UNIVERSIDADE ESTADUAL DE CAMPINAS
INSTITUTO DE BIOLOGIA

MAGDA SILVA CARNEIRO

EFEITO DA COMPOSIÇÃO DE UMA PAISAGEM FRAGMENTADA SOBRE A RIQUEZA E DIVERSIDADE DE ESPÉCIES E A DIVERSIDADE FUNCIONAL DAS COMUNIDADES REMANESCENTES

COMPOSITION OF A FRAGMENTED LANDSCAPE EFFECT ON RICHNESS AND DIVERSITY OF SPECIES AND FUNCTIONAL DIVERSITY OF THE REMAINING COMMUNITIES

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Thesis presented to the Institute of Biology of the University of Campinas in partial fulfillment of the requirements for the degree of Doctor in Ecology

Tese apresentada ao Instituto de Biologia da Universidade Estadual de Campinas como parte dos requisitos exigidos para a obtenção do Título de Doutora em Ecologia

ESTE ARQUIVO DIGITAL CORRESPONDE À VERSÃO FINAL DA TESE DEFENDIDA PELA ALUNA MAGDA SILVA CARNEIRO E ORIENTADA PELO PROF. DR. FLAVIO ANTONIO MAES DOS SANTOS.

Orientador: Flavio Antonio Maes dos Santos

Co-Orientador: Flavio Nunes Ramos

CAMPINAS

2017

Agência(s) de fomento e nº(s) de processo(s): Não se aplica.

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

	Carneiro, Magda Silva, 1987-
C215c	Composition of a fragmented landscape effect on richness and diversity of species and functional diversity of the remaining communities / Magda Silva Carneiro. – Campinas, SP : [s.n.], 2017. Orientador: Flavio Antonio Mäes dos Santos. Coorientador: Flavio Nunes Ramos. Tese (doutorado) – Universidade Estadual de Campinas, Instituto de Biologia. 1. Fragmentação florestal. I. Santos, Flavio Antonio Mäes dos,1958-. II. Ramos, Flavio Nunes. III. Universidade Estadual de Campinas. Instituto de Biologia. IV. Título.

Informações para Biblioteca Digital

Título em outro idioma: Efeito da composição de uma paisagem fragmentada sobre a riqueza e diversidade de espécies e a diversidade funcional das comunidades remanescentes
Palavras-chave em inglês:
Forest fragmentation
Área de concentração: Ecologia
Titulação: Doutora em Ecologia
Banca examinadora:
Flavio Antonio Mäes dos Santos [Orientador]
Marco Antonio Assis
Márcia Cristina Mendes Marques
Marcelo Trindade Nascimento
Simone Aparecida Vieira
Data de defesa: 21-08-2017
Programa de Pós-Graduação: Ecologia

Campinas, 21 de agosto de 2017.

COMISSÃO EXAMINADORA

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Profa. Dra. Marcia Cristina Mendes Marques

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

Aos meus pais, Miguel e Isabel,
exemplos de amor e dedicação!
Ao Anézio,companheiro de
vida!

Agradecimentos

À Deus por estar ao meu lado, mesmo quando eu acreditava estar sozinha.

Ao Prof. Flavio Antonio Maes dos Santos pelos ensinamentos, paciência e confiança.

Ao Prof. Flavio Nunes Ramos pelo apoio incondicional em todas as horas, pelo incentivo e principalmente por não me deixar desistir.

À Thais e ao Caio, pela amizade, carinhoe apoio. Amigos queridos, que saudade eu sinto de vocês!

À Carolina e ao Maurício pelas correções, scripts e ideias compartilhadas.

À Nathália, ao Renan e ao João pelas longas horas na salinha, pelos cafés, e por me aguentarem nos momentos de angústia.

À Marina por me acolher quando eu mais precisei.

À Angelita pelas longas conversas ao telefone e por ser um exemplo de resiliência. Sua calma me acalma!

Ao Miguel Júnior, irmão, amigo, exemplo de profissional! Obrigada por me ouvir e incentivar.

Ao Anézio, por tudo! Você acreditou em mim quando eu não acreditava, obrigada por estar ao meu lado e me ver melhor do que eu sou. Te amo muito!

Aos meus pais, Miguel e Isabel, por dedicarem suas vidas à minha felicidade. Vocês são meus exemplos de vida, fé e amor.Amo vocês incondicionalmente!

Aos membros da pré-banca e aos membros da banca pelas valiosas considerações.

À UNIFAL-MG por fazer eu me sentir em casa.

À UNICAMP pela oportunidade.

À Capes pelo apoio financeiro.

MUITO OBRIGADA!

Resumo

Aproximadamente 2,3 x 10^6 ha da cobertura global das florestas tropicais são alterados anualmente pela fragmentação. O processo de fragmentação tem múltiplos efeitos negativos sobre esses ecossistemas, podendo alterar a diversidade, composição e os processos ecológicos. Entretanto, os efeitos da fragmentação não são homogêneos para todas as espécies, a intensidade desses efeitos sobre as populações depende da biologia das espécies e das características da paisagem. Estudos com diferentes abordagens podem auxiliar muito no entendimento dos mecanismos e processos ecológicos atuantes em sistemas fragmentados, contribuindo para o planejamento do uso da terra e para uma política da conservação da natureza. Diante disso, o objetivo principal deste estudo foi tentar entender como são organizadas as comunidades arbóreas em uma paisagem fragmentada, assim como tentar inferir os processos responsáveis por essa organização. O estudo foi realizado em nove fragmentos florestais de Mata Atlântica semi decídua, ao redor da cidade de Alfenas-MG, inseridos em uma paisagem altamente fragmentada. Dentro de cada fragmento foram instaladas 10 parcelas e todos os indivíduos com altura superior a 1,3 metros foram amostrados. Nossos resultados indicam que mesmo fragmentos localizados em paisagens com baixa cobertura florestal e inseridos em matrizes antrópicas podem ter alta riqueza e diversidade de espécies (capítulo 1). Além disso, verificamos que esses fragmentos apresentam alta diversidade beta e que a variação na diversidade beta é devido à substituição de espécies (“turnover”), e não ao aninhamento. Esses resultados foram encontrados tanto para árvores do dossel quanto para as árvores e arbustos de sub-bosque, oferecendo indícios de que os fragmentos podem manter a alta diversidade ao longo dos anos (capítulo 2). Quanto ao efeito da paisagem sobre os atributos das comunidades de árvores adultas e regenerantes, verificamos que não há relação entre os atributos das árvores adultas e os atributos da composição da paisagem. Por outro lado, os atributos da comunidade de indivíduos regenerantes tem uma relação positiva com a porcentagem de matriz de pasto, contrariando as expectativas. Esse resultado pode ser atribuído ao baixo manejo desenvolvido nas áreas de pastagens da região (capítulo 3). Já quando buscamos entender a variação da diversidade funcional desses fragmentos e como esta é afetada,verificamos que

os indivíduos adultos apresentaram baixa diversidade beta,está havendo uma perda ordenada de funções mais sensíveis, ehá um aumento na abundância das funções mais generalistas (aninhamento). Já o estrato regenerante apresenta menor diversidade funcional que o estrato adulto e alta abundância de indivíduos em funções mais tolerantes. Além disso, a porcentagem de matriz de pasto tem uma relação positiva com a abundância de indivíduos nas funções mais generalistas (capítulo 4). Em outras palavras, quando analisamos apenas a diversidade e riqueza de espécies, os fragmentos apresentam um bom estado de conservação. Entretanto, quando analisamos a diversidade funcional, verificamos que, apesar de haver muitas espécies (alta diversidade de espécies), essas desempenham as mesmas funções mais generalistas (baixa diversidade funcional), indicando que em termos de diversidade funcional, o estado de conservação dos fragmentos é preoucupante.

Abstract

Approximately 2.3 x 10^6 ha of tropical forests are altered annually through fragmentation. Fragmentation has multiple negative effects on forest ecosystems, impacting their diversity, composition, and ecological processes. The effects of fragmentation are not homogeneous for all species, however, and the intensities of those effects on populations will depend on the biology of the species and on landscape composition. Studies using different approaches can greatly aid our understanding of ecological mechanisms and processes in fragmented systems and contribute to land use planning and nature conservation policies. We examined how tree communities in eight Atlantic forest fragments in a highly fragmented landscape are organized, and the processes responsible for that organization. Ten 10x20 m plots were installed in each fragment, and all individuals above 1.3 m in height were sampled. Our results indicated that even fragments within landscapes with low forest cover and within anthropogenically modified matrices can show high species richness and diversity (chapter 1), that they show high beta diversity, and that variations in beta diversity are due to species substitutions (turnover) and not nesting. These results were valid for both canopy and understory plants, suggesting that forest fragments are capable of maintaining high diversity for many years (Chapter 2). In terms of landscape effects on the attributes of adult and sapling tree communities, we found that adult trees were not affected by landscape composition and, contrary to our expectations, saplings were benefited by the pasture matrix. These results can be attributed to the low management intensity of pasture areas in the region (chapter 3). Close examinations of how the functional diversity of these fragments varies and how it is affected by fragmentation, showed that adult individuals demonstrated low beta diversity and were experiencing an orderly loss of more sensitive functions and increases in more tolerant functions (nesting). The sapling layer showed lower functional diversity than the adult layer and high abundance of individuals with more tolerant functions that were benefited by a high pasture matrix percentage (chapter 4). Thus, when we analyzed only species diversity and richness, the fragments appeared to be well-conserved (higt diversity of species), but when their functional diversity was examined we saw that although there are many species, they displayed the same more tolerant functions (low functional diversity).

Sumário

Introdução Geral

As florestas tropicais são os ecossistemas com maior biodiversidade da Terra, reunindo de 20% a 40% de todas as espécies de plantas e animais (Myers 1991). Entretanto, estes ecossistemas encontram-se ameaçados. Aproximadamente 2,3 x 10^6 ha da cobertura global das florestas tropicais são alterados anualmente pela fragmentação (Mayaux *et al*. 2005). O processo de fragmentação tem múltiplos efeitos negativos sobre esses ecossistemas, podendo alterar a diversidade, a composição e os processos ecológicos (Laurence & Vasconcelhos 2009), porém os efeitos da fragmentação não são homogêneos para todas as espécies. A intensidade desses efeitos sobre as populações, tanto de plantas quanto de animais, depende da biologia das espécies, das características do ambiente e da paisagem como, por exemplo, a cobertura florestal e o tipo de matriz (Viana & Tabanez 1996).

Espécies intolerantes à luminosidade direta e ventos fortes, que necessitam de grandes áreas de vida, que produzem frutos grandes e são dipersas por grandes mamíferos são normalmente mais sensíveis aos efeitos da fragmentação que aquelas mais generalistas, que produzem grande número de frutos pequenos e dipersas pelo vento. Essas espécies são as primeiras as serem extintas nos fragmentos ou tem sua distribuição restritaaos interiores dos fragmentos, onde as condições ambientais são mais semelhantes às condições das florestas contínuas. Já as características do ambiente são determinantes para o estabelecimento das espécies, é sabido que independente da fragmentação, características de solo, umidade do ar, relevo, altitute e estágio de sucessão determinam a composição de espécies nos ecossistemas.

Enquanto vários estudos tem investigado os efeitos da fragmentação nas características ambientais e consequentemente na composição de espécies (por exemplo, trabalhos sobre o efeito de borda), os estudos sobre o efeito da composição da paisagem sobre comunidades localizadas em fragmentos florestais ainda são recentes e controversos. De forma geral, fragmentos maiores são mais similares às florestas contínuas e, por esse motivo, tendem a ser menos afetados pela fragmentação (Laurence *et al.* 2012). A redução na cobertura florestal e consequente perda de heterogeneidade de habitats em uma paisagem fragmentada pode diminuir o tamanho das populações (Prugh *et al.* 2009) e o número de espécies encontradas naquele ambiente (Gaston & Blackburn 2000). Já o efeito da matriz depende principalmente da estrutura e do uso da terra ao redor dos fragmentos (Fahrig 2003, Deconchat *et al.* 2009). Matrizes mais abertas, com vegetação rasteira, muito diferente estruturalmente dos fragmentos, como as matrizes de pasto, tendem a ter efeitos negativos sobre os fragmentos florestais. Isso acontece porque essas matrizes podem ampliar o efeito de borda, devido à diferença microclimática entre fragmento e matriz (Mesquita *et al.* 1999) e impedir a movimentação dos animais entre fragmentos, uma vez queessas matrizes não oferecem abrigo e recursos para os animais (Mesquita *et al.* 1999, Hernandez-Ruedas *et al.* 2014). Já as matrizes mais semelhantes aos fragmentos, ou seja, matrizes arbustivas ou arbóreas, como as matrizes de café, tendem a ter efeitos positivos sobre as comunidades nelas inseridas. Essas matrizes podem agir como uma zona tampão, amenizando o efeito de borda, e funcionar como uma extensão do fragmento, fornecendo abrigo e recursos para os animais (Hernandez-Ruedas *et al.*2014). Além desses efeitos, estudos revelam que o tipo de manejo desenvolvido nas matrizes podeafetar os fragmentos (Karp *et al.* 2012, Lindborg *et al.* 2014). De forma geral, cultivos muito mecanizados, em que são utilizadas grandes quantidades de agrotóxicos, têm um efeito negativomaior sobre os fragmentosque cultivos menos manejados (Karp *et al.* 2012, Lindborg *et al.* 2014).

Para as comunidades de plantas, o processo de fragmentação e, consequentemente as mudanças na paisagem, podem gerar dois diferentes cenários: homogeneização de espécies por aninhamento ou diferenciação florística através do “turnover” (troca de espécies) (Baselga 2010, Arroyo-Rodríguez *et al.* 2013, Hernández-Ruedas *et al.* 2014). Homogeneização de espécies por aninhamento é quando os fragmentos possuem espécies muito semelhantes (homogeinização de espécies) e a composição de espécies em um fragmento menos diverso é um subconjunto da composição de espécies nos fragmentos mais diversos (aninahmento). A hipótese da homogeneização prevê que fragmentos inseridos em uma mesma paisagem, com pressões abióticas e bióticas semelhantes (condições ambientais, tipo matriz), terão dinâmicas e trajetórias similares na mudança da composição de espécies (Laurance *et al.* 2007). As espécies sensíveis serão extintas localmente e quanto maior as pressões sobre os fragmentos, mais espécies desaparecerão de forma ordenada(Laurance *et al*. 2007).

Consequentemente, ao longo do tempo, os fragmentos se tornarão homogêneos e aninhados.

Por outro lado, a diferenciação florística através do turnover é ocorre quando os fragmentos possuem composição de espécies distinta (diferenciação florística) em consequêcia da troca de espécies (turnover). Esse processo ocorre em fragmentos que sofrem diferentes pressões ambientais e/ou estão localizados em uma paisagem heterogênea (Laurance *et al.* 2007). Neste cenário, a diferença inicial na composição florística dos fragmentos será amplificada ao longo do tempo (Laurence *et al.* 2007). A heterogeneidade original, combinada com a heterogeneidade da paisagem e os diferentes tipos de perturbações, irão gerar comunidades com diferentes composições florísticas (Baselga 2010).

Um estudo recente mostrou homogeinização da flora em fragmentos localizados em paisagens modificadas (Arroyo Rodriguez *et al.* 2013) colocando em dúvida a manutenção da diversidade em paisagens fragmentadas ao longo do tempo e o custo benfício de se preservar esses fragmentos. Entretanto, outros estudos (Santos *et al.* 2006, Hernández-Ruedas *et al.* 2014) mostram que a homogeneização florística nem sempre é predominante e que pequenos fragmentos inseridos em paisagens humanas modificadas podem manter a biodiversidade. Diante desse contexto, conhecer a ecologia das localizadas em fragmentos florestais e a influência da paisagem nessas comunidades é de extrema importância, tanto para um entendimento teórico dos processos de homogeneização e diferenciação florística quanto para estabelecer ações para conservação da biodiversidade.

Uma maneira adequada de buscar respostas iniciais de organização em comunidades inseridas em ambientes fragmentados é por meio de estudos fitossociológicos. Estudos de florística e fitossociologia buscam conhecer as espécies presentes em dada área e a estrutura desta vegetação, ou seja, esses estudos quantificam a riqueza, o número de indivíduos por espécie, suas respectivas classes de tamanho e o arranjo espacial dos mesmos. Em florestas tropicais este tipo de informação, assim como dados sobre a riqueza e diversidade, reflete não só fatores sucessionais, evolutivos e biogeográficos, como também o histórico de perturbação das respectivas áreas (Joly *et al.* 2012). O conhecimento gerado pelos estudos florísticos em comunidades arbóreas fragmentadas é considerado fundamental e prévio para embasar qualquer outro estudo, bem como medidas de manejo, recuperação e conservação das florestas tropicais (Ribas *et al.* 2003).

Uma vez compreendidos os padrões iniciais de organização das comunidades fragmentadas, é necessário compreender como os padrões de diversidade variam através de diferentes escalas espaciais (Summerville *et al.* 2003). Uma maneira de abordar este problema é através da partição aditiva de diversidade (Lande 1996), em que a diversidade total (gama) é dividida entre a diversidade dentro (alfa) e entre (beta) amostras. A maneira como a diversidade de espécies se distribui em diferentes escalas pode ter uma grande relevância para testar teorias ecológicas e entender os fenômenos de homogeneização e diferenciação florística (Laurance *et al.* 2007, Baselga 2010). De forma geral, fragmentos homogeneizados apresentam baixa diversidade beta, enquanto na diferenciação florística a diversidade beta é alta (Laurance *et al.* 2007, Arroyo-Rodríguez *et al.* 2013).

Além disso, a partição aditiva da diversidade permite que se amplie o conhecimento dinâmica de troca de espécies em ambientes fragmentados, fornecendo bases consistentes para a criação de unidades de conservação. Em um cenário de homogeneização, a melhor estratégia para a conservação é preservar os remanescentes mais diversos, uma vez que os fragmentos menos diversos são apenas subcoinjuntos (Wright e Reeves 1992, Laurence *et al.* 2007). Já em um cenário de diferenciação florística, a conservação da biodiversidade depende da proteção do maior número possível de fragmentos, uma vez que cada fragmento possui uma composição única de espécies e são, portanto, complementares (Wright & Reeves 1992, Laurance *et al.* 2007).

Para evitar perda de informação em estudos sobre as comunidades fragmentadas dois aspectos, pouco abordados, precisam ser considerados: o estudo do estrato regenerante e da diversidade funcional. O componente florestal adulto é frequentemente o mais enfatizado em estudos ecológicos porque é onde se concentra a maior parte da biomassa florestal, e a de maior importância econômica (Meira Neto & Martins 2003). Entretanto, árvores de florestas tropicais têm um longo tempo de vida (Martinez-Ramos & Alvarez-Buylla 1998) e os indivíduos adultos presentes num fragmento podem ser representativos de um tempo em que as condições ambientais eram outras, como por exemplo, quando o ambiente ainda não havia sido fragmentado. Dessa forma, efeitos de alterações ambientais em árvores em ambientes fragmentados podem ser mais bem expressos pelas populações do estrato regenerante (Hill & Curan 2003).A regeneração natural é um importante processo na evolução das florestas tropicais (Narvaes *et al.*2008), este estrato representa indivíduos muitas vezes mais vulneráveis a impactos antrópicos. Estudos sobre o estrato regenerante também permitem inferir sobre o estado de conservação dos fragmentos, bem como a resposta do fragmento às perturbações naturais ou antropogênicas (Silva *et al.* 2007), uma vez que, em muitos casos, os indivíduos jovens se estabeleceram nesses ambientes sob os efeitos da fragmentação. A regeneração natural ainda constitui um indicador de suma importância na avaliação e monitoramento de áreas em processo de recuperação (Rodrigues *et al*. 2004).

Já a diversidade funcional tem um maior significado biológico que as medidas tradicionais de diversidade, pois é um conceito que liga indivíduos ao ambiente em que estão inseridos (Petchey & Gaston 2006). Uma forma de acessar a diversidade funcional é classificando as espécies presentes nas comunidades em grupos funcionais relacionados à sobrevivência, crescimento e reprodução (Sobral & Cianciaruso 2012). Essas informações são úteis para inferir sobre os processos de organização das comunidades, uma vez que características funcionais fornecem informações sobre o uso de recursos e requerimentos de habitat (Nason *et al.* 2007). Além de ser uma abordagem que nos permite compreender osprocessos responsáveis pela composição e estrutura das comunidades(Gomez 2010), a diversidade funcional também pode ser usada para fazer previsões ecológicas além da escala de regiões. O estudo das respostas vegetais às mudanças ambientais em uma escala global não pode se basear em espécies, uma vez que a maioria delas apresenta distribuição geográfica limitada. Para este fim, o uso de grupos funcionais é melhor, uma vez que agrega um grupo de plantas que afetam de modo similar o ambiente ou que apresentam uma resposta similar às mesmas variações ambientais em qualquer escala (Pillar 1999).

Entretanto, apesar de muitas vantagens, o uso da diversidade funcional, assim como o uso das medidas tradicionais de diversidade possui limitações. Espécies de um mesmo grupo não são completamente redundantes e espécies de grupos diferentes não são completamente diferentes, o que determina a diferença é a característica/função estudada. Não havendo redundância absoluta entre espécies, os agrupamentos não permitem prever o impacto real da perda ou acréscimo de espécie no funcionamento da comunidade, pois caracteres não incluídos na construção dos grupos produzem efeitos que vão para além dos previstos, criandoconsequências comunitárias inesperadas. Dessa forma, para um resultado mais próximo da realidade, o ideal são estudos com diferentes abordagens.

Diante deste contexto, o objetivo principal deste estudo é tentar compreender como a composição de uma paisagem fragmentada tem alterado as comunidades arbóreas dentro dequatro abordagens: descrição fitossociológica da comunidade, partição aditiva da diversidade,comunidades regenerantes e diversidade funcional. Buscamos também entender como são organizadas comunidades arbóreas em paisagens fragmentadas, assim como tentar inferir os processos responsáveis por essa organização.Florestas tropicais são ecossistemas complexos e ameaçados (Myers 1991), e estudos com diferentes abordagens são necessários para o entendimento dos mecanismos e processos ecológicos atuantes nesses sistemas, contribuindo para o planejamento do uso da terra e para uma política da conservação da natureza (Gomes *et al.* 2004).

A tese foi dividida em quatro capítulos. No Capítulo 1 intitulado “A importância ecológica para conservação da biodiversidade tropical de fragmentos florestais inseridos em paisagens humana modificadas” o nosso objetivo foi verificar o valor ecológico de pequenos fragmentos de floresta para a conservação da biodiversidade considerando a riqueza e diversidade de espécies, assim como a estrutura da vegetação(área basal, altura e diâmetro dos indivíduos). Nossa hipótese era que dada as características dos fragmentos estudados (pequenos [<100 ha], isolados [distância média de 28 km entre fragmentos] e inseridos em paisagens muito manejadas) nós encontraríamos riqueza e diversidade de espécies baixa e baixo número de espécies exclusivas e raras. Da mesma forma a estrutura da vegetação apresentaria características de perturbação recente, tais como área basal baixa e alta densidadede indivíduos curtos e finos. Entre as contribuições desse estudo podemos destacar: (i) informações sobre a flora regional; (ii) aumento do conhecimento sobre a distribuição geográfica e abundância das espécies e (iii) importância da amostragem do sub-bosque.

No Capítulo 2 intitulado “Substituição de espécies mantém alta diversidade em assembleias de árvores em uma paisagem tropical fragmentada” nosso objetivo foi analisar a partição aditiva de diversidade dos indivíduos de dossel e sub-bosque e decompor a diversidade beta em seus componentes de “nestedness” (aninhamento) e “turnover” (substituição de espécies) para responder as seguintes questões: (i) o padrão de partição aditiva da diversidade dos indivíduos de dossel é semelhante ao padrão dos indivíduos de sub-bosque?; (ii) a diversidade beta dos indivíduos de dossel e sub-boque é devido ao “nestedness” ou “turnover”?; e (iii) a diversidade beta é explicada por alguma característica da paisagem?. Nós acreditamos que os resultados deste estudo possam contribuir paramelhorar o entendimento sobre (i) o processo de homogeneização florestal em fragmentos florestais tropicais e (ii)os processos responsáveis pela variação da beta diversidade.

No Capítulo 3 intitulado “Em uma paisagem fragmentada, a presença de pasto modifica os atributos da comunidade de árvores regenerantes, sem causar homogeneização de espécies”, nosso objetivo foi investigar como a configuração da paisagem (cobertura florestal e tipos de matriz)está relacionada aos atributos (riqueza, abundância e diversidade) das comunidades regenerantes e adultas. Nossa hipótese é que matrizes estruturalmente diferentes dos fragmentos, como pastagem e cana-de-açúcar, tenham uma relação negata com os atributos das comunidades,enquanto que as matrizes estruturalmente semelhantes ao fragmento, como a matriz de café, afetem tenham uma relação positiva com os atributos. A principal contribuição deste trabalho é tentar compreender como a composição da paisagem se relacionacomos atributos das comunidades adultas e regenerantes localizadas em fragmentos florestais.

No Capitulo 4 intitulado “Perda aninhada de diversidade funcional devido a pastagem em uma paisagem altamente fragmentada” nosso objetivo foi analisar e comparar a diversidade funcional do estrato adulto e regenerante e verificar como esse atributo está relacionado à configuração da paisagem. Nossa expectativa é que os estratos sejam diferentes em termos de composição funcional. Acreditamos ainda que a fragmentação irá afetar negativamente as funções mais específicas (dispersão por grandes mamíferos, grandes frutos, sistemas sexuais auto-incompatíveis) e favorecer o estabelecimento de funções mais generalistas (pequenos frutos, polinização por insetos generalistas, árvores pioneiras) na comunidade regenerante.

Área de estudo

O estudo foi realizado em nove fragmentos florestais localizados no sul de Minas Gerais, ao redor da cidade de Alfenas (21º 25’ 45’’ S e 45º 56’50’’ W) (Fig. 01).

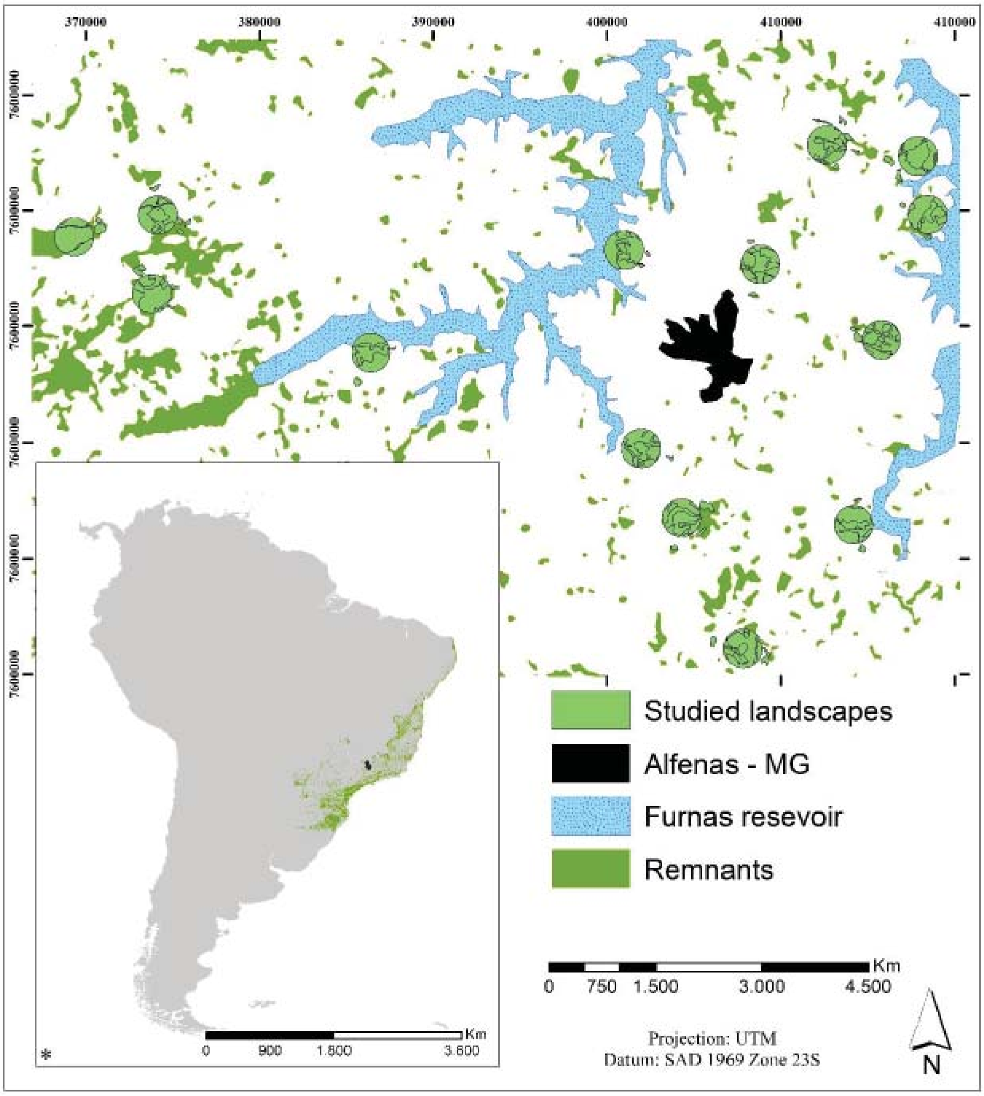

Figura 01- Localização dos noves fragmentos estudados. Fonte: Raniero 2015.

Minas Gerais é o estado com maior área da região Sudeste e o quarto maior do Brasil. Este estado abriga uma grande variedade de biomas, como Campos Rupestres, Cerrado e Mata Atlântica e, apesar da grande diversidade, somente 2,3% do seu território é legalmente protegido, sendo que apenas 0,65% está sob proteção integral.Alfenas é um dos 34 municípios que compõe o circuito dos Lagos da Represa de Furnas, a cidade é considerada um núcleo urbano de bastante importância na região sul mineira e tradicionalmente um grande centro produtor de café. A agricultura e a pecuária contribuem com 15% do produto interno bruto do município (IBGE 2017).

A fragmentação na região começou no final do século 18 e início do século 19 com a expansão das lavouras de café (Castilho 2009). Atualmente a região possui aproximadamente 9% de floresta nativa, localizada em fragmentos florestais pequenos (97% dos fragmentos são menores que 50 ha) e isolados (Fundação SOS Mata Atlântica). Esses fragmentos possuem diferentes graus de perturbação e estão localizados em paisagens com diferentes contextos.O uso do solo assim como a porcentagem de floresta nativa variou pouco nos últimos anos (1986-2011). A porcentagem de floresta nativa vaiou entre 9a 13 %, a porcentagem de matriz de cana variou de 3,4 a 4,5 %, de café variou de 8,8 a 12,6% e a porcentagem de pasto variou de 62,1 a 67,9%. (Olivetti 2015).

A seleção dos fragmentos foi realizada através de uma classificação das áreas florestais remanescentes no município de Alfenas, utilizando o processamento digital de imagens de satélite Sino-Brasileiro CBERS-2B do ano de 2011, com resolução de 20 metros. Os atributos considerados para a seleção foram: (a) similaridade no nível de degradação dos remanescentes (aspectos semelhantes de textura, forma, cor), (b) tamanho do fragmento entre 15 ha e 100 ha e (c) distância superior a 4 quilometros entre os fragmentos para garantir independência dos dados obtidos.

Os fragmentos selecionados são remanescentes de Mata Atlântica Semidecídua (Oliveira et al. 2001). As famílias e espécies mais abundantes são, respectivamente: Fabaceae, Myrtaceae, Lauraceae, Meliaceae, e Euphorbiaceae, e *Copaifera langsdorffii*, *Ocotea odorifera*, *Cryptocarya aschersoniana*, *Metrodorea stipularis*e *Miconia willdenowii*. Os fragmentosfazem parte da Bacia Hidrográfica do Rio Grande, cujo afluente direto é o Reservatório da Usina Hidrelétrica de Furnas. A região está situada nos limites meridionais da zona intertropical e, devido à elevada altitude da região, que varia de 720 to 1350 m, o clima é do tipo tropical mesotérmico(Alvares et al. 2014). A temperatura anual média é de 19 °C, o verão e a primavera são as estações mais quentes, com máximas diárias variando de 28 a 30 °C. No inverno a temperatura pode chegar a 0 °C, resultando em geadas. Em relação ao regime de chuvas, o clima é úmido, com precipitação média anual de aproximadamente 1 590 mm.

Todos os fragmentos estudados são pequenos (menores que 100 ha), estão localizados em propriedades privadas de pequeno a médio porte, estão inseridos em matrizes antrópicas e aparentemente não possuem uso, exceto o fragmento 5, que possui trilhas para caminhadas (Tabela 01).

Tabela 01: Área dos fragmentos e porcentagem de diferentes matrizes ao redor dos fragmentos

Frag-mento	Área (ha)	% de cobertura florestal	% de café	% de cana	% de pasto
1	20,91	13.39	15.72	31.39	33.14
2	81,55	9.38	51.83	0.00	14.02
3	56,05	24.54	16.83	45.13	10.5
4	36,85	13.66	62.87	0.00	13.74
5	37,05	37.05	0.00	58.33	0.00
6	22,99	25.98	21.19	36.96	8.18
7	87,18	26.43	1.17	0.00	51.56
8	24,80	27.75	0.00	32.51	2.3
9	28,57	26.00	25.00	49.00	0.00

Fonte: Dados trabalhados pelo autor.

Os fragmentos também não apresentam sinais aparentes de corte seletivo, queimadas ou caça (observação pessoal). A produção de café e cana de açúcar ao redor dos fragmentos é voltada para o comércio,d essa forma, essas culturas são muito manejadas. Isto é, são culturas mecanizadas, onde são usados agrotóxicos e fogo periodicamente (observação pessoal). Já as pastagens são para consumo próprio ou pequeno comércio. Essas matrizes possuem muitas árvores(Gonçalves 2015), poucos animais e baixo manejo (não há uso de máquinas, agrotóxicos e fogo) (observação pessoal).

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CAPÍTULO 01

The ecological importance for tropical biodiversity conservation of forest fragments embedded in human-modified landscapes [1]

[1] **Nos moldes da revista Biodiversity and Conservation**

The ecological importance for tropical biodiversity conservation of forest fragments embedded in human-modified landscapes

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Acknowledgements

We thank Felipe D., Pimenta E., Carvalho A., and Alcantara M. for help with the field work, and Stephen Lathan for correcting the English. This research was supported by grants from the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) to Magda Silva Carneiro, the Fundação de Amparo a Pesquisa do Estado de Minas Gerais (FAPEMIG)/Vale Company (RDP-00104-10), and the Conselho Nacional de Desenvolvimento Cientifico e Tecnológico (CNPq) (472250/2010-8 and 306595/2014-1).

Abstract

The ecological importance of large reserves for biodiversity conservation is unquestionable, however, as forests are increasingly being fragmented globally, it is important to consider the ecological importance for biodiversity conservation of small forest fragments. The aim of the present study was to assess the ecological importance for biodiversity conservation of small semideciduous forest fragments (< 100 ha) in south-eastern Brazilian Atlantic Forest. For this we inventoried all individuals taller than 1.3 m in nine Submontane Semideciduous forest fragments.Despite the presence of some exotic species, and the high numbers of individuals of a few dominant species, the forest remnants demonstrated high diversity, with many rare, exclusive, and threatened species. We demonstrated the importance and structural integrity of remnant forest areas, with the expectation that they will be incorporated into future conservation planning and initiatives.

Keywords: Biodiversity conservation, floristic composition, fragmentation processes, vegetation structure, tropical forests.

Introduction

Fragmentation studies increasingly focus on the impact of forest fragmentation on biodiversity conservation (Santos et al. 2006; Melo et al. 2013; Bregman et al. 2014). The reason for this growing interest is the finding that, while historically marginalized in conservation initiatives (Gibson et al. 2011), most biodiversity currently resides within small forest fragments on private lands (Perfecto and Vandermeer 2008). Fragmentation has multiple negative effects on forest ecosystems, altering their diversity and composition and impacting the ecological processes of the remnant communities (see Laurance et al. 2011 for a review). Studies have shown that fragmentation leads to impoverished communities with few dominant and generalist species, and have few rare and/or threatened taxa (Oliveira et al. 2004; Laurance et al. 2007; Arroyo Rodriguez et al. 2013).

The impoverishment of plant communities in these fragments is directly caused by area reduction (deforestation) and habitat losses, and indirectly by changes in the microclimate or mutualistic relationships, such as pollination and seed dispersal (Fahrig 2003). Additionally, as a response to changes in micro-climate, plants in fragmented environments have lower total heights and diameters, show greater distances between individuals, and show less canopy overlapping (Bruna and Kress 2002). Although fragmentation can cause all of these alterations that lead to degradation, some authors have argued that forest fragments nonetheless are important and merit conservation efforts (Santos et al. 2006; Hernadez-Ruedas et al. 2014). Small forest patches can serve as stepping stones and food sources for numerous species and contribute to maintaining a certain degree of regional spatial heterogeneity and thus species diversity (Hernandez-Ruedas et al. 2014). Small fragments, therefore play a key role in connecting larger fragments and continuous forest areas and contributing to gene flow between populations (Perfecto and Vandermeer 2010). Within this context, it will be important to conduct more studies that encompass landscapes with contrasting management histories and land use intensities. In particular, an assessment of community structure (richness, diversity, evenness, floristic composition) in small fragments and biodiversity conservation value (numbers of rare, threatened, or exclusive species) are required, especially in tropical biodiversity hotspots (Hernadez-Ruedas et al. 2014).

Most studies on forest fragmentes have sampled adult trees (DBH > 10 cm) (Meira Neto and Martins 2003), thus usually samling individuals that established before fragmentation (Martinez-Ramos and Alvarez-Buylla 1998). Although to better characterize diversity and better understand the long-term effects of fragmentation on biodiversity it is important to include seedlings and saplings in studies (Sales and Schiavini 2007). Seedlings and saplings represent the regenerative potential of the forest, making it possible to construct inferences about forest regeneration and the maintenance of biodiversity in the medium and long term (Alves and Metzger 2006; Salles and Schiavini 2007).

In this study, we assessed the ecological importance for biodiversity conservation of small fragments of Atlantic Semideciduous forest(< 100 ha).For this we sampled all individuals with DBH greater than 1.3 m, describe the floristic composition,calculate the richness and diversity of species and verify the presence of rare and endangered species. We also described the vegetation structure (diameter, height, basal area and density) of the and compared our results with data found in well-preserved areas of Semideciduous

Atlantic Forest. This survey intended to: (i) provide information concerning the regional flora; (ii) increase our knowledge of the geographical distributions and abundances of species; (iii) serve as a reference for future and more applied works; and; (iv) attempt to clarify the potential roles of forest fragments in biodiversity conservation.

Methods

Study area

Our study was undertaken in nine Semideciduous Atantic forest fragments in Alfenas, southern Minas Gerais State, Brazil (21º25'45"S x 45º56'50"W) (Table 1). The distance between fragments varied from 3.1 to 49.6 km and the area of fragments varied form 20.9 to 87.2 ha. The regional climate is classified as Cwa with average winter temperature is 16.9 ºC and 21.5 ºC in summer, and the average rainfall is 26 mm during the winter and 290 mm during the summer (Alvares et al. 2014). Regional elevations vary from 720 to 1350 m a.s.l. in a predominantly hilly landscape associated with mountains.The region retains only about 9% of its native forest cover, with the most common matrix types being pasture (51%), permanent cultivation (mainly coffee - 17%), and temporary crops (mainly sugar cane and corn - 7%) (Olivetti 2014).

Sample design

Ten randomly placed 10 x 20 m plots (0.2 ha per fragment) were installed in each forest fragment studiedat least 10 m from the edge into the fragment interior. All individuals taller than 1.3 m were sampled and measured within each plot. For practical reasons, the individuals were divided into two classes:

those with diameters at breast height (DBH) ≥ 5 cm (hereafter referred to as canopy individuals); and those with DBH < 5 cm (hereafter referred to as understory individuals). Although many young trees in the understory category will achieve forest canopy height when fully grown, and there were many tall adult shrubs placed in the canopy category, we established these two classes to facilitate the interpretation of our results.

Botanical identifications were made in the field; when this was not possible, samples were collected for posterior identification by specialists. We used the Angiosperm Phylogeny Group III (APG IV 2014) system for nomenclature and 97% of species are identified. Conservation status was evaluated by using the lists of endangered species of Minas Gerais State and Brazil (Biodiversitas 2007).

Statistical analyses

Phytosociological parameters were calculated using the Fitopac software (Shepherd 2010). Species frequencies (log10 transformed) were correlated with abundance, using the Pearson's correlation coefficient, in BioEstat 5.3 software (Ayres 2007). Species richness was estimated using rarefaction curves, employing the Bootstrap estimator in the Estimate software (Colwell 2013). This estimator is indicated for samples with large numbers of rare species (Magurran 2004).

Size structures (heights and diameters) were verified using the Gini coefficient and the Lorenz curve. The Gini coefficient (G) is a measure of inequality, which can vary from 0 to 1; values near 0 indicate that individuals in the community are of similar dimensions, while values near 1 indicate large differences between them. In the Lorenz curve, the data (Cumulative size x Cumulative number of individuals) is ordered from the lowest to the highest (height or diameter) in a cumulative manner. If all individuals are of the same size/diameter, the resulting plot will be a diagonal line; if the individuals are of unequal sizes/diameters the curve will fall below the diagonal. Greater spacing between the sizes/diameters reflects more unequal sizes/diameters (Wiener and Solbrig 1984). These analyses were made using the "redist" and "boot" packages in the R environment.

Results

We recorded 8338 individuals belonging to 60 families, 156 genera, and 310 species, only 3% of individuals were not identified. Of this, 167 (53.87%) species and 46 (76.67%) families were common to the canopy and understory layers. Large numbers of species (38.6%) and families (20.7%) occurred only in the understory, while two families (16.4%) (Caricaceae and Rhamnaceae) and 38 (18.5%) species occurred only in the canopy. The families Anacardiaceae, Annonaceae, Euphorbiaceae, Fabaceae, Lauraceae, Meliaceae, Myrtaceae, Salicaceae, and Rubiaceae, as well as the species *Duguetia lanceolata* and *Copaifera langsdorffii* occurred in both layers and in all fragments (Table 2).

Canopy stratum - Individuals with DBH ≥ 5 cm

We sampled 1985 individuals of the canopy stratum of the nine fragments with DBH ≥ 5 cm, distributed among 205 species, 115 genera, and 48 families. Twenty-one families (43.7%) were represented by only a single species, while the richest families were Fabaceae (14.15%), Myrtaceae

(10.73%), Lauraceae (9.27%), Meliaceae (5.85%), and Euphorbiaceae (5.85%). These five families greatly contributed to regional tree diversity, comprising 45.8% of all of the species (Table 2). Anacardiaceae, Annonaceae, Euphorbiaceae, Fabaceae, Lauraceae, Meliaceae, Myrtaceae, Rubiaceae, and Salicaceae (18.7%) were the most frequent families, occurring in almost all of the fragments (Table 2).

Copaifera langsdorffii, *Ocotea odorifera*, *Cryptocarya aschersoniana*, *Metrodorea stipularis*, and *Miconia willdenowii* had the highest IVI (Online Resource 1);*Duguetia lanceolata*, *Cryptocarya aschersoniana*, *Copaifera langsdorffii,*and *Ocotea corymbosa* were the most frequent species in the nine fragments, representing 1.9% of the total richness and 13.5% of the total number of individuals sampled. The most abundant species were also the most frequent ($rPearson_{203}$=0.94; p<0.0001). Some species, however, had high abundances but low frequencies such as *Vernonanthura divaricata, Platypodium elegans,* and *Piptadenia gonoacantha*, while others had low abundances and high frequencies, such as *Dalbergia villosa, Tapirira obtusa,* and *Guatteria australis* (Table 2). The fragments had high numbers of exclusive species (species occurring in only one fragment), with approximately 33% of the species in each fragment and 40.5% of the total number of species being exclusive.

The total species richness of the study area (1.8 ha), as estimated using the Bootstrap estimator, was 239.05 (Online Resource 2), while Shannon’s diversity index was 4.38, and Pielou’s evenness index was 0.82. The richness estimated by Bootstrap for each plot (0.2 ha) varied from 46.5 to 105.3, while Shannon’s diversity index (H’) ranged from 3.17 to 4.03 nats/ind, and Pielou’s evenness (J') index ranged from 0.82 to 0.91 (Table 3). In addition to retaining large numbers of species, the areas held many endangered species. *Euterpe edulis*, *Ficus cyclophylla,* and *Cedrela fissilis* are classified as vulnerable to extinction, while *Ocotea odorífera* is endangered in Brazil, and *Persea rufotomentosa* and *Ocotea odorífera* are classified as vulnerable in Minas Gerais State (Online Resource 3).

The canopy stratum of the fragments demonstrated generally low basal areas and high densities compared others studies. The total area studied (1.8 ha) had a mean total basal area of 28.10 m^2 ha^{-1} and an overall density of 1102.78 ind ha^{-1}. Among the different fragments, basal areas ranged from 17.8to 46.2 m^2 ha^{-1},while their densities ranged from 845 to 1400 ind ha^{-1} (Table 3). The sampled individuals were found to be generally low (mean = 9.2±3.9 m tall) and thin (mean = 13.8±11.6 cm in diameter); there were no large variations among individuals either in terms of their heights (G=0:26; CI=0.22 to 0.24) or diameters (G=0.36; IC=0.34 to 0.37) (Online Resource 4).

Understory stratum - Individuals with DBH ≤ 5 cm

The 6353 individuals recorded in the understory, belonged to 272 species, of 145 genera in 58 families. Eighteen families (31.0%) were represented by a single species. Conversely, 40.8% of the species belonged to five families: Myrtaceae (13.9%), Fabaceae (11.0%), Rubiaceae (5.5%), Lauraceae (5.1%), and Melastomataceae (5.1%) contributing (Table 2). Fabaceae, Myrtaceae, and Rubiaceae were the most frequent families, occurring in almost in every fragment (Table 1).

Siparuna guianensis, Psychotria vellosiana, Trichilia emarginata, Cupania zanthoxyloides, and *Ocotea odorifera* had the highest IVI (Online Resource 5). *Duguetia lanceolata*, *Croton floribundus*, *Copaifera langsdorffii, Ocotea odorifera,Mollinedia widgrenii, Amaioua guianensis, Psychotria vellosiana, Casearia decandra, Casearia sylvestris*, and *Siparuna guianensis* were the most frequent species in the nine fragments, representing 3.7% of the total richness and 33% of the total number of individuals surveyed. The most abundant species were also the most frequent (rPearson$_{270}$=0.73; p<0.0001). Some species showed high abundance but low frequency, including *Trichilia clausseni, Piper amalago, Galipea jasminiflora,* and *Trichilia elegans*; others showed low abundance and high frequency, such as *Tapirira obtuse*, *Cordia sellowiana*, *Eugenia florida,* and *Ilex cerasifolia* (Table 2). The fragments had large numbers of species that occurred in one fragment only), with approximately 27% of the species in single fragments and, on the whole, 34.2% of the total number of species being exclusive.

When the nine sites where considered as a single sample (1.8 ha), the total species richness, as estimated using the Bootstrap estimator, was 304.5 (Online Resource 2); Shannon's diversity index was 4.26 nats/ind, and Pielou's evenness index was 0.76. Richness, as estimated by Bootstrap, for each fragment (0.2 ha) ranged from 96.8 to 147.3, Shannon's diversity (H') ranged from 3.18 to 4.15 nats/ind, and Pielou's evenness (J') ranged from 0.67 to 0.85 (Table 3). Among the large numbers of species, the fragemntscontained numerous endangered species, such as species in a vulnerable conservation status (*Ficus cyclophylla, Myrcia diaphana, Rudgea jasminoides,* and *Cedrela fissilis*) or are threatened with extinction in Brazil(*Cariniana legalis, Eugenia* *malacantha,* and *E. malacantha*). In addition,*Persea rufotomentosa* and *Ocotea odorifera* are classified as vulnerable in Minas Gerais State (Online Resource 3)

The understory stratum of the fragments generally showed low basal areas and high densities compared others studies. The study area (1.8 ha) had a mean basal area of 1.38 m^2 ha^{-1} area and total density of 3529.4 ind ha^{-1}. The basal areas of the fragments ranged from 0.85 m^2 ha^{-1} to 2.2 m^2 ha^{-1},while plant density at each site (0.2 ha) ranged from 2085 ind ha^{-1} to 5770 ind ha^{-1}(Table 3). The individuals surveyed were generally short (mean = 3.3 ± 1.9 m tall) and thin (mean = 2.0 ± 0.98 cm in diameter) and there were no large variations between individuals either in terms of height (G=0.23; CI=0.23 to 0.24) or diameter (G=0.26; IC=0.25 to 0.26) (Online Resource 6).

Discussion

The fragments studied presented high species richness and diversity, with many rare and/or exclusive species, as well as endangered taxa; few species were very abundant, and some were exotic. In general, the fragments had low basal areas, with high densities of small and thin individuals,compared others studies, which indicated that these areas were in a recuperation phase following a recent disturbance. The understory contributed large numbers of exclusive species to the total species richness of the fragments, and the inclusion of this stratum in the sampling increased richness by 34%. More than half of canopy species (54%) were also encountered in the understory, indicating regeneration, so the fragments will be capable of attaining high diversity over time.

The small fragmented landscape patches in the present study showed high diversity and richness, compared with those in well-preserved areas or large fragments(Meira Neto and Martins 2000; Werneck et al. 2000; Botrel et al. 2002; Dalanesi et al 2004; Spósito and Stehmann 2006; Maragon et al. 2008). The fragments studied here showed large numbers of rare (one individual) and exclusive species (present in only a single fragment) that contributed to the high observed diversity. The species diversity (H'=4.38) and estimated richness (239) observed in the present study were similar to those of other relatively well-preserved areas of Semideciduous Atlantic Forest, whose diversity varies between 3.2 and 4.2(Meira Neto and Martins 2000; Werneck et al. 2000; Botrel et al. 2002; Dalanesi et al 2004; Spósito and Stehmann 2006; Maragon et al. 2008).

Additionally, the fragments still preserved many families and species known from the surrounding regional semideciduous forests. The families Myrtaceae, Fabaceae, and Lauraceae showed the highest numbers of species in both strata in almost all the fragments. These families are commonly encountered in mature stage semideciduous tropical forests (Oliveira Filho and Fontes 2001). Likewise, *Ocotea odorifera*and *Copaifera langsdorffi* which were most important species in almost all the fragments, are likewise found in large numbers in other large and well-preserved semideciduous forests (Werneck et al 2000; Dalanesi et al. 2004; Marangon et al. 2008). The fragments studied here still preserved nine endangered species in both their canopies and understories, demonstrating the importance of these fragments to the conservation of regional biodiversity; two of those species (*Euterpe edulis* and

Ficus cyclophylla), however, showed few individuals in the understory and none in the canopy stratum.

The fragments also presented characteristics of disturbed areas, however, such as invasions by exotic species and high densities of thin and short individuals (Nunes et al. 2003). One of the most abundant species in the understory stratum was *Coffea arabica* (coffee), an exotic plant. This species represents a biological invasion problem that has been described in other semideciduous forest sites (Durigan et al 2000; Martins and Rodrigues 2002). The principal concern is that this species interferes with the processes of natural succession through competition with native species (Guaratini et al 2008).

The size structure of the plant community studied here also indicated disturbance(Uhl and Murphy 1981; Parthasarathy 1999; Nunes et al. 2003). The basal areas and densities of individuals were among the largest found in Semideciduous Atlantic forest sites (Oliveira Filho *et al*. 1994; Botrel et al. 2002; Sousa et al. 2003; Dalanesi et al. 2004; Oliveira- Filho et al. 2004; Pinto et al. 2008; Dias Neto et al. 2009; Lopes et al. 2011) – although when *Ficus trigona* and *F. gomelleira* (species with extremely large DBH and only one individual each in the total sample) were removed from the analysis, the observed basal area was actually among the lowest yet reported. High tree densities with low basal areas indicate the densification of thinner individuals – characteristic of environments that have suffered recent disturbances and are in the process of regeneration (Uhl and Murphy 1981; Parthasarathy 1999; Nunes et al. 2003). Two other aspects of this study reinforce this conclusion: the average height of all individuals was 9.2±3.9 m, and only two fragments had emergent individuals (more than 20 m tall); most individuals had thin stems (mean: 13.8±11.6 cm) – high densities of small, thin trees are indicative of recent disturbance (Nunes et al. 2003). Some authors have argued that is possible to infer the regeneration status of a plant community based on its structure (Clark 1994; Hutchings 1997; Condit et al. 1998), as structure reflects the biotic and abiotic factors influencing the plant populations. Areas studied (Atlantic forest) by Nunes et al. (2003) that had suffered recent disturbances had basal areas, densities, heights and diameters similar to those observed in the present study. Neves and Peixoto studying fragments of Atlantic forest (2008) found that the canopy stratum of recently disturbed fragments (<20 years) had thinner and shorter trees than older disturbed sites (> 40 years).

The fragments studied here were structurally very similar to other tropical fragments in that they were relatively small, isolated, and embedded in anthropogenic matrices (Joly et al. 2014), although our results contrast with most of those findings. While many authors have reported fragmented communities with low diversities and that were dominated by pioneers species (Silva and Tabarelli 2000; Laurance et al. 2006; Santos et al. 2008), we found that our forest remnants had high levels of diversity, with many rare, exclusive and threatened species, despite the presence of some exotic taxa and many individuals of just a few dominant species. Our hypothesis for these results is than the landscape type and land-use history of the matrix in which these fragments are inserted are helping to maintain species diversity.Small-scale agricultural practices dominate, with different cultivation schemes on most properties and heterogeneous and structurally varied matrices (in terms of plant sizes and crown coverage) that are structurally similar to forest fragments would tend to attenuate edge effects. This type of matrix could function as a fragment extension, providing shelter and food for wildlife while facilitating dispersal between remnants (Mesquita et al. 1999), and the remnants themselves, being relatively close to one another, would facilitate the survival and maintenance of species and their exchange between fragments. Our fragments did not show evidence of severe disturbance, such as selective logging or the passage of fire, although cattle had apparently grazedsome areas. These factors, combined with the fact that the understories showed high diversity, reinforce the view that these fragments will be capable of maintaining their diversity for many years and thus contribute to biodiversity conservation.

Conservation implications

The preservation and restoration of forest fragments is currently a central issue in discussions of biodiversity conservation, as most tropical biodiversity is now located in these types of environments (Melo et al. 2013). We have shown that forest fragments can maintain high species richness and diversity and therefore have high conservation ecological value, with the nine endangered species identified in the study fragments reinforcing their conservation priority. Additionally, these fragments are representative of regional landscape, help maintain ecosystem services, serve as stepping stones, and provide resources for wildlife by increasing landscape connectivity (Ewers and Didham 2006; Arroyo Rodriguez et al. 2009; Hernández-Ruedas et al. 2014). The ecological value of large reserves for biodiversity conservation is unquestionable, although in a world facing a growing scenario of fragmentation, the importance and functionality of forest fragments should be recognized and incorporated into future conservation initiatives.

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Anexos

Table 01: Information (popular name, area, localization) of nine seasonal semideciduous forest fragments in southeastern Brazil analyzed.

Fragment	Popular name	Area (ha)	Localization
1	Matão	20,91	S 21°30'16.8" W 045°52'38.5"
2	Gaspar Lopes	81,55	S 21°22'43.8" W 045°55'46.7"
3	M	56,05	S 21°27'24.6" W 046°10'07.1"
4	Paraiso	36,85	S 21°21'46.5" W 045°50'26.4"
5	I	37,05	S 21°25'35.1" W 046°05'39.4"
6	Cemiterio	22,99	S 21°33'34.5" W 045°56'15.8"
7	Porto	87,18	S 21°25'16.3" W 046°07'22.3"
8	N	24,80	S 21°28'07.2" W 046°09'46.2"
9	São José	28,57	S 21°26'02.4"W 046°08'57.6"

Table 02: List of species with DAP< 5.0 and DAP> 5.0 cm sampled in nine sites of seasonal semideciduous forest of southeastern Brazil. T1= Sum of individuals with DBH ≥ 5 cm. T2= Sum of individuals with DBH < 5 cm. T3= Sum of all individuals.

Fragments	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**T1**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**T2**	**T3**
Families/Species	**≥5 cm**										**<5cm**										
Anacardiaceae	8	3	3	2	1	4	12	3	6	42	20	6	16	8	4	14	19	10	15	112	154
***Astronium fraxinifolium* Schott**	0	0	1	0	0	0	5	2	0	8	1	0	4	2	0	0	12	6	9	34	42
Astronium graveolens Jacq.	0	0	2	0	0	0	7	0	2	11	0	0	5	1	0	1	7	0	1	15	26
Lithrea molleoides (Vell.) Engl.	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Tapirira guianensis Aubl.	7	1	0	1	1	2	0	0	1	13	16	2	4	1	0	3	0	2	0	28	41
Tapirira obtusa (Benth.) J.D.Mitch.	1	1	0	1	0	2	0	1	3	9	3	4	3	4	4	10	0	2	5	35	44
Annonaceae	7	8	12	15	11	6	7	6	9	81	6	18	6	14	20	13	13	36	18	144	225
Anaxagorea dolichocarpa Sprague & Sandwith	0	0	0	0	0	0	0	0	0	0	0	3	0	1	0	0	0	0	0	4	4
Annona cacans Warm.	0	1	1	0	0	0	1	1	0	4	0	0	0	0	0	0	0	0	1	1	5
Annona emarginata (Schltdl.) H.Rainer	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Annona glabra L.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	2	3	3
Annona sylvatica A.St.-Hil.	1	3	2	0	0	0	0	0	1	7	0	10	0	1	1	1	0	0	0	13	20
Duguetia lanceolata A.St.-Hil.	5	1	5	9	10	4	5	2	5	46	3	3	4	7	16	11	11	1	10	66	112
Guatteria australis A.St.-Hil.	1	2	2	1	1	0	1	1	2	11	0	2	0	2	0	1	1	0	3	9	20
Xylopia brasiliensis Spreng.	0	0	2	5	0	2	0	2	1	12	2	0	2	3	3	0	1	35	2	48	60
Apocynaceae	0	1	3	0	2	1	1	0	3	11	1	2	6	0	5	5	8	0	2	29	40
Aspidosperma cylindrocarpon Mull. Arg.	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1
Aspidosperma discolor A.DC.	0	0	0	0	2	0	0	0	0	2	0	0	4	0	3	0	5	0	0	12	14
Aspidosperma dispermum Müll.Arg.	0	0	3	0	0	0	0	0	0	3	0	0	0	0	1	0	0	0	0	1	4
Aspidosperma olivaceum Müll.Arg.	0	1	0	0	0	0	0	0	1	2	0	1	1	0	1	0	1	0	0	4	6

Aspidosperma parvifolium A.DC.	0	0	0	0	0	1	0	0	0	1	1	0	1	0	0	4	0	0	0	6	7
Aspidosperma polyneuron Müll.Arg.	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	2	0	2	6	6
Aspidosperma spruceanum Benth. ex Müll.Arg.	0	0	0	0	0	0	1	0	1	2	0	0	0	0	0	0	0	0	0	0	2
Aquifoliaceae	6	2	1	4	0	0	0	0	2	15	11	2	2	4	4	9	1	1	8	42	57
Ilex affinis Gardner	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Ilex cerasifolia Reissek	1	0	0	0	0	0	0	0	2	3	2	2	0	1	4	1	0	1	5	16	19
Ilex paraguariensis A.St.-Hil.	5	2	1	4	0	0	0	0	0	12	9	0	2	3	0	8	0	0	2	24	36
Ilex sapotifolia Reissek	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1
Araliaceae	1	2	0	0	0	0	1	0	1	5	15	3	2	9	0	4	0	0	3	36	41
Aralia excelsa (Griseb.) J.Wen	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Dendropanax cuneatus (DC.) Decne. & Planch.	1	2	0	0	0	0	0	0	1	4	13	3	2	0	0	4	0	0	0	22	26
Schefflera calva (Cham.) Frodin & Fiaschi	0	0	0	0	0	0	0	0	0	0	2	0	0	6	0	0	0	0	2	10	10
Schefflera morototoni (Aubl.) Maguire et al.	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	1	4	4
Arecaceae	4	0	14	4	0	2	0	0	0	24	1	0	0	0	0	0	0	0	0	1	25
Euterpe edulis Mart.	0	0	1	0	0	2	0	0	0	3	0	0	0	0	0	0	0	0	0	0	3
Syagrus romanzoffiana (Cham.) Glassman	4	0	13	4	0	0	0	0	0	21	1	0	0	0	0	0	0	0	0	1	22
Asteraceae	1	9	2	3	0	0	19	7	0	41	0	1	0	0	0	3	3	1	0	8	49
Piptocarpha macropoda (DC.) Baker	1	1	2	2	0	0	13	2	0	21	0	0	0	0	0	3	2	1	0	6	27
Vernonanthura divaricata (Spreng.) H.Rob.	0	8	0	1	0	0	6	5	0	20	0	1	0	0	0	0	1	0	0	2	22
Bignoniaceae	1	0	0	1	0	0	1	0	1	4	3	5	0	3	1	5	2	1	1	21	25
Cybistax antisyphilitica (Mart.) Mart.	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	1	0	0	5	5
Handroanthus ochraceus (Cham.) Mattos	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	1
Handroanthus serratifolius	1	0	0	0	0	0	0	0	0	1	1	0	0	0	1	0	0	0	0	2	3

(Vahl) S.O.Grose																					
Jacaranda macrantha Cham.	0	0	0	1	0	0	1	0	1	3	1	0	0	3	0	0	1	0	1	6	9
Jacaranda puberula Cham.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Tabebuia roseoalba (Ridl.) Sandwith	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	5	0	0	0	6	6
Boraginaceae	3	0	0	0	2	0	1	0	0	6	0	3	5	6	2	6	1	5	1	29	35
Cordia sellowiana Cham.	3	0	0	0	2	0	1	0	0	6	0	2	5	6	2	6	1	5	1	28	34
Cordia trichotoma (Vell.) Arráb. ex Steud.	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	1
Burseraceae	3	7	3	0	8	5	0	1	16	43	8	29	15	9	12	28	0	5	11	117	160
Protium heptaphyllum (Aubl.) Marchand	0	5	0	0	2	2	0	1	3	13	1	16	4	7	3	1	0	3	1	36	49
Protium spruceanum (Benth.) Engl.	0	0	2	0	1	2	0	0	2	7	4	2	7	2	4	21	0	2	2	44	51
Protium widgrenii Engl.	3	2	1	0	5	1	0	0	11	23	3	11	4	0	5	6	0	0	8	37	60
Cannabaceae	0	1	1	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	2	2	4
Celtis brasiliensis (Gardner) Planch.	0	1	1	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	1	1	3
Celtis iguanaea (Jacq.) Sarg.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Cardiopteridaceae	0	1	0	0	0	0	0	1	0	2	0	3	0	0	0	0	0	0	2	5	7
Citronella paniculata (Mart.) R.A.Howard	0	1	0	0	0	0	0	1	0	2	0	3	0	0	0	0	0	0	2	5	7
Caricaceae	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Jacaratia spinosa (Aubl.) A.DC.	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Celastraceae	0	1	0	2	1	0	0	1	1	6	3	12	2	1	0	1	6	1	6	32	38
Cheiloclinium serratum (Cambess.) A.C.Sm.	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Chrysophyllum marginatum (Hook. & Arn.) Radlk.	0	0	0	0	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	7	7
Clethra scabra Pers.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	2	2
Hippocratea volubilis L.	0	1	0	1	0	0	0	0	0	2	0	1	0	0	0	0	0	0	0	1	3
Maytenus glaziovi Reissek	0	0	0	0	0	0	0	0	0	0	2	2	0	1	0	0	0	1	3	9	9
Maytenus ilicifolia Mart. ex Reissek	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	2	2

Maytenus robusta Reissek	0	0	0	0	0	0	0	0	1	1	1	0	1	0	0	0	5	0	1	8	9
Maytenus salicifolia Reissek	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Salacia elliptica (Mart. ex Schult.) G.Don	0	0	0	1	0	0	0	1	0	2	0	0	1	0	0	0	1	0	0	2	4
Combretaceae	0	0	0	0	3	0	0	1	0	4	0	1	0	1	1	0	0	0	0	3	7
Terminalia glabrescens Mart.	0	0	0	0	3	0	0	1	0	4	0	1	0	1	1	0	0	0	0	3	7
Connaraceae	0	0	0	0	0	0	0	1	0	1	0	1	0	0	1	0	0	1	0	3	4
Connarus regnellii G.Schellenb.	0	0	0	0	0	0	0	1	0	1	0	1	0	0	1	0	0	0	0	2	3
Connarus rostratus (Vell.) L.B.Sm.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Cunoniaceae	1	0	0	1	1	0	0	0	1	4	0	0	0	1	0	0	0	0	0	1	5
Lamanonia ternata Vell.	1	0	0	1	1	0	0	0	1	4	0	0	0	1	0	0	0	0	0	1	5
Ebenaceae	1	0	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	2	3
Diospyros burchellii Hiern	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Diospyros inconstans Jacq.	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	2	2
Elaeocarpaceae	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	1	0	3	3
Sloanea monosperma Vell.	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	1	0	3	3
Erythroxylaceae	0	0	0	0	0	0	0	0	0	0	10	2	0	5	16	2	5	0	3	43	43
Erythroxylum citrifolium A.St.-Hil.	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	1
Erythroxylum pelleterianum A.St.-Hil.	0	0	0	0	0	0	0	0	0	0	10	1	0	5	16	2	5	0	3	42	42
Euphorbiaceae	30	4	40	15	2	6	31	40	9	177	44	14	27	22	1	6	28	64	27	233	410
Alchornea glandulosa Poepp. & Endl.	0	0	3	0	0	1	1	0	0	5	1	0	1	0	0	1	0	5	0	8	13
Alchornea triplinervia (Spreng.) Müll.Arg.	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Aparisthmium cordatum (A.Juss.) Baill.	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0	0	0	0	0	8	8
Croton floribundus Spreng.	9	3	19	13	2	0	1	7	9	63	5	8	13	12	1	1	7	23	9	79	142
Croton rugosus Spreng.	4	0	13	0	0	0	0	1	0	18	13	0	3	0	0	0	0	0	0	16	34
Croton salutaris Casar.	0	0	3	0	0	0	0	0	0	3	0	0	1	0	0	0	0	0	0	1	4

***Croton urucurana* Baill.**	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	1
***Mabea fistulifera* Mart.**	2	0	0	1	0	1	22	26	0	52	5	0	0	2	0	0	12	29	3	51	103
***Mabea piriri* Aubl.**	0	0	0	0	0	1	1	0	0	2	0	0	0	1	0	0	2	0	0	3	5
***Mabea pohliana* (Benth.) Müll. Arg.**	0	0	0	0	0	0	4	0	0	4	0	0	0	5	0	0	0	3	2	10	14
***Manihot grahamii* Hook.**	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1
***Maprounea guianensis* Aubl.**	15	0	0	1	0	2	2	2	0	22	19	0	0	2	0	4	7	1	1	34	56
***Sapium glandulatum* (Vell.) Pax**	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
***Sebastiania brasiliensis* Spreng.**	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	2	2
***Sebastiania commersoniana* (Baill.) L.B.Sm. & Downs**	0	1	0	0	0	0	0	4	0	5	0	5	0	0	0	0	0	3	11	19	24
***Sorocea bonplandii* (Baill.) W.C.Burger et al.**	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Fabaceae	48	52	28	25	34	13	8	10	29	247	22	132	43	11	29	23	21	12	45	338	585
***Acacia polyphylla* DC.**	0	3	1	0	0	0	0	0	0	4	0	9	2	0	0	0	1	0	0	12	16
***Albizia polycephala* (Benth.) Killip ex Record**	0	0	2	0	0	0	0	0	2	4	0	4	4	0	0	0	1	1	4	14	18
***Andira fraxinifolia* Benth.**	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	1
***Andira ormosioides* Benth.**	0	0	0	0	2	0	0	0	0	2	0	2	2	0	0	0	1	0	0	5	7
***Apuleia leiocarpa* (Vogel) J.F.Macbr.**	1	1	0	0	0	0	2	0	0	4	2	0	3	0	2	0	2	1	1	11	15
***Bauhinia longifolia* (Bong.) Steud.**	0	7	0	0	0	0	0	0	0	7	1	9	0	0	0	0	0	0	0	10	17
***Bauhinia rufa* (Bong.) Steud.**	0	0	0	0	0	0	2	0	0	2	0	0	0	0	0	0	0	0	1	1	3
***Casearia arborea* (Rich.) Urb.**	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	2	2
***Copaifera langsdorffii* Desf.**	29	6	1	23	19	12	2	3	9	104	4	36	5	3	17	12	2	2	13	94	198
***Dalbergia villosa* (Benth.) Benth.**	1	0	1	2	1	0	0	1	0	6	0	0	0	0	1	3	4	0	3	11	17
***Enterolobium contortisiliquum* (Vell.) Morong**	0	0	1	0	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	2	3
***Hymenaea courbaril* L.**	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1

***Inga cylindrica* (Vell.) Mart.**	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
***Inga marginata* Willd.**	0	0	2	0	0	0	0	2	0	4	0	3	8	0	0	0	1	1	5	18	22
***Inga vera* Willd.**	2	0	9	0	0	0	0	0	0	11	1	1	3	2	0	1	0	2	0	10	21
***Machaerium acutifolium* Vogel**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
***Machaerium brasiliense* Vogel**	0	1	2	0	0	0	0	0	0	3	0	1	0	0	0	0	0	0	3	4	7
***Machaerium dimorphandrum* Hoehne**	0	0	0	0	0	0	0	0	1	1	3	2	0	0	1	1	0	2	1	10	11
***Machaerium hirtum* (Vell.) Stellfeld**	0	0	0	0	0	0	0	0	1	1	2	6	1	0	0	0	0	0	1	10	11
***Machaerium nyctitans* (Vell.) Benth.**	3	1	0	0	2	0	0	0	2	8	3	2	2	1	1	0	2	0	1	12	20
***Machaerium stipitatum* Vogel**	0	2	0	0	0	0	0	0	3	5	0	3	1	0	0	0	1	0	3	8	13
***Machaerium villosum* Vogel**	3	2	3	0	1	1	0	2	1	13	2	0	0	1	3	0	0	0	0	6	19
***Peltophorum dubium* (Spreng.) Taub.**	0	3	3	0	0	0	2	0	4	12	0	6	6	1	0	0	0	0	0	13	25
***Piptadenia gonoacantha* (Mart.) J.F.Macbr.**	0	9	0	0	0	0	0	0	0	9	0	9	0	0	0	0	2	0	0	11	20
***Piptocarpha macropoda* (DC.) Baker**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1
***Platycyamus regnellii* Benth.**	1	5	2	0	7	0	0	0	5	20	2	6	0	0	1	0	2	0	0	11	31
***Platypodium elegans* Vogel**	7	0	0	0	0	0	0	0	0	7	2	3	0	0	0	0	0	0	0	5	12
***Senegalia polyphylla* (DC.) Britton & Rose**	0	4	1	0	0	0	0	0	0	5	0	21	2	0	0	0	0	0	6	29	34
***Senna macranthera* (DC. ex Collad.) H.S.Irwin & Barneby**	0	7	0	0	0	0	0	0	0	7	0	2	0	3	0	6	0	1	0	12	19
***Senna multijuga* (Rich.) H.S.Irwin & Barneby**	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
***Sweetia fruticosa* Spreng.**	0	0	0	0	0	0	0	0	0	0	0	3	0	0	1	0	0	0	0	4	4
***Symplocos pubescens* Klotzsch ex Benth.**	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1
***Tachigali denudata* (Vogel) Oliveira-Filho**	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1

***Tachigali rugosa* (Mart. ex Benth.) Zarucchi & Pipoly**	0	0	0	0	0	0	0	2	0	2	0	0	0	0	2	0	0	2	0	4	6
***Zollernia ilicifolia* (Brongn.) Vogel**	0	0	0	0	0	0	0	0	0	0	0	3	1	0	0	0	0	0	2	6	6
Hypericaceae	0	0	0	0	0	0	0	5	0	5	1	0	0	0	0	1	0	0	1	3	8
***Vismia guianensis* (Aubl.) Choisy**	0	0	0	0	0	0	0	5	0	5	1	0	0	0	0	1	0	0	1	3	8
Lacistemataceae	0	0	0	0	0	0	0	0	0	0	4	6	0	0	1	28	0	1	42	82	82
***Lacistema hasslerianum* Chodat**	0	0	0	0	0	0	0	0	0	0	4	6	0	0	1	28	0	1	42	82	82
Lamiaceae	0	1	0	0	0	0	1	0	0	2	3	4	0	1	1	1	1	0	1	12	14
***Aegiphila integrifolia* (Jacq.) Moldenke**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1
***Aegiphila sellowiana* Cham.**	0	1	0	0	0	0	1	0	0	2	1	0	0	0	0	0	1	0	0	2	4
***Vitex polygama* Cham.**	0	0	0	0	0	0	0	0	0	0	2	4	0	1	1	0	0	0	1	9	9
Lauraceae	25	11	56	49	42	35	42	20	36	316	65	34	40	117	38	59	30	10	67	460	776
***Aiouea saligna* Meisn.**	0	0	0	1	1	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2
***Aniba canelilla* (Kunth) Mez**	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	1
***Aniba firmula* (Nees & Mart.) Mez**	0	1	0	1	0	0	1	1	3	7	0	0	1	0	0	0	0	0	5	6	13
***Cryptocarya aschersoniana* Mez**	5	1	15	19	20	8	4	3	9	84	7	2	6	20	10	4	3	0	8	60	144
***Cryptocarya moschata* Nees & Mart. ex Nees**	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	1
***Endlicheria paniculata* (Spreng.) J.F.Macbr.**	0	2	2	0	0	0	0	0	2	6	1	6	4	0	0	0	1	2	8	22	28
***Nectandra grandiflora* Nees**	7	0	1	2	0	7	0	1	0	18	31	5	6	5	1	13	0	0	1	62	80
***Nectandra oppositifolia* Nees**	0	1	2	0	0	0	0	0	0	3	0	0	4	2	1	0	1	0	0	8	11
***Ocotea bicolor* Vattimo-Gil.**	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
***Ocotea brachybotrya* (Meisn.) Mez**	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	4	0	0	1	5	7
***Ocotea corymbosa* (Meisn.) Mez**	4	1	2	11	1	4	3	3	6	35	20	12	6	59	6	17	0	6	10	136	171
***Ocotea diospyrifolia* (Meisn.)**	5	0	3	0	0	0	0	0	0	8	0	0	4	0	1	2	0	0	4	11	19

Mez																					
Ocotea elegans Mez	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Ocotea laxa (Nees) Mez	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Ocotea minarum (Nees & Mart.) Mez	0	1	0	0	0	1	0	0	0	2	0	1	0	0	0	0	0	0	0	1	3
Ocotea odorifera (Vell.) Rohwer	1	0	27	15	17	15	34	8	13	130	3	6	6	31	17	19	25	1	26	134	264
Ocotea pulchella (Nees & Mart.) Mez	3	3	1	0	0	0	0	0	1	8	2	0	0	0	2	0	0	1	3	8	16
Persea major L.E.Kopp	0	1	1	0	0	0	0	1	0	3	0	2	0	0	0	0	0	0	0	2	5
Persea rufotomentosa Nees & Mart. ex Nees	0	0	0	0	1	0	0	0	0	1	1	0	3	0	0	0	0	0	0	4	5
Persea willdenowii Kosterm.	0	0	2	0	0	0	0	1	0	3	0	0	0	0	0	0	0	0	0	0	3
Lecythidaceae	0	0	0	0	0	0	0	1	0	1	0	0	0	1	0	0	1	1	1	4	5
Cariniana estrellensis (Raddi) Kuntze	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	1
Cariniana legalis (Mart.) Kuntze	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1	3	3
Lecythis pisonis Cambess.	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	1
Loganiaceae	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	1	0	3	6	6
Strychnos brasiliensis Mart.	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	1	0	3	6	6
Lythraceae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Lafoensia glyptocarpa Koehne	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Malpighiaceae	0	0	1	1	2	4	0	0	2	10	10	0	1	7	4	2	5	0	3	32	42
Byrsonima crassifolia (L.) Kunth	0	0	1	1	2	4	0	0	0	8	0	0	0	2	3	2	3	0	1	11	19
Byrsonima laxiflora Griseb.	0	0	0	0	0	0	0	0	2	2	10	0	1	5	1	0	2	0	2	21	23
Malvaceae	3	5	0	2	2	0	0	0	5	17	6	7	0	0	1	0	0	0	0	14	31
Eriotheca candolleana (K.Schum.) A.Robyns	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	3	3
Guazuma ulmifolia Lam.	0	3	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	3
Luehea candicans Mart. & Zucc.	1	2	0	0	2	0	0	0	4	9	1	4	0	0	1	0	0	0	0	6	15
Luehea divaricata Mart.	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	2	2

&Zucc.																				
***Luehea grandiflora* Mart. & Zucc.**	2	0	0	2	0	0	0	0	1	5	1	1	0	0	0	0	0	0	0	2	7
***Luehea paniculata* Mart. & Zucc.**	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1
Melastomataceae	0	0	0	17	27	15	5	17	3	84	18	3	8	24	0	27	9	11	17	117	201
***Leandra scabra* DC.**	0	0	0	0	0	0	0	0	0	0	3	0	0	1	0	0	0	2	1	7	7
***Miconia affinis* DC.**	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1
***Miconia albicans* (Sw.) Triana**	0	0	0	0	0	0	0	0	0	0	1	1	0	1	0	3	0	0	0	6	6
***Miconia chartacea* Triana**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1
***Miconia cinerascens* Miq.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	4	4
***Miconia cubatanensis* Hoehne**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1
***Miconia hymenonervia* (Raddi) Cogn.**	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	3	5	5
***Miconia paulensis* Naudin**	0	0	0	0	0	0	0	0	1	1	3	0	0	3	0	1	0	0	1	8	9
***Miconia pepericarpa* DC.**	0	0	0	0	0	0	0	2	0	2	7	0	3	5	0	2	0	1	0	18	20
***Miconia prasina* (Sw.) DC.**	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	2	1	2	0	8	8
***Miconia pusilliflora* (DC.) Naudin**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1
***Miconia urophylla* DC.**	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	2	2
***Miconia willdenowii* Klotzsch ex Naudin**	0	0	0	17	27	15	5	13	2	79	2	0	2	13	0	17	7	5	7	53	132
***Tibouchina arborea* (Gardner) Cogn.**	0	0	0	0	0	0	0	2	0	2	0	0	0	0	0	0	0	1	1	2	4
Meliaceae	17	18	40	63	9	13	22	6	18	206	37	140	138	111	48	25	73	21	114	707	913
***Cabralea canjerana* (Vell.) Mart.**	1	0	4	0	2	0	1	1	1	10	3	4	7	1	0	1	0	1	3	20	30
***Cedrela fissilis* Vell.**	2	0	6	2	0	0	0	0	1	11	5	1	4	0	0	0	0	0	1	11	22
***Guarea guidonia* (L.) Sleumer**	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1
***Guarea kunthiana* A.Juss.**	0	0	1	0	0	0	0	2	0	3	0	0	15	0	0	0	0	5	0	20	23
***Guarea macrophylla* Vahl**	0	0	1	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	1	2
***Trichilia casaretti* C.DC.**	0	0	0	1	1	0	0	0	0	2	0	0	1	1	3	0	1	0	1	7	9

***Trichilia catigua* A.Juss.**	0	5	7	12	0	0	4	0	6	34	0	9	5	10	0	0	12	13	56	105	139
***Trichilia clausseni* C.DC.**	0	1	15	0	0	0	14	0	3	33	0	24	75	0	0	0	15	0	17	131	164
***Trichilia elegans* A.Juss.**	0	6	0	0	0	0	0	0	1	7	0	41	1	0	0	0	0	0	17	59	66
***Trichilia emarginata* (Turcz.) C.DC.**	14	1	0	46	4	13	3	3	4	88	18	1	10	80	26	24	42	0	10	211	299
***Trichilia pallens* C.DC.**	0	1	0	0	0	0	0	0	0	1	0	1	0	0	0	0	2	0	0	3	4
***Trichilia pallida* Sw.**	0	4	6	2	2	0	0	0	1	15	10	59	20	19	19	0	1	2	9	139	154
Monimiaceae	3	22	7	2	1	0	1	3	8	47	4	44	8	4	3	1	1	2	5	72	119
***Mollinedia widgrenii* A.DC.**	3	22	7	2	1	0	1	3	8	47	4	44	8	4	3	1	1	2	5	72	119
Moraceae	0	3	1	1	2	0	0	3	0	10	0	1	5	0	2	0	1	2	1	12	22
***Ficus adhatodifolia* Schott ex Spreng.**	0	1	0	0	0	0	0	1	0	2	0	0	1	0	0	0	0	0	0	1	3
***Ficus arpazusa* Casar.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1
***Ficus cyclophylla* (Miq.) Miq.**	0	0	0	0	0	0	0	2	0	2	0	0	0	0	0	0	0	2	0	2	4
***Ficus enormis* Mart. ex Miq.**	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
***Ficus gomelleira* Kunth & C.D.Bouché**	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
***Ficus guaranitica* Chodat**	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
***Ficus pertusa* L.f.**	0	1	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	1	2
***Ficus trigona* L.f.**	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
***Maclura tinctoria* (L.) D.Don ex Steud.**	0	0	1	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	2
***Sorocea bonplandii* (Baill.) W.C.Burger et al.**	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	1	0	1	6	6
Myrsinaceae	0	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0	0	0	1	2
***Myrsine guianensis* (Aubl.) Kuntze**	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1
***Myrsine umbellata* Mart.**	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	1
Myrtaceae	14	9	4	30	17	35	12	18	24	163	25	67	19	41	51	63	75	39	147	527	690
***Blepharocalyx salicifolius* (Kunth) O.Berg**	1	0	0	2	0	0	0	0	0	3	1	0	0	3	0	1	0	0	0	5	8
***Calyptranthes brasiliensis* Spreng.**	3	0	0	0	0	1	0	0	0	4	5	0	0	1	1	4	0	0	0	11	15
Calyptranthes clusiifolia	0	0	1	2	8	10	1	1	0	23	1	1	0	5	9	13	9	11	2	51	74

O.Berg																					
***Calyptranthes grandiflora* O.Berg**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
***Calyptranthes widgreniana* O.Berg**	0	0	0	0	0	0	0	1	1	2	0	2	1	0	0	1	3	1	4	12	14
***Campomanesia guazumifolia* (Cambess.) O.Berg**	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	1
***Campomanesia simulans* M.L.Kawas.**	1	0	0	2	0	0	0	0	0	3	0	0	0	2	0	0	0	0	0	2	5
***Campomanesia xanthocarpa* (Mart.) O.Berg**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	2	2
***Eugenia acutata* Miq.**	2	5	0	8	2	0	1	12	9	39	1	30	0	3	5	0	2	11	78	130	169
***Eugenia blastantha* (O.Berg) D.Legrand**	0	0	0	0	0	0	0	1	1	2	0	0	0	0	0	0	1	0	1	2	4
***Eugenia dodonaeifolia* Cambess.**	0	0	0	2	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2
***Eugenia florida* DC.**	0	1	1	3	0	0	4	1	3	13	9	5	4	1	2	2	0	1	3	27	40
***Eugenia involucrata* DC.**	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	1	1	3
***Eugenia leitonii* ined.**	0	0	0	1	0	0	2	0	2	5	0	0	0	1	1	0	7	0	8	17	22
***Eugenia malacantha* D.Legrand**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	7	0	1	9	9
***Eugenia paracatuana* O.Berg**	0	0	0	0	0	0	0	0	0	0	0	3	0	0	1	1	6	1	1	13	13
***Eugenia revoluta* Griseb.**	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1
***Eugenia sonderiana* DC.**	0	1	0	2	0	0	0	1	0	4	0	7	0	0	2	1	1	1	3	15	19
***Eugenia uniflora* L.**	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	1	1	2
***Eugenia widgrenii* Sond. ex O.Berg**	0	0	0	1	0	0	0	0	1	2	0	0	0	0	0	0	1	0	3	4	6
***Marlierea laevigata* (DC.) Kiaersk.**	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	1	0	4	4
***Marlierea racemosa* (Vell.) Kiaersk.**	0	0	0	0	5	0	1	0	0	6	0	0	0	7	1	2	5	9	13	37	43
***Myrceugenia miersiana* (Gardner) D.Legrand & Kausel**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1

***Myrcia diaphana* (O.Berg) N.Silveira**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	2	2
***Myrcia fenzliana* O.Berg**	0	0	0	0	0	0	0	0	0	0	0	3	0	0	2	0	0	0	1	6	6
***Myrcia guianensis* (Aubl.) DC.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	3	3
***Myrcia hebepetala* DC.**	0	0	0	0	0	0	0	0	2	2	0	0	0	0	4	0	0	0	3	7	9
***Myrcia pubiflora* DC.**	0	0	0	0	0	0	0	0	0	0	0	0	0	2	7	14	3	0	2	28	28
***Myrcia pulchra* (O.Berg) Kiaersk.**	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	1
***Myrcia splendens* (Sw.) DC.**	0	1	2	1	2	20	3	1	1	31	4	14	12	8	7	19	22	0	10	96	127
***Myrcia tomentosa* (Aubl.) DC.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1
***Myrcia venulosa* DC.**	0	0	0	0	0	0	0	0	1	1	0	0	0	2	0	0	0	0	0	2	3
***Myrciaria floribunda* (H.West ex Willd.) O.Berg**	0	1	0	0	0	0	0	0	0	1	0	0	0	0	4	0	0	0	5	9	10
***Myrsine coriacea* (Sw.) R.Br. ex Roem. & Schult.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
***Plinia cauliflora* (Mart.) Kausel**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1
***Psidium cattleianum* Sabine**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1
***Psidium rufum* Mart. ex DC.**	0	0	0	1	0	0	0	0	0	1	0	0	1	1	0	0	0	0	1	3	4
***Psidium sartorianum* (O.Berg) Nied.**	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	2	2
***Siphoneugena densiflora* O.Berg**	7	0	0	4	0	4	0	0	0	15	3	0	0	3	0	5	1	2	3	17	32
***Siphoneugena reitzii* D.Legrand**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1
Nyctaginaceae	0	0	0	0	0	0	0	0	0	0	1	4	4	0	4	3	1	0	5	22	22
***Guapira hirsuta* (Choisy) Lundell**	0	0	0	0	0	0	0	0	0	0	0	3	1	0	1	3	0	0	1	9	9
***Guapira opposita* (Vell.) Reitz**	0	0	0	0	0	0	0	0	0	0	1	1	3	0	3	0	1	0	4	13	13
Ochnaceae	0	0	0	0	1	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	2
***Ouratea semiserrata* (Mart. & Nees) Engl.**	0	0	0	0	1	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	2

Oleaceae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
***Chionanthus trichotomus* (Vell.) P.S.Green**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Opiliaceae	1	0	1	0	1	0	0	0	1	4	0	0	0	0	1	0	0	0	0	1	5
***Agonandra excelsa* Griseb.**	1	0	1	0	1	0	0	0	1	4	0	0	0	0	1	0	0	0	0	1	5
Peraceae	0	3	0	4	0	6	0	1	0	14	1	4	0	3	2	3	0	0	6	19	33
***Pera glabrata* (Schott) Poepp. ex Baill.**	0	3	0	4	0	6	0	1	0	14	1	4	0	3	2	3	0	0	6	19	33
Piperaceae	0	0	0	0	0	0	0	0	0	0	2	84	28	3	2	12	2	16	11	160	160
***Piper aduncum* L.**	0	0	0	0	0	0	0	0	0	0	0	6	2	0	0	1	0	11	0	20	20
***Piper amalago* L.**	0	0	0	0	0	0	0	0	0	0	0	68	15	1	0	0	0	0	11	95	95
***Piper arboreum* Aubl.**	0	0	0	0	0	0	0	0	0	0	2	10	11	2	2	11	2	5	0	45	45
Primulaceae	0	0	0	0	0	1	1	0	0	2	3	4	2	0	0	1	5	2	6	23	25
***Myrsine coriacea* (Sw.) R.Br. ex Roem. & Schult.**	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	1	0	0	3	7	7
***Myrsine umbellata* Mart.**	0	0	0	0	0	1	1	0	0	2	0	4	2	0	0	0	5	2	3	16	18
Proteaceae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	2	2
***Euplassa organensis* (Gardner) I.M.Johnst.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	0	2	2
Rhamnaceae	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1
***Rhamnidium elaeocarpum* Reissek**	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1
Rosaceae	1	0	1	0	1	0	0	1	1	5	1	1	0	2	0	2	0	0	3	9	14
***Prunus myrtifolia* (L.) Urb.**	1	0	1	0	1	0	0	1	1	5	1	1	0	2	0	1	0	0	3	8	13
***Rubus brasiliensis* Mart.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1
Rubiaceae	13	10	1	3	14	13	4	3	21	82	151	101	96	52	28	21	76	49	101	675	757
***Alseis floribunda* Schott**	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	2	2
***Amaioua guianensis* Aubl.**	12	3	1	2	0	10	3	1	2	34	34	10	2	12	11	6	10	5	14	104	138
***Coffea arabica* L.**	0	0	0	0	0	0	0	0	0	0	0	0	27	4	0	1	26	1	1	60	60
***Cordiera concolor* (Cham.) Kuntze**	0	0	0	0	0	0	1	0	0	1	0	12	0	1	0	0	1	0	2	16	17
***Cordiera elliptica* (Cham.) Kuntze**	0	0	0	0	0	1	0	0	1	2	0	0	0	0	0	0	0	0	0	0	2
***Cordiera sessilis* (Vell.)**	0	3	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	3

Kuntze																					
Coutarea hexandra (Jacq.) K.Schum.	0	0	0	0	0	0	0	0	3	3	0	3	0	0	0	0	0	0	2	5	8
Faramea latifolia (Cham. & Schltdl.) DC.	0	0	0	0	2	0	0	1	0	3	0	2	2	0	1	0	0	0	1	6	9
Faramea nigrescens Mart.	0	2	0	0	0	0	0	0	0	2	0	0	0	0	2	0	0	0	1	3	5
Guettarda uruguensis Cham. & Schltdl.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
Ixora brevifolia Benth.	1	1	0	1	12	2	0	1	14	32	0	3	1	0	5	7	6	3	14	39	71
Psychotria cephalantha (Müll.Arg.) Standl.	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	3	4	4
Psychotria hastisepala Müll.Arg.	0	0	0	0	0	0	0	0	0	0	10	0	21	1	4	0	12	0	6	54	54
Psychotria triflora Müll.Arg.	0	0	0	0	0	0	0	0	0	0	1	0	32	8	2	2	17	0	0	62	62
Psychotria vellosiana Benth.	0	0	0	0	0	0	0	0	0	0	105	69	10	23	1	5	4	40	52	309	309
Randia armata (Sw.) DC.	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1
Roupala montana Aubl.	0	1	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	0	0	2
Rudgea jasminoides (Cham.) Müll.Arg.	0	0	0	0	0	0	0	0	0	0	0	2	1	1	2	0	0	0	3	9	9
Rutaceae	9	2	24	0	3	0	46	18	7	109	15	8	79	1	5	2	88	46	56	300	409
Galipea jasminiflora (A.St.-Hil.) Engl.	0	0	0	0	0	0	0	0	0	0	0	0	41	0	0	0	48	0	0	89	89
Hortia arborea Engl.	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Metrodorea nigra A.St.-Hil.	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	1
Metrodorea stipularis Mart.	6	0	21	0	2	0	41	18	4	92	14	0	35	1	5	0	39	45	40	179	271
Zanthoxylum monogynum A.St.-Hil.	2	0	0	0	0	0	0	0	2	4	1	5	3	0	0	0	0	0	12	21	25
Zanthoxylum rhoifolium Lam.	1	1	3	0	1	0	3	0	1	10	0	3	0	0	0	2	1	0	4	10	20
Zanthoxylum riedelianum Engl.	0	1	0	0	0	0	1	0	0	2	0	0	0	0	0	0	0	0	0	0	2
Salicaceae	9	20	24	11	16	8	7	5	5	105	44	45	20	57	25	29	6	22	11	259	364
Banara axilliflora Sleumer.	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0	0	0	5	5
Casearia arborea (Rich.) Urb.	4	1	0	2	2	3	0	0	1	13	7	1	0	28	4	7	2	12	1	62	75

***Casearia decandra* Jacq.**	1	1	1	4	2	1	1	2	0	13	12	5	7	11	6	12	1	1	2	57	70
***Casearia gossypiosperma* Briq.**	0	2	0	0	0	0	0	0	0	2	0	1	0	0	0	0	0	0	0	1	3
***Casearia lasiophylla* Eichler**	0	2	0	0	0	2	4	0	0	8	2	6	0	2	0	2	0	0	0	12	20
***Casearia mariquitensis* Kunth**	0	1	0	1	2	0	0	0	0	4	0	0	0	0	0	0	0	0	1	1	5
***Casearia obliqua* Spreng.**	1	3	2	3	5	2	0	0	2	18	0	4	1	3	9	1	0	1	1	20	38
***Casearia sylvestris* Sw.**	3	10	21	1	5	0	2	3	2	47	21	26	12	8	6	7	3	8	4	95	142
***Prockia crucis* P.Browne ex L.**	0	0	0	0	0	0	0	0	0	0	2	1	0	0	0	0	0	0	1	4	4
***Xylosma prockia* (Turcz.) Turcz.**	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	2	2
Sapindaceae	0	17	6	0	3	0	0	4	13	43	37	189	37	18	41	11	13	12	71	429	472
***Allophylus edulis* (A.St.-Hil. et al.) Hieron. ex Niederl.**	0	0	0	0	0	0	0	0	0	0	1	7	0	0	0	0	0	0	8	16	16
***Allophylus racemosus* Sw.**	0	1	0	0	0	0	0	0	0	1	1	0	1	0	0	2	1	0	0	5	6
***Cupania vernalis* Cambess.**	0	14	2	0	3	0	0	2	6	27	25	24	8	2	15	9	1	2	3	89	116
***Cupania zanthoxyloides* Radlk.**	0	0	0	0	0	0	0	0	1	1	0	128	18	13	20	0	10	6	44	239	240
***Diatenopteryx sorbifolia* Radlk.**	0	0	0	0	0	0	0	0	3	3	1	0	0	0	0	0	0	0	4	5	8
***Matayba elaeagnoides* Radlk.**	0	1	0	0	0	0	0	0	0	1	0	0	0	2	0	0	1	1	2	6	7
***Matayba guianensis* Aubl.**	0	0	0	0	0	0	0	1	2	3	2	16	4	1	5	0	0	1	3	32	35
***Matayba juglandifolia* (Cambess.) Radlk.**	0	1	4	0	0	0	0	1	1	7	7	14	6	0	1	0	0	2	7	37	44
Sapotaceae	0	4	0	0	1	0	0	0	0	5	0	18	1	0	0	0	0	0	4	23	28
***Chrysophyllum gonocarpum* (Mart. & Eichler ex Miq.) Engl.**	0	2	0	0	1	0	0	0	0	3	0	6	1	0	0	0	0	0	4	11	14
***Chrysophyllum marginatum* (Hook. & Arn.) Radlk.**	0	2	0	0	0	0	0	0	0	2	0	12	0	0	0	0	0	0	0	12	14
Siparunaceae	0	0	0	0	0	0	0	0	0	0	269	133	26	230	62	197	16	113	43	1089	1089
***Siparuna brasiliensis* (Spreng.) A.DC.**	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	1	3	3

***Siparuna guianensis* Aubl.**	0	0	0	0	0	0	0	0	0	0	269	132	26	230	62	197	16	112	42	1086	1086
Solanaceae	0	0	0	0	0	0	0	1	0	1	0	1	3	1	0	2	1	3	1	12	13
***Brunfelsia uniflora* (Pohl.) D. Don**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1
***Cestrum laevigatum* Schltdl.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	1
***Solanum leucodendron* Sendtn.**	0	0	0	0	0	0	0	1	0	1	0	1	3	1	0	0	1	2	1	9	10
***Solanum pseudoquina* A.St.-Hil.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Styracaceae	0	2	0	0	1	0	0	0	0	3	1	4	0	3	1	0	0	0	2	11	14
***Styrax camporum* Pohl**	0	0	0	0	0	0	0	0	0	0	0	3	0	3	0	0	0	0	1	7	7
***Styrax latifolius* Pohl**	0	1	0	0	1	0	0	0	0	2	0	1	0	0	1	0	0	0	1	3	5
***Styrax pohlii* A.DC.**	0	1	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	1	2
Symplocaceae	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	1
***Symplocos pubescens* Klotzsch ex Benth.**	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	1
Thymelaeaceae	0	0	0	0	0	0	0	0	0	0	2	5	0	0	0	0	0	0	3	10	10
***Daphnopsis brasiliensis* Mart. & Zucc.**	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	1	4	4
***Daphnopsis fasciculata* (Meisn.) Nevling**	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	4	4
***Daphnopsis utilis* Warm.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	2	2
Urticaceae	0	3	6	2	0	1	6	3	1	22	2	0	2	5	0	0	4	0	0	13	35
***Cecropia glaziovii* Snethl.**	0	0	0	2	0	0	0	1	0	3	2	0	1	0	0	0	0	0	0	3	6
***Cecropia pachystachya* Trécul**	0	3	0	0	0	0	1	1	0	5	0	0	0	3	0	0	0	0	0	3	8
***Urera baccifera* (L.) Gaudich. ex Wedd.**	0	0	6	0	0	1	5	1	1	14	0	0	1	2	0	0	4	0	0	7	21
Verbenaceae	0	1	0	0	1	0	1	0	0	3	0	7	16	2	0	0	1	2	3	31	34
***Aloysia virgata* (Ruiz & Pav.) Juss.**	0	0	0	0	0	0	0	0	0	0	0	0	15	0	0	0	0	0	0	15	15
***Duranta vestita* Cham.**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	1
***Lippia brasiliensis* T.R.S.Silva**	0	0	0	0	0	0	0	0	0	0	0	7	1	2	0	0	0	2	1	13	13
***Vitex polygama* Cham.**	0	1	0	0	1	0	1	0	0	3	0	0	0	0	0	0	1	0	1	2	5

Vochysiaceae	1	0	1	0	1	1	0	1	1	6	2	0	0	0	0	3	0	0	3	8	14
***Callisthene major* Mart. & Zucc.**	1	0	0	0	1	1	0	0	0	3	0	0	0	0	0	2	0	0	0	2	5
***Qualea cryptantha* (Spreng.) Warm.**	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1	2
***Qualea dichotoma* (Mart.) Warm.**	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	2	2
***Vochysia magnifica* Warm.**	0	0	0	0	0	0	0	1	0	1	1	0	0	0	0	1	0	0	1	3	4
***Vochysia tucanorum* Mart.**	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	1

Table 03: Number of families (F), sampled species (S), estimated species by Bootstrap (ES), number of rare species (only one individual) (RS), number of individuals (I), Shannon diversity index (H'), Pielou evenness index (J'), density of individuals per hectare (ind ha^{-1}) (D), basal area (m^2 ha^{-1}) (BA), richest families (RF) and species with greater IVI (CS) from nine sites of seasonal semideciduous forest in southeastern Brazil.

Fragments	DBH	F	S	ES	RS	I	H'	J'	D	BA	RF	CS
1 - Porto	≥ 5 cm	24	57	69.77	25	210	3.54	0.88	1050.0	21.3	Fabaceae Lauraceae Myrtaceae Salicaceae Euphorbiaceae	*Copaifera langsdorffii* *Maprounea guianensis* *Trichilia emarginata* *Croton floribundus* *Amaioua guianensis*
	< 5 cm	38	103	124.81	46	848	3.18	0.67	4240.0	1.65	Fabaceae Myrtaceae Melastomataceae Lauraceae Euphorbiaceae	*Siparuna guianensis* *Psychotria vellosiana* *Amaioua guianensis* *Nectandra grandiflora* *Cupania vernalis*
2 – Gaspar Lopes	≥ 5 cm	28	85	105.26	46	222	4.00	0.90	1110.0	20.85	Fabaceae Lauraceae Salicaceae Meliaceae	*Platycyamus regnellii* *Mollinedia widgrenii* *Copaifera langsdorffii* *Cupania vernalis* *Luehea candicans*
	< 5 cm	45	127	147.29	34	1154	3.83	0.79	5770.0	2.2	Fabaceae Myrtaceae Meliaceae Salicaceae Rubiaceae	*Siparuna guianensis* *Cupania zanthoxyloides* *Mollinedia widgrenii* *Trichilia pallida* *Piper amalago*
3 - M	≥ 5 cm	24	68	81.62	27	280	3.61	0.86	1400.0	28.25	Fabaceae	*Ocotea odorifera*

										Lauraceae Meliaceae Euphorbiaceae Annonaceae	*Croton floribundus* *Metrodorea stipularis* *Casearia sylvestris* *Cryptocarya aschersoniana*
	< 5 cm	30	102	119.34	31	658	3.94	0.85	3290.0	1.35	Fabaceae Meliaceae Lauraceae Rubiaceae Euphorbiaceae	*Trichilia clausseni* *Metrodorea stipularis* *Galipea jasminiflora* *Siparuna guianensis* *Trichilia pallida*
4 – I	≥ 5 cm	22	56	68.17	23	257	3.28	0.82	1285.0	40.8	Myrtaceae Lauraceae Meliaceae Salicaceae	*Copaifera langsdorffii* *Trichilia emarginata* *Cryptocarya aschersoniana* *Ocotea odorifera* *Miconia willdenowii*
	< 5 cm	34	99	117.69	34	779	3.24	0.71	3895.0	1.55	Myrtaceae Rubiaceae Salicaceae Melastomataceae Fabaceae	*Siparuna guianensis* *Trichilia emarginata* *Ocotea corymbosa* *Ocotea odorifera* *Casearia arborea*
5 - Paraiso	≥ 5 cm	29	58	70.34	26	210	3.42	0.84	1050.0	46.25	Fabaceae Lauraceae Salicaceae Myrtaceae Meliaceae	*Ficus trigona* *Ficus gomelleira* *Miconia willdenowii* *Cryptocarya aschersoniana* *Copaifera langsdorffii*
	< 5 cm	32	87	103.15	37	417	3.75	0.84	2085.0	1	Myrtaceae Fabaceae	*Siparuna guianensis* *Trichilia emarginata*

											Rubiaceae Lauraceae	*Cupania zanthoxyloides* *Ocotea odorifera* *Duguetia lanceolata*
6 – São José	≥ 5 cm	18	39	46.54	14	169	3.17	0.87	845.0	26.85	Lauraceae Euphorbiaceae Myrtaceae Salicaceae	*Copaifera langsdorffii* *Ocotea odorifera* *Cryptocarya aschersoniana* *Miconia willdenowii* *Myrcia splendens*
	< 5 cm	34	83	98.04	31	612	3.25	0.74	3060.0	1.1	Myrtaceae Melastomataceae Lauraceae Salicaceae Rubiaceae	*Siparuna guianensis* *Ocotea odorifera* *Trichilia emarginata* *Myrcia splendens* *Lacistema hasslerianum*
7 – N	≥ 5 cm	22	52	63.58	24	230	3.17	0.80	1150.0	17.8	Euphorbiaceae Myrtaceae Rutaceae Lauraceae Meliaceae Fabaceae	*Metrodorea stipularis* *Ocotea odorifera* *Mabea fistulifera* *Piptocarpha macropoda* *Trichilia clausseni*
	< 5 cm	32	93	111.65	40	517	3.80	0.84	2585.0	1.15	Myrtaceae Fabaceae Rubiaceae Meliaceae	*Metrodorea stipularis* *Trichilia emarginata* *Galipea jasminiflora* *Ocotea odorifera* *Myrcia splendens*
8 - Cemiterio	≥ 5 cm	27	66	82.72	36	181	3.59	0.86	905.0	25.15	Lauraceae Myrtaceae	*Metrodorea stipularis* *Mabea fistulifera*

										Euphorbiaceae	*Miconia willdenowii*
										Fabaceae	*Eugenia acutata*
										Annonaceae	*Ocotea odorifera*
	< 5 cm	31	80	96.85	35	492	3.27	0.75	2460.0	0.85	Myrtaceae	*Siparuna guianensis*
										Fabaceae	*Metrodorea stipularis*
										Euphorbiaceae	*Xylopia brasiliensis*
										Sapindaceae	*Psychotria vellosiana*
										Melastomataceae	*Croton floribundus*
9 - Matão	≥ 5 cm	28	83	101.37	40	226	4.03	0.91	1130.0	25.65	Myrtaceae	*Ixora brevifolia*
										Fabaceae	*Copaifera langsdorffii*
										Meliaceae	*Platycyamus regnellii*
										Lauraceae	*Ocotea odorifera*
										Rubiaceae	*Protium widgrenii*
	< 5 cm	43	153	183.51	62	876	4.15	0.82	4380.0	1.6	Myrtaceae	*Eugenia acutata*
										Fabaceae	*Trichilia catigua*
										Rubiaceae	*Cupania zanthoxyloides*
										Lauraceae	*Ocotea odorifera*
										Salicaceae	*Siparuna guianensis*
Total	≥ 5 cm	48	205	239.05	56	1985	4.38	0.82	1102.78	28.10	Fabaceae	*Copaifera langsdorffii*
										Myrtaceae	*Ocotea odorifera*
										Lauraceae	*Cryptocarya aschersoniana*
										Meliaceae	*Metrodorea stipularis*
										Euphorbiaceae	*Miconia willdenowii*
	< 5 cm	58	272	304.52	62	6353	4.26	0.76	3529.44	1.38	Myrtaceae	*Siparuna guianensis*
										Fabaceae	*Psychotria vellosiana*
										Rubiaceae	*Trichilia emarginata*

Lauraceae	*Cupania zanthoxyloides*
Melastomataceae	*Ocotea odorifera*

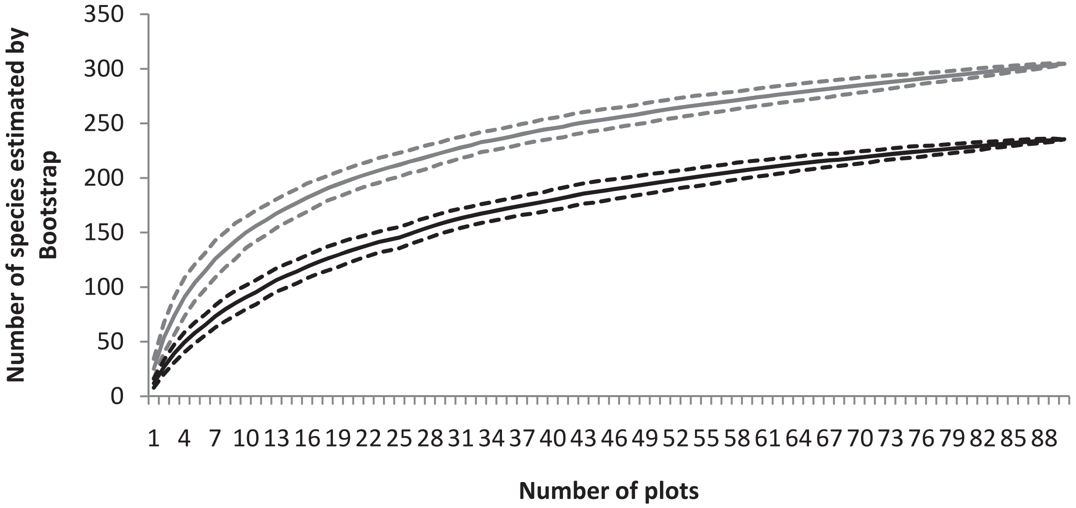
Online Resource 1: Phytosociological parameters of tree species with DAP> 5.0 cm sampled in nine sites of seasonal semideciduous forest of southeastern Brazil. N= number of individuals, AD= absolute density, RD= relative density, AF= absolute frequency, RF= relative frequency, ADo = absolute dominance, RDo = relative dominance and IVI = ecological value of importance.

Species	**N**	**AD**	**RD**	**AF**	**RF**	**ADo**	**RDo**	**IVI**
Copaifera langsdorffii Desf.	104	57.8	5.24	46.67	3.84	4.97	17.68	22.92
Ocotea odorifera (Vell.) Rohwer	130	72.2	6.55	54.44	4.48	1.42	5.07	11.62
Cryptocarya aschersoniana Mez	84	46.7	4.23	40.00	3.29	1.75	6.23	10.46
Metrodorea stipularis Mart.	92	51.1	4.63	18.89	1.56	1.48	5.27	9.90
Miconia willdenowii Klotzsch ex Naudin	79	43.9	3.98	35.56	2.93	0.88	3.15	7.13
Trichilia emarginata (Turcz.) C.DC.	88	48.9	4.43	32.22	2.65	0.64	2.27	6.70
Croton floribundus Spreng.	63	35.0	3.17	21.11	1.74	0.63	2.23	5.41
Ficus trigona L.f.	1	0.6	0.05	1.11	0.09	1.70	6.05	6.10
Duguetia lanceolata A.St.-Hil.	46	25.6	2.32	28.89	2.38	0.40	1.43	3.75
Casearia sylvestris Sw.	47	26.1	2.37	28.89	2.38	0.28	0.99	3.36
Mabea fistulifera Mart.	52	28.9	2.62	15.56	1.28	0.45	1.59	4.21
Mollinedia widgrenii A.DC.	47	26.1	2.37	22.22	1.83	0.34	1.20	3.57
Ocotea corymbosa (Meisn.) Mez	35	19.4	1.76	26.67	2.20	0.36	1.26	3.03
Ficus gomelleira Kunth & C.D.Bouché	1	0.6	0.05	1.11	0.09	1.31	4.68	4.73
Ixora brevifolia Benth.	32	17.8	1.61	16.67	1.37	0.48	1.69	3.30
Eugenia acutata Miq.	39	21.7	1.96	20.00	1.65	0.25	0.89	2.85
Platycyamus regnellii Benth.	20	11.1	1.01	13.33	1.10	0.66	2.34	3.35
Amaioua guianensis Aubl.	34	18.9	1.71	18.89	1.56	0.26	0.94	2.65
Trichilia catigua A.Juss.	34	18.9	1.71	18.89	1.56	0.14	0.51	2.22
Myrcia splendens (Sw.) DC.	31	17.2	1.56	17.78	1.46	0.15	0.53	2.09
Cupania vernalis Cambess.	27	15.0	1.36	16.67	1.37	0.14	0.49	1.85
Machaerium villosum Vogel	13	7.2	0.65	13.33	1.10	0.40	1.43	2.09
Trichilia clausseni C.DC.	33	18.3	1.66	11.11	0.91	0.13	0.46	2.12
Calyptranthes clusiifolia O.Berg	23	12.8	1.16	15.56	1.28	0.15	0.54	1.70
Protium widgrenii Engl.	23	12.8	1.16	15.56	1.28	0.12	0.42	1.58
Syagrus romanzoffiana (Cham.) Glassman	21	11.7	1.06	11.11	0.91	0.25	0.87	1.93
Luehea candicans Mart. & Zucc.	9	5.0	0.45	7.78	0.64	0.41	1.47	1.92
Pera glabrata (Schott) Poepp. ex Baill.	14	7.8	0.71	8.89	0.73	0.30	1.08	1.79
Maprounea guianensis Aubl.	22	12.2	1.11	12.22	1.01	0.10	0.37	1.47
Croton rugosus Spreng.	18	10.0	0.91	8.89	0.73	0.24	0.84	1.75
Nectandra grandiflora Nees	18	10.0	0.91	13.33	1.10	0.12	0.42	1.32
Casearia obliqua Spreng.	18	10.0	0.91	15.56	1.28	0.05	0.19	1.10
Piptocarpha macropoda (DC.) Baker	21	11.7	1.06	8.89	0.73	0.16	0.58	1.64
Vernonanthura divaricata (Spreng.) H.Rob.	20	11.1	1.01	8.89	0.73	0.14	0.49	1.50
Peltophorum dubium (Spreng.) Taub.	12	6.7	0.60	11.11	0.91	0.19	0.66	1.27
Eugenia florida DC.	13	7.2	0.65	11.11	0.91	0.16	0.58	1.23
Casearia decandra Jacq.	13	7.2	0.65	13.33	1.10	0.10	0.37	1.02
Xylopia brasiliensis Spreng.	12	6.7	0.60	11.11	0.91	0.15	0.55	1.16
Siphoneugena densiflora O.Berg	15	8.3	0.76	7.78	0.64	0.17	0.60	1.36
Casearia arborea (Rich.) Urb.	13	7.2	0.65	13.33	1.10	0.05	0.18	0.83
Trichilia pallida Sw.	15	8.3	0.76	10.00	0.82	0.09	0.30	1.06
Urera baccifera (L.) Gaudich. ex Wedd.	14	7.8	0.71	10.00	0.82	0.10	0.35	1.05
Machaerium nyctitans (Vell.) Benth.	8	4.4	0.40	5.56	0.46	0.28	1.00	1.41
Tapirira guianensis Aubl.	13	7.2	0.65	8.89	0.73	0.12	0.44	1.09

Guatteria australis A.St.-Hil.	11	6.1	0.55	10.00	0.82	0.10	0.36	0.92
Astronium graveolens Jacq.	11	6.1	0.55	7.78	0.64	0.15	0.54	1.10
Annona sylvatica A.St.-Hil.	7	3.9	0.35	7.78	0.64	0.21	0.74	1.09
Inga vera Willd.	11	6.1	0.55	7.78	0.64	0.15	0.53	1.08
Ilex paraguariensis A.St.-Hil.	12	6.7	0.60	8.89	0.73	0.07	0.23	0.84
Protium heptaphyllum (Aubl.) Marchand	13	7.2	0.65	8.89	0.73	0.05	0.16	0.82
Cabralea canjerana (Vell.) Mart.	10	5.6	0.50	8.89	0.73	0.07	0.24	0.74
Cedrela fissilis Vell.	11	6.1	0.55	8.89	0.73	0.05	0.19	0.74
Zanthoxylum rhoifolium Lam.	10	5.6	0.50	10.00	0.82	0.03	0.12	0.62
Astronium fraxinifolium Schott	8	4.4	0.40	5.56	0.46	0.16	0.58	0.98
Tapirira obtusa (Benth.) J.D.Mitch.	9	5.0	0.45	7.78	0.64	0.09	0.33	0.78
Ocotea diospyrifolia (Meisn.) Mez	8	4.4	0.40	5.56	0.46	0.11	0.38	0.79
Byrsonima crassifolia (L.) Kunth	8	4.4	0.40	8.89	0.73	0.03	0.10	0.50
Ocotea pulchella (Nees & Mart.) Mez	8	4.4	0.40	6.67	0.55	0.07	0.26	0.66
Trichilia elegans A.Juss.	7	3.9	0.35	5.56	0.46	0.11	0.38	0.73
Matayba juglandifolia (Cambess.) Radlk.	7	3.9	0.35	6.67	0.55	0.08	0.28	0.64
Dalbergia villosa (Benth.) Benth.	6	3.3	0.30	6.67	0.55	0.09	0.31	0.61
Machaerium brasiliense Vogel	3	1.7	0.15	3.33	0.27	0.20	0.72	0.87
Piptadenia gonoacantha (Mart.) J.F.Macbr.	9	5.0	0.45	4.44	0.37	0.08	0.29	0.74
Marlierea racemosa (Vell.) Kiaersk.	6	3.3	0.30	5.56	0.46	0.10	0.34	0.64
Aniba firmula (Nees & Mart.) Mez	7	3.9	0.35	7.78	0.64	0.03	0.10	0.45
Protium spruceanum (Benth.) Engl.	7	3.9	0.35	6.67	0.55	0.04	0.15	0.50
Cordia sellowiana Cham.	6	3.3	0.30	5.56	0.46	0.08	0.29	0.60
Casearia lasiophylla Eichler	8	4.4	0.40	6.67	0.55	0.03	0.10	0.50
Endlicheria paniculata (Spreng.) J.F.Macbr.	6	3.3	0.30	6.67	0.55	0.04	0.14	0.45
Persea willdenowii Kosterm.	3	1.7	0.15	2.22	0.18	0.17	0.62	0.77
Ficus cyclophylla (Miq.) Miq.	2	1.1	0.10	1.11	0.09	0.20	0.72	0.82
Eugenia leitonii ined.	5	2.8	0.25	5.56	0.46	0.06	0.20	0.45
Tachigali rugosa (Mart. ex Benth.) Zarucchi & Pipoly	2	1.1	0.10	1.11	0.09	0.20	0.70	0.80
Machaerium stipitatum Vogel	5	2.8	0.25	5.56	0.46	0.04	0.16	0.41
Platypodium elegans Vogel	7	3.9	0.35	2.22	0.18	0.09	0.32	0.67
Prunus myrtifolia (L.) Urb.	5	2.8	0.25	5.56	0.46	0.04	0.13	0.38
Annona cacans Warm.	4	2.2	0.20	4.44	0.37	0.07	0.26	0.46
Sebastiania commersoniana (Baill.) L.B.Sm. & Downs	5	2.8	0.25	4.44	0.37	0.06	0.20	0.46
Senna macranthera (DC. ex Collad.) H.S.Irwin & Barneby	7	3.9	0.35	3.33	0.27	0.05	0.19	0.54
Senegalia polyphylla (DC.) Britton & Rose	5	2.8	0.25	4.44	0.37	0.05	0.18	0.43
Alchornea glandulosa Poepp. & Endl.	5	2.8	0.25	4.44	0.37	0.04	0.14	0.39
Lamanonia ternata Vell.	4	2.2	0.20	4.44	0.37	0.05	0.18	0.38
Terminalia glabrescens Mart.	4	2.2	0.20	3.33	0.27	0.08	0.27	0.47
Eugenia sonderiana DC.	4	2.2	0.20	4.44	0.37	0.04	0.15	0.35
Casearia mariquitensis Kunth	4	2.2	0.20	4.44	0.37	0.04	0.14	0.34
Luehea grandiflora Mart. & Zucc.	5	2.8	0.25	4.44	0.37	0.02	0.09	0.34
Cecropia pachystachya Trécul	5	2.8	0.25	4.44	0.37	0.02	0.07	0.32
Apuleia leiocarpa (Vogel) J.F.Macbr.	4	2.2	0.20	3.33	0.27	0.06	0.21	0.41
Nectandra oppositifolia Nees	3	1.7	0.15	3.33	0.27	0.06	0.22	0.37
Agonandra excelsa Griseb.	4	2.2	0.20	4.44	0.37	0.02	0.08	0.28
Dendropanax cuneatus (DC.) Decne. & Planch.	4	2.2	0.20	4.44	0.37	0.02	0.06	0.26
Bauhinia longifolia (Bong.) Steud.	7	3.9	0.35	2.22	0.18	0.02	0.07	0.42
Vismia guianensis (Aubl.) Choisy	5	2.8	0.25	2.22	0.18	0.04	0.15	0.40
Acacia polyphylla DC.	4	2.2	0.20	3.33	0.27	0.02	0.08	0.28
Ocotea minarum (Nees & Mart.) Mez	2	1.1	0.10	2.22	0.18	0.08	0.27	0.37
Albizia polycephala (Benth.) Killip ex Record	4	2.2	0.20	3.33	0.27	0.02	0.07	0.27
Zanthoxylum monogynum A.St.-Hil.	4	2.2	0.20	3.33	0.27	0.02	0.05	0.25

Matayba guianensis Aubl.	3	1.7	0.15	3.33	0.27	0.03	0.10	0.25
Guarea kunthiana A.Juss.	3	1.7	0.15	2.22	0.18	0.05	0.18	0.33
Blepharocalyx salicifolius (Kunth) O.Berg	3	1.7	0.15	3.33	0.27	0.02	0.09	0.24
Faramea latifolia (Cham. & Schltdl.) DC.	3	1.7	0.15	3.33	0.27	0.02	0.08	0.23
Persea major L.E.Kopp	3	1.7	0.15	3.33	0.27	0.02	0.08	0.23
Tibouchina arborea (Gardner) Cogn.	2	1.1	0.10	2.22	0.18	0.06	0.21	0.31
Jacaranda macrantha Cham.	3	1.7	0.15	3.33	0.27	0.02	0.06	0.22
Ilex cerasifolia Reissek	3	1.7	0.15	3.33	0.27	0.02	0.06	0.21
Guazuma ulmifolia Lam.	3	1.7	0.15	3.33	0.27	0.02	0.06	0.21
Callisthene major Mart. & Zucc.	3	1.7	0.15	3.33	0.27	0.01	0.04	0.19
Aspidosperma dispermum Müll.Arg.	3	1.7	0.15	3.33	0.27	0.01	0.04	0.19
Vitex polygama Cham.	3	1.7	0.15	3.33	0.27	0.01	0.04	0.19
Chrysophyllum gonocarpum (Mart. & Eichler ex Miq.) Engl.	3	1.7	0.15	3.33	0.27	0.01	0.02	0.17
Miconia pepericarpa DC.	2	1.1	0.10	2.22	0.18	0.05	0.16	0.26
Vochysia magnifica Warm.	1	0.6	0.05	1.11	0.09	0.08	0.29	0.34
Roupala montana Aubl.	2	1.1	0.10	2.22	0.18	0.04	0.13	0.23
Inga marginata Willd.	4	2.2	0.20	2.22	0.18	0.01	0.02	0.22
Calyptranthes brasiliensis Spreng.	4	2.2	0.20	2.22	0.18	0.01	0.02	0.22
Euterpe edulis Mart.	3	1.7	0.15	2.22	0.18	0.02	0.07	0.22
Campomanesia simulans M.L.Kawas.	3	1.7	0.15	2.22	0.18	0.02	0.06	0.21
Hippocratea volubilis L.	2	1.1	0.10	2.22	0.18	0.03	0.11	0.21
Mabea pohliana (Benth.) Müll. Arg.	4	2.2	0.20	1.11	0.09	0.02	0.09	0.29
Aspidosperma spruceanum Benth. ex Müll.Arg.	2	1.1	0.10	2.22	0.18	0.03	0.09	0.20
Coutarea hexandra (Jacq.) K.Schum.	3	1.7	0.15	2.22	0.18	0.01	0.04	0.19
Ficus enormis Mart. ex Miq.	1	0.6	0.05	1.11	0.09	0.07	0.23	0.28
Diatenopteryx sorbifolia Radlk.	3	1.7	0.15	1.11	0.09	0.04	0.13	0.28
Ficus adhatodifolia Schott ex Spreng.	2	1.1	0.10	2.22	0.18	0.02	0.08	0.18
Eugenia dodonaeifolia Cambess.	2	1.1	0.10	2.22	0.18	0.02	0.08	0.18
Cordiera sessilis (Vell.) Kuntze	3	1.7	0.15	2.22	0.18	0.01	0.02	0.17
Cecropia glaziovii Snethl.	3	1.7	0.15	2.22	0.18	0.00	0.02	0.17
Myrsine umbellata Mart.	2	1.1	0.10	2.22	0.18	0.02	0.07	0.17
Eugenia widgrenii Sond. ex O.Berg	2	1.1	0.10	2.22	0.18	0.02	0.06	0.16
Zanthoxylum riedelianum Engl.	2	1.1	0.10	2.22	0.18	0.02	0.05	0.16
Celtis brasiliensis (Gardner) Planch.	2	1.1	0.10	2.22	0.18	0.01	0.05	0.15
Aspidosperma olivaceum Müll.Arg.	2	1.1	0.10	2.22	0.18	0.01	0.04	0.14
Aegiphila sellowiana Cham.	2	1.1	0.10	2.22	0.18	0.01	0.04	0.14
Byrsonima laxiflora Griseb.	2	1.1	0.10	2.22	0.18	0.01	0.04	0.14
Andira ormosioides Benth.	2	1.1	0.10	2.22	0.18	0.01	0.03	0.13
Trichilia casaretti C.DC.	2	1.1	0.10	2.22	0.18	0.01	0.03	0.13
Citronella paniculata (Mart.) R.A.Howard	2	1.1	0.10	2.22	0.18	0.01	0.03	0.13
Casearia gossypiosperma Briq.	2	1.1	0.10	2.22	0.18	0.01	0.03	0.13
Aspidosperma discolor A.DC.	2	1.1	0.10	2.22	0.18	0.01	0.03	0.13
Aiouea saligna Meisn.	2	1.1	0.10	2.22	0.18	0.01	0.02	0.12
Myrcia hebepetala DC.	2	1.1	0.10	2.22	0.18	0.01	0.02	0.12
Cordiera elliptica (Cham.) Kuntze	2	1.1	0.10	2.22	0.18	0.00	0.02	0.12
Inga cylindrica (Vell.) Mart.	1	0.6	0.05	1.11	0.09	0.04	0.16	0.21
Chrysophyllum marginatum (Hook. & Arn.) Radlk.	2	1.1	0.10	2.22	0.18	0.00	0.02	0.12
Mabea piriri Aubl.	2	1.1	0.10	2.22	0.18	0.00	0.01	0.12
Salacia elliptica (Mart. ex Schult.) G.Don	2	1.1	0.10	2.22	0.18	0.00	0.01	0.11
Eugenia blastantha (O.Berg) D.Legrand	2	1.1	0.10	2.22	0.18	0.00	0.01	0.11
Calyptranthes widgreniana O.Berg	2	1.1	0.10	2.22	0.18	0.00	0.01	0.11
Styrax latifolius Pohl	2	1.1	0.10	2.22	0.18	0.00	0.01	0.11
Croton salutaris Casar.	3	1.7	0.15	1.11	0.09	0.01	0.03	0.18
Enterolobium contortisiliquum (Vell.) Morong	1	0.6	0.05	1.11	0.09	0.04	0.13	0.18
Aniba canelilla (Kunth) Mez	1	0.6	0.05	1.11	0.09	0.04	0.13	0.18
Aspidosperma cylindrocarpon Mull. Arg.	1	0.6	0.05	1.11	0.09	0.03	0.12	0.17

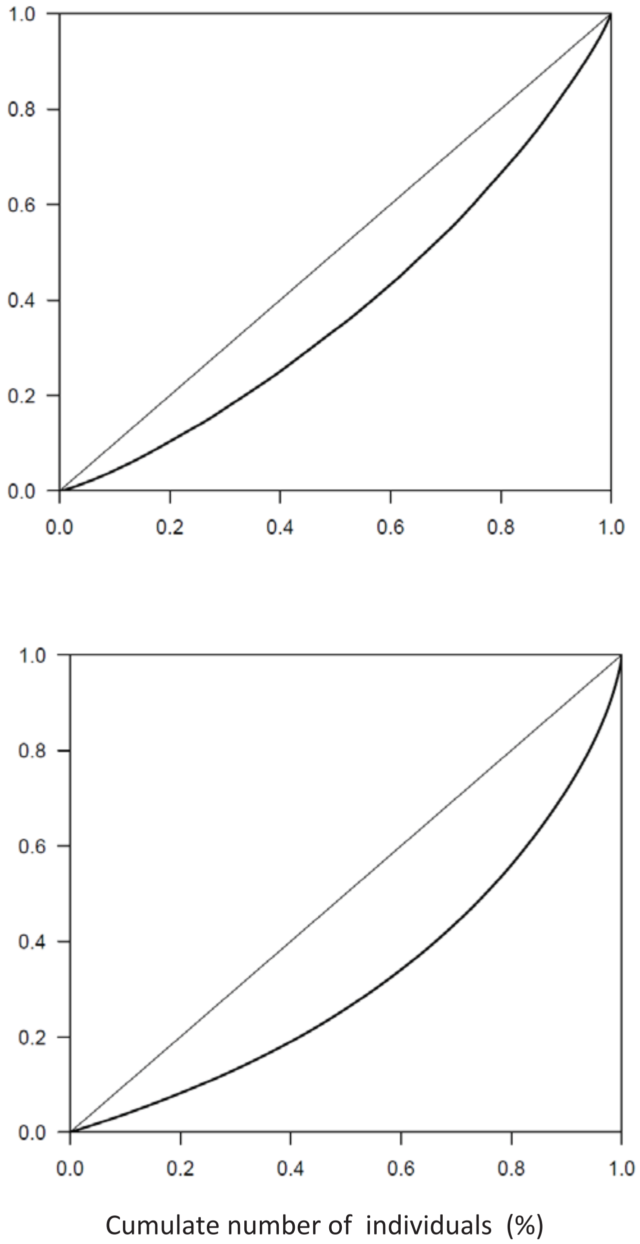
Eugenia involucrata DC.	2	1.1	0.10	1.11	0.09	0.02	0.06	0.16
Guarea macrophylla Vahl	1	0.6	0.05	1.11	0.09	0.03	0.11	0.16
Lecythis pisonis Cambess.	1	0.6	0.05	1.11	0.09	0.03	0.11	0.16
Lithrea molleoides (Vell.) Engl.	1	0.6	0.05	1.11	0.09	0.03	0.10	0.15
Maytenus robusta Reissek	1	0.6	0.05	1.11	0.09	0.03	0.09	0.14
Ocotea brachybotrya (Meisn.) Mez	2	1.1	0.10	1.11	0.09	0.01	0.03	0.13
Faramea nigrescens Mart.	2	1.1	0.10	1.11	0.09	0.01	0.03	0.13
Metrodorea nigra A.St.-Hil.	1	0.6	0.05	1.11	0.09	0.02	0.08	0.13
Solanum leucodendron Sendtn.	1	0.6	0.05	1.11	0.09	0.02	0.07	0.12
Rhamnidium elaeocarpum Reissek	1	0.6	0.05	1.11	0.09	0.02	0.07	0.12
Alchornea triplinervia (Spreng.) Müll.Arg.	1	0.6	0.05	1.11	0.09	0.02	0.07	0.12
Tachigali denudata (Vogel) Oliveira-Filho	1	0.6	0.05	1.11	0.09	0.02	0.06	0.11
Maclura tinctoria (L.) D.Don ex Steud.	1	0.6	0.05	1.11	0.09	0.02	0.06	0.11
Myrcia venulosa DC.	1	0.6	0.05	1.11	0.09	0.02	0.06	0.11
Bauhinia rufa (Bong.) Steud.	2	1.1	0.10	1.11	0.09	0.00	0.01	0.11
Handroanthus serratifolius (Vahl) S.O.Grose	1	0.6	0.05	1.11	0.09	0.02	0.06	0.11
Machaerium hirtum (Vell.) Stellfeld	1	0.6	0.05	1.11	0.09	0.01	0.04	0.09
Machaerium dimorphandrum Hoehne	1	0.6	0.05	1.11	0.09	0.01	0.03	0.08
Trichilia pallens C.DC.	1	0.6	0.05	1.11	0.09	0.01	0.03	0.08
Matayba elaeagnoides Radlk.	1	0.6	0.05	1.11	0.09	0.01	0.03	0.08
Eugenia revoluta Griseb.	1	0.6	0.05	1.11	0.09	0.01	0.03	0.08
Vochysia tucanorum Mart.	1	0.6	0.05	1.11	0.09	0.01	0.03	0.08
Allophylus racemosus Sw.	1	0.6	0.05	1.11	0.09	0.01	0.03	0.08
Diospyros burchellii Hiern	1	0.6	0.05	1.11	0.09	0.01	0.02	0.07
Ocotea laxa (Nees) Mez	1	0.6	0.05	1.11	0.09	0.01	0.02	0.07
Myrciaria floribunda (H.West ex Willd.) O.Berg	1	0.6	0.05	1.11	0.09	0.01	0.02	0.07
Ficus guaranitica Chodat	1	0.6	0.05	1.11	0.09	0.00	0.02	0.07
Ocotea bicolor Vattimo-Gil.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Psidium rufum Mart. ex DC.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Qualea cryptantha (Spreng.) Warm.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Symplocos pubescens Klotzsch ex Benth.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Sorocea bonplandii (Baill.) W.C.Burger et al.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Eugenia uniflora L.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Ficus pertusa L.f.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Cheiloclinium serratum (Cambess.) A.C.Sm.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Aralia excelsa (Griseb.) J.Wen	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Jacaratia spinosa (Aubl.) A.DC.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Senna multijuga (Rich.) H.S.Irwin & Barneby	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Guarea guidonia (L.) Sleumer	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Hymenaea courbaril L.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Styrax pohlii A.DC.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Persea rufotomentosa Nees & Mart. ex Nees	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Myrsine guianensis (Aubl.) Kuntze	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Annona emarginata (Schltdl.) H.Rainer	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Aspidosperma parvifolium A.DC.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Cordiera concolor (Cham.) Kuntze	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Miconia paulensis Naudin	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Connarus regnellii G.Schellenb.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Sapium glandulatum (Vell.) Pax	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Ouratea semiserrata (Mart. & Nees) Engl.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Cupania zanthoxyloides Radlk.	1	0.6	0.05	1.11	0.09	0.00	0.01	0.06
Cryptocarya moschata Nees & Mart. ex Nees	1	0.6	0.05	1.11	0.09	0.00	0.00	0.05



Online Resource 2: Species accumulation curve designed by Bootstrap estimator for the nine sites of seasonal semideciduous forest in southeastern Brazil. Gray lines: Species with DAP < 5.0 cm. Black lines: Species with DAP > 5.0 cm. Dashed line: standard deviation.

Online Resource 3: Species sampled in nine sites of seasonal semideciduous forest of southeastern Brazil which are endangered. VU: Vulnerable. ID: In danger. F: Frequency. A: Abundance. AT: Total Abundance.

Families/Species	**Brasil**	**Minas Gerais**	**F>5cm**	**F<5cm**	**A>5cm**	**A<5cm**	**AT**
Arecaceae							
Euterpe edulis	VU		2	0	3	0	3
Lauraceae							
Ocotea odorifera	ID	VU	8	9	130	134	264
Persea rufotomentosa	VU	1	2	1	4	5	
Lecythidaceae							
Cariniana legalis	ID		0	3	0	3	3
Meliaceae							
Cedrela fissilis	VU		4	4	11	11	22
Moraceae							
Ficus cyclophylla	VU		1	2	2	4	6
Myrtaceae							
Eugenia malacantha	ID		0	3	0	9	9
Myrcia diaphana	VU		0	1	0	2	2
Rubiaceae							
Rudgea jasminoides	VU		0	5	0	9	9



Online Resource 4: Lorenz curve for height (G=0.26; CI=0.22to0.24) e diameter (G=0.36; IC=0.34 to 0.37) of individuals with DAP ≥ 5cm sampled in nine sites of seasonal semideciduous forest of southeastern Brazil.

Online Resource 5: Phytosociological parameters of tree species with DAP< 5.0 cm sampled in nine sites of seasonal semideciduous forest of southeastern Brazil. N= number of individuals, AD= absolute density, RD= relative density, AF= absolute frequency, RF= relative frequency, ADo = absolute dominance, RDo = relative dominance and IVI = ecological value of importance.

Species	**N**	**AD**	**RD**	**AF**	**RF**	**ADo**	**RDo**	**IVI**
Siparuna guianensis Aubl.	1086	603.33	17.10	83.33	3.41	0.17	11.93	32.44
Psychotria vellosiana Benth.	309	171.67	4.86	45.56	1.86	0.03	2.31	9.04
Trichilia emarginata (Turcz.) C.DC.	211	117.22	3.32	45.56	1.86	0.05	3.78	8.97
Cupania zanthoxyloides Radlk.	239	132.78	3.76	48.89	2.00	0.04	2.62	8.38
Ocotea odorifera (Vell.) Rohwer	134	74.44	2.11	50.00	2.04	0.05	3.52	7.67
Metrodorea stipularis Mart.	179	99.44	2.82	21.11	0.86	0.04	2.99	6.67
Trichilia pallida Sw.	139	77.22	2.19	51.11	2.09	0.03	2.30	6.58
Ocotea corymbosa (Meisn.) Mez	136	75.56	2.14	46.67	1.91	0.03	2.04	6.09
Amaioua guianensis Aubl.	104	57.78	1.64	41.11	1.68	0.04	2.72	6.04
Trichilia clausseni C.DC.	131	72.78	2.06	22.22	0.91	0.03	2.40	5.37
Eugenia acutata Miq.	130	72.22	2.05	25.56	1.05	0.03	1.99	5.08
Myrcia splendens (Sw.) DC.	96	53.33	1.51	45.56	1.86	0.02	1.61	4.99
Cupania vernalis Cambess.	89	49.44	1.40	40.00	1.64	0.03	1.93	4.97
Trichilia catigua A.Juss.	105	58.33	1.65	38.89	1.59	0.02	1.60	4.84
Casearia sylvestris Sw.	95	52.78	1.50	36.67	1.50	0.02	1.77	4.77
Copaifera langsdorffii Desf.	94	52.22	1.48	36.67	1.50	0.02	1.53	4.51
Croton floribundus Spreng.	79	43.89	1.24	20.00	0.82	0.03	2.36	4.43
Mollinedia widgrenii A.DC.	72	40.00	1.13	26.67	1.09	0.03	2.02	4.25
Duguetia lanceolata A.St.-Hil.	66	36.67	1.04	36.67	1.50	0.02	1.37	3.90
Cryptocarya aschersoniana Mez	60	33.33	0.94	34.44	1.41	0.02	1.15	3.50
Nectandra grandiflora Nees	62	34.44	0.98	25.56	1.05	0.02	1.44	3.46
Casearia arborea (Rich.) Urb.	64	35.56	1.01	25.56	1.05	0.02	1.15	3.20
Piper amalago L.	95	52.78	1.50	18.89	0.77	0.01	0.88	3.15
Casearia decandra Jacq.	57	31.67	0.90	27.78	1.14	0.01	1.03	3.06
Miconia willdenowii Klotzsch ex Naudin	53	29.44	0.83	32.22	1.32	0.01	0.83	2.98
Galipea jasminiflora (A.St.-Hil.) Engl.	89	49.44	1.40	10.00	0.41	0.01	0.90	2.71
Calyptranthes clusiifolia O.Berg	51	28.33	0.80	21.11	0.86	0.01	1.01	2.68
Lacistema hasslerianum Chodat	82	45.56	1.29	18.89	0.77	0.01	0.53	2.59
Ixora brevifolia Benth.	39	21.67	0.61	22.22	0.91	0.01	1.06	2.58
Protium widgrenii Engl.	37	20.56	0.58	25.56	1.05	0.01	0.85	2.48
Protium spruceanum (Benth.) Engl.	44	24.44	0.69	27.78	1.14	0.01	0.63	2.46
Trichilia elegans A.Juss.	59	32.78	0.93	14.44	0.59	0.01	0.85	2.37
Mabea fistulifera Mart.	51	28.33	0.80	14.44	0.59	0.01	0.82	2.22
Erythroxylum pelleterianum A.St.-Hil.	42	23.33	0.66	25.56	1.05	0.01	0.51	2.22
Xylopia brasiliensis Spreng.	48	26.67	0.76	17.78	0.73	0.01	0.72	2.20
Tapirira obtusa (Benth.) J.D.Mitch.	35	19.44	0.55	26.67	1.09	0.01	0.55	2.19
Matayba juglandifolia (Cambess.) Radlk.	37	20.56	0.58	21.11	0.86	0.01	0.73	2.18
Marlierea racemosa (Vell.) Kiaersk.	37	20.56	0.58	24.44	1.00	0.01	0.60	2.18
Psychotria triflora Müll.Arg.	62	34.44	0.98	20.00	0.82	0.01	0.36	2.16
Coffea arabica L.	60	33.33	0.94	17.78	0.73	0.01	0.42	2.09

***Piper arboreum* Aubl.**	45	25.00	0.71	25.56	1.05	0.00	0.32	2.07
***Maprounea guianensis* Aubl.**	34	18.89	0.54	16.67	0.68	0.01	0.85	2.06
***Psychotria hastisepala* Müll.Arg.**	54	30.00	0.85	18.89	0.77	0.00	0.34	1.97
***Matayba guianensis* Aubl.**	32	17.78	0.50	17.78	0.73	0.01	0.73	1.96
***Protium heptaphyllum* (Aubl.) Marchand**	36	20.00	0.57	20.00	0.82	0.01	0.55	1.94
***Astronium fraxinifolium* Schott**	34	18.89	0.54	20.00	0.82	0.01	0.51	1.86
***Eugenia florida* DC.**	27	15.00	0.43	21.11	0.86	0.01	0.45	1.74
***Ilex paraguariensis* A.St.-Hil.**	24	13.33	0.38	17.78	0.73	0.01	0.52	1.62
***Cordia sellowiana* Cham.**	28	15.56	0.44	24.44	1.00	0.00	0.17	1.61
***Casearia obliqua* Spreng.**	20	11.11	0.31	16.67	0.68	0.01	0.61	1.61
***Endlicheria paniculata* (Spreng.) J.F.Macbr.**	22	12.22	0.35	17.78	0.73	0.01	0.52	1.60
***Byrsonima laxiflora* Griseb.**	21	11.67	0.33	15.56	0.64	0.01	0.45	1.41
***Pera glabrata* (Schott) Poepp. ex Baill.**	19	10.56	0.30	15.56	0.64	0.01	0.47	1.40
***Tapirira guianensis* Aubl.**	28	15.56	0.44	11.11	0.45	0.01	0.48	1.37
***Zanthoxylum monogynum* A.St.-Hil.**	21	11.67	0.33	15.56	0.64	0.00	0.32	1.29
***Dendropanax cuneatus* (DC.) Decne. & Planch.**	22	12.22	0.35	11.11	0.45	0.01	0.45	1.25
***Cabralea canjerana* (Vell.) Mart.**	20	11.11	0.31	14.44	0.59	0.00	0.35	1.25
***Myrsine umbellata* Mart.**	17	9.44	0.27	14.44	0.59	0.01	0.39	1.25
***Myrcia pubiflora* DC.**	28	15.56	0.44	12.22	0.50	0.00	0.29	1.24
***Siphoneugena densiflora* O.Berg**	17	9.44	0.27	14.44	0.59	0.01	0.36	1.22
***Ilex cerasifolia* Reissek**	16	8.89	0.25	16.67	0.68	0.00	0.27	1.20
***Astronium graveolens* Jacq.**	15	8.33	0.24	14.44	0.59	0.01	0.37	1.20
***Senegalia polyphylla* (DC.) Britton & Rose**	29	16.11	0.46	8.89	0.36	0.00	0.36	1.18
***Guarea kunthiana* A.Juss.**	20	11.11	0.31	10.00	0.41	0.00	0.28	1.00
***Eugenia sonderiana* DC.**	15	8.33	0.24	11.11	0.45	0.00	0.31	1.00
***Albizia polycephala* (Benth.) Killip ex Record**	14	7.78	0.22	13.33	0.55	0.00	0.23	1.00
***Casearia lasiophylla* Eichler**	12	6.67	0.19	11.11	0.45	0.00	0.35	0.99
***Annona sylvatica* A.St.-Hil.**	13	7.22	0.20	7.78	0.32	0.01	0.46	0.98
***Miconia pepericarpa* DC.**	18	10.00	0.28	11.11	0.45	0.00	0.23	0.96
***Machaerium nyctitans* (Vell.) Benth.**	12	6.67	0.19	12.22	0.50	0.00	0.27	0.96
***Byrsonima crassifolia* (L.) Kunth**	11	6.11	0.17	11.11	0.45	0.00	0.30	0.93
***Inga marginata* Willd.**	18	10.00	0.28	11.11	0.45	0.00	0.18	0.92
***Croton rugosus* Spreng.**	16	8.89	0.25	4.44	0.18	0.01	0.48	0.91
***Calyptranthes widgreniana* O.Berg**	12	6.67	0.19	10.00	0.41	0.00	0.29	0.88
***Piper aduncum* L.**	20	11.11	0.31	10.00	0.41	0.00	0.16	0.88
***Dalbergia villosa* (Benth.) Benth.**	11	6.11	0.17	7.78	0.32	0.00	0.36	0.85
***Chrysophyllum marginatum* (Hook. & Arn.) Radlk.**	19	10.56	0.30	6.67	0.27	0.00	0.27	0.84
***Senna macranthera* (DC. ex Collad.) H.S.Irwin & Barneby**	12	6.67	0.19	7.78	0.32	0.00	0.33	0.84
***Calyptranthes brasiliensis* Spreng.**	11	6.11	0.17	7.78	0.32	0.00	0.34	0.83
***Eugenia leitonii* ined.**	17	9.44	0.27	7.78	0.32	0.00	0.24	0.82
***Platycyamus regnellii* Benth.**	11	6.11	0.17	7.78	0.32	0.00	0.32	0.81
***Piptadenia gonoacantha* (Mart.) J.F.Macbr.**	11	6.11	0.17	5.56	0.23	0.01	0.41	0.81

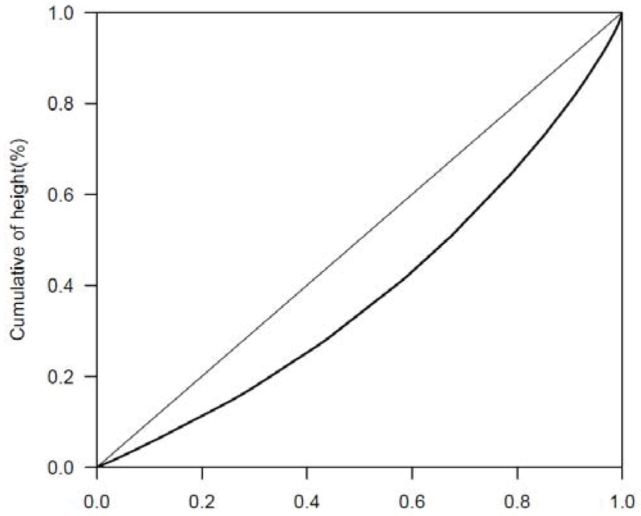
Chrysophyllum gonocarpum (Mart. & Eichler ex Miq.) Engl.	11	6.11	0.17	8.89	0.36	0.00	0.27	0.80
Cordiera concolor (Cham.) Kuntze	16	8.89	0.25	6.67	0.27	0.00	0.26	0.79
Cedrela fissilis Vell.	11	6.11	0.17	5.56	0.23	0.01	0.39	0.79
Guapira opposita (Vell.) Reitz	13	7.22	0.20	11.11	0.45	0.00	0.12	0.78
Ocotea diospyrifolia (Meisn.) Mez	11	6.11	0.17	8.89	0.36	0.00	0.24	0.78
Sebastiania commersoniana (Baill.) L.B.Sm. & Downs	19	10.56	0.30	5.56	0.23	0.00	0.24	0.77
Vitex polygama Cham.	11	6.11	0.17	8.89	0.36	0.00	0.23	0.77
Bauhinia longifolia (Bong.) Steud.	10	5.56	0.16	8.89	0.36	0.00	0.24	0.77
Aspidosperma discolor A.DC.	12	6.67	0.19	10.00	0.41	0.00	0.17	0.76
Eugenia paracatuana O.Berg	13	7.22	0.20	10.00	0.41	0.00	0.12	0.74
Solanum leucodendron Sendtn.	9	5.00	0.14	8.89	0.36	0.00	0.23	0.73
Apuleia leiocarpa (Vogel) J.F.Macbr.	11	6.11	0.17	10.00	0.41	0.00	0.14	0.72
Machaerium hirtum (Vell.) Stellfeld	10	5.56	0.16	8.89	0.36	0.00	0.19	0.71
Guatteria australis A.St.-Hil.	9	5.00	0.14	8.89	0.36	0.00	0.20	0.70
Trichilia casaretti C.DC.	7	3.89	0.11	7.78	0.32	0.00	0.25	0.68
Inga vera Willd.	10	5.56	0.16	8.89	0.36	0.00	0.14	0.67
Lippia brasiliensis T.R.S.Silva	13	7.22	0.20	8.89	0.36	0.00	0.08	0.65
Machaerium dimorphandrum Hoehne	10	5.56	0.16	10.00	0.41	0.00	0.07	0.64
Prunus myrtifolia (L.) Urb.	8	4.44	0.13	7.78	0.32	0.00	0.20	0.64
Zanthoxylum rhoifolium Lam.	10	5.56	0.16	6.67	0.27	0.00	0.21	0.64
Machaerium stipitatum Vogel	8	4.44	0.13	7.78	0.32	0.00	0.17	0.61
Guapira hirsuta (Choisy) Lundell	9	5.00	0.14	8.89	0.36	0.00	0.10	0.61
Allophylus edulis (A.St.-Hil. et al.) Hieron. ex Niederl.	16	8.89	0.25	5.56	0.23	0.00	0.13	0.61
Nectandra oppositifolia Nees	8	4.44	0.13	5.56	0.23	0.00	0.24	0.59
Acacia polyphylla DC.	12	6.67	0.19	5.56	0.23	0.00	0.17	0.59
Machaerium villosum Vogel	6	3.33	0.09	5.56	0.23	0.00	0.25	0.57
Maytenus glaziovi Reissek	9	5.00	0.14	8.89	0.36	0.00	0.06	0.56
Styrax camporum Pohl	7	3.89	0.11	6.67	0.27	0.00	0.17	0.55
Aniba firmula (Nees & Mart.) Mez	6	3.33	0.09	5.56	0.23	0.00	0.23	0.55
Myrciaria floribunda (H.West ex Willd.) O.Berg	9	5.00	0.14	7.78	0.32	0.00	0.09	0.55
Alchornea glandulosa Poepp. & Endl.	8	4.44	0.13	7.78	0.32	0.00	0.09	0.53
Maytenus robusta Reissek	8	4.44	0.13	6.67	0.27	0.00	0.12	0.52
Peltophorum dubium (Spreng.) Taub.	13	7.22	0.20	3.33	0.14	0.00	0.18	0.52
Ocotea pulchella (Nees & Mart.) Mez	8	4.44	0.13	6.67	0.27	0.00	0.11	0.51
Piptocarpha macropoda (DC.) Baker	7	3.89	0.11	5.56	0.23	0.00	0.17	0.51
Mabea pohliana (Benth.) Müll. Arg.	10	5.56	0.16	4.44	0.18	0.00	0.16	0.50
Urera baccifera (L.) Gaudich. ex Wedd.	7	3.89	0.11	5.56	0.23	0.00	0.16	0.50
Miconia prasina (Sw.) DC.	8	4.44	0.13	6.67	0.27	0.00	0.10	0.49
Eugenia malacantha D.Legrand	9	5.00	0.14	6.67	0.27	0.00	0.08	0.49
Miconia paulensis Naudin	8	4.44	0.13	5.56	0.23	0.00	0.14	0.49
Jacaranda macrantha Cham.	6	3.33	0.09	4.44	0.18	0.00	0.21	0.49
Rudgea jasminoides (Cham.) Müll.Arg.	9	5.00	0.14	6.67	0.27	0.00	0.07	0.48
Schefflera calva (Cham.) Frodin & Fiaschi	10	5.56	0.16	5.56	0.23	0.00	0.09	0.48

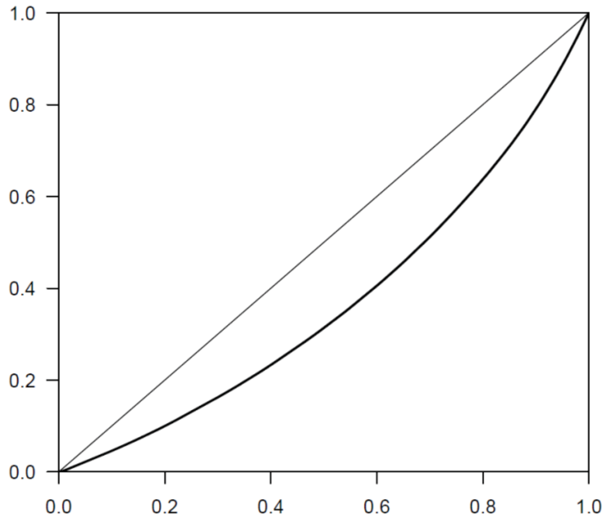
***Myrcia hebepetala* DC.**	7	3.89	0.11	4.44	0.18	0.00	0.19	0.48
***Aspidosperma polyneuron* Müll.Arg.**	6	3.33	0.09	6.67	0.27	0.00	0.10	0.46
***Zollernia ilicifolia* (Brongn.) Vogel**	6	3.33	0.09	5.56	0.23	0.00	0.14	0.46
***Myrsine coriacea* (Sw.) R.Br. ex Roem. & Schult.**	8	4.44	0.13	5.56	0.23	0.00	0.09	0.45
***Leandra scabra* DC.**	7	3.89	0.11	6.67	0.27	0.00	0.06	0.44
***Aparisthmium cordatum* (A.Juss.) Baill.**	8	4.44	0.13	5.56	0.23	0.00	0.09	0.44
***Strychnos brasiliensis* Mart.**	6	3.33	0.09	6.67	0.27	0.00	0.05	0.42
***Daphnopsis brasiliensis* Mart. & Zucc.**	4	2.22	0.06	3.33	0.14	0.00	0.22	0.42
***Aloysia virgata* (Ruiz & Pav.) Juss.**	15	8.33	0.24	1.11	0.05	0.00	0.13	0.41
***Coutarea hexandra* (Jacq.) K.Schum.**	5	2.78	0.08	5.56	0.23	0.00	0.10	0.40
***Miconia albicans* (Sw.) Triana**	6	3.33	0.09	5.56	0.23	0.00	0.08	0.40
***Matayba elaeagnoides* Radlk.**	6	3.33	0.09	5.56	0.23	0.00	0.08	0.40
***Sorocea bonplandii* (Baill.) W.C.Burger et al.**	6	3.33	0.09	5.56	0.23	0.00	0.07	0.39
***Faramea latifolia* (Cham. & Schltdl.) DC.**	6	3.33	0.09	5.56	0.23	0.00	0.07	0.39
***Luehea candicans* Mart. & Zucc.**	6	3.33	0.09	5.56	0.23	0.00	0.07	0.39
***Blepharocalyx salicifolius* (Kunth) O.Berg**	5	2.78	0.08	3.33	0.14	0.00	0.17	0.39
***Miconia hymenonervia* (Raddi) Cogn.**	5	2.78	0.08	4.44	0.18	0.00	0.12	0.38
***Aspidosperma parvifolium* A.DC.**	6	3.33	0.09	5.56	0.23	0.00	0.06	0.38
***Allophylus racemosus* Sw.**	5	2.78	0.08	4.44	0.18	0.00	0.12	0.38
***Myrcia fenzliana* O.Berg**	6	3.33	0.09	5.56	0.23	0.00	0.06	0.38
***Marlierea laevigata* (DC.) Kiaersk.**	4	2.22	0.06	4.44	0.18	0.00	0.12	0.36
***Faramea nigrescens* Mart.**	3	1.67	0.05	3.33	0.14	0.00	0.18	0.36
***Platypodium elegans* Vogel**	5	2.78	0.08	4.44	0.18	0.00	0.10	0.36
***Andira ormosioides* Benth.**	5	2.78	0.08	5.56	0.23	0.00	0.06	0.36
***Diatenopteryx sorbifolia* Radlk.**	5	2.78	0.08	5.56	0.23	0.00	0.05	0.35
***Aspidosperma olivaceum* Müll.Arg.**	4	2.22	0.06	4.44	0.18	0.00	0.10	0.35
***Sweetia fruticosa* Spreng.**	4	2.22	0.06	2.22	0.09	0.00	0.15	0.31
***Cecropia glaziovii* Snethl.**	3	1.67	0.05	2.22	0.09	0.00	0.17	0.31
***Tabebuia roseoalba* (Ridl.) Sandwith**	6	3.33	0.09	4.44	0.18	0.00	0.02	0.30
***Ocotea brachybotrya* (Meisn.) Mez**	5	2.78	0.08	3.33	0.14	0.00	0.08	0.30
***Eugenia widgrenii* Sond. ex O.Berg**	4	2.22	0.06	4.44	0.18	0.00	0.05	0.30
***Tachigali rugosa* (Mart. ex Benth.) Zarucchi & Pipoly**	4	2.22	0.06	3.33	0.14	0.00	0.09	0.29
***Citronella paniculata* (Mart.) R.A.Howard**	5	2.78	0.08	4.44	0.18	0.00	0.02	0.28
***Styrax latifolius* Pohl**	3	1.67	0.05	3.33	0.14	0.00	0.10	0.28
***Terminalia glabrescens* Mart.**	3	1.67	0.05	3.33	0.14	0.00	0.09	0.27
***Schefflera morototoni* (Aubl.) Maguire et al.**	4	2.22	0.06	3.33	0.14	0.00	0.07	0.27
***Daphnopsis fasciculata* (Meisn.) Nevling**	4	2.22	0.06	4.44	0.18	0.00	0.02	0.26
***Trichilia pallens* C.DC.**	3	1.67	0.05	2.22	0.09	0.00	0.12	0.26
***Vernonanthura divaricata* (Spreng.) H.Rob.**	2	1.11	0.03	2.22	0.09	0.00	0.13	0.25
***Cybistax antisyphilitica* (Mart.) Mart.**	5	2.78	0.08	3.33	0.14	0.00	0.04	0.25
***Prockia crucis* P.Browne ex L.**	4	2.22	0.06	3.33	0.14	0.00	0.04	0.24
***Vismia guianensis* (Aubl.) Choisy**	3	1.67	0.05	3.33	0.14	0.00	0.05	0.23
***Machaerium brasiliense* Vogel**	4	2.22	0.06	3.33	0.14	0.00	0.03	0.23

Psidium rufum Mart. ex DC.	3	1.67	0.05	3.33	0.14	0.00	0.04	0.23
Siparuna brasiliensis (Spreng.) A.DC.	3	1.67	0.05	3.33	0.14	0.00	0.04	0.23
Persea rufotomentosa Nees & Mart. ex Nees	4	2.22	0.06	2.22	0.09	0.00	0.07	0.23
Vochysia magnifica Warm.	3	1.67	0.05	3.33	0.14	0.00	0.04	0.22
Banara axilliflora Sleumer.	5	2.78	0.08	2.22	0.09	0.00	0.05	0.22
Psychotria cephalantha (Müll.Arg.) Standl.	4	2.22	0.06	3.33	0.14	0.00	0.02	0.22
Anaxagorea dolichocarpa Sprague & Sandwith	4	2.22	0.06	2.22	0.09	0.00	0.06	0.21
Mabea piriri Aubl.	3	1.67	0.05	2.22	0.09	0.00	0.07	0.21
Cecropia pachystachya Trécul	3	1.67	0.05	1.11	0.05	0.00	0.11	0.21
Salacia elliptica (Mart. ex Schult.) G.Don	2	1.11	0.03	2.22	0.09	0.00	0.08	0.20
Cariniana legalis (Mart.) Kuntze	3	1.67	0.05	3.33	0.14	0.00	0.02	0.20
Campomanesia simulans M.L.Kawas.	2	1.11	0.03	1.11	0.05	0.00	0.12	0.19
Annona glabra L.	3	1.67	0.05	2.22	0.09	0.00	0.05	0.19
Sloanea monosperma Vell.	3	1.67	0.05	2.22	0.09	0.00	0.05	0.19
Persea major L.E.Kopp	2	1.11	0.03	1.11	0.05	0.00	0.10	0.18
Handroanthus serratifolius (Vahl) S.O.Grose	2	1.11	0.03	2.22	0.09	0.00	0.05	0.17
Eriotheca candolleana (K.Schum.) A.Robyns	3	1.67	0.05	2.22	0.09	0.00	0.03	0.17
Ficus cyclophylla (Miq.) Miq.	2	1.11	0.03	1.11	0.05	0.00	0.09	0.17
Eugenia blastantha (O.Berg) D.Legrand	2	1.11	0.03	2.22	0.09	0.00	0.04	0.16
Miconia urophylla DC.	2	1.11	0.03	2.22	0.09	0.00	0.03	0.16
Maytenus ilicifolia Mart. ex Reissek	2	1.11	0.03	2.22	0.09	0.00	0.03	0.15
Xylosma prockia (Turcz.) Turcz.	2	1.11	0.03	2.22	0.09	0.00	0.03	0.15
Miconia cinerascens Miq.	4	2.22	0.06	1.11	0.05	0.00	0.04	0.15
Aegiphila sellowiana Cham.	2	1.11	0.03	2.22	0.09	0.00	0.02	0.15
Luehea divaricata Mart. & Zucc.	2	1.11	0.03	1.11	0.05	0.00	0.07	0.14
Sebastiania brasiliensis Spreng.	2	1.11	0.03	2.22	0.09	0.00	0.02	0.14
Tibouchina arborea (Gardner) Cogn.	2	1.11	0.03	2.22	0.09	0.00	0.02	0.14
Luehea grandiflora Mart. & Zucc.	2	1.11	0.03	2.22	0.09	0.00	0.02	0.14
Campomanesia xanthocarpa (Mart.) O.Berg	2	1.11	0.03	2.22	0.09	0.00	0.02	0.14
Croton salutaris Casar.	1	0.56	0.02	1.11	0.05	0.00	0.08	0.14
Qualea dichotoma (Mart.) Warm.	2	1.11	0.03	2.22	0.09	0.00	0.01	0.14
Alseis floribunda Schott	2	1.11	0.03	2.22	0.09	0.00	0.01	0.13
Diospyros inconstans Jacq.	2	1.11	0.03	2.22	0.09	0.00	0.01	0.13
Andira fraxinifolia Benth.	1	0.56	0.02	1.11	0.05	0.00	0.07	0.13
Casearia gossypiosperma Briq.	1	0.56	0.02	1.11	0.05	0.00	0.07	0.13
Cordia trichotoma (Vell.) Arráb. ex Steud.	1	0.56	0.02	1.11	0.05	0.00	0.07	0.13
Syagrus romanzoffiana (Cham.) Glassman	1	0.56	0.02	1.11	0.05	0.00	0.07	0.13
Euplassa organensis (Gardner) I.M.Johnst.	2	1.11	0.03	2.22	0.09	0.00	0.01	0.13
Clethra scabra Pers.	2	1.11	0.03	2.22	0.09	0.00	0.01	0.13
Enterolobium contortisiliquum (Vell.) Morong	2	1.11	0.03	2.22	0.09	0.00	0.01	0.13
Connarus regnellii G.Schellenb.	2	1.11	0.03	2.22	0.09	0.00	0.01	0.13

Myrcia guianensis (Aubl.) DC.	3	1.67	0.05	1.11	0.05	0.00	0.03	0.13
Callisthene major Mart. & Zucc.	2	1.11	0.03	1.11	0.05	0.00	0.04	0.12
Bauhinia rufa (Bong.) Steud.	1	0.56	0.02	1.11	0.05	0.00	0.05	0.11
Eugenia uniflora L.	1	0.56	0.02	1.11	0.05	0.00	0.05	0.11
Ocotea elegans Mez	1	0.56	0.02	1.11	0.05	0.00	0.05	0.11
Ocotea minarum (Nees & Mart.) Mez	1	0.56	0.02	1.11	0.05	0.00	0.05	0.11
Qualea cryptantha (Spreng.) Warm.	1	0.56	0.02	1.11	0.05	0.00	0.05	0.11
Guarea macrophylla Vahl	1	0.56	0.02	1.11	0.05	0.00	0.04	0.10
Lafoensia glyptocarpa Koehne	1	0.56	0.02	1.11	0.05	0.00	0.04	0.10
Psidium sartorianum (O.Berg) Nied.	2	1.11	0.03	1.11	0.05	0.00	0.02	0.10
Ficus adhatodifolia Schott ex Spreng.	1	0.56	0.02	1.11	0.05	0.00	0.04	0.10
Hippocratea volubilis L.	1	0.56	0.02	1.11	0.05	0.00	0.04	0.10
Miconia cubatanensis Hoehne	1	0.56	0.02	1.11	0.05	0.00	0.04	0.10
Solanum pseudoquina A.St.-Hil.	1	0.56	0.02	1.11	0.05	0.00	0.04	0.10
Myrcia diaphana (O.Berg) N.Silveira	2	1.11	0.03	1.11	0.05	0.00	0.02	0.10
Lamanonia ternata Vell.	1	0.56	0.02	1.11	0.05	0.00	0.04	0.10
Daphnopsis utilis Warm.	2	1.11	0.03	1.11	0.05	0.00	0.02	0.10
Agonandra excelsa Griseb.	1	0.56	0.02	1.11	0.05	0.00	0.03	0.09
Ficus pertusa L.f.	1	0.56	0.02	1.11	0.05	0.00	0.03	0.09
Jacaranda puberula Cham.	1	0.56	0.02	1.11	0.05	0.00	0.03	0.09
Myrcia venulosa DC.	2	1.11	0.03	1.11	0.05	0.00	0.01	0.09
Ficus arpazusa Casar.	1	0.56	0.02	1.11	0.05	0.00	0.03	0.09
Myrcia pulchra (O.Berg) Kiaersk.	1	0.56	0.02	1.11	0.05	0.00	0.02	0.09
Ilex sapotifolia Reissek	1	0.56	0.02	1.11	0.05	0.00	0.02	0.08
Brunfelsia uniflora (Pohl.) D. Don	1	0.56	0.02	1.11	0.05	0.00	0.02	0.08
Handroanthus ochraceus (Cham.) Mattos	1	0.56	0.02	1.11	0.05	0.00	0.02	0.08
Miconia affinis DC.	1	0.56	0.02	1.11	0.05	0.00	0.02	0.08
Manihot grahamii Hook.	1	0.56	0.02	1.11	0.05	0.00	0.02	0.08
Miconia chartacea Triana	1	0.56	0.02	1.11	0.05	0.00	0.02	0.08
Ouratea semiserrata (Mart. & Nees) Engl.	1	0.56	0.02	1.11	0.05	0.00	0.02	0.08
Aegiphila integrifolia (Jacq.) Moldenke	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Maclura tinctoria (L.) D.Don ex Steud.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Siphoneugena reitzii D.Legrand	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Plinia cauliflora (Mart.) Kausel	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Annona cacans Warm.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Aspidosperma dispermum Müll.Arg.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Campomanesia guazumifolia (Cambess.) O.Berg	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Hortia arborea Engl.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Machaerium acutifolium Vogel	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Calyptranthes grandiflora O.Berg	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Styrax pohlii A.DC.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Luehea paniculata Mart. & Zucc.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Casearia mariquitensis Kunth	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Duranta vestita Cham.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07

Miconia pusilliflora (DC.) Naudin	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Myrcia tomentosa (Aubl.) DC.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Ilex affinis Gardner	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Cariniana estrellensis (Raddi) Kuntze	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Eugenia involucrata DC.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Cestrum laevigatum Schltdl.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Chionanthus trichotomus (Vell.) P.S.Green	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Psidium cattleianum Sabine	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Symplocos pubescens Klotzsch ex Benth.	1	0.56	0.02	1.11	0.05	0.00	0.01	0.07
Erythroxylum citrifolium A.St.-Hil.	1	0.56	0.02	1.11	0.05	0.00	0.00	0.07
Croton urucurana Baill.	1	0.56	0.02	1.11	0.05	0.00	0.00	0.07
Rubus brasiliensis Mart.	1	0.56	0.02	1.11	0.05	0.00	0.00	0.07
Celtis brasiliensis (Gardner) Planch.	1	0.56	0.02	1.11	0.05	0.00	0.00	0.07
Celtis iguanaea (Jacq.) Sarg.	1	0.56	0.02	1.11	0.05	0.00	0.00	0.07
Connarus rostratus (Vell.) L.B.Sm.	1	0.56	0.02	1.11	0.05	0.00	0.00	0.06
Myrceugenia miersiana (Gardner) D.Legrand & Kausel	1	0.56	0.02	1.11	0.05	0.00	0.00	0.06
Guettarda uruguensis Cham. & Schltdl.	1	0.56	0.02	1.11	0.05	0.00	0.00	0.06
Maytenus salicifolia Reissek	1	0.56	0.02	1.11	0.05	0.00	0.00	0.06
Randia armata (Sw.) DC.	1	0.56	0.02	1.11	0.05	0.00	0.00	0.06





Online Resource 6: Lorenz curve for height (G=0.23; CI=0.23to0.24) e diameter (G=0.26; IC=0.25 to 0.26) of individuals with DAP ≤ 5cm sampled in nine sites of seasonal semideciduous forest of southeastern Brazil.

CAPÍTULO 02

Spatial species turnover maintains high diversities in a tree assemblage of a fragmented tropical landscape[1]

[1] Artigo publicado na revista Ecosphere em 19/10/2016

Spatial species turnover maintains high diversities in a tree assemblage of a fragmented tropical landscape

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Abstract

The fragmentation process has many negative effects on communities, particularly for plants. This process can generate two distinct scenarios: homogenization of species composition, due to assemblage nestedness, or flora differentiation, due to spatial species turnover. The aim of this study was to answer the following: (a) Is the tree canopy and understory community of a highly fragmented landscape (9% of forest cover) the result of species nestedness or turnover?; (b) Is the pattern of additive partitioning of diversity similar between the understory and canopy tree communities?; and (c) Are landscape characteristics responsible for diversity partitioning of the tree assemblage? The studied area has low remaining forest coverage (ca. 9%), caused by deforestation that started in the 18th century, and a very heterogeneous matrix around forest patches. Within this landscape context, we hypothesized that the tree assemblage (both canopy and understory strata) in the studied fragments would be homogeneous, as a consequence of a nested subset. The study wascarried out in ninefragmentsof submontane semideciduous Atlantic Forest. All individuals with a height > 1 m in 10 (200m^2) plots in each studied fragment were sampled, measured and divided into two strata (canopy and understory individuals). The study found that the high beta diversity among plots and among fragments in both strata was due to species turnover (avoiding species homogenization), and that the landscape characteristics tested were not responsible for this result. These fragments present many rare and exclusive species, and are not dominated by only a few species. In this scenario it is necessary to conserve as many fragments as possible to protect most of tree assemblage because each fragment has a unique species composition.

Keywords: Alpha diversity; beta diversity; flora differentiation; floristic homogenization; landscape context; nestedness.

Introduction

Currently, most of the remaining forests are located in human modified landscapes, and are small, disconnected fragments in anthropogenic matrices (Melo et al. 2013). The fragmentation process is considered one of the major threats to tropical biodiversity and may cause changes in the structure and composition of communities over time (Laurance et al. 2011 for revision). For plant communities, fragmentation may generate two different scenarios: species homogenization by nestedness or floristic differentiation by species turnover (Baselga 2010, Arroyo-Rodríguez et al. 2013, Hernández-Ruedas et al. 2014). Nestedness occurs when communities from poor sites are subsets of richer ones, reflecting non-random extinction. While spatial species turnover refers to the replacement of some species by different ones (Chase and Myers 2011), and is a consequence of environmental sorting or spatial and historical constraints (Baselga 2010).

The homogenization hypothesis predicts that fragments inserted in the same modified human landscape, with similar abiotic and biotic pressures (environmental conditions, anthropogenic matrix type), will have similar dynamics and trajectory of species composition change (Laurance et al. 2007). Sensitive species will be extinguished locally, and the greater the impact and stress on the fragment, the more species will disappear in an orderly way, creating a nested community (Laurance et al. 2007). Thus, the species composition in less diverse fragments will be a subset of the species composition from the richest one (Wright et al. 1998, Baselga 2010), and, over time, the fragments become homogeneous. On the contrary, floristic differentiation occurs in fragments that are suffering from different environmental pressures and/or is located in a heterogeneous landscape (Laurance et al. 2007). In this scenario, the initial flora difference is amplified over time due to differences in the disturbances (Laurence et al. 2007). The original heterogeneity, combined with landscape heterogeneity and the different types of disturbances, generate communities with different floristic compositions in each remnant, and consequently no nested communities (Baselga 2010). This heterogeneity generates dynamic fragments, where there is no orderly species extinction, but a replacement of some original species with other species (Wright and Reeves 1992, Baselga 2010).

A recent study of tropical landscapes in Mexico (Arroyo-Rodríguez et al. 2013) found flora homogenization in fragments in areas with high deforestation levels. These results put into question the diversity maintenance in these anthropogenic landscapes over time. However, other studies (Santos et al. 2006, Hernández-Ruedas et al. 2014) show that floristic homogenization is not always predominant, and that small fragments inserted in modified human landscapes can retain large levels of biodiversity. The justification for these different results is that the trajectory of fragments inserted in human modified landscapes depends on many factors, such as anthropogenic matrix type, forest cover, distance between fragments, disturbance history and fragmentation age (Santos et al. 2006, Laurance et al. 2007, Arroyo-Rodríguez et al. 2013, Hernández-Ruedas et al. 2014). Tree assemblage within close fragments, inserted in landscapes with high forest coverage and heterogeneous matrices, have a higher chance of exchanging pollen and seeds (Antongiovanni andMetzger2005) and are less affected by edge effect (Mesquita et al. 1999). For these reasons, they are less likely to become homogeneous and have high conservation ecological value (Hernández-Ruedas et al. 2014).

An efficient way to understanding the floristic homogenization and differentiation phenomena is through additive partitioning of diversity (Laurance et al. 2007, Baselga 2010). Fragments with homogenized composition present low beta diversity, both within and among fragments, while beta diversity is high among fragments with floristic differentiation (Laurance et al. 2007, Arroyo-Rodríguez et al. 2013). Through statistical analyses it is also possible and important to decompose the beta diversity in species nesting and turnover

(Baselga 2010), because the conservation implications for each result is different. In nested subset scenarios, the implication for conservation is to preserve the most diverse fragments, since the remaining fragments are smaller subsets with fewer, more tolerant species (Wright and Reeves 1992, Laurence et al. 2007). For the floristic differentiation scenario, biodiversity conservation depends on protecting the largest possible number of fragments, because each one has a unique species composition and is therefore complementary to the landscape gamma diversity (Wright and Reeves 1992, Laurance et al. 2007).

There are few empirical studies that have quantified the spatial variation of beta diversity based on tree nesting and turnover components; there are two studies about temperate European forests (Baeten et al. 2012, Kouba et al. 2014) and one about African coastal forests (Olivier and Aarde 2014). The present study is the first for tropical forest fragments. Beta diversity, even more than alpha diversity, gives a broader view of regional biodiversity. In addition, it allows conservation status (high beta diversity within fragments) and environmental disturbance (low beta diversity within fragments) to be assessed (Condit et al. 2002). In this context, the aim of this study was to apply the additive partitioning of diversity (as proposed by Laurance et al. 2007), in semideciduous forest fragments, by quantifying the proportion of the diversity that resulted from nesting and turnover processes on a tree assemblage (Baselga 2010). More specifically, we asked the following questions: (a) Is the pattern of additive partitioning of diversity of the tree assemblage similar between the understory and canopy strata?; (b) Did the tree assemblage pattern in the canopy and understory strata result from species nesting or replacement turnover?; (c) Are landscape characteristics responsible for diversity partitioning of the tree assemblage?

The studied area has low remaining forest coverage (ca. 9%), caused by deforestation that started in the 18th century, and forest fragments surrounded by pasture (~50%), coffee and sugar cane (~20%) matrices (see Methods). Within this landscape context, we hypothesized that the tree assemblage (both canopy and understory strata) in the studied fragments would be homogeneous, as a consequence of a nested subset (Baselga 2010, Arroyo-Rodríguez et al. 2013, Hernández-Ruedas et al. 2014). We expected this because areas with low forest cover in very disturbed landscapes (Laurence et al. 2007, Arroyo-Rodríguez et al. 2013) are more likely to suffer from floristic homogenization. In contrast, landscapes with high forest cover, low urbanization (Hernández-Ruedas et al. 2014) and little management (Santos et al. 2006) are able to maintain high diversity.

Methods

Study area

The study wascarried out in ninefragments in Alfenas, Minas Gerais, Brazil (21º25'45"S, 45º56'50"W)(Table 1). We choose to work with only one landscape because studiesreport that plant species richness is more strongly affected by local management and field margin composition and structure than by the landscape (Gabriel et al. 2006, Aavik et al. 2008, Marshall 2009). The distance among fragments ranged from 3.1 to49.6 km. These fragmentspreserveremnants of submontane semideciduous Atlantic Forest. The most dominant families in these fragments are Fabaceae, Myrtaceae, Lauraceae, Meliaceae and Euphorbiaceae. The most dominant species are *Copaifera langsdorffii*, *Ocotea odorífera*, *Cryptocarya aschersoniana*, *Metrodorea stipularis* and *Miconia willdenowii*.

The climate is classified asCwa(with a dry winter and temperate summer), the average temperature is 16.9 ºC in the winter and 21.5 ºC in the summer, and the average precipitation is 26 mm in the winter and 290 mm in the summer (Alvares et al. 2014). The region haselevationsranging from 720 to 1350m, and apredominantlyundulatingreliefassociated withhills andmountains.The region has only about 9% native forest habitat and the most common matrix types are pastures (51%), and permanent (mainly coffee - 17%) and temporary (mainly sugar cane and corn - 7%) crops (Olivetti 2015). The region excelled as a coffee producer in the XIX and XX centuries. However, it was used for traditional agropastoral practices in the XVIII and early XIX centuries (Castilho 2009).

Sample design

Ten plots of 10 x 20 m (0.2 ha) wererandomly installed in each fragment studied. We sampled and measured, within each plot, all individuals taller than 1.3 m. We divided the individuals into the following two classes: canopy individuals, those with a diameter of breast height (DBH) ≥ 5 cm; and understory individuals, those with a DBH ≤ 5 cm. We knew that there were many young trees in the understory category that would reach the forest canopy when mature, as well as that there were many tall mature shrubs in the canopy category. However, we named these two categories in order to facilitate the understanding of the readers.

Landscape composition and community attributes

We selected eight fragments and made a buffer of 1000 meters around each fragment. We selected this buffer size because we believe that the landscape configuration, such as matrix type, affects the edge effect (the transition between fragments and anthropogenic matrix), and also affects animal pollinators and seed dispersers over larger distances, which can influence mutualistic relationships between plants and animals and consequently community attributes. Within each buffer we calculated, using ArcGIS (Version 10.0, ESRI 2014), the percentage of forest cover (COVER) and percentage of different types of matrices (MATRIX; sugarcane, coffee and pasture) in a buffer of 1000 m around each fragment. COVER is the proportion of landscape covered by forest, which is calculated using $Af = \pi . r^2 * PF$, where Af is the total area of forest (forest), πr^2 is the area of the buffer circumference and *PF* is the total proportion of forest in each buffer (1000 m). MATRIX (Landscape index) quantifies the landscape proportion inside the buffer covered by each habitat matrix (coffee, pasture and sugarcane), which is calculated using the equation $Am = \pi .\, r^2 * PM$, where *Am* is the sum of anthropogenic habitat matrix areas and *PM* is the total proportion of habitat matrix in each buffer.

Statistical analysis

Taxonomic richness, bootstrap estimator, the Shannon-Wiener index and Pielou's evenness was used to evaluate the community diversity (Magurran 2004). Diversityprofiles were used to evaluate and comparethe diversity of speciesamong fragments (Tóthmérész 1995). Each diversity index differs in the weight given to rare species and by using the diversity profiles it is possible to observe whether the difference in diversity among fragments is due to rare or abundant species (Melo 2008). Diversityprofiles were calculated with Hill'sSeriesin the softwarePAST (Hammer and Harper 2014). In this serieswhenalpha=0, Hill's number=number of species(S), whenalpha=1, Hill's number=exp (H') and whenalpha=2, Hill"s number=inverseofSimpson'sDominanceIndex (1/D).

Theadditive partitioningof species diversity was used in order to verify how the variation of diversity occurs at different scales(i) among plots within the same fragment and (ii) among different fragments(Crist etal. 2003,Andersonetal. 2011). In theadditive partitioningof species diversity(here calculated byspecies richness), the regional ortotal diversity (γ) isdivided intoαandβcomponentsthatare expressedin the same unit. The totalspecies richness(γ) of a set of samplesis divided at therichnessaverage into the samples (α) andrichnessaverage absent ineachsample (β). That way, $\gamma = \alpha + \beta 1 + \beta 2 + ...\beta n$, where n=number oflevels, andβn=αn+1-αn(Crist etal. 2003). The average diversity in the plots (level 1) and in the fragments (level 2) were calculated as a component oftotal diversity, in order to verifythe spatial variation ofdiversity. Thisobserveddiversity was comparedwith a nulldistribution modelof diversityconsistingof the meandiversityobtained through1000randomizations. The1000randomizationsensuresthe expecteddiversityof the distributionof individualsisrandom. Theexpecteddiversity wascalculatedwith the software Partition(Crist et al. 2003). We used the additive approach because, unlike the multiplicative, the components of diversity remain in the same sample unit independent of the scale. Additionally, it is possible to calculate the relative contribution of alpha and beta diversity for gamma diversity at different scales (Lande 1996, Crist et al. 2003).

Presence/absence matrices and the Sorensen dissimilarity index were used for the decomposition of beta diversity in nestedness and turnover components for two scales (plots and fragments). This index ranges from 0 (completely similar) to 1 (completely dissimilar). Subsequently, we partitioned the total dissimilarities (B_{SOR}) in the proportions generated by nestedness (B_{NES}) and turnover (B_{SIM}), where $B_{SOR} = B_{NES} + B_{SIM}$ (Baselga 2010). These analyses were carried out in R (R Development Core Team 2007) using the functions “beta-multi.R” in package “betaparte.” A recent study (Baselga and Leprieur 2015) showed that with this method the nestedness-resultant component accounts only for richness differences derived from nested patterns, while in other methods richness difference dissimilarity accounts for all kinds of richness differences. Moreover, in the method proposed by Baselga (2010) the replacement component is independent of richness difference.

A partial redundancy analysis (PRDA) (Rao 1964) was performed in order to verify whether the landscape configurations, COVER and MATRIX, are able to explain the variation in the beta diversity among fragments. For this analysis we used two data matrices, a matrix with the number of individuals per species in each fragment and a matrix with the landscape parameters. The calculation of how landscape parameters selected are able to explain the variation in the beta diversity among the fragments was carried out according to

Legendre et al. (2005) using the vegan package (Oksanen et al. 2007) available in the software R (R Development Core Team 2007). The significance of the percentage of explanation was evaluated according to the permutation method described by Legendre and Legendre (1998), performing 9999 permutations.

Results

Canopy stratum

We sampled 1985 individuals with a DBH ≥ 5 cm, distributed in 205 species, 115 genera and 48 families, in the nine studied fragments. Although some fragments have twice the species than others (richness estimated by bootstrap ranged from 46.5 to 105.3), all fragments had high species diversity(Shannon's diversity index (H') ranged from 3.17 to 4.03 nats/ind), high evenness (Pielou's evenness index [J'] ranged from 0.82 to 0.91) and a large percentage of rare species (one individual in the sample) (39.7% to 54.5%) (Table 2). The diversity profilesshowedthat richerfragmentswerenot alwaysthe most diverse. In other words, the weight ofrare speciescandetermine diversity ecological valuesin each site. For example, fragment6 hadthe lowestnumber ofspecies; however, by reducing theweight ofrare species(alpha > 1), this remnantbecomesmore diversethan fragments2, 3, 4 and 5 (richest fragments) (Fig.1a). The total species richness based on Bootstrap estimator in the nine sites together was 239 and Shannon's diversity index (4.38 nats/ind) and Pielou's evenness index (0.82) remained high (Table 2). The frequent species, those occurring in all fragments, represent 1.9% of the total richness and 13.5% of thetotal individuals sampled. On the other hand, 40.5% (83) of the species and 7.7% of the total individuals occurin onlyone fragment.

The beta diversity from this stratum among fragments was higher than expected if the distribution was random (p=0.001), representing 70% of the regional diversity (gamma) (Fig. 2). Additionally, 92% of the beta diversity among fragments is generated by species turnover and only 8% by nestedness. Similarly, beta diversity among plots within the fragments was also higher than expected by chance (p=0.001), except for fragment nine, representing approximately 79% of diversity in all fragments. Approximately 95% of the beta diversity within all fragments is due to species turnover and about 5% is due to nestedness. None of landscape parameters analyzed for this stratum explained the variation in beta diversity ($F_{4,3}$=0.91; p=0.67).

Understory stratum

We sampled 6353 individuals with a DBH < 5 cm, distributed in 272 species, 145 genera and 58 families, in the nine studied fragments. All fragments have high richness (richness estimated by bootstrap ranged from 96.8 to 147.3), high diversity(Shannon's diversity index [H'] ranged from 3.18 to 4.15 nats/ind), high evenness (Pielou's evenness index [J'] ranged from 0.67 to 0.85) and the percentage of rare species did not exceed 50% in each of the fragments (Table 2). However, the weight ofrarespeciesdetermined the diversity in the fragments, except for fragment 9 that is the richest and the most diverse and did not change when we altered the alpha ecological value. For example, fragment 2has morespecies than fragments3 and 7,but whenwe reduced theweight of therare species the latter became morediverse (Fig.1b).

The total species richness based on bootstrap estimator, in the nine sites remained high (304.5), as well as Shannon's diversity index (4.26 nats/ind) and Pielou's evenness index (0.76) (Table 02). Frequent species, those occurring in all fragments, represent 3.7% of the total richness and 33% of thetotal individuals sampled. On the other hand, speciesthat occurin onlyone fragmentrepresent34.2% (93) of the total species and 2.4% of the individuals.

The beta diversity from this stratum among fragments was higher than expected if the distribution was random (p = 0.001), representing 77% of the regional diversity (gamma)

(Fig.3). Ninety percent of the beta diversity among fragments is due to species turnover and only 10% to nestedness. In turn, beta diversity within the fragments was, by chance, also higher than expected ($p = 0.001$) and represents approximately 76% of the gamma diversity in all fragments. Ninety-three percent of the beta diversity within all fragments is due to species turnover and 7% is due to nestedness. None of the landscape parameters analyzed for this stratum were able to explain the variation in beta diversity ($F_{4,3}=1.4$; $p=0.20$).

Discussion

In the present study we found that the high beta diversity within and among fragments in both strata was due to species turnover (avoiding species homogenization). The tested landscape characteristics did not influence the diversity partition of the tree assemblage of both strata. The partition of the diversity of the tree assemblage from the understory and canopy strata confirms the good conservation condition of the fragmented landscape studied, and suggests that these fragments are able to maintain diversity over time. Most of the trees studies were not included in the canopy stratum (the only stratum studied in most other works); however, they are important to consider (Salles and Schiavini 2007) because these individuals are more sensitive (smaller, thinner and younger individuals) and may have been recruited after the fragmentation process. The results for the understory stratum to recent disturbances are more accurate than the canopy stratum, since adult trees have long life cycles and could have been established before the fragmentation process, while most of the life cycle stages of the plants from the understory stratum happened during or after fragmentation (Martinez-Ramos and Alvarez-Buylla 1998). In this way it is possible to predict the change in the community structure over time.

It was expected that smaller and isolated fragments, inserted in a landscape with low forest coverage and in anthropogenic matrices, would be in a process of floristic homogenization. In other words, the tree assemblage would be slowly dominated by tolerant species and missing more sensitive species (Laurance et al. 2007, Arroyo-Rodríguez et al. 2013). However, despite the studied landscape showing all these characteristics, the tree assemblage in the fragments is not in a homogenization process. On the contrary, the tree assemblage has high beta and alpha diversity while the diversity profile shows that the diversity is mainly due to a large number of rare species. In addition, a previous study in the same area showed that these fragments still preserve high species diversity and richness and endangered species. We hypothesize that these unexpected results were from two main factors: the fragment heterogeneity and the landscape context where the fragments occur.

Local beta diversity patterns can be attributed to niche structure, biological interactions and environmental characteristics (Arroyo-Rodríguez et al. 2013). More heterogeneous environments present more resources and conditions, which would result in a greater number of niches, supporting greater diversity of species than simpler ones (MacArthur and MacArthur 1961). Forest fragments, even small ones, can present different internal characteristics, such as the presence or absence (and/or quantity) of streams, gaps and/or slopes, which create different micro-environments that could be occupied by different species, increasing the diversity (Chávez and Macdonald 2012, Sabatini et al. 2014). In fact, the studied fragments presented different characteristics among them, such as presence or absence of streams, slope degrees and different anthropogenic matrix types, which can generate intrinsic heterogeneity (*personal observation*). Another aspect of the study, reinforcing this hypothesis, is that high beta diversity within the fragments is mainly due to species turnover and not to richness nestedness, contrary to expectations generated by Laurance et al. (2007). One of the main factors responsible for species turnover is a heterogeneous environment (Wright et al. 1998). This result indicates that the differences in species composition among plots and among fragments are generated by different abiotic and biotic conditions within fragments or by different disturbance histories (Gaston et al. 2007, Baeten et al. 2012), and not by ordered loss of species. A study in a temperate forest reserve in Europe (Baeten et al. 2012) found similar results to the present study, and attributed the high species turnover to heterogeneity of environmental conditions within the reserve. Our results show that even forest fragments located in human modified landscapes can preserve some heterogeneous environmental conditions.

The second factor that could explain the floristic differentiation and high species turnover is the landscape context. Although we did not find a relationship between diversity partition and forest or anthropogenic matrix (pasture, sugar cane and coffee) coverage, other landscape characteristics (not tested by us) could influence the relationship between diversity partition. We suggest future studies test this relationship in fragmented landscapes with species turnover and other landscape characteristics. For example, the studied fragments are inserted in a more familiar agricultural system that, although commercial, is small-scale with little mechanization. Additionally, more than one crop type is produced in the landscape, mainly coffee and sugar cane, and there are pastures with some isolated trees (Gonçalves 2015, Olivetti 2015, Vergne 2015). Studies show that in landscapes where there are trees in pastures (Lindborg et al. 2014) and traditional agriculture (without mechanization) (Karp et al. 2012) the diversity and the plant species turnover is greater. Moreover, an attenuated edge effect could occur in landscapes with more heterogeneous and less managed matrices (Mesquita et al. 1999). Edge effect is one of the major factors leading to loss of biodiversity in forest fragments (Laurance and Peres 2006 for review). When this effect is softened, it is expected that diversity could be maintained or that species loss would occur slowly, avoiding the fragment homogenization (Hernández-Ruedas et al. 2014). The anthropogenic matrixheterogeneity in the present study could also contribute to the tree species turnover within and among areas (Fahrig et al. 2011). Another consideration based on our results is that some trees could have been recruited before and others after the fragmentation process, which would leave a confounding factor in the statistical analysis because only the trees recruited after fragmentation would have been influenced by the forest and matrix cover. This characteristic could explain the lack of a relationship between the tested landscape characteristics and the tree assemblage process. Tree assemblages have a delay in response to the fragmentation processes because they are long living (Helm et al. 2006, Vellend et al. 2006). Therefore, we suggest that future studies test if the turnover in landscapes with forest fragmentation is due to our two hypotheses explained above: (i) the fragment heterogeneity and (ii) the landscape context where the fragments occur.

Studies about beta diversity are relatively recent. The first work to verify beta diversity in tropical forests trees was published in 2002 (Condit et al. 2002). Beta diversity is more important than alpha diversity for biodiversity conservation, because it considers not only the diversity within a fragment, but its variation in the region (Condit et al. 2002). This work is one of the first to investigate the homogenization process at local scales for tropical forest fragments, and decompose the beta diversity in turnover and nestedness components. This decomposition is important because it has valuable implications for the conservation of diversity. When the fragments have a nested structure, the conservation of the richest fragment preserves all biodiversity in the region, since the other fragments are only a subset of the most diverse fragment. However, when there is large species turnover, as in the fragments studied, biodiversity conservation depends on the conservation of the largest possible number of fragments; since each fragment has a unique species composition, the loss of a fragment leads to the local extinction of exclusive species (Wright and Reeves 1992, Baselga 2010).

This study showsthat evenina modifiedhumanlandscape,thecanopyandunderstory plant communitiesare not in ahomogenizationprocess.In addition, we show for the first timethat betadiversity oftropicalforestfragmentsis due toturnoverand nottonesting. These results, in addition to practical implications, can help to: (a) understand the process of homogenization in tropical forest fragments; (b) show the importance of intrinsic heterogeneity and a landscape context for maintaining high beta diversity within the fragments; and (c) understand the processes responsible for beta diversity in fragmented environments. Further, because of the implications for conservation, we suggest that the additive partitioning of diversity and nested subsets calculations are incorporated into management and conservation plans. In addition, we especially suggestthat fragmentsand landscapeheterogeneitybe preservedin orderto maintain regional diversity.

Acknowledgements

We thank Derek Felipe , Erlon Pimenta, André Carvalho, Marcela Alcantara e Bruno Roberto Ribeiro for help with the fieldwork, Hisaias Almeida for identifying the species and Nathan Smith for correcting the English. This research was financially supported by a grant from the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) awarded to Magda Silva Carneiro. The project was also supported by the Fundação de Amparo a Pesquisa do Estado de Minas Gerais (FAPEMIG)/Vale Company (RDP-00104-10) and the Conselho Nacional de Desenvolvimento Cientifico e Tecnológico (CNPq) (472250/2010-8 and 306595/2014-1).We also thank Fundação de Amparo a Pesquisa do Estado de Minas Gerais (FAPEMIG) for financial assistance to pay the publication fee.

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Anexos

Table 01: Information (popular name, area, location) analyzed from nine seasonal semideciduous forest fragments in southeastern Brazil.

Fragment	Popular name	Area (ha)	COVER(%)†	Location
1	Matão	20,91	15.04	S 21°30'16.8" W 045°52'38.5"
2	Gaspar Lopes	81,55	26.43	S 21°22'43.8" W 045°55'46.7"
3	M	56,05	25.98	S 21°27'24.6" W 046°10'07.1"
4	Paraiso	36,85	11.73	S 21°21'46.5" W 045°50'26.4"
5	I	37,05	13.66	S 21°25'35.1" W 046°05'39.4"
6	Cemiterio	22,99	9.38	S 21°33'34.5" W 045°56'15.8"
7	Porto	87,18	27.75	S 21°25'16.3" W 046°07'22.3"
8	N	24,80	13.39	S 21°28'07.2" W 046°09'46.2"
9	São José	28,57	24.54	S 21°26'02.4"W 046°08'57.6"

†Cover: Percentage of forest cover.

Table 02: Number of families (F), sampled species (S), estimated species by bootstrap (ES), number of rare species (only one individual) (RS), number of individuals (I), Shannon's diversity index (H') and Pielou's evenness index (J') from nine sites of seasonal semideciduous forest in southeastern Brazil.

Fragments	DBH	F	S	ES	RS	I	H'	J'
1 - Porto	≥ 5 cm	24	57	69.77	25 (43.85%)	210	3.54	0.88
	≤ 5 cm	38	103	124.81	46 (44.66%)	848	3.18	0.67
2 – Gaspar Lopes	≥ 5 cm	28	85	105.26	46 (54.12%)	222	4.00	0.90
	≤ 5 cm	45	127	147.29	34 (26.77%)	1154	3.83	0.79
3 - M	≥ 5 cm	24	68	81.62	27 (39.71%)	280	3.61	0.86
	≤ 5 cm	30	102	119.34	31 (30.39%)	658	3.94	0.85
4 – I	≥5 cm	22	56	68.17	23 (41.07%)	257	3.28	0.82
	≤ 5 cm	34	99	117.69	34 (34.34%)	779	3.24	0.71
5 - Paraiso	≥ 5 cm	29	58	70.34	26 (44.82%)	210	3.42	0.84
	≤ 5 cm	32	87	103.15	37 (42.53%)	417	3.75	0.84
6 – São José	≥ 5 cm	18	39	46.54	14 (35.89%)	169	3.17	0.87
	≤ 5 cm	34	83	98.04	31 (37.35%)	612	3.25	0.74
7 – N	≥ 5 cm	22	52	63.58	24 (46.15%)	230	3.17	0.80

	≤ 5 cm	32	93	111.65	40 (43.01%)	517	3.80	0.84
8 - Cemiterio	≥ 5 cm	27	66	82.72	36 (54.54%)	181	3.59	0.86
	≤ 5 cm	31	80	96.85	35 (43.75%)	492	3.27	0.75
9 - Matão	≥ 5 cm	28	83	101.37	40 (48.19%)	226	4.03	0.91
	≤ 5 cm	43	153	183.51	62 (40.52%)	876	4.15	0.82
Total	≥ 5 cm	48	205	239.05	56 (27.31%)	1985	4.38	0.82
	≤ 5 cm	58	272	304.52	62 (22.80%)	6353	4.26	0.76

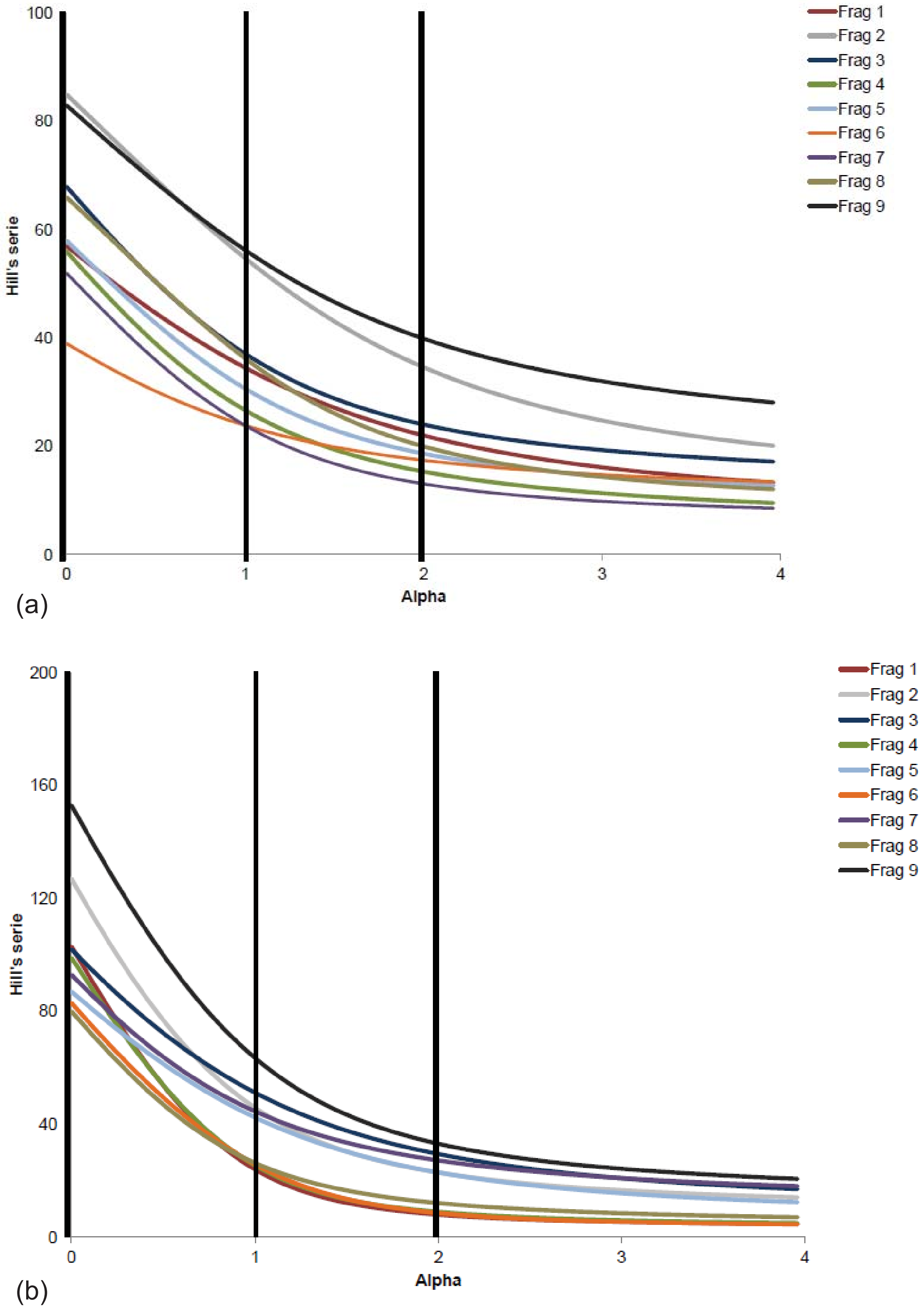

Figure 01: Diversity profiles for the nine fragments using Hill's Series. When the parameter alpha= 0, the diversity ecological value is equal to the number of species in the sample. For alpha tending to 1, the diversity ecological value is equivalent to the

Shannon Index (natural base). For alpha= 2, the ecological value is the same as obtained using the inverse of Simpson Index (1/D). (a) Individuals with DAP $\geq$ 5.0 cm. (b) Individuals with DAP $\leq$ 5.0 cm.

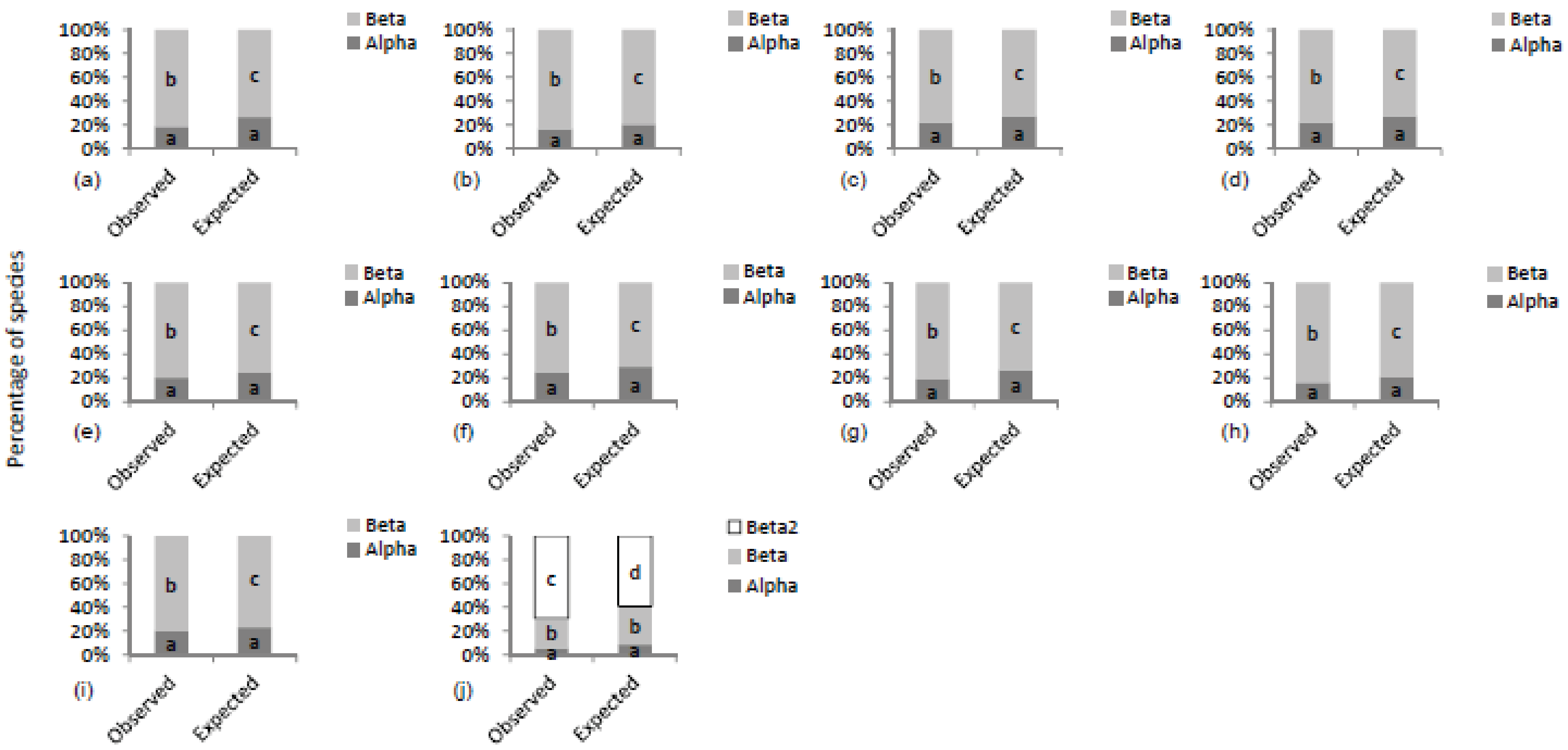

Figure 02: Additive partition of species diversity for species with DAP ≥ 5.0 cm. Alpha is the average richness within the plots. Beta is the average richness absent in the plots. Beta2 is the average richness absent in the fragments. The sum of alpha and beta components results in gamma diversity of each fragment. The sum of alpha, beta and beta2 components results in gamma diversity of the region. Different letters indicate observed ecological values significantly different than the expected ecological value, if the distribution were random. (a) Additive

partition of species diversity in $_{\text{fragment 1 (p}}$alpha$_{=0.99,\text{ p}}$beta=0.001). (b) Additive partition of species diversity in $_{\text{fragment 2 (p}}$alpha$_{=0.99,\text{ p}}$beta=0.001). (c) Additive partition of species diversity in $_{\text{fragment 3 (p}}$alpha$_{=0.99,\text{ p}}$beta=0.001). (d) Additive partition of species diversity in $_{\text{fragment 4 (p}}$alpha$_{=0.99,}$ $_{\text{p}}$beta=0.001). (e) Additive partition of species diversity in $_{\text{fragment 5 (p}}$alpha$_{=0.99,\text{ p}}$beta=0.001). (f) Additive partition of species diversity in $_{\text{fragment 6}}$ $_{\text{(p}}$alpha$_{=0.99,\text{ p}}$beta=0.001). (g) Additive partition of species diversity in $_{\text{fragment 7 (p}}$alpha$_{=0.99,\text{ p}}$beta=0.001). (h) Additive partition of species diversity in $_{\text{fragment 8 (p}}$alpha$_{=0.99,\text{ p}}$beta=0.001). (i) Additive partition of species diversity in $_{\text{fragment 9 (p}}$alpha$_{=0.99,\text{ p}}$beta=0.001). (j) Additive partition of species diversity in region (all fragments) $_{\text{(p}}$alpha$_{=0.99,\text{ p}}$beta$_{=0.99,\text{ p}}$beta2=0.001). Different letters indicate statistical difference.

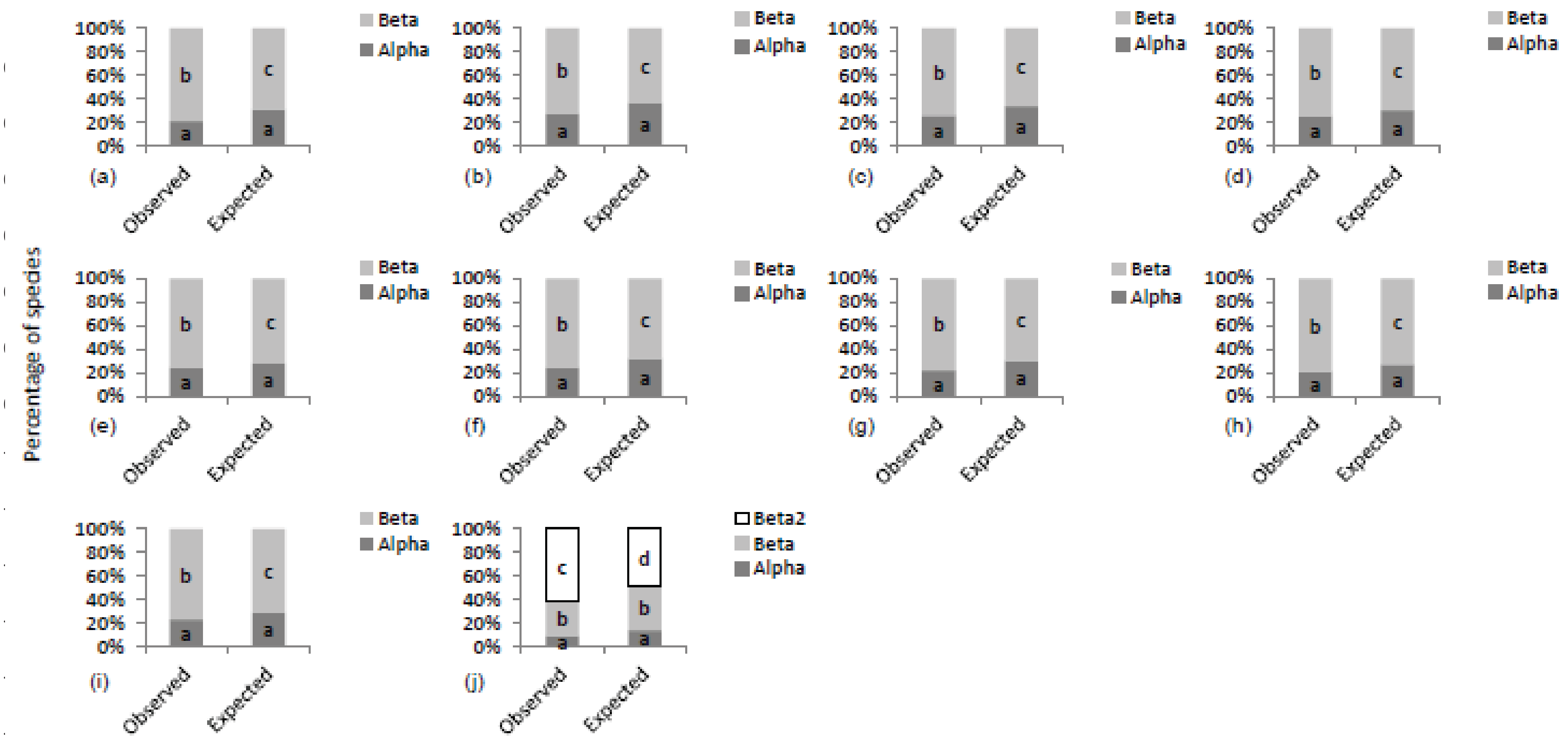

Figure 03: Additive partition of species diversity for species with DAP ≤ 5.0 cm. Alpha is the average richness within the plots. Beta is the average richness absent in the plots. Beta2 is the average richness absent in the fragments. The sum of alpha and beta components results in the gamma diversity of each fragment. The sum of alpha, beta and beta2 components results in the gamma diversity of the region. Different letters

indicate observed ecological values significantly different than the expected ecological value, if the distribution were random. (a) Additive partition of species diversity in fragment 1 (p_{alpha}=0.99, p_{beta}=0.001). (b) Additive partition of species diversity in fragment 2 (p_{alpha}=0.99, p_{beta}=0.001). (c) Additive partition of species diversity in fragment 3 (p_{alpha}=0.99, p_{beta}=0.001). (d) Additive partition of species diversity in fragment 4 (p_{alpha}=0.99, p_{beta}=0.001). (e) Additive partition of species diversity in fragment 5 (p_{alpha}=0.99, p_{beta}=0.001). (f) Additive partition of species diversity in fragment 6 (p_{alpha}=0.99, p_{beta}=0.001). (g) Additive partition of species diversity in fragment 7 (p_{alpha}=0.99, p_{beta}=0.001). (h) Additive partition of species diversity in fragment 8 (p_{alpha}=0.99, p_{beta}=0.001). (i) Additive partition of species diversity in fragment 9 (p_{alpha}=0.99, p_{beta}=0.001). (j) Additive partition of species diversity in region (all fragments) (p_{alpha}=0.99, p_{beta}=0.99, p_{beta2}=0.001). Different letters indicate statistical difference.

CAPÍTULO 03

In a fragmented landscape, the presence of pastures modifythe attributes of regenerating tree community, without leading to species homogenization [1]

[1] **Nos moldes da revista Conservation Biology**

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Article Impact Statement: Pasture matrices affect the sapling community in a fragmented landscape.

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Acknowledgements

We thank F. D., P.E., C.A. and A.M. for help with the field work and A.M. for correcting the English. This research was financially supported by a grant from Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) to M.S.C. The project was supported by Fundação de Amparo a Pesquisa do

Estado de Minas Gerais (FAPEMIG)/Vale Company (RDP-00104-10), as well as Conselho Nacional de Desenvolvimento Cientifico e Tecnológico (CNPq) (472250/2010-8 and 306595/2014-1).

In a fragmented landscape, the presence of pastures modify the attributes of regenerating tree community, without leading to species homogenization

Abstract

As in large parts of the tropics most biodiversity is concentrated in forest fragments, conservation strategies must take into consideration the distribution patterns of organisms in those fragmented landscapes to be able to effectively protect them. It is therefore necessary to understand how landscape configurations affect species diversity and composition within forest fragments, as well as the diversity that exists among different fragments. We investigated how landscape configurations affect community attributes of adult and sapling tree populations, and examined whether those configurations influenced beta diversity inside and among fragments. We expected that landscape parameters would affect all community attributes analyzed. We censused all adult trees (individuals with diameters at breast height (DBH) > 20 cm) and sapling trees (individuals with DBH < 1 cm and height > 1.30 m) in each of ten 10 x 20 m plots (0.2 ha) in each of eight fragments of Submontane Semideciduous Atlantic Forest. The adult and sapling layers showed high beta diversity (80% of gamma diversity), indicating that the fragments were not experiencing species homogenization processes. Comparisons of the assemblages in the two layers demonstrated that regional tree richness and diversity should remain high over time.There are indications, however, that alterations due to fragmentation, especially the presence of pastures, may be altering the composition of the new assemblage (sapling), making the fragments less heterogeneous.

Key-words: Additive partitioning of diversity, beta diversity, forest cover, forest fragments, matrix effect, and regeneration layer.

Introduction

Tropical rainforests are the most diverse ecosystems on our planet, harboring 20% to 40% of all plant and animal species (Myers 1991), however, this diversity is threatened by forest fragmentation (see Laurence et al. 2011 for a review). Although there is a general consensus that fragmentation has negative effects on the composition, diversity, and structure of plant communities (Laurence et al. 2011), these effects may not be identical for all communities. The intensities of the effects of fragmentation depend on the biology of the impacted species, environmental characteristics and landscape characteristics into which the fragments are inserted,such as matrix type and forest cover (Viana & Tabanez 1996). The environment characteristics are determinant for the species establishment, it is known that independent of the fragmentation, soil characteristics, air humidity, relief, altitute and succession stage determine the species composition of the environments (Laurence et al. 2011).

While several studies have investigated the effects of fragmentation on environmental characteristics and consequently on species composition (for example, studies about edge effect), studies on the effect of landscape changes on fragmented communities are still recent and sometimes controversial. In general, landscapes with high forest cover are more similar to natural environments. Anthropogenic matrices, on the other hand, have varying negative effects on fragments. Pesticides and herbicides used in plantations, for example, can reach the interiors of forest fragments, causing the disappearance of pollinating insects (Hester & Hobbs 1992) – which will modify mutualistic relationships and can trigger cascading effects of plant extinctions. Pesticide use can also contaminate the soil and alter biogeochemical cycles (Hester & Hobbs 1992). Matrix structures present another problem (Fahrig 2003, Deconchat et al. 2009). Matrices structurally more similar (height and leaf longevity) to forest fragments (such as shrub matrices like coffee) could soften edge effects and act as fragment extensions, serving as stepping stones and feeding stations for animals (Mesquita et al. 1999, Hernandez-Ruedas et al. 2014). Most herbaceous matrices, however, such as pastures, have the opposite effect on fragments, tend to isolate fragments and prevent animal movements (Mesquita et al. 1999).

As in the Atlantic Forest region most tropical biodiversity is contained within forest fragments (Melo et al. 2013), conservation strategies designed to effectively protect forest remnants must take into account the distribution patterns of organisms within those fragmented landscapes (Fahrig 2003). It is therefore necessary to better understand how landscape configurations affect the diversity and composition of species within the fragments, and examine the diversity between fragments (beta diversity) (Summerville *et al.* 2003). Beta diversity is central to understanding how species diversity is organized and maintained (Condit et al. 2002). The way in which species diversity varies from fragment to fragmentis of relevance to testing ecological theories that seek to understand the processes that generate those patterns, and which have practical applications in conservation biology (Veech *et al.* 2002). While some empirical studies have indicated that landscape configurations have a fundamental role in determining beta diversity (Arroyo Rodriguez et al. 2013, Hernandez-Ruedas et al. 2014), many landscape parameters, including the effects of matrix type, have not been examined; likewise, various parameters that have been tested (such as forest cover and geographical distance effects) have yielded conflicting results (Arroyo Rodriguez et al. 2013, Carneiro et al. 2016).

One problem related to these studies (as well as other studies of fragmentation effects on plant communities) is that most have addressed only the adult tree component (DBH> 5 cm). Rainforest often have very long-lived trees (Martinez-Ramos & Alvarez-Buylla 1998), and mature trees present in fragments may be representative of a time when environmental conditions were quite different (such as before the forest was fragmented). Thus, the effects of environmental alterations due to fragmentation may best be expressed in sapling populations (Hill & Curan 2003). Different from the adult tree component, saplings are young and more vulnerable to anthropogenic impacts (Silva *et al.* 2007), and studies of the sapling component should allow us to make inferences about the current (and future) conservation states of those fragments, and about their ability to preserve biodiversity over time (Silva et al. 2007). Thus, natural regeneration is an indicator of great relevance in evaluating and monitoring recovering areas (Rodrigues & Gandolfi 2004).

The present study therefore investigated the relationship between landscape configurations (percentage of forest cover and different land uses [matrix percentage of coffee, sugarcane, and pasture]) and community attributes (richness, abundance, numbers of rare species, and numbers of abundant species) of adult and sapling trees to determine whether those factors influence beta diversity within and among fragments. We therefore posed the following questions: (i) is the adult tree community similar, in terms of species composition and alpha diversity, to the regenerating tree community?; (ii) Is the additive partitioning of the diversity pattern of the adult community similar to that of the sapling community?; (iii) do matrices structurally similar to the fragment (percentage of coffee matrix) and forest cover have positive effects on beta diversity and on the community attributes of both mature and regenerating trees?; (iv) Do matrices that are structurally different from the fragments (percentages of pasture and sugar cane matrices) have negative effects on beta diversity and on the community attributes of adult and regenerating trees? This study is the first to examine the effects of matrix types on the beta diversities of both adult and regenerating trees, and one of the first to analyze the effects of landscape parameters on regenerating communities. The major contribution of this work is to provide useful information about the factors that control beta diversity in modified landscapes, especially in the regenerating component.

Methods

Study area

The present study was carried out in eight forest fragments in Alfenas, Minas Gerais State, in southeastern Brazil (Table 1). The forest fragments preserve remnants of Submontane Semideciduous Atlantic Forest. The most dominant families and species are, respectively:Fabaceae, Myrtaceae, Lauraceae, Meliaceae, and Euphorbiaceae and *Copaifera langsdorffii*, *Ocotea odorifera*, *Cryptocarya aschersoniana*, *Metrodorea stipularis* and *Miconia willdenowii* (Carneiro et al.unpublished data). The average distances between fragments are 28.1 ± 14.8 km. The regional climate is classified as Cwb with average winter temperature is 16.9 ºC, and 21.5 ºC in the summer. The average monthly precipitation is 26 mm in the winter and 290 mm in the summer (1500 total mm annually) (Alvares et al. 2014). The relief is predominantly mountainous with elevations range from 720 to 1350 m a.s.l.The region retains only approximately 9% of its former native forest habitat; the most common matrix types now are pastures (51%), coffee(17%) and sugar cane (7%) (Olivetti 2015).

Sample design

Fragment selection was carried out by classifying remnant forest areas by digital processing of CBERS-2B satellite images captured in 2009 (20 m resolution). The fragments were selected based on: (i) having similar degradation levels (through classifications of texture, shape, and color aspects); (ii) fragment size (between 15 and 100 ha); and, (iii) distance between fragments (exceeding 4 km, to ensure independence of the data). Ten 10 x 20 m plots (0.2 ha) were installed in each of the eight study fragments (total = 2.0 ha). We recorded and measured all individuals taller than 1.30 m within each plot. For statistical analyses, we divided the individuals into two classes: adult trees (individuals with diameters at breast height (DBH) > 20 cm) and saplings (individuals with DBH < 1 cm and height > 1 m). We selected individuals > 1 m tall because thesewould have passed the critical period of seedling mortality and survived the strong selective post-establishment environment filters (Sales & Schiavini 2007). We sampled only saplings as they would later constitute the adult component. Individuals of species that had already been recorded in the adult component and/or would potentially grow to 10 m in height (Vidal & Vidal 2000) were counted as saplings. The potential heights of the species were verified in the NeoTropTree database (Oliveira- Filho 2014). Botanical identifications were made in the field; when this was not possible, specimens were collected for posterior identification by specialists. The Angiosperm Phylogeny Group III (APG IV 2014) system was used for species identifications and classification.

Landscape composition and community attributes

A a buffer of 1,000 mfrom the center of each fragment was considered and the percent cover of landscape components within the buffer area were calculated using ArcGIS (Version 10.0, ESRI) and Fragstats software (McGarigal & Marks 1995). The proportion of forest cover was calculated using the formula: $Af = (\pi * r^2 * PF)/100$, where Af is the total area of the forest (forest), πr^2 is the area of the buffer around the central fragment, and PF is the total proportion of forest within radius r of each buffer. We used the equation: $Am = (\pi r^2 * PM)/100$, where Am is the sum of anthropogenic habitat matrix areas, and PM is the total proportion of habitat matrix within each buffer for calculate the proportions of the buffer area covered by each of coffee, pasture, and sugarcane. All landscape metrics were log10 transformed.

The community attributes were: species richness, abundance of individuals, number of rare species (singleton species), and evenness.

Statistical analyses

The Bray-Curtis Similarity Index was used to examine the floristic similarity between the adult and sapling layers (Magurran 2004); a PERMANOVA test was used to determine if the floristic compositions of the two layers differed.These analyses were performed using PC-ORD software. We used diversity profiles to evaluate and compare the diversity of species in the two layers and to determine if the differences between them were due to rare or abundant species (Tóthmérész 1995). We calculated diversity profiles using Hill’s Series in PAST software (Hammer & Harper 2014). In this series, when alpha=0, Hill’s number=number of species (S); when alpha=1, Hill’s number= exponential (H’); and when alpha=2, Hill's number= the inverse of the Simpson Dominance Index (1/D).

The additive partitioning of species diversity was used to determine how diversity variations occured at different scales (plots and fragments) for both layers (Crist *et al.* 2003, Anderson *et al.* 2011). In the additive partitioning of species diversity, the total species richness (γ) of a set of samples is divided by the average richness of the samples (α) and by the average richness absent in each sample (β). In that way, $\gamma = \alpha + \beta 1 + \beta 2 + \ldots \beta n$, where n = number of levels, and $\beta n = \alpha n + 1 - \alpha n$ (Crist *et al* 2003). The average diversities of the plots (level 1) and fragments (level 2) were calculated as components of total diversity in order to determine the spatial variations of diversity. This observed diversity was compared to a null diversity distribution model consisting of the mean diversity obtained through 1000 randomizations. These 1000 randomizations create the expected diversity if the distributions of individuals are random. The expected diversity was calculated using Partition software (Crist et al. 2003).

We performed a Partial Redundancy Analysis (PRDA) (Rao 1964) in order to determine whether landscape configurations (COVER and MATRIX) and geographical distances between fragments were able to explain the variations in the beta diversities of each layer among the different fragments. This analysis allowed the removal of the influence of a set of explanatory variables (for example, geographical distance) before evaluating a second set (for example, landscape configuration), that is, we were able toquantify this way the influence of landscape configuration independently of geographical distance, and vice versa (Legendre et al. 2005). For this analysis, we used three data matrices: the matrix with the numbers of individuals per species in each fragment, the matrix with the geographic coordinates of each fragment, and the matrix of landscape parameters. The matrix of landscape parameters was subjected to a selection of variables (stepwise CCA) to optimize the correlation between a subset of landscape variables and the matrix of individuals. The calculations of how landscape parameters were selected by stepwise CCA and how geographical distance can explain the variations in beta diversity between the fragments was performed according to Legendre et al. (2005) using the ‘vegan’ package (Oksanen et al. 2007) in the R environment (R Development Core Team 2007 version 3.1.3). The significance of the percent of variance explaned was evaluated according to the permutation method described by Legendre & Legendre (1998), performing 9999 permutations.

Generalized Linear Models (GLM) were used to examine the effects of each landscape parameter (percentage forest cover and the percentage of each matrix) on community attributes (species richness, abundance of individuals, numbers of rare species, and numbers of abundant species). We ran the GLM with a log function and Poisson distribution to model the response data. To assess the performance of the models, we employed the Akaike Information Criterion corrected for small sample sizes (AICc, Burnham and Anderson 2001).

The best model was based on the lowest AICc value (Burnham and Anderson 2001). We also considered other indices to evaluate model support, including the difference between model AICc and the minimum AICc (ΔAICc), where models with ΔAICc < 2 are considered as having substantial support, and Akaike weights (wAIC), which describe the strength of the evidence for the model. This can be interpreted as the probability that the model is the best model among the set of models being considered (Burnham and Anderson 2001). All models were validated by tests that verified the normality and independence of the residuals

Results

Composition, diversity, and community similarities

We sampled 335 individual adult trees (104 species and 30 families) (DBH ≥ 20 cm) in the eight fragments. The richest families were Fabaceae (16%), Lauraceae (10%), Myrtaceae (10%), and Meliaceae (8%). The most abundant and frequent species were *Copaifera langsdorffii, Cryptocarya aschersoniana, Miconia willdenowii,* and *Ocotea odorifera*. Sixty percent of the most abundant species were classified as climax species (Swaine & Whitmore 1988 – shade tolerance for seed germination and seedling growth).

A total of 450 individuals (93 species and 29 families) were sampled in the regenerating layer (DAP ≤ 1 cm, H ≥ 1.5 m). The richest families were Fabaceae (17%), Myrtaceae (12%), Meliaceae (6%), Lauraceae (6%), and Euphorbiaceae (6%). The most abundant and frequent species were *Cupania zanthoxyloides, Trichilia emarginata, Trichilia pallida,* and *Eugenia acutata*

(Table 2). Ninety percent of the most abundant species were classified as climax species.

The two layers had the same estimated richness, although the diversity of the regenerating species was greater than adult tree diversity, independent of the diversity index used (Fig.1). Individual species were better distributed, that is, there was no dominance of any species, in the sapling layer, while adult trees showed considerable species dominance. The 10 most abundant adult tree species represented 50.4% of all individuals, and 50% of the species were represented by only a single individual. In the regenerating layer, the 10 most abundant species comprised 39% of all individuals, with only 8% of species being rare (one individual).

Although the two layers had 61 species in common (58.6% of the adult trees and 59.2% of the regenerating trees – considering all eight fragments), species similarity between the layers was low (39%) and two well-defined groups of species were identified: adults and sapling trees ($F = 2.22$, $p = 0.002$) (Fig.2). When the fragments were analyzed separately, the floristic differences between the two layers was even more evident: with more than 70% of the species present in the fragments being exclusive to just one layer, with similarities ranging from 16 to 39% (Fig. 2). Furthermore, the proportions of the adult species occurring in the regenerating layer were also low – not exceeding 50% in any of the fragments (Fig. 3).

This high number of exclusive species was reflected in beta diversity. The beta diversity within fragments was higher than expected by chance ($p < 0.005$), representing more than 80% of diversity within the fragments for both layers (Fig. 4 and 5). The beta diversity between fragments was also high (and higher than expected by chance), but dissimilarity in the adult layer was 80% of gamma diversity, (Fig. 4) being greater than the 68% gamma diversity in the sapling layer (Fig. 5).

Effects of landscape parameters on community attributes

Analyses of the variations of the partitioning of beta 2 (beta diversity of fragments) diversity in purely spatial, purely environmental, and in the environment structured by space, revealed that the variations in beta diversity 2 of: (i) the sapling layer was related to the percentage of pasture in the matrix (19.75%) ($F_{1,6}$ = 1.48, p = 0.03), but, (ii) the adult layer it was not explained by any of the landscape parameters. This analysis also revealed that the distance between fragments did not affect beta 2 diversity either among adult trees ($F_{2,5}$ = 1.19, p = 0.25) or among saplings ($F_{2,5}$ = 1.21, p = 0.14).

Richness, abundance, and number of abundant species in the regenerating communitywere positively related with the percent cover of pasture in the matrix (Table 3). None of the attributes of the adult communitywere affected by any landscape parameter.

Discussion

Assemblage comparisons between the two layers demonstrated that trees richness and diversity in the fragments will probably persist for a long time, since the sapling (future) community has high diversity and richness of species. There are indications, however, that alterations due to landscape fragmentation, especially with the presence of pastures, may be altering the composition of the new fragment assemblies. The two layers showed high beta diversity both within and among fragments, indicating that these fragments did not appear to have experienced species homogenization. However, canopy and sapling layers were very dissimilar, in terms of their floristic compositions. The regenerating communities in the fragments, although more diverse and more equitable than the adult community, showed greater similarities in their compositions (lower beta diversity). Additionally, beta diversity, total richness and abundance of species in the sapling layer were positively affected by the percentage of pasture in the surrounding matrix, while the number of rare species in this layer was negatively related topasture cover.

The adult layer presented beta diversity both within and between fragments was very high, meaning that the fragments were heterogeneous and maintained high species diversity levels. Our analyses also showed that most of the abundant species were climax species, contrary the expectation that these environments would be dominated by pioneers species (Laurence et al. 2007). Some authors have suggested that landscape configurations could help maintain tree diversity in fragmented environments (Carneiro et al. 2016, Arroyo Rodriguez et al. 2014). High forest cover, for example, in addition to less inospitable matrices, could lessen edge effect and facilitate dispersal, thus helping to maintain species richness in forest remnants (Mesquita et al. 1999, Hernandez-Ruedas et al. 2014). In this study, however, the high observed beta diversity could not be attributed to forest cover and matrix type, as none of these parameters was related to adult community attributes. We believe that this occurbecause the adult trees, having long life spans, were already present before fragmentation occurred, and therefore were not affected by the current landscape configurations.However we can not test this hypothesis, since we do not have data offragments before the frgamentation process

We believe that fragmentation processes affect adult trees individually, with greater impacts on their growth and reproduction. Two facts reinforce this hypothesis: first, previous work in the same areas showed that the adult community demonstrated structural traits of recently disturbed communities, such as shorter and thinner individuals (Carneiro et al. unpublished data), although high species diversity was maintained, including some endangered species. Second, the results of the present study showed that the numbers of adult tree species in the sapling community were very low (< 39%) compared to others studies (Prata et al. 2011; Souza et al. 2012; Milhomem et al. 2013), suggesting problems with adult tree species reproduction.

The sapling layer had fewer numbers of dominant and rare species, which resulted in high diversity. The beta diversity among fragments was lower than that found for the adult layer, but was nonetheless very high (70%), suggesting that the flora of this layer is not experiencing homogenization. Furthermore, the most abundant species were also climax species. Contrary to our expectations, the pasture cover in the matrix benefited species diversity in the regenerating layer. The fragments with the highest percentages of pasture matrix (fragments 2 and 8) had the most diverse sapling communities. We expected that greater forest cover and coffee matrices (which are structurally more similar to forests) would benefit the community, while the sugarcane and pasture matrices would be more inhospitable and prejudicial (Mesquita et al. 1999) – because matrices structurally similar to fragments can soften edge effect and create buffer zones that help maintain a milder microclimate

(Hernandez-Ruedas et al. 2014). They can also act as fragment extensions providing food and shelter for wildlife (Hernandez-Ruedas et al. 2014). Pasture, however, represents a harsher herbaceous matrix that could have the opposite effect on forest fragments (Mesquita et al. 1999).

Our hypothesis to account for this unexpected result is that the management of those matrices has more impact on tree communities than simply the matrix structure (or forest cover). Pasture is the matrix with the lowest management intensity among the three land cover types in the matrix analyzed. Large amounts of fertilizer and pesticides are applied to coffee plantations two or three times a year (Carvalho 2006). Studies have shown that these chemical products decrease native plant richness (Stoate et al. 2001, Bengtsson et al. 2005, Gabriel et al. 2006, Liira et al. 2008, Kleijn et al. 2009). Coffee harvesting is also performed mechanically on some properties, and studies have also revealed that mechanized agriculture lowers native plant richness (Flohre et al. 2012, Lindborg et al. 2013). Pesticides are likewise applied in the sugar cane matrix, and fire is used to facilitate harvesting (Lehtone et al. 2009). Fire in the sugarcane matrices could reach the interiors of the fragments and cause significant damage, such as the death or displacement of animals and plants as well as seed bank damage (Klerton et a. 2011). Pastures in the study region, on the other hand, are only perfunctorily managed. There were usually very few animals in the pastures, and many isolated trees distributed throughout them (Vegner 2015, Gonçalves 2015). Additionally, no pesticide applications, machinery, or fires were observed in that matrix type. A study in Sweden showed that less-managed landscapes have high species richness and can maintain species pools at regional scales (Lindborg et al.

2013). Those authors also found that even small landscape traits, such as trees distributed in pastures, can serve as refuges and stepping stones for wildlife, and should therefore be considered functional parts of the fragments. Those scattered trees can contribute to dispersal between fragments, enhance colonization, and reduce local extinction rates, thus mitigating the negative effects of fragmentation (Tscharntke et al. 2005, Kleijn & Baldi 2005, Cousins 2006). Another study developed in a different part of Europe showed that agricultural intensification (mechanization, pesticides use) has reduced not only native plants richness, but also species turnover (Flohre et al. 2012). In other words, it was probably not the pasture itself that benefited the regenerating community, but rather its management regime, or the absence of management in those areas.

This study is one of the first to examine the effects of landscape configurations on sapling communities, and presents numerous conclusions: (a) the future tree community will probably be very different from the current adult community, although contrary to our predictions, it will be more diverse and will have large numbers of climax species, with greater similarities among the fragments and (b) adult community attributes are not good indicators of the effects of landscape changes, with regenerating layer attributes being better measures of the impacts of fragmentation effects.

Our work also has practical implications for conservation. We recommend that management and conservations projects consider, and monitor, sapling layer characteristics – because they respond more visibly to landscape alterations changes and represent the future adult community. Additionally, we suggest reducing or avoiding matrix management (mechanization, pesticides, and the use of fire) near fragments to maintain tree diversity.Despite the numerous contributions, our study did not consider the functional and phylogenetic diversity of the fragments, so we suggest that these approaches be considered in future works.

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Anexos

Table 01: Fragment location and the landscape parameter ecological values analyzed.

Fragment	Location	% COVER	% Coffee	% Sugarcane	% Pasture
1	21°29'13.13"S 46° 5'40.32"W	13.39	15.72	31.39	33.14
2	21°26'14.51"S 46° 8'46.93"W	9.38	51.83	0.00	14.02
3	21°34'42.37"S 45°58' 15.04"W	24.54	16.83	45.13	10.5
4	21°33' 44.68"S 45°56' 12.80"W	13.66	62.87	0.00	13.74
5	21°27'50.38"S 45°54'58.10"W	37.05	0.00	58.33	0.00
6	21°25'25.97"S 46° 5'8.03"W	25.98	21.19	36.96	8.18
7	21°28'16.28"S 46° 7'22.43"W	26.43	1.17	0.00	51.56
8	21°25'27.26"S 46° 9'35.66"W	27.75	0.00	32.51	2.3

Table 02: Number of families (F) and sampled species (S), estimated species by Bootstrap (ES), number of rare species (only one individual) (RS), number of individuals (I), Shannon diversity index (H’), Pielou evenness index (J’), density of individuals per hectare (ind ha^{-1}) (D), richest families (RF), species with greater numbers of individuals (AS), frequent species (FS), exclusive species (ES), common species (CE), and Bray Curtis similarity index between both layers (BC) from eight seasonal semideciduous forest sites in southeastern Brazil.

Sites	**DBH**	**F**	**S**	**I**	**H’**	**J’**	**D**	**RF**	**AS**	**FS**	**ES**	**CE**	**BC**
1 - Porto	≥ 20	9	17	41	2.29	0.81	205.00	Fabaceae Lauraceae Euphorbiaceae	*Copaifera langsdorffii* *Ocotea diospyrifolia* *Croton floribundus* *Siphoneugena densiflora*	*Copaifera langsdorffii* *Ocotea diospyrifolia* *Platypodium elegans* *Tapirira guianensis*	9 (52.94%)	8 (47.09%)	0.18
	< 1	17	29	49	3.15	0.94	272.22	Lauraceae Euphorbiaceae Meliaceae	*Ocotea corymbosa* *Nectandra grandiflora* *Cupania vernalis* *Byrsonima laxiflora* *Casearia sylvestris*	*Nectandra grandiflora* *Ocotea corymbosa* *Cupania vernalis* *Byrsonima laxiflora*	21 (72.41%)	8 (27.59%)	0.18
2 – Gaspar	≥ 20	10	21	29	2.93	0.96	145.00	Fabaceae Lauraceae Sapindaceae	*Copaifera langsdorffii* *Cupania vernalis* *Peltophorum dubium*	*Copaifera langsdorffii* *Cupania vernalis* *Platycyamus regnellii* *Machaerium villosum* *Peltophorum dubium*	16 (76.19%)	5 (23.81%)	0.12

	< 1	18	39	105	3.29	0.90	525.00	Fabaceae Sapindaceae Meliaceae	*Cupania zanthoxyloides* *Trichilia pallida* *Trichilia elegans* *Copaifera langsdorffii* *Mollinedia widgrenii*	*Cupania zanthoxyloides* *Trichilia pallida* *Trichilia elegans* *Copaifera langsdorffii*	34 (87.18%)	5 (12.82%)	0.12
3 - M	≥ 20	12	22	41	2.85	0.92	205.00	Fabaceae Euphorbiaceae Lauraceae Annonaceae Meliaceae	*Metrodorea stipularis* *Croton rugosus* *Croton floribundus* *Inga vera* *Syagrus romanzoffiana*	*Croton rugosus* *Metrodorea stipularis* *Inga vera*	17 (77.27%)	5 (22.73%)	0.21
	< 1	8	17	25	2.73	0.97		Meliaceae Fabaceae Sapindaceae	*Inga marginata* *Metrodorea stipularis*	*Inga marginata* *Trichilia clausseni* *Matayba juglandifolia* *Guarea kunthiana* *Trichilia catigua*	12 (70.59%)	5 (29.41%)	0.21
4 – I	≥ 20	11	19	57	2.46	0.84	285.00	Myrtaceae Lauraceae Fabaceae Annonaceae Salicaceae	*Copaifera langsdorffii* *Miconia willdenowii* *Cryptocarya aschersoniana* *Trichilia emarginata* *Xylopia brasiliensis*	*Copaifera langsdorffii* *Miconia willdenowii*	11 (57.89%)	8 (42.11%)	0.27
	< 1	13	22	41	2.80	0.90	205.00	Lauraceae	*Trichilia emarginata* *Ocotea corymbosa* *Casearia arborea* *Trichilia pallida*	*Trichilia emarginata* *Ocotea corymbosa* *Casearia arborea*	14 (63.64%)	8 (36.36%)	0.27
5 - Paraiso	≥ 20	11	16	44	2.4	0.87	220.00	Fabaceae Myrtaceae Lauraceae Moraceae	*Miconia willdenowii* *Cryptocarya aschersoniana* *Ocotea odorifera* *Ixora brevifolia*	*Miconia willdenowii* *Cryptocarya aschersoniana* *Ocotea odorifera* *Copaifera langsdorffii* *Ixora brevifolia*	15 (93.75%)	1 (6.25%)	0.16
	< 1	12	18	27	2.79	0.97	225.00	Myrtaceae	*Trichilia emarginata*	*Trichilia emarginata*	17	1	0.16

								Meliaceae Lauraceae Salicaceae Sapindaceae	*Metrodorea stipularis*	*Metrodorea stipularis* *Eugenia acutata* *Trichilia pallida* *Casearia decandra*	(94.44%)	(5.55%)	
6 – São José	≥ 20	8	12	30	2.08	0.84	150.00	Lauraceae Myrtaceae	*Copaifera langsdorffii* *Cryptocarya aschersoniana* *Ocotea odorifera* *Miconia willdenowii* *Trichilia emarginata*	*Copaifera langsdorffii* *Ocotea odorifera* *Cryptocarya aschersoniana* *Miconia willdenowii* *Trichilia emarginata*	6 (50.00%)	6 (50.00%)	0.24
	< 1	14	23	52	2.80	0.92	260.00	Myrtaceae Fabaceae	*Miconia willdenowii* *Protium spruceanum* *Cordia sellowiana* *Trichilia emarginata* *Tabebuia roseoalba*	*Protium spruceanum* *Cordia sellowiana*	17 (73.91%)	6 (26.09%)	0.24
7- Cemiterio	≥ 20	19	34	53	3.31	0.94	265.00	Lauraceae Fabaceae Euphorbiaceae Meliaceae	*Metrodorea stipularis* *Mabea fistulifera* *Ocotea odorifera* *Eugenia acutata*	*Metrodorea stipularis* *Mabea fistulifera* *Eugenia acutata* *Cryptocarya aschersoniana* *Miconia pepericarpa*	28 (82.36%)	6 (17.65%)	0.31
	< 1	14	24	44	2.91	0.92	244.44	Myrtaceae Euphorbiaceae Lauraceae Meliaceae Salicaceae	*Metrodorea stipularis* *Xylopia brasiliensis* *Casearia arborea* *Mabea fistulifera*	*Metrodorea stipularis* *Cordia sellowiana* *Trichilia catigua* *Casearia sylvestris*	18 (75.00%)	6 (25.00%)	0.31
8- Matão	≥ 20	15	28	41	3.13	0.94	205.00	Fabaceae Lauraceae Meliaceae Myrtaceae	*Ixora brevifolia* *Copaifera langsdorffii* *Platycyamus regnellii*	*Ixora brevifolia* *Copaifera langsdorffii*	15 (53.57%)	13 (46.43%)	0.21
	< 1	20	46	119	3.38	0.88	595.00	Myrtaceae Meliaceae	*Eugenia acutata* *Trichilia catigua*	*Eugenia acutata* *Cupania zanthoxyloides*	33 (71.74%)	13 (28.26%)	0.21

								Fabaceae Lauraceae Sapindaceae	*Metrodorea stipularis* *Cupania zanthoxyloides* *Trichilia elegans*	*Trichilia catigua*		
Total	≥ 20	30	104	335	3.85	0.84	212.03	Fabaceae Lauraceae Myrtaceae Meliaceae	*Copaifera langsdorffii* *Cryptocarya aschersoniana* *Miconia willdenowii* *Ocotea odorifera* *Metrodorea stipularis*	*Copaifera langsdorffii* *Cryptocarya aschersoniana* *Miconia willdenowii* *Ocotea odorifera*	43 (41.35%)	61 (58.65%)	0.39
	< 1	29	93	452	4.13	0.89	316.44	Fabaceae Myrtaceae Meliaceae Lauraceae Euphorbiaceae	*Eugenia acutata* *Metrodorea stipularis* *Cupania zanthoxyloides* *Trichilia emarginata* *Trichilia pallida*	*Ocotea corymbosa* *Cupania zanthoxyloides* *Trichilia emarginata* *Trichilia pallida* *Eugenia acutata*	42 (40.78%)	61 (59.22%)	0.39

Table 03: General linear models (GLM) predicting relationships between community attributes and landscape parameters. Models clearly worse (Δ AICc> 2.0) than the best model were rejected and not included in the table. K= Number of parameters. Δ AICc= Difference in AIC from one model to one with the lowest AIC ecological value. w= evidence weight.

Community attributes	Model	Estimate	p ecological value	Δ AICc	w
DAP < 1cm					
Abundance	Pasture	0.02	0.001	0.0	1
Richness	Pasture	0.02	0.0004	0.0	0.99
Evenness	Pasture	0.04	0.004	0.0	0.75

Figure 01: Diversity profiles for two layers using Hill's Series. When the parameter alpha= 0, the diversity ecological value is equal to the number of species in the sample. For alpha tending to 1, the diversity ecological value is equivalent to the Shannon Index (natural base). For alpha= 2, the ecological value is the same as obtained using the inverse of Simpson Index (1/D). Gray solid line= Individuals with DAP < 1.0 cm. Black solid line= Individuals with DAP ≥ 20.0 cm. Dotted lines= 95% confidence.

Figure 02: Dendrogram of similarity (Bray-Curtis Index) produced by cluster analysis (UPGMA connection method) of tree species compositions among the eight sites (1-8). (a) Species with DAP > 20.0 cm. (b) Species with DAP < 1.0 cm.

Figure 03: Proportions of adult tree species. Exclusive= Proportion of adult trees occuring only in the superior layer. Common= Proportion of adult trees occuring in both layers (adult and regenerative).

Figure 04: Additive partition of species diversity for trees species with DAP > 20.0 cm. Alpha is the average richness within the plots. Beta is the average richness absent in the plots. Beta2 is the average richness absent in the fragments. The sum of alpha and beta components results in the gamma diversity of each fragment. The sum of the alpha, beta and beta2 components results in the gamma diversity of the region. Different letters indicate observed ecological values significantly different than expected, if the distributions were random. (a) Additive partition of species diversity in fragment 1 (p_{alpha}=0.99,

p_{beta}=0.001). (b) Additive partition of species diversity in fragment 2 (p_{alpha}=0.33, p_{beta}=0.23). (c) Additive partition of species diversity in fragment 3 (p_{alpha}=0.94, p_{beta}=0.01). (d) Additive partition of species diversity in fragment 4 (p_{alpha}=0.93, p_{beta}=0.02). (e) Additive partition of species diversity in fragment 5 (p_{alpha}=0.99, p_{beta}=0.002). (f) Additive partition of species diversity in fragment 6 (p_{alpha}=0.99, p_{beta}=0.001). (g) Additive partition of species diversity in fragment 7 (p_{alpha}=0.99, p_{beta}=0.001). (h) Additive partition of species diversity in fragment 8 (p_{alpha}=0.86, p_{beta}=0.03). (i) Additive partition of species diversity in the region (all fragments) (p_{alpha}=0.99, p_{beta}=0.99, p_{beta2}=0.001). Different letters indicate statistical differences.

Figure 05: Additive partition of species diversity for trees species with DAP > 20.0 cm. Alpha is the average richness within the plots. Beta is the average richness absent in the plots. Beta2 is the average richness absent in the fragments. The sum of the alpha and beta components results in the gamma diversity of each fragment. The sum of the alpha, beta and beta2 components results in the gamma diversity of region. Different letters indicate observed ecological values significantly different than expected, if the distributions were random. (a) Additive partition of species diversity in fragment 1 (p_{alpha}=0.94, p_{beta}=0.01). (b) Additive partition of species diversity in fragment 2 (p_{alpha}=0.99, p_{beta}=0.003). (c) Additive partition of species diversity in fragment 3 (p_{alpha}=0.68, p_{beta}=0.32). (d) Additive partition of species diversity in fragment 4 (p_{alpha}=0.92, p_{beta}=0.01). (e) Additive partition of species diversity in fragment 5 (p_{alpha}=0.48, p_{beta}=0.21). (f) Additive partition of species diversity in fragment 6 (p_{alpha}=0.89, p_{beta}=0.04). (g) Additive partition of species diversity in fragment 7 (p_{alpha}=0.99,

p_{beta}=0.001). (h) Additive partition of species diversity in fragment 8 (p_{alpha}=0.86, p_{beta}=0.03). (i) Additive partition of species diversity in the region (all fragments) (p_{alpha}=0.99, p_{beta}=0.99, p_{beta2}=0.001). Different letters indicate statistical differences.

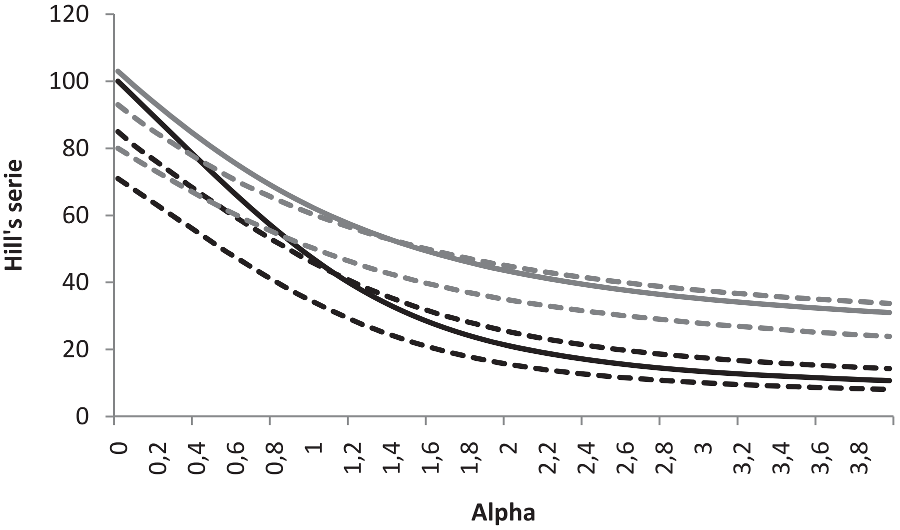

Figure 01: Diversity profiles for two layers using Hill's Series. When the parameter alpha= 0, the diversity ecological value is equal to the number of species in the sample. For alpha tending to 1, the diversity ecological value is equivalent to the Shannon Index (natural base). For alpha= 2, the ecological value is the same as obtained using the inverse of Simpson Index (1/D). Gray solid line= Individuals with DAP < 1.0 cm. Black solid line= Individuals with DAP ≥ 20.0 cm. Dotted lines= 95% confidence.

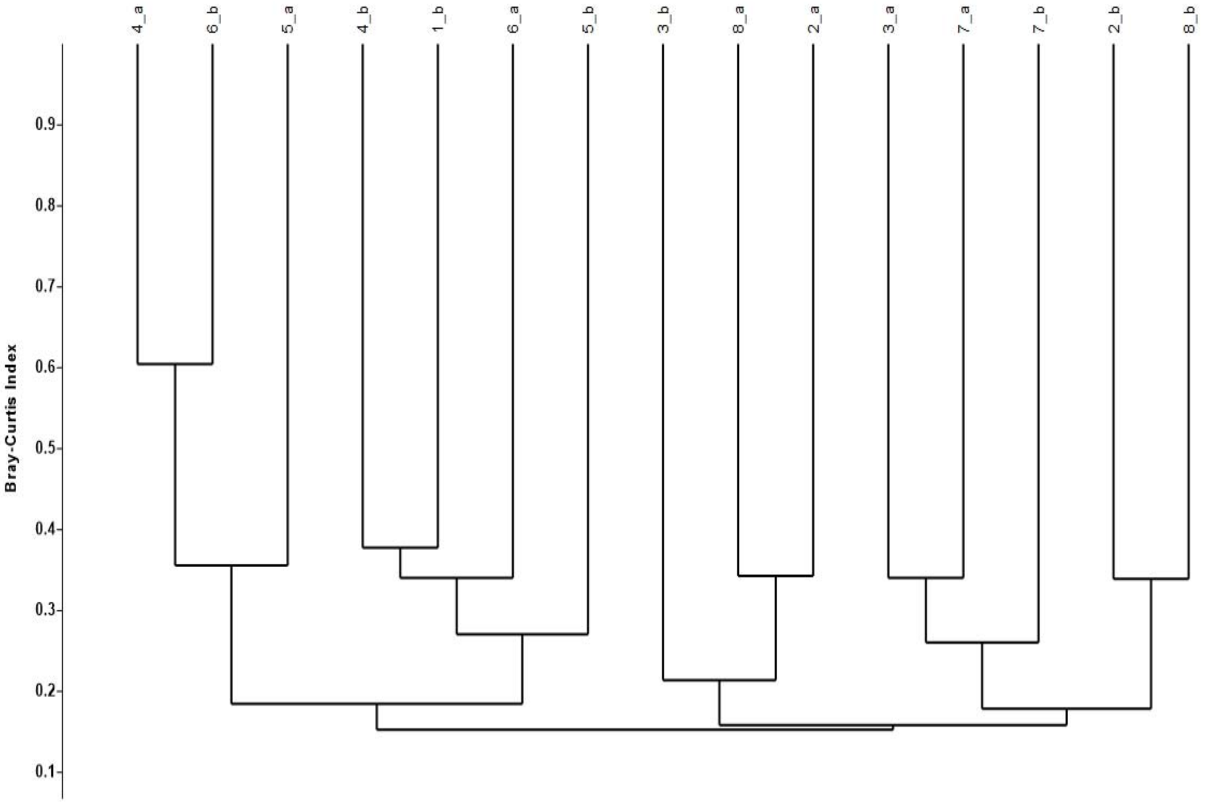

Figure 02: Dendrogram of similarity (Bray-Curtis Index) produced by cluster analysis (UPGMA connection method) of the tree species compositions among the eight sites (1-8). (a) Species with DAP > 20.0 cm. (b) Species with DAP < 1.0 cm.

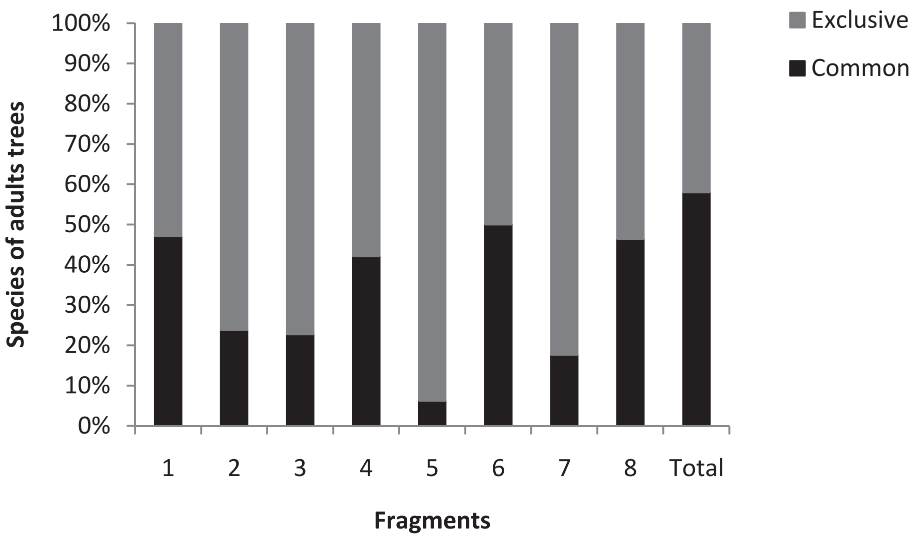

Figure 03: Proportions of adult tree species. Exclusive= Proportion of adult trees occuring only in the superior layer. Common= Proportion of adult trees occuring in both layers (adult and regenerative).

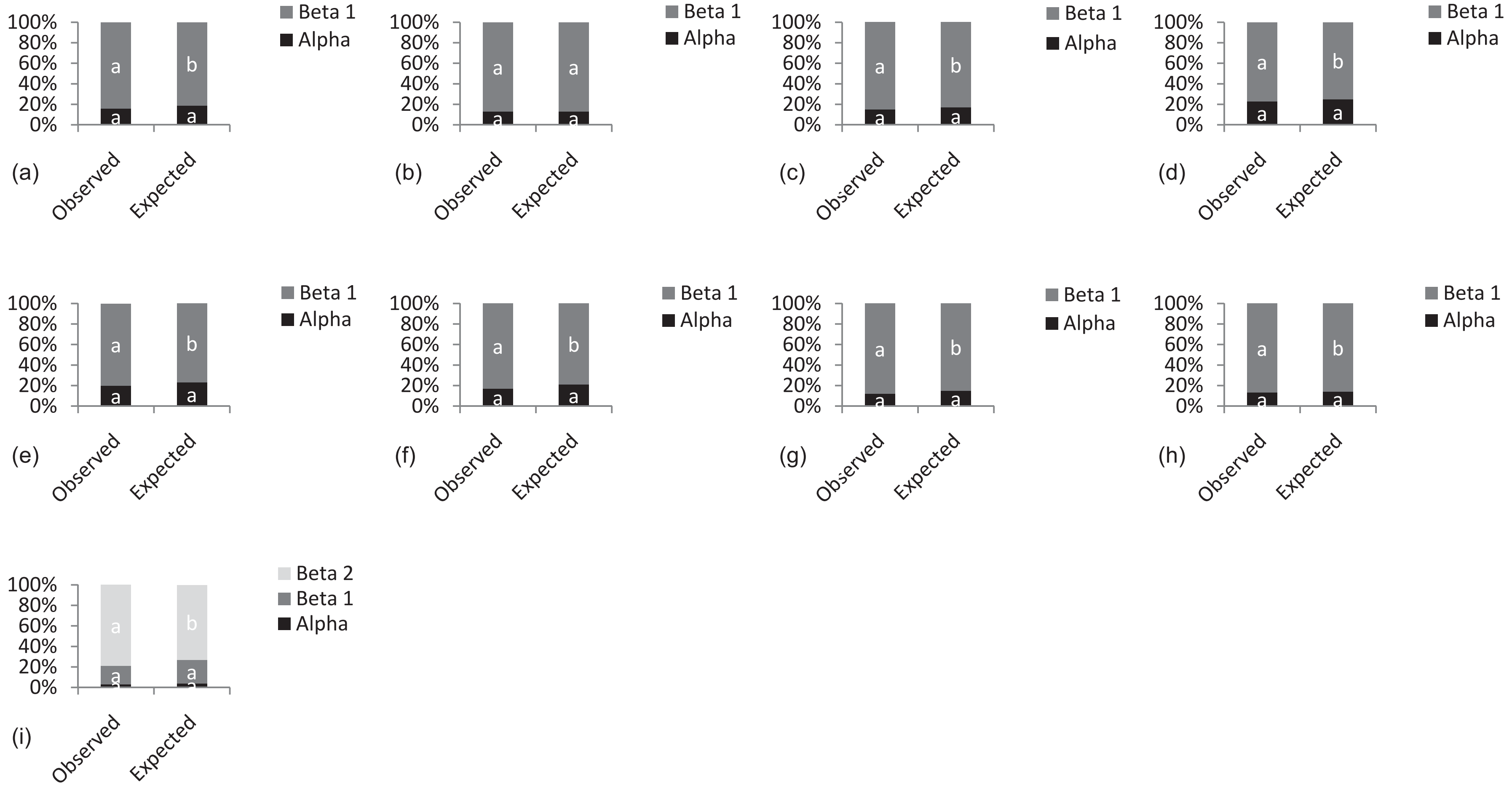

Figure 04: Additive partition of species diversity for tree species with DAP > 20.0 cm. Alpha is the average richness within the plots. Beta is the average richness absent in the plots. Beta2 is the average richness absent in the fragments. The sum of the alpha and

beta components results in the gamma diversity of each fragment. The sum of the alpha, beta and beta2 components results in the gamma diversity of the region. Different letters indicate observed ecological values significantly different than expected if the distribution were random. (a) Additive partition of species diversity in fragment 1 (p_{alpha}=0.99, p_{beta}=0.001). (b) Additive partition of species diversity in fragment 2 (p_{alpha}=0.33, p_{beta}=0.23). (c) Additive partition of species diversity in fragment 3 (p_{alpha}=0.94, p_{beta}=0.01). (d) Additive partition of species diversity in fragment 4 (p_{alpha}=0.93, p_{beta}=0.02). (e) Additive partition of species diversity in fragment 5 (p_{alpha}=0.99, p_{beta}=0.002). (f) Additive partition of species diversity in fragment 6 (p_{alpha}=0.99, p_{beta}=0.001). (g) Additive partition of species diversity in fragment 7 (p_{alpha}=0.99, p_{beta}=0.001). (h) Additive partition of species diversity in fragment 8 (p_{alpha}=0.86, p_{beta}=0.03). (i) Additive partition of species diversity in the region (all fragments) (p_{alpha}=0.99, p_{beta}=0.99, p_{beta2}=0.001). Different letters indicate statistical differences.

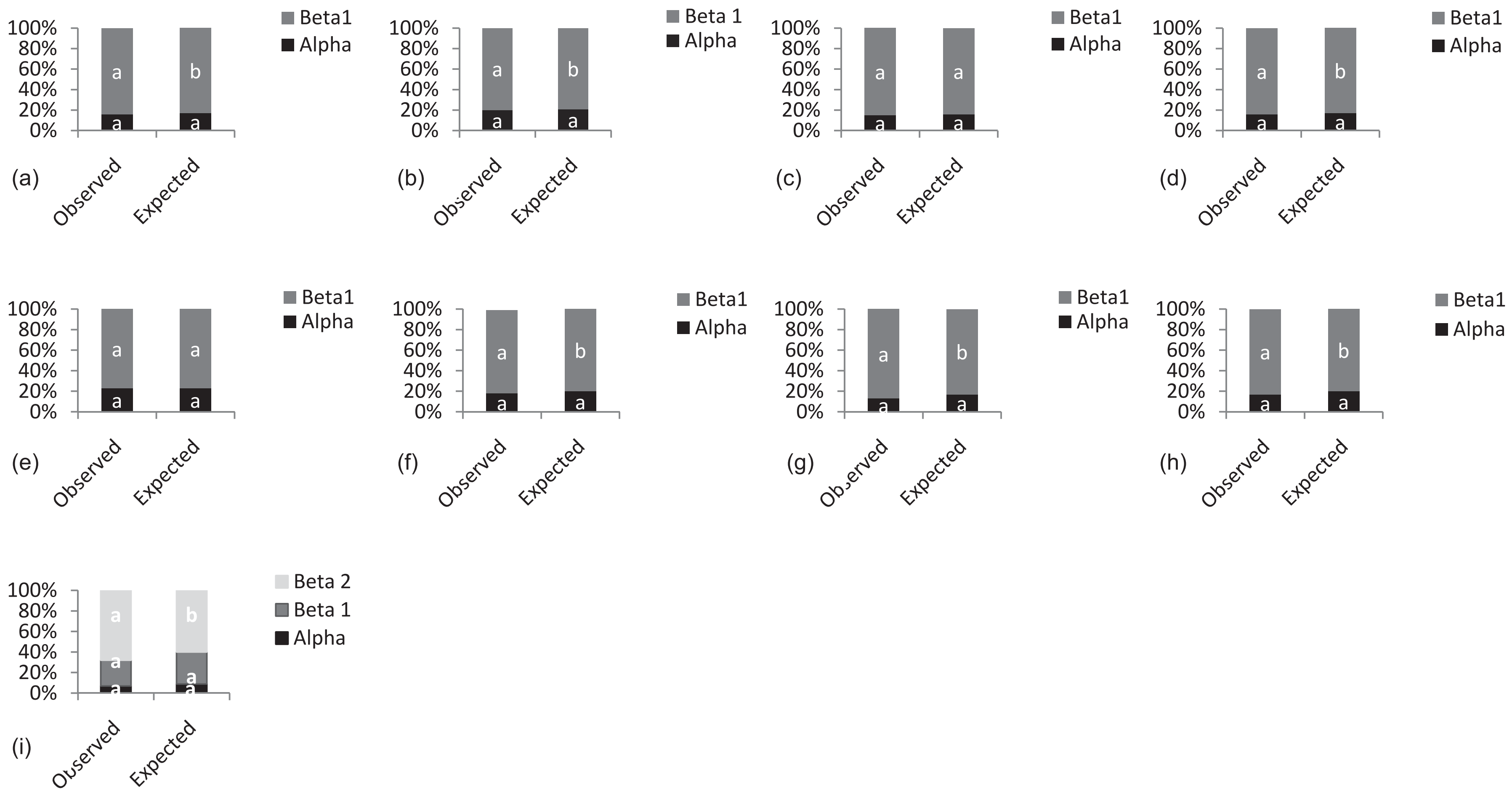

Figure 05: Additive partition of species diversity for trees species with DAP > 20.0 cm. Alpha is the average richness within the plots. Beta is the average richness absent in the plots. Beta2 is the average richness absent in the fragments. The sum of the alpha

and beta components results in the gamma diversity of each fragment. The sum of the alpha, beta and beta2 components results in the gamma diversity of region. Different letters indicate observed ecological values significantly different than expected if the distribution were random. (a) Additive partition of species diversity in fragment 1 (p_{alpha}=0.94, p_{beta}=0.01). (b) Additive partition of species diversity in fragment 2 (p_{alpha}=0.99, p_{beta}=0.003). (c) Additive partition of species diversity in fragment 3 (p_{alpha}=0.68, p_{beta}=0.32). (d) Additive partition of species diversity in fragment 4 (p_{alpha}=0.92, p_{beta}=0.01). (e) Additive partition of species diversity in fragment 5 (p_{alpha}=0.48, p_{beta}=0.21). (f) Additive partition of species diversity in fragment 6 (p_{alpha}=0.89, p_{beta}=0.04). (g) Additive partition of species diversity in fragment 7 (p_{alpha}=0.99, p_{beta}=0.001). (h) Additive partition of species diversity in fragment 8 (p_{alpha}=0.86, p_{beta}=0.03). (i) Additive partition of species diversity in the region (all fragments) (p_{alpha}=0.99, p_{beta}=0.99, p_{beta2}=0.001). Different letters indicate statistical differences.

CAPÍTULO 04

Functional diversity loss of sapling tree community in a highly fragmented landscape[1]

[1] Nos moldes da revista BiologicalConservation

Functional diversity loss of sapling tree community in a highly fragmented landscape

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Abstract

Conservation strategies that aim to effectively protect forest remnants have to take into account the effects of the surrounding landscape composition on the species composition and diversity of the fragments. In addition, functional diversity and distribution patterns of organisms need to be considered. The aim of this study was to analyze and compare the functional diversity of adult and sapling layers and to verify how this attribute is affected by landscape configuration. The study was conducted in eight forest fragments of a Semideciduous Atlantic Forest. We divided the sampled individuals into two classes, adult and sapling trees, and all sampled individuals were classified within 52 functional types belonging to 16 functional categories. The landscape composition effect on total functional diversity of both layers was evaluated using a generalized linear mixed model. The adult and sapling communities were found to have similar functional compositions and partitioning of diversity. However, the sapling layer (i) had a larger number of individuals in more generalist categories (white flowers, selfcompatible reproductive system, pollination by generalist insects), (ii) showed lower functional diversity, and (iii) the beta diversity variation was due to turnover, and not to nesting as with adult individuals. In addition, (iv) the sapling community showed a positive relationship with the percentage of pasture matrix. We interpreted this as the pasture benefiting the individual abundance of more generalist groups, therefore increasing the total functional diversity and diversity within functional categories. The fact that the adult community is not affected by any landscape parameter is an alert. Conclusions about fragmentation effects and conservation initiatives should not be made considering only adult species diversity. Another relevant point was the functional diversity study: we found that fragments near pasture matrices showed high diversities in terms of species composition, although the species that benefited most were those that performed more general functions.

Keywords: Additive partitioning of diversity, beta diversity, forest cover, forest fragments, functional diversity, matrix effect and sapling layer.

Introduction

Currently, most tropical biodiversity is found in modified human landscapes (Melo et al. 2013), and parks and reserves may only maintain 10% of the natural cover of tropical ecosystems (Gradwohl & Greenberg 1991). In this context, conservation strategies that aim to effectively protect forest remnants have to take into account the landscape composition effects on species composition and diversity. Land use around forest fragments influence abiotic and biotic effects within them (Fahrig 2003, Deconchat et al. 2009). A reduction in forest cover and consequent loss of habitat heterogeneity in a fragmented landscape may, for example, decrease population size (Prugh et al., 2009) and species numbers found in the fragments (Gaston & Blackburn 2000).

More open matrices, such as grassland, are very different and tend to have negative effects on forest fragments (Mesquita et al. 1999). In addition, matrices that are structurally more similar to fragments (e.g., shrub or arboreal matrices) may have positive effects on fragments (Mesquita et al. 1999, Hernandez-Ruedas et al. 2014). These matrices can act as a buffer zone by softening the edge effect and function as an extension of the fragment by providing shelter and resources for animals (Hernandez-Ruedas et al., 2014). Besides this, studies suggest that the management type developed for land use could have effects on fragments (Karp et al. 2012, Lindborg et al. 2014). In general, highly mechanized crops, where large amounts of pesticides are used, have a greater negative effect on fragments than less intensively managed crops (Karp et al. 2012, Lindborg et al. 2014).

It is relatively clear that landscape change may have negative effects on the composition, diversity and structure of plant communities (Laurence et al. 2011). However, there is an unavoidable loss of information when only species richness and diversity are considered in fragmentation studies. These indices ignore differences between species because traditional measures of diversity assume two premises. First, all species are considered equivalent, that is, they all have the same importance with respect to the amount of information they carry (Magurran 2004). Second, within the same species, individuals are assumed to be equivalent, regardless of differences in height or biomass, or whether they grow in more or less shaded locations. An alternative to these indices is functional diversity, which is the diversity of functional types, for example, plant groups that present similar responses to environmental conditions and have similar effects on ecosystem processes

(Cornelissen et al. 2003). This approach is important because it is a concept that links organisms and communities (Petchey & Gaston 2006). In addition to having greater biological significance, functional diversity measures can be used for ecological projections beyond a regional scale. The study of plant responses to environmental changes at a global scale cannot be based on species, since most of them have a limited geographical distribution (Pillar 1999). For ecological predictions, the use of functional groups is better because a functional type aggregates a plant group that similarly affects the environment or presents a similar response to the same environmental variations at any scale (Gomez 2010).

Another aspect often ignored in fragmentation studies is understanding the distribution patterns of organisms in a fragmented landscape (Fahrig 2003). One way to approach this problem is through additive partitioning of diversity (Lande 1996), where the total diversity (γ) is divided between the diversity within (α) and between (β) samples. Beta diversity is a central concept that can be used to understand how species diversity is organized and maintained, as well as to test ecological theories that try to understand the process that generated the patterns, and there is potential to apply this concept in conservation biology. Understanding beta diversity is important because it considers not only local diversity (alpha), but the difference in diversity between areas (Condit et al. 2002). Using beta diversity, it is possible to verify if the areas have many or few common species, and to investigate which landscape or environmental parameters are structuring the communities of the areas. The few studies about this indicate that landscape configuration plays a key role in beta species diversity (Arroyo Rodriguez et al. 2013, Hernandez-Ruedas et al. 2014, Carneiro et al. 2016); however, there are no studies about the effect of the landscape on functional beta diversity.

Unfortunately, despite the great importance of functional diversity and functional beta diversity, these aspects have not been considered in most studies. Another poorly considered aspect is the fragmentation effect on richness and diversity of regenerating individuals, and there are no studies that consider the landscape effect on regenerating functional diversity. One of the main questions in terms of forest fragment conservation is to understand if plant communities are viable over time (Melo et al. 2013). One of the most important ways to answer this question is to study young individuals (Silva et al. 2007). Studies of seedling and sapling layers allow inferences about the current state and conservation future of fragments, and about their viability to preserve biodiversity over the years (Silva et al. 2007).

The aim of this study was to analyze and compare the functional diversity of adult and saplings layers and to verify how this attribute is affected by landscape configuration. More specifically, we intend to answer the following questions: (a) Are the adult and sapling communities different in terms of functional composition and diversity?, (b) Does forest cover and percentage of coffee plantations have a positive effecton total functional diversity, functional diversity of each category, and beta diversity of adult and sapling communities?, (c) Does percentage of pasture and sugarcane have a negative effecton total functional diversity, functional diversity of each category, and beta diversity of adult and sapling communities?. We assumed that fragmentation might negatively affect the more specific functions (dispersal by large mammals, large fruits, self-incompatible sexual systems) and favor the establishment of more generalist functions (small fruits, general insect pollination, pioneer trees) in the both layers

Methods

Study area

The eight forest fragments in Alfenas, Minas Gerais, Brazil was studied(Table 1).The distance between fragments ranged from 3.1 to 49.6 km with mean of 28.1 ± 14.8 km. The average rainfall is 26 mm in the winter and 290 mm in the summer (1500 mm annually) and the average temperatures is 16.9ºC in the winter and 21.5ºC in the summer(Alvares et al. 2014). The region has a predominantly hilly relief with an elevation ranging from 720 to 1350 m. The most common matrix types are pastures (51%), perennial crops (mainly coffee - 17%) and annual crops (mainly sugarcane and corn - 7%).

The fragments studied are preserved remnants of submontane semideciduous Atlantic Forest. The region only has about 9% remnant native forest and (Olivetti et al. 2015).The most dominant families of trees are Fabaceae, Myrtaceae, Lauraceae, Meliaceae, and Euphorbiaceae and most tree species are *Copaifera langsdorffii* Desf., *Ocotea odorifera* (Vell.) Rohwer., *Cryptocarya aschersoniana* Mez., *Metrodorea stipularis* Mart., and *Miconia willdenowii*Klotzsch ex Naudin.

We selected the nine fragments by classifying remnant forest areas using images (20 m resolution) from 2009 that were taken by the Sino-Brazilian CBERS-2B satellite (INPE 2009). To select the fragments, the following criteria were applied: i) similarity of degradation by observing spectral attributes, such as color and texture; ii) a minimum distance of four kilometers between areas, to ensure sample independence; and iii) a central fragment size between 15 ha and 100 ha.

Sample design and species sampling

Ten plots of 10 x 20 m (0.2 ha) were demarcated in each of the nine fragments studied (resulting in a total of 1.8 ha sampled). We divided the sampled individuals into two classes: adult trees, individuals with a diameter at breast height (DBH) > 20 cm; and sapling trees, individuals with a DBH < 1 cm and height > 1.3 m. We selected individuals greater than 1.30 meter because they survived the critical period of mortality (seedlings) and surpassed strong environmental selective action (Felfili et al. 2000, Sales & Schiavini 2007). We excluded shrubs from the analysis, or species that will never become more than 6 m tall (Oliveira-Filho et al. 2004, Liebsch 2008), to guarantee that only trees were analyzed in the study.

All sampled individuals were classified within 52 functional types belonging to 16 functional categories (Table 2) (Cornelissen et al. 2003, Carneiro et al. 2016). The functional categories area more broads, suchas dipersion type and the functional types are more specific, such as plants zoocoric dispersion, each functional categorie has several functional types. The classification into functional types was based on data in the literature and field observations. The abundance of individuals within each functional type for each landscape was the sum of the values from the ten plots sampled.

Landscape composition

A buffer of 1,000 m was drawn from the center of each fragment and the percentage of each landscape composition (COVER and MATRIX) was calculated using the programs ArcGIS (Version 10.0, ESRI) and Fragstats (McGarigal & Marks 1995). COVER is the proportion of forest in the buffer, and was calculated using the formula $Af = (\pi * r^2 * PF)/100$, where Af is the total area of forest (forest), πr^2 is the area of the buffer around the central fragment, and PF is the total proportion of forest in radius r of each buffer. MATRIX (Landscape index) quantifies the proportion of buffer covered by each habitat matrix (coffee, pasture, and sugarcane), and was calculated using the equation $Am = (\pi r^2 * PM)/100$, where Am is the sum of anthropogenic habitat matrix areas and PM is the total proportion of habitat matrix in each buffer. All the landscape metrics were log10 transformed.

Functional diversity calculation and statistical analysis

To calculate the functional diversity (FD) the method described by Petchey & Gaston (2002) was used. FD measures the extent of complementarity between values of individual traits. Greater differences between these values represent greater complementarity and, therefore, a higher FD. A FD value was obtained for each fragment and for eachfunctional category. The FD calculation is based on a cluster analysis and has the following four steps: (I) creating a functional matrix (abundance of individuals × functional traits) for each fragment studied; (II) converting the functional matrix into a distance matrix; (III) producing a dendrogram from the distance matrix; and (IV) calculating the total length of the dendrogram branches. The distance measure used was the Gower distance (Podani & Schimera 2006). There are other functional diversity measures (Petchey et al. 2004); however, FD better explains the community functional differences because it is not affected by species number (Petchey & Gaston 2006). The analyses were performed with the software R (R Development Core Team) *2.5,* and the FD, ade4, and picante packages.

To describe the adult and sapling communities in terms of functional composition and to check for dominant functional types within each functional category, an ANOVA and Tukey's test were conducted when there were more than two functional types within the same functional category, and a T-test was conducted when there were only two functional types. This analysis helps verify, for example, if there is a dominance of functional type characteristics from more generalist species, such as pioneer trees, small fruits and self-compatibility. The functional similarity between the fragments was analyzed using the Bray-Curtis index. This index was selected because it considers the abundance of individuals (Magurran 2004).To verify if the proportion of adult individuals differs from the proportion of sapling individuals within each functional group we used the G Test.

Additive partitioning of species diversity was used to verify how the variation of diversity occurs at different scales (plots and fragments) for both life stages (Crist et al.2003). In the additive partitioning of species diversity, the diversity of functional types of the region or in each functional category (γ) is split into alpha and beta components that are expressed in the same units. Therefore, γ = alpha + beta 1 + beta 2 +...beta n, where n = number of levels, and beta n = alpha n+1 - alpha n (Podani & Schimera 2006). This observed diversity was compared with a null distribution model of diversity consisting of the mean diversity obtained through 1000 randomizations. The 1000 randomizations create the expected diversity if the distribution of individuals is random. This analysis also allowed us to compare the results observed to what would be expected if the distribution was random, that is, when there is no factor affecting the community structure (Crist et al.2003). A difference from the expected values may indicate dispersal limitation, environmental heterogeneity and/or effect of the landscape on the observed diversity (Crist et al.2003). The expected diversity was calculated with the software Partition (Crist et al.2003).

To verify whether the beta diversity observed was due to loss of ordinate functions (nestedness) or exchange functions (turnover), we decomposed the beta diversity of these two components. The index of beta diversity ranges from 0 (completely similar) to 1 (completely dissimilar). Subsequently, we partitioned the total dissimilarities (B_{SOR}) in the proportion generated by nestedness (B_{NES}) and in the proportion generated by turnover (B_{SIM}), where B_{SOR}= B_{NES} + B_{SIM} (Baselga 2010). These analyses were carried out in R (R Development Core Team 2007) using the functions “beta-multi.R” in the package “betaparte.” A recent study (Baselga & Leprieur 2015) showed that with this method the nestedness-resultant component accounts for only richness differences derived from nested patterns, while in other methods the richness difference dissimilarity accounts for all kinds of richness differences. Moreover, in the method proposed by Baselga (2010) the replacement component is independent of richness difference.

We performed a partial redundancy analysis (PRDA) (Rao 1964) to verify whether the landscape configuration (COVER and MATRIX) can explain the variation in the functional beta diversity of each layer among the fragments. For this analysis, we used two data matrices, the matrix with the number of functional types in each fragment, and the matrix with the landscape parameters. The matrix with the landscape parameters was subjected to a selection of variables (stepwise CCA) to optimize the correlation between a subset of landscape variables and the matrix of functional types. The calculation of how landscape parameters selected by stepwise CCA can explain the variation in the beta diversity between the fragments was carried out according to Legendre et al. (2005) using the vegan package available in the software R (R Development Core Team 2007). The significance of the percentage of explanation was evaluated according to the permutation method described by Legendre & Legendre (1998), performing 9999 permutations.

The landscape composition (COVER, percentage of coffee, pasture and sugar cane) effect on the total functional diversity (FD), functional diversity of categories and on the number of individuals within the functional types of both layers was evaluated using a generalized linear mixed model(glmer in R package lme4) and included the fragments as a random factor. The fragment was used as a random variable because it was not in our interest check the fragment effect on functional diversity (FD) and on the number of individuals within functional types. In other words, we considered the samples dependence within each fragment and increased the test power by subtracting the random factor error variation. A generalized linear mixed model was runto model the (i) abundance within functional types (response data), with Poisson distribution, and (ii) functional diversity (response data) and functional diversity of categories, with Gaussian distribution. The effect of isolated landscape parameters (independent variable) and the effect of parameters in pairs were checked through an additive relationship. An additive relationship means that the value of a variable does not depend (there is no interaction) on the value of another variable, but the variables are necessary for a good model fit and a more accurate estimation. Significant models were validated by normality and independence testing of residue. To assess the performance of models, the Akaike Information Criterion (AIC) was employed (Burnham & Anderson 2001). The best model was based on the lowest AIC value (Crist et al.2003, the ΔAIC and Akaike weights (wAIC). The ΔAIC is the difference between the model and the minimum AIC values, and the models with ΔAIC < 2 are considered as having substantial support, while the Akaike weights (wAIC) describe the probability of the model being the best model among those studied (Burnham & Anderson 2001). Only the results from valid models are presented.

Results

Functional composition and additive partition of functional diversity

We sampled 363 individuals belonging to 104 species in the adult layer and 462 individuals belonging to 103 species in the sapling layer. The two layers are very similar in terms of functional composition (Bray-Curtis=0.89). The functional groups within each functional category with the highest number of adult and sapling individuals, that is, the functional groups more abudants were the following: flowering in rainy season, white flowers, pollination by bees, resource of nectar floral, dispersion by bird, small number of seeds and slow growth. When we compare the proportion of adult and sapling individuals within each functional group, the sapling layer has a higher proportion of individuals in the following functional groups: flowering in rainy season, white flowers, pollination by bees and generalist insects, flower resource of nectar and pollen, fruiting in dry season, dispersal by birds, medium fruits, small seeds, average number of seeds, perennial individuals, self-compatible individuals and light wood density (Fig. 2).In addition, the regenerating individuals presented lower functional diversity (t_{134}=1.83, p=0.03).

When we analyze the additive partition of functional diversity, approximately 95% of the total diversity (gamma) for the adult and sapling layers is due to diversity within the fragments (alpha) (Fig. 3a, b). The beta diversity (between fragments) is responsible for approximately 5% of the gamma diversity and was higher than expected at random for adults (p=0.01) (Fig. 3a) and saplings (p=0.01) (Fig. 3b). The difference between the two layers was that while most (67%) of the adult beta diversity is generated by nesting, most (55%) of the beta diversity of the saplings is generated by turnover.

Effect of landscape configuration

The adult individuals were not affected by landscape parameters. There is no relationship between these parameters and (i) total functional diversity, (ii) functional diversity by category, (iii) and beta diversity of adult individuals.

On the other hand, sapling individuals are positively affected by the pasture matrix. The percentage of pasture matrix has a positive relation with (i) total functional diversity (Table 3) and (ii) functional diversity of all categories, except fruit size (Table 3). The large part (78%) of sapling beta diversity is explained by the percentage of pasture matrix ($F_{1,6}$=26.50; p=0.012).

The percentage of pasture matrix also has a positive relation with individual abundance of the sapling layer for the following functional types: flowering in rainy season, white flower, pollination by generalist insects and bees, floral reward of nectar and pollen, dispersion by birds, medium number of seeds, perennial individuals, self-compatible reproductive system and light wood density (Table 4).

Discussion

The adult and sapling communities present similar functional composition and partition of diversity, however, there are considerable differences because the sapling layer has a larger number of individuals in more generalist categories, such as white flowers, pollination by generalist insect, small seeds, self-compatible reproductive system and light wood density. This layer also presents lower functional diversity, and the beta diversity variation is due to turnover, not to nesting as found for the adult individuals. In addition, while the adult community is not affected by any landscape parameter, the sapling community has a positive relationship with the percentage of pasture.

We hypothesized that there would be a large difference in functional composition between layers. Despite that some results support our hypothesis, such as differences in individual abundance in some functional categories, the layers were similar in terms of functional composition. This result is unexpected because a study in the same region revealed that these two layers are very different in terms of floristic composition (Carneiro et al. unpublished data). In other words, layers have different species but similar functions. Our hypothesis for explaining this result is that although the species are different, those that belong to generalist functional groups are common in both layers. The result that supports this hypothesis is that the functional diversity of the adult community is nested. In other words, less diverse adult communities are a subset of the most diverse communities. Nesting is generated by the ordered loss of more sensitive functions and, consequently, an increase in dominance of tolerant species (Chase & Myers 2011). Thus, the less diverse communities, or those dominated by abundant species that have not been lost yet, are a sample of the most diverse communities, without sensitive and rare species (Laurance et al. 2007, Arroyo-Rodríguez et al. 2013).

Another similarity between the layers is the additive partition of functional diversity, where most of gamma diversity (95%) is due to diversity within plots (alpha diversity) both for adults and saplings. This result is also contrary to what was found for species composition in the same region (Carneiro et al. unpublished data). The additive partition of species diversity revealed that 80% of the gamma diversity is due to beta diversity (diversity between fragments) (Carneiro et al. unpublished data), not the alpha diversity as revealed by this study. This result is also contrary to what is usually found for species composition in tropical forests (Condit 2002). Continuous tropical forests have high alpha diversity, but most of gamma diversity is due to beta diversity (Condit 2002). This happens due to high environmental heterogeneity of forests and dispersion (MacArthur & MacArthur 1961, Butaye et al. 2001). The heterogeneity creates different niches that harbor different species performing different functions (MacArthur & MacArthur 1961), while seed and pollen dispersion leads to a greater species flow (Butaye et al., 2001). The fact that 95% of the gamma diversity is due to alpha diversityand the fragments presents small beta diversity indicates that there is a great functional redundancy between the fragments, that is, the fragments have very similar functions.

The difference between the layers is that while the functional diversity of the adult community is nested, the structure of the sapling community is generated by turnover, that is, there is an exchange of species between sapling communities in fragments (Laurence et al. 2007, Baselga 2010). Unfortunately, the results indicate that this species exchange does not seem to benefit the sapling community. When we compare the proportion of individuals in functional groups between layers, the sapling community presents a higher proportion of individuals in more generalist functional groups, such as white flowers, pollination by generalist insect, small seeds, self-compatible reproductive system, and light wood density. In addition, this layer has less functional diversity than the adult layer. These three facts, hight redundance functional between fragments, the dominance of generalist functions and less functional diversity, confirm our initial hypothesis that sapling individuals are being affected by the fragmentation processthat this process, particularly the landscape composition, is acting as an environmental filter, inhibiting the establishment of species that perform more sensitive functions, and favoring the establishment of more generalist species that, consequently, reduce functional diversity.

The percentage of pasture matrix has a positive relation with total functional diversity, functional diversity of most functional categories and the proportion of individuals in some functional groups. However, this result is contrary to our expectations. We expected that this anthropogenic matrix would have a negative effect on the sapling community, since pasture matrices have structure (plant height and density) and a microclimate very different from forest fragments (Mesquita et al. 1999). Inhospitable matrices, such as pastures, can increase microclimatic differences between the fragment and edge, intensifying the edge effect and making dispersal between fragments difficult (Galetti et al. 2003). Our hypothesis for this surprising finding is that pasture increases total functional diversity and within functional categories, because this anthropogenic matrix is benefiting the individual abundance of more generalist groups, a fact that is confirmed by our results. Our expectation is that over the years this effect will become negative, since there is a tendency for these fragments to be dominated by more generalist functions.

As expected, the adult community is not affected by any landscape parameter. This result is an alert. Conclusions about fragmentation effects and conservation initiatives should not be made considering only adult species diversity. A previous work in the region showed that, in terms of species composition, the fragments are very rich and have high diversity (Carneiro et al. submitted). If the sapling layer was not analyzed we could conclude that the fragments are in good condition, since the functional diversity of the adult layer is not affected by the landscape composition. Another relevant point is the functional diversity study: although we found that, in relation to species composition, the fragments have the pasture matrix benefits the high diversity of species composition. However, the species that benefit are those that perform more general functions.

This work is the first to partition functional diversity and verify the landscape effect on functional diversity. The results found are worrying. Fragment regeneration is being affected by fragmentation, and the pastures growing around the fragments are benefiting the individual abundance of more generalist groups. If there is no management, the new assemblage of trees in the fragments will change over time; they will become poorer and more generalist. Measures need to be taken in order to preserve these environments because most of the tropical biodiversity today is located in forest fragments (Perfecto & Vandermeer 2010). If nothing is done, the predictions are pessimistic. Functional diversity is extremely important for ecosystem balance and resilience (Diaz & Cabido 2001), and communities with low functional diversity may not be able to recover from environmental disturbances (resilience) because the loss of just a few species can represent the loss of many functions (Diaz & Cabido 2001, Enrique et al. 2016).

Acknowledgments

We thank Derek Felipe, Erlon Pimenta, André Carvalho, Marcela Alcantara and Bruno Roberto Ribeiro for helping with fieldwork, Hisaias Almeida for identifying the species, and Roy Funch for correcting the English.

Funding source

This research was financially supported by a grant from the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES) awarded to Magda Silva Carneiro. The project was also supported by the Fundação de Amparo a Pesquisa do Estado de Minas Gerais (FAPEMIG)/Vale Company (RDP-00104-
10) and the Conselho Nacional de Desenvolvimento Cientifico e Tecnológico (CNPq) (472250/2010-8 and 306595/2014-1). These agencies did not influence in study design, collection, analysis and interpretation of data, the writing of the report, and the decision to submit the article for publication.

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Anexos

Table 01: Fragment locations and landscape parameter ecological values analyzed.

Fragment	**Location**	**Area (ha)**	**% Coffee**	**% Sugarcane**	**% Pasture**
1	21°29'13.13"S 46° 5'40.32"W	20.91	15.72	31.39	**33.14**
2	21°26'14.51"S 46° 8'46.93"W	22.99	51.83	0.00	**14.02**
3	21°34'42.37"S 45°58' 15.04"W	28.57	16.83	45.13	**10.5**
4	21°33' 44.68"S 45°56' 12.80"W	36.85	62.87	0.00	**13.74**
5	21°27'50.38"S 45°54'58.10"W	37.05	0.00	58.33	**0.00**
6	21°25'25.97"S 46° 5'8.03"W	56.05	21.19	36.96	**8.18**
7	21°28'16.28"S 46° 7'22.43"W	81.55	1.17	0.00	**51.56**
8	**21°25'27.26"S 46° 9'35.66"W**	**87.18**	**0.00**	**32.51**	**2.3**

Table 2: Functional categories studied with their respective reproductive functional groups (RFGs) and community functions that can affect functional type changes.

Functional categories	Functional type	Affected function
Flowering season		Pollination Resource provision for wildlife
	Rainy (October-March)	
	Dry (April –September)	
Flower color		Pollinator attraction Reproduction success
	Yellow	
	White	
	Red	
	Green	
Pollination		Reproduction success Wildlife associated Disturbance tolerance
	Bees	
	Beetles	
	Butterflies	
	Flies	
	Generalist insects	
	Vertebrates	
	Wasps	
	Wind	
Floral reward		Resource provision for wildlife Wildlife associated
	Floral tissues	
	Nectar	
	Odor	
	Oil	
	Pollen	
	Shelter	
	Without resource	
Fruiting season		Dispersion Resource provision for wildlife
	Rainy (October- March)	
	Dry (April –September)	
Dispersal		Reproduction success Migration Dispersal distance
	Self-dispersal	
	Birds	
	Large mammals	
	Small mammals	
	Bats	

	Wind	
Fruit size	Small (< 0.5 cm) Medium (0.6-2.9 cm) Large (> 3.0cm)	Wildlife associated Dispersal distance Longevity in seed bank
Fruit type	Fleshy Dry	Wildlife associated Dispersal distance Longevity in seed bank
Seed size	Small (< 1.0 cm) Medium (1.1-2.9 cm) Large (> 3.0cm)	Wildlife associated Dispersal distance Longevity in seed bank Resource provision for wildlife Competitive ability
Seed type	Orthodox Recalcitrant	Longevity in seed bank Stress tolerance
Seed number	Few seeds (0-5 seeds) Medium number of seeds (6-10 seeds) Many seeds (> 10 seeds)	Competitive ability Reproduction success Resources provision for wildlife
Leaf phenology	Deciduous Perennial	Stress tolerance Disturbance avoidance
Successional class	Pioneer Climax	Disturbance tolerance Competitive ability Composition change Succession failed
Reproductive systems	Self -compatible Self -incompatible	Disturbance tolerance Reproduction success

Growth rate		Competitive ability Plant longevity
	Fast growth (> 0. 40 cm)	
	Slow growth (< 0.41 cm)	
Wood density		Competitive ability Carbon storage
	Light density (> 0.6 g/cm^3)	
	Moderate density (0.7-0.8 g/cm^3)	
	Heavy density (< 0.9 g/cm^3)	

Table 03: Relation between total functional diversity, functional diversity of the functional categories and the abundance of individuals within each functional type and the parameters of the landscape analyzed. Non-significant models are not presented.

	Models Estimate	p ecological value	ΔAICc	w
Total functional diversity				
Fixed effects				
Intercept	0.09	0.0001		
Pasture	0.002	0.01		
Random effect				
Fragment	0.03	-	-	-
Residuals	0.06	-	-	-
Functional diversity of categories				
Flowering season	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.37	0.0001	0.0	0.91
Pasture	0.03	0.009		
Random effect				
Fragment	0.0001	-	-	-
Residuals	0.16	-	-	-
Flower color	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.29	< 0.05	-	-
Pasture	0.05	0.01		
Random effect				
Fragment	0.07	-	-	-
Residuals	0.12	-	-	-
Pollination	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.27	0.0001	0.0	1
Pasture	0.03	0.006		
Random effect				
Fragment	0.005	-	-	-
Residuals	0.12	-	-	-
Floral reward	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.24	0.0001	0.0	0.1
Pasture	0.03	0.0001		

Random effect				
Fragment	0.09	-	-	-
Residuals	0.09	-	-	-
Fruiting season	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.43	0.0001	-	-
Pasture	0.03	0.005		
Random effect				
Fragment	0.03	-	-	-
Residuals	0.14	-	-	-
Dispersal	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.31	< 0.05	-	-
Pasture	0.02	0.008		
Random effect				
Fragment	0.05	-	-	-
Residuals	0.12	-	-	-
Fruit type	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.35	0.0001	0.0	1
Pasture	0.04	0.002		
Random effect				
Fragment	0.46	-	-	-
Residuals	0.16	-	-	-
Seed size	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.24	0.0001	0.0	1
Pasture	0.05	0.0003		
Random effect				
Fragment	0.55	-	-	-
Residuals	0.11	-	-	-
Seed type	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.33	0.0001	0.0	1
Pasture	0.04	0.003		
Random effect				
Fragment	0.02	-	-	-
Residuals	0.15	-	-	-
Seed number	Estimate	p ecological value	ΔAICc	w
Fixed effects				

Intercept	0.32	0.0001	0.0	1
Pasture	0.03	0.003		
Random effect				
Fragment	0.07	-	-	-
Residuals	0.12	-	-	-
Leaf phenology	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.41	0.0001	-	-
Pasture	0.04	0.004		
Random effect				
Fragment	0.005	-	-	-
Residuals	0.15	-	-	-
Successional class	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.41	0.0001	0.0	1
Pasture	0.04	0.004		
Random effect				
Fragment	0.02	-	-	-
Residuals	0.15	-	-	-
Reproductive systems	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.38	0.0001	0.0	1
Pasture	0.04	0.002		
Random effect				
Fragment	0.04	-	-	-
Residuals	0.15	-	-	-
Growth rate	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.42	0.0001	0.0	1
Pasture	0.04	0.003		
Random effect				
Fragment	0.04	-	-	-
Residuals	0.15	-	-	-
Wood density	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.29	0.0001	0.0	1
Pasture	0.03	0.003		
Random effect				
Fragment	0.12	-	-	-
Residuals	0.13	-	-	-
Individuals abundance in functional types				
Flowering in rainy season	Estimate	p	ΔAICc	w

		ecological value		
Fixed effects				
Intercept	0.93	0.0001	0.0	0.97
Pasture	0.02	0.0001		
Random effect				
Fragment	0.01	-	-	-
Residuals	0.10	-	-	-
White flowers	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.44	0.03	-	-
Pasture	0.02	0.001		
Random effect				
Fragment	0.15	-	-	-
Residuals	0.39	-	-	-
Pollination by generalist insects	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	-0.86	0.0001	0.0	0.87
Pasture	01.02	0.0008		
Random effect				
Fragment	0.01	-	-	-
Residuals	0.01	-	-	-
Pollination by bees	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.69	0.005	0.0	0.92
Pasture	0.02	0.0001		
Random effect				
Fragment	0.08	-	-	-
Residuals	0.28	-	-	-
Floral reward by nectar	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.98	0.0001	-	-
Pasture	0.01	0.003		
Random effect				
Fragment	0.11	-	-	-
Residuals	0.33	-	-	-
Floral reward by pollen	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	-0.99	0.0001	0.0	0.70
Pasture	0.03	0.0001		
Random effect				
Fragment	0	-	-	-

Residuals	0	-	-	-
Dispersal by birds	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.91	0.0001	0.0	0.94
Pasture	0.02	0.0001		
Random effect				
Fragment	0.04	-	-	-
Residuals	0.20	-	-	-
Medium number of seeds	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	-0.68	0.0001	0.0	0.68
Pasture	0.02	0.0001		
Random effect				
Fragment	0	-	-	-
Residuals	0	-	-	-
Perennial individuals	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.83	0.0001	0.0	0.81
Pasture	0.02	0.0001		
Random effect				
Fragment	0.06	-	-	-
Residuals	0.26	-	-	-
Self-compatible reproductive system	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.14	0.4	-	-
Pasture	0.02	0.0001		
Random effect				
Fragment	0.05	-	-	-
Residuals	0.24	-	-	-
Fixed effects				
Light density	Estimate	p ecological value	ΔAICc	w
Fixed effects				
Intercept	0.30	0.04	-	-
Pasture	0.20	0.008		
Random effect				
Fragment	0.04	-	-	-
Residuals	0.22	-	-	-

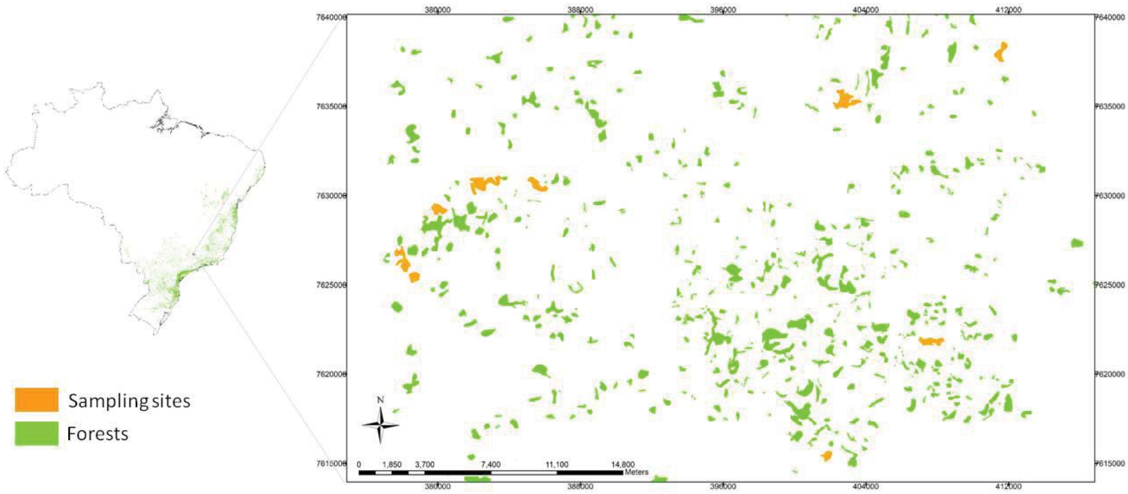

Figure 01: Location of the eight fragments studied in Alfenas, Minas Gerais. Orange polygons represent studied forest and green polygons represent other forests patches.

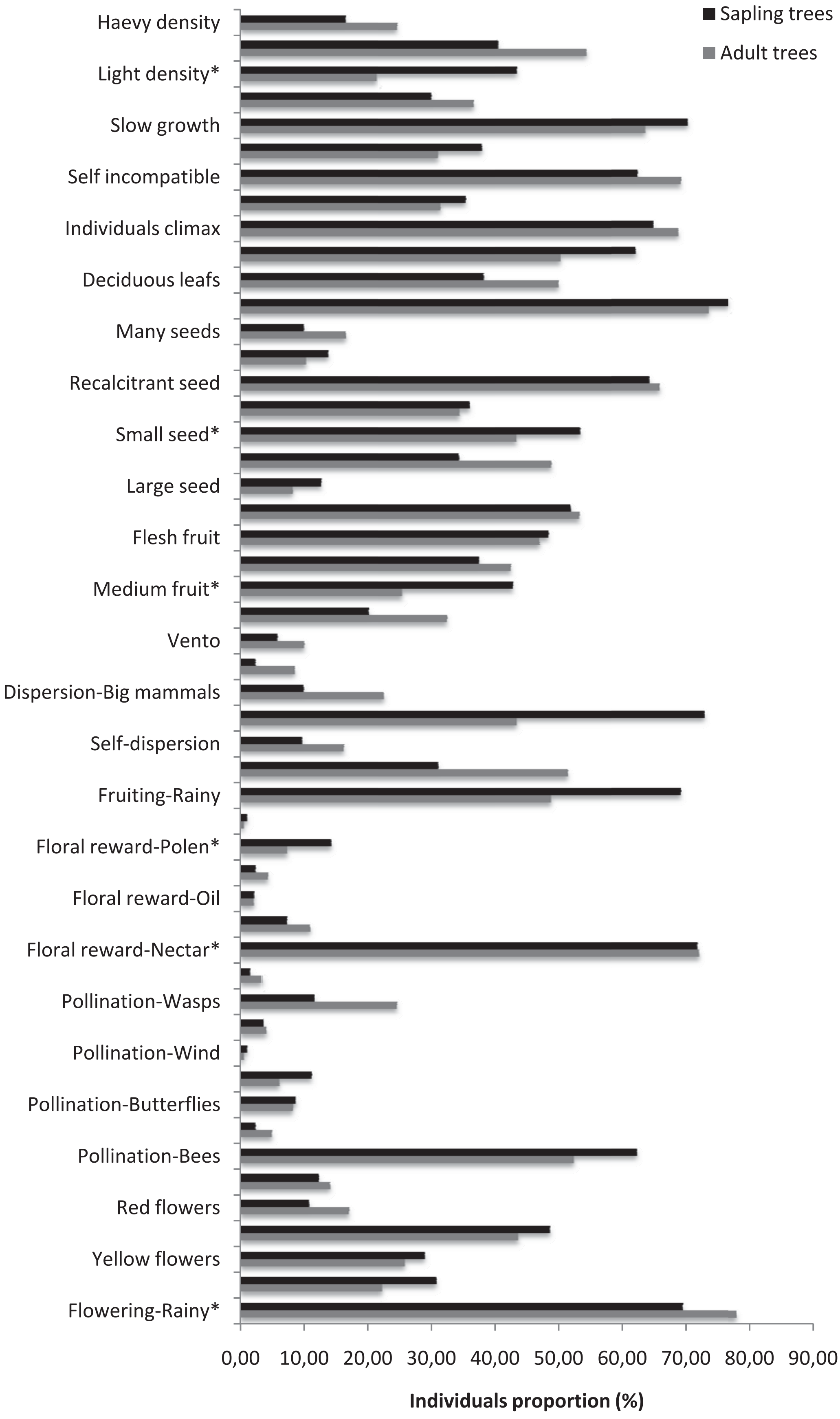
Haevy density
Light density*
Slow growth
Self incompatible
Individuals climax
Deciduous leafs
Many seeds
Recalcitrant seed
Small seed*
Large seed
Flesh fruit
Medium fruit*
Vento
Dispersion-Big mammals
Self-dispersion
Fruiting-Rainy
Floral reward-Polen*
Floral reward-Oil
Floral reward-Nectar*
Pollination-Wasps
Pollination-Wind
Pollination-Butterflies
Pollination-Bees
Red flowers
Yellow flowers
Flowering-Rainy*
Sapling trees
Adult trees
0,00
10,00
20,00
30,00
40,00
50,00
60,00
70,00
80,00
90,00
Individuals proportion (%)

Figure 02: Proportion of adult and regenerating individuals within each functional type. *= Significantly difference.

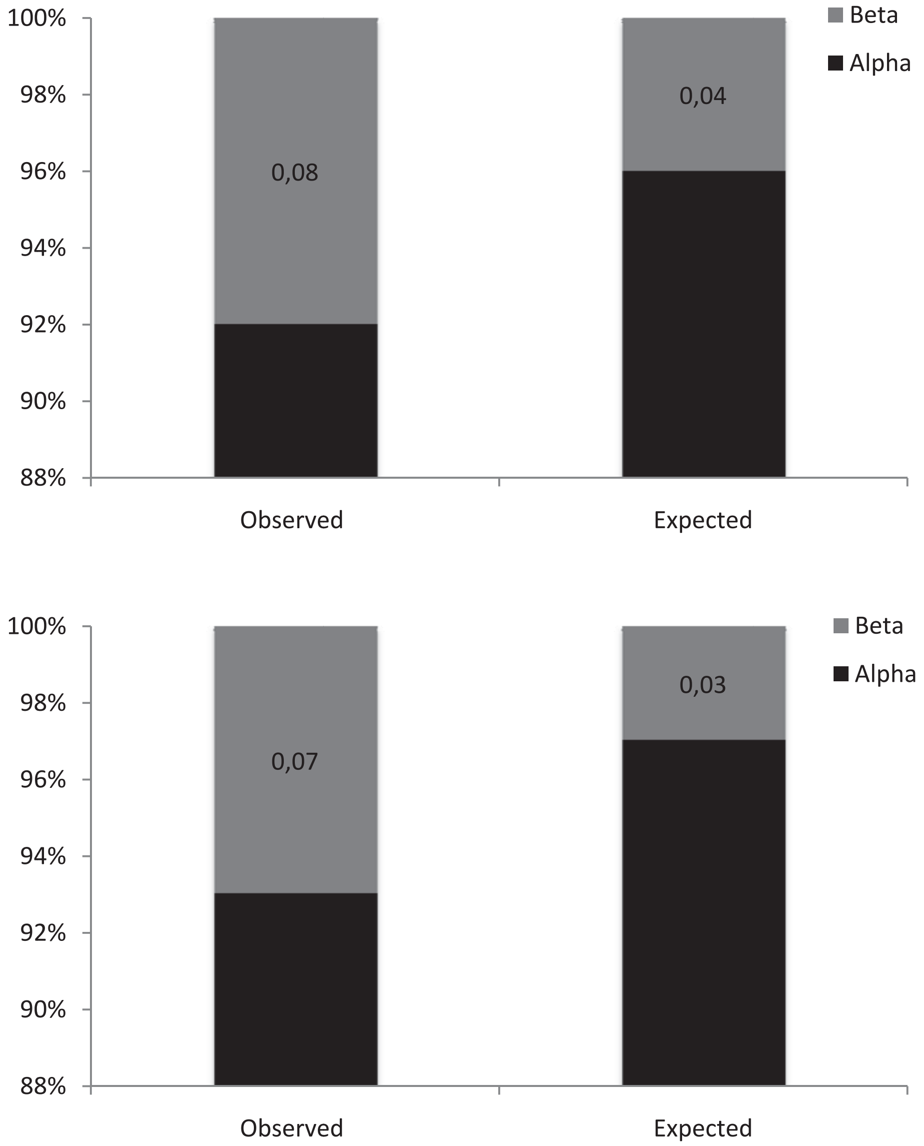

Figure 03:Additive partition of functional diversity for functional types using q=1 (all functional types have a weight proportional to their relative abundances). (a) Additive partition of functional diversity for sapling layer. (b) Additive partition of functional diversity for adult layer. Alpha is the average diversity within the plots. Beta is the average diversity absent in the plots. Beta2 is the average diversity absent in the fragments. The sum of alpha and beta components results in gamma diversity of each fragment. The sum of alpha, beta, and beta2 components results in gamma diversity of the region. Different letters indicate observed ecological values significantly different than the expected ecological value if the distribution were random.

Considerações finais

Considerando que a maior parte da biodiversidade tropical está localizada em fragmentos florestais inseridos em paisagens antropizadas, os objetivos principais desse estudo foram verificar: (i) se fragmentos florestais inseridos em paisagens antropizadas ainda preservam alta diversidade de espécies, e por isto têm valor ecológico ecológico para a conservação; (ii) como essa diversidade está organizada (partição da diversidade), e a trajetória de mudança na composição de espécies (homogeneização ou diferenciação florística); (iii) se os fragmentos são capazes de manter essa diversidade ao longo dos anos e; (iv) como a composição da paisagem afeta as comunidades remanescentes. Nós encontramos que mesmo fragmentos localizados em paisagens com baixa cobertura florestal e inseridos em matrizes antrópicas podem ter alta riqueza e diversidade de espécies (capítulo 1), assim como espécies exclusivas, raras e ameaçadas. Mesmo apresentando alguns sinais de perturbação, como alta densidade de indivíduos finos e baixos, nós acreditamos que, considerando apenas a composição florística, esses fragmentos têm alto valor ecológico ecológico para conservação.

Além disso, verificamos que esses fragmentos apresentam alta diversidade beta (capítulo 2), contrariando as expectativas da literatura. De acordo com trabalhos recentes, era esperado que fragmentos pequenos, isolados e inseridos em matrizes antrópicas estivessem passando por um processo de homogeneização florística (baixa diversidade beta). Nós encontramos, também, que a variação na diversidade beta entre os fragmentos é devida à substituição de espécies ("turnover"), e não ao aninhamento. Em outras palavras, está havendo uma troca de espécies nas áreas, e não uma perda ordenada de espécies mais sensíveis. Nós atribuímos esses resultados surpreendentes a dois fatores: (i) as áreas ainda mantêm certa heterogeneidade ambiental que mantém o "turnover" de espécies e; (ii) o tipo de manejo desenvolvido nas matrizes da região é baixo e ajuda a manter a diversidade dos fragmentos. Esses resultados foram encontrados, tanto para plantas de dossel, quanto para as de sub-bosque, oferecendo indícios de que os fragmentos são capazes de manter alta diversidade ao longo dos anos.

Testamos, também, o efeito da composição da paisagem sobre os atributos das comunidades de árvores adultas e regenerantes (capítulo 3), com o objetivo de verificar se essas áreas são capazes de manter a diversidade ao longo dos anos e como a composição da paisagem afeta os fragmentos florestais. A matriz de pasto aumenta a diversidade, riqueza e abundância de indivíduos regenerantes nos remanescentes. Atribuímos esse resultado ao baixo manejo desenvolvido nas áreas de pastagens da região, isto é, nas pastagens da região há poucos animais, pouca mecanização e baixo uso de agrotóxicos. Diante disso, nós acreditamos que mais importante que o tipo de matriz, é o tipo de manejo desenvolvido nas áreas. Conclusões importantes podem ser tiradas desses resultados: (i) indivíduos adultos não são bons indicadores dos efeitos da fragmentação sobre as comunidades de plantas; (ii) a fragmentação nem sempre leva a homogeneização florítica; (iii) tipo de manejo parece ser mais importante que o tipo de matriz e; (iv) a composição de espécies futura será diferente, e possivelmente a alta diversidade de espécies gama será mantida.

Com base nesses resultados, nós poderíamos concluir que as áreas são promissoras para a manutenção da biodiversidade. No entanto, nós também buscamos entender como varia a diversidade funcional desses ambientes e como esta é afetada. Infelizmente, para esses dados, nossos resultados são preocupantes. Os indivíduos adultos apresentam baixa diversidade beta e a comunidade apresenta uma estrutura aninhada. Esses resultados indicam que para comunidade adulta está havendo uma perda ordenada de funções mais sensíveis, e um aumento na abundância das funções mais tolerantes. Em outras palavras, as funções das plantas nos fragmentos menos diversos são um subconjunto das funções das plantas nos fragmentos mais diversos. Além disso, esses últimos fragmentos estão dominados por plantas que exercem funções mais tolerantes. Já o estrato regenerante apresenta menor diversidade funcional que o estrato adulto e alta abundância de indivíduos em funções mais tolerantes. E essas funções, estão sendo beneficiadas pela porcentagem de matriz de pasto ao redor dos fragmentos.

Em outras palavras, quando analisamos apenas a diversidade e riqueza de espécies, os fragmentos apresentam um bom estado de conservação. Entretanto, quando analisamos a diversidade funcional verificamos que, apesar de haver muitas espécies, essas desempenham as mesmas funções mais tolerantes. Acreditamos que se nada for feito, com o intuito de recuperar a diversidade funcional das áreas, o futuro desses fragmentos é incerto. Uma medida minimizadora seria o plantio direto de árvores com funções mais sensíveis.

Entre as contribuições do nosso estudo podemos destacar: (i) informações sobre a flora regional; (ii) aumento do conhecimento sobre a distribuição geográfica e abundância das espécies; (iii) referência para trabalhos futuros e mais aplicados; (iv) importância da amostragem do sub-bosque; (v) demonstrar o potencial dos pequenos fragmentos para conservação da biodiversidade; (vi) melhorar o entendimento sobre o processo de homogeneização florestal em fragmentos florestais tropicais; (vii) mostrar a importância da heterogeneidade intrínseca e do contexto da paisagem para manter a alta diversidade em áreas fragmentadas; (viii) melhor entendimento dos processos responsáveis pela variação da beta diversidade entre remanescentes florestais. Uma abordagem muito importante para analisar os processos ecológicos que organizam as comunidades que não foi considerada nesse estudo devido as dificuldades metológicas, é a diversidade filogenética. Dessa forma, sugerimos esta medida seja considerada em estudos futuros.

Além dessas contribuições, esse trabalho é um alerta. Futuros trabalhos sobre os efeitos da fragmentação não podem ser baseados apenas em diversidade de espécies e é necessário considerar o estrato regenerante. Caso esses dois aspectos não tivessem sido considerados no nosso trabalho, nós não conheceríamos o real estado de conservação das áreas. Mais estudos ainda precisam ser realizados para esclarecer questões como em que condições há homogeneização florística e funcional, como varia a diversidade beta em ambientes fragmentados e como a paisagem pode amplificar ou minimizar os efeitos da fragmentação. Sugerimos que trabalhos futuros considerem a intensidade do manejo como um fator importante para responder algumas dessas questões.

Anexos

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 **RIQUEZA E DIVERSIDADE DE ESPÉCIES E DIVERSIDADE FUNCIONAL DE COMUNIDADES DE ÁRVORES ADULTAS E REGENERANTES LOCALIZADAS EM FRAGMENTOS FLORESTAIS**, não infringem os dispositivos da Lei n.° 9.610/98, nem o direito autoral de qualquer editora.

Campinas, 30 de agosto de 2017

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