LUIZ FERNANDO SILVA MAGNAGO

FOREST FRAGMENTATION ON TREE COMMUNITIES, FUNCTIONAL DIVERSITY AND CARBON STORAGE IN A BRAZILIAN ATLANTIC RAIN FOREST

Tese apresentada à Universidade Federal de Viçosa, como parte das exigências do Programa de Pós-Graduação em Botânica, para obtenção do título de *Doctor Scientiae*.

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APROVADA: 15 de março de 2013.

Prof. Ernesto G. Reynaud Schaefer

Profª. Andreza Viana Neri

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(Orientador)

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BIOGRAFIA

LUIZ FERNANDO SILVA MAGNAGO, filho de Laerte José Magnago e Denise Silva Magnago, irmão de Luiz Claudio Magnago, nasceu em 25 de setembro de 1983, em Vitória, Espírito Santo.

Fez aulas de músicas com professores particulares, com ênfase em violão popular entre os anos de 1996 a 1999, passando para guitarra elétrica clássica em 1998 a 2001.

Em dezembro de 2001, concluiu o ensino médio no colégio Darwin: a evolução do ensino.

Em julho de 2005, graduou-se em Ciências Biológicas na Faculdade de Saúde e Meio Ambiente - FAESA.

Em março de 2007, ingressou no curso de Mestrado em Botânica, na área de concentração de Ecologia, com ênfase em Estrutura, Funcionamento e Manejo de Comunidades Vegetais, na Universidade Federal de Viçosa, Minas Gerais, Brasil, finalizando em Janeiro 2009.

Em março de 2009 ingressou no doutorado em Botânica também na UFV.

SUMÁRIO

RESUMO

MAGNAGO, Luiz Fernando Silva, D.Sc. Universidade Federal de Viçosa, março de 2013. A **fragmentação florestal em comunidades arbóreas, diversidade funcional e estoque de carbono na Floresta Atlântica Ombrófila no Brasil**. Orientador: Sebastião Venâncio Martins. Co-orientadores: William F. Laurance e Ainhoa Magrach.

A fragmentação das florestas tropicais é uma das maiores ameaças à biodiversidade global, uma vez que os efeitos após a fragmentação promovem alterações no meio abiótico e com consequências no meio biótico. Entre os efeitos abióticos estão o aumento dos distúrbios causados pelo vento e a dessecação microclimática e entre os efeitos bióticos podemos citar o aumento das taxas de mortalidade, mudanças na composição, estrutura e traços funcionais das espécies. Para investigar os efeitos da fragmentação na Floresta Atlântica focamos nas espécies arbóreas, tendo três objetivos gerais: (i) verificar os impactos da fragmentação nas mudanças abióticas (microclima e atributos do solo) na biomassa florestal acima do solo; (ii) verificar os impactos da fragmentação na riqueza, estrutura da comunidade e diversidade funcional de espécies arbóreas; e (iii) verificar a existência de co-benefícios entre biodiversidade e estoque de carbono para aplicação de mecanismos de conservação por meio do mercado de carbono (*Reducing Emissions from Deforestation and Forest Degradation* - REDD+). Nosso experimento foi desenvolvido em uma paisagem de floresta tropical brasileira conhecida como Florestas de Tabuleiro, onde o conhecimento sobre a fragmentação florestal ainda é incipiente. Amostramos 12 fragmentos de diferentes tamanhos (3 repetições/tamanho do fragmento) com 240 parcelas de 10mx10m, igualmente distribuídas entre borda e interior e entre quatro classes de tamanho de fragmentos, sendo pequenos, médios, grandes e controles. Em cada parcela nós coletamos dados sobre a riqueza de espécies arbóreas, estoque de biomassa acima do solo, estoque de carbono (estoque de carbono=biomassa/2), cipós e árvores mortas em pé, bem como dados sobre o microclima e atributos do solo. Nós classificamos as espécies quanto as suas características funcionais, endêmicas da Floresta Atlântica e ameaçadas de extinção (Lista Vermelha da IUCN). Os gradientes de dessecação (menores valores de umidade do ar e maiores valores de temperatura do ar) e aumento da velocidade do vento foram significativos e positivamente relacionados com a redução de tamanho do fragmento e com a criação de bordas, além disso, o habitat de borda apresentou um solo mais fértil e menos ácido. Os resultados também mostraram significativa redução da biomassa de árvores e um significativo aumento da biomassa de lianas

em habitats de borda e pequenos fragmentos, estando estes relacionados ás mudanças no microclima e no solo, o que indicou distúrbios na biomassa florestal. O habitat bordas promoveu mudanças marcantes na estrutura da comunidade de árvores e em suas características funcionais, reduzindo significativamente a riqueza de espécies e a diversidade funcional, de maneira tal que, os fragmentos maiores e o habitat de interior das florestas possuem maior potencial de fornecer recursos alimentares e interações com a fauna. Encontramos uma forte existência de co-benefícios entre a conservação da biodiversidade e estoque de carbono na paisagem fragmentada de floresta tropical. Além disso, esta relação de co-benefícios aumenta com o tamanho do fragmento, onde existe, significativamente, um maior estoque de carbono e significativamente mais espécies com elevado valor de conservação. Finalizando, temos conclusões notáveis sobre a fragmentação Florestas Tropicais do Brasil, sendo: (i) as mudanças no microclima e solo são afetadas pela fragmentação, promovendo um impacto negativo sobre a biomassa de espécies arbóreas e aumento da biomassa de lianas; (ii) em uma paisagem fragmentada a funcionalidade ecológica de espécies arbóreas existente em fragmentos maiores foram significativamente diferentes daquela existente em fragmentos pequenos; e (iii) o mecanismo REDD+ de co-benefícios pode ser utilizado em uma paisagem fragmentada, mesmo com um nível de fragmentação elevado, o que sugere que os fundos de REDD+ podem ser utilizados para beneficiar o estoque de carbono e o valor biológico dos fragmentos através de planos de manejo. No entanto, pequenos fragmentos têm um papel importante na manutenção dos serviços ecológicos, tornando-os indispensáveis para a conservação da biodiversidade, principalmente em um domínio fitogeográfico tão ameaçado quanto o da Floresta Atlântica.

ABSTRACT

MAGNAGO, Luiz Fernando Silva, D.Sc. Universidade Federal de Viçosa, march, 2013. **Forest fragmentation on tree communities, functional diversity and carbon storage in a Brazilian Atlantic Rain Forest**. Adviser: Sebastião Venâncio Martins. Co-advisers: William F. Laurance and Ainhoa Magrach.

The fragmentation of tropical forests is one of the greatest threats to global biodiversity, promoting both abiotic and biotic changes. Among the abiotic effects are the increased disturbance caused by wind and microclimatic desiccation whilst among the biotic effects there are increases in mortality rates, changes to species composition, forest structure and functional traits of the species. To investigate the effects of fragmentation in the Atlantic Forest we focus on tree species, with three main objectives: (i) to verify the impacts of fragmentation on the abiotic environment (microclimate and soil attributes) and on above ground forest biomass; (ii) to assess the impacts of fragmentation on richness, community structure and functional diversity of tree species; and (iii) to evaluate the existence of co-benefits between biodiversity and carbon stocks in order to implement conservation mechanisms through the carbon market (Reducing Emissions from Deforestation and Forest Degradation - REDD +). We conducted our experiments in a Brazilian Tableland Forest fragmented landscape , where the knowledge about forest fragmentation is still incipient. We sampled 12 fragments of different sizes (3 replicates/fragment size) with 240 10mx10m plots, equally distributed between edge and interior areas and in each of four fragment size classes: small (≤50 ha), medium (51-250 ha), large (250-1,500 ha) and control (≥10,000 ha). Inside each plot we recorded tree species richness, above ground biomass , carbon stocks (carbon stock=biomass/2), liana abundance and abundance of standing dead trees jointly with microclimate and soil attributes measurements . We also classified the species in relation to their functional traits, Atlantic Forest endemic character and level of threat (IUCN Red List). The gradients of desiccation (less air humidity and more air temperature) and increases of wind speed were significant and positively related with reductions in fragment size and edge habitat creation, moreover the edge habitat had more fertile soil and less acid soils. The results also showed significant reductions of tree biomass and an increase in lianas biomass in edge habitats and small fragments, following the microclimate and soil changes, both indicators of disturbance in forestry biomass. Edge habitats promoted remarkable changes in tree community structure and functional traits, significantly reducing species richness and functional diversity, with larger fragments and forest interiors having more potential to provide food resources and interactions with fauna. We found that biodiversity and carbon stock were highly spatially congruent in our study area. Also, these co-benefit relationship increased with fragment size, where significant increases in carbon stocks are coupled with species of high conservation value. Finally our results lead us to the following noteworthy conclusions about Brazilian Rainforest fragmentation: (i) microclimate and soil changes driven by fragmentation promoted negative impacts on tree biomass and an increase in liana biomass; (ii) in a fragmented landscape plant functionality of larger fragments was significantly different to that of smaller fragments; (iii) the REDD+ co-benefits can be used in fragmented landscape, even subjected to high fragmentation levels, suggesting that additional REDD+ funds could be used to enhance the carbon and biological value through the management of fragmented landscapes. Nonetheless, small fragments have an important role in the maintenance of ecological services making them indispensable to conservation of biodiversity within the highly threatened Atlantic forest biome.

I. Introdução Geral

As florestas tropicais são reconhecidas como o mais importante repositório da biodiversidade mundial (Ayres *et al*. 2005). Cobrindo apenas 7% da superfície terrestre, abrigam mais da metade das espécies biológicas do planeta (Myers 1997), sendo a maioria completamente desconhecida pela ciência. Há estimativas de que mais de 200.000 km^2 de florestas tropicais são destruídas por ano (Myers 1997), o que representa uma inestimável perda de diversidade biológica, principalmente quando se trata de florestas que ainda permanecem em seu estado primário (Gibson et al. 2011). Estas florestas têm diferentes funções na regulação climática, como a de sequestrar e estocar carbono da atmosfera em sua biomassa, tendo assim, significativas influências na regulação climática, que vão desde escalas locais até mundiais (Laurance 2004; Laurance et al. 2011).

A fragmentação das florestas tropicais é citada como uma das principais ameaças a biodiversidade de espécies e à funcionalidade ecológica dos ecossistemas (Pardini et al. 2010; Gibbs et al. 2010; Pütz et al. 2011). O processo de desmatamento, que transforma uma paisagem de florestas contínuas em uma paisagem de mosaicos de fragmentos florestais de diferentes tamanhos e geralmente imersos em matrizes antrópicas diferenciadas (Bennett & Saunders 2010), atua primeiramente na destruição do habitat, acarretando em uma perda imediata de espécies, de funcionalidade ecológica e da biomassa estocada. Posteriormente ao processo de fragmentação *per se* (veja Fahrig 2003), os efeitos de bordas trazem mais perdas de biodiversidade e biomassa, devido às alterações no meio abiótico (promovidos pelo vento, dessecação e mudanças na ciclagem de nutrientes), sendo estas seguidas por mudanças no meio biótico, como o aumento das taxas de mortalidade e proliferação de espécies tolerantes a luminosidade nos fragmentos remanescentes (Laurance et al. 2006; Haper et al. 2005; Pütz et al. 2011).

Estudar as alterações na riqueza e composição de espécies em função dos efeitos da criação de bordas e/ou da redução dos tamanhos dos fragmentos remanescentes têm sido um dos temas mais importantes em estudos sobre a fragmentação em florestas tropicais (Laurance et al. 2002; Tabarelli et al. 2010; Pardini et al. 2010; Pütz et al. 2011). Nesse contexto, a maior parte dos estudos com biodiversidade foi baseada na identidade taxonômica das espécies, revelando muitas informações sobre as interações entre as espécies e as mudanças do ambiente, sendo muito utilizados para indicar áreas com relevante importância para conservação (e.g. Myers 1988; Hill et al. 2003; Edwards et al. 2011). No entanto, as informações baseadas na identidade taxonômica muitas vezes mostram-se incompletas para demonstrar as mudanças na biodiversidade em relação ao ambiente, pois eles não levam em conta a identidade biológica e as diferenças funcionais entre as espécies, sendo muitas vezes insuficientes por si só para explicar os processos ecossistêmicos (Villéger et al. 2010). Como o estudo da fragmentação pode ser muito complexo, pois lida com um variedade imensa de variáveis ambientais, advindas das mudanças microclimáticas, no solo, na dinâmica florestal, do histórico de uso da paisagem e das interações biológicas (Murcia et al. 1995; Laurance et al. 2002; Fahrig 2003; Harper et al. 2005), o uso de metodologias que podem nos ajudar a reconhecer o papel funcional das espécies no ambiente é imprescindível (Chapin 2003).

Desta forma, as análises de diversidade funcional tem sido utilizadas em artigos recentes para determinar as respostas das funções ecossistêmicas das assembléias de espécies as mudanças ambientais (Cadotte et al. 2011), tendo mostrado resultados interessantes na descrição de distúrbios ambientais na funcionalidade ecossistêmica (Villéger,et al. 2010; Pakeman et al. 2011; Baraloto et al. 2012). As respostas obtidas pelas análises com os índices de diversidade funcional em relação às variações no ambiente tem sido mais expressivas que a dos os índices que descrevem a diversidade de espécies (Loreau et al. 2001), isso devido as diferenças na funcionalidade atribuídas a cada espécie (Petchey & Gaston 2002), o que determina qual a função de uma dada espécie dentro da comunidade.

As florestas tropicais mantêm uma alta produção de biomassa acima do solo pela vegetação, podendo contribuir para até um terço da produtividade primária líquida dos ecossistemas terrestres (Field et al. 1998), tendo assim uma importância ecológica imprescindível para manter o ciclo de carbono do planeta (Keeling & Phillips, 2007). Entretanto, a elevada taxa de desmatamento das florestas tropicas (veja Gibbs et al. 2010), libera esse grande estoque de carbono contido na biomassa na forma de gases estufa para atmosfera (Laurance 2006), trazendo consequências negativas e provavelmente irreversíveis para o clima do planeta (Solomon et al. 2009).

Desta forma, estudos que visam investigar como os impactos da fragmentação atuam na perda de biomassa são de suma importância, visto que biomassa florestal é considerada uma importante abordagem ecológica para a caracterização de um ecossistema florestal, já que a eficiência do armazenamento de carbono na matéria orgânica reflete a qualidade das condições ambientais existente em fragmentos remanescentes, tais como o clima e os atributos do solo (Chave et al. 2001).

Com objetivo de minimizar os efeitos dos gases estufa nas mudanças climáticas o mecanismo REDD (*Reduced Emissions from Deforestation and Degradation*) propõe que os países inseridos nas florestas tropicais seriam compensados por reduzirem suas taxas de desmatamento, e assim diminuir as emissões de gases de efeito estufa (Grainger et al. 2009). O mecanismo REDD evoluiu e passou a ter em seu escopo, a conservação de biodiversidade por meio da proteção do estoque de carbono (co-benefícios), sendo então designado como REDD+ (Grainger et al. 2009). Desta forma, a integração desses co-benefícios é atualmente o centro dos esforços das ações conservacionistas mundiais (Phelps et al. 2012).

Por ser um tema relativamente recente, essa relação de co-benefícios ainda carece de comprovação empírica, impossibilitando a aplicação confiável dos conceitos do REDD+ para conservação da biodiversidade por meio da proteção de florestas com potencial estoque de carbono (UNEP-WCMC 2008; Diaz et al. 2009; Talbot 2010; Phelps et al. 2012). Isto tem resultado na aplicação desse mecanismo com benefícios apenas para áreas com interesse para o estoque de carbono e não para a biodiversidade (Lindenmayer et al. 2012).

Quando pensamos em investigar a viabilidade teórica dos co-benefícios em paisagens fragmentadas, temos que considerar indubitavelmente, a importância dos efeitos ecológicos da fragmentação, associados principalmente à redução no tamanho dos fragmentos remanescentes e à criação do habitat de borda, já que a maior parte dos remanescentes florestais tropicais está impactada pelos efeitos do desmatamento (Gibbs et al. 2010).

Entre os ecossistemas mundiais que são considerados prioritários para conservação da biodiversidade está o *hotspot* de Floresta Atlântica (Myers et al. 2000). Quando pensamos na Floresta Atlântica brasileira as primeiras coisas que lembramos são: a elevada riqueza de espécies que se pode encontrar e o quão deflorestado está esse domínio fitogeográfico. Não é para menos que pensemos assim, pois na Floresta Atlântica é possível encontrar mais de 380 espécies arbóreas em apenas um hectare de floresta (Saiter et al. 2011), e devido ao deflorestamento toda essa riqueza de espécies está confinada a apenas 11.26% de cobertura florestal remanescente, com 80% dos fragmentos menores que 50 hectares (Ribeiro et al. 2009). Desta forma essa riqueza frequentemente ocorre em paisagens severamente fragmentadas, a ponto de impactar severamente a riqueza de espécies existentes nos fragmentos remanescentes (e.g. Pardini et al. 2010).

Contudo, na Floresta Atlântica, principalmente nos trechos mais próximos ao litoral, ainda existem paisagens florestais fragmentadas que podem apresentar uma biodiversidade de relativo valor para conservação (espécies ameaçadas de extinção e endêmicas) e ainda um elevado potencial para estocar carbono na biomassa vegetal (veja Rolim et al. 2005), configurando um elevado potencial para aplicações de mecanismos conservacionistas, e.g. REDD+ (Strassburg et al. 2010).

Focamos o nosso estudo na biodiversidade de espécies arbóreas, na funcionalidade ecológica existente nos fragmentos, na biomassa acima do solo e no estoque de carbono em uma paisagem de Floresta Atlântica, tendo como objetivos gerais: (i) verificar os impactos da fragmentação nas mudanças abióticas (microclima e atributos do solo) e na biomassa florestal acima do solo; (ii) verificar os impactos da fragmentação na riqueza e estrutura da comunidade de espécies arbóreas, bem como nas mudanças dos traços e diversidade funcional; e (iii) avaliar a existência de co-benefícios entre biodiversidade e estoque de carbono para aplicação de mecanismos de conservação por meio do mercado de carbono (REDD+).

Para isso selecionamos uma paisagem fragmentada de Floresta Atlântica de Tabuleiro no norte do Espírito Santo, Sudeste do Brasil. Essa paisagem apresenta uma elevada relevância para conservação devido à presença de dois fragmentos com tamanho acima dos 20.000 hectares, que representam apenas 0.08% dos remanescentes florestais existentes nesse bioma no Brasil (Ribeiro et al. 2009). As Florestas de Tabuleiro estudadas ainda são reconhecidamente detentoras de uma elevada diversidade de espécies vegetais e animais (Peixoto & Silva 1997; Chiarello et al. 1999; Masden et al. 2001).

Para melhor compreensão e atendimento dos objetivos propostos, a presente tese de doutorado foi dividida em três capítulos. Desta forma, cada capítulo traz em detalhes as informações sobre a área de estudo e metodologias aplicadas para coleta e tratamento dos dados.

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III. CAPÍTULO I

FOREST FRAGMENTATION EFFECTS DECREASE TREE BIOMASS AND INCREASE LIANA BIOMASS

ABSTRACT – Forest biomass has an important role on the maintenance of the carbon cycle, with the impact of fragmentation effects being considered an important ecological issue. Thus, we aim to study the existence of variations on microclimate and soil attributes in order to understand how changes in these abiotic resources can impact the biomass of trees and lianas. We conducted our experiment in a fragmented landscape of Tableland Atlantic Rain Forest, where the knowledge about forest fragmentation is still incipient. Data were obtained from 240 10x10m plots . Plots were equally distributed in 12 fragments classified into four size classes (small, medium, large and control) and in both the edges and the interior habitats of these fragments. We measured above ground biomass of trees and lianas, as well as several microclimatic variables (maximum air temperature, relative humidity and maximum wind speed) and soil attributes (phosphorous, bases sum, pH in H_2O and organic matter) on each plot. We sampled a total of 4,140 tree individuals and 8,236 liana individuals. The most parsimonious models showed that gradients of desiccation (low air humidity and high air temperature) and the increase of wind speed are positively related to the creation of edges and to the reduction in fragments size. Moreover, edge habitats presented the highest nutrient status and less acidity. Models also showed a significant reduction in tree biomass and an increase in liana biomass in edge habitats and small fragments following changes to microclimate and soil attributes. These results indicate that forest fragmentation leads to the disturbance of forest biomass. Thus, we concluded that changes to microclimate and soil attributes due to forest fragmentation promote negative impacts on tree biomass and an increase in liana biomass, leading to an overall decrease in above ground biomass in forest fragments.

Keywords: Carbon cycle; Biomass; Microclimate changes; Air Temperature; Desiccation; Soil fertility.

Introduction

Forest biomass is considered an important ecological approach to characterize a forest ecosystem due to the efficiency of carbon stored in organic matter to reflect the quality of environmental conditions, such as climate and soil attributes (Chave et al. 2001). Tropical forests present a high production of above ground biomass by vegetation, which can contribute to more than a third of the net primary productivity in terrestrial ecosystems (Field et al. 1998). This demonstrates the ecological importance of these forests in maintaining the global carbon cycle (Keeling & Phillips 2007).

In forest ecosystems, trees are responsible for more than 90% of the above ground biomass production (Laurance et al. 1997; Chave et al. 2005). Although, lianas contribute less than trees to the total forest biomass, they are the second highest contributors to wood biomass stocks in tropical forests ecosystems (Laurance et al. 1997; Chave et al. 2005).

Forest fragmentation promotes several abiotic changes in forest fragments, which can modify their biological functionality (Murcia 1995; Laurance et al. 2002). The reduction in fragment size and the creation of edge habitats can lead to an increase in the impacts of intense light conditions, high air temperature, reductions in air humidity, high wind exposure and to modifications on soil attributes (Kapos 1989; Chen et al. 1993; Camargo & Kapos 1995; Turton & Freiburger 1997; Chen et al. 1999; Culley et al. 2000; Laurance et al. 2002). Moreover, the response of trees and lianas to these changes in the surrounding abiotic conditions is different.

Tree biomass, especially that provided by big trees, is negatively influenced by climatic desiccation events, increases in wind speed, and high temperature (Rolim et al. 2005; Briant et al. 2010; Laurance 2012), with these impacts being intensified in fragmented forests (Laurance at al. 1997; Nascimento & Launrance 2004; Briant et al. 2010; Pütz et al. 2012). On the other hand, lianas show a higher abundance and biomass under abiotic stress conditions, like an increase in light intensity and soil fertility (Schnitzer & Bonger 2002), which makes this group more competitive than trees in fragmented tropical forests (Laurance et al. 2001). In fragmented ecosystems, lianas present a strong competition with trees, which usually increases rates of tree felling and limb breakage (Lowe and Walker 1977, Putz 1980, 1984), reducing forest biomass (Laurance et al. 2001; Schnitzer & Bonger 2002). Thus, we aim to study the existence of variations in microclimate and soil attributes, and the response of tree and liana biomass to these abiotic changes.

Our experiment was conducted in a landscape of Tableland Atlantic Rain Forest, which shows great potential to store biomass (Rolim et al. 2005; Strassburg et al. 2010) and to conserve biodiversity (Peixoto & Silva 1997; Chiarello 1999; Marsden & Whiffin 2003). However, how microclimatic parameters such as soil attributes, and biomass stocks are related has never been studied. To evaluate how the impacts of forest fragmentation on this landscape affect this relationship, we tested two hypotheses: (i) soil attributes and microclimate vary across fragmentation gradients (fragment size reduction and edge creation); (ii) tree and liana biomass change following the abiotic gradients of fragmentation.

Material and Methods

Study area

This study was carried out in the state of Espírito Santo, in Southeast Brazil. We focused on the municipalities of Sooretama, Linhares and Jaguaré (19^{o}04'05 "S and 39^{o}57'35" W, 28- 65 m.a.s.l) (Figure S1), which contain a landscape matrix composed mainly by grasslands and plantations of *Eucalyptus* spp., coffee and papaya (Rolim et al. 2005). Climate is tropical wet (Köppen classification) with an annual precipitation of 1,403 mm and a distinct dry season from May to September, when precipitation is only 33 mm per month (Peixoto & Gentry 1990). Predominant soil in the study region is Yellow Podzolic (IBGE 1987) with a low fertility due to low concentrations of exchangeable bases (Garay et al. 2004). This region is part of the phytogeographic domain Atlantic Forest and is officially classified as Lowland Rain Forest (IBGE 1987). However this ecosystem can also be called as Tertiary Tablelands Forest or just Tableland Forest (Peixoto & Silva 1997).

Tree sampling

Fieldwork was conducted from January 2011 to January 2012. We created permanent plots along transects on nine forest fragments differing in size (range=13.18 to 1318.26 ha; mean=333.9 ha) and on two control forests larger than 20,000 ha (Reserva Natural da Vale -

RNV, and Reserva Biológica de Sooretama - REBIO) (Table S1). We delimited two transects on each fragment: one approximately 5 m inside the fragment and parallel to the forest edge, and another in the interior of the fragment (≥300 m from the forest edge). Along each transect, we established ten 10 x 10 m plots located 20 m from each other, summing up 240 plots. Due to the absence of other control forests, we allocated one pair of transects in the RNV and two in the REBIO. Each pair was composed by 10 plots on the edge and 10 on the forest interior. The mean distance between transect pairs was 17.1 km (± 10.4). All plots were established on the same type of soil (Yellow Podzolic).

We sampled every living tree individual with a diameter ≥4.8 cm at breast height (DBH) measured at 1.3m above the ground. We also measured every liana larger than 1.6 cm at the height of 10 cm above soil height (DSH). Samples from each living tree were collected on all the plots. We identified trees according to references from the CVRD Herbarium of the Vale and the VIES Herbarium of the Federal University of Espírito Santo, and with the aid of taxonomic specialists for specific families (e.g. Myrtaceae and Sapotaceae). Botanical material collected in a fertile stage was deposited in the collection of CVRD Herbarium of the Vale, located in Linhares, ES.

Tree and liana biomass estimation

To estimate the amount of Tree Above Ground Biomass (AGBt) in each individual live and standing dead tree we used Chave et al.'s (2006) equation:

$$\mathrm{ABGt} = \mathrm{p}.\exp(-1.499 + 2.148\ln(\mathrm{Dt}) + 0.207(\ln(\mathrm{Dt}))^2 - 0.0281(\ln(\mathrm{Dt}))^3)$$

Where p = wood density (g/cm^3) and Dt = diameter at breast height (DBH).

For the Liana Above Ground Biomass (AGBl), was used Schnitzer et al.'s (2006) equation:

$$\mathrm{AGBl} = \exp(-1.484 + 2.657\ln(\mathrm{Dl}))$$

where Dl = DSH (lianas). We assume that 50% of AGB of each individual is represented by carbon (Laurance et al. 1997; Malhi et al. 2004; Chave et al. 2005; IPCC 2006; Paula et al. 2011). Thus the total carbon stock for each plot and each site was the sum of all individual components: total carbon stock = live tree carbon + dead tree carbon + liana carbon.

Tree species data for wood density on dry weight (g/cm^3) were obtained from *The Global Wood Density (GWD) database* in the subsection *Tropical South America* (http://hdl.handle.net/10255/dryad.235; Chave et al. 2009; Zanne et al. 2009). We made three adjustments (following Flores & Coomes 2011; Hawes et al. 2012): (i) for morphospecies only identified to the family or genus level, we used the average wood density of the taxonomic group; (ii) for species not in the GWD database, we used the average wood density for the species' genus; and (iii) for the standing dead trees individuals, we used the average wood density found for living trees in the same plot of each dead tree.

Microclimatic variables and soil sampling

To measure microclimatic variables we used two Kestrel 4,500 weather stations. Data of maximum air temperature (^{0}C), maximum wind speed (km/h) and relative air humidity (%) were collected in all sample plots. In order to standardize data collection among sample plots, all measures were recorded during 15 minutes at 1.5 m above the ground. Since there is a natural variation in microclimatic parameters among different days, we placed a Kestrel weather station in every fragment matrix. Thus, data collected at the edges and forest fragment interiors were standardized with the matrix values, which were considered as maximum values in order to minimize the effects of natural climate variability during the sampling days. Values obtained in each matrix were considered as 100%, being the percentage of increase and decrease of each microclimatic variable across edge and interior habitats calculated from the value obtained for its matrix (Table S3).

Three replicates of the top layer soil (0-10 cm) were collected in each sample unit for chemical analysis. Soil samples were mixed to form one sample per plot, totaling 240 samples. Samples were air-dried and sieved with a 2 mm diameter mesh, and analyzed in the Soil Analysis Laboratory, Department of Soils, Federal University of Viçosa (UFV). We analyzed available phosphorous, pH in H_2O, and base sum (SB=Ca+Mg+K), which were considered as soil fertility, and organic matter (Table S1).

Data Analysis

To investigate changes in microclimate and soil attributes due to fragmentation effects we created a global model which included the interaction between fragment size and habitat (edge and interior) for all fragments and controls. Moreover, to evaluate the relationship between tree and liana biomass and the changes in microclimatic and soil attributes we applied two models: (i) one considering fragments size, and including the interior of all fragments and controls, and (ii) another with the inclusion of all habitats, fragments and controls (global model).

We used the *glm* function from the R program in the model that just considered the size of the fragment. Mixed models were generated using the *lme* function from the *nlme* package. Random intercept models were estimated using maximum likelihood estimationsto allow comparisons between models. Each fragment was codified as a random variable in all analyses (Bolker et al. 2009). We used the *AICcmodavg* package to test all possible combinations of the variables included in the global model. However, to avoid multicollinearity between explanatory variables, we only considered variables with correlations (linear Pearson correlation) less or equal to 0.6 in each model (Table S2).

To determine the best models we used a theoretical information approach based on the Akaike Information Criterion of Second Order (AICc), indicated for small sample sizes. The best model was indicated by the lowest value of AICc (Burnham et al. 2011). The plausibility of alternative models was estimated by the differences in their AICc values in relation to the AICc of the most plausible model (ΔAICc), where a value of $\Delta\text{AICc}<2$ indicates equally plausible models (Tables S3, S4, S5). However, we only considered it as an important result when some variable of the model was significant ($p<0.05$). The Akaike weights (wi) express the relative likelihood of each model, in a scale of 0 to 1. All analyses were performed in the R version 2.15.1 (R Development Core Team 2012).

Results

Abiotic changes by fragmentation effects

Our models showed that fragmentation promotes significant changes in microclimate through reductions in fragment size and the creation of edges (Figure 1). Air temperature was significantly higher in small fragments (GLM; t=-3.06, p=0.01; Figure 1A) and edges (GLM; t=3.56, p<0.01; Figure 1B). On the other hand, air humidity showed a significant interaction with fragment size and habitat (GLM; t=-3.05, p=0.01; Figure 1C), showing a positively and significant influence on fragment interior (F=24.48, p<0.001) and edges as well (F=8.24, p<0.05). Air humidity also was significantly higher in the interior habitat (GLM; t=4.84, p=0.02; Figure 1D).

Wind speed showed a significant interaction with fragment size and habitat (GLM; t=2.28, p<0.05; Figure 1E), with a significant negative influence of fragments size for plots located at the interior of the fragments (F=8.79, p=0.01), but not at the edges (F=0.2, p=0.66). The model also indicated that wind incidence was significantly higher in edge habitats (GLM; t=5.06, p<0.001; Figure 1F).

Fragment size had no influence in our best model for soil attributes, but habitat was present in all of the best models. The creation of an edge habitat had a significant influence on soil fertility (GLM; t=3.22, p=<0.01; Figure 2A) and acidity - pH (GLM; t=2.45, p<0.05; Figure 2B), with soil in edge habitats being significantly more fertile and less acid than soils of interior habitats. However, the best models showed only a marginally significant influences of edge habitats on phosphorous (GLM; t=1.94, p=0.08) and organic matter (GLM; t=-1.94, p=0.08).

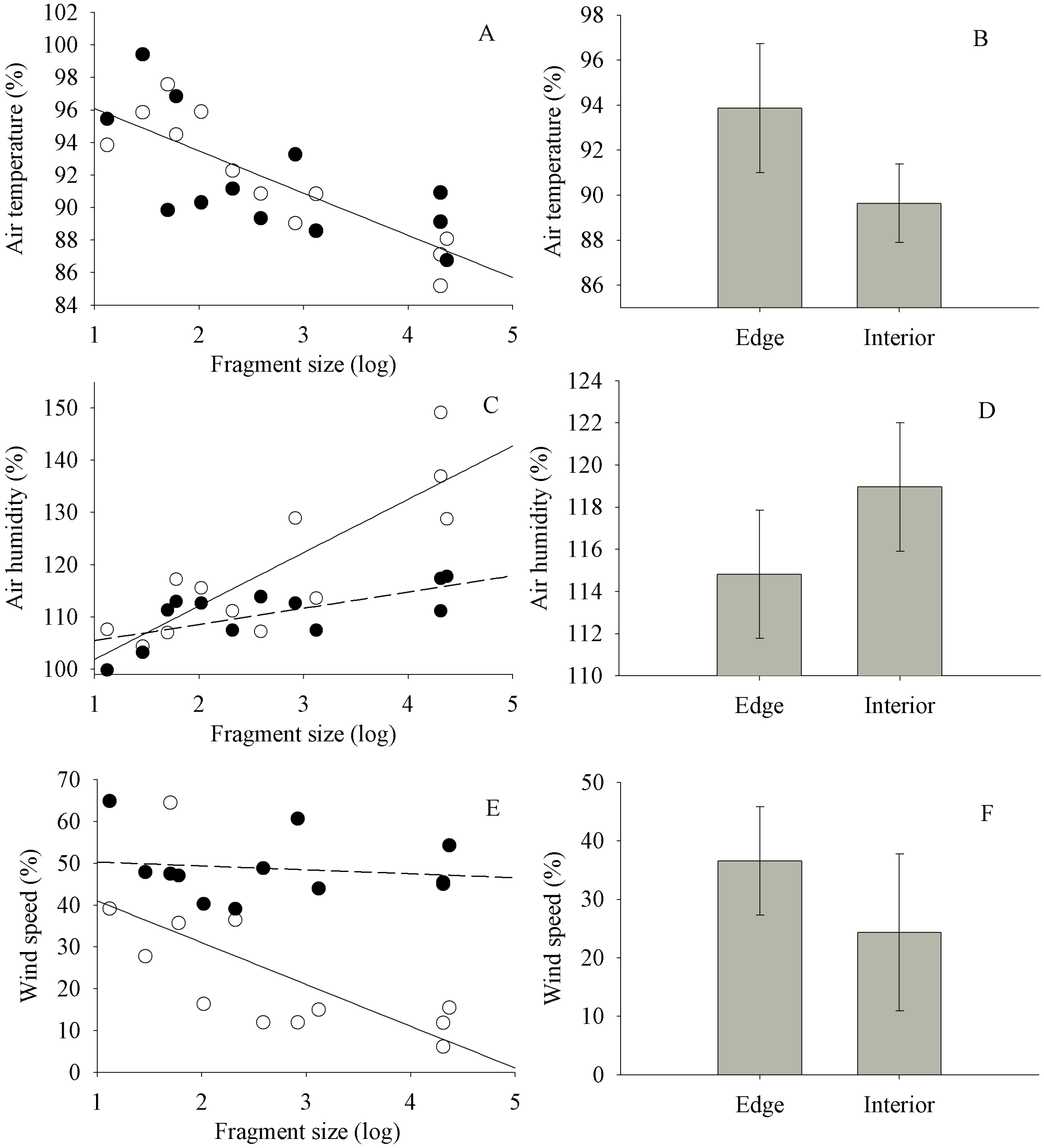

Figure 1 - Best model graphs for the effects of fragments size and habitats (global models) on microclimate variables. (A-B) Effects of fragment size and habitats on air temperature; (C-D) Effects of fragment size and habitat on air humidity; (E-F) Effects of fragment size and habitat on wind speed. Black circles = Edge; White circles = Interior. Circles represent values obtained

after the summation of raw residuals to the expected values for each variable, being assumed average values for other covariates.

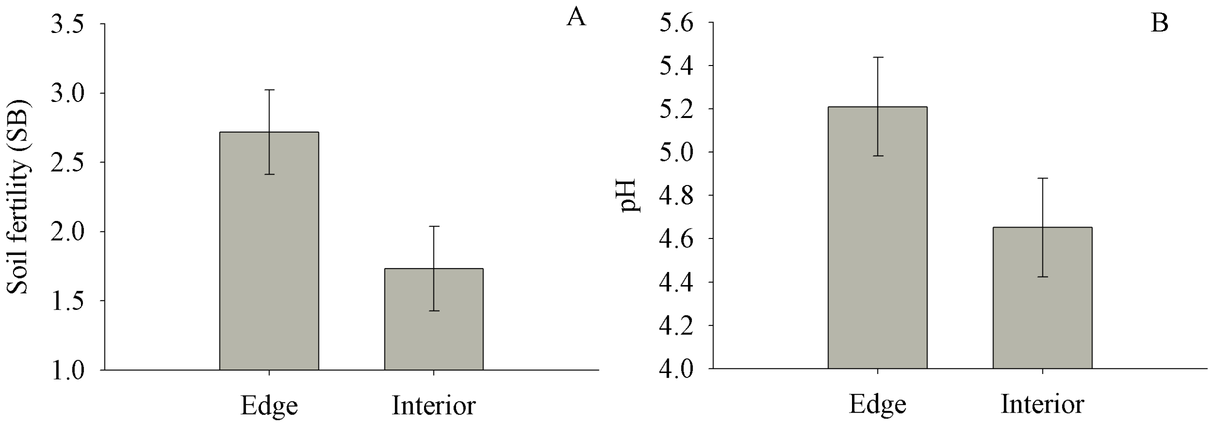

Figure 2 - Best model graphs generated for the results of habitat effect on soil attributes. (A) Habitat effect on soil fertility via bases sum (SB); (B) Habitat effect on soil acidity - pH in H_2O.

Microclimate changes impacting forestry biomass

A total of 4,140 tree individuals and 8,236 liana individuals were sampled during this study. We found that the biomass stored by trees was proportionally higher than the biomass stored by lianas in all fragments and habitats studied (Table S1). Considering only the fragment interiors, our best models indicated the highest tree biomass was found in the largest fragments where air humidity was high, and air temperature and wind speed were lower. Liana biomass increased in the areas with the lowest air humidity. We found that tree biomass was negative influenced by air temperature (GLM; t=-3.13, p=0.01; Figure 3A), showed a positive relation with air humidity (GLM; t=2.75, p=0.02; Figure 3B), and was negatively influenced by wind speed (GLM; t=-2.62, p<0.05; Figure 3C). Liana biomass was negatively influenced by air humidity (GLM; t=-2.67; p<0.05; Figure 3D). We did not find significant relationships between the biomass of trees and lianas and the soil attributes.

Considering the global models for microclimate changes, we found that tree biomass was negatively related to wind speed (GLM; t=-2.62, p<0.05; Figure 4A), but was not related to air temperature (GLM; t=-2.01, p=0.07) and air humidity (GLM; t=1.73, p=0.11). Liana biomass showed a positive relation with air temperature (GLM; t=3.57, p<0.01; Figure 4B) and a negative relation with air humidity (GLM; t=-3.55; p<0.01; Figure 4C). We did not find significant relationships between liana biomass and wind speed (GLM; t=1.39; p=0.19).

The best models for the relation between tree biomass and soil attributes selected the variables soil fertility (GLM; t=-1.61; p=0.13), soil acidity - pH (GLM; t=-0.94; p=0.37) and phosphorous (GLM; t=-0.83; p=0.42). But none of these variables showed significant relations with tree biomass. Nevertheless, liana biomass demonstrated significantly increases in the most fertile (GLM; t=3.14, p<0.01; Figure 5A) and less acid soils (GLM; t=2.81, p<0.05; Figure 5B).

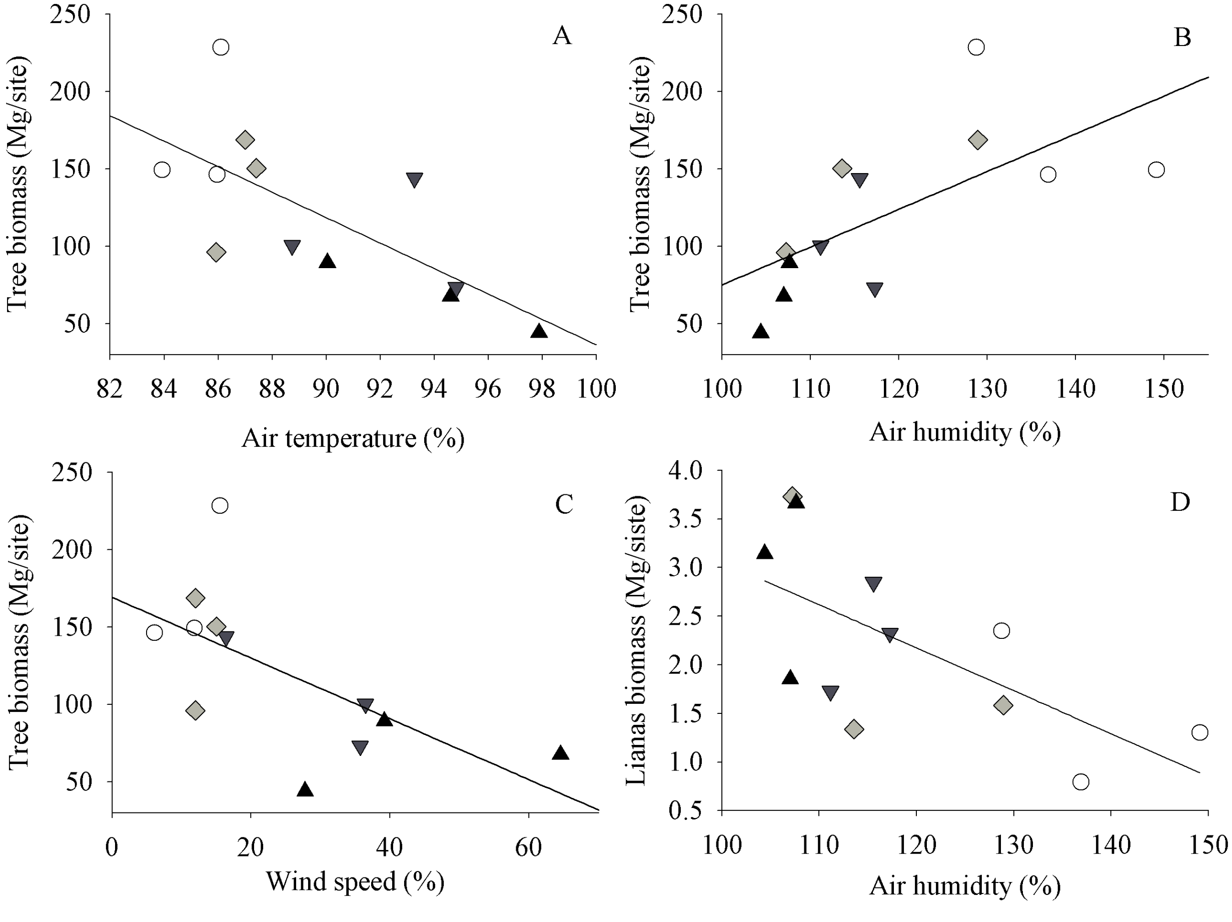

Figure 3 - Best model graphs for the effects of microclimate variables (models with fragment interiors) on the biomass of trees and lianas. (A) Effects of air temperature on tree biomass; (B) Effects of air humidity on tree biomass; (C) Effects of wind speed on tree biomass; (D) Effects

of air humidity on liana biomass. White circles = Control fragments; Gray diamonds = Large fragments; Inverse triangles = Medium fragments; Up-pointing triangles = Small fragments. All points represent values obtained after the summation of raw residuals to the expected values for each variable, being assumed average values for other covariates.

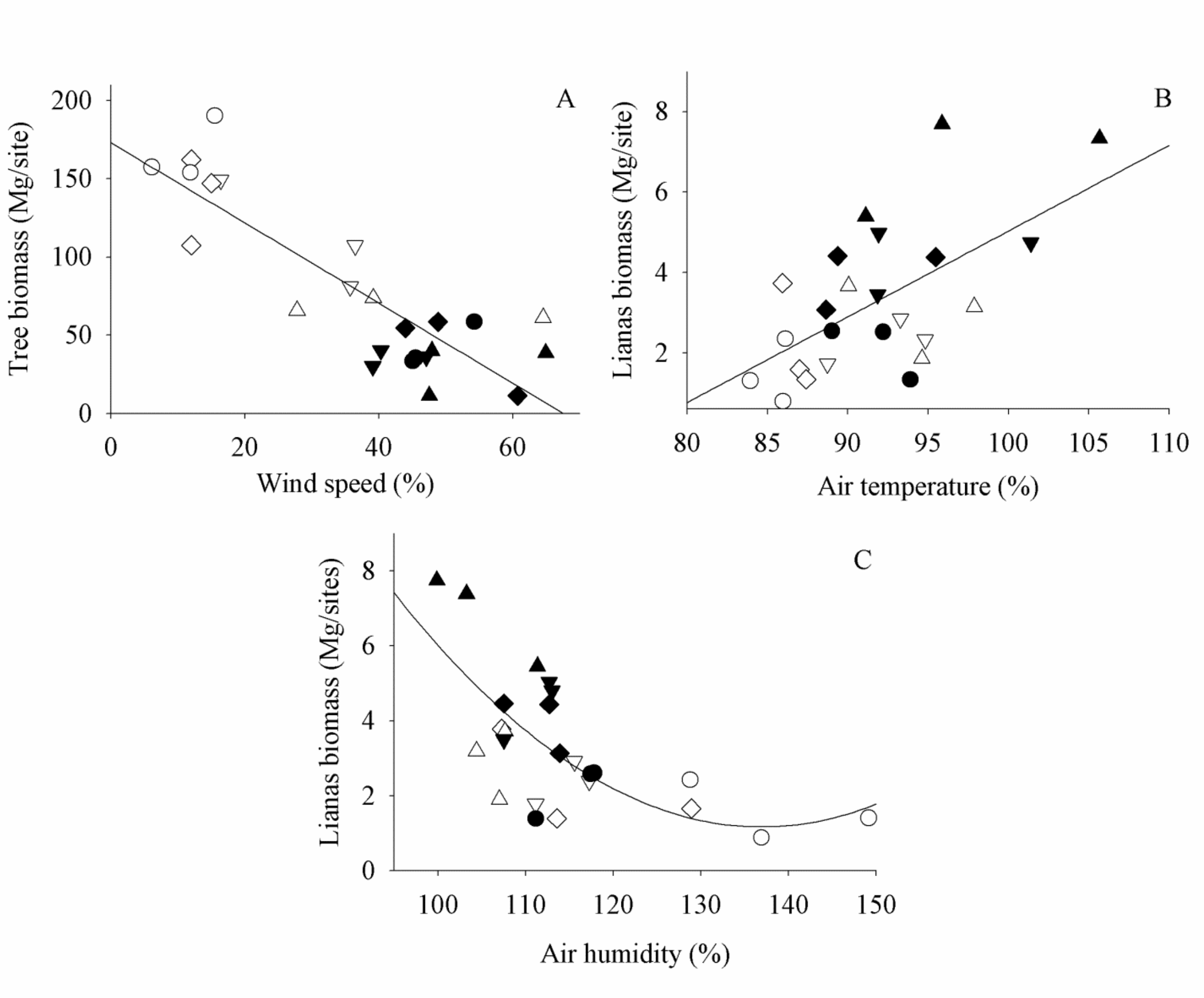

Figure 4 - Best model graphs for the effects of microclimate variables (global models) on the biomass of trees and lianas. (A) Effects of wind speed on tree biomass; (B) Effects of air temperature on liana biomass; (C) Effects of air humidity on liana biomass. Circles = Control fragments; Diamonds = Large fragments; Inverse triangles = Medium fragments; Up-pointing triangles = Small fragments; White geometric shapes= Fragment interiors; Black geometric shapes = Fragment edges. All geometric shapes represent values obtained after the summation of

raw residuals to the expected values for each variable, being assumed average values for other covariates.

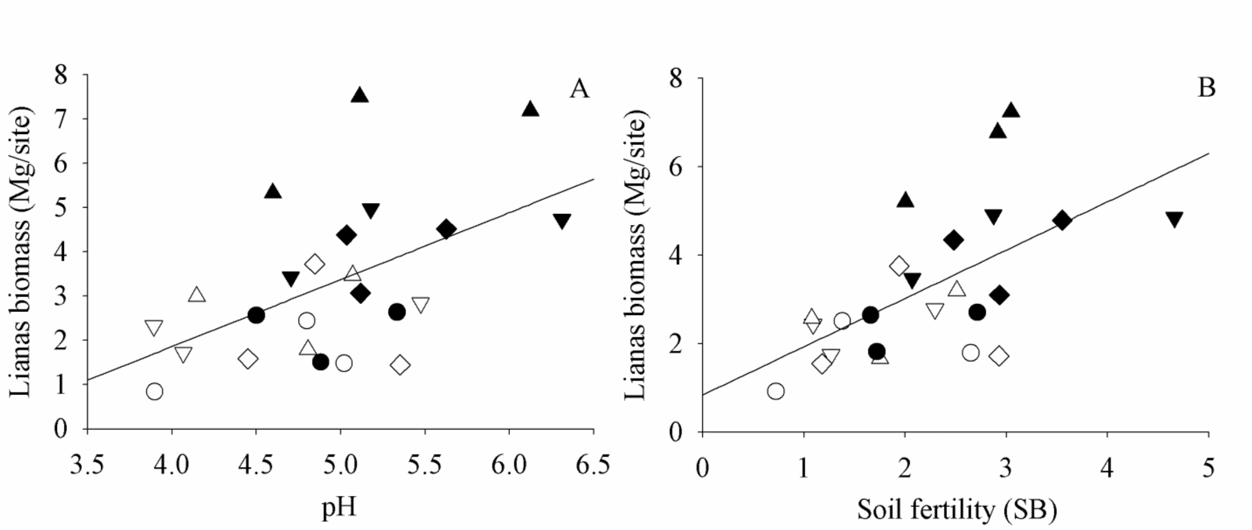

Figure 5 - Best model graphs for the effects of soil variables (global models) on liana biomass. (A) Effects of soil acidity (pH) on liana biomass; (B) Effects of soil fertility via bases sum (SB) on liana biomass. Circles = Control fragments; Diamonds = Large fragments; Inverse triangles = Medium fragments; Up-pointing triangles = Small fragments; White geometric shapes = Fragment interiors; Black geometric shapes = Fragment edges. All geometric shapes represent values obtained after the summation of raw residuals to the expected values for each variable, being assumed average values for other covariates.

Discussion

Abiotic changes in fragmented tropical forest fragments

Our results regarding abiotic changes in forest fragments in the tableland forest area sampled, showed significant differences in microclimatic conditions and soil attributes acorss the gradient of fragments sizes, and among edge and interior habitats. We also observed a positive relation between both gradients of desiccation (low air humidity and high air temperature) and an increase of wind speed, and the creation of edge habitats and the reduction in fragments size. Moreover, edge habitats presented the most fertile and less acid soils. These distinct features may be considered the main factors in understanding biomass changes in tropical forests due to fragmentation effects, since the changes in the microclimatic conditions and soil attributes have a direct effect on the structure and dynamics of vegetation (Laurance *et al*. 1998; Didham & Lawton 1999; Laurance *et al*. 2002; Fahrig 2003; Harper *et al*. 2005).

The differences in microclimatic variables across the fragmentation gradients we found show that the smallest fragments and edges were the most impacted by fragmentation effects (Kapos 1989; Chen *et al*. 1993; Camargo & Kapos 1995; Ramos & Santos 2006). According to Zhu et al. (2004), small fragments present smaller distances between their edges and interiors, resulting in more severe impacts here than in larger fragments. In agreement with these authors, we observed the values of wind speed and air humidity were more similar in edge and interior habitats in small fragments than in large fragments in our study (Figure 1C and 1E).

These results contrast with those found by Pinto *et al*. (2010) in an Atlantic Forest landscape, where no microclimatic differences were found between fragments of different sizes, edges and forest interior habitats. According to these authors, these changes can be minimized depending on the matrix type. In the landscape where our study was developed, fragments were limited by roads adjacent to *Eucalyptus* spp. plantations, pastures and other agricultural plantations such as coffee, papaya and banana. Thus, the diversity of agricultural matrices existent in this area may not provide a strong mitigation for the impacts arising from microclimatic changes (see Murcia 1995).

The results of our model showed that fragment size did not affect soil attributes, indicating that interior habitats have the same soil resources across the whole gradient of fragment sizes. However, edge creation had a significant impact on soil fertility and acidity. Other authors observed higher soil fertility and pH value near edges and disturbed areas in fragmented forests (Laurance et al. 2001; Zhu et al. 2004).

The increase in soil fertility and pH near edges can be related to some possible explanations, since our samples were collected in the same type of soil (see Material and Methods). First, given the high dynamism near forest edges promoted by the increase in fast growth plants (pioneer species), and by the high rates of trees mortality and turnover (Laurance et al. 1998; Laurance et al. 2002), a greater amount of nutrients returns to the top soil. Thus, soil nutrients can increase near edges due to the higher production of dead wood and leafs (Laurance et al. 2002). Second, the proximity to agricultural plantations where artificial fertilizers are usually applied can increase the amount of nutrients and the pH value through the increase in nutrients carried by air and water to the inside of the fragments (Selle 2007). This effect is probably greatest near the edges due to their proximity to the matrix, which makes them in general more susceptible to matrix effects than forest interiors (Laurance et al. 2011). Another possible explanation for the increasing of pH and bases sum near to the edges is by burming biomass transfering suplying ashes and burned OM to the top soil.

Biomass changes due to fragmentation effects

The great contribution of trees to above ground biomass followed by that of lianas seems to be common in tropical forests (Chaves et al. 2008), even in highly fragmented landscapes (e.g. Laurance et al. 1997; Nascimento & Laurance 2004). The significant reduction in tree biomass and the increase in liana biomass in edge habitats and small fragments, as well as the changes in microclimate and soil attributes (Figure 4 and 5), indicate disturbances in forest structure and biomass (see Laurance et al. 2001; Chaves et al. 2008).

The increase in lianas biomass and abundance promoted by significant changes in microclimate and soil attributes can interfere negatively with tree biomass due to the competition with tree species, resulting in an increase in tree mortality and impeding forest regeneration (Laurance et al. 2001). Moreover, lianas present morphological and physiological characteristics that can limit their potential to accumulate biomass (Schnitzer & Bongers 2002; Laurance et al. 1997). Thus, an increase in lianas abundance is generally associated to a decrease in trees biomass, resulting in significant reductions in total biomass and carbon stocks (see the Results of chapter 3).

Our results showed that changes in microclimatic values have the potential to promote deleterious effects on forest biomass (see Murcia 1995; Didham & Lawton 1999; Laurance et al. 2002). Tree biomass was influenced by the reduction in air humidity and the increase in wind speed and air temperature (see Results) in edges and small fragments. Reductions of tree biomass due to the wind speed are a classic association in fragmented landscapes (Laurance et al. 2000; Nascimento & Laurance 2004). Wind turbulences can impact forest structure even far from forest edges (Laurance et al. 1997), being the impact with the greatest potential to penetrate long distances inside the forest (Laurance et al. 2002), causing physical damage to the canopy, especially so for big trees due to their thicker and less flexible structure (Laurance et al. 2000; Laurance 2012).

We found a decreases in tree biomass due to desiccation was found across the gradient of fragment size (models considering only fragment interior, see Figure 3). Microclimate desiccations have a significant effect in the reduction of forest biomass (Briant et al. 2010) since these changes can promote a trade-off in functional traits (Tabarelli & Peres 2002; Laurance 2006). Edges and small fragments usually have a greater number of species adapted to develop under high light intensity conditions (and thus more desiccated; e.g.: pioneer tree species). These species show fast growth and low wood density, resulting in a reduced capability to incorporate high biomass stocks in fragments (Laurance et al. 2006). Therefore, microclimate desiccation favors the selection of species with physiological and morphological adaptations to survive and develop under low humidity and air temperature, which can result in a biomass loss.

Our results for liana biomass showed that this group increased in an opposite way than tree biomass when considering microclimate. Besides that, lianas were greatly influenced by soil attributes. An increase in liana expressiveness (abundance and biomass) is usually related to abiotic factors such as (i) an decrease in total rainfall and an increased in seasonality in macro scale samples; and (ii) an increase in soil fertility and on the disturbance level at local scales (see Schnitzer & Bongers 2002). Supporting these ideas we observed that liana biomass increased significantly in conditions of low air humidity and high air temperature, and with increases in soil fertility and pH. These results corroborate the findings of other studies in fragmented and disturbed forests, which show that lianas have a significant expressiveness in habitats with a higher desiccation impact and better soil conditions (Laurance et al. 2001; Schnitzer & Bongers 2002; Malizia et al. 2010).

In conclusion, our results support our two hypotheses. Microclimatic variables (wind speed, air humidity and air temperature) and soil attributes showed significant changes across fragmentation gradients (size reduction and edge creation). Tree and liana biomass were also significantly influenced by abiotic gradients of fragmentation. Summarizing, we found that changes in microclimate and soil attributes due to fragmentation effects can impact negatively tree biomass and favor an increase in liana biomass, which can store less carbon than trees in forest ecosystems. Thus, the impact of abiotic resources changes studied here can be the main factors to change the above ground biomass and carbon stored in tropical forests, promoting direct tree biomass losses impacting upon tree physiology functionality (less air humidity and high values of air temperature), in a physical way (increases of wind speed), and by the increase in competition between trees and lianas.

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SUPPLEMENTARY MATERIAL

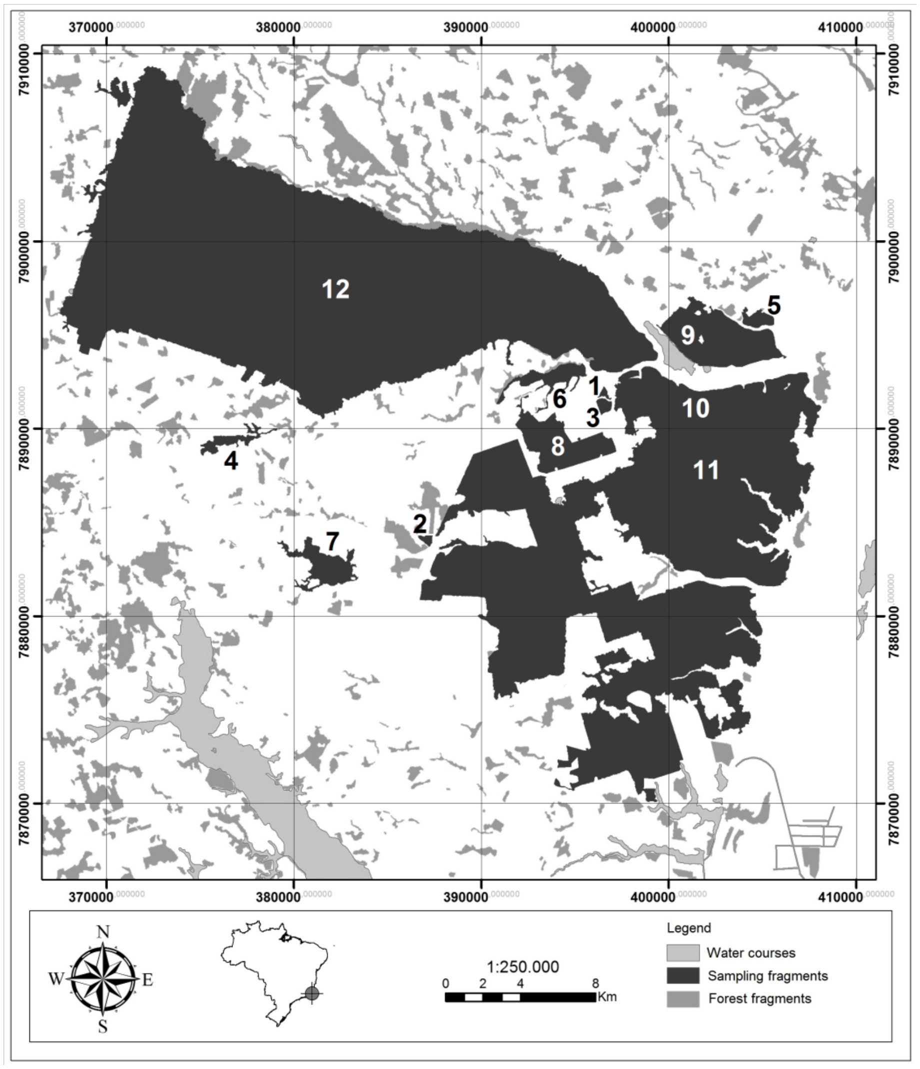

Figure S1- Study area and forest fragments sampled in Southeastern Brazil Brazil (Tableland Atlantic Rain Forest, Espírito Santo). To check the respective names and information about fragments see the Table S1.

Table S1 - Identification and variables measured of fragments sampled in the study area in Southeastern Brazil (Tableland Atlantic Rain Forest, Espírito Santo). MC=Map code (see Figure S1); RI=Regional identification; SC=Fragment size class; S=Fragment size; H=Habitat; TI= Tree abundance; LI=Lianas abundance; TB=Tree biomass (Mg/site); LB=Liana biomass (Mg/site); TBP= Tree biomass proportional (%); LBP= Liana biomass proportional (%); MDS=Non metric multidimensional scale (axis 2); t= Maximum air temperature (%); h= Relative air humidity (%); w=Maximum wind speed (%); pH=Soil acidity - pH in H_2O; P= Available phosphorous (mg/dm^3); BS=Bases sum ($cmol_c/dm^3$); OM=Organic matter (dag/kg).

MC	RI	SC	S	H	TI	LI	TB	LB	TBP	LBP	MDS	t	h	w	pH	P	BS	OM
1	Fazenda Cúpido	Small	13.18	Interior	170	220	89.20	3.66	96.1	3.9	1.274	90.04	107.63	39.17	5.07	0.41	2.51	3.03
				Edge	144	703	54.14	7.70	87.6	12.4	0.375	95.87	99.86	64.91	5.11	0.79	3.05	2.96
2	Reserva Natural Vale	Small	28.84	Interior	205	267	43.91	3.14	93.3	6.7	0.831	97.87	104.38	27.78	4.15	2.69	1.08	3.61
				Edge	196	775	18.18	7.33	71.3	28.7	0.343	105.66	103.24	47.89	6.12	0.20	2.92	2.17
3	RPPN Recando das Antas	Small	50.12	Interior	166	212	67.61	1.85	97.3	2.7	0.526	94.61	107.00	64.52	4.81	0.84	1.75	2.78
				Edge	108	636	17.62	5.39	76.6	23.4	1.039	91.10	111.34	47.52	4.60	0.35	2.00	3.06
4	Fazenda do Neb	Medium	60.26	Interior	171	261	73.20	2.32	96.9	3.1	0.696	94.81	117.26	35.70	3.89	3.21	1.09	4.28
				Edge	152	831	28.22	4.73	85.6	14.4	0.093	101.39	113.01	47.05	6.31	4.17	4.66	3.45
5	Fazenda do Marim	Medium	104.71	Interior	155	97	143.79	2.85	98.1	1.9	0.331	93.27	115.56	16.38	5.47	3.62	2.29	1.60
				Edge	168	422	34.75	4.97	87.5	12.5	-0.746	91.92	112.69	40.30	5.18	4.29	2.87	2.24
6	Fazenda Caliman	Medium	208.93	Interior	156	114	100.29	1.73	98.3	1.7	0.393	88.74	111.15	36.45	4.07	3.16	1.27	4.53
				Edge	159	506	23.01	3.45	87.0	13.0	0.192	91.86	107.50	39.10	4.70	4.03	2.07	2.67
7	Fazenda Rochedo	Large	389.05	Interior	215	194	96.07	3.73	96.3	3.7	0.453	85.93	107.24	11.98	4.85	0.79	1.94	3.78
				Edge	203	472	47.12	3.07	93.9	6.1	-1.015	88.64	113.89	48.82	5.12	5.28	2.93	3.32
8	RPPN Recando das Antas	Large	831.76	Interior	188	135	168.72	1.58	99.1	0.9	0.853	87.00	128.91	11.98	4.45	2.43	1.18	2.88
				Edge	145	562	17.71	4.38	80.2	19.8	-1.416	95.47	112.69	60.70	5.04	4.04	2.48	2.15
9	REBIO de Sooretama	Large	1318.26	Interior	182	117	150.33	1.34	99.1	0.9	0.500	87.41	113.57	15.00	5.35	1.17	2.93	3.12
				Edge	175	505	57.81	4.41	92.9	7.1	-0.901	89.36	107.50	43.95	5.63	7.52	3.55	3.13
10	REBIO de Sooretama	Control	20417.38	Interior	175	99	149.31	1.30	99.1	0.9	0.527	83.93	149.14	11.86	5.02	1.20	2.65	2.18
				Edge	173	233	30.69	1.33	95.9	4.1	-1.823	93.89	111.15	45.46	4.88	1.73	1.72	2.04
11	Reserva Natural Vale	Control	20417.38	Interior	178	53	146.24	0.79	99.5	0.5	0.354	85.96	136.91	6.11	3.90	0.78	0.72	4.03
				Edge	159	226	22.12	2.52	89.8	10.2	-2.114	92.20	117.39	45.05	4.50	1.38	1.66	2.97
12	REBIO de Sooretama	Control	23442.29	Interior	213	171	228.44	2.35	99.0	1.0	0.471	86.10	128.77	15.51	4.80	2.04	1.38	1.85
				Edge	184	425	96.66	2.54	97.4	2.6	-1.235	89.02	117.80	54.24	5.34	3.80	2.71	2.33

Table S2 - Pearson correlations between microclimate and soil variables. Temp=Maximum air temperature (%); Humid= Relative air humidity (%); Wind=Maximum wind speed (%); pH=Soil acidity - pH in H_2O; P=Available phosphorous (mg/dm^3); BS=Bases sum (cmol$_c$/dm^3); OM=Organic matter (dag/kg).

Microclimatic variables - Interiors dataset			
	Temp	**Humid**	**Wind**
Temp	1.00		
Humid	-0.59	1.00	
Wind	**0.83**	-0.57	1.00

Microclimatic variables - Global dataset			
	Temp	**Humid**	**Wind**
Temp	1.00		
Humid	-0.59	1.00	
Wind	0.57	**-0.61**	1.00

Soil variables - Interiors dataset				
	pH	**P**	**BS**	**OM**
pH	1.00			
P	**0.62**	1.00		
BS	**0.93**	**0.66**	1.00	
OM	-0.32	-0.07	-0.11	1.00

Soil variables - Global dataset				
	pH	**P**	**BS**	**OM**
pH	1.00			
P	0.22	1.00		
BS	**0.90**	0.35	1.00	
OM	-0.50	-0.002	-0.25	1.00

Table S3 - Model selection for microclimatic and soil variables in relation with fragment size and habitats (global model). K=number of parameters; AICc=Akaike Information Criterion for small samples; ΔAICc=difference between the AICc of a given model and that of the best model; AICcWt= Akaike weights (based on AIC corrected for small sample sizes); Cum.Wt= Cumulative Akaike weights; LL=maximum likelihood. Temp=Maximum air temperature (%); Humid= Relative air humidity (%); Wind=Maximum wind speed (%); pH=Soil acidity - pH in H_2O; P= Available phosphorous (mg/dm^3); BS=Bases sum (cmol$_c$/dm^3); OM=Organic matter (dag/kg).

Wind speed ~ Fragment size*Habitats

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
Size*Habitat	6	194.67	0	0.69	0.69	-88.87
Size+Habitat	5	196.64	1.97	0.26	0.94	-91.65
Habitat	4	199.72	5.04	0.06	1	-94.81
Size	4	212.58	17.91	0	1	-101.24

Air temperature ~ Fragment size*Habitats

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
Size+Habitat	5	140.44	0	0.71	0.71	-63.55
Size*Habitat	6	143.21	2.77	0.18	0.89	-63.13
Habitat	4	144.88	4.43	0.08	0.97	-67.39
Size	4	146.73	6.29	0.03	1	-68.31

Air humidity ~ Fragment size*Habitats

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
Size*Habitat	6	169.09	0	0.93	0.93	-76.07
Size+Habitat	5	174.65	5.56	0.06	0.99	-80.66
Size	4	178.69	9.6	0.01	1	-84.29
Habitat	4	187.75	18.67	0	1	-88.82

Bases sum ~ Fragment size*Habitats

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
Habitat	4	64.37	0	0.62	0.62	-27.13
Size+Habitat	5	66.01	1.64	0.27	0.9	-26.34
Size*Habitat	6	68.32	3.96	0.09	0.99	-25.69
Size	4	71.91	7.54	0.01	1	-30.9

pH in H2O ~ Fragment size*Habitats

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
Habitat	4	48.21	0	0.7	0.7	-19.05
Size+Habitat	5	50.71	2.5	0.2	0.9	-18.69
Size	4	53.43	5.22	0.05	0.95	-21.66
Size*Habitat	6	53.53	5.33	0.05	1	-18.3

Phosphorous ~ Fragment size*Habitats

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
Habitat	4	103.21	0	0.69	0.69	-46.55
Size+Habitat	5	106.36	3.16	0.14	0.83	-46.52
Size	4	106.69	3.48	0.12	0.96	-48.29
Size*Habitat	6	108.71	5.5	0.04	1	-45.88
Organic matter ~ Fragment size*Habitats						
Model	**K**	**AICc**	**ΔAICc**	**AICcWt**	**Cum.Wt**	**LL**
Habitat	4	59.67	0	0.6	0.6	-24.78
Size+Habitat	5	61.88	2.21	0.2	0.79	-24.27
Size	4	62.18	2.51	0.17	0.96	-26.04
Size*Habitat	6	65.29	5.62	0.04	1	-24.18

Table S4 - Model selection for tree biomass and lianas biomass in relation with microclimatic and soil variables considering only the fragments interior. K=number of parameters; AICc=Akaike Information Criterion for small samples; ΔAICc=difference between the AICc of a given model and that of the best model; AICcWt= Akaike weights (based on AIC corrected for small sample sizes); Cum.Wt= Cumulative Akaike weights; LL=maximum likelihood. Temp=Maximum air temperature (%); Humid= Relative air humidity (%); Wind=Maximum wind speed (%); pH=Soil acidity - pH in H_2O; P= Available phosphorous (mg/dm^3); BS=Bases sum (cmol$_c$/dm^3); OM=Organic matter (dag/kg).

Tree biomass ~ microclimate variables						
Model	**K**	**AICc**	**ΔAICc**	**AICcWt**	**Cum.Wt**	**LL**
Temp	3	128.65	0	0.44	0.44	-59.83
Humid	3	130.08	1.43	0.22	0.66	-60.54
Wind	3	130.58	1.92	0.17	0.83	-60.79
Temp+Humi	4	131.57	2.91	0.1	0.93	-58.93
Humid+Wind	4	132.47	3.81	0.07	1	-59.38
Liana biomass ~ microclimate variables						
Model	**K**	**AICc**	**ΔAICc**	**AICcWt**	**Cum.Wt**	**LL**
Humid	3	34.46	0	0.71	0.71	-12.73
Humid+Wind	4	38.08	3.62	0.12	0.83	-12.18
Temp	3	39.1	4.64	0.07	0.9	-15.05
Temp+Humi	4	39.12	4.66	0.07	0.97	-12.7
Wind	3	40.5	6.04	0.03	1	-15.75
Tree biomass ~ soil variables						
Model	**K**	**AICc**	**ΔAICc**	**AICcWt**	**Cum.Wt**	**LL**
OM	3	132.29	0	0.57	0.57	-61.65
pH	3	135.64	3.34	0.11	0.68	-63.32

BS+OM	4	136.3	4.01	0.08	0.75	-61.29
pH+OM	4	136.45	4.16	0.07	0.83	-61.37
BS	3	136.72	4.42	0.06	0.89	-63.86
P	3	136.85	4.56	0.06	0.95	-63.93
P+OM	4	137.01	4.71	0.05	1	-61.65

Liana biomass ~ soil variables

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
pH	3	40.25	0	0.28	0.28	-15.63
BS	3	40.64	0.39	0.23	0.52	-15.82
OM	3	40.91	0.65	0.2	0.72	-15.95
P	3	40.92	0.67	0.2	0.92	-15.96
P+OM	4	44.49	4.24	0.03	0.96	-15.39
SB+OM	4	45.35	5.09	0.02	0.98	-15.82
P+OM	4	45.6	5.35	0.02	1	-15.94

Table S5 - Model selection for tree biomass and lianas biomass in relation with microclimatic and soil variables (global model). K=number of parameters; AICc=Akaike Information Criterion for small samples; ΔAICc=difference between the AICc of a given model and that of the best model; AICcWt= Akaike weights (based on AIC corrected for small sample sizes); Cum.Wt= Cumulative Akaike weights; LL=maximum likelihood. Temp=Maximum air temperature (%); Humid= Relative air humidity (%); Wind=Maximum wind speed (%); pH=Soil acidity - pH in H_2O; P= Available phosphorous (mg/dm^3); BS=Bases sum (cmol$_c$/dm^3); OM=Organic matter (dag/kg).

Tree biomass ~ microclimate variables

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
Temp+Wind	5	251.36	0	0.36	0.36	-119.01
Wind	4	251.6	0.24	0.32	0.67	-120.75
Humid+Wind	5	251.63	0.28	0.31	0.99	-119.15
Temp	4	259.03	7.67	0.01	0.99	-124.46
Humid	4	259.55	8.19	0.01	1	-124.72

Liana biomass ~ microclimate variables

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
Temp+Humid	5	94.46	0	0.34	0.34	-40.56
Temp	4	95.01	0.55	0.26	0.59	-42.45
Humid	4	95.14	0.68	0.24	0.83	-42.52
Humid+Wind	5	96.69	2.23	0.11	0.94	-41.68
Wind	4	97.95	3.49	0.06	1	-43.92

Tree biomass ~ soil variables

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
BS	4	269.68	0	0.39	0.39	-129.79
pH	4	271.42	1.74	0.16	0.55	-130.66
P	4	271.62	1.94	0.15	0.69	-130.76
SB+OM	5	272.16	2.48	0.11	0.81	-129.42
OM	4	272.22	2.54	0.11	0.91	-131.06
pH+MO	5	273.61	3.93	0.05	0.97	-130.14
P+OM	5	274.7	5.02	0.03	1	-130.68

Liana biomass ~ soil variables

Model	K	AICc	ΔAICc	AICcWt	Cum.Wt	LL
BS	4	97.54	0	0.43	0.43	-43.72
pH	4	98.69	1.16	0.24	0.68	-44.29
BS+OM	5	99.76	2.23	0.14	0.82	-43.22
pH+OM	5	100.74	3.2	0.09	0.9	-43.7
OM	5	101.62	4.08	0.06	0.96	-44.14
P	6	103.37	5.83	0.02	0.98	-43.22
P+OM	4	105.44	7.9	0.01	0.99	-47.67

IV. CAPÍTULO II

COMMUNITY AND FUNCTIONAL IMPACTS OF FRAGMENTATION EFFECTS ON TREES SPECIES

ABSTRACT – The fragmentation of tropical forests is one of the greatest threats to global biodiversity. Some studies reveal that fragmentation can impact severely upon species richness and community structure. However, fewer studies have evaluated the potential loss in functional diversity in fragmented landscapes. We tested whether smaller fragments retain important biodiversity value and if forest fragmentation effects (fragment size and edge and interior habitats) impact negatively upon tree functional diversity on tree community. Our study was carried out in remnants of the Brazilian Atlantic forests. We used generalized linear mixed models to study fragmentation effects (size and edge effects),. which we parameterized with a functional traits dataset including food resources, seed dispersal, carbon storage and forest structure. A total of 4,140 individuals (DBH≥4.8 cm) belonging to 444 tree species were sampled across the study area. The most parsimonious model showed that forest edges promoted marked changes in tree community structure and functional traits, significantly reducing species richness and functional diversity. Our models also show that larger fragments and forest interiors have significantly more potential to provide food resources and interactions with fauna. We conclude that in a fragmented landscape the plant functionality in larger fragments is significantly different from that of smaller fragments, the result of differing functional traits. Nonetheless, small fragments have an important role in the maintenance of ecological services making them indispensable to conservation of biodiversity within the highly threatened Atlantic forest biome.

Keywords: Tableland Atlantic Rain Forest; Functional traits; Fragmented landscape; Species richness; Fauna resources; Wood density; Carbon.

Introduction

The fragmentation of tropical forests is one of the greatest threats to global biodiversity (Fahrig 2003; Laurance et al. 2006a,b). These threats are arising primarily from rapid habitat loss (Schroth et al. 2004), with some ~83 million hectares of tropical forest cleared for agriculture in the 1980's and 90's alone (Gibbs et al. 2010). Clearance isolates remnant blocks of forest and divides them into smaller parcels of forest, driving subsequent negative effects on wildlife populations, the severity of which is determined by the size, shape, isolation and edge effects of the fragments (Murcia et al. 1995; Fahrig 2003). While the biggest blocks of primary forest are irreplaceable for biodiversity conservation (Gibson et al. 2011), the sheer scale of forest fragmentation means that understanding the impacts on the biological and functional value of smaller fragments is of critical importance to sustaining biodiversity in fragmented landscapes (Santos et al. 2010).

Many studies have highlighted the changing patterns of species richness, diversity and community composition between contiguous forest and fragments, and across gradients of forest fragment sizes. These studies reveal that intense landscape-level fragmentation can impact severely upon species richness, with a pantropical reduction in richness compared to intact forests (e.g. Laurance 1994; Benítez-Malvido & Martínez-Ramos 2003; Watson et al. 2004; Benedick et al. 2006; Hillers et al. 2008; Arroyo-Rodríguez et al. 2008; Pardini et al. 2010; Tabarelli et al. 2010). They also indicate that there are severe edge effects, such as desiccation, wind disturbance, light and temperature increase, and decrease in air humidity, which can further complicate the biological impacts of fragmentation (Laurance et al. 2002; Tabarelli et al. 2010; Pütz et al. 2011). At forest edges, forest specialist species (e.g., shade-tolerant trees) are typically replaced by generalist or pioneer species, promoting losses of species richness, changes in community structure, and shifts in forest dynamics and functionality (Oliveira et al. 2004; Tabarelli et al. 2010; Laurance et al. 2006ab). Nevertheless, given the high species richness and spatial turnover in intact tropical forests, fragments that have apparently undergone severe declines in species richness can still retain a subset of species with high conservation value (Hill et al. 2011; Arroyo-Rodríguez et al. 2008; Gardner et al. 2009; Pardini et al. 2010; Santos et al. 2010).

Our understanding of the impacts of forest fragmentation on the functional roles performed by species, and thus on ecosystem functioning, is much more limited (Chapin 2003; Gardner *et al.* 2009). As an example, the sizes and dispersal types of fruits and seeds can be used to evaluate resource availability and the diversity of interactions between animals and plants (Moran & Catterall 2010). Most assessments use simple indices, such as the Shannon and Simpson diversity indices (e.g. Metzger 2000; Girão et al. 2007) or the number of functional traits observed per plot (Mayfield et al. 2005), to infer that communities have significantly lower functionality in fragments than in intact tropical forests (see Metzger 2000; Mayfield et al. 2005; Girão et al. 2007; Tabarelli & Peres 2002; Laurance et al. 2006ab; Michalski et al. 2007). However, these methods of quantifying the impacts of disturbance on functional roles are incapable of combining a variety of functional traits into a single overall measure of functional changes (Petchey & Gaston 2002). Further, they fail to consider variation in the functional impacts of other traits that vary within a particular functional group, for instance, large-fleshy fruits that contain one large to many small seeds.

An alternative approach to evaluating the effects of forest fragmentation on the functional roles performed by species is to examine functional diversity (Loreau 2001; Petchey & Gaston 2002; Villéger et al. 2008). Functional diversity quantifies a range of functional traits within multi-dimensional niche space, typically focusing on the physiological and morphological traits that define a species' ecological role in a community (Petchey & Gaston 2006; Villéger et al. 2008) and yielding a single continuous measure. This also allows one to assess how regularly species are distributed within functional space, weighted by relative abundances, and how the relative abundance of species are distributed within functional space, relative to the centre of gravity (Villéger et al. 2008). Such assessments can help us to understand the effects of disturbance on ecosystem functioning, particularly in the context of conservation of tropical biodiversity (Laliberté et al. 2010; Villéger, et al. 2010; Pakeman et al. 2011; Baraloto et al. 2012). Furthermore, functional diversity indices are typically more able to discern impacts of environmental disturbance than are basic measures of species diversity (Loreau et al. 2001), due to differences in functionality assigned to each species (Petchey & Gaston 2002).

In this study, we focus on forest fragmentation in the Atlantic Forest and on trees, which play critical functional roles in ecosystems, for instance, by providing shelter and food resources for fauna (Moran & Catterall 2010), energy transformation in live biomass (primary production)

(Barber 2007), and thus atmospheric carbon sequestration and climate regulation (Laurance 2004; Nascimento & Laurance 2004). The Brazilian Atlantic Forest is a hotspot of imperiled biodiversity (Myers et al. 1988). More than 380 tree species are found in just one hectare of Atlantic Forest (Saiter et al. 2011), making it some of the biologically most important real estate on Earth. Yet deforestation has been so widespread in the Atlantic Forest that just 11% of forest cover remains and 80% of the forest that does persist is within fragments smaller than 50 hectares (Ribeiro et al. 2009). We tested two hypotheses: (1) that smaller fragments retain important biodiversity value; and (2) that forest fragmentation effects (fragment size and edge and interior habitats) impact negatively upon tree functional diversity.

Material and Methods

Study area

This study was based in the state of Espírito Santo, in Southeast Brazil. Within the region, we focused on the municipalities of Sooretama, Linhares and Jaguaré (19^{o}04'05 "S and 39^{o}57'35" W, 28- 65 m.a.s.l) (Figure 1), which contain a landscape matrix composed mainly of *Eucalyptus* spp. plantations, grasslands, coffee and papaya plantations (Rolim et al. 2005). The climate is tropical wet (Köppen classification), with an annual precipitation of 1,403 mm and a distinct dry season from May to September, when precipitation is just 33 mm per month (Peixoto & Gentry 1990). The predominant soil in the study region is Yellow Podzolic (IBGE 1987).

This region is part of the phytogeographic domain Atlantic Forest and is officially classified as Lowland Rain Forest (IBGE 1987) or Tertiary Tabelands Forest according with Peixoto & Silva (1997). The study area is of high conservation importance due to the presence of two forest fragments larger than 20,000 hectares, which house a high diversity of plant and animal species (Peixoto & Silva 1997; Chiarello 1999; Masden et al. 2001).

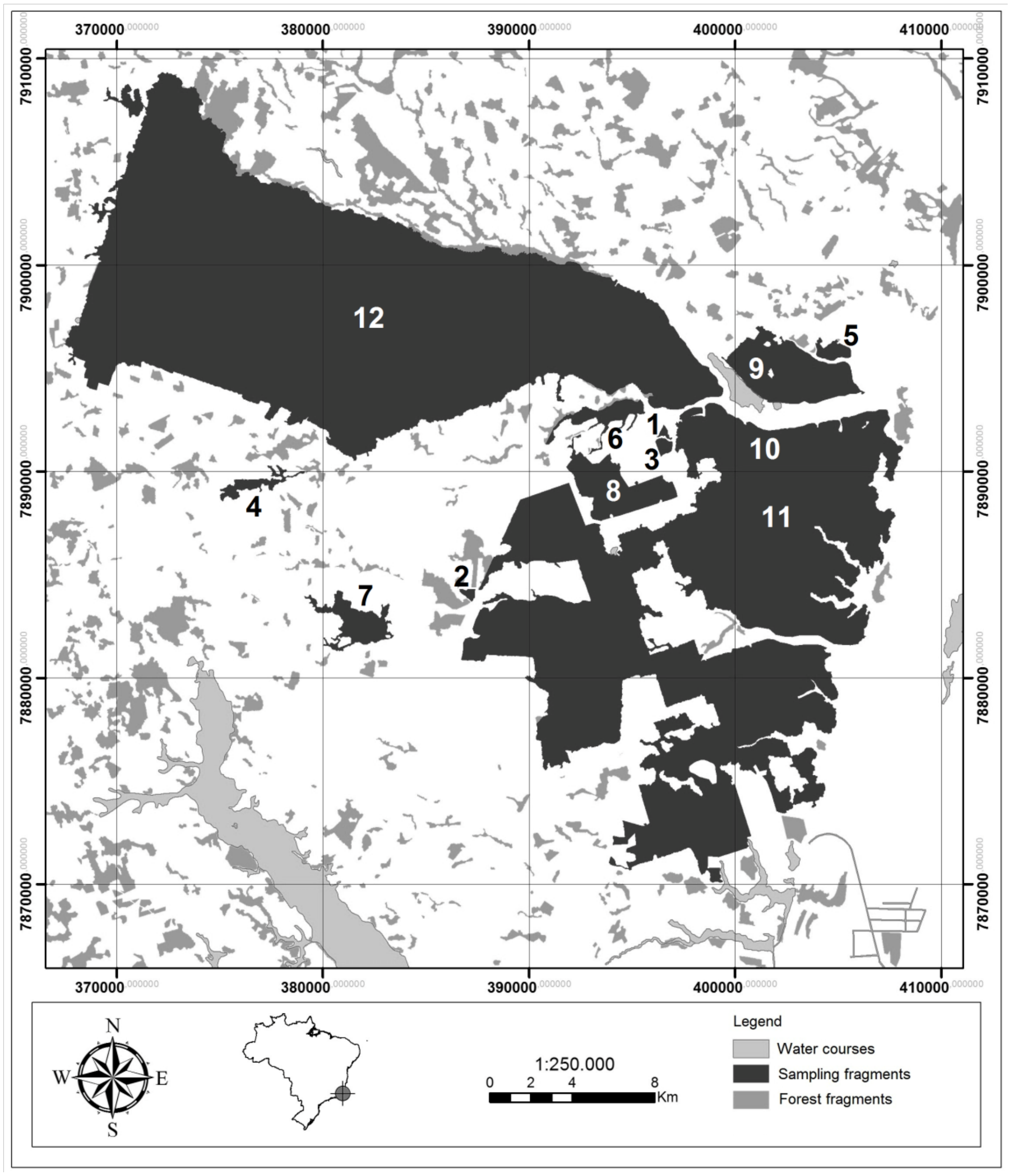

Figure 1 - Study area and forest fragments sampled in Southeastern Brazil. To check the respective names and information about fragments see the table S1.

Tree sampling

Fieldwork was conducted from January 2011 to January 2012. We created permanent plots along transects within each of nine fragments (range=13.18 to 1318.26 ha; mean=333.9 ha) and two control forest blocks larger than 20,000 ha in Reserva Natural da Vale (RNV) and

Reserva Biológica de Sooretama (REBIO) (Table S1). Within each fragment, we created two transects: one ~5 m into the fragment and parallel to the forest edge and one in the forest interior (≥300 m from the forest edge). Along each transect, we stationed ten 10 x 10 m plots positioned at 20 m intervals, totaling 240 plots. Due to the absence of other control forest blocks, we allocated three pairs (of 10 edge and 10 interior plots) to transects in RNV (one pair) and REBIO (two pairs), with a mean distance of 17.1 ± 10.4 km between transect pairs. All plots were on the same type of soil (Yellow Podzolic).

We sampled every living tree individual with a diameter at breast height ≥4.8 cm at 1.3 m above ground height in each plot, collecting samples from each tree individual. We identified this material with reference to collections at the CVRD Herbarium of the Vale and the VIES Herbarium of the Federal University of Espírito Santo, and with aid from taxonomic experts in plant species identification in specific families (e.g. Myrtaceae and Sapotaceae). Botanical material collected in the fertile stage was deposited in the collection of Vale Herbarium of the Reserva Natural Vale in Linhares, ES.

Functional trait matrix

We use functional traits that are relevant to the morphological and physical adaptations of trees in their role as food resources, their dispersal, and in carbon storage and forest structure (Tabarelli & Peres 2002; Bolmgren & Eriksson 2005; Laurance et al. 2006ab; Bongers et al. 2009; Moran & Catterall 2010; Tabarelli et al. 2010). Within these three broad types of functional role, we had five functional categories and one continuous functional trait, classified as: (1) fruit size, (2) seed size, and (3) fruit type, each relevant to food resource functions; (4) fruit dispersal syndrome, (5) successional group and (6) wood density (continuous variable) relevant to carbon storage and forest structure.

Food Resources: Fruit and seed sizes for each of the species identified were classified into four categories according to Tabarelli & Peres (2002): small (size values <0.6 cm in length), medium (size between 0.6 and 1.5 cm), large (size between 1.6 to 3.0 cm), and extremely large (size larger than 3.0 cm). We categorized the fruits into two types: (i) fleshy fruits (i.e., the pericarp can accumulate water and many organic compounds, see Coombe (1976)) and non-fleshy fruits.

Fruit dispersal syndrome: Fruits were classified as zoochoric or non-zoochoric following Van Der Pijl (1982). A zoochoric tree produces diaspores surrounded by fleshy pulp, an aryl, or other features that are typically associated with dispersal by animals, and a non-zoochoric tree has characteristics that indicate dispersal by abiotic means, such as winged seeds, feathers, or a lack of features that indicate dispersal via methods other than downfall or explosive indehiscence.

Carbon storage and forest structure: We classified species into the successional groups defined by Bongers et al. (2009). We considered as pioneers those trees that develop in conditions of high luminosity and generally do not occur in the understory, as initial secondary those trees that develop in intermediate shading conditions, and as late secondary those trees that develop exclusively in permanently shaded understory. Species were classified using the database by Jesus & Rolim (2005) from the Reserva Natural da Vale. Data for wood density in dry weight (g/cm^3) were obtained from *The Global Wood Density (GWD) database* in the subsection *Tropical South America* (http://hdl.handle.net/10255/dryad.235, Chave et al. 2009; Zanne et al. 2009). We made two adjustments (following Flores & Coomes 2011; Hawes et al. 2012): (i) for morphospecies only identified to the family or genus level, we used the average wood density of the taxonomic group; and (ii) for species not in the GWD database, we used the average wood density for the species' genus.

Species identified at morphospecies level represented only 1.13% of species richness and 0.22% of total abundance. These species were not treated in any of the functional traits described above, being considered only in the analysis of species richness and community structure.

Data Analysis

We used Nonmetric Multidimensional Scaling (NMS) ordination analysis in the PC-ORD 6 package (McCune & Mefford 2011) to identify changes in community structure between different-sized fragments, and between edge and interior habitats. We used the species abundance raw data from each plot for this analysis and the metric distance used was Sorensen (Bray-Curtis).We considered the NMS results arising from tree species abundance data as a measure of community structure (Barlow et al. 2010). We considered the number of individuals as abundance.

To analyze functional diversity we used three indices proposed by Villéger et al. (2008): functional richness (FRic), functional evenness (FEve) and functional divergence (FDiv). According to Villéger et al. (2008) the FRric represents the volume of space of a functional convex hull occupied by the community, FEve the regularity of the distribution in abundance on this volume, and FDiv the divergence in the distribution of species characteristics within the volume occupied by each functional trait. To calculate these three indices we used methods and scripts from Villéger et al. (2008), in R version 2.15.1 (R Development Core Team 2012).

To investigate fragmentation effects in the community and functionality of tree species in the studied landscape we considered three classic factors from fragmentation to compose models: (i) fragments size, (ii) edge and interior habitats, and (iii) the interaction between size and habitat.

Models were carried out using the *glmmadmb* function from the *glmmADMB* package. We used Negative Binomial error distributions for count data, since our data showed significant overdispersion. We used a Gaussian error distribution for the rest of the data. The sites (each fragment) were codified as a random variable in all analyses (Bolker et al. 2009). We used the *dredge* function from the *MuMIn* package to test all possible combinations of the variables included in the global model. To determine which of these factors were the most decisive in possible changes in species richness, community structure, functional traits and functional diversity we used an information theoretical approach based on the Akaike Information Criterion of Second Order (AICc), which is indicated for small sample sizes and the best model was indicated by the AICc lower value (Burnham et al. 2011). All analyses were performed in the R version 2.15.1 (R Development Core Team 2012).

Results

Fragments retain important biodiversity value

A total of 4,140 tree individuals belonging to 443 species were sampled in this study (Table S2). Considering only the best model selected on the basis of their AICc values, species richness was significantly influenced by the interaction between fragment size and edge habitat (GLZ: z=2.52; p = 0.03), whereby fragment size had a significant negative effect on species richness at fragment interiors (F=6.95: p = 0.02; Figure 2A) and no significant effects near edges

(F=1.89: p = 0.1982; Figure 2A). Comparing the habitats, the average richness was significantly higher within fragment interiors than near forest edges (GLZ: z=2.79; p = 0.02, Figure 2B).

NMS analysis of species composition and abundance (community structure) parameters indicated the existence of significant changes on community structure for the axes 1, 2, and 3 (p = 0.02; Figure S1). However, changes of community structure (axes scores) for axis 3 cannot be explained by the GLZ models tested. In contrast, significant influences of fragments size and of edge effects on tree community structure were shown by GLZ models for axis 1 and 2. The best model selected by AICc for axis 1 indicated that changes in community structure were strongly influenced by the creation of an edge habitat (GLZ: z=3.84; p <0.01; Figure 2C). For axis 2, the best model demonstrated that the interaction between fragment size and habitat altered community structure (GLZ: z=-4.05; p <0.01; Figure 2D), with changes significantly related to fragment size in the interior (F=95.95: p = 0.01). There were no significant effects of edges on community structure (F=7.76: p = 0.37).

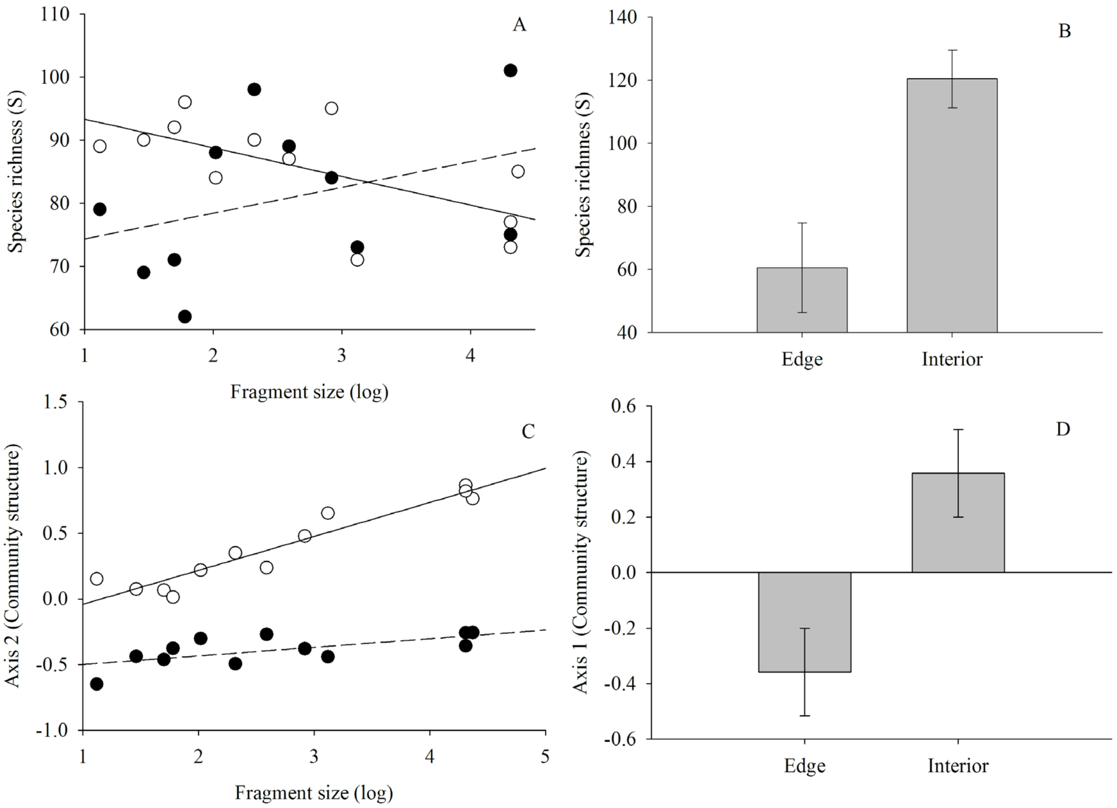

Figure 2 - Taxonomic changes as a function of fragments size and habitat. (A) The effect of the interaction between fragments size and habitat on total species richness, partial residuals plots;

(B) the effect of habitat on total species richness; (C) the effect of the interaction between fragment size and habitat on tree community structure (Axis 2 scores from NMS analysis), partial residuals plots; and (D) the effect of habitat on tree community structure (Axis 1 scores from NMS analysis). Filled (forest edge) and empty (forest interior) circles represent values obtained after summation of raw residuals with the expected values for each variable, assuming average values for other covariates.

Fragmentation, edge effects and tree functional diversity

The habitat variable had the largest effect on species richness within different functional groups, and is present in six of fifteen of the best models proposed by the AICc ordination for functional traits (Table 1). Proximity to forest edges had a significantly negative influence on the abundance of tree species with zoochoric dispersion, fleshy fruits and later secondary species and a positive influence on the abundance of non-zoochoric dispersed species, pioneers and initial secondary species (Table 1). Fragment size, and the interaction between fragment size and habitat variable, had a significant negative effect on the abundance of tree species with very large fruits (Table 1, see also Figure S2).

Habitat was also prominent for models based on species abundance data (present in 11 of 15 the best models), significantly and negatively influencing species with zoochoric dispersion, non-zoochoric dispersion, fleshy fruits, very large fruits, medium fruits, small fruits, large seeds, medium seeds, pioneers and late secondary species. Fragments (present in seven of fifteen of the best models) showed a negative influence on very large fruits and a positive one on zoochoric dispersion, fleshy fruits, medium fruits, large seeds and medium seeds. The interaction between fragments size and habitat (present in two of 15 models), influenced very large fruits and the initial secondary species (Table 1, Figure S3). Wood density did not respond significantly to fragment size nor to habitat.

Focusing on the impact of fragmentation on functional diversity, we found that forest habitats (GLZ: z=-0.46; p=0.66), a reduction in fragments size (GLZ: z=-0.78; p=0.45) and interactions between size and habitats (GLZ: z=0.38; p=0.71) did not result in significant effects on functional ricnhess . This result indicates that forest fragmentation causes no loss in the volume of functional richness in this landscape. Functional evenness was negatively related to fragment size (GLZ: z=-2.23; p <0.05; Figure 3A), indicating that the evenness traits are less heterogeneous in larger fragments. This parameter was also significantly higher in fragment interiors versus edges (GLZ: z=4.8; p <0.001, Figure 3B). The best model for functional divergence showed a significant negative relationship with forest patch size (GLZ: z=2.1; p <0.04; Figure 3C), demonstrating that smaller patches are more divergent (less functional redundant) than larger fragments in the landscape.

Table 1 – Results from Generalized Linear Mixed Models (only the best models according to their AICc values are shown) for the effects of fragment size, habitat and their interaction on species richness and abundance of different functional groups. Values show coefficient estimates and standard errors.

Functional trait	**Fragment size (log)**	**Habitats (Edge)**	**Size*habitats**
Trait richness			
Zoochoric dispersion		-10.08 (3.65)**	
Non-zoochoric dispersion		5.58 (1.94)**	
Fleshy fruits		-9.08 (-3.26)*	
Non-fleshy fruits		4.58 (2.57)ns	
Very large fruits	-4.05 (1.2)***	-8.55 (4.91)ns	4.24 (1.7)*
Large fruits		-4.92 (2.3)ns	
Medium fruits		-3.25 (1.7)ns	
Small fruits		0.92 (0.77)	
Very large seeds	-0.73 (0.47)ns		
Large seeds		-4.91 (2.3)ns	
Medium seeds		-5.25 (2.74)ns	
Small seeds		1.08 (1.92)ns	
Pioneers		2.17 (0.93)*	
Initial secondary		7.75 (2.3)**	
Later secondary		-14.42 (2.7)***	
Trait abundance			
Zoochoric dispersion	0.09 (0.02)***	-0.32 (0.06)***	
Non-zoochoric dispersion		0.41 (0.09)***	
Fleshy fruits	0.12 (0.03)***	-0.27 (0.07)***	
Non-fleshy fruits	-0.04 (0.07)ns		
Very large fruits	-0.15 (0.06)**	-0.53 (0.22)*	0.25 (0.08)***
Large fruits		-0.15 (0.1)ns	

Medium fruits	0.11 (0.05)*	-0.19 (0.09)*	
Small fruits		-0.39 (0.16)*	
Very large seeds		0.28 (0.21)ns	
Large seeds		-0.14 (0.08)ns	
Medium seeds	0.1 (0.03)**	-0.22 (0.08)**	
Small seeds		-0.07 (0.08)ns	
Pioneers		1.05 (0.32)**	
Initial secondary	0.13 (0.05)**	-0.14 (0.25)ns	0.25 (0.09)**
Later secondary	0.06 (0.04)ns	-0.48 (0.07)***	
Wood characteristic			
Wood density		-0.02 (0.01)ns	

Note we used an inverse link function, so positive parameters indicate negative effects and negative parameters indicate positive effects. N=12; * p> 0.05, ** p> 0.01, *** p> 0.001, ns = not significant.

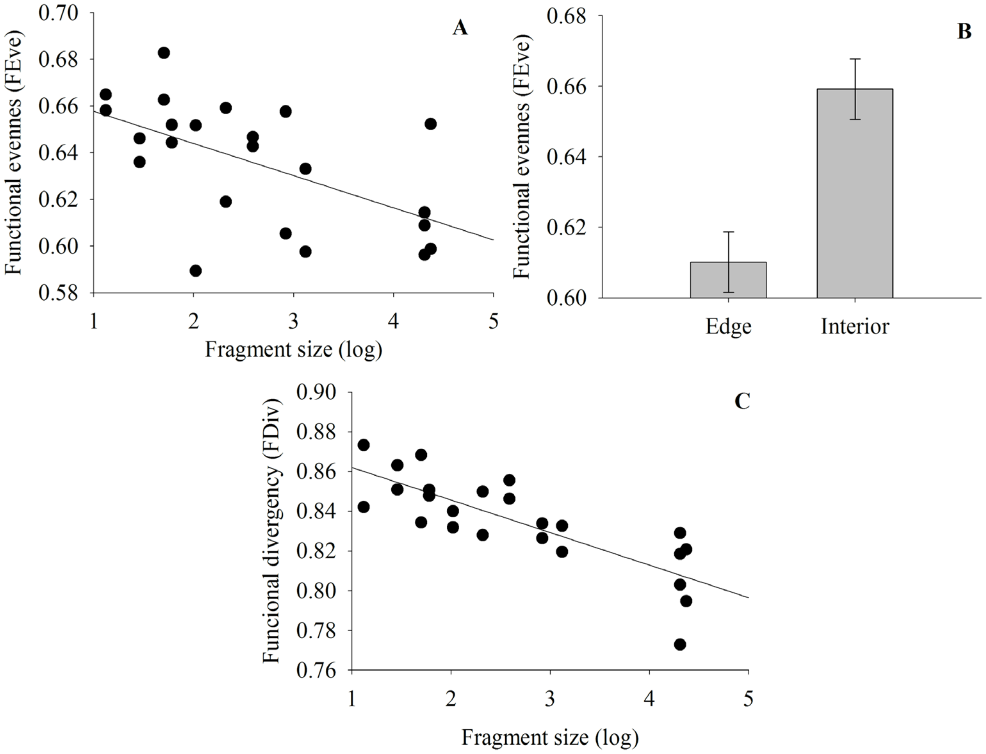

Figure 3 – Graphs of best models of functional diversity in relation to fragments size and habitat. (A) The effect of fragment size on Functional Evenness quability (FEve), partial residuals plots; (B) the effect of habitat on Functional Evenness (FEve); and (C) the effect of fragment size on Functional Divergence (FDiv), partial residuals plots. Filled circles represent values obtained

after the summation of raw residuals with the expected values for each variable, assuming average values for other covariates.

Discussion

Fragments and edges effects on biodiversity value

Our results show that species richness in the interior of forest fragments is negatively related to forest patch size (e.g. Michalski et al. 2007; Zipkin et al. 2009), in contrast to other studies that showed that larger fragments retain the highest values of species richness (Laurance & Vasconcelos 2009; Laurance et al. 2011). Since small fragments tend to be more intensely affected by edge effects, exhibiting both species typical from interior and from edge areas, our results are broadly consistent with the intermediate disturbance theory (Connell 1978), which predicts that communities with an intermediate level of disturbance will have the highest species richness. In particular, we found the abundance of species in the early secondary functional group, which generally establish in areas with intermediate light-intensity conditions within forests (e.g. Bongers et al. 2009), to decline in abundance as fragment size increased.

Across all fragment sizes, we found lower species richness at fragments edges than interiors. Forest edges are subjected to a range of impacts, such as increased wind speed, air temperature, and luminosity, and decreased humidity, compared to forest interiors (Laurance et al. 2002). Because of the high intensity and frequency of such disturbances at fragments edges, our study supports others (Oliveira et al. 2004; Lopes et al. 2009) in showing the number of species to be relatively lower at edges than in the forest interior.

We observed significant shifts in community structure with fragments size and near edges. It is likely that these shifts are related to the abundance of successional species (pioneer and initial secondary), since these species are faster-growing and have higher-mortality than species characteristic of intact forests (late secondary species). In fragmented areas, the increase in pioneer and initial secondary species leads to a rapid change in forest structure, species composition and ecological functionality (Laurance et al. 2006ab; Bongers et al. 2009; Tabarelli et al 2010). Such changes in successional groups are apparently driven by proximity to forest edges rather than by fragment size (Table 1). For instance, Amazonian forest fragments tend to have a greater number of successional species and in higher abundances near edges than in interiors of fragments (Laurance et al. 2006ab; Michalski et al. 2007), which may influence the structure and dynamics of forest patches (e.g. Laurance et al. 2006ab; Tabarelli et al. 2010).

Fragmentation, edge effects and tree functional diversity

Shifts in the abundance of different forest resources following fragmentation can modify the diversity of interactions between animals and plants (Moran & Catterall 2010). In this study, we found decreases in the species richness and abundance of species with fleshy fruits, zoochorous dispersal, and smaller fruit and seed sizes for plants at habitat edges and in smaller forest fragment. This indicates dramatic shifts in the key food resource for fauna, like frugivorous birds and mammals, and in the regeneration potential via dispersal (see Bolmgren & Eriksson 2005). Such a loss of ecological functionality is likely to have secondary consequences for faunal and floral richness (Tabarelli & Peres 2002; Laurance et al. 2002; Oliveira et al. 2004; Laurance et al. 2006b). In fact within our study landscape, the consequences of the reductions in fauna resources was indicated for mammals and birds, where in medium and large fragments have greater richness and abundance of medium and large frugivorous mammals than do smaller fragments, which are dominated by herbivorous mammals, promoting decreases in the structure mammals complexity (Chiarello 1999, see also Marsden & Whiffin 2003 for frugivorous birds).

Although there are significant increases in the species richness and abundance of pioneer and initial secondary trees at forest edges versus interiors and of initial secondary trees in the interior of smaller fragments, we found no effects of fragmentation on wood density within the fragments (see Results). Wood density are strongly affected by fragmentation (Laurance et al. 2006b), nevertheless, wood density can, sometimes, be a poor predictor of the responses of successional species to fragmentation effects (Laurance et al. 2006a, but see Michalski et al. 2007). The apparent lack of response of wood density to fragmentation suggests that even small fragments or fragment edges can be managed to play an important role in carbon storage. We do note that since wood density could vary with environmental characteristics, the average wood densities we used from the literature (Chave et al. 2009; Zanne et al. 2009) could however shift as the result of fragmentation (e.g. Thomas et al. 2007; Nock et al. 2009).

Our results suggest that among the three indices that describe functional diversity, those which account for the abundance of species (functional evenness) and dominance (by abundance) of a functional group over other functional groups (functional divergence) were more sensitive to fragmentation effects than functional richness, which is more influenced by species richness (see Villéger et al. 2008; Mouchet et al. 2010). In our study, functional richness remained constant with fragment size and at edges versus interiors (see Results), and thus fragmentation does not alter the volume of functional space occupied by species within different communities. However, the retention of functional richness with fragmentation is apparently underpinned by species exhibiting different functional traits, and thus there will be shifts in ecological functionality. At forest edges, the functional richness is maintained by a significantly higher richness of pioneers, early secondary, dried fruits and non-zoochorous dispersed species, whereas in forest interiors, the functional richness is maintained by significantly high species richness of late secondary, fleshy fruits and zoochoric dispersed species (Table 1).

A high intensity and frequency of disturbances at fragment edges (Murcia 1995) may explain reductions in functional evenness compared to fragment interiors, while this parameters and functional divergence had a negative relationship with increasing fragment size. Declines in functional evenness indicate that some parts of the functional space within edges and fragment size reduce or disappear (e.g. Mouchet et al. 2010). In turn, declines in functional divergence in larger fragments suggests niche homogenization between species (Mouchet et al. 2010), with most species being functionally similar and exhibiting higher competitiveness.

At fragment edges, the reductions of functional evenness were related to the loss of important functional traits, such as zoochoric dispersion, fleshy fruits and later secondary species, with the increase in the population of a few groups near forest edges (non-zoochoric and pioneers species). Such shifts in functional traits are likely to make edges functionally less heterogeneous and attractive for fauna. Thus, the reduction in functional evenness can be related to an increase in the intensity and frequency of disturbances near fragment edges. In contrast, the reductions in functional evenness and functional divergence within the largest fragments were accompanied by an increase of fleshy fruit, zoochoric dispersed species, as well as a reduction in the abundance of initial secondary species, indicating that larger fragments and interiors can provide more resources and interactions with fauna and have less disturbances (see Bolmgren & Eriksson 2005; Bongers et al. 2009).

Implications for conservation and conclusions

The changes in functional diversity, functional traits, species richness and community structure near forest edges and in smaller fragments have important implications for conservation in fragmented landscapes. Although our fragments are typically ~40 years old, we cannot rule out the possibility that there remains an extinction debt, especially given the long ages of many tree species (Chambers et al. 1998; Laurance et al. 2004), both in smaller fragments and at edges that could eventually degrade the functional value of these habitats. Additionally, tree populations in smaller fragments may be sustained to some extent by seed dispersal from larger blocks of natural habitat. Hence the functional diversity of our study system may change over time or if other forest fragments were removed from the landscape.

Nevertheless, our results support previous data, indicating that even small forest patches can retain high conservation value (e.g. Arroyo-Rodríguez et al. 2008; Gardner et al. 2009; Santos et al. 2010). First, our small forest patches retained similar communities to larger patches in the interiors, suggesting that they could represent important reservoirs of forest specialist trees and aid in seed dispersal or connectivity across landscapes. Second, the retention of functional diversity within small fragments and edges was maintained by non-zoochoric, pioneer and initial secondary tree species. These species are excellent dispersers and they could play important roles as sources of seeds in the recovery of early secondary forests in the event of agricultural abandonment or land purchases to reconnect forest fragments (Martínez-Garza & Howe 2003; Cortines & Valcarcel 2009; Simmons et al. 2011 see Pimm's savingspecies.org land purchases in Atlantic forests). Third, the absence of change in wood density across the fragmentation gradient suggests that fragments can represent important carbon stores, with potential co-benefits between carbon market, ecological services and biodiversity protection (Díaz et al. 2009; Phelps al. 2012).

The tree species functional diversity maintained in larger fragments, via higher abundance of species with fleshy fruits, suggests that they are able to provide resources for small to large-bodied fauna (see Bolmgren & Eriksson 2005). This is particularly important in larger fragments in the study area, where there are a number of endemic and IUCN red-listed bird and mammal species (e.g. Chiarello 1999; Marsden & Whiffin 2003) that are reliant upon these resources.

In conclusion, this is the first study that combines metrics of community change, with functional traits and functional diversity indices to explain the effects of fragmentation on the tree species community and ecosystem functioning within tropical forests. It shows that forest edges have a strong impact on tree community, drastically reducing species richness and functional diversity, and promoting changes in community structure and functional traits. In turn, larger fragments and forest interiors apparently have more food resources for fauna. We finish by noting that even the forest edge and small forest fragments within the deforestation scenario can provide some types of important ecological services and still harbour remnant populations of forest interior specialists, making them important in the conservation of biodiversity within the highly threatened Atlantic forest biome.

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SUPPLEMENTARY MATERIAL

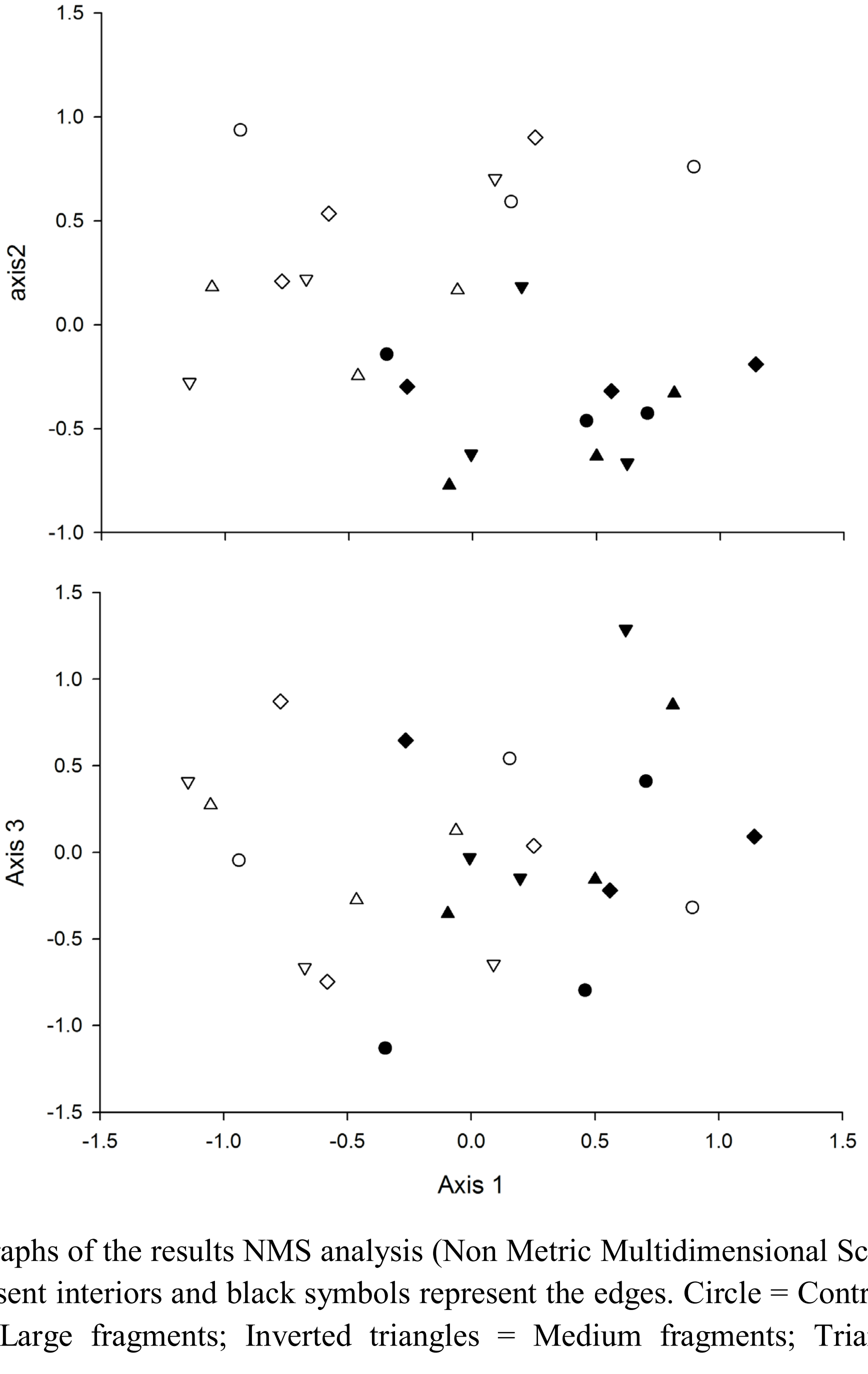

Figure S1 - Graphs of the results NMS analysis (Non Metric Multidimensional Scale). The white symbols represent interiors and black symbols represent the edges. Circle = Controls fragments ; Diamonds = Large fragments; Inverted triangles = Medium fragments; Triangles = Small fragments.

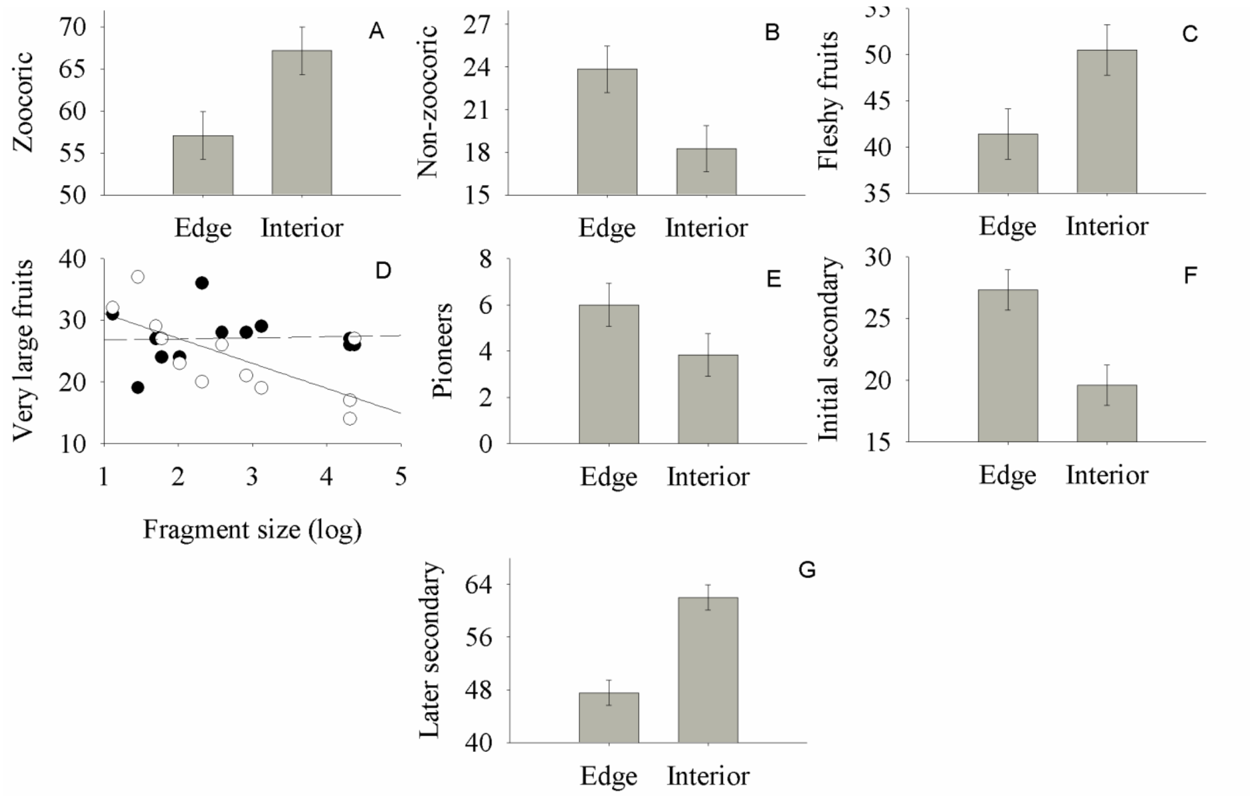

Figure S2 - Graphs of Generalized Linear Models results (only the best models according to AICc) for the fragment size and habitats effects on species richness per functional trait. Black (Edge) and white (Interior) circles represent values obtained after summation of raw residuals to the expected values for each variable, being assumed average values for other covariates. (A) Zoochoric dispersion; (B) Non-zoochoric dispersion; (C) Fleshy fruits; (D) Very large fruits; (E) Pioneers; (F) Initial secondary; (G) Later secondary.

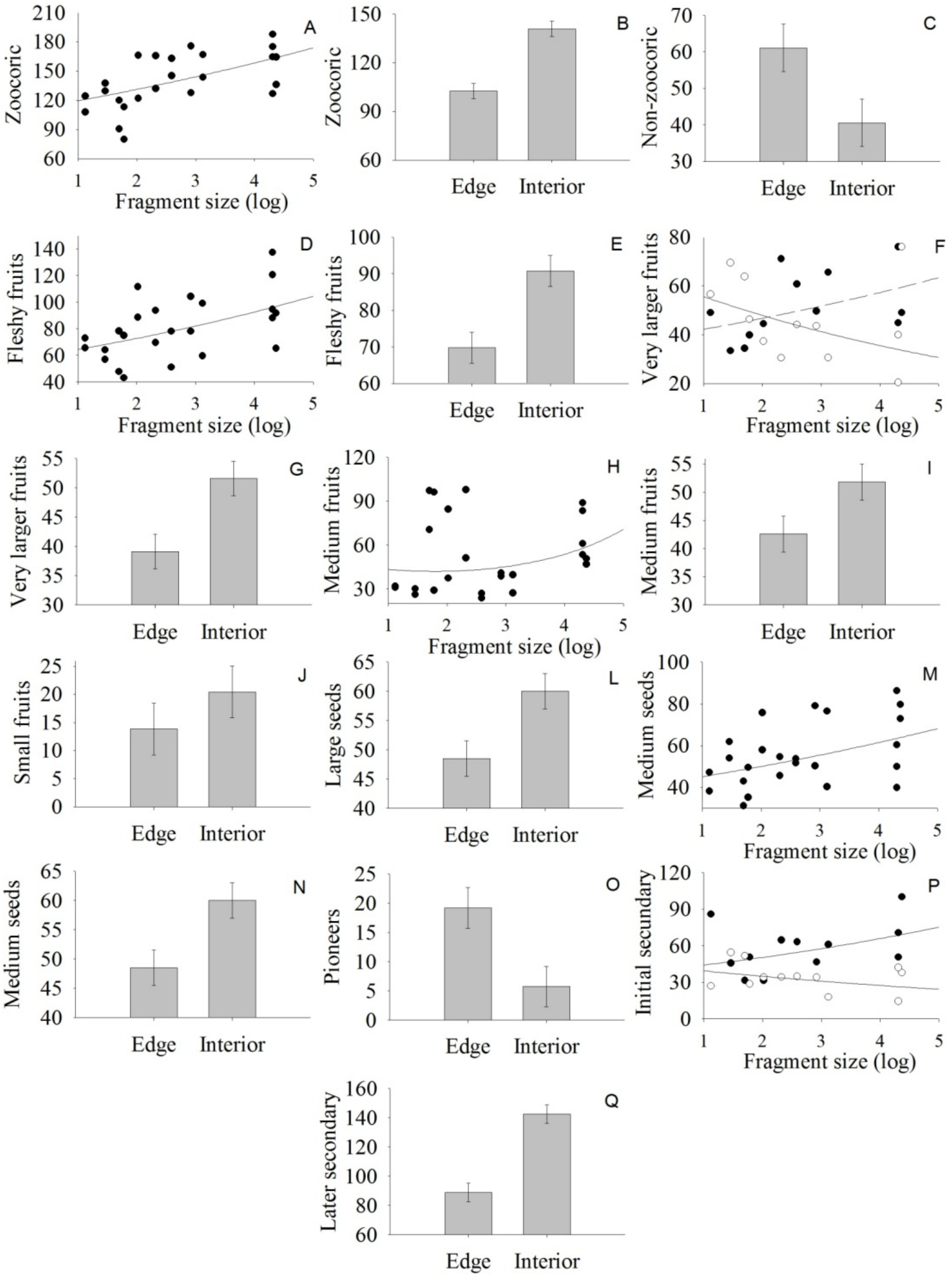

Figure S3 - Graphs of Generalized Linear Models results (only the best models according to AICc) for the fragment size and habitats effects on species abundance per functional trait. Black

(Edge) and white (Interior) circles represent values obtained after summation of raw residuals to the expected values for each variable, being assumed average values for other covariates. (A-B) Zoochoric dispersion; (C) Non-zoochoric dispersion; (D-E) Fleshy fruits; (G) Very large fruits; (H-I) Medium fruits; (J) Small fruits; (L) Larger seeds; (M-N) Medium seeds; (O) Pioneers; (P) Initial secondary; (Q) Later secondary.

Table S1 - Identification and size of fragments sampled in the study area in Southeastern Brazil.

Regional identification	**Size class**	**Size (ha)**
1. Fazenda Cúpido	Small	13.18
2. Reserva Natural Vale	Small	28.84
3. RPPN Recando das Antas	Small	50.12
4. Fazenda do Neb	Medium	60.26
5. Fazenda do Marim	Medium	104.71
6. Fazenda Caliman	Medium	208.93
7. Fazenda Rochedo	Large	389.05
8. RPPN Recando das Antas	Large	831.76
9. REBIO de Sooretama	Large	1318.26
10. REBIO de Sooretama	Control	20417.38
11. Reserva Natural Vale	Control	20417.38
12. REBIO de Sooretama	Control	23442.29

Table S2 - List of species and attributes that was used to construct the models. E=Edge species abundance; I=Interior species abundance; P=Pioneer species; I=Initial secondary species; L=Later secondary species; N=Non-zoochoric dispersion; Zoo=Zoochoric dispersion; F=Fleshy fruit; NF=Non-fleshy fruit; VL=Very large size; L=Large size; M=Medium size; S=Small size; NC=No classified.

Species	**Habitat**		**Successional classification**				**Dispersion type**			**Fruit type**			**Fruit size**					**Seed size**					**Wood density (g/cm^3)**
	E	**I**	**P**	**I**	**L**	**NC**	**N**	**Z**	**NC**	**F**	**NF**	**NC**	**VL**	**L**	**M**	**S**	**NC**	**VL**	**L**	**M**	**S**	**NC**	
Abarema cochliacarpos (B.A.Gomes) Barneby & J.W.Grimes	-	1	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.585
Acacia glomerosa Benth.	17	1	-	X	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.629
Acosmium lentiscifolium Spreng.	8	6	-	-	X	-	X	-	-	-	X	-	-	-	X	-	-	-	-	X	-	-	0.763
Actinostemon concolor (Spreng.) Müll. Arg.	-	1	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.907
Actinostemon estrellensis (Mull. Arg.) var. latifolius Pax	27	52	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.907
Aegiphila verticillata Vell.	-	1	X	-	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.657
Albizia pedicellaris (DC.) Barneby & J.W.Grimes	1	-	-	X	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.497
Albizia polycephala (Benth.) Killip	18	2	-	X	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.542
Alchornea sidifolia Klotzch.	-	2	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.378
Allophylus petiolulatus Radlk.	45	8	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.431
Alseis involuta K.Schum.	10	-	-	-	X	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.85
Amaioua intermedia (A.Rich.) Steyerm.	-	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.625
Ampelocera glabra Kuhlm.	1	4	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.674
Amphirrhox longifolia (A.St.-Hil.) Spreng	1	6	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.71
Anaxagorea silvatica R.E.Fr.	1	13	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.58
Andira fraxinifolia Benth.	1	-	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	X	-	-	-	-	0.722
Andira legalis (Vell.) Toledo	-	2	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	X	-	-	-	-	0.722
Andira ormosioides Benth.	-	1	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	X	-	-	-	-	0.722
Angostura bracteata (Nees. A. Mart.) Kallunki	1	-	-	X	-	-	X	-	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.642
Aniba canellila Mez	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.952
Aniba firmula (Nees & C. Mart.) Mez	1	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.669

Annona acutiflora Mart.	1	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.413
Annona cacans Warm.	-	2	X	-	-	-	-	X	-	X	-	-	X	-	-	-	-	-	-	X	-	-	0.413
Annona dolabripetala Raddi	3	3	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	-	X	-	-	0.413
Annona sp.	-	1	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	-	X	-	-	0.413
Aparisthmium cordatum (Juss.) Baill.	-	1	-	X	-	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.39
Apuleia leiocarpa (Vogel) J.F. Macbr.	3	10	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.788
Aspidosperma cylindrocarpon Müll. Arg.	3	1	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.637
Aspidosperma desmanthum Benth. ex Müll. Arg.	-	3	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.61
Aspidosperma discolor A.DC.	1	6	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.758
Aspidosperma illustre (Vell.) Kuhlm. & Piraja	2	3	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.739
Aspidosperma parvifolium A. DC.	1	3	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.737
Astrocaryum aculeatissimum (Schott) Burret	25	14	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	X	-	-	-	-	0.508
Astronium concinnum (Engl.) Schott	110	11	-	X	-	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.818
Astronium graveolens Jacq.	36	7	-	X	-	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.818
Bactris ferruginea Burret	2	-	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.426
Barnebydendron riedelii (Tul.) J.H. Kirkbride	1	4	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.681
Bauhinia forficata Link subsp. forficata	7	3	X	-	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.6
Bauhinia longifolia (Bong.) Steud.	1	-	X	-	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.6
Beilschmiedia linharensis Sachiko Nishida & H.van der Werff	4	4	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	X	-	-	-	-	0.563
Bixa arborea Huber	1	2	X	-	-	-	-	X	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.37
Blepharocalyx eggersii (Kiaersk.) Landrum	1	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.726
Brasiliocroton mamoninha P.E.Berry & Cordeiro	89	15	X	-	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.408
Brosimum glaucum Taub.	23	15	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.56
Brosimum guianense (Aubl.) Huber	-	3	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.843
Brosimum lactescens (S. Moore) C.C. Berg	-	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.656
Byrsonima cacaophila W.R. Anderson	-	2	X	-	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.646
Byrsonima stipulacea (Juss.) Nied.	1	3	X	-	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.709
Calycophyllum papillosum J.H. Kirkbr.	-	1	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.708

Calyptranthes lucida var. *polyantha* (Berg) C.D.Legrand	2	21	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.86
Campomanesia espiritosantensis Landrum	-	5	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.73
Campomanesia guazumifolia (Cambess.) O.Berg	3	7	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.73
Campomanesia lineatifolia Ruiz et Pav.	4	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.73
Cariniana estrellensis (Raddi.) Kuntze	-	1	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.565
Cariniana legalis (Mart.) Kuntze	1	11	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.483
Carpotroche brasiliensis (Raddi.) A. Gray	11	21	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	-	-	-	X	-	0.45
Caryodendron grandifolium Pax	-	4	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.65
Caryodendron janeirense Müll.Arg	1	-	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.65
Casearia arborea (L.C.Richard) Urban	1	-	-	X	-	-	-	X	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.595
Casearia commersoniana Cambess.	3	4	-	-	X	-	-	X	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.664
Casearia javitensis H.B. & K.	2	-	-	X	-	-	-	X	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.753
Casearia oblongifolia Cambess.	5	2	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.664
Casearia sp. new species.1	6	1	-	X	-	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.664
Casearia sp. new species.2	10	5	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.664
Casearia sp.1	-	1	-	X	-	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.664
Casearia sp.2	1	-	-	X	-	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.664
Casearia sylvestris Sw.	1	-	X	-	-	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.68
Casearia ulmifolia Vahl. ex Vent.	3	4	-	X	-	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.664
Cecropia glaziovi Snethl.	2	-	X	-	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.33
Cecropia hololeuca Miq.	1	-	X	-	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.33
Cedrela odorata Linn.	3	-	X	-	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.427
Ceiba pubiflora (A. St.-Hil.) K. Schum.	1	1	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.365
Centrolobium sclerophyllum Lima	1	-	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.655
Chamaecrista aspleniifolia (H.S.Irwin & B). H.S. Irwin & Barneby	1	3	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.903
Chamaecrista bahiae (Irwin) Irwin & Barneby	1	-	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.903
Chamaecrista ensiformis (Vell.) Irwin & Barneby	-	9	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.924
Chamaecrista sp.	1	-	X	-	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.903
Chomelia pubescens Cham. & Schltdl.	1	1	X	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.57

Chrysobalanaceae	3	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.799
Chrysophyllum gonocarpum (Mart. & Eichler ex Miq.) Engl.	7	10	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.775
Chrysophyllum januariense Eichler	3	4	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.775
Chrysophyllum lucentifolium Cronquist	11	5	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.787
Chrysophyllum sp.	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.775
Chrysophyllum splendens Spreng.	3	4	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.775
Clarisia ilicifolia (Spreng.) Lanj. & Rossb.	5	10	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.58
Clarisia racemosa Ruiz & Pav.	4	5	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.585
Cnidoscolus oligandrus (Mull. Arg.) Pax	6	-	X	-	-	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.552
Coccoloba tenuiflora Lindau	5	1	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.568
Coccoloba warmingii Meisn	2	1	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.568
Connarus detersus Planch.	2	1	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.52
Copaifera langsdorffii Desf.	-	1	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.6
Copaifera lucens Dwyer	7	21	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.615
Cordia acutifolia Fresen.	4	2	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.485
Cordia ecalyculata Vell.	10	2	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.485
Cordia magnoliaefolia Cham.	-	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.485
Cordia sp.1	2	-	X	-	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.485
Cordia sp.2	-	1	X	-	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.485
Cordia trichoclada DC.	-	1	X	-	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.485
Cordia trichotoma (Vell.) Arráb. ex Stend.	1	-	X	-	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.56
Couepia belemii Prance	1	-	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.789
Couepia schottii Fritsch	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.789
Couratari asterotricha Prance	20	11	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.51
Couratari macrosperma A.C. Smith	3	3	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.67
Coussapoa curranii Blake	-	3	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.461
Coussarea contracta (Walp.) Benth. & Hook. ex Mull. Arg.	-	4	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.61
Coutarea hexandra (Jacq.) K. Schum.	-	1	-	X	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.6
Crepidospermum atlanticum D.C. Daly	1	8	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.578

Cryptocarya citriformis (Vellozo) P.L.R. Moraes	1	-	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.597
Cryptocarya saligna Mez.	1	-	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.597
Cunuria sp.	2	2	-	X	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.552
Cupania cf. *scrobiculata* L.C. Rich.	14	9	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.628
Cupania emarginata Cambess.	1	1	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.622
Cupania oblongifolia Mart.	2	-	X	-	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.622
Cupania rugosa Radlk.	5	4	X	-	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.622
Cupania sp.	1	-	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.622
Dalbergia elegans A.M. Carvalho	1	-	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.8
Dalbergia nigra (Vell.) Allemao ex Benth.	-	4	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.749
Deguelia longeracemosa (Benth.) Az.-Tozzi	7	1	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.726
Dendropanax cuneatus (DC.) Decne. & Planch.	1	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.423
Dialium guianense (Aubl.) Sandwith	12	30	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.867
Dilodendron elegans (Radlk.) Gentry & Steyerm.	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.617
Dimorphandra sp. new species	-	6	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.742
Diospyros brasiliensis Mart. ex Miq.	2	1	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	-	X	-	-	0.573
Diplotropis incexis Rizzini & A.Mattos	-	2	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.75
Drypetes sp.	2	5	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.914
Duguetia chrysocarpa Maas	1	1	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	-	X	-	-	0.757
Dulacia sp.	-	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.569
Duroia valesca C. Persson & Delprete	2	1	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	-	-	-	X	-	0.772
Ecclinusa ramiflora Mart.	13	28	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.637
Emmotum aff. *nitens* (Benth.) Miers.	-	5	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.727
Ephedranthus sp. new species.1	1	-	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.585
Ephedranthus sp. new species.2	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.585
Eriotheca candolleana (K. Schum.) A. Robyns	3	6	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.46
Eriotheca macrophylla (K. Schum.) A. Robyns	19	27	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.46
Erythroxylum columbinum Mart.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.71
Erythroxylum pulchrum A. St.Hil.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.71

Eschweilera ovata (Cambess.) Miers	5	19	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	0.9
Esenbeckia grandiflora Mart. subsp. *grandiflora*	-	3	-	X	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.642
Eugenia bahiensis DC	-	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia batingabranca Sobral	-	8	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia beaurepaireana (Kiaersk.) C.D.Legrand	4	5	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia brasiliensis Lam.	-	3	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia cf. *badia* O.Berg	2	8	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia cf. *moonioides* Berg	1	3	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia cf. *tinguyensis* Cambess.	16	35	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia excelsa O.Berg	13	14	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.726
Eugenia fluminensis Berg	-	16	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia gemmiflora O. Berg	-	3	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	X	-	-	-	0.726
Eugenia handroi (Mattos) Mattos	-	2	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia itapemirimensis Cambess.	7	20	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia ligustrina Berg	1	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.726
Eugenia macrosperma DC.	3	7	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia platyphylla O.Berg	32	21	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.726
Eugenia platysema Berg	3	5	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia plicatocostata O.Berg	1	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia prasina O.Berg	7	18	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia sp.1	1	3	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia sp.2	-	2	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia sp.3	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia sp.4	1	-	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia sp.5	2	2	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.726
Eugenia sp.6	1	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.726
Eugenia sp.7	-	6	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Eugenia sp.8	1	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.726
Eugenia subterminalis DC.	3	2	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.726

Euphorbiaceae (new species)	2	13	-	X	-	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.557
Exellodendron gracile (Kuhlmann) Prance	1	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.707
Exostyles venusta Schott ex Spreng.	4	1	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.681
Ficus cyclophylla (Miq.) Miq.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.394
Ficus gomelleira Kunth & C.D. Bouché	1	2	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.394
Ficus mariae C.C. Berg, Emygdio & Carauta	1	5	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.394
Ficus nymphaeifolia Mill.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.415
Galipea cf. *laxiflora* Engl.	5	14	-	-	X	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.642
Geissospermum laeve (Vell.) Baill.	12	17	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.782
Glycydendron espiritosantense Kuhlm.	1	4	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.681
Gomidesia martiana O. Berg.	1	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.801
Goniorrhachis marginata Taub.	6	9	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.681
Guapira noxia (Netto) Lundell	6	4	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.492
Guapira opposita (Vell.) Reitz	20	16	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.492
Guapira venosa (Choisy) Lundell	4	4	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.492
Guarea aff. *juglandiformis* Pennington	1	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.606
Guarea penningtoniana Pinheiro	-	3	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.606
Guatteria macropus Mart.	1	-	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.54
Guatteria sellowiana Schltdl.	-	1	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.54
Guazuma crinita Mart.	8	1	X	-	-	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.44
Guettarda angelica Mart. ex Müell. Arg.	6	-	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.707
Handroanthus arianeae (A.H. Gentry) S. O. Grose	8	2	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.774
Handroanthus riodocensis (A.H. Gentry) S. O. Grose	5	2	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.774
Handroanthus serratifolius (Vahl) S. O. Grose	1	-	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.924
Heisteria cf. *ovata* Benth.	1	4	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.54
Heisteria sp.	1	-	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.704
Helicostylis tomentosa (Poep. et Endl.) Rusby	2	6	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.627

Himatanthus bracteatus (A. DC.) Woodson	2	1	X	-	-	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.53
Hirtella hebeclada Moric. ex A. P. DC.	1	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.793
Hirtella sprucei Benth.ex Hook.f.	-	3	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.793
Hornschuchia citriodora D. M. Johnson	2	-	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.585
Humiriastrum spiritu-sancti Cuatrec	-	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.668
Hydrogaster trinervis Kuhlm.	9	18	-	X	-	-	X	-	-	-	X	-	-	X	-	-	-	-	X	-	-	-	0.443
Hymenaea aurea Y.T.Lee & Langenheim	5	5	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.79
Hymenaea courbaril L.	3	1	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.787
Indet. 1	1	-	-	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-
Indet. 2	-	1	-	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-
Indet. 3	1	4	-	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-
Indet. 4	1	-	-	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-
Indet. 5	-	1	-	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-
Inga aff. *cylindrica* (Vell.) Mart.	-	2	-	X	-	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.576
Inga cabelo T.D. Penn.	4	1	-	X	-	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.592
Inga capitata Desv.	-	2	-	X	-	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.576
Inga exfoliata T.D. Penn. & F.C.P. García	-	2	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.576
Inga flagelliformis (Vell.) Mart.	9	10	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.576
Inga hispida Schott. ex Benth.	1	3	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.576
Inga striata Benth.	1	-	-	X	-	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.576
Inga thibaudiana subsp. *thibaudiana* T.D. Penn.	5	-	-	X	-	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.637
Ixora warmingii Mull. Arg.	4	2	X	-	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.382
Jacaranda puberula Cham.	4	7	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.265
Jacaratia heptaphylla (Vell.) A. DC.	4	4	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	-	-	X	-	0.39
Joannesia princeps Vell.	37	15	X	-	-	-	-	X	-	X	-	-	X	-	-	-	-	X	-	-	-	-	0.628
Kielmeyera occhioniana Saddi	2	-	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.597
Lauraceae (new species)	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.818
Lecythis lanceolata Poir.	9	5	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.83
Lecythis lurida (Miers) S.A.Mori	19	13	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.852

Lecythis pisonis Cambess.	-	4	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.818
Lecythis sp.	1	3	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.823
Licania belemii Prance	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.816
Licania heteromorpha Benth. var. *heteromorpha*	1	-	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.88
Licania kunthiana Hook.f.	3	6	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.823
Licania salzmannii (Hook.) Fritsch.	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.823
Licania sp.	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.815
Licaria bahiana Kutz	1	3	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.726
Lonchocarpus cultratus (Vell.) A.M.G. Azevedo & H.C. Lima	10	4	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.507
Luehea mediterranea (Vell.) Angely	13	3	X	-	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.616
Mabea cf. *fistulifera* Mart.	1	3	X	-	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.78
Machaerium fulvovenosum H.C.Lima	46	9	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.78
Machaerium ovalifolium Glaziou ex Rudd	2	1	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.604
Macrothumia kuhlmannii (Sleumer) M.H.Alford	9	7	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.884
Manilkara bella Monach.	4	4	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.884
Manilkara salzmannii (A.DC.) H.J.Lam	2	4	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.484
Margaritaria nobilis Linn.f.	4	2	-	X	-	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.936
Marlierea estrellensis Berg	-	4	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.936
Marlierea excoriata Mart.	1	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.936
Marlierea grandifolia O. Berg	2	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.936
Marlierea obversa Legrand.	3	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.936
Marlierea sucrei G.M. Barroso et Peixoto	3	7	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.801
Marlierea clausseniana (O.Berg) Kiaersk.	2	5	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.75
Matayba discolor Radlk.	1	-	-	X	-	-	-	X	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.82
Matayba guianensis Aubl.	3	2	-	X	-	-	-	X	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.745
Maytenus cestrifolia Reiss.	3	3	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.745
Maytenus multiflora Reiss.	4	3	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.745
Maytenus patens Reiss.	1	-	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.637
Melanopsidium nigrum Colla	1	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.9

Melanoxylon brauna Schott.	16	8	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.689
Melicoccus espiritosantensis Acev.-Rodr.	4	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.642
Metrodorea maracasana Kaastra	2	12	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.62
Miconia cf. *cinnamomifolia* (DC.) Naudin	1	-	X	-	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.62
Miconia cf. *rimalis* Naud.	-	1	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.75
Miconia lepidota Schrad. et Mart. ex DC.	1	-	X	-	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.71
Miconia prasina (Sw.) DC.	1	-	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.65
Micropholis aff. *gnaphaloclados* Pierre	3	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.65
Micropholis crassipedicellata (Mart. & Eichler.) Pierre	1	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.65
Micropholis cuneata Pierre ex Glaziou	2	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.65
Micropholis gardneriana (A.DC.) Pierre	1	4	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.681
Moldenhawera papillanthera L.P.Queiroz, G.P.Lewis & R.Allkin	5	8	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.665
Mollinedia marquetiana A.L. Peixoto	2	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.665
Mollinedia ovata Ruiz & Pav.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.637
Molopanthera paniculata Turcz.	-	1	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.691
Monilicarpa brasiliana (Banks ex DC.) Cornejo & Iltis	2	-	-	X	-	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.836
Mouriri arborea Gardner	-	4	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.836
Mouriri glazioviana Cogn.	-	3	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.801
Myrcia eumecephylla (O.Berg) Nied.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.81
Myrcia fallax DC.	3	2	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.801
Myrcia follii Barroso et Peixoto	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.801
Myrcia isaiana G.M. Barroso et Peixoto	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.801
Myrcia lineata (Berg) G.M. Barroso	4	4	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.801
Myrcia multiflora (L) DC.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.801
Myrcia riodocensis G.M. Barroso et Peixoto	2	2	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.801
Myrcia rostrata DC.	-	3	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.7
Myrciaria aureana Mattos	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.7
Myrciaria ferruginea O. Berg	1	-	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.755

Myrciaria floribunda (West. ex Willd.) O. Berg	6	20	-	-	X	-	-	X	-	X	-	-	-	X	-	X	-	-	-	X	X	-	0.7
Myrciaria tenella (DC.) O.Berg	1	-	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.775
Myrocarpus frondosus Allemao	3	2	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.743
Myrtaceae	2	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.651
Naucleopsis oblongifolia (Kuhlm.) Carauta	4	11	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.62
Neea floribunda Poepp. & Endl.	2	1	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.691
Neocalyptrocalyx nectarea (Vell.) Hutch.	1	3	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.743
Neomitranthes langsdorffii (O.Berg) J.R. Mattos	6	6	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.642
Neoraputia alba (Nees & Mart.) Emmerich	28	23	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.501
Ocotea argentea Mez	1	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.501
Ocotea conferta Coe Teixeira	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.501
Ocotea confertiflora (Meisn.) Mez	7	10	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.501
Ocotea elegans Mez	3	16	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.501
Ocotea lancifolia (Schott) Mez	3	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.462
Ocotea leucoxylon (Sw.) de Lanessan s.l.	1	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.501
Ocotea nitida (Meissn.) J.G.Rohwer	1	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.501
Ocotea nutans (Nees) Mez	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.77
Ocotea odorifera (Vell.) Rohwer	1	3	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.501
Ocotea pluridomatiata A. Quinet	1	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.501
Ocotea sp.	-	6	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.621
Ormosia arborea (Vell.) Harnu	1	1	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.621
Ormosia nitida Vogel	-	1	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.774
Ouratea cuspidata (A.St.-Hil.) Engl.	-	1	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.774
Ouratea sp.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.748
Oxandra martiana (Schltdl.) R.E.Fr.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.748
Oxandra nitida R.E. Fries	1	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.748
Oxandra reticulata Maas	-	3	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.448
Pachira stenopetala Casar.	1	3	-	X	-	-	-	X	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.78
Parapiptadenia pterosperma (Benth.) Brenan	10	5	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.704

Parinari excelsa Sabine	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.707
Parinari parvifolia Sandw.	4	4	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.59
Pausandra morisiana (Casar.) Radlk.	2	17	-	-	X	-	-	X	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.443
Pavonia crassipedicellata Krapov.	2	1	-	X	-	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.598
Paypayrola blanchetiana Tul.	1	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.792
Peltogyne angustiflora Ducke	-	5	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.647
Pera leandri Baill.	1	3	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.647
Pera sp.	-	2	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.395
Picramnia ramiflora Planch.	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.395
Picramnia sellowii Planch.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.78
Piptadenia paniculata Benth.	5	1	X	-	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.3
Pisonia aff. *ambigua* Heimerl	1	5	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.792
Platymiscium floribundum Vogel	1	-	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.7
Plinia grandifolia (Mattos) Sobral	1	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.7
Plinia involucrata (Berg) McVaugh.	3	34	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.7
Plinia renatiana G.M.Barroso & Peixoto	-	19	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.7
Plinia stictophylla Barroso & Peixoto	1	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.62
Poecilanthe falcata (Vell.) Heringer	2	-	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.833
Pogonophora schomburgkiana Miers ex Benth.	1	1	-	X	-	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.426
Polyandrococos caudescens (Mart.) Barb. Rodr.	30	9	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	X	-	-	-	-	0.584
Polygala pulcherrima Kuhlm.	2	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	-
Posoqueria latifolia (Rudge) Roem & Schult.	-	4	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	-	X	-	-	0.38
Pourouma guianensis Aubl. subsp. guianensis	1	-	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.39
Pourouma mollis Trécul ssp. mollis	-	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.783
Pouteria aff. *bapeba* T.D.Pennington	3	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.964
Pouteria aff. *filipes* Eyma	9	8	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.783
Pouteria bangii (Rusby) T.D.Pennington	7	4	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.783
Pouteria bullata (S.Moore) Baehni	1	3	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.783
Pouteria butyrocarpa (Kuhlm.) T.D. Penn.	1	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.783

Pouteria coelomatica Rizzini	5	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.58
Pouteria durlandii (Standl.) Baehni	-	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.874
Pouteria hispida Eyma	14	12	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.737
Pouteria macrophylla (Lam) Eyma	2	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.783
Pouteria macrostachiosa Pennington	2	7	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.76
Pouteria pachycalyx T.D. Penn.	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.783
Pouteria psammophila (Mart.) Radlk.	1	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.876
Pouteria reticulata (Engl.) Eyma	6	6	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.783
Pouteria sp.1	4	-	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.783
Pouteria sp.2	2	-	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.783
Pouteria sp.3	-	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.92
Pouteria venosa subsp. *amazonica* T.D.Pennington	6	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.731
Pradosia lactescens (Vellozo) Radlk.	14	6	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.572
Protium brasiliense (Spreng.) Engl.	1	-	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.629
Protium heptaphyllum (Aubl.) Marchand.	15	9	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.572
Protium warmingianum Marchand	16	20	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.8
Pseudima frutescens (Aubl.) Radlk.	6	8	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.278
Pseudobombax longiflorum (Mart. & Zucc.) A. Robyns	4	4	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.664
Pseudopiptadenia contorta (DC.) G.P.Lewis & M.P.M.de Lima	23	3	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.664
Pseudopiptadenia psilostachya (DC.) G.P. Lewis & M.P. Lima	8	5	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.37
Pseudoxandra spiritus-sancti Maas	2	23	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.52
Psicotria sp.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.684
Psidium cauliflorum Landrum & Sobral	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.684
Psidium longipetiolatum D.Legrand	1	-	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.684
Psidium oblongatum O.Berg	4	5	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.684
Psidium sartorianum (Berg) Nied.	-	2	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.427
Pterocarpus rohrii Vahl.	18	15	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.59
Pterygota brasiliensis Fr. All.	9	4	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.65
Qualea jundiahy Warm.	-	2	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.633

Qualea megalocarpa Stafleu	3	3	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.499
Quararibea penduliflora (A.St.Hil.) K. Schum.	15	25	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.69
Randia armata D.C.	7	4	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.482
Rauvolfia capixabae I. Koch & Kin.-Gouv.	-	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.642
Ravenia infelix Vell.	2	4	-	-	X	-	X	-	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.787
Rhamnidium glabrum Reissek	4	-	X	-	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.654
Rheedia gardneriana Triana & Planch.	1	5	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.652
Rinorea bahiensis (Moric.) Kuntze	36	123	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.652
Rinorea sp.	1	6	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.652
Rudgea sp.	-	1	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.689
Sapindaceae	1	-	-	X	-	-	X	-	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.421
Sapium glandulatum (Vell.) Pax.	2	3	-	X	-	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.453
Schefflera morototoni (Aubl.) Maguire, Steyermark & Frodin	4	2	X	-	-	-	-	X	-	X	-	-	-	-	-	X	-	-	-	-	X	-	0.723
Schoepfia brasiliensis A. DC.	4	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.723
Schoepfia obliquifolia Turcz.	3	10	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.78
Senefeldera multiflora Mart.	41	140	-	-	X	-	-	X	-	-	X	-	-	-	X	-	-	-	-	-	X	-	0.474
Simaba cedron Planchon	3	1	-	-	X	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.419
Simaba subcymosa A. St. Hil. & Tul.	1	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.378
Simaruba amara Aubl.	4	3	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.66
Simira glaziovii (K. Schum.) Steyermark	1	3	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.66
Simira grazielae A. L. Peixoto	3	2	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.66
Simira sampaioana (Standl.) Steyerm.	3	1	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.656
Siparuna reginae (Tul.) A. DC.	-	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.806
Sloanea aff. *granulosa* Ducke	-	3	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.75
Sloanea eichleri K. Schum.	4	3	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.806
Sloanea garckeana K. Schum.	3	2	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.28
Solanum sooretamum Carvalho	17	2	X	-	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.578
Sorocea guilleminiana Gaudich.	14	32	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.665

Sparattosperma leucanthum (Vell.) K. Schum.	1	1	X	-	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.395
Spondias macrocarpa Engl.	12	1	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.395
Spondias venulosa Mart. ex Engl.	2	-	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.661
Stephanopodium blanchetianum Baill.	-	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.419
Sterculia elata Ducke	1	1	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.51
Sterculia speciosa Ducke	16	20	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.34
Styrax glabratum Schott.	-	1	X	-	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.834
Swartzia acutifolia Vogel	1	3	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.834
Swartzia apetala Raddi	5	12	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.834
Swartzia linharensis Mansano	2	1	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.9
Swartzia myrtifolia var. *elegans* (Schott) R.S.Cowan	2	1	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.834
Swartzia simplex var. *continentalis* Urban	10	11	-	-	X	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.68
Sweetia fruticosa Spreng.	4	1	-	-	X	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.426
Syagrus botryophora (Mart.) Mart.	7	9	-	X	-	-	-	X	-	X	-	-	X	-	-	-	-	-	X	-	-	-	0.49
Symplocos pycnobotrya Mart. ex Miq.	1	-	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.774
Tabebuia cf. *elliptica* (DC.) Sandwith	1	-	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.774
Tabebuia obtusifolia (Cham.) Bureau	2	4	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.774
Tabebuia roseo-alba (Ridley) Sandwith	7	1	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	-	X	-	0.469
Tabernaemontana salzmanni A. DC.	2	1	X	-	-	-	-	X	-	X	-	-	X	-	-	-	-	-	-	X	-	-	0.56
Tachigali pilgeriana (Harms) Oliveira-Filho	7	2	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	X	-	-	-	-	0.775
Talisia intermedia Radlk.	6	2	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.457
Tapirira guianensis Aubl.	7	1	X	-	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.81
Terminalia argentea Mart.	3	1	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.73
Terminalia glabrescens Mart.	2	2	-	X	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.68
Terminalia kuhlmannii Alwan & Stace	14	16	-	X	-	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.54
Thyrsodium spruceanum Benth.	11	11	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	X	-	-	-	0.608
Toulicia patentinervis Radlk.	-	2	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.679
Tovomita brevistaminea Engl.	1	1	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	X	-	-	0.46
Trichilia aff. *surumuensis* C.DC.	1	8	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.635

Trichilia casaretti C.DC.	49	19	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.635
Trichilia elegans A. Juss. subsp. *elegans*	-	1	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.635
Trichilia lepidota subsp. *schumanniana* (Harms) T.D.Pennington	21	21	-	-	X	-	-	X	-	-	X	-	-	-	X	-	-	-	-	X	-	-	0.635
Trichilia pallens C. DC.	8	10	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.548
Trichilia quadrijuga Kunth. subsp. *quadrijuga*	3	13	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.635
Trichilia silvatica C. DC.	1	5	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.635
Trichilia sp.	16	6	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	X	-	-	0.635
Trigoniodendron spiritusanctense E.F. Guim. & Miguel	-	1	-	-	X	-	X	-	-	-	X	-	-	X	-	-	-	-	-	X	-	-	-
Unonopsis renati Maas & Westra	1	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	-	X	-	0.559
Vatairea heteroptera (Allem.) Ducke ex de Assis Iglesias	5	6	-	X	-	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.67
Vataireopsis araroba (Aguiar) Ducke	1	-	-	X	-	-	-	X	-	-	X	-	X	-	-	-	-	-	X	-	-	-	0.634
Virola gardneri (A.DC.) Warb.	12	27	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	X	-	-	-	0.45
Vitex aff. *megapotamica* (Spreng.) Moldenke	-	1	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.553
Vitex montevidensis Cham.	2	2	-	X	-	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	0.553
Vochysia angelica M.C. Vianna & Fontella	2	3	-	X	-	-	X	-	-	-	X	-	X	-	-	-	-	-	-	X	-	-	0.457
Xylopia ochrantha Mart.	1	-	-	-	X	-	-	X	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.57
Xylopia sericea A. St.-Hil.	-	1	-	X	-	-	-	X	-	-	X	-	-	X	-	-	-	-	-	-	X	-	0.57
Zanthoxylum aff. *retusum* (Albuq.) P.G. Waterman	1	-	-	X	-	-	-	X	-	-	X	-	-	-	-	X	-	-	-	-	X	-	0.601
Ziziphus glaziovii Warm.	3	6	-	-	X	-	-	X	-	X	-	-	-	-	X	-	-	-	-	-	X	-	0.838
Zollernia latifolia Benth.	3	1	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	1.05
Zollernia modesta A.M.de Carvalho & R.C.Barneby	4	6	-	-	X	-	-	X	-	X	-	-	-	X	-	-	-	-	-	X	-	-	1.005

V. CAPÍTULO III

CAN REDD+ PROVIDE CARBON AND BIODIVERSITY CO-BENEFITS IN A FRAGMENTED TROPICAL FOREST LANDSCAPE?

ABSTRACT – To mitigate the impact of greenhouse gases, the United Nations created a mechanism to advance forest carbon and biodiversity saving initiatives through Reducing Emissions from Deforestation and Forest Degradation (REDD+). However some methods to recognize the co-benefits among carbon and biodiversity have failed in highly fragmented landscapes. Therefore our study tested the potential for the existence of carbon-biodiversity co-benefits in a Tableland Atlantic Rain Forest in Brazil. Inside 240 10x10m plots , we measured three rsources of carbon present in forest fragments: trees, lianas and standing dead trees. We then related this carbon sources to species richness, community structure, endemic species richness and IUCN Red listed tree species. To evaluate these relationships we used generalized linear mixed models, selecting the best performing model on the basis of their corrected Akaike Information Criterion value, ideal for small sample sizes (AICc). We measured a total of 4,140 trees, 8,236 lianas and 277 standing dead tree individuals. We estimated that the forest fragments we sampled contain 424.39Mg ha^{-1} of carbon and 443 species of trees, of which 188 are Atlantic Forest endemic species and 36 are considered as threatened species by the IUCN. Our results showed there is a significant spatial congruence between biodiversity and carbon stocks in fragmented landscapes of tropical forest. This relationship, however, is strongerin larger fragments, where carbon stocks are significantly larger and the number of species with high conservation value is greater. In conclusion, the REDD+ co-benefits scheme could be use in a fragmented landscape, even one subjected to high fragmentation levels. This suggests that additional REDD+ funds could be used to enhance the carbon and biological value through the management of forest fragments

Keywords: REDD+; Safeguards; Carbon; Forest management; Biodiversity value; Threatened species; Biomass.

Introduction

Tropical forests play multiple roles in climate regulation, from local to global scales (Laurance 2004). In particular, they currently harbour the largest carbon stores on Earth (Lewis 2006; Laurance 2008). However, tropical forests are being rapidly degraded and converted, with ~83 million hectares of forest converted to agriculture alone between 1980's and 90's (Gibbs et al. 2010), and with the rate of forest loss increasing by 0.52% per year (Frédéric Achard et al. 2002). This deforestation is second only to the burning of fossil fuels as the key emitter of greenhouse gases (Laurance 1998a), driving potentially irreversible global climatic change (Solomon et al. 2008).

The United Nations, Reducing Emissions from Deforestation and Forest Degradation (REDD) mechanism was created to allow developed nations to pay developing nations to advance forest carbon saving initiatives in a bid to slow the rate of carbon release and thus climate change (Grainger et al. 2009). REDD has since been revised as REDD+ to incorporate payments that consider the conservation of biodiversity, via the protection of carbon stocks (Grainger et al. 2009). Thus, the integration of these co-benefits is currently the center of global efforts for conservation (Phelps et al. 2012a,b; Gardner et al. 2012).

Recent studies have sought to identify the strength of co-benefits that could be achieved under REDD+. Some have focused on the importance of mature forests to conserve biodiversity and carbon stocks (Díaz et al. 2009; Hatanaka et al. 2010). Others aimed to maximize resource allocation according to the values of biodiversity and carbon stocks (Venter et al. 2009; Strassburg et al. 2010). However, these co-benefit relations are typically identified using course-scale layers of global biodiversity and carbon, which can be unreliable at smaller spatial scales (Strassburg et al. 2010). For instance, biodiversity layers use species maximum range extents, but within those ranges there will be many degraded or over-hunted forests that now lack the full compliment of species.

What we now require are empirical studies that detail the precise nature of overlap between carbon and biodiversity at smaller spatial scales, and particularly so in ecosystems that have already undergone some kind of anthropogenic degradation (see Gardner et al. 2012). Without such assessments, it precludes the reliable application of REDD+ for biodiversity conservation and carbon stock protection (UNEP-WCMC 2008; Díaz et al. 2009; Talbot 2010;

Phelps et al. 2012a,b), and may result in benefits only for carbon stocks and not for biodiversity (Lindenmayer et al. 2012).

One of the key land-use transformations that could disrupt the apparent relationship of carbon and biodiversity co-benefits is forest fragmentation. As forests are converted to agriculture, remnant patches of forest that remain have different sizes, shapes, ages and degrees of isolation, each of which changes the biological community and the functional traits of species that persist. Such fragmentation effects can thus shift tree communities from hardwood to soft wood species, causing losses in biomass contained in arboreal species and reducing the forest ability to stock carbon (Laurance et al. 1997; Laurance et al. 2002; Nascimento & Laurance 2004; Paula et al. 2011). Furthermore, fragments suffer from edge effects such as desiccation, wind disturbance, and changes in micro-climate, which shift forest dynamics and functionality (Oliveira et al. 2004; Tabarelli et al. 2010; Laurance et al. 2006; (Laurance et al. 2002; Tabarelli et al. 2010; Pütz et al. 2011) and which may further shift the relationship between biodiversity and carbon storage.

Here we investigate the impacts of forest fragmentation on carbon-biodiversity co-benefits, using the Atlantic Forest landscape as our model system. The Atlantic Forest is the second largest forest domain of Brazil (Ribeiro et al. 2009), a global hotspot of threatened biodiversity and endemism, and has suffered a high degree of deforestation (Myers et al. 2000). The biological impacts promoted by fragmentation here mirror those in the Amazon and Southeast Asia, making it a valuable model system.

We previously found in this landscape that fragmentation causes an increase in species richness and functional diversity, followed by increases in non-zoochoric and initial secondary species, with reductions in fleshy fruited and zoochoric species, which promoted changes in community structure and functionality in smaller forest patches (see the results of chapter 2). With edge creation we found drastic reductions in functional diversity and species richness, followed by increases in pioneer, initial secondary, non-fleshy fruited and non-zoochoric dispersed species.

Our previous results suggest that there are biological losses with fragmentation, but we do not yet know how those losses are related to changes in carbon storage. In addition, such changes would need to be weighted against the most important species within communities from a conservation perspective, which we define as those that are endemic or IUCN Red-listed. Here we focus on tree species biodiversity value and on the carbon stored in above-ground trees and lianas, both of which reach high levels in this region (Strassburg et al. 2010). Thus, we test three hypotheses: (1) the reduction in fragment size and the creation of edges negatively affect carbon stocks; (2) fragment size reduction and edge creation negatively affect species with high conservation value (endemic and threatened species); and (3) in fragmented landscapes, fragments can have different relations between carbon stocks and biodiversity values of conservation concern , which can be managed by REDD+ mechanism.

Construction and assumptions of models

To investigate the effects of fragmentation on Atlantic forest tree endemics and Red-listed trees, and on the amount of carbon stored in live and dead trees and lianas as well as the total amount of carbon stored in the studied landscape, we considered three classic factors from the fragmentation literature to compose our models: (i) edge and interior habitats, including control fragments; (ii) fragment size, including the interior of all fragments and controls, and (iii) the interaction between fragment size and habitat (edge versus interior) for all fragments and controls(global model).

We built the models assuming that the effects of fragmentation are derived from those of the reduction in fragment size and the creation of edge habitats, given that they can lead to microclimatic changes such as increased wind, light intensity in the understory, and air temperature, and lower humidity inside these habitats and fragments (Kapos 1989, Chen et al. 1993; Camargo & Kapos 1995). We also assumed that the reduction in fragment size can directly increase the impacts of edge effects inside fragments.

We considered in the models the following metrics of conservation value: (1) total species richness (derived from chapter 2); (2) community structure (derived from chapter 2); (3) the richness and abundance of species endemic to the Atlantic Forest; and (4) the richness and abundance of threatened species on the IUCN list (see http://www.carbon-biodiversity.net).

We considered five types of the possible co-benefit relationships between biodiversity and carbon stocks, based upon those proposed for the implementation of REDD+ by Phelps et al. (2012a,b): (1) the metrics of biodiversity and total carbon stocks are synergistic, showing positive co-benefits relationships and suggesting that protection of biodiversity can be achieved through the protection of carbon stocks, at no extra cost for REDD+ funds (Figure S2A); (2) biodiversity and carbon stocks have a negative relationship in a fragmented landscape, thus to protect biodiversity implies extra costs than those anticipated by carbon protection under REDD+ (Figure S2B); (3) there is no difference in biodiversity with increasing carbon stock, such that the most important habitats for carbon storage are selected (Figure S2C); (4) there is no difference in carbon stocks with increasing biodiversity, such that the most important habitats for biodiversity are protected under REDD+ (Figure S2D); and (5) when biodiversity metrics and carbon stock do not have linear relations, such that each is measured concurrently in the same units of landscape, we can also indicate the existence of co-benefits between carbon and biodiversity for conservation (Figure S2E-H).

Material and Methods

Study area

This study was based in the state of Espírito Santo, in Southeast Brazil. Within the region, we focused on the municipalities of Sooretama, Linhares and Jaguaré (19^{o}04'05 "S and 39^{o}57'35" W, 28- 65 m.a.s.l) (Figure S1), which contain a landscape matrix composed mainly of *Eucalyptus* spp. plantations, grasslands, coffee and papaya plantations (Rolim et al. 2005). The climate is tropical wet (Köppen classification), with an annual precipitation of 1,403 mm and a distinct dry season from May to September, when precipitation is just 33 mm per month (Peixoto & Gentry 1990). The predominant soil in the study region is Yellow Podzolic (IBGE 1987).

This region is part of the phytogeographic domain Atlantic Forest and is officially classified as Lowland Rain Forest (IBGE 1987), or Tertiary Tabelands Forest (Peixoto & Silva 1997). The study area is of high conservation importance due to the presence of two forest fragments larger than 20,000 hectares, which house a high diversity of plant and animal species (Peixoto & Silva 1997; Chiarello et al. 1999; Masden et al. 2001).

Tree sampling

Fieldwork was conducted from January 2011 to January 2012. We created permanent plots along transects within each of nine fragments (range=13.18 to 1318.26 ha; mean=333.9 ha) and two control forest blocks larger than 20,000 ha in Reserva Natural da Vale (RNV) and Reserva Biológica de Sooretama (REBIO) (Table S1). Within each fragment, we created two transects: one ~5 m into the fragment and parallel to the forest edge and one in the forest interior (≥300 m from the forest edge). Along each transect, we stationed ten 10 x 10 m plots positioned at 20 m intervals, totaling 240 plots. Due to the absence of other control forest blocks, we allocated three pairs (of 10 edge and 10 interior plots) of transects in RNV (one pair) and REBIO (two pairs), with a mean distance of 17.1 ± 10.4 km between transect pairs. All plots were on the same type of soil (Yellow Podzolic).

We sampled every live and standing dead tree individual with a diameter at breast height of ≥4.8 cm at 1.3 m above ground height (DBH) and every liana with diameter ≥1.6 cm at 10 cm above soil height (DSH). We collected in each plot samples from each live tree individual. We identified this material (just live trees) with reference to collections at the CVRD Herbarium of the Vale and the VIES Herbarium of the Federal University of Espírito Santo, and with aid from taxonomic experts in plant species identification in specific families (e.g. Myrtaceae and Sapotaceae). Botanical material collected in the fertile stage was deposited in the collection of Vale Herbarium of the Reserva Natural Vale in Linhares, ES.

Tree carbon stock and conservation status

To estimate the amount of Above Ground Biomass (AGB) in each individual live and standing dead tree we used Chave et al.'s (2006) equation:

$$p \,.\, \exp(-1.499+2.148\ \ln(D) + 0.207(\ln(D))^2 - 0.0281\ (\ln(D))^3)$$

Where p = wood density (g/cm^3) and D = diameter at breast height (DBH).

For AGB of lianas, was used Schnitzer et al.'s (2006) equation::

$$AGB = \exp(-1.484 + 2.657\ \ln(D))$$

where D = DSH (lianas). We assume that 50% of AGB of each individual is represented by carbon (Laurance et al. 1997; Malhi et al. 2004; Chave et al. 2005; IPCC 2006; Paula et al. 2011). Thus the total carbon stock for each plot and each site was the sum of all individual components: total carbon stock = live tree carbon + dead tree carbon + liana carbon.

Data for wood density in dry weight (g/cm^3) for tree species, were obtained from *The Global Wood Density (GWD) database* in the subsection *Tropical South America* (http://hdl.handle.net/10255/dryad.235; Chave et al. 2009; Zanne et al. 2009). We made two adjustments (following Flores & Coomes 2011; Hawes et al. 2012): (i) for morphospecies only identified to the family or genus level, we used the average wood density of the taxonomic group; (ii) for species not in the GWD database, we used the average wood density for the species' genus; and (iii) for standing dead tree individuals, we used the average wood density we found for the live trees within the same plot.

To classify the species endemic to the Atlantic Forest domain we used the database Flora do Brazil (List of Species of the Brazilian Flora 2012, in http://floradobrasil.jbrj.gov.br/2012 in http://floradobrasil.jbrj.gov.br/2012). We classified threatened species as those listed on the IUCN Red List (IUCN 2012) as Vulnerable, Endangered or Critically Endangered.

Data Analysis

We used Nonmetric Multidimensional Scaling (MDS) ordination analysis in the PC-ORD 6 package (McCune & Mefford 2011) to identify changes in community structure between different-sized fragments, and between edge and interior habitats (as per Chapter 2). We used the species abundance raw data from each plot for this analysis and the metric distance used was Sorensen (Bray-Curtis). We considered the NMS results arising from tree species abundance data as a measure of community structure (Barlow et al. 2010).

For models including only controls and fragment size, we used the *glm* function from the R program. Mixed models were carried out using the *glmmadmb* function from the *glmmADMB* package. We used a Poisson error distribution with a log link function for count data and Quasipoisson and Negative Binomial with log link functions when the data showed significant overdispersion. We used a Gaussian error distribution with identity link function for the rest of the data. The sites (each fragment) were codified as a random variable in all analyses (Bolker et al. 2009). We used the *dredge* function from the *MuMIn* package to test all possible combinations of the variables included in the global model. To determine which factors were most related to species richness, community structure, functional traits and functional diversity, we used an information theoretical approach based on the Akaike Information Criterion of Second Order (AICc), which is robust to small sample sizes, and the best model was indicated by the AICc lower value (Burnham et al. 2011). All analyses were performed in the R version 2.15.1 (R Development Core Team 2012).

Results

To do all of our results, we made several models with different landscape scales and with different framework. To access all of our models results, see the Table S3 in the supplementary material.

How much carbon is in the intact forest?

A total of 4,140 tree individuals, 8,236 liana individuals and 277 standing dead tree individuals were sampled during this study (Table S2). We found a total carbon stock of 265.21Mg in the interior of control fragments. Live trees accounted for the majority of this carbon (98.8%), with lianas (0.8%) and dead trees (0.4%) accounting for the remainder. Considering all sampled fragments, we found 424.39Mg ha^{-1} of carbon. Again, live trees contributed the majority of carbon stocks (93.5%), followed by lianas (3.9%) and dead trees (2.6%).

What is the impact of fragmentation on carbon stocks?

Edges had a significant negative impact on carbon stocks, with edge habitats exhibiting 3.3-fold less carbon per plot than control fragments interior (GLM; t=4.08; p<0.001; Figure 1A). The proportional contribution of live trees, dead trees and lianas to the carbon stock was different between control fragment habitats. Live trees were the main contributors to carbon stocks in the interior of these fragments (GLM; t=-4.38; p<0.0001; Figure 1B), whereas standing dead trees

(GLM; t=2.54; p<0.02; Figure 1C) and lianas had significantly higher carbon stocks in edge habitats (GLM; t=3.59; p<0.001; Figure 1D).

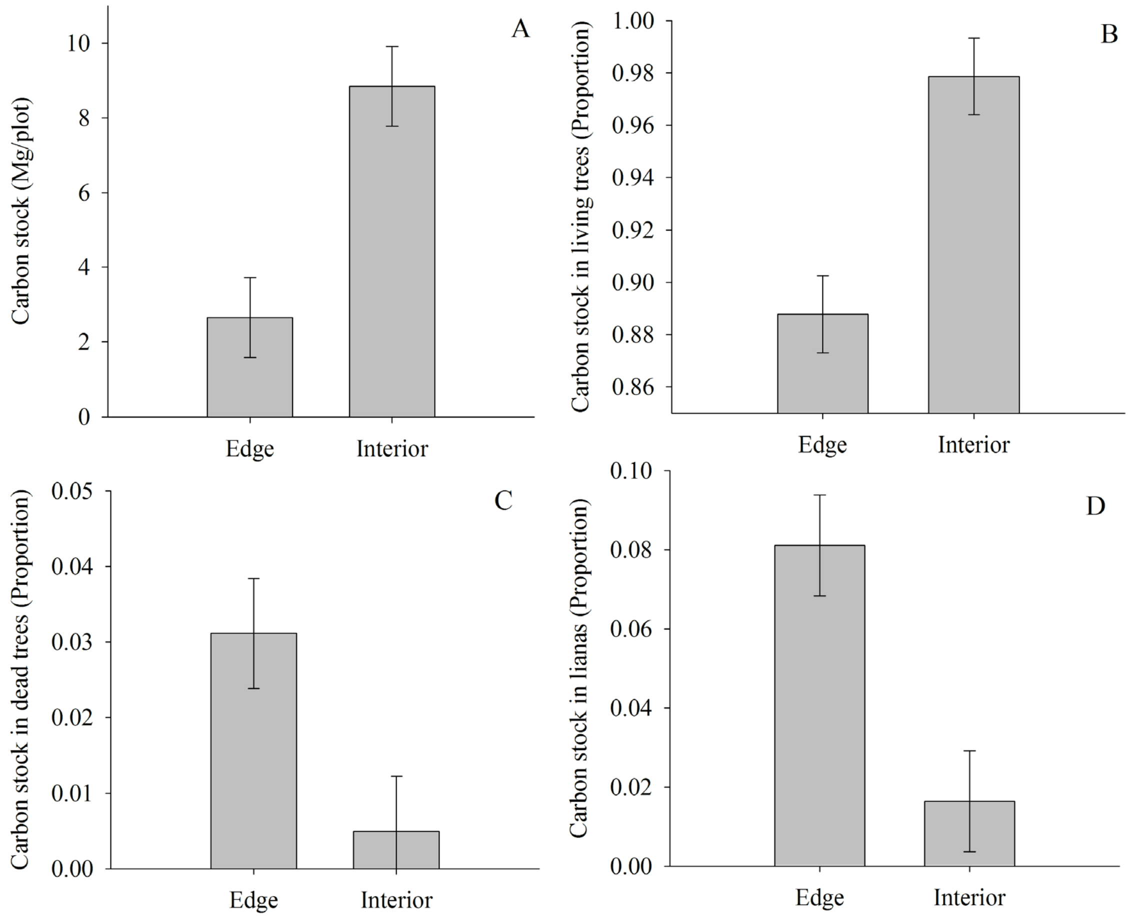

Figure 1 – Best model graphs generated for the results of habitat effect on control fragments carbon stock. (A) Habitat effect on total carbon stock; (B) Habitat effect on the proportional contribution of live trees to the total carbon stock; (C) Habitat effect on the proportional contribution of standing dead trees to the total carbon stock; (D) Habitat effect on the proportional contribution of lianas to the total carbon stock.

Considering only the impact of fragment size on carbon stocks found in the interiorof forest fragmenrs, our results show a significant negative effect of size reduction on total carbon stocks (GLM; t=4.03; p <0.01; Figure S3A). Results obtained by the model indicate a reduction of 41% on carbon stocked in small fragments biomass comparing to control fragments. The contribution of live trees to carbon stocks increased significantly with an increase in fragments size (GLM; t=3.23; p <0.01; Figure S3B). On the other hand, lianas significantly decreased their contribution to carbon stocks with an increase in fragments size (GLM; t=-3.32; p <0.01; Figure S3C), while carbon stocked in dead trees showed no influence of fragments size (GLM; t=1.94; p=0.08).

Considering the global models, our results showed that total carbon stocks were not significantly influenced only by habitat (interior and edge), but also by the interaction between habitat and fragment size (GLM; t=-3.82; p<0.01; Figure 2A). In other words, an increase in fragment size leads to a significant increase on carbon stocked in fragment interiors (F test, p<0.0001), but fragment size does not have a significant influence on carbon stocked on edges (F test, p=0.19). Carbon stocked on live trees changes significantly with a reduction in fragments size (GLM; z= 3.51; p<0.01; Figure 2B) and with edge versus interior habitat (GLM; z=-6.21; p<0.0001; Figure 2C). Thus, smallest fragments and edge habitats had the lowest proportion of carbon. Nevertheless, the carbon stocked in lianas had the highest values on edge habitats (GLM; z=-4.01; p <0.01; Figure 2D), and decreased significantly with an increase in fragment size (GLM; z=-3.71; p <0.01; Figure 2E). The proportion of carbon contained in standing dead trees did not change with fragments size (GLM; z=-0.75; p <0.35), habitat (GLM; z=-1.41; p <0.56), or their interaction (GLM; z=-1.18; p <0.12), indicating that carbon stored in dead trees is not influenced by fragmentation effects.

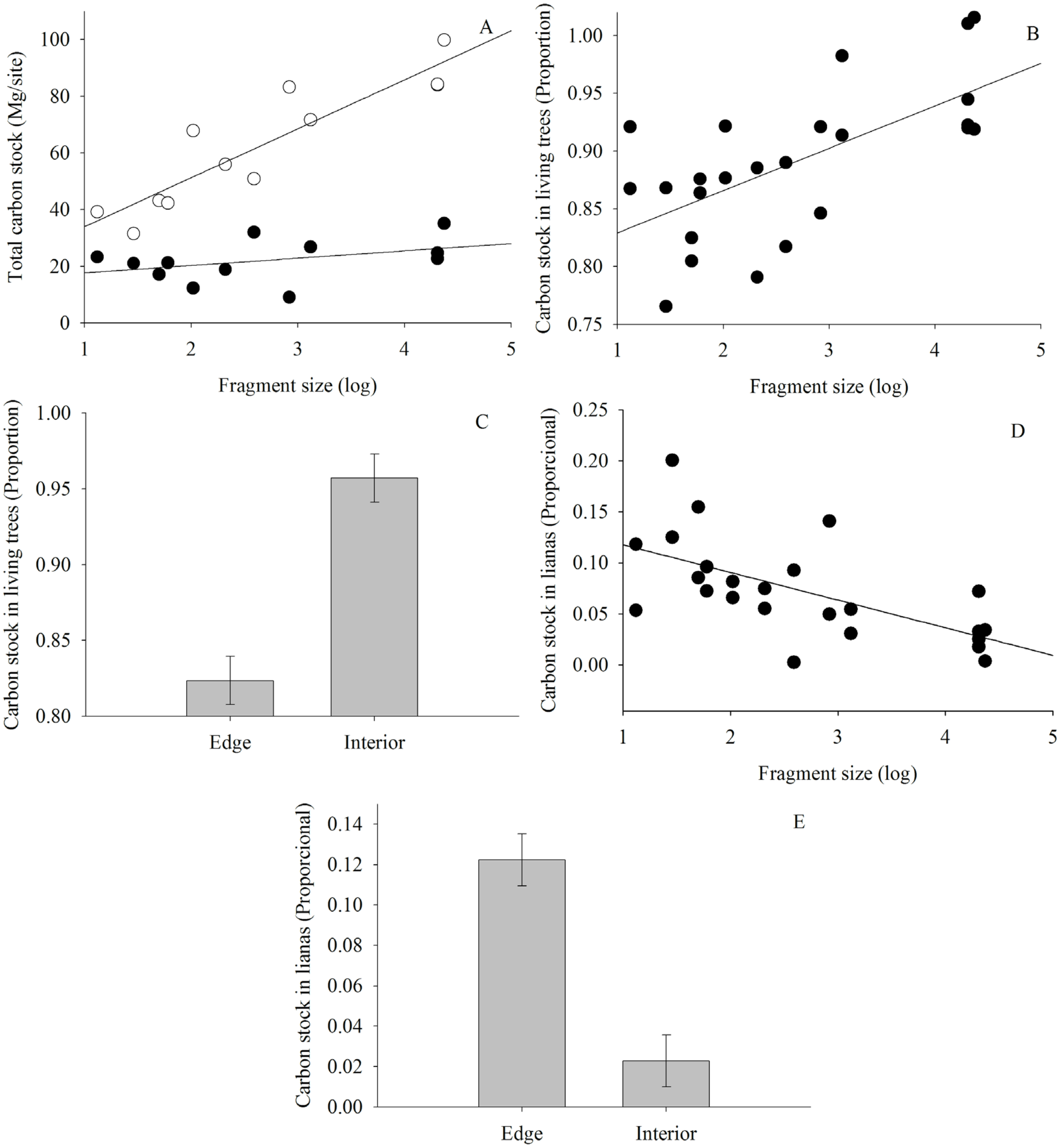

Figure 2 - Best model graphs for the effects of fragments size and habitats (global models) on carbon stock. (A) Partial residuals graphical representation showing the effects of fragments size and habitat interaction on total carbon stock - black circles = edge, white circles = interior; (B) Partial residuals graphical representation of fragment size effect on live trees proportional contribution to total carbon stock; (C) Habitat effect on live trees proportional contribution to total carbon stock; (D) Partial residual graphical representation of fragments size effect on lianas proportional contribution to total carbon stock; (E) Habitat effect on lianas proportional contribution to total carbon stock. Circles represent values obtained after the summation of raw

residuals to the expected values for each variable, being assumed average values for other covariates.

What are the impacts of fragmentation on biodiversity metrics?

We sampled a total of 4,140 tree individuals belonging to 443 species, including 3,912 individuals of 188 species that are endemic to the Atlantic Rain Forest domain and 443 individuals of 36 threatened species that are IUCN Red-listed (Table S2). Endemic species abundance was significantly higher in control fragments interior than in its edges (GLM; t=-2.27; p<0.05; Figure S4A). However, we did not find significant differences with endemic species richness (GLM; t=-0.36; p=0.72), with threatened species richness (GLM; t=-1.35; p=0.18), or with threatened abundance (GLZ; t=-0.939; p=0.35) between forest edge and interior.

The threatened species abundance (IUCN Red List) was positively related to fragment size (GLM; t=2.45; p<0.05; Figure S4B). Threatened species richness (GLM; t=-0.92 p=0.37), endemic species richness (GLM; t=-1.51; p=0.16), and endemic species abundance (GLM; t=0.88; p=0.39) showed no significant relationship with this variable.

Richness of endemic (GLM; z=0.7; p=0.5) and threatened species (GLM; z=-1.47; p=0.17) showed no significant relationship with fragmentation effects included in the global model. However, fragment size had a positive influence on threatened species abundance, with an increase in its abundance following an increase in fragment size (GLZ; z=2.41; p <0.02; Figure 3A). The best model for endemic species abundance included both fragment size (GLZ; 1.69; p=0.09) and habitat, but only habitat had a significant influence in the model (GLZ; -4.45; p <0.0001; Figure 3B), indicating that fragment interiors have greater abundance of endemic species than edge habitats.

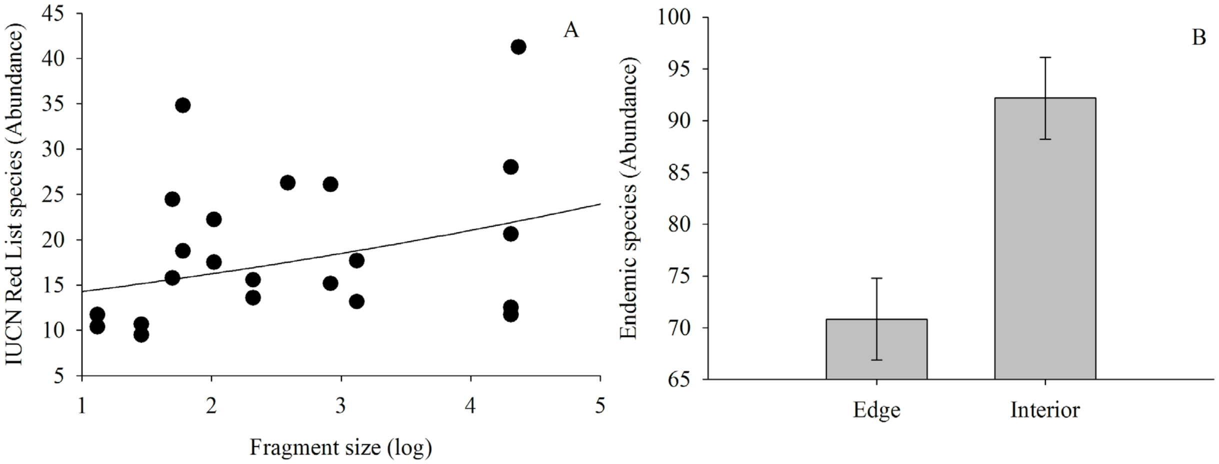

Figure 3 - Best model graphs (global models) for the results of fragmentation effects on endemic and endangered species (IUCN Red List). (A) Partial residuals graphical representation showing the effect of fragment size on endangered species abundance; (B) Habitat effect on endemic species abundance in forest fragments. Circles represent values obtained after the summation of raw residuals to the expected values for each variable, assuming average values for other covariates.

Are there co-benefits between carbon stock and biodiversity?

Our results show that species communities in fragment interiors had significantly higher carbon stocks than species communities on edges (GLM; t=3.24; p <0.01; Figure S5A) considering only control fragments. Evaluating only the effect of fragment size on the model, we found that tree community structure in the largest fragments had the highest carbon stocks (GLM; t=-2.28; p <0.05; Figure S5B).

Global models showed that changes in community structure and carbon stocks are significantly associated with habitat changes and fragments size (GLM; z=9.01; p<0.0001; Figure 4A). The global model of endemic species abundance and carbon stocks indicated a positive co-benefit, showing that areas with the highest endemic species abundance are also areas with more carbon stock (GLZ; z=3.52; p <0.01; Figure 4B). However, the carbon stock relationships with species richness (GLZ; z=0.16; p=0.87), threatened species richness (GLZ; z=0.71; p=0.49) and abundance (GLZ; z=1.01; p=0.34), and endemic species richness (GLZ; z=0.15; p=0.88) were not significant.

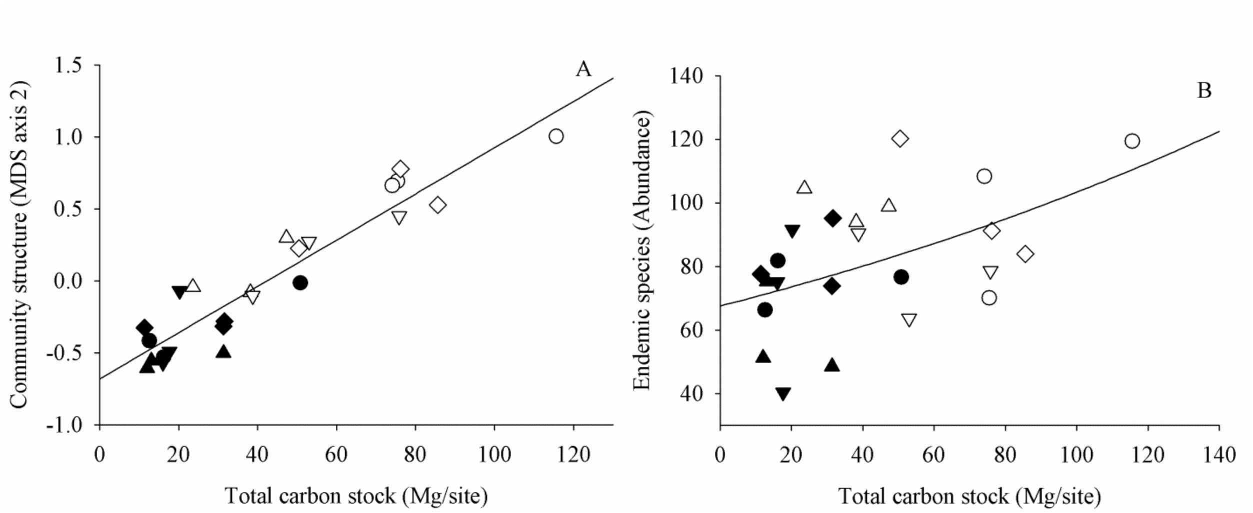

Figure 4 - Best model graphs for the results of the co-benefits relationship between biodiversity metrics and total carbon stock (global models). (A) Partial residuals graphical representation showing the relationship between community structure (MDS axis 2 - which was related with changes between community structure fragment size) and total carbon stock; (B) Partial residuals graphical representation showing the relationship between endemic species and total carbon stock. Symbols represent values obtained after the summation of raw residuals to the expected values for each variable, being assumed average values for other covariates. Black symbols=Edge; White symbols=Interior; Circle=Controls; Diamonds=Large fragments; Inverse triangle=Medium fragments; Up triangle=Small fragments.

Discussion

This is the first empirical research in any Neotropical rainforest to verify the synergisms between biodiversity conservation and carbon stocks, a vital step in understanding the potential for carbon-biodiversity co-benefits under REDD+ (e.g. UNEP-WCMC, 2008; Grainger et al. 2009; Phelps et al. 2012a,b; Talbot 2010; Gardner et al. 2012; Venter et al. 2012). It is also the first such study globally to use rigorous field data to show the existence of these co-benefits in fragmented tropical landscapes, revealing that such benefits can be found in other tropical forest ecosystems through the world. These results thus suggest that in fragmented landscapes, macro-scale models are inaccurate at detecting carbon and biodiversity co-benefits (Pfeifer et al. 2012), because these models fail to recognize the existence of high carbon stocks in highly fragmented lands (see Strassburg et al. 2010).

Impact of fragmentation on carbon stocks

Our estimates of carbon stocks in the above ground biomass of live trees within forest control and fragment interiors are similar to values obtained for most of the Amazon Forest (Houghton et al. 2000) and are high according to the scale proposed by UNEP-WCMC (2008), which considers forests with high carbon stock when they contain 273-769 Mg/ha^{-1}. Other studies conducted on Tableland Forests also showed the high structural development (basal area) of these forests (Souza et al. 1998; Jesus & Rolim 2005; De Paula et al. 2011), revealing a high productivity biomass of trees per hectare (Rolim et al. 2005) and similar structure to Terra Firme Amazon Forest (Heinsdijk et al. 1965).

Live trees made by far the largest contribution to carbon stocks, followed by lianas and dead trees, and this relationship was held across all fragment sizes. This pattern seems to be common in tropical forests (Chave et al. 2008), even in fragments under the impacts of edge effects (e.g. Laurance et al. 1997; Nascimento & Laurance 2004). However, the reduction in the proportion of live trees and the increase of lianas in carbon stocks in edge areas and small fragments (Figure 2) is indicative of a more disturbed forest structure (e.g. Laurance et al. 2001; Chave et al. 2008). Lianas interfere negatively with forest carbon stocks due to (i) resource competition with trees which leads to an increase in tree mortality and impedes forest regeneration (Laurance et al. 2001), and (ii) morphological and physiological characteristics that limit their potential to sequester carbon (Schnitzer & Bongers 2002; Laurance et al. 1997).

Although tree species with high wood density accumulate the most carbon stocks, we did not find a significant effect of fragmentation on wood density. This result suggests that small fragments can sequester and stock high levels of carbon in tree biomass. In spite of fragmentation impacts not acting directly on the wood density in our study, elsewhere the smallest fragments and edge habitats reduce the ability of trees to store carbon in live biomass (e.g. Laurance et al. 1997, Laurance et al. 1998a, Laurance et al. 2002; Nascimento & Laurance 2004; Paula et al. 2011). In these studies, reductions in the ability to accumulate carbon in tree biomass may be due to changes in forest microclimate, including increased wind, which leads to the death of large trees and impedes natural regeneration.

The impact of fragmentation on biodiversity of conservation concern

Endemic and threatened species were most prevalent in the largest fragments and in forest interiors, suggesting that these have the highest value for conservation. This result mirrors those from other tropical fragmented forests such as the Amazon Madagascar, Northeastern Australia and Borneo, ecosystems of high importance for the conservation of threatened and endemic species that have been widely cited as priority sites for biodiversity conservation (Myers 1988; Myers et al. 2000), even when considering regional scales of conservation (Ginsburg 2001). The prioritization of fragments to preserve endemic and IUCN Red List tree species thus increases with the size of the remnant in the landscape, but nevertheless, it is important to emphasize that small fragments can still harbour threatened and endemic flora, and in particular might be important in retaining landscape connectivity for populations of rare species.

In a previous study within these fragments, we also showed that the creation of forest edges resulted in a significant impact on tree community, drastically reducing species richness and functional diversity, and promoting changes in community structure and functional traits, reflected by increases in the abundances of pioneers and non-zoochoric dispersed species and decreases in abundances of shade tolerant and zoochoric dispersed species (see the results of chapter 2). Fragment size also showed significant changes on species richness, community structure and functional traits, with higher abundance of species with fleshy fruits (more resources for fauna) and more abundance of zoochoric species in larger fragments, whereas in small fragments, non-fleshy fruits and initial secondary species increased to promote significant changes on functional diversity (see the results of chapter 2). These results support the suggestion that the biological value is highest in larger fragments, with higher species richness, functional diversity and value for species of conservation concern (endemic and threatened species).

Co-benefits among carbon stock and biodiversity

Our results reveal important co-benefits between carbon stocks and biodiversity in fragmented landscapes (see Grainger et al. 2009; Venter et al. 2012; Phelps et al . 2012b). This is particularly so within the largest forest fragments, which have the greatest biodiversity and highest estimated carbon stocks per hectare. Thus, by preferentially protecting such big fragments, biodiversity protection could be implemented without costing additional funds, which is the priority assumption for REDD+ investments (Scenario 1, Figure S2A; see also Phelps et al. 2012b).

Our results also suggest that to maximise co-benefits in fragmented systems, the direction of funds to protect carbon stocks can be assessed using biodiversity as a key indicator of higher tree carbon stocks in fragments, since the protection of forest fragments with the greatest biological integrity (i.e. forests less affected by fragmentation effects, and those with more endemic and threatened species) ensures the maintenance of high carbon stocks (see Diaz et al. 2009). To do so would require tree biodiversity measurements through field work in permanent plot inventories, but using DBH as an inclusion criterion of trees in plot sampling would provide datasets to work with carbon and biodiversity at the same time, helping to construct a solid framework to REDD+ assessment, principally to further verify and measure the impacts of forest degradation on this relationship (Gardner et al. 2012).

Once the integrity of tree biodiversity and high carbon stocks in rainforests above ground biomass are maintained of (Laurance et al. 1997; Nascimento & Laurance 2004, Laurance et al. 2006; Strassburg et al. 2010; Diaz et al. 2009; Paula et al. 2011), there are potentially other important environmental services of forest fragment protection, which need to be quantified in future REDD+ co-benefit analyses. In particular, forest fragments with high biodiversity can increase the ecological services such as provisioning services (e.g. food, fibre and medicinal and cosmetic products), regulating services (e.g. local climate, soil and water regulation, and water and air purification) and cultural services (Fischlin et al. 2007), could yield significant benefit, and in some instances could themselves raise forest protection payments to supplement economic gains from carbon payments.

Implications for future assessments of carbon and biodiversity co-benefits

Our co-benefits results have important implications for our understanding of biodiversity and carbon co-benefits in fragmented tropical forest. First, our results show that macro-scale analyses introduce serious sources of error when trying to assess co-benefits under REDD+

(Hatanaka et al. 2011; Venter et al. 2012). By using coarse-scale analyses, biomes like the Atlantic Rainforest global biodiversity hotspot do not show co-benefits between biodiversity and carbon stocks (Strassburg et al. 2010; Pfeifer et al. 2012), probably because the high level of land converted to non-forest cover (e.g. agriculture) means that these biomes have low carbon stores per cell. As we show, however, forest fragments within such cells can be both carbon-rich and exceptionally biologically diverse (see Strassburg et al. 2010). Our study reveals that future remote-sensing studies must consider far more effectively the potential high value of forest fragments. To do so, we need further studies such as this work to calibrate a next generation of more reliable coarse-scale REDD+ co-benefit analyses (see Gardner et al. 2012).

Second, our results suggest that in a fragmented forest landscape, it is the assessment of biodiversity conservation value that is most important to ensure co-benefits. This is because high conservation value begets high stocks of carbon, but the same is not necessarily the case when valuing carbon stocks (Talbot 2010; Lindenmayer et al. 2012), with high carbon in planted forests that are far less valuable for biodiversity. Such an assessment inversion has the likely added bonus of providing other important environmental services (Diaz et al. 2006; Fischlin et al. 2007; Gamfeldt 2013) that correlate with high biodiversity, in addition to live tree biomass (Laurance et al. 1997; Nascimento & Laurance 2004, Laurance et al. 2006; Diaz et al. 2009; Paula et al. 2011).

Third, management to increase carbon stocks in smaller fragments can be done, since there were no differences in wood density, like selective cutting of lianas and forest restoration, which could increase the production of live tree biomass (Rey Benayas et al. 2009; Edwards et al. 2011). Indeed the smaller fragments usually belong to small-scale farmers or other civilians, which means that the increased management to enhance the potential to improve the carbon stocks of trees biomass can amplify the importance of conservation of these fragments categories via REDD+ (see Chazdon 2008). Furthermore, forest management increases opportunities for local employment in poor communities (Edwards et al. 2011), increasing the possibility thar locals will buy-into forest conservation (Dietz et al. 2003; Smith & Scherr 2003; Sachs et al. 2009), which can encourage the long-term conservation of forests fragments and their ecological services by the local stakeholders (Gardner et al. 2012).

Moreover, the existence of small fragments in highly fragmented landscapes has a recognized importance, promoting more functional connectivity than any agroforestry plantations, playing a key role in biodiversity conservation via retaining metapopulation dynamics (Laurance 2004b), and certainly increasing total carbon stocks and the amount of biodiversity retained in a fragmented landscape.

We conclude that in highly fragmented forests within hotspots of threatened biodiversity there are important synergies between carbon and biodiversity, which could protect high levels of carbon stocks while providing globally important benefits for conservation. Yet worryingly, to date these co-benefits have been overlooked by macro-scale models of co-benefits under REDD+, indicating that we urgently require a new breed of model that includes the likely biodiversity and carbon value of individual forest patches rather than averaged coarse-grained cells. Finally, our study suggests that in areas with threatened fragments, carbon payments to protect larger blocks would have stronger co-benefits, while we finish by noting that additional funds for forest protection could be gained via carbon enhancements within fragments and restoration of the surrounding matrix (REDD+) or via other Payments for Ecosystem Service mechanisms that might well also correlate with high carbon-biodiversity fragments.

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SUPPLEMENTARY MATERIAL

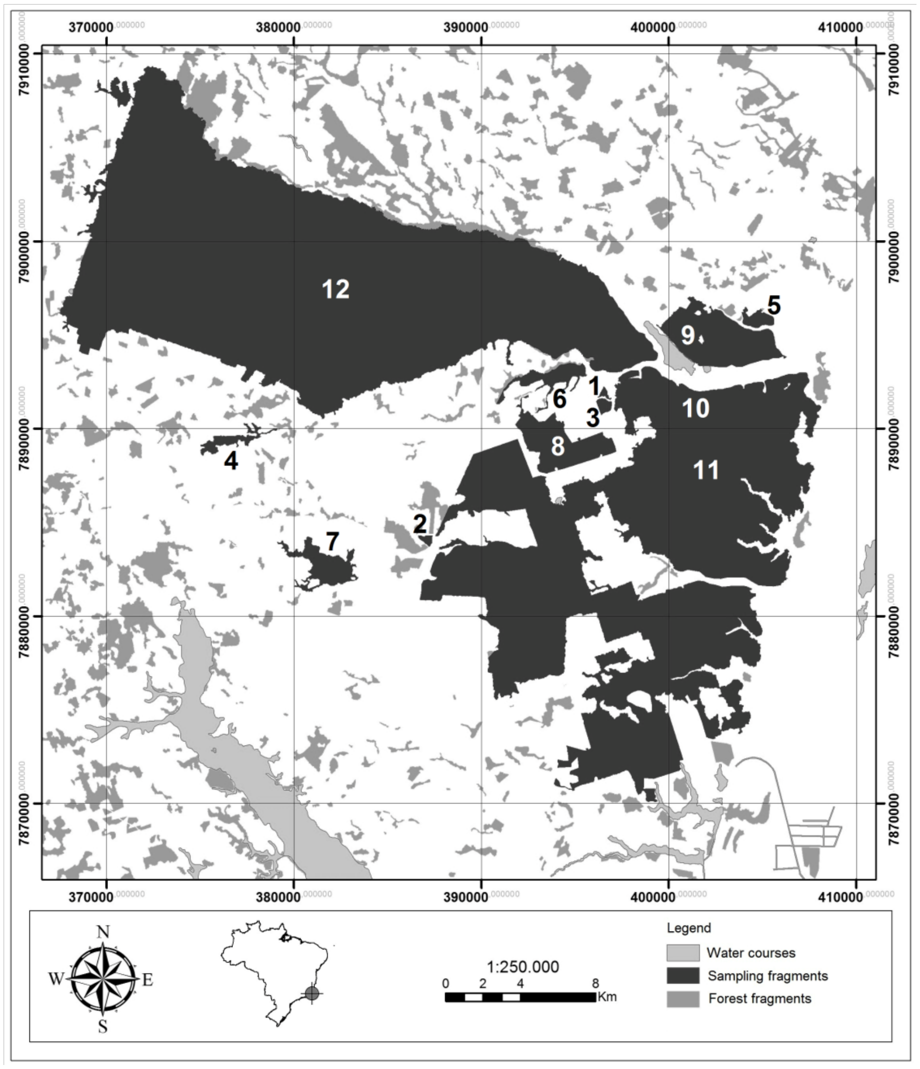

Figure S1- Study area and forest fragments sampled in Southeastern Brazil. To check the respective names and information about fragments see the table S1.

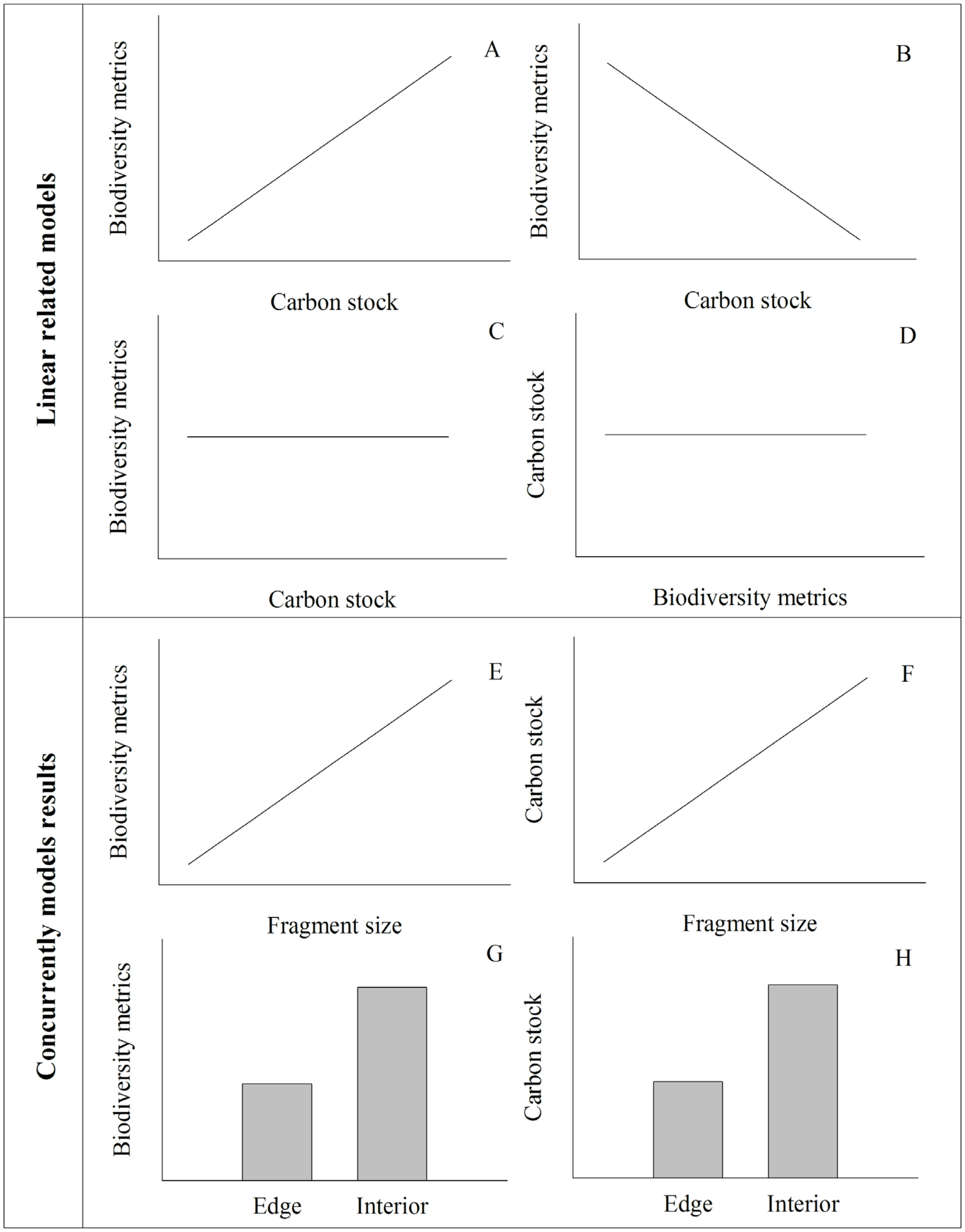

Figure S2 - Conceptual models for interpreting the relationship between biodiversity and carbon stock. (A) the metrics of biodiversity and the total carbon stock are synergistic, showing co-benefits relationships; (B) biodiversity and carbon stock are as different attributes in a fragmented landscape; (C) no differences in the metrics of biodiversity; (D) no differences in carbon stock; (E-F) concurrently results of biodiversity metrics and carbon stock in the same units of landscape.

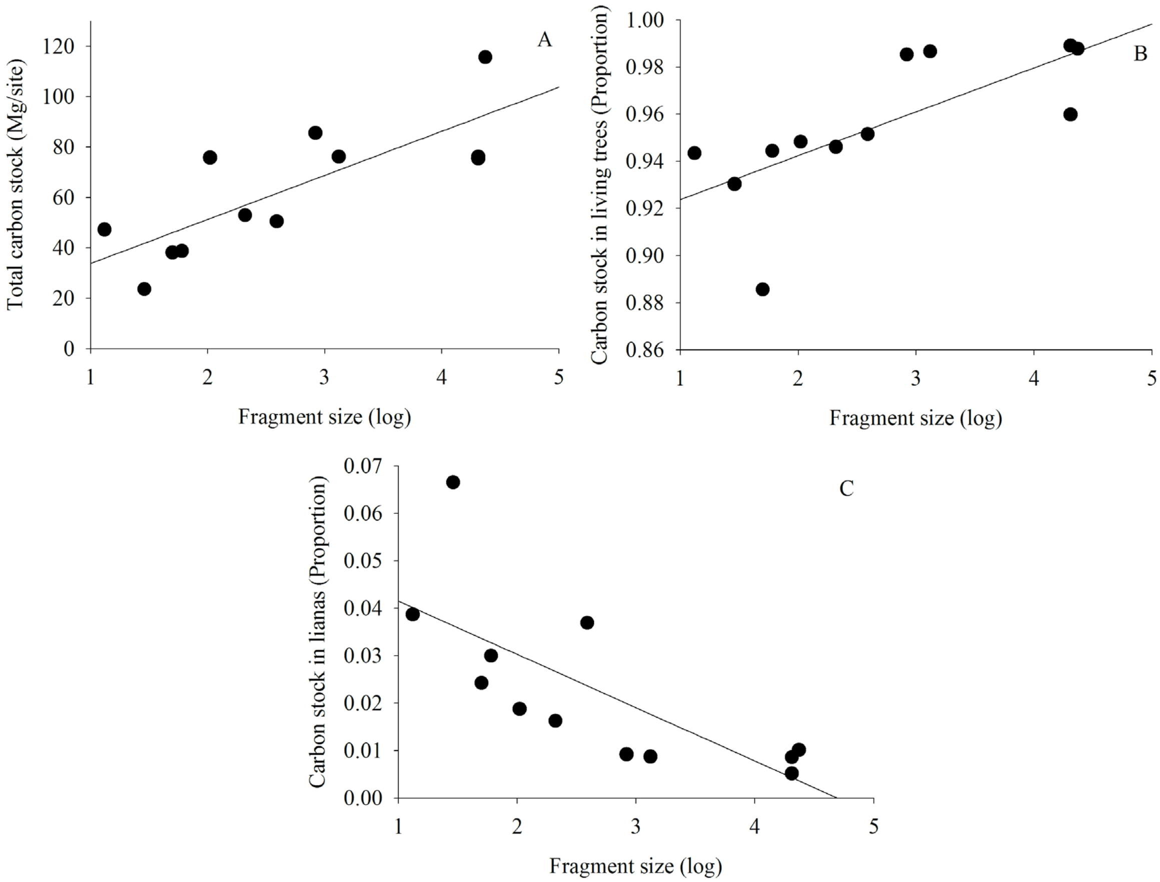

Figure S3 – Best model graphs generated for the results of fragment size effect on carbon stock. (A) Partial residuals graphical representation showing the effects of fragment size on total carbon stock; (B) Partial residual graphical representation of fragment size effect on the proportional contribution of live trees to total carbon stock; (C) Partial residual graphical representation of fragment size effect on the proportional contribution of lianas to total carbon stock. Black circles represent values obtained after the summation of raw residuals to the expected values for each variable, being assumed average values for other covariates.

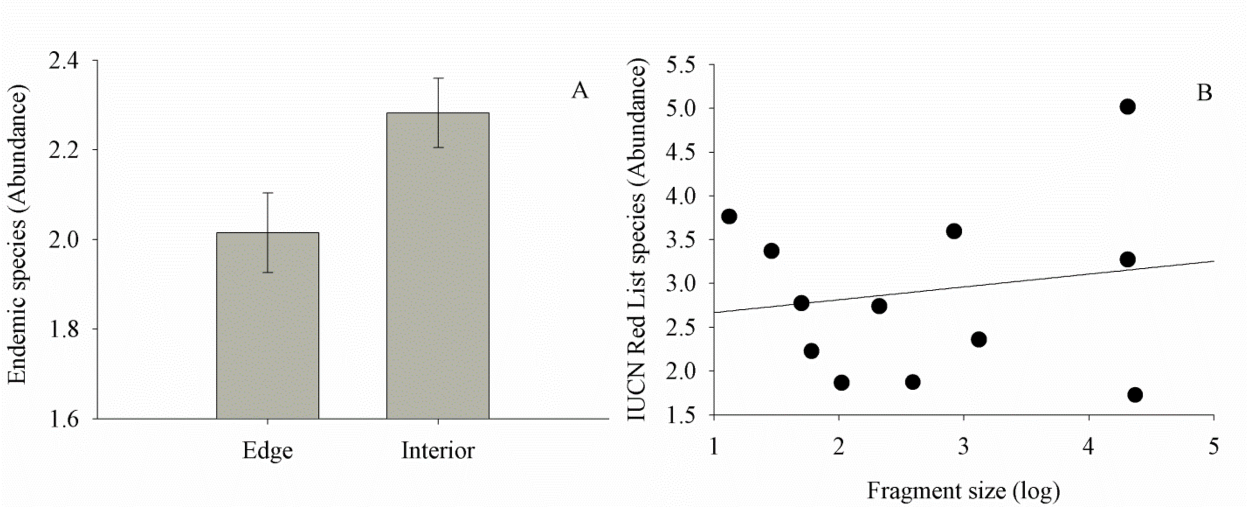

Figure S4 – Best model graphs for the results of fragmentation effects on species richness, and on endemic and endangered species (IUCN Red List). (A) Habitat effect on endemic species abundance in control fragments; (B) Partial residuals graphical representation showing the effect of fragment size on endangered species abundance. Circles represent values obtained after the summation of raw residuals to the expected values for each variable, being assumed average values for other covariates.

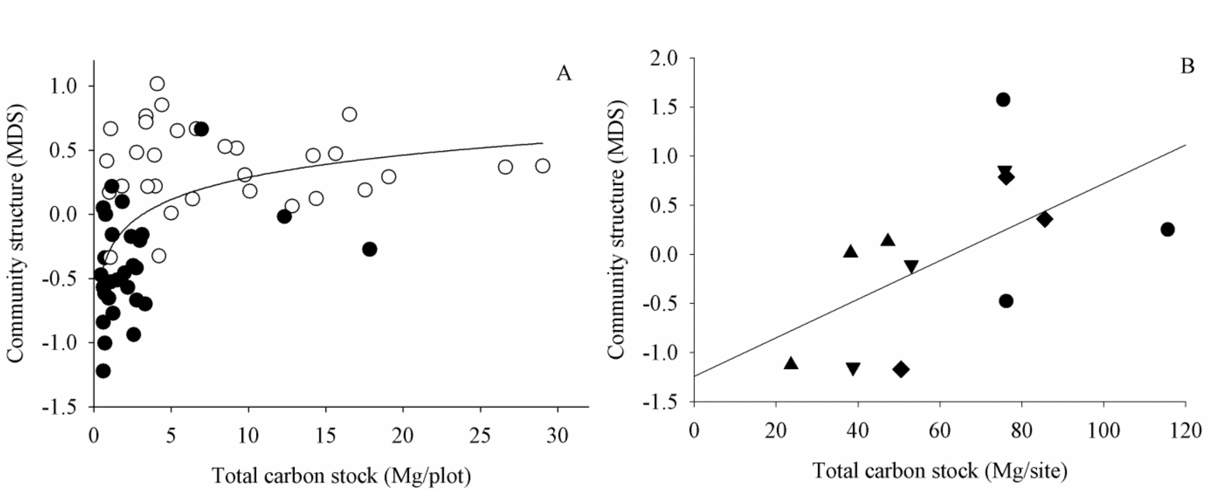

Figure S5 - Best model graphs for the results of the co-benefits relationship between biodiversity metrics and total carbon stock. (A) Partial residuals graphical representation showing the relationship between community structure and total carbon stock for control fragments; (B) Partial residuals graphical representation showing the relationship between community structure and total carbon stock for fragments interior.

Table S1 - Identification and size of fragments sampled in the study area in Southeastern Brazil.

Regional identification	Size class	Size (ha)
13. Fazenda Cúpido	Small	13.18
14. Reserva Natural Vale	Small	28.84
15. RPPN Recando das Antas	Small	50.12
16. Fazenda do Neb	Medium	60.26
17. Fazenda do Marim	Medium	104.71
18. Fazenda Caliman	Medium	208.93
19. Fazenda Rochedo	Large	389.05
20. RPPN Recando das Antas	Large	831.76
21. REBIO de Sooretama	Large	1318.26
22. REBIO de Sooretama	Control	20417.38
23. Reserva Natural Vale	Control	20417.38
24. REBIO de Sooretama	Control	23442.29

Table S2 - List of species and attributes that was used to construct the models. E=Edge habitat abundance; I=Interior habitat abundance; Pi=Pioneer species; Zoo=Zoochoric dispersion; IUCN= IUCN Red List of Threatened species; End= Endemic species of Atlantic Forest.

Species	E	I	Pi	Zoo	IUCN	End	Wood density (g/cm^3)	Carbon (Mg)
Abarema cochliacarpos (B.A.Gomes) Barneby & J.W.Grimes	-	1	-	-	X	-	0.585	1.315
Acacia glomerosa Benth.	17	1	-	-	-	-	0.629	5.972
Acosmium lentiscifolium Spreng.	8	6	-	-	-	X	0.763	5.039
Actinostemon concolor (Spreng.) Müll. Arg.	-	1	-	-	-	-	0.907	0.013
Actinostemon estrellensis (Mull. Arg.) var. latifolius Pax	27	52	-	-	-	-	0.907	2.777
Aegiphila verticillata Vell.	-	1	X	X	-	-	0.657	0.028
Albizia pedicellaris (DC.) Barneby & J.W.Grimes	1	-	-	-	-	-	0.497	2.798
Albizia polycephala (Benth.) Killip	18	2	-	-	-	-	0.542	3.746
Alchornea sidifolia Klotzch.	-	2	-	X	-	X	0.378	0.154
Allophylus petiolulatus Radlk.	45	8	-	X	-	X	0.431	1.557
Alseis involuta K.Schum.	10	-	-	-	-	-	0.850	0.247
Amaioua intermedia (A.Rich.) Steyerm.	-	1	-	X	-	-	0.625	0.007
Ampelocera glabra Kuhlm.	1	4	-	X	-	-	0.674	2.676
Amphirrhox longifolia (A.St.-Hil.) Spreng	1	6	-	X	-	-	0.710	0.030
Anaxagorea silvatica R.E.Fr.	1	13	-	-	-	X	0.580	0.515
Andira fraxinifolia Benth.	1	-	-	X	-	-	0.722	0.067
Andira legalis (Vell.) Toledo	-	2	-	X	-	X	0.722	0.163
Andira ormosioides Benth.	-	1	-	X	-	X	0.722	0.010
Angostura bracteata (Nees. A. Mart.) Kallunki	1	-	-	-	-	X	0.642	0.006
Aniba canellila Mez	-	1	-	X	-	-	0.952	0.006

Aniba firmula (Nees & C. Mart.) Mez	1	1	-	X	-	-	0.669	0.011
Annona acutiflora Mart.	1	1	-	X	-	X	0.413	0.006
Annona cacans Warm.	-	2	X	X	-	X	0.413	0.024
Annona dolabripetala Raddi	3	3	-	X	-	X	0.413	0.706
Annona sp.	-	1	-	X	-	-	0.413	0.002
Aparisthmium cordatum (Juss.) Baill.	-	1	-	-	-	-	0.390	0.018
Apuleia leiocarpa (Vogel) J.F. Macbr.	3	10	-	-	-	-	0.788	1.464
Aspidosperma cylindrocarpon Müll. Arg.	3	1	-	-	-	-	0.637	1.434
Aspidosperma desmanthum Benth. ex Müll. Arg.	-	3	-	-	-	-	0.610	1.693
Aspidosperma discolor A.DC.	1	6	-	-	-	-	0.758	0.251
Aspidosperma illustre (Vell.) Kuhlm. & Piraja	2	3	-	-	-	X	0.739	0.718
Aspidosperma parvifolium A. DC.	1	3	-	-	-	-	0.737	0.547
Astrocaryum aculeatissimum (Schott) Burret	25	14	-	X	-	X	0.508	2.278
Astronium concinnum (Engl.) Schott	110	11	-	-	-	-	0.818	32.524
Astronium graveolens Jacq.	36	7	-	-	-	-	0.818	8.092
Bactris ferruginea Burret	2	-	-	X	-	X	0.426	0.018
Barnebydendron riedelii (Tul.) J.H. Kirkbride	1	4	-	-	-	-	0.680	5.764
Bauhinia forficata Link subsp. forficata	7	3	X	-	-	X	0.600	0.441
Bauhinia longifolia (Bong.) Steud.	1	-	X	-	-	-	0.600	0.027
Beilschmiedia linharensis Sachiko Nishida & H.van der Werff	4	4	-	X	-	X	0.563	0.378
Bixa arborea Huber	1	2	X	X	-	-	0.370	0.181
Blepharocalyx eggersii (Kiaersk.) Landrum	1	-	-	X	-	-	0.726	0.548
Brasiliocroton mamoninha P.E.Berry & Cordeiro	89	15	X	-	-	-	0.408	2.702
Brosimum glaucum Taub.	23	15	-	X	-	X	0.560	7.173
Brosimum guianense (Aubl.) Huber	-	3	-	X	-	-	0.843	0.428
Brosimum lactescens (S. Moore) C.C. Berg	-	1	-	X	-	-	0.656	0.315
Byrsonima cacaophila W.R. Anderson	-	2	X	X	-	X	0.646	0.088
Byrsonima stipulacea (Juss.) Nied.	1	3	X	X	-	-	0.709	0.481
Calycophyllum papillosum J.H. Kirkbr.	-	1	-	-	-	X	0.708	2.241
Calyptranthes lucida var. *polyantha* (Berg) C.D.Legrand	2	21	-	X	-	-	0.860	0.976
Campomanesia espiritosantensis Landrum	-	5	-	X	X	X	0.730	0.568
Campomanesia guazumifolia (Cambess.) O.Berg	3	7	-	X	-	-	0.730	0.080
Campomanesia lineatifolia Ruiz et Pav.	4	1	-	X	-	-	0.730	0.190
Cariniana estrellensis (Raddi.) Kuntze	-	1	-	-	-	-	0.565	0.016
Cariniana legalis (Mart.) Kuntze	1	11	-	-	X	-	0.483	34.729
Carpotroche brasiliensis (Raddi.) A. Gray	11	21	-	X	-	-	0.450	0.565
Caryodendron grandifolium Pax	-	4	-	X	-	X	0.650	0.533
Caryodendron janeirense Müll.Arg	1	-	-	X	-	X	0.650	0.006
Casearia arborea (L.C.Richard) Urban	1	-	-	X	-	-	0.595	0.010
Casearia commersoniana Cambess.	3	4	-	X	-	-	0.664	0.203
Casearia javitensis H.B. & K.	2	-	-	X	-	-	0.753	0.015

Casearia oblongifolia Cambess.	5	2	-	X	-	X	0.664	0.122
Casearia sp. new species.1	6	1	-	X	-	X	0.664	0.133
Casearia sp. new species.2	10	5	-	X	-	X	0.664	0.428
Casearia sp.1	-	1	-	X	-	-	0.664	0.099
Casearia sp.2	1	-	-	X	-	-	0.664	0.005
Casearia sylvestris Sw.	1	-	X	X	-	-	0.680	0.012
Casearia ulmifolia Vahl. ex Vent.	3	4	-	X	-	-	0.664	0.490
Cecropia glaziovi Snethl.	2	-	X	X	-	X	0.330	0.041
Cecropia hololeuca Miq.	1	-	X	X	-	-	0.330	0.128
Cedrela odorata Linn.	3	-	X	-	X	-	0.427	0.033
Ceiba pubiflora (A. St.-Hil.) K. Schum.	1	1	-	-	-	-	0.365	2.327
Centrolobium sclerophyllum Lima	1	-	-	-	-	-	0.655	1.817
Chamaecrista aspleniifolia (H.S.Irwin & B). H.S. Irwin & Barneby	1	3	-	-	-	-	0.903	0.861
Chamaecrista bahiae (Irwin) Irwin & Barneby	1	-	-	-	-	-	0.903	0.025
Chamaecrista ensiformis (Vell.) Irwin & Barneby	-	9	-	-	-	-	0.924	0.207
Chamaecrista sp.	1	-	X	-	-	-	0.903	0.009
Chomelia pubescens Cham. & Schltdl.	1	1	X	X	-	X	0.570	0.017
Chrysobalanaceae	3	-	-	X	-	-	0.799	0.876
Chrysophyllum gonocarpum (Mart. & Eichler ex Miq.) Engl.	7	10	-	X	-	-	0.775	2.316
Chrysophyllum januariense Eichler	3	4	-	X	X	X	0.775	0.097
Chrysophyllum lucentifolium Cronquist	11	5	-	X	-	-	0.787	1.694
Chrysophyllum sp.	-	1	-	X	-	-	0.775	0.004
Chrysophyllum splendens Spreng.	3	4	-	X	X	X	0.775	5.462
Clarisia ilicifolia (Spreng.) Lanj. & Rossb.	5	10	-	X	-	-	0.580	0.747
Clarisia racemosa Ruiz & Pav.	4	5	-	X	-	-	0.585	1.110
Cnidoscolus oligandrus (Mull. Arg.) Pax	6	-	X	X	-	-	0.552	0.476
Coccoloba tenuiflora Lindau	5	1	-	X	-	-	0.568	0.146
Coccoloba warmingii Meisn	2	1	-	X	-	-	0.568	0.021
Connarus detersus Planch.	2	1	-	X	-	X	0.520	0.015
Copaifera langsdorffii Desf.	-	1	-	X	-	-	0.600	0.869
Copaifera lucens Dwyer	7	21	-	X	-	X	0.615	19.663
Cordia acutifolia Fresen.	4	2	-	X	-	X	0.485	0.148
Cordia ecalyculata Vell.	10	2	-	X	-	X	0.485	0.415
Cordia magnoliaefolia Cham.	-	1	-	X	-	X	0.485	0.004
Cordia sp.1	2	-	X	X	-	-	0.485	0.005
Cordia sp.2	-	1	X	X	-	-	0.485	0.005
Cordia trichoclada DC.	-	1	X	X	-	X	0.485	0.023
Cordia trichotoma (Vell.) Arráb. ex Stend.	1	-	X	X	-	-	0.560	0.009
Couepia belemii Prance	1	-	-	X	-	X	0.789	0.017
Couepia schottii Fritsch	-	1	-	X	X	X	0.789	2.210
Couratari asterotricha Prance	20	11	-	-	X	X	0.510	8.607

Couratari macrosperma A.C. Smith	3	3	-	-	-	-	0.670	3.760
Coussapoa curranii Blake	-	3	-	X	X	-	0.461	4.773
Coussarea contracta (Walp.) Benth. & Hook. ex Mull. Arg.	-	4	-	X	-	-	0.610	0.030
Coutarea hexandra (Jacq.) K. Schum.	-	1	-	-	-	-	0.600	0.084
Crepidospermum atlanticum D.C. Daly	1	8	-	X	-	X	0.578	1.269
Cryptocarya citriformis (Vellozo) P.L.R. Moraes	1	-	-	X	-	X	0.598	0.016
Cryptocarya saligna Mez.	1	-	-	X	-	X	0.598	0.010
Cunuria sp.	2	2	-	-	-	-	0.552	1.156
Cupania cf. *scrobiculata* L.C. Rich.	14	9	-	X	-	-	0.628	0.918
Cupania emarginata Cambess.	1	1	-	X	-	X	0.622	0.013
Cupania oblongifolia Mart.	2	-	X	X	-	-	0.622	1.080
Cupania rugosa Radlk.	5	4	X	X	-	-	0.622	0.282
Cupania sp.	1	-	-	X	-	-	0.622	0.013
Dalbergia elegans A.M. Carvalho	1	-	-	-	-	X	0.800	0.010
Dalbergia nigra (Vell.) Allemao ex Benth.	-	4	-	-	X	X	0.749	0.130
Deguelia longeracemosa (Benth.) Az.- Tozzi	7	1	-	-	-	X	0.726	0.185
Dendropanax cuneatus (DC.) Decne. & Planch.	1	1	-	X	-	-	0.423	0.299
Dialium guianense (Aubl.) Sandwith	12	30	-	-	-	-	0.867	29.703
Dilodendron elegans (Radlk.) Gentry & Steyerm.	-	1	-	X	-	-	0.617	0.008
Dimorphandra sp. new species	-	6	-	-	-	X	0.742	7.147
Diospyros brasiliensis Mart. ex Miq.	2	1	-	X	-	X	0.573	0.018
Diplotropis incexis Rizzini & A.Mattos	-	2	-	-	-	X	0.750	0.111
Drypetes sp.	2	5	-	X	-	-	0.914	2.086
Duguetia chrysocarpa Maas	1	1	-	X	-	X	0.757	0.012
Dulacia sp.	-	1	-	X	-	-	0.569	0.007
Duroia valesca C. Persson & Delprete	2	1	-	X	-	X	0.772	0.969
Ecclinusa ramiflora Mart.	13	28	-	X	-	-	0.637	2.060
Emmotum aff. *nitens* (Benth.) Miers.	-	5	-	X	-	-	0.727	1.854
Ephedranthus sp. new species.1	1	-	-	X	-	X	0.585	0.011
Ephedranthus sp. new species.2	-	1	-	X	-	X	0.585	0.087
Eriotheca candolleana (K. Schum.) A. Robyns	3	6	-	-	-	-	0.460	0.386
Eriotheca macrophylla (K. Schum.) A. Robyns	19	27	-	-	-	X	0.460	14.183
Erythroxylum columbinum Mart.	-	1	-	X	-	X	0.710	0.005
Erythroxylum pulchrum A. St.Hil.	-	1	-	X	-	-	0.710	0.005
Eschweilera ovata (Cambess.) Miers	5	19	-	X	-	-	0.900	8.924
Esenbeckia grandiflora Mart. subsp. *grandiflora*	-	3	-	-	-	-	0.642	0.012
Eugenia bahiensis DC	-	2	-	X	-	X	0.726	0.018
Eugenia batingabranca Sobral	-	8	-	X	-	X	0.726	0.094
Eugenia beaurepaireana (Kiaersk.) C.D.Legrand	4	5	-	X	X	X	0.726	0.542
Eugenia brasiliensis Lam.	-	3	-	X	-	X	0.726	0.063
Eugenia cf. *badia* O.Berg	2	8	-	X	-	X	0.726	0.130
Eugenia cf. *moonioides* Berg	1	3	-	X	-	X	0.726	1.558

Eugenia cf. *tinguyensis* Cambess.	16	35	-	X	-	X	0.726	0.426
Eugenia excelsa O.Berg	13	14	-	X	-	-	0.726	0.497
Eugenia fluminensis Berg	-	16	-	X	-	X	0.726	0.224
Eugenia gemmiflora O. Berg	-	3	-	X	-	-	0.726	5.241
Eugenia handroi (Mattos) Mattos	-	2	-	X	-	X	0.726	0.205
Eugenia itapemirimensis Cambess.	7	20	-	X	-	X	0.726	0.670
Eugenia ligustrina Berg	1	1	-	X	-	-	0.726	0.029
Eugenia macrosperma DC.	3	7	-	X	-	X	0.726	0.264
Eugenia platyphylla O.Berg	32	21	-	X	-	X	0.726	2.146
Eugenia platysema Berg	3	5	-	X	-	X	0.726	0.366
Eugenia plicatocostata O.Berg	1	-	-	X	-	X	0.726	0.004
Eugenia prasina O.Berg	7	18	-	X	X	X	0.726	0.158
Eugenia sp.1	1	3	-	X	-	-	0.726	0.017
Eugenia sp.2	-	2	-	X	-	-	0.726	0.007
Eugenia sp.3	-	1	-	X	-	-	0.726	0.098
Eugenia sp.4	1	-	-	X	-	-	0.726	0.022
Eugenia sp.5	2	2	-	X	-	-	0.726	0.025
Eugenia sp.6	1	-	-	X	-	-	0.726	0.010
Eugenia sp.7	-	6	-	X	-	-	0.726	0.213
Eugenia sp.8	1	1	-	X	-	-	0.726	0.395
Eugenia subterminalis DC.	3	21	-	X	-	-	0.726	0.272
Euphorbiaceae (new species)	2	3	-	-	-	X	0.552	0.028
Exellodendron gracile (Kuhlmann) Prance	1	1	-	X	-	X	0.707	0.231
Exostyles venusta Schott ex Spreng.	4	1	-	X	-	X	0.680	0.292
Ficus cyclophylla (Miq.) Miq.	-	1	-	X	X	X	0.394	11.521
Ficus gomelleira Kunth & C.D. Bouché	1	2	-	X	-	-	0.394	20.185
Ficus mariae C.C. Berg, Emygdio & Carauta	1	5	-	X	-	X	0.394	4.562
Ficus nymphaeifolia Mill.	-	1	-	X	-	-	0.415	1.685
Galipea cf. *laxiflora* Engl.	5	14	-	-	-	X	0.642	0.155
Geissospermum laeve (Vell.) Baill.	12	17	-	X	-	-	0.782	4.534
Glycydendron espiritosantense Kuhlm.	1	4	-	X	-	X	0.681	1.461
Gomidesia martiana O. Berg.	1	-	-	X	-	X	0.801	0.023
Goniorrhachis marginata Taub.	6	9	-	-	-	-	0.680	7.944
Guapira noxia (Netto) Lundell	6	4	-	X	-	-	0.492	0.186
Guapira opposita (Vell.) Reitz	20	16	-	X	-	-	0.492	2.635
Guapira venosa (Choisy) Lundell	4	4	-	X	-	-	0.492	1.955
Guarea aff. *juglandiformis* Pennington	1	1	-	X	X	-	0.606	0.013
Guarea penningtoniana Pinheiro	-	3	-	X	-	X	0.606	0.089
Guatteria macropus Mart.	1	-	-	X	-	X	0.540	0.003
Guatteria sellowiana Schltdl.	-	1	-	X	-	-	0.540	0.480
Guazuma crinita Mart.	8	1	X	-	-	-	0.440	0.407
Guettarda angelica Mart. ex Müell. Arg.	6	-	-	X	-	-	0.707	0.166

Handroanthus arianeae (A.H. Gentry) S. O. Grose	8	2	-	-	-	X	0.774	1.161
Handroanthus riodocensis (A.H. Gentry) S. O. Grose	5	2	-	-	-	X	0.774	3.402
Handroanthus serratifolius (Vahl) S. O. Grose	1	-	-	-	-	-	0.924	0.064
Heisteria cf. *ovata* Benth.	1	4	-	X	-	-	0.540	0.515
Heisteria sp.	1	-	-	X	-	-	0.704	0.004
Helicostylis tomentosa (Poep. et Endl.) Rusby	2	6	-	X	-	-	0.627	0.819
Himatanthus bracteatus (A. DC.) Woodson	2	1	X	-	-	-	0.530	0.087
Hirtella hebeclada Moric. ex A. P. DC.	1	1	-	X	-	-	0.793	0.058
Hirtella sprucei Benth.ex Hook.f.	-	3	-	X	-	-	0.793	0.039
Hornschuchia citriodora D. M. Johnson	2	-	-	X	-	X	0.585	0.008
Humiriastrum spiritu-sancti Cuatrec	-	2	-	X	-	X	0.668	1.310
Hydrogaster trinervis Kuhlm.	9	18	-	-	-	X	0.443	18.046
Hymenaea aurea Y.T.Lee & Langenheim	5	5	-	X	-	X	0.790	5.129
Hymenaea courbaril L.	3	1	-	X	-	-	0.787	0.383
Indet. 1	1	-	-	-	-	-	-	-
Indet. 2	-	1	-	-	-	-	-	-
Indet. 3	1	4	-	-	-	-	-	-
Indet. 4	1	-	-	-	-	-	-	-
Indet. 5	-	1	-	-	-	-	-	-
Inga aff. *cylindrica* (Vell.) Mart.	-	2	-	X	-	-	0.576	0.105
Inga cabelo T.D. Penn.	4	1	-	X	X	X	0.592	0.037
Inga capitata Desv.	-	2	-	X	-	-	0.576	0.015
Inga exfoliata T.D. Penn. & F.C.P. García	-	2	-	X	X	X	0.576	0.008
Inga flagelliformis (Vell.) Mart.	9	10	-	X	-	-	0.576	0.462
Inga hispida Schott. ex Benth.	1	3	-	X	X	X	0.576	0.024
Inga striata Benth.	1	-	-	X	-	-	0.576	0.004
Inga thibaudiana subsp. *thibaudiana* T.D. Penn.	5	-	-	X	-	-	0.637	0.353
Ixora warmingii Mull. Arg.	4	2	X	X	-	-	0.382	0.070
Jacaranda puberula Cham.	4	7	-	-	-	X	0.265	0.134
Jacaratia heptaphylla (Vell.) A. DC.	4	4	-	X	-	-	0.390	0.244
Joannesia princeps Vell.	37	15	X	X	X	-	0.628	11.642
Kielmeyera occhioniana Saddi	2	-	-	-	-	X	0.598	0.255
Lauraceae (new species)	-	1	-	X	-	X	0.818	5.240
Lecythis lanceolata Poir.	9	5	-	X	-	X	0.830	27.863
Lecythis lurida (Miers) S.A.Mori	19	13	-	X	-	-	0.852	39.888
Lecythis pisonis Cambess.	-	4	-	X	-	-	0.818	65.153
Lecythis sp.	1	3	-	X	-	-	0.823	25.234
Licania belemii Prance	-	1	-	X	-	X	0.816	0.016
Licania heteromorpha Benth. var. *heteromorpha*	1	-	-	X	-	-	0.880	0.039
Licania kunthiana Hook.f.	3	6	-	X	-	-	0.823	1.586
Licania salzmannii (Hook.) Fritsch.	-	1	-	X	-	X	0.823	4.090
Licania sp.	-	1	-	X	-	-	0.815	0.251

Licaria bahiana Kutz	1	3	-	X	-	X	0.726	0.106
Lonchocarpus cultratus (Vell.) A.M.G. Azevedo & H.C. Lima	10	4	-	-	-	-	0.507	0.809
Luehea mediterranea (Vell.) Angely	13	3	X	-	-	-	0.616	1.674
Mabea cf. *fistulifera* Mart.	1	3	X	X	-	-	0.780	0.038
Machaerium fulvovenosum H.C.Lima	46	9	-	-	-	X	0.780	2.587
Machaerium ovalifolium Glaziou ex Rudd	2	1	-	-	-	-	0.604	0.056
Macrothumia kuhlmannii (Sleumer) M.H.Alford	9	7	-	X	-	X	0.884	1.332
Manilkara bella Monach.	4	4	-	X	X	X	0.884	3.701
Manilkara salzmannii (A.DC.) H.J.Lam	2	4	-	X	-	X	0.484	2.137
Margaritaria nobilis Linn.f.	4	2	-	-	-	-	0.936	0.071
Marlierea estrellensis Berg	-	4	-	X	-	X	0.936	0.890
Marlierea excoriata Mart.	1	1	-	X	-	X	0.936	0.014
Marlierea grandifolia O. Berg	2	1	-	X	-	X	0.936	0.136
Marlierea obversa Legrand.	3	1	-	X	-	X	0.936	0.052
Marlierea sucrei G.M. Barroso et Peixoto	3	7	-	X	-	X	0.801	0.112
Marlierea clausseniana (O.Berg) Kiaersk.	2	5	-	X	-	-	0.750	0.380
Matayba discolor Radlk.	1	-	-	X	-	X	0.820	0.038
Matayba guianensis Aubl.	3	2	-	X	-	-	0.745	0.294
Maytenus cestrifolia Reiss.	3	3	-	X	-	X	0.745	0.169
Maytenus multiflora Reiss.	4	3	-	X	-	X	0.745	0.785
Maytenus patens Reiss.	1	-	-	X	-	X	0.637	0.050
Melanopsidium nigrum Colla	1	1	-	X	-	X	0.900	0.011
Melanoxylon brauna Schott.	16	8	-	-	-	-	0.689	3.319
Melicoccus espiritosantensis Acev.-Rodr.	4	1	-	X	-	X	0.642	3.988
Metrodorea maracasana Kaastra	2	12	-	-	-	-	0.620	0.538
Miconia cf. *cinnamomifolia* (DC.) Naudin	1	-	X	X	-	X	0.620	0.623
Miconia cf. *rimalis* Naud.	-	1	-	X	-	-	0.750	0.004
Miconia lepidota Schrad. et Mart. ex DC.	1	-	X	X	-	-	0.710	0.067
Miconia prasina (Sw.) DC.	1	-	-	X	-	-	0.650	0.019
Micropholis aff. *gnaphaloclados* Pierre	3	1	-	X	-	-	0.650	0.204
Micropholis crassipedicellata (Mart. & Eichler.) Pierre	1	1	-	X	-	X	0.650	7.546
Micropholis cuneata Pierre ex Glaziou	2	1	-	X	-	X	0.650	0.255
Micropholis gardneriana (A.DC.) Pierre	1	4	-	X	-	-	0.680	0.073
Moldenhawera papillanthera L.P.Queiroz, G.P.Lewis & R.Allkin	5	8	-	-	-	X	0.665	2.215
Mollinedia marquetiana A.L. Peixoto	2	2	-	X	X	X	0.665	0.038
Mollinedia ovata Ruiz & Pav.	-	1	-	X	-	-	0.637	0.005
Molopanthera paniculata Turcz.	-	1	-	X	-	-	0.691	0.261
Monilicarpa brasiliana (Banks ex DC.) Cornejo & Iltis	2	-	-	X	-	-	0.836	0.008
Mouriri arborea Gardner	-	4	-	X	-	X	0.836	0.141
Mouriri glazioviana Cogn.	-	3	-	X	-	-	0.801	1.360
Myrcia eumecephylla (O.Berg) Nied.	-	1	-	X	-	X	0.810	0.005
Myrcia fallax DC.	3	2	-	X	-	-	0.801	0.138

Myrcia follii Barroso et Peixoto	-	1	-	X	-	X	0.801	0.004
Myrcia isaiana G.M. Barroso et Peixoto	-	1	-	X	X	X	0.801	0.006
Myrcia lineata (Berg) G.M. Barroso	4	4	-	X	X	X	0.801	0.379
Myrcia multiflora (L) DC.	-	1	-	X	-	-	0.801	0.015
Myrcia riodocensis G.M. Barroso et Peixoto	2	2	-	X	-	X	0.801	0.101
Myrcia rostrata DC.	-	3	-	X	-	-	0.700	0.195
Myrciaria aureana Mattos	-	1	-	X	-	X	0.700	0.007
Myrciaria ferruginea O. Berg	1	-	-	X	-	X	0.756	0.007
Myrciaria floribunda (West. ex Willd.) O. Berg	6	20	-	X	-	-	0.700	0.128
Myrciaria tenella (DC.) O.Berg	1	-	-	X	-	-	0.775	0.149
Myrocarpus frondosus Allemao	3	2	-	-	-	X	0.743	0.492
Myrtaceae	2	-	-	X	-	-	0.651	0.021
Naucleopsis oblongifolia (Kuhlm.) Carauta	4	11	-	X	X	-	0.620	0.354
Neea floribunda Poepp. & Endl.	2	1	-	X	-	-	0.691	0.063
Neocalyptrocalyx nectarea (Vell.) Hutch.	1	3	-	X	-	-	0.743	0.017
Neomitranthes langsdorffii (O.Berg) J.R. Mattos	6	6	-	X	X	X	0.642	0.209
Neoraputia alba (Nees & Mart.) Emmerich	28	23	-	-	-	X	0.501	2.474
Ocotea argentea Mez	1	-	-	X	-	X	0.501	0.006
Ocotea conferta Coe Teixeira	-	1	-	X	-	X	0.501	0.421
Ocotea confertiflora (Meisn.) Mez	7	10	-	X	-	X	0.501	0.560
Ocotea elegans Mez	3	16	-	X	-	X	0.501	2.350
Ocotea lancifolia (Schott) Mez	3	2	-	X	-	-	0.462	0.301
Ocotea leucoxylon (Sw.) de Lanessan s.l.	1	-	-	X	-	-	0.501	0.005
Ocotea nitida (Meissn.) J.G.Rohwer	1	1	-	X	-	-	0.501	0.016
Ocotea nutans (Nees) Mez	-	1	-	X	-	-	0.770	0.006
Ocotea odorifera (Vell.) Rohwer	1	3	-	X	-	-	0.501	2.200
Ocotea pluridomatiata A. Quinet	1	-	-	X	-	X	0.501	0.006
Ocotea sp.	-	6	-	X	-	-	0.621	0.049
Ormosia arborea (Vell.) Harnu	1	1	-	X	-	-	0.621	0.053
Ormosia nitida Vogel	-	1	-	X	-	X	0.774	0.008
Ouratea cuspidata (A.St.-Hil.) Engl.	-	1	-	X	-	-	0.774	0.008
Ouratea sp.	-	1	-	X	-	-	0.748	0.011
Oxandra martiana (Schltdl.) R.E.Fr.	-	1	-	X	-	X	0.748	0.316
Oxandra nitida R.E. Fries	1	-	-	X	-	X	0.748	0.072
Oxandra reticulata Maas	-	3	-	X	-	-	0.448	0.210
Pachira stenopetala Casar.	1	3	-	X	-	-	0.780	0.392
Parapiptadenia pterosperma (Benth.) Brenan	10	5	-	-	-	X	0.704	1.258
Parinari excelsa Sabine	-	1	-	X	-	-	0.707	0.466
Parinari parvifolia Sandw.	4	4	-	X	-	-	0.590	3.197
Pausandra morisiana (Casar.) Radlk.	2	17	-	X	-	X	0.443	0.212
Pavonia crassipedicellata Krapov.	2	1	-	-	-	X	0.598	0.011
Paypayrola blanchetiana Tul.	1	2	-	X	-	X	0.792	0.023

Peltogyne angustiflora Ducke	-	5	-	-	-	X	0.647	6.090
Pera leandri Baill.	1	3	-	X	-	-	0.647	0.092
Pera sp.	-	2	-	X	-	-	0.395	0.021
Picramnia ramiflora Planch.	-	1	-	X	-	-	0.395	-
Picramnia sellowii Planch.	-	1	-	X	-	-	0.780	-
Piptadenia paniculata Benth.	5	1	X	-	-	X	0.300	0.322
Pisonia aff. *ambigua* Heimerl	1	5	-	X	-	-	0.792	0.467
Platymiscium floribundum Vogel	1	-	-	-	-	-	0.700	0.009
Plinia grandifolia (Mattos) Sobral	1	1	-	X	-	X	0.700	0.076
Plinia involucrata (Berg) McVaugh.	3	34	-	X	-	-	0.700	0.777
Plinia renatiana G.M.Barroso & Peixoto	-	19	-	X	-	X	0.700	0.597
Plinia stictophylla Barroso & Peixoto	1	1	-	X	-	X	0.620	0.131
Poecilanthe falcata (Vell.) Heringer	2	-	-	-	-	X	0.833	0.010
Pogonophora schomburgkiana Miers ex Benth.	1	1	-	-	-	-	0.426	0.111
Polyandrococos caudescens (Mart.) Barb. Rodr.	30	9	-	X	-	X	0.584	1.159
Polygala pulcherrima Kuhlm.	2	2	-	X	-	X	-	-
Posoqueria latifolia (Rudge) Roem & Schult.	-	4	-	X	-	-	0.380	0.018
Pourouma guianensis Aubl. subsp. guianensis	1	-	-	X	-	-	0.390	0.013
Pourouma mollis Trécul ssp. mollis	-	1	-	X	-	-	0.783	0.026
Pouteria aff. *bapeba* T.D.Pennington	3	1	-	X	X	X	0.964	0.352
Pouteria aff. *filipes* Eyma	9	8	-	X	-	-	0.783	0.403
Pouteria bangii (Rusby) T.D.Pennington	7	4	-	X	-	-	0.783	0.356
Pouteria bullata (S.Moore) Baehni	1	3	-	X	X	X	0.783	0.050
Pouteria butyrocarpa (Kuhlm.) T.D. Penn.	1	1	-	X	X	X	0.783	1.991
Pouteria coelomatica Rizzini	5	1	-	X	X	X	0.580	0.881
Pouteria durlandii (Standl.) Baehni	-	2	-	X	-	-	0.874	0.023
Pouteria hispida Eyma	14	12	-	X	-	-	0.737	7.703
Pouteria macrophylla (Lam) Eyma	2	1	-	X	-	-	0.783	0.361
Pouteria macrostachiosa Pennington	2	7	-	X	-	X	0.760	2.731
Pouteria pachycalyx T.D. Penn.	-	1	-	X	X	X	0.783	0.079
Pouteria psammophila (Mart.) Radlk.	1	2	-	X	X	X	0.876	0.731
Pouteria reticulata (Engl.) Eyma	6	6	-	X	-	-	0.783	0.196
Pouteria sp.1	4	-	-	X	-	-	0.783	0.044
Pouteria sp.2	2	-	-	X	-	-	0.783	0.017
Pouteria sp.3	-	1	-	X	-	-	0.920	0.067
Pouteria venosa subsp. *amazonica* T.D.Pennington	6	2	-	X	-	-	0.731	1.166
Pradosia lactescens (Vellozo) Radlk.	14	6	-	X	-	X	0.572	0.390
Protium brasiliense (Spreng.) Engl.	1	-	-	X	-	-	0.629	0.051
Protium heptaphyllum (Aubl.) Marchand.	15	9	-	X	-	-	0.572	5.769
Protium warmingianum Marchand	16	20	-	X	-	-	0.800	4.977
Pseudima frutescens (Aubl.) Radlk.	6	8	-	X	-	-	0.278	0.165
Pseudobombax longiflorum (Mart. & Zucc.) A. Robyns	4	4	-	-	-	-	0.664	1.734

Pseudopiptadenia contorta (DC.) G.P.Lewis & M.P.M.de Lima	23	3	-	-	-	-	0.664	2.284
Pseudopiptadenia psilostachya (DC.) G.P. Lewis & M.P. Lima	8	5	-	-	-	-	0.370	8.606
Pseudoxandra spiritus-sancti Maas	2	23	-	X	-	X	0.520	0.390
Psicotria sp.	-	1	-	X	-	-	0.684	0.005
Psidium cauliflorum Landrum & Sobral	-	1	-	X	-	X	0.684	0.049
Psidium longipetiolatum D.Legrand	1	-	-	X	-	X	0.684	0.006
Psidium oblongatum O.Berg	4	5	-	X	-	X	0.684	0.050
Psidium sartorianum (Berg) Nied.	-	2	-	X	-	-	0.427	0.061
Pterocarpus rohrii Vahl.	18	15	-	-	-	-	0.590	34.152
Pterygota brasiliensis Fr. All.	9	4	-	-	-	X	0.650	21.933
Qualea jundiahy Warm.	-	2	-	-	-	-	0.633	0.219
Qualea megalocarpa Stafleu	3	3	-	-	-	X	0.499	4.437
Quararibea penduliflora (A.St.Hil.) K. Schum.	15	25	-	X	-	X	0.690	0.365
Randia armata D.C.	7	4	-	X	-	-	0.482	0.091
Rauvolfia capixabae I. Koch & Kin.-Gouv.	-	2	-	X	-	X	0.642	0.065
Ravenia infelix Vell.	2	4	-	-	-	X	0.787	0.035
Rhamnidium glabrum Reissek	4	-	X	X	-	-	0.654	0.082
Rheedia gardneriana Triana & Planch.	1	5	-	X	-	-	0.652	0.060
Rinorea bahiensis (Moric.) Kuntze	36	123	-	X	-	X	0.652	28.979
Rinorea sp.	1	6	-	X	-	-	0.652	0.034
Rudgea sp.	-	1	-	X	-	-	0.689	0.007
Sapindaceae	1	-	-	-	-	-	0.421	0.014
Sapium glandulatum (Vell.) Pax.	2	3	-	X	-	-	0.453	0.149
Schefflera morototoni (Aubl.) Maguire, Steyermark & Frodin	4	2	X	X	-	-	0.723	0.326
Schoepfia brasiliensis A. DC.	4	1	-	X	-	-	0.723	0.213
Schoepfia obliquifolia Turcz.	3	10	-	X	-	-	0.780	0.993
Senefeldera multiflora Mart.	41	140	-	X	-	X	0.474	6.987
Simaba cedron Planchon	3	1	-	X	-	-	0.419	0.171
Simaba subcymosa A. St. Hil. & Tul.	1	1	-	X	-	X	0.378	0.448
Simaruba amara Aubl.	4	3	-	X	-	-	0.660	1.296
Simira glaziovii (K. Schum.) Steyermark	1	3	-	-	-	X	0.660	0.183
Simira grazielae A. L. Peixoto	3	2	-	-	-	X	0.660	0.399
Simira sampaioana (Standl.) Steyerm.	3	1	-	-	-	-	0.656	0.279
Siparuna reginae (Tul.) A. DC.	-	2	-	X	-	-	0.806	0.044
Sloanea aff. *granulosa* Ducke	-	3	-	X	-	-	0.750	26.813
Sloanea eichleri K. Schum.	4	3	-	X	-	-	0.806	1.520
Sloanea garckeana K. Schum.	3	2	-	X	-	-	0.280	0.108
Solanum sooretamum Carvalho	17	2	X	X	-	X	0.578	0.099
Sorocea guilleminiana Gaudich.	14	32	-	X	X	-	0.666	2.027
Sparattosperma leucanthum (Vell.) K. Schum.	1	1	X	-	-	-	0.395	0.011
Spondias macrocarpa Engl.	1	1	-	X	-	X	0.395	0.770

	2							
Spondias venulosa Mart. ex Engl.	2	-	-	X	-	-	0.660	0.053
Stephanopodium blanchetianum Baill.	-	1	-	X	-	X	0.419	0.009
Sterculia elata Ducke	1	1	-	X	-	-	0.510	11.236
Sterculia speciosa Ducke	16	20	-	X	-	-	0.340	24.404
Styrax glabratum Schott.	-	1	X	X	-	X	0.834	0.034
Swartzia acutifolia Vogel	1	3	-	X	-	-	0.834	7.953
Swartzia apetala Raddi	5	12	-	X	-	-	0.834	1.803
Swartzia linharensis Mansano	2	1	-	X	-	X	0.900	2.560
Swartzia myrtifolia var. *elegans* (Schott) R.S.Cowan	2	1	-	X	-	-	0.834	0.636
Swartzia simplex var. *continentalis* Urban	10	11	-	X	-	X	0.680	0.187
Sweetia fruticosa Spreng.	4	1	-	-	-	-	0.426	0.091
Syagrus botryophora (Mart.) Mart.	7	9	-	X	-	X	0.490	0.291
Symplocos pycnobotrya Mart. ex Miq.	1	-	-	X	-	X	0.774	0.006
Tabebuia cf. *elliptica* (DC.) Sandwith	1	-	-	-	-	-	0.774	0.045
Tabebuia obtusifolia (Cham.) Bureau	2	4	-	-	-	-	0.774	0.238
Tabebuia roseo-alba (Ridley) Sandwith	7	1	-	-	-	-	0.469	2.220
Tabernaemontana salzmanni A. DC.	2	1	X	X	-	-	0.560	0.019
Tachigali pilgeriana (Harms) Oliveira-Filho	7	2	-	-	-	X	0.775	0.648
Talisia intermedia Radlk.	6	2	-	X	-	-	0.457	3.267
Tapirira guianensis Aubl.	7	1	X	X	-	-	0.810	1.965
Terminalia argentea Mart.	3	1	-	-	-	-	0.730	0.328
Terminalia glabrescens Mart.	2	2	-	-	-	-	0.680	11.726
Terminalia kuhlmannii Alwan & Stace	14	16	-	-	X	X	0.540	43.308
Thyrsodium spruceanum Benth.	11	11	-	X	-	-	0.608	0.829
Toulicia patentinervis Radlk.	-	2	-	-	-	-	0.679	0.845
Tovomita brevistaminea Engl.	1	1	-	X	-	-	0.460	0.012
Trichilia aff. *surumuensis* C.DC.	1	8	-	X	X	-	0.635	0.670
Trichilia casaretti C.DC.	49	19	-	X	X	-	0.635	0.753
Trichilia elegans A. Juss. subsp. *elegans*	-	1	-	X	-	-	0.635	0.010
Trichilia lepidota subsp. *schumanniana* (Harms) T.D.Pennington	21	21	-	X	X	X	0.635	2.072
Trichilia pallens C. DC.	8	10	-	X	-	-	0.548	0.178
Trichilia quadrijuga Kunth. subsp. *quadrijuga*	3	13	-	X	-	-	0.635	1.299
Trichilia silvatica C. DC.	1	5	-	X	X	-	0.635	0.110
Trichilia sp.	16	6	-	X	-	-	0.635	0.376
Trigoniodendron spiritusanctense E.F. Guim. & Miguel	-	1	-	-	-	X	-	-
Unonopsis renati Maas & Westra	1	1	-	X	-	X	0.559	0.114
Vatairea heteroptera (Allem.) Ducke ex de Assis Iglesias	5	6	-	X	-	X	0.670	1.321
Vataireopsis araroba (Aguiar) Ducke	1	-	-	X	-	X	0.634	0.059
Virola gardneri (A.DC.) Warb.	12	27	-	X	-	X	0.450	26.985
Vitex aff. *megapotamica* (Spreng.) Moldenke	-	1	-	X	-	-	0.553	0.003

Vitex montevidensis Cham.	2	2	-	X	-	-	0.553	0.066
Vochysia angelica M.C. Vianna & Fontella	2	3	-	-	-	X	0.457	1.219
Xylopia ochrantha Mart.	1	-	-	X	-	X	0.570	0.004
Xylopia sericea A. St.-Hil.	-	1	-	X	-	-	0.570	0.131
Zanthoxylum aff. *retusum* (Albuq.) P.G. Waterman	1	-	-	X	-	X	0.601	0.013
Ziziphus glaziovii Warm.	3	6	-	X	-	X	0.838	3.973
Zollernia latifolia Benth.	3	1	-	X	-	-	1.050	0.118
Zollernia modesta A.M.de Carvalho & R.C.Barneby	4	6	-	X	-	X	1.005	5.111

Table S3 **-** Effects included in each of the analyses performed as well as type of error distribution and link function used. We also include information on random variables used for the different parts of the study: a) Models with control fragments; b) Models content the interior of fragments to verify the effect of fragment size; c) Global models with all fragment size and habitats; d) Models to verify co-benefits among biodiversity metrics and total carbon stock in all landscape scales, being control fragments, fragments interiors and all fragments and habitats. ns = non significant; * = p≤0.05; ** p ≤ 0.01;*** = p ≤ 0.05.

a)	**Habitat (edge)**	**Error distribution**	**Link function**
Total carbon stock (Mg/plot)	-6.184*	Gaussian	Identity
Tree carbon stock (Mg/plot)	-0.09*	Gaussian	Identity
Liana carbon stock (Mg/plot)	0.06*	Gaussian	Identity
Standing dead trees carbon stock (Mg/plot)	0.03*	Gaussian	Identity
Species richness	-0.3ns	Gaussian	Identity
Richness of threatened species (IUCN Red List)	-0.36ns	Gaussian	Identity
Abundance of threatened species (IUCN Red List)	-0.16ns	Poisson	Log
Richness of endemic species	-0.73ns	Gaussian	Identity
Abundance of endemic species	-0.27*	Quasipoisson	Log

MDS scores (axis 1)	-0.80*	Gaussian	Identity

b)	**Fragment size (log)**	**Error distribution**	**Link function**
Total carbon stock (Mg/plot)	17.50*	Gaussian	Identity
Tree carbon stock (proportion of total)	0.02**	Gaussian	Identity
Liana carbon stock (proportion of total)	-0.01**	Gaussian	Identity
Standing dead trees carbon stock (proportion of total)	-0.41ns	Gaussian	Identity
Species richness	-4.54*	Gaussian	Identity
Richness of threatened species (IUCN Red List)	-0.54ns	Gaussian	Identity
Abundance of threatened species (IUCN Red List)	0.15*	Poisson	Log
Richness of endemic species	-1.54ns	Gaussian	Identity
Abundance of endemic species	0.03ns	Quasipoisson	Log

C)	**Habitat (edge)**	**Fragment size (log)**	**Fragment size*habitat (edge)**	**Error distribution**	**Link function**	**Random**
Total carbon stock (Mg/plot)	-1.63ns	17.27***	-14.68***	Gaussian	Identity	Sites
Tree carbon stock (proportion of total)	-0.13***	0.04***	-	Gaussian	Identity	Sites
Liana carbon stock (proportion of total)	-0.18***	-0.04***	-	Gaussian	Identity	Sites

Standing dead trees carbon stock (proportion of total)	-	-0.06ns	-	Gaussian	Identity	Sites
Richness of threatened species (IUCN Red List)	-1.33ns	-	-	Gaussian	Identity	Sites
Abundance of threatened species (IUCN Red List)	-	0.13*	-	Negative binomial	Log	Sites
Richness of endemic species	-1.33ns	-	-	Gaussian	Identity	Sites
Abundance of endemic species	-0.26ns	0.052ns	-	Negative binomial	Log	Sites

d)	**Landscape scale**	**Total carbon stock (Mg/site)**	**Error distribution**	**Link function**	**Random**
Species richness	Control level	0.08ns	Gaussian	Identity	-
	Across fragment size (interiors)	-0.14ns	Gaussian	Identity	-
	Global model (Fragment size and habitats)	0.012ns	Gaussian	Identity	Sites
Richness of endemic species	Control level	0.07ns	Gaussian	Identity	-
	Across fragment size (interiors)	-0.04ns	Gaussian	Identity	-
	Global model (Fragment size and habitats)	-0.01ns	Gaussian	Identity	Sites
Abundance of endemic species	Control level	0.01	Poisson	Log	-
	Across fragment size (interiors)	0.002ns	Poisson	Log	-

	Global model (Fragment size and habitats)	0.004***	Negative binomial	Log	Sites
Richness of threatened species (IUCN Red List)	Control level	0.04ns	Gaussian	Identity	-
	Across fragment size (interiors)	-0.02ns	Gaussian	Identity	-
	Global model (Fragment size and habitats)	0.01ns	Gaussian	Identity	Sites
Abundance of threatened species (IUCN Red List)	Control level	0.02	Poisson	Log	-
	Across fragment size (interiors)	0.003ns	Poisson	Log	-
	Global model (Fragment size and habitats)	0.002ns	Negative binomial	Log	Sites
NMS (scores)	Control level	0.03**	Gaussian	Identity	-
	Across fragment size (interiors)	-0.02*	Gaussian	Identity	-
	Global model (Fragment size and habitats)	0.02***	Gaussian	Identity	Sites

VI. CONCLUSÕES GERAIS

A partir dos resultados obtidos nos três capítulos pôde-se concluir que:

(i) as mudanças no microclima e nos atributos do solo foram afetadas pela fragmentação, promovendo um impacto negativo sobre a biomassa das espécies arbóreas e aumentando a biomassa de lianas em direção aos fragmentos menores e mais intensivamente para o habitat de borda;

(ii) em uma paisagem fragmentada a funcionalidade das espécies arbóreas em fragmentos maiores foi diferente daquela existente em fragmentos menores e nas bordas, sendo que nesses ambientes houve uma maior expressividade de espécies sucessionais e com menores recursos para a fauna;

(iii) existem co-benefícios entre o estoque de carbono e biodiversidade na paisagem estudada, estando estes relacionados principalmente com aumento de tamanho dos fragmentos existentes na paisagem. Isso sugere que os fundos de REDD+ podem ser utilizados para promover a conservação dos remanescentes existentes e ainda amplificar o carbono e o valor biológico por meio de projetos de manejo e restauração florestal de fragmentos.

No entanto, como consideração final, ressaltamos que os pequenos fragmentos têm um papel importante na manutenção dos serviços ecológicos, tornando-os indispensáveis para a conservação da biodiversidade em um domínio fitogeográfico intensamente explorado e fragmentando como o da Floresta Atlântica, uma vez que esses fragmentos ainda abrigam uma parcela significativa da riqueza de espécies arbóreas, da biomassa e do carbono estocado na paisagem, além da sua funcionalidade ecológica, que por abrigar espécies com maior tolerância a distúrbios e com dispersão independente da fauna, se tornam imprescindíveis em projetos de restauração florestal, bem como no auxílio da sucessão natural de áreas degradadas.