UNIVERSIDADE ESTADUAL DE CAMPINAS

INSTITUTO DE BIOLOGIA

CRISTINA YURI VIDAL

THE IMPORTANCE OF FOREST FRAGMENTS' BIODIVERSITY FOR CONSERVATION AND ECOLOGICAL RESTORATION WITHIN AGRICULTURAL LANDSCAPES

A IMPORTÂNCIA DE FRAGMENTOS FLORESTAIS PARA A CONSERVAÇÃO E RESTAURAÇÃO ECOLÓGICA EM PAISAGENS AGRÍCOLAS

CAMPINAS

2018

CRISTINA YURI VIDAL

THE IMPORTANCE OF FOREST FRAGMENTS' BIODIVERSITY FOR CONSERVATION AND ECOLOGICAL RESTORATION WITHIN AGRICULTURAL LANDSCAPES

A IMPORTÂNCIA DE FRAGMENTOS FLORESTAIS PARA A CONSERVAÇÃO E RESTAURAÇÃO ECOLÓGICA EM PAISAGENS AGRÍCOLAS

Thesis presented to the Biology Institute of the University of Campinas in partial fulfillment of the requirements for the PhD degree in *Plant Biology.*

Tese apresentada ao Instituto de Biologia da Universidade Estadual de Campinas como parte dos requisitos exigidos para a obtenção do título de Doutora em *Biologia Vegetal.*

ESTE ARQUIVO DIGITAL CORRESPONDE À VERSÃO FINAL DA TESE DEFENDIDA PELA ALUNA CRISTINA YURI VIDAL E ORIENTADA PELO RICARDO RIBEIRO RODRIGUES.

Orientador: *Prof. Dr. RICARDO RIBEIRO RODRIGUES*

CAMPINAS

2018

Agência(s) de fomento e nº(s) de processo(s): CAPES; CNPq, 870360/1997-3; FAPESP, 2013/50718-5
ORCID: https://orcid.org/0000-0003-0295-5348

Ficha catalográfica
Universidade Estadual de Campinas
Biblioteca do Instituto de Biologia
Mara Janaina de Oliveira - CRB 8/6972

V667i

Vidal, Cristina Yuri, 1980-
The importance of forest fragments' biodiversity for conservation and ecological restoration within agricultural landscapes / Cristina Yuri Vidal. – Campinas, SP : [s.n.], 2018.

Orientador: Ricardo Ribeiro Rodrigues.
Tese (doutorado) – Universidade Estadual de Campinas, Instituto de Biologia.

1. Biodiversidade - Conservação. 2. Ecologia de restauração. 3. Diversidade beta. 4. Florestas secundárias. 5. Viveiros florestais. I. Rodrigues, Ricardo Ribeiro. II. Universidade Estadual de Campinas. Instituto de Biologia. III. Título.

Informações para Biblioteca Digital

Título em outro idioma: A importância da biodiversidade dos fragmentos florestais para a conservação e restauração ecológica em paisagens agrícolas
Palavras-chave em inglês:
Biodiversity conservation
Restoration ecology
Beta diversity
Secondary forests
Forest nurseries
Área de concentração: Biologia Vegetal
Titulação: Doutora em Biologia Vegetal
Banca examinadora:
Ricardo Ribeiro Rodrigues [Orientador]
Letícia Couto Garcia
Adriana Maria Zanforlin Martini
Ingrid Koch
Carlos Alfredo Joly
Data de defesa: 13-12-2018
Programa de Pós-Graduação: Biologia Vegetal

COMISSÃO EXAMINADORA

Prof. Dr. Ricardo Ribeiro Rodrigues (orientador)

Profa. Dra. Letícia Couto Garcia

Profa. Dra. Adriana Maria Zanforlin Martini

Profa. Dra. Ingrid Koch

Prof. Dr. Carlos Alfredo Joly

Os membros da Comissão Examinadora acima assinaram a Ata de defesa, que se encontra no processo de vida acadêmica do aluno.

AGRADECIMENTOS

Ouvi dizer que “a gratidão é a memória do coração”, e se assim for, o meu guarda muitas memórias. Agradecer significa reconhecer que esta tese só foi possível porque ao longo da caminhada – que começou bem antes da minha matricula na Pós-Graduação – muitas pessoas influenciaram o que sou e o que me tornei.

Agradeço primeiramente à minha família e em especial aos meus pais, Mauro Vidal e Emiko Sato Vidal, que são os meus maiores exemplos: de caráter e senso ético, de generosidade e respeito às pessoas, de dedicação ao trabalho. São eles que sempre batalharam e investiram em minha educação e formação. São eles que torcem e comemoram por cada conquista em minha vida e é para eles que eu gostaria de retribuir um pouco do orgulho que tenho em ser filha.

Ao Tadeu Siqueira: não é tarefa fácil encontrar palavras para te agradecer. O coração aperta de tanta emoção…afinal esse doutorado, do começo ao fim, só foi possível graças ao seu apoio incondicional. Não só como o meu companheiro de vida, mas como o meu cientista favorito, pois conheço como ninguém a sua dedicação e conduta profissional. Se por vezes isso me fez reconher minhas inúmeras limitações, também me fez tentar melhorar, e no fim só agradeço a oportunidade de tê-lo como minha inspiração. Obrigada por todo o amor e carinho e por cuidar de mim com toda a paciência e compreensão durante as fases mais tensas do doutorado.

Agradeço por fazer parte do Laboratório de Ecologia e Restauração Florestal (LERF-ESALQ/USP), onde encontrei o sentido da minha profissão. Em especial, agradeço aos meus queridos orientadores, prof. Ricardo R. Rodrigues (doutorado) e prof. Sergius Gandolfi (mestrado), dois gigantes guerreiros do universo da restauração ecológica. Poucas vezes vi o cansaço no ombro deles, sempre empolgados e determinados na missão de propagar suas ideias em defesa do valor das florestas em pé, num processo constante de ensino e aprendizagem. Por me permitirem acompanhar e aprender com a atuação de vocês e por serem exemplos de como espalhar o conhecimento acadêmico e semeá-lo por aí...vou leva-los sempre comigo. Um agradecimento especial ao prof. Ricardo R. Rodrigues, que desde sempre demonstrou acreditar em mim muito mais do que eu mesma. Ricardo: obrigada por confiar no meu potencial, ainda que tantas vezes me pareceu loucura atingir às suas expectativas. Obrigada por todas as infinitas oportunidades de crescimento profissional que você me proporcionou e proporciona. Obrigada por ser a pessoa mais pragmática que eu conheço e por agregar tantas pessoas e ideias em prol da conservação e restauração das florestas.

Agradeço aos funcionários e colaboradores do LERF: Vanderlei Miranda, Francisco Antonioli, Giovana Oliveira, André G. Nave.

Aos amigos que fiz no LERF e na ESALQ desde o dia em que cheguei: vocês fazem parte de mim e ocupam um espaço gigante no meu coração. Que sorte a minha

as nossas histórias terem se cruzado em algum momento; tenho certeza de que vez ou outra nos encontraremos por aí. Vocês são muitos – afinal essa lista começou a ser formada em 2004 – e não cabe aqui a vontade de agradecer cada um com as palavras especiais que merecem (senão vira um romance!); mas vocês sabem quem são e o carinho que tenho por vocês. Obrigada por todo o aprendizado dentro e fora da bolha acadêmica; por tantas parcerias, discussões, risadas e lembranças juntos...vocês fazem tudo ser mais leve e divertido! Em especial, agradeço aos que acompanharam de perto as derrotas e conquistas do doutorado: Ana Paula Liboni, Julia R.S.A. Mangueira, Laíne C. Silveira, Rafaela P. Naves, Nino Amazonas, Cinthia Montibeller, Allan Camatta Mônico, Marina M. Duarte, Mariana M. Pardi, Thais M. Haddad, Débora C. Rother, Mariana Piva da Silva, Vivian Nasser Vilela.

Sem esquecer das muitas etapas para a conclusão da tese, reforço aqui os agradecimentos a todos os que compartilharam dados dos fragmentos florestais amostrados em seus projetos de pesquisa ou no Programa de Adequação Ambiental (LERF/ESALQ-USP) – em especial ao Fabiano T. Farah, Jeanette I. Miachir, Julia Raquel S.A. Mangueira, Ana Paula Liboni, Natália M. Ivanauskas, Vinicius C. Souza e Andréia A.Rezende. Agradeço também a todos os viveiristas participantes da minha pesquisa; vocês possuem um conhecimento valioso e cuidam de verdadeiros berçários de “arvores bebês”, tão importantes para o nosso futuro.

Um agradecimento especial à professora Marinez Ferreira de Siqueira, que me recebeu de braços abertos no Jardim Botânico do Rio de Janeiro e que elevou o meu projeto de pesquisa com as suas contribuições. Guardo uma profunda admiração pela sua generosidade, energia e pela sua competência profissional. Foi um prazer e uma honra passar 2 meses sob sua tutela. Agradeço também ao Diogo Souza Bezerra Rocha, que mesmo sem aviso prévio abraçou “a causa” e trabalhou intensamente nos modelos de distribuição das espécies.

Agradeço os colegas e amigos que me acolheram em Campinas (Carolina Potascheff, Ivan J.S. Diogo, Gustavo Shimizu, Elisa Cândido, Carol Devides, Larissa S. Pereira); no Rio de Janeiro (Polyana M. Nogueira) e infinitas vezes na “Piracicaba que eu adoro tanto” (Vivian Nasser Vilela, Aline Kamiya e Matheus P. Chagas, Cinthia Montibeller e Thais M. Hadadd, Ana Paula Liboni, Marta R.A. Muniz e Ronaldo, Marina M. Duarte, Mariana Piva da Silva, Débora C. Rother e Anani M. Zanini).

Por fim, agradeço ao Programa de Pós-Graduação em Biologia Vegetal (UNICAMP), à CAPES (código de financiamento 001) e CNPq (processo 870360/1997-3) pelas bolsas concedidas e à FAPESP (projeto temático processo 2013/507185). Aos professores que participaram das etapas de qualificação (Flavio A. Maes Santos, Sergius Gandolfi, Pedro H.S. Brancalion, Ingrid Koch), pré-banca (Ingrid Koch, Marinez F. Siqueira, Ricardo Dobrovolski) e banca final (Letícia C. Garcia, Adriana M.Z. Martini, Ingrid Koch e Carlos A. Joly): obrigada pelo tempo precioso investido na leitura crítica do manuscrito e pelas valiosas contribuições.

RESUMO

A conservação da biodiversidade em paisagens modificadas pelo homem é um dos maiores desafios da atualidade, sobretudo em regiões tropicais, onde a taxa de conversão de florestas em áreas agrícolas é elevada. O Estado de São Paulo (Brasil) representa um cenário típico de ocupação humana intensa, que resultou na redução de ecossistemas naturais de um dos *hotspots* do planeta: a Mata Atlântica. Particularmente no interior do estado, onde a cobertura florestal é extremamente reduzida e fragmentada, e onde predomina uma matriz agrícola intensa, com poucas Unidades de Conservação, as ações de conservação dependem dos fragmentos florestais em áreas privadas e da restauração ecológica de ecossistemas. Nessa tese, investigamos o papel dos fragmentos florestais em paisagens agrícolas para a conservação da biodiversidade e a restauração de áreas degradadas. No capítulo 1, investigamos a contribuição desses fragmentos em áreas privadas para as Unidades de Proteção Integral (UCS), descrevendo a representatividade e a distribuição da diversidade regional de espécies arbustivas e arbóreas. No capítulo 2, investigamos se os fragmentos florestais demonstram sinais de homogeneização ou heterogeneização biológica. Para tanto, modelamos a distribuição de 663 espécies (modelagem de nicho e distribuição potencial de espécies), assumindo ausência de fatores de fragmentação e isolamento dos habitats. Então comparamos a variação na composição de espécies (diversidade β) das comunidades modeladas *versus* as observadas. No capítulo 3, avaliamos a diversidade disponível para restauração ecológica em viveiros de espécies nativas, descrevendo (i) a variação da composição de espécies entre viveiros; e (ii) a representatividade das espécies produzidas localmente em relação as listas regionais. Para os capítulos 1 e 2 utilizamos dados de levantamentos florísticos de 367 fragmentos em propriedades privadas e 20 UCS, enquanto no capítulo 3 utilizamos dados detalhados sobre as espécies e quantidades de mudas produzidas por 54 viveiros de nativas. Os fragmentos florestais em áreas privadas possuem riqueza muito menor (56 ± 18) do que as UCS (260 ± 110), porém contemplam 59% de todas as espécies registradas em nosso estudo, incluindo raras e ameaçadas. Demonstramos uma enorme variação na composição de espécies entre os fragmentos (diversidade β elevada), e que essa variação é maior do que a registrada entre as comunidades modeladas, indicando o processo de heterogeneização biológica. Esses resultados sugerem que a heterogeneidade natural das florestas está aumentando, possivelmente em resposta ao isolamento e diferentes históricos de perturbação dos fragmentos florestais. Os viveiros de espécies nativas produzem um total de 561 espécies nativas, com média de 86.4 espécies por viveiro. Apesar dessa elevada diversidade, alguns grupos estão sub-representados, como as formas de vida não arbóreas, as espécies típicas de cerrado e as zoocóricas. A composição de espécies produzidas pelos viveiros é bastante dissimilar entre si, refletindo a variação constatada nos fragmentos florestais, de onde os propágulos para a produção de mudas são coletados. Os resultados dessa tese sugerem que a conservação e restauração da biodiversidade em paisagens agrícolas dependem de

abordagens amplas e inclusivas, que considerem todos os elementos da paisagem, sem negligenciar o valor dos fragmentos em propriedades privadas.

Palavras-chave: conservação da biodiversidade; restauração ecológica; paisagens agrícolas

ABSTRACT

Biodiversity conservation within human modified landscapes is one of the greatest challenges we face today, especially in tropical regions, where conversion of forests to agriculture prevails. São Paulo state in Brazil represents a typical scenario of intense human-induced occupation, which resulted on severe reduction of natural ecosystems of one of the planet's hotspots: the Atlantic Forest. Particularly in the countryside of the state, where forest cover is extremely reduced and fragmented, and where intensive agriculture prevails, with few Strictly Protected Areas (SPAs), conservation initiatives depend on forest fragments within private lands and on ecological restoration of ecosystems. In this study, we investigated the role of forest fragments within agricultural landscapes for biodiversity conservation and restoration. In chapter 1, we evaluated the contribution of forest fragments within private lands to conventional SPAs, describing the representativeness and distribution of regional tree and shrub diversity. In chapter 2, we investigated whether forest fragments are under a taxonomic homogenization or heterogenization process. To do so, we modelled the distribution of 663 species (environmental niche and potential species distribution modeling), assuming the lack of habitat fragmentation and isolation. Then, we compared the species composition variation (β-diversity) considering observed *versus* modelled data. In chapter 3 we evaluated the diversity available for ecological restoration in native plant nurseries, describing (i) the species composition variation among plant nurseries; and (ii) the representativeness of locally produced species in relation to regional floras. In chapters 1 and 2 we used floristic surveys of 367 forest fragments in private lands and 20 in SPAs, while in chapter 3 we used detailed information on the species and quantities of seedlings produced by 54 native plant nurseries. Forest fragments in private lands have much lower species richness (56 ± 18) than in SPAs (260 ± 110), but they harbor 59% of all species registered in our study, including threatened and rare species. We showed a great species composition variation among forest fragments (high β diversity) and that this variation is much higher than the variation registered among modelled communities, indicating a biotic heterogenization process. These results suggest the natural heterogeneity of forests is increasing, possibly in response to isolation and unique disturbance histories of forest fragments. Native plant nurseries produced a total of 561 native species, with 86.4 average per nursery. Despite the high diversity, some groups are underrepresented, such as non-tree, savanna specialists and animal-dispersed species. Species composition is very dissimilar among plant nurseries, reflecting the variation observed on forest fragments, where propagules are collected for seedlings' production. Our results suggest that biodiversity conservation and restoration within agricultural landscapes depend on wide and inclusive approaches, which comprehend all elements of a landscape, without neglecting the role of forest fragments in private lands.

Key-words: biodiversity conservation; ecological restoration; agricultural landscapes

SUMÁRIO

INTRODUÇÃO GERAL

As **florestas tropicais** representam os ecossistemas terrestres mais complexos e diversos do nosso planeta, onde se concentram mais da metade da biodiversidade e um terço de toda a produtividade terrestre (Malhi et al. 2014). No entanto, possuem um histórico de **conversão, fragmentação e degradação** de habitats proporcional à sua magnitude: mais de 50% das florestas tropicais já foram desmatadas e convertidas, enquanto o padrão de desmatamento nos trópicos prevalece crescente até hoje (Hansen et al. 2013). As interações e modificações causadas pelo homem começaram há dezenas ou centenas de anos com o intuito de promover a habitação humana e o cultivo agrícola, tornando-se mais acentuadas ao longo do século XIV e atingindo seu auge no final do século XX (Malhi et al. 2014; Newbold et al. 2015). Uma das principais consequências da perda de habitat nos trópicos é a atual crise de biodiversidade, decorrente do efeito combinado da extinção das espécies nativas e da disseminação das exóticas, sendo considerada por alguns autores como a 6ª extinção em massa (Dirzo et al. 2014; Alroy 2017; Ceballos et al. 2017).

Nesse cenário, **as florestas tropicais** que sobraram não estão livres da **influência humana**, já que em boa parte são pequenas, isoladas e sob efeitos de borda: 50% das florestas do mundo estão localizadas a uma distância de até 500 metros da borda e 20% estão a 100 metros (Haddad et al. 2015). Nessas condições, as florestas – protegidas ou não - estão sujeitas às mais variadas formas de perturbação: extração seletiva de espécies, defaunação, incidência de fogo, invasões biológicas, modificações microclimáticas (luz, temperatura e umidade) etc. (Malhi et al. 2014; Haddad et al. 2015; Bello et al. 2015; Jones et al. 2018; Barlow et al. 2018). O efeito combinado desses distúrbios com o uso do solo do entorno pode ter diferentes intensidades, sendo mais severo em regiões com atividades intensivas como os cultivos agrícolas, as pastagens e as áreas urbanas (Gibson et al. 2011; McGill et al. 2015; Mendenhall et al. 2016). Entre as diversas consequências relatadas na literatura, destacamos a **redução local da riqueza de espécies** como uma das variáveis mais avaliadas em resposta à perda e/ou degradação dos habitats, embora com grandes variações entre os tipos de impacto, as regiões geográficas, os biomas e os grupos taxonômicos avaliados (Gibson et al. 2011; Murphy & Romanuk 2014; Newbold et al. 2015). A perda da riqueza biológica resulta em alterações na

distribuição e composição de espécies em paisagens modificadas pelo homem, onde dois processos principais têm sido descritos na literatura. O primeiro deles é a **homogeneização biótica**, definida como a convergência, no espaço e no tempo, de comunidades que gradualmente sofrem uma simplificação das suas diversidades genética, taxonômica e/ou funcional (Olden & Rooney 2006). Isso ocorre porque algumas espécies são mais vulneráveis à extinção local, sobretudo aquelas que possuem capacidade limitada de dispersão, populações pequenas, ciclos de vida muito longos ou especialização à habitats específicos (Gibson et al. 2011; McGill et al. 2015; Mendenhall et al. 2016; Beca et al. 2017; Pfeifer et al. 2017). Essas espécies são amplamente conhecidas como espécies "*perdedoras*", enquanto um subconjunto de espécies generalistas com características que favorecem sua proliferação em habitats perturbados são conhecidas como "*ganhadoras*" (McKinney & Lockwood 1999; Silva & Tabarelli 2000; Lôbo et al. 2011; Tabarelli et al. 2012). O segundo processo descrito em paisagens modificadas pelo homem é a **heterogeneização biótica**, em que a composição das comunidades torna-se mais divergente ao longo do espaço e do tempo (Laurance et al. 2007; Socolar et al. 2016; Olden et al. 2018). Essa divergência pode ser explicada pelas diferentes frequências e intensidades de distúrbio combinadas à heterogeneidade ambiental típica de florestas tropicais, que são reforçadas por eventuais limitações de dispersão das espécies, decorrentes do isolamento dos habitats fragmentados (Arroyo-Rodríguez et al. 2013; Solar et al. 2015; Sfair et al. 2016).

Considerando essas recentes evidências registradas em paisagens modificadas pelo homem, um dos maiores **desafios para a conservação** na atualidade é manter a biodiversidade remanescente em condições tão alteradas, sobretudo em **paisagens agrícolas** (Gardner et al. 2009; Seppelt et al. 2016; Dobrovolski, Diniz-Filho, et al. 2011; Mendenhall et al. 2016). A abordagem convencional baseia-se na criação de **Unidades de Conservação** (Mendenhall et al. 2011; Barlow et al. 2018) , prática que começou na década de 70 e que cresceu substancialmente ao redor do mundo a partir da década de 90 (Jenkins & Joppa 2009; Oliveira et al. 2017). Apesar da valiosa contribuição dessas áreas protegidas para a preservação de espécies e de ecossistemas (Andam et al. 2008; Joppa et al. 2008; Carranza et al. 2014; Coetzee et al. 2014; Gray et al. 2016), elas ainda não são suficientes ou representativas considerando-se as diferentes dimensões da

diversidade (*e.g*. taxonômica, funcional e filogenética) e as mudanças climáticas, sobretudo em regiões de elevado potencial econômico (Rodrigues et al. 2004; Andam et al. 2008; Joppa et al. 2008; Jenkins & Joppa 2009; Lemes et al. 2014; Bartonova et al. 2016; Bergamin et al. 2017; Oliveira et al. 2017; Jones et al. 2018; Saraiva et al. 2018). Além disso, as unidades de conservação estendem-se por apenas 13% dos biomas terrestres, com menos de 6% em categorias de proteção restrita (Jenkins & Joppa 2009), o que deixa claro que o futuro da biodiversidade terrestre depende em grande parte das áreas não protegidas (Gardner et al. 2009; Mendenhall et al. 2016).

Na região das florestas tropicais, os **fragmentos florestais** fora das Unidades de Conservação são compostos por remanescentes de florestas primárias e por florestas secundárias, caracterizados por diversos níveis de perturbação (Malhi et al. 2014). Apesar da constatação de forte redução da riqueza local nesses fragmentos florestais em relação às florestas preservadas (Gibson et al. 2011; Canale et al. 2012; McGill 2015; Gray et al. 2016; Mendenhall et al. 2016; Sfair et al. 2016; Alroy 2017; Beca et al. 2017; Farah et al. 2017; Saraiva et al. 2018; Solar et al. 2015), muitos estudos reconhecem o valor que esses fragmentos desempenham no suporte e manutenção da diversidade em paisagens modificadas pelo homem (Silva & Tabarelli 2000; Arroyo-Rodríguez et al. 2008; Mendenhall et al. 2011; Joly et al. 2014; Solar et al. 2015; Sfair et al. 2016; Beca et al. 2017; Farah et al. 2017). Em regiões onde a cobertura vegetal foi muito reduzida, os objetivos de conservação dos ecossistemas e dos seus serviços associados (*e.g.* sequestro de carbono, proteção do solo, provisão de água etc.) dependem não só da manutenção do que restou, mas também da **restauração** desses **ecossistemas** (Calmon et al. 2011; Holl & Aide 2011; Rodrigues et al. 2011; Brancalion et al. 2013; Martínez-Ramos et al. 2013; Vidal et al. 2016; Meli, Herrera, et al. 2017). A relevância dessa abordagem é constatada pelo enorme desenvolvimento da Ecologia da Restauração nas últimas décadas, além dos exemplos recentes de iniciativas e compromissos de ações de restauração nos âmbitos subnacionais, nacionais e internacionais (Rodrigues et al. 2009; Suding 2011; Aronson & Alexander 2013; Holl 2017): em 2006 o movimento PACTO pela restauração da Mata Atlântica estabeleceu a meta de restaurar 15 milhões de hectares di bioma até 2050; em 2011 foi lançado o Desafio de Bonn, que possui a meta de restaurar 350 milhões de hectares de áreas degradadas no mundo até 2030; e em 2017 o governo brasileiro anunciou o Plano Nacional de Recuperação da Vegetação

Nativa (PLANAVEG), com meta de restaurar 12 milhões de hectares até 2030 (Scaramuzza et al. 2017). O cumprimento dessas metas ambiciosas de restauração, sobretudo de florestas tropicais, representa um enorme desafio para a ciência e para a prática da restauração, principalmente quando a recuperação da biodiversidade é um dos objetivos almejados (Rodrigues et al. 2009; Wright et al. 2009; Aerts & Honnay 2011; Rey Benayas & Bullock 2012; Brancalion & Holl 2015; Mayfield 2016).

No contexto delineado até aqui, o estado de São Paulo representa uma oportunidade muito interessante de estudo de caso, pois possui um **longo histórico de ocupação** e conversão de uso do solo que resultou em extensas **paisagens agrícolas** (*e.g.* canaviais e pastagens), onde originalmente se distribuíam dois biomas que, hoje, são considerados ***hotspots*** de diversidade: a Mata Atlântica e o Cerrado (Myers et al. 2000; Metzger 2009; Ribeiro et al. 2009; Sparovek et al. 2010; Dobrovolski, Loyola, et al. 2011). Devido a esse potencial agrícola, as regiões mais interioranas do estado possuem poucas Unidades de Conservação de proteção integral (Durigan et al. 2006; Ribeiro et al. 2009; Joly et al. 2014) e os fragmentos florestais em propriedades privadas são predominantemente compostos por florestas secundárias de tamanho pequeno (*i.e.* <50ha) (Ribeiro et al. 2009; Beca et al. 2017; Farah et al. 2017), sujeitos às perturbações recorrentes que os mantêm em estado de sucessão estagnada (Brancalion et al. 2013; Arroyo-Rodriguez et al. 2015; Ghazoul et al. 2015). Muitos estudos indicam que, ao se reduzir a cobertura vegetal a aproximadamente 30%, as paisagens atingem um **limiar de fragmentação,** a partir do qual ocorrem perdas de resiliência e biodiversidade potencialmente irreversíveis (Pardini et al. 2010; Tabarelli 2010; Martensen et al. 2012; Estavillo et al. 2013; Rigueira et al. 2013; Banks-Leite et al. 2014; Lima & Mariano-Neto 2014; Benchimol et al. 2017). Nas paisagens hiper-fragmentadas do estado de SP, com menos de 30% de cobertura florestal (Ribeiro et al. 2009; Beca et al. 2017; Farah et al. 2017), existe pouco espaço para uma expansão significativa das Unidades de Conservação de uso restrito, o que impõe um grande desafio ao estabelecimento de uma estratégia conservacionista alinhada com o potencial agrícola e com os interesses econômicos do estado com o maior Produto Interno Bruto (PIB) do Brasil (Joly et al. 2010; Sayer et al. 2013; Vidal et al. 2016). Além da criação de Unidades de Conservação, existem pelo menos duas outras ações possíveis nas propriedades rurais inseridas nessas paisagens modificadas (Vidal et al. 2016): i) o manejo adaptativo para retomar e/ou

redirecionar as trajetórias sucessionais de fragmentos estagnados e áreas em regeneração natural (Brancalion et al. 2012; Arroyo-Rodriguez et al. 2015); ii) o aumento da cobertura vegetal e da conectividade dos habitats por meio das ações de restauração, que podem ter foco exclusivo na restauração ecológica da diversidade e, em alguns casos, podem ser consorciados com propósitos econômicos (Brancalion et al. 2013; Garcia et al. 2013; Martínez-Ramos et al. 2013; Ghazoul et al. 2015; Amazonas et al. 2018).

O principal dispositivo legal que regulamenta a conservação e restauração em propriedades rurais é a **Lei de Proteção da Vegetação Nativa** (Lei 12.651/2012), que modificou o antigo Código Florestal (Lei 4.771/1965) e resultou em avanços e retrocessos ambientais (Sparovek et al. 2010; Garcia et al. 2013; Soares-Filho et al. 2014; Brancalion et al. 2016a; Scaramuzza et al. 2016). As discussões ao longo dessas mudanças foram conduzidas sob grande pressão dos diferentes grupos envolvidos, expondo não só o clássico embate entre o setor agrícola e o ambiental, mas também a dificuldade ou falta de diálogo entre os cientistas e os tomadores de decisão, traduzidos em falhas e lacunas nas políticas públicas, mesmo quando as evidências estiveram disponíveis na literatura acadêmica (Joly et al. 2014; Loyola 2014; Young et al. 2014; Brancalion et al. 2016a). Para transpor esse gargalo, uma alternativa promissora é o desenvolvimento de estudos em escalas espaciais próximas ou proporcionais às escalas administrativas em que as tomadas de decisão são executadas (*e.g.* municipal, estadual) (Gardner et al. 2013).

ESCOPO e ESTRUTURA da TESE

Apesar do potencial que os fragmentos florestais em áreas privadas possuem no suporte à conservação e restauração da biodiversidade em paisagens agrícolas, esses fragmentos geralmente são estudados pontualmente e de forma dispersa (Lima et al. 2015), com pouco conhecimento a respeito do seu valor coletivo e em contextos geográficos amplos. A disponibilidade de uma grande quantidade de dados florísticos amostrados fora das Unidades de Conservação pelo Laboratório de Ecologia e Restauração Florestal (LERF/ESALQ-USP) foi um dos grandes motivadores desta pesquisa, que teve o intuito de aproveitar uma boa oportunidade de se gerar conhecimento para o refinamento das políticas públicas do estado. Com

o objetivo de investigar o potencial de contribuição dos fragmentos em propriedades privadas para a conservação e restauração da biodiversidade regional, três estudos foram desenvolvidos nesta tese.

No **capítulo 1,** descrevemos como a diversidade estava distribuída entre os fragmentos florestais em Unidades de Conservação de Proteção Integral (n=20) e em propriedades privadas (n=367), localizados nas regiões Sudeste, Centro e Oeste do estado de São Paulo. Utilizamos dados de levantamentos florísticos de espécies arbustivas e arbóreas para descrever a ocorrência (exclusiva ou compartilhada) e a frequência das espécies (raras ou comuns), além de quantificar as ameaçadas de extinção. Também avaliamos a variação da composição de espécies entre as comunidades (diversidade β) e os seus componentes *turnover* e aninhamento, sendo que o *turnover* representa as diferenças decorrentes da substuição das espécies entre comunidades enquanto o aninhamento representa as diferenças entre os sítios mais ricos e os mais pobres em espécies, considerando que os mais pobres são compostos por sub-conjuntos de espécies dos sítios mais ricos. Nas paisagens hiper-fragmentadas deste estudo, encontramos valores reduzidos de riqueza local (diversidade α) nas propriedades privadas, porém com valores elevados de diversidade β e do seu componente *turnover*; o que indicou uma grande variação da composição das espécies entre os fragmentos, dando suporte ao valor coletivo dos mesmos para a conservação da biodiversidade regional.

No **capítulo 2**, utilizando o mesmo conjunto de dados do capítulo 1, investigamos qual processo está em curso nas paisagens hiper-fragmentadas da nossa região de estudo: homogeneização ou heterogeneização biótica. Devido à inexistência de registros florísticos consistentes antes do amplo desmatamento do estado, aplicamos a modelagem de distribuição potencial de espécies (SDM) para estimar a riqueza e a composição sem a influência do processo de fragmentação dos habitats – portanto esses valores potenciais, estimados pela modelagem, nos serviu como um equivalente à uma réplica temporal (*i.e.* pré-desmatamento). Dessa forma, foi possível comparar as mudanças na diversidade β ao longo do espaço e do “tempo”, além de ter permitido comparações de eventuais padrões nas diferentes regiões do estado e entre as propriedades privadas e Unidades de Conservação. Registramos uma riqueza local (diversidade α) bastante reduzida (cerac de 3.8 vezes menor) em

relação à riqueza estimada pela SDM, sobretudo nos fragmentos em propriedades privadas. A diversidade β, no entanto, foi maior para os dados observados em relação aos estimados pela SDM, indicando uma tendência geral à heterogeneização biótica. A comparação entre o observado e a SDM revelou ainda que, em nossas paisagens hiper-fragmentadas, tanto a diferenciação quanto a homogeneização estão ocorrendo em consequência da perda local de espécies, reforçando a complexidade de respostas e dos processos que afetam a biodiversidade. As diferenças entre diversidade β observada e SDM foram significativas e consistentes em todas as regiões avaliadas e para as propriedades privadas, sendo as Unidades de Conservação a única exceção. Esses resultados reforçaram que, além da manutenção das Unidades de Conservação, é necessário mitigar a perda de espécies e o isolamento dos fragmentos florestais (*i.e.* transpor as limitações de dispersão) por meio de i) ações de manejo adequadas às particularidades dos habitats remanescentes (*e.g.* enriquecimento com espécies e/ou grupos funcionais ausentes, controle de espécies hiper-abundantes e/ou invasoras); e ii) restauração da conectividade entre os fragmentos.

Para o desenvolvimento do **capítulo 3**, assumimos a relevância dos fragmentos florestais como fonte de propágulos, tanto para o processo de regeneração natural, quanto para a restauração ativa (*e.g.* manejo de fragmentos, restauração de áreas degradadas), uma vez que a produção de mudas para esse propósito depende da coleta de frutos e sementes em remanescentes nas propriedades privadas. A restauração ativa é o principal método recomendado para aumentar a cobertura vegetal nativa em paisagens hiper-fragmentadas, onde o isolamento dos fragmentos dificulta ou impede a dispersão das espécies. Para entender melhor o potencial das ações de restauração ecológica no estado, avaliamos a diversidade disponível nos viveiros florestais de nativas, aproveitando para fazer um diagnóstico atualizado deles. Avaliamos a proporção da diversidade disponível nos viveiros em relação às listas regionais oficiais - fornecidas pela Secretaria do Meio Ambiente e pelo Instituto de Botânica do estado de São Paulo - e em relação ao conjunto de dados do capítulo 1. Também calculamos a variação da composição de espécies entre os viveiros e a relação da diversidade disponível com fatores como a capacidade produtiva, a cobertura vegetal e os tipos de vegetação do entorno. Na maior cadeia produtiva do país, encontramos uma riqueza de espécies surpreendente,

embora ainda parcialmente representativa da flora regional e com um viés de produção das espécies arbustivas e arbóreas. Destacamos ainda que a produção dos viveiros é bastante dissimilar entre si, refletindo a variação e a heterogeneidade dos fragmentos florestais do entorno, conforme constatado nos capítulos anteriores (elevada diversidade β e *turnover*).

Encerramos a tese com uma **discussão geral** e abrangente dos resultados gerados por esta tese, com as considerações sobre suas implicações para as políticas públicas do estado. Além da importância das Unidades de Conservação de Proteção Integral, os dados gerados demonstraram que a diversidade remanescente está distribuída entre os fragmentos florestais nas propriedades privadas, que devem ser protegidos em seu conjunto, por meio da aplicação efetiva da Lei de Proteção da Vegetação Nativa (Lei 12.651/2012). Nesse contexto, as Áreas de Preservação Permanente e as Reservas Legais merecem maior atenção, por meio de incentivos e programas específicos para o seu manejo e conservação. Recomendamos que as ações de restauração sejam inclusivas, considerando tanto i) o manejo de fragmentos florestais com o intuito de potencializar seu papel de conservação, quanto ii) a restauração de áreas degradadas, para reestabelecer corredores que aumentem a cobertura florestal e promovam a conectividade funcional dos remanescentes. Por fim, destacamos que a qualidade das ações de restauração está fortemente associada i) aos fragmentos florestais de onde os propágulos para a produção de mudas nativas são coletados e ii) do desempenho do setor de propagação e produção de mudas, que possui grande demanda por políticas efetivas de fomento, capacitação e assistência técnica permanente.

REFERÊNCIAS

Aerts R, Honnay O (2011) Forest restoration, biodiversity and ecosystem functioning. BMC Ecol 11:1472–6785

Alroy J (2017) Effects of habitat disturbance on tropical forest biodiversity. Proceedings of the National Academy of Sciences 114:6056–6061

Amazonas NT et al. (2018) High diversity mixed plantations of Eucalyptus and native trees: an interface between production and restoration for the tropics. Forest Ecology and Management 417:247–256

Andam KS et al. (2008) Measuring the effectiveness of protected area networks in reducing deforestation. Proceedings of the National Academy of Sciences of the United States of America 105:16089–16094

Aronson J, Alexander S (2013) Ecosystem restoration is now a global priority: Time to roll up our sleeves. Restoration Ecology 21:293–296

Arroyo-Rodriguez V et al. (2015) Multiple successional pathways in human-modified tropical landscapes: New insights from forest succession, forest fragmentation and landscape ecology research. Biological Reviews

Arroyo-Rodríguez V et al. (2013) Plant β-diversity in fragmented rain forests: Testing floristic homogenization and differentiation hypotheses. Journal of Ecology 101:1449–1458

Arroyo-Rodríguez V et al. (2008) Value of small patches in the conservation of plant-species diversity in highly fragmented rainforest. Conservation Biology 23:729–39

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

Barlow J et al. (2018) The future of hyperdiverse tropical ecosystems. Nature

Bartonova A et al. (2016) How universal are reserve design rules? A test using butterflies and their life history traits. Ecography 39:456–464

Beca G et al. (2017) High mammal species turnover in forest patches immersed in biofuel plantations. Biological Conservation

Bello C et al. (2015) Defaunation affects carbon storage in tropical forests. Science Advances 1:1–11

Benchimol M et al. (2017) Translating plant community responses to habitat loss into conservation practices: Forest cover matters. Biological Conservation 209:499–507

Bergamin RS et al. (2017) Linking beta diversity patterns to protected areas: lessons from the Brazilian Atlantic Rainforest. Biodiversity and Conservation

Brancalion PHS et al. (2016) A critical analysis of the Native Vegetation Protection Law of Brazil (2012): Updates and ongoing initiatives. Natureza e Conservacao 14:1–15

Brancalion PHS et al. (2012) Estratégias para auxiliar na conservação de florestas tropicais secundárias inseridas em paisagens alteradas. Bol. Mus. Para. Emílio Goeldi 900:219–234

Brancalion PHS, Holl KD (2015) Functional composition trajectory: A resolution to the debate between Suganuma, Durigan, and Reid. Restoration Ecology 24:1–3

Brancalion PHS et al. (2013) Biodiversity persistence in highly human- modified tropical landscapes depends on ecological restoration. Tropical Conservation Science 6:705–710

Calmon M et al. (2011) Emerging Threats and Opportunities for Large-Scale Ecological Restoration in the Atlantic Forest of Brazil. Restoration Ecology 19:154–158

Canale GR et al. (2012) Pervasive defaunation of forest remnants in a tropical biodiversity hotspot. PLoS ONE 7

Carranza T et al. (2014) Protected area effectiveness in reducing conversion in a rapidly vanishing ecosystem: The Brazilian Cerrado. Conservation Letters 7:216–223

Ceballos G, Ehrlich PR, Dirzo R (2017) Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. Proceedings of the National Academy of Sciences 201704949

Coetzee BWT, Gaston KJ, Chown SL (2014) Local scale comparisons of biodiversity as a test for global protected area ecological performance: A meta-analysis. PLoS ONE 9

Dirzo R et al. (2014) Defaunation in the Anthropocene. Science 345:401–406

Dobrovolski R, Diniz-Filho JAF, et al. (2011) Agricultural expansion and the fate of global conservation priorities. Biodiversity and Conservation 20:2445–2459

Dobrovolski R, Loyola RD, et al. (2011) Agricultural expansion can menace Brazilian protected areas during the 21 st century. Natureza e Conservacao 9:208–213

Durigan G et al. (2006) Seleção De Fragmentos Prioritários Para a Criação De Unidades De Conservação Do Cerrado No Estado De São Paulo. Revista do Instituto Florestal 18:23–37

Estavillo C, Pardini R, Da Rocha PLB (2013) Forest loss and the biodiversity threshold: An evaluation considering species habitat requirements and the use of matrix habitats. PLoS ONE 8:1–10

Farah FT et al. (2017) Integrating plant richness in forest patches can rescue overall biodiversity in human-modified landscapes. Forest Ecology and Management 397:78–88

Garcia LC et al. (2013) Restoration Challenges and Opportunities for Increasing Landscape Connectivity under the New Brazilian Forest Act. Natureza&Conservação 11:181–185

Gardner T a. et al. (2009) Prospects for tropical forest biodiversity in a human-modified world. Ecology Letters 12:561–582

Ghazoul J et al. (2015) Conceptualizing Forest Degradation. Trends in Ecology and Evolution 30:622–632

Gibson L et al. (2011) Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478:378–81

Gray CL et al. (2016) Local biodiversity is higher inside than outside terrestrial protected areas worldwide. Nature Communications 7

Haddad NM et al. (2015) Habitat fragmentation and its lasting impact on Earth ' s ecosystems. Science 1–9

Hansen MC et al. (2013) High-resolution global maps of 21st-century forest cover change. Science (New York, N.Y.) 342:850–3

Holl KD (2017) Restoring tropical forests from the bottom up. Science 355:455–456

Holl KD, Aide TM (2011) When and where to actively restore ecosystems? Forest Ecology and Management 261:1558–1563

Jenkins CN, Joppa L (2009) Expansion of the global terrestrial protected area system. Biological Conservation 142:2166–2174

Joly CA et al. (2010) Biodiversity Conservation Research, Training, and Policy in Sao Paulo. Science 328:1358–1359

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

Jones KR et al. (2018) One-third of global protected land is under intense human pressure. Science 360:788–791

Joppa LN, Loarie SR, Pimm SL (2008) On the protection of 'protected areas'. Proceedings of the National Academy of Sciences 105:6673–6678

Laurance WF et al. (2007) Habitat fragmentation, variable edge effects, and the landscape-divergence hypothesis. PLoS ONE 2

Lemes P, Melo AS, Loyola RD (2014) Climate change threatens protected areas of the Atlantic Forest. Biodiversity and Conservation 23:357–368

Lima MM, Mariano-Neto E (2014) Extinction thresholds for Sapotaceae due to forest cover in Atlantic Forest landscapes. Forest Ecology and Management 312:260–270

Lima RAF de et al. (2015) How much do we know about the endangered Atlantic Forest? Reviewing nearly 70 years of information on tree community surveys. Biodiversity and Conservation

Lôbo D et al. (2011) Forest fragmentation drives Atlantic forest of northeastern Brazil to biotic homogenization. Diversity and Distributions 17:287–296

Loyola R (2014) Brazil cannot risk its environmental leadership. Diversity and Distributions 20:1365–1367

Malhi Y et al. (2014) Tropical Forests in the Anthropocene. Annual Review of Environment and Resources 39:125–159

Martensen AC et al. (2012) Associations of Forest Cover, Fragment Area, and Connectivity with Neotropical Understory Bird Species Richness and Abundance. Conservation Biology 26:1100–1111

Martínez-Ramos M et al. (2013) Natural forest regeneration and ecological restoration in human modified tropical landscapes. Biotropica 48:745–757

Mayfield MM (2016) Restoration of tropical forests requires more than just planting trees, a lot more... Applied Vegetation Science 19:553–554

McGill B (2015) Land use matters. Nature 520:38–39

McGill BJ et al. (2015) Fifteen forms of biodiversity trend in the anthropocene. Trends in Ecology and Evolution 30:104–113

McKinney ML, Lockwood JL (1999) Biotic homogenization: A few winners replacing many losers in the next mass extinction. Trends in Ecology and Evolution 14:450–453

Meli P et al. (2017) Four approaches to guide ecological restoration in Latin America. Restoration Ecology 25:156–163

Mendenhall CD et al. (2011) Predictive model for sustaining biodiversity in tropical countryside. Proceedings of the National Academy of Sciences 108:16313–16316

Mendenhall CD et al. (2016) Quantifying and sustaining biodiversity in tropical agricultural landscapes. Proceedings of the National Academy of Sciences 113:14544–14551

Metzger JP (2009) Conservation issues in the Brazilian Atlantic forest. Biological Conservation 142:1138–1140

Murphy GEP, Romanuk TN (2014) A meta-analysis of declines in local species richness from human disturbances. Ecology and Evolution 4:91–103

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

Newbold T et al. (2015) Global effects of land use on local terrestrial biodiversity. Nature

Olden JD, Comte L, Giam X (2018) The Homogocene: a research prospectus for the study of biotic homogenisation. NeoBiota 37:23–36

Olden JD, Rooney TP (2006) On defining and quantifying biotic homogenization. Global Ecology and Biogeography 15:113–120

Oliveira U et al. (2017) Biodiversity conservation gaps in the Brazilian protected areas. Scientific Reports 7:1–9

Pardini R et al. (2010) Beyond the fragmentation threshold hypothesis: regime shifts in biodiversity across fragmented landscapes. PloS one 5:e13666

Pfeifer M et al. (2017) Creation of forest edges has a global impact on forest vertebrates. Nature 551:187–191

Rey Benayas JM, Bullock JM (2012) Restoration of Biodiversity and Ecosystem Services on Agricultural Land. Ecosystems 15:883–899

Ribeiro MC et al. (2009) The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biological Conservation 142:1141–1153

Rigueira DMG, da Rocha PLB, Mariano-Neto E (2013) Forest cover, extinction thresholds and time lags in woody plants (Myrtaceae) in the Brazilian Atlantic Forest: Resources for conservation. Biodiversity and Conservation 22:3141–3163

Rodrigues ASL et al. (2004) Effectiveness of the global protected area network in representing species diversity. Nature 428:9–12

Rodrigues RR et al. (2011) Large-scale ecological restoration of high-diversity tropical forests in SE Brazil. Forest Ecology and Management 261

Rodrigues RR et al. (2009) On the restoration of high diversity forests: 30 years of experience in the Brazilian Atlantic Forest. Biological Conservation 142:1242–1251

Saraiva DD et al. (2018) How effective are protected areas in conserving tree taxonomic and phylogenetic diversity in subtropical Brazilian Atlantic Forests? Journal for Nature Conservation 42:28–35

Sayer J et al. (2013) Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. Proceedings of the National Academy of Sciences 110:8349–8356

Scaramuzza CA de M et al. (2016) Elaboração da proposta do plano nacional de recuperação da vegetação nativa. In: Mudanças no Código Florestal Brasileiro: desafios para implementação da nova lei. Silva, APM Da, Marques, HR, & Sambuichi, RHR, editors. Brasilia pp. 185–208.

Scaramuzza CA de M et al. (2017) A Política Nacional de Recuperação da Vegetação Nativa: Lições apendidas. In: Economia da Restauração Florestal. Benini, R de M & Adeodat, editors. The Nature Conservancy, São Paulo p. 135

Seppelt R et al. (2016) Harmonizing Biodiversity Conservation and Productivity in the Context of Increasing Demands on Landscapes. BioScience 66:890–896

Sfair JC et al. (2016) Taxonomic and functional divergence of tree assemblages in a fragmented tropical forest. Ecological Applications 2:1816–1826

Silva JMC da, Tabarelli M (2000) Tree species impoverishment and the future flora of the Atlantic forest of northeast Brazil. Nature 404:72–74

Soares-filho B et al. (2014) Cracking Brazil ' s Forest Code. Science 344:363–364

Socolar JB et al. (2016) How Should Beta-Diversity Inform Biodiversity Conservation? Trends in Ecology and Evolution 31:67–80

Solar RRDC et al. (2015) How pervasive is biotic homogenization in human-modified tropical forest landscapes? Ecology Letters n/a-n/a

Sparovek G et al. (2010) Brazilian agriculture and environmental legislation:status and future challenges. Environmental science & technology 44:14637–14641

Suding KN (2011) Toward an Era of Restoration in Ecology: Successes, Failures, and Opportunities Ahead. Annual Review of Ecology, Evolution, and Systematics 42:465–487

Tabarelli M (2010) Tropical Biodiversity in Human-Modified Landscapes: What is our Trump Card? Biotropica 42:553–554

Tabarelli M, Peres C a., Melo FPL (2012) The 'few winners and many losers' paradigm revisited: Emerging prospects for tropical forest biodiversity. Biological Conservation 155:136–140

Vidal CY et al. (2016) Biodiversity Conservation of Forests and their Ecological Restoration in Highly-modified Landscapes. In: Biodiversity in Agricultural Landscapes of Southeastern Brazil. Gheler-Costa, C, Lyra-Jorge, MC, & Verdade, LM, editors. De Gruyter Open Ltd, Warsaw/Berlin p. 342.

Wright J et al. (2009) Restoring biodiversity and ecosystem function: will an integrated approach improve results? In: Biodiversity, ecosystem functioning, & human wellbeing. An ecological and economic perspective. Naeem, S, Bunker, DE, Hector, A, Loreau, M, & Perrings, C, editors. Oxford University Press pp. 167–177.

Young JC et al. (2014) Improving the science-policy dialogue to meet the challenges of biodiversity conservation: having conversations rather than talking at one-another. Biodiversity and Conservation 23:387–404

CHAPTER 1. TROPICAL FOREST CONSERVATION WITHIN AGRICULTURAL LANDSCAPES: PRIVATE LANDS AND THEIR SUPPORT TO STRICTLY PROTECTED AREAS

Cristina Yuri Vidal* [1, 2]; Ana Paula Liboni[1,2], Julia Raquel de Sá Abílio Mangueira[1,2], Fabiano Turini Farah[3], Jeanette Inamine Miachir[4], Natalia Macedo Ivanauskas[5], Vinicius Castro Souza[2], Débora Cristina Rother[1,2], Ricardo Ribeiro Rodrigues[2]

[1] Universidade Estadual de Campinas (UNICAMP), Programa de Pós-Graduação em Biologia Vegetal, Instituto de Biologia, Campinas - SP, Brazil.

[2] Universidade de São Paulo (USP), Escola Superior de Agricultura “Luiz de Queiroz”, Departamento de Ciências Biológicas, Piracicaba - SP, Brazil.

[3] Universidade Estadual Paulista (UNESP), Departamento de Ecologia, Rio Claro - SP, Brasil

[4] Parque Ecológico de Paulínia “Armando Müller”, Paulínia – SP, Brasil

[5] Instituto Florestal, Seção de Ecologia Florestal, São Paulo - SP, Brasil

ABSTRACT

Strictly Protected Areas (SPAs) have limited non-random distribution that fail to capture the conservation needs in the tropics, especially in agricultural landscapes. In the Brazilian Atlantic Forest, over 90% of forest fragments occur on private lands (PLs), representing a substantial complement to support biodiversity persistence over modified landscapes. In this study, we evaluated the relative contribution of forest fragments in private lands to Strictly Protected Areas in São Paulo state, Brazil. To do so, we used occurrence data from i) floristic surveys performed on PLs (N=367) and ii) available checklists for SPAs (N=20), considering tree and shrub species composition of forest fragments distributed across three regions (West, Center and Southeast). We analyzed species' occurrence between SPAs and PLs and their species composition variation (β diversity) and components (turnover and nestedness). From 1,558 tree or shrub species registered in this study, we found that PLs encompasses 59%, including 48% of rare and 41% of threatened species. We found that SPAs sites have higher local diversity than PLs, up to four orders of magnitude. Despite these marked differences among forest fragments, we registered high values of β diversity, particularly due to its turnover component, indicating species replacement among forest fragments. Overall, β diversity was similar between protection categories as well as among different regions, suggesting the impacts of the fragmentation process on regional diversity are similar across all forest fragments – protected or not. The major contribution from the turnover component (>94%) highlight the collective value of forest fragments, reinforcing conservation efforts must target multiple sites and habitat types to capture the variation along space and regional diversity.

INTRODUCTION

Degradation of natural ecosystems is the main driver of the current biodiversity loss (Haddad et al. 2015; McGill 2015), describing a scenario that is particularly threatening for the tropical region, which support over half of the world's terrestrial biodiversity (Malhi et al. 2014). Emerging in early 70's, the creation of Protected Areas represent the main conservation strategy to preserve natural ecosystems and protect biodiversity (Rodrigues et al. 2004; Jenkins & Joppa 2009; Laurance et al. 2012). Despite the worldwide substantial increase in the percentage of land protection over the last three decades (Jenkins & Joppa 2009; Oliveira et al. 2017), the limited number, non-random distribution and representativeness of PAs most likely fail to capture the conservation needs in the tropics (Margules & Pressey 2000; Putz et al. 2001; Rodrigues et al. 2004; Andam et al. 2008).

The designation of PAs has evoked discussion among conservation scientists: the debate as whether single large reserves are better than several small ones (SLOSS) has never came to a resolution, since it depends on a great variety of local factors and targeted species (Ovaskainen 2002; Tjørve 2010). More recently, the recognition that agriculture expansion is a substantial menace to natural habitats (Dobrovolski, Loyola, et al. 2011; Laurance et al. 2014; Mendenhall et al. 2016) has led to the debate as whether land-sparing (*i.e.* spatial delimitation of intensive agriculture and conservation) or land-sharing (*i.e.* biodiversity-friendly agricultural practices) promote better outcomes for local and regional biodiversity (Phalan et al. 2011; Laurance et al. 2014; Kremen 2015). Both debates have evidence suggesting that a variety of approaches combined may be the best option, whenever conservation of pristine ecosystems is no longer an option. Single large and several small PAs are important in different ways and perspectives, considering several taxonomic groups and landscapes (Ovaskainen 2002; Tjørve 2010; Thomas et al. 2012; Fahrig 2013; Bartonova et al. 2016), as well as land-sharing and land-sparing (Phalan et al. 2011; Laurance et al. 2014; Kremen 2015).

Regardless of the relative contributions of these contrasting strategies, protected areas are cannot be isolated and therefore depend, to some extent, on their surrounding non-protected matrix, especially on habitat patches within private lands (Chazdon, Harvey, et al. 2009; Prevedello & Vieira 2010; Laurance et al. 2012;

Bartonova et al. 2016). On broad intensive or extensive agricultural landscapes, where matrix quality may pose a barrier for some taxonomic groups, the reduced habitat patches most likely play an essential role supporting PAs (Bergamin et al. 2017; Farah et al. 2017). Composed by a variety of different sized secondary forests or disturbed and degraded old-forest remnants (Malhi et al. 2014), these forest fragments recently have been recognized as valuable for conservation purposes, harboring an impoverished but significant fraction of biodiversity (Morante-Filho, Arroyo-Rodríguez, et al. 2015; Sfair et al. 2016; Beca et al. 2017; Farah et al. 2017; Emer et al. 2018). Several studies discuss the value of secondary forests (Arroyo-Rodríguez et al. 2008; Chazdon et al. 2009) and the importance of a distributed network of forest fragments to support biodiversity persistence over modified landscapes (Silva & Tabarelli 2000; Arroyo-Rodríguez et al. 2008; Solar et al. 2015; Sfair et al. 2016; Beca et al. 2017; Bergamin et al. 2017).

The Brazilian Atlantic Forest is a largely deforested landscape with over three centuries of land-use conversion and human occupancy (Metzger 2009; Ribeiro et al. 2009; Haddad et al. 2015), where persisting agriculture expansion threatens protected and unprotected natural vegetation (Sparovek et al. 2010; Dobrovolski et al. 2011). With less than 10% of total remaining forests classified as Strictly Protected Areas (hereafter SPAs) (*i.e.*, Integral Protection Protected Areas by the Brazilian Ministry of Environment - MMA, 2007) (Ribeiro et al. 2009; Dobrovolski, Loyola, et al. 2011), it is reasonable to evaluate the distribution of diversity and composition variability (β diversity) among forest fragments in regional spatial scales, including the massive representation (>90%) of those in private lands. Even though measures of β diversity can be hard to translate, it might give us relevant information on the mechanisms of regional diversity maintenance (Socolar et al. 2016), underpinning and contributing to conservation strategies within hyper-fragmented landscapes.

In this study, Atlantic forest will be used for a particular study model using an extensive data set of tree and shrub species of 367 private lands and 20 SPAs. Our main objective is to evaluate the relative contribution of forest fragments in private lands to Strictly Protected Areas in São Paulo state, Brazil, with particular interest on species' exclusive occurrence between the distinct protection categories of forest fragments and within distinct regions of the state (Southeast, Center, West). We

specifically investigated the degree of change in species composition among communities (β diversity) and its components turnover and nestedness, which reflect the differences resulting from species replacement among sites (turnover) and differences among sites when the poorest sites represent sub sets of the richest sites and the regional species pool (nestedness). In these hyper-fragmented landscapes we expect to find (1) low mean values of α-diversity for forest fragments on private lands in comparison to SPAs and (2) high values of β-diversity in a broader scale, on account of its turnover component, supporting the collective value of forest fragments for conservation and highlighting the importance of multiple sites to encompass regional diversity.

METHODS

Study Region

The state of São Paulo is located in the Southeastern of Brazil and encompasses two worldwide *hotspots* – Cerrado (Brazilian savannas) and Atlantic Forest (Myers et al. 2000). In the interior plateaus of the state, both Atlantic Forest and Cerrado comprises several vegetation types, as a reflection of variable topography, geological history, soil types and environmental gradients, resulting in a complex and heterogeneous transition and replacement of species (Morellato & Haddad 2000; Oliveira-Filho & Fontes 2000). In this region, the main Atlantic Forest vegetation types are: i) the predominant Seasonal Semideciduous Forests (SSF), characterized by partial deciduousness during dry season (usually from April to September) (Morellato & Haddad 2000; Oliveira-Filho & Fontes 2000); ii) Seasonal Deciduous Forests (SDF), characterized by total deciduousness during dry season; iii) Alluvial and swamp forests, located along riversides, with floristic composition influenced by eventual or permanent flooding (Kurtz et al. 2015); iv) Atlantic Forest *sensu stricto* (AFSS), comprising the coastal rain forests (Morellato & Haddad 2000; Oliveira-Filho & Fontes 2000) that occasionally advance to the interior plateaus. Cerrado vegetation types are typically open non-forest ecosystems that comprises several structural forms (see details in Durigan & Ratter 2006), but in its broad definition, Cerrado *sensu lato* also includes forests occurring in more fertile soils, where trees may form a closed canopy

(often 8-12m) that shade and reduce the ground vegetation. This forest is known as *Cerradão* and it can be considered a forest-savanna transition that share large extents with Seasonal Semideciduous Forests (SSF) (Oliveira-Filho & Fontes 2000; Durigan & Ratter 2006).

The great variety of vegetation types originally present in the interior plateaus of the state were heavily deforested during the long history of land conversion for agricultural purposes (Metzger 2009; Ribeiro et al. 2009; Joly et al. 2014). Indiscriminate deforestation resulted on hyper-fragmented landscapes especially in the uttermost countryside of the state, where sugarcane, cattle pastures, and *Eucalyptus ssp.* plantations currently dominate the agricultural matrix (Metzger 2009). Strictly Protected Areas safeguard only 4% of the Atlantic forest original area and 0,3 to 0,5% of Cerrado's original area (Durigan et al. 2006).

For the purpose of this study, we considered regions in the countryside of the state – West, Center, Southeast (**Figure 1**) - where mean forest cover is below 30% and agriculture prevails in the landscape. Because the study region encompasses several vegetation types distributed along a continuum of environmental gradients, where species replacement may occur gradually and it is difficult to distinguish them apart (Morellato & Haddad 2000; Oliveira-Filho & Fontes 2000), we selected forest fragments classified as Seasonal Semideciduous Forests (SSF) or their ecotonal sites, excluding Atlantic Forest s.s and Cerrado.

Occurrence data collection – woody plant assemblages/ Vegetation data sets

We compiled data from floristic surveys performed on private lands by the Forest Ecology and Restoration Laboratory (University of São Paulo) (N=367) (see details in Rodrigues et al. 2011) and available checklists for Strictly Protected Areas (N=20) (**Figure 1**). To overcome different floristic survey methods we performed all analysis with occurrence data (presence-absence) of tree and shrub species. Names and synonyms were standardized through the Plantminer web tool (www.plantminer.com) (Carvalho et al. 2010) based on Flora do Brasil (www.floradobrasil.jbrj.gov.br) and The Plant List (www.theplantlist.org/) (Flora_do_Brasil_2020). Complementary queries were performed on The Missouri

Botanical Gardens (www.tropicos.org) and NeoTropTree (Oliveira-Filho 2017). According to these databases, we excluded any exotic and unidentified species from final compilation. We considered as rare any species occurring in less than 5% of the sites and threatened species were classified as vulnerable, endangered, or critically endangered as determined by the CNC Flora (http://cncflora.jbrj.gov.br) according to the IUCN Convention, and to the State of São Paulo list of endangered species (SMA_57/ 2016).

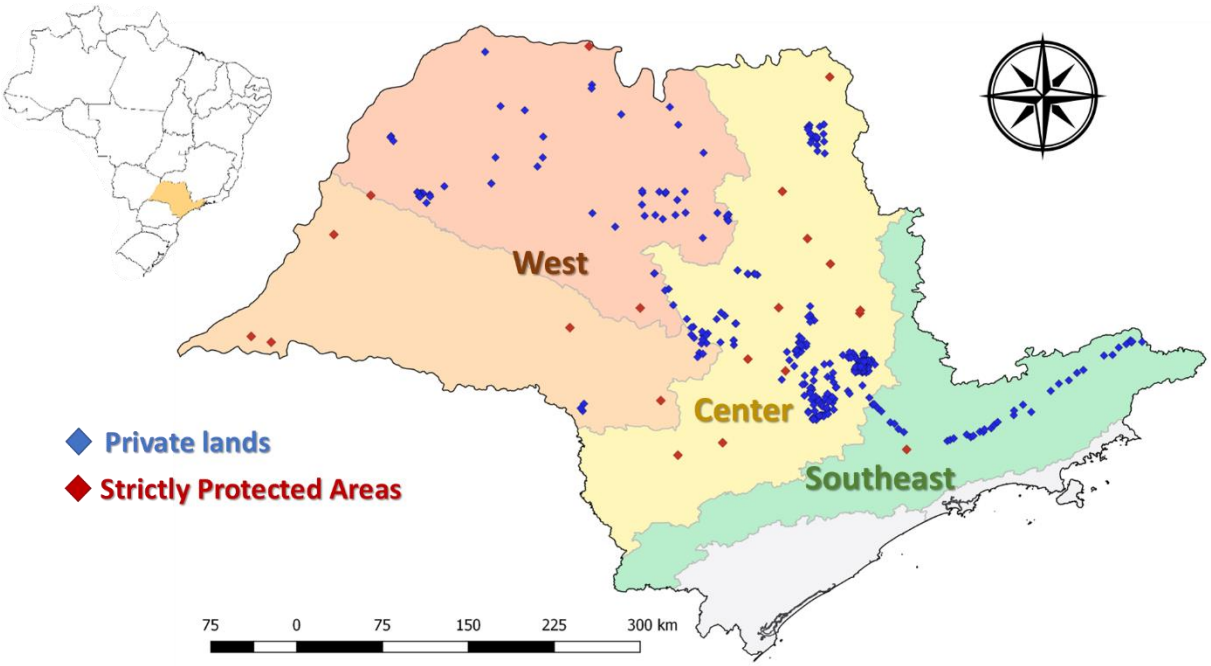


Figure 01: Distribution of forest fragments among regions in São Paulo state (Southeastern Brazil), considering distinct categories: private lands (n=367) and strictly protected areas (n=20). Mean forest cover based on São Paulo State Forest Inventory (2011): West = 6.5%, Center= 10.9%, Southeast= 27.7%.

Data analysis

We assessed overall differences on sampling efforts by calculating the proportion of observed species from the total estimated richness, based on the average of three frequency-based non-parametric estimators: Chao 2, Jacknife 1 and Jacknife 2 (Magurran 2013). We performed all estimations on EstimateS (Colwell 2013).

In order to estimate the turnover among communities, *i.e.,* the change on their species compositions (Anderson et al. 2010), we considered measures based on resemblances among sample units. We calculated the multi-site Sorensen (β_{SOR}) and Simpson indices (β_{SIM}) to evaluate the contributions of β diversity components (*i.e.,*

nestedness and turnover) (Baselga 2010; Baselga et al. 2015). β_{SOR} includes variation from both replacement and nestedness while β_{SIM} only measures turnover (Socolar et al. 2016). Therefore, the nestedness ($\boldsymbol{\beta_{NEST}}$) component of β diversity is the difference of β_{SOR}- β_{SIM}. Particularly, β_{SIM} is a choice of dissimilarity metric that is widely recommended for presence-absence data since it is nearly as insensitive to sample size as the best abundance-based measures (Koleff et al. 2003) and is appropriate to identify spatial and environmental gradients where rare-species occurs, even when sampling is sparse or uneven (Socolar et al., 2016). It is also a measure that focus on compositional differences more than differences in species richness (Koleff et al. 2003), alleviating distinct sampling efforts. Additionally, to evaluate and compare the beta diversity structure within groups – the distinct regions (West, Center, Southeast) or protection categories (private lands or strictly protected areas) - we measured their average dissimilarity in relation to the group centroid in a multivariate space ($\boldsymbol{\beta_{DISPER}}$). This allowed us to properly test for the homogeneity of multivariate dispersions, *i.e.*, to test the null hypothesis of no difference in β diversity among groups (Anderson et al. 2006, 2010). We applied this analysis computing an incidence-based Jaccard dissimilarity index.

We calculated all β diversity indices for each region and for protection categories separately, standardizing the number of sites based on the minimum amount of samples, that is, 20 samples for each protection category and 38 samples for each region. We repeated this random sampling procedure 1,000 times, extracting the mean and standard deviation for each β diversity (*i.e.,* $\beta_{SOR,}$ β_{SIM}, β_{NEST} and β_{DISPER}). All analysis were perfomed on R (R Development Core Team 2007) function “beta-multi.R” in package “betapart” (Baselga & Orme 2012), function “betadisper” and “permutest” in package VEGAN (Oksanen 2015).

RESULTS

Species distribution/occurence among private lands and protected areas

We registered 1,558 tree or shrub species in 387 sites, representing 452 genera and 98 botanical families. Considering the mean estimated richness (Chao 2, Jacknife 1, and Jacknife 2), sampling effort captured on average 72% of species (range 47 – 84%) (**Table 1**).

Table 1: Observed (S_obs) and mean estimated richness (S_est) - based on Chao2, Jackknife 1 and Jackknife 2 estimators - for distinct regions and protection categories (PL=private lands and SPA=strictly protected areas), with an overall percentage of observed/estimated richness (%_obs_est). N indicates sample size.

Region	Categories	N	S_obs	Chao 2	Jack 1	Jack 2	S_est (mean ± SD)	%_obs_est
Southeast	PL	37	539	690	688	765	714 ± 33.63	75
	SPA	1	253	-	-	-	-	-
Center	PL	265	639	728	759	800	762 ± 25	84
	SPA	11	902	1191	1198	1342	1244 ± 65	72
West	PL	65	476	556	582	620	586 ± 23	81
	SPA	8	895	2465	1430	1821	1905 ± 374	47

Total species richness occurring in SPAs or PLs vary amongst different regions of the state (bar graph, **Figure 2**), but in general, SPAs sum the largest portion of regional species, except by the Southeastern region where we only sampled one SPA. Overall, most of the species (1,360 species or 87%) occur in SPAs, while PLs encompasses a total of 913 species (59%) with 198 exclusive species (13%) (Venn diagram, **Figure 2**). Average species richness per site was much higher for SPAs (260+-110 species per site) than for PLs (56+-18 species per site), with an overall ratio 5 times higher for SPAs than PLs (Boxplots, **Figure 2**) (**APPENDICES 1 and 2**).

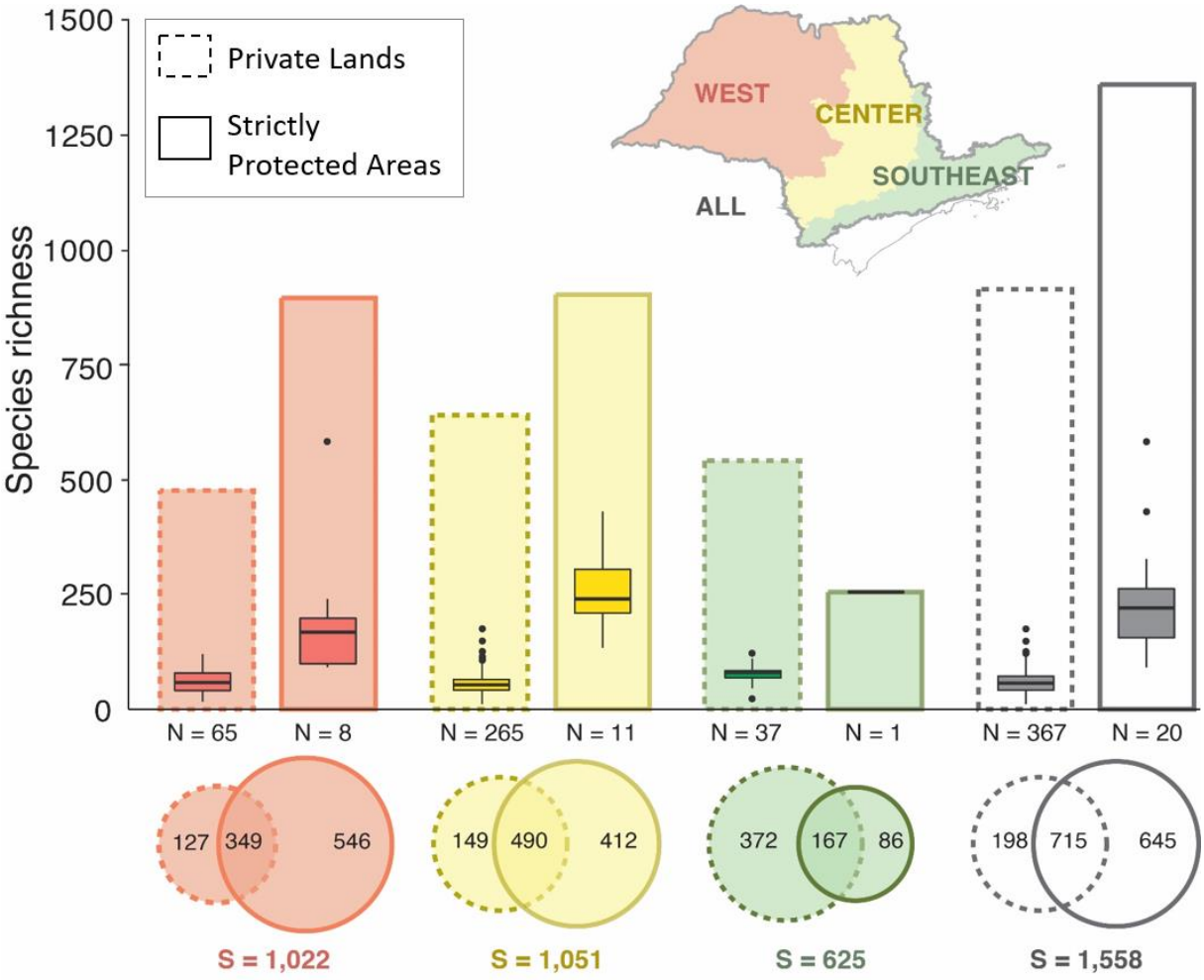


Figure 2: Distribution of species richness between protection categories and among regions of São Paulo state (West = orange, Center = yellow, Southeast = green, All = gray/white). Bar graph indicates the total accumulated species richness and Venn diagrams indicate shared and exclusive species; Private Lands are indicated by dashed-line bars/circles and Strictly Protected Areas by solid-line bars/circles. Boxplots are based on the average species richness per site.

Species vary greatly regarding relative frequencies among sampled sites: over three quarters (1,252 species, 80.3%) are rare (i.e. registered in less than 5% of sites). From the total rare species, 15% (190 species) occured exclusively in PLs , 33% (417 species) occured both in PLs and SPAs and 51% (645 species) occured exclusively in SPAs (**Figure 3**). From the 69 threatened species – according to CNC Flora and the IUCN (Flora do Brasil 2020) or by the Sao Paulo state's Red List (SMA 57/2016) - we registered 59% (41 species) exclusively on SPAs, 16% (11 species) exclusively in private lands and 25% (17 species) on both (**Figure 3**).

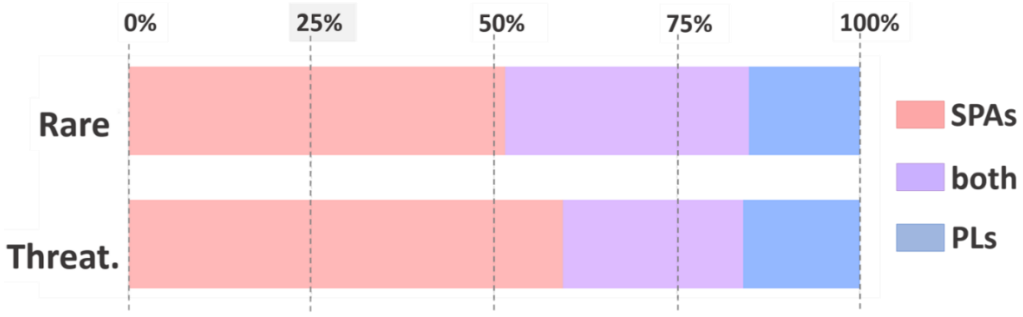


Figure 3: Proportion of rare and threatened species occurring in Strictly Protected Areas, Private Lands or both. Rare species are those occurring in less than 5% of all sites. Threatened species are those classified as extinct, extinct in the wild, critically endangered, endangered or vulnerable by the CNC Flora and IUCN, and São Paulo state's red list.

Variation in species composition: Beta diversity

Beta diversity of tree and shrub species revealed that the turnover (β_{SIM}) is the major component of overall beta diversity (β_{SOR}), regardless of which subset is evaluated: it represents over 96% among regions and over 94% between PAs and FFs (**Table 2**). The variation in species composition ($\boldsymbol{\beta_{DISPER}}$) was similar within regions or types, ranging from 0.55 to 0.60.

Table 2: Mean total β diversity and its components turnover and nestedness for distinct regions and categories. * Fixed number of samples (N), randomly selected 10000 times.

Region	**N**	$\boldsymbol{\beta_{SOR}}$	$\boldsymbol{\beta_{NES}}$	$\boldsymbol{\beta_{SIM}}$	$\boldsymbol{\beta_{disper}}$
Southeast	38	0.9385 (100%)	0.0187 (3%)	0.9198 (98%)	0.5963
Center	38*	0.9427 (100%)	0.0344 (4%)	0.9082 (96%)	0.5890
West	38*	0.9425 (100%)	0.0353 (4%)	0.9072 (96%)	0.6056
Category	**N**	$\boldsymbol{\beta_{SOR}}$	$\boldsymbol{\beta_{NES}}$	$\boldsymbol{\beta_{SIM}}$	$\boldsymbol{\beta_{disper}}$
Private Lands	20*	0.9169 (100%)	0.0318 (3%)	0.8851 (97%)	0.5964
Strictly Protected Areas	20	0.9011 (100%)	0.0568 (6%)	0.8443 (94%)	0.5526
TOTAL	387	0.9937 (100%)	0.0042 (0.5%)	0.9895 (99.5%)	0.6134

DISCUSSION

Our dataset represented a high percentage of overall estimated richness, revealing that forest fragments in private lands harbor a significant portion of regional species richness (41 to 88%), including threatened and rare species. We registered

high values of β diversity, particularly due to its turnover component, indicating species replacement among forest fragments (Baselga 2010; Soininen et al. 2018). Variation in species composition was similar among different regions and between strictly protected areas and private lands, suggesting that the impacts of the fragmentation process on regional diversity distribution may be similar across all forest fragments – protected or not. These results altogether highlight the importance of forest fragments in private lands, supporting biodiversity maintenance at the regional scale.

The proportion of observed/estimated richness was high for both SPAs and PLs, but the striking differences on local diversity may be related to sampling procedures. SPAs in the Atlantic Forest biome are among the most densely sampled in Brazil (Oliveira et al. 2017) and their available official species lists often represent a compilation of several assessments, ratifying our assumption on their higher sampling effort. On the other hand, the forest fragments within PLs evaluated in this study had a much greater sampling size with less sampling effort, as the species lists were taken to fulfill quick assessments of the regional vegetation (see details in Rodrigues et al. 2011). Therefore, we must acknowledge, to some degree, that the higher local diversity registered in the studied SPAs reflect these sampling procedures. Indeed, beyond these differences, it is reasonable to expect higher local diversity in SPAs when considering human-modified landscapes, as they reduce tropical forest deforestation both inside (Andam et al. 2008) and on their surroundings (Joppa et al. 2008), and most likely prevent these areas from other human impacts such as fire, hunting, selective logging etc. (Gray et al. 2016). Recent studies confirm higher species richness within SPAs for several taxa, ecosystems, and regions (Coetzee et al. 2014; Gray et al. 2016), but some contrasting outcomes indicate non-significant differences regarding plants, the South America continent (Coetzee et al. 2014) and rarefied richness (i.e., the number of species for each site adjusted by the number of individuals) (Gray et al. 2016). These inconsistencies among studies relates to the fact that these comparisons are context-specific, depending on a complex of factors that include surrounding forest cover and land use (Joppa et al. 2008; Coetzee et al. 2014; McGill 2015; Gray et al. 2016). In human-modified landscapes, there is clear evidence on the structural changes and impoverishment of tropical forests (Farah et al. 2014; Chazdon et al. 2016; Rocha-Santos et al. 2016), where more sensitive or more ecologically specialized plant guilds (e.g. large-seeded old growth trees) are replaced

by a sub-set of generalists such as pioneer or early secondary species, with higher dispersal abilities (Silva & Tabarelli 2000; Gibson et al. 2011; Laurance et al. 2012; Solar et al. 2015; Emer et al. 2018).

While PLs have lower species richness than SPAs at the site level, compositional variation is consistently high for both SPAs and PLs as well as for different regions of the state. Confirming the expected high levels of β-diversity and the major contribution from the turnover component, we corroborated several studies for distinct taxonomic groups within the Atlantic Forest Domain (Silva et al. 2014; Machado et al. 2016; Beca et al. 2017; Bergamin et al. 2017; Farah et al. 2017) and other tropical ecosystems (Solar et al. 2015; Collins et al. 2017; Soininen et al. 2018). High β-diversity and turnover is attributable to species replacement (Baselga 2010) and indicate forest fragments in our study region are very heterogeneous, which is in accordance to the well-known trend of rare species' prevalence in tropical forests (Caiafa & Martins 2010; Hubbell 2013; Slik et al. 2015). However, these results may be confounded by the previously discussed overall low levels of α diversity, which may create a sampling effect that result in inflated β diversity (Karp et al. 2012; Newbold et al. 2015). Despite this caveat, our results contributes to relevant insights regarding conservation strategies in hyper-fragmented agricultural landscapes: to effectively maintain the regional diversity, conservation efforts should embrace SPAs and PLs, targeting multiple sites (Silva & Tabarelli 2000; Solar et al. 2015; Sfair et al. 2016; Socolar et al. 2016; Bergamin et al. 2017).

Beyond the overall effectiveness of SPAs regarding local diversity (Rodrigues et al. 2004; Coetzee et al. 2014; Gray et al. 2016), they also have positive effects protecting species' range (Rodrigues et al. 2004; Thomas et al. 2012), preserving forest cover and reducing deforestation (Andam et al. 2008; Joppa et al. 2008; Carranza et al. 2014). However, under several aspects the SPAs network have limited performance, reflecting their spatially biased distribution and representation (Rodrigues et al. 2004; Andam et al. 2008; Jenkins & Joppa 2009; Bergamin et al. 2017; Oliveira et al. 2017), with further adverse consequences under climate-change scenarios (Lemes et al. 2014). In addition to the gaps on taxonomic diversity (Oliveira et al. 2017), functional and phylogenetic diversity may also be underrepresented (Bartonova et al. 2016; Saraiva et al. 2018). As an aggravating factor of all of the

above, SPAs are often poorly effective when embedded in regions with economic value or potential (Joppa et al. 2008; Dobrovolski et al. 2011) – such as the agricultural landscapes of this study. In tropical agricultural landscapes, the strongly reduced site-level richness is well documented (Gibson et al. 2011; Canale et al. 2012; McGill 2015; Gray et al. 2016; Mendenhall et al. 2016; Sfair et al. 2016; Alroy 2017; Beca et al. 2017; Farah et al. 2017; Saraiva et al. 2018; Solar et al. 2015), but our results on the significant portion of regional species harbored in private lands' forest fragments - almost 60% - is thriving. In a partial extent of our study area, Farah *et al.* (2017) concluded that the potential additional richness of forest fragments in relation to SPAs is considerable (up to 90%), even when located in low forest cover landscapes (<30%). Although we did not evaluate size-related issues in our study, a relevant remark is the lack of clear dependency between species richness and patch size when considering a "local landscape", that is, the area within an appropriate distance of the sample site - as presented by Fahrig (2013). That is because usually, smaller habitat patches are spread over larger extents than one big patch, therefore reflecting more heterogeneous habitats (Fahrig 2013); that would be a natural expectation in our study area, considering the great variety of vegetation types occurring in the Atlantic Forest domain/biome (Morellato & Haddad 2000; Oliveira-Filho & Fontes 2000; Bergamin et al. 2017). Moreover, these forest fragments – even the small ones (<50ha) - are extremely valuable to reduce isolation and promote connectivity in the hyper-fragmented landscapes of the Atlantic Forest (Santos et al. 2007, Ribeiro et al. 2009; Farah et al. 2017), where species persistence depend on the maintenance of plant and animal diversity and interactions (Howe 2014; Emer et al. 2018).

Our findings indicate that forest fragments in private lands are crucial to support strictly protected areas in agricultural landscapes, corroborating other studies under similar scenarios (Carneiro et al. 2016; Beca et al. 2017; Farah et al. 2017; Emer et al. 2018). Despite being mostly composed (83.4%) by fragments with less than 50ha (Ribeiro et al. 2009) and with presumed impoverished site-level richness, the importance of these unprotected forests is pivotal: taken together they represent 93.2% of remaining Interior Atlantic Forests (Ribeiro et al. 2009) and, as shown in this study, harbor at least 59% of regional diversity, including rare and threatened species. Recognizing that pristine forests are irreplaceable for conserving biodiversity (Gibson et al. 2011; Alroy 2017) but also that agricultural expansion in the tropics is somewhat

inevitable (Dobrovolski, Loyola, et al. 2011; Laurance et al. 2014; Malhi et al. 2014), we advocate that both spared (*e.g.* protected areas) and shared forest fragments within biodiversity-friendly land uses (*e.g*. private lands) could be combined to enhance conservation outcomes (Tjørve 2010; Melo, Arroyo-Rodríguez, et al. 2013). That is particularly true in agricultural landscapes with long period of intensive use, where original vegetation have nearly vanished and remnants are often modified to varying degrees, resulting in stagnant or arrested succession (Laurance et al. 2014; Malhi et al. 2014). While it is unlikely to expand the Protected Area's network in our study region (Dobrovolski et al. 2011; Oliveira et al. 2017), we highlight the need for strengthening protected areas policies (Bernard et al. 2014; De Marques & Peres 2015) and improving protection of vegetation within private lands.

In this sense, the Brazilian government must guarantee enforcement of the current legislation – the Native Vegetation Protection Law (NVPL) (see details in Brancalion et al. 2016) – in order to fulfill the conservation potential of these forest fragments, which have never been fully exploited nor protected. Our results provide relevant evidence on the current value of forests in private lands for conservation and on the need for active restoration under local and regional-scale approaches, in order to enhance connectivity, forest cover, foster habitat heterogeneity, and enable species mobility (Brancalion et al. 2013; Melo, Arroyo-Rodríguez, et al. 2013; Howe 2014; Beca et al. 2017; Emer et al. 2018; Rother et al. 2018). To sustain a more biodiversity friendly agricultural landscape (Arroyo-Rodríguez et al. 2008; Phalan et al. 2011; Laurance et al. 2014) it is necessary to broaden the dominant idea that only primary forests or larger fragments are able to promote biodiversity conservation, and embrace the full range of size and conservation status of unprotected forest fragments in private lands.

ACKNOWLEDGEMENTS: This study was financed in part by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior – Brasil (CAPES) – Finance Code 001, by the National Council for Scientific and Technological Development (CNPq grant 870360/1997-3) and by The São Paulo Research Foundation (FAPESP grant 2013/50718-5).

REFERENCES

Alroy J (2017) Effects of habitat disturbance on tropical forest biodiversity. Proceedings of the National Academy of Sciences 114:6056–6061

Andam KS et al. (2008) Measuring the effectiveness of protected area networks in reducing deforestation. Proceedings of the National Academy of Sciences of the United States of America 105:16089–16094

Anderson MJ et al. (2010) Navigating the multiple meanings of β diversity: A roadmap for the practicing ecologist. Ecology Letters 14:19–28

Anderson MJ, Ellingsen KE, McArdle BH (2006) Multivariate dispersion as a measure of beta diversity. Ecology Letters 9:683–693

Arroyo-Rodríguez V et al. (2008) Value of small patches in the conservation of plant-species diversity in highly fragmented rainforest. Conservation Biology 23:729–39

Bartonova A et al. (2016) How universal are reserve design rules? A test using butterflies and their life history traits. Ecography 39:456–464

Baselga A (2010) Partitioning the turnover and nestedness components of beta diversity. Global Ecology and Biogeography 19:134–143

Baselga A, Orme CDL (2012) Betapart: An R package for the study of beta diversity. Methods in Ecology and Evolution 3:808–812

Baselga A et al. (2015) betapart: Partitioning beta diversity into turnover and nestedness components. In: R Package Version 1.3 1–26

Beca G et al. (2017) High mammal species turnover in forest patches immersed in biofuel plantations. Biological Conservation

Bergamin RS et al. (2017) Linking beta diversity patterns to protected areas: lessons from the Brazilian Atlantic Rainforest. Biodiversity and Conservation

Bernard E, Penna LAO, Araújo E (2014) Downgrading, douwnsizing, degazettement, and reclassification of Protected Areas in Brazil. Conservation Biology 28:939–950

Brancalion PHS et al. (2016) A critical analysis of the Native Vegetation Protection Law of Brazil (2012): Updates and ongoing initiatives. Natureza e Conservacao 14:1–15

Brancalion PHS et al. (2013) Biodiversity persistence in highly human- modified tropical landscapes depends on ecological restoration. Tropical Conservation Science 6:705–710

Caiafa AN, Martins FR (2010) Forms of rarity of tree species in the southern Brazilian Atlantic rainforest. Biodiversity and Conservation 19:2597–2618

Canale GR et al. (2012) Pervasive defaunation of forest remnants in a tropical biodiversity hotspot. PLoS ONE 7

Carneiro MS et al. (2016) Spatial species turnover maintains high diversities in a tree assemblage of a fragmented tropical landscape. Ecosphere 7:1–12

Carranza T et al. (2014) Protected area effectiveness in reducing conversion in a rapidly vanishing ecosystem: The Brazilian Cerrado. Conservation Letters 7:216–223

Carvalho GH, Cianciaruso MV, Batalha MA (2010) Plantminer: A web tool for checking and gathering plant species taxonomic information. Environmental Modelling and Software 25:815–816

Chazdon RL, Harvey CA, et al. (2009) Beyond Reserves : A Research Agenda for Conserving Biodiversity in Human-modified Tropical Landscapes. Biotropica 41:142–153

Chazdon RL, Peres C a, et al. (2009) The potential for species conservation in tropical secondary forests. Conservation biology 23:1406–17

Chazdon RL et al. (2016) When is a forest a forest ? Forest concepts and definitions in the era of forest and landscape restoration. 1–36

Coetzee BWT, Gaston KJ, Chown SL (2014) Local scale comparisons of biodiversity as a test for global protected area ecological performance: A meta-analysis. PLoS ONE 9

Collins CD et al. (2017) Fragmentation affects plant community composition over time. Ecography 40:119–130

Colwell R (2013) EstimateS: Statistical estimation of species richness and shared species from samples. Version 9. User's Guide and application.

Dobrovolski R et al. (2011) Agricultural expansion can menace Brazilian protected areas during the 21 st century. Natureza e Conservacao 9:208–213

Durigan G, Ratter JA (2006) Successional changes in cerrado and cerrado/forest ecotonal vegetation in western São Paulo State, Brazil, 1962-2000. Edinburgh Journal of Botany 63:119–130

Durigan G et al. (2006) Seleção De Fragmentos Prioritários Para a Criação De Unidades De Conservação Do Cerrado No Estado De São Paulo. Revista do Instituto Florestal 18:23–37

Emer C et al. (2018) Seed-dispersal interactions in fragmented landscapes – a metanetwork approach. Ecology Letters 21:484–493

Fahrig L (2013) Rethinking patch size and isolation effects: The habitat amount hypothesis. Journal of Biogeography 40:1649–1663

Farah FT et al. (2014) Forest destructuring as revealed by the temporal dynamics of fundamental species - Case study of Santa Genebra Forest in Brazil. Ecological Indicators 37:40–44

Farah FT et al. (2017) Integrating plant richness in forest patches can rescue overall biodiversity in human-modified landscapes. Forest Ecology and Management 397:78–88

Flora_do_Brasil_2020 Jardim Botânico do Rio de Janeiro. Available on http://floradobrasil.jbrj.gov.br/.

Gibson L et al. (2011) Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478:378–81

Gray CL et al. (2016) Local biodiversity is higher inside than outside terrestrial protected areas worldwide. Nature Communications 7

Haddad NM et al. (2015) Habitat fragmentation and its lasting impact on Earth ’ s ecosystems. Science 1–9

Howe HF (2014) Diversity Storage: Implications for tropical conservation and restoration. Global Ecology and Conservation 2:349–358

Hubbell SP (2013) Tropical rain forest conservation and the twin challenges of diversity and rarity. Ecology and Evolution 3:3263–3274

Jenkins CN, Joppa L (2009) Expansion of the global terrestrial protected area system. Biological Conservation 142:2166–2174

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

Joppa LN, Loarie SR, Pimm SL (2008) On the protection of ‘protected areas’. Proceedings of the National Academy of Sciences 105:6673–6678

Karp DS et al. (2012) Intensive agriculture erodes B-diversity at large scales. Ecology Letters 15:963–970

Koleff P, Gaston KJ, Lennon JJ (2003) Measuring beta diversity for presence – absence data. Journal of Animal Ecology 72:367–382

Kremen C (2015) Reframing the land-sparing/land-sharing debate for biodiversity conservation. Annals of the New York Academy of Sciences 1355:52–76

Kurtz BC, Valentin JL, Scarano FR (2015) Are the Neotropical Swamp Forests a Distinguishable Forest Type? Patterns From Southeast and Southern Brazil. Edinburgh Journal of Botany 72:191–208

Laurance WF et al. (2012) Averting biodiversity collapse in tropical forest protected areas. Nature 489:290–4

Laurance WF, Sayer J, Cassman KG (2014) Agricultural expansion and its impacts on tropical nature. Trends in Ecology and Evolution 29:107–116

Lemes P, Melo AS, Loyola RD (2014) Climate change threatens protected areas of the Atlantic Forest. Biodiversity and Conservation 23:357–368

Machado FS et al. (2016) Tree diversity of small forest fragments in ecotonal regions: why must these fragments be preserved? Biodiversity and Conservation 525–537

Magurran AE (2013) Measuring biological diversity.

Malhi Y et al. (2014) Tropical Forests in the Anthropocene. Annual Review of Environment and Resources 39:125–159

Margules CR, Pressey RL (2000) Systematic conservation planning. Nature 405:243–53

De Marques AAB, Peres CA (2015) Pervasive legal threats to protected areas in Brazil. Oryx 49:25–29

McGill B (2015) Land use matters. Nature 520:38–39

Melo FPL et al. (2013) On the hope for biodiversity-friendly tropical landscapes. Trends in ecology & evolution 28:462–8

Mendenhall CD et al. (2016) Quantifying and sustaining biodiversity in tropical agricultural landscapes. Proceedings of the National Academy of Sciences 113:14544–14551

Metzger JP (2009) Conservation issues in the Brazilian Atlantic forest. Biological Conservation 142:1138–1140

Morante-Filho JC, Arroyo-Rodríguez V, Faria D (2015) Patterns and predictors of beta diversity in the fragmented Brazilian Atlantic forest: A multiscale analysis of forest specialist and generalist birds. Journal of Animal Ecology 85:240–250

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

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

Newbold T et al. (2015) Global effects of land use on local terrestrial biodiversity. Nature

Oksanen J (2015) Multivariate analysis of ecological communities in R: vegan tutorial. R documentation 43

Oliveira-Filho AT (2017) NeoTropTree - Tree flora of the Neotropical Region: A database involving biogeography, diversity and conservation. Universidade Federal de Minas Gerais http://www.neotroptree.info

Oliveira-Filho A, Fontes M (2000) Patterns of Floristic Differentiation among Atlantic Forests in Southeastern Brazil and the Influence of Climate1. Biotropica 32:793–810

Oliveira U et al. (2017) Biodiversity conservation gaps in the Brazilian protected areas. Scientific Reports 7:1–9

Ovaskainen O (2002) Long-term persistence of species and the SLOSS problem. Journal of Theoretical Biology 218:419–433

Phalan B et al. (2011) Reconciling Food Production and Biodiversity Conservation: Land sharing and Land sparing compared. Science 333:1289–1291

Prevedello JA, Vieira M V. (2010) Does the type of matrix matter? A quantitative review of the evidence. Biodiversity and Conservation 19:1205–1223

Putz FE et al. (2001) Tropical Forest Management and Overview Conservation of Biodiversity : Conservation biology 15:7–20

Ribeiro MC et al. (2009) The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biological Conservation 142:1141–1153

Rocha-Santos L et al. (2016) The shrinkage of a forest: Landscape-scale deforestation leading to overall changes in local forest structure. Biological Conservation 196:1–9

Rodrigues ASL et al. (2004) Effectiveness of the global protected area network in representing species diversity. Nature 428:9–12

Rodrigues RR et al. (2011) Large-scale ecological restoration of high-diversity tropical forests in SE Brazil. Forest Ecology and Management 261

Rother DC et al. (2018) How Legal-Oriented Restoration Programs Enhance Landscape Connectivity? Insights From the Brazilian Atlantic Forest. Tropical Conservation Science 11:194008291878507

Santos K, Kinoshita LS, Santos FAM (2007) Tree species composition and similarity in semideciduous forest fragments of southeastern Brazil. Biological Conservation 135(2): 268-277

Saraiva DD et al. (2018) How effective are protected areas in conserving tree taxonomic and phylogenetic diversity in subtropical Brazilian Atlantic Forests? Journal for Nature Conservation 42:28–35

Sfair JC et al. (2016) Taxonomic and functional divergence of tree assemblages in a fragmented tropical forest. Ecological Applications 2:1816–1826

Silva FR, Almeida-Neto M, Arena MVN (2014) Amphibian Beta Diversity in the Brazilian Atlantic Forest : Contrasting the Roles of Historical Events and Contemporary Conditions at Different Spatial Scales. PLoS ONE 9:1–9

Silva JMC da, Tabarelli M (2000) Tree species impoverishment and the future flora of the Atlantic forest of northeast Brazil. Nature 404:72–74

Slik JWF et al. (2015) An estimate of the number of tropical tree species. Proceedings of the National Academy of Sciences 112:201512611

SMA_57/ (2016) Resolução SMA 57/2016 Lista oficial das espécies da flora ameaçadas de extinção no Estado de São Paulo.

Socolar JB et al. (2016) How Should Beta-Diversity Inform Biodiversity Conservation? Trends in Ecology and Evolution 31:67–80

Soininen J, Heino J, Wang J (2018) A meta-analysis of nestedness and turnover components of beta diversity across organisms and ecosystems. Global Ecology and Biogeography 27:96–109

Solar RRDC et al. (2015) How pervasive is biotic homogenization in human-modified tropical forest landscapes? Ecology Letters n/a-n/a

Sparovek G et al. (2010) Brazilian agriculture and environmental legislation:status and future challenges. Environmental science & technology 44:14637–14641

Thomas CD et al. (2012) Protected areas facilitate species' range expansions. Proceedings of the National Academy of Sciences 109:14063–14068

Tjørve E (2010) How to resolve the SLOSS debate: Lessons from species-diversity models. Journal of Theoretical Biology 264:604–612

APPENDICES

Appendix 1: Total accumulated species richness (S total) from sampled forest fragments (N) in distinct protection categories (PL=private lands and SPA=Strictly Protected Areas) and among regions of São Paulo state (S obs), with the average species richness per site (S_mean +- sd). Exclusive and shared species' proportions based on S total.

Region	Categories	N	S total	S obs	S_mean +- sd	Exclusive	Shared
Southeast	PL	37	625	539 (86%)	74 +- 15	372 (59%)	167 (27%)
	SPA	1		253 (41%)	-	86 (14%)	
Center	PL	265	1,051	639 (61%)	54 +- 17	149(14%)	490 (47%)
	SPA	11		902 (86%)	277 +- 82	412 (39%)	
West	PL	65	1,022	476 (47%)	56 +- 20	127 (13%)	349 (34%)
	SPA	8		895 (88%)	212 +- 115	546 (53%)	
All	PLs	367	**1,558**	913 (59%)	56 +- 18	**198 (13%)**	**715 (46%)**
	SPA	20		1,360 (87%)	260 +- 110	**645 (41%)**	
		387		**1558**	**67 +- 30**		

Appendix 2: List of all forest fragments (Frag_id) among regions of São Paulo state and protection categories (Frag_cat), with the sampling method and sampling effort when the information was available. S_richness indicate que number of registered tree and shrub species. Vegetation types include all fragments classified as SSF = Seasonal Semideciduous Forests or as their ecotones: AFSS = Atlantic Forest *sensu stricto*, A/SF = Alluvial and swamp forests, SDF = Seasonal Deciduous Forests and CE = *Cerradão*.

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Sudeste	private	10TAV	Franco da Rocha	30min_walks	77 min	SSF_AFSS	87
Sudeste	private	11TAV	Franco da Rocha	30min_walks, fito	95 min	SSF_AFSS	76
Sudeste	private	14TAV	Guarulhos	30min_walks, fito	160 min	SSF_AFSS	89
Sudeste	private	15TAV	Guarulhos	30min_walks, fito	181 min	SSF_AFSS	111
Sudeste	private	16TAV	Arujá	30min_walks, fito	145 min	SSF_AFSS	80
Sudeste	private	17TAV	Santa Isabel	30min_walks	97 min	SSF_AFSS	71
Sudeste	private	18TAV	Santa Isabel	30min_walks	150 min	SSF_AFSS	78
Sudeste	private	19TAV	Santa Isabel	30min_walks, fito	107 min	SSF_AFSS	92
Sudeste	private	20TAV	Santa Isabel	30min_walks, fito	140 min	SSF_AFSS	94
Sudeste	private	21TAV	Jacareí	30min_walks, fito	145 min	SSF_AFSS	44
Sudeste	private	22TAV	Jacareí	30min_walks	156 min	SSF_AFSS	76
Sudeste	private	23TAV	Jacareí	30min_walks	180 min	SSF_AFSS	79
Sudeste	private	24TAV	Jacareí	30min_walks, fito	135 min	SSF_AFSS	121
Sudeste	private	25TAV	Jacareí	30min_walks	110 min	SSF_AFSS	83
Sudeste	private	26TAV	São José dos Campos	30min_walks	100 min	SSF_AFSS	23
Sudeste	private	27TAV	São José dos Campos	30min_walks, fito	90 min	SSF_AFSS	50
Sudeste	private	28TAV	Caçapava	30min_walks	165 min	SSF_AFSS	66
Sudeste	private	29TAV	Caçapava	30min_walks	120 min	SSF_AFSS	84
Sudeste	private	30TAV	Tremembé	30min_walks	180 min	SSF_AFSS	51
Sudeste	private	31TAV	Pindamonhangaba	30min_walks, fito	160 min	SSF_AFSS	47
Sudeste	private	32TAV	Pindamonhangaba	30min_walks	70 min	SSF_AFSS	44

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Sudeste	private	33TAV	Guaratinguetá	30min_walks	140 min	SSF_AFSS	79
Sudeste	private	34TAV	Guaratinguetá	30min_walks, fito	80 min	SSF_AFSS	81
Sudeste	private	36TAV	Cachoeira Paulista	30min_walks	172 min	SSF_AFSS	84
Sudeste	private	38TAV	Cruzeiro	30min_walks	140 min	SSF_AFSS	59
Sudeste	private	39TAV	Lavrinhas	30min_walks	70 min	SSF_AFSS	66
Sudeste	private	40TAV	Lavrinhas	30min_walks, fito	160 min	SSF_AFSS	80
Sudeste	private	41TAV	Queluz	30min_walks	105 min	SSF_AFSS	68
Sudeste	private	42TAV	Queluz	30min_walks	131 min	SSF_AFSS	66
Sudeste	private	43TAV	Queluz	30min_walks	126 min	SSF_AFSS	77
Sudeste	private	44TAV	Queluz	30min_walks, fito	160 min	SSF_AFSS	71
Sudeste	private	4TAV	Itupeva	30min_walks	123 min	SSF	55
Sudeste	private	5TAV	Itupeva	30min_walks	150 min	SSF	50
Sudeste	private	6TAV	Itupeva	30min_walks	150 min	SSF	96
Sudeste	private	7TAV	Jundiaí	30min_walks, fito	128 min	SSF	87
Sudeste	private	8TAV	Jundiaí	30min_walks	130 min	SSF	81
Sudeste	private	9TAV	Jundiaí	30min_walks, fito	90 min	SSF	94
Sudeste	protected	PE do Jaragua	São Paulo	non available	2nd. data	SSF_AFSS	253
Centro	private	100PAU	Paulínia	15minute_walks	90 min	SSF	59
Centro	private	101PAU	Paulínia	15minute_walks	75 min	SSF	73
Centro	private	103PAU	Paulínia	15minute_walks	30 min	SSF	36
Centro	private	104PAU	Paulínia	15minute_walks	45 min	SSF	50
Centro	private	105PAU	Paulínia	15minute_walks	135 min	SSF	78
Centro	private	106PAU	Paulínia	15minute_walks	105 min	SSF_A/SF	45
Centro	private	108PAU	Paulínia	15minute_walks	45 min	SSF	30
Centro	private	109PAU	Paulínia	15minute_walks	105 min	SSF_A/SF	49
Centro	private	10PAU	Paulínia	15minute_walks	270 min	SSF_A/SF	83
Centro	private	110PAU	Paulínia	15minute_walks	45 min	SSF	31
Centro	private	111PAU	Paulínia	15minute_walks	60 min	SSF	69

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	112PAU	Paulínia	15minute_walks	60 min	SSF	42
Centro	private	114PAU	Paulínia	15minute_walks	75 min	SSF	58
Centro	private	115PAU	Paulínia	15minute_walks	30 min	SSF_A/SF	28
Centro	private	116PAU	Paulínia	15minute_walks	75 min	SSF	61
Centro	private	11PAU	Paulínia	15minute_walks	105 min	SSF	62
Centro	private	12PAU	Paulínia	15minute_walks	30 min	SSF	21
Centro	private	13PAU	Paulínia	15minute_walks	105 min	SSF	87
Centro	private	14PAU	Paulínia	15minute_walks	75 min	SSF	45
Centro	private	15PAU	Paulínia	15minute_walks	60 min	SSF	42
Centro	private	16PAU	Paulínia	15minute_walks	180 min	SSF_CE	45
Centro	private	17PAU	Paulínia	15minute_walks	540 min	SSF	126
Centro	private	19PAU	Paulínia	15minute_walks	120 min	SSF	75
Centro	private	1PAU	Paulínia	15minute_walks	1185 min	SSF	176
Centro	private	20PAU	Paulínia	15minute_walks	150 min	SSF	99
Centro	private	21PAU	Paulínia	15minute_walks	165 min	SSF_A/SF	62
Centro	private	22PAU	Paulínia	15minute_walks	135 min	SSF	72
Centro	private	23PAU	Paulínia	15minute_walks	120 min	SSF	60
Centro	private	24PAU	Paulínia	15minute_walks	120 min	SSF	60
Centro	private	25PAU	Paulínia	15minute_walks	75 min	SSF	64
Centro	private	26PAU	Paulínia	15minute_walks	135 min	SSF_A/SF	44
Centro	private	28PAU	Paulínia	15minute_walks	135 min	SSF	50
Centro	private	29PAU	Paulínia	15minute_walks	135 min	SSF	86
Centro	private	2PAU	Paulínia	15minute_walks	630 min	SSF	149
Centro	private	30PAU	Paulínia	15minute_walks	120 min	SSF_A/SF	68
Centro	private	31PAU	Paulínia	15minute_walks	45 min	SSF	46
Centro	private	32PAU	Paulínia	15minute_walks	75 min	SSF	56
Centro	private	33PAU	Paulínia	15minute_walks	195 min	SSF	109
Centro	private	34PAU	Paulínia	15minute_walks	135 min	SSF	81

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	35PAU	Paulínia	15minute_walks	90 min	SSF	80
Centro	private	36PAU	Paulínia	15minute_walks	75 min	SSF	60
Centro	private	37PAU	Paulínia	15minute_walks	60 min	SSF	58
Centro	private	39PAU	Paulínia	15minute_walks	75 min	SSF	59
Centro	private	3PAU	Paulínia	15minute_walks	180 min	SSF	86
Centro	private	40PAU	Paulínia	15minute_walks	90 min	SSF	69
Centro	private	41PAU	Paulínia	15minute_walks	45 min	SSF_CE	42
Centro	private	42PAU	Paulínia	15minute_walks	135 min	SSF	101
Centro	private	43PAU	Paulínia	15minute_walks	75 min	SSF	41
Centro	private	44PAU	Paulínia	15minute_walks	120 min	SSF_A/SF	67
Centro	private	45PAU	Paulínia	15minute_walks	105 min	SSF	84
Centro	private	46PAU	Paulínia	15minute_walks	60 min	SSF_A/SF	40
Centro	private	48PAU	Paulínia	15minute_walks	45 min	SSF_A/SF	44
Centro	private	49PAU	Paulínia	15minute_walks	120 min	SSF_A/SF	52
Centro	private	4PAU	Paulínia	15minute_walks	420 min	SSF	125
Centro	private	50PAU	Paulínia	15minute_walks	75 min	SSF	69
Centro	private	53PAU	Paulínia	15minute_walks	90 min	SSF_A/SF	50
Centro	private	54PAU	Paulínia	15minute_walks	75 min	SSF	88
Centro	private	55PAU	Paulínia	15minute_walks	90 min	SSF	67
Centro	private	56PAU	Paulínia	15minute_walks	150 min	SSF	86
Centro	private	57PAU	Paulínia	15minute_walks	75 min	SSF	63
Centro	private	58PAU	Paulínia	15minute_walks	30 min	SSF_A/SF	38
Centro	private	59PAU	Paulínia	15minute_walks	60 min	SSF	31
Centro	private	5PAU	Paulínia	15minute_walks	360 min	SSF	115
Centro	private	60PAU	Paulínia	15minute_walks	75 min	SSF	70
Centro	private	61PAU	Paulínia	15minute_walks	90 min	SSF	75
Centro	private	62PAU	Paulínia	15minute_walks	75 min	SSF	62
Centro	private	63PAU	Paulínia	15minute_walks	30 min	SSF	24

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	64PAU	Paulínia	15minute_walks	60 min	SSF	48
Centro	private	65PAU	Paulínia	15minute_walks	75 min	SSF	52
Centro	private	66PAU	Paulínia	15minute_walks	105 min	SSF	85
Centro	private	67PAU	Paulínia	15minute_walks	45 min	SSF	45
Centro	private	68PAU	Paulínia	15minute_walks	75 min	SSF	44
Centro	private	69PAU	Paulínia	15minute_walks	60 min	SSF	59
Centro	private	6PAU	Paulínia	15minute_walks	150 min	SSF	105
Centro	private	70PAU	Paulínia	15minute_walks	30 min	SSF_A/SF	34
Centro	private	71PAU	Paulínia	15minute_walks	60 min	SSF	41
Centro	private	72PAU	Paulínia	15minute_walks	135 min	SSF	73
Centro	private	73PAU	Paulínia	15minute_walks	120 min	SSF	64
Centro	private	76PAU	Paulínia	15minute_walks	75 min	SSF	72
Centro	private	77PAU	Paulínia	15minute_walks	90 min	SSF_A/SF	82
Centro	private	78PAU	Paulínia	15minute_walks	60 min	SSF_A/SF	39
Centro	private	79PAU	Paulínia	15minute_walks	75 min	SSF	61
Centro	private	80PAU	Paulínia	15minute_walks	45 min	SSF_A/SF	31
Centro	private	82PAU	Paulínia	15minute_walks	30 min	SSF_A/SF	30
Centro	private	83PAU	Paulínia	15minute_walks	75 min	SSF_A/SF	48
Centro	private	84PAU	Paulínia	15minute_walks	30 min	SSF	26
Centro	private	85PAU	Paulínia	15minute_walks	45 min	SSF	27
Centro	private	86PAU	Paulínia	15minute_walks	75 min	SSF	56
Centro	private	87PAU	Paulínia	15minute_walks	30 min	SSF	54
Centro	private	88PAU	Paulínia	15minute_walks	45 min	SSF	48
Centro	private	89PAU	Paulínia	15minute_walks	60 min	SSF	45
Centro	private	8PAU	Paulínia	15minute_walks	90 min	SSF	91
Centro	private	9 PAU	Paulínia	15minute_walks	465 min	SSF	125
Centro	private	90 PAU	Paulínia	15minute_walks	45 min	SSF	35
Centro	private	91 PAU	Paulínia	15minute_walks	75 min	SSF	41

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	93 PAU	Paulínia	15minute_walks	75 min	SSF	71
Centro	private	94 PAU	Paulínia	15minute_walks	30 min	SSF	34
Centro	private	95 PAU	Paulínia	15minute_walks	90 min	SSF	65
Centro	private	96 PAU	Paulínia	15minute_walks	105 min	SSF	32
Centro	private	97 PAU	Paulínia	15minute_walks	60 min	SSF	56
Centro	private	99 PAU	Paulínia	15minute_walks	90 min	SSF_A/SF	55
Centro	private	101FAB	Itaju	random_walks	90 min	SSF_SDF	57
Centro	private	104FAB	Dois Córregos	random_walks	75 min	SSF	70
Centro	private	105FAB	Dois Córregos	random_walks	60 min	SSF_A/SF	31
Centro	private	106FAB	Dois Córregos	random_walks	90 min	SSF_CE	75
Centro	private	107FAB	Dois Córregos	random_walks	65 min	SSF_CE	47
Centro	private	108FAB	Dois Córregos	random_walks	90 min	SSF_A/SF	42
Centro	private	10ESTER	Cosmopolis	15minute_walks	60 min	SSF	57
Centro	private	10FAB	Porto Feliz	random_walks	45 min	SSF	18
Centro	private	11ESTER	Cosmopolis	15minute_walks	135 min	SSF	85
Centro	private	11FAB	Porto Feliz	random_walks	83 min	SSF	62
Centro	private	12ESTER	Cosmopolis	15minute_walks	45 min	SSF	39
Centro	private	12FAB	Porto Feliz	random_walks	98 min	SSF	46
Centro	private	132FAB	Araraquara	random_walks	60 min	SSF_CE	53
Centro	private	133FAB	Araraquara	random_walks	60 min	SSF_CE	41
Centro	private	134FAB	Araraquara	random_walks	75 min	SSF_A/SF	18
Centro	private	135FAB	Araraquara	random_walks	90 min	SSF_CE	54
Centro	private	136FAB	Araraquara	random_walks	90 min	SSF_SDF	40
Centro	private	137FAB	Araraquara	random_walks	45 min	SSF	23
Centro	private	138FAB	Araraquara	random_walks	40 min	SSF_CE	27
Centro	private	139FAB	Santa Ernestina	random_walks	45 min	SSF_A/SF	24
Centro	private	13ESTER	Cosmopolis	15minute_walks	60 min	SSF	54
Centro	private	13FAB	Porto Feliz	random_walks	44 min	SSF	32

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	145FAB	Porto Feliz	random_walks	110 min	SSF	45
Centro	private	146FAB	Capivari	random_walks	90 min	SSF	51
Centro	private	147FAB	Capivari	random_walks	156 min	SSF	30
Centro	private	148FAB	Rio das Pedras	random_walks	90 min	SSF	27
Centro	private	14ESTER	Cosmopolis	15minute_walks	45 min	SSF	73
Centro	private	14FAB	Porto Feliz	random_walks	16 min	SSF_SDF	12
Centro	private	15ESTER	Cosmopolis	15minute_walks	30 min	SSF	38
Centro	private	15FAB	Porto Feliz	random_walks	124 min	SSF	60
Centro	private	16ESTER	Cosmopolis	15minute_walks	60 min	SSF	65
Centro	private	16FAB	Porto Feliz	random_walks	7 min	SSF	12
Centro	private	17ESTER	Cosmopolis	15minute_walks	60 min	SSF_A/SF	49
Centro	private	17FAB	Porto Feliz	random_walks	66 min	SSF	50
Centro	private	18ESTER	Cosmopolis	15minute_walks	30 min	SSF	20
Centro	private	18FAB	Porto Feliz	random_walks	40 min	SSF_SDF	19
Centro	private	19ESTER	Cosmopolis	15minute_walks	30 min	SSF	23
Centro	private	19FAB	Porto Feliz	random_walks	50 min	SSF	58
Centro	private	1ESTER	Cosmopolis	15minute_walks	195 min	SSF	25
Centro	private	1FAB	Porto Feliz	random_walks	81 min	SSF	54
Centro	private	20ESTER	Cosmopolis	15minute_walks	45 min	SSF	37
Centro	private	20FAB	Porto Feliz	random_walks	80 min	SSF	42
Centro	private	21ESTER	Cosmopolis	15minute_walks	90 min	SSF	60
Centro	private	21FAB	Porto Feliz	random_walks	35 min	SSF	28
Centro	private	22ESTER	Cosmopolis	15minute_walks	120 min	SSF	83
Centro	private	22FAB	Porto Feliz	random_walks	120 min	SSF	62
Centro	private	23ESTER	Cosmopolis	15minute_walks	75 min	SSF	53
Centro	private	23FAB	Porto Feliz	random_walks	72 min	SSF	44
Centro	private	24ESTER	Cosmopolis	15minute_walks	105 min	SSF	75
Centro	private	24FAB	Porto Feliz	random_walks	135 min	SSF	61

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	25FAB	Porto Feliz	random_walks	85 min	SSF	42
Centro	private	26FAB	Rafard	random_walks	105 min	SSF	59
Centro	private	27FAB	Rafard	random_walks	108 min	SSF	45
Centro	private	28FAB	Rafard	random_walks	68 min	SSF	40
Centro	private	29FAB	Rafard	random_walks	76 min	SSF	54
Centro	private	2ESTER	Cosmopolis	15minute_walks	105 min	SSF	91
Centro	private	2FAB	Porto Feliz	random_walks	45 min	SSF	38
Centro	private	30FAB	Rafard	random_walks	102 min	SSF	45
Centro	private	31FAB	Rafard	random_walks	80 min	SSF	55
Centro	private	32FAB	Capivari	random_walks	75 min	SSF	67
Centro	private	33FAB	Capivari	random_walks	156 min	SSF	35
Centro	private	34FAB	Capivari	random_walks	90 min	SSF	58
Centro	private	35FAB	Capivari	random_walks	90 min	SSF	14
Centro	private	36FAB	Capivari	random_walks	145 min	SSF	67
Centro	private	37FAB	Capivari	random_walks	40 min	SSF_A/SF	14
Centro	private	38FAB	Capivari	random_walks	71 min	SSF	50
Centro	private	39FAB	Capivari	random_walks	95 min	SSF	71
Centro	private	3ESTER	Cosmopolis	15minute_walks	60 min	SSF	69
Centro	private	3FAB	Porto Feliz	random_walks	94 min	SSF	57
Centro	private	40FAB	Capivari	random_walks	28 min	SSF	26
Centro	private	41FAB	Rio das Pedras	random_walks	72 min	SSF	47
Centro	private	42FAB	Rio das Pedras	random_walks	79 min	SSF	54
Centro	private	43FAB	Rio das Pedras	random_walks	90 min	SSF	47
Centro	private	44FAB	Rio das Pedras	random_walks	90 min	SSF	46
Centro	private	45FAB	Rio das Pedras	random_walks	90 min	SSF	54
Centro	private	46FAB	Rio das Pedras	random_walks	60 min	SSF	49
Centro	private	47FAB	Rio das Pedras	random_walks	105 min	SSF	55
Centro	private	48FAB	Rio das Pedras	random_walks	101 min	SSF	71

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	49FAB	Rio das Pedras	random_walks	16 min	SSF	28
Centro	private	4ESTER	Cosmopolis	15minute_walks	105 min	SSF	52
Centro	private	50FAB	Santa Bárbara D'Oeste	random_walks	86 min	SSF	67
Centro	private	51FAB	Santa Bárbara D'Oeste	random_walks	75 min	SSF	59
Centro	private	52FAB	Santa Bárbara D'Oeste	random_walks	88 min	SSF	57
Centro	private	53FAB	Santa Bárbara D'Oeste	random_walks	89 min	SSF	60
Centro	private	54FAB	Santa Bárbara D'Oeste	random_walks	86 min	SSF	69
Centro	private	55FAB	Santa Bárbara D'Oeste	random_walks	63 min	SSF	69
Centro	private	56FAB	Santa Bárbara D'Oeste	random_walks	48 min	SSF	59
Centro	private	57FAB	Santa Bárbara D'Oeste	random_walks	53 min	SSF	42
Centro	private	58FAB	Santa Bárbara D'Oeste	random_walks	44 min	SSF_A/SF	24
Centro	private	59FAB	Piracicaba	random_walks	77 min	SSF	48
Centro	private	5ESTER	Cosmopolis	15minute_walks	75 min	SSF	67
Centro	private	5FAB	Porto Feliz	random_walks	38 min	SSF	48
Centro	private	60FAB	Piracicaba	random_walks	99 min	SSF	50
Centro	private	61FAB	Piracicaba	random_walks	144 min	SSF	65
Centro	private	62FAB	Piracicaba	random_walks	52 min	SSF	53
Centro	private	63FAB	Piracicaba	random_walks	52 min	SSF	13
Centro	private	64FAB	Monte Mor	random_walks	82 min	SSF	59
Centro	private	65FAB	Santa Bárbara D'Oeste	random_walks	92 min	SSF	37

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	66FAB	Piracicaba	random_walks	118 min	SSF	70
Centro	private	67FAB	Piracicaba	random_walks	75 min	SSF	58
Centro	private	68FAB	Águas de São Pedro	random_walks	66 min	SSF	67
Centro	private	69FAB	Iracemápolis	random_walks	104 min	SSF	65
Centro	private	6ESTER	Cosmopolis	15minute_walks	30 min	SSF	37
Centro	private	6FAB	Porto Feliz	random_walks	26 min	SSF	34
Centro	private	70FAB	Iracemápolis	random_walks	93 min	SSF	74
Centro	private	71FAB	Piracicaba	random_walks	105 min	SSF	71
Centro	private	75FAB	Charqueada	random_walks	120 min	SSF	112
Centro	private	76FAB	Charqueada	random_walks	83 min	SSF	56
Centro	private	7ESTER	Cosmopolis	15minute_walks	165 min	SSF	92
Centro	private	7FAB	Porto Feliz	random_walks	75 min	SSF	59
Centro	private	82FAB	Charqueada	random_walks	105 min	SSF	92
Centro	private	83FAB	Piracicaba	random_walks	30 min	SSF	34
Centro	private	84FAB	Barra Bonita	random_walks	70 min	SSF	47
Centro	private	85FAB	Barra Bonita	random_walks	78 min	SSF	49
Centro	private	86FAB	Barra Bonita	random_walks	95 min	SSF	56
Centro	private	87FAB	Barra Bonita	random_walks	45 min	SSF	41
Centro	private	88FAB	Barra Bonita	random_walks	45 min	SSF	27
Centro	private	89FAB	Barra Bonita	random_walks	75 min	SSF_A/SF	24
Centro	private	8ESTER	Cosmopolis	15minute_walks	30 min	SSF	38
Centro	private	8FAB	Porto Feliz	random_walks	40 min	SSF	50
Centro	private	90FAB	Barra Bonita	random_walks	80 min	SSF	65
Centro	private	91FAB	Barra Bonita	random_walks	105 min	SSF	71
Centro	private	92FAB	Barra Bonita	random_walks	35 min	SSF	34
Centro	private	93FAB	Barra Bonita	random_walks	90 min	SSF_A/SF	32
Centro	private	94FAB	Barra Bonita	random_walks	90 min	SSF	73

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	95FAB	Barra Bonita	random_walks	50 min	SSF_SDF	34
Centro	private	96FAB	Jaú	random_walks	60 min	SSF	47
Centro	private	97FAB	Jaú	random_walks	90 min	SSF	49
Centro	private	98FAB	Jaú	random_walks	90 min	SSF	58
Centro	private	99FAB	Pederneiras	random_walks	60 min	SSF	41
Centro	private	9ESTER	Cosmopolis	15minute_walks	45 min	SSF	48
Centro	private	9FAB	Porto Feliz	random_walks	45 min	SSF	29
Centro	private	C01G	Ipeúna	fito	1200 m^2 / frag	SSF	46
Centro	private	C01P	Charqueada	fito	1200 m^2 / frag	SSF	28
Centro	private	C02G	Piracicaba	fito	1200 m^2 / frag	SSF	30
Centro	private	C02P	Charqueada	fito	1200 m^2 / frag	SSF	29
Centro	private	C03G	Piracicaba	fito	1200 m^2 / frag	SSF	34
Centro	private	C03P	Piracicaba	fito	1200 m^2 / frag	SSF	25
Centro	private	P01G	Rio Claro	fito	1200 m^2 / frag	SSF	40
Centro	private	P01P	Corumbataí	fito	1200 m^2 / frag	SSF	31
Centro	private	P02G	Rio Claro	fito	1200 m^2 / frag	SSF	32
Centro	private	P02P	Rio Claro	fito	1200 m^2 / frag	SSF	46
Centro	private	P03G	Rio Claro	fito	1200 m^2 / frag	SSF	32
Centro	private	P03P	Rio Claro	fito	1200 m^2 / frag	SSF	41
Centro	private	ANA1	Batatais	fito	1000 m^2 / frag	SSF_CE	46
Centro	private	ANA10	Batatais	fito	1000 m^2 / frag	SSF_CE	50
Centro	private	ANA11	Batatais	fito	1000 m^2 / frag	SSF_CE	63
Centro	private	ANA12	Batatais	fito	1000 m^2 / frag	SSF_CE	39
Centro	private	ANA13	Batatais	fito	1000 m^2 / frag	SSF_CE	65
Centro	private	ANA14	Batatais	fito	1000 m^2 / frag	SSF_CE	63
Centro	private	ANA15	Batatais	fito	1000 m^2 / frag	SSF_CE	37
Centro	private	ANA16	Batatais	fito	1000 m^2 / frag	SSF_CE	39
Centro	private	ANA17	Batatais	fito	1000 m^2 / frag	SSF_CE	24

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Centro	private	ANA18	Batatais	fito	1000 m^2 / frag	SSF_CE	41
Centro	private	ANA2	Batatais	fito	1000 m^2 / frag	SSF_CE	57
Centro	private	ANA3	Batatais	fito	1000 m^2 / frag	SSF_CE	45
Centro	private	ANA4	Batatais	fito	1000 m^2 / frag	SSF_CE	55
Centro	private	ANA5	Batatais	fito	1000 m^2 / frag	SSF_CE	56
Centro	private	ANA6	Batatais	fito	1000 m^2 / frag	SSF_CE	51
Centro	private	ANA7	Batatais	fito	1000 m^2 / frag	SSF_CE	47
Centro	private	ANA8	Batatais	fito	1000 m^2 / frag	SSF_CE	56
Centro	private	ANA9	Batatais	fito	1000 m^2 / frag	SSF_CE	59
Centro	private	1TAV	Campinas	30min_walks	90 min	SSF	40
Centro	private	3TAV	Campinas	30min_walks	148 min	SSF	66
Centro	protected	EE de Angatuba	Angatuba	non available	2nd. data	SSF	429
Centro	protected	EE de Paranapanema	Paranapanema	non available	2nd. data	SSF_AFSS	288
Centro	protected	EE de Ribeirao Preto	Ribeirão Preto	non available	2nd. data	SSF	216
Centro	protected	EE do Barreiro Rico	Anhembi	non available	2nd. data	SSF	134
Centro	protected	EE Ibicatu	Piracicaba	non available	2nd. data	SSF	198
Centro	protected	EE Itirapina	Itirapina	non available	2nd. data	SSF_CE	239
Centro	protected	EE Mogi Guacu	Mogi Guaçu	non available	2nd. data	SSF_CE	247
Centro	protected	PE das Furnas do Bom Jesus	Pedregulho	non available	2nd. data	SSF_CE	146
Centro	protected	PE de Porto Ferreira	Porto Ferreira	non available	2nd. data	SSF_CE	226
Centro	protected	PE de Vassununga	Santa Rita do Passa Quatro	non available	2nd. data	SSF_CE	322
Centro	protected	RB Mogi-Guacu	Mogi Guaçu	non available	2nd. data	SSF_CE	328
Oeste	private	10COL	Itajobi	15minute_walks	non available	SSF_CE	70
Oeste	private	11COL	Santa_Adelia	15minute_walks	non available	SSF	34
Oeste	private	12COL	Santa_Adelia	15minute_walks	non available	SSF_CE	45
Oeste	private	13COL	Santa_Adelia	15minute_walks	non available	SSF	41
Oeste	private	2COL	Pindorama	15minute_walks	non available	SSF	31

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Oeste	private	4COL	Santa_Adelia	15minute_walks	non available	SSF	26
Oeste	private	5COL	Santa_Adelia	15minute_walks	non available	SSF_CE	81
Oeste	private	6COL	Santa_Adelia	15minute_walks	non available	SSF_CE	41
Oeste	private	7COL	Itapolis	15minute_walks	non available	SSF	12
Oeste	private	8COL	Santa_Adelia	15minute_walks	non available	SSF_CE	69
Oeste	private	9COL	Santa_Adelia	15minute_walks	non available	SSF	38
Oeste	private	100FAB	Iacanga	random_walks	145 min	SSF	88
Oeste	private	102FAB	Iacanga	random_walks	45 min	SSF_A/SF	27
Oeste	private	103FAB	Arealva	random_walks	90 min	SSF_SDF	50
Oeste	private	109FAB	Araçatuba	random_walks	50 min	SSF_SDF	31
Oeste	private	110FAB	Araçatuba	random_walks	45 min	SSF	34
Oeste	private	111FAB	Valparaiso	random_walks	135 min	SSF_SDF	72
Oeste	private	112FAB	Valparaíso	random_walks	45 min	SSF_SDF	41
Oeste	private	114FAB	Valparaiso	random_walks	70 min	SSF_CE	71
Oeste	private	116FAB	Valparaiso	random_walks	85 min	SSF_CE	77
Oeste	private	117FAB	Valparaiso	random_walks	105 min	SSF_SDF	56
Oeste	private	118FAB	Valparaiso	random_walks	55 min	SSF_SDF	38
Oeste	private	120FAB	Valparaiso	random_walks	72 min	SSF_SDF	40
Oeste	private	121FAB	Valparaiso	random_walks	21 min	SSF	15
Oeste	private	122FAB	Valparaiso	random_walks	45 min	SSF_A/SF	15
Oeste	private	123FAB	Valparaiso	random_walks	66 min	SSF_SDF	26
Oeste	private	124FAB	Valparaiso	random_walks	53 min	SSF_A/SF	22
Oeste	private	125FAB	Valparaiso	random_walks	72 min	SSF_SDF	45
Oeste	private	126FAB	Valparaiso	random_walks	40 min	SSF	18
Oeste	private	127FAB	Valparaiso	random_walks	60 min	SSF_SDF	42
Oeste	private	128FAB	Valparaiso	random_walks	35 min	SSF	28
Oeste	private	129FAB	Andradina	random_walks	91 min	SSF_CE	75
Oeste	private	130FAB	Andradina	random_walks	33 min	SSF_SDF	30

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Oeste	private	131FAB	Andradina	random_walks	135 min	SSF_CE	64
Oeste	private	140FAB	Santa Ernestina	random_walks	90 min	SSF_A/SF	30
Oeste	private	141FAB	Santa Ernestina	random_walks	128 min	SSF_CE	77
Oeste	private	142FAB	Santa Ernestina	random_walks	105 min	SSF_CE	80
Oeste	private	143FAB	Santa Ernestina	random_walks	75 min	SSF_CE	67
Oeste	private	144FAB	Santa Ernestina	random_walks	45 min	SSF_CE	56
Oeste	private	72FAB	Ipaussu	random_walks	84 min	SSF	78
Oeste	private	73FAB	Ipaussu	random_walks	39 min	SSF	54
Oeste	private	74FAB	Ipaussu	random_walks	17 min	SSF	21
Oeste	private	77FAB	Igaraçu do Tietê	random_walks	109 min	SSF	53
Oeste	private	78FAB	Igaraçu do Tietê	random_walks	105 min	SSF	62
Oeste	private	79FAB	Igaraçu do Tietê	random_walks	90 min	SSF	54
Oeste	private	80FAB	Igaraçu do Tietê	random_walks	120 min	SSF	70
Oeste	private	81FAB	Igaraçu do Tietê	random_walks	60 min	SSF	38
Oeste	private	PabG1	Novo Horizonte	fito	10000 m^2 / frag	SSF_CE	88
Oeste	private	PabG2	Sales	fito	10000 m^2 / frag	SSF_CE	67
Oeste	private	PabG3	Planalto	fito	10000 m^2 / frag	SSF_CE	83
Oeste	private	PabG4	União Paulista	fito	10000 m^2 / frag	SSF_CE	76
Oeste	private	PabG5	São João de Iracema	fito	10000 m^2 / frag	SSF	77
Oeste	private	PabG6	Nova Granada	fito	10000 m^2 / frag	SSF_CE	69
Oeste	private	PabG7	Barretos	fito	10000 m^2 / frag	SSF_CE	80
Oeste	private	PabG8	Bebedouro	fito	10000 m^2 / frag	SSF_CE	45
Oeste	private	PabG9	Matão	fito	10000 m^2 / frag	SSF	123
Oeste	private	PabP1	Santo Antônio do Aracanguá	fito	10000 m^2 / frag	SSF_CE	92
Oeste	private	PabP2	Macaubal	fito	10000 m^2 / frag	SSF_CE	80
Oeste	private	PabP3	Votuporanga	fito	10000 m^2 / frag	SSF_CE	77

Region	Frag_cat	Frag_id	Municipality	Sampling_method	Sampling effort	Vegetation	S_richness
Oeste	private	PabP4	Turmalina	fito	10000 m^2 / frag	SSF_CE	82
Oeste	private	PabP5	Palestina	fito	10000 m^2 / frag	SSF_CE	77
Oeste	private	PabP6	Palestina	fito	10000 m^2 / frag	SSF	66
Oeste	private	PabP7	Barretos	fito	10000 m^2 / frag	SSF	54
Oeste	private	PabP8	Taquaritinga	fito	10000 m^2 / frag	SSF	83
Oeste	private	PabP9	Pindorama	fito	10000 m^2 / frag	SSF	65
Oeste	protected	EE Avare	Avaré	non available	2nd. data	SSF_CE	183
Oeste	protected	EE de Bauru	Bauru	non available	2nd. data	SSF	175
Oeste	protected	EE de Paulo de Faria	Paulo de Faria	non available	2nd. data	SSF_CE	95
Oeste	protected	EE dos Caetetus	Assis	non available	2nd. data	SSF	240
Oeste	protected	EE Mico Leao Preto	Teodoro Sampaio	non available	2nd. data	SSF	156
Oeste	protected	PE do Aguapei	Castilho	non available	2nd. data	SSF	90
Oeste	protected	PE do Morro do Diabo	Teodoro Sampaio	non available	2nd. data	SSF	584
Oeste	protected	PE do Rio Peixe	Dracena	non available	2nd. data	SSF	100

CHAPTER 2. HETEROGENIZATION OF TREE/SHRUB ASSEMBLAGES IN AGRICULTURAL LANDSCAPES

Cristina Yuri Vidal* [1, 2]; Diogo Souza Bezerra Rocha[3]; Marinez Ferreira de Siqueira[3]; Tadeu Siqueira[4], Ricardo Ribeiro Rodrigues[2],

[1] Universidade Estadual de Campinas (UNICAMP), Programa de Pós-Graduação em Biologia Vegetal, Instituto de Biologia, Campinas-SP, Brazil.

[2] Departamento de Ciências Biológicas, Universidade de São Paulo, Escola Superior de Agricultura “Luiz de Queiroz”, Piracicaba-SP, Brazil.

[3] Jardim Botânico do Rio de Janeiro, Rio de Janeiro-RJ, Brazil.

[4] Instituto de Biociências, Universidade Estadual Paulista (UNESP), Rio Claro - SP, Brasil

ABSTRACT

There is an increasing worldwide interest on the conservation of tropical forests, since over 50% of their area has been converted into agricultural lands and other uses. Understanding how the remaining biodiversity is distributed along agricultural landscapes is an essential task to guide future conservation strategies. To understand the long-term effects of fragmentation on biodiversity, we investigated whether forest fragments in southeastern Brazil are under a taxonomic homogenization or heterogenization process. We estimated pre-deforestation species richness and composition based on a Species Distribution Modelling approach, and compared them to the observed patterns of α- and β-diversity. In particular, we asked (i) if changes in β-diversity reveal convergence or divergence on species composition; (ii) if these changes are similar between forest fragments in Strictly Protected Areas (SPAs) (n=20) and within private lands (n=367) and in different regions of the state (West, Center, and Southeast). We detected steep reductions in observed local species richness in relation to our modeled predictions, and this was particularly true among forest fragments in non-protected private lands. The higher observed β diversity indicated an overall biotic heterogenization process, which is consistent with the idea that the originally diverse vegetation is now reduced to small and isolated patches, with unique disturbance histories and impoverished communities derived from a large regional species pool. Recognizing that conservation of biodiversity extends far beyond the boundaries of strictly Protected Areas, we advocate that forest fragments are valuable for conservation in agricultural landscapes, with particular relevance for private lands, which represent the most exposed and neglected share of what is left.

INTRODUCTION

There is an increasing worldwide interest on the conservation and restoration of tropical forests, known for holding a substantial portion of the world's terrestrial biodiversity (Myers et al. 2000; Chazdon, Harvey, et al. 2009; Gardner et al. 2010; Slik et al. 2015), and yet subject to extensive land use conversion (Gibson et al. 2011; Laurance et al. 2014; Mendenhall et al. 2016). With over 50% of tropical forest converted into agricultural lands or other uses, deforestation rates are still on the rise as much as the prospect for agricultural expansion in tropical developing countries (Hansen et al. 2013; Laurance et al. 2014). Understanding how the remaining biodiversity is distributed along human modified landscapes (HMLs) and what have changed after forest conversion are essential questions to guide future conservation strategies (Tabarelli et al. 2010; Melo, Arroyo-Rodríguez, et al. 2013; Laurance et al. 2014; Socolar et al. 2016). Within HMLs, tropical forest fragments comprehend a variety of different sized habitats, including forests that never experienced clear cutting or severe impacts (*i.e.* primary forests), and the full spectrum of degraded forests that are regenerating after extraction, fire or abandonment of croplands and pastures, among other previous land-uses (*i.e.* secondary forests) (Gibson et al. 2011; Melo, Arroyo-Rodríguez, et al. 2013; Malhi et al. 2014; Arroyo-Rodriguez et al. 2015). A very narrow fraction of these forest fragments are under restrictive categories of protected areas, where biodiversity conservation is tangible to a limited extent (Andam et al. 2008; Joppa et al. 2008; Coetzee et al. 2014; Gray et al. 2016). In this context, the variety of forest fragments located within private lands not only represent the largest share of what is left (Gardner et al. 2009; Sparovek et al. 2012; Soares-Filho et al. 2014; Mendenhall et al. 2016), but also the most neglected. These fragments are rarely explicitly targeted in conservation programs, as the focus is usually on avoiding deforestation, failing to go beyond and avert the anthropogenic disturbances (Chazdon, Harvey, et al. 2009; Barlow et al. 2016). Several studies have shown that secondary forests may play an important role in conservation (Santos et al. 2007, Chazdon et al. 2009; Dent & Wright 2009; Tabarelli et al. 2012), as they hold a depleted but relevant portion of biodiversity even within HMLs. Abundant evidence is available for birds (Karp et al. 2012; Morante-Filho, Faria, et al. 2015; Emer et al. 2018), mammals (Galetti

et al. 2009; Pardini et al. 2010; Beca et al. 2017) and plants (Arroyo-Rodríguez et al. 2008; Norden et al. 2009; Lima et al. 2015; Carneiro et al. 2016; Machado et al. 2016; Sfair et al. 2016; Farah et al. 2017).

Effects of habitat loss and fragmentation over biodiversity have been intensively studied over the past three decades, with primarily focus at more local scales (Gibson et al. 2011; Karp et al. 2012; Vellend et al. 2013; Dornelas et al. 2014; Malhi et al. 2014; Murphy & Romanuk 2014; Newbold et al. 2015; Barlow et al. 2016). There have been an increase on studies focusing on broader extensions, based on the assumption that we cannot properly understand the consequences of deforestation if disregarding the influence of entire landscapes over local processes (Tscharntke et al. 2012; Malhi et al. 2014). Also, beyond the intuitive interest on species richness loss, an emerging issue of interest is how community composition responds to fragmentation along spatial gradients and periods of time (Karp et al. 2012; Arroyo-Rodríguez et al. 2013; Solar et al. 2015; Morante-Filho, Arroyo-Rodríguez, et al. 2015; França et al. 2016; Collins et al. 2017; Olden et al. 2018). Measures of the local species diversity (α) coupled with the variation in species composition among sites (β) can indicate if communities are converging or diverging in response to fragmentation, providing relevant information on the mechanisms responsible for the maintenance of regional diversity (Socolar et al. 2016).

Some studies have demonstrated that forest fragmentation and degradation result into biotic homogenization (Vellend et al. 2007; Lôbo et al. 2011; Karp et al. 2012; Marcelo Tabarelli et al. 2012; Püttker et al. 2015; Zwiener et al. 2017), *i.e.* the convergence of biotas in time and space, in which communities may suffer a simplification of their genetic, taxonomic and functional diversities (McKinney & Lockwood 1999; Olden & Rooney 2006). The rationale is that more ecologically specialized species (“losers”) are locally extinct, while a much narrower sub-set of generalists, with high dispersal abilities (“winners”), override them (Silva & Tabarelli 2000; Lôbo et al. 2011; Marcelo Tabarelli et al. 2012; Siqueira et al. 2015; Mendenhall et al. 2016). This process results in impoverished communities that represent sub-sets of a larger pool of species, translated by reduced β-diversity and high contribution of the nestedness

component on β diversity. The predominance of nestedness suggests conservation efforts might focus on the richest sites, as long as they are connected to allow and support viable communities (Martensen et al. 2008; Howe 2014; Socolar et al. 2016; Emer et al. 2018). An opposite consequence to fragmentation and degradation occurs when communities diverge on composition over time and space (*i.e.* enhanced β diversity) because they suffer different frequencies and levels of disturbances combined with dispersal limitations and environmental heterogeneity, resulting into biotic heterogenization (Dornelas et al. 2014; Solar et al. 2015; Sfair et al. 2016; Catano et al. 2017; Collins et al. 2017). In this scenario, widespread regional diversity conservation is only possible if targeting multiple sites.

Very few studies of plant community changes in response to fragmentation evaluate β-diversity patterns based on temporal replicates (Lôbo et al. 2011; Dornelas et al. 2014; Haddad et al. 2015; Collins et al. 2017); most of them adopt a space-for-time approach (*e.g.,* disturbed x undisturbed) regardless of the fact that distinct sites could reflect distinct pre-disturbance conditions (França et al. 2016; Collins et al. 2017). In addition to the lack of temporal replicates, severely deforested landscapes may not be suitable for the space-for-time approach when in the absence of large forest remnants or high forest-covered regions to represent undisturbed ecosystems. For that matter, environmental niche modeling (ENM) and species distribution modeling (SDM) can be useful to provide species' spatial occurrence disregarding the effects of anthropogenic disturbances, which is a major driver of community composition changes (Malhi et al. 2014, Catano et al. 2017). Since ENM is based on the niche concept and considers environmental conditions as the primarily influence over the establishment of a given species (De Marco Junior & Siqueira 2009), it results in maps representing the geographic space where the abiotic conditions are appropriate (Peterson & Soberon 2012). Distinctly from ENM, which disregards dispersal/colonization limitations and biotic interactions, SDM restrict the model calibration to accessible areas, incorporating dispersal issues into analyses and producing maps where a focal species may potentially occur, with varying degrees of suitability (De Marco Junior & Siqueira 2009; Peterson & Soberon 2012). For community-level modeling, further methodologies are available to

adjust the over-prediction of the number of species coexisting at a given location (Guisan & Rahbek 2011; Calabrese et al. 2014; Gavish et al. 2017). Therefore, these approaches combined can be used to generate expectations of what communities would look like, in terms of species composition, if they were not disturbed by land use conversion.

In recent years, Brazil has stood out among tropical developing countries for its environmental engagement, which resulted on the exceptional decline in deforestation rates during 2000-2012 (Hansen et al. 2013; Loyola 2014), despite the fact that it already has 30% of its total area occupied by agricultural lands (Martinelli et al. 2010). In Sao Paulo state, southeastern Brazil, which includes two biodiversity hotspots (Atlantic Forest and Cerrado) (Myers et al. 2000; Laurance 2009), deforestation has taken place during the last three centuries (Metzger 2009; Joly et al. 2014), and surveys only became a common practice in the last 30 years (Haddad et al. 2015; Renato Augusto Ferreira de Lima et al. 2015). With a very long history of land conversion for agricultural purposes, most remaining vegetation is comprised by small forest fragments (i.e. <50ha) (Ribeiro et al. 2009), representing an unique opportunity to understand the long-term effects of fragmentation on biodiversity. The purpose of our study was to evaluate whether the woody assemblages on forest fragments are under a taxonomic homogenization or heterogenization process in response to habitat fragmentation. For that matter, we estimated pre-deforestation species richness and composition based on a Species Distribution Modelling approach, and compared them to the observed patterns of α- and β-diversity. In particular, we asked (i) if changes in β-diversity indicate convergent or divergent composition; (ii) if these changes are similar between forest fragments under strict protection or within private lands and in different regions of the state. In the hyper-fragmented landscapes of this study, we expected to find lower mean values of α-diversity within private lands relative to strictly Protected Areas. Additionally, because of intrinsic environmental heterogeneity strengthened by fragmentation disturbances, we expected an overall increase in β-diversity, indicating a taxonomic differentiation process. This pattern should be particularly more evident for unprotected forest fragments located within private lands, where

fragments are more susceptible to a broader range of recurrent disturbances (Laurance et al. 2014; Malhi et al. 2014).

METHODS

Study region

The state of Sao Paulo is located within the range of two current global hotspots, the Atlantic Forest and Cerrado (tropical savannas) (Myers et al. 2000). With a long history of deforestation caused by timber extraction and agricultural cycles (coffee, pasture, orange, sugarcane) (Metzger 2009), the São Paulo state case-study may provide relevant insight about the long-term effects of habitat loss and fragmentation on biodiversity, which can be useful for other tropical regions facing the same threats (Laurance 2009; Taubert et al. 2018). Both hotspots are poorly protected, with only 1.6% and 0.5% of Atlantic Forest and Cerrado's original area protected as strictly Protected Areas (Durigan et al. 2006; Ribeiro et al. 2009; Carranza et al. 2014). The remaining vegetation cover in the interior plateau (*i.e.* excluding coastal areas) ranges from 1 to 30% (São Paulo State Forest Inventory 2011), mostly located within private rural properties (Gardner et al. 2009; Sparovek et al. 2012; Soares-Filho et al. 2014; Mendenhall et al. 2016).

In order to facilitate analyses' interpretation along the extent region of this study, we adopted the ecological regions defined by Setzer (1966), which divide the state in 6 sub-regions based on climate, soil, topography and vegetation variables (Setzer 1966). We excluded the south and north coastal areas because their forest cover is well above the rest of the state. To meet a minimum of 30 localities per sub-region, we joined Southwest and Northwest into one single "West" sub-region (**Figure 1**).

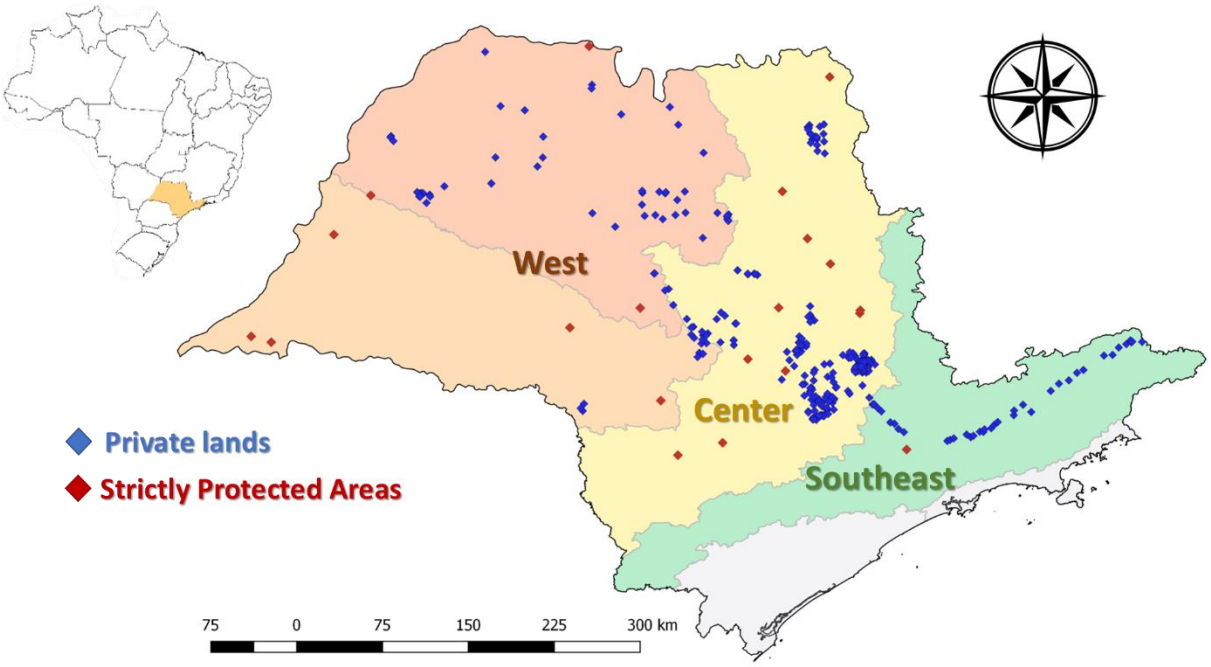


Figure 1: Distribution of forest fragments among regions in São Paulo state (Southeastern Brazil), considering those in private lands (n=367) and strictly protected areas (n=20). Mean forest cover based on São Paulo State Forest Inventory (2011): West = 6.5%, Center= 10.9%, Southeast= 27.7%.

Regarding vegetation, we focused on the predominant seasonal semi-deciduous forest (SSF), considering its transition to evergreen forests or forested savannas (Cerradão) (Oliveira-Filho & Fontes 2000; Durigan & Ratter 2006), and all other forest ecosystems included in this extension: swamp, alluvial and deciduous forests. Despite the fact that these forests are influenced and determined by soil, altitude and climatic conditions (Morellato & Haddad 2000; Oliveira-Filho & Fontes 2000), their floristic composition are strongly influenced by the surrounding vegetation (*e.g.* SSF), creating complex transitional mosaics and continuum distribution of species (Kurtz et al. 2015; Oliveira-Filho & Fontes 2000).

Woody plant species occurrence data

We chose to use woody plant species (*i.e.* trees and shrubs) in this study because they represent a fundamental structure and functional component of forest ecosystems, as they support food webs and represent a substantial

proportion of tropical diversity (Arroyo-Rodriguez et al. 2015; Magnago et al. 2015; Slik et al. 2015). Based on a compilation of 367 floristic surveys in private lands developed by the Forest Ecology and Restoration Laboratory (University of São Paulo) under the scope of "Environmental Planning Programs" (more details in Rodrigues et al. 2011), we initially defined a species pool of 921 species (20,662 records). To enhance our sample of the environmental space occupied by these species and to improve modelling outcomes, we retrieved complementary national occurrence data for these species from SpeciesLink (http://splink.cria.org.br/) and NeoTropTree (http://prof.icb.ufmg.br/treeatlan/). Finally, we selected 726 species occurring in more than 10 localities along the Brazilian territory.

We also gathered available checklists for 20 strictly Protected Areas within the study region (**Figure 1**) following the same criteria regarding forest types as those applied to select forests in private lands. We did not use the strictly Protected Areas localities to build species distribution models due to the lack of precision on their geographical coordinates, varying from a random point within their boundaries to the municipality centroid. We compiled the floristic surveys available at Fundação Florestal (http://fflorestal.sp.gov.br) and Instituto Florestal (http://iflorestal.sp.gov.br), both related to the São Paulo State Environmental Secretariat, and WWF Protected Areas' Observatory (http://observatorio.wwf.org.br/). Species names were standardized using the Plantminer web tool (www.plantminer.com) (Carvalho et al. 2010), based on Flora do Brasil (www.floradobrasil.jbrj.gov.br) (Flora_do_Brasil_2020) and The Plant List (www.theplantlist.org/). Complementary queries were performed on The Missouri Botanical Gardens (www.tropicos.org). According to these databases, we excluded any exotic and unidentified species from final compilation.

Environmental data

We compiled 22 environmental predictors with spatial resolution of 1km² and summarized them by using a Principal Component Analysis (PCA) considering 1,000 randomly distributed points within Sao Paulo state. Pairs of variables with scores > I1I (absolute value) were verified to avoid multicollinearity

(correlation <0.7) finally selecting 6 variables with the highest PCA scores (**Table 1**).

Table 1: Environmental layers used in the Principal Component Analysis (PCA) and (*) selected for modelling

Description	**Short name**		**Source**
Slope (declividade)	declividade_br		Ambdata
Height above nearest drain (HAND 50)	hand50_br		Ambdata
Altitude	altitude_br		Ambdata
Depth to bedrock (R horizon) up to 200 cm	Depth		SoilGrids
Soil organic carbon content (fine earth fraction) g/ kg	Carbon_cnt	*	SoilGrids
Clay content (0–2 µm) mass fraction (%)	Clay		SoilGrids
Silt content (2–50 µm) mass fraction (%)	Silt	*	SoilGrids
Sand content (50–2000 µm) mass fraction (%)	Sand		SoilGrids
Cation exchange capacity of soil in cmolc/kg	Cation		SoilGrids
Soil organic carbon stock in tonnes per ha	Carbon_stc		SoilGrids
Soil pH x 10 in H2O	pH_H2O	*	SoilGrids
Soil pH x 10 in KCl	pH_KCl		SoilGrids
Aridity Index	AI		CGIAR CSI
Actual Evapotranspiration	AET		CGIAR CSI
Potential Evapotranspiration	PET		CGIAR CSI
Precipitation (mm)	precip	*	WorldClim v.2
Solar radiation (kJ .m^{-2} .day^{-1})	srad	*	WorldClim v.2
Average temperature (°C)	tavg	*	WorldClim v.2
Maximum temperature (°C)	tmax		WorldClim v.2
Minimum temperature (°C)	tmin		WorldClim v.2
Water vapor pressure (kPa)	vapr		WorldClim v.2
Wind speed (m .s^{-1})	wind		WorldClim v.2

Environmental Niche Modeling and Species Distribution Modeling

Environmental Niche Models (ENM) are statistical models that relate focal species occurrence to associated environmental conditions, generating correlative rules that allow extrapolation and prediction of occupancy patterns over wide geographic extents, representing a valuable tool for conservationist purposes (De Marco Junior & Siqueira 2009; Paglia et al. 2012; Angelieri et al. 2016; Gavish et al. 2017; Guisan et al. 2013). We applied a Species Distribution Model (SDM) approach by restricting ENM to accessible areas, aiming to presume the distribution range of species if they were not affected by habitat loss, fragmentation and disturbance. In other words, if species' distribution were primarily defined by abiotic conditions (*i.e.,* environmental niche), in the lack of constraints imposed by altered habitat and landscape structure (*e.g.,* fragmentation and patch isolation) (Peterson & Soberon 2012). As ENM and SDM can be based on the same sets of mathematical algorithms, occurrence data and environmental variables (Peterson & Soberon 2012), they are near-synonymous: the main difference is that SDM implies on some sort of restriction over ENM, which will be further detailed; for the purpose of this study, we hereafter will refer to our modeling approach only as SDM.

We built the SDM for each species using the Model-R framework (Sánchez-Tapia et al. 2018) with a three-fold cross validation procedure, meaning that two partitions were used for parameter estimation and algorithm training, and one to evaluate the model's accuracy. Random pseudo-absence points (n_{back} = 1000) were sorted within a mean distance buffer, where the radius of the buffer was the mean geographic distance between the occurrence points. If one species' records were less than 20 km apart, they were rarefied to reduce effects of sampling bias and avoid modelling overfitting (Elith et al. 2006; Zwiener et al. 2017).

In the Model-R framework, for each partition and algorithm a model was built and its performance was tested by their True Skill Statistics (TSS) (Allouche et al. 2006). We previously tested several algorithms - BioClim, GLM, SVM, Random Forest, MaxEnt – and selected the last two based on their overall performance (**APPENDIX 1**), which was consonant to the results found by Diniz-

filho et al. (2009). We then selected Random Forest and MaxEnt partitions with TSS>0.4, and applied a threshold that maximizes two error types: sensitivity (*i.e.* true presences) and specifity (*i.e.* true absences) (Sánchez-Tapia et al. 2018). The resulting binary models were averaged into a final model for each algorithm, an then combined into a final ensemble model with an average threshold that maximizes TSS values (Sánchez-Tapia et al. 2018), resulting in a final map indicating areas of probable presence.

Species richness and community composition

Our analyses were based on two different type of information: the observed species richness and community composition in each site and the species richness and community composition predicted by SDM. The observed metrics were adjusted to consider only modelled species, that is, species occurring in more than ten localities, at least 20km apart and with a final ensemble model derived from Random Forest and Maxent algorithms, using their partitions with TSS>0.4 (**Figure 2a**), as previously detailed. This adjustment was necessary in order to make proper comparisons between observed and predicted richness and compositions, and considered a final sub-set of 663 woody species.

Species richness based on SDM can be predicted either by stacking individual species-level models (Stacking SDM, S-SDM) or by modelling α-diversity itself (Macroecological Models, MEM) (Gavish et al. 2017; Calabrese et al. 2014; D'Amen et al. 2015; Guisan & Rahbek 2011). Since stacking binary presence/absence SDM tend to overpredict richness, as it does not account for biotic interactions or filters (Gavish et al. 2017; Guisan & Rahbek 2011), Calabrese *et al.* (2014) proposed S-SDM corrections to reduce these overpredictions and concluded that if stacked correctly, S-SDM are no worse than MEM. According to their findings, a corrected S-SDM approach involves summing-up the raw predicted suitabilities for each locality instead of summing-up their binary values (Calabrese et al. 2014; D'Amen et al. 2015) (**Figure 2b**). Hence, we inputted an average of the raw suitability values into the areas of probable presence defined on the final ensemble models; the predicted

community species richness is the sum of these values considering any given location (i.e. pixel).

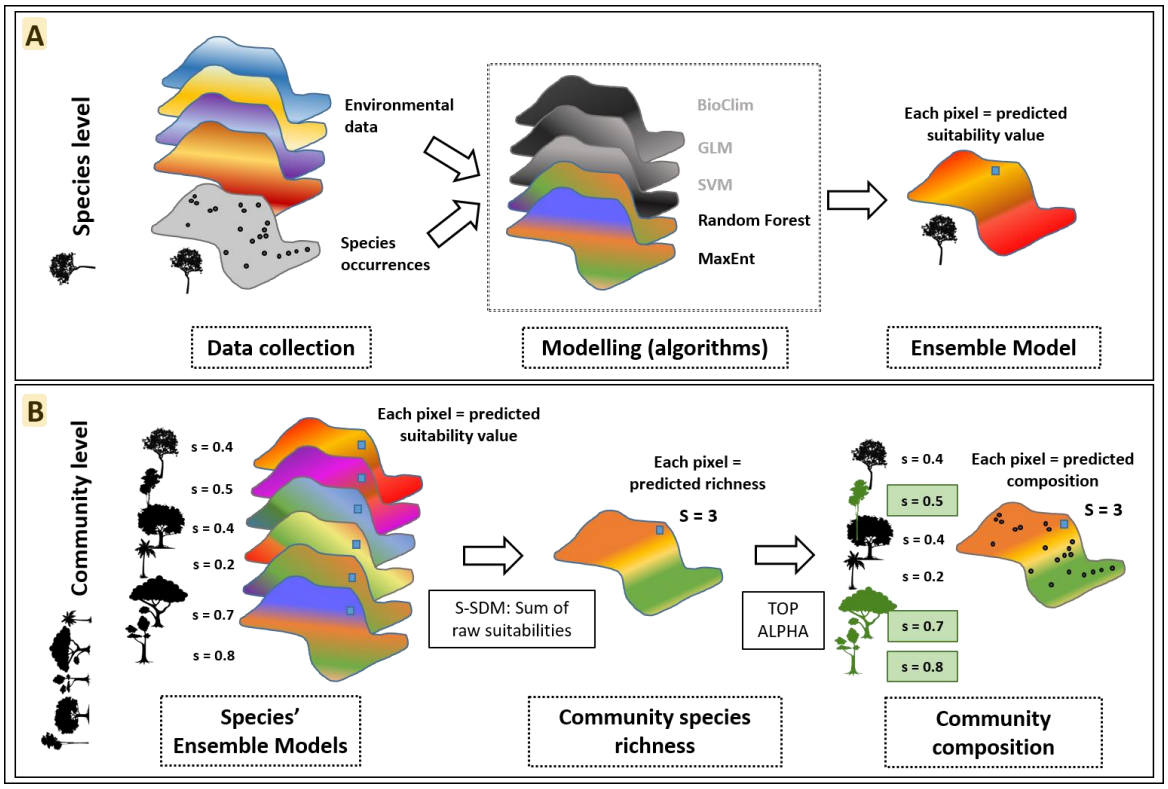


Figure 2: Steps for species distribution modeling and for predicting community richness and composition at a given point (i.e. pixel). (A) Step 1: Species distribution modeling for each of the 663 species, considering Random Forest and MaxEnt algorithms and their partitions with True Skill Statistics (TSS) ≥0.4. The final ensemble model indicate, for each pixel, the suitability of occurrence for one given species. (B) Step 2: The predicted community species richness at a given point results from the sum of the raw probabilities of occurrence (*i.e.,* suitabilities not converted to binary values by any threshold) of targeted species. Step 3: The community composition results from the TOP ALPHA approach, which consists on ranking the species according to their suitabilities of occurrences and selecting species with the highest values until attaining the predicted richness (step 2).

Following the estimation of potential richness, we predicted site-level composition by adopting the "top alpha" approach (Gavish et al. 2017), where we ranked the species' suitabilities of occurrence per site from the highest to the lowest values – based on their individual ensemble SDM – and then selected the top number of species that equals to the predicted potential richness per site (D'Amen et al. 2015; Gavish et al. 2017) (**Figure 2b**).

β-diversity analyses

The compositional variation among communities from site-to-site (β-diversity) relates local diversity (α) to the regional species pool (γ) (Anderson et al. 2010). When evaluating the effects of habitat loss and fragmentation, changes on the organization of biodiversity over space and time can reveal if biological homogenization or heterogenization is taking place, an essential information to guide conservation planning over regional diversity (Socolar et al. 2016; Arroyo-Rodríguez et al. 2013; Püttker et al. 2015). Despite the valuable contribution from evaluating β-diversity, its interpretation must be very cautious, as there are several ways to measure and compare it (Koleff et al. 2003; Baselga et al. 2007; Jost 2007; Chao et al. 2012; Jost et al. 2010; Anderson et al. 2006, 2010; Tuomisto 2010).

There is extensive debate regarding the interrelationships among α, β and γ-diversity, in addition to measures for partitioning it (*i.e.* multiplicative or additive) and statistical approaches to properly analyze β-diversity (Anderson et al. 2010). A particular concern for our study is the fact that a great variety of metrics to estimate β-diversity depend on α and γ diversity – and therefore on scale and sample size. Considering that our samples represent a compilation of floristic surveys using distinct methods and sampling effort, we decided to use a β-diversity metric that weights on composition dissimilarities more than on richness differences (Koleff et al. 2003). For that matter, we calculated pairwise Sorensen ($\boldsymbol{\beta_{SOR}}$) and Simpson indices ($\boldsymbol{\beta_{SIM}}$) among sites, which indicate the overall variation on the species composition between pairs of sites ($\boldsymbol{\beta_{SOR}}$) and the variation related to its turnover component ($\boldsymbol{\beta_{SIM}}$), reflecting the replacement of species (Baselga 2010; Baselga et al. 2015; Socolar et al. 2016).

To evaluate differences between observed and predicted β-diversity, we: (i) ran a Principal Coordinate Analysis (PCoA) based on the Simpson ($\boldsymbol{\beta_{SIM}}$) observed dissimilarity matrix (as the turnover component was the main contribution to total beta diversity; see bellow); (ii) estimated the distance of each site (forest fragment) to the group centroid in the multivariate ordination space generated by the PCoA ($\boldsymbol{\beta_d}$, **Figure 3**); (iii) repeated steps (i) and (ii) using the SDM predicted species composition; and (iv) compared mean observed $\boldsymbol{\beta_d}$ with

mean predicted β_d with permutational paired t-test. In this test, observed β_d values were paired with predicted β_d ones. β_d is analogous to the local contribution to beta diversity (LCBD) proposed by Legendre and De Cáceres (2013), since higher values of β_d represent higher distinctiveness of one site (forest fragment) within a group. This metric with standard effect size allowed us to test the null hypothesis that beta diversity does not differ among observed and predicted communities.

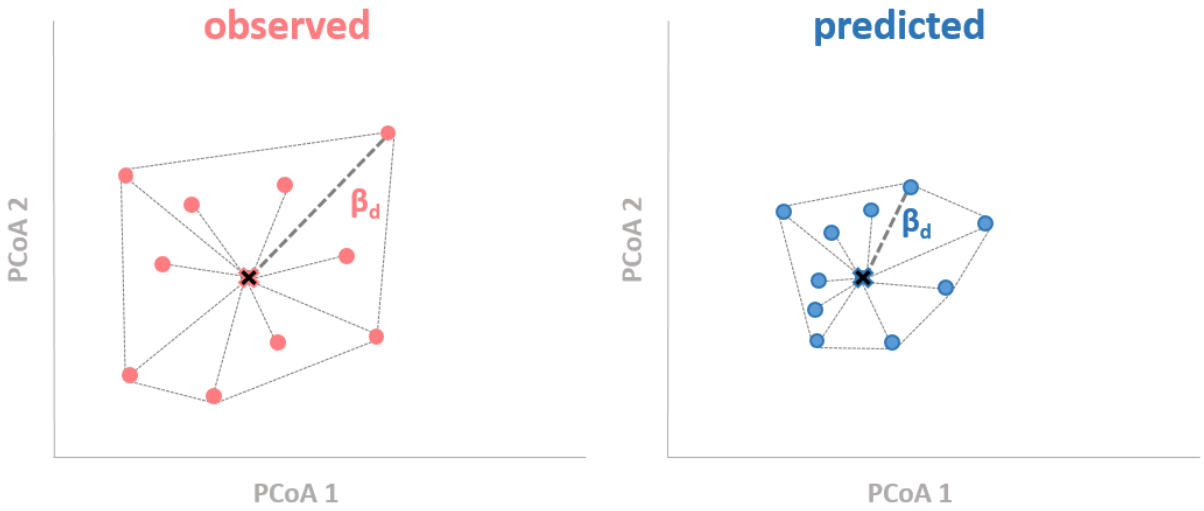


Figure 3: Hypothetical example of a multivariate ordination space describing the distance (β_d) of a unit to the group centroid (central cross), considering the observed (red) and the predicted (blue) species composition.

RESULTS

For the 663 woody species considered in this study, overall comparisons revealed that predicted richness at the site level (α-diversity) was 3.8 times higher than observed richness. This ratio was much lower when considering only strictly Protected Areas, ranging around 1.0. In fact, the few forest fragments that presented increased observed species richness in relation to predicted richness were mostly represented by Protected Areas (**Figure 4**) (**APPENDICES 2 and 3**).

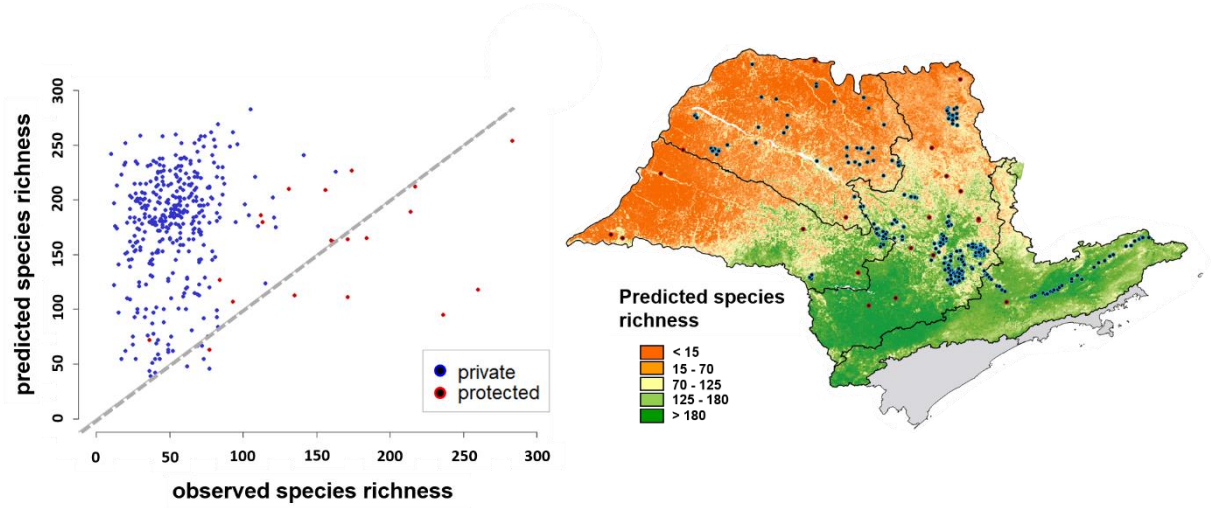


Figure 4: (a) Scatterplot relating the predicted and observed species richness at the site level, considering both protection types (strictly Protected Areas and Private lands). Dots below the line represent forest fragments with higher observed richness than predicted. **(b)** Predicted species richness map, calculated by summing the raw suitabilities of occurrence for each of the 663 woody species of this study regions.

Partition of total beta diversity (β_{SOR}) indicated a consistent higher contribution of turnover (β_{SIM}) to overall dissimilarity within regions and protection categories (**Table 2**) (**APPENDIX 4**).

Table 2: Mean ± standard deviation of total beta diversity (Bsor) and its components turnover (Bsim) and nestedness (Bnes) for distinct regions and categories, considering the observed (red) and the predicted (blue) species composition.

		Bsor	Bsim	Bnes	Bsor	Bsim	Bnes
Regions	Center	0.73 ± 0.12	0.63 ± 0.17	0.10 ± 0.11	0.35 ± 0.21	0.27 ± 0.19	0.08 ± 0.07
	Southeast	0.74 ± 0.09	0.68 ± 0.11	0.05 ± 0.06	0.48 ± 0.16	0.43 ± 0.16	0.04 ± 0.03
	West	0.76 ± 0.12	0.65 ± 0.16	0.11 ± 0.11	0.48 ± 0.16	0.32 ± 0.16	0.16 ± 0.11
Protection categories	Protected	0.61 ± 0.15	0.48 ± 0.16	0.13 ± 0.11	0.58 ± 0.17	0.48 ± 0.18	0.11 ± 0.08
	Private	0.76 ± 0.12	0.69 ± 0.15	0.07 ± 0.08	0.46 ± 0.23	0.36 ± 0.21	0.10 ± 0.09

We found that sites with lower predicted than observed β_{SOR} (ratio < 1) also had higher predicted than observed species richness (ratio > 1) (**Figure 5**), which indicates a correlation between local species loss and heterogenization.

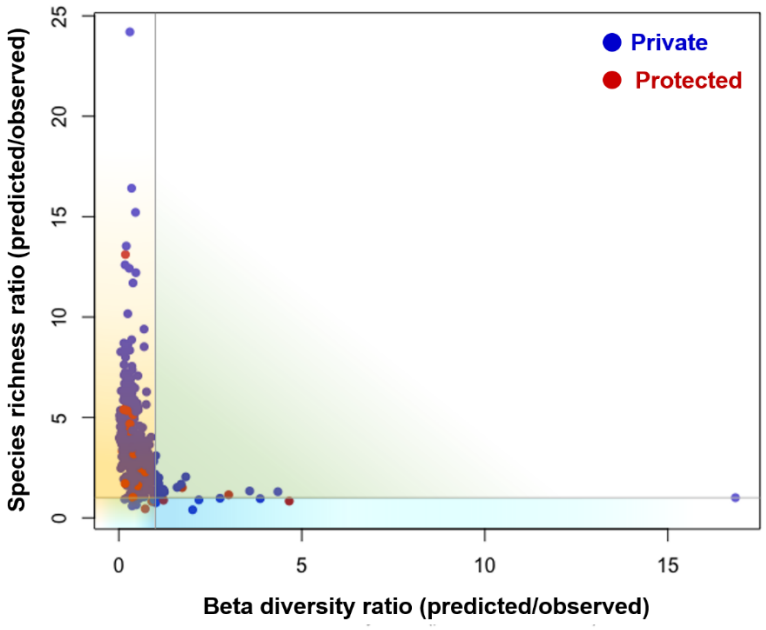


Figure 5: Scatterplot relating the predicted/observed species richness ratio (vertical axis) and total beta diversity (β_{SOR}) ratio (horizontal axis). Sites (dots) above vertical value=1 and below horizontal value=1 (yellow quadrant) experienced local species loss and heterogenization (higher observed β_{SOR}) in comparison to predictions. Green quadrant represent sites that experienced local species loss and homogenization (lower observed β_{SOR}).

The mean observed distance to centroid (β_d) was significantly higher than the predicted β_d for all regions and for private lands, with strictly protected areas being the only exception (**Table 3**; **Figure 6**).

Table 3: Minimum, mean ± standard deviation and maximum values of distance to centroid (β_d) for distinct regions and categories, considering the observed (red) and the predicted (blue) species composition.

		Min.	Mean ± SD	Max.	Min.	Mean ± SD	Max.	*t*-value	P
	Center	0.03	0.45 ± 0.12	0.79	0.01	0.18 ± 0.15	0.72	26.18	<0.0001
Regions	**Southeast**	0.08	0.47 ± 0.09	0.63	0.16	0.32 ± 0.08	0.52	9.8	<0.0001
	West	0.05	0.46 ± 0.10	0.65	0.02	0.22 ± 0.13	0.69	12.28	<0.0001
Protection	**Protected**	0.16	0.37 ± 0.16	0.64	0.13	0.44 ± 0.19	0.75	0.29	0.75
categories	**Private**	0.02	0.39 ± 0.15	0.78	0.01	0.34 ± 0.20	0.73	31.79	<0.0001

Individual β_d values varied within regions and protection categories from almost zero to almost 0.8, especially in the central region (**Figure 6a**). Despite this variation, most observed-predicted pairs of sites showed an increase in β_d from predicted to observed values (**Figure 6b**), indicating a clear trend for biotic heterogenization in the different regions. After splitting the data into

protection categories (all regions pooled), we found that the higher beta-diversity in the observed data was due to an increase in β_d values in private lands, as there was no difference between observed and predicted values in strictly protected areas. These results were confirmed by the paired t-test (**Table 3**).

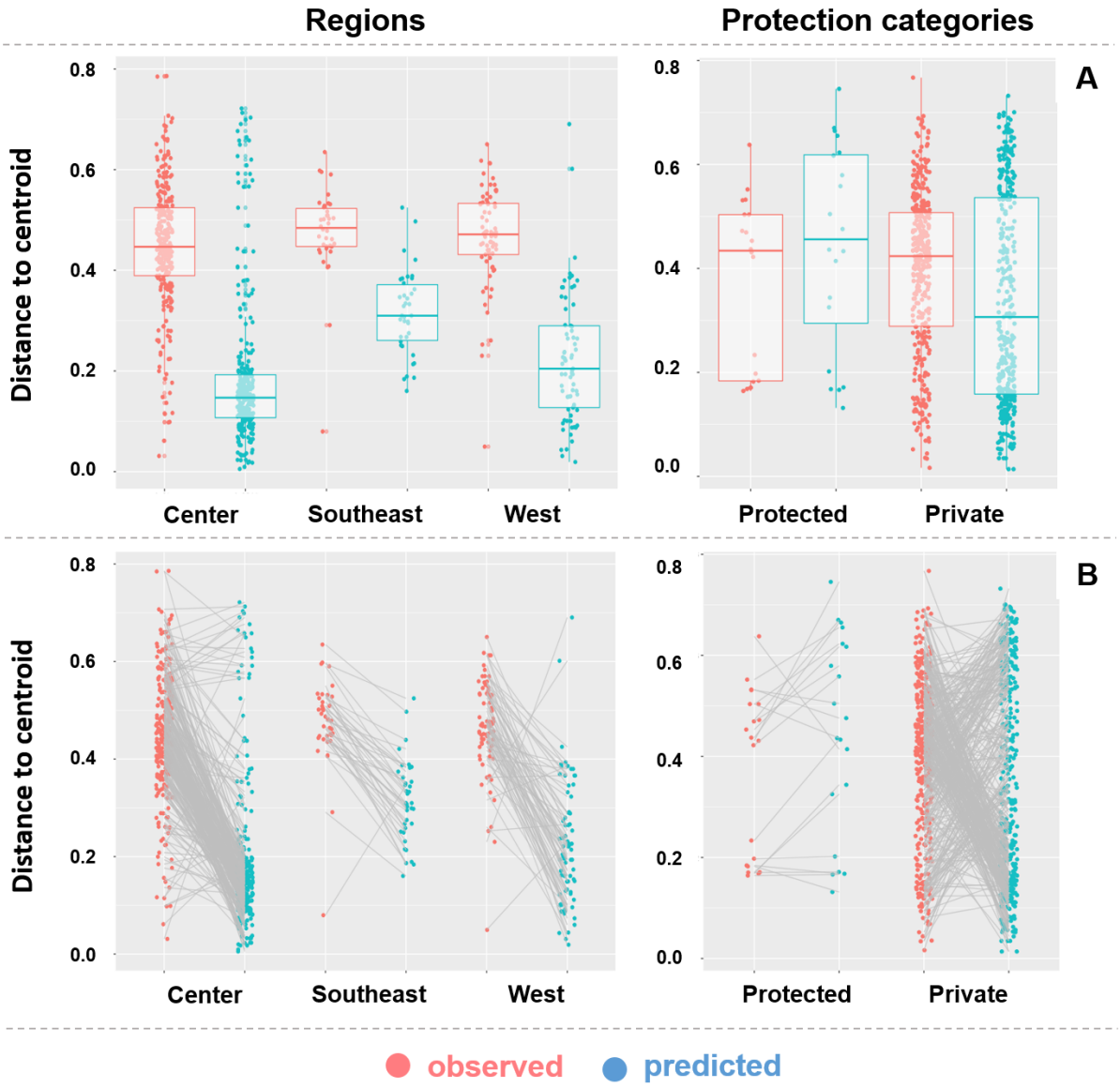


Figure 6: **(a)** Boxplots representing individual distances to centroid (**βd** values) considering the observed (red) and predicted (blue) datasets of different regions and protection categories. Each point represents the distance from one site (forest fragment) to the group centroid in in a βsim-based ordination space. **(b)** Grey lines connect pairs of observed and predicted **βd** values.

DISCUSSION

We detected a general loss on observed local diversity in relation to our modeled predictions and this was particularly true among forest fragments in private lands, where we registered consistent reductions in species richness. The higher β diversity registered for the observed dataset imply an overall biotic heterogenization. However, our study also exposes the complexity of this process, with evidence indicating that both homogenization (positive observed-predicted pairwise slopes) and heterogenization (negative observed-predicted pairwise slopes) are taking place in these hyper-fragmented landscapes. Even though distinct regions of São Paulo state were gradually occupied and converted along time – from the eastern coast towards western countryside – all of them presented the same pattern: observed and predicted beta diversity were significantly different, with lower mean values for the latter. This is probably because of the results found among private lands, which represent almost 95% of our samples and drove the pattern registered for all state regions (*i.e.,* heterogenization). As an exception, strictly Protected Areas had lower local species' loss with no significant differences between observed and predicted β diversity, suggesting they may be fulfilling, to some extent, their protection purpose.

Biodiversity changes in tropical ecosystems are extremely complex to evaluate and understand, as they are scale and context dependent, differ among taxonomic groups and ecosystems, and often respond differently to similar environmental changes (Vellend et al. 2013; Dornelas et al. 2014; Newbold et al. 2015; McGill 2015; McGill et al. 2015; Boesing et al. 2018; Catano et al. 2017; Magurran et al. 2018). For instance, recent studies found no evidence for systematic loss in local diversity (Vellend et al. 2013; Dornelas et al. 2014), while several others indicate this is not true for the tropics, where a variety of taxa experienced steep local species decreases in human modified landscapes (Haddad et al. 2015; Mendenhall et al. 2016; Beca et al. 2017; Ceballos et al. 2017; Farah et al. 2017; Galetti et al. 2017; Barlow et al. 2018; Bovendorp et al. 2018). Whereas the unprecedented level of forest degradation, fragmentation and intensive land use have an undeniable contribution to immediate and long-term local diversity loss (Haddad et al. 2015; Barlow et al. 2018), less is known

regarding how these disturbances modify and drive the community composition along time and space, regardless of the recent growing interest and evidence on these compositional shifts (Dornelas et al. 2014; Haddad et al. 2015; McGill et al. 2015; Collins et al. 2017; Olden et al. 2018).

The comparison between observed and predicted species composition revealed the idiosyncratic responses of β-diversity (i.e. homogenization and heterogenization) in agricultural landscapes, which can be explained by several mechanisms suggested by the literature. Studies that observed biotic homogenization associate β diversity reduction to niche-selection processes, suggesting ecological filtering overrides environmental heterogeneity (Vellend et al. 2007; Lôbo et al. 2011; Marcelo Tabarelli et al. 2012; Arroyo-Rodríguez et al. 2013; Morante-Filho, Arroyo-Rodríguez, et al. 2015; Püttker et al. 2015; Zwiener et al. 2017). This statement assumes non-random local species extinctions occur because habitat fragmentation affects species differently, according to traits such as rarity, life span, dispersal, and reproductive mode (Haddad et al. 2015), supporting the proliferation of widespread, short-lived and small-seeded species (*e.g.* pioneer species, generalists or “winners” as defined by Tabarelli et al. 2012a) in detriment of rare and shade-tolerant species (*e.g.* specialists or “losers”) (Lôbo et al. 2011; Marcelo Tabarelli et al. 2012; Arroyo-Rodríguez et al. 2013; Morante-Filho, Arroyo-Rodríguez, et al. 2015; Zwiener et al. 2017). Additionally, non-random plant extinctions may also be related to selective logging, which overharvest valuable hardwood species, and to the disappearance of large and medium frugivores through overhunting and habitat loss, with cascading effects over plant-frugivore interactions, species persistence, ecosystem services and functioning in human-modified landscapes (Bello et al. 2015; Bovendorp et al. 2018). All of these mechanistic explanations may be related to the homogenization registered in our study region, where forest fragments are usually small and therefore exposed to edge effects, with depleted plant-animal interactions – especially large-sized species (Beca et al. 2017, Emer et al. 2018), and subject to recurrent fire and other disturbances (Farah et al. 2017).

Overall, however, our results showed that biotic heterogenization is the predominant process in our study region, accordingly to studies that found

compositional shifts heading towards divergent communities (Smart et al. 2006; Dornelas et al. 2014; Solar et al. 2015; Sfair et al. 2016; Collins et al. 2017). In fact, a meta-analysis carried by Catano *et al.* (2017) found 21 cases of heterogenization among 22 studies evaluating herbaceous plants in disturbed/undisturbed grasslands and savannas. Assuming the long history of forest degradation and fragmentation in our study region as the main driver acting upon forest fragments, we refer to some mechanisms that may explain why the studied communities were more heterogeneous than compared to our modeled predictions. First, there is a combination of long-term disturbances that impose constant selection pressures (*e.g.,* edge effects) with occasional and contingent perturbations (*e.g.,* fires, windstorms etc.), resulting in unique disturbance histories and shifts in the physical environment (*e.g.* microhabitat conditions), that most likely enhance pre-disturbance compositional differences (Haddad et al. 2015; Catano et al. 2017). Second, forest fragments in private lands are more susceptible to disturbances due to the lack of formal and effective protection, proven by their altogether smaller patch sizes (Ribeiro et al. 2009). They also represent a greater variety of conditions that range from mature forests experiencing post-fragmentation changes to regenerating secondary forests (Laurance et al. 2014; Malhi et al. 2014; Farah et al. 2017). In common, they share reduced local species richness, which together with a large regional species pool (*i.e.,* γ diversity) may create a sampling effect; *i.e.*, a higher probability of more distinct composition between sites when a small portion of the species pool (*i.e.,* low α diversity) is expected to occur in any random community, inflating β diversity (Karp et al. 2012; Newbold et al. 2015). The third explanation is particularly relevant in the studied hyper-fragmented landscapes, where dispersal limitation due to patch isolation might play a dominant role in making those communities such heterogeneous. This is supported by the strong positive relation between local species richness and seed arrival in plant communities (Myers & Harms 2009) and because dispersal limitation play a stronger role in determining community assembly in tropical forests (Myers et al. 2013). More specifically, Catano's *et al*. (2017) findings on how disturbance and dispersal interact and alter community composition support that increased β-diversity in disturbed landscapes occurs when dispersal is limited, challenging the hypothesis that disturbances always homogenizes communities compositions

through deterministic environmental filtering, that is, selecting those species best able to survive within HMLs (Vellend et al. 2007; Lôbo et al. 2011; Arroyo-Rodríguez et al. 2013; Püttker et al. 2015). Finally, other plausible mechanisms acting upon these forest fragments may be related to the reduction in the number of individuals and thus in community size, turning them more susceptible to ecological drift and other stochastic forces (Orrock & Watling 2010), and to competitive release arising from the removal of dominant species (Catano et al. 2017).

Given that, by definition, proper evaluation of biotic homogenization or heterogenization processes depend on quantifying changes in β diversity through space and time (McKinney & Lockwood 1999; Olden & Rooney 2006; Olden et al. 2018), the use of Species Distribution Model proved valuable for predicting community composition in the absence of habitat loss and fragmentation, serving as a temporal surrogate in our study. However, we must acknowledge that our modeling approach imposed some restrictions, notably the non-inclusion of rare or poorly-sampled species and biotic interactions. That said, we do not expect that the overall trend registered here - biotic heterogenization - would be affected by the absence of rare species because their inclusion would most likely increase the differences among communities, while biotic interactions were addressed by choosing a method to adjust or at least reduce an overprediction bias related to the lack of biotic interactions (Calabrese et al. 2014; Gavish et al. 2017; Guisan & Rahbek 2011; D'Amen et al. 2017). Another caveat is that the reduced local species richness among private lands may be related, to some extent, to sampling effort. Since the floristic assessments that compose most of the dataset used here aimed to quickly characterize the regional flora for restoration purposes (Rodrigues et al. 2011), we applied preliminary analysis of incidence-based estimated richness (e.g., Chao 2, Jacknife 1 and Jacknife 2 (Magurran 2013)) that indicated satisfactory sampling effort for both private lands and strictly protected areas. Furthermore, we chose β diversity metrics that focused on compositional changes to alleviate the contribution of α diversity and eventual uneven sampling effort (Koleff et al. 2003). With those considerations, we are confident that our results are consistent and would not be much different from what we have shown here.

Our study highlights the complexity and idiosyncrasies of community compositional shifts in hyper-fragmented landscapes, where both homogenization and heterogenization processes were detected, with the latter prevailing as an overall trend, especially in non-protected private lands. From an applied perspective, the implication of biotic homogenization or heterogenization alone is not sufficient to underpin conservation strategies, as its interpretation is not straightforward – human disturbances can cause β diversity to increase, decrease or remain unchanged (Socolar et al. 2016; Olden et al. 2018). However, the heterogenization process in our study is coupled with a scenario where (i) the originally diverse vegetation is now extremely reduced and fragmented, with small and isolated patches distributed within an intensive agricultural matrix; (ii) forest fragments, especially in private lands, represent unique disturbance histories that result in varying quality habitats, and often in reduced local diversity derived from a large regional species pool (γ diversity); (iii) community composition accumulate great variation among patches (high β diversity), predominantly from turnover (*i.e.* replacement of species). Bringing these facts together and recognizing that conservation of biodiversity extends far beyond the boundaries of strictly Protected Areas, we advocate that all forest fragments are valuable for conservation in HMLs, with particular relevance for private lands, which represent the most exposed and neglected share of what is left (Gardner et al. 2009; Mendenhall et al. 2016; Farah et al. 2017). Based on our results and supported by many other studies, we understand there is enough information to develop an evidence-based approach that should be considered in future management and conservation plans. To foster and sustain biodiversity conservation in HMLs, we thus recommend: (i) effective protection of strictly Protected Areas, which usually represent the largest regional core areas (Joppa et al. 2008) and where compositional shifts apparently are more stable; (ii) active restoration of forest fragments to enhance their alpha diversity, through the management of hyper abundant species (*e.g.,* lianas) (César et al. 2016; Estrada-Villegas & Schnitzer 2018) and reintroduction of lacking groups of species (Garcia et al. 2014; Viani et al. 2015) (iii) active restoration of corridors where the vegetation is degraded and natural regeneration is unlikely, aiming to enhance forest cover and connectivity among forest fragments, allowing species to disperse and persist (Howe 2014; Emer et al. 2018). Finally, considering the

growing development of more sustainable agricultural practices (Ferreira et al. 2012, Gonthier et al. 2014) and alternatives for ecological restoration with profitable purposes (Pedro H. S. Brancalion et al. 2012), we encourage the establishment of policies that foster a feasible production model, aligned with the conservation of the remaining biodiversity.

ACKNOWLEDGEMENTS: This study was financed in part by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior – Brasil (CAPES) – Finance Code 001, by the National Council for Scientific and Technological Development (CNPq grant 870360/1997-3) and by The São Paulo Research Foundation (FAPESP grant 2013/50718-5).

REFERENCES

Allouche O, Tsoar A, Kadmon R (2006) Assessing the accuracy of species distribution models: Prevalence, kappa and the true skill statistic (TSS). J Appl Ecol 43:1223–1232. doi: 10.1111/j.1365-2664.2006.01214.x

Andam KS, Ferraro PJ, Pfaff A, et al (2008) Measuring the effectiveness of protected area networks in reducing deforestation. Proc Natl Acad Sci U S A 105:16089–16094. doi: 10.1073/pnas.0800437105

Anderson MJ, Crist TO, Chase JM, et al (2010) Navigating the multiple meanings of β diversity: A roadmap for the practicing ecologist. Ecol Lett 14:19–28. doi: 10.1111/j.1461-0248.2010.01552.x

Anderson MJ, Ellingsen KE, McArdle BH (2006) Multivariate dispersion as a measure of beta diversity. Ecol Lett 9:683–693. doi: 10.1111/j.1461-0248.2006.00926.x

Angelieri CCS, Adams-Hosking C, Ferraz KMPM de B, et al (2016) Using Species Distribution Models to Predict Potential Landscape Restoration Effects on Puma Conservation. PLoS One 11:1–18. doi: 10.1371/journal.pone.0145232

Arroyo-Rodriguez V, Melo FPL, Martinez-Ramos M, et al (2015) Multiple successional pathways in human-modified tropical landscapes: New insights from forest succession, forest fragmentation and landscape ecology research. Biol Rev. doi: 10.1111/brv.12231

Arroyo-Rodríguez V, Pineda E, Escobar F, Benítez-Malvido J (2008) Value of small patches in the conservation of plant-species diversity in highly fragmented rainforest. Conserv Biol 23:729–39. doi: 10.1111/j.1523-1739.2008.01120.x

Arroyo-Rodríguez V, Rös M, Escobar F, et al (2013) Plant β-diversity in fragmented rain forests: Testing floristic homogenization and differentiation hypotheses. J Ecol 101:1449–1458. doi: 10.1111/1365-2745.12153

Barlow J, França F, Gardner T, et al (2018) The future of hyperdiverse tropical ecosystems. Nature. doi: 10.1038/s41586-018-0301-1

Barlow J, Lennox GD, Ferreira J, et al (2016) Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. Nature 1–16. doi: 10.1038/nature18326

Baselga A (2010) Partitioning the turnover and nestedness components of beta diversity. Glob Ecol Biogeogr 19:134–143. doi: 10.1111/j.1466-8238.2009.00490.x

Baselga A, Jiménez-Valverde A, Niccolini G (2007) A multiple-site similarity measure independent of richness. Biol Lett 3:642–645. doi: 10.1098/rsbl.2007.0449

Baselga A, Orme D, Villeger S, et al (2015) betapart: Partitioning beta diversity into turnover and nestedness components. R Packag. Version 1.3 1–26

Beca G, Vancine MH, Carvalho CS, et al (2017) High mammal species turnover in forest patches immersed in biofuel plantations. Biol Conserv. doi: 10.1016/j.biocon.2017.02.033

Bello C, Galetti M, Pizo MA, et al (2015) Defaunation affects carbon storage in tropical forests. Sci Adv 1:1–11. doi: 10.1126/sciadv.1501105

Boesing AL, Nichols E, Metzger JP (2018) Biodiversity extinction thresholds are modulated by matrix type. Ecography (Cop) 1–14. doi: 10.1111/ecog.03365

Bovendorp RS, Brum FT, McCleery RA, et al (2018) Defaunation and fragmentation erode small mammal diversity dimensions in tropical forests. Ecography (Cop). doi: 10.1111/ecog.03504

Brancalion PHS, Viani RAG, Strassburg BBN, Rodrigues RR (2012) Finding the money for tropical forest restoration. Unasylva 63:41–50

Calabrese JM, Certain G, Kraan C, Dormann CF (2014) Stacking species distribution models and adjusting bias by linking them to macroecological models. Glob Ecol Biogeogr 23:99–112. doi: 10.1111/geb.12102

Carneiro MS, Campos CCF, Ramos FN, Dos Santos FAM (2016) Spatial species turnover maintains high diversities in a tree assemblage of a fragmented tropical landscape. Ecosphere 7:1–12. doi: 10.1002/ecs2.1500

Carranza T, Balmford A, Kapos V, Manica A (2014) Protected area effectiveness in reducing conversion in a rapidly vanishing ecosystem: The Brazilian Cerrado. Conserv Lett 7:216–223. doi: 10.1111/conl.12049

Carvalho GH, Cianciaruso MV, Batalha MA (2010) Plantminer: A web tool for checking and gathering plant species taxonomic information. Environ Model Softw 25:815–816. doi: 10.1016/j.envsoft.2009.11.014

Catano CP, Dickson TL, Myers JA (2017) Dispersal and neutral sampling mediate contingent effects of disturbance on plant beta-diversity: a meta-analysis. Ecol Lett 20:347–356. doi: 10.1111/ele.12733

Ceballos G, Ehrlich PR, Dirzo R (2017) Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. Proc Natl Acad Sci 201704949. doi: 10.1073/pnas.1704949114

César RG, Holl KD, Girão VJ, et al (2016) Evaluating climber cutting as a strategy to restore degraded tropical forests. Biol Conserv 201:309–313. doi: 10.1016/j.biocon.2016.07.031

Chao A, Chiu CH, Hsieh TC, Inouye BD (2012) Proposing a resolution to debates on diversity partitioning. Ecology 93:2037–2051. doi: 10.1890/11-1817.1

Chazdon RL, Harvey CA, Komar O, et al (2009a) Beyond Reserves : A Research Agenda for Conserving Biodiversity in Human-modified Tropical Landscapes. Biotropica 41:142–153

Chazdon RL, Peres C a, Dent D, et al (2009b) The potential for species conservation in tropical secondary forests. Conserv Biol 23:1406–17. doi: 10.1111/j.1523-1739.2009.01338.x

Coetzee BWT, Gaston KJ, Chown SL (2014) Local scale comparisons of biodiversity as a test for global protected area ecological performance: A

meta-analysis. PLoS One 9:. doi: 10.1371/journal.pone.0105824

Collins CD, Banks-Leite C, Brudvig LA, et al (2017) Fragmentation affects plant community composition over time. Ecography (Cop) 40:119–130. doi: 10.1111/ecog.02607

D'Amen M, Pradervand J-N, Guisan A (2015) Predicting richness and composition in mountain insect communities at high resolution : a new test of the SESAM framework. Glob Ecol Biogeogr 24:1443–1453. doi: 10.1111/geb.12357

D'Amen MD, Rahbek C, Zimmermann NE, Guisan A (2017) Spatial predictions at the community level : from current approaches to future frameworks. Biol Rev 92:169–187. doi: 10.1111/BRV.12222

De Marco Junior P, Siqueira MF (2009) Como determinar a distribuição potencial de espécies sob uma abordagem conservacionista? Megadiversidade 5:65–76

Dent DH, Joseph Wright S (2009) The future of tropical species in secondary forests: A quantitative review. Biol Conserv 142:2833–2843. doi: 10.1016/j.biocon.2009.05.035

Diniz-filho AF, Bini LM, Rangel TF, et al (2009) Partitioning and mapping uncertainties in ensembles of forecasts of species turnover under climate change. Ecography (Cop) 32:897–906. doi: 10.1111/j.1600-0587.2009.06196.x

Dornelas M, Gotelli NJ, McGill B, et al (2014) Assemblage Time Series Reveal Biodiversity Change but Not Systematic Loss. Science (80-) 344:296–299. doi: 10.1126/science.1248484

Durigan G, Ratter JA (2006) Successional changes in cerrado and cerrado/forest ecotonal vegetation in western São Paulo State, Brazil, 1962-2000. Edinburgh J Bot 63:119–130. doi: 10.1017/S0960428606000357

Durigan G, Siqueira MF, Franco GADC, Ratter JA (2006) Seleção De

Fragmentos Prioritários Para a Criação De Unidades De Conservação Do Cerrado No Estado De São Paulo. Rev do Inst Florest 18:23–37. doi: 10.1007/s13398-014-0173-7.2

Elith J, Graham C, Anderson R, et al (2006) Novel methods improve prediction of species' distributions from occurrence data. Ecography (Cop) 29:129–151. doi: 10.1111/j.2006.0906-7590.04596.x

Emer C, Galetti M, Pizo MA, et al (2018) Seed-dispersal interactions in fragmented landscapes – a metanetwork approach. Ecol Lett 21:484–493. doi: 10.1111/ele.12909

Estrada-Villegas S, Schnitzer SA (2018) A comprehensive synthesis of liana removal experiments in tropical forests. Biotropica 0:1–11. doi: 10.1111/btp.12571

Farah FT, Muylaert R de L, Ribeiro MC, et al (2017) Integrating plant richness in forest patches can rescue overall biodiversity in human-modified landscapes. For Ecol Manage 397:78–88

Flora_do_Brasil_2020 Jardim Botânico do Rio de Janeiro. Available on http://floradobrasil.jbrj.gov.br/

França F, Louzada J, Korasaki V, et al (2016) Do space-for-time assessments underestimate the impacts of logging on tropical biodiversity? An Amazonian case study using dung beetles. J Appl Ecol 53:1098–1105. doi: 10.1111/1365-2664.12657

Galetti M, Brocardo CR, Begotti RA, et al (2017) Defaunation and biomass collapse of mammals in the largest Atlantic forest remnant. Anim Conserv 20:270–281. doi: 10.1111/acv.12311

Galetti M, Giacomini HC, Bueno RS, et al (2009) Priority areas for the conservation of Atlantic forest large mammals. Biol Conserv 142:1229–1241. doi: 10.1016/j.biocon.2009.01.023

Garcia LC, Hobbs RJ, Mäes dos Santos F a., Rodrigues RR (2014) Flower and Fruit Availability along a Forest Restoration Gradient. Biotropica 46:114–

123. doi: 10.1111/btp.12080

Gardner T a., Barlow J, Chazdon R, et al (2009) Prospects for tropical forest biodiversity in a human-modified world. Ecol Lett 12:561–582. doi: 10.1111/j.1461-0248.2009.01294.x

Gardner T a., Barlow J, Sodhi NS, Peres C a. (2010) A multi-region assessment of tropical forest biodiversity in a human-modified world. Biol Conserv 143:2293–2300. doi: 10.1016/j.biocon.2010.05.017

Gavish Y, Marsh CJ, Kuemmerlen M, et al (2017) Accounting for biotic interactions through alpha-diversity constraints in stacked species distribution models. Methods Ecol Evol. doi: 10.1111/2041-210X.12731

Gibson L, Lee TM, Koh LP, et al (2011) Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478:378–81. doi: 10.1038/nature10425

Gray CL, Hill SLL, Newbold T, et al (2016) Local biodiversity is higher inside than outside terrestrial protected areas worldwide. Nat Commun 7:. doi: 10.1038/ncomms12306

Guisan A, Rahbek C (2011) SESAM – a new framework integrating macroecological and species distribution models for predicting spatio-temporal patterns of species assemblages. J Biogeogr 38:1433–1444. doi: 10.1111/j.1365-2699.2011.02550.x

Guisan A, Tingley R, Baumgartner JB, et al (2013) Predicting species distributions for conservation decisions. Ecol Lett 16:1424–1435. doi: 10.1111/ele.12189

Haddad NM, Brudvig LA, Clobert J, et al (2015) Habitat fragmentation and its lasting impact on Earth ’ s ecosystems. Science (80-) 1–9

Hansen MC, Potapov P V, Moore R, et al (2013) High-resolution global maps of 21st-century forest cover change. Science 342:850–3. doi: 10.1126/science.1244693

Howe HF (2014) Diversity Storage: Implications for tropical conservation and

restoration. Glob Ecol Conserv 2:349–358. doi: 10.1016/j.gecco.2014.10.004

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

Joppa LN, Loarie SR, Pimm SL (2008) On the protection of “protected areas.” Proc Natl Acad Sci 105:6673–6678

Jost L (2007) Partitioning diversity into independent alpha and beta components. Ecology 88:2427–2439. doi: 10.1890/06-1736.1

Jost L, Chao A, Chazdon RL (2010) Compositional similarity and B (beta) diversity. In: Biological diversity: frontiers in measurement and assessment. p 368

Karp DS, Rominger AJ, Zook J, et al (2012) Intensive agriculture erodes B-diversity at large scales. Ecol Lett 15:963–970. doi: 10.1111/j.1461-0248.2012.01815.x

Koleff P, Gaston KJ, Lennon JJ (2003) Measuring beta diversity for presence – absence data. J Anim Ecol 72:367–382. doi: 10.1046/j.1365-2656.2003.00710.x

Kurtz BC, Valentin JL, Scarano FR (2015) Are the Neotropical Swamp Forests a Distinguishable Forest Type? Patterns From Southeast and Southern Brazil. Edinburgh J Bot 72:191–208. doi: 10.1017/S096042861400033X

Laurance WF (2009) Conserving the hottest of the hotspots. Biol Conserv 142:1137. doi: 10.1016/j.biocon.2008.10.011

Laurance WF, Sayer J, Cassman KG (2014) Agricultural expansion and its impacts on tropical nature. Trends Ecol Evol 29:107–116. doi: 10.1016/j.tree.2013.12.001

Legendre P, De Cáceres M (2013) Beta diversity as the variance of community data: Dissimilarity coefficients and partitioning. Ecol Lett 16:951–963. doi: 10.1111/ele.12141

Lima RAF de, Mori DP, Pitta G, et al (2015a) How much do we know about the endangered Atlantic Forest? Reviewing nearly 70 years of information on tree community surveys. Biodivers Conserv. doi: 10.1007/s10531-015-0953-1

Lima PB, Lima LF, Santos B a., et al (2015b) Altered herb assemblages in fragments of the Brazilian Atlantic forest. Biol Conserv 191:588–595. doi: 10.1016/j.biocon.2015.08.014

Lôbo D, Leão T, Melo FPL, et al (2011) Forest fragmentation drives Atlantic forest of northeastern Brazil to biotic homogenization. Divers Distrib 17:287–296. doi: 10.1111/j.1472-4642.2010.00739.x

Loyola R (2014) Brazil cannot risk its environmental leadership. Divers Distrib 20:1365–1367. doi: 10.1111/ddi.12252

Machado FS, Fontes MAL, dos Santos RM, et al (2016) Tree diversity of small forest fragments in ecotonal regions: why must these fragments be preserved? Biodivers Conserv 525–537. doi: 10.1007/s10531-016-1063-4

Magnago LFS, Magrach A, Laurance WF, et al (2015) Would protecting tropical forest fragments provide carbon and biodiversity cobenefits under REDD+? Glob Chang Biol 21:3455–3468. doi: 10.1111/gcb.12937

Magurran AE (2013) Measuring biological diversity

Magurran AE, Deacon AE, Moyes F, et al (2018) Divergent biodiversity change within ecosystems. Proc Natl Acad Sci 201712594. doi: 10.1073/pnas.1712594115

Malhi Y, Gardner T a., Goldsmith GR, et al (2014) Tropical Forests in the Anthropocene. Annu Rev Environ Resour 39:125–159. doi: 10.1146/annurev-environ-030713-155141

Martensen AC, Pimentel RG, Metzger JP (2008) Relative effects of fragment size and connectivity on bird community in the Atlantic Rain Forest: Implications for conservation. Biol Conserv 141:2184–2192. doi: 10.1016/j.biocon.2008.06.008

Martinelli LA, Joly CA, Nobre CA, Sparovek G (2010) A falsa dicotomia entre a preservação da vegetação natural e a produção agropecuária. Biota Neotrop 10:323–330. doi: 10.1590/S1676-06032010000400036

McGill B (2015) Land use matters. Nature 520:38–39. doi: 10.1038/520038a

McGill BJ, Dornelas M, Gotelli NJ, Magurran AE (2015) Fifteen forms of biodiversity trend in the anthropocene. Trends Ecol Evol 30:104–113. doi: 10.1016/j.tree.2014.11.006

McKinney ML, Lockwood JL (1999) Biotic homogenization: A few winners replacing many losers in the next mass extinction. Trends Ecol Evol 14:450–453. doi: 10.1016/S0169-5347(99)01679-1

Melo FPL, Arroyo-Rodríguez V, Fahrig L, et al (2013) On the hope for biodiversity-friendly tropical landscapes. Trends Ecol Evol 28:462–8. doi: 10.1016/j.tree.2013.01.001

Mendenhall CD, Shields-Estrada A, Krishnaswami AJ, Daily GC (2016) Quantifying and sustaining biodiversity in tropical agricultural landscapes. Proc Natl Acad Sci 113:14544–14551. doi: 10.1073/pnas.1604981113

Metzger JP (2009) Conservation issues in the Brazilian Atlantic forest. Biol Conserv 142:1138–1140. doi: 10.1016/j.biocon.2008.10.012

Morante-Filho JC, Arroyo-Rodríguez V, Faria D (2015a) Patterns and predictors of beta diversity in the fragmented Brazilian Atlantic forest: A multiscale analysis of forest specialist and generalist birds. J Anim Ecol 85:240–250. doi: 10.1111/1365-2656.12448

Morante-Filho JC, Faria D, Mariano-Neto E, Rhodes J (2015b) Birds in anthropogenic landscapes: The responses of ecological groups to forest loss in the Brazilian Atlantic forest. PLoS One 10:1–18. doi: 10.1371/journal.pone.0128923

Morellato PC, Haddad CFB (2000) Introduction: the Brazilian Atlantic Forest. Biotropica 32:786–792. doi: 10.1111/j.1744-7429.2000.tb00618.x

Murphy GEP, Romanuk TN (2014) A meta-analysis of declines in local species

richness from human disturbances. Ecol Evol 4:91–103. doi: 10.1002/ece3.909

Myers J a., Harms KE (2009) Seed arrival, ecological filters, and plant species richness: A meta-analysis. Ecol Lett 12:1250–1260. doi: 10.1111/j.1461-0248.2009.01373.x

Myers JA, Chase JM, Jiménez I, et al (2013) Beta-diversity in temperate and tropical forests reflects dissimilar mechanisms of community assembly. Ecol Lett 16:151–157. doi: 10.1111/ele.12021

Myers N, Mittermeier RA, Mittermeier CG, et al (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858. doi: 10.1038/35002501

Newbold T, Hudson LN, Hill SLL, et al (2015) Global effects of land use on local terrestrial biodiversity. Nature. doi: 10.1038/nature14324

Norden N, Chazdon RL, Chao A, et al (2009) Resilience of tropical rain forests: Tree community reassembly in secondary forests. Ecol Lett 12:385–394. doi: 10.1111/j.1461-0248.2009.01292.x

Olden JD, Comte L, Giam X (2018) The Homogocene: a research prospectus for the study of biotic homogenisation. NeoBiota 37:23–36. doi: 10.3897/neobiota.37.22552

Olden JD, Rooney TP (2006) On defining and quantifying biotic homogenization. Glob Ecol Biogeogr 15:113–120. doi: 10.1111/j.1466-822X.2006.00214.x

Oliveira-Filho A, Fontes M (2000) Patterns of Floristic Differentiation among Atlantic Forests in Southeastern Brazil and the Influence of Climate1. Biotropica 32:793–810. doi: 10.1111/j.1744-7429.2000.tb00619.x

Orrock JL, Watling JI (2010) Local Community size mediates ecological drift and competition in metacommunities. Proc R Soc B Biol Sci 277:2185–2191. doi: 10.1098/rspb.2009.2344

Paglia a P, de Rezende DT, Koch I, et al (2012) Species distribution models (SDM) in biodiversity conservation strategies and climate change

ecosystem based adaptation [Modelos de Distribuição de Espécies em Estratégias para a Conservação da Biodiversidade e para Adaptação Baseada em Ecossistemas Frent. Nat a Conserv 10:231–234. doi: 10.4322/natcon.2012.031

Pardini R, Bueno ADA, Gardner T a, et al (2010) Beyond the fragmentation threshold hypothesis: regime shifts in biodiversity across fragmented landscapes. PLoS One 5:e13666. doi: 10.1371/journal.pone.0013666

Peterson AT, Soberon J (2012) Species Distribution Modeling and Ecological Niche Modeling: Getting the Concepts Right. Nat Conserv 10:102–107. doi: 10.4322/natcon.2012.019

Püttker T, Bueno A de A, Prado PI, Pardini R (2015) Ecological filtering or random extinction? Beta-diversity patterns and the importance of niche-based and neutral processes following habitat loss. Oikos 124:206–215. doi: 10.1111/oik.01018

Ribeiro MC, Metzger JP, Martensen AC, et al (2009) The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biol Conserv 142:1141–1153. doi: 10.1016/j.biocon.2009.02.021

Rodrigues RR, Gandolfi S, Nave AG, et al (2011) Large-scale ecological restoration of high-diversity tropical forests in SE Brazil. For Ecol Manage 261:1605–1613. doi: 10.1016/j.foreco.2010.07.005

Sánchez-Tapia A, de Siqueira MF, Lima RO, et al (2018) Model-R: A framework for scalable and reproducible ecological niche modeling. Commun Comput Inf Sci 796:218–232. doi: 10.1007/978-3-319-73353-1_15

Santos K, Kinoshita LS, Santos FAM (2007) Tree species composition and similarity in semideciduous forest fragments of southeastern Brazil. Biological Conservation 135(2): 268-277

Setzer J (1966) Atlas climático e ecológico do estado de São Paulo. 61

Sfair JC, Arroyo-Rodriguez V, B. A. S, Tabarelli M (2016) Taxonomic and

functional divergence of tree assemblages in a fragmented tropical forest. Ecol Appl 2:1816–1826. doi: 10.1890/15-1673.1

Silva JMC da, Tabarelli M (2000) Tree species impoverishment and the future flora of the Atlantic forest of northeast Brazil. Nature 404:72–74. doi: 10.1038/35003563

Siqueira T, Lacerda CGT, Saito VS (2015) How Does Landscape Modification Induce Biological Homogenization in Tropical Stream Metacommunities ? Biotropica 0:1–8

Slik JWF, Arroyo-Rodríguez V, Aiba S-I, et al (2015) An estimate of the number of tropical tree species. Proc Natl Acad Sci 112:7472–7477. doi: 10.1073/pnas.1512611112

Smart SM, Thompson K, Marrs RH, et al (2006) Biotic homogenization and changes in species diversity across human-modified ecosystems. Proc Biol Sci 273:2659–2665. doi: 10.1098/rspb.2006.3630

Soares-Filho B, Rajão R, Macedo M, et al (2014) Cracking Brazil ' s Forest Code. Science (80-) 344:363–364

Socolar JB, Gilroy JJ, Kunin WE, Edwards DP (2016) How Should Beta-Diversity Inform Biodiversity Conservation? Trends Ecol Evol 31:67–80. doi: 10.1016/j.tree.2015.11.005

Solar RRDC, Barlow J, Ferreira J, et al (2015) How pervasive is biotic homogenization in human-modified tropical forest landscapes? Ecol Lett n/a-n/a. doi: 10.1111/ele.12494

Sparovek G, Berndes G, Barretto AGDOP, Klug ILF (2012) The revision of the brazilian forest act: Increased deforestation or a historic step towards balancing agricultural development and nature conservation? Environ Sci Policy 16:65–72. doi: 10.1016/j.envsci.2011.10.008

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

10.1016/j.biocon.2010.02.005

Tabarelli M, Peres C a., Melo FPL (2012a) The “few winners and many losers” paradigm revisited: Emerging prospects for tropical forest biodiversity. Biol Conserv 155:136–140. doi: 10.1016/j.biocon.2012.06.020

Tabarelli M, Santos B a., Arroyo-rodríguez V, Melo F (2012b) Secondary forests as biodiversity repositories in human-modified landscapes: insights from the Neotropics. Bol do Mus do Pará Emílio Goeldi 7:319–328

Taubert F, Fischer R, Groeneveld J, et al (2018) Global patterns of tropical forest fragmentation. Nature. doi: 10.1038/nature25508

Tscharntke T, Tylianakis JM, Rand T a., et al (2012) Landscape moderation of biodiversity patterns and processes - eight hypotheses. Biol Rev 87:661–685. doi: 10.1111/j.1469-185X.2011.00216.x

Tuomisto H (2010) A diversity of beta diversities: Straightening up a concept gone awry. Part 1. Defining beta diversity as a function of alpha and gamma diversity. Ecography (Cop) 33:2–22. doi: 10.1111/j.1600-0587.2009.05880.x

Vellend M, Baeten L, Myers-Smith IH, et al (2013) Global meta-analysis reveals no net change in local-scale plant biodiversity over time. Proc Natl Acad Sci U S A 110:19456–9. doi: 10.1073/pnas.1312779110

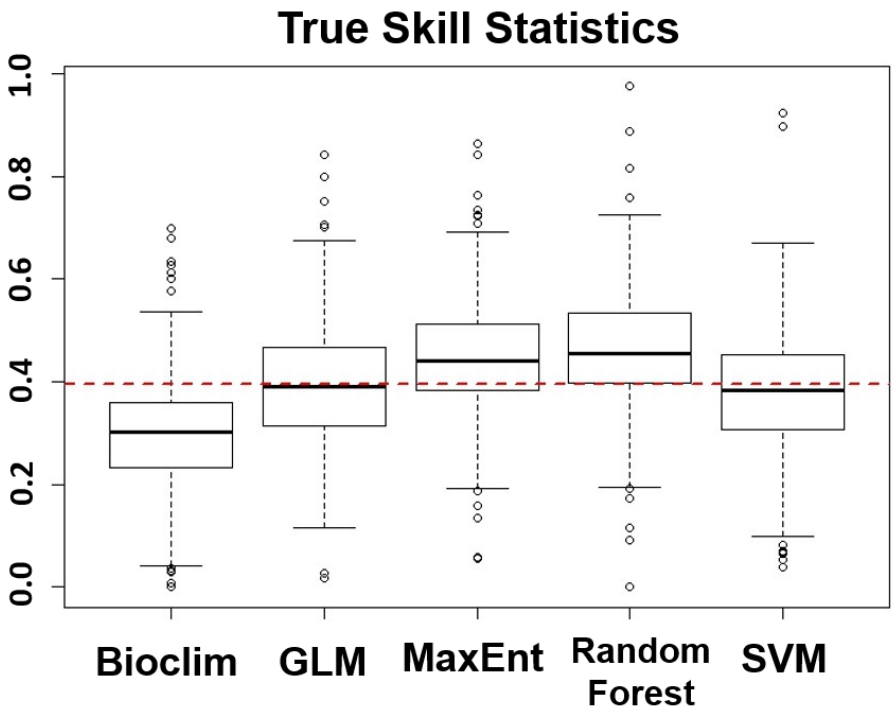
Vellend M, Verheyen K, Flinn KM, et al (2007) Homogenization of forest plant communities and weakening of species – environment relationships via agricultural land use. J Ecol 95:565–573. doi: 10.1111/j.1365-2745.2007.01233.x

Viani RAG, Mello FNA, Chi IE, Brancalion PHS (2015) A new focus for ecological restoration: management of degraded forest remnants in fragmented landscapes. GPL news november:5–9

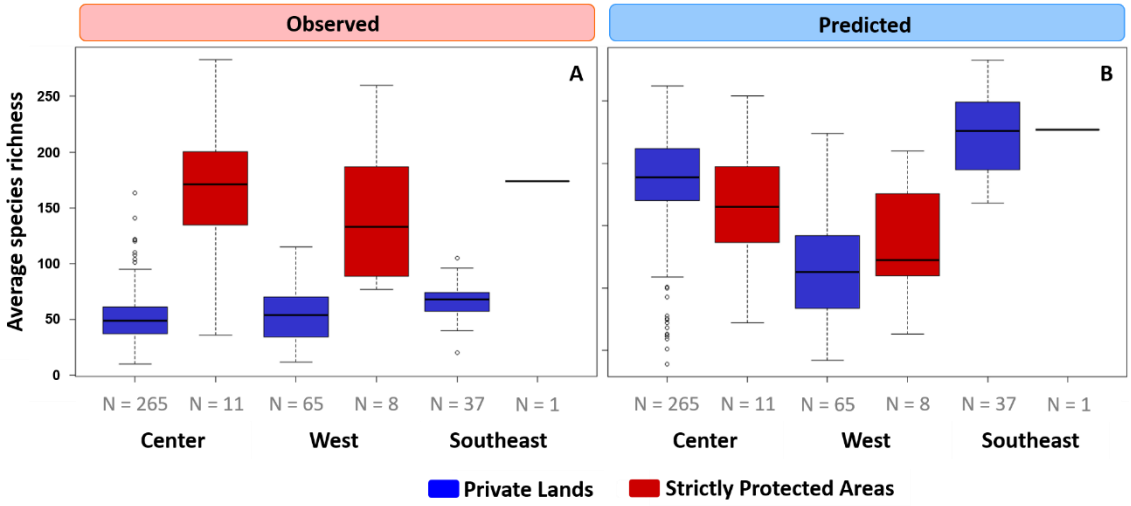
Zwiener VP, Lira-Noriega A, Grady CJ, et al (2017) Climate change as a driver of biotic homogenization of woody plants in the Atlantic Forest. Glob Ecol Biogeogr 1–12. doi: 10.1111/geb.12695

APPENDICES

Appendix 1: Boxplots (median ± quartiles) of the True Skill Statistics (TSS) comparing partitions of previously tested algorithms used to generate the Species Distribution Models. Dashed red line indicates the selection criteria (TSS=0.4) applied to choose algortihms that composed the final ensemble model (MaxEnt and Random Forest).



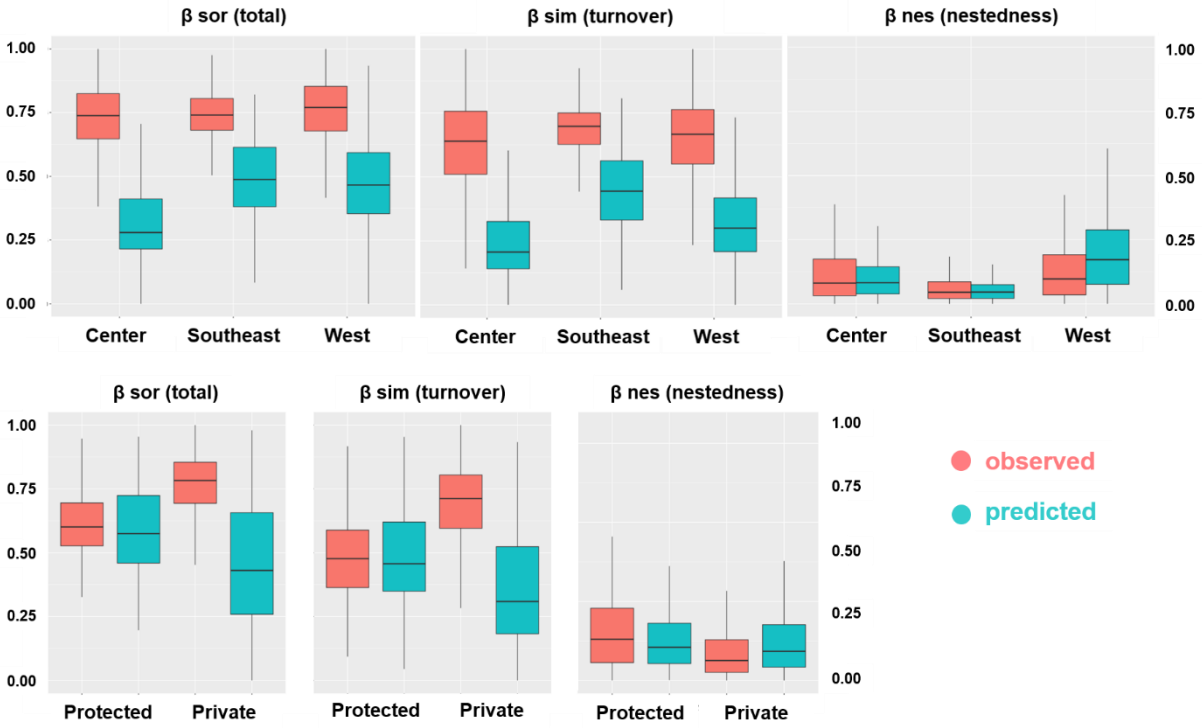
Appendix 2: Boxplots (median ± quartiles) comparing protection categories and regions of São Paulo state considering (a) the observed and (b) the predicted species richness per forest fragment. N = sampled forest fragments.



Appendix 3: Minimum, mean ± standard deviation and maximum values for species richness for distinct regions and categories, considering the observed (red) and the predicted (blue) species composition, and their ratio (grey). N = sampled forest fragments.

			S_Obs			S_SDM			ratio SDM/OBS		
Region	Type	N	min	mean ± SD	max	min	mean ± SD	max	min	mean ± SD	max
Southeast	Private	37	20	65 ± 17	105	168	221 ± 30	283	2	4 ± 2	12
	Protected	1	-	174	-	-	227	-	-	1	-
	ALL	38	20	68 ± 24	174	168	221 ± 30	283	1	4 ± 2	12
Center	Private	265	10	51 ± 22	163	39	185 ± 41	262	1	4 ± 3	24
	Protected	11	36	167 ± 67	283	72	165 ± 54	254	0	1 ± 0	2
	ALL	276	10	56 ± 34	283	39	184 ± 41	262	0	4 ± 3	24
West	Private	65	12	52 ± 22	115	42	115 ± 47	224	1	3 ± 2	10
	Protected	8	77	144 ± 65	260	63	136 ± 48	210	0	1 ± 0	2
	ALL	73	12	62 ± 41	260	42	118 ± 47	224	0	3 ± 2	10

Appendix 4: Boxplots (median ± quartiles) of total beta diversity (Bsor) and its components turnover (Bsim) and nestedness (Bnes) for distinct regions and categories, considering the observed (red) and the predicted (blue) species composition.



CHAPTER 3. ECOLOGICAL RESTORATION IN SAO PAULO, BRAZIL: HOW MUCH DIVERSITY CAN WE REPLICATE AT PLANT NURSERIES?

Cristina Yuri Vidal [1, 2], Rafaela Pereira Naves[3], Ricardo Augusto Gorne Viani[4], Ricardo Ribeiro Rodrigues[2]

[1] Universidade Estadual de Campinas (UNICAMP), Programa de Pós-Graduação em Biologia Vegetal, Instituto de Biologia, Campinas-SP, Brazil.

[2] Departamento de Ciências Biológicas, Universidade de São Paulo, Escola Superior de Agricultura “Luiz de Queiroz”, Piracicaba-SP, Brazil.

[3] Departamento de Ciências Florestais, Universidade de São Paulo, Escola Superior de Agricultura “Luiz de Queiroz”, Piracicaba-SP, Brazil.

[4] Departamento de Biotecnologia e Produção Vegetal e Animal, Universidade Federal de São Carlos, Centro de Ciências Agrárias, Araras-SP, Brazil.

ABSTRACT

Brazil has been committed to fulfill international restoration goals and to enforce an environmental legislation that will require restoration of 21 million hectares of degraded and deforested landscapes, with explicit requirement for undertaking active restoration rather than simply promoting spontaneous forest regeneration. To support a broad range of restoration practices, a consolidated supply chain able to represent and replicate regional plant diversity is essential. This study investigated seedling diversity available on native plant nurseries in São Paulo state, southeastern Brazil and evaluated their geographic distribution, similarity of their production, the proportion of species represented from regional floras and the relation of diversity descriptors with production capacity, surrounding forest cover and number of vegetation types. Despite the lack of technical assistance and the presence of exotic species (126 species, average 7.5 species/nursery), we found still more impressive native species richness in plant nurseries (561 species, average 86.4species/nursery) from both the Atlantic Forest and Cerrado domains, representing 38 to 44% of regional floras, depending on evaluated references. There is a huge bias toward tree and shrub species (96.6%) and absence or underrepresentation of other growth forms as well as of savanna specialists, animal-dispersed and threatened species. The diversity available in plant nurseries was positively related to its production capacity. The great dissimilarity of species offered in the surveyed nurseries undersocres the importance of regional seed collection practices. Effective assistance and capacitation are essential to address issues related to misidentification of species, underrepresentation of functional plant groups and the presence of exotic species, as well as to support the supply chain, currently undergoing market downturn.

INTRODUCTION

Recent studies argue it is unlikely that many or most tropical countries will be able to achieve their international commitments to restore ecosystems without natural and assisted regeneration (Chazdon & Uriarte 2016; Crouzeilles et al. 2017), a less costly alternative to scale-up restoration efforts (Holl & Aide 2011; Melo, Arroyo-Rodríguez, et al. 2013; Brancalion, Schweizer, et al. 2016; Latawiec et al. 2016). However, in landscapes with a long history of land conversion and extensive deforestation and defaunation, resilience is low (Rodrigues et al. 2009; Brancalion et al. 2012; Bello et al. 2015; Crouzeilles et al. 2017) and vegetation recovery depends on active restoration through direct seeding or planting seedlings (Holl & Aide 2011; Crouzeilles et al. 2017; Holl 2017; Meli, Holl, et al. 2017). Indeed, planting trees is the most common tropical forest restoration technique, despite being expensive, and time consuming (Rodrigues et al. 2011; Palma & Laurance 2015; Brancalion, Schweizer, et al. 2016; Holl et al. 2017; Meli, Holl, et al. 2017).

While opportunities for forest restoration are widespread within the tropics (Laestadius et al. 2012), Brazil has set a role model regarding restoration initiatives (Aronson et al. 2011; Calmon et al. 2011; Brancalion et al. 2013; Melo, Pinto, et al. 2013; Chaves et al. 2015; Holl 2017; Viani et al. 2017). The country has been committed to fulfill international restoration goals and to enforce a recently revised environmental legislation (*i.e.,* Native Vegetation Protection Law no. 12.651/2012 - NVPL) that applies on private lands, where *ca.* 53% of Brazil's remaining native vegetation is located (Soares-Filho et al. 2014; Brancalion et al. 2016a). To comply with the NVPL, about 21 million hectares will need restoration, including areas where active restoration is recommended; building up capacity and a supply chain to meet this demand is a major challenge, common to many other tropical countries worldwide.

Particularly on severely deforested scenarios, active restoration depends on the production of native seeds and seedlings, a bottleneck when intending to represent a large pool of species and genotypes (Brancalion et al. 2012; Nevill et al. 2016). There are 40 to 53 thousand tree species within the tropics (Slik et al. 2015) and at least 30 thousand seed plant species in Brazil

(Forzza et al. 2012; BFG 2015); as expected, there is not enough knowledge on their biology and current distribution. The challenge goes further when considering the process of harvesting propagules for viable seeds and/or seedlings' production, restrained by the reduced and degraded forest cover, lack of information on species reproductive biology and phenological patterns, unskilled labor and deficient technical capacity and assistance (Gregorio et al. 2004; Viani & Rodrigues 2009; Palma & Laurance 2015; Dedefo et al. 2017; White et al. 2018). Despite these setbacks, replication of plant diversity from regional pools is essential to consolidate a native plant market that enables a broad range of restoration goals, from ecological restoration as defined by the Society for Ecological Restoration (SER 2004) to mixed forestry systems with native and exotic species aiming timber production (Brancalion et al. 2012; Amazonas et al. 2018).

In fact, there are few examples of established supply chain for native species aiming tropical ecosystems' restoration on a large scale (Gregorio et al. 2004; Ladouceur et al. 2017; White et al. 2018). As an exceptional example, São Paulo state, Brazil developed a supply chain to fulfill their ecological restoration goals, with notable advances on the establishment of plant nurseries during the last 30 years (Barbosa et al. 2003; Martins 2011; Silva et al. 2015, 2017) and of a legal framework for ecological restoration (Durigan et al. 2010; Aronson et al. 2011; Chaves et al. 2015). Besides, São Paulo state represents an unique opportunity for a case study because i) it is composed by two of the hottest global hotspots, Atlantic Forest and Cerrado (Myers et al. 2000; Forzza et al. 2012) and ii) about 75% of the state has vegetation cover below the 30% threshold (Pardini et al. 2010; Estavillo et al. 2013; Banks-Leite et al. 2014; Boesing et al. 2018), reinforcing the demand on active restoration. Even though previous assessments and reports investigated plant nurseries' structure, production capacity and related difficulties, little or none attention has focused on the composition of the species available for restoration and the ecological aspects regarding taxonomic and functional approaches.

This study evaluated seedlings for active restoration on native plant nurseries in São Paulo State, Brazil. We investigated native plant nurseries production capacity (number of seedlings), with information about richness and

abundances distributions among species. We also examined their distribution along the state, how similar are their production and the proportion of species produced from regional floras. Finally, we explored the relation of diversity descriptors with possible explanatory variables such as production capacity, surrounding forest cover and number of vegetation types. We predicted native tree species composition is similar among plant nurseries and with an average richness around 80 species, in accordance to state recommendations established since 2008 (see details in Aronson et al. 2011; Chaves et al. 2015). We also predicted a compatible but limited representation of regional floras – both under taxonomic and functional approaches - and a positive influence of production capacity, surrounding forest cover and number of vegetation types on overall nurseries' richness or diversity.

METHODS

Data surveys and sampling

We compiled all São Paulo state plant nurseries listed on previous official assessments (Barbosa & Martins 2003; Martins 2011; Silva et al. 2015) plus new or unlisted nurseries indicated by restoration practitioners, totaling 347 plant nurseries. While contacting them by email/website, telephone or in person, we discharged duplicates (n=18) and those we could not reach by any means after five attempts (n=55) or that do not produce native species (n=33). Considering 241 eligible plant nurseries, we divided our sample in: i) quick surveys to assess their current production (2015 to 2017); ii) detailed surveys to assess relevant information on the origin of propagules, infra-structure, and market related issues (questionnaire adapted from Oliveira & Zakia 2010), as well as their species and abundance production (raining season 2015/2016) (**APPENDIX 1**).

For the quick surveys, we considered all regions of the São Paulo state, while for detailed surveys we sampled regions with mean forest cover below 30% (i.e., Southwest, Northwest, Center, Southeast) (Figure 1), where active restoration is usually recommended (Holl & Aide 2011; Tambosi et al.

2013). Additionally, detailed surveys' sampling depended on plant nurseries' willingness to provide the requested information.

Data on species abundance included any nursery-grown seedling available for planting in the field, regardless of the plant container or size. We emphasized the list of available species should consider only those appropriate for ecological restoration projects (i.e., excluding urban afforestation, silviculture, etc.), giving the nursery's staff free will to choose native species based on their judgment - a common real life practice that can mistakenly lead to the misuse of exotic species.

We dismissed morphospecies identified only to the family or genus levels and standardized species names using the Plantminer tool (www.plantminer.com, Carvalho et al. 2010), according to Flora do Brasil 2020 (http://reflora.jbrj.gov.br/) and The Plant List (www.theplantlist.org). From Flora do Brasil 2020 we retrieved information on growth forms, occurrence (Atlantic Forest and/or Cerrado) and origin (native, exotic), with further evaluation of problematic exotic species according to Sartorelli et al. (2018). For species occurring in the Cerrado (Brazilian Savanna), we refined the classification of occurrence according to their habitat preferences: i) savanna specialist, ii) forest specialist and iii) generalists (Mendonça et al. 2008; Abreu et al. 2017). For a functional grouping approach, we classified native species into the following functional guilds: i) pioneer, ii) fast-growing shading (Rodrigues et al. 2009), iii) understory non-pioneer and iv) canopy non-pioneer. Additionally, species were classified by dispersal syndromes and sub-syndromes (Bello et al. 2017).

For each plant nursery, we calculated the percentage of forest cover and the number of vegetation types (ordinal) in a surrounding 100 km buffer, extracting the information available on official shapefiles provided by the São Paulo State Forest Inventory (2011) with ArcGIS software (University of Campinas license). The six vegetation types occur both within Atlantic Forests (Seasonal Semideciduous Forests (SSF), Atlantic Forest *sensu stricto* (AFSS), Mixed Temperate Araucaria Forests (MTAF), Alluvial and Swamp Forests (A/SF)) and within Cerrado (Cerrado *sensu stricto* and Cerradão).

Data analysis

We compared native species and families available on plant nurseries with the list of species officially recommended for restoration in different regions of São Paulo State, provided by the state's Botanical Institute (hereafter SP-IBt) and available at www.botanica.sp.gov. We also compared our data with a dataset of floristic surveys performed by the Forest Ecology and Restoration Laboratory (LERF/University of São Paulo), describing the occurrence of shrub/tree species across forest fragments (N=371) in the studied regions (Rodrigues et al. 2011). Comparisons focused on evaluation of shared and exclusive species, proportions of functional guilds and ranking the botanical families' richness, in order to detect eventual mismatches or lacking groups in plant nurseries.

To describe the diversity among plant nurseries and within ecological regions we used species abundance distribution (SAD) for native species (McGill et al. 2007). SAD models provide a powerful way to understand the abundance structure of nurseries' production, revealing the evenness – or lack thereof - of their "communities" (Magurran 2013). We fitted log series and Poisson log normal distributions to the species abundance data using the maximum-likelihood tools with the sads package for R 3.1 version (Prado et al. 2016). We compared the models based on Akaike's information criterion (AIC) (Hilborn & Mangel 1997) and for every plant nursery, Poisson log normal provided the best fit to our data. Therefore, we used its parameter sigma (σ) as a diversity metric (Sæther et al. 2013).

We fitted linear models to analyze the influence of explanatory variables - production capacity, forest cover, number of vegetation types, ecological regions - on richness and sigma diversity descriptors. We tested all models to meet assumptions of normality and homogeneity of variance and then compared them based on AIC. We ranked the models according to the lowest AIC value; models with a difference in AIC (Δ) ≤ 2 can be considered to have equivalently strong empirical support and similar plausibility (Hilborn & Mangel 1997; Bolker 2008).

To evaluate how the available diversity is distributed among the plant nurseries studied, we first examined compositional resemblance using incidence-based Jaccard's dissimilarity index, considering two communities similar when

values were above 0.25 (Durigan 2012). We also calculated regional beta diversity in order to measure the contribution of its components' relative nestedness and turnover: nestedness results from differences between the richest and the poorest assemblages, considering they represent sub sets of the regional species pool, while turnover results from the actual replacement of species between local species assemblages (Baselga 2010; Socolar et al. 2016). For practical purposes, highly nested β diversity means species-rich plant nurseries are sufficient to represent regional diversity, while high turnover means that distinct plant nurseries represent the regional diversity when gathered. We calculated multi-site Sorensen index (β_{SOR}), representing total β diversity, and the Simpson index (β_{SIM}) that only measures turnover; therefore, the nestedness (β_{NEST}) component is $\beta_{SOR} - \beta_{SIM}$ (Baselga 2010). All analysis were perfomed on R (R Development Core Team 2011) function "beta-multi.R" in package "betapart" (Baselga & Orme 2012).

RESULTS

Plant nurseries assessment

We contacted 241 registered native plant nurseries with a "quick" survey, and confirmed that 64.3% (n=155) were still active, while 35.7% were either currently deactivated (n=69) or retailing seedlings from other nurseries (n=17). Spatial distribution of active nurseries was concentrated in the Center (n=55) and Southeast (n=57) regions of Sao Paulo state (**Figure 1**); combined they constituted 72.3% of all plant nurseries.

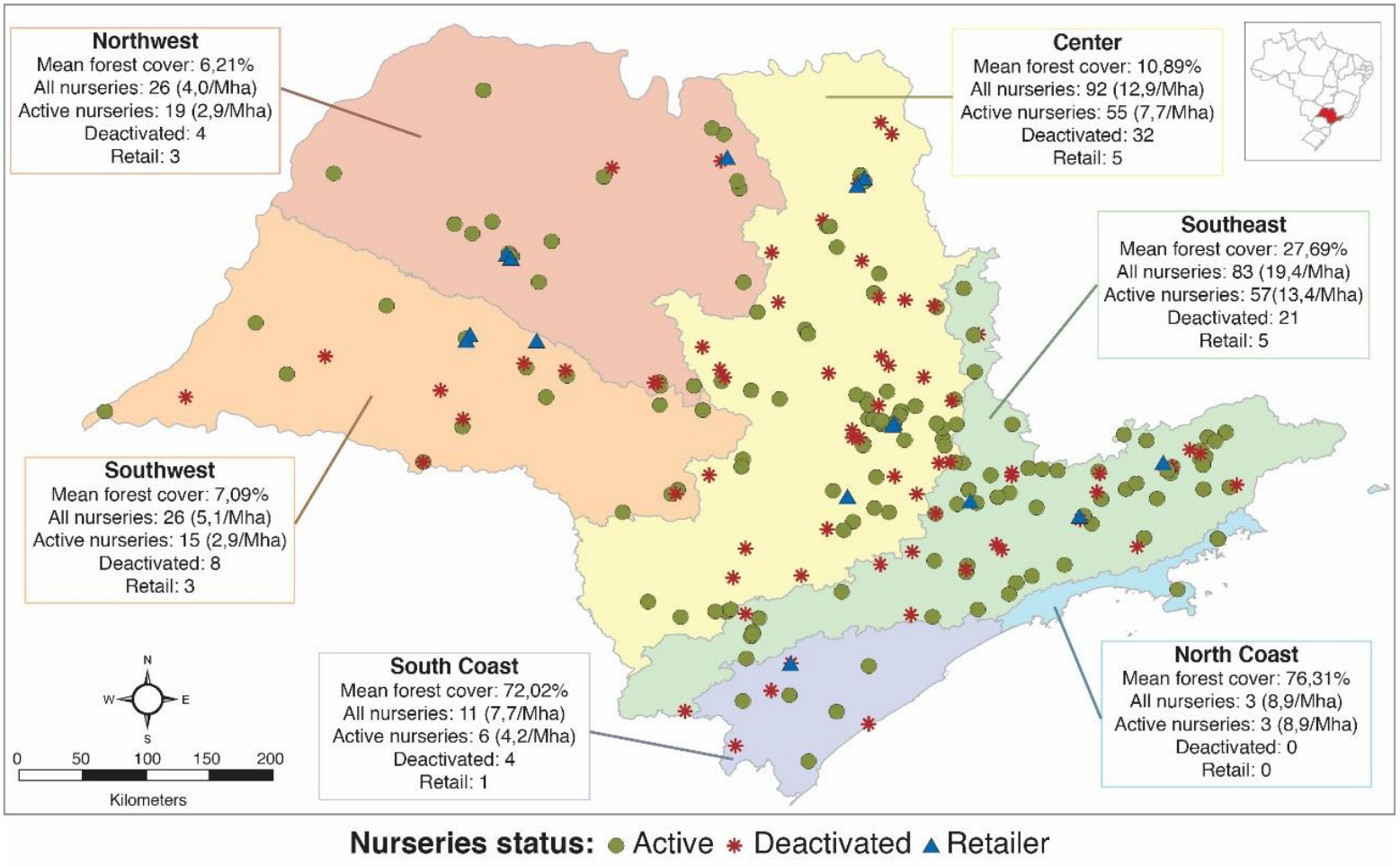


Figure 1. Distribution of native plant nurseries among the ecological regions defined by São Paulo state's Botanical Institute (SP-IBt). Each region is described by their mean forest cover and by the quantity and density of all assessed native plant nurseries and active nurseries only, as well as the quantity of deactivated and retailing nurseries. Mean forest cover based on São Paulo State Forest Inventory (2011) and density calculated by million hectares (Mha).

From detailed surveys made with 54 plant nurseries, we noted that 87% (n=47) collected their own propagules (i.e., seeds and/or fruits), harvesting from the surrounding forest fragments within an average distance of 100 km radius. Over half of them (57%, n=31) also purchased additional seeds from other sources - even from out of the state - to enhance diversity. In the rainy season of 2015/2016 the nurseries we surveyed produced approximately 9.3 million seedlings, with individual production ranging from 3,500 to 1,800,000 seedlings. Regarding identification practices, most active nurseries (90%) kept track of species using both popular and scientific names. However, for botanical identification, they relied on their own expertise and/or illustrated guides such as "*Brazilian Trees*" (Lorenzi 2002), as most of them lack access to qualified botanical experts.

Species diversity

From 687 plant species, 561 (81.1%) are native from São Paulo and 126 (17.8%) are exotic (**APPENDICES 2 and 3**). Among natives, there were 542 shrub/tree species (96.6%), five sub-shrubs, seven palm trees and seven lianas, with average richness of 86.4 per nursery, ranging from 18 to 194 species (**Table 1**). Exotic species represented 4.8% of the total number of seedlings; eight of the 10 most abundant species occur in Brazil, but in other regions and/or vegetation types. According to Sartorelli et al. (2018), 10 species should be avoided in restoration projects due to their invasive potential.

Table 1. Native and exotic species richness registered for different regions of the of São Paulo state, where we sampled N plant nurseries. Threatened species included extinct, extinct in the wild, critically endangered, endangered or vulnerable species according to CNC Flora and the IUCN or the State of Sao Paulo's red list. Regions: SE= southeast, CT=center, NW=northwest, SW=southwest.

		NATIVE SPECIES					EXOTIC SPECIES			
Regions	**N**	**all**	**min**	**mean**	**max**	**threat**	**all**	**min**	**mean**	**max**
Souteast	17	326	18	74.4	124	12	44	1	5.4	18
Center	27	472	29	92.5	194	16	87	1	8.3	22
Northwest	6	227	27	87.4	116	7	40	2	10.2	27
Soutwest	4	193	54	95	122	7	21	4	7.5	11
Total	**54**	**561**	**18**	**86.4**	**194**	**19**	**126**	**1**	**7.5**	**27**

Considering only native shrub/tree species, plant nurseries produced 86 exclusive species, i.e. not listed on references but native to São Paulo; their production encompassed 37.9% of species recommended for restoration by regions in São Paulo State and 44.2% of species registered within surveys in São Paulo state forest remnants (**Figure 2A**). From all species in the plant nurseries, 462 occur in the Atlantic Forest biome and 396 species in the Cerrado biome, but for the latter, only 94 are savanna specialist species (23.7%), while 250 are forest specialists or generalists (63.1%).

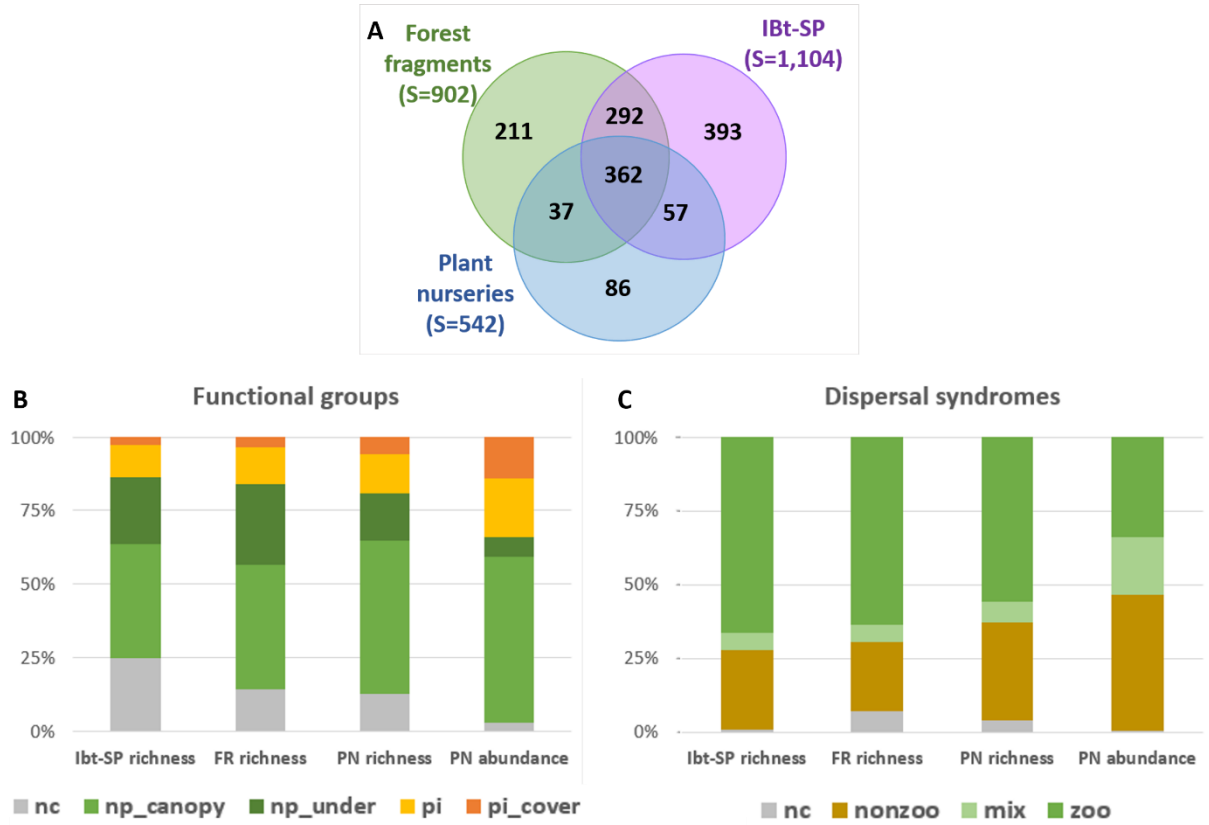


Figure 2. Comparison of floristic composition among plant nurseries (PN), sampled forest fragments (FR) and species officially recommended for restoration by the state's Botanical Institute (IBt-SP). (A) Shared and exclusive species richness; (B) Proportion of functional groups/guilds: nc = non-classified, np_canopy = canopy non-pioneer, np_under = understory non-pioneer pi = pioneer and fg_shad = fast-growing shading species; (c) Proportion of dispersal syndromes: nc= non-classified, nonzoo = non zoochoric, mix= mixed (both non-zoo and zoo), zoo = zoochoric species.

Plant nurseries partially replicated species richness and maintained overall proportions of functional groups and dispersal syndromes observed on references (**Figure 2B, 2C**). When considering abundances of plant produced in the plant

nurseries surveyed, non-pioneer (canopy and understory) were approximately two times more abundant than pioneer and fast-growing shading species (**Figure 2B**) while animal-dispersed species represented over half (56%) of species and one third (34%) of produced seedlings (**Figure 2C**). We observed the same rationale among the richest families, as they were fairly represented in plant nurseries but with some depleted families such as Lauraceae, Melastomataceae and Rubiaceae (**APPENDIX 4**).

The SAD revealed that nurseries presented very uneven communities, with 35 species (6.2%) representing half of all produced seedlings, while the other half included 526 species (93.7%) (**APPENDIX 5A**). Regarding overall abundance, the Central region concentrated 55% of the nurseries production capacity, followed by Southeast (14%) (**APPENDIX 5B**). Species frequency indicated only 12 species (2%) as common (i.e. in more than 75% nurseries), while 440 species (78.6%) were considered rare (i.e. less than 25%).

The 35 most abundant species – 12 of which were also among the most frequent - represented half of seedlings available. They were mostly non-pioneer canopy (23 species), with five fast-growing shading and five pioneer species. They were predominantly abiotic-dispersed species (19), with 10 dispersed either by abiotic or biotic factors while six were strictly dependent on animals.

In agreement with what the above-cited frequency patterns suggest, Jaccard's index of dissimilarity revealed plant nurseries were very different regarding species composition, since this index varied from 0 (similar) to 1 (dissimilar) and the overall mean value is 0.75 (**Table 2**). In addition, βsor diversity for distinct regions is mainly due to species turnover (βsim) (**Table 2**).

Table 2. Jaccard dissimilarity index (mean +- standard deviation) and beta diversity (β_{SOR}), decomposed into turnover (β_{SIM}) and nestedness (β_{NEST}) components for all distinct regions. N is the number of sampled plant nurseries.

Regions	N	Jacc _index	β_{SOR}	β_{SIM}	β_{NEST}
Southeast	17	0.77+-0.07	0.87 (100%)	0.78 (89.6%)	0.09 (10.4%)
Center	27	0.73+-0.10	0.90 (100%)	0.83 (92.2%)	0.07 (7.8%)
Northwest	6	0.71+-0.09	0.70 (100%)	0.58 (82.8%)	0.12 (17.2%)
Soutwest	4	0.64+-0.07	0.58 (100%)	0.45 (77.5%)	0.13 (22.5%)
Total	**54**	**0.75+-0.09**	**0.95** (100%)	**0.91** (95.8%)	**0.04** (4.2%)

There was a positive correlation between the production capacity of a plant nursery and its species richness and sigma (**Figure 3**). Models considering forest cover, number of vegetation types, ecological regions were no better than expected by chance (null model) (**APPENDIX 6**).

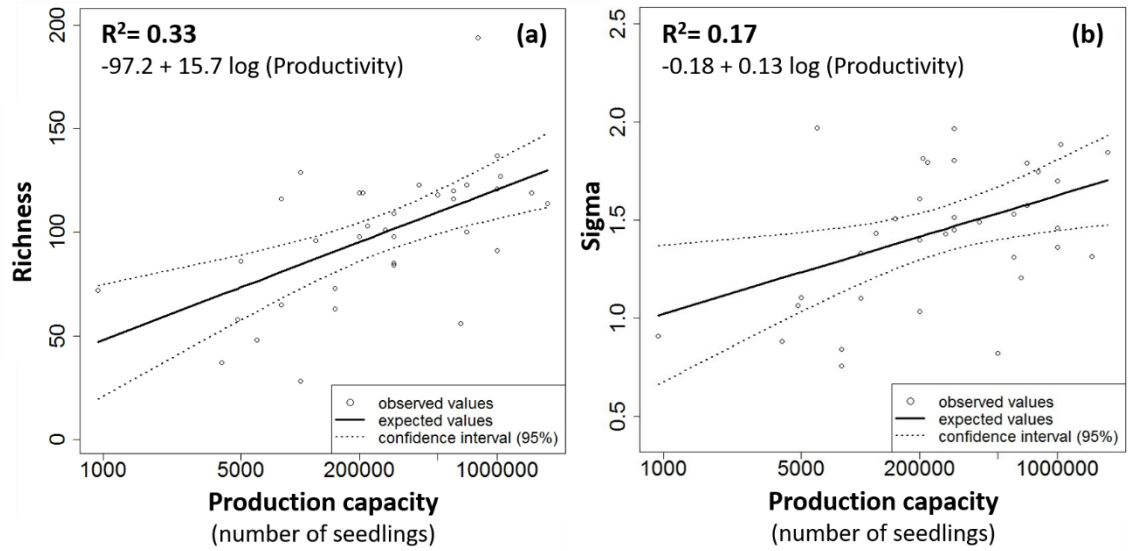


Figure 3. Best fitting linear models for diversity descriptors considering production capacity as an explanatory variable for (a) richness (R^2= 0.33) and (b) sigma (R^2 = 0.34).

DISCUSSION

Our study on the largest native supply chain in Brazil (Silva et al. 2017) revealed that plant nurseries were propagating and offering for sale approximately 38% of native tree and shrub species recommended for restoration in Sao Paulo state. We registered high overall native species richness (561) and average per nursery (86.4), which is above Brazilian national standards (63) (Silva et al. 2015) and above previous averages recorded for plant nurseries in São Paulo state (Barbosa et al. 2003; Martins 2011). The only explanatory variable positively related to diversity in plant nurseries is their production capacity, suggesting that nurseries producing more seedlings tend to be more organized and professional, which might be related to higher input resources on a wide-range seed collection and other seed acquisition strategies (Brancalion et al. 2012a). A remarkable result of our study is the singularity of plant nurseries' production, proven by the high values of β-diversity (i.e. high dissimilarity among nurseries' composition). Since most plant nurseries collect propagules from the

surroundings, we presume that not only they represent the regional taxonomic diversity but also the populations' genotypes (Zucchi et al. 2017; White et al. 2018). These results altogether reinforce the importance of public policies and incentives addressing the restoration supply chain, especially in those regions where restoration cannot rely on natural regeneration process and where a well-established native plant market may contribute to high diversity ecological restoration initiatives.

While representing surrounding vegetation, both Atlantic Forest and Cerrado species are available in plant nurseries, but the underrepresentation of savannas' specialists most likely lead to afforestation of grassy biomes, with negative consequences for the native biodiversity of the savanna biome (Overbeck et al. 2013; Veldman et al. 2015a, 2015b; Abreu et al. 2017; Buisson et al. 2018; Pilon et al. 2018). Available species represent a narrow spectrum of growth forms that lack or underrepresent lianas, epiphytes and herbs, which should exceed 2 to 7 times the number of tree species in the Atlantic Forest and Cerrado biomes (BFG 2015) and are an important flora component as they increase resource availability to fauna (Garcia et al. 2014, Garcia et al. 2015). Although production bias towards woody species exists because they are the main structural components of forests – which in turn is the main target of most Brazilian restoration initiatives - awareness should be raised as to the importance of other growth forms, especially for nonforest biomes, where restoration demand is increasing and propagation knowledge is still challenging (Overbeck et al. 2013; Campbell et al. 2015; Veldman et al. 2015a; Garcia et al. 2016; Mayfield 2016; Duarte & Gandolfi 2017; Buisson et al. 2018; Pilon et al. 2018). Considering only tree and shrub species, studied plant nurseries were offering less than 50% of official regional species lists (SMA-IBt) and remaining diversity in forest fragments; these numbers are less than we could expect from the most established supply chain in Brazil (Silva et al. 2015, 2017), even when accounting for the lack of knowledge on species' biology and distribution (White et al. 2018). The situation is even more critical when considering that only 2.3% (19 spp) of São Paulo state's threatened plant species were found in plant nurseries, falling short of the objectives of the Global Strategy for Plant Conservation in Brazil, which defines a goal of making 20% of threatened species available for restoration efforts by 2020 (Martins et al. 2017). Since threatened species offer a greater challenge for conservationists, specific recovery plans would be necessary to achieve this particular goal (Durigan et al. 2010; Martins et al. 2017).

Representation of functional groups in nurseries should also include fast-growing shading species that boost soil coverage and shade exotic weeds (Rodrigues et al. 2009) and a greater variety and quantity of canopy non-pioneer species that will presumably persist in restored sites over time (Rodrigues et al. 2011; Brancalion et al. 2012a). However, the overall variety and quantities of animal-dispersed species are below those expected for Atlantic Forest where it varies from 70 to 94% of woody species (Almeida-Neto et al. 2008; Bello et al. 2017). As shown by Brancalion et al. (2018), large-seeded animal-dispersed are particularly underrepresented, with consequences on carbon storage and restoration outcomes. We recommend enhancement on the proportion of animal-dispersed species in plant nurseries, since plants consumed and dispersed by animals are notably important in degraded landscapes, where maintenance of plant-animal interactions are essential to enable restoration of ecological processes and biological fluxes over time(Peña-Domene et al. 2014; Howe 2016; Brancalion et al. 2018).

Tropical ecosystems are typically characterized by skewed species-abundance distribution, with some very common species and many rare or very rare (Caiafa & Martins 2010; Hubbell 2013). Reflecting this pattern, we found half of total seedling production represented by only 35 species (6%), while almost 80% of species were considered rare. This very high proportion of rare species is consonant with our results that indicated high variability among plant nurseries' species composition, with particular contribution from turnover (i.e. species replacement) (Baselga 2010; Socolar et al. 2016). Aligned with the fact that turnover is consistently the larger component of β-diversity within the tropics (Soininen et al. 2018), these results are also a reflection of plant nurseries' practice on collecting propagules from surrounding forest fragments, representing their highly variable species composition. Following this rationale, if nurseries are well distributed, so is the taxonomic diversity, maximizing the chances of representing local specimens well adapted for regional restoration projects (White et al. 2018). Thus, the biased spatial distribution of plant nurseries raise an issue to be addressed by public policy makers. For example, an argument could be made to govern nursery practice and corrective and supportive measures such as the implementation of regional seed exchange programs (Brancalion et al. 2012a).

Contrary to the results we found for plant nurseries representing the surrounding vegetation and regional flora, we did not find evidence supporting the influence of surrounding forest cover percentage and number of vegetation types over available diversity in plant nurseries. That is probably because all nurseries evaluated on the detailed surveys are located in regions under similar conditions (less than 30% forest cover), with little variation on vegetation types surrounding them. Beyond the positive influence of production capacity over diversity availability, we consider that overall, high species richness most likely derive from the enforcement of São Paulo state legislation (Brancalion et al. 2010), which used to establish the minimum amount of reintroduced species (Aronson et al. 2011) and currently focuses on monitoring the achievement of goals related both to structure and diversity of sites undergoing restoration (Chaves et al. 2015). Regardless of the discussion on whether it is positive or negative to standardize the amount of species on a restoration project (Aronson et al. 2011), we must recognize that these legal instruments have pushed plant nurseries to enhance their diversity (Brancalion et al. 2012a; Silva et al. 2017), placing São Paulo state native trees' seedling production at a very high level, far higher than elsewhere in Brazil, and possibly worldwide.

As much as we are thrilled to know native plant nurseries - with all their reservations - are replicating and producing such a large variety of native tree and shrub species, we must consider some caveats of the results registered in this study. First, we highlight the alarming market downturn that have been affecting the production of native seedlings since the initial discussions to revise the main environmental legislation in Brazil (*i.e.* Native Vegetation Protection Law no. 12.651/2012) (details in Brancalion et al. 2016a). Second, we considered only the rainy season of 2015/2016 and species richness may be even higher if a longer period is evaluated, as flowering and fruiting periods have interannual variability (Morellato et al. 2001; Viani & Rodrigues 2009; Brancalion et al. 2012a). Third, few nurseries adopt good identification practices such as the collection of samples for vouchers specimens for depositing in herbaria and examination by professional botanists. Mistaken identification in plant nurseries can mislead to over- or under-estimations of the actual diversity available on nurseries and it also explains the production of exotic species, a common issue in restoration sites (Barbosa et al. 2003; Assis et al. 2013; Brancalion et al. 2016b). While some argue the inclusion of exotics is acceptable under specific

circumstances (Brancalion et al. 2012b; Amazonas et al. 2018), there is general consensus that species with invasive behavior must be excluded (*e.g. Syzigium cumini* (Myrtaceae), *Azadirachta indica* (Meliaceae), and *Eriobotrya japonica* (Rosaceae), as they could lead do deleterious effect on native communities (Sartorelli et al. 2018).

Our results underscore that native plant nurseries in São Paulo replicate a considerable portion of tree and shrub diversity, but how that affects success of ecological restoration depends on whether we consider biodiversity introduced in restoration projects as a goal or a driver of the recovery process (Naeem 2016). There is ongoing debate in Brazil regarding the benefits of using a high or low diversity in restoration efforts (Brancalion et al. 2010; Durigan et al. 2010; Aronson et al. 2011), considering cost reductions, field performance, and definition of supposed "framework species" (Suganuma & Durigan 2015) as well as compelling evidence associating biodiversity and ecosystem functioning (BEF) (Wright et al. 2009; Aerts & Honnay 2011; Cardinale et al. 2012; Tilman et al. 2014; Brockerhoff et al. 2017). Despite the lack of consistent evidence relating the amount of reintroduced diversity and restoration success, pursuing and promoting higher diversity in the production of native species is essential to foster a wider variety of restoration approaches, including those with economic benefits (e.g., mixed plantations) (Brancalion et al. 2012b; Amazonas et al. 2018), and those for conventional and alternative conservation purposes (e.g. landscape gardening, restoration of degraded forest remnants, etc) (Viani et al. 2015).

The impressive levels of species richness registered in this study represent, to our knowledge, the most diverse tropical native tree seedling production and supply chain anywhere in the world. Plant nurseries play an extremely important role on replicating remaining biodiversity, as they collect most of their seed from local provenance and represent local populations and communities, contributing to their uniqueness regarding species composition. Nevertheless, the partial representation of the great tropical ecosystems' diversity confirm active restoration cannot fully restore all dimensions of biodiversity (Crouzeilles et al. 2016; Naeem 2016; Meli et al. 2017), highlighting the complementary role it plays supporting the conservation of native vegetation remnants. We found nurseries concentrate their production on shrub and tree species and are sub-representing other growth forms and some functional groups, such as savanna specialists, animal-dispersed and threatened species. In that sense,

our recommendation is to provide effective assistance and capacity building to address issues related to misidentification, underrepresentation of functional groups, and the presence of exotic and invasive species, as well as to support the supply chain, currently under market downturn.

ACKNOWLEDGEMENTS: We thank all plant nurseries' owners and staff for sharing their information and thoughts; Letícia S. Santos, Bruno H. Guastala, Fernando H. Silva and Sergio Lozano-Baez for contacting nurseries during quick surveys. This study was financed in part by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior – Brasil (CAPES) – Finance Code 001, by the National Council for Scientific and Technological Development (CNPq grant 870360/1997-3) and by The São Paulo Research Foundation (FAPESP grant 2013/50718-5).

REFERENCES

Abreu RCR, Hoffmann WA, Vasconcelos HL, Pilon NA, Rossatto DR, Durigan G (2017) The biodiversity cost of carbon sequestration in tropical savanna. Science Advances 3:e1701284

Aerts R, Honnay O (2011) Forest restoration, biodiversity and ecosystem functioning. BMC Ecology 24:11-29

Aronson J, Brancalion PHS, Durigan G, Rodrigues RR, Engel VL, Tabarelli M, et al. (2011) What Role Should Government Regulation Play in Ecological Restoration? Ongoing Debate in São Paulo State, Brazil. Restoration Ecology 19:690–695

Assis GB, Suganuma MS, Melo ACG, Durigan G (2013) Uso de espécies nativas e exóticas na restauração de matas ciliares no Estado de São Paulo (1957 - 2008). Revista Árvore 37:599–609

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

Barbosa LM, Barbosa JM, Barbosa KC, Potomati A, Martins SE, Asperti LM et al. (2003) Recuperação Florestal Com Espécies Nativas No Estado De São Paulo: Pesquisas Apontam Mudanças Necessárias. Florestar estatístico 6:28–34

Barbosa LM and Martins SE (2003) Diversificando o reflorestamento no estado de São Paulo: espécies disponíveis por região e ecossistema. Secretaria de Meio Ambiente do Estado de São Paulo, São Paulo

Baselga A (2010) Partitioning the turnover and nestedness components of beta diversity. Global Ecology and Biogeography 19:134–143

Baselga A, Orme CDL (2012) Betapart: An R package for the study of beta diversity. Methods in Ecology and Evolution 3:808–812

Bello C, Galetti M, Montan D, Pizo MA, Mariguela TC, Culot L, et al. (2017) Atlantic frugivory : A plant – frugivore interaction data set for the Atlantic Forest. Ecology 0:0

Bello C, Galetti M, Pizo MA, Magnago LFS, Rocha MF, Lima RAF, Peres CA, Ovaskainen O, Jordano P (2015) Defaunation affects carbon storage in tropical forests. Science Advances 1:1–11

BFG The Brazil Flora Group (2015) Growing knowledge: an overview of Seed Plant diversity in Brazil. Rodriguesia 66:1-29

Bolker BM (2008) Ecological models and data in R. Princeton University Press, New Jersey

Brancalion PHS, Bello C, Chazdon RL, Galetti M, Jordano P, Lima RAF, et al. (2018) Maximizing biodiversity conservation and carbon stocking in restored tropical forests. Conservation Letters:e12454

Brancalion PHS, Garcia LC, Loyola R, Rodrigues RR, Pillar VD, Lewinsohn TM (2016a) A critical analysis of the Native Vegetation Protection Law of Brazil (2012): Updates and ongoing initiatives. Natureza e Conservacao 14:1–15

Brancalion PHS, Melo FPL, Tabarelli M, Ricardo R (2013) Biodiversity persistence in highly human-modified tropical landscapes depends on ecological restoration. Tropical Conservation Science 6:705–710

Brancalion PHS, Rodrigues RR, Gandolfi S, Kageyama PY, Nave AG, Gandara FB, Barbosa LM, Tabarelli M (2010) Instrumentos legais podem contribuir para a restauração de florestas tropicais biodiversas. Revista Árvore 34:455–470

Brancalion PHS, Schweizer D, Gaudare U, Mangueira JR, Lamonato F, Farah FT, Nave AG, Rodrigues R (2016b) Balancing economic costs and ecological outcomes of passive and active restoration in agricultural landscapes : the case of Brazil. Biotropica 48:856–867.

Brancalion PHS, Viani RAG, Aronson J, Rodrigues RR, Nave AG (2012a) Improving Planting Stocks for the Brazilian Atlantic Forest Restoration through

Community-Based Seed Harvesting Strategies. Restoration Ecology 20:704–711

Brancalion PHS, Viani RAG, Strassburg BBN, Rodrigues RR (2012b) Finding the money for tropical forest restoration. Unasylva 63:41–50

Brockerhoff EG, Barbaro L, Castagneyrol B, Forrester DI, Gardiner B, Lyver POB et al. (2017) Forest biodiversity , ecosystem functioning and the provision of ecosystem services. Biodiversity and Conservation 26:3005

Buisson E, Le Stradic S, Silveira FAO, Durigan G, Overbeck GE, Fidelis A et al. (2018) Resilience and restoration of tropical and subtropical grasslands, savannas, and grassy woodlands. Biological Reviews

Caiafa AN and Martins FR (2010) Forms of rarity of tree species in the southern Brazilian Atlantic rainforest. Biodiversity and Conservation 19:2597–2618

Calmon M, Brancalion PHS, Paese A, Aronson J, Castro P, Silva SC, Rodrigues RR (2011) Emerging Threats and Opportunities for Large-Scale Ecological Restoration in the Atlantic Forest of Brazil. Restoration Ecology 19:154–158

Cardinale BJ, Duffy E, Gonzalez A, Hooper DU, Perrings C, Venail P, et al. (2012) Biodiversity loss and its impact on humanity. Nature 489:326–326

Carvalho GH, Cianciaruso MV, Batalha MA (2010) Plantminer: A web tool for checking and gathering plant species taxonomic information. Environmental Modelling and Software 25:815–816

Chaves RB, Durigan G, Brancalion PHS, Aronson J (2015) On the need of legal frameworks for assessing restoration projects success: new perspectives from São Paulo state (Brazil). Restoration Ecology 23:754–759

Chazdon RL and Uriarte M (2016) Incorporating natural regeneration in forest landscape restoration in tropical regions : synthesis and key research gaps. Biotropica 48:915–924

Crouzeilles R, Curran M, Ferreira MS, Lindenmayer DB, Grelle CE V, Rey Benayas JM (2016) A global meta-analysis on the ecological drivers of forest restoration success. Nature communications 7:1–8

Crouzeilles R, Ferreira MS, Chazdon RL, Lindenmayer DB, Sansevero JBB, Monteiro L, et al. (2017) Ecological restoration success is higher for natural regeneration than for active restoration in tropical forests. Science Advances 3:e1701345

Dedefo K, Derero A, Tesfaye Y, Muriuki J (2017) Tree nursery and seed procurement characteristics influence on seedling quality in Oromia, Ethiopia. Forests Trees and Livelihoods 26:96–110

Durigan G (2012) Estrutura e diversidade de comunidades florestais. Pages 294–319 In:Martins SV (editor)Ecologia de florestas tropicais do Brasil. Universidade Federal de Viçosa, Viçosa, Minas Gerais

Durigan G, Engel VL, Torezan JM, Melo ACG, Marques MCM, Martins SV, et al. (2010) Normas jurídicas para a restauração ecológica: uma barreira a mais a dificultar o êxito das iniciativas? Revista Árvore 34:471–485

Estavillo C, Pardini R, Rocha PLB (2013) Forest loss and the biodiversity threshold: An evaluation considering species habitat requirements and the use of matrix habitats. PLoS ONE 8:1–10

Flora do Brasil 2020 Jardim Botânico do Rio de Janeiro. Available on http://floradobrasil.jbrj.gov.br/ (acessed 20 July 2018)

Forzza RC, Baumgratz JF, Bicudo CEM, Canhos DAL, Carvalho Jr AA, Coelho MAN et al. (2012) New Brazilian Floristic List Highlights Conservation Challenges. BioScience 62:39–45

Garcia LC, Hobbs RJ, Mäes dos Santos F a., Rodrigues RR (2014) Flower and Fruit Availability along a Forest Restoration Gradient. Biotropica 46:114–123. doi: 10.1111/btp.12080

Garcia LC, Cianciaruso MV, Ribeiro DB, Santos FAM, Rodrigues RR (2015) Flower functional trait responses to restoration time. Applied Vegetation Science

Gregorio N, Herbohn J, Harrison S (2004) Small-scale forestry development in Leyte, Philippines: The central role of nurseries. Small-scale Forest Economics, Management and Policy 3:337–351

Hilborn R and Mangel M (1997) The Ecological Detective Confronting Models with Data. Princeton University Press, New Jersey

Holl KD (2017) Restoring tropical forests from the bottom up. Science 355:455–456

Holl KD and Aide TM (2011) When and where to actively restore ecosystems? Forest Ecology and Management 261:1558–1563

Holl KD, Reid JL, Chaves-Fallas JM, Oviedo-Brenes F, Zahawi RA (2017) Local tropical forest restoration strategies affect tree recruitment more strongly than does landscape forest cover. Journal of Applied Ecology 54:1091–1099

Hubbell SP (2013) Tropical rain forest conservation and the twin challenges of diversity and rarity. Ecology and Evolution 3:3263–3274

Ladouceur E, Jim B, Marin M, Vitis M De, Abbandonato H, Iannetta PPM, Bonomi C, Pritchard HW (2017) Native Seed Supply and the Restoration Species Pool. Conservation Letters 00:1–9

Laestadius L, Maginnis S, Minnemeyer S, Potapov P, Saint-Laurent C, Sizer N (2012) Mapping opportunities for forest landscape restoration. Unasylva 62:47–48

Latawiec AE, Crouzeilles R, Brancalion PHS, Rodrigues RR, Sansevero JB, Santos JS, et al. (2016) Natural regeneration and biodiversity: a global meta-analysis and implications for spatial planning. Biotropica 48:844–855

Lorenzi H (2002) Árvores Brasileiras: manual de identificação e cultivo de plantas arbóreas do Brasil. Instituto Plantarum, Nova Odessa, São Paulo

Magurran AE (2013) Measuring biological diversity. Blackwell Publishing Company

Martins E, Loyola R, Martinelli G (2017) Challenges and Perspectives for Achieving the Global Strategy for Plant Conservation Targets in Brazil. Annals of the Missouri Botanical Garden 102:347–356

Martins RB (2011) Diagnóstico dos produtores de mudas florestais nativas do Estado de São Paulo. Secretaria de Meio Ambiente do Estado de São Paulo, São Paulo

McGill BJ, Etienne RA, Gray JS, Alonso D, Anderson MJ, Benecha HK, et al. (2007) Species abundance distributions: Moving beyond single prediction theories to integration within an ecological framework. Ecology Letters 10:995–1015

Meli P, Holl KD, Benayas JMR, Jones HP, Jones PC, Montoya D, Mateos DM (2017) A global review of past land use, climate, and active vs. passive restoration effects on forest recovery. PLoS ONE 12:1–17

Melo FPL, Pinto SRR, Brancalion PHS, Castro PS, Rodrigues RR, Aronson J, Tabarelli M (2013) Priority setting for scaling-up tropical forest restoration projects: Early lessons from the Atlantic forest restoration pact. Environmental Science and Policy 33:395–404

Mendonça RC, Felfili JM, Walter BM, Silva Junior MC, Rezende AV, Filgueiras TS, Nogueira P (2008) Flora vascular do bioma Cerrado. Pages 422–442 In Sano SM, Almeida SP, Ribeiro JF (eds.) Flora vascular do bioma Cerrado: checklist com 12.356 espécies. EMBRAPA Cerrados, Brasília, Distrito Federal

Morellato LPC, Talora DC, Takahasi A, Bencke CC, Romera EC, Zipparro VB (2001) Phenology of Atlantic rain forest trees: A comparative study. Biotropica 32:811–823

Myers N, Mittermeier RA, Mittermeier CG, Fonseca GA, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858

Naeem S (2016) Biodiversity as a Goal and Driver of Restoration. Pages 57–89 In Palmer, MA, Zedler JB, Falk DA (eds.) Foundations of Restoration Ecology. Second edition. Island Press, Washington

Nevill PG, Tomlinson S, Elliott CP, Espeland EK, Dixon KW, Merritt DJ (2016) Seed production areas for the global restoration challenge. Ecology and Evolution 6:7490–7497

Oliveira RE and Zakia MJB (2010) Guia para análise de viveiros de mudas nativas - Checklist para verificação da adequação legal, socioambiental e ecológica de viveiros de mudas florestais nativas. Piracicaba, São Paulo

Overbeck GE, Hermann JM, Andrae BO, Boldrini II, Kiehl K, Kirmer A, et al. (2013) Restoration ecology in Brazil-time to step out of the forest. Natureza a Conservacao 11:92–95

Palma AC and Laurance SGW (2015) A review of the use of direct seeding and seedling plantings in restoration: What do we know and where should we go? Applied Vegetation Science 18:561–568

Pardini R, Bueno ADA, Gardner TA, Prado PI, Metzger JP (2010) Beyond the fragmentation threshold hypothesis: regime shifts in biodiversity across fragmented landscapes. PloS one 5:e13666

Peña-Domene M, Martínez-Garza C, Palmas-Pérez S, Rivas-Alonso E, Howe HF (2014) Roles of birds and bats in early tropical-forest restoration. PLoS ONE 9:1–6

Pilon NAL, Buisson E, Durigan G (2018) Restoring Brazilian savanna ground layer vegetation by topsoil and hay transfer. Restoration Ecology 26:73–81

Prado PI, Miranda MD, Chalom A (2016) sads: Maximum Likelihood Models for Species Abundance Distributions R package version 0.3.1. Available from https://cran.r-project.org/package=sads

Rodrigues RR, Gandolfi S, Nave AG, Aronson J, Barreto TE, Vidal CY, Brancalion PHS (2011) Large-scale ecological restoration of high-diversity tropical forests in SE Brazil. Forest Ecology and Management 261: 1605-1613

Rodrigues RR, Lima RAF, Gandolfi S, Nave AG (2009) On the restoration of high diversity forests: 30 years of experience in the Brazilian Atlantic Forest. Biological Conservation 142:1242–1251

Sæther BE, Engen S, Grøtan V (2013) Species diversity and community similarity in fluctuating environments: Parametric approaches using species abundance distributions. Journal of Animal Ecology 82:721–738

Sartorelli PAR, Benedito ALD, Campos Filho EM, Sampaio AB, Ana S, Gouvêa APML (2018) Guia de plantas não desejáveis na restauração florestal.

Agroicone, São Paulo. Available from http://www.inputbrasil.org/wp-content/uploads/2018/03/guia-plantas-nao-desejaveis.pdf

SER Society for Ecological Restoration International Science & PolicyWorking Group (2004) The SER International Primer on Ecological Restoration. Tucson, Arizona.

Silva APM, Marques HR, Santos TVMN, Teixeira AMC, Luciano MSF, Sambuichi RHR (2015) Diagnóstico da Produção de Mudas Florestais Nativas no Brasil. IPEA, Brasília, Distrito Federal

Silva APM, Schweizer D, Marques HR, Cordeiro AM, Nascente TVM, Sambuichi RHR, et al. (2017)Can current native tree seedling production and infrastructure meet an increasing forest restoration demand in Brazil? Restoration Ecology 25:509–515

Slik JWF et al. (2015) An estimate of the number of tropical tree species. Proceedings of the National Academy of Sciences 112:7472-7477

Soares-Filho B, Rajão R, Macedo M, Carneiro A, Costa W, Coe M, et al. (2014) Cracking Brazil ' s Forest Code. Science 344:363–364

Socolar JB, Gilroy JJ, Kunin WE, Edwards DP (2016) How Should Beta-Diversity Inform Biodiversity Conservation? Trends in Ecology and Evolution 31:67–80

Soininen J, Heino J, Wang J (2018) A meta-analysis of nestedness and turnover components of beta diversity across organisms and ecosystems. Global Ecology and Biogeography 27:96–109

Suganuma MS and Durigan G (2015) Indicators of restoration success in riparian tropical forests using multiple reference ecosystems. Restoration Ecology 23:238–251

Tambosi LR, Martensen AC, Ribeiro MC, Metzger JP (2013) A framework to optimize biodiversity restoration efforts based on habitat amount and landscape connectivity. Restoration Ecology 22:169–177

R Development Core Team (2011) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria

Tilman D, Isbell F, Cowles JM (2014) Biodiversity and ecosystem functioning. Annual Review of Ecology, Evolution, and Systematics 45:471–493

Veldman JW, Overbeck GE, Negreiros D, Mahy G, Le Stradic S, Fernandes GW, et al. (2015a) Tyranny of trees in grassy biomes. Science 347:484–485

Veldman JW, Overbeck GE, Negreiros D, Mahy G, Stradic SLE, Fernandes GW, et al. (2015b) Where Tree Planting and Forest Expansion are Bad for Biodiversity and Ecosystem Services. BioScience 65:1011–1018

Viani RAG, Holl KD, Padovezi A, Strassburg BBN, Farah FT, Chaves RB, et al. (2017) Protocol for Monitoring Tropical Forest Restoration: Perspectives From the Atlantic Forest Restoration Pact in Brazil. Tropical Conservation Science 10:1–8

Viani RAG, Mello FNA, Chi IE, Brancalion PHS (2015) A new focus for ecological restoration:Management of degraded forest remnants in fragmented landscapes. GPL news November:5–9

Viani RAG and Rodrigues RR (2009) Potential of the seedling community of a forest fragment for tropical forest restoration. Scientia Agricola 66:772–779

White A, Fant J, Havens K, Skinner M, Krammer A (2018) Restoring species diversity: assessing capacity in the United States native plant industry. Restoration Ecology 26: 605-611

Wright J, Symstad A, Bullock JM, Engelhardt K, Jackson L (2009) Restoring biodiversity and ecosystem function: will an integrated approach improve results? Pages 167–177 In Naeem,S, Bunker DE, Hector A, Loreau M, Perrings C (eds.) Biodiversity, ecosystem functioning, & human wellbeing. An ecological and economic perspective. Oxford University Press

Zucchi MI, Suiji PS, Mori GM, Viana JPG, Grando C, Silvestre EA, et al. (2017) Genetic diversity of reintroduced tree populations in restoration plantations of the Brazilian Atlantic Forest. Restoration Ecology 26:694-701

APPENDICES

Appendix 1. Survey adapted from “Native plant nurseries’ evaluation guide” (IPEF, 2010), available at: http://www.ipef.br/pcsn/documentos/guia_viveiro.pdf

A. General data

Plant nursery name and address:

Phone number and email:

Designated technician (name and email):

Plant nursery type: ( ) commercial ( ) governmental ( ) NGO ( ) private

Years of existence:

Member of National Registration for Seeds and Seedlings (RENASEM)? ( ) yes ( ) no

Number of employees (current): ( ) men ( ) women

B. Genetic and Ecological quality of the seeds

Fruit or seed harvesting by:

( ) own staff ( ) independent harvesters

( ) acquisition from other nurseries ( ) seed exchange

Harvesting on:

( ) regional forest remnants ( ) urban areas

( ) other. Please, specify: _____

Within a mean distance of: _____km

If you acquire or exchange seed with other nurseries, institutions or independent harvesters, who are they and where are they located (city/region)?_____

Do you apply any seed origin control or registration? Ex: Provenance region and/or geographical coordinates, sampled site description, minimum number of individuals?

( ) yes ( ) no

Please, specify: _____

Mother trees are mapped and/or identified? ( ) yes ( ) no

C. Genetic and Ecological quality of the seedlings

Do you produce you seedlings from:

( ) seeds (____% of total production)

( ) acquired young seedlings from other nurseries (____% of total production)

If you acquire young seedlings from other nurseries, please list their names and location: _____

How do you identify your seedlings?

( ) popular name only ( ) scientific name only ( ) scientific and popular name

Do you apply any ecological classification on the species you produce?

( ) yes, successional groups (pioneer, non-pioneer)

( ) sim, planting groups (recovery and diversity)

D. Production

Annual total production (estimate):

Number of produced species (average):

Seedlings production of:

( ) natives ( ) ornamentals, fructiferous ( ) exotics (Pinus, Eucaliptus ssp. etc)

Native tree species production's purpose:

( ) sales (____% of total production)

( ) donation (____% of total production)

( ) own use (____% of total production)

( ) other (____% of total production)

Native tree species production's destination:

( ) forest restoration (____% of total production)

( ) urban afforestation (____% of total production)

( ) landscaping (____% of total production)

( ) other (____% of total production)

E. Difficulties related to native seedlings production

( ) seed acquisition

( ) lack of information about native species

( ) breaking seed dormancy, germination

( ) skilled labor

( ) structure (ex: watering system, greenhouse), equipment, supplies

( ) meet legal regulations (ex: minimum number of species, successional groups, endangered species etc)

( ) meet National Registration for Seeds and Seedlings (RENASEM) regulations

( ) sales

Appendix 2. Native species ordered by quantities available for ecological restoration in plant nurseries in Sao Paulo, Brazil (raining season 2015-2016). Functional guilds PI=pioneer, NPi=non-pioneer. CNC Flora/IUCN and SP threatened species VU=vulnerable, EN=endangered, CR=critically endangered, DD=data deficient, LC=low concern, NT=nearly threatened. Relative frequency considering 54 plant nurseries.

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Anacardiaceae	*Schinus terebinthifolia*	tree/shrub	Pi	zoo			0.93	282,009
Verbenaceae	*Citharexylum myrianthum*	tree/shrub	Pi	zoo			0.83	244,245
Euphorbiaceae	*Croton urucurana*	tree/shrub	covering Pi	mix			0.72	224,999
Malvaceae	*Ceiba speciosa*	tree/shrub	Canopy NPi	nonzoo			0.91	218,486
Malvaceae	*Guazuma ulmifolia*	tree/shrub	covering Pi	mix			0.81	217,720
Fabaceae	*Anadenanthera colubrina*	tree/shrub	Canopy NPi	nonzoo			0.63	216,681
Fabaceae	*Peltophorum dubium*	tree/shrub	Canopy NPi	nonzoo			0.78	215,670
Lythraceae	*Lafoensia pacari*	tree/shrub	Canopy NPi	mix	LC		0.93	189,163
Malvaceae	*Heliocarpus popayanensis*	tree/shrub	covering Pi	nonzoo			0.61	165,607
Phytolaccaceae	*Gallesia integrifolia*	tree/shrub	Canopy NPi	nonzoo			0.78	160,384
Bignoniaceae	*Handroanthus impetiginosus*	tree/shrub	Canopy NPi	nonzoo	NT		0.74	149,494
Fabaceae	*Inga laurina*	tree/shrub	Pi	zoo	LC		0.61	148,269
Fabaceae	*Parapiptadenia rigida*	tree/shrub	Canopy NPi	nonzoo			0.46	143,013
Fabaceae	*Inga vera subsp. affinis*	tree/shrub	Pi	mix			0.43	135,950
Bignoniaceae	*Handroanthus heptaphyllus*	tree/shrub	Canopy NPi	nonzoo	LC		0.63	133,810
Meliaceae	*Cedrela fissilis*	tree/shrub	Canopy NPi	nonzoo	VU	VU	0.85	133,407
Euphorbiaceae	*Croton floribundus*	tree/shrub	covering Pi	mix			0.67	126,025
Bignoniaceae	*Tabebuia roseoalba*	tree/shrub	Canopy NPi	nonzoo			0.76	123,572
Myrtaceae	*Eugenia uniflora*	tree/shrub	UnderStory NPi	zoo			0.85	120,929
Fabaceae	*Enterolobium contortisiliquum*	tree/shrub	Canopy NPi	mix			0.72	114,874
Lecythidaceae	*Cariniana legalis*	tree/shrub	Canopy NPi	nonzoo	EN	VU	0.72	110,922
Boraginaceae	*Cordia trichotoma*	tree/shrub	Canopy NPi	nonzoo			0.57	107,124
Polygonaceae	*Triplaris americana*	tree/shrub	Canopy NPi	nonzoo			0.54	104,443
Fabaceae	*Senna multijuga*	tree/shrub	covering Pi	mix			0.70	102,862
Fabaceae	*Inga vera*	tree/shrub	Pi	mix			0.43	97,585

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Fabaceae	*Mimosa bimucronata*	tree/shrub	nc	nonzoo			0.39	97,322
Bignoniaceae	*Handroanthus chrysotrichus*	tree/shrub	Canopy NPi	nonzoo			0.70	93,835
Anacardiaceae	*Myracrodruon urundeuva*	tree/shrub	Canopy NPi	nonzoo	LC		0.61	91,990
Bignoniaceae	*Jacaranda cuspidifolia*	tree/shrub	Canopy NPi	nonzoo			0.44	90,588
Lecythidaceae	*Cariniana estrellensis*	tree/shrub	Canopy NPi	nonzoo			0.72	89,016
Fabaceae	*Senegalia polyphylla*	tree/shrub	Canopy NPi	mix			0.54	88,361
Moraceae	*Maclura tinctoria*	tree/shrub	Canopy NPi	zoo			0.37	85,892
Fabaceae	*Anadenanthera peregrina*	tree/shrub	Canopy NPi	nonzoo			0.31	84,860
Malvaceae	*Luehea divaricata*	tree/shrub	Pi	nonzoo			0.67	78,066
Boraginaceae	*Cordia superba*	tree/shrub	Canopy NPi	zoo			0.56	76,374
Rhamnaceae	*Colubrina glandulosa*	tree/shrub	covering Pi	zoo	LC		0.61	75,859
Fabaceae	*Albizia niopoides*	tree/shrub	Canopy NPi	nonzoo	LC		0.61	75,065
Urticaceae	*Cecropia pachystachya*	tree/shrub	Pi	zoo			0.57	74,027
Fabaceae	*Hymenaea courbaril*	tree/shrub	Canopy NPi	zoo	LC		0.83	73,936
Fabaceae	*Poecilanthe parviflora*	tree/shrub	Canopy NPi	nonzoo	LC		0.59	70,258
Solanaceae	*Acnistus arborescens*	tree/shrub	covering Pi	zoo			0.19	68,204
Myrtaceae	*Psidium cattleianum*	tree/shrub	UnderStory NPi	zoo			0.76	68,089
Calophyllaceae	*Calophyllum brasiliense*	tree/shrub	Canopy NPi	zoo			0.59	67,105
Rutaceae	*Esenbeckia leiocarpa*	tree/shrub	Canopy NPi	mix	LC		0.69	64,042
Fabaceae	*Pterogyne nitens*	tree/shrub	Canopy NPi	mix	LC		0.61	61,975
Verbenaceae	*Aloysia virgata*	tree/shrub	Pi	mix			0.43	60,157
Fabaceae	*Libidibia ferrea*	tree/shrub	Canopy NPi	nonzoo			0.57	58,555
Rubiaceae	*Genipa americana*	tree/shrub	Canopy NPi	zoo	LC		0.59	58,498
Fabaceae	*Bauhinia forficata*	tree/shrub	covering Pi	mix			0.67	57,791
Fabaceae	*Myroxylon peruiferum*	tree/shrub	Canopy NPi	nonzoo	LC		0.70	57,438
Myrtaceae	*Eugenia involucrata*	tree/shrub	UnderStory NPi	zoo			0.54	57,202
Fabaceae	*Senna macranthera*	tree/shrub	covering Pi	mix			0.57	55,747
Euphorbiaceae	*Mabea fistulifera*	tree/shrub	Pi	nonzoo			0.37	55,724
Lamiaceae	*Aegiphila integrifolia*	tree/shrub	Pi	mix			0.57	54,525

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Meliaceae	*Cedrela odorata*	tree/shrub	Canopy NPi	nonzoo	VU	VU	0.35	53,463
Anacardiaceae	*Astronium graveolens*	tree/shrub	Canopy NPi	nonzoo	LC		0.44	53,111
Anacardiaceae	*Tapirira guianensis*	tree/shrub	Pi	zoo			0.30	53,071
Solanaceae	*Solanum pseudoquina*	tree/shrub	covering Pi	zoo	LC		0.20	51,526
Fabaceae	*Copaifera langsdorffii*	tree/shrub	Canopy NPi	zoo			0.70	51,193
Bignoniaceae	*Handroanthus vellosoi*	tree/shrub	Canopy NPi	nonzoo			0.43	49,516
Apocynaceae	*Tabernaemontana hystrix*	tree/shrub	covering Pi	zoo			0.43	48,717
Melastomataceae	*Pleroma granulosa*	tree/shrub	nc	nonzoo			0.56	47,139
Rutaceae	*Dictyoloma vandellianum*	tree/shrub	Canopy NPi	nonzoo			0.37	46,731
Boraginaceae	*Cordia americana*	tree/shrub	Canopy NPi	nonzoo			0.37	46,414
Fabaceae	*Inga edulis*	tree/shrub	Pi	zoo			0.41	44,816
Primulaceae	*Myrsine coriacea*	tree/shrub	Pi	zoo			0.54	44,692
Euphorbiaceae	*Alchornea glandulosa*	tree/shrub	covering Pi	zoo			0.35	44,572
Sapindaceae	*Sapindus saponaria*	tree/shrub	Canopy NPi	zoo			0.50	44,415
Bignoniaceae	*Sparattosperma leucanthum*	tree/shrub	Pi	nonzoo			0.22	42,260
Anacardiaceae	*Lithrea molleoides*	tree/shrub	Pi	zoo			0.35	39,312
Myrtaceae	*Plinia cauliflora*	tree/shrub	UnderStory NPi	zoo			0.19	38,808
Arecaceae	*Euterpe edulis*	palm	Canopy NPi	zoo	VU	VU	0.52	36,747
Moraceae	*Ficus guaranitica*	tree/shrub	Canopy NPi	zoo			0.39	36,455
Myrtaceae	*Eugenia pyriformis*	tree/shrub	UnderStory NPi	zoo			0.63	36,052
Boraginaceae	*Cordia ecalyculata*	tree/shrub	Canopy NPi	zoo			0.57	35,456
Malvaceae	*Luehea grandiflora*	tree/shrub	Canopy NPi	nonzoo			0.48	35,061
Fabaceae	*Mimosa caesalpiniifolia*	tree/shrub	Pi	nonzoo	LC		0.09	34,840
Cannabaceae	*Trema micrantha*	tree/shrub	covering Pi	mix			0.37	34,731
Fabaceae	*Schizolobium parahyba*	tree/shrub	Pi	nonzoo			0.57	34,707
Myrtaceae	*Psidium rufum*	tree/shrub	Canopy NPi	zoo			0.33	34,691
Primulaceae	*Myrsine guianensis*	tree/shrub	Canopy NPi	zoo			0.19	34,562
Bixaceae	*Bixa orellana*	tree/shrub	Pi	zoo			0.41	34,199
Bignoniaceae	*Handroanthus umbellatus*	tree/shrub	Canopy NPi	nonzoo			0.22	33,620

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Fabaceae	*Machaerium villosum*	tree/shrub	Canopy NPi	nonzoo	LC		0.24	33,478
Salicaceae	*Casearia gossypiosperma*	tree/shrub	Canopy NPi	zoo	LC		0.13	32,513
Asteraceae	*Moquiniastrum polymorphum*	tree/shrub	Canopy NPi	nonzoo			0.31	32,108
Fabaceae	*Dalbergia nigra*	tree/shrub	Canopy NPi	nonzoo	VU	CR	0.35	31,853
Phytolaccaceae	*Seguieria langsdorffii*	tree/shrub	Canopy NPi	nonzoo	LC		0.20	31,758
Fabaceae	*Inga cylindrica*	tree/shrub	Pi	zoo			0.06	30,846
Bignoniaceae	*Handroanthus ochraceus*	tree/shrub	Canopy NPi	nonzoo			0.30	30,712
Primulaceae	*Myrsine gardneriana*	tree/shrub	UnderStory NPi	zoo			0.11	30,580
Caricaceae	*Jacaratia spinosa*	tree/shrub	Pi	zoo	LC		0.43	30,218
Myrtaceae	*Psidium guineense*	tree/shrub	UnderStory NPi	zoo			0.09	29,719
Fabaceae	*Senna pendula*	tree/shrub	Pi	mix			0.13	29,153
Rubiaceae	*Alseis floribunda*	tree/shrub	Canopy NPi	nonzoo			0.02	27,230
Myrtaceae	*Eugenia brasiliensis*	tree/shrub	UnderStory NPi	zoo	LC		0.52	26,639
Bignoniaceae	*Zeyheria tuberculosa*	tree/shrub	Canopy NPi	nonzoo	VU	VU	0.37	26,479
Solanaceae	*Solanum granulosoleprosum*	tree/shrub	covering Pi	zoo	LC		0.11	26,334
Bignoniaceae	*Cybistax antisyphilitica*	tree/shrub	Canopy NPi	nonzoo			0.41	26,011
Fabaceae	*Inga marginata*	tree/shrub	Pi	zoo			0.19	25,922
Fabaceae	*Pterocarpus rohrii*	tree/shrub	Canopy NPi	nonzoo			0.52	25,915
Malvaceae	*Guazuma crinita*	tree/shrub	Pi	nonzoo			0.17	25,175
Solanaceae	*Solanum mauritianum*	tree/shrub	covering Pi	zoo			0.09	25,143
Myrtaceae	*Psidium myrtoides*	tree/shrub	UnderStory NPi	zoo			0.22	25,008
Fabaceae	*Dahlstedtia muehlbergiana*	tree/shrub	Pi	nonzoo			0.37	25,004
Rhamnaceae	*Rhamnidium elaeocarpum*	tree/shrub	Canopy NPi	zoo			0.39	24,871
Fabaceae	*Piptadenia gonoacantha*	tree/shrub	Canopy NPi	nonzoo	LC		0.44	24,139
Malvaceae	*Apeiba tibourbou*	tree/shrub	covering Pi	zoo			0.26	24,134
Lamiaceae	*Vitex megapotamica*	tree/shrub	Canopy NPi	zoo			0.35	23,801
Fabaceae	*Paubrasilia echinata*	tree/shrub	Canopy NPi	nonzoo			0.28	23,466
Arecaceae	*Syagrus romanzoffiana*	palm	Canopy NPi	zoo	LC		0.57	23,293
Salicaceae	*Casearia sylvestris*	tree/shrub	Pi	zoo			0.39	22,854

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Fabaceae	*Machaerium aculeatum*	liana	liana	nonzoo			0.22	22,677
Magnoliaceae	*Magnolia ovata*	tree/shrub	Canopy NPi	zoo	LC		0.41	22,629
Apocynaceae	*Aspidosperma polyneuron*	tree/shrub	Canopy NPi	nonzoo	NT		0.39	22,422
Fabaceae	*Erythrina speciosa*	tree/shrub	Pi	mix			0.39	22,302
Fabaceae	*Cassia leptophylla*	tree/shrub	Canopy NPi	nonzoo			0.39	22,208
Malvaceae	*Pseudobombax grandiflorum*	tree/shrub	Canopy NPi	nonzoo	LC		0.41	21,968
Phytolaccaceae	*Phytolacca dioica*	tree/shrub	Canopy NPi	zoo			0.17	21,827
Fabaceae	*Platypodium elegans*	tree/shrub	Canopy NPi	nonzoo			0.46	21,696
Fabaceae	*Lonchocarpus cultratus*	tree/shrub	Canopy NPi	nonzoo			0.37	21,639
Chrysobalanaceae	*Licania tomentosa*	tree/shrub	nc	zoo			0.28	20,734
Rutaceae	*Helietta apiculata*	tree/shrub	Canopy NPi	nonzoo			0.33	20,544
Fabaceae	*Ormosia arborea*	tree/shrub	Canopy NPi	mix			0.44	19,677
Solanaceae	*Solanum lycocarpum*	tree/shrub	covering Pi	zoo			0.17	19,667
Myrtaceae	*Psidium oblongatum*	tree/shrub	UnderStory NPi	zoo			0.04	18,822
Combretaceae	*Terminalia argentea*	tree/shrub	Canopy NPi	nonzoo	LC		0.24	18,815
Fabaceae	*Enterolobium timbouva*	tree/shrub	nc	nc			0.07	18,798
Meliaceae	*Guarea guidonia*	tree/shrub	Canopy NPi	zoo			0.28	18,194
Anacardiaceae	*Astronium fraxinifolium*	tree/shrub	Canopy NPi	nonzoo	LC		0.07	17,587
Fabaceae	*Cassia grandis*	tree/shrub	Canopy NPi	nonzoo			0.26	17,235
Fabaceae	*Poincianella pluviosa var. peltophoroides*	tree/shrub	nc	nonzoo			0.30	16,256
Fabaceae	*Bauhinia longifolia*	tree/shrub	Pi	nonzoo			0.24	15,924
Apocynaceae	*Aspidosperma cylindrocarpon*	tree/shrub	Canopy NPi	nonzoo	LC		0.43	15,770
Rosaceae	*Prunus myrtifolia*	tree/shrub	Canopy NPi	zoo			0.35	15,336
Myrtaceae	*Myrciaria trunciflora*	tree/shrub	UnderStory NPi	zoo			0.19	14,492
Solanaceae	*Solanum paniculatum*	tree/shrub	Pi	zoo			0.04	14,400
Urticaceae	*Cecropia hololeuca*	tree/shrub	Pi	zoo			0.22	14,072
Fabaceae	*Apuleia leiocarpa*	tree/shrub	Canopy NPi	nonzoo	VU	VU	0.13	13,784
Araliaceae	*Dendropanax cuneatus*	tree/shrub	Canopy NPi	zoo	LC		0.22	13,665
Bignoniaceae	*Jacaranda macrantha*	tree/shrub	Canopy NPi	nonzoo	LC		0.17	13,343

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Bignoniaceae	*Handroanthus albus*	tree/shrub	Canopy NPi	nonzoo	LC		0.06	13,086
Rubiaceae	*Simira sampaioana*	tree/shrub	Canopy NPi	nonzoo			0.11	12,452
Fabaceae	*Machaerium stipitatum*	tree/shrub	Canopy NPi	nonzoo			0.33	12,419
Boraginaceae	*Cordia sellowiana*	tree/shrub	Canopy NPi	zoo			0.28	12,288
Fabaceae	*Erythrina verna*	tree/shrub	Canopy NPi	nonzoo			0.22	11,828
Solanaceae	*Solanum variabile*	tree/shrub	Pi	zoo			0.07	11,766
Loganiaceae	*Strychnos brasiliensis*	liana/shrub	liana	zoo			0.11	11,725
Annonaceae	*Annona mucosa*	tree/shrub	nc	zoo			0.24	11,074
Fabaceae	*Hymenaea stigonocarpa*	tree/shrub	Canopy NPi	zoo			0.09	10,978
Clusiaceae	*Garcinia gardneriana*	tree/shrub	UnderStory NPi	zoo			0.22	10,977
Fabaceae	*Clitoria fairchildiana*	tree/shrub	Pi	nonzoo			0.17	10,608
Fabaceae	*Senna spectabilis*	tree/shrub	Pi	nonzoo			0.15	10,454
Apocynaceae	*Aspidosperma parvifolium*	tree/shrub	Canopy NPi	nonzoo			0.37	10,255
Fabaceae	*Machaerium nyctitans*	tree/shrub	Canopy NPi	nonzoo	LC		0.41	10,130
Moraceae	*Ficus luschnathiana*	tree/shrub	Canopy NPi	zoo			0.07	9,997
Araucariaceae	*Araucaria angustifolia*	tree/shrub	Canopy NPi	mix	EN		0.28	9,917
Malvaceae	*Bastardiopsis densiflora*	tree/shrub	covering Pi	zoo			0.07	9,742
Fabaceae	*Centrolobium tomentosum*	tree/shrub	Canopy NPi	nonzoo	LC		0.46	9,644
Rutaceae	*Balfourodendron riedelianum*	tree/shrub	Canopy NPi	nonzoo	NT		0.31	9,561
Rubiaceae	*Psychotria carthagenensis*	tree/shrub	UnderStory NPi	zoo			0.09	9,445
Fabaceae	*Holocalyx balansae*	tree/shrub	Canopy NPi	zoo			0.31	9,290
Lamiaceae	*Vitex polygama*	tree/shrub	Canopy NPi	zoo			0.20	9,288
Sapindaceae	*Allophylus edulis*	tree/shrub	Canopy NPi	zoo			0.28	9,191
Bignoniaceae	*Jacaranda micrantha*	tree/shrub	Canopy NPi	nonzoo			0.15	9,109
Malvaceae	*Luehea candicans*	tree/shrub	Canopy NPi	nonzoo	LC		0.11	9,088
Anacardiaceae	*Spondias mombin*	tree/shrub	Canopy NPi	zoo			0.13	8,920
Euphorbiaceae	*Alchornea triplinervia*	tree/shrub	covering Pi	zoo			0.20	8,894
Fabaceae	*Inga sellowiana*	tree/shrub	UnderStory NPi	zoo	NT		0.04	8,580
Myrtaceae	*Myrcia tomentosa*	tree/shrub	UnderStory NPi	zoo			0.09	8,392

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Fabaceae	*Swartzia oblata*	tree/shrub	Canopy NPi	nc			0.02	8,292
Fabaceae	*Cassia ferruginea*	tree/shrub	Pi	mix			0.22	8,122
Myrtaceae	*Campomanesia xanthocarpa*	tree/shrub	Canopy NPi	zoo	LC		0.22	8,067
Polygonaceae	*Ruprechtia laxiflora*	tree/shrub	Canopy NPi	nonzoo			0.07	8,063
Fabaceae	*Machaerium paraguariense*	tree/shrub	Canopy NPi	nonzoo	LC		0.15	7,518
Lauraceae	*Nectandra megapotamica*	tree/shrub	Canopy NPi	zoo			0.35	7,421
Malvaceae	*Eriotheca gracilipes*	tree/shrub	nc	nonzoo			0.02	7,242
Rutaceae	*Zanthoxylum riedelianum*	tree/shrub	Canopy NPi	zoo			0.13	7,153
Annonaceae	*Annona cacans*	tree/shrub	Canopy NPi	zoo	LC		0.28	7,135
Fabaceae	*Inga sessilis*	tree/shrub	Canopy NPi	zoo			0.15	6,900
Fabaceae	*Albizia polycephala*	tree/shrub	Canopy NPi	nonzoo			0.15	6,779
Fabaceae	*Chloroleucon tortum*	tree/shrub	Canopy NPi	nonzoo	NT		0.15	6,764
Fabaceae	*Leucochloron incuriale*	tree/shrub	Canopy NPi	nonzoo			0.20	6,748
Bignoniaceae	*Tabebuia aurea*	tree/shrub	Canopy NPi	nonzoo			0.15	6,673
Erythroxylaceae	*Erythroxylum argentinum*	tree/shrub	UnderStory NPi	zoo			0.02	6,601
Fabaceae	*Senna alata*	tree/shrub	Pi	nonzoo			0.09	6,507
Fabaceae	*Muellera campestris*	tree/shrub	Canopy NPi	nonzoo			0.11	6,466
Myrtaceae	*Eugenia myrcianthes*	tree/shrub	UnderStory NPi	zoo			0.22	6,444
Anacardiaceae	*Tapirira obtusa*	tree/shrub	Pi	zoo			0.06	6,381
Rubiaceae	*Posoqueria acutifolia*	tree/shrub	UnderStory NPi	zoo			0.07	6,377
Fabaceae	*Erythrina cristagalli*	tree/shrub	Pi	nonzoo			0.19	6,370
Myrtaceae	*Myrciaria glazioviana*	tree/shrub	nc	zoo			0.17	6,364
Bignoniaceae	*Tabebuia insignis*	tree/shrub	Canopy NPi	nonzoo			0.07	6,301
Fabaceae	*Platycyamus regnellii*	tree/shrub	Canopy NPi	nonzoo			0.17	6,241
Styracaceae	*Styrax ferrugineus*	tree/shrub	Canopy NPi	zoo			0.02	6,000
Bignoniaceae	*Handroanthus serratifolius*	tree/shrub	Canopy NPi	nonzoo			0.06	5,987
Apocynaceae	*Aspidosperma ramiflorum*	tree/shrub	Canopy NPi	nonzoo	LC		0.19	5,982
Sapindaceae	*Diatenopteryx sorbifolia*	tree/shrub	Canopy NPi	nonzoo			0.09	5,961
Annonaceae	*Annona montana*	tree/shrub	Canopy NPi	zoo			0.06	5,901

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Erythroxylaceae	*Erythroxylum deciduum*	tree/shrub	nc	zoo			0.09	5,851
Fabaceae	*Samanea tubulosa*	tree/shrub	Canopy NPi	nonzoo			0.15	5,815
Myrtaceae	*Eugenia florida*	tree/shrub	UnderStory NPi	zoo	LC		0.24	5,765
Myrtaceae	*Campomanesia guaviroba*	tree/shrub	Canopy NPi	zoo			0.02	5,649
Fabaceae	*Erythrina falcata*	tree/shrub	Canopy NPi	nonzoo			0.22	5,603
Rutaceae	*Zanthoxylum rugosum*	tree/shrub	Canopy NPi	nc			0.02	5,600
Sapindaceae	*Dilodendron bipinnatum*	tree/shrub	Pi	zoo	LC		0.09	5,455
Fabaceae	*Luetzelburgia auriculata*	tree/shrub	Canopy NPi	nc			0.07	5,453
Fabaceae	*Machaerium acutifolium*	tree/shrub	Canopy NPi	nonzoo			0.13	5,395
Myrtaceae	*Myrciaria dubia*	tree/shrub	UnderStory NPi	zoo			0.07	5,390
Euphorbiaceae	*Joannesia princeps*	tree/shrub	covering Pi	zoo	LC		0.28	5,202
Myrtaceae	*Psidium longipetiolatum*	tree/shrub	UnderStory NPi	zoo	LC		0.07	5,177
Moraceae	*Ficus enormis*	tree/shrub	Canopy NPi	zoo			0.09	5,131
Meliaceae	*Guarea kunthiana*	tree/shrub	Canopy NPi	zoo			0.06	5,092
Annonaceae	*Annona sylvatica*	tree/shrub	Canopy NPi	zoo			0.13	4,958
Fabaceae	*Senna occidentalis*	tree/shrub	nc	nonzoo			0.04	4,880
Lauraceae	*Ocotea puberula*	tree/shrub	Canopy NPi	zoo	NT		0.07	4,851
Rutaceae	*Esenbeckia febrifuga*	tree/shrub	UnderStory NPi	mix			0.13	4,812
Malvaceae	*Ceiba glaziovii*	tree/shrub	nc	nonzoo			0.06	4,624
Malvaceae	*Eriotheca candolleana*	tree/shrub	Canopy NPi	nonzoo			0.04	4,619
Verbenaceae	*Citharexylum solanaceum*	tree/shrub	Pi	zoo			0.06	4,589
Fabaceae	*Ateleia glazioveana*	tree/shrub	nc	nonzoo			0.11	4,562
Lauraceae	*Nectandra lanceolata*	tree/shrub	Canopy NPi	zoo			0.13	4,513
Bignoniaceae	*Jacaranda puberula*	tree/shrub	Canopy NPi	nonzoo	LC		0.15	4,474
Celastraceae	*Maytenus gonoclada*	tree/shrub	UnderStory NPi	zoo			0.04	4,446
Fabaceae	*Poincianella pluviosa*	tree/shrub	nc	nonzoo			0.02	4,413
Combretaceae	*Terminalia glabrescens*	tree/shrub	Canopy NPi	nonzoo			0.20	4,384
Fabaceae	*Machaerium brasiliense*	liana/shrub	Canopy NPi	nonzoo			0.07	4,244
Fabaceae	*Amburana cearensis*	tree/shrub	nc	nonzoo	NT		0.11	4,240

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Moraceae	*Ficus eximia*	tree/shrub	Canopy NPi	zoo	LC		0.07	4,210
Sapindaceae	*Cupania vernalis*	tree/shrub	Canopy NPi	zoo			0.22	4,182
Fabaceae	*Dalbergia brasiliensis*	tree/shrub	Canopy NPi	nonzoo			0.09	4,082
Myrtaceae	*Myrcia citrifolia*	tree/shrub	UnderStory NPi	zoo			0.04	4,080
Sapindaceae	*Magonia pubescens*	tree/shrub	Canopy NPi	nonzoo	LC		0.04	4,068
Malpighiaceae	*Galphimia brasiliensis*	sub_shrub	nc	nc			0.02	3,937
Meliaceae	*Trichilia hirta*	tree/shrub	UnderStory NPi	zoo	LC		0.06	3,761
Apocynaceae	*Aspidosperma olivaceum*	tree/shrub	Canopy NPi	nonzoo			0.04	3,638
Fabaceae	*Stryphnodendron adstringens*	tree/shrub	nc	nonzoo	LC		0.06	3,630
Vochysiaceae	*Qualea grandiflora*	tree/shrub	Canopy NPi	nonzoo			0.04	3,599
Fabaceae	*Dipteryx alata*	tree/shrub	Canopy NPi	zoo	LC		0.11	3,540
Malvaceae	*Taliparitipernambucense*	tree/shrub	Pi	nonzoo			0.09	3,536
Fabaceae	*Myrocarpus frondosus*	tree/shrub	Canopy NPi	nonzoo	LC		0.17	3,474
Phyllanthaceae	*Savia dictyocarpa*	tree/shrub	Canopy NPi	mix	LC		0.04	3,465
Lauraceae	*Nectandra grandiflora*	tree/shrub	Canopy NPi	zoo	LC		0.02	3,350
Apocynaceae	*Aspidosperma subincanum*	tree/shrub	Canopy NPi	nonzoo			0.09	3,197
Chrysobalanaceae	*Licania octandra*	tree/shrub	Canopy NPi	zoo			0.02	3,180
Fabaceae	*Albizia edwallii*	tree/shrub	Canopy NPi	nonzoo	LC		0.02	3,113
Fabaceae	*Bowdichia virgilioides*	tree/shrub	nc	nonzoo	NT		0.04	3,108
Fabaceae	*Parapiptadenia pterosperma*	tree/shrub	Canopy NPi	nonzoo			0.02	3,092
Nyctaginaceae	*Guapira opposita*	tree/shrub	UnderStory NPi	zoo			0.06	3,036
Vochysiaceae	*Vochysia tucanorum*	tree/shrub	Canopy NPi	nonzoo			0.02	3,020
Melastomataceae	*Pleroma mutabilis*	tree/shrub	nc	nonzoo			0.20	3,007
Meliaceae	*Guarea macrophylla*	tree/shrub	Canopy NPi	zoo			0.04	3,007
Meliaceae	*Trichilia clausseni*	tree/shrub	UnderStory NPi	zoo			0.07	2,946
Annonaceae	*Annona coriacea*	tree/shrub	nc	zoo	LC		0.11	2,905
Apocynaceae	*Rauvolfia sellowii*	tree/shrub	Canopy NPi	zoo			0.17	2,887
Burseraceae	*Protium heptaphyllum*	tree/shrub	Canopy NPi	zoo			0.09	2,781
Myrtaceae	*Eugenia sonderiana*	tree/shrub	UnderStory NPi	zoo			0.04	2,737

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Lythraceae	*Physocalymma scaberrimum*	tree/shrub	nc	nonzoo	LC		0.11	2,718
Lauraceae	*Persea willdenovii*	tree/shrub	Canopy NPi	zoo	LC		0.07	2,712
Peraceae	*Pera glabrata*	tree/shrub	Canopy NPi	zoo			0.11	2,691
Fabaceae	*Leptolobium elegans*	tree/shrub	Canopy NPi	nonzoo			0.06	2,610
Ebenaceae	*Diospyros inconstans*	tree/shrub	Canopy NPi	zoo	LC		0.15	2,598
Urticaceae	*Coussapoa microcarpa*	tree/shrub	Canopy NPi	zoo			0.02	2,572
Myrtaceae	*Myrciaria glomerata*	tree/shrub	nc	zoo			0.09	2,558
Myrtaceae	*Eugenia acutata*	tree/shrub	Canopy NPi	zoo			0.06	2,551
Rubiaceae	*Posoqueria latifolia*	tree/shrub	UnderStory NPi	zoo	LC		0.07	2,510
Solanaceae	*Solanum argenteum*	tree/shrub	covering Pi	zoo			0.02	2,468
Malvaceae	*Sterculia apetala*	tree/shrub	Canopy NPi	zoo			0.09	2,434
Malvaceae	*Ceiba crispiflora*	tree/shrub	nc	nc			0.02	2,390
Sapindaceae	*Cupania racemosa*	tree/shrub	Canopy NPi	zoo			0.04	2,344
Myrtaceae	*Myrcia ilheosensis*	tree/shrub	UnderStory NPi	zoo			0.02	2,322
Fabaceae	*Senna rugosa*	tree/shrub	nc	nonzoo			0.02	2,315
Solanaceae	*Vassobia breviflora*	tree/shrub	Pi	zoo			0.02	2,292
Celastraceae	*Maytenus ilicifolia*	tree/shrub	UnderStory NPi	zoo	LC	VU	0.06	2,287
Euphorbiaceae	*Gymnanthes klotzschiana*	tree/shrub	UnderStory NPi	mix			0.09	2,284
Ebenaceae	*Diospyros brasiliensis*	tree/shrub	Canopy NPi	zoo			0.02	2,250
Myrtaceae	*Eugenia candolleana*	tree/shrub	UnderStory NPi	zoo			0.06	2,244
Lauraceae	*Ocotea serrana*	tree/shrub	Canopy NPi	zoo	EN	EN	0.02	2,172
Salicaceae	*Xylosma ciliatifolia*	tree/shrub	Canopy NPi	zoo			0.04	2,166
Myrtaceae	*Campomanesia pubescens*	tree/shrub	Canopy NPi	zoo	LC		0.07	2,150
Clusiaceae	*Garcinia brasiliensis*	tree/shrub	Canopy NPi	zoo			0.04	2,144
Moraceae	*Ficus adhatodifolia*	tree/shrub	Canopy NPi	zoo			0.02	2,127
Sapindaceae	*Cupania oblongifolia*	tree/shrub	Canopy NPi	zoo			0.04	2,041
Fabaceae	*Pterodon emarginatus*	tree/shrub	Canopy NPi	nonzoo			0.06	2,026
Myrtaceae	*Plinia edulis*	tree/shrub	UnderStory NPi	zoo	VU		0.07	2,006
Calophyllaceae	*Kielmeyera coriacea*	tree/shrub	nc	nonzoo			0.02	2,000

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Fabaceae	*Enterolobium gummiferum*	tree/shrub	nc	mix			0.02	2,000
Melastomataceae	*Miconia ferruginata*	tree/shrub	nc	zoo			0.02	2,000
Melastomataceae	*Tibouchina sellowiana*	tree/shrub	Pi	nonzoo			0.02	2,000
Vochysiaceae	*Qualea multiflora*	tree/shrub	Canopy NPi	nonzoo			0.02	2,000
Rubiaceae	*Cordiera sessilis*	tree/shrub	nc	zoo			0.04	1,998
Apocynaceae	*Tabernaemontana catharinensis*	tree/shrub	Pi	zoo			0.04	1,994
Myrtaceae	*Myrcia splendens*	tree/shrub	UnderStory NPi	zoo			0.06	1,968
Euphorbiaceae	*Pleradenophora membranifolia*	tree/shrub	UnderStory NPi	nc			0.06	1,938
Primulaceae	*Myrsine umbellata*	tree/shrub	UnderStory NPi	zoo			0.11	1,910
Meliaceae	*Cabralea canjerana*	tree/shrub	Canopy NPi	zoo			0.13	1,887
Monimiaceae	*Mollinedia widgrenii*	tree/shrub	UnderStory NPi	zoo			0.04	1,883
Fabaceae	*Inga striata*	tree/shrub	Pi	zoo			0.02	1,872
Lamiaceae	*Aegiphila verticillata*	tree/shrub	Pi	zoo			0.04	1,870
Annonaceae	*Annona glabra*	tree/shrub	nc	zoo	LC		0.06	1,865
Rubiaceae	*Randia armata*	liana/shrub	liana	zoo			0.07	1,836
Moraceae	*Ficus organensis*	tree/shrub	Canopy NPi	zoo			0.04	1,807
Myrtaceae	*Eugenia luschnathiana*	tree/shrub	UnderStory NPi	zoo			0.09	1,773
Asteraceae	*Vernonanthura polyanthes*	tree/shrub	Pi	nonzoo			0.06	1,752
Bignoniaceae	*Jacaranda caroba*	tree/shrub	Canopy NPi	nonzoo			0.02	1,725
Fabaceae	*Dimorphandra mollis*	tree/shrub	nc	nonzoo			0.04	1,716
Annonaceae	*Duguetia lanceolata*	tree/shrub	Canopy NPi	zoo	LC		0.09	1,714
Styracaceae	*Styrax pohlii*	tree/shrub	Canopy NPi	zoo			0.02	1,700
Lauraceae	*Cryptocarya aschersoniana*	tree/shrub	Canopy NPi	zoo			0.07	1,663
Annonaceae	*Annona crassiflora*	tree/shrub	nc	zoo			0.07	1,662
Rutaceae	*Zanthoxylum rhoifolium*	tree/shrub	Canopy NPi	zoo			0.13	1,659
Rubiaceae	*Psychotria vellosiana*	tree/shrub	UnderStory NPi	zoo			0.02	1,644
Lauraceae	*Ocotea odorifera*	tree/shrub	Canopy NPi	zoo	EN	EN	0.04	1,640
Myrtaceae	*Myrcia multiflora*	tree/shrub	UnderStory NPi	zoo			0.06	1,635
Rubiaceae	*Cordiera macrophylla*	tree/shrub	UnderStory NPi	zoo			0.04	1,600

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Fabaceae	*Vatairea macrocarpa*	tree/shrub	Canopy NPi	nonzoo			0.04	1,580
Meliaceae	*Trichilia pallida*	tree/shrub	UnderStory NPi	zoo			0.07	1,518
Fabaceae	*Chamaecrista debilis*	tree/shrub	nc	nc			0.02	1,500
Meliaceae	*Trichilia catigua*	tree/shrub	UnderStory NPi	zoo			0.04	1,500
Lecythidaceae	*Lecythis pisonis*	tree/shrub	Canopy NPi	zoo			0.17	1,485
Fabaceae	*Calliandra brevipes*	tree/shrub	nc	nonzoo			0.04	1,478
Myrtaceae	*Eugenia francavilleana*	tree/shrub	UnderStory NPi	zoo			0.02	1,470
Fabaceae	*Chloroleucon tenuiflorum*	tree/shrub	Canopy NPi	nonzoo			0.02	1,440
Melastomataceae	*Miconia cinerascens*	tree/shrub	Pi	zoo			0.02	1,400
Fabaceae	*Senna silvestris*	tree/shrub	Pi	mix			0.04	1,394
Myrtaceae	*Plinia peruviana*	tree/shrub	UnderStory NPi	zoo			0.04	1,365
Malvaceae	*Sterculia striata*	tree/shrub	Canopy NPi	zoo			0.06	1,342
Fabaceae	*Plathymenia reticulata*	tree/shrub	Canopy NPi	nonzoo	LC		0.04	1,330
Lauraceae	*Ocotea silvestris*	tree/shrub	Canopy NPi	zoo	LC		0.02	1,323
Bignoniaceae	*Jacaranda brasiliana*	tree/shrub	Canopy NPi	nonzoo			0.06	1,307
Bignoniaceae	*Paratecoma peroba*	tree/shrub	Canopy NPi	nonzoo	EN		0.02	1,300
Myrtaceae	*Eugenia itaguahiensis*	tree/shrub	UnderStory NPi	zoo			0.02	1,300
Winteraceae	*Drimys brasiliensis*	tree/shrub	Canopy NPi	zoo	LC		0.02	1,300
Apocynaceae	*Aspidosperma australe*	tree/shrub	Canopy NPi	nonzoo	LC		0.04	1,294
Myrtaceae	*Campomanesia guazumifolia*	tree/shrub	Canopy NPi	zoo			0.07	1,270
Clusiaceae	*Clusia criuva*	tree/shrub	Canopy NPi	zoo	LC		0.02	1,260
Solanaceae	*Solanum caavurana*	tree/shrub	Pi	zoo			0.02	1,248
Melastomataceae	*Pleroma fissinervia*	tree/shrub	nc	nonzoo			0.02	1,226
Rutaceae	*Metrodorea stipularis*	tree/shrub	Canopy NPi	mix			0.02	1,201
Fabaceae	*Machaerium scleroxylon*	tree/shrub	Canopy NPi	nonzoo			0.04	1,195
Myrtaceae	*Calyptranthes clusiifolia*	tree/shrub	Canopy NPi	zoo			0.04	1,189
Sapotaceae	*Pouteria torta*	tree/shrub	nc	zoo	LC		0.09	1,180
Fabaceae	*Erythrina velutina*	tree/shrub	nc	nonzoo			0.06	1,170
Fabaceae	*Senna obtusifolia*	tree/shrub	nc	nonzoo			0.02	1,164

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Aquifoliaceae	*Ilex cerasifolia*	tree/shrub	UnderStory NPi	zoo			0.06	1,137
Myrtaceae	*Eugenia sulcata*	tree/shrub	Canopy NPi	zoo			0.02	1,136
Euphorbiaceae	*Micrandra elata*	tree/shrub	Canopy NPi	nonzoo			0.04	1,126
Monimiaceae	*Mollinedia uleana*	tree/shrub	UnderStory NPi	zoo			0.04	1,088
Melastomataceae	*Pleroma candolleana*	tree/shrub	nc	nonzoo			0.02	1,072
Melastomataceae	*Miconia pusilliflora*	tree/shrub	UnderStory NPi	zoo			0.04	1,062
Clethraceae	*Clethra scabra*	tree/shrub	Canopy NPi	nonzoo	LC		0.04	1,057
Sabiaceae	*Meliosma sellowii*	tree/shrub	Canopy NPi	zoo			0.02	1,045
Moraceae	*Sorocea bonplandii*	tree/shrub	Canopy NPi	zoo			0.02	1,037
Fabaceae	*Senna cana*	tree/shrub	nc	nonzoo			0.02	1,000
Sapotaceae	*Chrysophyllum gonocarpum*	tree/shrub	Canopy NPi	zoo			0.06	986
Urticaceae	*Cecropia glaziovii*	tree/shrub	Pi	zoo			0.13	966
Rubiaceae	*Guettarda viburnoides*	tree/shrub	UnderStory NPi	zoo			0.06	960
Malpighiaceae	*Byrsonima laxiflora*	tree/shrub	Pi	zoo			0.02	953
Malvaceae	*Ceiba erianthos*	tree/shrub	nc	nonzoo			0.02	940
Euphorbiaceae	*Mabea piriri*	tree/shrub	Pi	nonzoo			0.04	924
Meliaceae	*Trichilia casaretti*	tree/shrub	UnderStory NPi	zoo	LC		0.02	906
Bignoniaceae	*Pyrostegia venusta*	liana	liana	nonzoo			0.02	900
Fabaceae	*Dalbergia miscolobium*	tree/shrub	nc	nonzoo			0.06	881
Moraceae	*Ficus obtusifolia*	tree/shrub	Canopy NPi	zoo			0.04	869
Myrtaceae	*Eugenia brevistyla*	tree/shrub	UnderStory NPi	zoo	LC		0.02	864
Arecaceae	*Mauritia flexuosa*	palm	nc	zoo		VU	0.06	850
Annonaceae	*Annona neosericea*	tree/shrub	nc	zoo			0.04	847
Fabaceae	*Vachellia farnesiana*	tree/shrub	nc	nc			0.04	835
Sapindaceae	*Matayba guianensis*	tree/shrub	Canopy NPi	zoo			0.04	833
Sapindaceae	*Matayba elaeagnoides*	tree/shrub	Canopy NPi	zoo			0.07	823
Myrtaceae	*Eugenia monosperma*	tree/shrub	Canopy NPi	zoo			0.02	802
Sapotaceae	*Pouteria caimito*	tree/shrub	Canopy NPi	zoo			0.09	795
Sapotaceae	*Chrysophyllum marginatum*	tree/shrub	Canopy NPi	zoo			0.04	787

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Meliaceae	*Trichilia pallens*	tree/shrub	UnderStory NPi	zoo	LC		0.02	786
Sapotaceae	*Pouteria gardneri*	tree/shrub	Canopy NPi	zoo			0.02	770
Lauraceae	*Endlicheria paniculata*	tree/shrub	UnderStory NPi	zoo			0.02	756
Lauraceae	*Ocotea divaricata*	tree/shrub	Canopy NPi	zoo			0.02	720
Myrtaceae	*Calyptranthes concinna*	tree/shrub	UnderStory NPi	zoo	LC		0.04	700
Asteraceae	*Stifftia parviflora*	tree/shrub	Canopy NPi	nonzoo	DD		0.02	688
Sapotaceae	*Pouteria ramiflora*	tree/shrub	nc	zoo			0.04	672
Euphorbiaceae	*Alchornea sidifolia*	tree/shrub	covering Pi	zoo			0.04	652
Monimiaceae	*Mollinedia schottiana*	tree/shrub	UnderStory NPi	zoo			0.02	648
Euphorbiaceae	*Sapium glandulosum*	tree/shrub	Canopy NPi	zoo			0.07	634
Lecythidaceae	*Gustavia augusta*	tree/shrub	Canopy NPi	nc			0.02	626
Rubiaceae	*Guettarda uruguensis*	tree/shrub	UnderStory NPi	zoo			0.04	619
Lauraceae	*Beilschmiedia emarginata*	tree/shrub	Canopy NPi	zoo			0.02	611
Myrtaceae	*Plinia rivularis*	tree/shrub	UnderStory NPi	zoo			0.04	603
Apocynaceae	*Hancornia speciosa*	tree/shrub	Canopy NPi	mix			0.02	600
Moraceae	*Ficus pertusa*	tree/shrub	Canopy NPi	zoo			0.02	600
Lauraceae	*Ocotea velutina*	tree/shrub	Canopy NPi	zoo			0.06	596
Melastomataceae	*Miconia chamissois*	tree/shrub	nc	zoo			0.04	595
Malvaceae	*Eriotheca pentaphylla*	tree/shrub	Canopy NPi	nonzoo			0.02	585
Anacardiaceae	*Astronium concinnum*	tree/shrub	Canopy NPi	nonzoo			0.04	552
Fabaceae	*Dalbergia frutescens*	liana/shrub	liana	nonzoo			0.04	550
Combretaceae	*Terminalia triflora*	tree/shrub	Canopy NPi	nonzoo			0.13	546
Myrtaceae	*Campomanesia hirsuta*	tree/shrub	Canopy NPi	zoo	EN		0.02	545
Celastraceae	*Maytenus subalata*	tree/shrub	UnderStory NPi	zoo			0.02	533
Fabaceae	*Calliandra tweedii*	tree/shrub	Pi	nonzoo			0.02	515
Myrtaceae	*Pimenta pseudocaryophyllus*	tree/shrub	UnderStory NPi	zoo			0.04	501
Fabaceae	*Machaerium hirtum*	tree/shrub	Canopy NPi	nonzoo			0.02	500
Fabaceae	*Luetzelburgia guaissara*	tree/shrub	Canopy NPi	nonzoo	LC		0.02	493
Malvaceae	*Luehea paniculata*	tree/shrub	Canopy NPi	nonzoo			0.02	490

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Euphorbiaceae	*Sebastiania brasiliensis*	tree/shrub	UnderStory NPi	mix			0.02	481
Boraginaceae	*Cordia magnoliifolia*	tree/shrub	Canopy NPi	zoo			0.02	474
Rubiaceae	*Randia ferox*	tree/shrub	UnderStory NPi	zoo			0.02	465
Nyctaginaceae	*Bougainvillea glabra*	liana/shrub	liana	nonzoo			0.04	460
Boraginaceae	*Cordia alliodora*	tree/shrub	nc	nc			0.02	450
Euphorbiaceae	*Tetrorchidium rubrivenium*	tree/shrub	Canopy NPi	zoo	LC		0.02	450
Rubiaceae	*Coutarea hexandra*	tree/shrub	UnderStory NPi	nonzoo			0.02	450
Lauraceae	*Cryptocarya micrantha*	tree/shrub	Canopy NPi	zoo		VU	0.04	446
Fabaceae	*Zollernia ilicifolia*	tree/shrub	Canopy NPi	zoo			0.02	436
Malpighiaceae	*Byrsonima basiloba*	tree/shrub	nc	zoo			0.02	432
Myrtaceae	*Marlierea eugeniopsoides*	tree/shrub	Canopy NPi	zoo			0.02	432
Primulaceae	*Myrsine parvifolia*	tree/shrub	Pi	zoo			0.04	421
Myrtaceae	*Myrciaria delicatula*	tree/shrub	nc	zoo			0.02	410
Calophyllaceae	*Kielmeyera variabilis*	sub_shrub	nc	nonzoo			0.02	400
Erythroxylaceae	*Erythroxylum suberosum*	tree/shrub	nc	zoo			0.02	400
Fabaceae	*Leptolobium dasycarpum*	tree/shrub	Canopy NPi	nonzoo			0.02	400
Melastomataceae	*Miconia flammea*	tree/shrub	nc	zoo			0.02	394
Asteraceae	*Stifftia chrysantha*	tree/shrub	Canopy NPi	nonzoo			0.02	387
Myrtaceae	*Eugenia pruinosa*	tree/shrub	UnderStory NPi	zoo	EN		0.02	383
Fabaceae	*Copaifera trapezifolia*	tree/shrub	Canopy NPi	zoo			0.02	379
Sapotaceae	*Chrysophyllum viride*	tree/shrub	Canopy NPi	zoo	NT		0.04	369
Myrtaceae	*Eugenia dysenterica*	tree/shrub	Canopy NPi	zoo			0.06	360
Proteaceae	*Roupala montana*	tree/shrub	nc	mix			0.04	360
Anacardiaceae	*Anacardium occidentale*	tree/shrub	nc	zoo			0.07	350
Lauraceae	*Ocotea corymbosa*	tree/shrub	Canopy NPi	zoo			0.02	350
Fabaceae	*Platymiscium floribundum*	tree/shrub	Canopy NPi	nonzoo			0.02	338
Arecaceae	*Syagrus hoehnei*	palm	nc	zoo			0.02	337
Fabaceae	*Sweetia fruticosa*	tree/shrub	Canopy NPi	nonzoo			0.02	330
Myrtaceae	*Myrcia obovata*	tree/shrub	Canopy NPi	zoo	LC		0.02	329

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Fabaceae	*Dalbergia villosa*	tree/shrub	nc	nonzoo			0.04	325
Araliaceae	*Schefflera morototoni*	tree/shrub	Pi	zoo			0.07	310
Myrtaceae	*Eugenia mattosii*	tree/shrub	UnderStory NPi	zoo	EN		0.02	310
Myrtaceae	*Myrciaria tenella*	tree/shrub	nc	zoo	DD		0.04	307
Araliaceae	*Schefflera vinosa*	tree/shrub	nc	zoo			0.02	300
Erythroxylaceae	*Erythroxylum cuneifolium*	tree/shrub	UnderStory NPi	zoo			0.02	300
Nyctaginaceae	*Neea theifera*	tree/shrub	nc	zoo			0.02	300
Phyllanthaceae	*Hyeronima alchorneoides*	tree/shrub	covering Pi	zoo			0.02	299
Fabaceae	*Andira anthelmia*	tree/shrub	Canopy NPi	zoo			0.06	290
Fabaceae	*Mimosa tenuiflora*	tree/shrub	Pi	nonzoo			0.02	280
Lamiaceae	*Vitex cymosa*	tree/shrub	Canopy NPi	zoo			0.04	280
Myrtaceae	*Acca sellowiana*	tree/shrub	Canopy NPi	zoo			0.04	278
Rhamnaceae	*Rhamnus sphaerosperma*	tree/shrub	Canopy NPi	zoo			0.02	270
Urticaceae	*Urera baccifera*	tree/shrub	Pi	zoo			0.02	256
Sapindaceae	*Dodonaea viscosa*	tree/shrub	Canopy NPi	nonzoo			0.02	252
Myrtaceae	*Campomanesia eugenioides*	tree/shrub	UnderStory NPi	zoo	LC		0.02	250
Euphorbiaceae	*Sapium haematospermum*	tree/shrub	Canopy NPi	zoo			0.02	240
Annonaceae	*Xylopia aromatica*	tree/shrub	Canopy NPi	zoo	LC		0.06	237
Celastraceae	*Maytenus evonymoides*	tree/shrub	UnderStory NPi	zoo			0.02	236
Solanaceae	*Solanum crinitum*	tree/shrub	Pi	zoo			0.02	234
Malvaceae	*Eriotheca pubescens*	tree/shrub	nc	zoo	LC	VU	0.02	224
Myrtaceae	*Campomanesia phaea*	tree/shrub	Canopy NPi	zoo	LC		0.06	220
Boraginaceae	*Cordia glabrata*	tree/shrub	UnderStory NPi	mix			0.02	216
Rubiaceae	*Ixora gardneriana*	tree/shrub	UnderStory NPi	zoo			0.02	216
Myrtaceae	*Psidium ovale*	tree/shrub	Canopy NPi	zoo	LC		0.02	209
Myrtaceae	*Myrcianthes pungens*	tree/shrub	UnderStory NPi	zoo	LC		0.06	203
Bixaceae	*Cochlospermum regium*	tree/shrub	covering Pi	nonzoo	LC		0.02	200
Caryocaraceae	*Caryocar brasiliense*	tree/shrub	nc	zoo	LC		0.02	200
Myrtaceae	*Psidium striatulum*	tree/shrub	UnderStory NPi	zoo			0.02	200

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Vochysiaceae	*Qualea jundiahy*	tree/shrub	Canopy NPi	nonzoo			0.02	200
Vochysiaceae	*Qualea parviflora*	tree/shrub	Canopy NPi	nonzoo			0.02	200
Lauraceae	*Nectandra barbellata*	tree/shrub	Canopy NPi	zoo	VU	VU	0.02	177
Fabaceae	*Guibourtia hymenaeifolia*	tree/shrub	nc	nc			0.02	170
Malpighiaceae	*Byrsonima verbascifolia*	tree/shrub	nc	zoo			0.02	169
Erythroxylaceae	*Erythroxylum ambiguum*	tree/shrub	UnderStory NPi	zoo	LC		0.02	162
Sapotaceae	*Manilkara salzmannii*	tree/shrub	Canopy NPi	zoo			0.02	156
Rutaceae	*Esenbeckia grandiflora*	tree/shrub	Canopy NPi	mix			0.09	155
Chrysobalanaceae	*Licania apetala*	tree/shrub	nc	zoo			0.02	150
Euphorbiaceae	*Croton piptocalyx*	tree/shrub	covering Pi	nonzoo			0.02	150
Solanaceae	*Cestrum corymbosum*	tree/shrub	Pi	zoo			0.02	150
Salicaceae	*Casearia decandra*	tree/shrub	Canopy NPi	zoo			0.04	146
Caricaceae	*Vasconcellea quercifolia*	tree/shrub	Pi	zoo			0.02	143
Clusiaceae	*Tovomitopsis paniculata*	tree/shrub	Canopy NPi	zoo			0.02	139
Fabaceae	*Tachigali denudata*	tree/shrub	Canopy NPi	nonzoo	NT		0.02	139
Asteraceae	*Piptocarpha rotundifolia*	tree/shrub	Canopy NPi	nonzoo			0.02	130
Euphorbiaceae	*Pachystroma longifolium*	tree/shrub	Canopy NPi	nonzoo			0.02	130
Euphorbiaceae	*Croton macrobothrys*	tree/shrub	covering Pi	nonzoo			0.02	121
Arecaceae	*Syagrus oleracea*	palm	Canopy NPi	zoo			0.04	120
Fabaceae	*Cyclolobium brasiliense*	tree/shrub	Canopy NPi	nonzoo			0.07	115
Myrtaceae	*Campomanesia neriiflora*	tree/shrub	Canopy NPi	zoo	LC		0.06	114
Moraceae	*Ficus catappifolia*	tree/shrub	Canopy NPi	nc	LC		0.02	110
Fabaceae	*Mimosa laticifera*	tree/shrub	Pi	nonzoo			0.02	108
Polygonaceae	*Ruprechtia exploratricis*	tree/shrub	Canopy NPi	nc			0.02	104
Ochnaceae	*Ouratea spectabilis*	tree/shrub	nc	zoo	LC		0.02	100
Annonaceae	*Annona rugulosa*	tree/shrub	Pi	zoo			0.02	98
Araliaceae	*Dendropanax monogynus*	tree/shrub	UnderStory NPi	zoo			0.02	90
Elaeocarpaceae	*Sloanea hirsuta*	tree/shrub	Canopy NPi	zoo	LC		0.02	90
Moraceae	*Ficus gomelleira*	tree/shrub	Canopy NPi	zoo			0.02	90

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Fabaceae	*Dalbergia decipularis*	tree/shrub	Canopy NPi	nc			0.02	85
Fabaceae	*Swartzia flaemingii*	tree/shrub	Canopy NPi	zoo	LC		0.02	85
Fabaceae	*Sesbania virgata*	tree/shrub	Pi	nonzoo			0.02	84
Fabaceae	*Piptadenia paniculata*	tree/shrub	Canopy NPi	nonzoo			0.02	82
Fabaceae	*Zygia selloi*	tree/shrub	Canopy NPi	nc			0.02	80
Myrtaceae	*Eugenia neoverrucosa*	tree/shrub	Canopy NPi	zoo			0.02	79
Sapindaceae	*Cupania tenuivalvis*	tree/shrub	Canopy NPi	zoo			0.02	75
Myrtaceae	*Blepharocalyx salicifolius*	tree/shrub	UnderStory NPi	zoo	LC		0.02	74
Fabaceae	*Lonchocarpus latifolius*	tree/shrub	Canopy NPi	nc			0.04	71
Melastomataceae	*Miconia albicans*	tree/shrub	Pi	zoo			0.02	71
Lauraceae	*Ocotea pulchella*	tree/shrub	Canopy NPi	zoo	LC		0.02	70
Lauraceae	*Nectandra leucantha*	tree/shrub	Canopy NPi	zoo			0.02	68
Solanaceae	*Cestrum intermedium*	tree/shrub	Pi	zoo			0.02	64
Arecaceae	*Syagrus picrophylla*	palm	nc	nc	VU		0.02	63
Fabaceae	*Stryphnodendron rotundifolium*	tree/shrub	Canopy NPi	nonzoo			0.02	63
Connaraceae	*Connarus regnellii*	tree/shrub	nc	zoo			0.02	61
Arecaceae	*Attalea phalerata*	palm	nc	zoo	LC		0.02	56
Opiliaceae	*Agonandra brasiliensis*	tree/shrub	Canopy NPi	zoo			0.02	54
Lauraceae	*Nectandra oppositifolia*	tree/shrub	Canopy NPi	zoo			0.02	53
Apocynaceae	*Himatanthus obovatus*	tree/shrub	Pi	nonzoo			0.02	50
Cardiopteridaceae	*Citronella gongonha*	tree/shrub	UnderStory NPi	zoo			0.02	50
Fabaceae	*Andira humilis*	tree/shrub	nc	zoo			0.02	50
Myrtaceae	*Psidium firmum*	tree/shrub	Canopy NPi	zoo			0.02	50
Rubiaceae	*Amaioua guianensis*	tree/shrub	Canopy NPi	zoo			0.02	50
Fabaceae	*Mimosa scabrella*	tree/shrub	Pi	mix			0.04	40
Malvaceae	*Helicteres lhotzkyana*	tree/shrub	Pi	nonzoo			0.04	38
Phyllanthaceae	*Margaritaria nobilis*	tree/shrub	Canopy NPi	zoo	LC		0.02	32
Annonaceae	*Annona dolabripetala*	tree/shrub	nc	zoo			0.02	31
Malvaceae	*Callianthe rufinerva*	tree/shrub	nc	nc			0.02	31

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Apocynaceae	*Aspidosperma tomentosum*	tree/shrub	Canopy NPi	mix	LC		0.02	30
Araliaceae	*Schefflera macrocarpa*	tree/shrub	Pi	zoo			0.02	30
Thymelaeaceae	*Daphnopsis fasciculata*	tree/shrub	Canopy NPi	zoo			0.02	30
Melastomataceae	*Tibouchina multiceps*	tree/shrub	nc	nc			0.02	24
Piperaceae	*Piper aduncum*	tree/shrub	UnderStory NPi	zoo			0.02	24
Combretaceae	*Buchenavia tetraphylla*	tree/shrub	Canopy NPi	zoo			0.02	20
Euphorbiaceae	*Manihot caerulescens*	tree/shrub	nc	mix			0.02	20
Lacistemataceae	*Lacistema hasslerianum*	tree/shrub	UnderStory NPi	zoo			0.02	20
Lauraceae	*Cryptocarya mandioccana*	tree/shrub	Canopy NPi	zoo			0.02	20
Malpighiaceae	*Byrsonima sericea*	tree/shrub	covering Pi	zoo			0.02	20
Malvaceae	*Pseudobombax longiflorum*	tree/shrub	Canopy NPi	nonzoo			0.02	20
Myrtaceae	*Eugenia livida*	tree/shrub	UnderStory NPi	zoo			0.02	20
Rutaceae	*Zanthoxylum caribaeum*	tree/shrub	Canopy NPi	zoo			0.02	20
Fabaceae	*Bauhinia pentandra*	tree/shrub	nc	nc			0.02	19
Asteraceae	*Raulinoreitzia crenulata*	tree/shrub	Pi	nonzoo			0.02	18
Fabaceae	*Cenostigma macrophyllum*	tree/shrub	nc	nc			0.02	18
Annonaceae	*Guatteria australis*	tree/shrub	UnderStory NPi	zoo	LC		0.02	17
Melastomataceae	*Miconia ligustroides*	tree/shrub	nc	zoo			0.02	17
Aquifoliaceae	*Ilex paraguariensis*	tree/shrub	UnderStory NPi	zoo	LC		0.02	12
Myrtaceae	*Plinia phitrantha*	tree/shrub	UnderStory NPi	zoo			0.02	12
Lecythidaceae	*Eschweilera nana*	tree/shrub	Canopy NPi	nc			0.02	10
Rutaceae	*Zanthoxylum acuminatum*	tree/shrub	Canopy NPi	zoo			0.02	10
Fabaceae	*Inga vulpina*	tree/shrub	Pi	zoo			0.02	5
Melastomataceae	*Miconia cabucu*	tree/shrub	covering Pi	zoo			0.02	5
Lauraceae	*Ocotea teleiandra*	tree/shrub	Canopy NPi	zoo			0.02	2
Apocynaceae	*Aspidosperma riedelii*	tree/shrub	Canopy NPi	nonzoo	LC	EN	0.02	1
Bignoniaceae	*Tabebuia cassinoides*	tree/shrub	nc	nonzoo	EN	EN	0.04	1
Celastraceae	*Maytenus aquifolia*	tree/shrub	UnderStory NPi	zoo	LC		0.02	1
Fabaceae	*Bauhinia ungulata*	tree/shrub	nc	nonzoo			0.02	1

Family	Species (561)	Growth form	Func. Guilds	dispersion	IUCN_threat	SP_threat	Relative F	quantity
Fabaceae	*Tachigali rugosa*	tree/shrub	Canopy NPi	nonzoo	NT		0.02	1
Lauraceae	*Ocotea catharinensis*	tree/shrub	Canopy NPi	zoo	VU	VU	0.02	1
Melastomataceae	*Miconia cinnamomifolia*	tree/shrub	covering Pi	zoo			0.02	1
Melastomataceae	*Tibouchina pulchra*	tree/shrub	Pi	nonzoo			0.02	1
Myrtaceae	*Campomanesia sessiliflora*	tree/shrub	UnderStory NPi	zoo	LC		0.02	1
Myrtaceae	*Psidium humile*	tree/shrub	UnderStory NPi	zoo			0.02	1
Solanaceae	*Solanum swartzianum*	tree/shrub	covering Pi	zoo			0.02	1

Appendix 3. Exotic species ordered by quantities available for ecological restoration in plant nurseries in Sao Paulo, Brazil (raining season 2015-2016). Exotic type according to Flora do Brasil 2020 EXO_cult = cultivated, EXO_nat = naturalized, Exotic_reg = regional, that is, species that are native to Brazil but do not occur in Sao Paulo state. Relative frequency considering 54 plant nurseries. ** indicate species that should be avoided in restoration projetcs according to Sartorelli et al. (2018).

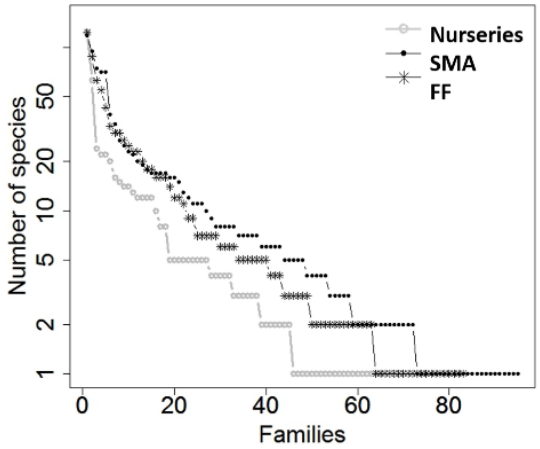
Family	Species (126)	Growth form	EXO type	Relative F	Quantity
Myrtaceae	*Psidium guajava*	tree/shrub	EXO_nat	0.65	113,739
Lythraceae	*Lafoensia glyptocarpa*	tree/shrub	EXO_reg	0.63	55,333
Anacardiaceae	*Schinus molle*	tree/shrub	EXO_reg	0.48	28,065
Malvaceae	*Pachira glabra*	tree/shrub	EXO_nat	0.37	21,651
Malvaceae	*Ochroma pyramidale*	tree/shrub	EXO_reg	0.11	20,093
Moraceae	*Ficus insipida*	tree/shrub	EXO_reg	0.28	18,800
Malvaceae	*Pachira aquatica*	tree/shrub	EXO_reg	0.22	18,522
Meliaceae	*Swietenia macrophylla*	tree/shrub	EXO_reg	0.17	16,738
Malvaceae	*Basiloxylon brasiliensis*	tree/shrub	EXO_reg	0.17	13,476
Polygonaceae	*Triplaris weigeltiana*	tree/shrub	EXO_reg	0.09	12,801
Arecaceae	*Roystonea oleracea*	palm	EXO	0.07	12,687
Bignoniaceae	*Jacaranda mimosifolia*	tree/shrub	EXO	0.09	12,062
Arecaceae	*Euterpe oleracea*	palm	EXO_reg	0.13	12,031
Fabaceae	*Senna bicapsularis*	tree/shrub	EXO	0.13	8,345
Rubiaceae	*Calycophyllum spruceanum*	tree/shrub	EXO_reg	0.13	7,688
Bignoniaceae	*Tabebuia rosea*	tree/shrub	EXO_cult	0.04	6,810
Rhamnaceae	*Frangula purshiana*	tree/shrub	EXO	0.02	6,800
Lecythidaceae	*Couroupita guianensis*	tree/shrub	EXO_reg	0.09	5,766
Fabaceae	*Tamarindus indica*	tree/shrub	EXO_cult	0.07	5,198
Fabaceae	*Caesalpinia spinosa*	tree/shrub	EXO	0.04	5,048
Malpighiaceae	*Malpighia emarginata*	tree/shrub	EXO_cult	0.06	4,764
Fabaceae	*Schizolobium parahyba var. amazonicum*	tree/shrub	EXO_reg	0.04	4,586
Malvaceae	*Ceiba boliviana*	tree/shrub	EXO	0.04	4,330
Rutaceae	*Murraya paniculata*	tree/shrub	EXO	0.04	4,167
Myrtaceae	***Syzygium cumini *****	tree/shrub	EXO_nat	0.07	3,830

Family	Species (126)	Growth form	EXO type	Relative F	Quantity
Lythraceae	*Punica granatum*	tree/shrub	EXO_cult	0.04	3,373
Sapindaceae	*Koelreuteria paniculata*	tree/shrub	EXO	0.04	3,014
Meliaceae	*Toona ciliata*	tree/shrub	EXO_cult	0.02	2,858
Meliaceae	*Khaya ivorensis*	tree/shrub	EXO	0.04	2,736
Solanaceae	*Solanum grandiflorum*	tree/shrub	EXO	0.02	2,600
Meliaceae	***Azadirachta indica*** **	tree/shrub	EXO_cult	0.04	2,338
Moraceae	*Morus nigra*	tree/shrub	EXO_cult	0.11	1,925
Rosaceae	***Eriobotrya japonica*** **	tree/shrub	EXO_nat	0.06	1,899
Rhamnaceae	***Hovenia dulcis*** **	tree/shrub	EXO_nat	0.06	1,833
Fabaceae	*Tipuana tipu*	tree/shrub	EXO_cult	0.07	1,824
Sapindaceae	*Koelreuteria bipinnata*	tree/shrub	EXO	0.04	1,817
Sapotaceae	*Mimusops coriacea*	tree/shrub	EXO_cult	0.02	1,764
Lamiaceae	*Calicarpa reevesii*	tree/shrub	EXO	0.04	1,640
Malvaceae	*Ceiba samauma*	tree/shrub	EXO_reg	0.02	1,625
Rhamnaceae	*Ziziphus joazeiro*	tree/shrub	EXO_reg	0.04	1,600
Oleaceae	***Ligustrum lucidum*** **	tree/shrub	EXO	0.02	1,500
Myrtaceae	*Callistemon viminalis*	tree/shrub	EXO	0.04	1,409
Myrtaceae	*Syzygium malaccense*	tree/shrub	EXO	0.04	1,300
Magnoliaceae	*Magnolia grandiflora*	tree/shrub	EXO	0.02	1,275
Annonaceae	*Annona muricata*	tree/shrub	EXO_reg	0.09	1,246
Fabaceae	*Cassia fistula*	tree/shrub	EXO	0.02	1,196
Fabaceae	***Albizia lebbeck*** **	tree/shrub	EXO	0.04	1,175
Fabaceae	*Parkinsonia aculeata*	tree/shrub	EXO_nat	0.04	1,175
Fabaceae	*Bauhinia variegata*	tree/shrub	EXO	0.02	1,160
Urticaceae	*Cecropia sciadophylla*	tree/shrub	EXO_reg	0.04	1,146
Moraceae	***Artocarpus heterophyllus*** **	tree/shrub	EXO_nat	0.02	1,107
Lauraceae	*Cinnamomum verum*	tree/shrub	EXO	0.02	1,100
Lythraceae	*Lagerstroemia indica*	tree/shrub	EXO	0.02	1,072
Malvaceae	*Hibiscus rosa-sinensis*	tree/shrub	EXO	0.04	965

Family	Species (126)	Growth form	EXO type	Relative F	Quantity
Myrtaceae	*Eugenia glazioviana*	tree/shrub	EXO	0.07	959
Proteaceae	*Grevillea banksii*	tree/shrub	EXO	0.04	812
Solanaceae	*Solanum erianthum*	tree/shrub	EXO	0.06	800
Salicaceae	*Salix babylonica*	tree/shrub	EXO	0.02	720
Myrtaceae	*Melaleuca leucadendra*	tree/shrub	EXO_cult	0.02	710
Fabaceae	*Bauhinia purpurea*	tree/shrub	EXO	0.02	700
Myristicaceae	*Virola surinamensis*	tree/shrub	EXO_reg	0.02	700
Malpighiaceae	*Lophanthera lactescens*	tree/shrub	EXO_reg	0.09	629
Oleaceae	*Ligustrum japonicum*	tree/shrub	EXO	0.02	585
Myrtaceae	*Eugenia leitonii*	tree/shrub	EXO	0.04	540
Fabaceae	*Inga macrophylla*	tree/shrub	EXO_reg	0.04	500
Rosaceae	*Prunus campanulata*	tree/shrub	EXO	0.02	500
Myrtaceae	*Eugenia stipitata*	tree/shrub	EXO_reg	0.09	485
Meliaceae	***Melia azedarach*** **	tree/shrub	EXO_nat	0.02	480
Arecaceae	*Phoenix roebelenii*	palm	EXO	0.02	450
Arecaceae	*Bactris gasipaes*	palm	EXO_reg	0.02	390
Fabaceae	*Caesalpinia pulcherrima*	tree/shrub	EXO_nat	0.02	378
Fabaceae	*Dipteryx odorata*	tree/shrub	EXO_reg	0.02	360
Acanthaceae	*Thunbergia grandiflora*	tree/shrub	EXO	0.02	336
Euphorbiaceae	*Hevea brasiliensis*	tree/shrub	EXO_reg	0.06	329
Moraceae	*Morus celtidifolia*	tree/shrub	EXO_cult	0.02	311
Boraginaceae	***Cordia africana*** **	tree/shrub	EXO	0.02	288
Fabaceae	*Adenanthera pavonina*	tree/shrub	EXO	0.02	280
Clusiaceae	*Garcinia cochinchinensis*	tree/shrub	EXO	0.02	270
Arecaceae	*Licuala grandis*	palm	EXO	0.02	240
Muntingiaceae	*Muntingia calabura*	tree/shrub	EXO_reg	0.04	231
Lauraceae	*Nectandra rigida*	tree/shrub	EXO	0.06	202
Cupressaceae	*Cupressus lusitanica*	tree/shrub	EXO	0.02	200
Fabaceae	*Machaerium floridum*	tree/shrub	EXO_reg	0.02	180

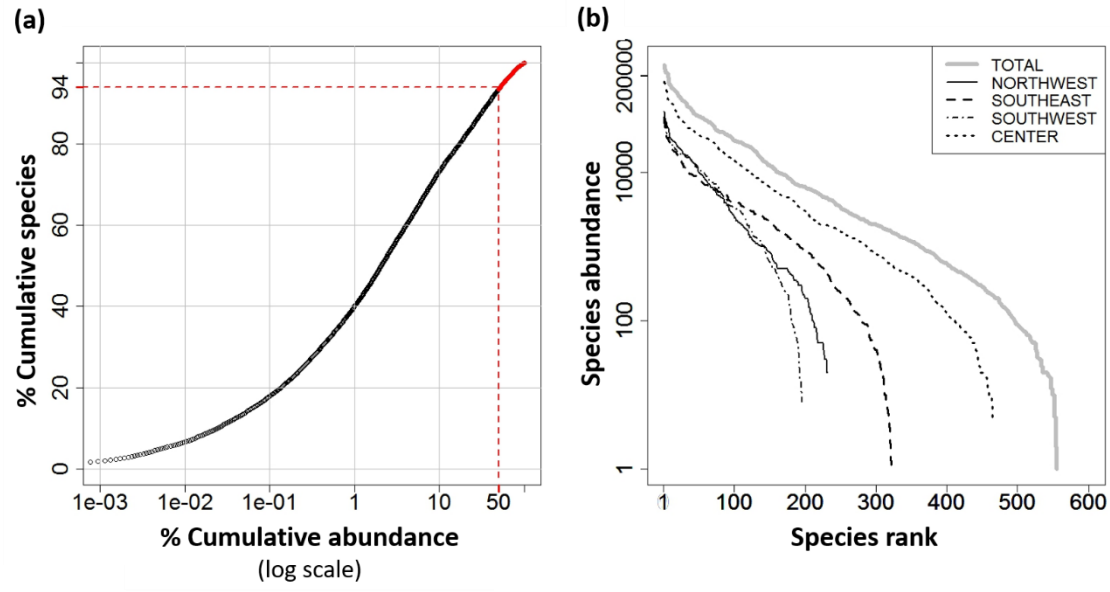
Family	Species (126)	Growth form	EXO type	Relative F	Quantity
Sapotaceae	*Manilkara zapota*	tree/shrub	EXO_cult	0.02	180
Asteraceae	*Cabobanthus polysphaerus*	tree/shrub	EXO	0.04	175
Bignoniaceae	***Tecoma stans***	tree/shrub	EXO_nat	0.02	175
Rosaceae	*Cydonia oblonga*	tree/shrub	EXO_cult	0.02	174
Malpighiaceae	*Bunchosia armeniaca*	tree/shrub	EXO	0.04	165
Arecaceae	*Cocos nucifera*	palm	EXO_nat	0.02	150
Arecaceae	*Roystonea regia*	palm	EXO	0.02	150
Arecaceae	*Sabal mexicana*	palm	EXO	0.02	150
Fabaceae	***Acacia mangium***	tree/shrub	EXO	0.02	150
Fabaceae	*Delonix regia*	tree/shrub	EXO_cult	0.02	150
Araliaceae	*Aralia excelsa*	palm	EXO	0.02	100
Arecaceae	*Trithrinax brasiliensis*	palm	EXO_reg	0.02	100
Winteraceae	*Drimys winteri*	tree/shrub	EXO	0.02	81
Apocynaceae	*Aspidosperma discolor*	tree/shrub	EXO_reg	0.04	63
Asteraceae	*Eremanthus arboreus*	tree/shrub	EXO_reg	0.02	63
Malvaceae	*Bombax ceiba*	tree/shrub	EXO	0.02	60
Sapotaceae	*Planchonella obovata*	tree/shrub	EXO_reg	0.02	50
Moraceae	*Morus alba*	tree/shrub	EXO_cult	0.06	48
Araucariaceae	*Araucaria columnaris*	tree/shrub	EXO	0.02	45
Fabaceae	*Acacia xanthophloea*	tree/shrub	EXO	0.02	40
Euphorbiaceae	*Hura crepitans*	tree/shrub	EXO_reg	0.02	30
Fabaceae	*Cassia javanica*	tree/shrub	EXO_cult	0.02	30
Annonaceae	*Guatteria citriodora*	tree/shrub	EXO_reg	0.02	24
Euphorbiaceae	*Jatropha curcas*	tree/shrub	EXO_nat	0.02	24
Arecaceae	*Dypsis decaryi*	palm	EXO	0.02	20
Euphorbiaceae	*Euphorbia pulcherrima*	tree/shrub	EXO	0.02	20
Myrtaceae	*Syzygium aromaticum*	tree/shrub	EXO	0.02	20
Myrtaceae	*Psidium acutangulum*	tree/shrub	EXO_reg	0.04	17
Fabaceae	*Albizia gummifera*	tree/shrub	EXO	0.02	14

Family	Species (126)	Growth form	EXO type	Relative F	Quantity
Moringaceae	*Moringa oleifera*	tree/shrub	EXO_cult	0.02	12
Arecaceae	*Dyctiosperma alba rubra*	palm	EXO	0.02	10
Anacardiaceae	*Spondias dulcis*	tree/shrub	EXO	0.02	8
Platanaceae	*Platanus acerifolia*	tree/shrub	EXO	0.02	5
Bignoniaceae	*Tabebuia gemmiflora*	tree/shrub	EXO	0.02	1
Fabaceae	*Enterolobium schomburgkii*	tree/shrub	EXO_reg	0.02	1
Fabaceae	*Parkia multijuga*	tree/shrub	EXO_reg	0.02	1
Fabaceae	*Tachigali multijuga*	tree/shrub	EXO_reg	0.02	1
Lecythidaceae	*Bertholletia excelsa*	tree/shrub	EXO_reg	0.02	1
Lecythidaceae	*Couratari asterotricha*	tree/shrub	EXO_reg	0.02	1
Malvaceae	*Hibiscus tiliaceus*	tree/shrub	EXO	0.02	1
Malvaceae	*Pachira insignis*	tree/shrub	EXO_reg	0.02	1
Sapindaceae	*Melicoccus oliviformis*	tree/shrub	EXO_reg	0.02	1
Sapotaceae	*Manilkara bidentata*	tree/shrub	EXO_reg	0.02	1



Plant nurseries	Forest fragments	IBt - SP
Fabaceae (125)	Fabaceae (124)	Fabaceae (118)
Myrtaceae (63)	Myrtaceae (88)	Myrtaceae (95)
Malvaceae (24)	**Rubiaceae (63)**	**Melastomataceae (75)**
Bignoniaceae (23)	**Melastomataceae (55)**	**Lauraceae (71)**
Lauraceae (22)	**Lauraceae (44)**	**Rubiaceae (71)**
Euphorbiaceae (20)	Solanaceae (33)	Asteraceae (39)
Melastomataceae (16)	Euphorbiaceae (30)	Solanaceae (34)
Rubiaceae (16)	Piperaceae (30)	Malvaceae (27)
Apocynaceae (14)	Asteraceae (27)	Bignoniaceae (25)
Solanaceae (14)	Malvaceae (25)	Euphorbiaceae (23)

Appendix 4. Comparison of floristic composition among plant nurseries, forest fragments and SP-IBt. Ranking of families richness, indicating the top 10 most rich families.



Appendix 5. Empirical cumulative distribution function for total abundance and species (a) and abundance rank curves for native species produced by plant nurseries (n=54) in four ecological regions and their total (b).

Appendix 6: Comparisons of fitted linear models based on Akaike Information Criterion (AIC); models with a difference in delta AIC ≤ 2 have equivalently strong empirical support and similar plausibility. -LL= negative log likelihood, df= degrees of freedom.

Models	Response variale	Explanatory variable	-LL	df	AIC	delta AIC
Model 1	Richness	**production capacity**	-168.5	3	343	**0**
Mode 2	Richness	forest cover	-174	3	354	10.9
Model 4	Richness	number veg. types	-174	3	354	11
Null Model	Richness	(none)	-175.7	2	356	12.4
Model 3	Richness	ecological regions	-173.1	5	356	13.2
Model 1	Sigma	**production capacity**	-11.8	3	30	**0**
Model 3	Sigma	ecological regions	-11.8	5	34	4.1
Null Model	Sigma	(none)	-15.1	2	34	4.7
Model 2	Sigma	forest cover	-14.8	3	36	6.1
Model 4	Sigma	number veg. types	-15	3	36	6.4

DISCUSSÃO GERAL

A importância dos fragmentos florestais inseridos na matriz agrícola

Nosso estudo compõe uma avaliação abrangente dos fragmentos florestais distribuídos por uma grande extensão geográfica do estado de São Paulo, onde predomina uma matriz agrícola altamente tecnificada. Nessas condições, ressaltamos a importância dos fragmentos remanescentes para a conservação e restauração da biodiversidade.

As **Unidades de Conservação de Proteção Integral** são os mais emblemáticos e tradicionais componentes das estratégias conservacionistas, sobretudo em regiões sob intensa ocupação humana. Sua distribuição espacial é bastante irregular e revela uma natureza residual, geralmente delimitadas em áreas de baixo potencial agrícola (Loyola 2016, Bergamin et al. 2017; Oliveira et al. 2017). Apesar desse viés, elas frequentemente representam os maiores fragmentos e portanto as maiores áreas "*core*" de suas regiões (Joppa et al. 2008), sendo então natural que essas áreas representem os pilares para a conservação da diversidade regional, onde a riqueza de espécies permanece mais elevada (**Capítulo 1**) e onde as alterações na composição de espécies se mostraram mais estáveis, sem diferenças significativas entre um cenário atual e um cenário hipotético antes da fragmentação (**Capítulo 2**). Vale lembrar, no entanto, que isoladamente essas áreas não garantem a proteção efetiva das várias dimensões da diversidade, como por exemplo a diversidade taxonômica, a filogenética, a funcional (Bergamin et al. 2017; Oliveira et al. 2017; Saraiva et al. 2018), sobretudo quando localizadas em regiões sob intensa ocupação humana (Joppa et al. 2008; Laurance et al. 2012, 2014). Em outras palavras, as Unidades de Conservação de Proteção Integral dependem dos fragmentos ao seu redor, incluindo os fragmentos dentro de propriedades privadas, representados pelas Áreas de Proteção Permanente (APPs) e pelas Reservas Legais (RLs) e as Unidades de Conservação classificadas sob categorias menos restritivas, como as Unidades de Uso Sustentável - que na prática podem se aproximar mais de um mecanismo de manejo e uso da terra do que de fato proteção (*e.g*. Áreas de Proteção Ambiental ou APAs) (Crouzeilles et al. 2013).

Além dos limites das Unidades de Conservação, os **fragmentos em propriedades privadas** estão diretamente expostos à fragmentação e degradação recorrente, o que explica o fato de estarem empobrecidos localmente, embora em conjunto ainda abriguem porção expressiva e relevante da biodiversidade remanescente (**Capítulo1**). Os fragmentos inseridos em propriedades privadas compreendem uma grande variedade de situações, que inclui desde florestas secundárias em processo de regeneração até florestas primárias degradadas ou mais conservadas (Farah et al. 2017). Esses fragmentos também refletem a heterogeneidade ambiental e biológica típica da Floresta Atlântica, composta por diferentes tipos vegetacionais (Morellato & Haddad 2000; Oliveira-Filho & Fontes 2000). Nossos resultados expressam todas essas diferenças através dos elevados valores de diversidade β e *turnover* registrados nas regiões estudadas (**Capítulo 1**), revelando ainda que essas diferenças entre fragmentos, de maneira geral, são cada vez maiores em resposta ao longo histórico de distúrbio e degradação (**Capítulo 2**). A heterogeneidade na composição desses fragmentos também se reflete na heterogeneidade de espécies nativas disponíveis nos viveiros de mudas, que costumam coletar propágulos nos fragmentos da sua região; por esse motivo, a composição de espécies produzidas pelos viveiros é bastante diferente entre si, conforme constatamos no **Capítulo 3**. Portanto, a diversidade de espécies disponíveis para as ações de restauração florestal depende da diversidade presente nos fragmentos próximos; esses, por sua vez, também dependem de ações de restauração ecológica para potencializar seu papel de conservação da biodiversidade (enriquecimento artificial) e para refazer a conectividade entre eles (corredores ecológicos), aumentando as chances de persistência das espécies nessas paisagens hiper-fragmentadas.

Aplicação para as políticas públicas de conservação e restauração

O reconhecimento e consideração da heterogeneidade dos fragmentos florestais é importante para elaborar planos de conservação mais abrangentes e inclusivos (Vidal et al. 2016), que contemplem além das Unidades de Conservação e considerem de fato os remanescentes naturais existentes nas propriedades privadas. Nas paisagens agrícolas do nosso estudo, partimos do princípio que esses fragmentos

florestais devem prioritariamente cumprir seu papel de conservação da biodiversidade remanescente, através de ações que potencializem esse propósito (Viani et al. 2015). Sem desconsiderar que a lei prevê, em alguns casos, a possibilidade de exploração econômica desses fragmentos (*e.g.* Reserva legal e Áreas de Preservação Permanente de pequenas propriedades rurais), discutiremos também recomendações específicas para essas situações.

A Lei de Proteção da Vegetação Nativa (LPVN) (popularmente conhecida como o "Novo Código Florestal") (leis 12.561/2012 e 12.727/2012) é a principal regulamentação legal para conciliar a exploração e a conservação da vegetação nativa em propriedades privadas (Brancalion, Garcia, et al. 2016b). Junto com a Política Nacional de Recuperação da Vegetação Nativa (PROVEG) (decreto no.8.972/2017) e seu instrumento de implementação, o Plano Nacional de Recuperação da Vegetação Nativa (PLANAVEG) (portaria interministerial no. 230 de 2017), esses dispositivos legais reconhecem a necessidade de conservar, recuperar ou compensar os desmatamentos que foram incentivados durante uma ocupação territorial mal planejada (Benini et al. 2017; Scaramuzza et al. 2017). No entanto, o conteúdo desses dispositivos possui um forte viés para a restauração de áreas já desmatadas e/ou muito degradadas, uma área do conhecimento relativamente recente mas bem estabelecida no Brasil (Rodrigues et al. 2009). Menções explícitas sobre a restauração dos fragmentos degradados (*i.e.* manejo para conservação) são raras e pouco detalhadas (Brancalion et al. 2012) e essa abordagem superficial se justifica pela falta de evidências consistentes a respeito da eficácia dessas ações, tanto do ponto de vista ecológico quanto econômico (Viani et al. 2015).

Recentemente um grupo de pesquisadores organizou um documento técnico para embasar políticas públicas e ações de melhoria nos remanescentes florestais, considerando aspectos conceituais, legais e práticos (Assis et al. 2018, no prelo). A partir de uma avaliação capaz de reconhecer as florestas degradadas e de diagnosticar a necessidade de ações de restauração (Liboni et al. 2018, no prelo), as principais recomendações são (Brancalion et al. 2012; Viani et al. 2015; Mangueira et al. 2018, no prelo): 1) erradicação de espécies invasoras, 2) controle de espécies hiperabundantes (*e.g.* lianas) e 3) plantios de adensamento nas áreas mais abertas e de enriquecimento nas mais fechadas, com o intuito de cicatrizar trechos muito

abertos e de reintroduzir ou favorecer espécies de grupos funcionais comprometidos, como por exemplo espécies raras, ameaçadas de extinção ou climácicas (*e.g.* espécies de crescimento lento e com sementes grandes dispersas por grandes vertebrados ausentes na paisagem) (Bello et al. 2015; Beca et al. 2017).

Conforme discutido no **Capítulo 2**, a heterogeneização biótica dos fragmentos florestais pode ser resultante, entre outros fatores, do grau de isolamento desses habitats e das limitações de dispersão entre eles (Catano et al. 2017). Nessas paisagens hiper-fragmentadas, com cobertura de vegetação natural abaixo dos limiares de fragmentação (Pardini et al. 2010; Estavillo et al. 2013; Banks-Leite et al. 2014) além da restauração dos fragmentos degradados é preciso restaurar corredores e trampolins ecológicos, interligando esses fragmentos na paisagem, de forma a sustentar as redes de interações e viabilizar o fluxo biológico entre eles (Howe 2014; Emer et al. 2018). Nesse cenário, a principal recomendação para a reconstrução da cobertura vegetal entre os fragmentos remanescentes é a adoção de ações de restauração (Rodrigues et al. 2009; Brancalion et al. 2013; Vidal et al. 2016). Considerando a expectativa pelo cumprimento dos passivos ambientais definidos pela LPVN e a consequente demanda por mudas de espécies nativas, destacamos a importância da cadeia e/ou mercado da restauração (Benini et al. 2016; Benini & Adeodato 2017; Silva et al. 2017). No **Capítulo 3** discutimos sobre a relevância da diversidade de espécies vegetais produzida pelos viveiros de mudas para a restauração ecológica, mas devemos expandir a discussão para as implicações e os benefícios que essa diversidade tem sobre outras possibilidades de restauração, como os plantios de espécies nativas com fins econômicos (*e.g.* silvicultura ou sistemas agroflorestais) (Brancalion et al. 2012; Batista et al. 2017; WRI Brasil 2017).

Segundo a LPVN, as propriedades rurais são divididas em áreas agrícolas, onde é permitido a produção agrossilvipastoril, e áreas de vegetação nativa, que devem ser protegidas ou utilizadas de forma sustentável (APPs de pequenas propriedades rurais e RLs); nos casos em que a vegetação já foi desmatada, a lei prevê a sua recomposição. Embora haja situações específicas que permitem a exploração das **Áreas de Preservação Permanente** (*e.g.* propriedade rurais abaixo de 4 módulos fiscais), essas porções da propriedade devem atender sua função primordial de proteger as nascentes, os cursos d`água, as áreas de recarga hídrica e

evitar os processos erosivos, cumprindo também o papel de corredores ecológicos. Nas **Reservas Legais**, onde a **exploração econômica sustentável** dos fragmentos florestais é permitida por lei, nossa recomendação é de que, em paisagens hiper-fragmentadas, todo fragmento florestal seja destinado prioritariamente à conservação da biodiversidade remanescente (Beca et al. 2017; Farah et al. 2017). Recomendamos, portanto, que haja o estabelecimento de critérios para identificar os fragmentos inelegíveis a esse tipo de intervenção, poupando, por exemplo, aqueles mais conservados, onde o manejo florestal pode comprometer seu papel atual de conservação da biodiversidade. A exploração econômica pode ainda ser restrita à borda dos fragmentos ou a grupos menos impactantes, como os produtos não madeireiros. Com base nesses critérios seriam indicados quais os fragmentos passíveis de manejo e qual o tipo de manejo possível para cada fragmento. Assim, a exploração econômica das Reservas Legais ficaria restrita aos casos em que há a necessidade de recomposição da vegetação, ou então, limitadas às condições de borda ou aos trechos mais degradados dos fragmentos remanescentes, reduzindo o impacto sobre a vegetação nativa (Brancalion et al. 2012; Putz & Romero 2014; WRI Brasil 2017). Vale lembrar aqui que a tendência para os mecanismos de Compensação de Reserva Legal (*e.g.* Cota de Reserva Ambiental) tem sido o de estimular a compensação dentro do próprio estado, atendendo as demandas dos serviços ecossistêmicos afetados local ou regionalmente (*e.g.* proteção dos recursos hídricos e do solo) e obedecendo o critério de equivalência ou similaridade ecológica dentro de um mesmo bioma (Silva & Ranieri 2014)cf.(Soares-Filho et al. 2016). Para cumprir o déficit de Reservas Legais através da restauração ou recomposição da vegetação, diversas atividades de exploração econômica podem ser praticadas nessas áreas. Algumas alternativas estão sendo compiladas e avaliadas pelo projeto VERENA (WRI Brasil 2017), cuja iniciativa tem o propósito de demonstrar a viabilidade e os gargalos técnicos e econômicos da restauração e reflorestamento com espécies nativas visando a exploração econômica. Essa iniciativa organiza informações valiosas para fomentar e fortalecer uma economia florestal de baixo carbono, ao mesmo tempo que pode contribuir para a ampliação da cobertura florestal (WRI Brasil 2017).

Apesar de predominar no Brasil um descompasso entre o conhecimento gerado no ambiente acadêmico e sua aplicação nas políticas públicas ambientais

(Karam-Gemael et al. 2018), o estado de São Paulo possui alguns exemplos de políticas públicas pautadas em evidências científicas e com participação de diversos setores (Aronson et al. 2011; Chaves et al. 2015; Joly et al. 2010). Através da discussão e de uma construção participativa envolvendo órgãos ambientais e agrícolas, universidades, ONGs e consultorias especializadas, o governo do estado elaborou um arcabouço legal sobre temas específicos – notadamente sobre restauração ecológica - além de diversos programas e ferramentas relacionados à restauração e conservação de modo geral (*e.g.*: Programa Nascentes, Programa Estadual Microbacias, Mapa de Zoneamento Agroecológico, Mapa de Áreas Prioritárias para a Conservação e Restauração, Programa Recuperação de Matas Ciliares; Programa RPPN Paulistas, Crédito Ambiental Paulista para as RPPN, Sistema de Apoio à Restauração Ecológica (SARE), Banco de Áreas, DATAGEO, SICAR-SP, etc.) (http://www.ambiente.sp.gov.br/ e http://sigam.ambiente.sp.gov.br/sigam3/). No entanto, ainda existem muitas lacunas a serem preenchidas e as evidências expostas nesta tese podem servir para aprimorar e nortear algumas políticas públicas voltadas à conservação em paisagens agrícolas.

Em suma, o conhecimento científico gerado nesse trabalho nos permite fazer as seguintes considerações finais:

- Nós valorizamos e reforçamos o papel do conhecimento científico no suporte às políticas públicas ambientais;
- Com base nas evidências geradas neste estudo, reconhecemos o papel das propriedades privadas às convencionais abordagens de conservação (Unidades de Conservação de Proteção Integral), destacando sua essencial contribuição à restauração;
- Conservação em paisagens agrícolas é um grande desafio e depende de abordagens amplas e inclusivas que considerem todos os elementos da paisagem, sem negligenciar o valor dos fragmentos em propriedades privadas. No nosso caso, os fragmentos devem ser efetivamente protegidos, manejados e conservados, as áreas a serem restauradas devem aumentar a cobertura e o fluxo biológico e, quando destinadas à exploração econômica, devem priorizar modelos alternativos menos impactantes;

- A ideia de conflito entre agricultura e conservação é um paradigma difícil de ser quebrado, mas um modelo que combine a produção eficiente numa paisagem de elevada diversidade natural, através de ações de conservação e restauração da biodiversidade, é possível e deve ser buscado como modelo a ser replicado para todo o Brasil e para o mundo.

REFERÊNCIAS

Aronson J et al. (2011) What Role Should Government Regulation Play in Ecological Restoration? Ongoing Debate in São Paulo State, Brazil. Restoration Ecology 19:690–695

Assis LS de, Campos M, Girão VJ (2018) Documento técnico para manejo de fragmentos florestais degradados. Fundação JO'se Pedro de Oliveira (FJPO) e The Nature COnservancy (TNC)

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

Batista A et al. (2017) Reflorestamento com espécies nativas para fins econômicos. In: Economia da restauração florestal. Benini, R de M & Adeodato, S, editors. The Nature Conservancy pp. 74–91.

Beca G et al. (2017) High mammal species turnover in forest patches immersed in biofuel plantations. Biological Conservation

Bello C et al. (2015) Defaunation affects carbon storage in tropical forests. Science Advances 1:1–11

Benini R de M, Adeodato S (2017) O desafio econômico de recobrir o Brasil. In: Economia da Restauração Florestal. Benini, R de M & Adeodato, S, editors. The Nature Conservancy, São Paulo.

Benini R de M, Brancalion PHS, Rodrigues RR (2017) O futuro da restauração no contexto econômico. In: Economia da Restauração Florestal. Benini, R de M & Adeodato, S, editors. The Nature Conservancy, São Paulo p. 135.

Benini R de M et al. (2016) Plano Estratégico da Cadeia da Restauração Florestal : O caso do. In: Mudanças no Código Florestal Brasileiro: desafios para implementação da nova lei. Silva, APM Da, Marques, HR, & Sambuichi, RHR, editors. IPEA, Brasilia pp. 209–234.

Bergamin RS et al. (2017) Linking beta diversity patterns to protected areas: lessons from the Brazilian Atlantic Rainforest. Biodiversity and Conservation

Brancalion PHS et al. (2016) Análise crítica da Lei de Proteção da Vegetação Nativa (2012), que substituiu o antigo Código Florestal: Atualizações e ações em curso. Natureza e Conservacao 14:e1–e16

Brancalion PHS et al. (2012) Estratégias para auxiliar na conservação de florestas tropicais secundárias inseridas em paisagens alteradas. Bol. Mus. Para. Emílio Goeldi 900:219–234

Brancalion PHS et al. (2013) Biodiversity persistence in highly human- modified tropical landscapes depends on ecological restoration. Tropical Conservation Science 6:705–710

Brancalion PHS et al. (2012) Finding the money for tropical forest restoration. Unasylva 63:41–50

Catano CP, Dickson TL, Myers JA (2017) Dispersal and neutral sampling mediate contingent effects of disturbance on plant beta-diversity: a meta-analysis. Ecology Letters 20:347–356

Chaves RB et al. (2015) On the need of legal frameworks for assessing restoration projects success: new perspectives from São Paulo state (Brazil). Restoration Ecology 23:754–759

Crouzeilles R, Lorini ML, Grelle CEV (2013) The importance of using sustainable use protected areas for functional connectivity. Biological Conservation 159:450–457

Emer C et al. (2018) Seed-dispersal interactions in fragmented landscapes – a metanetwork approach. Ecology Letters 21:484–493

Estavillo C, Pardini R, Da Rocha PLB (2013) Forest loss and the biodiversity threshold: An evaluation considering species habitat requirements and the use of matrix habitats. PLoS ONE 8:1–10

Farah FT et al. (2017) Integrating plant richness in forest patches can rescue overall biodiversity in human-modified landscapes. Forest Ecology and Management 397:78–88

Howe HF (2014) Diversity Storage: Implications for tropical conservation and restoration. Global Ecology and Conservation 2:349–358

Joly CA et al. (2010) Biodiversity Conservation Research, Training, and Policy in Sao Paulo. Science 328:1358–1359

Joppa LN, Loarie SR, Pimm SL (2008) On the protection of ‘protected areas’. Proceedings of the National Academy of Sciences 105:6673–6678

Karam-Gemael M et al. (2018) Poor alignment of priorities between scientists and policymakers highlights the need for evidence-informed conservation in Brazil. Perspectives in Ecology and Conservation 16:125–132

Laurance WF et al. (2012) Averting biodiversity collapse in tropical forest protected areas. Nature 489:290–4

Laurance WF, Sayer J, Cassman KG (2014) Agricultural expansion and its impacts on tropical nature. Trends in Ecology and Evolution 29:107–116

Liboni AP et al. (2018) Diagnóstico de fragmentos florestais fegradados como subsídio para o manejo adaptativo: Proposta de avaliação ecológica rápida para a Floresta Estacional Semidecidual. In: Documento técnico para manejo de fragmentos florestais degradados. Fundação José Pedro de Oliveira (FJPO) e The Nature COnservancy (TNC) p. 172.

Mangueira JR de SA et al. (2018) Métodos de manejo de fragmentos florestais: Revisão da literatura e propostas para orientar a prática. In: Documento técnico para manejo de fragmentos florestais degradados. Fundação José Pedro de Oliveira (FJPO) e The Nature COnservancy (TNC) p. 172.

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

Oliveira-Filho A, Fontes M (2000) Patterns of Floristic Differentiation among Atlantic Forests in Southeastern Brazil and the Influence of Climate1. Biotropica 32:793–810

Oliveira U et al. (2017) Biodiversity conservation gaps in the Brazilian protected areas. Scientific Reports 7:1–9

Pardini R et al. (2010) Beyond the fragmentation threshold hypothesis: regime shifts in biodiversity across fragmented landscapes. PloS one 5:e13666

Putz FE, Romero C (2014) Futures of Tropical Forests (sensu lato). Biotropica 46:495–505

Ribeiro MC et al. (2009) The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biological Conservation 142:1141–1153

Rodrigues RR et al. (2009) On the restoration of high diversity forests: 30 years of experience in the Brazilian Atlantic Forest. Biological Conservation 142:1242–1251

Saraiva DD et al. (2018) How effective are protected areas in conserving tree taxonomic and phylogenetic diversity in subtropical Brazilian Atlantic Forests? Journal for Nature Conservation 42:28–35

Scaramuzza CA de M et al. (2017) A Política Nacional de Recuperação da Vegetação Nativa: Lições apendidas. In: Economia da Restauração Florestal. Benini, R de M & Adeodat, editors. The Nature Conservancy, São Paulo p. 135.

Silva APM da et al. (2017) Can current native tree seedling production and infrastructure meet an increasing forest restoration demand in Brazil ? Restoration Ecology 25:509–515

Silva JS Da, Ranieri VEL (2014) O mecanismo de compensação de reserva legal e suas implicações econômicas e ambientais. Ambiente & Sociedade 17:115–132

Soares-Filho B et al. (2016) Brazil's Market for Trading Forest Certificates. Plos One 11:e0152311

Viani RAG et al. (2015) A new focus for ecological restoration: management of degraded forest remnants in fragmented landscapes. GPL news november:5–9

Vidal CY et al. (2016) Biodiversity Conservation of Forests and their Ecological Restoration in Highly-modified Landscapes. In: Biodiversity in Agricultural Landscapes of Southeastern Brazil. Gheler-Costa, C, Lyra-Jorge, MC, & Verdade, LM, editors. De Gruyter Open Ltd, Warsaw/Berlin p. 342.

WRI Brasil (2017) Relatório de atividades da rede de parceiros Projeto VERENA.

APÊNDICES

APÊNDICE 1: Folder de divulgação dos resultados.

Principais resultados gerados no capítulo 3 da tese, sintetizado com o intuito de informar os viveiristas participantes. *** Relatório completo disponível em www.lerf.esalq.usp.br (> Publicações > Material LERF > Manuais Técnicos e Relatórios)

APRESENTAÇÃO

Graças às discussões entre pesquisadores. ONGs e profissionais do mercado desde meados da década de 80, o estado de São Paulo possui hoje um arcabouço legal específico para as ações de restauração ecológica e a maior cadeia produtiva voltada à restauração de florestas nativas.

Apesar disso, o cumprimento do Novo Código Florestal (lei 12.651 de 2012) em concordância com os requisitos definidos pela legislação estadual é um desafio; um diagnóstico detalhado dessa cadeia permite detectar os gargalos e consolidar os avanços.

Nesse contexto, o objetivo principal do trabalho foi avaliar a diversidade disponível para a restauração ecológica ativa no interior do estado de São Paulo, onde predomina uma paisagem agrícola que dificulta a recuperação natural da vegetação.

Apresento aqui alguns resultados interessantes e relevantes aos viveiristas e outros tomadores de decisão, expondo a situação atual e a relevância do papel que esse setor desempenha na qualidade das ações de restauração ativa

Ma. **Cristina Yuri Vidal**
Programa de Pós-Graduação em Biologia Vegetal (UNICAMP)
Laboratório de Ecologia e Restauração Florestal (LERF) - Escola Superior de Agronomia "Luiz de Queiroz" (ESALQ), Universidade de São Paulo (USP)

AUTORIA
Ma. **Cristina Yuri Vidal**
Programa de Pós-Graduação em Biologia Vegetal do Instituto de Biologia da Universidade Estadual de Campinas.

Orientação:
Prof. Dr. **Ricardo Ribeiro Rodrigues**
Laboratório de Ecologia e Restauração Florestal (LERF), Departamento de Ciência Biológicas, Escola Superior de Agronomia "Luiz de Queiroz" (ESALQ), da Universidade de São Paulo (USP)

Colaboração - coleta de dados:
Letícia S. Santiago, **Bruno Henrique Guastala** , **Sergio Esteban Lozano-Baez**, **Fernando Henrique Silva**
Laboratório de Ecologia e Restauração Florestal (LERF)

Colaboração - análise de dados:
Dra. **Rafaela Pereira Naves**
Laboratório de Métodos Quantitativos (LMQ), Departamento de Ciências Florestais, Escola Superior de Agronomia "Luiz de Queiroz" (ESALQ), da Universidade de São Paulo (USP)

Colaboração (fornecimento dos dados):
Viveiros: Associação de Reposição Florestal Cantareira, Ambiental mudas nativas, AEHDA-Araras, Capivari Monos, Flora Vale, Flora Sandrin, Flora Brasília, Fundação José Lazzarini, Usina Batatais, Associação de Recuperação Florestal e Ecológica de Bauru, Jardim Botânico de Bauru, Fábrica de Árvores, Verde Nativo, Instituto Refloresta, Municipal de Capivari, Mudas da Mata, Florestal Jequitibá, Flora Paulista, Oiti, Camará, Dois Córregos, Vicente mudas, VM mudas, CATI Itaberá, Mata Atlântica Viveiro Florestal, SOS Mata Atlântica, Da Serra, Alameda, Bilato Bosco, Bioverde Árvores do Brasil, Sequinato, Ipês, Instituto Pesquisas Ecológicas IPÊ, Núcleo Pederneiras, Flora Gerais, Flora Tietê, Bioflora, Municipal de Piracicaba, Associação de recuperação florestal do Paranapanema, Multiverde, AES Tietê Energia S/A, Schmidt mudas, Tecplant, Arboreto Siriema, Viveirão Agrícola, Equilíbrio Verde, VidaVerde, Carobinha, Viveiro Florestal de Taubaté, Manacá, Instituto Árvore, Léo, Chácara Pau Brasil, Instituto Oikos, Árvores Brasileiras, Plantae, Viverde Ambiental, Associação Florestal Copaíba.

Apoio:
CAPES CNPq Processo 870360/1997-3 UNICAMP FAPESP Processo 2013/50718-5 IB Instituto de Biologia ESALQ LERF

RESTAURAÇÃO DA DIVERSIDADE: OS VIVEIROS DO ESTADO DE SÃO PAULO

OBJETIVOS

Descrever os viveiros quanto à:
(1) Distribuição e status
(2) Capacidade produtiva e diversidade
(3) Representatividade da riqueza regional

AMOSTRAGEM

Concentramos a amostragem dos viveiros nas regiões **Noroeste, Sudoeste, Centro e Sudeste** (maior demanda por restauração ativa).

Questionários rápidos
Venda de nativas (sim ou não)
Status (ativos, inativos, revenda)

Questionários detalhados
Lista de espécies e quantidades
Questionário completo

Utilizamos a lista de espécies nativas regionais recomendadas pela SMA 32/2014 e elaboradas pelo Instituto de Botânica (IBt) para avaliar a representatividade da riqueza regional (apenas para espécies arbustivas ou arbóreas).

Calculamos os descritores (status, capacidade produtiva, diversidade, riqueza produzida e recomendada) de forma geral e por região ecológica. Avaliamos as frequências e abundâncias das espécies (questionário detalhado).

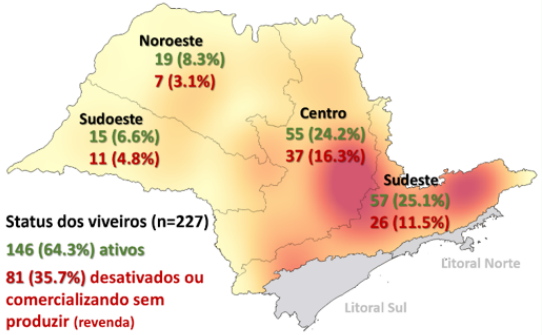
Total de viveiros listados em SP: 296

Amostragem questionário rápido: 227 (76%)

Amostragem questionário detalhado: 54 (18%)

DISTRIBUIÇÃO E STATUS

Distribuição dos viveiros em São Paulo e status regionais



Status dos viveiros (n=227)
146 (64.3%) ativos
81 (35.7%) desativados ou comercializando sem produzir (revenda)

Grande **concentração** nas regiões **Centro e Sudeste**.

CARACTERIZAÇÃO PRODUTIVA e da DIVERSIDADE

Redução de **5 a 15 milhões de mudas** e **780 empregos** em relação a 2011 (relatório SMA-SP)

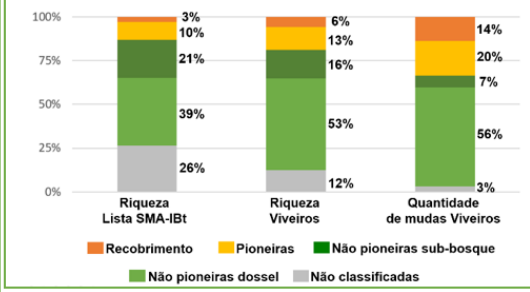
Riqueza total: 561 espécies nativas (81.6%)
+ 126 espécies exóticas (18.4%) (incluindo naturalizadas e cultivadas)

Riqueza média: 86.4 nativas/viveiro
Min.= 18 Max.= 194

Formas de vida: 542 arbustivas e arbóreas
+ 7 palmeiras, 5 subarbustos, 7 lianas

Diversidade (σ) média: 1,44 +- 0,36
(mín.=0,76 máx.=2,32)

Grupos funcionais de espécies

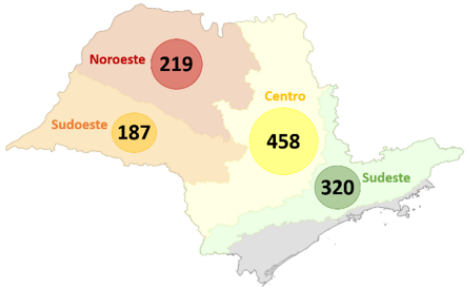


REPRESENTATIVIDADE DA RIQUEZA REGIONAL

85% coleta as próprias sementes (total ou parcialmente) num raio de 100km >>> **amostra das populações locais**

Representam 38% das espécies arbustivas ou arbóreas indicadas pela SMA 32/14
+ 123 não listadas

Espécies arbustivas/arbóreas nativas produzidas em cada região



Viveiros são **bem diferentes entre si**, considerando a frequência das espécies:
10 espécies (1.7%) produzidas em **mais de 75%** dos viveiros
448 espécies (80%) produzidas em **menos de 25%** dos viveiros

RECOMENDAÇÕES para POLITICAS PUBLICAS

- **Planejamento** regional, **capacitação** e **assessoria** técnica **constante,** com ênfase em **gestão administrativa** e **identificação botânica**

- **Recessão** do setor é **preocupante,** considerando a demanda para o cumprimento do "Novo Código Florestal"

RECOMENDAÇÕES para os VIVEIRISTAS

- Coletar sementes nos **fragmentos do entorno,** importante para **representar a riqueza regional**
- **Revisar e eliminar** a produção de espécies **exóticas**
- Outros materiais e vídeos em ***www.lerf.esalq.usp.br***

ANEXOS

ANEXO 1 : Declaração sobre Bioética e Biossegurança

UNICAMP
COORDENADORIA DE PÓS-GRADUAÇÃO
INSTITUTO DE BIOLOGIA
Universidade Estadual de Campinas
Caixa Postal 6109. 13083-970, Campinas, SP, Brasil
Fone (19) 3521-6378. email: cpgib@unicamp.br
IB

DECLARAÇÃO

Em observância ao **§5º do Artigo 1º da Informação CCPG-UNICAMP/001/15**, referente a Bioética e Biossegurança, declaro que o conteúdo de minha Tese de Doutorado, intitulada "***The importance of forest fragments' biodiversity for conservation and ecological restoration within agricultural landscapes*", desenvolvida no Programa de Pós-Graduação em Biologia Vegetal do Instituto de Biologia da Unicamp, não versa sobre pesquisa envolvendo seres humanos, animais ou temas afetos a Biossegurança.

Assinatura: Cristina Yuri Vidal
Nome do(a) aluno(a): Cristina Yuri Vidal

Assinatura: Ricardo Ribeiro Rodrigues
Nome do(a) orientador(a): prof. Dr. Ricardo Ribeiro Rodrigues

Data: Campinas, 07 de Janeiro de 2019

ANEXO 2 : Declaração sobre direitos autorais

Declaração

As cópias de artigos de minha autoria ou de minha co-autoria, já publicados ou submetidos para publicação em revistas científicas ou anais de congressos sujeitos a arbitragem, que constam da minha Dissertação/Tese de Mestrado/Doutorado, intitulada **"The importance of forest fragments' biodiversity for conservation and ecological restoration within agricultural landscapes"**, não infringem os dispositivos da Lei n.° 9.610/98, nem o direito autoral de qualquer editora.

Campinas, 07 de Janeiro de 2019

Assinatura : _____
Nome do(a) autor(a): **Cristina Yuri Vidal**
RG n.° 27.721.576-6

Assinatura : _____
Nome do(a) orientador(a): **Ricardo Ribeiro Rodrigues**
RG n.° 10.954.040