

Fernando Ravanini Gardon

Serviços ecossistêmicos de carbono, composição e
estrutura florestal em uma cronosequência de
plantios de restauração

Carbon ecosystem services, forest composition and
structure in a chronosequence of restoration
plantings

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Orientadora: Dra. Rozely F. dos Santos

Co-orientador: Dr. Ricardo R. Rodrigues

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Comissão Julgadora:

Prof(a). Dr(a).

Prof(a). Dr(a).

Prof(a). Dr(a).

Prof(a). Dr(a).

Profa. Dra. Rozely Ferreira dos Santos

Orientadora

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Dedico esta tese à Rozely Ferreira dos Santos,
uma grande amiga e inspiração.

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“In the planting of the seeds of most trees, the best gardeners do no more than follow Nature....”

The Succession of Forest Trees - Henry David Thoreau (1860)

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Resumo

A restauração florestal é uma estratégia amplamente utilizada para reverter o atual cenário de perda de biodiversidade, degradação das funções ecológicas e intensificação das mudanças climáticas. Entretanto, é preciso atestar se as ações de restauração florestal já executadas estão efetivamente cumprindo esses papéis. Neste contexto, objetivamos avaliar tanto o potencial de florestas restauradas em acumular carbono e reestabelecer as funções ecológicas envolvidas neste processo, quanto sua contribuição para a conservação da flora nativa. Primeiramente, nós realizamos uma revisão de literatura para fornecer uma visão em nível nacional das estimativas de carbono acima do solo em florestas restauradas por diferentes métodos (restaurações ativa e passiva), discutindo as tendências e lacunas no conhecimento acumulado e nas pesquisas conduzidas no Brasil até o ano de 2019. Os resultados mostram que restaurações florestais brasileiras apresentam alto potencial em contribuir para os serviços de carbono, sendo que a restauração ativa se mostra mais promissora para este fim, pelo menos em um período de 30 anos. Contudo, somente 23% dos estudos de biomassa e carbono foram conduzidos em restaurações ativa. Ademais, apenas 9% dos estudos avaliaram a relação entre o estoque de carbono e a diversidade florística das restaurações. Devido a este desequilíbrio no conhecimento acumulado, nós buscamos fornecer estimativas robustas da contribuição de restaurações ativa tanto para os serviços ecossistêmicos de carbono, quanto para a conservação da flora nativa. Assim, nós avaliamos estes parâmetros em 16 plantios de restauração com idades entre 5 e 30 anos, de forma a prover uma análise temporal dos resultados. As estimativas mostram que plantios com 30 anos de idade apresentam estoque de carbono similar a remanescentes florestais maduros. Apesar de indivíduos de pequeno porte serem numericamente dominantes independentemente da idade das restaurações, os resultados indicam que plantas de grande porte passam a deter estoque de carbono superior antes do fim da primeira década após o plantio. Entre os grupos sucessionais, espécies secundárias passam a ser mais abundantes que pioneiras somente após 20 anos, porém muito antes

disso o estoque de carbono em espécies secundárias já se mostra similar ao das pioneiras. Observamos que a dinâmica que se estabelece nessas comunidades é marcada pela transição de plantas de diferentes classes de tamanho e grupos ecológicos, sendo que os padrões observados promovem a trajetória sucessional e, possivelmente, a perpetuação dos processos envolvidos no acúmulo de carbono ao longo do tempo. Mesmo que os resultados relacionados a estrutura, grupos funcionais e estoque de carbono das restaurações sejam bastante positivos, o potencial destas comunidades em contribuir para a conservação de espécies vegetais poderia ser melhor desenvolvido. Assim, apesar dos plantios de restauração contribuírem para a manutenção da diversidade florística em regiões antropizadas, seu papel na conservação de espécies prioritárias é menos destacável. Contudo, visto que constatamos nos plantios a presença de diferentes espécies relevantes para o funcionamento das comunidades e para a oferta de serviços ecossistêmicos de carbono, assumimos que plantios de restauração florestal inseridos em paisagens agrícolas representam uma grande oportunidade de promover resultados sinérgicos. Por outro lado, este potencial não se mostra totalmente aproveitado, pois espécies prioritárias para conservação, principalmente aquelas ameaçadas e endêmicas, não são frequentes ou abundantes nestas florestas. Por fim, nós fornecemos diretrizes que podem amenizar as limitações identificadas e maximizar o caráter multifuncional das restaurações, garantindo que os objetivos previstos no planejamento destas ações sejam efetivamente alcançados.

Abstract

Forest restoration is a strategy widely used to reverse the current scenario of biodiversity loss, ecological functions degradation, and climate change intensification. However, it is necessary to certify whether the implemented forest restorations are effectively fulfilling these roles. In this context, we aimed to evaluate the potential of active forest restorations in accumulating carbon and reestablishing the ecological functions involved in this process, in addition to their contribution to the conservation of native plant species. First, we conducted a literature review to provide a national-level view of the aboveground carbon estimates in forests restored by different methods (active and passive restorations). We discussed trends and gaps in accumulated knowledge and the researches conducted in Brazil until the year 2019. The results show that Brazilian forest restorations present a high potential to contribute to carbon services regardless of the restoration method, but active restoration shows more promising results, at least over a period of 30 years. However, only 23% of the biomass and carbon studies in Brazil were conducted in active restorations. In addition, the relationship between the carbon stock of the restorations and their respective plant diversity was present in only 9% of the studies. Due to this bias in the accumulated knowledge, we aimed to provide robust estimates of the contribution of active restorations for carbon ecosystem services and the conservation of native flora. Thus, we evaluated these parameters in 16 restoration plantings aging from 5 to 30 years-old, to provide a temporal analysis of the results. Our estimates show that 30y-restorations can present carbon stocks similar to mature forest remnants. Although small individuals are numerically dominant regardless of the restorations' age, the results indicate that large plants hold the higher carbon stock before the end of the first decade after planting. Among the ecological groups, secondary species become more abundant than pioneers after 20 years, but carbon stock in secondary species is already similar to the pioneer ones earlier on the chronosequence. We observed that the dynamics established in these communities is marked by the transition among plants from

different size classes and ecological groups. We can infer that the observed patterns promote the successional trajectory and, possibly, the perpetuation of the processes involved in the carbon accumulation over time. Although the outcomes related to structure, functional groups, and carbon stocks of the restorations are quite positive, their role in plant conservation could be better developed. Thus, although restoration plantations contribute to the maintenance of floristic diversity in anthropized regions, their role in conserving priority plant species is less remarkable. However, with the presence of different species relevant to the ecosystem functioning and the provision of carbon ecosystem services within the restorations, we assume that active restorations embedded in agricultural landscapes provide a huge opportunity to promote synergistic outcomes. On the other hand, this potential is not fully employed, as priority species for conservation, especially those threatened with extinction and endemic, are not frequent or abundant in these forests. Finally, we provide guidelines that could overcome the identified limitations and maximize the multifunctional character of the active restorations, ensuring that the objectives foreseen in the planning of these actions are effectively achieved.

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Introduction

Historically, the substitution of natural ecosystems by human activities has led regions of extensive and continuous tropical forests to a fragmented condition, where some of these landscapes are now almost absent of native vegetation (Chazdon, 2014; McFarland, 2018). It is known that the supply of ecosystem services (ES) depends on ecological processes and functions sustained by biodiversity, their interactions, and the biotic/abiotic elements of the landscapes (Aerts and Honnay, 2011; Cardinale et al., 2012; Naeem et al., 2012; Wu, 2013). In the tropics, this interaction network has been jeopardized for centuries, but after the year 1920, the deforestation rates increased substantially due to the expansion of agricultural areas and the overexploitation of natural resources for economic purposes (Chazdon, 2014; McFarland, 2018). Thus, the intense change in the land-use/cover conditions reduced the availability of habitat for native fauna and flora, threatening both biodiversity and the supply of vital ES, as climate regulation and natural resources provisioning (Aerts and Honnay, 2011; Wu, 2013; Brockerhoff et al., 2017; McFarland, 2018; Grass et al., 2019).

According to McDonald et al. (2016), if we consider the extent to which the degradation of natural ecosystems has proceeded and continues to expand, protecting remaining ecosystems is no longer sufficient to safeguard biodiversity, ecosystems functioning and the ES lost by human activities. In this context, ecological restoration represents one of the main global tools to overcome the current scenario of environmental negligence. The “Society for Ecological Restoration - SER”, defines ecological restoration as “*the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed*” (Gann et al., 2019). This international

community emphasizes that ecological restoration diverges from other methods of ecosystem recovery because it focuses on the restoration of native ecosystems and their ecological integrity, directing the recovering path towards a persistent and resilient system, where the species can adapt and interact (SER, 2004; McDonald et al., 2016; Gann et al., 2019). Not surprisingly, the importance of forest restoration programs for biodiversity and ES has been a focal point on the global committees and conferences on environmental conservation and climate change, such as the COP 21, United Nations Climate Change Conference, Bonn Challenge, and the Initiative 20x20. Moreover, the United Nations (UN) declared the period between 2021 and 2030 as the restoration decade, which leveraged the restoration programs to an outstanding position, raising the expectations surrounding the success of these actions.

In this context, it is imperative to assess, at both local and regional scales and over time, the role of forest restoration in biodiversity conservation, the reestablishment of the ecological functions and ES supply, aiming to prove the true contribution of the restoration initiatives. Among the several ES sustained by forest ecosystems, their ability in sequestering atmospheric carbon and storing it in plant biomass constitutes one of the most relevant regulating ES, which contributes to atmospheric greenhouse gas reduction (Jobbágy and Jackson, 2000; Cerri et al., 2004). Such ability is, in fact, an ecological function that can and should be promoted through forest restoration, as a means for countries to achieve their goals toward mitigating the effects of climate change. Regarding this, it is essential to know the trajectory of the carbon accumulation in forest restorations overtime, concerning the influence of structural and compositional aspects of the restored communities on the carbon-related process.

Although the role of restored forests in supplying carbon ES is a global consensus, and studies have contributed to the expansion of this knowledge, one of the most used arguments to bring attraction to forest restoration programs is related to its potential in recovering biodiversity, including different groups of both fauna and flora threatened with extinction and/or endemic (Benayas and Bullock. 2012; IUCN, 2016; Brancalion et al., 2018; Lindenmayer, 2019). Thus, it is expected that ecological restoration, when successful, be able to promote synergistic effects,

contributing both to the supply of ES and to biological conservation (IUCN, 2016; McDonald et al., 2016; Crouzeilles et al., 2017; Gann et al., 2019). However, in forests undergoing restoration, plant diversity/composition presents a slow recovering process comparing to forests structure and ES reestablishment (Aide et al., 2000; Martin et al., 2013; Suganuma and Durigan, 2015). This implies that assessments should be conducted to provide a robust database related to the role of forest restoration projects in plant conservation, which could subsidy restoration practitioners, planners, and decision-makers in the interpretation of the responses of plant diversity to the progress of the restored communities over time. This is necessary to safeguard the expected outcomes and to reliably estimate the potential of forest restorations in contributing to both plant species diversity and climate change mitigation issues.

We must recognize that forest restoration is an old human activity, historically practiced to solve applied problems as erosion control, but restoration ecology is a relatively new area of knowledge, well established only at the end of the twentieth century (Young, 2005). In consequence, restoration methodologies as planting seedlings grounded on ecological principles began to be practiced consistently in some tropical regions only in the last few decades (Rodrigues et al., 2009). This condition hampers assessments of the restoration outcomes in a broad temporal context, being necessary to adopt strategies that allow predicting the results over time. In tropical forests, most studies of the trajectory of recovering communities are derived from chronosequences, where the analyses are based on sampling several forests of different ages, allowing to obtain results on a temporal scale (Chazdon et al., 2007; Feldpausch et al., 2007). Some authors have questioned, or have been careful about, the validity of chronosequence analyzes since confusing factors can be easily missed on the site selection phase (Chazdon et al., 2007; Feldpausch et al., 2007; Letcher and Chazdon, 2009). However, in the context of forest restoration, especially the active restoration methods practiced by techniques as planting seedlings, several characteristics support the use of this approach to predict the outcomes. Suganuma and Durigan (2015) state that, restoration plantings of the same region may present a standard and predictable trajectory for attributes as structure, richness, and functional guilds.

Considering the arguments presented and the eminent scenario of biodiversity loss and climate change, creating a robust scientific knowledge towards the dynamics established in restored communities is necessary to communicate the outcomes obtained. This even more important when laborious and costly forest restoration programs are implemented, as the active restoration projects (Benini and Adeodato, 2017). Therefore, we aimed to answer questions related to the contribution of restored forests to carbon ES, the recovery of ecosystem functioning, and plant species conservation. We selected four questions concerning this subject: (i) What is the state of the art of the knowledge related to biomass and carbon estimates in Brazilian forest restorations?; (ii) How the temporal pattern of carbon accumulation relates to forest structure in a chronosequence of restoration plantings?; (iii) Do species functional attributes and plant community dynamics affects carbon accumulation in restoration plantings over time?; and (iv) Can forest restoration plantings contribute to floristic diversity and priority plant species conservation, providing synergistic outcomes in agricultural landscapes?

The proposed questions were addressed in the different chapters of this thesis. We designed the thesis to provide a national overview of the proposed subject, and to reliable evaluate the outcomes from active restorations embedded in Brazilian agricultural landscapes.

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Chapter 1.

Brazil's forest restoration, biomass and carbon stocks: A critical review of the knowledge gaps

Fernando Ravanini Gardon^a; Rozely Ferreira dos Santos^a; Ricardo Ribeiro Rodrigues^b

^a Department of Ecology, Institute of Biosciences, University of São Paulo, São Paulo, Brazil

^b Laboratory of Forest Ecology and Restoration, Department of Biological Sciences, University of São Paulo, Escola Superior de Agricultura “Luiz de Queiroz”, Piracicaba, São Paulo, Brazil

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1.1. ABSTRACT

Throughout tropical regions, forest restoration is a priority in conservation programs to reestablish ecological processes, as biomass accumulation, and the delivery of ecosystem services, as climate change mitigation by sequestering atmospheric carbon. Forest restoration has been an important tool for governments to reach their goals and compromises related to biodiversity conservation and carbon emissions reduction. In large countries as Brazil, to reliably estimate biomass accumulation and carbon sequestration potential in forest restorations can be hampered by the high variability of forests, soil types, territorial extension, landscape conditions and land-use history, where different restoration methods are expected to present different outcomes. In this context, we conducted a critical review of the theoretic-methodological scientific knowledge accumulated in the field of aboveground biomass recovery and carbon sequestration in Brazilian restored forests. We observed that the main knowledge limitations are related to territorial distribution and methodological issues encompassed by the biomass studies. Studies of biomass in Brazil focus on early (<30y), passive and Amazon biome restoration sites. Methodological criteria presented several limitations among studies, ranging from the allometric equation choice to the inclusion of non-tree life-forms on biomass stocks estimates. Biomass recovery by restorations is weakly addressed for some Brazilian regions, either by the absence of studies or by the weakness of the methods used to estimate biomass and carbon stocks. We concluded that efforts need to be applied to improve stated knowledge, and developing standardized protocols for biomass monitoring is crucial to determine the real contribution of Brazilian forest restorations to carbon sequestration and climate change over time.

Keywords: Aboveground biomass; Carbon sequestration; Ecological Restoration; Tropical Forest; Ecosystem Services.

1.2. INTRODUCTION

Globally, tropical forests have been regarded as diversity pools and carbon sinks, and its conservation plays an important role in the context of climate change mitigation (IPCC, 2005; Chazdon et al., 2016; Poorter et al., 2016). Estimates suggest that forests store up to 80% of terrestrial carbon (Houghton, 2008), and throughout tropical regions, forest restoration has become a conservation tool to reestablish ecological processes as biomass and carbon accumulation, and the delivery of regulating ecosystem services, as reducing the effects of climate change (Chazdon, 2008; Locatelli et al., 2015; Meli et al., 2017).

The potential of restored forests in sequestering atmospheric carbon is closely attained to ecological aspects, as forest type, restoration method and management, landscape context, and environmental conditions (Chazdon, 2014; Chazdon et al., 2016; Crouzeilles et al., 2017; James et al., 2018). Despite the particular biotic and abiotic features of each restoration, restoring forests by any method (planting seedlings or assisting natural regeneration) drives to a local increment in biomass and carbon stocks (Benini and Adeodato, 2017). Since biomass accumulation can be the main driver of plant community changes during tropical forest succession (Lohbeck et al., 2015), and forest restoration is a global strategy for climate change mitigation, ensuring the reestablishment of this ecological processes could afford both restoration success and vital regulating ecosystem services.

In Brazil, country which holds the second largest amount of natural forest in the globe (SFB, 2013), many restoration projects have been implemented due to the legal obligation imposed by the Forest Code (Federal Law, 12.651), and this activity became a potential agribusiness productive chain instead of just a visionary and romantic concern (Benini and Adeodato, 2017). Thus, a robust scientific knowledge related to biomass accumulation in forest restoration is expected to support robust evaluations of the real contribution of these systems to carbon ecosystem services (Van der Gaast et al., 2018).

In this context, we aimed to conduct a systematic review in the field of aboveground biomass recovery and carbon sequestration by forest restorations in Brazil. We highlight the limitations of the accumulated theoretic-methodological scientific knowledge for generalizations of the biomass stock in restored forests throughout the Brazilian territory.

1.3. METHODS

1.3.1. STUDIES SEARCHES AND SELECTION

We conducted a systematic and integrative review to search for biomass or carbon evaluations conducted in Brazilian Passive Restoration (PR) and Active Restoration (AR) sites. A systematic qualitative and quantitative literature review is a robust and replicable method, based on a comprehensive and detailed search strategy, able to identify boundaries around generalizations and important geographic, scalar, theoretical and methodological gaps in the literature (Pickering and Byrne, 2014; Torraco, 2016). We compiled information related to the geographic distribution of the studies, the methods used to estimate AGB/carbon stocks, existing bias in the researches, the main factors influencing AGB/carbon recovery, and trends and limitations of the researches carried out in Brazil. We selected only scientific studies available in indexed bibliographic databases and published in English or Portuguese.

The searches were performed in Scopus and Web of Science databases for all years available, with the last view on October 2019. We selected the search terms to account for studies that evaluated AGB/carbon stocks in restoration sites implemented by active (planting seedlings and seeding) and passive (natural regeneration and assisted regeneration) restoration methods in Brazil. The selected terms were *forest** (ecosystem), *regrowth OR regener* OR restor* OR recover* OR reforest* OR reveget** (processes involved in the ecosystem recovery), *biomass OR carbon* (targeted ecological process), and *Brazil* (studies geographic distribution). As Wortley et al (2013), the terms “*reclamation*” or “*rehabilitation*” were not included, in order to focus on studies that were in line with the *Society for Ecological Restoration* (SER) definition of ecological restoration. Besides that, standardization of restoration terms still in process (McDonald et al.,

2016), and, for this reason, terms as “reforestation” and “recovery” were also included to account for more published studies. Because we used the query “forest”, we highlight that our results are representative of the Brazilian forest formations embedded in Amazon, Atlantic Forest, Caatinga (Tropical Dry Forest) and Cerrado (Savanna) biomes.

We refined the search by identifying the studies that addressed the following criteria: i) studies carried out exclusively in the Brazilian territory; ii) empirical studies, field surveys of observational data (excluding reviews, meta-analyses, perspectives, modeling, GIS assessments, and scenarios papers); iii) studies that directly or indirectly estimated biomass or carbon in restoration sites; iv) studies conducted in areas previously forested, where native vegetation was removed for human interventions (agriculture, pasture, clear-cut, etc) and then naturally regenerated or restored by other methods; and v) excluded areas of selective logging, assessment of gap dynamics, studies conducted in greenhouses, experimental sites, commercial plantations, monocultures, and agroforestry.

We selected only the studies that evaluated live and/or dead AGB and/or carbon (i.e. litterfall and plants aerial parts). We decided to restrict our study to aboveground stock because most of the belowground (soil) studies found did not included roots contribution, but soil carbon contents and microbial biomass. For this reason, we kept our review aligned with aboveground compartments, since plants structures were not the main target of belowground biomass studies.

1.3.2. STUDIES CLASSIFICATION

Studies included in the final list were classified within criteria enclosed in Table 1.

Table 1. Categories and subcategories of studies classification.

Publication detail	Site information	Evaluation methods	Factors of influence on AGB recovery
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Subcategory	Year of publication; Journal of publication; Author's institutions	Location;	Life-forms evaluated (e.g. trees, shrubs, palms, lianas, and herbaceous plants);	Landscape;
		Forest Biome;	AGB or carbon equation;	Forest attributes;
		n° areas evaluated;	Plot size and total area sampled/site;	Restoration implementation;
		Restoration age;	Criteria of plant individuals' inclusion (minimum diameter or height);	AGB estimate;
		Restoration method (passive active);	Wood specific gravity and carbon factor usage	Past land-use;
		Past land-use		Soil properties;
				Plant interaction;
				Climate;

The geographic position of all sites sampled in the selected studies was mapped to the most detailed scale provided in the article (coordinates, locality, municipality, state). Where it was possible to identify the same site in more than one article it was mapped once, thus we found 35 replicated sites. The biome was determined based on the Brazilian Biomes Map provided by the Brazilian Institute of Geography and Statistics (IBGE 2004) in 1:5 000 000 scale. All mapping procedures were conducted in QGIS (Version 3.4.12). Past land-use was determined only if the information contained in the article was local, not the surroundings. Some articles defined thresholds in the plant individuals' size (diameter and height) to be included in their database, and we obtained this information for all articles. We defined as passive restorations sites areas where forest was cleared, used or not for human activities, and then abandoned for natural regeneration recovery, that is, secondary forests. On the other hand, active restorations were defined as sites previously forested, used or not for human activities, but recovered by human interventions, as planting seedlings and direct seeding.

1.3.3. ABOVEGROUND BIOMASS TEMPORAL ANALYSIS

We extracted the AGB/carbon values and age of the restoration sites from the texts, tables, and figures available in the selected publications. We included in the analyses AGB values

obtained by the use of allometric and volume equations or destructive methods to estimate the AGB stored in the plant individuals. We only attained information from restorations of age <30y, since this is the most common age class of the restoration sites found among the studies (>90%).

Due to the impossibility in determining the exact age or AGB stock for some of the restorations sites studied in the articles, the data set is composed of individual and grouped (average age or AGB) values. Thus, we obtained both AGB and age for 177 events. Data presented as carbon values were converted to biomass by the carbon conversion factor given in the particular study.

Because AGB recovery over time is a non-linear process and is expected to be different among restoration methods (Holl and Zahawi, 2014), we estimate sites' AGB stocks ($\text{MgAGB}\cdot\text{ha}^{-1}$) as a function of age ensuring an asymptotical shape in the AGB accumulation pattern. We built second-degree individual polynomial models for each restoration method (passive and active). The quality of the model was evaluated based on the adjusted R squared (R^2), residuals standard error (RSE), and F-statistics (p-value).

All data and graphics were manipulated using R 3.5.1 (R Core Team, 2018).

1.4. RESULTS

1.4.1. PUBLICATION'S DETAILS

We found 843 studies, whose 792 studies that did not match the adopted criteria of inclusion were removed and 51 studies that evaluated aboveground biomass and/or carbon stocks in AR and PR sites in Brazil were retained (Figure A.1). The first article found was published in 1988 (Uhl et al., 1988), thus displaying a range of 31 years of publications (Figure 1), with an average of 1.6 article per year, with 65% of the articles published between the years 2009-2019. The biome Amazon was the first Brazilian forest formation to be evaluated for AGB (Uhl et al., 1988). Only after the year 2000, biomass was evaluated at restorations of different forest types, as the Atlantic Forest, Cerrado, and Caatinga biomes (Figure 1).

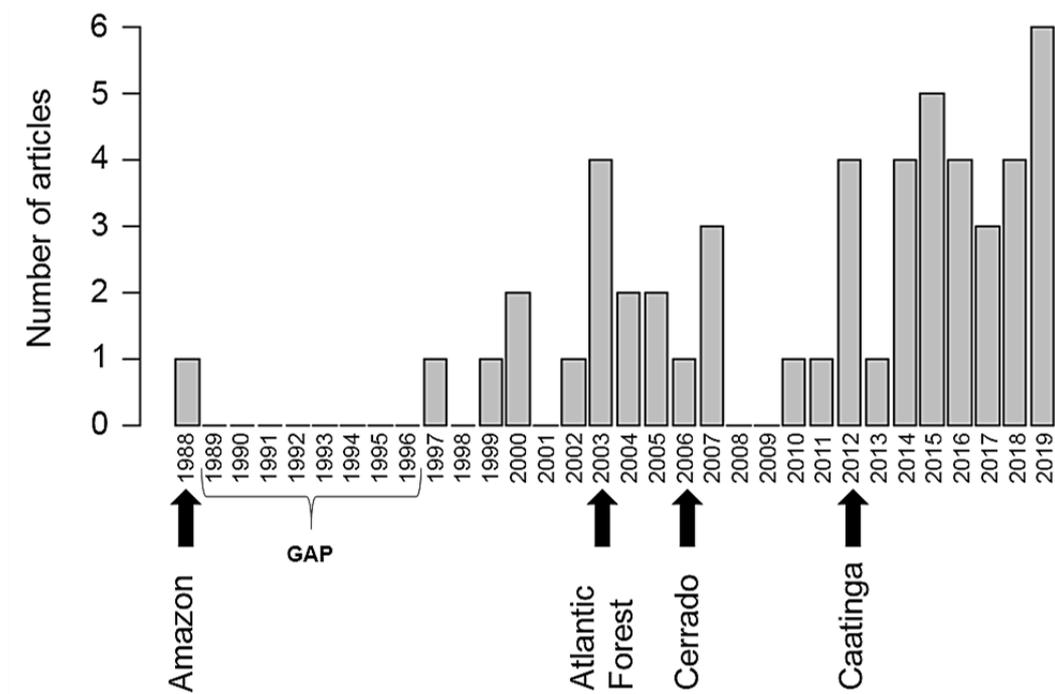


Figure 1. Publications per year evidencing a 10y gap of published studies and the year that each biome was first evaluated.

Over the last three decades (1988-2019), 31 journals published the 51 studies included in our database, and only 21% of them published more than one article (Table A.1; Figure A.2). The journal “Forest Ecology and Management” had the highest number of publications, almost 14% of the articles (n=7). Articles published by Brazilian journals represent 23% (Table A.1). We identified 73 institutions linked to the authors of the articles (Table A.1). These institutions are distributed in eight countries of three continents (America, Europe, and Oceania). Brazilian (57%), American (20%), and British (11%) institutions are the main contributors to the scientific knowledge accumulated.

1.4.2. RESTORATION SITE’S INFORMATION

The 51 articles found encompassed 582 “restoration sites”, that is, individual areas under restoration process. Restorations ranged from 1 to 70 years after its implementation, but only 35% of the articles studied restorations older than 30y. Sites are distributed among four (Atlantic Forest, Amazon, Caatinga, and Cerrado) of the six Brazilian biomes (Figure 2). Amazon is by far

the biome with more studied sites (68%; n=400), followed by the Atlantic Forest (20%; n=116) (Figure 2). Cerrado and Caatinga were the biomes with less studied restorations, comprising only 12% of the restoration sites evaluated for biomass (Figure 2).

PR sites recovered by natural regeneration comprise almost 68% of the studied sites and are distributed throughout the four biomes identified by the systematic review (Figure 2). Besides almost 70% of the studied restorations are in the Amazon range, only one study of AGB in ARs was found in this biome. Atlantic Forest is the biome where AR have been more studied, comprising >50% of the whole AR sites identified (Figure 2). Only 8% of the articles evaluated ARs and PRs of similar conditions and at the same time (Ferreira et al., 2015; César et al., 2018).

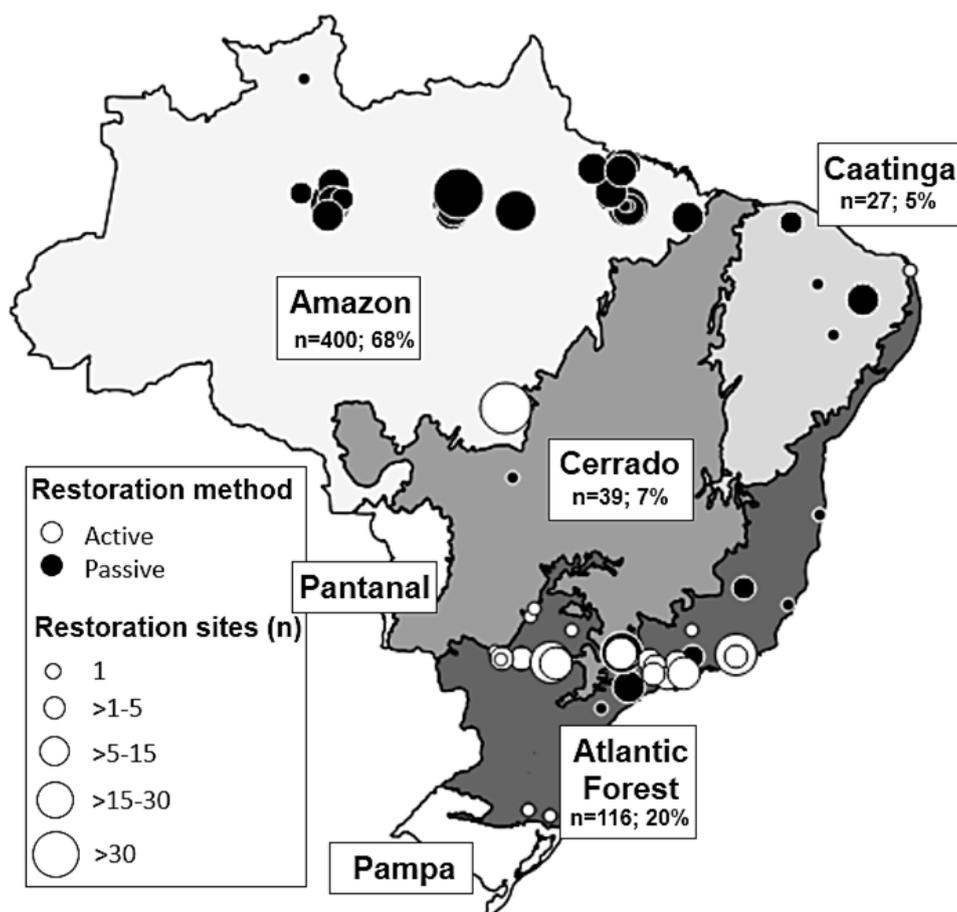


Figure 2. Distribution of the studied restoration sites found among Brazilian biomes and the restoration method implemented. The color of the circles represents the restoration method

evaluated, where white circles are the location of active restoration, and black circles are passive restorations. Restoration sites of the same study, located in the same biome and close to each other are grouped to improve sites' distribution visualization. The size of the circles represents the number of restoration sites.

We identified a variety of human land-uses practiced among sites before restorations implementation, ranging from clear-cut to a combination of agriculture, pasture, and burning (Table A.2). Agriculture and pasture were the most common past land-uses among restoration sites, handled in 57% of the articles. Almost 23% of the studies evaluated burned sites, and all of them were conducted in PRs, mainly in Amazon (75%). PRs derived from silviculture as past land-use activity were also found in Atlantic Forest (César et al., 2018; Almeida et al., 2019; Rosenfield and Müller, 2019a; Rosenfield and Müller, 2019b), and an actively restored site previously degraded by hydropower dam construction was also found in this biome (Silva et al., 2015).

1.4.3. METHODOLOGICAL CRITERIA ADOPTED BY THE STUDIES TO ESTIMATE AGB

To estimate AGB of plants aerial parts, studies used an averaged total area sampled per site of 2363.2 m². Among studies, the total area sampled per site ranged from 75 m² to 13000 m² (Table A.3). Some of them used additional sample plots to calculate the biomass of shrubs, herbs and small plants (<1 m height) by destructive methods. The criteria for inclusion of the plant individuals used in the articles were diameter and height. For plant diameter, the minimum size ranges from >0.2 to ≥ 10 cm, and for height from >1 to >2 m (Table A.3). Moreover, 37% of the articles that evaluated AGB in plant aerial parts included only individuals of DBH ≥ 5 cm.

Fifty-five allometric equations were used among studies to estimate AGB, and one to direct estimate carbon (Table A.4). Of these allometric equations, 45% were developed based on weighing mixed species and 54% of them derived from Brazilian forests, distributed in the Amazon (n=6), Atlantic Forest (n=3), Caatinga (n=4), and Cerrado (n=1) biomes (Table A.4).

Only 28% of the mixed species equations used among the studies derived from restored forests, all of them were developed in PRs (naturally regenerated) of Amazon and Atlantic Forest biomes. We observed that Amazon was the Brazilian biome with the largest set of equations, with 43% of the Brazilian mixed species equations found in the studies, and the two mixed species equations most used among the articles were derived from this biome (Uhl et al., 1988; Nelson et al., 1999). We found seven mixed species equations developed in PRs, and no equation derived from sites restored by AR methods (Table A.4).

We also found 30 equations specific for 16 species, four plant genus, different life-forms (lianas, palms, and shrubs), and the Araliaceae botanical family (Table A.4). However, only 41% of the studies included non-tree life-forms in the AGB estimates (e.g. shrubs, lianas, palms, bamboo, and herbaceous plants) (Table A.3). Almost all studies that accounted for different life-forms were conducted in PR sites, with only one study conducted in ARs. We also found an equation to estimate the leaf biomass in Caatinga plants (de Souza et al., 2012).

The allometric equations used to estimate AGB in the articles are based in plant's attributes, as trunk diameter, diameter plus height, basal area plus height, basal area, crown area or a combination of diameter, height and wood specific gravity (Table A.4). Among the mixed species equations used in the articles, the diameter of the felled and weighed plants ranged from >0 to 212 cm and the number of individuals from 14 to 4004, but only 32% of these equations presented plant individuals with diameter ≥ 100 cm, and 56% felled and weighed ≥ 100 plant individuals.

Just 31% of the articles that estimated AGB in plant individuals accounted for wood specific gravity (Table A.3), as a constant factor (Lindner and Sattler, 2012; da Silva et al., 2014; Galvão et al., 2015; Silva et al., 2016; da Silva et al., 2017), or averaged values for specific species, genus and botanical family (Uhl et al., 1988; Steininger, 2000; de Souza et al., 2011; Imaña-Encinas et al., 2012; Robinson et al., 2015; Silva et al., 2015; Sansevero et al., 2017; César et al., 2018).

Carbon stored in plants biomass was evaluated only in 25% of the studies, whose assumed that 45%, 47%, or 50% of the AGB values correspond to carbon elements (Markewitz et al., 2004; de Melo and Durigan, 2006; de Souza et al., 2011; Berenguer et al., 2014; Sattler et al., 2014; Robinson et al., 2015; Moura et al., 2016). Destructive methods and laboratory procedures to calculate carbon concentration in different species were also observed (Feldpausch et al., 2004; Fearnside et al., 2007; Silva et al., 2015; Pereira et al., 2016).

Biomass stored in litter compartment has also being evaluated at different forest types, conditions and restoration methods. We found that litter biomass was evaluated in 23% of the articles (Table A.3), 75% of them conducted in PRs (Gama-Rodrigues et al., 2003; Markewitz et al., 2004; Gama-Rodrigues et al., 2007; Vendrami et al., 2012; Berenguer et al., 2014; Moura et al., 2016; Pereira et al., 2016; Peixoto et al., 2017; de Azevedo et al., 2018; Froufe et al., 2019; Rosenfield and Müller, 2019ab). Litter stocks were most evaluated in Atlantic Forest, at both PRs and ARs.

1.4.4. FACTORS OF INFLUENCE ON AGB RECOVERY

We observed that 70% of the articles related AGB to one, or more, biotic and abiotic factors. Forest attributes were the factors of influence most related to AGB among the studies, including forest age, vegetation type, plant community condition and composition (Table 2). Restoration age, that is, the time elapsed since restoration implementation, was the factor most evaluated among restoration studies, and our temporal analysis of AGB revealed that sites restored by different methods present divergent patterns of AGB accumulation in the middle/long term (Figure 3).

Table 2. Factors of influence and parameters related to AGB recovery found in the articles.

Factors of influence on AGB recovery	Parameters	% articles
Landscape	edge effect; amount of adjacent forest; slope, and elevation	10% (n=5)

Forest attributes	age; forest type; successional phase; diversity (richness, Simpson, and Shannon); plant density; and life-form	45% (n=23)
Restoration implementation	restoration method (passive, active); and nutrient addition	8% (n=4)
Past land-use	type and time of land-use and site degradation level	16% (n=8)
Soil	soil type and soil properties	16% (n=8)
Plant interaction	herbivory	2% (n=1)
Climate	Annual precipitation	2% (n=1)

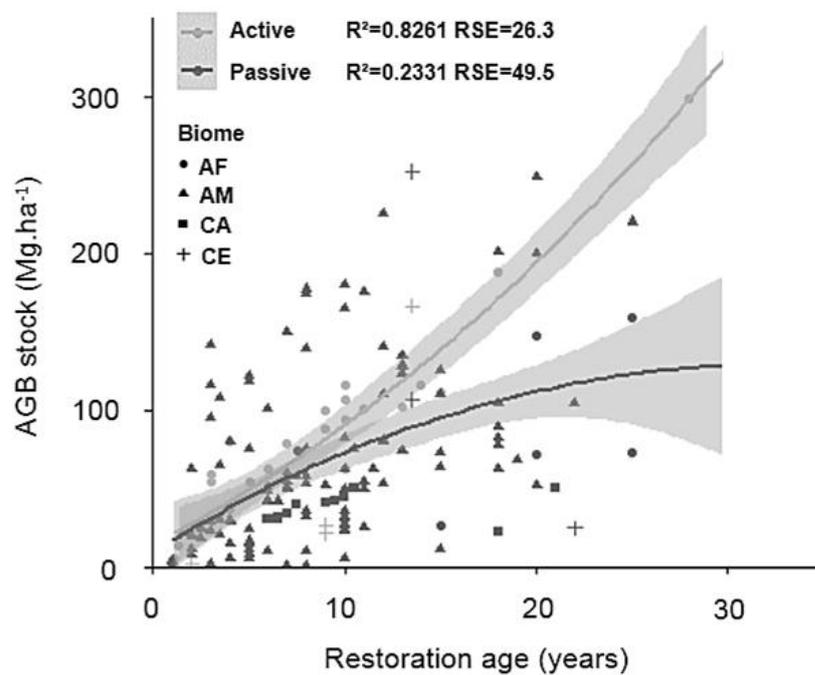


Figure 3. Predictive polynomial models fitted to estimate AGB stock (MgAGB.ha⁻¹) in active (adjusted R² = 0.82; p<0.05; Residual Standard Error = 26.3 MgAGB.ha⁻¹; n=37; AGB = -19.35058 + 4.00302 × age + 0.21641 × age²) and passive (adjusted R² = 0.23; p<0.05; Residual Standard Error - RSE = 49.5 MgAGB.ha⁻¹; n=140; AGB = 10.0731 + 7.3956 × age - 0.1158 × age²) restorations as a response of age. The dark shaded line is the trend of AGB stock recovery over time observed in passive restorations among biomes, and the lightly shaded line is the trend for active restorations. The shaded areas along the predictive lines represent the confidence

interval (95%) for the model. Data points were categorized by biome: AF (circle) – Atlantic Forest; AM (triangle) – Amazon; CA (square) – Caatinga; and CE (cross) – Cerrado.

1.5. DISCUSSION

1.5.1. DISTRIBUTION OF AGB RECOVERY STUDIES AMONG BRAZILIAN BIOMES AND RESTORATION METHODS: THE IMPACTS TO NATIONAL-SCALE GENERALIZATIONS

International commitments for the recovery of degraded lands, the rise of carbon credit markets, the pressure to reduce CO₂ emissions by forest conservation and restoration has been on the government's agendas for more than 40 years, since the First United Nations Conference on the Human Environment in 1972, and the First World Climate Conference in 1979. Along this period, political marks as the United Framework Convention on Climate Change (1992), the Kyoto Protocol (1997) and the Intergovernmental Panel on Climate Change (IPCC) lead global governments to prioritize climate change mitigation and to preserve and recovery natural ecosystems. For this reason, we were expecting to observe an earlier increase in the number of studies related to AGB recovery by restored forests, improving the knowledge and supporting political decisions over the years, instead of only in the last decade.

In 2008-2009 the National Plan on Climate Change and the National Politic on Climate Change were implemented in Brazil (Brazil, 2008; 2009). This can explain a late and slight increase in the number of publications/year, reaching the peak in 2019 (Figure 1). Although ecological restoration in Brazil present a solid and growing scientific production, with almost 300 articles published over the last 30 years (Guerra et al., 2020), specific evaluations of AGB/carbon recovery in these systems does not keep up with the increasing political decisions and appealing discussions related to forest restoration and climate change. Furthermore, because the Brazilian post-graduation network present solely almost 300 programs directly related to biodiversity and environmental sciences researches (Souza and Fernandes, 2013) we expected to find a more diversity of institutions improving scientific knowledge related to AGB recovery.

We argue that the continuous improvement of the general knowledge could be attained to the broad range geographic distribution of scientific studies, allowing to predict AGB recovery patterns of different forests restored by different methods. However, we observed an unbalance in the restoration methods evaluated among biomes, where outcomes of AGB recovery from restored forests lack for some parts of Brazil and only 23% of the studies evaluated AGB recovery in ARs.

Guerra et al. (2020) observed that AR (seedling planting) is the most used restoration method among Brazilian biomes, but we found an expressive predominance of PR sites among the studies of AGB/carbon (Figure 2). In addition, studies have shown that 78% of the land allocated for restoration projects in the Atlantic Forest, Atlantic Forest/Cerrado ecotone and Amazon was designed for AR (Brancalion et al., 2016). Together, these arguments evidence that scientific knowledge related to AGB recovery is not aligned with the restoration methods most needed and practiced in Brazil.

The fact that we found only one study that evaluated AGB stock in ARs implemented by direct seeding, an efficient restoration method cheapest than planting seedlings and able to trigger initial successional processes (Freitas et al., 2019), evidence another bottleneck in the field. However, this result can be a consequence of the focus of our study - forests, seen that direct seeding is a restoration method used in most of the Brazilian biomes, but mainly implemented in non-forest ecosystems.

Due to agricultural expansion, Brazil is one of the major contributors to deforestation and to land-use change emissions (Zarin et al., 2016) and the trend is that Brazilian biomes will continue to be deforested (Soares-Filho et al., 2014; Rochedo et al., 2018), increasing the need for restoration efforts. In this sense, our results support the need for scientific evaluations of AGB accumulation in regions with weak information about the recovery of this ecological process, beyond Amazon and Atlantic Forest biomes. This is crucial to the effective achievement of the restoration goals, based on realistic site conditions. Better understanding temporal changes in the

process of AGB/carbon accumulation among restored sites, could leverage the improvement of AGB/carbon monitoring protocols, allowing a confident interpretation about the large-scale potential of forest restorations in mitigating climate change.

1.5.2. METHODOLOGICAL DIVERGENCIES IN AGB AND CARBON STOCKS ESTIMATES IN BRAZILIAN RESTORED FORESTS

The rigorous selection of the appropriate allometric equation is fundamental to a reliable AGB estimate and studies have shown that the main source of error is attained to the choice of the allometric model, once not representative equations of a targeted forest can lead to substantial errors in AGB estimates. Even in areas with similar conditions, as climate, soil, forest type, restoration method, and age, major errors can be introduced (Chave et al., 2004; Lindner and Sattler, 2012; Sileshi, 2014; Silva et al., 2015). In tropical forests from Central America, Chave et al. (2004) found that different allometric models can estimate AGB with a variation of 246 MgAGB.ha⁻¹, even when applied to the same plot. In Amazonian PR sites, using eight different models to estimate AGB stock, Wandelli and Fearnside (2015) observed that the mean error of estimates for accumulated biomass varied from 5.6% to 57.5%. These studies highlight the need for a representative set of allometric equations of the many tropical forest formations.

Thus, it is to be assumed that these issues were taken into account by AGB recovery studies, but our results show major divergences. Studies of AGB recovery in restored forests of Brazil have used equations derived from Brazilian natural forests and also from tropical forests of other countries to estimate AGB (Table A.4), forests which may present evident differences from the targeted one. The few equations derived from restored forests (28%) are not representative of the many forest formations and restoration methods used in Brazil, and we did not find in our database any equation derived from AR sites.

The similarity between the ecosystem that the equation derived and where it will be applied is important, but the equation inputs must also be considered. The allometric equations used in the studies are based on plant's trunk diameter and height, but basal area, crown area or a

combination of diameter, height and wood specific gravity are also used to estimate AGB (Saldarriaga et al., 1988; Deans et al., 1996; Chave et al., 2005; Chave et al., 2014; Robinson et al., 2015). Chave et al. (2005) observed that the inclusion of height in the equations reduced the error of individual tree biomass estimation from 16% to 6%. In contrast, Hunter et al. (2013) argue that height measurements in tropical forests are labor and present potentially large errors, related to forest conditions (e.g. dense understory vegetation, tall canopies, and closed-canopies), observer experience, and the equipment used.

The performance of allometric models is also highly associated with the diameter range and the number of weighted plants included in their database, and equations developed from a small number of individuals and small diameter range should be avoided (Chave et al., 2004). Among studies, we found seven equations encompassing ≥ 100 weighed plants and individuals with large diameters (≥ 100 cm), but only one of these equations was derived from a Brazilian forest, an Amazonian natural forest (dos Santos, 1996; Ferreira and Prance, 1999). Despite some equations enable the inclusion of small-sized plants, 37% of the studies included in their AGB estimations only plant individuals ≥ 5 cm diameter.

Of course, individuals within this range represent the most share of AGB in mature tropical forests, but in young restorations (2-4y regeneration) 100% of the AGB can be stored in the 1-5 cm DBH class, decreasing to less than 10% in older restorations (12-14y) (Feldpausch et al., 2005). Thus, biomass evaluations in restored forests should be aware of the contribution of these small-sized individuals to AGB recovery and, depending on restoration method and the successional stage, neglecting small plants can cause misleading interpretations of the AGB stored in restored forests.

This issue can be extended to the limitation of using wood specific gravity, due to the scarce availability of experts able to identify species in field campaigns. In spite of that, besides wood specific gravity be also recognized as an important specie's attribute to reliably estimating AGB

stock on forest stands (Brown et al., 1989; Chave et al., 2006; Chave et al., 2014), the majority of the studies in Brazilian restored forests have neglected this attribute.

Another highlight we found is that the number of species used to develop the Brazilian's mixed species allometric equations is not even close to the plant diversity of these tropical forests. The importance of including more species and life-forms in the allometric models relies on the processes of weighing plant's components (trunk, canopy, leaves), while species store different amount of biomass among their components. The concern of including more species in the allometric equations also involves accounting for different life-forms. Besides trees share the highest proportion of AGB stocks in tropical forests (Ligot et al., 2018; Meakem et al., 2018), other life-forms can play important roles in ecological processes and functions, as forest productivity (Gerwing and Farias, 2000; Schnitzer and Bongers, 2002; Gehring et al., 2005; Alves et al., 2012). For example, in Amazon, shrubs can correspond to 74% of the total AGB stored in 2-4y regenerated pastures (Feldpausch et al., 2005), and among different forests in the Atlantic Forest biome, lianas, palms, tree ferns, bamboo, and epiphytes can share more than 10% of the AGB (Vieira et al., 2008).

Despite the share of different life-forms on AGB stocks, only four equations and less than half of the studies included non-tree life-forms in their database. Moreover, no AR site was evaluated within this concern. The contribution of different species and life-forms to AGB accumulation in tropical forests cannot be neglect, and this issue is accentuated by the urgent need to include more diversity of species and life-forms in AR projects (Rodrigues et al., 2009).

Studies are selecting equations not well fitted with their objectives, or even their forest types or stage, suggesting the lack of multiple equations adjusted for restored forests and at different conditions, which can lead to unreliable AGB estimates. In a first effort to overlap this gap, wide and robust allometric models must be available for generalizations in each biome. Whenever possible, these general models should encompass data from the different vegetation types of each biome, as the Atlantic Forest formations recognized as Ombrophilous (Dense,

Mixed, and Open) and Seasonal (Semideciduous and Deciduous) forests, or even the forested Savannas formations (Cerradão). Other studies could regard the inclusion of allometric equations fitted for vegetation associated with water courses, and also mangroves located at costal zones (Ferreira et al., 2015; Ferreira et al., 2019), since this proximity directly influence the vegetation patterns (structure and composition) at different biomes (Coutinho, 2016; IBGE, 2012).

Because evaluations of AGB stocks in forests are based on the extrapolation of data collected at plot scale, plot size and plot position are also important issues to ensure robust AGB estimates. Chave et al. (2004) proposed that plots should size 2500 m² to estimate AGB with good confidence, representing the spatial variability of forests AGB. In general, the average plot size found is smaller than that for studies conducted in Brazil, and this pattern seems to be widespread among tropical restoration sites - 947 m², average size for ~1468 plots of Neotropical secondary forests (Poorter et al., 2016).

The conversion of AGB on carbon stocks also need to be highlighted. We observed that carbon concentration in dry AGB, used as an carbon conversion factor among studies, varied from 45% to 50%. However, it is known that different forests, with different species composition, can present disparities in average values of carbon (IPCC, 2006). Obviously, generalizations are important, especially to conduct large-scale studies, but these methodological aspects might be rigorously thought, once it can result in not confident estimates of carbon stocks and CO₂ sequestration potential by restorations.

It is known that technical and legal issues can bar building allometric equations in restored forests and, for a while, authors are restricted to the few equations indicated for AGB estimations in these areas. For example, costs related to destructive methods of AGB estimation can reach US\$ 11.00 per plant individual (Silva, 2007), affecting the viability of fitting new equations based on large datasets. Nevertheless, some advances for estimating AGB stocks in AR sites has risen in the Atlantic Forest biome (Miranda et al., 2011; Nogueira Jr et al., 2014; Ferez et al., 2015; Ré et al., 2015; Zanini, 2019). We state that our goal was not to evaluate the whole set of equations

available for Brazilian forests, but those that have been most used among studies to estimate AGB in restoration sites.

1.5.3. THE MAIN DRIVERS OF AGB RECOVERY STUDIED AMONG BRAZILIAN RESTORED SITES

Forest attributes (45%), past land-use (16%), soil properties (16%), and landscape metrics (10%), were the most factors related to biomass, comprised in 65% of the studies. Among the forest attributes, time elapsed since restoration starts (e.g. age) was related to AGB stocks in 29% of the studies conducted in Brazil. Restoration age is important because enough time is needed to forests succession and ecological processes, as AGB accumulation, to reestablish (Crouzeilles et al., 2016). Besides that, age is more correlated to AGB stocks in ARs ($R^2=0.82$) than PRs ($R^2=0.22$) (Figure 3), suggesting that in PR other factors present high influences on the AGB stock.

Plant community composition has a relevant importance on AGB stocks in restoration, since fast-growing species present a higher contribution during the early years (<37 years), but the contribution of slow-growing is significant at later stages of succession (Shimamoto et al., 2014). This suggests that AGB accumulation patterns is not constant over time, and is related to the changes in the plant composition along the restorations successional trajectory.

In this context, forest type and successional stage are also related to AGB stocks (Lu et al., 2003; Santos et al., 2003). Costa et al. (2014) observed that Caatinga PR sites of dense forests and at advanced successional stages accumulate more than 2-fold the AGB stock of open forests at early regeneration stages (45.8 and 20.7 Mg ha⁻¹, respectively). Similar results were also observed for different vegetation types at different successional stages in Cerrado biome, where sites of Cerradão (the Brazilian Forested Savanna) presented more AGB stocks than grasslands and typical Savannas (de Miranda et al., 2014; Roquette, 2018). Because Brazilian biomes comprise a mosaic of vegetation types, these results imply in a huge challenge to understand how AGB is recovered by restoration actions in a full range of situations.

The influence of plant diversity (species richness, Shannon and Simpson index) on AGB stocks was accessed in 9% of the studies, conducted in Amazon, Atlantic Forest and Caatinga biomes, pointing to a positive relationship with AGB recovery (Vieira et al., 2003; de Aguiar et al., 2013; Robinson et al., 2015; Rosenfield et al., 2019b). The positive relationship between plant community diversity and biomass productivity occurs due to the coexistence of species that ensure a greater complementarity and facilitation in forested ecosystems, and consequently a better efficiency in the use of resources (Wright, 2002; Nakamura, 2008; de Aguiar et al., 2013). For example, de Aguiar et al (2013) estimated that enhancing species richness from 1 to 10 species, PR sites of Caatinga (6-9y) annual AGB increment can increase $5.1 \text{ Mg}\cdot\text{ha}^{-1}\cdot\text{y}^{-1}$. These results have a direct effect on restoration strategies, providing the subsidy for restoration ecology, which is the basis for any ecological restoration.

The AGB recovery rate also depends on the site's conditions, as past land-use, adjacent forest cover, climatic water deficit, and other social and political drivers (Chazdon et al., 2016; Crouzeilles et al., 2016; Chazdon et al., 2017). Studies have found that, in restoration sites at flat terrains and embedded in landscapes with high forest coverage, the AGB recovery is more efficient for both PR and AR methods (Sattler et al., 2014; Robinson et al., 2015; Toledo et al., 2018). Sattler et al. (2014) found evidence that AR sites located on sloped terrains stock less than half the carbon ($17.5 \text{ Mg}\cdot\text{ha}^{-1}$) stored by similar restorations on flat terrains ($37.6 \text{ Mg}\cdot\text{ha}^{-1}$) in Atlantic Forest biome.

Throughout the landscapes, the amount and distribution of surrounding forests are the main regulators of seed dispersal with a strong effect in early-successional trajectories, while flat terrains are associated with high water retention, being both relevant drivers of AGB accumulation (Osman and Barakbah, 2011; Uriarte et al., 2011; Holl and Zahawi 2014; Sattler et al., 2014; Robinson et al., 2015). In 30y Atlantic Forest PRs, Robinson et al. (2015) observed that the amount of adjacent forest and slope are predictors with the large effects on biomass stock, positively and negatively correlated to AGB, respectively.

Edge effects were also studied, and sample plots near to the forest edge presented low AGB stocks (Berenguer et al., 2014), result that can be explained by the decline observed in AGB within 100 m from forest fragments edges, where tree mortality is increased by microclimatic changes and wind disturbances (Laurance et al., 1997). Regarding these evidences and the fact that a significant part of the environmental debt encompasses legally protected forests at riparian and sloped areas (Federal Law 12.651; Soares-Filho et al., 2014), which can be limited to strips of 5 m width and steep slopes, restorations success and goals can be hampered.

Land-use type, degradation level, time since last disturbance and under land-use, were observed to have significant effects on AGB. Time since last fire event presented positive effects on AGB recovery, while grazing, time under land-use, and land-use intensities negatively affected AGB recovery in Amazonian PR sites (Uhl et al., 1988; Feldpausch et al., 2007; Gehring et al., 2005; Berenguer et al., 2014). In Amazon, PR sites (<15y) previously used for agriculture presented faster AGB accumulation than PR sites regenerated from pasture lands (Wandelli and Fearnside 2015). However, 7-20y PRs regenerated from pastures in Atlantic Forest can store 43% more AGB of native species than regenerated sites of similar age derived from *Eucalyptus* sp. plantations (César et al., 2018). The number of burns also negatively affect AGB (Wandelli and Fearnside 2015; Sansevero et al., 2017).

Soil parameters presented important effects on AGB stocks among Brazilian restored forests. In PRs derived from pastures in Atlantic Forest, soil sand and clay content presented negative and positive effects on AGB recovery, respectively (Robinson et al., 2015; Toledo et al., 2018). These properties are related to soil water retention capacity, an important driver for plant growth and AGB recovery (Lu et al., 2002). Higher AGB stocks were observed in Atlantic Forest ARs at fertile and clayish soils than the same restorations at sandy and poor soils (de Melo and Durigan, 2006). Moran et al. (2000) also found significant effects of soil type on AGB stocks in passively restored forests in Amazon. Besides soil fertilization play an important role on any plant growth, undesired outcomes as benefiting aggressive and invasive species may negatively affect

AGB accumulation, but control interventions present positive effects in these cases (Siddique et al., 2010, Ferez et al., 2015)

The litter compartment and its contribution to total biomass stocks in restoration sites also depends on factors as restoration age, structure and composition. In Amazon (19y secondary forest), Atlantic Forest (40y secondary forest), and Cerrado (22y secondary forest) biomass stocks in litterfall was estimated to reach 67 Mg.ha⁻¹, 7.3 Mg.ha⁻¹, and 3.9 Mg.ha⁻¹, respectively (Markewitz et al., 2004; Gama-Rodrigues et al., 2007; Peixoto et al., 2017). In Caatinga (30y regenerated forest), annual production of litterfall biomass can reach 6.1 MgAGB.ha⁻¹.y⁻¹ (Pereira et al., 2016). In secondary Atlantic Forests at different regeneration stages, annual production of litterfall biomass was higher in early stages of natural regeneration than in more advanced forests, and this is attained to the high proportion of pioneer species in young secondary forests, where species present higher leaf turnover than late successional species (Vendrami et al., 2012). Besides we found a few studies of litter stock, they are also high concentrated in PRs. Only three studies evaluated litter stock in ARs, exclusively in the Atlantic Forest biome (de Azevedo et al., 2018; Rosenfield and Müller, 2019a; Rosenfield and Müller, 2019b).

1.5.4. TEMPORAL ANALYSIS OF AGB RECOVERY IN BRAZILIAN FOREST RESTORATIONS

In recent years, studies have focused on to determine which one, PR or AR methods, is more successful in recovering AGB stocks, vegetation structure and biodiversity (Brancalion et al., 2016; Crouzeilles et al., 2017, Meli et al., 2017). We found only two articles that compared AGB of AR and PR sites at similar conditions (Ferreira et al., 2015; César et al., 2018). César et al. (2018) studying AGB recovery in Atlantic Forest, found that PR sites derived from pasture stock 45% (91.3 MgAGB.ha⁻¹) less AGB of native species than AR sites (132.2 MgAGB.ha⁻¹) from 7 to 20-year-old. In an Atlantic Forest mangrove, Ferreira et al. (2015) also found this pattern of AGB recovery in 5-year-old restored sites, but AGB stock in AR (60 Mg.ha⁻¹) was 3-fold the

AGB of PR sites (20 Mg.ha⁻¹). Our model estimated that PRs of 30y can stock 38% (127 Mg.ha⁻¹) of the AGB estimated for ARs of the same age (334 Mg.ha⁻¹) (Figure 3).

Other studies have found that AGB recovery in 30y PR sites can reach 98.5 MgAGB.ha⁻¹ in Amazon and 43.3 MgAGB.ha⁻¹ in Caatinga sites, while 25y PRs in Atlantic Forest can accumulate 158.8 MgAGB.ha⁻¹ (de Souza et al., 2011; Galvão et al., 2015; Pereira et al., 2016). ARs in Atlantic Forest and Cerrado can present AGB stock higher than PRs of the region and similar age (de Melo and Durigan, 2006; de Souza et al., 2011; César et al., 2018). While AGB stocks of AR sites of the Atlantic Forest can reach 222 MgAGB.ha⁻¹ at 25 years after restoration implementation (Durigan et al., 2016), 25 years-old PRs of the same biome can stock from 73 to 159 MgAGB.ha⁻¹ (de Souza et al., 2011).

The results corroborates the pattern of AGB accumulation between restoration methods, where ARs present better outcomes than PRs. In general, it can be observed that the processes of AGB accumulation in the first years of restoration are similar at both active and passive restorations (Figure 3), but after 5-10 year AGB accumulation restoration methods start to present divergences. However, to compare PR and AR outcomes depends on how restorations data sets are categorized as passive and active methods, and controlling for key factors as past disturbance, landscape metrics, climatic patterns, and restoration age is fundamental (Crouzeilles et al., 2017; Reid et al., 2018). Nevertheless, comparative studies of different restoration methods are important to know what to expect from each strategy, but the selection of the restoration method for a particular site must be based on its natural resilience, past land-use history, and the surrounding landscape matrix (Holl and Aide, 2011).

It is important to highlight that remote sensing (e.g. LiDAR and optical sensors) are useful tools that have been utilized among Brazilian biomes to the monitoring of AGB recovery, especially due to the possibility of long-term temporal analysis (Luckman et al., 1997; Santos et al., 2003; Vieira et al., 2003; da Silva et al., 2014; Galvão et al., 2015; Almeida et al., 2019). Also,

although allowing quick data collection of forest structural attributes, remote methods are supported by field measures, needed to the validation of the estimates.

1.6. ADVANCING THE FIELD AND IMPLICATIONS FOR FUTURE STUDIES

We observed main researches bottlenecks related to studies distribution and methodological issues, especially the lack of studies across the whole Brazilian territory and equations fitted for AGB estimation in restored sites of different forest types. The real contribution of forest restorations to climate change mitigation still weakly addressed for some Brazilian regions, either by the absence of AGB studies from different biomes and restoration methods, or by the weakness of the methodology used to estimate AGB and carbon stocks.

Regarding the ecological role of biomass as a driver for restoration success, studies of AGB should be conducted at restored forests in a wide range of situations (restoration method, landscape forest coverage, soil and forest formation). AGB researches must be intensified in restored forests, beyond Amazon, and at both passive and active restorations. A robust set of allometric equations representative of Brazilian restored forests, and a database on species wood specific gravity and carbon concentration, available for scientists and decision-maker usage is urgently need for Brazilian ecosystems. In addition, we defend that depending on the restoration method, environmental conditions and the successional phase of the recovering sites, different life-forms should be included, leading to an accurate estimate of AGB recovery. Yet, studies are limited to the methodological alternatives that account for this issue, and efforts must be applied to overcome this barrier.

An effective path to overcome this barrier would be the development of Federal or State standardized monitoring protocols for forest restoration, where aboveground biomass estimates would be attached in their routines. A minimum set of allometric equations should be available for different forest formations, at least at biome level, regarding the particular ecological aspects of each ecosystem, as plants' architecture, size, wood specific gravity and carbon concentration. Together, these approaches could result in a qualified database, helpful to develop robust

scenarios able to predict the real contribution of different restoration methods to AGB recovery, carbon sequestration and climate change mitigation in Brazilian lands.

Because Brazilian biomes present different demands for restoration methods, but the need for both passive and active restorations exists in every biome comprised in this study, restoring these lands performs an unprecedented opportunity to address the knowledge gaps highlighted here. Finally, forest restoration programs are global tools to mitigate biodiversity loss and the climate change effects, and, for this reason, the outcomes of these approaches must be deeply understood, and science must continually move the knowledge forward, otherwise, already set and future efforts can fail in achieving the expected targets.

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Chapter 2.

Forest restoration plantings and carbon stocks in Brazilian agricultural landscapes: the role of forest's age and structure

Fernando Ravanini Gardon^a; Rozely Ferreira dos Santos^a; Ricardo Ribeiro Rodrigues^b

^a Department of Ecology, Institute of Biosciences, University of São Paulo, São Paulo, Brazil

^b Laboratory of Forest Ecology and Restoration, Department of Biological Sciences, University of São Paulo, Escola Superior de Agricultura “Luiz de Queiroz”, Piracicaba, São Paulo, Brazil

2.1. ABSTRACT

Large-scale forest restoration programs represent an unprecedented opportunity to expand the forest cover of the tropical landscapes and ensure the provision of vital ecosystem services for the human population, such as climate change mitigation. However, in the scope of active restoration, particularly the methods based on planting native seedlings, this knowledge is still scarce. In this context, the objective of this study was to evaluate the trajectory of carbon accumulation in forests of different ages implanted by active restoration, as well as the influence of the forest structure in this ecological process over time. By using allometric equations, we estimated the biomass of 16 restoration plantings aged from 5 to 30 years old, and related the carbon results to the age and the structural attributes of the communities. We found that restoration plantings can be effective carbon sinks and after 30 years these communities can present a carbon stock similar to local native forests. The results show that large plants directly influence the carbon stock of these planted communities, while total plant's density presents secondary effects. The carbon stock and density of small plants are also significantly related to the total stock. We conclude that, when ecologically planned active forest restorations are efficient in recovering important ecological processes related to carbon sequestration and its accumulation in forest biomass. In this sense, despite active forest restoration is a method applied to the recovery of degraded lands of low resilience, its significant contribution to climate change mitigation is evidenced.

Keywords: Ecological Restoration; Structural Attributes; Chronosequence; Ecosystem Services; Tropical Forests

2.2. INTRODUCTION

Historically, tropical ecosystems were intensely degraded or replaced by anthropic uses (Chazdon, 2014) and these changes led to a scenario where just protecting the remaining ecosystems is no longer sufficient to safeguard biodiversity and the ecosystem services (ES) (McDonald et al., 2016). Thus, to mitigate the effects of deforestation it is necessary to link environmental conservation policies to ecological restoration programs. The ecological restoration of degraded tropical lands has been encouraged as a fundamental complementary strategy for conservation, able to promote the integrity of the landscapes and the recovery of the expected ES (Wu, 2013; Chazdon, 2014; Kueffer and Bunbury, 2014; Suding et al., 2015).

Among these ES, restored forests are widely recognized as carbon sinks (Poorter et al., 2016; Lewis et al., 2019; Gardon et al., 2020), providing support for climate change mitigation through the sequestration of atmospheric carbon and its storage in plant biomass (Bullock et al., 2011; Houghton et al., 2015; Crouzeilles et al., 2016; Suarez et al., 2019). In this perspective, forest restoration has become an important tool for intergovernmental programs aiming reduction of the atmospheric carbon concentrations and, for this reason, restored forests represent a focal point of discussion in the main global committees, conferences, and agreements related to global climate change, such as COP 21, United Nations Conference on Climate Change, Bonn Challenge, and 20x20 Initiative. Thus, robust evaluations of carbon in restored forests are fundamental elements for reliable greenhouse gas inventories, effective forest management, and solid carbon markets and policy.

If, on the one hand, the accumulation of carbon in plant's biomass makes it possible to interpret the potential of forest restorations in mitigating the effects of climate change, on the other, this ecological process represents one of the main drivers of the successional trajectory of recovering forests (Lohbeck et al., 2015), being relevant to both the plant community itself and human society. Although studies have shown the high potential of tropical forests restored by natural regeneration in sequestering carbon, this knowledge is less consolidated in communities restored by active restoration methods, such as the planting of seedlings and/or seeds (Chazdon

et al., 2016; Poorter et al., 2016; Gardon et al., 2020). This is a problem because, depending on the remaining resilience of a degraded site, different restoration methods are more suitable, and the application of the appropriate method ensures the total recovery of the forest ecosystem (Holl and Aide, 2011). Thus, stimulating assessments of active restorations at different ages would help to quickly determine the patterns of carbon accumulation in these man-made systems and predict the consequences over time. Despite the limitations, the chronosequence approach has been a strategy widely applied to evaluate the temporal dynamics of recovering forests (Chazdon et al., 2007; Feldpausch et al., 2007; Letcher and Chazdon, 2009; Suganuma and Durigan, 2015).

The reliability of the chronosequence approach in the evaluation of the trajectory of active restorations lies in the fact that, planting seedlings, is a technique indicated to recover forest cover and ecological functions and processes at low resilience sites (Holl and Aide, 2011; Brancalion et al., 2015; Meli et al., 2017). Thus, the low forest cover of the landscapes where these restorations are commonly implemented may not sustain the arrival of new propagating materials, which could balance the seed bank impoverished by soil degradation and influence the community assemblage (Holl and Aide, 2011; Reid et al., 2015). This condition restricts the factors influencing the restoration trajectory to the restoration method (seedling spacing, fertilization, species selection, among others), climatic variation, and stochastic events (drought, fire, among others) (Holl and Aide, 2011; Toledo et al., 2018), the latter being avoided or controlled along with the restoration progress (Rodrigues et al., 2009; Brancalion et al., 2015). Another feature that contributes to the use of chronosequence in restoration plantings assessments is the fact that species composition planted in these systems depends on their availability in the local and regional nurseries. In this context, in the state of São Paulo - Brazil, for example, 155 nurseries produce a total of 561 species, but about 35 species account for approximately 50% of the total seedlings produced (Vidal et al., 2020). This is evidence that this set of species is commonly introduced in the restoration plantings of the São Paulo state, which ensures a higher control in the chronosequence.

Considering that the implantation of new forests in intensely degraded landscapes depends on the active restoration, methods that lack scientific studies and require high financial support (Benini and Adeodato, 2017; Brancalion et al., 2019), understanding the temporal trajectory of carbon is fundamental to ensure the ecological outcomes expected by these systems, contributing to the global targets related to climate change mitigation. In this context, our objective was to evaluate the contribution of forest restoration plantings to carbon ecosystem services based on a chronosequence of sites ranging from 5 to 30 years old, while investigating the relationships between the trajectory of the carbon stock and the forest structure.

2.3. MATERIAL AND METHODS

2.3.1. STUDY AREA

The chronosequence used in our study is composed of 16 restorations implanted in old and degraded agricultural areas, whose restoration method was active forest restoration, based on the same technique - planting seedlings with a high diversity of species. The factors of influence over the restorations trajectories, such as the restoration method, the land-use history, the composition of the adjacent matrix, and climate aspects, were controlled to satisfy the assumptions of the chronosequence approach.

To site selection, we initially evaluated the active restorations identified within the São Paulo State - Brazil, regarding the planting date, information obtained in the literature (Garcia, 2012; Duarte, 2013; Silva 2013; Suganuma, 2013; Vasconcellos et al., 2013; Pardi 2014; Suganuma & Durigan, 2015; Domene, 2018), from historical satellite images from the National Institute for Space Research - INPE (www.dgi.inpe.br) and Google Earth Pro, and confirmed by the landowners. To determine the exact age of plantings, we consider T0 the starting year of the seedlings planting and T1 the sampling year. Subsequently, we obtained the land-use history and the composition of the matrix adjacent to the restoration sites by using the annual maps of land-use/cover of the Brazilian territory since 1985, made available on a 30m scale on the public

platform MapBiomas (www.mapbiomas.org), which were reviewed by comparison with satellite images.

The selected restorations are distributed along with watercourses and small water reservoirs, that is, in riparian areas protected by law (Brazilian Forest Code - Law 12.651/2012). Such restorations are located in six municipalities in the State of São Paulo - Brazil (Figure 1, Table B.1). The predominant plant formation in the study region is the Seasonal Semideciduous Forest (SSF) of the Atlantic Forest biome, but contact zones with Cerradão, a forest formation of the Cerrado biome, are also observed (Coutinho, 2016; IBGE, 2019), where 50% of the studied restoration are in these regions.

The landscapes adjacent to the restoration sites present low coverage of native vegetation - 12% on average (Table B.1), being dominated by mechanized sugar cane cultivation (Coutinho, 2016; IBGE 2019; MapBiomas; Google Earth Pro). Regarding the restoration method, among the restorations, the average density of planted seedlings and the diversity of species used per hectare were 1660 plant individuals and 74 species (Garcia, 2012; Duarte, 2013; Silva, 2013; Sugauma, 2013; Vasconcellos et al., 2013; Pardi, 2014; Sugauma and Durigan, 2015; Domene, 2018).

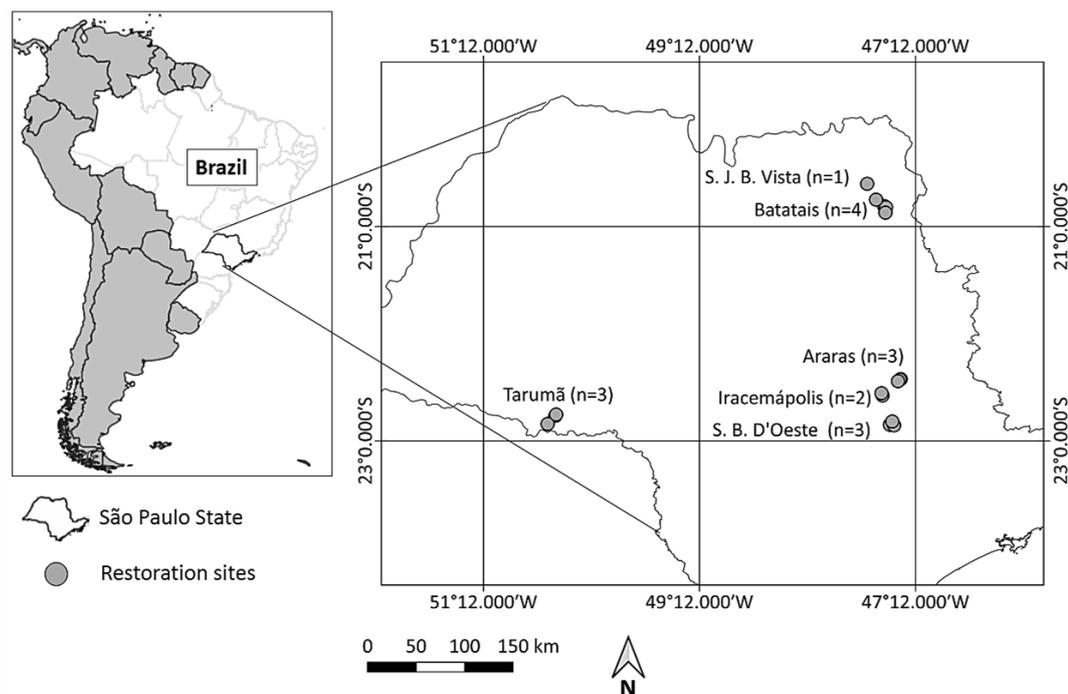


Figure 1. Geographic location of the study areas in six municipalities of the state of São Paulo - Brazil.

2.3.2. DATA SAMPLING

Between the years 2018 and 2019, we installed 10 transects of 100 m² (4 x 25 m) equidistantly along the length and obliquely to the drainage system of each plantation (Figure B.1). We fixed the center of the transects in the medial portion of the width of the restorations. The distances between transects varied from 30 m to 312 m. This strategy was adopted due to the dimensional characteristics of the restoration sites, which generally comprise 1:22 ratio polygons (1300 m length and 60 m width, on average).

In total, we installed 160 transects, comprising a sampling area of 1.6 ha of restoration plantings (1000 m² per site). In these transects, we identified at the most possible detailed botanical level all individuals of woody species (trees, palms, and lianas) with the diameter at breast height (DBH - measured at 1.30 m in height from the ground) higher than 3 cm. Species that could not be identified in the field were collected and herborized for later identification by comparison with materials deposited in the E.S.A. herbarium (Thiers, 2019, continuously updated). We used the Flora do Brasil 2020 (<http://floradobrasil.jbrj.gov.br/>) database and the INCT - Virtual Herbarium of Flora and Fungi (<http://inct.splink.org.br/>) as support for species identification. From the “*flora*” package (Carvalho, 2017) available for the R program (R Core Team, 2018), the scientific names of the species were corrected and synonyms standardized according to the Flora do Brasil 2020. The DBH of each individual was obtained from the measurement of the trunk perimeter by a measuring tape and its subsequent conversion to diameter (cm).

2.3.3. BIOMASS AND CARBON EVALUATIONS

We estimate the aboveground biomass (AGB) of each plant individual by allometric equations based on DBH, height and wood density (g.cm⁻³). The list of identified species was inserted in the “*getWoodDensity*” function of the “*BIOMASS*” package (Réjou-Méchain et al.,

2017) to obtain the wood density values. When this data was not found for a given species, we used the average density of the genus, if there was no data for the genus, then the average of the botanical family was used. We also adopt this criterium when an individual has been identified only at the genus or family level. The wood density of individuals not identified at any botanical level was considered as the average of the species included in the study database. Thus, we obtained wood density at species, genus, family and database levels for 60.1%, 25.5%, 11.8%, and 2.6% of the measured individuals, respectively. Plants with multiple or bifurcated stems were considered as a single trunk, based on the calculation of the equivalent diameter, as proposed by Batista et al. (2014).

To estimate the AGB stock we selected allometric equations built from different species and a large range of sizes. The equation proposed by Chave et al. (2014) was selected to estimate the AGB of tree species (Table 1, Eq.1), since this equation allows the most robust generalization, due to the large sample used, composed of tropical and subtropical forests at different successional stages, including the Atlantic Forest. We applied this equation to estimate AGB in trees with DBH ≥ 5 cm, as small trees are outside the threshold proposed by the authors (DBH: 5-212 cm). Due to this limitation, to estimate the AGB of trees with DBH between ≥ 3 and < 5 cm we used the equation proposed by Nogueira Jr et al. (2014) (Table 1, Eq.2), which derives from forest restorations of the state of São Paulo and from species commonly observed in the studied sites. We also used specific equations for different life-forms, such as lianas (Gehring et al., 2004), and palm trees (Tiepolo et al., 2002) (Table 1, Eq. 3 and 4). To estimate the AGB of liana species, DBH was converted to the diameter at the ground level, as proposed by Gehring et al. (2004; 2008).

Table 1. Allometric equations used to estimate de AGB of the plant individuals.

Reference	Application	Equation
Chave et al. (2014)	Trees (DBH \geq 5 cm)	1 $AGB = 0.0673 \times (WD \times DBH^2 \times H)^{0.976}$
Nogueira Jr et al. (2014)	Trees (3 \geq DBH < 5 cm)	2 $\log(AGB) = -1.305 + 1.055 \times \log(DBH^2) + 0.34 \times \log(H) + 1.077 \times \log(WD)$
Gehring et al. (2004)	Liana (DBH \geq 3 cm)	3 $\ln(AGB) = -6.105 + 1.413 \times \ln(D) + 0.997 \times \ln(H)$
Tiepolo et al. (2002)	Palms (DBH \geq 3 cm)	4 $AGB = 0.3999 + 7.907 \times H$

AGB: Aboveground Biomass (Kg); WD: wood density (g.cm⁻³); DBH: diameter at breast height (cm); D: diameter at the ground level (cm); H: total height (m).

To understand the contribution of active restorations to the carbon ES we converted the stock values from AGB to aboveground carbon stock (AGC). Firstly, the AGB of the individuals were added and converted to tons per hectare (MgAGB.ha⁻¹). The stock of AGB were transformed into AGC (MgAGC.ha⁻¹), assuming that the dry biomass of the plants is constituted by 50% carbon (IPCC, 2006).

2.3.4. RESTORATION'S STRUCTURAL ATTRIBUTES AND CARBON STOCKS

In an attempt to understand the influence of the forest structure on the carbon stock in restorations at different ages, we selected attributes of density and size of the plants that are related to the carbon accumulation along the successional trajectory of forest ecosystems (Slik et al., 2013; Rozendaal et al., 2017; Ligot et al., 2018). In this sense, we calculated the total density of individuals - D_{Total} , the density of medium/large sized individuals (DBH \geq 30 cm) - $D_{\geq 30}$, and the density of small individuals (3 \geq DBH < 30 cm) - $D_{< 30}$. This subdivision into size classes follows the guidelines for tropical forests proposed by Slik et al. (2013), where three levels of DBH are defined: i) plants up to 30 cm (small size); ii) > 30-70 cm (medium); and iii) > 70 cm (large). As few individuals sampled fit into the last class, the medium and large classes were aggregated

(DBH \geq 30 cm). For each class adopted, the total AGC stock was calculated from the values obtained in each restoration, that is, carbon stored in medium/large plants - AGC $_{\geq 30}$, and small plants - AGC $_{<30}$. We also calculated the density of dead plants - D $_{\text{dead}}$, and its carbon stock (AGC $_{\text{dead}}$), since plants senescence affect the forest dynamics and influence the total carbon stock (Rozendaal and Chazdon, 2015; Rozendaal et al., 2017).

2.3.5. DATA ANALYSIS

To assess the temporal trajectory of carbon accumulation in restoration plantings, we analyzed AGC stocks concerning the restorations age. We estimate the potential for carbon storage by plantations using a regression model, generated from the carbon stock (MgAGC.ha $^{-1}$) as a function of the restoration age (5-30 years). Linear, polynomial and logarithmic models were tested and compared using the F test, with the linear model ensuring the best fit to the data. The quality assessment of the adjusted models was based on the coefficient of determination (R 2), the p-value and standard error of the regression. It is important to highlight that, for the construction of the models, we use only the carbon values stored in living plants, as dead plants no longer contribute to the processes of carbon sequestration and accumulation

We used the Pearson correlation test to assess the linear relationships among the AGCTotal stock (MgAGC.ha $^{-1}$) observed in the restorations, the calculated forest attributes (D $_{\text{Total}}$, D $_{\geq 30}$, D $_{<30}$, AGC $_{\geq 30}$, AGC $_{<30}$) and the age of the plantations. We evaluated the correlations and data dispersion graphs using the “*chart.Correlation*” function of the “*Performance Analytics*” package (Peterson et al., 2019; R Core Team, 2018). Data normality was checked by Shapiro-Wilk tests and when this presupposed was violated data were transformed by natural logarithm before the correlation test. Correlations were categorized according to the level of significance of the test (p-value: 0 to 0.001 ***, > 0.001 to 0.01 **, > 0.01 to 0.05 *, and > 0.05). According to the data dispersal pattern (linear or polynomial), we draw tendency lines for the carbon stock and abundance of the small and large size plants. All statistical analyzes were conducted in the R program, at the 5% ($\alpha = 0.05$) level of significance.

Finally, we obtained carbon stock values for mature FES remnants from the literature and used them as reference ecosystems. Such data were collected from studies conducted in forest remnants occurring in the study region, including a conservation unit (Barreiro Rico Ecological Station), whose data were obtained by methods similar to those used in the present study (Ferez et al., 2015; César et al 2018).

2.4. RESULTS

In total, we measured 2238 individuals ($DBH \geq 3$ cm) in the 16 restorations. Among the living individuals, we observed 236 taxa, of which 97.5% ($n = 230$ spp.) were identified at some botanical level (species, genus or family). We obtained the wood density of the different species observed, a parameter that presented a high amplitude, varying from 0.17 g.cm^{-3} (*Urera baccifera* (L.) Gaudich. ex Wedd.) to 1.12 g.cm^{-3} (*Machaerium acutifolium* Vogel), while the average value observed among the individuals is equal to 0.61 g.cm^{-3} . The individuals' DBH and height ranged from 3-100.7cm and 1.30-22m, respectively. The average value of AGC_{Total} observed in the living plants of the restorations from 5 to 30 years was $76 \pm 51 \text{ Mg.ha}^{-1}$. The AGC_{Total} values ranged from 17.0 Mg.ha^{-1} to 176.4 Mg.ha^{-1} . We observed that the stock of AGC_{Total} has a positive linear relationship with the age of planting (Figure 2; $R^2 = 0.46$, $p = 0.004$, Standard Error = 39.2).



Figure 2. Predictive model of carbon stock for restorations implanted by planting seedlings ($AGC \text{ stock} = 4.982 + 4.507 \times \text{age}$). Circles represent restorations at different ages, while the square

element represents the average AGC stock observed in reference ecosystems (Ferez et al., 2015; César et al., 2018).

Regarding the relative contribution to AGC_{Total} , individuals of medium/large size ($DBH \geq 30$ cm) represented only 8.4% of the plant's density observed in restorations of 5-30 years, but, on average, 61% of the carbon is allocated in this DBH class. In general, these individuals contribute to the majority of the AGC_{Total} at the end of the first decade after planting (Figure 3a), even though the density of these individuals remains very low over time (Figure 3b). Despite the AGC stock in both size classes increased over time, this effect was more pronounced for large plants. In addition, the contribution of these individuals to AGC_{Total} reached minimum values of 3% in young restorations (<15y), but never less than 40% in older restorations (≥ 15 y). The results also showed that the AGC stock in dead plants varied from 0.1 to 10.8 $MgAGC.ha^{-1}$ (2.7 ± 2.9 $MgAGC.ha^{-1}$ on average), which represents 3.5% of the AGC stock observed in live plants.

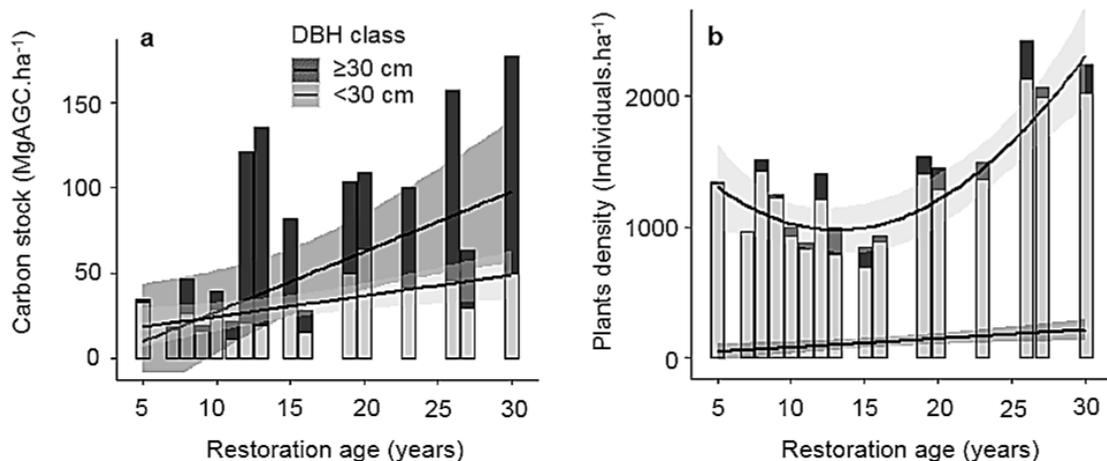


Figure 3. (a) Pattern of carbon accumulation in small ($DBH < 30$ cm) and medium/large plants ($DBH \geq 30$ cm) and (b) density of individuals in response to the age of the restorations. The colors of the bars and shaded areas of the regression lines represent the AGC_{Total} stock allocated in small (light gray; $DBH < 30$ cm) and medium/large plants (dark gray; $DBH \geq 30$ cm).

Pearson's correlation test indicated that the restorations age has a positive effect on every selected variable, that is, just as the AGC_{Total} , both the density of individuals and the carbon stock in plants of different sizes increase with time (Figure 4a). The dispersion of D_{Total} and $D_{<30}$ data

indicates that these attributes decrease in a first moment, increasing 10-15 years after planting, and the similarity in their tendency over time indicates an interdependent relationship. We observed that all the attributes evaluated are correlated to the total stock, but the density and stock in medium/large sized individuals ($DBH \geq 30\text{cm}$) showed the most significant correlations with total AGC ($D_{\geq 30} = 0.94$; $AGC_{\geq 30} = 0.86$) (Figure 4b), being the parameters more linearly correlated with the carbon stocks in the active restorations.

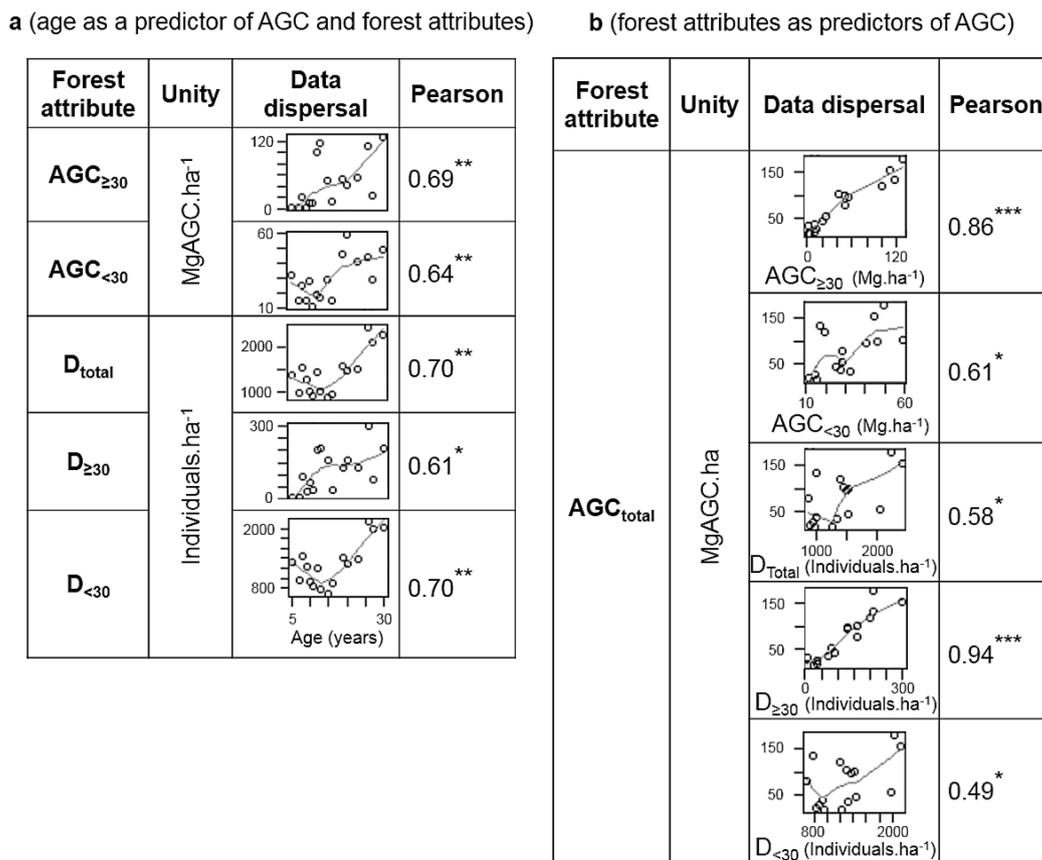


Figure 4. Data dispersal and Pearson's correlation. (a) Influence of restoration age in the AGC stock and the forest attributes and (b) influence of the forest attributes on the AGC stock.

2.5. DISCUSSION

2.5.1. CARBON STOCKS: RESTORATION PLANTINGS AS CARBON SINKS

Understanding the contribution of active forest restoration to carbon ES is essential to predict the consequences, advantages and disadvantages of this activity in the long term, assisting

the definition of monitoring and adaptive management protocols for these complex environments with multiple trajectories. Despite the potential for carbon sequestration and storage being extensively studied in natural forests (Chazdon et al., 2007; Chazdon, 2014; Lohbeck et al., 2015; Chazdon et al., 2016; Poorter et al., 2016; Rozendaal et al., 2017), these processes still lack knowledge in the context of communities implanted by planting seedlings.

The assessment of the carbon stock in the active restorations of different ages allowed to determine the pattern of carbon accumulation over time, evidencing that restorations aging from 5 to 30y present 76 MgAGC.ha⁻¹, on average (Table B.2), and the total carbon stocks responds linearly to restoration age. In regions of transition between SSF and Cerradão, César et al. (2018) observed that restoration plantings aging from 7 to 20y, implemented in sites previously used for the cultivation of sugarcane, can stock 83 MgAGC.ha⁻¹, on average. Also, in 4-25y restoration plantings within the extent of the SSF, Durigan et al. (2016) estimated carbon stocks equal to 76 MgAGC.ha⁻¹. Despite the particularities of each restoration site, these results show convergence in the pattern of carbon accumulation in active restorations of similar forest formations.

Regarding the temporal pattern of carbon accumulation, the literature points out that, along the secondary succession of tropical forests, the AGC stock has an initial pattern of rapid and linear increment, which stabilizes and reaches a plateau over time (Chazdon, 2014). In fact, we observed that the increase in the carbon stock in response to the age of the restoration plantings has a linear pattern, at least up to 30 years of age. This result indicates that the ecological processes involved in the carbon accumulation of restoration plantations correspond to the early phases of the forest succession, since, considering three decades past the implantation, we did not observe a leveling in the pattern of carbon accumulation, which is eventually expected in more advanced forests (Chazdon, 2014).

Compared to reference forests, the AGC results obtained in the restoration plantings are highly promising. Our model estimates that active restorations implanted with a high species diversity can store 140.2 MgAGC.ha⁻¹ in 30 years (Figure 2). According to Ferez et al. (2015) and César et al. (2018), the carbon stock estimated among remnants of SSF in the state of São Paulo

averages $153.7 \text{ MgAGC.ha}^{-1}$. This shows that more than 90% of the carbon stock observed in remaining forest fragments can be recovered by planting native seedlings after 30 years. For humanity, the relevance of this rapid recovery of carbon stocks by restored forests can be expressed in terms of atmospheric carbon sequestration. Thus, converting the carbon stock to carbon dioxide sequestration by a factor equal to 3.67 (IPCC, 2005), the estimated carbon stock for 30y restoration plantings is equivalent to 514.5 tons of CO₂ sequestered for each hectare of degraded lands restored. Regarding that the use of fossil fuels involved in the annual processes of planting, cultural treatments and mechanized harvesting of sugar cane emits around $1.1 \text{ MgCO}_2.\text{ha}^{-1}$ (Soares et al., 2009; Souza, 2016), the amount of carbon sequestered by restored forests represent a significant offsetting tool. We can roughly estimate that, after 30y, restoring one hectare of forests by planting seedlings could mitigate carbon emissions from agricultural activities carried out on approximately 16 hectares used for sugar cane production during the same period, an offsetting relationship of 1:13.

Such results corroborate that restoration plantings present a high potential to sequester atmospheric carbon and store it in the forest biomass, being able to reach, relatively quickly, the original stock of native vegetation. The results strengthen restoration actions as a tool to reduce global warming, which also represents an unprecedented opportunity to transform millions of hectares of degraded and unproductive lands around the world into ES-providing ecosystems (Chazdon, 2014; Bastin et al., 2019).

2.5.2. CARBON STOCKS AND FOREST STRUCTURE ALONG THE RESTORATION'S TRAJECTORY

We know that the pattern of carbon accumulation in tropical forests results from changes in the structure of the plant community over time (Marin-Spiotta et al., 2007; Kenzo et al., 2010; Fonseca et al., 2011; Lohbeck et al., 2015; Rozendaal and Chazdon, 2015; Rozendaal et al., 2017). In this context, we observed a positive correlation between the age of the restorations, the large trees and the total carbon stock (Figure 4b). In restorations $\geq 15\text{y}$ ($n = 8$), both parameters of abundance and carbon stock in these large individuals ($151 \pm 79 \text{ individuals.ha}^{-1}$ and 59.3 ± 40

MgAGC.ha⁻¹, respectively) represent almost 2-fold the values obtained in restorations <15y (n = 8; 83 ± 80 individuals. ha⁻¹ and 33 ± 47 MgAGC.ha⁻¹). Also, medium/large sized individuals are more frequent in restorations ≥15y, since they were observed in 76% of the sample plots allocated in this age class, while in younger restorations these individuals were found in less than 50% of the transects.

This can be explained by the fact that, as the surviving individuals grow, most of the total stock is transferred to superior diameter classes. Nevertheless, the time which a given species reaches a large size depends both on the availability of resources in the environment and on the growth rate of the species itself, parameters that are, in turn, affected by the competition that is intensified along the succession (Ferez et al., 2015; Rozendaal et al., 2017; Ligot et al., 2018). Regarding the behavior of the species, the medium/large individuals observed in the plantations are, in decreasing order of frequency, the species *Schizolobium parahyba* (Vell.) Blake, *Ceiba speciosa* (A.St.-Hil.) Ravenna, *Citharexylum myrianthum* Cham., *Enterolobium contortisiliquum* (Vell.) Morong, and *Inga vera* Willd.. These species correspond to trees that present rapid growth, and are widely implemented in restoration programs of the study region (Vidal et al., 2020), which supports the observed pattern of early carbon accumulation. The results corroborate studies conducted in natural forests, indicating that plant's size is one of the most important drivers in the carbon stock of forest ecosystems (Ligot et al., 2018). However, although we do not perceived negative effects of large trees on the restored communities, it is necessary to highlight the cases in which the excessive dominance of resources by these large individuals can trigger negative consequences in the long term, an effect already observed in restorations of our study region (Oliveira et al., 2019).

In a broader sense of the importance of large plants for the assembly of forest communities, the rapid growth of some species at the initial phases of the restorations' trajectory influences the reestablishment of ecological interactions that involve competition for resources, affecting the growth of the planted seedlings and the recruitment of new individuals (Reid et al., 2015; Toledo et al., 2018). Also, the small individuals residing in the understory, or moving to the forest

canopy, are extremely important for the restorations success, as they replace the individuals that gradually die due to senescence or competitive exclusion (Connel, 1978; Hubbel et al., 1999; Chazdon, 2014). Although small plants (DBH < 30cm) hold only 40% of the total carbon stock, these individuals dominate the studied communities, representing 92% of the total plant's density, on average. These results show that active restorations present a good development of the community structure that, probably, will sustain the replacement of senescent individuals, supporting the perpetuation of the carbon accumulation process over time.

An interesting result is that, in the first decades (10-20y) after the active restorations planting, the density of the communities tends to remain constant, or even decrease. However this pattern shifts after this period and the plant's density increases significantly (Figure 4a). This is explicit if we consider that the density of individuals in restorations <20y (1154 ± 263 individuals.ha⁻¹) is 30% lower than the average density of planted seedlings (1660 ± 251 individuals.ha⁻¹) among sites, while in restorations ≥ 20 y this value is about 15% higher (1938 ± 442 individuals.ha⁻¹). Thus, we can assume that new plants are recruited in the active restorations over time, composing the inferior stratum of these forests together with other small individuals of slow-growing species already established. Also, as individuals with DBH < 30 cm are dominant within the studied restorations, the total plant's density of these communities are directly related to the dynamics of these small plants and, if the recruitment process mentioned did not occur, there would be no significant relationship between the density of individuals and the age of the restorations. Furthermore, the representativeness of mortality in the carbon stock would be even more evident than the observed (3.5% of the AGC_{Total}, on average) if there was no recruitment, where there would be no replacement of the senescent individual.

In summary, the evaluated attributes are likely to be interpreted as indicators of forest recovery and, consequently, of the continuous and effective contribution of these environments to the carbon ES. In this context, the results suggest that the financial investment applied in active restorations can be justified in the medium/long term, at least concerning the reestablishment of processes involved in the forest carbon accumulation. Despite the limitations associated with the

use of chronosequence to assess restorations' trajectories, the observed patterns provide relevant insights regarding the ecological dynamics established in the forest structure of restoration plantings. The results allowed us to effectively evaluate the responses of the carbon stock to the age and structure of active restorations, contributing to the understanding of the functional dynamics of these communities undergoing restoration.

2.6. CONCLUSIONS

Our results show that forest restoration plantings inserted in landscapes degraded by agricultural activities, can be considered effective sinks of atmospheric carbon, since the stocks in the studied restorations increase with time and, after 30 years, reaches the stock observed in native forest remnants. We observed that medium/large plants represent the main structural elements related to the carbon stock of the restoration plantings. The fact that the total plant's density is higher in the older restorations, allow us to infer about the trajectory of these communities and their potential in sustaining the carbon accumulation process over time. We highlight that, besides being significant correlated to the carbon stocks and/or the restorations' age, some of the attributes may not present a linear response, especially plant's density. An important finding is that, even under unfavorable conditions, as sites of low local resilience and embedded in landscapes with few sources of propagating material, the ecological processes involved in the carbon accumulation of natural forest ecosystems can be restored in degraded lands by active restoration. Finally, the results show that, in general, forest restoration projects when ecologically planned and well-monitored, are effective in restoring functional ecosystems, being useful tools to contribute to the urgent and ambitious global goals toward climate change mitigation.

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Chapter 3.

The role of plant species and functional traits on the carbon stocks of active forest restorations

Fernando Ravanini Gardon^a; Alexandre Adalardo de Oliveira^a; Rozely Ferreira dos Santos^a; Ricardo Ribeiro Rodrigues^b

^a Department of Ecology, Institute of Biosciences, University of São Paulo, São Paulo, Brazil

^b Laboratory of Forest Ecology and Restoration, Department of Biological Sciences, University of São Paulo, Escola Superior de Agricultura “Luiz de Queiroz”, Piracicaba, São Paulo, Brazil

3.1. ABSTRACT

Active forest restoration is a global activity that attempts to recover the structure, composition, and ecological processes observed in natural ecosystems, contributing to ecosystem services as climate change mitigation. To ensure the effectiveness of these actions, it is necessary to assess the trajectory of the carbon stock concomitantly to the plant community dynamics over time. We aimed to evaluate the relationships among the carbon stock, the species abundance, and plant functional traits in active restorations. We selected a chronosequence of 16 active restorations aging from 5 to 30 years old, where individuals of woody plant species were identified and measured. We estimated biomass by allometric equations and converted to carbon stock. We related the carbon stocks to functional traits to understand the potential of carbon accumulation by different plant groups (life form, ecological group, and wood density). We assessed the species' abundances within the communities as a function of the restorations' age to evaluate dominance effects. We observed similar carbon stocks and abundances of pioneer and secondary species, which presented higher values than the late-successional ones. However, while late-successional species show a continuous increase in their relative contribution to carbon stocks, the secondary and pioneer species present a decreasing relative contribution over time. It evidences that the established dynamics favor the carbon stock in species of higher wood density, relevant elements for the perpetuation of the carbon accumulation over time. Although a higher asymmetry in the plant species abundance is expected in older restorations, we observed no negative effects on the total carbon stock. Our study highlights the transitional dynamics among the different ecological groups during active restorations trajectory, providing relevant results for the restoration practice.

Keywords: Restoration Trajectory; Restoration Age; Life form; Ecological Group; Wood Density; Tropical Forest.

3.2. INTRODUCTION

Forest restoration is a strategy applied to expand forest cover and recover the biodiversity and the ecological processes lost by deforestation (Brancalion et al., 2015; McDonald et al., 2016), contributing to the integrity of tropical landscapes and the delivery of several ecosystem services (ES) (Bullock et al., 2011; Wu, 2013; Crouzeilles et al., 2017). Worldwide, the ability of forests undergoing restoration to sequester atmospheric carbon and mitigate climate change effects have been treated as one of the key ES provided by these systems (Chazdon et al., 2016; Poorter et al., 2016; Crouzeilles et al., 2017; Phillipson et al., 2020). Studies have shown that planting seedlings of native species (active restoration), planned in the principles of ecological restoration, is a promising strategy to support the international goals related to the reduction of the atmospheric CO₂ concentrations (Suganuma and Durigan, 2015; César et al., 2018; Brancalion et al., 2019; Gardon et al., 2020). Notwithstanding, if the purpose is to implement restorations that contribute to regulating ES in the long term, it is necessary to evaluate the patterns of carbon accumulation according to the ecological dynamics that establishes in the different phases of the restoration process.

Several researchers recognize that, throughout the successional trajectory of tropical forests, the changes in the plant community affect the recovering of ecological processes, which depends directly on specific characteristics of the occurring species and their populations (Westoby et al., 2002; Cornelissen et al., 2003; Violle et al., 2007; Lavorel et al., 2007; Webb et al., 2010; Missio et al., 2017). Over time, groups of species with diverging characteristics interpose themselves according to the changes in the environment, filling and complementing the different niches available within the recovering communities, which results in a resilient and stable system where the successional trajectory can move forward (Chazdon, 2014; Arroyo-Rodríguez et al., 2017). Such processes are associated with morpho-physio-ecological aspects of the plant species named functional traits, which determine their performance within the communities (Violle et al., 2007). Since these traits can influence the dynamics of the community at different organizational levels (Orians et al., 1996; Violle et al., 2007; Gamfeldt et al., 2008),

we can expect effects on ecological processes relevant to humanity, such as the atmospheric carbon sequestration (Cornelissen et al., 2003; Lavorel et al., 2007; Chazdon et al., 2014).

In the context of active forest restoration, the applicability of functional approaches lies in the fact that this can be useful in species selection, leading to the planting of a set of species that will promote the quick recovery of specific ecological processes, or the desired species composition (Suganuma et al., 2014; Rayome et al., 2019; Gastauer et al., 2020). Thus, functional trait approaches assist the planning and the determination of the effectiveness of restorations in offering carbon ES (Carlucci et al., 2020).

Among the traits that affect the carbon stock in forest ecosystems, the life form (tree, liana, palm, among others) is related to the plants' resource acquisition strategies, which results in species with different sizes and architectures that influence the carbon stratification within the plant community (Pugnaire and Valladares, 1999; Jirka et al., 2007; Van der Heijden and Phillips, 2009). Also, the contribution of species from different ecological groups (pioneer, secondary, and late-successional species) to the carbon stock depends on the longevity, growth rate, and habitat requirements of the species, thus including the availability of resources in the environment (Orians et al., 1996; Chazdon, 2014). Such aspects determine the phase of the ecological succession in which a specific group of species dominates the community, and this turnover in the ecological groups is one of the mechanisms that ensure the continuity of the carbon accumulation by the forest ecosystems (Chazdon, 2014). However, the wood density may be more directly related to the carbon stock of forest ecosystems, since it expresses the dry nutrient content of the species, including carbon content, being an attribute that is tied to plants' life history and successional pathways (Chave et al., 2006; Poorter et al., 2019).

Although the literature shows that plant species have traits that are linked to their role in the forest carbon stock, these evaluations are still scarce in active restorations. In addition, if we consider that, unlike passive restoration, active restorations are based on the choice of species that are introduced in the planting sites, it is essential to evaluate the assertiveness of these decisions in a temporal context. This is crucial to determine if the restoration plantings can contribute to

carbon ES in a stable and resilient manner. For these reasons, we aimed to evaluate the relationships between the carbon stock, the functional traits, and the composition of plant species in forest restoration plantings of different ages.

3.3. MATERIAL AND METHODS

3.3.1. STUDY AREA

We based our analysis on a database composed of the aboveground carbon stock, species composition, and age of 16 forest restorations (aging from 5 to 30 years) distributed in six municipalities of the state of São Paulo - Brazil (see chapter 2). The active restorations were implemented by planting 1660 individual.ha⁻¹ and 74 species.site⁻¹, on average (Garcia, 2012; Duarte, 2013; Silva, 2013; Suganuma, 2013; Vasconcellos et al., 2013; Pardi, 2014; Suganuma and Durigan, 2015; Domene, 2018). The study region is embedded in the range of the Seasonal Semideciduous Forest (SSF), including ecotones of forest formations - SSF/Cerradão (Coutinho, 2016; IBGE, 2019). The landscape adjacent to the restorations presents low native forest cover (12% on average) and the matrix is constituted of mechanized sugar cane cultivation (Coutinho, 2016; IBGE, 2019; Mapbiomas).

3.3.2. DATA SAMPLING

In 160 transects systematically installed within the restorations (10/restoration; 100 m² transects - 4 x 25 m) all individuals of woody species (trees, shrubs, lianas, and palm trees) with a diameter at breast height (DBH) \geq 3 cm were identified, and its DBH (cm) and height (m) were measured. The biomass of the sampled woody individuals (tree, lianas, and palm species) was estimated by specific allometric equations adjusted for the different life forms, being transformed into the carbon amounts by a conversion factor (see chapter 2).

To this database, we added functional traits of the species related to carbon dynamics in forest ecosystems: the life form, the ecological group (species' successional category), and the wood density. We confirmed the life form (trees, shrubs, lianas, and palm trees) of each species in the Flora do Brasil 2020 database (<http://floradobrasil.jbrj.gov.br/>), by the “*flora*” package

(Carvalho, 2017) available in the R software environment (R Core Team, 2018). Data related to the ecological group of the identified species were obtained in the literature (Table C.1), where pioneer and secondary-early species were respectively classified as “pioneer” and “secondary”, while the secondary-late/climax ones were classified as “late”. From the “*BIOMASS*” package in R (Rejou-Mechain et al., 2017), we obtained the wood density of the species. If this data was not available for a particular species the average wood density of the genus was provided, and if there was no data for the genus, then the average of the botanical family was used. We also adopt this criterium when an individual has been identified only at the genus or family level. The wood density of individuals not identified at any botanical level was considered as the average of the species included in the study database. Thus, we obtained wood density at species, genus, family and database levels for 57.9 %, 27.9%, 11.5%, and 2.7% of the measured individuals, respectively.

We assessed the relationship of dominance among the species present in the communities undergoing restoration by the Pielou Index, using the “vegan” package in R (Oksanen et al., 2019). This parameter holds the result of the ratio between the Shannon-Weiver diversity index (H') and the natural logarithm of the species richness observed in each restoration. This index compared among active restorations of different ages could help to explain the dominance relationships among the planted species, contributing to the understanding of the carbon stock accumulation in these human-made systems (Smith and Wilson, 1996; Siddique et al., 2008; Pütz et al., 2014; Podadera et al., 2015; Oliveira et al., 2019; Pontes et al., 2019).

3.3.3. DATA ANALYSES

To identify the role of trees, lianas, and palm trees in the carbon stock of the planted communities over time, we calculated the carbon stored in each life form and presented the results as mean values and standard errors, procedure also applied to the ecological groups.

To understand the temporal dynamics of the ecological groups, the carbon stocks and the density of individuals (response variables) of the pioneer, secondary, and late-successional species in each restoration were related to the restorations' ages (explanatory), through multiple

regression analyses including linear and polynomial relationships. We used the ecological group as a categorical explanatory variable in these models. To control for the differences observed in the total plant's density of the restoration plantings, this parameter was included as an explanatory variable. We considered both the additive (intercept change) and the interaction (slope change) effects of the explanatory variables. We used the F-test (variance ratio) to check the explanatory ability of nested models and select the ones that best fit the data. This procedure was conducted in a backward elimination approach based on the principle of parsimony (Occam's Razor). We also applied the F-test to evaluate the significance of the terms included in the selected models. We used the t-test to verify differences in the coefficients estimated for the different ecological groups. We evaluated the selected models based on their coefficient of determination (R^2) and the p-value of the overall F-test, with the level of significance equal to 5%.

The relative contribution of the different ecological groups to the total carbon stock and total plants' density was calculated by the ratio of the predicted values for each ecological group and the restorations' total carbon stock or total density. The models adjusted for total carbon and total density are available in the supplementary material (Figure C.1).

To relate the average wood density - g.cm^{-3} (response) and the Pielou Index (response) of each community to the restorations' age (explanatory) we conducted simple regression analyses and F tests to select the models.

In an attempt to identify not only the groups but the species that, in general, are more representative in terms of carbon stock and density of individuals, we evaluate the accumulated carbon stock and the overall density of each species, by summing the values obtained in the 16 restorations. We also calculate the carbon stock from exotic species, as the occurrence of these species can harm the plant community in ecosystems undergoing restoration (Sartorelli et al., 2018). The origin (native or exotic) of the identified species was confirmed according to the Flora do Brasil 2020 database. Posteriorly, we related the estimated carbon stock of the species identified in each restoration separately for native and exotic species.

All statistical analyses were conducted and graphics were generated in R (R Core Team, 2018)

3.4. RESULTS

In total, we measured 2323 individuals of 250 taxa, of which 92.5% were identified at the species level, 1.7% at the genus, and 3.3% at the family level, the remaining 2.5% represent non-identified species. In total, 92.5% (n = 197) are species native to Brazil and 7.5% (n = 16) exotic, being 4.2% (n = 9) naturalized and 3.3% (n = 7) cultivated. Among the naturalized and cultivated plant species, *Cordia myxa* L., *Leucaena leucocephala* (lam.) de Wit, *Syzygium cumini* (L.) Skeels, *Melia azedarach* L. and *Psidium guajava* L. are species that store significant amounts of carbon. Among the native tree species, we observe that the dominant species in the total carbon stock are, in decreasing order, *Anadenanthera colubrina* (Vell.) Brenan, *Platypodium elegans* Vogel, *Piptadenia gonoacantha* (Mart.) J.F.Macbr., *Citharexylum myrianthum* Cham., *Inga vera* Willd., *Schizolobium parahyba* (Vell.) Blake, *Gallesia integrifolia* (Spreng.) Harms, *Ceiba speciosa* (A.St.-Hil.) Ravenna, *Schinus terebinthifolia* Raddi and *Senegalia polyphylla* (DC.) Britton & Rose (Table C.1). We found 18 liana individuals, distributed in 11% of the 160 sampled transects. The genera *Senna*, *Dolioscarpus*, *Fridericia*, *Paullinia*, and *Serjania* are representative of the carbon stored by lianas in the restoration plantings. We observed only two individuals of palm trees, in two plots of different restoration sites, which were identified at the family level. Shrub species, on the other hand, occurred in 33% of the 160 transects, with species from the *Piper* genus being, by far, the most frequent within this life form.

Among the evaluated life forms, trees represented 90.8% (227 species) of the species pool, followed by shrub - 5.6% (n = 14), lianas - 2.8% (n = 7) and palm species - 0.8% (n = 2). The carbon stored in tree species reached, on average, $75.5 \pm 12.8 \text{ MgC.ha}^{-1}$, while palm trees showed an average stock of $0.29 \pm 0.15 \text{ MgC.ha}^{-1}$, similar to liana species - $0.28 \pm 0.09 \text{ MgC.ha}^{-1}$ (Figure 1). Regardless of restorations' age, the carbon stored in the tree species shows an increasing pattern, which is harsh to observe in the other life forms given its low abundances, frequencies, and contribution to carbon stock.

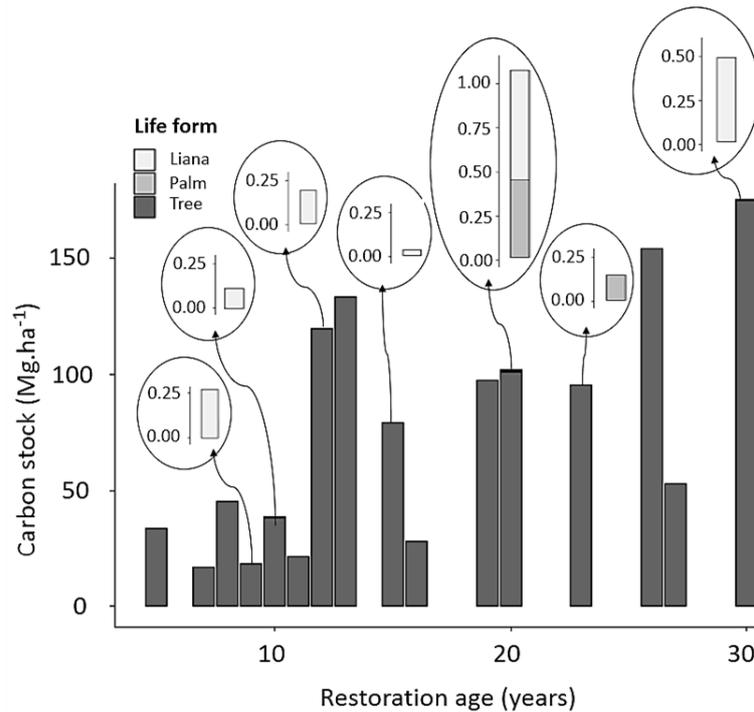


Figure 1. Carbon stock in the different life forms of the 16 restoration plantings. The bars represent the carbon stored in the tree, palm, and liana species. The circles above the bars detail the carbon stored in lianas and palm trees.

We were able to classify the ecological group into 83% of the identified species, or 92% of the measured plant individuals. Pioneer species represents 25% ($n = 62$) of the species, 34% ($n = 85$) secondary, and 24% ($n = 61$) late-successional ones (Table C.1). On average, secondary species constitute the main contributors to carbon ES ($31.3 \pm 5.8 \text{ Mg.ha}^{-1}$, which represents 45% of the total carbon stock) in the active restorations aging from 5 to 30y, with carbon stocks similar to pioneer species ($30.0 \pm 7.4 \text{ MgC.ha}^{-1}$; 43%), but higher than the late-successional ones ($9.0 \pm 2.3 \text{ MgC.ha}^{-1}$; 13%). The results show that the amount of carbon stored in the pioneer and secondary species is significantly different from the late-successional species (Table C.2). We observed a positive and linear response of the carbon stocks to the restorations' age for the three ecological groups ($p = 0.00$; $R^2 = 0.35$), with an increment equivalent $1.3 \text{ MgC.ha}^{-1}.\text{year}^{-1}$ (Figure 2a). The relative carbon stock from pioneer and secondary species is higher in the young restorations, while the late-successional ones present a continuous increase in their relative

contribution over time, being more representative of the total carbon stock in older restorations (Figure 2b).

On average, pioneer species represent the most abundant ecological group in the active restoration sites (529 ± 50 plants.ha⁻¹; which represents 40% of the total density of individuals), followed by the secondary (504 ± 68 plants.ha⁻¹; 38%) and late-successional species (296 ± 49 plants.ha⁻¹; 22%) (Figure 2d). As for the carbon results, we found that the abundance of pioneer and secondary species is significantly different from the late-successional species (Table C.3). In general, the abundance of the ecological groups presents a polynomial response to the restorations' age ($p = 0.00$; $R^2 = 0.68$). While late-successional species show a continuous, positive, and almost constant response to the restorations' age, pioneer species present a decrease in their abundances over the first years after planting, shifting about 15 years. This pattern was also observed for the secondary species, but a steep increase in their abundances can be expected after the mentioned period (Figure 2c). Along with the chronosequence, pioneer species are dominant in the active restorations during two decades after planting, a pattern that changes when secondary species become more representative within the communities, presenting a higher relative contribution to the total density of individuals (Figure 2d). The late-successional species is the only ecological group where an increase in its relative density can be observed in the first years after planting.

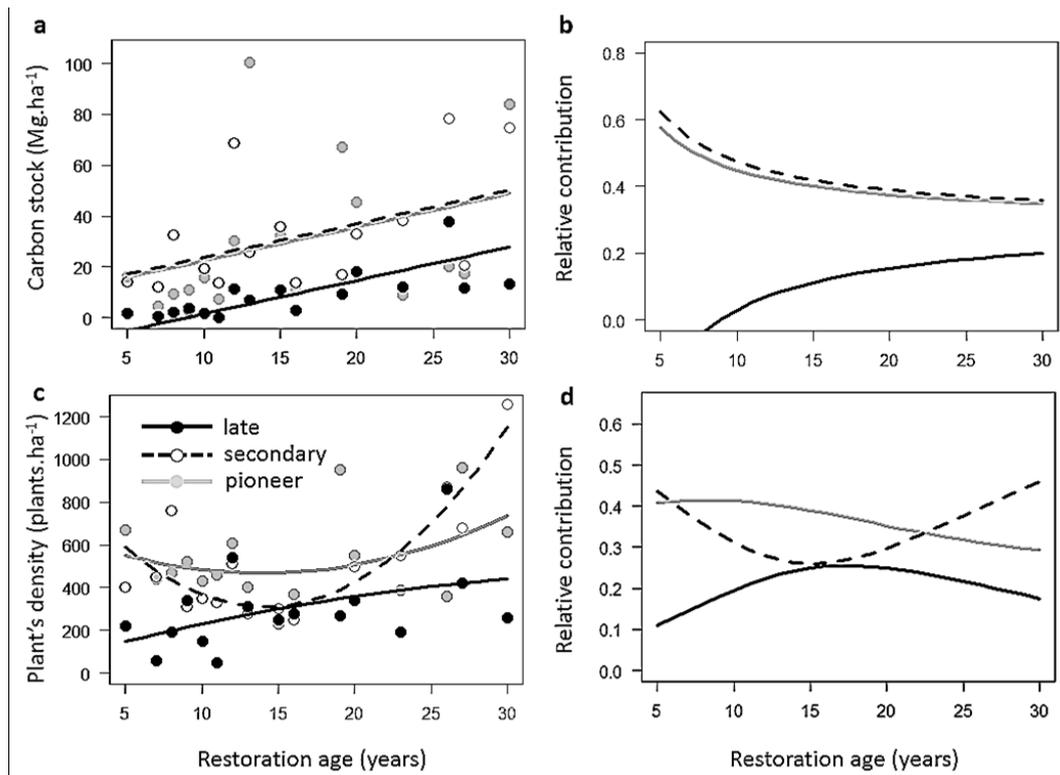


Figure 2. (a) Carbon stored in the different ecological groups ($\text{Mg}\cdot\text{ha}^{-1}$) observed in restoration plantings of different ages and (b) their predicted relative contribution to the total carbon stock of the restoration sites; (c) Plant's density (or abundance) of the different ecological groups ($\text{plants}\cdot\text{ha}^{-1}$) and (d) their predicted relative contribution to the total density of individuals.

Regarding the restorations' trajectories, the results indicate a positive effect of the restorations' age on the average wood density ($\text{g}\cdot\text{cm}^{-3}$) of the species identified in the communities undergoing restoration (Figure 3a). Also, the Pielou Index presented a polynomial response to the restorations' age, providing evidence that the symmetry in the abundance of the species increases in the first 15 years after seedling planting, followed by a decrease (Figure 3b).

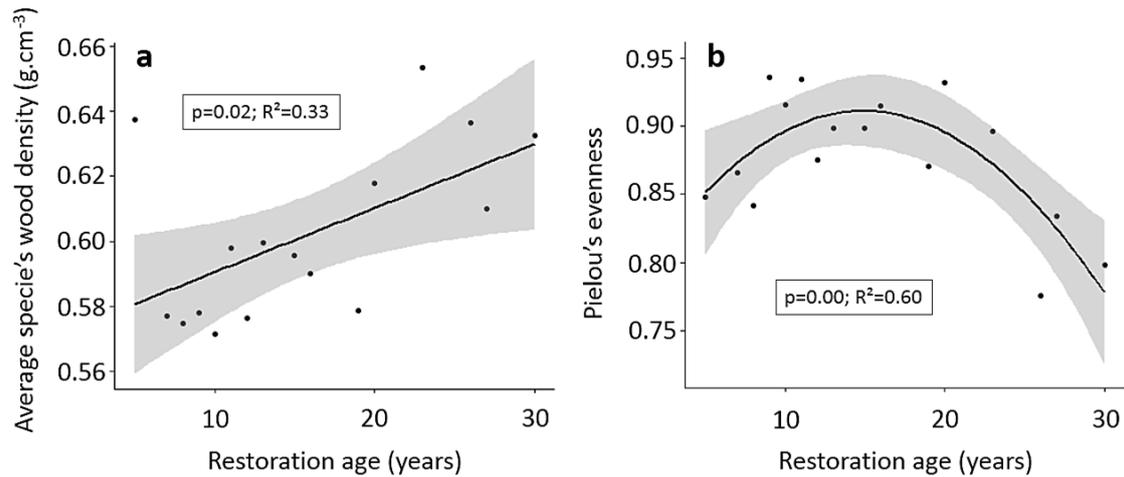


Figure 3. Effect of the restorations' age on the (a) average wood density (g.cm⁻³) and (b) the Pielou Index.

3.5. DISCUSSION

Our study shows that the restoration plantings aging from 5 to 30 years old presented carbon stocks ranging from 17 to 176 Mg. ha⁻¹ (Figure 1), evidence that restoring forests by planting seedlings is a strategy that contributes to offsetting human carbon emissions. However, the different plant life forms occurring in the restorations exhibit specific contributions to the total carbon stock. As expected, tree species represent the most relevant life form for carbon ES in the studied communities, regardless of the restorations' age (Figure 1). In contrast, lianas and palm species have a small contribution in the carbon stock of the restorations, being absent in more than half of the restorations.

We observed that the carbon stored in the tree species is almost three-fold (75.5 Mg.ha⁻¹ on average) the value obtained in tree species from naturally regenerated SSF of the Atlantic Forest (n = 13) also aging from 5 to 30 years - 26.8 Mg.ha⁻¹ (Matos et al., 2020). This evidences that the performance of tree species in restoration plantings can be superior to the observed in sites abandoned for passive restoration methods. On the other hand, the carbon stored in lianas and palm trees reached less than 1% of the total stock observed in their communities. Notwithstanding, the literature shows that palm trees, ferns, bamboos, and epiphytes can store more than 10% of the total carbon stock in Atlantic Forest fragments (Vieira et al., 2008).

Particularly, in forest remnants of the state of São Paulo, the carbon stored in liana species can reach from 3.3 to 7.5 Mg.ha⁻¹ (Burger and Delitti, 1999; Alves et al., 2012). Thus, if we consider the unfavorable conditions (low local and landscape resilience) in which the studied restorations are embedded, the carbon stored in tree species exhibits promising results, but other life forms present contributions far below the expected outcomes.

These results corroborate the need for efforts aiming at the production of seedlings of non-tree life forms in the nurseries and their use in restoration projects (Vidal et al., 2020), which would assist achieving the ecological principles that drive restoration success (McDonald et al., 2016). Despite this, we must pay attention to the presence of lianas or palms presenting invasive behavior on the restoration sites, which can negatively affect the successional progress (Dislich et al., 2002; Christianini, 2006; Cardinelli et al., 2017; Marshall et al., 2020). Thus, the ecological knowledge related to these species is of utmost importance to define when and how to introduce them into restoration plantings, avoiding undesirable consequences at both short and long term.

Regarding the ecological groups, the early carbon accumulation resultant from the pioneer and secondary species (about 80% of the total stock at 5y) does not seem to affect the carbon accumulation among late-successional species, on the contrary. This result is probably related to the changes in the availability of resources in the active restoration sites over time. After planting, the sites' conditions change from an exposed soil and a high incidence of sunlight to a continuous forest cover with substantial biomass, initially triggered by pioneer and early secondary species, which favors the development of the late-successional species, as in naturally regenerated forests (Chazdon, 2014). In fact, the fast shading is one of the main ecological processes intentionally reproduced in active restorations, aiming to inhibit the intense competition promoted by invasive species and create conditions for plants from different successional groups to establish (Rodrigues et al., 2009). Although time is necessary until forest biomass accumulates to the extent of changing environmental conditions, consequently affecting the distribution of the ecological groups within the communities (Chazdon, 2014; Lohbeck et al., 2015; Villa et al., 2020), the observed dynamics seems to facilitate late-successional species developing. On the other hand,

as the pioneer and secondary species are equivalent in carbon stocks and their relative contribution to carbon stocks present a continuous decrease over time, especially in the first 10y after planting, we presume that competing effects can be intense among these species in a 30y period.

While the carbon increment over time is constant to pioneer, secondary, and late-successional species, the increment in their abundances shows specific patterns. The finding that secondary species overwhelm the abundance of the pioneers can be interpreted as a “turnover” in the dominant ecological group. Notwithstanding, the literature highlights that it is not rare the cases of recovering forests that do not reach the “turnover” phases among ecological groups, leading to the stagnation or setback of their successional processes (Souza and Batista, 2004; Arroyo-Rodríguez et al., 2015; Villa et al., 2020). In general, we observed the opposite situation, where the dynamic of succeeding pioneer-secondary species suggests that the ecological processes ruling the recovering trajectory of natural forests can be reestablished or replicate in the restorations plantings (Guariguata and Ostertag, 2001; Chazdon, 2014).

A result that contributes to this discussion is that, in addition to the continuous increase in the carbon stock, late-successional species represent the only ecological group that shows an almost constant increase in its abundance over time. From this, we can presume that in active forest restorations aging from 5 to 30y late-successional species find well-established conditions for its development. The fact that the relative abundance of late-successional species changes its increment pattern 15 years after planting can be a consequence of the high recovering rate expressed by secondary species beyond this point, which results in a lower relative contribution from the other ecological groups.

Our results suggest that, over a 30y period, pioneer and secondary species are the main drivers of the carbon accumulation in restoration plantings. In contrast, the plant's density of these restorations is most related to the abundance of pioneer species in a first moment, a dominance that shifts towards secondary species over time, similar to the successional phases observed in naturally regenerated forests (Guariguata and Ostertag, 2001; Chazdon, 2014; Pinã- Rodrigues et al., 2014; Shimamoto et al., 2018; Bonner et al., 2020). These findings contribute to better

understanding the dynamics of the different ecological groups on a temporal scale. Thus, we can infer how the carbon accumulation process is affected by the successional trajectory established in active forest restorations.

We highlight that the relevance of late-successional species to the carbon accumulation in forest ecosystems is also associated with their wood properties, as the high wood density commonly observed in this ecological group influences fundamental processes related to carbon sequestration (Chave et al., 2006; Cavanaugh et al., 2014; Poorter et al., 2017). Thus, the rising of the average restorations' wood density along with the chronosequence indicates that species playing a relevant role in the functional aspects of the carbon ES supply tend to be more abundant in the older restorations. It is important to note that, in addition to the high wood density, late-successional species are commonly long-lived plants and are more resistant to unfavorable conditions, which assist the community to sustain carbon accumulation in both the short and long term (Maciel et al., 2003; Chazdon, 2014; Shimamoto et al., 2014). Thus, the positive increment observed in both the carbon stocks and the abundance of these species over time points to the potential of the communities undergoing restoration in perpetuating their contribution to carbon ES.

From the analysis of the species' abundance evenness, we found that some species tend to be dominant within the restorations, indicating a stronger effect among the young (<10y) and older (>20y) restorations, while within the middle age ones (aging from $\geq 10y$ to $\leq 20y$, approximately) the plant species present more equitable abundances (Figure 3b). Although this dominance effect was not detrimental to the carbon stocks of the evaluated restorations, it can result in negative consequences to the recovery of ecological processes, as biomass accumulation, in the face of extreme dominance of few plant species (Siddique et al., 2008; Pütz et al., 2014; Podadera et al., 2015; Oliveira et al., 2019; Pontes et al., 2019). Thus, it is necessary to assess the causes of the observed dominance patterns, which could assist in determining whether it is related to the planting method and/or the competing abilities among the introduced species (Orians et al., 1996; Suganuma et al., 2014; Hallett et al., 2017), supplying relevant information to define

possible management actions. Despite that, Saldarriaga et al. (1988) already argued that, along with the secondary succession of tropical forests naturally regenerated, a high degree of species evenness can be expected by 20y, which is similar to our results obtained in the restorations plantings.

We must highlight that plant species presenting expressive contributions to the carbon stock are not necessarily the most abundant. For example, the species *Calliandra foliolosa* Benth., *Mabea fistulifera* Mart., *Tapirira guianensis* Aubl., *Guarea Guidonia* (L.) Sleumer, and *Senna multijuga* (Rich.) H.S.Irwin & Barneby, are abundant in the restorations but present a low carbon stock accumulated. In contrast, the species *Cariniana estrellensis* (Raddi) Kuntze, *Vitex polygama* Cham., *Anadenanthera peregrine* (L.) Speg, *Poincianella pluviosa* (DC.) L.P.Queiroz, and *Peltophorum dubium* (Spreng.) Taub. present lower abundances but high contributions to carbon ES. On the other hand, the species *Cecropia pachystachya* Trécul, besides having low wood density (0.41 g.cm^{-3}) and small sizes (DBH = 11.9 cm; height = 7.3 m, on average), is abundant in the active restorations (4% of the total plants sampled), which results in a significant contribution to the carbon stock. Such results strengthen the importance of assessing the carbon stored by the different species in compliance with their abundances (i.e. based on the species' accumulated carbon stock), assisting the identification of the species that play relevant roles on the carbon accumulation in forests undergoing restoration and, consequently, on the communities' dynamics.

This issue can be extended to the occurrence of exotic species and their contribution to the restorations' carbon stocks, result that requires special attention. Together, naturalized and cultivated species showed an average accumulated carbon stock ranging from 0.1 and 50 Mg.ha^{-1} among the restorations, and the species. The problem is that some of these species (*C. myxa*, *L. leucocephala*, *M. azedarach*, and *S. cumini*) are recognized as undesirable species in forests undergoing restoration due to their competitiveness and persistence in the environment, which triggers several adverse effects on the communities' dynamics (Sartorelli et al., 2018; Castro-Díez et al., 2019). However, exotic species do not seem to be detrimental to the carbon accumulation

of active restorations aging from 5 to 30y. Despite this, we observe that exotic species can, in some cases, contribute to 20-30% of the total carbon stock of the restorations. However, even if the presence of exotic species does not demonstrate to be harmful to the carbon stock in the restoration plantings up to 30y, the continuous monitoring of these communities is the only path to determine the magnitude of these possible effects over time.

In a broader relevance of our study, the findings are important as, differently from passive restorations, the active restoration is grounded on the selection of the species planted in the focal sites, and these decisions can impact the restoration progress. In this context, priority species for active restoration can be selected based on their roles in the plant community, as relevant species for successional processes or carbon accumulation, aiming to promote the recovery of specific aspects of the ecosystem (Charles et al., 2018; Rayome et al., 2019; Carlucci et al., 2020; Tsujii et al., 2020). Therefore, restoration plantings hold the opportunity to introduce a set of key species that could maximize the expected outcomes. Successfully, we provided a list of the species that could guide restorations towards the recovery of carbon ES in agricultural landscapes (Table C.1). Despite that, planning the set of species towards a single goal can result in a tradeoff relationship among the ecological processes that lead to the full recovery of the ecosystems (Mansourian et al., 2017; Charles et al., 2018; Chazdon, 2019; Tsujii et al., 2020), and the decision-makers must take it on board.

3.6. CONCLUSIONS

Our study provides evidence that the trajectory of the carbon accumulation in active forest restorations is closely related to the functional traits of the species occurring within the communities. The results show that the carbon stock in restorations aging from 5 to 30 years old ranged from 17 to 176 Mg.ha⁻¹. This result is directly dependent on tree species since lianas and palm trees are not frequent or abundant in the active restorations and represent less than 1% of the aboveground carbon stock estimated in the restorations. We observed that the carbon dynamic established in these forests is closely related to the early successional species, which, in turn, favors late-successional species of higher wood density. Secondary species represent the most

diverse ecological group, store the highest amount of carbon already in the first years of the chronosequence, and overwhelm the abundance of pioneer species about 20 years after planting. Although some species tend to be dominant over the restoration progress, it was not detrimental to the carbon stock. We observed that abundant species do not necessarily present an expressive contribution to carbon stocks, but due to high abundances, small sized and low wood density species can play a relevant role in the carbon accumulation process.

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Chapter 4.

Plant diversity and priority species for conservation, community functioning and ecosystem services in active forest restorations

Fernando Ravanini Gardon^a; Rozely Ferreira dos Santos^a; Ricardo Ribeiro Rodrigues^b

^a Department of Ecology, Institute of Biosciences, University of São Paulo, São Paulo, Brazil

^b Laboratory of Forest Ecology and Restoration, Department of Biological Sciences, University of São Paulo, Escola Superior de Agricultura “Luiz de Queiroz”, Piracicaba, São Paulo, Brazil

4.1. ABSTRACT

Forest restoration projects have been encouraged to expand the native forest cover of the landscapes, in the hope of preserving plant species, recovering ecological functions, and reestablishing the supply of ecosystem services. For this reason, we aimed to assess the contribution of forest restoration plantings to plant species diversity and its potential in harboring priority species to conservation programs, community functioning and ecosystem services. Thus, in 16 plantings aging from 5 to 30 years old, we evaluated the total richness and classified the identified species in three categories based on five priority classes: i) conservation – endemism and threatened of extinction status, ii) community functioning – wood density and dispersal syndrome, and iii) ecosystem services – potential for carbon accumulation. We also assessed the density and richness of the plant species within the priority classes. These parameters were related to the restoration's age to evaluate the results in a temporal context. Also, we crossed the priority classes to identify the species included in more than one, which constituted the priority species to be planted in active restoration projects. Individually, the restorations presented an average richness of 43 species, but together the 16 restorations presented 250 species. Zoochoric species represent 51% of the identified species. Also, 15% are endemic, 12% have hardwood, 8% have a high carbon stock and 3% are among the most concerning threat levels. We observed that the density of species tends to increase with planting age only for zoochoric and high carbon stock species. We did not observe any effect of the restorations age on species richness regardless of the priority class. However, we found that few species are included in more than one priority class. We conclude that although restoration plantings can provide co-benefits, their potential is not fully employed, and possible solutions were discussed to improve the selection of species introduced in active forest restorations.

Keywords: Forest Restoration; Threatened Species; Endemism; Dispersal Syndrome; Wood Density; Ecosystem Services

4.2. INTRODUCTION

In tropical regions, forest restoration programs have been encouraged due to the multiple benefits that can be generated, such as the mitigation of the effects of biodiversity loss and global warming resulting from the degradation of natural ecosystems (Rodrigues et al., 2009; Crouzeilles et al., 2016; Poorter et al. 2016; Meli et al., 2017; Trentin et al., 2018; Brancalion et al., 2019; Rozendaal et al., 2019). These benefits have often been presented as Ecosystem Services (ES), a powerful tool for environmental planning and a guide to resource management (Balvanera et al., 2012). Although there is a continuous and fundamental growth in knowledge related to forest recovery and its role in the supply of ES, the implications of these actions for plant species conservation are less clear, especially in forests restored by active restoration methods (Gardner et al., 2011; Phelps et al., 2012; Suganuma et al., 2014; Poorter et al., 2016; Chazdon et al., 2016; MMA, 2017; Ferreira et al., 2018; Matos et al., 2019; Safar et al., 2020).

In forests undergoing restoration, studies have shown that there is an increase in species diversity over time and that forest restoration can recover a significant part of the species composition of the original ecosystems in the long term (Letcher and Chazdon, 2009; Chazdon, 2014; Suganuma et al., 2014). In this context, it is necessary to consider that different species have specific characteristics that determine their role in restoration projects, such as their importance for conservation (e.g. endemism and endangered status), their function within the community (e.g. successional processes and dispersion syndrome), as well as their ES supply potential (e.g. carbon accumulation) (Carter et al., 2000; Chave et al., 2006; Brehm et al., 2010; Kricsfalusy and Trevisan, 2014; Liu et al., 2019). Thus, it is reasonable to consider that species with multiple roles can be recognized as priority species, which should fill an outstanding position in the planning and monitoring of restoration projects (Beatty et al., 2018).

In this context, assessing both plant diversity and composition, as well as the presence and dynamics of species that are indicative of co-benefits is fundamental to direct the recovery and maintenance of key species in restorations embedded in anthropized landscapes. This information is particularly important when practicing active restoration methods, as planting seedlings, a

technique underpinned by the selection of the species introduced, able to achieve significant ecological outcomes in the low resilient sites (Rodrigues et al., 2009; Brancalion et al., 2015; Vidal et al., 2020). Thus, if species priority guides these decisions, restoration plantings could represent an opportunity to complement the “*in situ*” preservation practiced in protected forest remnants, ensuring synergistic outcomes by recovering ecological functions and supplying ES. Therefore, we aimed to investigate the contribution of forest restoration plantings to plant diversity and its potential in harboring priority species for restoration programs. We assessed the richness and abundance of plant species concerning their relevance to conservation programs, the recovery of ecological functions, and the delivery of carbon ES. We also provide a list of the potential priority species for restoration projects and discuss the mediations that could be adopted to achieve the expected multifunctional aspects of the restorations.

4.3. MATERIAL AND METHODS

4.3.1. STUDY AREA

The study was conducted in 16 active forest restorations aging from 5 to 30 years, distributed along with dams and watercourses in the state of São Paulo - Brazil. The study region was originally covered by the Seasonal Semideciduous Forest (SSF), including ecotones - SSF/Cerradão (Coutinho, 2016; IBGE, 2019). These forest formations are representative of the Atlantic Forest and the Cerrado biomes, recognized as biodiversity hotspots due to the high number of endemic and endangered species (Myers et al., 2000; Strassburg et al., 2017). The landscape adjacent to the restorations presents low native forest cover (12% on average) and the matrix is constituted of mechanized sugar cane cultivation (Coutinho, 2016; IBGE, 2019; Mapbiomas). Regarding the restoration technique, the average density of seedlings and the diversity of species planted among the study areas were 1660 individuals.ha⁻¹ and 74 species.site⁻¹, respectively (Garcia, 2012; Duarte, 2013; Silva, 2013; Sukanuma, 2013; Vasconcellos et al., 2013; Pardi, 2014; Sukanuma and Durigan, 2015; Domene, 2018).

4.3.2. DATA SAMPLING

Data were collected in 160 transects (4 x 25 m) installed in the 16 restorations (10.site⁻¹). In each transect, all individuals of woody species (trees, lianas, shrubs, and palms) with a diameter at breast height (DBH, measured at 1.30 m above the ground) ≥ 3 cm were identified at the most detailed botanical level and their DBH and height were measured (chapter 2). Species names were corrected and standardized by the function “*get.taxa*” from the “*flora*” package available for the R program (Carvalho, 2017; R Core Team, 2018).

We evaluated the species according to different characteristics related to their role in restoration projects. For this, we defined attributes of three categories to set groups of species representing key elements to i) forest conservation programs, ii) plant community functionality, and/or (iii) carbon ecosystem services.

4.3.3. PRIORITY SPECIES FOR CONSERVATION PROGRAMS

Regarding the priority species for conservation programs, we selected endemism and the conservation status at the national level of species as indicators of restoration's contribution to floristic conservation in anthropized landscapes. Endemic species have occurrence restricted to a given territory, being solid components of the conservation strategies (Myers et al., 2000; Lamoreux et al., 2006). This information was obtained from the Flora do Brasil 2020 database (<http://floradobrasil.jbrj.gov.br/>).

Threatened species are included in the red list of the Brazilian flora due to human exploitation, reduction of available habitat, or climate change, which justify its assessment as a priority species for conservation. The conservation status was obtained through the “*flora*” package, which accesses the database of the “National Center for Conservation of Flora” - CNCFlora (<http://cncflora.jbrj.gov.br/>), and provide conservation status in accordance to the “International Union for Conservation of Nature” - IUCN. We evaluated the species in the different categories proposed by IUCN (categories with adequate data: least concern, near threatened, vulnerable, endangered, critically endangered, regionally extinct, and extinct), but the

analyses were conducted considering only species classified in the threatened categories (vulnerable, endangered, or critically endangered), the ones that need to be urgently protected.

4.3.4. PRIORITY SPECIES FOR THE REESTABLISHMENT OF ECOLOGICAL FUNCTIONS

The dispersion syndrome was used to indicate plant species that interact with local fauna. In the context of forest restoration, the presence of these species drives restoration success by promoting seed dispersal and assisting in the recruitment of new individuals and species (DeWalt et al., 2003; Pardini et al., 2005; Reid et al., 2015). We obtained the species dispersion syndrome from the literature, being the species categorized into zoochoric, anemochoric, and autochoric (Table D.1).

We selected the species' wood density (g.cm^{-3}) because this attribute is associated with the reestablishment of ecological functions and successional processes. It is known that species of high wood density present slow growth, are long-lived plants, and more resistant to unfavorable conditions, recognized as important components of the communities' dynamics and related to the functional aspects of the carbon accumulation process over time (Chave et al., 2006; Slik et al., 2008; Lohbeck et al., 2015; Charles et al., 2018). Therefore, we categorized this functional trait as relevant to community functioning. To obtain the species wood density we used the function “*getWoodDensity*” of the “*BIOMASS*” package (Réjou-Méchain et al., 2017) available for the R program. When there was no information about a given species, the average wood density of the botanical genus was used. In cases of no information available for the genus, the average wood density of the botanical family was used, and for unidentified species (<3% of the plant individuals measured), the average wood density of all species observed in the restorations was used. We then define as hardwood species those with a wood density $\geq 0.774 \text{ g.cm}^{-3}$. This criterion was defined considering the average wood density of the identified species (0.611 g.cm^{-3}) plus one standard deviation (0.163 g.cm^{-3}).

4.3.5. PRIORITY SPECIES FOR CARBON ECOSYSTEM SERVICES

To understand the potential of restoration plantings in sustaining species that play a relevant role in the provision of ecosystem services, we used the total carbon accumulated in each species identified in the 16 restorations. For this, the biomass of the individuals sampled was estimated by specific allometric equations for trees, palms, and lianas, and the value obtained was transformed into carbon amounts by a conversion factor (see chapter 2). To calculate the accumulated stock, the carbon stock obtained for each individual of the species identified in the 160 sampled transects were summed (1.6 hectares). However, it is necessary to define the species that play a leading role in the reestablishment of carbon ES. Thus, we used the average of the accumulated carbon stock of the identified species (0.50 Mg) plus one standard deviation (1.20 Mg) to define species of high potential for carbon ES, that is, those that presented a total carbon stock (accumulated in the 16 restorations) equal to or greater than 1.7 Mg.

4.3.6. DATA ANALYSIS

We calculated the species richness of each restoration as a measure of the local contribution of these communities to plant diversity. We also assessed species richness on a regional scale, based on the pool of species observed in the 16 restorations. To assess the contribution of restoration plantings to plant species diversity in agricultural landscapes, we built a species accumulation curve. The species abundance in each of the 16 plantings was calculated and inserted in the "*specaccum*" function of the "*vegan*" package (Oksanen et al., 2019) to express the increment of species in the regional pool of species as a response to the sampled area.

To assess the plant diversity in the restoration plantings in a temporal context, we related the species richness of the restoration to their respective ages. We also evaluated the responses of the richness and the density of individuals of endemic, zoochoric, threatened with extinction (vulnerable, endangered, or critically endangered), high wood density, and high carbon stock species to the age of the restorations. These temporal relationships were expressed by regression analysis. We selected the models (linear and polynomial) that best fit the different data by

applying the F test to the residuals of the regressions. The quality of the adjusted models was checked based on the coefficient of determination (R^2) and the p-value of the regressions. Statistical analyzes were conducted in the R software environment (R Core Team, 2018), and p-values <0.05 were considered significant.

To determine the most important species for restoration programs, we also evaluated the priority classes simultaneously. Thus, we crossed the priority classes by a Venn diagram to highlight those that combine different classes, which were set as priority species for restoration as they present more reliable chances of promoting co-benefits.

4.4. RESULTS

We sampled 2323 individuals on the 160 transects sampled. We observed that, individually, restoration plantings present an average richness of 43 ± 10 species, and among the restorations, we identified 250 species, including trees, shrubs, palms, and lianas. Of these species, 20% ($n = 49$) are classified in the list of red list of Brazilian flora (CNCFlora), with 14% of them included in the most worrying levels of threat (vulnerable and endangered). In total, we found five species categorized as near threatened, five as vulnerable, and two others as endangered. Regarding the dispersion syndrome, 51% ($n = 127$) of the species are zoochoric, 25% ($n = 63$) anemochoric, and 16% ($n = 39$) autochoric. We observed that 15% ($n = 37$) are species endemic to the national territory, 12% ($n = 31$) present high wood density ($\geq 0.774 \text{ g.cm}^{-3}$), and 8% ($n = 20$) high carbon stock ($\geq 1.7 \text{ Mg}$).

The pattern of the species accumulation curve shows a high slope at the initial samples, with a decrease in the increment of species between 0.5 ha and 1 ha of sampling, where the accumulation of species becomes more linear, but remains positive (Figure 1a). Considering the temporal assessment of the species richness in the restorations, the regression analysis did not show any evidence of age effects on the species richness of the restorations (Figure 1b).

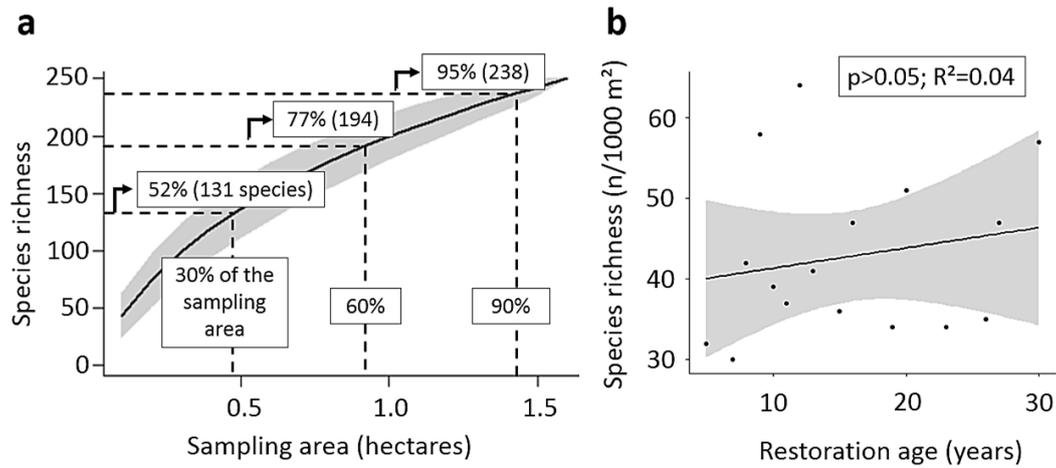


Figure 1. Responses of the (a) species accumulation curve as a function of the sampled area and the (b) species richness as a function of restoration age.

We also did not observe any effect of age on species richness, regardless of the priority class (Figure 2a). However, the results show positive relationships between the restorations age and the density of individuals of zoochoric and hardwood species, while for individuals of endangered species, endemic species, and high carbon stock, this effect is not significant (Figure 2b).

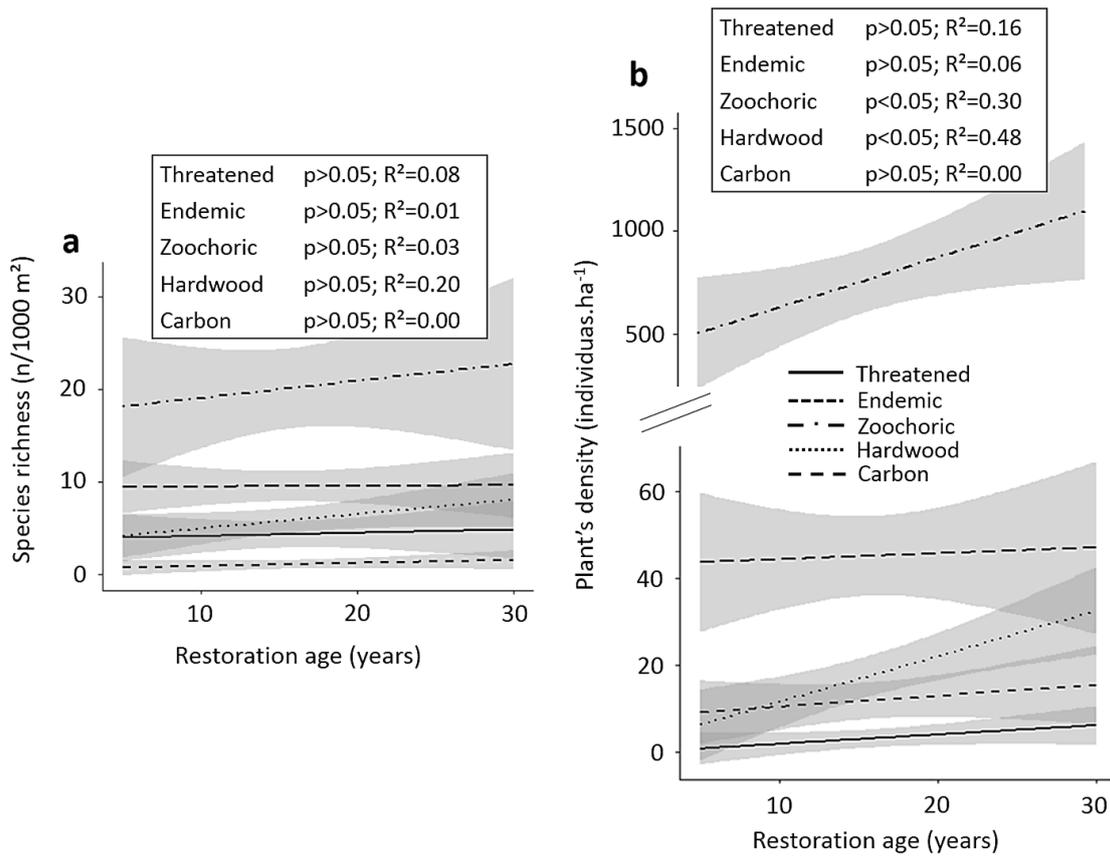


Figure 2. Relationship of the (a) richness and (b) density of priority species with the restoration age. The solid, dashed, dot-dashed, dotted, and long-dashed lines represent endangered, endemic, zoochoric, hardwood, and high carbon stock species, respectively.

The “Venn” diagram shows that some species share more than one class of priority, but none present more than three (Figure 3). It is possible to observe that the species included in the class of high potential for carbon stock are those that least share other relevant characteristics to restoration projects, which can also be observed for threatened species, but to a lesser extent.

state of São Paulo, including private lands and protected areas, can present an average richness of 177 species (Guaratini et al., 2008; Pinheiro and Monteiro, 2008; Sartori et al., 2015).

Studying a chronosequence of SSF regenerated from 7 to 33 years old, Safar et al. (2020) found that the species richness tends to increase with the age of the forest, pattern also observed along with a temporal scale analyses of a natural remnant of the same vegetation type (Farah et al., 2014). However, this effect does not seem to exist in restoration plantings aging from 5 to 30 years (Figure 1b). The hampered connectivity and the low forest cover (<30%) of our studied landscapes can be one of the causes of this result since the arrival of species via dispersion can be quite limited in these conditions (Holl et al., 2017). Another explanation for this result may be the fact that the past land-use of the studied areas was predominantly mechanized agriculture, an activity that harms the seed bank in the long term and contributes to the formation of low resilience sites (Rodrigues et al., 2009; Chazdon, 2014; Brancalion et al., 2015). Thus, the results indicate that the species richness observed in the restoration plantings can be directly dependent on the set of species introduced in the planting event, a condition that can persist for 30 years.

Our results corroborate the study conducted by Holl et al. (2017), which demonstrated that the restoration method and the site conditions are the main drivers of the ecological dynamics in active restorations implemented in regions of low forest coverage. It strengthens the importance of considering a high diversity of species when planting seedlings because, considering the poor local and landscapes conditions where these actions are usually implemented, there is no certainty that an expressive set of new species will arrive in the community via dispersion over time or seed bank germination (Rodrigues et al., 2009; Brancalion et al., 2015; Holl et al., 2017; Rother et al., 2019).

Although in the individual context the restoration plantings are far from presenting the species richness of the original forest formations, their combined contribution to the regional pool of species needs to be highlighted. Considering the pattern of the species accumulation curve, we can expect a less intense increase in the species pool from a sampling effort corresponding to 0.5 ha, which in our sample represents only five restoration plantings (Figure 2c). However, as new

restorations are added to the database, there is a tendency for a continuous increase in the total number of species. This argument is supported by the fact that we identified a total of 250 species, but the average richness among restorations is much lower (43 species). Moreover, 27% of the species ($n = 68$) were found in only one of the 16 restorations evaluated, showing that almost 1/4 of the species present very low frequency. Regarding the total pool of species and the species accumulation curve, the results reflect the high potential of restoration plantings in contributing to regional plant diversity in agricultural landscapes. Since this pattern is similar to that observed among 147 natural remnants of SSF embedded in São Paulo State – Brazil (Farah et al., 2017), we can expect active forest restorations to complement the plant diversity at the landscape level.

Worldwide, huge efforts have been applied to reintroduce and sustain endangered and endemic species in human-dominated landscapes, where forest restoration is one of the main tools indicated for this purpose (CBD, 2013; MMA, 2017). In Brazil, studies on naturally regenerated SSF aging from 5-33 years found an average of 2-3 species threatened of extinction (among the vulnerable, threatened, and critically endangered categories) and 9-10 species endemic in each restoration site (Matos et al., 2019; Safar et al., 2020). In a preserved remnant of SSF, Rocha et al. (2017) identified nine threatened and 44 endemic species. Considering that we found an average of only one threatened and four endemic species among the studied restorations, we can assume that in terms of priority species for conservation, restoration plantings do not achieve results similar to regenerated and/or preserved forests. Although we found a few threatened species in the restorations ($n = 7$), we can assume that the planning of these restorations did not significantly include the most important threat categories, as we did not find any species in the critically endangered category. We suggest that more efforts are needed to reintroduce vulnerable, endangered, and critically endangered species in restoration plantings, so that ecological restoration becomes an integral part of conservation programs (Volis et al., 2019). However, such changes are conditioned to the advancement of the knowledge related to the domestication of wild plant species and the improvement of the technical capacity of nurseries (see Vidal et al., 2020).

Another finding that draws attention is related to the hardwood species. The fact that the density of individuals of hardwood species tends to increase over time, is indicative that the dynamics established in the restoration plantings favor these species, promoting the trajectory of communities towards the advance of the ecological succession (Slik et al., 2008). These species, which arrive or develop later in the succession process, because of their higher wood density and their long life span are important elements of the carbon accumulation sustainability in the long-term (Chave et al., 2006; Chazdon, 2014; Shimamoto et al., 2014). Furthermore, hardwood species are more resistant to the scarcity of resources and to pests and pathogens, so the introduction of hardwood species in active restorations can benefit the dynamics of the communities, since these species present high survival rates even in unfavorable conditions (Charles et al., 2018). Among the hardwood species, *Machaerium acutifolium* Vogel., *Cordia ecalyculata* Vell., *Erythroxylum pulchrum* A.St.-Hil., and *Parapiptadenia rigida* (Benth.) Brenan are those presenting the highest values of wood density ($\geq 1.0 \text{ g.cm}^{-3}$).

In a broader sense, the high abundance of zoochoric species observed in the restoration plantings can be attractive to the remaining local fauna that, in turn, could facilitate the seeds dispersal, promoting the success of restorations (Reid et al., 2015). The substantial presence of zoochoric species observed among the restorations (47% of the restorations' species richness, on average) indicates well planned restorations, since this pattern has also been observed in SSF forest fragments (Metzger et al., 1997) and successfully replicated in the studied plantings. The abundancy and richness of these species in the restorations evidence the great importance of restored forests for the maintenance of native fauna in anthropized landscapes, since, in addition to shelter, recovering forests also provide food for many birds and small mammals (DeWalt et al., 2003; Pardini et al., 2005). In this context, zoochoric species such as *Genipa americana* L., *Guazuma ulmifolia* Lam., *Hymenaea courbaril* L., *Schinus terebinthifolia* Raddi, and *Trema micrantha* (L.) Blume, as well as those of the genera *Cecropia*, *Guarea*, *Inga*, *Myrsine*, and *Nectandra*, are important food resources commonly observed in the restoration plantings.

Regarding the capacity of the restorations in harboring species of high potential for carbon ES, the 20 species identified in this priority class represent more than 60% of the total carbon measured in the 16 restorations. In a decreasing order, the species *Anadenanthera colubrina* Vell. Brenan, *Platypodium elegans* Vogel, *Piptadenia gonoacantha* (Mart.) J.F.Macbr., *Citharexylum myrianthum* Cham., *Inga vera* Willd., *Schizolobium parahyba* (Vell.) Blake and *Gallesia integrifolia* (Spreng.) Harms presented the highest accumulated carbon stocks (≥ 4 Mg), equivalent to 38% of the total carbon measured in the restorations. These species can be regarded as key elements for restoration plantings where the objective is to offset atmospheric carbon emissions and mitigate the effects of climate change (Gaspar et al., 2014). It must be highlighted that in this priority class we used the values of carbon stock accumulated by species in the 16 restorations. Thus, if presenting high abundance and frequency, even species of low wood density and small sizes could be placed in a privileged position in terms of carbon ES due to cumulative stock. For example, we observed that the species *Cecropia pachystachya* Trécul has a high potential for carbon ES, despite its wood density being far below the criteria used in this study (Table D.1). This approach is important because it encompasses both species identity and populational aspects, constituting a more robust evaluation of the species classified as of high potential for carbon ES.

In the context of co-benefits, we observed that several species present only one of the priority attributes, but species included in at least two of the five priority classes evaluated are few and not abundant. We found that even in young restorations (<10 years), the density of individuals and species richness of different priority classes are very low, which indicates that these issues are not well established in the restoration planning, except for zoochoric species, which are abundant regardless of the restoration age. These results show that updating the concepts underpinning the selection of the set of species introduced in forest restorations could be positive (Gaspar et al., 2014). It could lead to an improvement in the synergistic gains promoted by these systems. Despite this, we recognize the dependence that the planning of these actions has on the availability of species in the local and regional nurseries (Vidal et al., 2020).

We observed that the species that represent key elements for restoration programs (species included in at least three priority classes) are not substantially present in the active restoration. In this sense, *Esenbeckia Leiocarpa* Engl., *E. pulchrum*, *Magnolia ovata* (A.St.-Hil.) Spreng., *Myrcia guianensis* (Aubl.) DC., *Ocotea puberula* (Rich.) Nees, *Psidium cattleyanum* Sabine, and *Trichilia casaretti* C.DC., are species that could offer synergistic gains, but which were observed in less than 20% of the restorations, being found one to five individuals of these species in the 160 sample plots. We must admit that there are rare species, of non-aggregated distribution, where few individuals naturally occurs in a given space (Metzger et al., 1997; Wilson et al., 2004; Maçaneiro et al., 2018), a pattern that might be replicated in ecological restoration plantings. However, the low abundance of these species in the restorations, together with the low forest cover of the studied regions, are detrimental factors for these populations in the long run, regarding the isolation context that these populations are embedded (Wilson et al., 2004; Guisan et al., 2006; Maçaneiro et al., 2018; Safar et al., 2020). Thus, to ensure the perpetuation of priority plant species in restored forests, it is necessary to monitor and manage their populations tied to the species natural behavior. This effort depends on assessments in preserved forest fragments distributed in the adjacencies of the restorations, to determine not only which species occur in the region, but their natural abundances and frequencies, a relevant information to restoration planning.

4.5.1. IMPLICATIONS FOR ACTIVE RESTORATION PRACTICE

The results show that restoration plantings have the potential to harbor species that supply different demands of restoration projects, but this potential has not been fully manipulated. Considering that the species richness in the different priority classes does not change with restoration age, well-planned enrichment actions, applied over time and focusing on priority species could improve the restorations contributions. Another possible action is the introduction of individuals of species included in a certain priority class that is not very abundant in the plantings, which would contribute to the maintenance of viable populations of priority species in anthropized landscapes. However, we must pay attention to the fact that priority classes, as wood

density, dispersion syndrome, and biomass are closely related to the species performance within the community (Reid et al., 2015; Charles et al., 2018). Thus, it is necessary to rigorously consider the identity and functional aspects of the species selected for planting, to correctly plan their introduction in time and space, avoiding negative effects and promoting synergistic results.

To the proposed management actions be effective in supporting multiple gains, defining a set of species that merge priority classes could be a reasonable action, able to leverage the ecological value of the forest restorations, harnessing its multifunctional character. However, this implies in the adaptation of the seedling nurseries for the production of different groups of species, which depends among other factors on developing restoration policies aiming the promotion of this supply chain that is undergoing a market downturn (Vidal et al., 2020).

While improvements in the production and selection of species for ecological restorations could assist to achieve synergistic outcomes, we consequently must avoid any tradeoff relationship. In this sense, we observed that the most relevant species in terms of carbon stocks rarely share other priority aspects. Thus, the prioritization of these species aiming at maximizing carbon sequestration could harm the multifunctional character of restorations and the synergistic gains. In fact, what we expect from a good restoration is to achieve the opposing situation. In this context, the study conducted by Slik et al. (2008) showed that forest sites that contain species of high wood density may present low floristic diversity due to dominance features, but still harboring many endemic species. Therefore, we suggest that the choice of the set of species to be introduced in restoration plantings should embrace at least the three restorations demands proposed (plant conservation, community functioning and ecosystem services), to ensure the success of the restoration plantings and to achieve the expected targets.

Finally, we must highlight the presence of exotic species in the restoration plantings. This information is relevant not only because of the presence of exotic species in the species pool, but because these species are also included in more than one priority class. Thus, *Cordia myxa* L., *Eriobotrya japonica* (Thunb.) Lindl., and *Syzygium cumini* (L.) Skeels, are species included among the priority ones due to their potential for carbon accumulation and/or zoochoric dispersion

and/or high wood density. However, since these are properties related to the performance and function of the species within the communities (Reid et al., 2015; Charles et al., 2018), monitoring their populations is necessary to avoid negative effects on the successional dynamics (Assis et al., 2013).

4.6. CONCLUSIONS

Our study reveals that the restoration plantings contribute to the regional plant diversity in highly anthropized landscapes, but, individually, these communities present low species richness compared to natural forests, even after 30 years. Although we observed the occurrence of species classified as threatened, endemic, hardwood, zoochoric, and of high potential for carbon ES, the results show that the species richness within the different priority classes does not change over time. In addition, zoochoric and hardwood species were the only classes in which we observed a positive relationship between the density of individuals and the restorations age. Despite the fact that restoration planting can complement “*in situ*” conservation practiced in protected areas, the opportunity to generate synergistic outcomes through the plantings is not being fully exploited. We provide evidence that selecting species for planting focusing on a single objective can produce a tradeoff relationship among the different demands of the restorations. Our study highlights the contribution of restoration plantings in sustaining floristic diversity in agricultural landscapes, but provide evidence that the contribution of these forests to the conservation of priority plant species and to multiple gains could be improved by better planning the selection and the management of the plant species.

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Final Considerations

The results reveal the high potential of Brazilian forest restorations (passive and active) in contributing to carbon ES, constituting an important tool to offset national carbon emission and mitigate the effects of climate change. Despite this, several limitations and the lack of standardization of the estimates, affect the power of generalization of the results obtained in different studies. Among these limitations, we observed biases related to the geographic distribution of biomass studies - which predominates in the Amazon, as well as in relation to the restoration method - where passive restoration (natural regeneration) predominates, promoting the disproportionality in the accumulated knowledge.

In this context, we attempt to contribute to a robust evaluation of the benefits provided by Brazilian restorations, more specifically concerning active restorations implemented in anthropized regions. Thus, in agricultural landscapes of the São Paulo state countryside, we observed that forest restoration plantings aging from 5 to 30 years old, present a linear pattern of carbon accumulation over time. This means that these communities sustain high productivity even after 30 years. We estimate that 30y forest restoration plantings can store carbon similar to preserved native forests, evidencing the relevant role of the restorations for the supply of climate regulating ES. However, we highlight that the changes occurring over time in both the structure and the functional aspects of the communities are directly related to the ecological process involved in the carbon accumulation. Over time, individuals of $DBH \geq 30$ cm and secondary-successional species hold higher amounts of aboveground carbon stock of the restorations, succeeding small plants and pioneer species. This is indicative that the forest structure recovers earlier than functional aspects along with the restoration trajectory. These findings reflect the

restorations' progress, indicating not only the importance of plant growth to carbon accumulation, but also the turnover on the ecological groups. Despite that, the carbon stored in small individuals and late-successional species responds positively to the restorations' age. Even that these plants did not present a leading role in the carbon stock, they hold the potential of the communities in sustaining, over time, the successional processes that afford stability to the supply of carbon ES. However, it is also important to highlight that early-successional species (pioneer and secondary) are dominant in richness, abundance, and carbon stock among the restorations.

Concerning the multiple benefits expected for restoration projects, our study shows that restoration plantings contribute to the promotion of floristic diversity in agricultural landscapes. However, the potential of these communities in providing synergistic outcomes by harboring priority species for conservation programs, ecosystems functioning, and the supply of carbon ES has not been adequately employed. For example, we observed a scarce presence of plant species threatened with extinction. The only priority class presenting expressive species richness and plant abundance is related to species that promote plant-animal interactions, a promising result for the restorations. Furthermore, despite the opportunity of selecting and introducing key species in the restoration plantings, we identified few species sharing more than one priority class, that is, species that contribute to the multifunctional nature of the ecological restoration. As a contribution, we provided a list of priority species for forest restoration, as well as recommendations for the maintenance of these species in restoration plantings embedded in agricultural landscapes.

In summary, we found that the dynamic established in the restoration plantings is similar to that observed in naturally regenerated forests, leading the communities towards successional progress and restoration success, at least for carbon ES. However, not all that glitters is gold, and the results indicate the need for improvement so that the restoration plantings could contribute to the different demands of the restoration programs. We hope our study assists decision-makers in better planning restoration plantings, increasing the opportunities for co-benefits generation.

Supplementary Material

Supplementary Material A (Chapter 1)

The information disclosed in this section refer to the database used to generate the results of our study. The four tables contain information related to: the publications details (institutions involved with the studies and journal of publication - Table A.1); sites description (site's age and biome, past land-use, and restoration method – Table A.2); methodological criteria selected to estimate aboveground biomass/carbon (life forms evaluated, adopted plot size and total area sample per site, criteria of plant individuals' inclusion, wood specific gravity (or wood density) usage, carbon conversion, and factors of influence – Table A.3); and the allometric equations used among the selected articles (Table A.4). Also, we present two figures related to the screening process and articles selection (Figure A.1), and (Figure A.2) the number of publication per scientific journal.

Table A.1. Publication's details collected in the articles included in the final list

Publication detail		
Authors	Author's institutions	Journal
Almeida et al., 2019	University of São Paulo; Michigan State University; University of Connecticut; National Amazonia Research Institute; Federal University of Vales do Jequitinhonha e Mucuri; University of Cambridge; NASA	FOREST ECOLOGY AND MANAGEMENT
Berenguer et al., 2014	Museu Paraense Emílio Goeldi; EMBRAPA; University of São Paulo; National Institute for Space Research; University of Exeter; Stockholm Environment Institute; International Institute for Sustainability; University of Cambridge; Lancaster University	GLOBAL CHANGE BIOLOGY
César et al., 2018	University of São Paulo; University of Campinas; University of Connecticut	ECOLOGICAL APPLICATIONS
Costa et al., 2014	Associação Plantas do Nordeste; EMBRAPA; Federal University of Pernambuco; Federal Rural University of Pernambuco	PLANT AND SOIL
da Silva et al., 2014	National Institute for Space Research	GISCIENCE & REMOTE SENSING
da Silva et al., 2017	State University of Maranhão; National Amazonia Research Institute	FOREST ECOLOGY AND MANAGEMENT
de Aguiar et al., 2013	EMBRAPA; State University of Ceará; Instituto Federal de Educação Ciência e Tecnologia do Piauí; Federal University of Ceará	AGROFORESTRY SYSTEMS
de Azevedo et al., 2019	Federal Rural University of Rio de Janeiro; Federal University of Viçosa	FLORESTA
de Lima et al., 2018	Federal Rural University of Pernambuco; State University of Amapa; Federal University of Alagoas	SCIENTIA FORESTALIS
de Melo and Durigan, 2006	Instituto Florestal	SCIENTIA FORESTALIS
de Souza et al., 2011	Celulose Nipo Brasileira; Federal University of Viçosa	ARVORE

de Souza et al., 2012	Federal Rural University of Pernambuco; Federal University of Pernambuco	NUTRIENT CYCLING IN AGROECOSYSTEMS
Durigan et al., 2016	Instituto Florestal	SCIENTIA FORESTALIS
Fearnside et al., 2007	National Amazonia Research Institute	FOREST ECOLOGY AND MANAGEMENT
Feldpausch et al., 2004	EMBRAPA; Centro Internacional de Agricultura Tropical; Cornell University	ECOLOGICAL APPLICATIONS
Feldpausch et al., 2005	Cornell University	EARTH INTERACTIONS
Feldpausch et al., 2007	The World Bank; University of Bristol; University of California; Cornell University	GLOBAL CHANGE BIOLOGY
Ferreira and Prance, 1999	Royal Botanic Gardens; National Amazonia Research Institute	BOTANICAL JOURNAL OF THE LINNEAN SOCIETY
Ferreira et al., 2015	Federal University of Rio Grande do Norte	OCEAN & COASTAL MANAGEMENT
Ferreira et al., 2019	Federal University of Ceara	WETLANDS ECOLOGY AND MANAGEMENT
Freitas et al., 2019	University of Brasilia; EMBRAPA - Brazilian Agricultural Research Corporation; Instituto Socioambiental	FOREST ECOLOGY AND MANAGEMENT
Froufe et al., 2019	EMBRAPA - Brazilian Agricultural Research Corporation; Federal University of Parana; Pontifical Catholic University of Parana; Chico Mendes Biodiversity Institute - ICMBio	AGROFORESTRY SYSTEMS
Galvao et al., 2015	Federal University of Santa Maria; National Institute for Space Research	INTERNATIONAL JOURNAL OF REMOTE SENSING
Gama-Rodrigues et al., 2003	Federal University of Viçosa; State University of North Fluminense	REVISTA BRASILEIRA DE CIENCIA DO SOLO
Gama-Rodrigues et al., 2007	University of Florida; Federal University of Viçosa; State University of North Fluminense	REVISTA BRASILEIRA DE CIENCIA DO SOLO
Gehring et al., 2005	University of Bonn; State University of Maranhão	JOURNAL OF TROPICAL ECOLOGY
Imaña-Encinas et al., 2012	Federal University of Mato Grosso do Sul; University of Brasília	FLORESTA

Lindner and Sattler 2012	University of Leipzig	NEW FORESTS
Lu et al., 2002	Indiana State University; Indiana University	LAND DEGRADATION & DEVELOPMENT
Lu et al., 2003	Indiana State University; Indiana University	FOREST ECOLOGY AND MANAGEMENT
Luckman et al., 1997	Federal University of Pernambuco; National Institute for Space Research; British National Space Centre	REMOTE SENSING OF ENVIRONMENT
Markewitz et al., 2004	Instituto de Pesquisa Ambiental da Amazônia (IPAM); The Woods Hole Research Center	ECOLOGICAL APPLICATIONS
Moran et al., 2000	EMBRAPA; University of Arizona; Indiana University	FOREST ECOLOGY AND MANAGEMENT
Moura et al., 2016	Federal University of Campina Grande; Federal University of Pernambuco	NUTRIENT CYCLING IN AGROECOSYSTEMS
Peixoto et al 2017	University of Oxford; Federal University of Acre; State University of Mato Grosso; Rede de Biodiversidade e Biotecnologia da Amazônia Legal	ACTA OECOLOGICA-INTERNATIONAL JOURNAL OF ECOLOGY
Pereira et al., 2016	University of Boston; Instituto Federal de Educação do Ceará; Federal University of Ceará	REVISTA CIENCIA AGRONOMICA
Robinson et al., 2015	Federal University of Lavras; Lancaster University	BIODIVERSITY AND CONSERVATION
Rosenfield and Müller, 2019a	Federal University of Rio Grande do Sul	RESTORATION ECOLOGY
Rosenfield and Müller, 2019b	Federal University of Rio Grande do Sul	ECOSYSTEMS
Sansevero et al., 2017	Federal Rural University of Rio de Janeiro; Instituto de Pesquisas Jardim Botânico do Rio de Janeiro	NEW FORESTS
Santos et al., 2003	National Institute for Space Research	REMOTE SENSING OF ENVIRONMENT
Sattler et al., 2014	Technische Universität Dresden; Winrock International; University of Leipzig	ECOLOGICAL ENGINEERING
Siddique et al., 2010	The Woods Hole Research Center; EMBRAPA; Museu Paraense Emílio Goeldi; University of Queensland	ECOLOGY

Silva et al., 2015	Federal University of Lavras; Federal University of Acre; Federal University of Piauí	SCIENTIA FORESTALIS
Silva et al., 2016	National Institute for Space Research	ACTA AMAZONICA
Steininger 2000	University of Maryland	JOURNAL OF TROPICAL ECOLOGY
Toledo et al., 2018	University of São Paulo; Ghent University; University of Western Australia;	APPLIED VEGETATION SCIENCE
Uhl et al., 1988	EMBRAPA; University of Georgia	JOURNAL OF ECOLOGY
Vendrami et al., 2012	Federal University of Rio Grande do Sul; University of São Paulo	BIOTA NEOTROPICA
Vieira et al., 2003	Instituto de Pesquisa Ambiental da Amazônia (IPAM); EMBRAPA; The Woods Hole Research Center; Museu Paraense Emílio Goeldi	REMOTE SENSING OF ENVIRONMENT
Wandelli and Fearnside, 2015	National Amazonia Research Institute; EMBRAPA	FOREST ECOLOGY AND MANAGEMENT

Table A.2. Site's information collected in the articles included in the final list

Site information					
Authors	Restoration method	Biome	Age	n° evaluated areas	Past land-use
Almeida et al., 2019	passive	Atlantic Forest and Cerrado	11-45y	19	pasture; silviculture
Berenguer et al., 2014	passive	Amazon	6-22y	50	clear cut
César et al., 2018	active; passive	Atlantic Forest and Cerrado	7-20y	28	pasture; silviculture; agriculture
Costa et al., 2014	passive	Caatinga	18-60y	9	agriculture + pasture; agriculture; clear cut
da Silva et al., 2014	passive	Amazon	<5 to >15y	28	NA
da Silva et al., 2017	passive	Amazon	5-10y	10	NA
de Aguiar et al., 2013	passive	Caatinga	6 and 9y	2	agriculture + pasture
de Azevedo et al., 2019	active	Atlantic Forest	3-7y	3	NA
de Lima et al., 2018	passive	Caatinga	2y	1	clear cut
de Melo and Durigan, 2006	active	Atlantic Forest and Cerrado	1-28y	25	agriculture + pasture
de Souza et al., 2011	passive	Atlantic Forest	20y	2	NA
de Souza et al., 2012	passive	Caatinga	16 and 38y	2	agriculture
Durigan et al., 2016	active	Atlantic Forest and Cerrado	4-53y	26	NA
Fearnside et al., 2007	passive	Amazon	6y	1	agriculture + burn
Feldpausch et al., 2004	passive	Amazon	0-14y	10	pasture

Feldpausch et al., 2005	passive	Amazon	0-14y	10	pasture
Feldpausch et al., 2007	passive	Amazon	1-14y	10	pasture
Ferreira and Prance, 1999	passive	Amazon	40y	3	agriculture + burn
Ferreira et al., 2015	active; passive	Atlantic Forest	5y	2	clear cut
Ferreira et al., 2019	active; passive	Atlantic Forest	10;10	2	shrimp production
Freitas et al., 2019	active; passive	Amazon	1-10y	81	NA
Froufe et al., 2019	passive	Atlantic Forest	10y	1	agriculture + pasture + burn
Galvao et al., 2015	passive	Amazon	30y	1	agriculture
Gama-Rodrigues et al., 2003	passive	Atlantic Forest	40y	1	NA
Gama-Rodrigues et al., 2007	passive	Atlantic Forest	40y	1	agriculture + pasture + burn
Gehring et al., 2005	passive	Amazon	2-25y	16	agriculture + burn
Imaña-Encinas et al., 2012	passive	Atlantic Forest	33y	1	NA
Lindner and Sattler 2012	passive	Atlantic Forest	60y	1	clear cut
Lu et al., 2002	passive	Amazon	NA	20	NA
Lu et al., 2003	passive	Amazon	1 to >30y	52	NA
Luckman et al., 1997	passive	Amazon	2-22y	13	NA
Markewitz et al., 2004	passive	Amazon	19y	1	pasture + burn
Moran et al., 2000	passive	Amazon	0 to >10y	20	NA
Moura et al., 2016	passive	Caatinga	1-57y	12	agriculture + pasture; clear cut + agriculture
Peixoto et al 2017	passive	Cerrado	22y	1	pasture

Pereira et al., 2016	passive	Caatinga	30y	1	clear cut
Robinson et al., 2015	passive	Atlantic Forest	30y	2	pasture
Rosenfield and Müller, 2019a	active	Atlantic Forest	8-12y	2	silviculture; agriculture
Rosenfield and Müller, 2019b	active	Atlantic Forest	8-12y	2	silviculture; agriculture
Sansevero et al., 2017	passive	Atlantic Forest	15 and 36y	2	pasture; pasture + burn
Santos et al., 2003	passive	Amazon	<5 to >15y	12	NA
Sattler et al., 2014	active	Atlantic Forest	4y	18	pasture
Siddique et al., 2010	passive	Amazon	15y	1	pasture + burn
Silva et al., 2015	active	Atlantic Forest	21y	1	area degraded by hydropower dam construction
Silva et al., 2016	passive	Amazon	<5-15y	3	NA
Steininger 2000	passive	Amazon	4-30y	20	clear cut; agriculture + burn; agriculture + burn + pasture
Toledo et al., 2018	active	Atlantic Forest	7y	32	pasture
Uhl et al., 1988	passive	Amazon	1.5-8y	13	pasture + burn
Vendrami et al., 2012	passive	Atlantic Forest	25 and >=65y	6	NA
Vieira et al., 2003	passive	Amazon	3 - 70y	12	agriculture + burn
Wandelli and Fearnside, 2015	passive	Amazon	1-15y	24	pasture; agriculture + burn

Table A.3. Methods used to estimate aboveground biomass/carbon (plant's aerial parts) and factors of influence observed among studies

Evaluation methods								Factors of influence on AGB recovery
Authors	Life forms evaluated	Allometric equation reference	Plot size (m ²)	Total area sampled/site (m ²)	Criteria of plant individuals' inclusion - diameter (D; centimeters) and height (h; meters)	Wood density	Carbon conversion factor	
Almeida et al., 2019	all plants	Chave et al., 2014	900	900; 1800; 2700; 3600	$D \geq 5$	yes	no	NA
Berenguer et al., 2014	trees, palms, lianas and litter	NA	2500	NA	NA	no	0.5	past land-use; landscape; soil
César et al., 2018	trees	Chave et al., 2014 (impossible to identify the exactly equation used in the article)	900	900	$D \geq 1$	yes	no	past land-use; restoration implementation
Costa et al., 2014	all plants	Sampaio and Silva, 2005	28.27	113.8	$D \geq 3$	no	no	forest attribute; soil
da Silva et al., 2014	trees	Uhl et al., 1988	1000; 2500	10000; 11000; 13000	$D \geq 5$	yes (constant 0.69 g/cm ³)	no	NA
da Silva et al., 2017	trees, shrubs, bamboo, lianas, herbs and grasses	Chave et al., 2005; Dulcey et al., 2009	330; 500	330; 500	$h > 1$	yes (constant 0.583 g/cm ³ for	no	soil

						unknown species)		
de Aguiar et al., 2013	trees, woody shrub and herbs	Silva and Sampaio, 2008	100	700	$D > 3$	no	no	forest attribute
de Azevedo et al., 2019	trees and litter	Brown, 1997	600	2400; 3600; 6000	$D \geq 3.2$	no	0.5	NA
de Lima et al., 2018	trees, shrubs	Silva and Sampaio, 2008	400	7200	$D \geq 0.3$	no	no	NA
de Melo and Durigan, 2006	trees	Brown, 1997	NA	NA	$h \geq 1.3$	no	0.5	soil
de Souza et al., 2011	trees	Based on volume and wood density - Soares et. al. 2011	500	3000; 8000	$D \geq 5$	yes	0.5	NA
de Souza et al., 2012	woody plants leaf	Silva and Sampaio, 2008	1000	1000	NA	no	no	Forest attribute
Durigan et al., 2016	trees	Brown et al., 1989 (impossible to identify the exactly equation used in the article)	225	2250	$D \geq 5$	no	no	forest attribute
Fearnside et al., 2007	all plants	Destructive (direct weighing) - Fearnside et al., 2007	60	360	NA	no	destructive	NA
Feldpausch et al., 2004	trees	Nelson et al., 1999 (impossible to identify the exactly equation used in the article)	35; 225	NA	$D \geq 5$	no	destructive	forest attribute
Feldpausch et al., 2005	trees, shrubs and lianas	Nelson et al., 1999	20; 35; 100	240; 420; 1200	$D \geq 1$	no	no	forest attribute

Feldpausch et al., 2007	trees	Nelson et al., 1999; Mesquita (unpublished data)	180; 300; 315; 675	180; 300; 315; 675	$D \geq 1$	no	no	Forest attribute
Ferreira and Prance, 1999	trees	dos Santos, 1996	400	10000	$D > 10$	no	no	plot location; forest attribute
Ferreira et al., 2015	trees	Medeiros and Sampaio 2008; Ferreira et al., 2015	25	250	$D \geq 2; h \geq 1.8$	no	no	restoration implementation
Ferreira et al., 2019	trees	Ferreira et al., 2015	25	250	NA	no	no	forest attribute, restoration implementation, plant interaction
Freitas et al., 2019	trees and saplings	Chave et al., 2014	100; 500	100; 500	$h > 1.30$	yes	0.5	forest attribute, soil, landscape, climate
Froufe et al., 2019	litter	NA	NA	NA	NA	no	no	NA
Galvao et al., 2015	trees	Uhl et al., 1988	2500	5000	$D > 10$	yes (constant 0.69 g/cm ³)	no	NA
Gama-Rodrigues et al., 2003	litter	NA	NA	NA	NA	no	no	NA
Gama-Rodrigues et al., 2007	litter	NA	NA	NA	NA	no	no	NA
Gehring et al., 2005	trees, lianas and palms	Nelson et al., 1999 (impossible to identify the exactly equation used in the article)	25; 56.25; 100	75; 300; 400; 425; 450; 506.25; 562.5; 600; 787.5; 956.25; 1000; 1500; 1518.75; 2100	$D > 0.2$	no	no	past land-use; forest attribute

Imaña-Encinas et al., 2012	trees and palms	Based on volume and wood density - Imaña-Encinas et al., 2012	400	2400	$D > 5$	yes	no	NA
Lindner and Sattler, 2012	trees and palms	Based on volume, biomass expansion factor and wood density- Garzuglia and Saket, 2003; Chave et al., 2011	400	400	$D \geq 10$	yes (constant 0.603 g/cm ³)	no	past land-use
Lu et al., 2002	trees and saplings	Overman et al., 1994; Nelson et al., 1999	150	1500	$D \geq 2$	no	no	soil
Lu et al., 2003	trees and saplings	Overman et al., 1994; Nelson et al., 1999	150	1500	$D \geq 2$	no	no	forest attribute
Luckman et al., 1997	trees	Deans, 1993	100; 500	100; 500	$D > 3$	no	no	NA
Markewitz et al., 2004	trees and litter	Uhl et al., 1988 (impossible to identify the exactly equation used in the article)	500; 750	500; 750	NA	no	0.5	NA
Moran et al., 2000	trees and saplings	Uhl et al., 1988; Brown et al., 1989 (impossible to identify the exactly equation used in the article)	150	750; 1500	$D \geq 2$	no	no	forest attribute
Moura et al., 2016	litter	NA	NA	NA	NA	no	0.45	NA
Peixoto et al 2017	trees and litter	Rezende et al., 2006	100	5000	$D \geq 5$	no	no	NA

Pereira et al., 2016	woody species, herbs and litter	Sampaio and Silva, 2005	100	10000	$D > 9; h > 1$	no	destructive	life-form
Robinson et al., 2015	trees and shrubs	Chave et al., 2014	400	2000	$D \geq 3$	yes	0.5	landscape; soil; forest attribute
Rosenfield and Müller, 2019a	litter	NA	NA	NA	NA	no	no	NA
Rosenfield and Müller, 2019b	trees and litter	Burger and Delitti, 2008	100	1500	$D \geq 5$	no	no	forest attribute
Sansevero et al., 2017	trees and palms	Chave et al., 2014	100	300	$D \geq 5$	yes	no	past land-use
Santos et al., 2003	trees	Uhl et al., 1988	1000	1000	$D > 5$	no	no	forest attribute
Sattler et al., 2014	trees	Brown and Iverson, 1992	100	100	$D \geq 1$	no	0.47	landscape
Siddique et al., 2010	trees	Uhl et al., 1988	400	1600	$D \geq 2$	no	no	restoration implementation
Silva et al., 2015	trees	Scolforo et al., 2008	400	2400	$D \geq 5$	yes	destructive	calculating method (equation)
Silva et al., 2016	trees	Uhl et al., 1988	1000; 2500	10000; 11000; 13000	$D \geq 5$	yes (constant 0.69 g/cm ³)	no	forest attribute
Steininger, 2000	trees	Saldarriaga et al., 1988; Scatena et al., 1993	100; 700	100; 700	$D > 5$	yes	no	forest attribute
Toledo et al., 2018	trees	Chave et al., 2014	200	800	$D \geq 5$	yes	no	soil, landscape

Uhl et al., 1988	trees, shrubs, vines, herbs and grasses	Uhl et al., 1988	100	100	$h \geq 2$	yes	no	past land-use
Vendrami et al., 2012	litter	NA	NA	NA	NA	no	no	forest attribute
Vieira et al., 2003	all plants	Uhl et al., 1988	250	500	$D \geq 5$	no	no	forest attribute
Wandelli and Fearnside, 2015	trees, shrubs and lianas	Wandelli and Fearnside, 2015	100	100	$D \geq 1$	no	no	past land-use; forest attribute

Table A.4. Base information related to the allometric equations used among articles to estimate AGB

Allometric equations base information										
Article reference	Equation reference	Equation	Equation type	Plant's size criteria	Equation constructed from	Sample size (n individuals)	Number of species	Forest type	Continent or country	Brazilian biome
Almeida et al., 2019	Chave et al., 2014	$AGB = \exp(-2.024 - 0.896 * E + 0.920 * \ln(p) + 2.795 * \ln(D) - 0.0461 * (\ln(D))^2)$	mixed species	D = 5-212	trees	4004	NA	Tropical and Subtropical forests, and Woodland Savannas	America, Africa, Asia, Europe	Amazon, edges Cerrado/ Amazon, Atlantic Forest
Berenguer et al., 2014	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
César et al., 2018	Chave et al., 2014	NA	mixed species	NA	trees	NA	NA	Tropical and Subtropical Forests, and Woodland Savannas	America, Africa, Asia, Europe	Amazon, edges Cerrado/ Amazon, Atlantic Forest
Costa et al., 2014	Sampaio and Silva, 2005	$AGB = 0.173 D^{2.295}$	mixed species	$D \geq 3; H > 1$	trees, shrub, Cactaceae	NA	9	Caatinga	Brazil	Caatinga

da Silva et al., 2014; Galvao et al., 2015; Santos et al., 2003; Siddique et al., 2010; Silva et al., 2016; Vieira et al., 2003; Uhl et al., 1988;	Uhl et al., 1988	$\ln AGB = -2.17 + 1.02 \ln D + 0.39 \ln H$	mixed species	$H \geq 2$	trees	30	NA	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
da Silva et al., 2017	Chave et al., 2005	$AGB = \exp(-2.997 + \ln(pD^2H))$	mixed species	$D = 5-156$	trees	2410	NA	Secondary and old growth Tropical Forests (dry, moist and wet and mangrove forests)	Cross tropics (America, Asia, Oceania)	Amazon
da Silva et al., 2017	Ducey et al., 2009	$AGB = 0.18598 * DBH^{2.3155}$	mixed species	$D \geq 1$	trees	82	26	Secondary Amazonian forests (abandoned for 15y)	Brazil	Amazon
da Silva et al., 2017	Gehring et al., 2011	$AGB = 95.1 + 49.68 * H$	life form	$H = 1.9-17.7$	palms	25	1	Secondary Amazonian forests (2 to >40y)	Brazil	Amazon

da Silva et al., 2017; Uhl et al., 1988	Uhl et al., 1988	$\ln\text{AGB} = -3.78 + 0.95 \ln D + 1.00 \ln H$	genus	$H \geq 2$	Cecropia	16	1	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
da Silva et al., 2017; Wandelli and Fearnside, 2015	Gehring et al., 2004	$\ln\text{AGB} = 7.114 + 2.276 \ln(D)$	life form	$D = 0.1-13.8$	lianas	439	26	Secondary and primary Amazonian forests (primary forests and 2-25y secondary forests)	Brazil	Amazon
de Aguiar et al., 2013	Silva and Sampaio, 2008	$\text{AGB} = 0.2368 * D^{2.219}$ (for large sized species)	mixed species	$D = 3-30$	plants	NA	6	Caatinga	Brazil	Caatinga
de Aguiar et al., 2013; de Lima et al., 2018	Silva and Sampaio, 2008	$\text{AGB} = 0.2627 * D^{1.9010}$ (for small sized species)	mixed species	$D = 3-30$	plants	NA	3	Caatinga	Brazil	Caatinga
de Azevedo et al., 2018	Brown, 1997	$\text{AGB} = \exp(-2.134 + 2.53 * \ln\text{DAP})$	mixed species	$D = 5-148$	trees	170	NA	Moist tropical forest	Tropical Forest	NA
de Melo and Durigan, 2006	Brown, 1997	$\text{AGB} = \exp\{-1.996 + 2.32 * \ln(\text{DAP})\}$	mixed species	$D = 5-40$	trees	28	NA	Dry forest	India	NA

de Souza et al., 2011	Soares et al., 2011	Based on volume and wood density	mixed species	$D \geq 5$	trees	NA	NA	Seasonal Semideciduous Forest	Brazil	Atlantic Forest
de Souza et al., 2012	Silva and Sampaio, 2008	Leaves biomass = $0.0681 \cdot D^{1.5829}$ (for large sized species)	specific leaf	$D = 3-30$	plants leaf	NA	6	Caatinga	Brazil	Caatinga
de Souza et al., 2012	Silva and Sampaio, 2008	Leaves biomass = $0.1900 \cdot D^{2.0515}$ (for small sized species)	specific leaf	$D = 3-30$	plants leaf	NA	3	Caatinga	Brazil	Caatinga
Durigan et al., 2016	Brown et al., 1989	NA	mixed species	NA	NA	NA	NA	NA	NA	NA
Fearnside et al., 2007	Fearnside et al., 2007	Destructive (direct weighing)	mixed species	NA	All plants	NA	NA	Secondary Amazonian forest	Brazil	Amazon
Feldpausch et al., 2004; Feldpausch et al., 2005; Feldpausch et al., 2007; Gehring et al., 2005	Nelson et al., 1999	$\ln AGB = 1.9968 + 2.4128 \ln(D)$	mixed species	$D = 1.2-28.6$	trees	132	7	Secondary Amazonian forests (8-15y)	Brazil	Amazon
Feldpausch et al., 2004; Feldpausch et al., 2007	Nelson et al., 1999	$\ln AGB = -2.3706 + 2.5392 \ln(D)$	specific	$D = 1.6-21.8$	<i>Vismia cayanensis</i>	20	1	Secondary Amazonian forest	Brazil	Amazon

Feldpausch et al., 2004; Feldpausch et al., 2007	Nelson et al., 1999	$\ln\text{AGB} = -1.7972 + 2.4206 \ln(D)$	specific	D = 1.5-12.2	Goupia glabia	17	1	Secondary Amazonian forest	Brazil	Amazon
Feldpausch et al., 2004; Feldpausch et al., 2007	Nelson et al., 1999	$\ln\text{AGB} = -2.224 + 2.5105 \ln(D)$	specific	D = 1.6-24.8	Laetia procera	22	1	Secondary Amazonian forest	Brazil	Amazon
Feldpausch et al., 2004; Feldpausch et al., 2007	Nelson et al., 1999	$\ln\text{AGB} = -1.7829 + 2.3651 \ln(D)$	specific	D = 1.5-28.6	Vismia japurensis	23	1	Secondary Amazonian forest	Brazil	Amazon
Feldpausch et al., 2004; Feldpausch et al., 2007; Robinson et al., 2015; Silva et al., 2016	Nelson et al., 1999	$\text{AGB} = \exp(-2.5118 + 2.4257 \ln D)$	specific	D = 5.1-38.2	Cecropia sciadophylla	27	1	Secondary Amazonian forest	Brazil	Amazon
Feldpausch et al., 2005	Putz, 1983	$\log\text{AGB} = 0.12 + 0.91 \log \text{Basal area}$	life form	D = 0.9-12	lianas	17	12	Evergreen Terra Firme Forest	Venezuela	NA
Feldpausch et al., 2007	Nelson et al., 1999	$\ln\text{AGB} = -1.8158 + 2.37 \ln(D)$	genus	D = 1.2-26.8	Bellucia	27	NA	Secondary Amazonian forest	Brazil	Amazon

Ferreira and Prance, 1999	dos Santos, 1996	$AGB = (\exp\{3.232 + \ln[2.546(D/100)]\}) * 600$	mixed species	D = 5-120	trees	319	NA	Amazonian Lowland Terra Firme forest	Brazil	Amazon
Ferreira et al., 2015	Medeiros and Sampaio, 2008	$AGB = 0.1442 * D^{2.325}$	specific	D = 2.1-17.8; H = 1.6-11.8	Laguncularia racemosa	35	1	Secondary Atlantic Forest (5y Restored Mangroove)	Brazil	Atlantic Forest
Ferreira et al., 2015	Medeiros and Sampaio, 2008	$AGB = 0.2938 * D^{2.384}$	specific	D = 2.5-20.7; H = 1.8-14	Rhizophora mangle	36	1	Secondary Atlantic Forest (5y Restored Mangroove)	Brazil	Atlantic Forest
Ferreira et al., 2015; Ferreira et al., 2019	Ferreira et al., 2015	$AGB = 427.26(D^2) - 544.45(D) + 994.63$	specific	H ≥ 1.8	Rhizophora mangle	10	1	Secondary Atlantic Forest (5y Restored Mangroove)	Brazil	Atlantic Forest
Ferreira et al., 2015; Ferreira et al., 2019	Ferreira et al., 2015	$AGB = 299.43(D^2) - 486.06(D) + 393.04$	specific	H ≥ 1.8	Laguncularia racemosa	10	1	Secondary Atlantic Forest (5y Restored Mangroove)	Brazil	Atlantic Forest

Imaña-Encinas et al., 2012	Imaña-Encinas et al., 2012	Based on volume and wood density	mixed species	$D > 5$	trees and palms	195	17	Ombrophilous Dense	Brazil	Atlantic Forest
Lindner and Sattler, 2012	Chave et al., 2001	$AGB = -2.19 + 2.54 \ln(D)$	mixed species	$D = 10-150$	trees	105	NA	Lowland Wet Tropical Forest	New Guinea, Puerto Rico, Indonesia, Costa Rica, Brazil and Cambodia	Amazon
Lindner and Sattler, 2012	Garzuglia and Saket, 2003	Based on volume, biomass expansion factor and wood density	mixed species	$D > 10$	trees	NA	NA	many	Africa, America, Oceania, Europe, Asia	NA
Lindner and Sattler, 2012	Hughes et al., 1999 modified by Vieira et al., 2008	$AGB = \exp(((5.7236+09285 \ln(D^2))*1.05001)/10^3)$	life form	NA	palms	15	1	Secondary Evergreen forests (0.5-50y)	Mexico	NA
Lu et al., 2002; Lu et al., 2003	Nelson et al., 1999	$\ln AGB = 2.5202 + 2.1400 \ln D + 0.4644 \ln H$	mixed species	$D = 1.2-28.6$	trees	132	7	Secondary Amazonian forests (8-15y)	Brazil	Amazon

Lu et al., 2002; Lu et al., 2003	Overman et al., 1994	$\ln \text{AGB} = -3.843 + 1.035 \ln(D^2 * H)$	mixed species	$D = 8-100$	trees	54	NA	Amazonian mature Lowland Terra Firme Forest	Colombia	NA
Luckman et al., 1997	Deans et al., 1996	$\text{AGB} = 0.40 + 0.0406 * (D^2) * H * p$	mixed species	$D \geq 15$	trees	14	5	Semideciduous Moist Forest	Cameroon	NA
Luckman et al., 1997	Uhl et al., 1988	$\text{AGB} = 0.0298 * (D^2)^{0.950} * H$	genus	$H \geq 2$	Cecropia	16	NA	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
Luckman et al., 1997	Uhl et al., 1988	$\text{AGB} = 0.0290 * (D^2)^{1.130} * H^{0.770}$	specific	$H \geq 2$	Vismia guianensis	10	1	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
Markewitz et al., 2004	Markewitz et al., 2004	NA	NA	Inga	NA	1	Amazonian forest	Amazon	Brazil	genus
Markewitz et al., 2004	Markewitz et al., 2004	NA	NA	Ocotea	NA	1	Amazonian forest	Amazon	Brazil	genus

Markewitz et al., 2004	Uhl et al., 1988	NA	mixed species	$H \geq 2$	trees	NA	NA	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
Moran et al., 2000	Brown et al., 1989	NA	mixed species	NA	NA	NA	NA	NA	NA	NA
Moran et al., 2000	Uhl et al., 1988	NA	mixed species	$H \geq 2$	trees	NA	NA	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
Peixoto et al., 2017	Rezende et al., 2006	$AGB = -0.4913 + 0.0291 (Db^2 * H)$	mixed species	$D = 5-55$	trees	174	49	Cerrado sensu stricto	Brazil	Cerrado
Pereira et al., 2016	Sampaio and Silva, 2005	$AGB = 0.0292 * (DB * H)^{1.6371}$	mixed species	$D \geq 3; H > 1$	trees, shrub, Cactaceae	270	9	Caatinga	Brazil	Caatinga
Pereira et al., 2016	Sampaio and Silva, 2005	$AGB = 0.2455 * D^{1.7726}$	specific	$D \geq 3; H > 1$	Aspidosperm a pyriformis	NA	1	Caatinga	Brazil	Caatinga
Pereira et al., 2016	Sampaio and Silva, 2005	$AGB = 0.1482 * D^{1.8741}$	specific	$D \geq 3; H > 1$	Croton sonderianus	NA	1	Caatinga	Brazil	Caatinga

Pereira et al., 2016	Sampaio and Silva, 2005	$AGB = 0.0689 * D^{2.0557}$	specific	$D \geq 3; H > 1$	<i>Jatropha mollissima</i>	NA	1	Caatinga	Brazil	Caatinga
Pereira et al., 2016	Sampaio and Silva, 2005	$AGB = 0.0596 * D^{2.4878}$	specific	$D \geq 3; H > 1$	<i>Myracrodruon urundeuva</i>	NA	1	Caatinga	Brazil	Caatinga
Robinson et al., 2015	Robinson et al., 2015	$AGB = 0.605 + (0.088 * \text{Crown area})$	life form	NA	shrubs	40	NA	Atlantic Forest	Brazil	Atlantic Forest
Robinson et al., 2015	Saldarriaga et al., 1988	$\ln AGB = -6.3789 - 0.877 \ln(1/D^2) + 2.151 \ln(H)$	life form	NA	palms	19	NA	Amazonian forests	Colombia and Venezuela	NA
Robinson et al., 2015; Sansevero et al., 2017; Toledo et al., 2018; Freitas et al., 2019	Chave et al., 2014	$AGB = 0.0673 * (pD^2H)^{0.976}$	mixed species	$D = 5-212$	trees	4004	NA	Tropical and Subtropical forests, and Woodland Savannahs	America, Africa, Asia, Europe	Amazon, edges Cerrado/ Amazon, Atlantic Forest
Rosenfield and Müller, 2019b	Burger and Delitti, 2008	$\ln(AGB) = -3.676 + 0.951(\ln(D^2H))$	mixed species	$D = 1.6-47.8; H = 1.9-27.9$	trees	82	NA	Secondary Atlantic Forest (30 year regeneration)	Brazil	Atlantic Forest
Sansevero et al., 2017	Tiepolo et al., 2002	$AGB = 0.3999 + 7.907 * H$	life form	$D = 1-33$	palms	NA	NA	Atlantic Forest	Brazil	Amazon

Sattler et al., 2014	Brown and Iverson, 1992	$AGB = 21.297 - 6.953(D) + 0.74(D^2)$	mixed species	$D = 4-116$	trees	169	NA	Wet Tropical Forests	New Guinea, Puerto Rico, Indonesia, Costa Rica	NA
Silva et al., 2015	Scolforo et al., 2008	$\ln AGB = -10.9532786932 + 2.5464820134 \ln(D) + 0.4667754371 \ln(H)$	mixed species	$D = 5.0-45.0$	trees	161	NA	Seasonal Semideciduous Forest	Brazil	Atlantic Forest
Silva et al., 2015	Scolforo et al., 2008	$\ln AGC = -12.3034390630 + 2.6584231780 \ln(D) + 0.5711719721 \ln(H)$	mixed species	$D = 5.0-45.0$	trees	161	NA	Seasonal Semideciduous Forest	Brazil	Atlantic Forest
Steininger et al., 2000	Folster et al., 1976	$AGB = 0.0101(D^2)^{1.136} + 0.0379(D^2)^{0.712}$	botanical family	NA	Araliaceae	NA	NA	Secondary and primary Tropical Evergreen Forest	Colombia	NA
Steininger et al., 2000	Saldarriaga et al., 1988	$AGB = \exp(-1.09 + 0.876 \ln(D^2) + 0.604 \ln(H) + 0.871 \ln(p))$	mixed species	$D \geq 1; H > 2$	trees	43	NA	Amazonian forest	Colombia and Venezuela	NA
Steininger et al., 2000	Scatena et al., 1993	$AGB = \exp(2.475 \ln(D) - 2.40)$	mixed species	$D = 2.5-57$	trees	63	NA	Tropical Forest	Puerto Rico	NA

Uhl et al., 1988	Uhl et al., 1988	$\ln AGB = -1.90 + 1.11 \ln D + 0.05 \ln H$	specific	$H \geq 2$	<i>Banara guianensis</i>	10	1	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
Uhl et al., 1988	Uhl et al., 1988	$\ln AGB = -2.95 + 1.02 \ln D + 0.83 \ln H$	specific	$H \geq 2$	<i>Solanum crinitum</i>	8	1	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
Uhl et al., 1988	Uhl et al., 1988	$\ln AGB = -3.06 + 0.89 \ln D + 1.11 \ln H$	specific	$H \geq 2$	<i>Solanum rugosum</i>	10	1	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
Uhl et al., 1988	Uhl et al., 1988	$\ln AGB = -3.54 + 1.13 \ln D + 0.77 \ln H$	specific	$H \geq 2$	<i>Vismia guianensis</i>	10	1	Secondary Amazonian forests (abandoned pastures 1-8y)	Brazil	Amazon
Wandelli and Fearnside, 2015	Wandelli and Fearnside, 2015	$AGB = -0.253 + 0.36 \ln D$	life form	$D = 1.0-4.0$	shrubs	74	NA	Secondary Amazonian forests (1-15y)	Brazil	Amazon

Wandelli and Fearnside, 2015	Wandelli and Fearnside, 2015	$\ln \text{AGB} = 1.869 + 2.231 \ln(D)$	mixed species	D = 1.0-23.0	trees	1128	NA	Secondary Amazonian forests (1-15y)	Brazil	Amazon
Wandelli and Fearnside, 2015	Wandelli and Fearnside, 2015	$\ln \text{AGB} = -4.173 + 1.477 \ln(D)$	specific	D = 1.0-7.0	Cecropia ulei	138	1	Secondary Amazonian forests (1-15y)	Brazil	Amazon

NA = Not available data

p = Wood density (g.cm^{-3})

H = Height

D = Trunk diameter

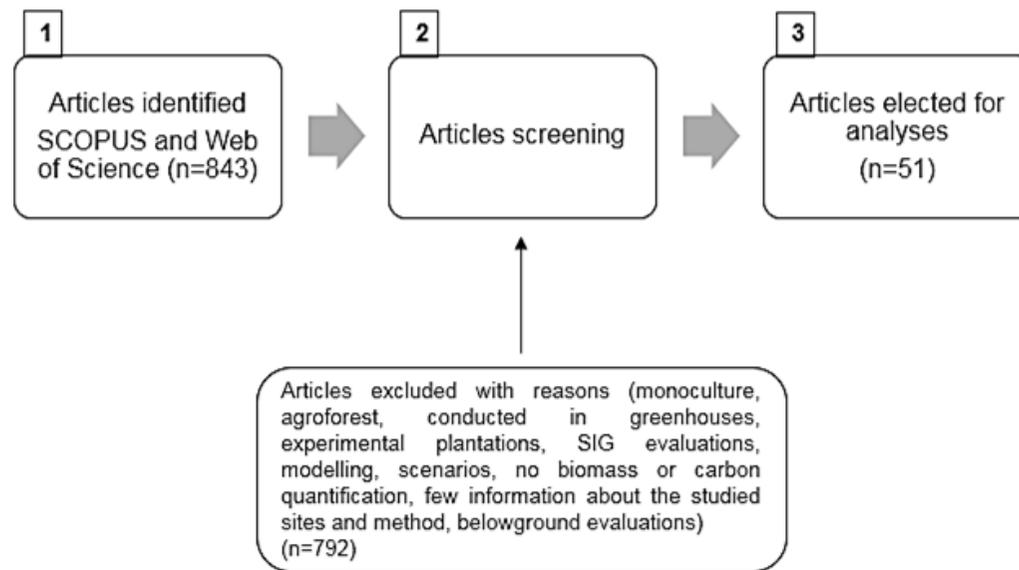


Figure A.1. Diagram of screening process flux

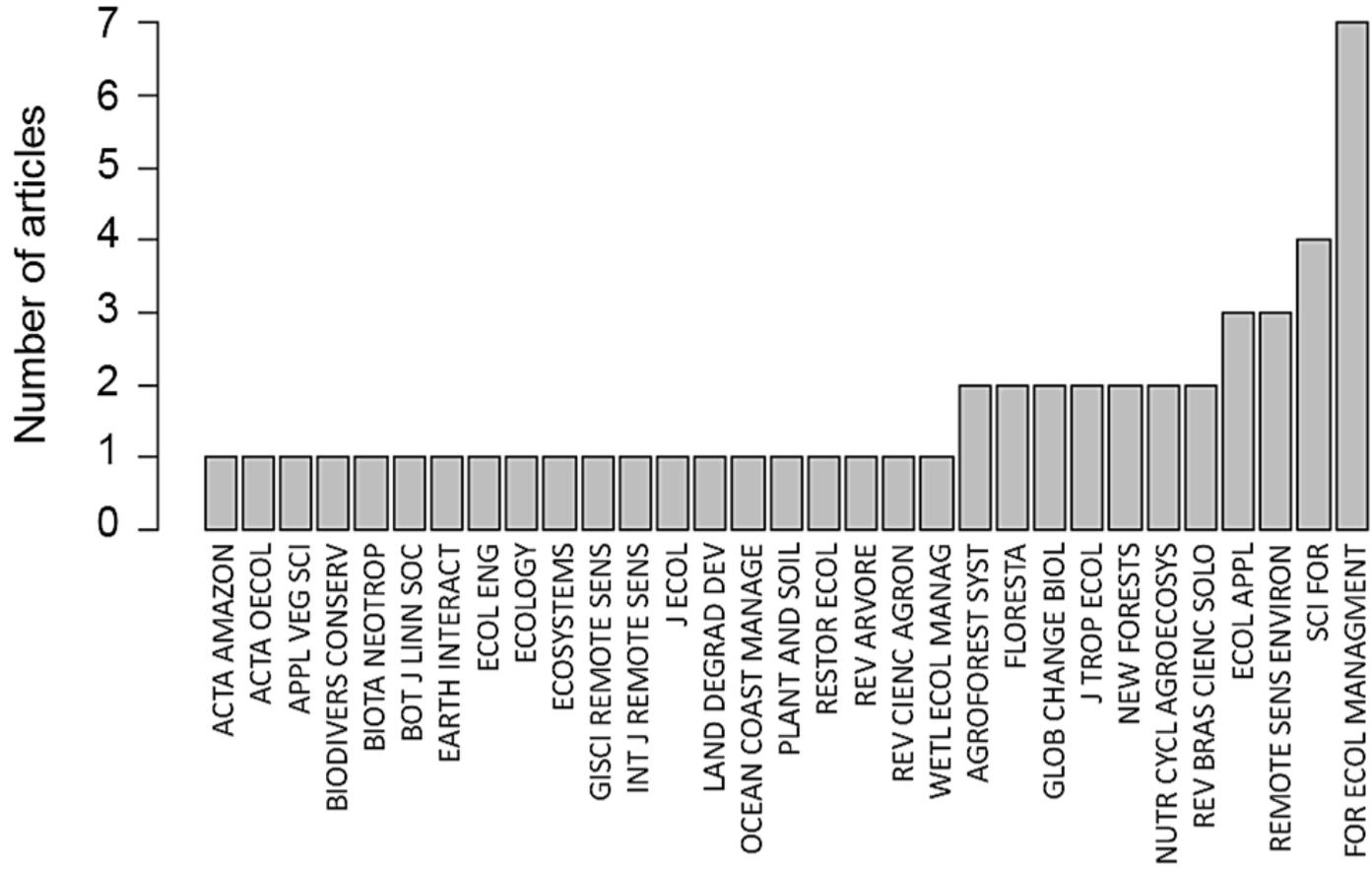


Figure A.2. Number of publications per Journal

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Supplementary Material B (Chapter 2)

This section contains a figure with the methodological representation of the sample design applied in the study (Figure B.1), a table with the restoration sites characteristics (Table B.1), and a table with the summary of the data related to carbon stocks and individuals density (Table B.2).

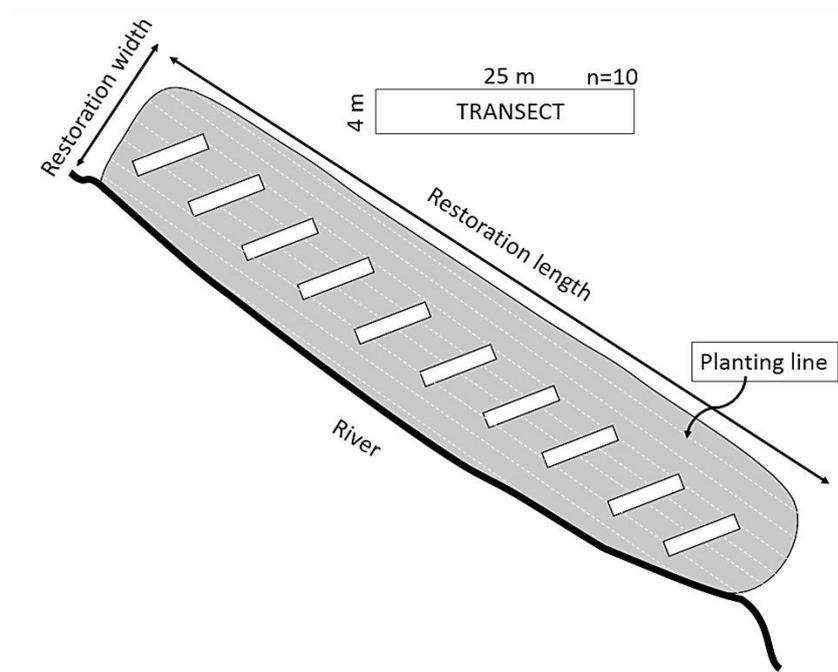


Figure B.1. Transects disposal in the studied active restoration sites, which were installed in the oblique direction and in a equidistant manner along with the drainage system (river/reservoir) of each site, with the center of transects positioned at the center of the restoration width.

Table B.1. General characteristics of the 16 active forest restoration sites and their respective surrounding landscapes.

Restoration age at survey date (years)	Survey year	Municipality	Restoration area (hectare)	Distance between transects (m)	Restoration width (m)	Restoration length (m)	Soil type ¹	Annual temperature (°C) ²	Annual precipitation (mm) ²	Elevation (m) ³	Forest cover in the landscape (%) (buffer 500 m)	Past land-use just before restoration implementation ^{3,4}	Matrix ^{3,4}
5	2018	Santa Bárbara D'Oeste	9.8	172	52	1891	Red Latosol	20.4	1261	600	9	Agriculture	Sugar-cane
7	2019	São José da Bela Vista	1.4	50	26	534	Red Latosol	19.6	1548	676	21	Agriculture	Sugar-cane
8	2018	Santa Bárbara D'Oeste	2.8	58	42	614	Red-Yellow Argisol	20.4	1261	559	15	Agriculture	Sugar-cane
9	2019	Batatais	3.6	120	37	1209	Red Latosol	20.2	1565	589	13	Agriculture	Sugar-cane

10	2019	Batatais	4.4	116	30	1185	Red Latosol	20.2	1565	755	14	Agriculture	Sugarcane
11	2019	Batatais	12.7	199	43	2040	Red Latosol	20.2	1565	729	12	Agriculture	Sugarcane
12	2019	Batatais	10.7	70	50	718	Red Latosol	20.2	1565	695	6	Agriculture	Sugarcane
13	2018	Iracemápolis	8.2	236	33	2616	Red Latosol	20.4	1294	631	11	Agriculture	Sugarcane
15	2019	Araras	13.7	68	180	695	Red Latosol	20.3	1312	653	22	Agriculture	Sugarcane
16	2019	Araras	13.4	106	109	1181	Red Latosol	20.3	1312	651	20	Agriculture	Sugarcane
19	2018	Araras	9.1	97	82	1092	Red Latosol	20.3	1312	673	7	Agriculture	Sugarcane
20	2018	Santa Bárbara D'Oeste	13.8	205	68	1729	Red Latosol	20.4	1261	554	4	Agriculture	Sugarcane
23	2019	Taruma	3.8	49	70	459	Clayey Nitosol	21.3	1295	418	13	Agriculture	Sugarcane
26	2019	Taruma	3.2	109	24	1117	Haplic Gleysol	21.3	1295	361	5	Agriculture	Sugarcane
27	2019	Taruma	2.5	30	65	350	Red Argisol	21.3	1295	434	12	Agriculture	Sugarcane

30	2018	Iracemápolis	30.4	312	71	3500	Red Latosol	20.4	1294	612	13	Agriculture	Sugar- cane
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¹ Rossi, M. 2017. Mapa pedológico do Estado de São Paulo: revisado e ampliado. São Paulo: Instituto Florestal, 1, 118.

² <https://pt.climate-data.org/>. Accessed in 20/04/2020

³ Google Earth Pro. Elevation Data/Terrain layer -Shuttle Radar Topography Mission (SRTM – NASA) and historical images of the planting date of the restoration sites. Accessed in 27/04/2020

⁴ Projeto MapBiomias – Coleção 4.1 da Série Anual de Mapas de Cobertura e Uso de Solo do Brasil. Accessed in 20/04/2020 <http://mapbiomas.org>

Table B.2. Structural attributes and carbon stocks observed in the 16 restorations sites.

Age (Years after plantation)	Density living plants (inds.ha ⁻¹)			Dead density (inds.ha ⁻¹)		AGB living plants	Carbon living plants			Dead carbon stock
	D _{total}	D _{<30}	D _{≥30}	D _{dead}		AGB _{total} (Mg.ha ⁻¹)	AGC _{total} (Mg.ha ⁻¹)	AGC _{<30} (Mg.ha ⁻¹)	AGC _{≥30} (Mg.ha ⁻¹)	AGC _{dead} (Mg.ha ⁻¹)
5	1340	1330	10	50		66.8	33.4	32.4	1.0	0.7
7	970	960	10	70		34.0	17.0	15.4	1.6	1.4
8	1520	1430	90	100		90.3	45.1	25.3	19.9	1.4
9	1250	1220	30	150		36.8	18.4	15.3	3.1	0.7
10	1000	930	70	70		76.9	38.5	27.5	11.0	1.1
11	880	840	40	20		42.7	21.4	11.1	10.2	0.1
12	1410	1210	200	270		239.6	119.8	19.2	100.6	1.6
13	1000	790	210	80		266.8	133.4	17.0	116.4	2.1
Mean	1171	1089	83	101		106	53	20	33.0	1

(5-13)										
15	850	690	160	100		158.4	79.2	28.8	50.4	2.6
16	930	890	40	100		55.9	27.9	15.0	12.9	0.3
19	1540	1410	130	280		194.4	97.2	46.0	51.3	6.3
20	1450	1290	160	120		204.0	102.0	58.9	43.1	6.9
23	1500	1370	130	60		191.1	95.6	40.6	54.9	4.2
26	2430	2130	300	120		308.4	154.2	44.3	109.9	3.2
27	2070	1990	80	150		105.7	52.9	28.6	24.3	10.8
0	2240	2030	210	80		352.7	176.4	49.1	127.2	0.5
Mean (>13-30)	1626	1475	151	126		196	98	39	59	4
Mean	1399	1282	117	114		151	76	30	46	2
Minimum	850	690	10	20		34	17	11	1.0	0.1
Maximum	2430	2130	300	280		352	176	59	127	10
Mature Forest (average biomass and carbon stock among five forest remnants) (Ferez et al., 2015; César et al., 2017)						307	154			

Supplementary Material C (Chapter 3)

This section contains one figure (Figure C.1) and three tables (Table C.1, C.2, and C.3). The Figure C.1 show the models adjusted for total carbon stock and total plant's density as a function of the restorations' age, which were used to calculate the relative carbon stock and plant's density of the different ecological group. The Table C.1 presents the functional attributes selected in function of their influence on the ecological processes related to carbon ecosystem services. The ecological group (i.e. sucessional group) was obtained on the literature (Durigan and Nogueira, 1990; Fonseca and Rodrigues, 2000; Carneiro, 2002; Nappo, 2002; Gomes et al., 2003; Silva, 2003; Veiga et al., 2003; Vieira, 2004; Melo, 2004; Rego, 2008; Aquino and Barbosa, 2009; Castanho, 2009; Dan, 2009; Donadio et al., 2009; Sobrinho et al., 2009; Almeida et al., 2010; Bastos, 2010; Demarchi, 2010; Freitas, 2010; Laurito, 2010; Polisel and Franco, 2010; Lopes et al., 2011; Rodrigues et al., 2011; Santos et al., 2011; Fehlaer and Rêgo, 2012; Gris, 2012; Mangueira, 2012; Rosa, 2012; da Costa, 2013; Marcuzzo et al., 2013; Cabacinha and Fontes, 2014; Guimarães et al., 2014; Garcia, 2015; Blagitz et al., 2016; Fetter, 2016; Freitas et al., 2016; Silva et al., 2016; Kunz and Martins, 2016; Motta, 2016; Lima et al., 2017; Magalhães et al., 2017; de Souza, 2018; Abaurre, 2019; Griebeler, 2019; Santana et al., 2019; Secco et al., 2019; Webber, 2019; Wheatley, 2019; da Silva et al., 2020). The life forms and the wood density were obtained through the packages "flora" and "BIOMASS" available for the R program, respectively (Carvalho, 2017; Rejou-Mechain et al., 2017; R Core Team, 2018). Species' abundance and accumulated carbon stock were calculated based on the sum of the data obtained in the 16 restoration plantings. The Tables C.2 (carbon stock of the ecological groups) and C.3 (abundance of the ecological groups) present the results of the F-tests and t-tests applied to the selected models.

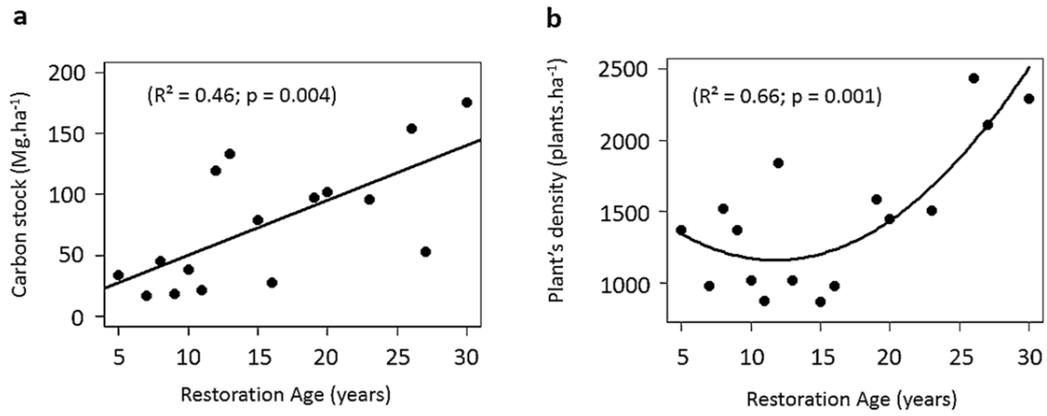


Figure C.1. Models adjusted for (a) total carbon stock (carbons stock = $4.982 + 4.507 \times \text{Age}$) and (b) total plant's density (plant's density = $1724.38 - 95.311 \times \text{Age} + 4.052 \times \text{Age}^2$) as a function of the restorations' age

Table C.1. List of species identified in the restoration plantings, its origins, functional attributes, abundance and accumulated carbon stock.

Species	Origin	Ecological group	Life form	Wood density (g.cm ⁻³)	Accumulated abundance (number of individuals sampled in the 160 transects)	Accumulated Carbon Stock (Mg sampled in the 160 transects)
<i>Aegiphila integrifolia</i>	native	P	tree	0.860	10	0.2528
<i>Albizia lebbek</i>	naturalized	P	tree	0.597	1	0.0104
<i>Albizia niopoides</i>	native	L	tree	0.555	16	1.3990
<i>Alchornea glandulosa</i>	native	S	tree	0.378	16	0.2074
<i>Alchornea sidifolia</i>	native	P	tree	0.408	1	0.2088
<i>Allophylus edulis</i>	native	S	tree	0.650	1	0.0003
<i>Aloysia virgata</i>	native	P	tree	0.643	9	0.0879
<i>Anadenanthera colubrina</i>	native	S	tree	0.866	39	10.6438
<i>Anadenanthera peregrina</i>	native	S	tree	0.883	8	0.9209
<i>Annona sylvatica</i>	native	S	tree	0.519	1	0.0004
<i>Apeiba tibourbou</i>	native	P	tree	0.200	9	0.4190
<i>Araliaceae sp.</i>	NA	NA	tree	0.423	3	0.0205
<i>Araucaria angustifolia</i>	native	L	tree	0.480	1	0.0003
<i>Arecaceae sp.1</i>	NA	NA	palm	0.552	1	0.0437

<i>Arecaceae sp.2</i>	NA	NA	palm	0.552	1	0.0145
<i>Aspidosperma cylindrocarpon</i>	native	L	tree	0.637	1	0.0004
<i>Aspidosperma olivaceum</i>	native	L	tree	0.750	1	0.0593
<i>Aspidosperma polyneuron</i>	native	L	tree	0.738	2	0.0347
<i>Aspidosperma ramiflorum</i>	native	L	tree	0.710	1	0.0005
<i>Aspidosperma subincanum</i>	native	S	tree	0.820	4	0.0105
<i>Astronium fraxinifolium</i>	native	S	tree	0.850	2	0.2125
<i>Astronium graveolens</i>	native	S	tree	0.868	2	0.0081
<i>Balfourodendron riedelianum</i>	native	L	tree	0.666	6	0.4373
<i>Basiloxylon brasiliensis</i>	native	L	tree	0.489	2	0.3349
<i>Bastardiopsis densiflora</i>	native	P	tree	0.651	4	0.0159
<i>Bathysa australis</i>	native	S	tree	0.640	1	0.0062
<i>Bauhinia forficata</i>	native	S	tree	0.755	4	0.0672
<i>Bauhinia longifolia</i>	native	S	tree	0.670	4	0.0254
<i>Bauhinia monandra</i>	cultivated	P	tree	0.755	1	0.0023
<i>Bauhinia variegata</i>	cultivated	P	tree	0.653	1	0.0174
<i>Bixa orellana</i>	native	S	tree	0.360	5	0.0431
<i>Cabralea canjerana</i>	native	S	tree	0.532	2	0.0103
<i>Calliandra foliolosa</i>	native	S	tree	0.816	49	0.2976
<i>Calophyllum brasiliense</i>	native	L	tree	0.589	6	0.0142
<i>Cariniana estrellensis</i>	native	S	tree	0.637	9	1.3934
<i>Cariniana legalis</i>	native	L	tree	0.495	9	0.2225
<i>Casearia gossypiosperma</i>	native	S	tree	0.625	4	0.0275
<i>Casearia sylvestris</i>	native	P	tree	0.705	11	0.0860
<i>Cassia ferruginea</i>	native	S	tree	0.866	4	0.4193
<i>Cecropia glaziovii</i>	native	P	tree	0.340	4	0.0409
<i>Cecropia hololeuca</i>	native	P	tree	0.430	5	0.0484
<i>Cecropia pachystachya</i>	native	P	tree	0.410	96	2.1569
<i>Cedrela fissilis</i>	native	L	tree	0.467	25	0.5678

<i>Cedrela odorata</i>	native	S	tree	0.447	2	0.0247
<i>Ceiba speciosa</i>	native	P	tree	0.392	16	3.4338
<i>Celtis brasiliensis</i>	native	P	tree	0.568	1	0.0034
<i>Celtis iguanaea</i>	native	P	tree	0.655	6	0.1411
<i>Centrolobium tomentosum</i>	native	S	tree	0.665	24	1.0913
<i>Cestrum axillare</i>	native	S	tree	0.538	2	0.0067
<i>Cestrum intermedium</i>	native	P	tree	0.538	1	0.0004
<i>Cestrum mariquitense</i>	native	P	shrub	0.538	2	NA
<i>Chloroleucon tortum</i>	native	S	tree	0.667	2	0.0177
<i>Chrysophyllum gonocarpum</i>	native	L	tree	0.672	1	0.0136
<i>Chrysophyllum marginatum</i>	native	L	tree	0.704	1	0.0003
<i>Citharexylum myrianthum</i>	native	P	tree	0.667	30	5.0148
<i>Clausena excavata</i>	cultivated	S	tree	0.482	45	0.3573
<i>Colubrina glandulosa</i>	native	S	tree	0.739	12	0.1397
<i>Conchocarpus pentandrus</i>	native	L	tree	0.717	1	0.0214
<i>Copaifera langsdorffii</i>	native	L	tree	0.650	3	0.0432
<i>Cordia americana</i>	native	S	tree	0.688	1	0.0004
<i>Cordia ecalyculata</i>	native	S	tree	1.076	2	0.0443
<i>Cordia myxa</i>	cultivated	S	tree	0.363	28	3.1878
<i>Cordia sellowiana</i>	native	S	tree	0.542	1	0.0172
<i>Cordia superba</i>	native	P	tree	0.542	10	0.0790
<i>Coutarea hexandra</i>	native	S	tree	0.600	2	0.1207
<i>Croton floribundus</i>	native	S	tree	0.600	19	1.9702
<i>Croton urucurana</i>	native	P	tree	0.528	53	1.2601
<i>Cupania vernalis</i>	native	S	tree	0.663	1	0.0004
<i>Dahlstedtia muehlbergiana</i>	native	S	tree	0.704	2	0.0247
<i>Dalbergia frutescens</i>	native	P	tree	0.690	4	0.1695
<i>Dalbergia nigra</i>	native	S	tree	0.812	9	0.0905
<i>Dendropanax cuneatus</i>	native	L	tree	0.420	2	0.0408

<i>Dictyoloma vandellianum</i>	native	S	tree	0.717	6	0.0320
<i>Dipteryx alata</i>	native	L	tree	0.978	1	0.0007
<i>Doliocarpus dentatus</i>	native	P	liana	0.633	1	0.0480
<i>Enterolobium contortisiliquum</i>	native	S	tree	0.397	40	2.7842
<i>Eriobotrya japonica</i>	naturalized	L	tree	0.880	4	0.0276
<i>Eriotheca gracilipes</i>	native	S	tree	0.438	3	0.1308
<i>Erythrina dominguezii</i>	native	S	tree	0.258	1	0.2108
<i>Erythrina sp.</i>	NA	NA	tree	0.289	1	0.0032
<i>Erythrina speciosa</i>	native	P	tree	0.289	7	0.0109
<i>Erythroxylum pulchrum</i>	native	S	tree	1.071	1	0.0005
<i>Esenbeckia leiocarpa</i>	native	L	tree	0.964	1	0.0087
<i>Eugenia ramboi</i>	native	L	tree	0.723	1	0.0007
<i>Eugenia uniflora</i>	native	L	tree	0.828	8	0.0414
<i>Fabaceae sp.</i>	NA	NA	tree	0.704	21	2.8788
<i>Fabaceae sp.1</i>	NA	NA	tree	0.704	2	0.0887
<i>Fabaceae sp.2</i>	NA	NA	tree	0.704	9	0.1176
<i>Fabaceae sp.3</i>	NA	NA	tree	0.704	2	0.0058
<i>Ficus guaranitica</i>	native	S	tree	0.406	2	0.0071
<i>Ficus insipida</i>	native	S	tree	0.377	3	0.0502
<i>Ficus sp.</i>	NA	NA	tree	0.406	7	0.0710
<i>Ficus trigona</i>	native	P	tree	0.406	1	0.1672
<i>Fridericia rego</i>	native	NA	liana	0.566	5	0.0168
<i>Fridericia speciosa</i>	native	NA	liana	0.566	2	0.0025
<i>Gallesia integrifolia</i>	native	S	tree	0.510	18	4.2226
<i>Genipa americana</i>	native	L	tree	0.634	24	1.6326
<i>Guapira hirsuta</i>	native	L	tree	0.671	16	0.1176
<i>Guarea guidonia</i>	native	L	tree	0.565	25	0.1801
<i>Guarea kunthiana</i>	native	L	tree	0.616	8	0.0148
<i>Guazuma ulmifolia</i>	native	P	tree	0.519	56	2.8732

<i>Handroanthus chrysotrichus</i>	native	S	tree	0.566	9	0.1800
<i>Handroanthus heptaphyllus</i>	native	S	tree	0.566	3	0.0406
<i>Handroanthus impetiginosus</i>	native	L	tree	0.566	21	1.3663
<i>Handroanthus ochraceus</i>	native	S	tree	0.566	8	0.0604
<i>Handroanthus serratifolius</i>	native	L	tree	0.566	8	0.0337
<i>Heliocarpus popayanensis</i>	native	P	tree	0.240	21	0.7531
<i>Holocalyx balansae</i>	native	L	tree	0.859	1	0.0145
<i>Hovenia dulcis</i>	naturalized	P	tree	0.543	3	0.1009
<i>Hymenaea courbaril</i>	native	L	tree	0.792	14	1.3241
<i>Hymenaea stigonocarpa</i>	native	L	tree	0.900	1	0.0351
<i>Inga edulis</i>	native	S	tree	0.587	6	0.4426
<i>Inga laurina</i>	native	S	tree	0.665	40	1.9332
<i>Inga sessilis</i>	native	S	tree	0.430	1	0.0168
<i>Inga striata</i>	native	S	tree	0.578	2	0.1503
<i>Inga vera</i>	native	S	tree	0.575	37	4.7215
<i>Jacaranda cuspidifolia</i>	native	P	tree	0.473	8	0.0428
<i>Jacaranda mimosifolia</i>	cultivated	S	tree	0.490	8	0.6196
<i>Joannesia princeps</i>	native	S	tree	0.511	2	0.5213
<i>Lafoensia pacari</i>	native	L	tree	0.800	13	0.1105
<i>Lauraceae sp.</i>	NA	NA	tree	0.564	10	0.4347
<i>Leucaena leucocephala</i>	naturalized	P	tree	0.683	32	2.3079
<i>Libidibia ferrea</i>	native	L	tree	0.704	7	0.6654
<i>Licania tomentosa</i>	native	S	tree	0.98	3	0.1784
<i>Lithraea molleoides</i>	native	P	tree	0.474	4	0.0474
<i>Lonchocarpus cultratus</i>	native	S	tree	0.761	3	0.0405
<i>Luehea candicans</i>	native	S	tree	0.568	12	0.2412
<i>Luehea divaricata</i>	native	S	tree	0.563	1	0.0003
<i>Luehea grandiflora</i>	native	P	tree	0.568	3	0.0445
<i>Mabea fistulifera</i>	native	P	tree	0.608	29	0.1454

<i>Mabea piriri</i>	native	S	tree	0.598	2	0.3751
<i>Machaerium acutifolium</i>	native	S	tree	1.120	3	0.3114
<i>Machaerium hirtum</i>	native	P	tree	0.660	1	0.0402
<i>Machaerium nyctitans</i>	native	P	tree	0.732	1	0.0144
<i>Machaerium stipitatum</i>	native	S	tree	0.732	2	0.0138
<i>Machaerium villosum</i>	native	L	tree	0.78	2	0.0278
<i>Maclura tinctoria</i>	native	S	tree	0.795	7	0.4807
<i>Magnolia ovata</i>	native	L	tree	0.525	1	0.0002
<i>Mangifera sp.</i>	NA	NA	tree	0.501	5	0.2386
<i>Melia azedarach</i>	naturalized	P	tree	0.438	5	0.9835
<i>Michelia champaca</i>	cultivated	NA	tree	0.528	4	0.1182
<i>Mimosa bimucronata</i>	native	P	tree	0.610	3	0.1974
<i>Mimosa caesalpiniiifolia</i>	native	P	tree	0.841	14	0.2960
<i>Mollinedia widgrenii</i>	native	L	tree	0.556	4	0.0085
<i>Moquiniastrum polymorphum</i>	native	S	tree	0.590	2	0.1213
<i>Myracrodruon urundeuva</i>	native	L	tree	0.566	12	0.3868
<i>Myrcia guianensis</i>	native	S	tree	0.816	1	0.0006
<i>Myrocarpus frondosus</i>	native	L	tree	0.818	5	0.1559
<i>Myroxylon peruiferum</i>	native	L	tree	0.824	26	1.4503
<i>Myrsine coriacea</i>	native	P	tree	0.647	7	0.3197
<i>Myrsine guianensis</i>	native	P	tree	0.741	45	0.8965
<i>Myrsine lancifolia</i>	native	L	tree	0.741	1	0.0034
<i>Myrsine umbellata</i>	native	S	tree	0.741	8	0.1741
<i>Myrtaceae sp.</i>	NA	NA	tree	0.766	5	0.0510
<i>Myrtaceae sp.2</i>	NA	NA	tree	0.766	1	0.0024
<i>Myrtaceae sp.3</i>	NA	NA	tree	0.766	5	0.5158
<i>Myrtaceae sp.4</i>	NA	NA	tree	0.766	2	0.0085
<i>Myrtaceae sp.5</i>	NA	NA	tree	0.766	1	0.0003
<i>Myrtaceae sp.6</i>	NA	NA	tree	0.766	2	0.0055

<i>Myrtaceae sp.7</i>	NA	NA	tree	0.766	1	0.0005
<i>Myrtaceae sp.8</i>	NA	NA	tree	0.766	4	0.0082
<i>Myrtaceae sp.9</i>	NA	NA	tree	0.766	1	0.0061
<i>Nectandra angustifolia</i>	native	L	tree	0.473	56	1.4141
<i>Nectandra megapotamica</i>	native	L	tree	0.562	30	0.8829
<i>Ocotea corymbosa</i>	native	L	tree	0.539	11	0.0945
<i>Ocotea puberula</i>	native	L	tree	0.500	1	0.0003
<i>Ocotea sp.</i>	NA	NA	tree	0.539	2	0.0698
<i>Ormosia sp.</i>	NA	NA	tree	0.579	15	0.0336
<i>Pachira glabra</i>	native	S	tree	0.479	5	0.1105
<i>Parapiptadenia rigida</i>	native	S	tree	1.070	21	2.3734
<i>Paubrasilia echinata</i>	native	L	tree	0.704	2	0.0565
<i>Paullinia sp.</i>	NA	NA	liana	0.734	1	0.0107
<i>Peltophorum dubium</i>	native	S	tree	0.744	10	0.6528
<i>Piper aduncum</i>	native	P	shrub	0.394	4	NA
<i>Piper amalago</i>	native	L	shrub	0.394	14	NA
<i>Piper arboreum</i>	native	S	shrub	0.394	11	NA
<i>Piper gaudichaudianum</i>	native	L	shrub	0.394	32	NA
<i>Piper glabratum</i>	native	S	shrub	0.394	1	NA
<i>Piper mollicomum</i>	native	S	shrub	0.394	4	NA
<i>Piper sp.</i>	NA	NA	shrub	0.394	6	NA
<i>Piper umbellatum</i>	native	S	shrub	0.394	2	NA
<i>Piptadenia gonoacantha</i>	native	P	tree	0.715	17	5.1323
<i>Platypodium elegans</i>	native	P	tree	0.806	18	6.1023
<i>Poincianella pluviosa</i>	native	L	tree	0.704	9	0.6728
<i>Pouteria ramiflora</i>	native	L	tree	0.688	5	0.1093
<i>Protium heptaphyllum</i>	native	L	tree	0.629	2	0.0006
<i>Pseudobombax grandiflorum</i>	native	S	tree	0.390	7	0.0674
<i>Pseudobombax majus</i>	native	NA	tree	0.289	1	0.0190

<i>Psidium cattleianum</i>	native	L	tree	0.855	4	0.0147
<i>Psidium guajava</i>	naturalized	P	tree	0.652	51	0.7313
<i>Psychotria sp.</i>	NA	NA	tree	0.564	1	0.0021
<i>Pterocarpus rohrii</i>	native	L	tree	0.456	3	0.0081
<i>Pterogyne nitens</i>	native	P	tree	0.693	13	2.6619
<i>Randia armata</i>	native	S	tree	0.668	1	0.0078
<i>Rhamnidium elaeocarpum</i>	native	S	tree	0.742	8	0.0538
<i>Ricinus communis</i>	cultivated	P	shrub	0.543	1	NA
<i>Sapindaceae sp.</i>	NA	NA	tree	0.734	2	0.0098
<i>Sapindus saponaria</i>	native	L	tree	0.712	6	0.1033
<i>Sapium glandulosum</i>	native	S	tree	0.415	2	0.0005
<i>Schefflera morototoni</i>	native	S	tree	0.456	2	0.0267
<i>Schefflera sp.</i>	NA	NA	tree	0.395	1	0.0021
<i>Schinus molle</i>	native	S	tree	0.525	1	0.0004
<i>Schinus terebinthifolia</i>	native	P	tree	0.587	96	3.3890
<i>Schizolobium parahyba</i>	native	P	tree	0.346	28	4.7011
<i>Sebastiania brasiliensis</i>	native	L	tree	0.674	1	0.0092
<i>Seguiera langsdorffii</i>	native	S	tree	0.590	3	0.2950
<i>Senegalia polyphylla</i>	native	S	tree	0.704	65	3.1224
<i>Senna macranthera</i>	native	P	tree	0.603	9	0.1331
<i>Senna multijuga</i>	native	S	tree	0.582	18	0.6409
<i>Senna sp.</i>	NA	NA	tree	0.603	1	0.0026
<i>Senna splendida</i>	native	S	liana	0.603	1	0.0579
<i>Senna velutina</i>	native	NA	shrub	0.603	1	NA
<i>Serjania lethalis</i>	native	NA	liana	0.734	2	0.0051
<i>Solanaceae sp.</i>	NA	NA	tree	0.538	1	0.0156
<i>Solanum granulosoleprosum</i>	native	P	tree	0.412	13	0.0698
<i>Solanum swartzianum</i>	native	P	tree	0.412	1	0.0023
<i>Solanum torvum</i>	native	NA	shrub	0.412	2	NA

<i>sp.1</i>	NA	NA	tree	0.590	1	0.0055
<i>sp.2</i>	NA	NA	tree	0.590	1	0.0021
<i>sp.4</i>	NA	NA	tree	0.590	1	0.0369
<i>sp.5</i>	NA	NA	liana	0.590	6	0.0223
<i>sp.6</i>	NA	NA	tree	0.590	2	0.0008
<i>sp.7</i>	NA	NA	tree	0.590	1	0.0045
<i>Spondias dulcis</i>	native	P	tree	0.370	2	0.0057
<i>Spondias mombin</i>	native	P	tree	0.391	1	0.0011
<i>Sterculia apetala</i>	native	NA	tree	0.932	1	0.0482
<i>Styrax pohlii</i>	native	L	tree	0.405	3	0.0069
<i>Syzygium cumini</i>	naturalized	P	tree	0.673	15	1.8752
<i>Syzygium jambos</i>	naturalized	NA	tree	0.700	1	0.0058
<i>Tabebuia roseoalba</i>	native	S	tree	0.762	7	0.0879
<i>Tabernaemontana catharinensis</i>	native	P	tree	0.526	13	0.0514
<i>Tapirira guianensis</i>	native	S	tree	0.457	27	0.2064
<i>Tecoma stans</i>	naturalized	P	tree	0.466	23	0.1168
<i>Terminalia sp.</i>	NA	NA	tree	0.630	3	0.0476
<i>Terminalia triflora</i>	native	L	tree	0.748	2	0.0706
<i>Trema micrantha</i>	native	P	tree	0.319	13	0.0977
<i>Trichilia casaretti</i>	native	L	tree	0.644	1	0.0003
<i>Trichilia catigua</i>	native	L	tree	0.688	2	0.0157
<i>Trichilia clauseni</i>	native	L	tree	0.644	2	0.0054
<i>Trichilia elegans</i>	native	L	tree	0.644	6	0.0352
<i>Triplaris americana</i>	native	S	tree	0.49	51	1.4011
<i>Urera baccifera</i>	native	P	tree	0.165	1	0.0017
<i>Vernonanthura polyanthes</i>	native	P	shrub	0.540	1	NA
<i>Vernonanthura sp.</i>	NA	NA	shrub	0.540	4	NA
<i>Vitex polygama</i>	native	S	tree	0.565	10	1.3369
<i>Xylopia aromatica</i>	native	P	tree	0.561	2	0.0331

<i>Zanthoxylum caribaeum</i>	native	S	tree	0.760	4	0.1928
<i>Zanthoxylum fagara</i>	native	P	tree	0.651	1	0.0006
<i>Zanthoxylum rhoifolium</i>	native	P	tree	0.569	8	0.1186
<i>Zeyheria tuberculosa</i>	native	S	tree	0.770	4	0.0726

P: Pioneer; S: Secondary; L: Late successional species; NA: Not Available data

Table C.2. Results of the model adjusted for the carbon stocks of the ecological groups

> summary()					
lm(formula = carbon ~ age + successional.group, data = carb)					
Coefficients	Estimate	Std. Error	t value	Pr(> t)	
(Intercept)	-11.7689	7.9193	-1.486	0.14438	
Age	1.3227	0.3896	3.395	0.00146 **	
pioneer	20.9952	7.1227	2.948	0.00511 **	
secondary	22.3203	7.1227	3.134	0.00307 **	
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					
Residual standard error: 20.15 on 44 degrees of freedom					
Multiple R-squared: 0.3519 Adjusted R-squared: 0.3077					
F-statistic: 7.963 on 3 and 44 DF, p-value: 0.0002382					
Analysis of Variance Table					
Response: carbon					
	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Age	1	4678.7	4678.7	11.5277	0.001463 **
successional.group	2	5017.3	2508.7	6.1811	0.004308 **
Residuals	44	17858	405.9		
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					

Table C.3. Results of the model adjusted for the abundances (i.e. plant's density) of the ecological groups

> summary()					
lm(formula = density ~ age + successional.group + I(age^2) + d.tot + age: successional.group + I(age^2):successional.group, data = carb)					
Coefficients	Estimate	Std. Error	t value	Pr(> t)	
(Intercept)	-436.831	251.5628	-1.736	0.09058 .	
Age	47.00035	28.76753	1.634	0.11056	
pioneer	615.4016	298.6535	2.061	0.04624 *	
secondary	918.8061	298.6535	3.076	0.00387 **	
age^2	-1.38731	0.8481	-1.636	0.11014	
d.tot	0.28504	0.07929	3.595	0.00092 ***	
age:pioneer	-48.7621	39.25451	-1.242	0.22177	
age:secondary	-113.821	39.25451	-2.9	0.00618 **	
pioneer:age^2	1.27001	1.10998	1.144	0.25971	
secondary:age^2	3.56353	1.10998	3.21	0.00270 **	

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					
Residual standard error: 155.5 on 38 degrees of freedom					
Multiple R-squared: 0.6752 Adjusted R-squared: 0.5982					
F-statistic: 8.775 on 9 and 38 DF, p-value: 5.501e-07					
Analysis of Variance Table					
Response: density					
	Df	Sum Sq	Mean Sq	F value	Pr(>F)
age	1	490196	490196	20.2821	6.175e-05 ***
successional.group	2	527150	263575	10.9055	0.0001807 ***
age^2	1	223805	223805	9.26	0.0042340 **
d.tot	1	312343	312343	12.9233	0.0009200 ***
age:successional.group	2	99361	49680	2.0555	0.142022
age^2:successional.group	2	255957	127979	5.2952	0.0093635 **
Residuals	38	918419	24169		
Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1					

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Supplementary Material D (Chapter 4)

This section contains one table showing the accumulated density of individuals of different species and their respective functional attributes (Table D.1). The endemism was obtained from the Flora do Brasil 2020 database (<http://floradobrasil.jbrj.gov.br/>). The threat status was consulted in the Brazilian red list of flora species, made available by the “National Center for Plant Conservation” - CNCFlora (<http://cncflora.jbrj.gov.br/>). The wood density was obtained through the “*BIOMASS*” package from the R environment (Réjou-Méchain et al., 2017). The dispersal syndrome was obtained in the literature (Penhalber e Mantovani, 1997; CABI; Carvalho, 2003; Vieira, 2004; Gressler et al., 2006; Carvalho, 2007; Amaral, 2009; Castanho, 2009; Borgo, 2010; Dike e Agugom, 2010; Mauad, 2010; Moura et al., 2011; Preiskorn, 2011; Oliveira, 2012; Schievenin et al., 2012; Zama et al., 2012; Abreu, 2013; Alves, 2013; Colmanetti e Barbosa, 2013; da Silva, 2013; Magnago et al., 2014; Cunto, 2015; do Carmo, 2015; Garcia, 2015; da Silva, 2015; Meira-Junior et al., 2015; Batista et al., 2016; Frazão e Somner, 2016; Peres, 2016; Santos-Filho et al., 2016; Barbosa et al., 2017; Seubert et al., 2017; Vidal et al., 2020).

Table D.1. List of species with their respective abundances and the attributes of ecological interest.

Species	Abundance (species accumulated density)	Life Form	Endemic	Conservation Status	Wood Density (g.cm ⁻³)	Dispersal Syndrome	Carbon stock accumulated (Mg in 1.6 ha of active restorations)
<i>Aegiphila integrifolia</i>	10	tree	no	-	0.860	Zoo	0.2528
<i>Albizia lebbek</i>	1	tree	no	-	0.597	Autoc	0.0104
<i>Albizia niopoides</i>	16	tree	no	LC	0.555	Autoc	1.3990
<i>Alchornea glandulosa</i>	16	tree	no	-	0.378	Zoo	0.2074
<i>Alchornea sidifolia</i>	1	tree	no	-	0.408	Zoo	0.2088
<i>Allophylus edulis</i>	1	tree	no	-	0.650	Zoo	0.0003
<i>Aloysia virgata</i>	9	tree	no	-	0.643	Anemoc	0.0879
<i>Anadenanthera colubrina</i>	39	tree	no	-	0.866	Autoc	10.6438
<i>Anadenanthera peregrina</i>	8	tree	no	-	0.883	Autoc	0.9209
<i>Annona sylvatica</i>	1	tree	endemic	-	0.519	Zoo	0.0004
<i>Apeiba tibourbou</i>	9	tree	no	-	0.200	Zoo	0.4190
<i>Araliaceae sp.</i>	3	tree	NA	-	0.423	NA	0.0205
<i>Araucaria angustifolia</i>	1	tree	no	EN	0.480	Zoo	0.0003
<i>Areaceae sp.1</i>	1	palm	NA	-	0.552	NA	0.0437
<i>Areaceae sp.2</i>	1	palm	NA	-	0.552	NA	0.0145

<i>Aspidosperma cylindrocarpon</i>	1	tree	no	LC	0.637	Anemoc	0.0004
<i>Aspidosperma olivaceum</i>	1	tree	endemic	-	0.750	Anemoc	0.0593
<i>Aspidosperma polyneuron</i>	2	tree	no	NT	0.738	Anemoc	0.0347
<i>Aspidosperma ramiflorum</i>	1	tree	no	LC	0.710	Anemoc	0.0005
<i>Aspidosperma subincanum</i>	4	tree	no	-	0.820	Anemoc	0.0105
<i>Astronium fraxinifolium</i>	2	tree	no	LC	0.850	Anemoc	0.2125
<i>Astronium graveolens</i>	2	tree	no	LC	0.868	Anemoc	0.0081
<i>Balfourodendron riedelianum</i>	6	tree	no	NT	0.666	Anemoc	0.4373
<i>Basiloxylon brasiliensis</i>	2	tree	endemic	-	0.489	Anemoc	0.3349
<i>Bastardiopsis densiflora</i>	4	tree	no	-	0.651	Zoo	0.0159
<i>Bathysa australis</i>	1	tree	no	LC	0.640	Autoc	0.0062
<i>Bauhinia forficata</i>	4	tree	no	-	0.755	Zoo	0.0672
<i>Bauhinia longifolia</i>	4	tree	no	-	0.670	Autoc	0.0254
<i>Bauhinia monandra</i>	1	tree	no	-	0.755	Autoc	0.0023
<i>Bauhinia variegata</i>	1	tree	no	-	0.653	Anemoc	0.0174
<i>Bixa orellana</i>	5	tree	no	-	0.360	Zoo	0.0431
<i>Cabrlea canjerana</i>	2	tree	no	-	0.532	Zoo	0.0103
<i>Calliandra foliolosa</i>	49	tree	no	-	0.816	Anemoc	0.2976
<i>Calophyllum brasiliense</i>	6	tree	no	-	0.589	Zoo	0.0142
<i>Cariniana estrellensis</i>	9	tree	no	-	0.637	Anemoc	1.3934
<i>Cariniana legalis</i>	9	tree	endemic	EN	0.495	Anemoc	0.2225
<i>Casearia gossypiosperma</i>	4	tree	no	LC	0.625	Zoo	0.0275
<i>Casearia sylvestris</i>	11	tree	no	-	0.705	Zoo	0.0860
<i>Cassia ferruginea</i>	4	tree	no	-	0.866	Autoc	0.4193
<i>Cecropia glaziovii</i>	4	tree	endemic	-	0.340	Zoo	0.0409
<i>Cecropia hololeuca</i>	5	tree	endemic	-	0.430	Zoo	0.0484
<i>Cecropia pachystachya</i>	96	tree	no	-	0.410	Zoo	2.1569
<i>Cedrela fissilis</i>	25	tree	no	VU	0.467	Anemoc	0.5678
<i>Cedrela odorata</i>	2	tree	no	VU	0.447	Anemoc	0.0247

<i>Ceiba speciosa</i>	16	tree	no	-	0.392	Anemoc	3.4338
<i>Celtis brasiliensis</i>	1	tree	no	-	0.568	Zoo	0.0034
<i>Celtis iguanaea</i>	6	tree	no	-	0.655	Zoo	0.1411
<i>Centrolobium tomentosum</i>	24	tree	endemic	LC	0.665	Anemoc	1.0913
<i>Cestrum axillare</i>	2	tree	no	-	0.538	Zoo	0.0067
<i>Cestrum intermedium</i>	1	tree	no	-	0.538	Zoo	0.0004
<i>Cestrum mariquitense</i>	2	shrub	no	-	0.538	Zoo	NA
<i>Chloroleucon tortum</i>	2	tree	endemic	NT	0.667	Autoc	0.0177
<i>Chrysophyllum gonocarpum</i>	1	tree	no	-	0.672	Zoo	0.0136
<i>Chrysophyllum marginatum</i>	1	tree	no	-	0.704	Zoo	0.0003
<i>Citharexylum myrianthum</i>	30	tree	no	-	0.667	Zoo	5.0148
<i>Clausena excavata</i>	45	tree	no	-	0.482	Zoo	0.3573
<i>Colubrina glandulosa</i>	12	tree	no	LC	0.739	Zoo	0.1397
<i>Conchocarpus pentandrus</i>	1	tree	no	-	0.717	Autoc	0.0214
<i>Copaifera langsdorffii</i>	3	tree	no	-	0.650	Zoo	0.0432
<i>Cordia americana</i>	1	tree	no	-	0.688	Anemoc	0.0004
<i>Cordia ecalyculata</i>	2	tree	no	-	1.076	Zoo	0.0443
<i>Cordia myxa</i>	28	tree	no	-	0.363	Zoo	3.1878
<i>Cordia sellowiana</i>	1	tree	endemic	-	0.542	Zoo	0.0172
<i>Cordia superba</i>	10	tree	endemic	-	0.542	Zoo	0.0790
<i>Coutarea hexandra</i>	2	tree	no	-	0.600	Anemoc	0.1207
<i>Croton floribundus</i>	19	tree	no	-	0.600	Autoc	1.9702
<i>Croton urucurana</i>	53	tree	no	-	0.528	Autoc	1.2601
<i>Cupania vernalis</i>	1	tree	no	-	0.663	Zoo	0.0004
<i>Dahlstedtia muehlbergiana</i>	2	tree	no	-	0.704	Autoc	0.0247
<i>Dalbergia frutescens</i>	4	tree	no	-	0.690	Anemoc	0.1695
<i>Dalbergia nigra</i>	9	tree	endemic	VU	0.812	Autoc	0.0905
<i>Dendropanax cuneatus</i>	2	tree	no	LC	0.420	Zoo	0.0408
<i>Dictyoloma vandellianum</i>	6	tree	no	-	0.717	Anemoc	0.0320

<i>Dipteryx alata</i>	1	tree	no	LC	0.978	Zoo	0.0007
<i>Doliocarpus dentatus</i>	1	liana	no	VU	0.633	Zoo	0.0480
<i>Enterolobium contortisiliquum</i>	40	tree	no	-	0.397	Autoc	2.7842
<i>Eriobotrya japonica</i>	4	tree	no	-	0.880	Zoo	0.0276
<i>Eriotheca gracilipes</i>	3	tree	no	-	0.438	Anemoc	0.1308
<i>Erythrina dominguezii</i>	1	tree	no	-	0.258	Autoc	0.2108
<i>Erythrina sp.</i>	1	tree	NA	-	0.289	NA	0.0032
<i>Erythrina speciosa</i>	7	tree	endemic	-	0.289	Autoc	0.0109
<i>Erythroxylum pulchrum</i>	1	tree	endemic	LC	1.071	Zoo	0.0005
<i>Esenbeckia leiocarpa</i>	1	tree	endemic	LC	0.964	Autoc	0.0087
<i>Eugenia ramboi</i>	1	tree	no	-	0.723	Zoo	0.0007
<i>Eugenia uniflora</i>	8	tree	no	-	0.828	Zoo	0.0414
<i>Fabaceae sp.</i>	21	tree	NA	-	0.704	NA	2.8788
<i>Fabaceae sp.1</i>	2	tree	NA	-	0.704	NA	0.0887
<i>Fabaceae sp.2</i>	9	tree	NA	-	0.704	NA	0.1176
<i>Fabaceae sp.3</i>	2	tree	NA	-	0.704	NA	0.0058
<i>Ficus guaranitica</i>	2	tree	no	-	0.406	Zoo	0.0071
<i>Ficus insipida</i>	3	tree	no	-	0.377	Zoo	0.0502
<i>Ficus sp.</i>	7	tree	NA	-	0.406	Zoo	0.0710
<i>Ficus trigona</i>	1	tree	no	-	0.406	Zoo	0.1672
<i>Fridericia rego</i>	5	liana	endemic	-	0.566	Anemoc	0.0168
<i>Fridericia speciosa</i>	2	liana	endemic	-	0.566	Anemoc	0.0025
<i>Gallesia integrifolia</i>	18	tree	endemic	-	0.510	Anemoc	4.2226
<i>Genipa americana</i>	24	tree	no	LC	0.634	Zoo	1.6326
<i>Guapira hirsuta</i>	16	tree	endemic	LC	0.671	Zoo	0.1176
<i>Guarea guidonia</i>	25	tree	no	-	0.565	Zoo	0.1801
<i>Guarea kunthiana</i>	8	tree	no	-	0.616	Zoo	0.0148
<i>Guazuma ulmifolia</i>	56	tree	no	-	0.519	Zoo	2.8732
<i>Handroanthus chrysotrichus</i>	9	tree	no	-	0.566	Anemoc	0.1800

<i>Handroanthus heptaphyllus</i>	3	tree	no	LC	0.566	Anemoc	0.0406
<i>Handroanthus impetiginosus</i>	21	tree	no	NT	0.566	Anemoc	1.3663
<i>Handroanthus ochraceus</i>	8	tree	no	-	0.566	Anemoc	0.0604
<i>Handroanthus serratifolius</i>	8	tree	no	-	0.566	Anemoc	0.0337
<i>Heliocarpus popayanensis</i>	21	tree	no	-	0.240	Anemoc	0.7531
<i>Holocalyx balansae</i>	1	tree	no	-	0.859	Autoc	0.0145
<i>Hovenia dulcis</i>	3	tree	no	-	0.543	Zoo	0.1009
<i>Hymenaea courbaril</i>	14	tree	no	LC	0.792	Zoo	1.3241
<i>Hymenaea stigonocarpa</i>	1	tree	no	-	0.900	Autoc	0.0351
<i>Inga edulis</i>	6	tree	no	-	0.587	Zoo	0.4426
<i>Inga laurina</i>	40	tree	no	LC	0.665	Zoo	1.9332
<i>Inga sessilis</i>	1	tree	endemic	-	0.430	Zoo	0.0168
<i>Inga striata</i>	2	tree	no	-	0.578	Zoo	0.1503
<i>Inga vera</i>	37	tree	no	-	0.575	Zoo	4.7215
<i>Jacaranda cuspidifolia</i>	8	tree	no	-	0.473	Anemoc	0.0428
<i>Jacaranda mimosifolia</i>	8	tree	no	-	0.490	Anemoc	0.6196
<i>Joannesia princeps</i>	2	tree	endemic	LC	0.511	Autoc	0.5213
<i>Lafoensia pacari</i>	13	tree	no	LC	0.800	Anemoc	0.1105
<i>Lauraceae sp.</i>	10	tree	NA	-	0.564	Zoo	0.4347
<i>Leucaena leucocephala</i>	32	tree	no	-	0.683	Autoc	2.3079
<i>Libidibia ferrea</i>	7	tree	endemic	-	0.704	Anemoc	0.6654
<i>Licania tomentosa</i>	3	tree	endemic	-	0.980	Zoo	0.1784
<i>Lithraea molleoides</i>	4	tree	no	-	0.474	Zoo	0.0474
<i>Lonchocarpus cultratus</i>	3	tree	no	-	0.761	Anemoc	0.0405
<i>Luehea candicans</i>	12	tree	no	LC	0.568	Anemoc	0.2412
<i>Luehea divaricata</i>	1	tree	no	-	0.563	Anemoc	0.0003
<i>Luehea grandiflora</i>	3	tree	no	-	0.568	Anemoc	0.0445
<i>Mabea fistulifera</i>	29	tree	no	-	0.608	Autoc	0.1454
<i>Mabea piriri</i>	2	tree	no	-	0.598	Autoc	0.3751

<i>Machaerium acutifolium</i>	3	tree	no	-	1.120	Anemoc	0.3114
<i>Machaerium hirtum</i>	1	tree	no	-	0.660	Anemoc	0.0402
<i>Machaerium nyctitans</i>	1	tree	no	LC	0.732	Anemoc	0.0144
<i>Machaerium stipitatum</i>	2	tree	no	-	0.732	Anemoc	0.0138
<i>Machaerium villosum</i>	2	tree	no	LC	0.780	Anemoc	0.0278
<i>Maclura tinctoria</i>	7	tree	no	-	0.795	Zoo	0.4807
<i>Magnolia ovata</i>	1	tree	endemic	LC	0.525	Zoo	0.0002
<i>Mangifera sp.</i>	5	tree	NA	-	0.501	Zoo	0.2386
<i>Melia azedarach</i>	5	tree	no	-	0.438	Zoo	0.9835
<i>Michelia champaca</i>	4	tree	no	-	0.528	Autoc	0.1182
<i>Mimosa bimucronata</i>	3	tree	no	-	0.610	Autoc	0.1974
<i>Mimosa caesalpiniiifolia</i>	14	tree	endemic	LC	0.841	Autoc	0.2960
<i>Mollinedia widgrenii</i>	4	tree	endemic	-	0.556	Zoo	0.0085
<i>Moquiniastrum polymorphum</i>	2	tree	no	-	0.590	Anemoc	0.1213
<i>Myracrodruon urundeuva</i>	12	tree	no	LC	0.566	Autoc	0.3868
<i>Myrcia guianensis</i>	1	tree	no	LC	0.816	Zoo	0.0006
<i>Myrocarpus frondosus</i>	5	tree	no	LC	0.818	Anemoc	0.1559
<i>Myroxylon peruiferum</i>	26	tree	no	LC	0.824	Anemoc	1.4503
<i>Myrsine coriacea</i>	7	tree	no	-	0.647	Zoo	0.3197
<i>Myrsine guianensis</i>	45	tree	no	-	0.741	Zoo	0.8965
<i>Myrsine lancifolia</i>	1	tree	endemic	-	0.741	Zoo	0.0034
<i>Myrsine umbellata</i>	8	tree	no	-	0.741	Zoo	0.1741
<i>Myrtaceae sp.</i>	5	tree	NA	-	0.766	Zoo	0.0510
<i>Myrtaceae sp.2</i>	1	tree	NA	-	0.766	Zoo	0.0024
<i>Myrtaceae sp.3</i>	5	tree	NA	-	0.766	Zoo	0.5158
<i>Myrtaceae sp.4</i>	2	tree	NA	-	0.766	Zoo	0.0085
<i>Myrtaceae sp.5</i>	1	tree	NA	-	0.766	Zoo	0.0003
<i>Myrtaceae sp.6</i>	2	tree	NA	-	0.766	Zoo	0.0055
<i>Myrtaceae sp.7</i>	1	tree	NA	-	0.766	Zoo	0.0005

<i>Myrtaceae sp.8</i>	4	tree	NA	-	0.766	Zoo	0.0082
<i>Myrtaceae sp.9</i>	1	tree	NA	-	0.766	Zoo	0.0061
<i>Nectandra angustifolia</i>	56	tree	no	LC	0.473	Zoo	1.4141
<i>Nectandra megapotamica</i>	30	tree	no	-	0.562	Zoo	0.8829
<i>Ocotea corymbosa</i>	11	tree	no	-	0.539	Zoo	0.0945
<i>Ocotea puberula</i>	1	tree	endemic	NT	0.500	Zoo	0.0003
<i>Ocotea sp.</i>	2	tree	NA	-	0.539	Zoo	0.0698
<i>Ormosia sp.</i>	15	tree	NA	-	0.579	NA	0.0336
<i>Pachira glabra</i>	5	tree	no	-	0.479	Zoo	0.1105
<i>Parapiptadenia rigida</i>	21	tree	no	-	1.070	Autoc	2.3734
<i>Paubrasilia echinata</i>	2	tree	endemic	-	0.704	Autoc	0.0565
<i>Paullinia sp.</i>	1	liana	NA	-	0.734	NA	0.0107
<i>Peltophorum dubium</i>	10	tree	no	-	0.744	Autoc	0.6528
<i>Piper aduncum</i>	4	shrub	no	-	0.394	Zoo	NA
<i>Piper amalago</i>	14	shrub	no	-	0.394	Zoo	NA
<i>Piper arboreum</i>	11	shrub	no	-	0.394	Zoo	NA
<i>Piper gaudichaudianum</i>	32	shrub	no	-	0.394	Zoo	NA
<i>Piper glabratum</i>	1	shrub	no	-	0.394	Zoo	NA
<i>Piper mollicomum</i>	4	shrub	no	-	0.394	Zoo	NA
<i>Piper sp.</i>	6	shrub	NA	-	0.394	Zoo	NA
<i>Piper umbellatum</i>	2	shrub	no	-	0.394	Zoo	NA
<i>Piptadenia gonoacantha</i>	17	tree	no	LC	0.715	Autoc	5.1323
<i>Platypodium elegans</i>	18	tree	no	-	0.806	Anemoc	6.1023
<i>Poincianella pluviosa</i>	9	tree	no	-	0.704	Autoc	0.6728
<i>Pouteria ramiflora</i>	5	tree	no	-	0.688	Zoo	0.1093
<i>Protium heptaphyllum</i>	2	tree	no	-	0.629	Zoo	0.0006
<i>Pseudobombax grandiflorum</i>	7	tree	endemic	LC	0.390	Anemoc	0.0674
<i>Pseudobombax majus</i>	1	tree	endemic	-	0.289	Anemoc	0.0190
<i>Psidium cattleianum</i>	4	tree	endemic	-	0.855	Zoo	0.0147

<i>Psidium guajava</i>	51	tree	no	-	0.652	Zoo	0.7313
<i>Psychotria sp.</i>	1	tree	NA	-	0.564	Zoo	0.0021
<i>Pterocarpus rohrii</i>	3	tree	no	-	0.456	Anemoc	0.0081
<i>Pterogyne nitens</i>	13	tree	no	LC	0.693	Anemoc	2.6619
<i>Randia armata</i>	1	tree	no	-	0.668	Zoo	0.0078
<i>Rhamnidium elaeocarpum</i>	8	tree	no	-	0.742	Zoo	0.0538
<i>Ricinus communis</i>	1	shrub	no	-	0.543	Autoc	NA
<i>Sapindaceae sp.</i>	2	tree	NA	-	0.734	NA	0.0098
<i>Sapindus saponaria</i>	6	tree	no	-	0.712	Zoo	0.1033
<i>Sapium glandulosum</i>	2	tree	no	-	0.415	Zoo	0.0005
<i>Schefflera morototoni</i>	2	tree	no	-	0.456	Zoo	0.0267
<i>Schefflera sp.</i>	1	tree	NA	-	0.395	NA	0.0021
<i>Schinus molle</i>	1	tree	no	-	0.525	Zoo	0.0004
<i>Schinus terebinthifolia</i>	96	tree	no	-	0.587	Zoo	3.3890
<i>Schizolobium parahyba</i>	28	tree	no	-	0.346	Autoc	4.7011
<i>Sebastiania brasiliensis</i>	1	tree	no	-	0.674	Autoc	0.0092
<i>Seguiera langsdorffii</i>	3	tree	endemic	LC	0.590	Anemoc	0.2950
<i>Senegalia polyphylla</i>	65	tree	no	-	0.704	Autoc	3.1224
<i>Senna macranthera</i>	9	tree	no	-	0.603	Autoc	0.1331
<i>Senna multijuga</i>	18	tree	no	-	0.582	Zoo	0.6409
<i>Senna sp.</i>	1	tree	NA	-	0.603	NA	0.0026
<i>Senna splendida</i>	1	liana	no	-	0.603	Autoc	0.0579
<i>Senna velutina</i>	1	shrub	no	-	0.603	Zoo	NA
<i>Serjania lethalis</i>	2	liana	no	-	0.734	Anemoc	0.0051
<i>Solanaceae sp.</i>	1	tree	NA	-	0.538	NA	0.0156
<i>Solanum granulosoleprosum</i>	13	tree	no	LC	0.412	Zoo	0.0698
<i>Solanum swartzianum</i>	1	tree	no	-	0.412	Zoo	0.0023
<i>Solanum torvum</i>	2	shrub	no	-	0.412	Zoo	NA
<i>sp.1</i>	1	tree	NA	-	0.590	NA	0.0055

<i>sp.2</i>	1	tree	NA	-	0.590	NA	0.0021
<i>sp.4</i>	1	tree	NA	-	0.590	NA	0.0369
<i>sp.5</i>	6	liana	NA	-	0.590	NA	0.0223
<i>sp.6</i>	2	tree	NA	-	0.590	NA	0.0008
<i>sp.7</i>	1	tree	NA	-	0.590	NA	0.0045
<i>Spondias dulcis</i>	2	tree	no	-	0.370	Anemoc	0.0057
<i>Spondias mombin</i>	1	tree	no	-	0.391	Zoo	0.0011
<i>Sterculia apetala</i>	1	tree	no	-	0.932	Zoo	0.0482
<i>Styrax pohlii</i>	3	tree	no	-	0.405	Zoo	0.0069
<i>Syzygium cumini</i>	15	tree	no	-	0.673	Zoo	1.8752
<i>Syzygium jambos</i>	1	tree	no	-	0.700	Zoo	0.0058
<i>Tabebuia roseoalba</i>	7	tree	no	-	0.762	Anemoc	0.0879
<i>Tabernaemontana catharinensis</i>	13	tree	no	-	0.526	Zoo	0.0514
<i>Tapirira guianensis</i>	27	tree	no	-	0.457	Zoo	0.2064
<i>Tecoma stans</i>	23	tree	no	-	0.466	Anemoc	0.1168
<i>Terminalia sp.</i>	3	tree	NA	-	0.630	NA	0.0476
<i>Terminalia triflora</i>	2	tree	no	-	0.748	Anemoc	0.0706
<i>Trema micrantha</i>	13	tree	no	-	0.319	Zoo	0.0977
<i>Trichilia casaretti</i>	1	tree	endemic	LC	0.644	Zoo	0.0003
<i>Trichilia catigua</i>	2	tree	endemic	-	0.688	Zoo	0.0157
<i>Trichilia clauseni</i>	2	tree	endemic	-	0.644	Zoo	0.0054
<i>Trichilia elegans</i>	6	tree	endemic	-	0.644	Zoo	0.0352
<i>Triplaris americana</i>	51	tree	no	-	0.490	Anemoc	1.4011
<i>Urera baccifera</i>	1	tree	no	-	0.165	Zoo	0.0017
<i>Vernonanthura polyanthes</i>	1	shrub	no	-	0.540	Anemoc	NA
<i>Vernonanthura sp.</i>	4	shrub	NA	-	0.540	Anemoc	NA
<i>Vitex polygama</i>	10	tree	endemic	-	0.565	Zoo	1.3369
<i>Xylopia aromatica</i>	2	tree	no	LC	0.561	Zoo	0.0331
<i>Zanthoxylum caribaeum</i>	4	tree	no	-	0.760	Zoo	0.1928

<i>Zanthoxylum fagara</i>	1	tree	no	-	0.651	Zoo	0.0006
<i>Zanthoxylum rhoifolium</i>	8	tree	no	-	0.569	Zoo	0.1186
<i>Zeyheria tuberculosa</i>	4	tree	no	VU	0.770	Anemoc	0.0726

LC: “Least Concern”; NT: “Near Threatened”; VU: Vulnerable; EN: “Endangered”; Zoo: Zoochorous; Anemoc: Anemochorous; Autoc: Autochorous; NA: Not Available data.

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