the state of Rio de Janeiro, thus occurring inside the biome's limit. Additionally, risk assessments were also carried out for newly described species within the biome, and lastly, risk assessments of 800 endemic trees were conducted in a partnership with the Global Tree Assessment (GTA) from the Botanic Garden Conservation International (BGCI), out of which 406 (51%) were pointed as Atlantic Forest species. All assessments which are not yet included in the National Red List (MMA 2014) are being submitted constantly toward the Ministry of Environment and will eventually be included in such a crucial normative tool.

As expected, the Brazilian Atlantic Forest had the highest number of species assessed to date, followed by the Cerrado. From almost 18,200 plant species occurring in the Atlantic Forest, 4350 species have their risk assessments undertaken by CNCFlora. From these, 1875 (43%) are considered threatened, and 1675 (38%) correspond to single-biome endemics. From those identified under a given threat category, 415 (22%) are considered critically endangered (CR), 1034 (55%) endangered (EN), and 426 (23%) deemed as vulnerable (VU). Finally, a great deal of species (702% and 16%) were classified as data deficient (DD), as there is not enough information available to robustly define its conservation status. Further studies regarding DD species distribution, population trends, and threats are critical to assess its extinction risk, hence promoting the implementation of accurate conservation measures for species which are frequently disclosed as threatened once further data come out to light (Bland et al. 2017).

### *20.4.1 Priority Areas for Plant Conservation and National Actions Plans*

The implementation of tailored conservation actions is essential to be continued to ensure effective conservation of threatened plants. Conservation planning in Brazil must be compatible with the current socioeconomic reality, considering the great territory, the high biological diversity, the presence of key areas for regional and/or global conservation, and the limited availability of human and financial resources to act directly in the design, implementation, and monitoring of actions (Loyola et al. 2014). Thus, the designation of priority areas for threatened plant species conservation is an efficient tool to select those areas in which conservation actions targeting endangered flora is most urgently needed.

In this scenario, CNCFlora conducts the process of elaboration, implementation, and monitoring of national action plans for the conservation of endangered flora. The design of such a guiding tool can be planned upon different planning units (from species-specific to relevant taxonomic groups or even for entire ecosystems and territories), but usually, a territorial approach is applied as it is among the most cost-effective mechanism to protect simultaneously several threatened species and their habitats. By this means, this approach reduces financial costs and enables the development and implementation of actions consistent with combating and mitigating threats affecting populations of multiple species.

Adopting as a planning unit of micro-watersheds, an initiative was conducted in 2014 by CNCFlora and Biogeography Laboratory of Goiás University with a total of 70 micro-watersheds within the Atlantic Forest which are a priority for plant conservation, where 27 areas are designated as with extremely high priority, 21 with very high priority, and 22 with high priority. The 27 extremely high-priority micro-watersheds are located mainly in the state of Rio de Janeiro and Espírito Santo, as well as off the coastline of São Paulo, a portion in Bahia, and in the transition region between the Atlantic Forest and the Cerrado hotspots in Minas Gerais. In total, priority areas for conservation of endangered flora correspond to 15.7% of the total area of the Atlantic Forest, highlighting the pertinence of these ecosystems for full conservation actions to be taken (Loyola et al. 2014).

Considering the importance to conserve Rio de Janeiro's flora as a whole due to its high concentration of priority areas for plant conservation within the Atlantic Forest, CNCFlora conducted, in partnership with the State Environmental Secretary (SEA), the conservation assessments of all Rio de Janeiro State's endemic plants (Martinelli et al. 2018), also ranking areas of conservation priority for actions targeting these endemics (Loyola et al. 2018; Pougy et al. 2018). The action plan was elaborated in a participatory process with relevant stakeholders that act in the conservation arena in the state, planned upon thematic lines and also indicating actors, specific legislation and public policies, research, and species and habitat management and awareness pertinent to the implementation of outlined mitigation actions.

The arboretum program has been working since 2010 in carrying efforts to conserve, restore, and raise awareness to the importance of the *Hiléia Baiana* Flora (a widely recognized hot-point within the Atlantic rainforest hotspot) and have on CNCFlora a crucial partner so its mission can be achieved and whose technical team has been supporting many of their activities. The program's mission is to conserve and restore forest diversity as a strategy to value forest resources, generating and disseminating sound scientific knowledge. The program spams from threatened plants seed collection, seedling cultivation, and forest restoration in order to support the development of productive chains integrated with the forest and local communities. Both activities revert in economic outcomes for the locals. To date, 1178 tree individuals from 524 species were selected, tagged, and had germplasm gathered by Arboretum staff, to be cultivated in one of the many local nurseries found in the region, including many species threatened with extinction. For instance, Arboretum nurseries produced together 342,556 seedlings of Atlantic Forest native tree species. Further work foreseen by this joint project will produce an action plan for the conservation of threatened trees occurring in southern Bahia, integrating plant conservation and forest restoration programs.

CNCFlora also established in 2017 a partnership with Rio de Janeiro State Secretary of Environment in order to trigger restoration initiatives in the state by selecting and tagging native tree species for prospective germplasm collection, cultivation, and utilization in restoration programs across the state. The project consisted of an innovative approach since it was the first of this kind to mark trees inside protected areas, following the publication of the state Resolution INEA-RJ 139/2016 and, from that, allowing seed collectors and nurseries to gather germplasm inside state's strict protected Areas. In this context, 849 native tree species from distinct ecological succession stages were tagged, geolocalized, and collected for taxonomic identification in five states' strict protected areas and are now included in a database of support for guiding state's nurseries future work in produce seedlings to be used in restoration efforts in Rio de Janeiro. From the species tagged in this work, 21 are considered threatened and are therefore key elements for ecosystem enrichment of restored areas. Additionally, 161 specimens gathered during the work represent the first botanical record held in one of the five protected areas, directly contributing to the consolidation of botanic knowledge inside these protected areas.

Additionally, to the aforementioned contributions, CNCFlora also participated in several conservation initiatives in the Atlantic Forest Hotspot, as described below:

1. National Action Plan of Faveiro-de-Wilson (*Dimorphandra wilsonii* Rizzini)

   An important effort undertaken in 2014 resulted in the publication of the Action Plan of the *Faveiro-de-Wilson*, a critically endangered (CR) tree endemic to transitional forest formations between the Atlantic Forest and the Cerrado in the state of Minas Gerais. The aim of this joint initiative among CNCFlora and the Botanic Garden of Municipal Parks Foundation and Zoobotânica (FPMZB) was to promote the effective conservation of this iconic forest resource, also enhancing the conservation status of further 42 threatened plant species occurring sympatrically. Before the initiative started in 2014, only 246 trees were known from wild populations. To date, the *Faveiro-de-Wilson* is now known 420 mature individuals, highlighting the impact coming from tailored research and action which amplified considerably its known population size, its data availability, and consequently its accurate conservation status and effective protection of its habitat.

2. The "Wanted Campaign": In Search of Long-Lost Species in Rio de Janeiro State

   Along the extinction risk assessment effort undertaken for Rio de Janeiro endemic plant species (Martinelli et al. 2018), half of the 884 taxa considered restricted to the state's border were assessed as data deficient (DD). To deal with such an obstacle imposed by the lack of robust information, CNCFlora started the "Wanted Campaign," as a complement to the Project "Protected Areas – PAs of the state of Rio de Janeiro: analyses and strategies for the conservation of threatened endemic flora." The campaign aimed to obtain more data in the field and in non-digitized herbaria on these DD species and, thus, provide sufficient information for extinction risk assessment. Species collected and identified on DD-targeted expeditions plus exsiccates consulted in herbaria resulted in 44 targeted species found. Of these, 25 are from active searches in the field, 22 are records from non-digitized herbaria and private collections, and 3 were found on two campaign's areas of action. This result shows the importance of fieldwork targeting gap species and points out the need to encourage research with such species even in areas where collection efforts are often regarded as sufficient. Despite the success of the campaign, what can be spanned throughout the whole country, hundreds of species endemic to the Rio de Janeiro still in need of data to be adequately assessed (Rosa et al. 2018) in order to prevent thousands of DD species slipping unnoticed toward extinction.

### 20.4.2 Current and Prospective Contributions

It is evident the necessity of adopts a participatory approach, since the beginning until delivery stages of any conservation action. Therefore, it is clear that multiple, committed stakeholders, with the appropriate scientific and political support, can promote sound conservation interventions in this iconic biome. Results and findings herein present would not be feasible without the full adherence of a broad network of scientists, decision-makers, and society as a whole.

With a guiding mission and well-defined and consequential positive outcomes after more than 10 years of innovative conservation measures, CNCFlora is steadily working on several national and international fronts to move forward so its mission can be fully accomplished. Just recently, CNCFlora updated the joint agreement with BGCI and set up new targets to be addressed between 2019 and 2020 regarding the completion of further 1000 trees full conservation assessment, which will double the current figure of evaluated Brazilian trees. We expect that this project represents the much-needed ignition triggering a complete analysis of the current conservation status of Brazilian trees – a step further toward the understanding of the conservation status of the whole Brazilian flora.

Moreover, CNCFlora and partners are also engaged since the end of 2018 with the Global Environmental Facility – GEF PRO-SPECIES project, a multi-stakeholder initiative aiming to improve tools to prevent extinction, recover populations, and promote knowledge and sustainable use of the Brazilian biodiversity. The project offsets conservation efforts from the simple detection of species likely to be threatened to actually plan concrete strategies to move such species out of red lists.

## 20.5 Private Protected Areas as Biome-Scale Strategy

Brazil has the largest and one of the best structured private protected area systems in the world. Private Natural Heritage Reserves (RPPNs in Brazil acronym) currently protect almost 780,000 hectares, distributed in 1536 reserves, located in almost 800 municipalities of the 27 units of the federation.

Comparing the total protected area and the average area of the units between public units and RPPNs may lead to the hasty conclusion that RPPNs have little relative importance in the nature conservation scenario. After all, although more than half of Brazilian protected areas are RPPNs, areas where they represent only 0.31% of the total protected by the system or about 0.5% if we exclude all public marine and ocean protected areas. The average area of the RPPNs (508 hectares) is equivalent to 0.37% of the average area of the continental public protected areas (138,563 hectares). Even with their rather modest size, RPPNs have been important in ensuring the protection of endangered species populations, endemic and rare species, and portions of ecosystems that are not adequately protected by the public network (Pinto et al. 2012; Crouzeilles et al. 2013). RPPNs also play a potentially relevant role in protecting ecosystem services, since in many Brazilian municipalities, as well as some river basins that supply small- and medium-sized cities in the country, the only existing PA is an RPPN.

### *20.5.1 RPPNs in the Atlantic Forest*

The Atlantic Forest is the biome with the largest number of RPPNs. Seven out of every 10 RPPNs in the country are located within the Atlantic Forest. Several authors have noted the importance of voluntary initiatives to create private protected areas in the Atlantic Forest, mainly because it is a region where more than 80% of the natural remnants are in private lands (Costa et al. 2004; Rambaldi et al. 2005;

Pinto et al. 2012). As recorded in Mesquita (2014), there are at least four factors that explain the predominance of the biome in RPPN statistics.

The first factor refers to the average socioeconomic profile of biome owners, especially in the states of the Southeast, which house half of the Atlantic Forest RPPNs. According to Mesquita (2004), Vieira et al. (2004) and Oliveira et al. (2010), a significant portion of the rural owners of the region have personal or family income from activities that have no relation to the rural property where the reserve is inserted and even less with the management of the RPPN itself. They are mostly small- and medium-sized entrepreneurs and traders, self-employed or successful career employees, civil servants, executives, or people who have inherited a family property and have been able to ensure the perpetuation of its existing natural heritage. This factor, combined with a higher level of education and greater sensitivity to the planet's challenges to environmental and climate issues, makes it possible to assume that rural owners of the Atlantic Forest are, on average, citizens with a more prominent environmental conservation awareness and culture. Thus, the combination of better informed and more conscious citizens with landowners who are not economically dependent on their land may explain, at least in part, a greater interest in the creation of private protected areas in this biome.

The second factor may be related to the land tenure network found in the Atlantic Forest. In the region, the proportion of micro, small, and medium properties is much higher than in any other biome, which explains the fact that the Atlantic Forest RPPNs present the smallest average area when compared to the reserves of other biomes. A larger proportion of small properties results in a larger number of properties. The existence of more real estate certainly influences the number of RPPNs, although the interest in the creation of reserves was proportionally equivalent to that found in other biomes.

The third fact is related to public policies and state government programs, which were decisive for the growth of the number of reserves in the biome. State environmental agencies in Minas Gerais, Paraná, Rio de Janeiro, and São Paulo – four out of the five states with the most RPPNs – have implemented programs that have encouraged and supported their creation. More than 70% of the Atlantic Forest RPPNs are located in these four states.

The fourth factor, which also explains the existence of various RPPN clusters, is the performance of civil society organizations, which since the last decade of the last century have promoted the creation of RPPNs in priority areas for biodiversity. In Mesquita (2014), we find a detailed analysis of this factor, passing through the Atlantic Forest Central Corridor and the region of occurrence of the golden lion tamarin (detailed later in this section).

Still on the role of these initiatives, we cannot fail to note the impact of the Atlantic Forest RPPN Incentive Program, a pioneering initiative led by the partnership between the SOS Mata Atlântica Foundation and Conservation International. Over the 13 years of this program, more than 200 new RPPNs were created, in addition to supporting the management and protection of another 130 reserves (Costa 2014).

Several authors have highlighted the role of private reserves in complementing government nature protection efforts, allowing in many situations to maintain a higher degree of connectivity of the natural landscape (Lees 1995; Morsello 2001; Langholz 2005; Pinto et al. 2004; Vieira et al. 2004). There is a lot of published evidence about the increased representation of priority conservation areas, not yet covered by the network of public protected areas (Mesquita and Leopoldino 2002; Mesquita and Vieira 2004).

The keyword is complementarity. If we consider the need for the formation of ecological corridors and mosaic landscapes, presenting protected areas combined with different land uses that are permeable to the transit of animals and seeds, the existence of several RPPNs in the interstices of parks and public reserves is undoubtedly one of the most efficient strategies. The public authority has the task of establishing large protected areas that ensure the protection of representative samples of ecosystems and the maintenance of ecological processes. It is incumbent upon the RPPNs, in a complementary manner, to form a network of protection or cushioning of impacts around public areas, promoting ecological permeability in strongly fragmented landscapes, such as the Atlantic Forest.

In some cases and regions, however, the role of RPPNs becomes central and preponderant. There are many river basins and hundreds of municipalities where the only existing conservation unit is an RPPN. Some species of the Atlantic Forest depend very heavily on RPPNs for their survival. This is the case with the muriquis and the golden lion tamarin.

The Feliciano Miguel Abdala RPPN, located in municipality of Caratinga at Minas Gerais state, is home to the largest population of northern muriquis, one of the 25 most endangered primates on the planet. It is estimated that the nearly 1000 hectares of this reserve house one-third of the entire remaining population of this species. In addition to the muriquis, the reserve also protects significant populations of howler monkeys (*Alouatta guariba*) and the rare tamarin (*Callithrix flaviceps*), two other endangered species. Over the past three decades, the owners' effective protection of the area has resulted in a tripling of the muriqui population.

The Mata do Sossego RPPN, located in the municipality of Simonesia at Minas Gerais state, about 80 km from Caratinga, is one of them. Created more than 20 years ago by the Biodiversitas Foundation (a distinguished conservationist NGO), it houses a scientific center that has monitored muriquis. In addition, the organization promotes various actions and partnerships along with the forest fragments between the two RPPNs. The objective is to encourage the adoption of land use practices that favor the formation of an ecological corridor between them, including the restoration of permanent preservation areas.

The golden lion tamarin (*Leontopithecus rosalia*), a highly endangered primate, which has the last habitat in the Atlantic Forest of north central Rio de Janeiro, also has the necessary reinforcement of the RPPN for its protection. Although there are two federal biological reserves in the municipalities of Silva Jardim and Casemiro de Abreu, the nearly 13,000 hectares of Poço das Antas and União is not sufficient to ensure the long-term preservation of the species. Applying the method called "population assessment and habitat viability," scientists estimated that at least 2000 golden lion tamarins would be required to live freely in at least 25,000 hectares of well-preserved and connected forests to ensure the survival of the species.

## 20.6 NGO SOS Mata Atlântica

SOS Mata Atlântica Foundation is a Brazilian environmental NGO that acts on the development of public policies for the conservation of this biome through environmental monitoring, production of studies, demonstrative projects, dialogue with public and private sectors, improvement of environmental legislation, communication, and society engagement. The SOS Mata Atlântica seeks to engage people, generate knowledge, and mobilize resources for our mission to inspire society in the defense of the Brazilian Atlantic Forest, transforming values and attitudes across the biome.

However, we live in an outdated model of development sustained in the myth of abundance. All the main economic cycles in the country's history followed the logic that growth occurs through territorial expansion and deforestation, from the exploitation of Pau-Brasil (*Caesalpinia echinate*), agriculture, and cattle raising to the industrialization and expansion of cities. Starting at the coast, our society ended up with about 90% of the original Atlantic Forest area. Therefore, in addition to conserving, we need to restore our forests, especially in springs, and riparian forest areas to mitigate the current and future risks of water shortages. There are many initiatives on restoration ongoing in Brazilian Atlantic Forest (De Siqueira et al. 2021 Chap. 18), and SOS Mata Atlântica has already planted more than 40 million seedlings of native Atlantic Forest species in more than nine states and 550 municipalities, helping to remove 6.5 million tons of CO2 from the atmosphere. This corresponds to an area of 23 thousand restored hectares, equivalent to the city of Recife, Pernambuco state. The Forests of the Future program, responsible for this result, brings together organized civil society, private initiative, landowners, and the public authorities in participatory forest restoration projects.

The SOS Mata Atlântica has a nursery that meets the Forests of the Future project with the capacity to produce 750,000 seedlings of 110 native Atlantic Forest species per year. The nursery is located at the SOS Mata Atlântica Forest Experiment Center – Heineken, Brazil, located in Itu, São Paulo state, which is a reference in Atlantic Forest restoration. This restoration initiative consists of not just planting seedlings of native species in the region, but actually reproducing a functional native environment, with the presence of regional biodiversity and providing ecosystem services – such as carbon sequestration, improvement of water and quality and amount, and restoration of natural landscapes – services that the forest exerted in its original state.

With the Atlantic Forest Atlas, SOS Mata Atlântica identifies and monitors the biome constantly update the number of forest remnants and natural areas. Held in partnership with the National Institute for Space Research (INPE in the Portuguese acronym) and with the technical implementation of Arcplan, SOS Mata Atlântica made possible, in the early 1990s, a diagnose of the situation of the Atlantic Forest. This initiative is fundamental for the development of new studies and vital strategies to ensure the protection of the biome, having subsidized, among other things, the construction and approval of the Atlantic Forest Law (11.428/2006).

An achievement of society, this law regulates the protection and utilization of the biodiversity and resources of this forest. It aims to ensure the rights and duties of citizens and public agencies to exploit it consciously and sustainably without harming its ecosystem. In addition, it creates financial incentives for ecosystem restoration, with donations from the private initiative for conservation projects, regulates the article of the Constitution that defines the Brazilian Atlantic Forest as a National Heritage, delimits its domain, prohibits the deforestation of primary forests, and creates rules for economic exploitation.

The Atlantic Forest Atlas contributes to environmental management and the improvement of legislation and public policies aimed at the conservation and restoration of the biome, the protection of water, biodiversity, and associated marine environments. Over the years, the Atlas has evolved with the advancement of information technology and geoprocessing, methodology, and quality of satellite imagery.

The Brazilian Atlantic Forest has is the only biome ensured by a specific law that, associated with other laws, regulates its use and conservation. To avoid setbacks and make improvements possible, SOS Mata Atlântica works with the legislature, executive, judiciary, prosecutors, and various channels of civil society participation – such as councils, commissions, public hearings, and regulatory agencies. Partnerships with other NGOs in networks, movements, observatories, and coalitions are also established. In this policy action, SOS Mata Atlântica seeks to influence and manage opportunities for formulating, implementing, and defending the priority laws and public policies for the Brazilian Atlantic Forest in favor of forest restoration, enhancement of parks and reserves, clean water, and protection of the sea. Also, the Brazilian Atlantic Forest offers services essential for our survival and well-being, as well as for activities important to our economy (Pires et al. 2021 – Chap. 16).

The SOS Mata Atlântica has also initiatives to clean up the Atlantic Forest rivers by collecting water quality data from volunteers and mobilizing civil society. Of the freshwater available in the world, 12% is in Brazil, but the distribution is very uneven, and the main rivers of the country have worrying rates regarding the quality and availability of water. By observing the rivers, SOS Mata Atlântica organizes volunteers and mobilize them to monitor the water quality of the Atlantic Forest rivers. The results are periodically released as a way to alert society and the public power and contribute to the improvement of legislation around this theme.

## 20.7 Academic Actions

### *20.7.1 Reintroduction of Vertebrate Populations in the Atlantic Forest Biome*

Most of the Atlantic Forest remnants nowadays consist of defaunated forests (sensu Dirzo et al. (2014)). Therefore, the Atlantic Forest biome is badly in need of reintroductions of locally extinct vertebrate populations, in order to recover species in the brink of extinction, to rebuild native faunas, and also to restore ecological interactions (such as seed dispersal and pollination), thus keeping ecosystem health in the remaining forest blocks (Fernandez et al. 2017).

The need for active interventions against the emptying of the Atlantic Forest had been perceived by some of Brazil's pioneer conservationists as early as the 1960s; therefore, the Atlantic Forest has the longest history of animal reintroduction of any biome in Brazil. Adelmar Coimbra-Filho and Antonio Aldrighi tried to restore the impoverished vertebrate fauna of Tijuca National Park, within Rio de Janeiro city, in the early 1970s, reintroducing 25 bird species, seven mammals, and one reptile (Coimbra-Filho and Aldrighi 1971). There was little know-how about reintroductions at that time, the reintroduced populations were not monitored, and most of these reintroductions failed. However, at least two of them, the channel-billed toucan (*Ramphastos vitellinus*) and boa constrictor (*Boa constrictor*), have succeeded.

Other early initiatives concerned two species of lion tamarins, the golden lion tamarin (*Leontopithecus rosalia*) in Rio de Janeiro and the black lion tamarin (*L. chrysopygus*) in São Paulo state. Coimbra-Filho was the one who sounded the alarm that *L. rosalia* was in a critical situation in the early 1970s, leading to the establishment of Poço das Antas Biological Reserve, where its last large population was found. Thereafter, the newly founded Golden Lion Tamarin Association reintroduced *L. rosalia* to several forest fragments in northern Rio de Janeiro state, starting in the 1980s (Kierulff et al. 2012). The reintroduced animals came from international zoos and translocations from remaining populations – including Poço das Antas – in order to improve the species' conservation *status*. The largest area to receive reintroduced goldens became in 1998 an important protected area in its own right (União Biological Reserve), primarily because of its acquired importance for the tamarins. Meanwhile, in the Pontal do Paranapanema region in western São Paulo state, the Ecological Research Institute (IPÊ), led by Claudio Valladares-Pádua, used reintroductions to improve the perspectives of survival of *L. chrysopygus*. Like its golden cousin, the species had been mostly confined to the last stronghold – Morro do Diabo State Park – but it was also reintroduced, to several smaller forest remnants of the region.

Another long-standing, important project is the reintroduction of the vinaceous-breasted amazon (*Amazona vinacea*) to the Guaragueçaba region in Paraná state, part of the largest remaining Atlantic Forest block. This program has been carried out by the Wildlife Research Society (SPVS) and "Instituto Espaço Silvestre" for over two decades now and has successfully blended research, management, and community involvement, increasing considerably the situation of this endangered parrot.

The onset of the new millennium witnessed an explosive worldwide increase in the use of reintroductions and rewilding as conservation tools (Seddon et al. 2007). This global perception of the need for a more active instance toward conservation would sooner or later reach Brazil, and so it happened. Therefore, the pace of reintroductions in the Atlantic Forest has dramatically increased in these last decades.

An important initiative, for example, was the reintroduction of red-billed curassows (mutums, *Crax blumembachii*) and jacutingas (*Pipile jacutinga*) to Reserva

Biológica Guapiaçú (REGUA), a private reserve in northern Rio de Janeiro state, carried out by Christine Steiner São Bernardo and Mauro Galetti. Although the reintroduction stood on the verge of success for several years, it eventually suffered from sponsorship problems and failed. Recently, the idea of reintroducing jacutingas to REGUA has been resuscitated under the guidance of the ornithologist Pedro Develey and SAVE Brasil.

Meanwhile, the Refauna project started in 2010 with a different and even more ambitious initiative: not to reintroduce a single species, but to reconstruct, as far as possible, the whole native vertebrate fauna of an empty forest (Fernandez et al. 2017). The area chosen was Tijuca National Park, a 3953 ha area surrounded by Rio de Janeiro city, a good "natural laboratory" due to its easy access for the researchers and isolation from other natural areas. The reintroduced populations are all monitored post-release, as well as their role in restoring missing ecological interactions. The Refauna project has been carried out by a consortium led by several researchers – Fernando Fernandez, Alexandra Pires, Marcelo Rheingantz, and Maron Galliez – and composed by several partner institutions such as Universidade Federal do Rio de Janeiro, Universidade Federal Rural do Rio de Janeiro, Instituto Federal do Rio de Janeiro, Rio de Janeiro Primatology Center, Rio de Janeiro Zoo, FIOCRUZ, Center for Alocation of Wildlife – Seropédica, and others.

The first species reintroduced to Tijuca National Park by Refauna was the agouti (*Dasyprocta leporina*) in 2010; the animals came from semi-captivity in an urban park and the reintroduction was considered successful (Cid et al. 2014; Kenup et al. 2018). The second species was the howler monkey (*Alouatta guariba*), reintroduced in 2015, from a variety of captive sources. The howlers' reintroduction has run into several problems, mostly from animals getting too addicted to human contact and from health concerns linked to the yellow fever outbreak in Brazil. Although the population persists, success is by no means sure. Both the reintroduced howlers (Genes et al. 2019) and the agoutis have successfully restored ecological interactions in Tijuca National Park. The third species being reintroduced by Refauna, the yellow-footed tortoise *Chelonoidis denticulata*, was brought from captive populations to Tijuca National Park starting in 2019, and the project intends to bring more species in sequence.

A welcome offshoot of the Refauna project was the reintroduction of the lowland tapir, *Tapirus terrestris*, to REGUA. Tapirs had been extinct in Rio de Janeiro state for more than a century, since 1914. TNP would be too small for a viable population of tapirs, but REGUA – with 7000 ha and part of a 70,000 ha block of Atlantic Forest including Três Picos State Park – provides a better option. This initiative has been led by Maron Galliez, and since 2017, 10 tapirs have been brought to REGUA to start the new population and monitored post-release. The population is thriving well, but it is still too early to evaluate the tapir's reintroduction success.

The developments along this last decade have been quite encouraging, and hopefully, population reintroduction will be a very useful tool to mitigate the effects of defaunation in the Atlantic Forest in the foreseeable future.

## 20.8 Landscape Connectivity and Perceptual Ranges

Landscape connectivity, the degree to which a landscape facilitates or prevents movements of organisms (Taylor et al. 1993), is a central concept to understanding effects of habitat loss and fragmentation (Haddad et al. 2017), viability and conservation of metapopulations (Hatfield et al. 2018), metacommunity dynamics (Monteiro et al. 2017), and ecosystem functioning (Staddon et al. 2010; Thompson et al. 2017). Ultimately, landscape connectivity depends on the successful movements of organisms through the landscape, between habitat patches, generally fragments of native vegetation, and should be used in strategies for conservation in biomes such as Brazilian Atlantic Forest. Recent advances in two recent areas of study have opened new perspectives on predicting probabilities of movement thought the landscape: understanding of perceptual ranges and the use of scattered trees, plantation rows, and matrix elements in general as guiding structures for animals. Prediction allows action and design of landscape management strategies to improve connectivity, biodiversity conservation, and maintenance of ecosystem services in human-altered landscapes.

The perceptual range is empirically defined and measured as “the distance from which a particular landscape element can be perceived as such (or detected) by a given animal” (Lima and Zollner 1996). The success in finding suitable habitat patches when moving in the landscape depends to a great extent on their perceptual range (Zollner and Lima 1999; Vuilleumier and Perrin 2006; Pe’er and Kramer-Schadt 2008). Frequent movements connecting local populations are mainly dependent on perceptual ranges, which may be within the home range of individuals, differing from occasional long-distance dispersal movements beyond their home ranges (Pe’er et al. 2011). Recent studies by perceptual ranges of nonflying vertebrates of the Atlantic Forest have opened a new perspective on predicting probabilities of routine movements between habitat fragments (Forero-Medina and Vieira 2009; Prevedello et al. 2010, 2011).

The general experimental design to measure perceptual ranges involves translocation experiments: individuals are removed from their home ranges and released in an unfamiliar landscape at varying distances from a habitat patch; the maximum distance where most individuals are considered oriented toward the habitat patch when released is considered its perceptual range (Goodwin et al. 1999; Zollner 2000; Forero-Medina and Vieira 2009). More complex designs are possible to separate potential confounding effects, such as innate bias in movement direction (Olden et al. 2004; Fletcher Jr et al. 2013). Empirical estimates of perceptual range based on this basic experimental design were made for small mammals in temperate (Zollner and Lima 1997; Zollner 2000; Mech and Zollner 2002; Schooley and Branch 2005; Flaherty et al. 2008) and tropical landscapes such as Brazilian Atlantic Forest (Forero-Medina and Vieira 2009; Prevedello et al. 2010, 2011). Estimates of perceptual ranges can now be used to simulate loss or gain of patches of native vegetation in landscapes, including corridors, restoration of riparian habitats, and their effects on landscape connectivity (e.g. Pe’er and Kramer-Schadt 2008).

Animals also have other strategies to orient themselves beyond perceptual ranges, frequently using elements of the matrix between forest patches, an information that can be used to manage landscapes to improve connectivity. Animals frequently use natural or man-made linear structures to quickly cross distances in an unfamiliar matrix, such as plantations rows in Brazilian Atlantic Forest (Prevedello and Vieira 2010; Sozio et al. 2013). If these linear structures were established connecting forest patches, populations otherwise isolated could become connected. Scattered trees in pastures or plantations are another matrix element used by forest animals to move between remaining forest patches (Manning et al. 2009; Fischer et al. 2010; Le Roux et al. 2018; Prevedello et al. 2018). The distribution of scattered trees could be managed to improve connectivity between forest patches.

The previous strategies to increase landscape connectivity are implemented within the spatial scale of a river catchment area and can have great impact on a larger geographical scale, a "bottom-up" strategy to enhance biodiversity conservation. A "top-down" strategy is also possible, for example, setting priorities for reforestation throughout the whole Atlantic Forest biome (Banks-Leite et al. 2014). Native species richness is always affected by habitat loss, but there seem to be thresholds of habitat loss beyond which species richness reduces more drastically, at a faster rate (Pardini et al. 2010; Estavillo et al. 2013; De Coster et al. 2015; Roque et al. 2018). These thresholds are points that should not be crossed, beyond which the costs of habitat restoration increase dramatically; thus, efforts and resources should be applied to landscapes whose native forest cover was reduced to levels close to these thresholds (Banks-Leite et al. 2014). Connectivity estimates at such geographical scales, larger than the landscapes of river catchments, can also be used to infer habitat availability and areas that are vital or more effective to reforest, which have a larger importance connecting larger areas, or to decide on the best strategy for forest restoration, active or passive (Crouzeilles et al. 2011, 2015).

## 20.9 Concluding Remarks

Natural resources can be a plentiful capital in megadiverse countries and important for human welfare (Constanza et al. 1997). It is from this natural resource (hereafter natural capital) that humans derive a wide range of ecosystem services, which make human life possible. Therefore, the conservation of biomes such as Brazilian Atlantic Forest is fundamental, and it is possible with effective environmental policy, including protected areas and conservation planning (Bustamante et al. 2019).

Obviously, to achieve these objectives, a country needs commitment, knowledge, and budget funding for science among other factors. As science-based decisions are crucial, scientific production and an integrated decision between researchers and policy makers are necessary. This dialogue among local communities, companies, and NGOs should be the goals of all megadiverse countries, and the present chapter – with authors from the Brazilian government, NGOs, and universities – shows how it is happening in Brazil. However, to run these initiatives, some budget is needed.

Unfortunately, these recent trends are paralleled by budget cuts in science and conservation funding. However, some authors argue that such cuts may have consequences for the commitments of the Brazilian government, such as sustainable development goals, Aichi targets, and Nationally Determined Contribution (Fernandes et al. 2017; Dobrovolski et al. 2018). Therefore, with potential consequences for global biodiversity and the ecosystem services (Overbeck et al. 2018) but the links between the socioecological science produced in the country and policy-making remain far from established. As expected, sustainable and environmental development depends on investments in science and technology (e.g., Tallis et al. 2008; Scarano 2017).

Recently, a report showed that there was a cut off funding of Brazilian Environmental Ministry, with a drastic reduction of the budget in the last years (WWF 2018). This cut of funding and the drastic reduction in budget for research in Brazil affect all initiatives described above in this chapter.

On the other approach, in the last 20 years, there was a flourish in scientific production in Brazil (Fernandes et al. 2017) that has a 13th position in the global ranking of scientific production (Clarivate Analytics 2017). The period evaluated in this report of Clarivate (between 2011 and 2016) reflects the grew up in budget funding in Brazil, with a peak in 2013 (Fernandes et al. 2017). Besides this and according to the Clarivate report (Clarivate Analytics 2017), environmental/ecology is one of the fields of knowledge in which Brazil can emerge as an international leader. Thus, there are evidences that Brazilian scientists are ready to help in the definition of conservation strategies and resource management.

In this chapter, we list a myriad of conservation initiatives, showing a governmental and nongovernmental concern on the future of Brazilian Atlantic Forest. The good news is a scientific maturity and a wish to work together in the design and implementation of strategies of conservation of the Brazilian Atlantic Forest.

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## References

Banks-Leite C, Tambosi LR, Pearse W, Bueno AA, Bruscagin RT, Condez TH, Dixo M, Igari AT, Martensen AC, Metzger JP (2014) Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot. Science 345:1041–1045

Bland LM, Bielby J, Kearney S, Orme CD, Watson JE, Collen B (2017) Toward reassessing data-deficient species. Conserv Biol 31:531–539

Bustamante M, Metzger JP, Scariot A, Bager A, Turra A, Barbieri A, Neves A, Boesing AL, Agostinho AA, Marques AC, Dias B, Grelle CEV, Caixeta D, Sawyer D, Scarano F, Sousa FD, Fernandes GW, Queiroz H, Miranda HS, Schongart J, Quintão JMB, Martinelli LA, Gomes LC, Cunha MC, Piedade MTF, Sato MN, Vale MM, Aquino MFS, Vogt N, May P, Fearnside P, Prado RP, Rodrigues RR, Thomaz SM, Pivello VR, Imperatriz Fonseca VL, Farjalla VF (2019) Tendências e impactos dos vetores de degradação e restauração da biodiversidade e dos serviços ecossistêmicos. In: Joly CA, Scarano FR, Seixas CS, Metzger JP, Ometto JP, Bustamante MMC, Padgurschischi MCG, Pires APF, Castro PFD, Gadda T, Toledo P (eds) 1° Diagnóstico brasileiro de biodiversidade e serviços ecossistêmicos. São Carlos, SP, Editora Cubo, pp 93–213

Brazil Flora Group (BFG) (2018) Brazilian Flora 2020: innovation and collaboration to meet Target 1 of the Global Strategy for Plant Conservation (GSPC). Rodriguésia 69(4):1513–1527

Brooks TM, Tobias JA, Balmford A (1999) Deforestation and bird extinctions in the Atlantic forest. Anim Conserv 2:211–222

Cid B, Figueira L, Mello AFT, Pires AS, Fernandez FAS (2014) Short-term success in the reintroducion of the agouti *Dasyprocta leporina* in Brazilian Atlantic Forest. Tropical Conserv Sci 7:796–810

Clarivate Analytics (2017) Research in Brazil: A report for CAPES by Clarivate Analytics. Report

Coimbra-Filho AF, Aldrighi AD (1971) A restauração da fauna do Parque Nacional da Tijuca. Publicações Avulsas do Museu Nacional 57:1–30

Constanza R d'A, de Groot R, Farber S, Grasso M, Hannon B, Naeem S, Limburg K, Paruelo J, O'Neill RV, Raskin R, Sutton P, van den Belt M (1997) The value of the world's ecosystem services and natural capital. Nature 387:253–260

Costa CMR (2014) Programa de Incentivo às RPPN da Mata Atlântica: dez anos que fizeram a diferença. Relatório Interno. Conservação Internacional e Fundação SOS Mata Atlântica

Costa CMR, Hirota MM, Pinto LPS, Fonseca MT, Lamas IR, Brito MCW, Mesquita CAB (2004) Incentivo à criação e gestão de Reservas Particulares do Patrimônio Natural (RPPN): uma estratégia para a conservação da Mata Atlântica. In: Castro RE, Borges MC (eds) RPPN – Conservação em Terras Privadas: desafios para a sustentabilidade. Confederação Nacional de Reservas Particulares do Patrimônio Natural, Edições CNRPPN, Curitiba, pp 69–77

Crouzeilles R, Lorini ML, Grelle CEV (2011) Applying graph theory to design networks of protected areas: using inter-patch distance for regional conservation planning. Natureza Conservação 9:219–224

Crouzeilles R, Beyer HL, Mills M, Grelle CEV, Possingham HP (2015) Incorporating habitat availability into systematic planning for restoration: a species-specific approach for Atlantic Forest mammals. Divers Distrib 21:1027–1037

Crouzeilles R, Vale MM, Cerqueira R, Grelle CEV (2013) Increasing strict protection through protected areas on Brazilian private lands. Environ Conserv 40:209–210

De Coster G, Banks-Leite C, Metzger JP (2015) Atlantic forest bird communities provide different but not fewer functions after habitat loss. Proceedings of the Royal Society B: Biological Sciences 282:20142844

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

Dobrovolski R, Loyola R, Rattis L, Gouveia SF, Cardosoa D, Santos-Silva R, Gonçalves D, Alves-Souza LMB, Diniz-Filho JAF (2018) Science and democracy must orientate Brazil's path to sustainability. Perspect Ecol Conserv 16:121–124

Estavillo C, Pardini R, da Rocha PLB (2013) Forest Loss and the Biodiversity Threshold: An Evaluation Considering Species Habitat Requirements and the Use of Matrix Habitats. PLoS ONE 8:e82369

Fernandes GW, Vale MM, Overbeck GE, Bustamante MMC, Grelle CEV et al (2017) Dismantling Brazil's science threatens global biodiversity heritage. Perspect Ecol Conserv 15:239–243

Fernandez FAS, Rheingantz ML, Genes L, Kenup CF, Galliez M, Cezimbra T, Cid B, Macedo L, Araujo BBA, Moraes BS, Monjeau A, Pires AS (2017) Rewilding the Atlantic Forest: restoring the fauna and ecological interactions of a protected area. Perspect Ecol Conserv 15:308–314

Fischer J, Stott J, Law BS (2010) The disproportionate value of scattered trees. Biol Conserv 143:1564–1567

Flaherty EA, Smith WP, Pyare S, Bem-David (2008) Experimental trials of the northern flying squirrel (*Glaucomys sabrinus*) traversing managed rainforest landscapes: perceptual range and fine-scale movements. Can J Zool 86:1050–1058

Fletcher RJ Jr, Maxwell CW, Andrews JE, Helmey-Hartman WL (2013) Signal detection theory clarifies the concept of perceptual range and its relevance to landscape connectivity. Landsc Ecol 28:57–67

Flora do Brasil (2020) Jardim Botânico do Rio de Janeiro. Available at: http://reflora.jbrj.gov.br/reflora/floradobrasil/FB128482. Accessed on: 01 Apr 2019

Fonseca GAB (1985) The vanishing Brazilian Atlantic Forest. Biol Conserv 34:17–34

Forero-Medina G, Vieira MV (2009) Perception of a fragmented landscape by neotropical marsupials: effects of body mass and environmental variables. J Trop Ecol 25:53–62

Fundação SOS Mata Atlântica, INPE and Instituto Socio Ambiental (1998) Atlas da Evolução dos Remanescentes Florestais e Ecossistemas Associados no Domínio da Mata Atlântica no período 1990–1995. SOS Mata Atlântica, São Paulo. 54p

Galindo-Leal C, Câmara IG (2003) The Atlantic Forest of South America: biodiversity status, threats, and outlook. CABS Island Press, Washington. 488p

Genes L, Fernandez FAS, Vaz-de-Mello FZ, Rosa P, Fernandez E, Pires AS (2019) Effects of howler monkey reintroduction on ecological interactions and processes. Conserv Biol 33:88–98

Goodwin BJ, Bender BJ, Contreras TA, Fahrig L, Wegner JF (1999) Testing for habitat detection distances using orientation data. Oikos:160–163

Grelle CEV, Alves MAS, Bergallo HG, Geise L, Rocha CFD, Van Sluys M, Caramaschi U (2005) Prediction of threatened tetrapods based on the species-area relationship in Atlantic Forest. Brazil J Zool Lond 265:359–364

Grelle CEV, Fonseca GAB, Fonseca MT, Costa L (1999) The question of scale in threat analysis: a case study with Brazilian mammals. Anim Conserv 2:149–152

Haddad NM, Holt RJ, Fletcher RJ, Loureau M, Clobert J (2017) Connecting models, data, and concepts to understand fragmentation's ecosystem-wide effects. Ecography 40:1–8

Hatfield JH, Orme CDL, Banks-Leite C (2018) Using functional connectivity to predict potential meta-population sizes in the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:215–220

Joly CA, Metzger JP, Tabarelli M (2014) Experience from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. New Phytol 204:459–473

Kenup CF, Alves RS, Kreischer C, Fernandez FAS (2018) Walking on their own legs: unassisted population growth of agoutis reintroduced to restore seed dispersal in an Atlantic Forest reserve. Oryx 52:571–578

Kierulff MCM, Ruiz-Miranda CR, Oliveira PP, Beck BB, Martins A, Dietz JM, Rambaldi DM, Baker AJ (2012) The Golden lion tamarin *Leontopithecus rosalia*: a conservation success story. Int Zoo Yearbook 46:36–45

Langholz J (2005) Seven myths about private protected areas. Parks 15:14–16

Le Roux DS, Ikin K, Lindenmayer DB, Manning AD, Gibbons P (2018) The value of scattered trees for wildlife: contrasting effects of landscape context and tree size. Divers Distrib 24:69–81

Lees A (1995) Innovative partners: the value of nongovernment organizations in establishing and managing protected areas. In: Mcneely JA (ed) Expanding partnerships in conservation. IUCN, USA, pp 188–196

Lima RAF, Mori DP, Pita G et al (2015) How much do we know about the endangered Atlantic Forest? Reviewing nearly 70 years of information on tree community surveys. Biodivers Conserv 24:2135–2148

Lima SL, Zollner PA (1996) Towards a behavioral ecology of ecological landscapes. Trends Ecol Evol 11:131–135

Loyola R, Machado N, Vila-Nova D, Martins E, Martinelli G (2014) Áreas Prioritárias para Conservaçao e Uso Sustentavel da Flora Brasileira Ameaçada de Extinção. Andrea Jakobsson Estúdio, Instituto de Pesquisas Jardim Botânico, Rio de Janeiro

Loyola R, Machado N, Ribeiro BR, Martins E, Martinelli G (2018) Áreas prioritárias para a conservação da flora endêmica do estado do Rio de Janeiro. Instituto de Pesquisas Jardim Botânico do Rio de Janeiro, Secretaria de Estado do Ambiente, Rio de Janeiro. 60p.

MMA – Ministério do Meio Ambiente (2014) Lista Nacional Oficial de Espécies da Flora Ameaçadas de Extinção. Portaria MMA no 443/2014. http://www.dados.gov.br/dataset/portaria_443. Acessed 10 Mar 2019

Manning AD, Gibbons P, Lindenmayer DB (2009) Scattered trees: a complementary strategy for facilitating adaptive responses to climate change in modified landscapes? J Appl Ecol 46:915–919

Margules CR, Sarkar S (2007) Systematic conservation planning. Cambridge University Press, Cambridge

Margules CR, Pressey RL (2000) Systematic conservation planning. Nature 405:243–253

Martinelli G, Moraes MA (orgs) (2013). Livro vermelho da flora do Brasil. Andrea Jakobsson Estúdio, Instituto de Pesquisas Jardim Botânico do Rio de Janeiro, Rio de Janeiro. 1100p

Martinelli G, Martins E, Moraes M, Loyola R, Amaro R (2018) Livro Vermelho da Flora Endêmica do Estado do Rio de Janeiro. Andrea Jakobsson Estúdio, Instituto de Pesquisas Jardim Botânico do Rio de Janeiro/ SEA- Secretaria de Estado do Ambiente, Rio de Janeiro

Mech SG, Zollner PA (2002) Using body size to predict perceptual range. Oikos 98:47–52

Mesquita CAB (2014) A natureza como o maior patrimônio: Desafios e perspectivas da conservação voluntária em áreas protegidas privadas no Brasil. Tese de Doutorado, Programa de Pós-Graduação em Ciências Ambientais e Florestais, Instituto de Florestas, UFRRJ. Seropédica, Rio de Janeiro, 188p

Mesquita CAB (2004) RPPN da Mata Atlântica: um olhar sobre as reservas particulares dos corredores de biodiversidade Central e da Serra do Mar. RPPN Mata Atlântica, Vol. 1. Belo Horizonte. Aliança para a Conservação da Mata Atlântica. 48p

Mesquita CAB, Leopoldino FS (2002) Incentivando e apoiando criação, manejo e integração entre Reservas Particulares do Patrimônio Natural (RPPNs). Anais do III Congresso Brasileiro de Unidades de Conservação. Rede Pró-Unidades de Conservação e Fundação O Boticário de Proteção à Natureza. p. 789–798

Mesquita CAB, Vieira MCW (2004) RPPN: Reservas Particulares do Patrimônio Natural da Mata Atlântica. São Paulo, Conselho Nacional da Reserva da Biosfera da Mata Atlântica. Caderno da Reserva da Biosfera da Mata Atlântica: série conservação e áreas protegidas, 28. 96p.

Monteiro VF, Paiva PC, Peres-Neto PR (2017) A quantitative framework to estimate the relative importance of environment, spatial variation and patch connectivity in driving community composition. J Anim Ecol 86:316–326

Morsello C (2001) Áreas protegidas públicas e privadas: seleção e manejo. Annablume: Fapesp, São Paulo. 343p

Muilaert RL, Vancine MH, Bernardo R, Oshima JEF, Sobral-Souza T, Tonetti VR, Niebuhr BB, Ribeiro MC (2018) Uma nota sobre os limites territoriais da Mata Atlântica. Oecol Autralis 22:302–311

Myers N, Mittermeier RA, Mittermeier CG, Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Olden JD, Schooley RL, Monroe JB, Poff NL (2004) Context-dependent perceptual ranges and their relevance to animal movements in landscapes. J Anim Ecol 73:1190–1194

Oliveira VB, Paglia AP, Fonseca M, Guimarães E (2010) RPPN e biodiversidade: o papel das reservas particulares na proteção da biodiversidade da Mata Atlântica. RPPN Mata Atlântica, Vol. 4. Belo Horizonte: Conservação Internacional, São Paulo: Fundação SOS Mata Atlântica, Curitiba: The Nature Conservancy. 48p

Oliveira-Filho AT, Fontes MAL (2000) Patterns of floristic differentiation among Atlantic forests in southeastern Brazil and the influence of climate. Biotropica 32:793–810

Overbeck GE, Bergallo H, Grelle CEV, Freddy Bravo AA, Colli GR, Magnusson WE, Moraes W, Fernandes GW (2018) Global biodiversity threatened by science budget cuts in Brazil. Bioscience 68(1):11–12

Pardini R, de A. Bueno A, Gardner TA, Prado PI, Metzger JP (2010) Beyond the Fragmentation Threshold Hypothesis: Regime Shifts in Biodiversity Across Fragmented Landscapes. PLoS ONE 5:e13666

Pe'er G, Kramer-Schadt S (2008) Incorporating the perceptual range of animals into connectivity models. Ecol Model 213:73–85

Pe'er G, Henle K, Dislich C, Frank K (2011) Breaking functional connectivity into components: a novel approach using an individual-based model, and first outcomes. PLoS One 6:e22355

Pinto LP, Bedê LC, Fonseca MT, Lamas IR, Mesquita CAB, Paglia AP, Pinheiro TC, Sá MB (2012) Mata Atlântica. In: Scarano et al (eds) Biomas Brasileiros: Retratos de um país plural. Conservação Internacional e Casa da Palavra, Rio de Janeiro, pp 16–55

Pinto LP, Bedê LC, Paese A, Fonseca MT, Paglia AP, Lamas IR (2006) Mata Atlântica Brasileira: Os desafios para a conservação da Biodiverisdade em um *hotspot* mundial. In: Rocha CFD, Bergallo HG, Van Sluys M, Alves MAS (orgs) Biologia da Conservação: Essências. Editora rima. São Carlos/SP, pp 91–118

Pinto LP, Paglia A, Paese A, Fonseca M (2004) O papel das reservas privadas na conservação da biodiversidade. In: Castro R, Borges ME (eds) RPPN – Conservação em terras privadas: desafios para a sustentabilidade. Memórias do II Congresso Brasileiro de RPPN. Edições CNRPPN, Planaltina do Paraná, PR, pp 14–27

Pinto MP, Grelle CEV (2009) Reserve selection and persistence: complementing the existing Atlantic Forest reserve system. Biodivers Conserv 18:957–968

Pougy N, Martins E, Verdi M, Fernandez E, Loyola R, Silveira-Filho TB, Martinelli G (2018) Plano de Ação Nacional para Conservação da Flora Endêmica Ameaçada de Extinção do Estado do Rio de Janeiro. Secretaria de Estado do Ambiente-SEA, Andrea Jakobsson Estúdio, Rio de Janeiro, 80 p

Prevedello JA, Vieira MV (2010) Plantation rows as dispersal routes: A test with didelphid marsupials in the Atlantic Forest, Brazil. Biol Conserv 143:131–135

Prevedello JA, Forero-Medina G, Vieira MV (2010) Movement behaviour within and beyond perceptual ranges in three small mammals: effects of matrix type and body mass. J Anim Ecol 79:1315–1323

Prevedello JA, Forero-Medina G, Vieira MV (2011) Does land use affect perceptual range? Evidence from two marsupials of the Atlantic Forest. J Zool 284:53–59

Prevedello JA, Almeida-Gomes M, Lindenmayer DB (2018) The importance of scattered trees for biodiversity conservation: a global meta-analysis. J Appl Ecol 55:205–214

Rambaldi DM, Fernandes RV, Schmidt MAR (2005) Private protected areas and their key role in the conservation of the Atlantic Forest biodiversity hotspot Brazil. Parks 15:30–38

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Strassburg BBN, Tabarelli M, Fonseca GA, Mittermeier RA (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:208–214

Rocha CFD, Bergallo HG, Van Sluys M, Alves MAS, Jenkins C (2006) Corredores Ecológicos e Conservação: um Estudo de Caso na Mata Atlântica. In: Rocha CFD, Bergallo, HG, van Sluys M, Alves MAS (orgs) Biologia da Conservação: Essências. Editora rima. São Carlos/SP, pp 317–342

Rosa P, Baez C, Martins E, Martinelli G (2018) Guia Procura-se: flora endêmica do Estado do Rio de Janeiro. Instituto de Pesquisas Jardim Botânico do Rio de. Janeiro, Graficci, Rio de Janeiro. 200 p

Roque FDO, Menezes JFS, Northfield T, Ochoa-Quintero JM, Campbell V, Laurance WF (2018) Warning signals of biodiversity collapse across gradients of tropical forest loss. Scientific Reports 8:1622

Scarano F (2017) Ecosystem-based adaptation to climate change: concept, scalability and a role for conservation science. Perspect Ecol Conserv 15:65–73

Schooley RL, Branch LC (2005) Limited perceptual range and anemotaxis in marsh rice rats Oryzomys palustris. Acta Theriol 50:59–66

Seddon PJ, Armstrong DP, Maloney RF (2007) Developing the science of reintroduction biology. Conserv Biol 21:303–312

Sobral M, Stehmann JR (2009) An analysis of new Angiosperm species discoveries in Brazil (1990–2006). Taxon 58:227–232

SOS Mata Atlântica and INPE (2017) Atlas dos remanescentes florestais da Mata Atlântica: período 2015–2018. SOS Mata Atlântica and INPE, São Paulo

Sozio G, Mortelliti A, Boitani L (2013) Mice on the move: wheat rows as a means to increase permeability in agricultural landscapes. Biol Conserv 165:198–202

Staddon P, Lindo Z, Crittenden PD, Gilbert F, Gonzalez A (2010) Connectivity, non-random extinction and ecosystem function in experimental metacommunities. Ecol Lett 13:543–552

Strassburg BBN, Beyer HL, Crouzeilles R, Iribarrem A, Barros F, de Siqueira MF, Sánchez-Tapia A, Balmford A, Sansevero JBB, Brancalion PHS, Broadbent EN, Chazdon RL, Filho AO, Gardner TA, Gordon A, Latawiec A, Loyola R, Metzger JP, Mills M, Possingham HP, Rodrigues RR, Scaramuzza CA d M, Scarano FR, Tambosi L, Uriarte M (2019) Strategic approaches to restoring ecosystems can triple conservation gains and halve costs. Nat Ecol Evol 3:62–70

Tallis H, Kareiva P, Marvier M, Chang A (2008) An ecosystem services framework to support both practical conservation and economic development. Proc Natl Acad Sci U S A 105:9457–9464

Taylor PD, Fahrig L, Henein K, Merriam G (1993) Connectivity is a vital element of landscape structure. Oikos 68:571–573

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

Vieira MCW, Silva JEL, Silva MCL (2004) O papel das RPPN na conservação de ecossistemas e na sua restauração. In: Mesquita CAB, e Vieira MCW (eds) RPPN: Reservas Particulares do Patrimônio Natural da Mata Atlântica, São Paulo, Conselho Nacional da Reserva da Biosfera da Mata Atlântica, Série Conservação e Áreas Protegidas, vol 28. Caderno da Reserva da Biosfera da Mata Atlântica, pp 35–40

Vuilleumier S, Perrin N (2006) Effects of cognitive abilities on metapopulation connectivity. Oikos 113:139–147

WWF (2018) Financiamento público em meio ambiente: um balanço da década e perspectivas. Report available in: https://www.wwf.org.br/informacoes/noticias_meio_ambiente_e_natureza/?63822/uniao-estados-municipios-orcamento-meio-ambiente

Zollner PA (2000) Comparing the landscape level perceptual abilities of forest sciurids in fragmented agricultural landscapes. Landsc Ecol 15:523–533

Zollner PA, Lima SL (1997) Landscape-level perceptual abilities in white-footed mice: perceptual range and the detection of forested habitat. Oikos 80:51–60

Zollner PA, Lima SL (1999) Search strategies for landscape-level interpatch movements. Ecology 80:1019–1030

Zwiener VP, Padial AA, Marques MCM, Faleiro FV, Loyola R, Peterson AT (2017) Planning for conservation and restoration under climate and land use change in the Brazilian Atlantic Forest. Divers Distrib:1–12

# Chapter 21
# Financing Conservation in the Brazilian Atlantic Forest

**Carlos Eduardo Frickmann Young and Biancca Scarpeline de Castro**

**Abstract** The Atlantic Forest is one of the world's biodiversity hotspots, due to a long term process of deforestation that must be reversed. Financial resources are necessary to foster the protection of remaining forests and restore the native habitats. This paper presents some alternatives to finance the conservation of the Atlantic Forest in Brazil and some economic instruments to enhance sustainable development activities in the biome, combining public and private sector efforts. The mechanisms discussed in the paper are sustainable public procurement, fiscal incentives, concessions for forestry and public use in protected areas, payments for environmental services, funds to boost avoided deforestation projects that reduce carbon emissions, and tradable environmental reserve quotas (CRA). We conclude that the involvement of the private sector is crucial for financing conservation projects in the Atlantic Forest, but it is also imperative to strengthen the public sector capacity to enforce the environmental legislation and to coordinate the actions of the many stakeholders in order to fulfill the objectives of the biodiversity conservation policy.

**Keywords** Financial mechanisms · Environmental policy · Atlantic Forest · Brazil

## 21.1 Introduction

The Atlantic Forest is a hotspot for biodiversity conservation worldwide (Myers et al. 2000). Given its importance, deforestation that still persists in the biome needs to be halted, and large-scale native vegetation restoration policies need to be

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implemented. Crouzeilles et al. (2019) estimate that, in order to comply with the Brazilian forest legislation, it is necessary to recover five million hectares of native vegetation in the Atlantic Forest.[1] This would prevent the extinction of 26% of species in the biome and sequester one billion tons of $CO_2$ equivalent. For these reasons, Brancalion et al. (2019) elected the Atlantic Forest as the priority hotspot for biodiversity conservation with the largest potential area for restoration, highlighting the great opportunity for sustainable business in this biome.

The present chapter contributes to this discussion pointing out different financing mechanisms for the conservation and restoration of the Brazilian Atlantic Forest. These instruments are not intended to exclude command and control policies that ensure compliance with regulatory frameworks established in recent years. Instead, they seek to contribute as complementary instruments to protect and restore natural habitats and to foster economic growth compatible with environmental protection and social justice.

Mechanisms based on public funding, market-driven, and privately funded actions will be addressed, as well as some public-private partnerships capable of securing standing forests. Some of these mechanisms are already in place in Brazil, highlighting the adaptability of fundraising initiatives to maintain biodiversity. Other mechanisms are not yet being implemented, but are included in the legislation or are used in other contexts.

The chapter summarizes a bibliographic survey carried out in documents related to environmental policies and biodiversity financing in different Brazilian biomes. The final considerations highlight that the most desired strategy is to combine private financing efforts for conservation and restoration of the Atlantic Forest with solid public sector support, since both are essential to ensure the success of these mechanisms.

## 21.2 Public Spending on Environmental Protection in Brazil

Despite advances in private actions aimed at environmental protection, nature conservation spending in Brazil is mostly from the public sector (Young et al. 2018). Because of this dependence on public spending, conservation actions are vulnerable to situations of fiscal crisis and consequent reduction in government financing capacity. Therefore, the fiscal crisis that Brazil is going through in the second half of the 2010s represents an increasing scarcity of financial resources, becoming a significant obstacle to environmental conservation policies.

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[1] Brazilian legislation, especially Law 12.651/2012, establishes specific requirements for forest conservation and management on private lands, including requirements that landowners permanently maintain native forests as “Legal Reserves,” based on a proportion of the private land (80% in the Amazon biome, 35% in the Cerrado biome in the Amazon region, and 20% in the rest of the country) and areas of permanent forest protection due to technical characteristics, such as proximity to water bodies, slope, and specific habitats.

This is a phenomenon that is also perceived in Latin America (IPBES 2018). In the Brazilian case, the lack of resources is well described in a series of studies showing that increasing environmental awareness was not accompanied by an equivalent enhance in spending on nature protection actions in Brazil (Young and Roncisvalle 2002; Medeiros and Young 2011; Young et al. 2014; Young and Medeiros 2018; Alvarenga Jr 2019). Most of these studies showed that environmental management expenditures represent a very small share of total government expenditure, and there is a long-term decline trend on environmental public spending.

The total public spending on environmental management, including wages and all subsectors (water, climate change, forest protection, etc.), for the 2013–2018 period (Table 21.1) shows a decline over time: in the federal government, environmental management expenditures fell from 0.20% to 0.14% from the total, while the aggregate for all of the 27 Brazilian states declined from 0.62% to 0.53%.

The public environmental spending is expected to decrease further in the coming years due to Constitutional Amendment N.95/2016, which established a freeze on federal public spending until 2035. According to this amendment, the total expenses of the Brazilian Federal Government cannot exceed what was observed in 2016, even with the population growth forecast for more than 11 million people (11.3%) in the 2016–2035 period (IBGE 2018).

Declining public resources for forest conservation and restoration brings many negative consequences for the management and monitoring of the Atlantic Forest and other Brazilian biomes: less monitoring of protected areas and law enforcement, less environmental education, less incentives to programs aiming at deforestation reduction, etc. This also negatively affects the provision of environmental services, reducing the quality of life for the population and the potential for sustainable businesses, such as forestry concessions or public use.

Involving the private sector in activities that contribute to environmental conservation is relevant, as it increases the number of economic actors interested in preserving the common good in the different areas in which these businesses are

**Table 21.1** Federal and state environmental management expenditures, in US$ millions (2018 prices) and % of total governmental expenditures

| | Federal government | | State governments | |
|---|---|---|---|---|
| Year | Total environmental management expenditures (2018 US$ Million) | Environmental management/total expenditures (%) | Total environmental management expenditures (2018 US$ Million) | Environmental Management/total expenditures (%) |
| 2013 | 1310 | 0.20% | 1574 | 0.62% |
| 2014 | 1292 | 0.17% | 1500 | 0.59% |
| 2015 | 982 | 0.14% | 1278 | 0.53% |
| 2016 | 1016 | 0.13% | 1248 | 0.54% |
| 2017 | 877 | 0.12% | 1284 | 0.54% |
| 2018 | 1007 | 0.14% | 1295 | 0.53% |

Source: Own elaboration, based on the Brazilian Integrated System of Budget and Planning (SIOP). Values in constant prices of 2018; exchange rate: R 3.259/*US* conducted. In this case, it is in the investor's own interest that the environmental conditions are the best possible, because the environmental quality is a crucial element for the profitability of the business. Activities such as environmental education and protected areas surveillance are no longer in the public interest alone, and collaboration with the private sector allows for increased resources and greater effectiveness in these actions without the need for additional budgetary resources.

On the other hand, even with increased private participation, the public sector will continue to be prominent in coordinating environmental policy. Despite financial difficulties, it is essential that public administrations continue to adopt environmental management and enforcement policies with their own budget resources. These policies will ensure legal certainty for other sectors to invest in environmental preservation and sustainable business.

## 21.3 Financial Mechanisms

Many mechanisms are already in place to finance conservation and restore native habitats in the Atlantic Forest and other Brazilian biomes. This section presents a wide range of possibilities combining public and private agents in different schemes. Some of them are initiatives undertaken by the public sector in different federative spheres, such as sustainable criteria in public procurement, environmental offsets to compensate potential damage in the licensing process, and the conversion of fines into environmental conservation projects. There are also initiatives started at the state and municipal levels to use fiscal instruments to promote environmental conservation.

Public use and forestry concessions are being implemented in Brazil. These are schemes in which the public administration allows private sector activities within protected areas under specific sustainable rules and raising funds for environmental agencies. Another private-public partnership with strong advance is the payment for environmental services programs, especially in forest conservation or restoration projects associated with watershed protection.

Carbon projects are being implemented mainly in the Amazon as a consequence of the reduction of emissions due to avoided deforestation until 2018, but this initiative can be used in other contexts. Finally, there is great expectation for future private investments in the market for environmental reserve quotas (CRA), a mechanism that allows trading rights to comply with the Brazilian forest legislation, detailed at the end of this section.

The initiatives discussed here are not homogeneously developed. They are in different degrees of implementation and distribution over the Brazilian territory. However, they have the potential to expand and generate important flows of financial resources and active participation of the private sector in the conservation of the Atlantic Forest and other biomes.

The environmental financing mechanisms are listed below and summarized in Table 21.2.

**Table 21.2** Environmental financing mechanisms available in Brazil and initiatives that can be used in the conservation of the Atlantic Forest

| Initiative | Federative level at which the mechanism can be implemented | Source of the financial resources |
|---|---|---|
| Sustainable public procurement | All | Public budget |
| Conversion of environmental fines | All | Private, imposed by the public sector |
| Environmental compensation | All | Private, imposed by the public sector |
| Ecological ICMS (ICMS-E) | States transfers to municipalities | Public budget |
| Green IPTU (urban property tax) | Municipalities | Public budget (reduction of property tax) |
| Public use (visiting and recreation) concessions | All | Public-private partnership |
| Forest concessions | Federal and states | Public-private partnership |
| Payments for environmental services | All | Public, private and international sources |
| Projects to prevent emissions from deforestation and carbon capture | Mainly federal government | Public, private and international sources |
| Environmental reserve quotas (CRA) | Federal and states | Private, imposed by the public sector |

### *21.3.1 Sustainable Public Procurement*

Sustainable public procurement refers to the use of the purchasing power of government entities to foster the procurement of goods and services from suppliers. This requires processes or products in accordance with environmentally responsible practices, with economic viability and social justice.

There is great potential for the use of sustainable public procurement in Brazil, while government spending represents between 10% and 15% of GDP (MMA 2009; Ribeiro and Júnior 2014; Cabral 2019). However, sustainable government procurement programs in Brazil are still rather timid. According to a report by the Federal Court of Auditors (TCU), sustainable purchases do not reach 1% of the total purchases made by the Federal Public Administration (TCU 2018). Thus, the expansion of this initiative can make the public administrations a strong inducer of sustainability in their supply chains.

In the Atlantic Forest biome, public procurement may rely on selective instruments that provide greater savings in energy, water, and other natural resources, recycled and recyclable products, purchase of products from agroforestry, etc. It is also possible to avoid the purchase of noncertified materials, which do not guarantee sustainable production.

### *21.3.2 Conversion of Environmental Fines*

The environmental fine, provided by Law N°. 9605/1998 (Brasil 1998), is a penalty imposed on agents who perform illegal acts regarding the protection and maintenance of the environment. This is an incentive to conservation since fines impose a cost on those who damage the environment. Besides being punitive, it is also educational, aiming to avoid further environmental damage in the future.

The conversion of environmental fines allows the defendant to have the penalty replaced by actions to restore or improve environmental conservation. This modality does not exempt the fined agent to repair the damage done, but allows the execution of activities and works linked to projects directly related to social and environmental policies.

In the federal government, IBAMA is responsible for proposing the conversion of fines into environmental projects (of direct execution by the offender or indirect execution, through actions taken by nongovernmental organizations). Its purpose is to avoid administrative and judicial appeals that delay the payment of fines, putting their effectiveness in check. In the case of subnational governments, their environmental agencies are responsible for these agreements, generally supported by the public prosecution service.

One of the problems with this type of financial instrument is the low effectiveness of fines collection in Brazil. Borges (2017) estimates that the total fines imposed by IBAMA between 2011 and 2016 reached US$ 9.2 billion, of which only US$ 0.2 billion (2.6% of the total) were effectively paid. A survey carried by the Brazilian Federal Court of Accounts (TCU 2015) also found that, out of 17 federal institutions, IBAMA ranked second to the last in its ability to collect fines. Other studies show that the same problem manifests itself in the states (Mariano 2018).

This is related to the judicialization of environmental fines in Brazil, and the delay in the prosecution of fines results in the loss of their effectiveness over time. Judicialization is a complex problem, related to the lack of coordination between the institutions involved in the many processes (monitoring, policing, enforcement), lack of technical or human resources, and little social perception of the importance of preserving common goods.

The fines are the result of irregular activities that cause damage to the environment, and to expand these resources would require even greater environmental damage. Therefore, it is not possible to design policies that need continued action or that are long term in scope for these initiatives, since the expectation is the reduction of environmental damage and not its continuity over time.

However, given the current scarcity of budgetary resources for conservation, this instrument must be applied as predicted by the legislation. The resources thus obtained should be allocated to implement environmental projects: a recent effort by the federal government (Decree N°. 9179/ 2017) established new possibilities to convert the environmental fines into conservation actions, like the recovery of native vegetation and degraded areas, biodiversity protection, and environmental education. Such conversion can anticipate the resolution of a conflict more quickly than if there were legal action and has a high potential to mobilize considerable resources for environmental projects with social justice, especially if civil society organizations are involved in their execution.

### 21.3.3 Environmental Compensation

Environmental compensation is an instrument established in Article 36 of Law No. 9985/2000 (Brasil 2000), which regulates the National System of Conservation Units (environmental protection areas). This rule stipulates that all projects with significant environmental impact (typified as a project that requires an environmental license for its acceptance) have to pay an environmental compensation, to be destined to the implementation or maintenance of conservation units. In other words, environmental compensation is an offset for environmental degradation, in which the social and environmental costs identified in the licensing process are incorporated into the overall costs of the entrepreneur (O Eco 2015).

The implementation of the environmental compensation, with the definition of the value and destination of resources, is the responsibility of the federative entity (federal, state, or municipal) responsible for the licensing of the project. This mechanism is implemented throughout the national territory, but it takes quite different forms since each federative entity can develop particular strategies for its realization.

Environmental compensation is an implicit form of the polluter pays principle since it establishes financial transfers from environmentally damaging projects to conservation. However, the currently implemented methodologies state that the environmental compensation must be proportional to the total financial cost of the project – at the federal level, there is a maximum limit of 0.5% of the value of the project to be charged as environmental compensation. This means that projects with significant environmental impact can cause damages whose economic values are much larger than the financial compensation to be paid. Therefore, this instrument usually implies in net losses to environmental conservation, since the financial compensation can be much lower than the social value of the problems that the projects may cause.

As in the case of environmental fines, it is not desirable that the volume of compensations increase in the long run, as this would mean a worsening of environmental conditions. Hence, this instrument should be thought of as a transition mechanism and that environmental externalities fees should be charged directly on those that cause environmental damage. Nevertheless, in the short run, the financial resources generated from environmental compensation contribute positively to the management and implementation of conservation units, especially in the current situation of constrain in the public budget for protected areas.

The Nature Conservancy (TNC 2015) estimated at US$ 486 million the amount of funds allocated by the Federal Environmental Compensation Committee (CCAF) accumulated between 2011 and 2014, to be invested in federal, state, and municipal conservation units, arising from environmental licensing projects under federal management. Also, according to TNC (2015), the main allocations of these funds were land property regularization (43%) and other uses defined by the managing body (42%). The third-largest use was the implementation of new protected areas (11%), followed by funding of protected area management plans (3%).

Using information from 19 states up to 2013, TNC (2015) estimated that the total accumulated volume of environmental compensation was approximately US$ 0.7 billion, but only 15% of this total was effectively executed. This low efficacy in using the resources is a consequence of operational difficulties and bureaucratic problems faced by public environmental agencies to enforce the environmental compensation payments.

In order to make this process less problematic, resources can be allocated directly to environmental compensation funds, managed by nongovernmental entities, which becomes responsible for the application of resources and the execution of associated projects. The constitution of compensation funds enables faster execution of resources than the direct execution by the public environmental agency. A pioneering experience of the environmental compensation fund is the Rio de Janeiro Atlantic Forest Fund (FMA). The FMA was established in 2009 by an agreement between the State of Rio de Janeiro and the NGO FUNBIO, which manages the resources as instructed by the Rio de Janeiro Environmental Compensation Chamber (Ilha and Albuquerque 2011). The FMA received US$ 124 million between 2009 and 2016 and executed US$ 48 million in conservation projects in protected areas in the State of Rio de Janeiro (FUNBIO 2018).

### *21.3.4 Ecological ICMS (ICMS-E)*

The tax on the circulation of goods and services (ICMS) is a value-added tax (VAT) collected by state governments, and part of these revenues must be redistributed among the municipalities. Three-quarters of this redistribution is defined by the federal constitution, but the remaining 25% is allocated according to each state's legislation. The ecological ICMS (ICMS-E), also called green ICMS, was developed by some Brazilian states to establish environmental criteria to distribute part of the ICMS among their municipalities (Castro et al. 2018).

Municipal resources obtained through ICMS-E transfers are not necessarily allocated to environmental expenditures and may be used for other purposes, such as supplementing municipal accounts and carrying out social policies. However, by establishing environmental criteria that need to be met for municipalities to receive a greater share of state resource transfers, the ICMS-E encourages positive competition among municipalities, which eventually improve their environmental management voluntarily (Castro et al. 2018).

The states in the Atlantic Forest that have active ICMS-E legislation are Rio Grande do Sul, Paraná, São Paulo, Goiás, Mato Grosso do Sul, Rio de Janeiro, Minas Gerais, Pernambuco, Ceará, and Piauí. The criteria for the distribution of ICMS-E in these states are quite varied and may include the size and characteristics of protected areas, water supply sources, waste collection and recycling systems, the characteristics and quality of environmental management, among others. The calculations for the distribution of state resources are also very diverse, with distinct methodologies to evaluate the municipal environmental performance (Castro et al. 2018).

According to Young et al. (2018), the 10 Atlantic Forest states with active ICMS-E legislation transferred US$ 1.7 billion by ICMS-E between 2012 and 2016. These funds were allocated to municipalities that met the environmental criteria established in state legislations, rewarding the results achieved and encouraging the maintenance of environmental quality. If the other seven Atlantic Forest states which do not have active ICMS-E legislation (Santa Catarina, Espírito Santo, Bahia, Alagoas, Sergipe, Paraiba, Rio Grande do Norte) instituted them, the volume of resources would be even bigger.

The ICMS-E has pros and cons in its application in the Atlantic Forest. One of the advantages is that ICMS-E does not increase public spending and does not impose a new tax on society, since it only redistributes taxes already collected based on criteria related to environmental performance. On the other hand, the calculations for the ICMS-E breakdown need to be clear and easy to understand, requiring greater transparency in the disclosure of criteria and the value of the transfers. The lack of knowledge of ICMS-E by local managers hinders the engagement of municipalities in more active environmental protection practices, reducing the effectiveness of this policy instrument.

### *21.3.5 Green IPTU (Urban Property Tax)*

The urban property tax (IPTU) is a municipal tax whose regulatory and collection structure is already implemented in all municipalities in Brazil. The green IPTU is a mechanism whereby discounts, or even exemption, are established on the amounts to be charged to the property owner according to environmental criteria (Dantas 2014).

This is an indirect form of funding for conservation, as it encourages private agents to improve environmental conditions in their properties that are of interest to the municipal manager in exchange for the reduction in property tax. Among the projects valued by environmental legislation are the maintenance of native trees and forests, rainwater harvesting, waste separation, and use of renewable energy, among other available technologies that help in environmental sustainability.

As this is a municipal initiative, there is no systematic record of these experiences, nor standardized legislation among the 5570 municipalities in Brazil. However, several municipalities located in the Atlantic Forest biome have green property taxes: Maringá and Curitiba (Paraná), Rio de Janeiro (Rio de Janeiro), Belo Horizonte (Minas Gerais), Araraquara, São Bernardo do Campo, São Carlos, and Guarulhos (São Paulo), among others (MMA 2015).

As the property tax is already charged in all Brazilian municipalities, it can be expected that other municipalities will adopt environmental criteria to promote discounts in the collection of the urban property tax. This can bring important benefits to the Atlantic Forest, as it is the biome with the highest urbanization rate in the country.

### *21.3.6 Public Use (Visiting and Recreation) Concessions*

Public use concessions for protected areas are contracts signed between the environmental protection authority and companies or civil society organizations, so that the latter perform a public service, at their own risk, by charging visitation and other tariffs or selling products in protected areas. Companies or social organizations are allowed to charge for tickets, transportation of visitors, restaurants, and convenience stores, among others, under rules defined by the public administration that usually requests a financial return to the government.

Besides financing the National System of Conservation Units (SNUC), there is also the objective to improve the quality of services provided to the visitors. The public environmental agencies are designed to focus on conservation, and do not have the skills or funding required to deal with services associated with public demands, such as access to lodging and food services (ICMBIO 2018).

On the other hand, increasing sales of goods and services locally produced benefit economically the communities around the protected area. Young et al. (2017b) estimated that 69% (US$ 5 million/year) of the total of the purchases made by the main private concessionary of the Iguaçu National Park (in Paraná) were produced in municipalities around the park, resulting in US$ 5 million in taxes collection per year. Together with other National Parks in the Atlantic Forest (Tijuca and Serra dos Órgãos, both in Rio de Janeiro), they generated US$ 130 million in revenues for the national environmental agency (ICMBIO) in the 2011–2018 period.

Public use concessions have been submitted by all federative entities. Besides the three parks mentioned above, other national parks in the Atlantic Forest (Itatiaia, in Rio de Janeiro, and Pau Brasil, in Bahia) had established public use concessions by early 2019, and bids for concession in other parks were under study or in process. State parks also present great potential for tourism and recreation. In São Paulo, for example, concessions were established for the state parks of Capivari and Campos do Jordão.

Rodrigues et al. (2018) estimated that the public use of all Brazilian protected areas could generate between US$ 0.7 and 1.7 billion per year, considering a number of 17 million visitors per year (base year 2016) and the multiplier effect of their expenses. Considering only the national parks, including those that do not have Atlantic Forest, the number of visitors is seven million per year, and the impact for local economies is estimated at between US$ 0.4 and 0.8 billion per year.

Given these high numbers, it is evident that public use concessions bring new sources of funding for maintaining or expanding protected areas, in a context where the fiscal crisis limits the budgetary capacity to execute these investments. These resources can establish a virtuous circle characterized by a situation in which improved quality in environmental management increases visitation, which also increases revenue from concessions, which generates more investments in environmental management, which in turn attracts new visitors, contributing to the protected area being a privileged locus of environmental education.

In addition, the concessions have accelerated the investment process in national and state parks in partnership with the private sector, which contributes to the economic and financial balance of environmental management in Brazil. However, the terms of the concession must strike a balance between private economic and financial results and the objectives of the public service related to the conservation of the area. It is also important to ensure the democratization of protected area access, since the management of these areas must remain with the public environmental authority (Rodrigues and Godoy 2013).

### *21.3.7 Forest Concessions*

Forest concession is a type of contract whereby companies or communities are entitled to manage public forests to extract timber and non-timber products, in exchange for the payment of a fee for the use of these natural resources, to be defined during the bidding process. The objective is to conserve the native vegetation cover while at the same time stimulating the economic viability of the forest and the quality of life of the population living in its surroundings. It is a way to keep the forest standing with reduced impact logging, allowing federal, state, and local governments to manage their forest assets. It should be noted that, in Brazil, it is guaranteed the free access of the local community to the concession area for the collection of non-timber products considered essential for their subsistence.

The revenues arising from these concession agreements are distributed to different agents, in addition to the environmental agencies. Part of the funds is earmarked for states, municipalities, and the National Forest Development Fund to finance sustainable projects that benefit local populations (Imaflora; SFB 2018).

The legal commercial exploitation of forest products allows their appreciation in the market and create jobs for the nearby communities. This strengthens the management of concession areas and reduces the illegal exploitation of natural resources, as the concentration of logging activities in formally legalized companies facilitates the work of monitoring compliance with environmental, labor, and fiscal standards.

However, forest concessions can only be thought of in a context of reduced deforestation, control, and enforcement of environmental legislation in the agreed activities. Without oversight of these activities, followed by penalization if illegal actions are undertaken, there is no incentive for the private agent to enter into concession agreements with the public sector.

Since the size of native forest remnants in the Atlantic Forest is relatively small, there is little room to implement forest concessions, which in Brazil are concentrated in the Amazon. However, there are exotic species planted in Atlantic Forest conservation units that can be commercially exploited, helping the plantation of native species in place of those that are eliminated. A new feature of the 2019 Forest Grant Plan (SFB 2018) is the inclusion of the Três Barras National Forest (Santa Catarina State) as a concession area to exploit exotic species. This will be the first conservation unit in the Atlantic Forest to receive studies for structuring a sustainable forest management project. If successful, this experience can pave the way for sustainable exploitation in other protected areas of the Atlantic Forest that have a vocation for such.

### *21.3.8 Payments for Environmental Services*

There are several ways to implement a payment for environmental services (PES) system. In general, it can be described as a voluntary or legally agreed arrangement whereby actions that result in the conservation of ecosystem services are awarded financially or through some other economic incentive (Wunder 2005; Young and Bakker 2014).

In the Atlantic Forest, the most developed PES schemes are those of forest conservation or restoration that result in the protection of water resources (Young et al. 2016). There is a wide variety in the source of funds used for payment and in the allocation criteria, but these PES experiences have in common some kind of compensation to landowners who decide to conserve or expand native forest areas. The main sources of financial resources are public funds and donations, but it is expected that in the future, it will be possible to establish a user-pay scheme, where those who benefit from the environmental services pay to those who protect them, establishing a stable source of funding.

The impact of PES programs can be significant. Considering a hypothetic payment for water use throughout all Brazilian territory, Young et al. (2016) estimate that a 2% charge on water bill could generate US$ 215 million per year, capable of financing PES systems aimed at restoring up to 363,000 hectares of native forests, capturing up to 125 million tons of $CO_2$, and avoiding erosion of up to 3.7 million tons per year of soil.

However, the creation of a national regulatory framework on the subject is still necessary. There are different bill proposals in the National Congress (Brasil 2013, 2015), and in September 2019, the Chamber of Deputies approved the PL N°. 312/15 proposing to institute the National Policy for Payment for Environmental Services (yet to be approved by the Senate). However, the costs of implementing and maintaining this policy, including the expenses with inspection and monitoring of the properties involved, as well as the rules for granting resources need to be clarified.

### *21.3.9 Projects to Prevent Emissions from Deforestation and Carbon Capture*

Deforestation is the main source of greenhouse gas (GHG) emissions in Brazil. As a result, there is a strong emphasis on proposing programs that reduce carbon emissions through actions that prevent deforestation and conserve biodiversity, known in the literature as REDD+. The two most important actions in this area are in the Amazon biome: the Amazon Fund and the Green Climate Fund.

The Amazon Fund is a joint initiative of the federal governments of Brazil, Norway, and Germany that transfers international cooperation resources to sustainable development projects. The guidelines and monitoring of project results are the responsibility of the fund's management led by the National Bank for Economic and Social Development (BNDES). The granting of funds is linked to the Brazilian commitment to reduce deforestation in the Amazon. Until mid-2019, the Amazon Fund had a portfolio of 103 projects supported, with US$ 484 million already disbursed, and another US$ 236 million available for new disbursements (Fundo Amazônia, http://www.fundoamazonia.gov.br/). Although the main focus is on the Amazon, up to 20% of the fund's resources can go to developing deforestation monitoring and control systems in the rest of Brazil and other tropical forest countries. For this reason, projects located in the Atlantic Forest in Bahia, Sergipe, and Mato Grosso do Sul have already benefited from the Amazon Fund. However, since 2019, there is a dispute between the Brazilian Federal Government and the donors that has blocked the disbursement of resources from the Amazon Fund, in which the future remains uncertain.

The Green Climate Fund (GCF) was created after the 2015 Paris Agreement and has a total portfolio of US$ 5.2 billion already committed (Green Climate Fund, s/d). The FP 100 Project was the first avoided deforestation and forest conservation (REDD+) project eligible to obtain GCF resources, having received US$ 96 million for reducing deforestation in the Brazilian Amazon between 2014 and 2015. Although 80% of the resources must be destined for the conservation and restoration of native vegetation in the Amazon (Forest + Program), the remainder can be used to strengthen the implementation of the National REDD+ Strategy (ENREDD+). In addition, FP 100 shows the way for similar REDD+ projects in other biomes. Until January 2020, these resources have not yet been used, since the Brazilian Federal Government has not, so far, implemented the institutional requirements established in the contract.

The Atlantic Forest is, by far, the most deforested biome in Brazilian history. Nevertheless, in the last decades, its deforestation rate (and potential for REDD+ projects) is smaller than deforestation in the Amazon and Cerrado biomes. On the other hand, there is a huge potential for forest recovery. Restoration projects are eligible for most carbon markets, but the values are often much lower than those for energy emission reduction credits. As a result, there are few large-scale carbon market-based native vegetation recovery projects. However, these projects have great potential if we consider the combination of other conservation benefits, especially the protection of water resources. Thus, even if the isolated value of carbon capture payments is monetarily low, it can be combined with payments from water-oriented PES systems or other forms of conservation stimulation (as in CRA transactions), so as to increase compensation to economic agents who decide in favor of forest recovery.

### *21.3.10 Environmental Reserve Quotas (CRA)*

The Native Vegetation Protection Law N°. 12.561 (Brasil 2012) sets minimum standards for native forest protection on private properties, called the legal reserve (RL). The law also states that properties that do not meet the minimum percentages of RL may compensate for their deficit in other properties, provided that the areas are ecologically equivalent (Young et al. 2017a).

The environmental reserve quota (CRA) is an economic mechanism used to offset RL deficits in private properties. This is a negotiable legal title representative of areas with intact or regenerating native vegetation that exceeds the minimum RL requirements of a property. This creates the possibility that landowners who have forest assets (beyond the legally mandated threshold) will negotiate with those who have forest liabilities by establishing a CRA market (Young et al. 2017a).

Since they are private transactions between CRA buyers and sellers, there is no need for public financing, and public intervention is restricted to the regulatory framework, with the supervision and monitoring of the conditions imposed by the legislation. A positive incentive is created for the maintenance of native vegetation on farms above the minimum levels legally required for RL compliance, producing environmental corridors and protected areas on a smaller number of properties, but with larger dimensions.

From a legal point of view, the basic mechanisms needed to define a CRA market are already in place. However, relevant administrative actions are still needed, such as the completion of the Rural Environmental Registry, which maps the deficit or surplus of native forests in each property; a federative framework that establishes the competencies of the various entities involved (federal, state and eventually, municipal); and the liaison with the financial market for transactions to be carried out.

The biggest challenge is the expected resistance of most landowners who have legal reserve deficits, especially in the Atlantic Forest, where the opportunity cost of land is the highest in the country. That is, in areas where agricultural productivity is higher, it will be more difficult to find surpluses of native vegetation for the generation of CRAs that, therefore, will tend to increase, forming resistances to the use of this mechanism. On the other hand, the larger the supply of CRAs, with the possibility of acquiring securities in other states or biomes, their price will tend to decrease, and, consequently, there will be less incentive for private forest conservation actions (Gasparinetti and Vilela 2018).

Therefore, it is necessary to define whether the CRA market will be nationally covered or restricted to transactions within the biome itself. This has implications for the value of CRAs to be negotiated: CRA prices will be lower in the Atlantic Forest in a national market that allows transactions in different biomes. The simulations of Young et al. (2017a) corroborate this perception. Considering a CRA market restricted to the Atlantic Forest, they estimate that up to 2.2 million hectares of legal reserve deficit could be offset, moving US$ 8.8 billion, with an average CRA value of US$ 4.1 thousand/ha, capturing 710 million tons of $CO_2$. If an unrestricted CRA market is established, the average price would be much lower (US$ 1.3 thousand/ha), but very little restoration would occur in the Atlantic Forest, as it would be cheaper to recover forests in Caatinga, Amazônia, and Cerrado.

## 21.4 Final Remarks

As seen throughout the chapter, there are several funding possibilities for Atlantic Forest conservation. Some of the experiences analyzed are aimed at improving the allocation of resources within the public sector itself, as in the case of sustainable public procurement and the adoption of environmental criteria in fiscal policy, both in tax collection (green IPTU) and in the distribution of fiscal resources (ecological ICMS).

Another part of the initiatives requires the protagonist role of the private initiative, which has a greater capacity to mobilize resources, but always with the need to comply with environmental legislation. Examples of this kind of public-private partnership are public use concessions and forest concessions. In these experiments, the public sector sets rules and operating conditions, but the dynamism is centered on private agents. Although profit-oriented, such activities can result in synergistic benefits from private investments for biodiversity conservation and ecosystem services, as well as resource generation for public agencies responsible for environmental management.

In other words, the search for solutions that coordinate public and private actions in the same direction seems to be the way to overcome the lack of resources for the conservation of the Atlantic Forest. The fiscal crisis and the limitation of command and control instruments point to the need for more space for private agents to act in the management of forest remnants. Even in actions primarily based on private actions, as CRA, the participation of the government in the success of these financing solutions is essential. Compliance with environmental legislation and contracts, including to monitor and penalize violators, is a prerequisite for the legal certainty and stability environment necessary for private agents to invest in sustainable business.

The identification of various experiences implemented or being implemented is a positive sign that environmental management in the Atlantic Forest has advanced in incorporating economic instruments. However, the current fiscal crisis and the poor operating conditions of environmental management bodies put these advances at risk. The political priority and the volume of investment, both private and public, need to be increased to encourage the much-needed restoration of the Atlantic Forest and the other endangered biomes in Brazil.

## References

Alvarenga M Jr (2019) Payment for ecosystem services: an alternative for the Brazilian Amazon. Final paper: 42nd UNEP/UNESCO/BMU International Postgraduate Course on Environmental Management for Developing Countries (EM42). Technische Universität Dresden (TUD), Dresden

Borges A (2017) Governo publica decreto que converte multa do Ibama em ação socioambiental. O Estado de São Paulo, 17/08/2017

Brasil (2000) Lei Federal n° 9.985/00, de 18 de julho de 2000

Brasil (1998) Lei N° 9.605, de 12 de fevereiro de 1998

Brasil (2015) PL N° 312/2015. Institui a Política Nacional de Pagamento por Serviços Ambientais e dá outras providências

Brasil (2013) PL n° 276/2013. Institui a Política Nacional de Pagamento por Serviços Ambientais

Brasil (2012) Lei Federal N° 12.561, de 25 de maio de 2012

Brancalion PHS et al (2019) Global restoration opportunities in tropical rainforest landscapes. Sci Adv 2019, EAAV3223. 11p

Cabral VN (2019) Implementação da política de compras públicas sustentáveis em uma instituição federal de ensino: uma análise a partir da Matriz 5C de Najam. Programa de Pós-Graduação em Administração. UFRRJ, Seropédica. 119 p

Castro BS, Correa MG, Costa DS, Costa LAN, Medeiros R, Young CEF (2018) Geração de receitas tributárias municipais. In: Young CEF, Medeiros R (org) Quanto vale o verde: a importância econômica das unidades de conservação brasileiras. 1ed. Rio de Janeiro: Conservação Internacional, vol 1. pp. 148–173

Crouzeilles R, Rodrigues RR, Strassburg BBN (2019) Relatório temático sobre restauração de paisagens e ecossistemas (livro eletrônico). Editora Cubo, São Carlos, SP

Dantas GT (2014) IPTU verde e o direito à cidade sustentável. Revista do Programa de Pós-Graduação em Direito da Universidade Federal da Bahia 24(26)

FUNBIO (2018) Relatório Anual FUNBIO 2017. Disponível em: https://www.funbio.org.br/wp-content/uploads/2019/02/Funbio_Relat%C3%B3rio-Anual_2017_Portugu%C3%AAs.pdf

Fundo Amazônia. Available in: www.fundoamazonia.gov.br/pt/home

Gasparinetti P, Vilela T (2018) Implementando Mercados de Cotas de Reserva Ambiental (CRA) nos Estados Brasileiros: Desafios e Oportunidades para as Regulamentações Estaduais. Conservação Estratégica

Green Climate Fund. Available in: www.greenclimate.fund/home

Ilha AS, Albuquerque DP (2011) O Fundo da Mata Atlântica do Rio de Janeiro. In: Congresso Brasileiro de Unidades de Conservação, 7. Fundação Grupo Boticário de Proteção à Natureza, Natal. Anais... Curitiba, pp 1–10

IBGE (2018). Projeção da população por sexo e idades simples, em 1° de julho - 2010/2060. Disponível em: https://www.ibge.gov.br/estatisticas/sociais/populacao/9109-projecao-da-populacao.html?=&t=resultados

Imaflora; Serviço Florestal Brasileiro - SFB (2018) Concessões Florestais Federais: participação, transparência e efetividade no uso dos recursos dos estados, municípios e comunidades locais

Instituto Chico Mendes de Conservação Da Biodiversidade – ICMBIO (2018) Parques do Brasil: Visitar é proteger! Estratégias de implementação da visitação em unidades de conservação federais: prioridades de execução 2018-2020. Coordenação Geral de Uso Público e Negócios – ICMBio

IPBES (2018) The IPBES regional assessment report on biodiversity and ecosystem services for the Americas. In: Rice J, Seixas CS, Zaccagnini ME, Bedoya-Gaitán M, Valderrama N (eds) Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Bonn, Germany. 656 p

Mariano, R. (2018) Estado pune empresas por danos ambientais, mas só 7% das multas são pagas. Portal de Notícias Hoje em dia. 28/03/2018

Medeiros R, Young CEF (2011) Contribuição das unidades de conservação brasileiras para a economia nacional: Relatório Final. 1 – Brasília: UNEP-WCMC, 120p

MMA (2009) Agenda Ambiental na Administração Pública (A3P). Cartilha da Secretaria de Articulação Institucional e Cidadania Ambiental. Departamento de Cidadania e Responsabilidade Socioambiental. Brasília, 5ª Edição

MMA (2015) Serviços de consultoria de pessoa jurídica – Relatório final, classificação das iniciativas de aplicação de instrumentos econômicos para a implementação de política ambiental. Revisão dos dados e contribuição dos operadores. Brasília, Ministério do Meio Ambiente

Myers N, Mittermeier RA, Mittermeier CG, Da Fonseca GA, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403(6772):853

O Eco (2015) O que é a Compensação Ambiental. 30 janeiro 2015. Disponível em: https://www.oeco.org.br/dicionario-ambiental/28899-o-que-e-a-compensacao-ambiental/. Acessado em: jan. 2020

Ribeiro C, Júnior E (2014) Mensurando o mercado de compras governamentais brasileiro. Caderno de Finanças Públicas 14:265–287

Rodrigues CGO, Godoy LRC (2013) Atuação pública e privada na gestão de unidades de conservação: aspectos socioeconômicos da prestação de serviços de apoio à visitação em parques nacionais. Desenvolvimento e Meio Ambiente (UFPR) 28(p):75–88

Rodrigues CGO, Fontoura LM, Rosa CR, Medeiros R, Young CEF (2018) Turismo e Uso Público. In: Young CEF, Medeiros R (eds) Quanto vale o verde? A Importância Econômica das Unidades de Conservação Brasileiras. Conservação Internacional, Rio de Janeiro, pp 79–103

Serviço Florestal Brasileiro (SFB) (2018) Plano de Outorga Florestal 2019 [recurso eletrônico]/ Serviço Florestal Brasileiro. MMA, Brasília, DF

Tribunal de Contas da União (TCU) (2018) Relatório Anual de Atividades: 2017 – Brasília

Tribunal de Contas da União (TCU) (2015) Relatório e parecer prévio sobre as contas do governo da república 2013. Brasília: TCU

The Nature Conservancy (TNC) (2015) Compensação Ambiental. Um retrato sobre o cenário brasileiro. Brasília: TNC, 37p

Wunder S (2005) Payments for environmental services: some nuts and bolts. CIFOR Occasional Paper N°. 42. Center for International Forestry Research, Jakarta

Young CEF, Castro BS, Castro RR, Santanna AA, Erazo JA, Bakker LB, Alvarenga Junior M, Young MB, Pereira V, Aguiar C, Tornaghi L, Costa DS, Mendes M, Costa LAN (2016) Relatório 05 (Final). Análise consolidada e apresentação dos subsídios técnicos para a construção de Política Nacional de Pagamento por Serviços Ambientais. (Relatório de pesquisa). MMA/PNUD

Young CEF, Castro BS, Mathias JFC, Penna C, Pereira GS, Ferraz C, Alvarenga M Jr, Machado M, Gatto DB, Batista AK, Arrellaga MM, Aguiar C, Duque D, Araújo KS, Jordão CS, Goncalves R, Fares LR, Manzatto LHR, Costa DS (2018) Relatório 4 (Final): Resultado 7. Análise conjuntural sobre ODS e efetividade das estruturas de financiamento públicas, privadas e mistas para a promoção do desenvolvimento sustentável aprimoradas. (Relatório de pesquisa). MMA/PNUD

Young CEF, Bakker LB (2014) Payments for ecosystem services from watershed protection: a methodological assessment of the Oasis Project in Brazil. Natureza Conservação 12:7178

Young CEF, Alvarenga M Jr, Gandra FM, Costa LNA, Mendes MP (2017a) Custos e Benefícios da Implementação de um Mercado de Cotas de Reserva Ambiental (CRA) no Brasil. XII Encontro Nacional da Sociedade Brasileira de Economia Ecológica

Young CEF, Medeiros R (org) (2018) Quanto vale o verde: a importância econômica das unidades de conservação brasileiras. 1ed.Rio de Janeiro: Conservação Internacional

Young CEF, Rocha ERP, Bakker LB, Santoro AF (2014) How green is my budget? Public environmental expenditures in Brazil. In: Dantas AT, Nowak AZ, Siuda-Ambroziak R (org) Brazil-Poland. Focus on economy, 1st ed. Warsaw: University of Warsaw. pp. 199–210

Young CEF, Roncisvalle CA (2002) Expenditures, investment and financing for sustainable development in Brazil, 1st edn. United Nations Publications, Santiago de Chile. 58p

Young CEF, Alvarenga M Jr, Sousa FH, Costa LAN, Mendes MP (2017b) Conservação ambiental, concessões privadas e dinamismo econômico: estudo de caso do Parque Nacional do Iguaçu. In: Encontro Nacional da Sociedade Brasileira de Economia Ecológica, vol 12. SBEE, Uberlândia. Anais... Uberlândia

# Chapter 22
# Integrating Researchers for Understanding Biodiversity in Atlantic Forest

**Helena Godoy Bergallo, Cecília Cronemberger, Juliana Hipólito, William E. Magnusson, and Carlos Frederico Duarte Rocha**

**Abstract** Long-term and interdisciplinary studies are needed to understand biodiversity and unravel ecological processes, test ecological hypotheses, propose public policies, and help decision-makers, especially with regard to human impacts and their consequences on the environment. These studies began with the Long-Term Ecological Research Program (LTER), generating knowledge about Brazilian ecosystems and their biodiversity. However, an integrative methodology was lacking in LTER sites, which was obtained using the RAPELD method developed in an Amazonian LTER site. In 2004, the Biodiversity Research Network Program (PPBio) was created in the Amazon and semiarid biomes, based on the strategy of creating regional hubs that served less favored regions in terms of human resources training and with great potential for biodiversity. In 2012, PPBio was expanded to the other Brazilian biomes. In a relatively short time, PPBio achieved consistent results, largely due to the use of RAPELD as an integrative methodology, which led to greater integration of information on different biotic and abiotic factors, as well as the availability of data in public repositories. LTER and PPBio are complementary networks of great relevance for the knowledge and conservation of Brazilian biodiversity, and their integration needs to be strengthened in the Atlantic Forest.

**Keywords** Biodiversity · PELD · Integrative Methodology · RAPELD · Network research · PPBio

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## 22.1 From Single to Integrated Studies

Modern science is increasingly collaborative in all research fields (Sidone et al. 2016), and research in ecology has long been collaborative due to the complexities that it deals with, but interdisciplinarity is becoming increasingly important as the complexity of the ecological problems facing humanity increases (Goring et al. 2014; Hobbie et al. 2003). Formulating and testing ecological theory demands long-term interdisciplinary study and long-term experiments, particularly considering recent global changes (Barbosa 2013; Hobbie et al. 2003).

Long-term collaborative research projects have increased since the establishment of the US Long-Term Ecological Research (US LTER) in 1980. By the 1990s, a consensus was established that many ecological questions require a particular study site to be investigated for a long time (Gosz 1998). In 1993, scientists from 16 countries meeting in the USA identified a "growing need for global communication and collaboration among long-term ecological researchers and to capture ecological phenomena in the context of global change." As a result, international LTER (ILTER) was founded during a meeting of researchers from the US Long-Term Ecological Research (US LTER) (Forsberg 1993). Strayer et al. (1986) pointed out that designing and operating such studies requires careful consideration of the potential problems if they are to be effective in the long term. Some of the challenges of interdisciplinary collaborative projects include the need for clear governance schemes and an explicit data-sharing policy that includes standard protocols for metadata, data quality, intellectual property, and other issues (Penman et al. 2011).

## 22.2 PELD, the Long-Term Research Program of Brazil

Until about 40 years ago, most researchers in Brazil were concerned with relatively short-term ecological questions, and long-term research was virtually nonexistent. Most ecological studies at that time were primarily involved with questions related to organismal-level biology (e.g., ecophysiology, population dynamics, development, and parasitism) or descriptive community studies (e.g., succession, species interactions, and productivity). These generally did not involve questions that require long-term studies to obtain answers, but they were the building blocks over which long-term studies would later develop. In addition, long-term research requires continuous funding, which was not available at the time.

In 1996, the Brazilian Forum of Coordinators of the Graduate Programs in Ecology proposed to establish an integrated program in Ecology, the Brazilian PIE (Barbosa et al. 1998, 2000). As a result, the Brazilian National Science Foundation (Conselho Nacional do Desenvolvimento Científico e Tecnológico – CNPq) adopted PIE as one of its programs, which was structured with a subprogram focusing on capacity building and research, and another subprogram specifically on long-term research, the “Pesquisas Ecológicas de Longa Duração” (PELD). Later, PIE was incorporated in PELD (Barbosa 2013; Mamede et al. 2013). This was the starting point for long-term studies in Brazil, and the PELD became a member of ILTER. This Brazilian initiative occurred soon after ILTER was established in 1993, and the Brazilian program was one of the pioneers in long-term ecological research. An initial set of nine potential sites were identified to establish long-term studies in Brazil (Barbosa 2013).

PELD/CNPq aims to generate knowledge on Brazilian ecosystems and their biodiversity and to promote the transference of such knowledge to civil society. Its priorities are to propose policies for the development of ecology in Brazil (thus including interactions and processes as well as organismal-level or descriptive studies), to develop research and information networks, to support international cooperation, to propose methodological standardization, to increase human-resource training and institutional development, and to act harmoniously with other government programs, such as the sustainable development policy committee and the National Agenda 21 (Barbosa 2013).

PELD has had an increasing number of sites since the first call for the establishment of sites was made by CNPq in 1997. Nine sites were established in 1999 with relatively restricted funding, with no guarantee of future support. However, in 2000, CNPq included a specific budget for this program in the Brazilian Pluriannual Plan (PPA). In 2001, CNPq made a second call, and three additional sites were aggregated to the program, then with 12 sites. However, for a megadiverse country, such as Brazil, encompassing at least six biomes, the number of LTER sites was still small and not representative of Brazilian ecosystems. In 2009, supplementary funds allowed a third call, which brought 14 additional sites into the program. By then, there were a total of 26 sites. This represented a marked expansion of PELD which improved its distribution among Brazilian landscapes. Since then, PELD has aggregated funds from 11 state funding agencies (states of Amazonas, Bahia, Goiás, Minas Gerais, Mato Grosso do Sul, Mato Grosso, Paraná, Rio Grande do Sul, São Paulo, Rio de Janeiro, and Pará). With these additional funds, CNPq made a fourth call in 2012, and another four sites were aggregated, giving the program’s present 31 LTER sites (Mamede et al. 2013). Of these, a large portion (15 sites or about 48%) was established in the Atlantic Forest Biome or in ecotone ecosystems adjacent to the Atlantic Forest (one site). There is a bias in the number of sites for the Atlantic Forest biome compared to the Amazon and Cerrado that have larger territorial extensions. This bias reflects the concentration of Brazil’s scientific institutions and, consequently, most of the country’s science production in all research fields, in the Atlantic Forest Domain (Motta et al. 2002; Sidone et al. 2016). These research sites housed studies on a wide range of issues, including ecosystems, habitats, environments, and landscapes, such as Atlantic Forest physiognomies at different altitudes, Araucaria forests, restingas, mangroves, floodplains, estuaries, coastal lagoons, and transitions between Atlantic Forest and other ecosystems (Table 22.1 and Fig. 22.1).

The initial challenges of the first PELD sites were to not only produce scientific products in quantity but also analyze how the information generated shed light on

**Table 22.1** PELD projects in the Atlantic Forest region

| Acronym | Full name | Main institution | State | Beginning year | Ecosystem |
|---|---|---|---|---|---|
| MLRD | Atlantic Forest and lacustrine system of the middle Rio Doce | Universidade Federal de Minas Gerais | Minas Gerais | 1999 | Atlantic Forest, freshwater lakes |
| ELFA | Araucaria forests of Southern Brazil | Universidade Federal do Paraná | Paraná | 1999 | Araucaria forest, Atlantic Forest-araucaria transition |
| PIAP | The Upper Paraná River floodplain | Universidade Estadual de Maringá | Paraná | 1999 | Rivers, lakes, floodplains |
| RLaC | Restingas and coastal lagoons of the northern Rio de Janeiro | Universidade Federal do Rio de Janeiro | Rio de Janeiro | 1999 | Restinga, coastal lagoons |
| ELPA | Patos lagoon estuary and adjacent coast | Universidade Federal do Rio Grande | Rio Grande do Sul | 1999 | Coastal lagoon, estuary, beaches |
| PEBG | Guanabara Bay, Rio de Janeiro | Universidade Federal do Rio de Janeiro | Rio de Janeiro | 2009 | Marine ecosystem, mangroove, beaches |
| CSUL | South fields of Pampa and Atlantic Forest biomes | Universidade Federal do Rio Grande do Sul | Rio Grande do Sul | 2009 | South fields, Atlantic Forest |
| BROA | The ecology of UHE Carlos Botelho (Lobo-Broa Reservoir) and its watershed, São Paulo, Brazil | Instituto Internacional de Ecologia | São Paulo | 2009 | Lakes, riparian forest |
| FGAF | Functional gradient of Atlantic Forest | Universidade Estadual de Campinas | São Paulo | 2009 | Atlantic Forest |
| CRSC | Rock outcrops of Serra do Cipó-MG | Universidade Federal de Minas Gerais | Minas Gerais | 2012 | Rock outcrops, Cerrado-Atlantic Forest transition |
| LAG | Lagamar | Universidade Federal do Paraná | Paraná | 2012 | Atlantic Forest |
| MCF | Central fluminense mosaic of parks and reserves | Universidade Federal do Rio de Janeiro | Rio de Janeiro | 2012 | Atlantic Forest |
| ITA | Structure and dynamics of communities in coastal rivers and streams from the Atlantic Forest – Itanhaém river basin | Universidade Estadual Paulista | São Paulo | 2012 | Rivers, riparian forest |

(continued)

**Table 22.1** (continued)

| Acronym | Full name | Main institution | State | Beginning year | Ecosystem |
|---|---|---|---|---|---|
| MAAM | Forest restoration in Atlantic Forest and Amazonia | USP-ESALQ | São Paulo | 2012 | Atlantic Forest |
| MANP | Brazilian orth Parana state seasonal Atlantic Forest | Universidade Estadual de Londrina | Paraná | 2014 | Atlantic forest |

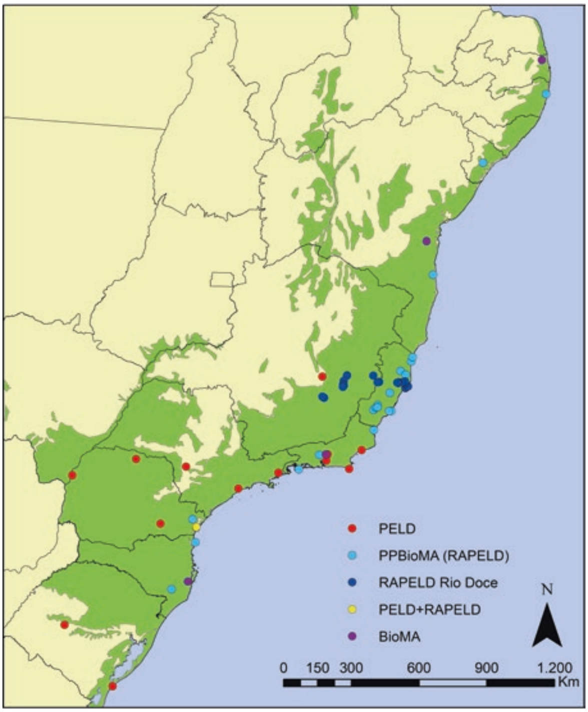

**Fig. 22.1** Location of PELD sites (red circles), RAPELD (blue circles; light blue: PPBio sites of the PPBioMA network; dark blue: monitoring sites along Rio Doce), PELD site that applies RAPELD (yellow circle) and PPBio sites of BioM.A. network (purple circles) throughout the Atlantic Forest and transition ecosystems (green area), as delimited by the Atlantic Forest Law. (Federal Law 11,428/2006, IBGE 2012)

long-term issues. Long-term ecological research involves the understanding of processes, which requires the identification of mechanisms that affect the processes and the interactions between them. Also, human populations exert increasing pressures on natural resources, leading to the erosion of biodiversity and generating global effects, such as climate change and species extinction. All this requires time to be understood (Mamede et al. 2013; Tabarelli et al. 2013). During the initial stages, data produced in some sites were not linked to long-term issues, but over the years, with the implementation of new sites and evaluation of PELD by CNPq, this culture is changing, resulting in each site increasingly reflecting on the information base it has generated and how this can be used to address long-term questions. Nevertheless, there is a persistent need for standardization of methodologies among sites. Ideally, this will facilitate data collection following the same methodological procedures and spatial scales, guarantee the statistical independence of the observations, and allow comparisons among sites, even those in other countries or continents. This does not imply that only standardized studies can be undertaken at each site. Rather, a subset of studies in each site should use standardized methodologies to permit within and among biome comparisons. We have come a long way in standardizing methods, but we need to understand that science is a culture, and to change culture we need to teach young people the ethical, scientific, and financial values of integration. We would go a long way as well if standardized methodologies were required or recommended in public calls and environmental-impact studies. One method for such standardization is the RAPELD method, which was developed within one of the original PELD sites, and this could form the basis for part of the standardization within the program (Magnusson et al. 2005, 2008).

After nearly two decades, most sites have generated information that increases possibilities for biodiversity conservation, comprehension of ecological processes, their function and dynamics, long-term ecological responses to pluriannual variation, knowledge of human impacts on different ecosystems, and knowledge of hydrological systems. These give guidance for sustainable regional development, public policies, dialogue with multiple stakeholders, and training of human resources within the scope of each project (Mamede et al. 2013; Tabarelli et al. 2013). PELD has been strategic not only to improve the knowledge of biodiversity and conservation and to provide subsidy for stakeholders, but also to stimulate a scientific culture embracing long-term ecological research as fundamental to conciliate human well-being and the sustainable use of natural resources. Together with the other PELD sites in Brazil, the Atlantic Forest-biome sites are contributing to an important set of long-term actions and integrated research in the Atlantic Forest biome and with other biodiversity programs, such as the Biodiversity Research Program (Programa de Pesquisas em Biodiversidade – PPBio) of the Ministry of Science, Technology and Innovations (MCTI). The PPBio adopted the RAPELD method in most of its field sites, so integration between PELD and PPBio promises to increase greatly the number of LTER sites in the Atlantic Forest and the possibility of comparisons among them.

## 22.3 Searching for an Integrated Methodology: RAPELD

There was a proliferation of biodiversity monitoring schemes in the 1980s and 1990s, which collectively gained the name of environmentally distributed ecological networks (EDENs) (Craine et al. 2007), and a group of Amazonian researchers started to search for field methods that would allow integration of data from multiple sites. Most previous attempts at standardization had focused on capture or measurement protocols and ignored the problem of spatial standardization. Some plot-based monitoring schemes were designed primarily for plant studies, and their scales are generally not suitable for other organisms (e.g., CTFS, RAINFOR Projects), or when designed to monitor multiple organisms, they are not standardized at different scales (e.g., TEAM), or the scales are small and not relevant to managers (e.g., GLORIA) (Magnusson et al. 2013). Most researchers only considered spatial aspects within their local study site, did not realize the difficulty of integrating studies that have no common spatial scale, and reported their results as though they could be generalized to landscapes, regions, or continents.

The problem is that the effect of most variables is strongly and nonlinearly related to spatial scale (Koblitz et al. 2017). For instance, variation in soil attributes might have very little effect on the composition of plant assemblages within study sites covering several meters, large effects in sites covering kilometers, no detectable effect on comparisons among regions, and variable effects in comparisons among continents. Also, the results for most academic studies were for spatial scales largely irrelevant for managers of protected areas and decision-makers in municipalities, states, or countries. It is not possible to standardize at all scales, but the objective was to have a sampling design that would allow comparisons among sites at a few spatial scales of interest to land managers and that would allow calibration and validation of remote-sensing methods that could allow scaling up at little additional cost (Magnusson et al. 2013).

Most biodiversity-monitoring projects focused on only a few taxa (e.g., woody plants), on a limited range of habitats (e.g., mountain tops) or limited to a particular technology (e.g., camera-trapping). Their designs normally also made it difficult to include data on abiotic drivers, such as soils, hydrology, and topography, which are essential to allow the evaluation of human-induced changes and natural variation across landscapes. Many designs were investigated over a period of more than a decade (Koehler 2000, Jucevica and Melecis 2005, Magnusson et al. 2005, 2013, among other examples below) before the group settled on a compromise arrangement that allowed the inclusion of almost all taxa and environmental drivers, while deviating as little as possible from traditional methods to facilitate the integration of previous studies when possible (Magnusson et al. 2005).

The system, called RAPELD (rapid assessment surveys (RAP) + PELD – from the Brazilian acronym for LTER), was first installed in Reserva Ducke, which is part of the Long-Term Ecological Research (LTER) site Amazon Forest – Manaus, which was then Site #1 of the Brazilian LTER. The design is basically a modular network of trails and permanent plots that can be accommodated to sample all elements of biodiversity at scales of 1 km or greater (Costa and Magnusson 2010).

The first study using RAPELD methodology from Reserva Ducke was published in 2005 (Kinupp and Magnusson 2005), but there are now over 50 papers published using RAPELD methodology that included data from the reserve. These cover many taxa and life forms, including mites, frogs, butterflies, bats, fish, herbs and trees, and processes such as groundwater availability and carbon storage. Data are made available locally and in international networks, such as Data One, RAINFOR, and ATDN. The number of publications from Reserva Ducke had been relatively limited before the installation of the RAPELD grid, but by 2008, it was considered one of the most productive research sites in the neotropics (Pitman et al. 2011).

RAPELD methodology is required by the Brazilian Environmental Authority (IBAMA) in environmental impact assessments for some types of infrastructure development, such as hydroelectric dams and highways. These have resulted in information on the status of trees (Moser et al. 2014), harpy eagles (Aguiar-Silva et al. 2015; Sanaiotti et al. 2015), bats (Bobrowiec and Tavares 2017), frogs (Carneiro et al. 2016; Koblitz et al. 2017; Ortiz et al. 2018), lizards (Morães et al. 2016), snakes (Fraga et al. 2014, 2017, 2018), and processes affecting multiple taxa (Santorelli et al. 2018).

RAPELD modules installed in Australia have revealed information on taxa ranging from koalas to forest trees (Hero et al. 2010, 2013; Lollback et al. 2017). Experimental modules have also been installed in Nepal and Liberia, but the only country besides Brazil with an actively expanding network of RAPELD modules at the moment is Argentina (https://ppbio.inpa.gov.br/en/PPBio_International/PPBio_Argentina). Within Brazil, the Program for Biodiversity Research (PPBio) has installed RAPELD modules in long-term ecological research sites in all biomes except the semiarid, and there are now more than 130 distributed across the country (Peixoto et al. 2016, Fig. 22.2). The biome with the largest number of RAPELD modules outside of Amazonia is the Atlantic Forest.

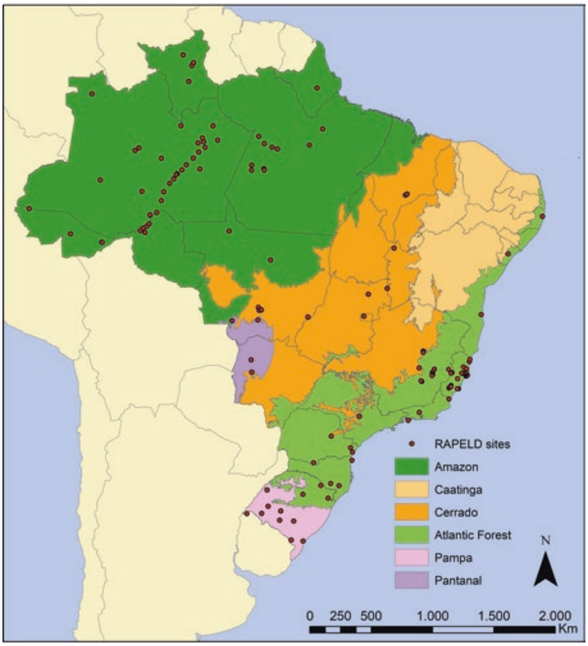

**Fig. 22.2** Location of RAPELD sites throughout the different biomes in Brazil. (IBGE 2019)

## 22.4 Integrating Research: The Brazilian Program for Biodiversity Research (PPBio)

Brazil hosted the United Nations Conference on Environment and Development in 1992, in Rio de Janeiro, the so-called RIO-92. One of the most important documents generated during RIO-92 was the Convention on Biological Diversity (CBD). Although Brazil was a signatory, Brazil implemented the National Biodiversity Policy in line with the CBD only 10 years later, with the general objective of "promoting in an integrated way, conservation of biodiversity and sustainable use of its components, with the fair and equitable sharing of the benefits derived from the use of genetic resources, components of genetic heritage and traditional knowledge associated with such resources" (Decree No. 4.339, 22 August 2002, Diário Oficial da União). However, biodiversity research was generally conducted as isolated surveys which were not converted into chains of knowledge production, nor was the knowledge generated shared with the different sectors of society or other researchers.

In 2004, to align biodiversity research with the National Biodiversity Policy, after several meetings with researchers and biodiversity stakeholders, the then Ministry of Science and Technology created the Brazilian Program for Biodiversity Research, the PPBio (Ordinance 268 of June 18, 2004, MCT, Diário Oficial da União) (Pezzini et al. 2012). The objectives of PPBio were to support, maintain, and expand the inventory networks and biological collections of Brazil; support research in thematic areas; disseminate the results for different purposes, such as environmental management and education; and develop strategic actions to implement biodiversity-research policies, addressing the issues raised by the CDB (Overbeck et al. 2018). PPBio was based on a strategy of creation of regional hubs that attended less-favored regions in terms of training of human resources and that had great biodiversity potential (Baccaro et al. 2008). Thus, the first research networks were created in the Amazon and in the Brazilian semiarid region.

In a relatively short time, the PPBio obtained consistent results, in large part due to use of RAPELD as an integrative methodology, which led to further integration of information on different biotic and abiotic factors (Baccaro et al. 2008), such as vertebrates and invertebrates (Pereira et al. 2019; Graça et al. 2015), topography (Norris et al. 2014), distance to the water table (Schietti et al. 2014), soil bacteria and fungi (Braga-Neto et al. 2008), and soil chemical elements (Moura et al. 2015).

PPBio was expanded in 2012 by seeking the formation of new research networks and expanding its research area to include the other Brazilian biomes. Today, it is the largest biodiversity network in Brazil encompassing more than 600 researchers in 90 institutions (Fernandes et al. 2017). Two networks were established in the Atlantic Forest biome, BioM.A. and PPBioMA. This biome suffers from degradation throughout its distribution, with only about 28% of remaining forest remnants and still conserving one of the largest samples of biodiversity on the planet (Rezende et al. 2018), which led the Atlantic Forest to be indicated as one of the 34 world biodiversity hotspots (Myers et al. 2000). Integration within the networks took place in different ways. The first network (BioM.A.) had been operating for about 20 years in the Serra dos Órgãos region (within the PELD Mosaico da Mata Atlântica Central Fluminense), with monitoring in permanent plots (e.g., Vieira et al. 2018; Brigatti et al. 2016), and was structured in themes (inventories, collections, research) covering three other sites (Guaribas Biological Reserve, PB; Pratigi Environmental Protect Area, BA; and Serra do Tabuleiro State Park, SC) (Fig. 22.1). The second network was established as an expansion of PPBio to the Atlantic Forest with distinct projects in the regional hubs (Northeast, Southeast, and South regions) and with articulation through an executive hub, with the use of the RAPELD methodology in each sampling site of this network, in order to facilitate the integration of data in different regions.

The first RAPELD modules in the Atlantic Forest were installed in the Una Biological Reserve in the State of Bahia and in the Ilha Grande State Park in the State of Rio de Janeiro, prior to the official start of the program in the Atlantic Forest biome. Subsequently, with the formation of the PPBioMA network, there was an expansion of RAPELD modules to other areas of the biome. Today, there are more than 21 sites with RAPELD modules covering coastal sand dunes, semideciduous seasonal forest, dense rainforest, mixed rainforest, and high-altitude fields (Table 22.2 and Fig. 22.1). Twenty-two RAPELD modules are also being used to monitor the environmental impact along the Doce River in the states of Minas Gerais and Espírito Santo, caused by the world's largest mine-waste-dam rupture environmental disaster (IBAMA 2017).

The accumulated knowledge from those years has already generated several papers within the network (e.g., Ferreguetti et al. 2018; Oliveira et al. 2019; Tromboni et al. 2018; Figueiredo et al. 2017), among the networks (e.g., Fernandes et al. 2017; Overbeck et al. 2018), data papers with other partners (e.g., Ramos et al. 2019), and dissemination materials, such as field guides and catalogs (e.g., Santos et al. 2017). While the scientific papers mainly reach the scientific community, the field guides are fundamental to unveil Brazilian biodiversity, unknown to most Brazilian citizens and local stakeholders.

Nevertheless, integration of a research network goes beyond sample sites with standardized methodology; internalization is necessary to ensure the expansion of knowledge of biodiversity and its use in multidisciplinary knowledge-production chains. This has taken place via the training of researchers, students, technicians, environmental analysts, and the public in general in several regions throughout the Atlantic Forest (Marques et al. 2016). PPBioMA has trained more than 530 people in technical and scientific courses, such as survey and monitoring of biodiversity, taxonomy of Atlantic Forest tree species, scientific photography, environmental legislation, scientific writing and data analysis, functional diversity, ecological networks, and data management.

Freely available data is necessary to ensure integration among researchers and demands, as well as to help decision-makers and managers in relation to public policies, effectively making the research results tools for conservation. The information

**Table 22.2** Location of RAPELD modules in the Atlantic Forest. The main institution corresponds to the one that coordinated the project in one of the three hubs (NE, northeast; SE, southeast; and S, south) of the PPBioMA network

| Acronym | Full name | Main institution | Hub/state | Ecosystem |
|---|---|---|---|---|
| PEDI | Parque Estadual Dois Irmãos | Universidade Federal Rural de Pernambuco | NE/ Pernambuco | Atlantic Forest |
| PNSI | Parque Nacional Serra de Itabaiana | Universidade Federal de Sergipe | NE/Sergipe | Atlantic Forest and Caatinga |
| RBU | Reserva Biológica Una | Universidade Estadual de Santa Cruz | NE/Bahia | Atlantic Forest |
| RBS | Reserva Biológica Sooretama | Universidade Federal do espírito Santo | SE/Espírito Santo | Coastal plain forest |
| RNV | Reserva Natural Vale | Universidade do Estado do Rio de Janeiro | SE/Espírito Santo | Coastal plain forest |
| PEMF | Parque Estadual Mata das Flores | Instituto Estadual do Meio Ambiente | SE/Espírito Santo | Atlantic Forest |
| PEPCV | Parque Estadual Paulo César Vinha | Instituto Estadual do Meio Ambiente | SE/Espírito Santo | Restinga, coastal lagoons |
| APAS | Área de Proteção Ambiental de Setiba | Instituto Estadual do Meio Ambiente | SE/Espírito Santo | Restinga, coastal plain |
| PEPA | Parque Estadual Pedra Azul | Instituto Estadual do Meio Ambiente | SE/Espírito Santo | Atlantic Forest |
| PEFG | Parque Estadual Forno Grande | Instituto Estadual do Meio Ambiente | SE/Espírito Santo | Atlantic Forest |
| RPPNAB | Reserva Particular do Patrimônio Natural Águia Branca | Instituto Estadual do Meio Ambiente | SE/Espírito Santo | Atlantic Forest |
| APACB | Área de Proteção Ambiental Conceição da Barra | Instituto Estadual do Meio Ambiente | SE/Espírito Santo | Restinga, coastal plain |
| RBAR | Reserva Biológica Augusto Ruschi | Instituto Nacional da Mata Atlântica | SE/Espírito Santo | Atlantic Forest |
| PEI | Parque Estadual Itaúnas | Instituto Estadual do Meio Ambiente | SE/Espírito Santo | Restinga, coastal plain |
| EEEG | Estação Ecológica Estadual Guaxindiba | Universidade Estadual do Norte Fluminense | SE/Rio de Janeiro | Coastal plain forest |
| RBT | Reserva Biológica do Tinguá | Jardim Botânico do Rio de Janeiro | SE/Rio de Janeiro | Atlantic Forest |
| PEIG | Parque Estadual da Ilha Grande | Universidade do Estado do Rio de Janeiro | SE/Rio de Janeiro | Atlantic Forest, restinga |
| RNRC | Reserva Natural do Rio Cachoeira | Universidade Federal do Paraná | S/Paraná | Atlantic Forest |
| RG | Rio Guaraguaçu | Universidade Federal do Paraná | S/Paraná | Coastal plain |

(continued)

**Table 22.2** (continued)

| Acronym | Full name | Main institution | Hub/state | Ecosystem |
|---|---|---|---|---|
| PEA | Parque Estadual Acaraí | Universidade Regional de Joinville | S/Santa Catarina | Restinga |
| PNSJ | Parque Nacional São Joaquim | Universidade Federal de Santa Catarina | S/Santa Catarina | Araucaria forest, cloud forest, altitude fields |

available on biotic and abiotic aspects, as well as the logistics of the research sites, make the research sites attractive, increasing the integration among researchers (Magnusson et al. 2013). Furthermore, the relevance of data and metadata availability in network projects is the integration of research, specially in times of budget cuts. PPBio has a data policy (MCT 2009) to ensure open access to robust and persistent data, and these have been made available on the DataOne platform (https://www.dataone.org/) and the Information System on Brazilian Biodiversity – SiBBr (http://www.sibbr.gov.br/).

## 22.5 Integrating Research and Biodiversity Conservation

Prioritization of new areas for protection is often undertaken based on a combination of field biological data, current and/or predicted patterns of forest loss, socioeconomic factors, infrastructural and institutional capacity issues, and politics, because not all sites are or can be conserved (e.g., private areas, areas used for expansion of agriculture or pasture) (MMA 2007). That means that scientists need to provide the best evidence they can to maintain at least some areas being developed under criteria that establish conservation priorities.

Within this perspective, long-term studies in Brazil, especially those related to PELD, RAPELD, and PPBio, offer an integrated and multiscale approach based on a combination of field data, current and/or predicted patterns of forest loss, and socioeconomic factors, as well as involving actors at local to national levels, infrastructural and institutional capacity issues, and politics. Almost all Brazilian LTER sites and PPBio have projects dealing with environmental education within different ecosystems (Barbosa et al. 2004). However, PELD and RAPELD sites are still mostly concentrated in the south and southeast of the Atlantic Forest and should be expanded to the Northeast (Fig. 22.1). Furthermore, there is no overlap between Atlantic Forest PELD sites and PPBio sites with the standardized RAPELD infrastructure, except for a small overlap in the Lagamar PELD where there is one site with a RAPELD module (Fig. 22.1). The standardized methodology is a need that the PELD has not yet been able to incorporate in its sites, and RAPELD has shown its suitability for integrating research in and among sites, as well as with other initiatives (e.g., RAINFOR, ATDN). Standardized methods must also be expanded to studies developed in marine environments, and RAPELD could be the model for this.

Political commitments and policy instruments to halt biodiversity loss require robust data and a diverse indicator set to monitor and report on biodiversity trends and gaps in data availability, and narrowly based indicator sets are significant information barriers to fulfilling these needs (Geijzendorffer et al. 2016). Brazil will likely fail to reach the National Targets for Biodiversity 2011–2020, and it will be difficult to fulfill the restoration target of the Brazilian Nationally Determined Contribution (NDC) and to advance with the sustainable development goals, especially due underfunding (Fernandes et al. 2017). Therefore, scarce resources have to be applied strategically.

Financial constraints represent an important challenge that needs to be addressed. The investment needed for long-term studies must be constant in terms of flux, and a relatively low amount has been destined for all types of research each year (less than 1% from the Brazilian GDP). The low-investment policies will be harmful not only to science and technology (Angelo 2016) but also to sustainable development and nature conservation (Overbeck et al. 2018). PELD and PPBio represent strategic actions that need to be financed if Brazil is to meet its international obligations and provide the data necessary to provide economic and environmental security for its citizens.

Applying scientific knowledge to biodiversity-conservation practice and decision-making is a challenge around the globe (Pullin et al. 2004; Kueffer et al. 2012), and the situation is no different regarding research in the Atlantic Forest. Often, the scale at which research is done is different from the scale of interest to decision-makers (Bacellar et al. 2020). That is why the use of a standardized method that considers multiple scales by PELD and PPBio may contribute to filling the research-implementation gap.

PELD and PPBio are complementary network programs of great relevance for the knowledge and conservation of Brazilian biodiversity, and their integration needs to be strengthened in the Atlantic Forest.

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## References

Aguiar-Silva FH, Junqueira TG, Sanaiotti TM, Guimarães VY, Mathias PVC, Mendonça CV (2015) Resource availability and diet in harpy eagle breeding territories on the Xingu River, Brazilian Amazon. Brz J Biol 75:181–189

Angelo C (2016) Brazilian scientists reeling as federal funds slashed by nearly half. Nature. https://doi.org/10.1038/nature.2017.21766

Baccaro FB, Drucker DP, Vale J, Oliveira ML, Magalhães C, Lepsch-Cunha N, Magnusson WE (2008) A Reserva Ducke. In: Oliveira ML, Baccaro FB, Braga-Neto R, Magnusson WE (eds) Reserva Ducke, a biodiversidade amazônica através de uma grade. Manaus. Instituto Nacional de Pesquisas da Amazônia, Áttema Design Editorial, pp 11–20. 166 p

Bacellar AEF, Salzo I, Ribeiro KT, Cronemberger C, Brito MA, Oliveira D, Mamede MA, Lacerda FS (2020) Parceria entre CNPq e Instituto Chico Mendes no fortalecimento dos sítios PELD em Unidades de Conservação federais. Oecologia Australis 24(2):266–270

Barbosa FAR, Scarano FR, Sabará MG, Esteves FA (2004) Brazilian LTER: ecosystem and biodiversity information in support of decision-making. Environ Monit Assess 90:121–133

Barbosa FAR (2013) Uma breve história do Programa de Pesquisas ecológicas de longa duração (PELD-CNPq) do Brasil. Da semente ao fruto. In: Tabarelli M, Rocha CFD, Romanowski HP, Rocha O, Lacerda LD (eds) PELD-CNPq Dez anos do Programa de Pesquisas Ecológicas de Longa Duração no Brasil: achados, lições e perspectivas. Editora Universitária UFPE, Recife, pp 13–27

Barbosa FAR, da Fonseca GAB, Cavalcanti RB, da Fonseca MT (1998) The Brazilian long-term ecological research program: present status and perspectives. In: US LTER Network Office. University of New Mexico, pp 7–13

Barbosa FAR, da Fonseca GAB, Cavalcanti RB, da Fonseca MT (2000) The Brazilian long-term ecological research program: present status and perspectives for the new decade. In: US LTER Network Office. University of New Mexico, pp 72–75

Bobrowiec PE, Tavares VC (2017) Establishing baseline biodiversity data prior to hydroelectric dam construction to monitoring impacts to bats in the Brazilian Amazon. PLoS One 12:e0183036

Braga-Neto R, Luizão RCC, Magnusson WE, Zuquim G, Castilho CV (2008) Leaf litter fungi in a central Amazonian forest: the influence of rainfall, soil and topography on the distribution of fruiting bodies. Biodivers Conserv 17(11):2701–2712

Brigatti E, Vieira MV, Kajin M, Almeida PJAL, De Menezes MA, Cerqueira R (2016) Detecting and modelling delayed density-dependence in abundance time series of a small mammal (*Didelphis aurita*). Sci Rep 6:19553

Carneiro LRA, Lima AP, Machado RB, Magnusson WE (2016) Limitations to the use of species-distribution models for environmental-impact assessments in the Amazon. PLoS One 11:e0146543

Costa FRC, Magnusson WE (2010) The need for large-scale integrated studies of biodiversity – the experience of the program for biodiversity research in the Brazilian Amazon. Natureza Conservação 8:1–5

Craine JM, Battersby J, Elmore AJ, Jones AW (2007) Building EDENs: the rise of environmentally distributed ecological networks. Bioscience 57:45–54

Fernandes GW, Vale MM, Overbeck GE, Bustamante MMC, Grelle CEV, Bergallo HG, Magnusson WE, Akama A, Alves SS, Amorim A, Araújo J, Barros CF, Bravo F, Carim MJV, Cerqueira R, Collevatti RG, Colli GR, Da Cunha CN, D'Andrea PS, Dianese JC, Diniz S, Estrela PC, Fernandes MRM, Fontana CS, Giacomin LL et al (2017) Dismantling Brazil's science threatens global biodiversity heritage. Perspect Ecol Conserv 15:1–5

Ferreguetti AC, Pereira-Ribeiro J, Prevedello JA, Tomás WM, Rocha CFD, Bergallo HG (2018) One step ahead to predict potential poaching hotspots: Modeling occupancy and detectability of poachers in a neotropical rainforest. Biol Conserv 227:133–140

Figueiredo MS, Barros CS, Delciellos AC, Guerra EB, Cordeiro-Estrela P, Kajin M, Alvarez MR, Asfora PH, Astúa D, Bergallo HG, Cerqueira R, Geise L, Gentile R, Grelle CEV, Iack-Ximenes GE, Oliveira LC, Weksler M, Vieira MV (2017) Abundance of small mammals in the Atlantic Forest (ASMAF): a data set for analyzing tropical community patterns. Ecology 98(11):2981

Forsberg BR (1993) Brazil-LTER. In: US LTER Network Office, University of Washington, College of Forest Resources, pp. 42–44

Fraga R, Stow AJ, Magnusson WE, Lima AP (2014) The cost of evaluating species densities and composition of snakes to assess development impacts in Amazonia. PLoS One 9:e105453

Fraga R, Lima AP, Magnusson WE, Ferrão M, Stow AJ (2017) Contrasting patterns of gene flow for Amazonian snakes that actively forage and those that wait in ambush. J Hered 108:534–534

Fraga R, Ferrão M, Stow AJ, Magnusson WE, Lima AP (2018) Different environmental gradients affect different measures of snake ß-diversity in the Amazon rainforests. PeerJ 6:e5628

Geijzendorffer IR, Regan EC, Pereira HM, Brotons L, Brummitt N, Gavish Y, Haase P, Martin CS, Mihoub JB, Secades C, Schmeller DS, Stoll S, Wetzel FT, Walters M (2016) Bridging the gap between biodiversity data and policy reporting needs: an essential biodiversity variables perspective. J Appl Ecol 53(5):1341–1350

Goring SJ, Weathers KC, Dodds WK, Soranno PA, Sweet LC, Cheruvelil KS, Kominoski JS, Rüegg J, Thorn AM, Utz RM (2014) Improving the culture of interdisciplinary collaboration in ecology by expanding measures of success. Front Ecol Environ 12:39–47. https://doi.org/10.1890/120370

Gosz JR (1998) Long-term research on a global scale. In: US ILTER Network. University of New Mexico, pp 3–6

Graça MB, Morais JW, Franklin E, Pequeno PACL, Souza JLP, Bueno AS (2015) Combining taxonomic and functional approaches to unravel the spatial distribution of an Amazonian butterfly community. Environ Entomol 183:1–9

Hero JM, Castley JG, Malone M, Lawson B, Magnusson WE (2010) Long-term ecological research in Australia: innovative approaches and future benefits. Aus Zool 35:90–102

Hero JM, Castley JG, Butler SA, Lollback GW (2013) Biomass estimation within an Australian eucalypt forest: mesoscale spatial arrangement and the influence of sampling intensity. For Ecol Manag 310:547–554

Hobbie JE, Carpenter SR, Grimm NB, James R, Seastedt TR (2003) The US long term ecological research program. Bioscience 53:21–32

IBAMA (2017) Parecer Técnico n° 1/2017-COREC/CGBIO/DBFLO. Análise do Plano de Trabalho apresentado pela Fundação RENOVA em atendimento a Notificação IBAMA/ES n.° 678322-E. Processo IBAMA n.° 02009.001474/2015-17

IBGE (2019) Biomas e sistema costeiro-marinho do Brasil: compatível com a escala 1:250 000. IBGE, Rio de Janeiro. Available a4t https://biblioteca.ibge.gov.br/visualizacao/livros/liv101676.pdf

IBGE (2012) Mapa da Área de Aplicação da Lei n° 11.428 de 2006. Escala 1: 5.000.000. Projeção Policônica. Datum Horizontal SIRGAS2000. Available at http://geoftp.ibge.gov.br/informacoes_ambientais/estudos_ambientais/biomas/mapas/lei11428_mata_atlantica.pdf

Jucevica E, Melecis V (2005) Long-term effects of climate warming on Forest soil Collembola. Acta Zoologica Lituanica 15:124–126

Kinupp VF, Magnusson WE (2005) Spatial patterns in the understory shrub genus *Psychotria* in central Amazonia: effects of distance and topography. J Trop Ecol 21:1–12

Koblitz RV, Lima AP, Menin M, Rojas D, Condrati LH, Magnusson WE (2017) Effect of species-counting protocols and the spatial distribution of effort on rarefaction curves in relation to decision making in environmental-impact assessments. Austral Ecol 42:723–731

Koehler HH (2000) Natural regeneration and succession — results from a 13 years study with reference to mesofauna and vegetation, and implications for management. Landsc Urban Plan 51(2–4):123–130

Kueffer C, Underwood E, Hadorn GH, Holdegger R, Lehning M, Pohl C, Schirmer M, Schwarzenbach R, Stauffacher M, Wuelser G, Edwards P (2012) Enabling effective problem-oriented research for sustainable development. Ecol Soc 17(4):8. https://doi.org/10.5751/ES-05045-170408

Lollback GW, Castley GJ, Mossaz AC, Hero JM (2017) Fine-scale changes in spatial habitat use by a low-density koala population in an isolated periurban forest remnant. Austral Mammal 40:84–92

Magnusson WE, Lima AP, Luizão R, Costa FRC, Castilho CV, Kinupp VF (2005) RAPELD: a modification of the Gentry method for biodiversity surveys in long-term ecological research sites. Biota Neotropica 5:19–24

Magnusson WE, Costa F, Lima A, Baccaro F, Braga-Neto R, Romero RL (2008) A Program for monitoring biological diversity in the Amazon: an alternative perspective to threat-based monitoring. Biotropica 40(4):409–411

Magnusson WE, Braga-Neto R, Pezzini F, Baccaro F, Bergallo H, Penha J, Rodrigues D, Verdade LM, Lima A, Al A, Hero JM, Lawson B, Castilho C, Galdino G, Castley G, Zuanon J, Vale J, Santos JLC, Luizão R, Cintra R, Barbosa RI, Lisboa A, Koblitz R, Cunha CN, Pontes ARM (2013) Biodiversity and integrated environmental monitoring. Attema, Manaus

Mamede AM, Pinheiro FC, Tabarelli M (2013) Dez anos de gestão do PELD: Resultados, lições e perspectivas. In: Tabarelli M, Rocha CFD, Romanowski HP, Rocha O, Lacerda LD (eds) PELD-CNPq Dez anos do Programa de Pesquisas Ecológicas de Longa Duração no Brasil: achados, lições e perspectivas. Editora Universitária UFPE, Recife, pp 415–436

Marques MCM, Silva ACLE, Rajão H, Rosado BHP, Barros CF, Oliveira JA, Finotti R, Neckel-Oliveira S, Amorim A, Cerqueira R, Bergallo HG (2016) Mata Atlântica - O desafio de transformar um passado de devastação em um futuro de conhecimento e conservação. In: Peixoto AL, Luz JRP, Brito MC (eds) Conhecendo a Biodiversidade, 1ed. MCTIC, CNPq, PPBio, Brasília, pp 50–67

MCT (2009) Ministério da Ciência e Tecnologia, 2009 Diário Oficial da União, Portaria N° 693, 20th August 2009

MMA (2007) Áreas Prioritárias para Conservação, Uso Sustentável e Repartição de Benefícios da Biodiversidade Brasileira: Atualização. Portaria MMA n°9, 23th January 2007. Ministério do Meio Ambiente, Secretaria de Biodiversidade e Florestas, Brasília

Morães LJCL, Pavan D, Barros MC, Ribas CC (2016) The combined influence of riverine barriers and flooding gradients on biogeographical patterns for amphibians and squamates in south-eastern Amazonia. J Biogeogr 43:2113–2124

Moser P, Oliveira WL, Medeiros MB, Pinto JR, Eisenlohr PV, Lima IL, Silva GP, Simon MF (2014) Tree species distribution along environmental gradients in an area affected by a hydroelectric dam in southern Amazonia. Biotropica 46:367–376

Motta E, Albuquerque E, Simões R, Baessa A, Campolina B, Silva L (2002) A distribuição espacial da produção científica e tecnológica brasileira: uma descrição de estatísticas de produção local de patentes e artigos científicos. Revista Brasileira Inovação 1:225–251. https://doi.org/10.20396/rbi.v1i2.8648860

Moura QLD, Ruivo MDLP, Rodrigues HJB, Rocha EJP, Silva Junior JDA, Vasconcelos SS, Andrade MC, Manes CLDO (2015) Variação sazonal da população de bactérias e fungos e dos teores de nitrato e amônio do solo nos sítios do LBA e PPBIO, na Amazônia Oriental. Revista Brasileira de Meteorologia

Myers N, Mittermeier RA, Mittermeier CG, Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403(6772):853–858. https://doi.org/10.1038/35002501

Norris D, Fortin MJ, Magnusson WE (2014) Towards monitoring biodiversity in Amazonian forests: how regular samples capture meso-scale altitudinal variation in 25 km$^2$ plots. PLoS One 9:e106150

Oliveira RA, Marques R, Marques MC (2019) Plant diversity and local environmental conditions indirectly affect litter decomposition in a tropical forest. Appl Soil Ecol 134:45–53

Ortiz DA, Lima AP, Werneck FP (2018) Environmental transition zone and rivers shape intraspecific population structure and genetic diversity of an Amazonian rain forest tree frog. Evol Ecol 32:359–378

Overbeck GE, Bergallo HG, Grelle CEV, Akama A, Bravo F, Colli GR, Magnusson WE, Tomas WM, Fernandes GW (2018) Global biodiversity threatened by science budget cuts in Brazil. Bioscience 68:11–12

Peixoto AL, Luz LJP, Brito MA (2016) Conhecendo a Biodiversidade. MCTIC, CNPq, PPBio, Brasília, DF

Penman D, Pearce A, Morton M (2011) From competition to collaboration: challenges for New Zealand science. Policy Q 7:44–49. https://doi.org/10.26686/pq.v7i4.4395

Pereira LGA, Capavede UD Jr, Tavares VC, Magnusson WE, Bobrowiec PED, Baccaro FB (2019) From a bat's perspective, protected riparian areas should be wider than defined by Brazilian laws. J Environ Manag 232:37–44

Pezzini FF, Melo PHA, Oliveira DMS, Amorim RX, Figueiredo FOG, Drucker DP, Rodrigues FRO, Zuquim G, Emilio T, Costa FRC, Magnusson WE, Sampaio AF, Lima AP, Garcia ARM, Manzatto AG et al (2012) The Brazilian program for biodiversity research (PPBio) information system. Biodivers Ecolo 4:265–274. https://doi.org/10.7809/b-e.00083

Pitman NCA, Widmer J, Jenkins CN, Stocks G, Seales L, Paniagua F, Bruna EM (2011) Volume and geographical distribution of ecological research in the Andes and the Amazon 1995-2008. Tropi Conserv Sci 4:64–81

Pullin AS, Knight TM, Stone DA, Charman K (2004) Do conservation managers use scientific evidence to support their decision-making? Biol Conserv 119:245–252. https://doi.org/10.1016/j.biocon.2003.11.007

Ramos FN, Mortara SR, Monalisa-Francisco N, Elias JPC, Neto LM, Freitas L et al (2019) Atlantic Epiphytes: a data set of vascular and non-vascular epiphyte plants and lichens from the Atlantic Forest. Ecology 100(2):e02541

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Strassburg BBN, Tabarelli M, Fonseca GA, Mittermeier RA (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:208–214. https://doi.org/10.1016/j.pecon.2018.10.002

Sanaiotti TM, Junqueira TG, Palhares V, Aguiar-Silva FH, Oliveira LMP, Guimarães VY, Castro V, Mota D, Trambin DF, Villar DNA, Lara KM, Fernandes D, Larangueiras L, Mathias PC, Mendonça CV (2015) Abundance of Harpy Eagles from a reservoir-impact area in the Low- and Mid-Xingu River. Braz J Biol 75:S190–S204

Santorelli S, Magnusson WE, Deus CP (2018) Most species are not limited by an Amazonian river postulated to be a border between endemism areas. Sci Rep 8:2294

Santos EM, Barbosa VN, Correia JS (2017) Guia de Répteis do Parque Estadual de Dois Irmãos, 1st edn. Editora UFRPE, Recife

Schietti J, Emilio T, Rennó CD, Drucker DP, Costa FR, Nogueira A, Baccaro FB, Figueiredo F, Castilho CV, Kinupp V, Guillaumet JL, Garcia ARM, Lima AP, Magnusson WE (2014) Vertical distance from drainage drives floristic composition changes in an Amazonian rainforest. Plant Ecol Divers 7(1–2):241–253. https://doi.org/10.1080/17550874.2013.783642

Sidone OJG, Haddad EA, Mena-Chalco JP (2016) A ciência nas regiões brasileiras: evolução da produção e das redes de colaboração científica. Transinformação 28:15–31. https://doi.org/10.1590/2318-08892016002800002

Strayer D, Glitzenstein JF, Jones CJ, Kolasa J, Likens GE, McDonnell MJ, Parker G, Picket STA (1986) Long-term ecological studies: an illustrated account of their design, operation and importance to ecology. Occasional Publication of the Institute of Ecosystem Studies, New York Botanical Garden, 2

Tabarelli M, Rocha CFD, Barbosa FAR, Hay JD, Romanowiski HP, Rocha O, Lacerda LD (2013) PELD-CNPq Dez anos do Programa de Pesquisas Ecológicas de Longa Duração no Brasil: achados, lições e perspectivas. Editora Universitária UFPE, Recife

Tromboni F, Thomas SA, Gücker B, Neres-Lima V, Lourenço-Amorim C, Moulton TP, Silva-Junior EF, Feijó-Lima R, Boëchat IG, Zandonà E (2018) Nutrient limitation and the stoichiometry of nutrient uptake in a tropical rain Forest stream. J Geophys Res Biogeo 123:2154–2167

Vieira MV, Almeida-Gomes M, Delciellos AC, Cerqueira R, Crouzeilles R (2018) Fair tests of the habitat amount hypothesis require appropriate metrics of patch isolation: an example with small mammals in the Brazilian Atlantic Forest. Biol Conserv 226:264–270

# Chapter 23
# The Future of the Brazilian Atlantic Forest

Carlos E. V. Grelle, Henrique Rajão, and Marcia C. M. Marques

**Abstract** The Atlantic Forest, one of the most diverse biomes in the world, is possibly one of the best places to understand how the evolutionary process develops in tropical systems. This characteristic made it recognized as one of the most important biomes for the world, which was ratified through the construction of a legal framework that guarantees its preservation in the near future. However, the long history of human occupation throughout the centuries has resulted in vulnerabilities, due to threats with regional effects (such as continuous changes in land use, hunting) or global (climate change, biological invasion). Even in this unfavorable current scenario, opportunities arise and are able to reconcile the conservation of biodiversity and ecosystem services with the social and economic development in Brazil. In order to envision a future for this important biome, we put, side by side, the internal strengths and weaknesses, together with the threats and opportunities, and discussed possible scenarios for this important world biome.

**Keywords** SWOT analysis · Atlantic Forest · Brazil · Conservation · Biodiversity

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## 23.1 Introduction

At the beginning of the twenty-first century, humanity finds itself in a period that was conventionally called postmodernity, characterized by the advance of capitalism, with profound social and cultural marks in societies. Globalization, initially thought of as an economic phenomenon, brought the production system of countries closer together while distributing poverty, human tragedies, and environmental problems. This new world order imposed in the Anthropocene dramatically accelerated the use of natural resources, seriously compromising the future of the planet.

In this current scenario, what can we expect from the future of humanity? Some contemporary philosophers envision a future that is not optimistic, with societies disrupted by the lack of work, limitations on freedoms, and the rise of nationalism. Others believe that the extremes imposed on postmodern societies may reflect improvements in working relationships, growth in gender equality, and greater global engagement, among others (de Masi 2019; Harari 2018). In all future scenarios, nature, science, and the humanities have an important role, either to provide resources necessary for human survival in all its cultural diversity, or to develop new technologies that allow sustainable production chains, or even to think on ethical principles compatible with the existence of all forms of life.

In this context, what can we expect from the future of the Brazilian Atlantic Forest? The chapters in this book show that although the recent past of destruction has left an indelible signature on the biome (Solórzano et al. 2021; Lins-e-Silva et al. 2021; Faria et al. 2021; Carlucci et al. 2021), the identity of the Atlantic Forest as one of the main biodiversity centers in the world is still preserved, following the updated lists of species richness in this book (Zwiener et al. 2021; Feitosa et al. 2021; Ramos et al. 2021; Figueiredo et al. 2021). Considering the multiple social, economic, and environmental effects of postmodern society, is it possible to envision an optimistic future for the Brazilian Atlantic Forest? The answer to this question is not trivial. As conservationists and scientists, we can hope that biodiversity and ecosystem services in the Atlantic Forest will be conserved in the future. However, this desire is useless if we are not prepared to face real and imagined challenges. At the same time, looking at the present should be a realistic exercise in identifying weaknesses and strengths, in addition to a constant duty of looking for opportunities, so that the future would be less uncertain than it naturally is for Brazilian Atlantic Forest.

In this chapter, we discuss the possible future of the Atlantic Forest, by identifying its weaknesses and strengths and external opportunities and risks. Based on the rereading of the synthesis chapters presented in this book, complemented with subjects not yet addressed, we seek to provide a realistic panorama of the future of the Atlantic Forest that is useful for the management of this important world biome.

## 23.2 Identifying Factors Affecting the Knowledge and Conservation of the Atlantic Forest

In order to achieve a favorable future, we can rely on a strategy capable of mapping the current situation and predicting future complications, in order to avoid undesirable paths. For this, we identify the strengths (characteristic of the biome that gives it an advantage over others), weaknesses (characteristic of the biome that gives it disadvantages), opportunities (elements that can better exploit the advantages and strengths of the biome), and threats (elements which can cause disadvantages or increase weaknesses) in the Atlantic Forest, considering the current knowledge about the biome, explained in the chapters of this book. Considering that this book is a document produced mainly by ecologists, conservationists, taxonomists, and other specialists in the natural sciences, we aimed to complement the information with empirical and theoretical knowledge in other fields (socio-environmentalism, economics, and politics, among others). The result of this search is presented in Fig. 23.1 and discussed in the following sections and can be used to strategically plan the futures of Atlantic Forest.

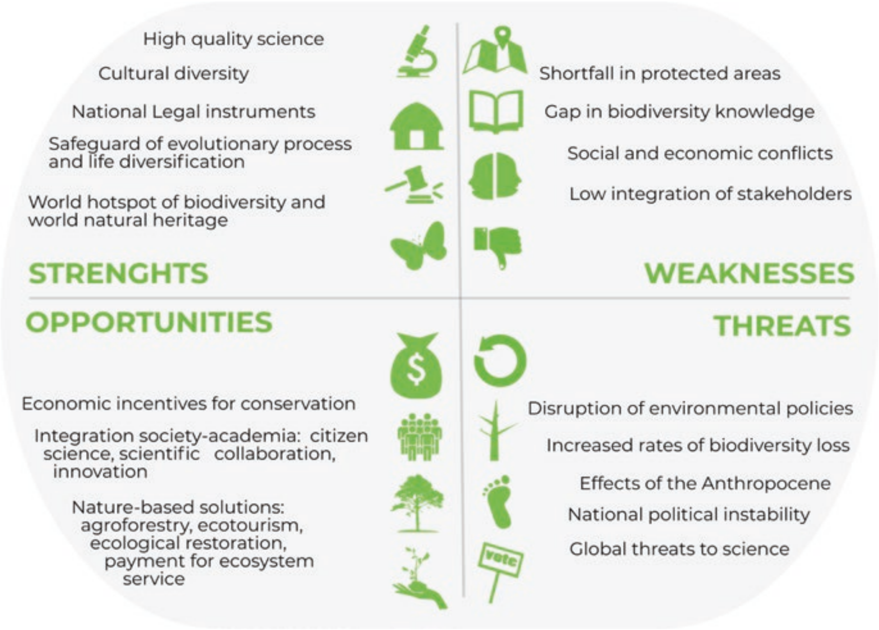

**Fig. 23.1** Synthesis of internal strengths and weaknesses and external opportunities and threats to the Atlantic Forest. The SWOT analysis, the technique for identifying strengths, weaknesses, opportunities, and threats, previously used for strategic business planning, is here used for planning the conservation strategies. (Gao and Peng 2011)

## 23.3 Strengths

The Atlantic Forest is certainly one of the most diverse biomes in the world. Considering only selected taxa, the biome is home of 2960 species of trees (Zwiener et al. 2021), 2227 species of epiphytes (Ramos et al. 2021), 1545 species of microalgae (Padial et al. 2021), 1401 species of social insects (Feitosa et al. 2021), 2645 species of Tetrapoda, being 719 species of amphibians, 517 of reptiles, 1025 species of birds, and 384 of mammals (Figueiredo et al. 2021). In addition, there is a high level of endemism in the Brazilian Atlantic Forest. For example, around 28% of mammals, 21% of birds, 24% of reptiles, and 70% of amphibians occurring in the biome are endemic (Figueiredo et al. 2021).

The high diversity is determined by the great variation of habitats in all its geographic extension, which includes woody and herbaceous vegetation formations, distributed in a variety of reliefs and landscapes (Marques et al. 2021). Throughout this variety of habitats, endemism (resulted from evolutionary processes) seems to be concentrated in five main centers: Pernambuco, Central Bahia, Coastal Bahia, Serra do Mar, and Paraná/Araucaria (Peres et al. 2020). Some of these areas are particularly important for diversifying a variety of taxa along the geological time. For example, the *Serra do Mar* region is possibly one of the main areas of species diversification, recorded in studies with birds, mammals, amphibians, butterflies, harvestmen, tiger moths, and Myrtaceae (Peres et al. 2020). Although most of the species of the Atlantic Forest biota arose from processes that occurred in the last four million years, much more recent events may also have favored species diversification. For example, during the Holocene (~10 k years BP), there was speciation of some groups (e.g., rodents; Freygang et al. 2004). Therefore, to some degree, the actual preserved refuges of the Atlantic Forest possibly still represent important centers for the diversification of life on Earth.

Due to the high diversity and endemism, the Atlantic Forest has been recognized as one of the most important points on the planet for the conservation of biodiversity (Myers et al. 2000). It is also one of UNESCO's biosphere reserves. The worldwide interest in the Atlantic Forest occurred almost 20 years after the Amazon was the focus of conservationists around the world. From the 1970s onward, the Atlantic Forest entered the agendas of large nature conservation organizations, motivated mainly by threats to the golden lion tamarin (*Leontopithecus rosalia*) and the muriqui (*Brachyteles* spp) (see the exciting story described by Russel Mittermeier and Gustavo Fonseca in the Forward of this book). The worldwide recognition of the biome's biological importance motivated the creation of a series of legal instruments, important for the conservation of the Atlantic Forest, especially the Law of Atlantic Forest, which not only established the limits of the biome but also created rules for the sustainable use in the region (Marques et al. 2021). Therefore, by being recognized as an important place in the world, the Atlantic Forest gains a political dimension, which reaches various sectors and interests of society.

Another aspect that puts the Atlantic Forest in a better position compared to other biomes is the large number of research centers that are located throughout its extension. Some of the most important universities and research centers in Brazil are distributed in the Atlantic Forest region, where science and technology is made with international quality standards (Clarivate Analytics 2018). Also, some of the main Brazilian research and monitoring programs focused on the knowledge of biodiversity, such as PPBio and PELD (Bergallo et al. 2021), in addition to previous programs financed by international agencies (e.g., the Atlantic Forest Program, sponsored by German and Brazilian governments), among others, have produced, in recent years, a large volume of knowledge that significantly helped to fill gaps on biodiversity distribution and drivers of biodiversity loss. Therefore, the future management of the biome, whether in the implementation of policies for the conservation of biodiversity or in the development of technologies that combine economic and social growth, is favored by this body of knowledge and scientists whose subject is the Atlantic Forest.

In addition to the amazing biological diversity, the Atlantic Forest is also marked by great cultural diversity. The population that currently occupies the territory of the biome is of European, African, and indigenous origin (Pinheiro et al. 2014). The indigenous peoples of the Atlantic Forest today represent 29 ethnic groups that are present in 196 indigenous lands. The "quilombos," the rural black communities descended from slaves, are also numerous throughout the Atlantic Forest, and they maintain cultural manifestations strongly linked to the past of African slaves coming to the country. In addition to these, other populations such as the "caiçaras" (native population of the coast) and the communities of European immigrants (that arrived in Brazil in the nineteenth and the beginning of the twentieth centuries) (Pinheiro et al. 2014) added a diversity of traditions and cultural wealth that are confused with their own identity of Brazil.

On technical capacity, we need scientific knowledge and trained people to perform the actions. Along with the chapters of this book, a diverse team composed of 149 researchers (from 65 universities) and governmental and nongovernmental environmental analysts and directors worked together to write the 23 chapters. In addition, there are 145 postgraduate courses distributed along all states and training students in the area of biodiversity in Brazil (https://www.capes.gov.br/avaliacao/sobre-as-areas-de-avaliacao/73-dav/caa1/4653-biodiversidade). This task force gives advanced knowledge to postgraduate students, helping in the educational formation of future professionals that will act in universities, government agencies, and NGOs. It is interesting to note that all principal authors of the chapters of this book got their master/Ph.D. degrees in Brazilian universities and that half of them finished their Ph.D. courses in the last 10 years. Therefore, thanks to the investment in the training of human resources in the last two decades, the continuity of research on the Atlantic Forest is ensured, at least for one generation.

## 23.4 Internal Weaknesses

The long history of occupation of the Atlantic Forest and the deep human disturbances resulted in an extremely threatened biome (Solórzano et al. 2021; Faria et al. 2021; Lins-e-Silva et al. 2021; Carlucci et al. 2021). Especially in biomes with such characteristics described above, protected areas are the main instrument to prevent even more dramatic losses of biodiversity and ecosystem services. Protected areas are cornerstones of biodiversity conservation, and its implementation is recognized as an efficient strategy (Boucher et al. 2013). In Brazil, there was an increase in number and extension of protected areas in the last 40 years in all biomes, but with a bias to Amazon with the largest areas and Brazilian Atlantic Forest with a great number of protected areas (Vieira et al. 2019). However, it is important to note that in spite of the quantity of protected areas, only 10.1% of the Brazilian Atlantic Forest territory has some protection (Vieira et al. 2019), which is less than the 17% previously established by Aichi Targets to conserve terrestrial ecosystems (Convention for Biological Diversity 2010), and much less than is really necessary to guarantee the conservation of biodiversity (Convention for Biological Diversity 2020). This protection is distributed in many categories, with only 9% of vegetation cover is inside strictly protected areas (categories I-IV of IUCN) and 21% in sustainable-use protected areas (categories V–VI of IUCN) (Rezende et al. 2018). In essence, strictly protected areas recognized as more effective to biodiversity conservation than sustainable-use protected areas, and most of the pioneer studies on systematic planning in the world use only them (Margules and Pressey 2000). On the other hand, sustainable use has a great objective to involve local communities, adding a human dimension. Therefore, strict and sustainable are clearly important and with complementary roles. In fact, both types of protected areas should be used in systematic planning analyses, as well indigenous land, such as already performed in the Amazon (Pinto et al. 2014; Nolte et al. 2013) and Cerrado (Carranza et al. 2014; Brum et al. 2019) but still unexplored in the Brazilian Atlantic Forest.

With a complex relief, the Brazilian Atlantic Forest has lowlands and mountains, with distinct biotas (Marques et al. 2021). A large amount of protected areas are located in moderate to very steep slopes, meaning less protection of lowland forests (Vieira et al. 2019). This biased distribution of protected areas has historical and opportunistic reasons, since first national parks launched 80 years ago are in inland and mountainous regions (National Park of Itatiaia, National Park of Serra dos Orgãos, National Park of Iguaçu) due to an intention to protect the landscape, and the lowlands were always intensively explored since the sixteenth century.

An additional weakness is the size of Brazilian Atlantic Forest protected areas that is influenced by the small size of forest remnants in this biome. Some years ago, Ribeiro et al. (2009) estimated that more than 80% of forest remnants have less than 50 ha (Ribeiro et al. 2009). The minimum size of protected areas to the maintenance of viable populations is relevant from a temporal perspective. Usually, conservation planning is performed under a spatial view, and might not be effective in an eco-evolutionary perspective. In fact, there are few studies on the viability of populations inside protected areas in the Brazilian Atlantic Forest (Brito and Grelle 2004; Brito et al. 2008; Eduardo et al. 2012) and only one using the minimum size of protected areas in systematic conservation planning analyses (Pinto and Grelle 2009).

The success of conservation planning is minimizing costs and conflicts, and cost-effective decisions are based on information such as human density and land cost. In an ideal world, there would be no spatial congruence between, for example, species richness and these socioeconomic factors cited above. This can be especially problematic if the highly speciose biome is also the main industrial center, represents 70% of the gross domestic product of Brazil, and harbors the largest cities of Brazil (São Paulo and Rio de Janeiro) (Joly et al. 2014). Therefore, cost-effective decisions are essential, and not necessarily simple, in the Brazilian Atlantic Forest, and this theme was explored in few studies (Pinto and Grelle 2011; Crouzeilles et al. 2015; Vale et al. 2018a, b; Marcilio-Silva et al. 2018).

Although knowledge about the biodiversity of the Atlantic Forest has increased substantially in the last century (Marques et al. 2021), for all taxa covered in this book, it is evident that a large knowledge gap still remains (Ramos et al. 2021; Zwiener et al. 2021; Feitosa et al. 2021; Figueiredo et al. 2021; Padial et al. 2021). Such gaps happen, in part, because the surveys are made, in general, close to large urban centers or, else, restricted to a few protected areas with greater logistical facility for research, compromising the spatial range of the species distribution knowledge (Wallacean shortfall). In addition, for almost all groups, there is still a deficit between the number of existing species and the number of species already described (Linnean shortfall). Thus, the number of species, by counting new species and new taxa from taxonomic revisions, is still increasing, which demonstrates that the survey of biodiversity is a long ongoing process in super diverse regions such as the Atlantic Forest.

In a biome as widely distributed as the Atlantic Forest, with such biodiversity and involving the majority of the Brazilian population, it is expected that the interests among stakeholders can be multiple. With the exception of some initiatives that seek to integrate actors on a more regional scale (Grelle et al. 2021; Viveiros-de-Castro et al. 2021) or on a broader scale (De Siqueira et al. 2021), the conservation and production sectors present their idiosyncrasies that, often, hinder a convergence of actions. For example, while the productive areas and most of the protected areas are found in rural areas, the population of the cities are the ones that demand a good part of ecosystem services. This spatial distance between these actors generates a degree of ignorance about the importance of the Atlantic Forest natural areas. Also, the main Brazilian environmental law, the Law on the Protection of Native Vegetation (Law 12.651/2012), establishes the rule for land use in rural properties, which is a private asset. The enforcement of this law generally comes up against the private interests of the owner, which causes a historical conflict between the conservation and production sectors (Brancalion et al. 2016). In addition, the implementation of the protected areas system since 2000 (Law 9.985/2000) has never been completely finished, since land tenure regularization and the implementation of management plans have been hampered by the lack of investment by countless governments. Therefore, there is a generalized climate of discontent, which demands communication between the stakeholders in order for conservation to be effective.

## 23.5 Threats

According to the recent global report on biodiversity and ecosystem services, nature and its vital contributions to people, which together embody biodiversity and ecosystem functions and services, are deteriorating worldwide (IPBES 2019a, b). The main drivers of changes, namely, land/sea use changes, species direct exploitation, climate change, pollution, and invasive alien species, have accelerated during the past 50 years (IPBES 2019a, b). In Atlantic Forest, all these drivers are known to alter ecosystems, with possible drastic future effects on biodiversity and ecosystem services. For example, climate change and invasive species are large-scale processes with biodiversity loss in regional and small scale (Vale et al. 2021; Vitule et al. 2021). Changes in land use, despite having been reduced with the implementation of legal provisions (Atlantic Forest Law and Native Vegetation Protection Law), have still been marked by a recurring occurrence of illegal deforestation (Lira et al. 2021). Species overexploitation, for wood, palm hearth (the edible apex of *Euterpe edulis*), animal traffic, and hunting are still continually bringing new species to the Red lists of flora and fauna (Brasil 2018; Martinelli and Moraes 2013; Galetti et al. 2021).

In addition to the threats per se, other cascading effects tend to intensify species losses (Silva and Tabarelli 2000; Lôbo et al. 2011). For example, a common effect of climate change is the redistribution of species, including disease vectors (e.g., Carvalho et al. 2017). Therefore, emergence and reemergence of some diseases can be influenced by climate change. In fact, there is a clear and strong relationship between biodiversity and disease, and reduction of biodiversity affects the transmission of infectious diseases of humans (Keesing et al. 2010). In a forecast view, a recent meta-analysis shows that a high risk of zoonotic diseases in forested tropical regions changes in land-use and mammals' species richness (Allen et al. 2017). There is a new concept, namely, ecohealth, which is a multidiscplinary and emerging field research to understand the complex relationship between the environment – including biological and socioeconomic approaches – and human health. Consequently, ecohealth studies should include the synergy among biodiversity, climate change, and human health (e.g., many chapters in Marselle et al. 2019).

Another threat factor for the Brazilian Atlantic Forest is the continuity of biodiversity research efforts. Biodiversity monitoring is a goal of national governments, including Brazil, that is a signatory of multilateral agreement to map and perform long-term studies on organisms and ecosystems. However, national institutions need to be equipped with capacity and budget enough for a country to map, know, and put in action strategies to conserve biodiversity, without a necessary economic return in the short and medium term. It is expected to be more common in democratic nations (Dobrovolski et al. 2018; Rydén et al. 2019). The disruption of the political and financial system in Brazil that has been underway since 2016 has dramatically affected the main search engines (Fernandes et al. 2017). Without knowledge of biodiversity and its importance in supplying ecosystem services, it is impossible to implement more effective and long-term conservation actions (Magnusson et al. 2018). In addition, the current deepening in the Brazilian political crisis, implacable since 2019 by an extreme right-wing, authoritarian, and anti-environmentalist government (Abessa et al. 2019), causes enormous insecurity and uncertainty about the future of the Atlantic Forest.

## 23.6 Opportunities

The challenge of protecting a megadiverse and densely populated biome in one of the countries with the greatest social inequalities in the world requires strategies that manage to balance social development, with income distribution and nature conservation. In this sense, the greatest opportunities for the conservation of the Atlantic forest are invariably associated with the concept of sustainability.

The idea that human well-being is inextricably linked to biodiversity is the basis of the concept of ecosystem services. This concept is key to thinking about a future aligned with the United Nations' Sustainable Development Goals, which seek for solutions based on nature. For a megadiverse biome such as the Atlantic Forest and with a direct relationship with 60% of the Brazilian population, identifying ecosystem services and considering it in environmental agendas facilitate the dialogue between the stakeholders while allowing more effective results for sustainability to be achieved. The Atlantic Forest provides a significant part of the ecosystem services delivered for the Brazilian population. However, these services are spatially structured, with large portions of regulating services (water balance, carbon stock, and soil productive capacity) occurring in protected areas and provision services related to crop (sugarcane, soybean, and corn) and cattle productions distributed mainly in inland areas of non-protected areas (Pires et al. 2021). It is important to note that, although the provision of some services is located outside protected areas, many agricultural crops depend directly on the services of pollinating animals, which use neighboring forest remnants as refuge and habitat (BPBES/RIBBIP 2019; Varassin et al. 2021). Therefore, a promising future for the Atlantic Forest must consider ways to guarantee the delivery of ecosystem services, through incentives for the implementation and maintenance of protected areas, incentives for restoration, or even economic mechanisms that value such ecosystem services. Specifically, in scenarios of global changes, the adaptation based on ecosystem integrates the use of biodiversity and ecosystem services into an overall strategy to help people adapt to the adverse impacts of climate change (Scarano and Ceotto 2015; Vale et al. 2021).

Ecological restoration is one of the more promising ways to guarantee a sustainable future for the Atlantic Forest. Whether recovering the provision of ecosystem services or biodiversity, ecological restoration has become a mandatory issue on the global environmental agenda. For this reason, the period from 2021 to 2030 is being considered the decade of the restoration of ecosystems by the United Nations. The existence of this global agenda, coupled with others that are also underway (e.g., the

Bonn initiative; Forest Landscape Restoration initiative), creates a favorable environment for the implementation of major restoration programs of forests around the world. For the Atlantic Forest, the Atlantic Forest Restoration Pact is the main large-scale initiative that aims to recover 15 million hectares of forests (Calmon et al. 2011). It is an initiative that is already very well structured, with different actions at different scales, with a great potential to reverse part of the losses arising from the long and historic fragmentation process (De Siqueira et al. 2021). In addition to restoring degraded ecosystems, encouraging an agricultural system that is based on sustainability, such as agroforestry systems, landscape management, and low-impact agriculture, can also guarantee the sustainability of the Atlantic Forest. Agroforestry systems enhance ecosystem service provision, contribute to biodiversity conservation, improve food security, contribute to poverty alleviation, and enhance human well-being (Tubenchlak et al. 2021). Thus, there is a real possibility of aggregating economic and social values, as this initiative encourages the creation of productive chains, with the participation of many stakeholders, guaranteeing livelihoods for the rural population.

The high flow of capital, especially in the country's richest states located in the core of Brazilian Atlantic Forest, creates opportunities for financing and investments in sustainable projects. On the national scale, the mechanisms include sustainable public procurement, fiscal incentives, concessions for forestry and public use in protected areas, payments for environmental services, funds to boost avoided deforestation projects that reduce carbon emissions, and tradable environmental reserve quotas (Young and Castro 2021). International financial sources can also be important to develop conservation initiatives. In all situations, greater engagement by the private sector is necessary, as much as maintaining a public strategy to guarantee economic incentives, with strong environmental and social impact.

Many of the main present and future problems can be overcome with scientific improvement and technology. As a region strengthened by the presence of some of the main research centers in Brazil, it is possible to imagine that many solutions to the threats and internal weaknesses of the Atlantic Forest come from these research centers and universities. Technological innovations can play an important role in the conservation of biodiversity and ecosystem services. For example, technologies involving unmanned aerial vehicles can be useful for the inspection of protected areas, for monitoring endangered species, restoration areas, etc. Also, technologies that increase production in agricultural areas (reducing the pressure on the forest remnants) that control pollution in urban areas, etc. could be the new frontiers of knowledge in the Atlantic Forest.

A promising opportunity to fill knowledge gaps is the incentive of citizen science actions, as a strategy to acquire a massive amount of data continuously throughout the entire biome. Citizen science actions and projects are beneficial to scientific research, to biodiversity conservation, and to society. These programs have the potential of attaining large amounts of data throughout long time series and in several geographic scales, increasing the sampling effort and broadening the databases. They are also a very important way of enticing the public to interact with the scientific academy, which enhances their belonging for this activity and creates a bond between society, science, and conservation (Cooper et al. 2007; Dickinson and Booney 2012; Silvertown 2009; Wiggins and Crowston 2011; Roy et al. 2012; Cohn 2008). Promising sources of data to be exploited as a tool for biodiversity monitoring in the Brazilian Atlantic Forest are the *WikiAves* portal and the *Vem Passarinhar* program, organized by state environmental organizations and nongovernmental programs related to conservation and bird watching. The *WikiAves* portal has currently over 33 thousand collaborators and over 300 thousand avian records of 1890 Brazilian species (this encompasses about 99% of all Brazilian species).

## 23.7 A Promised Future

Brazilian Atlantic Forest has a long-standing history of changes in land use and land cover, monitored scientifically since 1985, as explained in some chapters of this book (Marques et al. 2021; Solórzano et al. 2021; Lira et al. 2021). Consequently, this monitoring allows first international scientific articles on the consequences of habitats loss at the biome scale in the 1990s (e.g., Brooks et al. 1999; Grelle et al. 1999). Recently, Rezende et al. (2018) estimated, with high-quality images, the current forest cover of 28% that is more than twice the forest cover previously estimated. Additionally, some regional studies have argued that Brazilian Atlantic Forest is in an advanced stage of the forest transition process (sensu Rudel et al. 2010), with reduction of habitat loss in the last years (Baptista and Rudel 2006; Costa et al. 2017; Calaboni et al. 2018). Taking only these numbers into consideration, one could imagine a future of hope for the biome, as enthusiastically announced by Rezende et al. (2018). However, analyses performed by Lira et al. (2021; Figure 2) using the temporal series of MapBioma Project (http://mapbiomas.org/) show a light evolution of forest cover between 1985 and 2017, with a forest cover of 30% in 1985 and 28% in 2017 and no evidence of forest cover increase in the last 30 years. Thus, at the moment, there is no evidence of forest transition in the Brazilian Atlantic Forest, at least in a single analysis for the whole of Biome.

Although the forest transition is in doubt, the hope placed on ecological restoration can be more promising to project a future for the Atlantic Forest (Rezende et al. 2018, De Siqueira et al. 2021). Recent large-scale restoration initiatives, based on inexpensive strategies such as natural regeneration (Crouzeilles et al. 2019, De Siqueira et al. 2021, Crouzeilles et al. 2021), can be promising in terms of reversing the effects of fragmentation and loss of species and ecosystem services on the biome. Evidently, the recovery of part of the biome's distribution area per se does not solve all the problems of the Atlantic Forest, as noted above. An alignment between public policies, with society's initiatives, making up all the social, economic, and ecological dimensions of sustainability, is a necessity to guarantee the certainty of the persistence of this important biome for future generations.

## 23.8 Conclusion

Along with this chapter and with the contribution of the 149 authors of 23 chapters of this book, we offer a pluralistic view of Brazilian Atlantic Forest. Whether describing the physical geography, history, patterns, and process of biodiversity or highlighting the several threats (changes in land use and land cover, global warming and climate changes, invasive species and biotic homogenization, and hunting) and opportunities (the potential of ecosystem service concept, the agroforestry, restoration ecology, the ecological economy vision, the initiatives to connect people and biodiversity conservation, and the appraisal of monitoring programs of biodiversity), we present an overview of the biome, synthesized in this chapter.

With a horizontal and vertical approach, although devoid of the coordination of an integrated research program (sensu Lakatos), knowledge about the Atlantic Forest has significantly increased over the last decades. In the last 35 years, we saw many taxonomists describing species (and still counting), physiologists understanding details of organism function, population ecologists monitoring spatial and temporal dynamics, community ecologists looking for the role of interspecific relationships and regional factors, and ecosystem ecologists studying the natural systems. In spite of many shortfalls of knowledge, Brazilian scientists advanced identifying aim threats of biodiversity, and search for solutions to conserve biodiversity and increase economic activities, such as agriculture. Furthermore, Brazilian scientists are concern with global problems such as climate change and ecosystem service, increasing the importance of Brazilian Atlantic Forest from a biodiversity hotspot to a place with people searching for solutions to local, regional, and global problems.

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## References

Abessa D, Famá A, Buruaem L (2019) The systematic dismantling of Brazilian environmental laws risks losses on all fronts. Nat Ecol Evol 3:510–511

Allen T, Murray KA, Zambrana C, Torrelio SS, Morse C, Rondinini M, Marco N, Breit KJ, Olival PD (2017) Global hotspots and correlates of emerging zoonotic diseases. Nat Commun 8:1124

Baptista SR, Rudel TK (2006) A re-emerging Atlantic Forest? Urbanization, industrialization and the forest transition in Santa Catarina, southern Brazil. Environ Conserv 33:195–202

BPBES/REBIPP (2019) Relatório temático sobre Polinização, Polinizadores e Produção de Alimentos no Brasil. Marina Wolowski; Kayna Agostini; André Rodrigo Rech; Isabela Galarda Varassin; Márcia Maués; Leandro Freitas; Liedson Tavares Carneiro; Raquel de Oliveira Bueno; Hélder Consolaro; Luisa Carvalheiro; Antônio Mauro Saraiva; Cláudia Inês da Silva. Maíra C. G. Padgurschi (org) 1ª edição, São Carlos, SP: Editora Cubo. 184 páginas. https://doi.org/10.4322/978-85-60064-83-0

Bergallo HG, Cronemberger C, Hipólito J, Magnusson WE, Rocha CFD (2021) Integrating researchers for understanding biodiversity in Atlantic Forest. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Boucher TM, Spalding M, Revenga C (2013) Role and trends of protected areas in conservation. In: Levin S (ed) Encyclopedia of biodiversity, vol 6, 2nd edn. Academic Press, pp 485–503

Brancalion PHS, Garcia LC, Loyola R et al (2016) A critical analysis of the native vegetation protection law of Brazil (2012): updates and ongoing initiatives. Nat Conserv 14S:1–15

Brasil (2018) Livro Vermelho da Fauna Brasileira Ameaçada de Extinção, vol I. ICMBio/MMA, Brasília, DF, p 1st. 492 pp

Brito D, Grelle CEV, Boubli JP (2008) Is the Atlantic Forest protected area efficient in maintaining viable populations of Brachyteles hypoxanthus? Biodivers Conserv 17:3255–3268

Brito D, Grelle CEV (2004) Effectiveness of a reserve network for the conservation of an endemic marsupial in Atlantic forest. Biodivers Conserv 13:2519–2536

Brooks TM, Tobias JA, Balmford A (1999) Deforestation and bird extinctions in the Atlantic forest. Anim Conserv 2:211–222

Brum FT, Pressey RL, Mauricio L, Loyola R (2019) Forecasting conservation impact to pinpoint spatial priorities in the Brazilian Cerrado. Biol Conserv 240:108283. https://doi.org/10.1016/j.biocon.2019.108283

Calaboni A, Tambosi LR, Igari AT, Metzger JP, Uriarte M (2018) The forest transition in São Paulo, Brazil: historical patterns and potential drivers. Ecol Soc 23:7

Calmon M, Brancalion PHS, Paese A, Aronson J, Castro P, da Silva SC, Rodrigues RR (2011) Emerging threats and opportunities for large-scale ecological restoration in the Atlantic Forest of Brazil. Restor Ecol 19(2):154–158

Carlucci MB, Marcilio-Silva V, Torezan JM (2021) The southern Atlantic Forest: use, degradation, and perspectives for conservation. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Carranza T, Balmford A, Kapos V, Manica A (2014) Protected area effectiveness in reducing conversion in a rapidly vanishing ecosystem: the Brazilian Cerrado. Conserv Lett 7:216–223. https://doi.org/10.1111/conl.12049

Carvalho BM, Rangel EF, Vale MM (2017) Evaluation of the impacts of climate change on disease vectors through ecological niche modelling. Bull Entomol Res 107(4):419–430

Clarivate Analytics 2018. Research in Brazil: A report for CAPES by Clarivate Analytics. Report

Cohn JP (2008) Citizen science: can volunteers do real research? Bioscience 58(3):192–197

Convention of Biological Diversity (2010) Strategic Plan for Biodiversity 2011–2020 and the Aichi Targets. Available at: https://www.cbd.int/sp/

Convention of Biological Diversity (2020) Post-2020 global biodiversity framework: discussion paper. Note by the executive secretary. Available at: https://www.cbd.int/conferences/post2020/post2020-prep-01/documents

Cooper CB, Dickinson J, Phillips T, Bonney R (2007) Citizen science as a tool for conservation in residential ecosystems. Ecol Soc 12(2)

Costa R, Prevedello JA, Souza BG, Cabral DC (2017) Forest transitions in tropical landscapes: a test in the Atlantic Forest biodiversity hotspot. Appl Geogr 82:93–100

Crouzeilles R, Beyer HL, Mills M, Grelle CEV, Possingham HP (2015) Incorporating habitat availability into systematic planning for restoration: a species-specific approach for Atlantic Forest mammals. Diver Distrib 21:1027–1037

Crouzeilles R, Beyer HL, Monteiro LM, Feltran-Barbieri R, Pessôa ACM, Barros FSM, Lindenmayer DB, Lino EDSM, Grelle CEV, Chazdon RL, Matsumoto M, Rosa M, Latawiec AE, Strassburg BBN (2021) Achieving cost-effective landscape-scale forest restoration through targeted natural regeneration. Conserv Lett:e12709

Crouzeilles R, Santiami E, Rosa M, Pugliese L, Brancalion PHS, Rodrigues RR, Metzger JP, Calmon M, Scaramuzza CADM, Matsumoto MH, Padovezi A, Benini R d M, Chaves RB, Metzker T, Fernandes RB, Scarano FR, Schmitt J, Lui G, Christ P, Vieira RM, Senta MMD, Malaguti GA, Strassburg BBN, Pinto S (2019) There is hope for achieving ambitious Atlantic Forest restoration commitments. Perspect Ecol Conserv 17(2):80–83

De Masi D (2019) O mundo ainda é jovem. Conversas sobre o futuro próximo. Ed. Vestígio, São Paulo

De Siqueira LP, Tedesco AM, Rodrigues RR et al (2021) Engaging people for large-scale forest restoration: governance lessons from the Atlantic Forest of Brazil. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Dickinson JL, Booney R (2012) Citizen science: public participation in environmental research, 1st edn. Comstock Publishing Associates

Dobrovolski R, Loyola R, Rattis L, Gouveia SF, Cardosoa D, Santos-Silva R, Gonçalves D, Alves-Souza LMB, Diniz-Filho JAF (2018) Science and democracy must orientate Brazil's path to sustainability. Perspect Ecol Conserv 16:121–124

Eduardo AA, Nunes AV, Brito D (2012) Do the protected areas network of the state of Minas Gerais maintain viable populations of the lowland tapir (Tapirus terrestris)? NatConserv 10:27–33

Faria D, Delabie JHC, Dias MH (2021) The Hileia Baiana – an assessment of natural and historical aspects of the land use and degradation of the central corridor of the Brazilian Atlantic Forest. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Feitosa RM, Morini MS, Martins AC et al (2021) Social insects of the Atlantic Forest. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Fernandes GW, Vale MM, Overbeck GE, Bustamante MMC, Grelle CEV et al (2017) Dismantling Brazil's science threatens global biodiversity heritage. Perspect Ecol Conserv 15:239–243

Figueiredo MSL, Weber MM, Brasileiro CA et al (2021) Tetrapod diversity in the Atlantic Forest: maps and gaps. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Freygang CC, Marinho JR, Freitas TRO (2004) New caryotypes and some considerations about chromosomal diversification in Ctenomys minutus (Rodentia: Ctenomydae) on the coastal plain of the Brazilian State of Rio Grande do Sul. Genetica 121:125–132

Galetti M, Gonçalves F, Villar N et al (2021) Causes and consequences of large-scale defaunation in the Atlantic forest. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Gao CY, Peng DH (2011) Consolidating SWOT analysis with nonhomogeneous uncertain preference information. Knowl-Based Syst 24:796–808

Grelle CEV, Bayma AP, Lourenço LR et al (2021) Conservation initiatives in the Brazilian Atlantic forest. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Grelle CEV, Fonseca GAB, Fonseca MT, Costa L (1999) The question of scale in threat analysis: a case study with Brazilian mammals. Anim Conserv 2:149–152

Harari YN (2018) 21 lições para o século 21. Companhia das Letras, São Paulo

IPBES (2019a) Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. In: Brondizio ES, Settele J, Díaz S, Ngo HT (eds) IPBES secretariat, Bonn, Germany

IPBES (2019b) Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. In: Díaz S, Settele J, Brondízio ES, Ngo HT, Guèze M, Agard J, Arneth A, Balvanera P, Brauman KA, Butchart SHM, Chan KMA, Garibaldi LA, Ichii K, Liu J, Subramanian SM, Midgley GF, Miloslavich P, Molnár Z, Obura D, Pfaff A, Polasky S, Purvis A, Razzaque J, Reyers B, Chowdhury RR, Shin YJ, Visseren-Hamakers IJ, Willis KJ, Zayas CN (eds) IPBES secretariat, Bonn, Germany. 56 p. https://doi.org/10.5281/zenodo.3553579

Joly CA, Metzger JP, Tabarelli M (2014) Experience from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. New Phytol 204:459–473

Keesing F, Belden LK, Daszak P, Dobson A, Harvell CD, Holt RD, Hudson P, Jolles A, Jones KE, Mitchell CE, Myers SS, Bogich T, Ostfeld RS (2010) Impacts of biodiversity on the emergence and transmission of infectious diseases. Nature (London) 468:647–652

Lins-e-Silva ACB, Ferreira PSM, Rodal MJN (2021) The North-eastern Atlantic Forest: biogeographical, historical and current aspects in the Sugarcane Zone. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Lira PK, Portela RCQ, Tambosi LR (2021) Land-cover changes and an uncertain future: will the Brazilian Atlantic Forest lose the chance to become a *hopespot*? In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Lôbo D, Leão T, Melo FPL, Santos AMM, Tabarelli M (2011) Forest fragmentation drives Atlantic forest of northeastern Brazil to biotic homogenization. Divers Distrib 17:287–296

Magnusson WE, Grelle CEV, Marques MCM et al (2018) Effects of Brazil's political crisis on biodiversity conservation. Frontiers Ecol Evol 6:163

Marcilio-Silva V, Marques M, Cavender-Bares J (2018) Land-use trade-offs between tree biodiversity and crop production in the Atlantic Forest. Conserv Biol 32:1074–1084

Margules CR, Pressey RL (2000) Systematic conservation planning. Nature 405:243–253

Marques MCM, Trindade W, Bohn A, Grelle CEV (2021) The Atlantic Forest: an introduction to the megadiverse forest of Southern America. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Marselle MR, Stadler J, Korn H, Irvine KN (2019) Biodiversity and health in the face of climate change. Springer Open. 481p

Martinelli G, Moraes MA (2013) Livro Vermelho da Flora Brasileira. Andrea Jakobson Estúdio; Instituto de Pesquisas Jardim Botânico do Rio de Janeiro, Rio de Janeiro. 1100p

Myers N, Mittermeier RA, Mittermeier CG et al (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Nolte C, Agrawal A, Silvius KM, Soares-filho BS (2013) Governance regime and location influence avoided deforestation success of protected areas in the Brazilian Amazon. Proc Natl Acad Sci 110:4956–4961. https://doi.org/10.1073/pnas.1214786110

Padial AA, Costa APL, Bonecker CC et al (2021) Freshwater studies in Atlantic Forest: general overview and prospects. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Peres EA, Pinto-da-Rocha R, Lohmann G et al (2020) Patterns of species and lineage diversity in the Atlantic rainforest of Brazil. In: Rull V, Carnaval AC (eds) Neotropical diversification: patterns and processes. Springer, Switzerland

Pinheiro RS, Mellinger L, Scarano FR (2014) Os povos da Mata Atlântica. In: Scarano FR (org) Mata Atlântica: uma história do futuro. Série Biomas Brasileiros. Edições de Janeiro., Conservação Internacional, Rio de Janeiro

Pinto MP, Grelle CEV (2011) Minimizing conservation conflict for endemic primate species in Atlantic forest and uncovering knowledge bias. Environ Conserv 39:30–37

Pinto MP, Grelle CEV (2009) Reserve selection and persistence: complementing the existing Atlantic Forest reserve system. Biodivers Conserv 18:957–968

Pinto MP, Silva-Júnior JS, Lima AA, Grelle CEV (2014) Multi-scales analysis of primate diversity and protected areas at a megadiverse region. PLoS One 9(8):e105205. https://doi.org/10.1371/journal.pone.0105205

Pires APF, Shimamoto CY, Padgurschi MCG, Scarano FR, Marques MCM (2021) Atlantic Forest: ecosystem services linking people and biodiversity. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Ramos FN, Mortara SR, Elias JPC (2021) Vascular Epiphytes of the Atlantic Forest: diversity and community ecology. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Rezende CL, Scarano FR, Assad ED, Joly CA, Metzger JP, Strassburg BBN, Tabarelli M, Fonseca GA, Mittermeier RA (2018) From hotspot to hopespot: an opportunity for the Brazilian Atlantic Forest. Perspect Ecol Conserv 16:208–214

Ribeiro MC, Metzger JP, Martensen AC, Ponzoni FJ, Hirota MM (2009) The Brazilian Atlantic Forest: how much is left, and how is the remaining forest distributed? Implications for conservation. Biol Conserv 142:1144–1156

Roy HE, Pocock MJO, Preston CD, Roy DB, Savage J, Tweddle JC, Robinson LD (2012) Understanding Citizen Science & Environmental Monitoring. Final Report on behalf of UK-EOF. NERC

Rudel TK, Schneider L, Uriarte M (2010) Forest transitions: Na introduction. Land Use Policy 27:95–97

Rydén O, Zizka A, Jagers SC, Lindberg SI, Antonelli A (2019) Linking democracy and biodiversity conservation: empirical evidence and research gaps. Ambio:1–15

Scarano FR, Ceotto P (2015) Brazilian Atlantic forest: impact, vulnerability and adaptation to climate change. Biodivers Conserv 24:2319–2331

Silva JMC, Tabarelli M (2000) Tree species impoverishment and the future flora of the Atlantic forest of Northeast Brazil. Nature 404(6773):72–74

Silvertown J (2009) A new dawn for citizen science. Centre for Ecology & Hydrology and Natural History Museum. Trends Ecol Evol 24(9):467–471

Solórzano A, Brasil LSCA, Oliveira RR (2021) The Atlantic Forest ecological history: from prehistory to the Anthropocene. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Tubenchlak F, Badari CG, Strauch GF et al (2021) Changing the agriculture paradigm in Brazilian Atlantic Forest: the importance of agroforestry. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

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

Vale MM, Tourinho L, Lorini ML, Rajão HB, Figueiredo MSL (2018b) Endemic birds of the Atlantic Forest: traits, conservation status, and patterns of biodiversity. J Field Ornithol 89(3):193–206

Vale MM, Arias PA, Ortega G et al (2021) Climate change and biodiversity in the Atlantic Forest: best climatic models, predicted changes and impacts, and adaptation options. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Varassin IG, Agostini K, Wolowski M, Freitas L (2021) Pollination systems in the Atlantic Forest: characterisation, threats, and opportunities. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Vieira RRS, Pressey RL, Loyola R (2019) The residual nature of protected areas in Brazil. Biol Conserv 233:152–161

Vitule JRS, Occhi TVT, Carneiro L et al (2021) Non-native species introductions, invasions, and biotic homogenization in the Atlantic Forest. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Viveiros-de-Castro EB, Lanna AM, Lobo AC et al (2021) The Atlantic Forest trail: reconnecting people, biodiversity and protected areas. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Wiggins A, Crowston K (2011) From conservation to crowdsourcing: A typology of citizen science. 44th Hawaii International Conference on System Sciences, Kauai, HI. pp. 1–10

Young CEF, Castro BS (2021) Financing conservation in the Brazilian Atlantic forest. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

Zwiener VP, Lima RAF, Sánchez-Tapia A et al (2021) Tree diversity in the Brazilian Atlantic Forest: biases and general patterns using different sources of information. In: Marques MCM, Grelle CEV (eds) The Atlantic Forest: history, biodiversity, threats and opportunities of the megadiverse forest. Springer, Switzerland

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