MARIANA FERREIRA ROCHA

ACESSANDO A IMPORTÂNCIA DE CARACTERÍSTICAS ESTRUTURAIS E DA CONFIGURAÇÃO ESPACIAL DE REMANESCENTES LINEARES PARA CONSERVAÇÃO

LAVRAS-MG
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Orientador
Dr. Marcelo Passamani

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Dr. Luiz Fernando Silva Magnago

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APROVADA em 04 de abril de 2014.

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"What about sunrise? What about rain? What about all the things that you said we were to gain....What have we've done to the world, look what we've done! What about all the peace that you pledge your only son?... What about flowering fields? Is there a time, What about all the dreams that you said was yours and mine? Did you ever stop to notice, all the blood we've shed before? Did you ever stop to notice, this crying Earth, its' weeping shore?... I used to dream, I used to glance beyond the stars. Now I don't know where we are, although I know we've drifted far... What about yesterday? What about the seas? Heavens are falling down! I can't even breath...What about nature's worth? It's our planet's womb! What about animals? Have we lost their trust? What about forest trails? Burnt despite our pleas. What about the holy land? Torn apart by greed. What about everything? Where did we go wrong? What about us?"

Trechos da música Earth Song (Michael Jackson)

RESUMO GERAL

Globalmente, as florestas tropicais estão experimentando um aumento na taxa de perda de habitat maior do que qualquer outro ecossistema. A fragmentação florestal e perda de habitat causam diversas alterações na paisagem como aumento do isolamento entre remanescentes florestais, diminuição do tamanho dos remanescentes e substituição da floresta nativa por áreas agrícolas. Estas alterações na paisagem tem consequências drásticas, no que se refere a perda de espécies e disrupção de processos ecológicos, como alteração nas interações planta-animais, o que leva a alterações no funcionamento do ecossistema. Dessa forma, a sobrevivência a longo prazo das espécies em florestas tropicais depende de uma rede de remanescentes com um elevado nível de conexão, seja por meio de corredores de vegetação ou através de matrizes com alta permeabilidade. Assim, a conectividade da paisagem e as características estruturais dos remanescentes florestais são componentes-chave para a manutenção de espécies e processos ecológicos em paisagens fragmentadas e por isso, precisam ser melhor compreendidos. Desta forma, nosso estudo avaliou a influência de um conjunto de variáveis - relacionadas a características de fragmentos e conectividade estrutural - a nível de comunidade (composição, estrutura e diversidade), sob uma perspectiva funcional e por fim, a um nível de interação, considerando as espécies arbóreas com dispersão zoocórica. Mais especificamente, nosso objetivo foi avaliar o efeito das características estruturais e configuração espacial dos remanescentes lineares sobre essas variáveis para guiar estratégias de conservação relacionadas à importância e ao *design* de corredores de vegetação em paisagens fragmentadas, uma vez que poucas informações estão disponíveis. Para a análise de diversidade funcional, consideramos três índices: riqueza funcional, equabilidade funcional e divergência funcional. Consideramos como traços funcionais, àqueles relacionados à senibilidade das espécies à fragmentação, como tamanho do corpo, tamanho da prole, dieta e locomoção. Para as espécies arbóreas, consideramos o tipo de dispersão e o tamanho dos frutos e sementes. Nosso estudo foi realizado em uma área de Floresta Atlântica localizada no sudeste do Brasil. Amostramos cinco áreas, dentre: 1) interior de fragmento controle, 2) borda de fragmento controle, 3) remanescente lineare conectado à floresta controle, próximo a área de conexão, 4) remanescente linear conectado à floresta controle, distante da área de conexão e 5) remanescente linear não conectado. Para análise de dados, construímos modelos mistos e utilizamos o critério de informação Akaike para selecionar os melhores modelos. Nossos resultados mostraram que tanto a distância dos remanescentes lineares em relação à floresta contínua como a composição da matriz, influenciaram significativamente a maior parte dos atributos analisados. Por exemplo, a composição e estrutura de pequenos mamíferos se diferenciou nos tratamentos a medida que a distância a floresta controle aumenta. Além disso, tanto a conexão dos remanescentes lineares como a matriz circundante, influenciaram a abundância de indivíduos zoocóricos, bem como a abundância de indivíduos com frutos secos e carnosos de diferentes tamanhos, mais do que características estruturais dos remanescentes. Remanescentes lineares circundados por matrizes agrícolas abrigaram elevada abundância de generalistas e de indivíduos com baixa sensibilidade a fragmentação. Além disso, nossos resultados mostraram pela primeira vez que a diversidade funcional de pequenos mamíferos pode ser deteriorada mais pela influência de espécies exóticas do que pelas características estruturais e conectividade dos remanescentes florestais. Assim, concluímos que a implantação de remanescentes lineares em paisagens fragmentadas precisa ser cuidadosamente avaliada. A matriz do entorno dos remanescentes lineares deve ser considerada/manejada quando avaliamos o *design* dos corredores de vegetação em paisagens fragmentadas. Os remanescentes lineares conectados a fragmentos de maior tamanho devem ser considerados como prioridade para conservação em relação aos fragmentos isolados. Finalmente, concluímos que se adequadamente manejados os remanescentes lineares podem funcionar como importantes componentes para a conservação e precisam ser considerados em futuras decisões de conservação em paisagens onde eles já existem.

Palavras-chave: Ecologia de paisagem. Corredores de vegetação. Conservação. Manejo. Conexão. Composição de matriz. Pequenos mamíferos. Diversidade funcional. Espécie exotica. Dispersão.

ABSTRACT

Globally, tropical rainforests are experiencing a higher rate of habitat loss than any other ecosystem. The forest fragmentation and habitat loss cause diverse alterations in the landscape as increase of isolation among forest remnants, decrease of remnants size and replacement of native forest by agricultural matrices. These alterations drive drastic biodiversity loss in terms of species and ecological disruption as plant-animals interaction, changing the ecosystem functioning. Nevertheless, the long-term survival of many tropical species depends on a network of remnants with a high level of connectivity due to the presence of wildlife corridors and/or the permeable matrices. Thus, landscape connectivity and the structural features of forest remnants are key components for maintaining species and ecological processes in fragmented landscapes. In this way, we accessed the influence of the set of variables - fragment characteristics and structural connectivity on a community perspective (composition, structure and diversity), functional diversity perspective and a interaction level, evaluating the arboreal species with zoochoric dispersion. The main objective of our study was to infer about the spatial arrangement and configuration of linear remnants, providing a guideline to enhance the conservation value of these structures in fragmented landscapes and also, to bring information about the management of these structures, once these are still little understood. For the functional diversity analysis, we considered three indices: functional richness, functional evenness and functional divergence. For the functional traits analysis, we considered those related to fragmentation sensitivity, such as body and litter size, diet and locomotion. To the arboreal species, we considered the dispersion type and the fruit/seed size. Our study was carried out in an Atlantic Forest placed in southeastern Brazil. We sampled five rainforest habitats: 1) control forest fragment interior, 2) control forest edge, 3) linear remnants connected to the control forest, near the connection area and 4) linear remnants connected to the control forest, far from the connection area and 5) unconnected linear remnants. We used mixed models for data analysis and the Akaike Information Criterion to find the best models. Our results showed that both, the distance of linear remnants until the source fragment and the matrix composition have significant influences on the most of attributes. For example, the small mammals composition and structure in the treatments change with the increase of isolation from the source forest fragment. Moreover both, linear remnants connected and/or the surrounding matrix, influenced the abundance of zoochoric individuals and the abundance of species with fleshy and non-fleshy fruits of different sizes, more than structural characteristics. Linear remnants surrounded by the most forest matrices harbor high generalist abundance and individuals with low fragmentation sensitivity. Also, our results showed at the first time, that the small mammal functional diversity can be decayed by the influence of exotic species more than drive by structural characteristics and connectivity of forest remnants. As such, we concluded that the implantation of linear remnants in fragmented landscapes needs to be carefully designed. The matrix around the linear remnants needs to be considered in the vegetation corridor design in fragmented landscapes and in most cases, managed in conservation programs.The linear remnants connected to the largest fragments need to be consider as important targets for conservation in relation to the most isolated patches. In conclusion, the linear remnants if management, can work to conservation and need to be implemented in conservation plans in landscapes where their are.

Key-words: Landscape ecology. Vegetation corridors. Conservation. Management. Connection. Matrix composition. Small mammals. Functional diversity. Exotic species. Dispersion.

SUMÁRIO

PRIMEIRA PARTE

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1 INTRODUÇÃO GERAL

A fragmentação de habitats pode ser entendida como um "processo no qual uma grande quantidade de habitat é transformada em habitats remanescentes de menor tamanho, isolados um do outro por uma matriz de habitats diferente do original" (WILCOVE; MCLELLAN; DOBSON, 1986, p. 237). Seguindo esse conceito, pode-se distinguir quatro efeitos diretos desse processo: (I) subdivisão da vegetação remanescente em fragmentos e consequente aumento no número de fragmentos florestais; (II) perda na quantidade de vegetação original; (III) diminuição do tamanho dos remanescentes florestais e (IV) aumento da distância entre esses remanescentes (FAHRIG, 2003; BENNET; SAUNDERS, 2010).

As consequências negativas dos efeitos da fragmentação florestal na diversidade biológica já são bastante conhecidas e podem ser entendidas como a perda de espécies e mudanças na estrutura da comunidade de diversos grupos biológicos (LAURANCE et al., 2002; FAHRIG, 2003; OLIVEIRA; GRILLO; TABARELLI, 2004; EWERS; DIDHAM, 2006; PERES; PALACIOS, 2007; MAGNAGO et al., 2014). No entanto, o impacto da fragmentação pode ser ainda mais drástico quando consideramos os efeitos indiretos desse processo (LAURANCE, 2001) como as consequências advindas da exposição aos efeitos de borda (MURCIA, 1995; LAURANCE et al., 2002; OLIVEIRA; GRILLO; TABARELLI, 2004; EWERS; DIDHAM, 2008; MAGNAGO et al., 2014), caça (REDFORD, 1992; PERES, 2000; PERES; PALACIOS, 2007; CANALE et al., 2012), invasão de espécies exóticas (GIBSON et al., 2013) e substituição da cobertura vegetal por matrizes não florestais (LAURANCE, 2008; GARDNER et al., 2009; PARDINI et al., 2010; LAURANCE; SAYER; CASSMAN., 2014).

Ainda mais alarmante, é saber que estes efeitos podem ser mais drásticos do que os já relatados, uma vez que os impactos da fragmentação podem promover mudanças no funcionamento dos ecossistemas, alterando a diversidade funcional e as características funcionais das espécies em paisagens intensamente fragmentadas (ver FLYNN et al., 2009; BARRAGÁN et al., 2011; CADOTTE et al., 2011; MAGNAGO et al., 2014). No entanto, os estudos que mostram esse tipo de relação ainda são escassos (HAGEN et al., 2012).

Em paisagens fragmentadas dominadas por matrizes com baixa permeabilidade ao deslocamento dos organismos, os corredores de vegetação têm se mostrado uma alternativa para conectar (ou reconectar) espécies e populações nos fragmentos florestais isolados (BEIER; NOSS, 1998; PARDINI et al., 2005). Apesar da utilidade dos corredores variar dependendo da espécie considerada, em termos gerais, atualmente, a importância dos corredores de vegetação em paisagens fragmentadas é inquestionável (GIBERT-NORTON et al., 2009). No entanto, estudos têm mostrado que algumas características estruturais dos corredores de vegetação, como largura, comprimento e estrutura florestal são determinantes do seu uso pelas espécies com diferentes requerimentos de habitat e são importantes na avaliação do potencial destas estruturas para conservação (LAURANCE; LAURANCE, 1999; LAURANCE, 2004; HAWES et al., 2008; LEES; PERES, 2008; BARLOW et al., 2010). Também já é sabido que, além das características estruturais, as características da paisagem e o arranjo espacial dos fragmentos - como a composição da matriz do entorno, a proximidade com fragmentos fontes - influenciam a manutenção de espécies e processos ecológicos em paisagens fragmentadas (FAHRIG, 2003; LAURANCE, 2004; PARDINI et al., 2005; DAMSCHEN et al., 2006; EWERS; DIDHAM, 2006; HAWES et al., 2008; LEES; PERES, 2008; MARTENSEN; PIMENTEL; METZGER, 2008; BRUDVIG et al., 2009; BARLOW et al., 2010; PARDINI et al., 2010; MAGRACH; LARRINAGA; SANTAMARIA, 2011; MARTENSEN et al., 2011; PASSAMANI; FERNANDEZ, 2011; GARMENDIA et al., 2013).

Apesar disso, nenhum estudo avaliou juntamente, a influência da conectividade, das características estruturais e do arranjo espacial dos corredores de vegetação para a conservação em paisagens fragmentadas. Esse conhecimento é importante, pois a partir dele, pode-se, guiar estratégias efetivas de manejo em paisagens fragmentadas, considerando o arranjo espacial e a configuração dos corredores de vegetação. Estudos desse tipo têm sido bastante recomendados (EWERS; DIDHAM, 2006; LEES; PERES, 2008), principalmente quando realizados em florestas tropicais, que atualmente são representadas por poucos fragmentos que de fato garantem proteção à biodiversidade (RIBEIRO et al., 2009; LAURANCE et al., 2012) e situados em uma paisagem constituída em sua maior extensão, por diferentes tipos de interferência antrópica (RIBEIRO et al., 2009; BENNET; SAUNDERS, 2010; LAURANCE; SAYER; CASSMAN, 2014). Apesar disso, poucos estudos com essa abordagem foram realizados e o conhecimento ainda é muito limitado (HAWES et al., 2008; LEES; PERES, 2008; MARTENSEN et al., 2011).

Desta forma, o objetivo do presente trabalho foi avaliar a influência das características estruturais de fragmentos florestais - como tamanho e formato - e da conectividade estrutural da paisagem na comunidade de pequenos mamíferos, na sua diversidade funcional e nos traços funcionais das espécies. Além disso, esses efeitos foram testados na abundância de espécies arbóreas com dispersão zoocórica e na produtividade primária das florestas, uma vez que estes são considerados importantes preditores para avaliar os efeitos da fragmentação na comunidade arbórea e também descrevem as interações de recursos com a fauna (HAGEN et al., 2012). Mais especificamente, objetivo-se avaliar o efeito das características estruturais e configuração espacial dos remanescentes lineares sobre essas variáveis para guiar estratégias de conservação relacionadas à importância e ao *design* de corredores de vegetação em paisagens fragmentadas.

2 REFERENCIAL TEÓRICO

2.1 A Teoria da Biogeografia de Ilhas (MACARTHUR; WILSON, 1963, 1967) e sua influência nos estudos em paisagens fragmentadas

Na década de 60, MacArthur & Wilson (1963; 1967) propuserem um modelo para explicar o número de espécies presentes em ilhas insulares. De acordo com esse modelo, ilhas maiores teriam maior riqueza de espécies do que ilhas menores, seja, pelo próprio tamanho da ilha ou pela menor probabilidade de extinção das populações por eventos estocásticos. Também, ilhas mais próxima ao continente, teriam maior probabilidade de serem colonizadas ou re-colonizadas (via imigração de espécies a partir do continente) do que ilhas mais distante. Essas predições mostraram a importância do tamanho da ilha e da distância destas até o continente (ou isolamento) para prever a extinção e colonização de espécies em uma área.

A Teoria da Biogeografia de Ilhas inspirou diversas pesquisas em ecossistemas fragmentados (LAURANCE, 2008), uma vez que o processo de fragmentação florestal leva à redução do tamanho dos fragmentos florestais e aumento do isolamento entre eles (FARHIG, 2003; BENNET; SAUNDERS, 2010), variáveis já conhecidas por influenciar o número de espécies em ilhas insulares. Diversos estudos foram realizados para explicar o impacto da fragmentação florestal sobre diversos grupos biológicos (TURNER, 1996; CHIARELLO, 1999; PARDINI, 2004; VIEIRA et al., 2009; PASSAMANI; FERNANDEZ, 2011). Apesar de alguns estudos terem confirmado as predições da Teoria da Biogeografia de Ilhas para "ilhas terrestres" ou fragmentos florestais (ver VIEIRA et al., 2009; PREVEDELLO; VIEIRA, 2010) outros encontraram resultados que contradiziam essa Teoria (ver LAURANCE et al., 2002; PARDINI, 2004; PASSAMANI; FERNANDEZ, 2011).

Dessa forma, os estudos avaliando os impactos da fragmentação sobre a diversidade biológica começaram a verificar que outros fatores, que não só o tamanho e o isolamento *per se* dos remanescentes florestais, influenciavam a diversidade de espécies e a dinâmica em paisagens fragmentadas. Assim, importante atenção foi e ainda é dada para a composição da matriz no entorno dos fragmentos (LAURANCE et al., 2002; PIRES et al., 2002; LAURANCE, 2008; UMETSU; PARDINI, 2007; PASSAMANI; RIBEIRO, 2009; WATLING et al., 2011), aos efeitos de borda (MURCIA, 1995; LAURANCE et al., 2007; EWERS; DIDHAM, 2008), pressão de caça (REDFORD, 1992; CHIARELLO, 1999; PERES, 2000; PERES; PALACIOS, 2007; CANALE et al., 2012) e impacto de espécies exóticas (GIBSON et al., 2013) governando a riqueza e abundância de espécies em paisagens fragmentadas. Desse modo, a aplicação da Teoria da Biogeografia de Ilhas em estudos com paisagens fragmentadas possui limitações, uma vez que importantes fenômenos que ocorrem nessas paisagens não são levados em consideração (ver LAURANCE, 2008 para uma breve revisão). Apesar disso, é inegável a importância dessa teoria na discussão de tamanhos de reservas e da conectividade na manutenção da diversidade de espécies em paisagens fragmentadas (LAURANCE, 2008).

2.2 Corredores de vegetação e sua importância no contexto da fragmentação florestal

O conceito de fragmentação florestal proposto por Wilcove, McLellan e Dobson (1986) mostra que a fragmentação provoca efeitos diretos no contexto de paisagem, alterando a estrutura e o arranjo espacial dos remanescentes florestais (FARIGH, 2003; LAURANCE, 2008; BENNET; SAUNDERS, 2010). Dentre esses efeitos, o isolamento efetivo entre fragmentos florestais têm consequências negativas para a diversidade biológica e para a manutenção de processos ecológicos, como interações entre plantas zoocóricas e seus dispersores (JORDANO et al., 2006; HAGEN et al., 2012; MAGRACH; LARRINAGA; SANTAMARIA, 2012).

Dessa forma, a conectividade funcional da paisagem, dada pela capacidade de movimentação dos indivíduos em paisagens fragmentadas (TISCHENDORF; FAHRIG, 2000), é associada à persistência das espécies em paisagens fragmentadas (LAURANCE, 1994; PIRES et al., 2002; VIVEIROS DE CASTRO; FERNANDEZ, 2004; PARDINI, 2004; UMETSU; PARDINI, 2007; PASSAMANI; RIBEIRO, 2009).

Em paisagens fragmentadas dominadas por matrizes com baixa permeabilidade ao deslocamento dos organismos, os corredores de vegetação têm se mostrado uma alternativa para conectar (ou reconectar) espécies e populações nos fragmentos florestais isolados (BEIER; NOSS, 1998; PARDINI et al., 2005). No entanto, apesar dos corredores de vegetação contribuírem para a conexão estrutural da paisagem (ver definição de conectividade estrutural em TISCHENDORF; FAHRIG, 2000), sua presença não garante a conectividade funcional de paisagens fragmentadas. Para serem reconhecidos como componentes da conectividade da paisagem, os corredores de vegetação devem permitir/facilitar o movimento de espécies entre fragmentos florestais (TISCHENDORF; FAHRIG, 2000).

A importância dos corredores como elemento conector era controversa e bastante discutida em meados da década de 1990. Segundo alguns autores, os corredores facilitavam a propagação de distúrbios, como patógenos e incêndios, aumentavam a exposição a predadores e, quando possuíam pequena largura, poderiam facilitar a entrada de espécies exóticas e serem mais suscetíveis aos efeitos negativos associados à borda, aumentando a abundância de espécies generalistas e restringindo seu uso para espécies mais restritas a florestas (SIMBERLOFF e COX, 1987; HOBBS, 1992). Apesar do utilidade dos corredores variar dependendo da espécie considerada, em termos gerais, atualmente, a importância dos corredores de vegetação em paisagens fragmentadas é inquestionável. Diferentes estudos mostraram que estas estruturas, além de contribuirem para o deslocamento de espécies entre fragmentos florestais (BENNETT, 1990; HOBBS, 1992), aumentam a diversidade alfa em fragmentos florestais conectados (PARDINI et al., 2005; DAMSCHEN et al., 2006), compartilham espécies em comum com os fragmentos fontes à que estão conectados (LIMA; GASCON, 1999; HAWES et al., 2008; LEES; PERES, 2008; BARLOW et al., 2010; ROCHA; PASSAMANI; LOUZADA, 2011) e ainda contribuem para a manutenção de interações mutualísticas entre fragmentos conectados, permitindo a conectividade ecológica de paisagens fragmentadas (TEWKSBURY et al., 2002; MAGRACH; LARRINAGA; SANTAMARIA, 2012). Ainda, os corredores de vegetação funcionam para a retenção do estoque de carbono e podem abrigar alta riqueza de espécies e endemismos, a um baixo custo econômico (JANTZ; LAPORTE, 2014).

Dessa forma, pode-se concluir que o estudo com corredores de vegetação contribuiu bastante para o entendimento sobre a importância da conectividade para a conservação de paisagens fragmentadas. No entanto, o conhecimento sobre essas estruturas e sobre a paisagem em que estão inseridas é muito limitado, o que pode subestimar sua efetividade para conservação no manejo de paisagens fragmentadas.

3 CONCLUSÕES GERAIS

A partir dos resultados obtidos nos três capítulos, foi possível concluir que:

1) A conectividade estrutural da paisagem, provida pela presença de remanescentes florestais conectados à floresta controle e pela composição da matriz, exerceram influências significativas na diversidade beta, na composição e estrutura da comunidade de pequenos mamíferos e na abundância de espécies especialistas e generalistas. Também influenciaram significativamente a distribuição de espécies zoocóricas na paisagem e também a abundância de diferentes atributos relacionados à zoocoria. Esses resultados mostram a importância da conectividade estrutural da paisagem na redução dos efeitos negativos do isolamento;

2) Remanescentes florestais lineares conectados à floresta controle são mais importantes para conservação do que remanescentes lineares não conectados e por isso devem ser priorizados. Pode-se afirmar isso uma vez que a composição e estrutura da comunidade de pequenos mamíferos nos remnanescentes florestais lineares conectados não diferiram significativamente da comunidade amostrada na floresta controle. No entanto, os remnanescentes florestais lineares não conectados apresentaram uma comunidade de pequenos mamíferos mais homogêneos (menor diversidade beta), com maior abundância de generalistas e distinta da floresta controle. Além disso, os remanescentes lineares conectados apresentaram atributos relacionados à zoocoria que foram encontrados também no interior da floresta controle, como maior abundância de espécies com frutos carnosos e espécies com frutos médios e carnosos. Alguns atributos relacionados à comunidade de pequenos mamíferos e a zoocoria, foram diferentes no mesmo remanescente linear, quando considerado a distância de

conexão até a floresta fonte. Dessa forma, a distância entre os remanescentes lineares e a floresta fonte (ou grandes blocos florestais) é uma importante variável a ser considerada em estratégias de manejo e conservação em paisagens fragmentadas;

3) Remanescentes lineares circundados por matrizes compostas de áreas agrícolas tiveram maior abundância de espécies generalistas e de indivíduos com traços funcionais relacionados à baixa sensibilidade à fragmentação. De forma oposta, remanescentes lineares circundados por matrizes mais florestadas apresentam menor abundância de generalistas e maior riqueza e abundância de indivíduos com traços funcionais relacionados a maior sensibilidade à fragmentação. Assim, em longo prazo, é esperado que a substituição de matrizes florestais por matrizes agrícolas tenha consequências negativas, causando a perda de grupos funcionais mais sensíveis à fragmentação. Dessa forma, o manejo da matriz no entorno dos remanescentes lineares é extremamente necessário para aumentar o valor de conservação desses remanescentes em paisagens fragmentadas;

4) Características estruturais dos remanescentes florestais lineares, como forma, área e largura também precisam ser levados em consideração no delineamento ou escolha dos corredores de vegetação para serem conservados;

5) A presença e alta abundância de espécies exóticas têm influências negativas na diversidade funcional (representada pela riqueza funcional, equabilidade funcional e divergência funcional) e nos traços funcionais de pequenos mamíferos;

6) Dessa forma, os resultados mostram que os corredores de vegetação se adequadamente manejados, são importantes para conservação em paisagens fragmentadas e precisam ser considerados em futuras decisões de conservação em paisagens onde eles já existem.

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SEGUNDA PARTE - ARTIGOS

ARTIGO 1

Patch characteristics and structural connectivity evaluation to assess the effectiveness of linear remnants in a Tropical Landscape

ABSTRACT

Globally, tropical rainforests are experiencing a higher rate of habitat loss than any other ecosystem. Nevertheless, the long-term survival of many tropical species depends on a network of remnants with a high level of connectivity due to the presence of wildlife corridors and/or the permeable matrices. Thus, landscape connectivity and the structural features of forest remnants are key components for maintaining species and ecological processes in fragmented landscapes. Our main objective was to infer about the spatial arrangement and configuration of linear remnants, providing a guideline to enhance the conservation value of these structures in fragmented landscapes, using the small mammal group. For this, we accessed the influence of the set of variables - fragment characteristics and structural connectivity - on the alpha and beta diversity, species composition and community structure. We also verified if there is a difference in the habitat preferences of small mammal specialist and generalist species. Our study was carried out in an Atlantic Forest placed in southeastern Brazil. We sampled five rainforest habitats: 1) control forest fragment interior, 2) control forest edge, 3) linear remnants connected to the control forest, near the connection area and 4) linear remnants connected to the control forest, far from the connection area and 5) unconnected linear remnants. We sampled at total 25 sites, using an effort of 12,000 trap-nights. We used mixed models for data analysis and the Akaike Information Criterion to find the best models. Our results showed a strong influence of fragments size and connectivity on the alpha and beta diversity. Also, the linear connected remnants shared a small mammal community more similar to that of the control forest, whilst the unconnected linear remnants had a significantly different small mammal community. We found strongest effects of the linear remnant spatial arrangements, provided by the surrounding matrix, on the small mammal attributes and the specialist and generalist abundances. As such, we concluded that the implantation of linear remnants in fragmented landscapes needs to be carefully designed. The linear remnants connected to the largest fragments need to be consider as important targets for conservation in relation to the most isolated patches. Also, the width and shape of linear remnants and the management of the surrounding matrix have to be considered in the vegetation corridor design in fragmented landscapes.

Key-words: Brazilian Atlantic Forest. Isolation. Matrix. Vegetation corridors. Conservation. Management. Small mammals. Fragmented landscapes.

1 INTRODUCTION

The importance of Tropical Forests for biodiversity conservation is well known since they are the biologically richest ecosystems on Earth (GENTRY, 1992; GARDNER et al., 2009). These environments also provide local, regional and global benefits for humans through the provision of economic benefits and ecosystem services (GARDNER et al., 2009). Primary forests are the main repository of the tropical biodiversity (GIBSON et al., 2011). However, the current number of primary and large control forests still preserved is too small (Lairana, 2005), and half of the protected areas around the world are not effective for biodiversity conservation (LAURANCE et al., 2012).

Worse and more alarming is that, in consequence of the fragmentation process, most of the biodiversity is living in forest fragments of different sizes and shapes with different degrees of isolation (RIBEIRO et al., 2009; BENNET; SAUNDERS, 2010; ELLIS et al., 2010, HILL et al., 2011). Species composition in these remnants and in the surrounding non-forest habitats are a subset of species found in primary forests (LOUZADA et al., 2010; HILL et al., 2011).

As reported by MacArthur and Wilson (1967) for the insular biota, changes in island size and isolation also affect the terrestrial biota in tropical forest fragments (see LAURANCE et al., 1998; CHIARELLO, 1999; PARDINI, 2004; FERRAZ et al., 2007; PASSAMANI; FERNANDEZ, 2011; VIEIRA et al., 2009; PREVEDELLO; VIEIRA, 2010; GARMENDIA et al., 2013; GIBSON et al., 2013). However the terrestrial biota losses are mainly influenced by secondary fragmentation effects, related to high edge exposition (MURCIA, 1995; LAURANCE et al., 2002; EWERS; DIDHAM, 2008), hunting pressure (REDFORD, 1992; CHIARELLO, 1999; PERES, 2000; PERES; PALACIOS, 2007; CANALE et al., 2012), logging and fire increases (LAURANCE et al., 2012), exotic species invasion (FERREIRA et al., 2012) and introduction of new land-use forms in the landscape which may not be permeable to animal displacement (FOLEY et al., 2005; LAURANCE, 2008; DIXO; METZGER, 2009; GARDNER et al., 2009; MORRIS, 2010; MAGRACH; LARRINAGA; SANTAMARIA, 2012; LAURANCE; SAYER; CASSMAN, 2014). For this reason the most effective conservation strategies are those focused on providing information regarding how biodiversity is affected and maintained in fragmented landscapes, recovering and managing ecosystems embedded in anthropogenic lands (PARDINI et al., 2005; EWERS; DIDHAM, 2006; GARDNER et al., 2009; ELLIS et al., 2010).

In the last decades the importance of landscape connectivity for reducing biodiversity loss was highlighted (LUQUE; SAURA; FORTIN, 2012). This connectivity is provided through the association of structural connectivity (for example, by the presence of vegetation corridors and matrix composition) and functional connectivity, related to the permeability of these landscape elements to the movement/dispersion of individuals (TISCHENDORF; FAHRIG, 2000; LAURANCE et al., 2002; LAURANCE, 2004; PARDINI, 2004; PARDINI et al., 2005; UMETSU; PARDINI, 2007; MARTENSEN; PIMENTEL; METZGER, 2008; DIXO; METZGER, 2009; MAGRACH; LARRINAGA; SANTAMARIA, 2011; ROCHA; PASSAMANI; LOUZADA, 2011; MESQUITA; PASSAMANI, 2012; CASTRO; VAN DEN BERG, 2013).

In this way the structural features of remnants - like size and shape - and the structural connectivity - given by the distance to the source forest fragment, the composition of matrix surrounding the fragments and the structural connection by linear remnants - are key components for maintaining species and ecological processes in fragmented landscapes (to remnants size, see LAURANCE, 2004; EWERS; DIDHAM, 2006; LEES; PERES, 2008; MARTENSEN; PIMETEL; METZGER, 2008; MARTENSEN et al., 2012; GARMENDIA et al., 2013; to remnants shape, see LAURANCE, 2004;

MAGRACH; LARRINAGA; SANTAMARIA, 2011; GARMENDIA et al., 2013; to remnants width, LAURANCE, 2004; LEES; PERES, 2008; ASKINS; FOLSOM-O'KEEFE; HARDY, 2012; to forest cover, connectivity and matrix quality, see LAURANCE, 1994; GASCON et al., 1999; LAURANCE, 2004; PARDINI et al., 2005; HAWES et al., 2008; MARTENSEN; PIMENTEL; METZGER, 2008; BARLOW et al., 2010; PARDINI et al., 2010; MAGRACH; LARRINAGA; SANTAMARIA, 2011 and 2012; ROCHA; PASSAMANI; LOUZADA, 2011; WATLING et al., 2011; MARTENSEN et al., 2012; GARMENDIA et al., 2013). Moreover, these characteristics are important to infer about the best spatial arrangement and configuration of linear remnants for conservation in human-dominated landscapes and are poorly understood, mainly in tropical areas (HAWES et al., 2008; LEES; PERES, 2008; MARTENSEN et al., 2012). Also, the synergistic effects of the structural features and connectivity to evaluate the effective value of linear remnants have never been tested before, which hinders any decision for the protection/implantation of linear remnants in fragmented landscapes. Furthermore, studies with this focus have been recommended as broadly relevant to conservation in managed landscapes (EWERS; DIDHAM, 2006; LEES; PERES, 2008; FERREIRA et al., 2012).

We evaluated the effects of fragment structures and landscape connectivity on the small mammal group, considering the alpha and beta diversity, species composition and community structure. We also wanted to verify if there are differences in the responses of specialist and generalist species to these effects. More specifically, our main objective was to to assess the influences of structural characteristics and spatial configuration of linear remnants on the small mammal community to guide the conservation strategies and the vegetation corridor design in fragmented landscapes.

2 MATERIAL AND METHODS

2.1 Study area

Our study was carried out in southeastern Brazil (19° 11 '52 "S and 40° 5' 29" W - 18° 54 '18 "S and 40° 5' 19 "W). The study area is located in one of the most important global hotspots (MYERS et al., 2000) in a keystone biodiversity area (PAESE et al., 2010). The landscape studied comprises a large forest of 46,000 ha belonging to the Companhia Vale S.A., a privately-owned company, and to the federal government (Reserva Biológica de Sooretama) surrounded by a matrix composed mainly of *Eucalyptus* spp., papaya and coffee plantations and pasture (PEIXOTO et al., 2008; ROLIM et al., 2005) and by forest fragments of different sizes, shapes, widths and degrees of isolation. This forest is the second largest reserve of Tableland Forest (PEIXOTO et al., 2008; PEIXOTO; SIMONELLI, 2007; SBREK-ARAÚJO; CHIARELLO, 2008) and the one of the largest forest remnants of the Atlantic Forest (RIBEIRO et al., 2009). Furthermore it is considered one of the 14 centers with the highest plant diversity in Brazil (PEIXOTO; GENTRY, 1990; PEIXOTO; SILVA, 1997), the second most important area for mammal conservation in the Brazilian Atlantic Forest (GALLETI et al., 2009) and a refuge for threatened bird species (MARSDEN; WHIFFIN; GALETTI, 2001; SRBEK-ARAÚJO; CHIARELLO, 2006) and mammals (CHIARELLO, 1999).

The forest in the region is classified as Lowland Rain Forest (IBGE, 1987) or Tertiary Tableland because of its occurrence on Cenozoic sediments from the Barreiras group, with altitudes ranging from 28 to 65 m (PEIXOTO et al., 2008). The lowland forest is characterized with trees up to 40 m tall, girths up to 400 cm and a sparse understory, with the dominance of the Fabaceae, Myrtaceae, Rubiaceae, Annonaceae, Sapotaceae and Bignoniaceae tree families

(JESUS; ROLIM, 2005; PEIXOTO et al., 2008; PEIXOTO; SIMONELLI, 2007).

2.2 Sampling design

We selected the large forest previously mentionated as control and we sampled the small mammals within five rainforest habitats or treatments: 1) interior of control forest, 2) edge of control forest, 3) unconnected linear forest remnants (termed "unconnected linear remnants") and 4) linear forest remnants connected to the control forest (termed "connected linear remnants"). We separated the last treatment in two categories according to the distance until the structural connection as i) linear remnants connected near the control forest (placed after the edge), and ii) linear remnants connected far from the control forest, with the sample transect located along connected remnants and with a minimum distance of 400 m until the control forest. We chose these treatments considering the similarity in the composition of the surrounding matrix and a minimum distance of 400 m from the interior to the nearest edge of the control forest (range = 400 to 2,642 m) (see Figure1).

Our study included five sites per treatment totaling 25 sampling sites with a mean distance of 7,314 m between them. We established a 100 m transect in each sampling site, composed of six capture stations disposed in 20 m intervals. Each capture station received one large cage trap (45 x 16 x 16 cm) or one large Sherman (45 x 12.5 x 14.5 cm) on the ground, and one small Sherman trap (25 x 8 x 9 cm) in the understory vegetation, at a height of two meters. Traps were baited daily with a mixture of banana, peanut crumbs and sardine (fresh fish). We used the mark-recapture methods to evaluate the small mammal community in each treatment. All captured individuals were identified, weighed, measured, sexed, received a unique numbered ear tag (National Band and Tag

Inc.), and were released at the same capture station. Voucher specimens of all species were collected and deposited in the mammal collection at the Federal University of Espírito Santo (UFES-MAM). All procedures regarding the capture and marking of animals were conducted under the legal approval and consent of the Brazilian Federal Authority (IBAMA license number 27369-4).

We sampled each site for a total of 40 days distributed between April 2012 and May 2013, providing a combined total sampling effort of 12,000 trap-nights. We randomized the sites to be sampled and selected fifteen sites to sample for 10 days in each month.

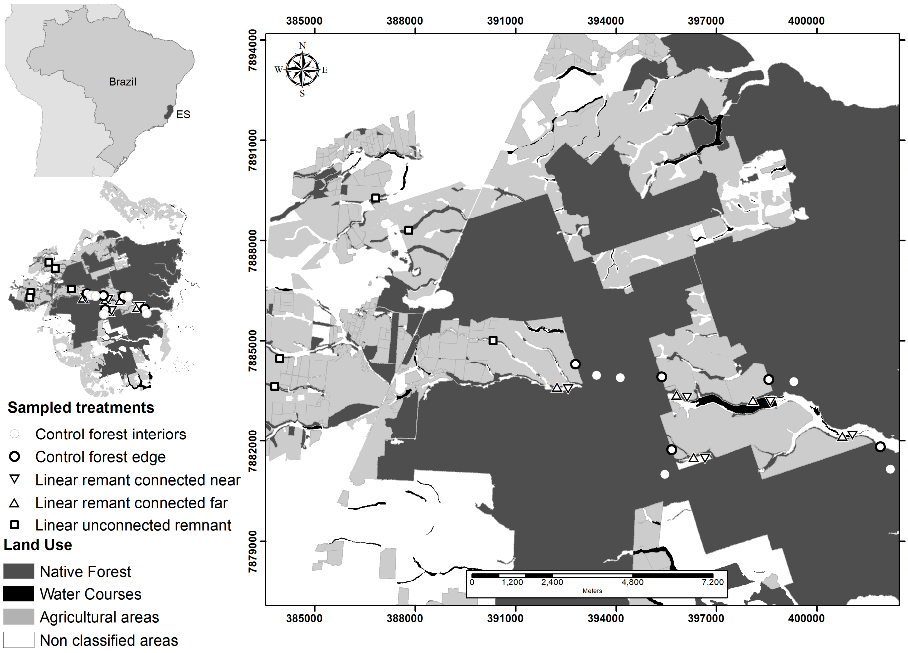

Figure 1 Study area and sampled treatments in a Tableland Forest in southeastern Brazil.

2.3 Land cover analysis and Independent variables

The metrics utilized to characterize the landscape changes and connectivity were extracted from a land cover map produced with an image with a high spatial resolution classification. We used an image with resolution of one meter, acquired in the year 2008. The orthorectified images and with atmospheric correction and visual evaluation of image registration, was obtained through the Vale Natural Reserve.

To classify the land cover we an used image based on multiscale segmentation The segmentation partitioned the image into groups of pixels spectrally similar and spatially adjacent (DESCLÉE; BORGAERT; DEFOURNY, 2006; DUVEILLER et al., 2008), using a "trial-and-error" attempt to find an fragmentation scale appropriate value. Once a successfully segmented image was obtained using 40 as a scale factor, we applied an object-based classification using Nearest Neighborhood (NN). We used 20 trained samples obtained in the field to apply the NN classification algorithm. The result is a class label for each of the segments in each class. A few wrongly-classified image objects were reassigned manually to the correct classes based on field knowledge and on visual interpretation of the image. Classification validation was obtained using 150 independent data sources as reference, randomly distributed over each class. User accuracy, producer accuracy, overall accuracy and kappa coefficient obtained high values, above 85%.

The resulting map was converted to vector format and we computed seven continuous variables using ArcGis (Table A.1, Supplementary material). For each sample treatment, we obtained their structural characteristics, such as size (hectare) and shape, using the ratio between area and perimeter according to Helzer and Jelinski (1999) and the mean width. For the mean width calculation, we obtained three widths for each treatment and considered the average among them. Also, to access the structural connectivity, we constructed a buffer with 2 km around each sampling treatment (total of 25). We quantified the amount of agricultural areas in the buffer (representing by coffee, *Eucalyptus* spp. and papaya plantations), the amount of native forest in the matrix and the amount of native forest of each treatment. Also, we measured the minimum distance between the sample treatment and the nearest source fragment and also, the mean distance to the neighbors nearest fragments. For this, we considered the four fragments nearest the sample treatment. We used these variables to characterize the spatial arrangement of linear remnants, as well. We chose these variables since they are key components to maintain species and ecological processes in fragmented landscapes, are essential to infer about the best spatial arrangement and to the evaluate the configuration of linear remnants for conservation in human-dominated landscape (see LAURANCE, 2004; PARDINI et al., 2005; EWERS; DIDHAM, 2006; HAWES et al., 2008; LEES; PERES, 2008; MARTENSEN; PIMENTEL; METZGER, 2008; BARLOW et al., 2010; MAGRACH; LARRINAGA; SANTAMARIA, 2011; ROCHA; PASSAMANI; LOUZADA, 2011; MARTENSEN et al., 2012; GARMENDIA et al., 2013).

We used two categorical variables to evaluate the effects of the structural connection among linear remnants and control forest (connected and unconnected), and the distance of the connection to the remnants (connected near and far). Furthermore, we used the width of linear remnants connected and unconnected as structural features. We considered the same variables used for evaluate the structural connectivity (described above) to characterize the spatial arrangement of linear remnants.

Thus, we verified the influence of habitat alteration, habitat features, structural connectivity and the effects of the structural features and spatial arrangement of linear remnants on the small mammal community (species composition and structure) and on the specialist and generalist abundances.

2.4 Dependent variables

We chose the small mammal group (rodents and opossums) since they are sensitive to landscape changes and can provide answers to important questions related to the biodiversity maintenance in fragmented ecosystems (PARDINI et al., 2005). Forest-specialist and endemic species are negatively affected by the size and isolation of Atlantic Forest fragments (VIEIRA et al., 2009), the loss of vegetation cover and the conversion of native forests into agricultural areas (UMETSU; PARDINI, 2007; PARDINI et al., 2010; PASSAMANI; FERNANDEZ, 2011). We considered the small mammal composition and structure, the alpha and beta diversity, and the specialist and generalist abundances as dependent variables.

We classified the captured species into specialist and generalist categories (Table B.1, Supplementary Material), based on species distribution in relation to the major neotropical bioma and habitat type, considering the information available in the literature (see PARDINI, 2004; PARDINI et al., 2005; UMETSU; PARDINI, 2007; PASSAMANI; RIBEIRO, 2009; PARDINI et al., 2010; PASSAMANI; FERNANDEZ, 2011). Thus, species which occur in the Atlantic Forest biome and were captured exclusively in the interior of fragments and/or that presented a high abundance in this habitat, were considered as "specialists". Otherwise, we considered the most captured species or most abundant species in the edge of fragments or in the agricultural matrix surrounding the forest fragments as “generalists”. *Nectomys squamipes* was considered generalist in our study since this species presents a preference for aquatic habitats, independent of being placed in the interior or forest edge.

2.5 Data analysis

As the sampling effort was equal for all treatments, each transect was considered a sampling unit or replicate. In this way, we obtained the abundance of specialist and generalist species by the number of individuals captured in each replicate.

We obtained the alpha and beta diversity values in each treatment trough the linear regression of log-species and log-individuals (HUBBEL, 1997), using the linearized power-law model of Arrhenius (1922) where $\mathrm{Log}S = \mathrm{Log}c + z\mathrm{Log}A$ and considered the concept of alpha and beta diversity suggested by Hubbel (1997). Thus, in a linear regression model, alpha diversity is represented by the regression intercept and corresponds to the number of species added considering a minimum number of samples (individuals). The beta diversity is represented by the line slope (z-slope) and a steeper slope indicates that a greater number of species is pooled as more individuals are sampled. We compared the diversities through the F test and their confidence intervals and considered a significance of $p \leq 0.05$.

We assessed the small mammal composition and the community structure (abundance of individuals) using a Non-metric Multi-dimensional Scaling (NMDS) ordination index based on Jaccard (for composition) and Bray-Curtis (for community structure) similarities with 1,000 restarts in the Primer Program (v.6). We used these analyzes to identify changes in species composition and community structure between the sampled treatments. We then used the two axis values generated by the NMDS to verify if species composition and community structure is affected by the fragment structural features and structural connectivity, considering all treatments. For these analyses, we used the R package version 2.15.1 (R Development Core Team 2012), and constructed generalized mixed models to solve the pseudoreplication problem between replicates placed in the same treatment. The same framework was used for specialist and generalist species analysis, but in this case we used the abundance data.

We also ran the model analyses considering only the connected/unconnected linear remnants to evaluate the influence of the structural features and the spatial arrangement of linear remnants on the small mammal composition and community structure. Moreover, we wanted to verify if the habitat preferences of specialist and generalist species are influenced by these effects. Within the models considering just the linear remnants, we used an interaction with three levels (connected near, connected far and unconnected) to check the effects of linear remnant connections (connected and unconnected) and to assess the influences of the distance effect within connected remnants (connected near and far) between these levels separately. The analyses regarding structural features and structural connectivity were run separately to verify the influence of each variable on the small mammal composition and structure and on the specialists and generalists abundances.

For the analyses with count data (abundance), we constructed mixed models using the *glmmadmb* function from the *glmmADMB* package and Negative Binomial error distributions for count data, since all data showed significant overdispersion. For uncountable data (NMDS axis) we used an *lmer* function from the *lme4* package with Gaussian family distribution. The sites (each treatment) 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 variables included in the global model. We excluded the same model variables with autocorrelations (linear Pearson correlation higher or equal to 0.5) to avoid multicollinearity.

To select the best model for both analyses (with all treatments and for linear remnants only) we used a theoretical information approach based on the

Akaike Information Criterion of Second Order, which is indicated for small sample sizes (AICc) and chose the models according to the lowest AICc value (BURNHAM; ANDERSON; HUYVAERT, 2011). The plausibility of alternative models was given by the differences in their AICc values in relation to the AICc of the most plausible model (ΔAICc). We considered as plausible models those with a value of $\Delta AICc<2$. When the models showed the ΔAICc value <2 and the variable included in the models was significant (considering $p\leq0.05$), we considered the variable appropriate to induce changes in species composition, community structure, and specialist and generalist abundances.

3 RESULTS

3.1 Small mammal community in a fragmented landscape

We captured 194 small mammal individuals of a total of 11 native and one exotic species, represented by eight marsupials and four rodents (Table B.1, Supplementary Material). The abundance across the treatments ranged from 17 individuals in the interior of the control forest to 65 individuals in unconnected linear remnants. In the edges of the control forest we recorded 45 individuals, 44 individuals in the far linear connected remnants and 23 individuals in the near linear remnants connected.

3.2 The alpha and beta diversity and influences of the structural characteristics of fragments and the structural connectivity on the small mammal community

The alpha diversity was significantly lower in the control forest compared to connected and unconnected linear remnants (Figure 2, Table 1).

The highest alpha diversity was observed to the unconnected remnants, as illustrated by the linear regression intercept. We found the opposite result for the beta diversity, as showed by the slope of log-species and log-individual in the linear regression in the graph (Figure 2, Table 1). The beta diversity was significantly higher in the interior of the control forest than in other treatments, did not differ between the edge of the control forest and the connected near linear remnants, decreased in the far connected linear remnants and was significantly lower in the unconnected linear remnants.

Table 1 Significance of *F* test for alpha (intercept) and beta diversity (slope) of linear regression between the treatments on a Tableland Forest in southeastern Brazil. Different letters denote significant differences at the $p \leq 0.05$ using pairwise comparisons of confidence intervals (CI). Label: interior of control forest (CFi), edge of control forest (CFe), linear remnants connected near (CRn), linear remnants connected far (CRf) and linear remnants unconnected (UC).

Estimator	**CFi**	**CFe**	**CRn**	**CRf**	**UC**
Log c (intercept)	0.021^a	0.025^a	0.142^b	0.181^b	0.28^c
Z (slope)	0.667^a	0.534^b	0.57^b	0.47^c	0.31^d

Z (slope) = b-diversity; Log c (intercept) = alpha-diversity

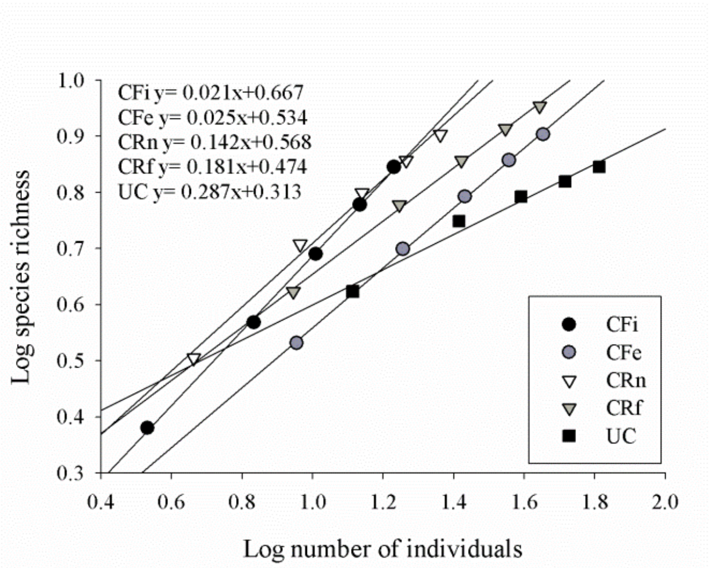

Figure 2 Linear regression showing the opposite changes in alpha and beta diversity of small mammal between the treatments on a Tableland Forest in southeastern Brazil. All the linear regression had $R^2 > 0.98$. Label: interior of control forest (CFi), edge of control forest (CFe), linear remnants connected near (CRn), linear remnants connected far (CRf) and linear remnants unconnected (UC).

The habitat alteration had influence on both species composition and small mammal community structure from the NMDS axis 1 analysis. Species composition was significantly different between unconnected linear remnants and the other treatments, and did not differ between the connected linear remnant and the control forest (Table C.1, Supplementary Material). A similar result was observed for the community structure. Although this parameter showed a significant difference between the unconnected linear remnants and the control fragment, the community structure did not differ among connected and unconnected linear remnants (Table C.1, Supplementary material). Changes in species composition and community structure for Axis 2 cannot be explained by the selected models (no significant results, Table C.1, Supplementary Material).

The fragment size and shape did not influence changes in species composition and community structure for both NMDS axis (Table D.1, Supplementary Material). In contrast, an increase in the distance to the control forest significantly influenced the species composition and community structure, as showed by the models for Axis 1 (Table D.2, Supplementary Material; Figure 3).

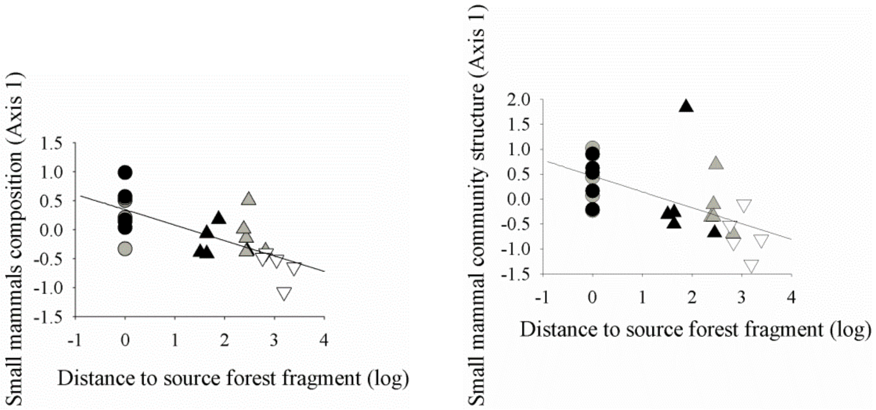

Figure 3 Best models results from Generalized linear mixed models showing the small mammal composition and community structure, obtained from NMDS axis 1, changes with the increase in the distance to the control forest, on a Tableland Forest in southeastern Brazil. Filled black and gray circles represent the interior and edge of control forest. Black and gray triangles represents the linear remnants connected near and far, respectively, and the empty triangles, the linear remnants unconnected.

In the sampled landscape, we captured 145 generalist individuals (5.8±3.77) and 49 specialists (1.96±2.26). The generalists abundance did not differ among the edge of the control forest, the far connected remnants and the unconnected remnants, and was significantly higher in these treatments than in the interior of the control forest and in the near connected remnants (Table E.1, Supplementary Material). Unexpectedly, the specialists abundance was higher in unconnected remnants than in other treatments, excepted for the edge of the control forest.

The fragment size and shape did not influence generalist and specialist abundances (Table D.1, Supplementary Material). However the best models selected showed that generalists abundance increased with an increase in the amount of agricultural areas (Figure 4, Table D.2, Supplementary Material), while the specialists abundance did not respond to the matrix in this fragmented landscape, with all results not significant (Table D.2, Supplementary Material).

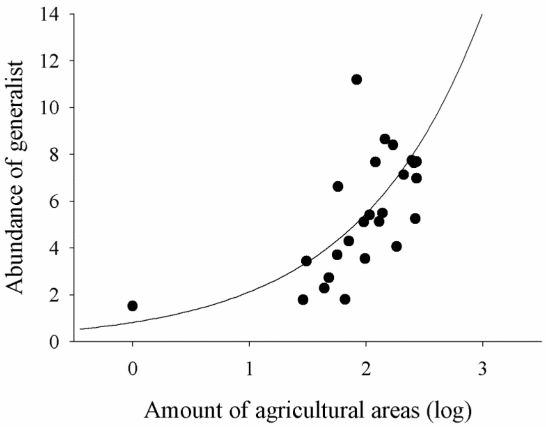

Figure 4 Best model result from Generalized Mixed Models showing that generalists abundance increase in fragments surrounded by highest amount of agricultural areas on a Tableland Forest in southeastern Brazil.

3.3 Influences of the structural features and spatial arrangement of linear remnants on the small mammal community

The species composition and community structure changed from the NMDS axis 2 and were explained by the selected models (Figure 5; Table F.1, Supplementary Material). Shifts in species composition happened as an isolation effect (GLMM; t= -3.56; p=0.004) and in consequence of the amount of agricultural areas in the matrix surrounding the linear remnants (GLMM; t= -3.49; p=0.003).

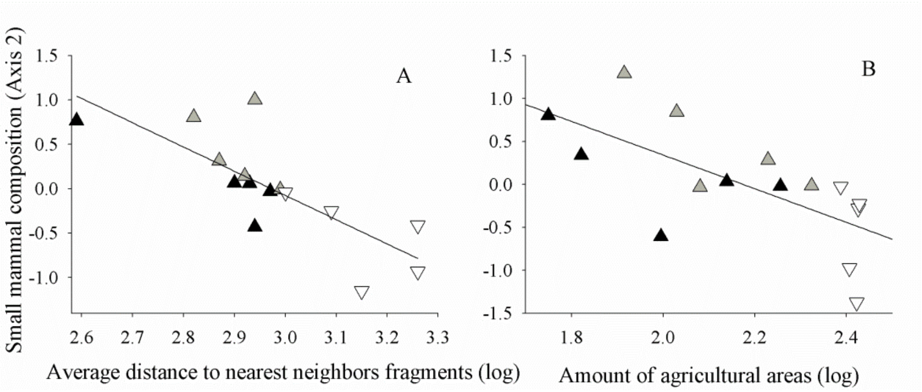

Figure 5 Best model result from Generalized linear mixed models showing that the changes on small mammal composition (from NMDS axis 2) are influenced by the average distance from the linear remnants to the nearest fragment (A), and by the amount of agricultural areas in matrices surrounding the linear remnants (B) on a Tableland Forest in southeastern Brazil. Black and gray triangles represents linear remnants connected near and far respectively, and empty triangles represents linear remnants unconnected.

The small mammal community structure changed in consequence of the linear remnant shape (GLMM; t= -3.87; p=0.02). Also, the community structure differed from the connected linear remnants to the unconnected linear remnants (GLMM; t=2.86; p=0.013, results from CRn and UC comparison and; t=4.1; p=0.001, results from CRf and UC comparison). As well as the species composition, the community structure was influenced by the amount of agricultural areas (GLMM; t=2.705; p=0.016) and native forests (GLMM; t= -3.49; p=0.016) in the surrounding matrix (Table F.1 Supplementary Material).

The generalist and specialist abundances responded differently to changes in the spatial arrangement and structural features of linear remnants. The generalist abundance was lower in remnants connected near the control forest compared to the linear remnants connected far (GLMM; z=2.65; p=0.08; Figure C.1, Supplementary Material) and the unconnected remnants (GLMM; z=2.22; p=0.026). Also, the highest generalist abundance was observed in the remnants more isolated from the source fragment (GLMM; z=3.05; p=0.002), surrounded by the largest amount of agricultural areas (GLMM; z=2.93; p=0.003) and with lowest amount of native forests (GLMM; z= -3.00; p=0.003), as demonstrated by the three best models selected (Table F.1, Supplementary Material; Figure 6).

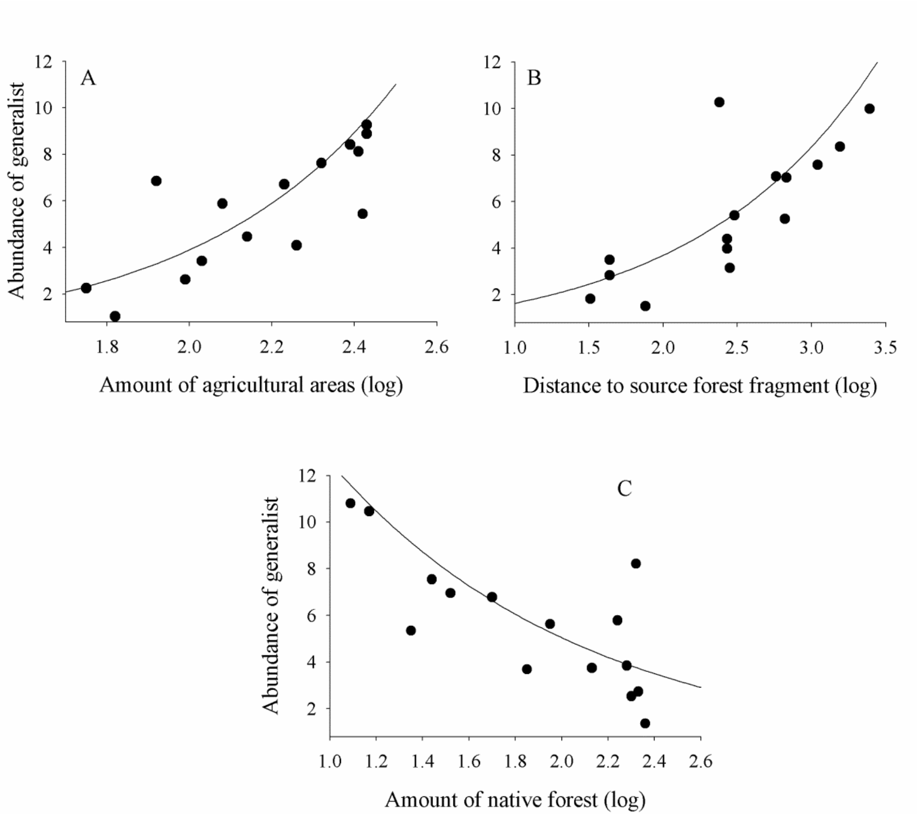

Figure 6 Best models results from Generalized linear mixed models showing that generalists abundance increase in linear remnants surrounded by most agricultural matrices (A), in linear remnants most distant to the control forest (B) in linear remnants surrounded by most deforested matrices (C), on a Tableland Forest at southeastern Brazil.

The specialist abundance was influenced negatively by the linear remnant sizes (GLMM; z=-2.44; p=0.01) and shapes (GLMM; z=-2.53; p=0.01), and positively influenced by the linear remnant widths (GLMM; z=2.72;

p=0.007). The unconnected linear remnants harbor a higher specialists abundance than the linear connected near (GLMM; z=2.22; p=0.027). However, the specialist abundance did not differ between the unconnected remnant and the linear remnant far (GLMM; z=0.32; p=0.75). Also, the specialist abundance was influenced positively by the isolation and increased in the more isolated linear remnants (GLMM; z=2.62; p=0.009).

4 DISCUSSION

4.1 Habitat changes and structural connectivity influences on the small mammal community

Our results showed that both, the size effect and the structural connectivity provides by linear remnants had strong influence on the alpha and beta diversity of small mammals. The higher alpha diversity in linear remnants compared to the control forest shows that, according to Hubbell's unified theory (1997) more species can be found in these linear remnants when analyzing the same number of captured individuals. It shows to be a reflection of the size effect, broadly demonstrated by the species-area relation (ROSENZWEIG, 1995). As the linear remnants have smaller size in comparison with the control forest, a high number of species is clustered in a small space, resulting in a higher alpha diversity. However, if we had considered the entire control forest in the samples, probably we would have found a higher alpha diversity within this habitat than in the linear remnants, according to the species-area relation (ROSENZWEIG, 1995). In this way, small fragments will have a subset of the species found in the large fragments and primary forests (LOUZADA et al., 2010; HILL et al., 2011; MENEZES; FERNANDEZ, 2013).

The alpha diversity in the unconnected linear remnants was higher than in connected linear remnants. However, beside this, the unconnected linear remnants harbor more species and a higher abundance of generalists - which are considered non-sensitive to habitat alterations (PARDINI, 2004; UMETSU; PARDINI, 2007; PARDINI et al., 2010) - than the interior of the control forest and the linear remnants connected near (see results).

The interior of the control forest had highest beta diversity, that means that it has a higher small mammal heterogeneous community. The beta diversity decreased significantly from the interior of the control forest and with the isolation increase, however it did not differ between the edges of the control forest and the linear remnants connected near to it. First of all, these results point out that the small mammal community in the linear remnants is influenced by the connectivity to the control forest; i.e., the control forest has a great influence on the small mammal community found in linear remnants connected near it and this can be explained through the spillover effect, as predicted by Brudvig et al. (2009). Second, the lowest beta diversity in the linear remnants unconnected in comparison to the other treatments can be a consequence of isolation, since species turnover is related to the animal displacements (HUBBELL, 1997; CONDIT et al., 2002). Thus, as the unconnected linear remnants are the most isolated areas, the small mammal community is more homogeneous there than in other treatments. Therefore, we can highlight the connectivity importance provided by linear remnants in reducing the isolation (CONDIT et al., 2002; PARDINI et al., 2005; ROCHA; PASSAMANI; LOUZADA, 2011; MESQUITA; PASSAMANI, 2012) and increasing the species turnover in fragmented landscapes.

The connected linear remnants shared the small mammal composition and community structure with the control forest, while the unconnected linear remnants harbor a composition and a community structure significantly different from the control forest. In fact, the small mammal composition and community structure changed with an increase in the distance to the control forest, and were not influenced by the fragment structural features (see results). Once again, these findings show the spillover effect from the control forest influencing the small mammal attributes in connected linear remnants more than in linear remnants far away (see COOK et al., 2002; BRUDVIG et al., 2009). Moreover, this means that connectivity provided by linear remnants has a great importance for native wildlife richness, composition and structure conservation, as observed by other studies (PARDINI et al., 2005; HAWES et al., 2008; MARTENSEN; PIMENTE; METZGER, 2008; BARLOW et al., 2010; ROCHA; PASSAMANI; LOUZADA, 2011; CASTRO and VAN DEN BERG, 2013), reducing the impact of fragment size reductions.

The generalists responded to landscape changes as we expected, showing lowest abundance in the interior of the control forest and in linear remnants connected near, and highest abundance in the other treatments. Also, our results showed that the generalists abundance was not affected by fragment isolation nor fragment size and shape reductions. However we found a strong influence of matrix composition on generalist abundance, which increased in fragments surrounded by a high amount of agricultural areas. These results corroborate with other studies which consider that the small mammal generalist species are not sensitive and can benefit from habitat alteration, such as fragment size reduction, forest loss and introduction of new land-uses in the landscape, like agricultural matrices (PARDINI, 2004; UMETSU; PARDINI, 2007; PARDINI et al., 2010; ESTAVILLO; PARDINI; ROCHA, 2013).

In contrast, the specialists abundance was higher in unconnected linear remnants than in the other treatments, and did not respond neither to fragment structural features nor to landscape connectivity. The specialists are found mainly in mature and control forests (PARDINI et al., 2010; ESTAVILLO;

PARDINI; ROCHA, 2013), being highly sensitive to fragment isolation (VIEIRA et al., 2009), vegetation cover loss and to the conversion of native forests into agricultural areas (UMETSU; PARDINI, 2007; PARDINI et al., 2010, PASSAMANI; FERNANDEZ, 2011). Thus, we believe that the studied landscape, althought fragmented, still retains some connectivity level, which explains the highest specialist abundance in the unconnected linear remnants and the absence of patch size influence on these species, as predicted by the conceptual model proposed for fragmented landscapes (see PARDINI et al., 2010). In fact, the unconnected linear remnants are surrounded by the forest fragments with different sizes (including large fragments) and mainly by matrices composed of *Eucalyptus* spp. plantations. The most of the forest matrices have shown to be the most permeable to animal displacement (UMETSU; PARDINI, 2007; PREVEDELLO; VIEIRA, 2010; WATLING et al., 2011). As such, both the forest fragments and eucalyptus matrices should be working as stepping-stones in this fragmented landscape, increasing the specialist abundance in non-connected habitats.

4.2 Influences of the structural features and spatial arrangement of linear remnants on the small mammal community

Our results showed that shifts in small mammal composition were strongly influenced by the spatial arrangement of linear remnants, changing with the isolation of the linear remnants and with the amount of agricultural areas in the surrounding matrix. Also, the small mammal community structure did not differ significantly from that in linear remnants connected near, and far from, the control forest, however it differed from that in unconnected linear remnants. These results highlight the effect of connectivity provided by connected linear remnants on the native biota community. Also, these results pointed out the importance of considering the isolation distance in vegetation corridor design, as demonstrated by other studies developed on birds (HAWES et al., 2008) and dung beetles (BARLOW et al., 2010).

Differently from the results found for the all the treatments sampled, the small mammal community structure changed in function of linear remnant shape. This structural characteristic determines the amount of area exposed to the edge (HELZER; ELINSKI, 1999) and has shown a great importance in determining the native biota distribution and density in fragmented landscapes (MAGRACH; LARRINAGA; SANTAMARIA, 2011; GARMENDIA et al., 2013), since edge effects have a high influence on forest biodiversity (MURCIA, 1995; STEVENS; HUSBAND, 1998; LIDICKER JR, 1999; LAURANCE et al., 2002; EWERS; DIDHAM, 2008).

We found a great effect of linear remnant features on the habitat preference of specialist species. The highest specialist abundance was observed in the smallest, most irregular and in the widest linear remnants. In fact, the width of linear remnants is related to specific characteristics of each species, this characteristic being mentioned as important for the evaluation of its conservation value (LAURANCE; LAURANCE, 1999; LAURANCE, 2004; LEES; PERES, 2008) since it determines the available habitat area and its vulnerability to edge effects.

The lower abundance of generalist species in linear remnants connected near, in comparison to linear connected far and unconnected linear remnants, showed that these landscape elements are not conducting/guiding generalist species to the control forest to which they are connected, as previously reported (SIMBERLOF; COX, 1987). This result can be enhanced by the fact of the highest generalist abundance was observed in linear remnants far away from the control forest (see results). In fact, recent studies show that connected linear remnants harbor native biota species sensitive to human-disturbance in the landscape (PARDINI et al., 2005; HAWES et al., 2008; LEES; PERES, 2008; BARLOW et al., 2010; ROCHA; PASSAMANI; LOUZADA , 2011; JANTZ; LAPORTE, 2014).

The input of our study, not mentioned in previous studies, was to show that both the connectivity and the spatial arrangement of linear remnants have significant influences on small mammal attributes. Furthermore, species responses to changes in the spatial arrangement of linear remnants varied as a function of the species habitat preferences (generalists and specialists). Therefore, the implantation of linear remnants in fragmented landscapes needs to be carefully designed.

5 CONCLUSIONS AND IMPLICATIONS FOR CONSERVATION

Our results showed the strong influence of the control forest on the connected linear remnants and matrix composition driving the shifts in the small mammal composition and community structure and on the generalists abundance. First, the beta diversity decreased significantly from the interior of the control forest to other treatments, with the increase of isolation. The linear remnants connected near the control forest had a more heterogeneous (higher beta diversity) small mammal community than linear remnants connected far and unconnected linear remnants, which had a lowest beta diversity. Second, the connected linear remnants shared a small mammal composition and structure with the control forest, whilst the small mammal community in unconnected linear remnants differed significantly from the control forest. In fact, the best models showed that the small mammal composition and community structure changed with an increase in the distance to the control forest. In fact a great spillover of biodiversity occur around reserves that harbor a large number of species (the largest reserves), in this case, the connected linear remnants, which increases its importance for conservation (BRUDVIG et al., 2009). Third, we found that the amount of agricultural areas in the matrix had significant effects on small mammal composition and structure in the linear remnants. The generalist abundance increased within linear remnants surrounded by most agricultural matrices and most distant from the control forest. Yet, the community structure and generalist abundance were influenced by the amount of native forest in the surrounding matrix.

These results highlight the linear remnants connected to large forests as important conservation targets in relation to the most isolated patches. In this way, we point out that these structures should be consider in management decisions regarding fragmented landscapes. The isolation distance between the linear remnants and the largest fragment in a landscape is a important variable to enhance the conservation value of linear remnants and should be included in vegetation corridor design. Yet, the surrounded matrix management has to be considered to ensure the conservation of the native biota assemblage, because the matrix composition influences the community attributes, as we verified, and drive different processes in fragmented landscapes (LAURANCE, 1994; GASCON et al., 1999; LAURANCE et al., 2002; UMETSU; PARDINI, 2007; FRANKLIN; LINDENMAYER, 2009; PARDINI et al., 2010; PREVEDELLO; FORERO-MEDINA; VIEIRA, 2010; PREVEDELLO and VIEIRA, 2010; WATLING et al., 2011).

Once again, we can point out the structural connectivity importance, provided by the matrix composition, to explain the highest specialist abundance in the unconnected linear remnants and the absence of patch size influence on these species. Also, our results showed that the structural characteristics of linear remnants, such as shape and width, need to be considered for the implantation or management of linear remnants. These features determine the available habitat area, its vulnerability to edges effects and consequently, its use by species more sensitive to fragmentation effects (LAURANCE; LAURANCE, 1999; LEES; PERES, 2008). Other important structural characteristics which need to be considered to enhance the biodiversity and functional connectivity of linear remnants are the vegetation structure inside them and its length (LAURANCE; LAURANCE, 1999; HAWES et al., 2008; BARLOW et al., 2010).

Finally, we known that primary forests are irreplaceable regarding the maintenance of tropical biodiversity (GIBSON et al., 2011). Also, they are important in fragmented landscapes, as they harbor the highest beta diversity and can have strong influence on the forest remnants around them (as we showed). However, the largest reserves represents less than 1% of Atlantic Forest remnants (RIBEIRO et al., 2009). As such, we need to think of other alternatives for biodiversity conservation and our study shows that if properly managed, the linear remnants can work towards conservation in fragmented landscapes.

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7 SUPPLEMENTARY MATERIAL

7.1 Tables

Table A.1 Mean values (±SD) of independent variables used to characterize the structural and landscape variables of the sampled treatments on a Tablelan Forest in southeastern Brazil. Label: Label: CFi=interior of control forest; CFe=edges of control forest; CRn=linear remnants connected near; CRf=linear remnants connected far; UC=linear remnants unconnected.

	Structural variables			Landscape variables			
Sampling treatments	**Size (ha)**	**Mean Width (m)**	**Shape (m)**	**Amount of agricultural areas (ha)**	**Amount of native forest in the matrix (m)**	**Distance to source fragment (m)**	**Mean distance to neighbors fragments (m)**
CFi	16,48±7,75	-	0.00149± 00007.8	29.4±18.3	1.9±0.9	-	-
CFe	16,48±7,75	-	0.00149± 00007.8	98.6±36.8	21.±37.3	-	-
CRn	22.8±14.27	75±18.58	0.038±0.00608	107.8±51.5	169±65.5	95.7±106.3	767.3±218.54
CRf	22.8±14.27	75±18.58	0.038±0.00608	138±51.6	139.3±74.8	347.9±156.2	815.8±122.17
UC	12.5±7.06	93±50.73	0.0365±0.0169	259.6±10	25.4±14.9	1,277±775	1,45±363.45

Table B.1 Small mammals species captured in each sampled tretatment on a Tableland Forest in southeastern Brazil, and their classification in relation of habitat preference. Label: CFi=interior of control forest; CFe=edge of control forest; CRn=linear remnants connected near; CRf=linear remnants connected far; UC=linear remnants unconnected.

	Sampled treatments					**Habitat preference**	
Species	**CFi**	**CFe**	**CRn**	**CRf**	**UC**	**Specialist**	**Generalist**
Marsupials							
Caluromys philander	1	0	1	3	0	X	
Didelphis aurita	8	24	4	12	11		X
Gracilinanus microtarsus	1	1	4	0	1	X	
Marmosa murina	1	4	4	14	10		X
Marmosa paraguayana	0	6	0	1	0	X	
Marmosops incanus	4	5	4	4	16		X
Metachirus nudicaudatus	0	1	0	0	1	X	
Monodelphis americana	1	1	1	0	0	X	
Rodents							
Nectomys squamipes	0	0	4	6	4		X
Rhipidomys mastacalis	0	0	0	2	0	X	
Trinomys setosus	0	0	0	1	22	X	
Rattus rattus	1	3	1	1	0		X
Total abundance (±SD)	**17 (±1.67)**	**45 (±3.24)**	**23 (±1.81)**	**44 (±3.11)**	**65 (±5.20)**	**7**	**5**

Table C.1 Results from Generalized Linear Models to compare the small mammal composition and community structure (obtained from both NMDS axis) between sampled treatments on a Tableland Forest in southeastern Brazil. Label: CFi=interior of control forest; CFe=edge of control forest; CRn=linear remnants connected near; CRf=linear remnants connected far; UC=linear remnants unconnected. Bold number indicate significat results.

	Species composition				**Community structure**			
	Axis 1		**Axis 2**		**Axis 1**		**Axis 2**	
Treatments	**t-value**	**p value**	**t-value**	**p value**	**t-value**	**p value**	**t-value**	**p value**
CFi x CFe	-0.35	0.74	-1.22	0.25	0.10	0.93	-0.10	0.92
CFi x CRn	-0.85	0.40	0.47	0.64	-0.78	0.44	-0.56	0.58
CFi x CRf	-0.50	0.63	-0.45	0.65	-1.22	0.23	1.36	0.19
CFi x UC	-3.19	**0.004**	0.13	0.90	-2.78	**0.01**	-1.36	0.19
CFe x CRn	-0.63	0.53	1.60	0.12	-0.87	0.39	0.46	0.65
CFe x CRf	-0.27	0.79	0.67	0.51	-1.31	0.20	1.46	0.16
CFe x UC	-2.97	**0.01**	1.25	0.22	-2.87	**0.01**	-1.26	0.22
CRn x CRf	0.56	0.53	-1.01	0.33	-0.48	0.64	1.92	0.07
CRn x UC	-2.33	**0.03**	-0.35	0.73	-2.00	0.06	-0.79	0.43
CRf x UC	-2.69	**0.01**	0.58	0.57	-1.56	0.13	-2.72	**0.01**

Table D.1 Best models results from Generalized linear mixed models showing that fragments size and shape did not influence changes in the small mammal attributes and on the specialist and generalist abundances, on a Tableland Forest in southeastern Brazil. Values inside the brackets show the coefficient estimates and standard errors for each selected model.

Small mammal attributes	**Size**	**Shape**
Species composition (NMDS axis 1)	0.165 (0.121) ^{ns}	- 7.647 (9.624)^{ns}
Species composition (NMDS axis 2)	- 0.110 (0.086) ^{ns}	6.005 (6.962) ^{ns}
Community structure (NMDS axis 1)	0.183 (0.107) ^{ns}	- 12.332 (8.496) ^{ns}
Community structure (NMDS axis 2)	0.023 (0.083) ^{ns}	3.269 (6.528) ^{ns}
Generalists abundance	- 0.052 (0.093) ^{ns}	2.381 (6.969) ^{ns}
Specialists abundance	- 0.238 (0.183) ^{ns}	1.911 (14.421) ^{ns}

ns = not significant results

Table D.2 Best models results from Generalized linear mixed models evaluating the structural connectivity influence on small mammal attributes and on the generalist and specialist abundances, on a Tableland Forest in southeastern Brazil. Values inside the brackets show the coefficient estimates and standard errors for each selected model. All treatments were considered in these analysis.

	Small mammal attributes			
	Species composition (axis 1)	**Community structure (axis 1)**	**Generalists abundance**	**Specialists abundance**
Distance to source fragment	-0.267 (0.117)*	-0.318 (0.108)**	-	0.367 (0.196)^{ns}
Average distance to neighbors fragments	-0.229 (0.116)^{ns}	-	-0.118 (0.096)^{ns}	0.257 (0.194)^{ns}
Amount of agricultural areas	-	-	0.946 (0.333)**	1.03 (0.581)^{ns}
Amount of native forest	-	-	-0.195 (0.168)^{ns}	-

* $p \leq 0.05$, ** $p \leq 0.01$, ns = not significant result.

Table E.1 Results from Generalized Linear Models to compare the abundance of specialist and generalist between sampled treatments on a Tableland Forest in southeastern Brazil. Label: CFi=interior of control forest; CFe=edge of control forest;

CRn=linear remnants connected near; CRf=linear remnants connected far; UC=linear remnants unconnected. Bold number indicate significat results.

	Specialist abundance		**Generalist abundance**	
Treatments	**z-value**	**p value**	**z-value**	**p value**
CFi x CFe	1.45	0.15	3	**0.0027**
CFi x CRn	0.61	0.54	0.39	0.7
CFi x CRf	0.89	0.37	2.46	**0.0138**
CFi x UC	2.56	**0.01**	2.79	**0.0053**
CFe x CRn	-0.75	0.45	-2.14	**0.03**
CFe x CRf	-0.52	0.61	-0.01	0.99
CFe x UC	1.81	**0.07**	0.35	0.73
CRn x CRf	0.28	0.78	2.65	**0.008**
CRn x UC	2.45	**0.01**	2.47	**0.014**
CRf x UC	2.21	**0.03**	0.36	0.72

Table F.1 The best models results from Generalized linear mixed models evaluating the influences of spatial arrangement of linear remnants on small mammal attributes and on the generalist and specialist abundances on a Tableland forest in southeastern Brazil. Values inside the brackets show the coefficient estimates and standard errors for each selected model.

	Small mammal attributes			
Spatial arrangement of remnants	**Species composition (axis 2)**	**Community structure (axis 2)**	**Generalists abundance**	**Specialists abundance**
Distance to source fragment	-	-	0.816 (0.267)**	-
Average distance to neighbors fragments	-2.729 (0.767)**	-	-	4.76 (1.81)**
Amount of agriculture areas	-1.960 (0.562)**	1.488 (0.550)*	2.076 (0.708)**	-
Amount of native forest	-	-0.824 (0.269)*	-0.914 (0.304)**	-

* p≤ 0.05 and ** p≤ 001

ARTIGO 2

Lessons about the management of linear remnants in Tropical Forest landscapes from a functional perspective

ABSTRACT

Besides the importance of studies that have evaluated the fragmentation impacts on the composition and abundances of species in different biological groups, they can bring out misunderstood interpretations about the real effect of forest fragmentation on biodiversity and important information can be lost. Thus, the evaluation of functional diversity and the shifts in species functional traits is a important step to guide biological conservation decisions about landscapes fragmented by humans, although still little understood. Our study was the first that has evaluated the impacts of patch characteristics and structural connectivity from a functional perspective of the small mammal community. Also, we considered the effects of configuration and spatial arrangement of linear remnants on the functional diversity and the species functional traits related to fragmentation sensitivity. Our study was carried out in a Brazilian Atlantic Forest and we sampled five habitats: 1) control forest interior, 2) control forest edge, 3) linear remnants connected to the control forest, near the connection area 4) linear remnants connected to the control forest, far from the connection area and 5) unconnected linear remnants. We sampled at total 15 sites, using a combinated effort of 7,200 trap-nights and 3,600 pitfall-nights. We used mixed models for data analysis, the Akaike Information Criterion (AICc) being used to find the best models. For the functional diversity analysis, we considered three indices: functional richness, functional evenness and functional divergence. For the functional traits analysis, we considered those related to fragmentation sensitivity, such as body and litter size, diet and locomotion. Our results showed that the functional richness and functional evenness not differ significantly from the interior of the control forest to other habitat types. Also, the functional divergence was lower in the control forest interior than edges and did not differ from the other sites. We associated these results to the presence of an exotic species, *Mus musculus*, because within the habitats where this species was most abundant, the functional richness was low and the functional redundancy was high. Thus, this result is the first step to understand the influence of exotic species on the functional diversity in tropical forests. The matrix composition around the fragments (linear or not) had great influence on the species functional traits. In general, most forest matrices harbor species and individuals into functional traits considered as being most sensitive to fragmentation effects, whilst most deforested matrices harbor those less sensitive to the fragmentation effects. Thus, over the long term, it is expected that the replacement of forest matrices by the agricultural, could lead to the loss of functional traits and harbor a more impoverished community from a conservation point of view. In this way, we can conclude that both the structural characteristics and matrix management should be considered in the planning of fragmented landscapes to ensure the conservation of species most sensitive to fragmentation effects. Also, our results highlight the importance of exotic species management in large fragments and in the most fragmented landscapes as well.

Key-words: Exotic species. Small mammal. Community disruption. Homogenization. Matrix. Vegetation corridor. Fragmented landscape. Structural connectivity. Management. Functional diversity.

1 INTRODUCTION

The negative impacts of direct and indirect effects of forest fragmentation are considered as one of the most important threats for biodiversity (TURNER, 1996; LAURANCE, 2001; LAURANCE et al., 2002; FARIGH, 2003; EWER; DIDHAM, 2006; GIBSON et al., 2011). These impacts are driven mostly by the loss of plant cover (see METZGER, 2010; PARDINI et al., 2010; PUTTKER et al., 2011; LIRA et al., 2012; MARTENSEN et al., 2011), forest fragment size reductions (see PÜTZ et al., 2011; GIBSON et al., 2013; MAGNAGO et al., 2014), edge effects (see MURCIA, 1995; OLIVEIRA; GRILLO; TABARELLI, 2004; LAURANCE et al., 2007; EWERS; DIDHAM, 2008; MAGNAGO et al., 2014) and introduction of new land-use forms on the landscape, such as agricultural areas (see FOLEY et al., 2005; LAURANCE, 2008; FLYNN et al., 2009; GARDNER et al., 2009; METZGER, 2010; MORRIS, 2010; LAURANCE; SAYER; CASSMAN, 2014). These habitat modifications contribute to increase hunting pressures (see REDFORD, 1992; CHIARELLO, 1999; PERES, 2000; PERES; PALACIOS, 2007; CANALE et al., 2012) and biological invasion by exotic species (see MCKINNEY, 2006; FERREIRA et al., 2012). Previous studies have reported the influences of these effects, changing the species composition and abundance of serveral biological groups (TURNER, 1996; CHIARELLO, 1999; LAURANCE et al., 2002; LAURANCE; SAYER; CASSMAN, 2014; PARDINI, 2004; FERRAZ et al., 2007; UMETSU; PARDINI, 2007; PARDINI et al., 2010; MARTENSEN et al., 2011; PASSAMANI; FERNANDEZ, 2011).

These effects can be more drastic than those already reported, since fragmentation impacts can promote changes in ecosystems functioning via strong shifts in functional traits and functional diversity inside the habitat remnants (see FLYNN et al., 2009; BARRÁGAN et al., 2011; CADOTTE;

CARSCADDEN; MIROTCHNICK, 2011; MAGNAGO et al., 2014). However, community changes considering the ecosystem functional level was not previously detected or poorly explored by the traditional diversity and composition analysis (PETCHEY; HECTOR; GASTON, 2004; CIANCIARUSO; SILVA; BATALHA, 2009). The initial step has been taken to understand the changes in functional traits composition and loss of important species traits facing intense human fragmented landscapes. Studies that use the functional group of analyses predict which of some species traits are most sensitive to fragmentation effects (TURNER, 1996; VIVEIROS DE CASTRO; FERNANDEZ, 2004; EWERS; DIDHAM, 2006; FORERO-MEDINA et al., 2009a). Others have shown that some functional traits – such as forest dependent species, species with more specialized diet and large body size, shade tolerant species, large trees, species with large seeds and large fruits, zoochoric species - are in general, found in the largest fragments, interior and mostly forested habitats, whilst the others – such as non-forest dependent species, species with less specialized diet and small body size, pioneer species, non-zoochoric species, species with small seeds and fruits - are found in mostly disturbed areas, like smaller forest fragments, habitat edges and agricultural matrices (CHIARELLO, 1999; OLIVEIRA; GRILLO; TABARELLI, 2004; UMETSU; PARDINI, 2007; FLYNN et al., 2009; BARGÁMAN et al., 2011; PARDINI et al., 2010; PREVEDELLO; FORERO-MEDINA; VIEIRA, 2010; PÜTZ et al., 2011; LUCK; CARTER; SMALLBONE, 2013; MAGNAGO et al., 2014). Nevertheless the majority of these studies did not directly evaluate the influence/impacts of disturbance (in our particular case, related to the fragmentation) on the functional traits or functional roles performed by species. Thus, our understanding about it is very limited (FLYNN et al., 2009; GARDNER et al., 2009; CADOTTE; CARSCADDEN; MIROTCHNICK, 2011).

Along these lines, the functional diversity measures proposed by Villéger et al. (2008) takes the multiple functional traits of species and individuals into consideration and in summary, evaluates the variety of functional roles played by the species and assesses the complementarity or redundancy on the functional traits performed by the individuals (see also MASON et al., 2005; CADOTTE; CARSCADDEN; MIROTCHNICK, 2011). Thus, the functional diversity measures (functional richness, functional evenness and functional divergence, see VILLÉGER; MASON; MOUILLOT, 2008) give us more precise information about how the conservation of biological diversity and the maintenance of functional process integrity within communities are being affected by environmental disturbances (VILLÉGER; MASON; MOUILLOT, 2008; CADOTTE; CARSCADDEN; MIROTCHNICK, 2011; MASON and DE BELLO, 2013), such as forest fragmentation and management (MAGNAGO et al., 2014) and other types of human impacts, like land-use alteration (FLYNN et al., 2009; BARGÁMAN et al., 2011; LUCK; CARTER; SMALLBONE, 2013). However, our knowledge about the influences of fragmentation effects (directs or indirect) on functional diversity and on the functional traits replacement in fragmented landscapes is still poorly understood.

Although some studies have shown and discussed the importance of linear remnants (or forest strips, vegetation corridors) to fragmented landscapes (LIMA; GASCON, 1999; HADDAD et al., 2003; PARDINI et al., 2005; HAWES et al., 2008; LEES; PERES, 2008; MARTENSEN; PIMENTEL; METZGER, 2008; BARLOW et al., 2010; ROCHA; PASSAMANI; LOUZADA, 2010; MARTENSEN et al., 2011; MESQUITA; PASSAMANI, 2012; CASTRO; VAN DEN BERG, 2013), none evaluated the functional diversity within them. More importantly, none was perfomed comparing the functional diversity in the control forest and in linear remnants connected or not and evaluate if the configuration and spatial arrangement of these remnants influence the functional diversity and also, the functional traits related to fragmentation sensitivity in fragmented and human disturbance landscapes.

Thus, this is the first study that has evaluated the effects of patch characteristics and structural connectivity from a functional perspective of the small mammal. Also, we evaluated the influence of linear remnants, their configuration and spatial arrangements on functional diversity index and functional traits of small mammals. Our main goal was to guide conservation strategies in fragmented landscapes through evaluation of more precise information about the conservation and maintenance of functional process integrity within communities.

2 MATERIAL AND METHODS

2.1 Study area

Our study was carried out in southeastern Brazil (19 ° 11 '52 "S and 40 ° 5' 29" W - 18 ° 54 '18 "S and 40 ° 5' 19 "W). The study area is located in one of the most important global hotspot (MYERS et al., 2000) in a keystone biodiversity area (PAESE et al., 2010). The landscape studied comprises of a large forest of 46,000 ha belonging to the Companhia Vale S.A., a privately-owned company, and to the federal government (Reserva Biológica de Sooretama) surrounded by a matrix composed mainly of *Eucalyptus* spp., papaya and coffee plantations and pasture (PEIXOTO et al., 2008; ROLIM et al., 2005) and by forest fragments of different sizes, shapes, width and degrees of isolation. This forest is the second largest reserve of Tableland Forest (PEIXOTO et al., 2008; PEIXOTO; SIMONELLI, 2007; SBREK-ARAÚJO; CHIARELLO, 2008) and the one of the largest forest remnant of the Atlantic Forest (RIBEIRO et al., 2009). Furthermore it is considered one of the 14 centers with the highest vegetal diversity in Brazil (PEIXOTO; GENTRY, 1990; PEIXOTO; SILVA, 1997), the second most important area for mammal conservation in the Brazilian Atlantic Forest (GALLETI et al., 2009) and a refuge for threatened bird species (MARSDEN; WHIFFIN; GALETTI, 2001; SRBEK-ARAÚJO; CHIARELLO, 2006) and mammals (CHIARELLO, 1999).

The forest in the region is classified as Lowland Rain Forest (IBGE, 1987) or Tertiary Tableland becaused its occurrence on Cenozoic sediments from the Barreiras group, with altitudes ranging from 28 to 65 m (PEIXOTO et al., 2008). The lowland forest is characterized with trees up to 40 m tall, girths up to 400 cm and a sparse understory, with the dominance of the Fabaceae, Myrtaceae, Rubiaceae, Annonaceae, Sapotaceae and Bignoniaceae tree families (JESUS; ROLIM, 2005; PEIXOTO et al., 2008; PEIXOTO; SIMONELLI, 2007).

2.2 Sampling design

We selected the large forest previously mentionated as control and we sampled the small mammals within five rainforest habitats or treatments: 1) interior of control forest, 2) edge of control forest, 3) unconnected linear forest remnants (termed “unconnected linear remnants”) and 4) linear forest remnants connected to the control forest (termed “connected linear remnants”). We separated the last treatment in two categories according to the distance until the structural connection as i) linear remnants connected near the control forest (placed after the edge), and ii) linear remnants connected far from the control forest, with the sample transect located along connected remnants and with a minimum distance of 400 m until the control forest. We chose these treatments considering the similarity in the composition of the surrounding matrix and a minimum distance of 400 m from the interior to the nearest edge of the control forest (range = 400 to 2,642 m) (see Figure1).

Our study included three replicates per treatment with a mean distance between sites of 6,391m between them and totalizing 15 sampling sites. We established a 100 m transect in each sampling site, composed by six capture stations disposed in 20 m intervals. Each capture station received one large cage trap (45 x 16 x 16 cm) or one large Sherman (45 x 12.5 x 14.5 cm) on the ground, and one small Sherman trap (25 x 8 x 9 cm) in the understory vegetation, at a height of two meters. Traps were baited daily with a mixture of banana, peanut crumbs and sardine (fresh fish). Using the same sample protocol, in each site, we set a 100 m transect sequence of six pitfall traps (plastic buckets of 60 liter volume) connected by plastic fence of one meter on height and without bait. We used both capture methods at the same time and we sampled each site for a total of 40 days between April 2012 and May 2013 providing a combined total sampling effort of 7,200 trap-nights and 3,600 pitfall-nights. We randomized the sites to be sampled for 10 days in each month.

All captured individuals were identified, weighed, measured, sexed, received a unique numbered ear tag (National Band and Tag Inc.), and were released at the same capture station. Voucher specimens of all species were collected and deposited in the mammal collection at the Federal University of Espírito Santo (UFES-MAM). All procedures regarding the capture and marking of animals were conducted under the legal approval and consent of the Brazilian Federal Authority (IBAMA license number 27369-4).

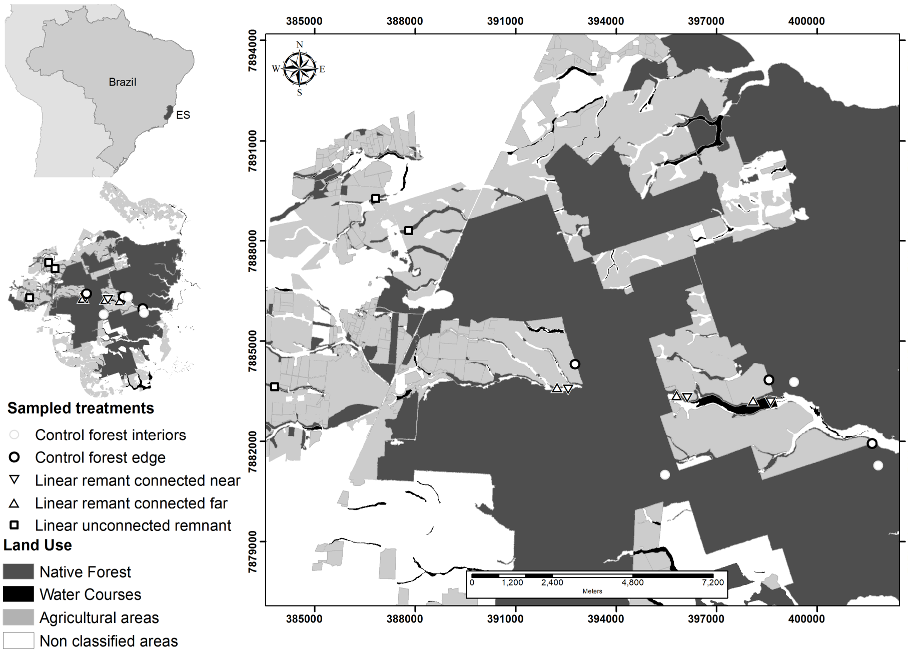

Figure 1 Study area and sampled treatments in a Tableland Forest in southeastern Brazil.

2.3 Land cover analysis and Independent variables

The metrics utilized to characterize the landscape changes and connectivity were extracted from a land cover map produced with an image with a high spatial resolution classification. We used an image with resolution of one meter, acquired in the year 2008. The orthorectified images and with atmospheric correction and visual evaluation of image registration, was obtained through the Vale Natural Reserve.

To classify the land cover we an used image based on multiscale segmentation The segmentation partitioned the image into groups of pixels spectrally similar and spatially adjacent (DESCLÉE; BOGAERT; DEFOURNY, 2006; DUVEILLER et al., 2008), using a "trial-and-error" attempt to find an fragmentation scale appropriate value. Once a successfully segmented image was obtained using 40 as a scale factor, we applied an object-based classification using Nearest Neighborhood (NN). We used 20 trained samples obtained in the field to apply the NN classification algorithm. The result is a class label for each of the segments in each class. A few wrongly-classified image objects were reassigned manually to the correct classes based on field knowledge and on visual interpretation of the image. Classification validation was obtained using 150 independent data sources as reference, randomly distributed over each class. User accuracy, producer accuracy, overall accuracy and kappa coefficient obtained high values, above 85%.

The resulting map was converted to vector format and we computed seven continuous variables using ArcGis (Table A.1, Supplementary material). For each sample treatment, we obtained their structural characteristics, such as size (hectare) and shape, using the ratio between area and perimeter according to Helzer and Jelinski (1999) and the mean width. For the mean width calculation, we obtained three widths for each treatment and considered the average among them. Also, to access the structural connectivity, we constructed a buffer with 2 km around each sampling treatment (total of 15). We quantified the amount of agricultural areas in the buffer (representing by coffee, *Eucalyptus* spp. and papaya plantations), the amount of native forest in the matrix and the amount of native forest of each treatment. Also, we measured the minimum distance between the sample treatment and the nearest source fragment and also, the mean distance to the neighbors nearest fragments. For this, we considered the four fragments nearest the sample treatment. We used these variables to characterize the spatial arrangement of linear remnants, as well. We chose these variables since they are key components to maintain species and ecological processes in fragmented landscapes, are essential to infer about the best spatial arrangement and to the evaluate the configuration of linear remnants for conservation in human-dominated landscape (see LAURANCE, 2004;

PARDINI et al., 2005; EWERS; DIDHAM, 2006; HAWES et al., 2008; LEES; PERES, 2008; MARTENSEN; PIMENTEL; METZGER, 2008; BARLOW et al., 2010; MAGRACH; LARRINAGA; SANTAMARIA, 2011; ROCHA; PASSAMANI; LOUZADA, 2011; MARTENSEN et al., 2012; GARMENDIA et al., 2013).

We used two categorical variables to evaluate the effects of the structural connection among linear remnants and control forest (connected and unconnected), and the distance of the connection to the remnants (connected near and far). Furthermore, we used the width of linear remnants connected and unconnected as structural features. We considered the same variables used for evaluate the structural connectivity (described above) to characterize the spatial arrangement of linear remnants.

Thus, we verified the influence of habitat alteration, habitat features, structural connectivity and the effects of structural features and spatial arrangement of linear remnants on the small mammal functional diversity (represented by functional richness, functional evenness and functional divergence index) and on the functional traits of small mammal community.

2.4 Dependent variables

We classified the captured species into four functional traits: (i) locomotion type, (ii) diet, (iii) body size and (iv) litter size (Table A.1, Suplementary material). We choose these functional traits due to the previous knowledge that these traits describe the species sensitivity to landscape changes and are related to species persistence in fragmented landscapes (DAVIES; MARGULES; LAWRENCE, 2000, HENLE et al., 2004). Arboreal species show the lowest rate of interfragment movements trough open areas or in less forested matrices than terrestrial species (PIRES et al., 2002; LIRA et al., 2007;

PASSAMANI; FERNANDEZ, 2011) and are consequently more dependent on forested areas and more sensitive to forest fragmentation. In the same way, the food supply is determinant for the species distribution, species with a more specialized diet having a more restrict territory (GRINNEL, 1917). The small mammal population size is also limited by the food supply (PREVEDELLO et al., 2013). As fruits are seasonal sources, species which mainly feed on fruit are less likely to persist in more disturbed habitats (see MILTON; MAY, 1976; CHIARELLO et al., 1999). On the other hand, the insectivore species are less vulnerable to the landscape modification, since insects are constantly available. Furthermore, the body size of small mammals is strictly related to their displacement ability between habitat patches, i.e., their perceptual range. Larger small mammal species have more perceptual range than smaller species and cross larger distances between forest fragments, having lower sensitivity to landscape modifications (FORERO-MEDINA; VIEIRA, 2009b; PREVEDELLO; FORERO-MEDINA;VIEIRA, 2010). Moreover, species with larger litter size are expected to maintain higher population sizes and to recover faster from population declines (MCKINNEY; LOCKWOOD, 1999) in human-disturbed areas. Otherwise, species with lower reproductive potential are more dependent on larger habitat amounts for persistence in fragmented landscapes (FAHRIG, 2001).

Below, we show the detailed classification of each functional group:

(1) Locomotion: the captured species were classified in three categories: arboreal, scansorial and terrestrial, according to the Paglia et al. (2012) classification and based on the available global data information for mammals, the PanTheria database (Jones et al., 2009), with some adaptations. For the exotic species *Mus musculus*, this information was unavailable, so we used the Shiels (2010) classification.

(2) Diet: we considered five categories: omnivore, insectivore/omnivore, frugivore/omnivore, frugivore/granivore and frugivore/seed predator. We adopted the classification used by Paglia et al. (2012) and the available information from the PanTheria database (JONES et al., 2009). However, we considered the diet of *Didelphis aurita* as omnivore. For *Mus musculus*, for which did not have available information, we used the Shiels (2010) classification.

(3) Body size: we used three categories: large, medium and small, considering the continuous measures available in the PanTheria database (Jones et al., 2009). For the species without available information we used used either genus or family values. We transformed the continuous variables into categorical variables, considering the body size difference between marsupials and rodents and using the rank adopted by Rossi (2011) (Table B.1).

(4) Litter size: we used continuous values according to the available information from the PanTheria database (JONES et al., 2009). The litter size was given by the number of offspring born per litter per female, either counted before birth, at birth or after birth. For the species without available information we used either genus or family mean values.

2.5 Data analysis

As the sampling effort was equal for all study sites, each sampled treatment was considered as replicate. Thus, we evaluated the data obtained for *live-trap* and pitfall together.

To calculate the three functional diversity indices we used methods and scripts from Villéger et al. (2008), who describe three functional indices: functional richness (FRic), functional evenness (FEve) and functional divergence (FDiv), according with Villéger et al. (2008). The first one 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. We got these values and we access if the small mammal functional diversity were affected by the fragment size, shape and structural connectivity. For this analysis, we considered all treatments together and constructed mixed models in R version 2.15.1 (R Development Core Team 2012). Secondarily, we constructed the models in the same package, however considering just the connected/unconnected linear remnants in the analysis. Thus, we evaluated the influence of structural features and spatial arrangement of the linear remnants on small mammal functional diversity and to each functional trait. Within these models, we used an interaction with three levels (connected near, connected far and unconnected) to check the effects of linear remnants connection (connected and unconnected) on functional diversity and to assess the influences of the distance effect within connected remnants (connected near and far) between these levels separately. We constructed the mixed models using lmer function from the lme4 package with Gaussian family, once all data were uncontable. The same framework was used to evaluate the richness by functional trait.

For the analysis with countable data (abundance by functional trait) we constructed the mixed models using *glmmadmb* function from the *glmmADMB* package with Poisson family. When these data showed overdispersion, we used the Negative Binomial family. We used the *dredges* function from the *MuMIn* package to test all possible combinations of variables included in the global model. However, to avoid multicollinearity between explanatory variables we not include in the same model the variables with autocorrelations (linear Pearson correlations large or equal to 0.6).

The sites (each treatment) were codified as a random variable in all analyses (BOLKER et al., 2009). To select the best model for both analyses (with all treatments and for linear remnants only) we used a theoretical information approach based on the Akaike Information Criterion of Second Order, which is indicated for small sample sizes (AICc) and chose the models according to the lowest AICc value (BURNHAM ; ANDERSON; HUYVAERT, 2011). The plausibility of alternative models was given by the differences in their AICc values in relation to the AICc of the most plausible model (ΔAICc). We considered as plausible models those with a value of $\Delta AICc<2$. However we only considered the variables in the models as an important to induce the changes in functional diversity and traits when the value of $\Delta AICc<2$ and when the variables included in the model was significant, considering $p\leq 0.05$. These analyses were performed in the R version 2.15.1 (R Development Core Team 2012).

3 RESULTS

3.1 Small mammal community and proliferation of an exotic species

In all, we captured 211 small mammal individuals from 16 species, including eight marsupials and eight rodents, and, two exotic species among them (Table B.1, Suplementary material).

Surprisingly the most abundant species, the exotic mouse *M.musculus*, also was the most abundant species in the interior of the control forest, accounting for more than half of the captured individuals (54.3%), and in linear remnants connected near (31.6% of captured individuals). Moreover, the abundance of this species was higher within the control forest and in linear connected near than in unconnected linear remnants (Figure A.1, Supplementary material). The interior of the control forest differs from linear connected far as well. Our best models confirmed these results (Figure A.2, Suplementary material), showing that the *M.musculus* abundance decreases with the distance to source fragments (GLMM; z= -2.16, p=0.024) and with fragments size (GLMM; z=0.247, p=0.024).

3.2 Do the habitat changes, forest fragments size reduction and structural connectivity impact on small mammal functional diversity and functional traits?

The model results showed that the functional richness did not differ significantly between the interior of the control forest and the other treatments (Figure 2). Also, our best model selected indicated no significant impact of fragment size reduction on functional richness (Table D.1, Suplementary material). Otherwise, the structural connectivity, given by the matrix features and distance to the source fragment, had significant effects on this index (Table D.1, Supplementary material; Figure 3). The functional richness increased in forest fragments with the lowest amount of native forest, surrounded by most agricultural matrices and apart from the control forest (Figure 3).

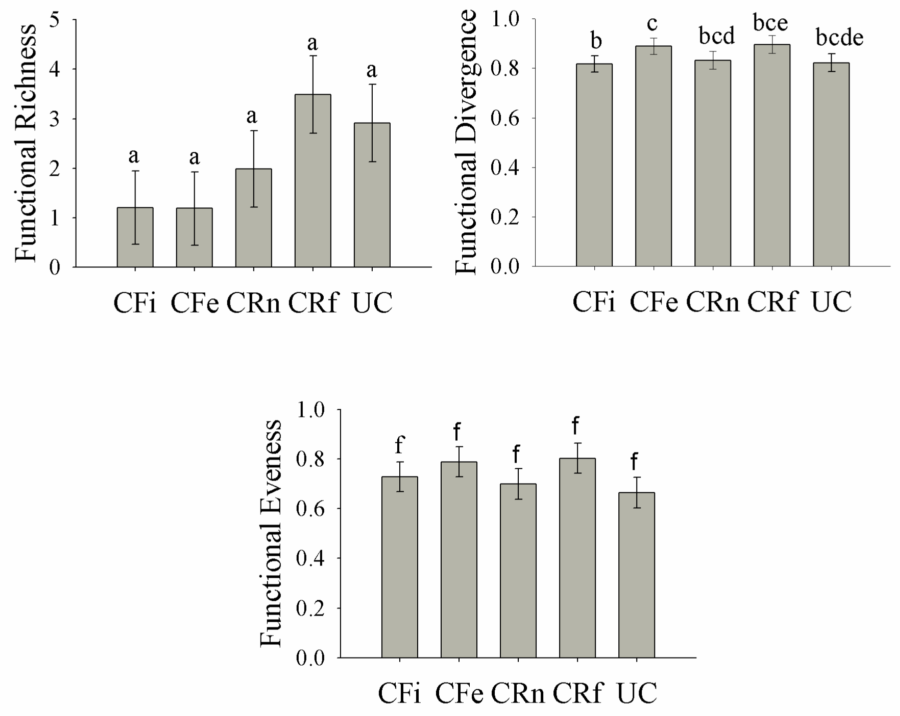

Figure 2 The graphs showing the influence of habitat alteration on small mammal functional divergence and no effects on the functional richness and functional evenness on a Tableland Forest in southeastern Brazil. Label: CFi=interior of control forest; CFe=edge of control forest; CRn=linear remnants connected near; CRf=linear remnants connected far and UC=linear remnants unconnected. Different letter denote significant differences at p≤0.02.

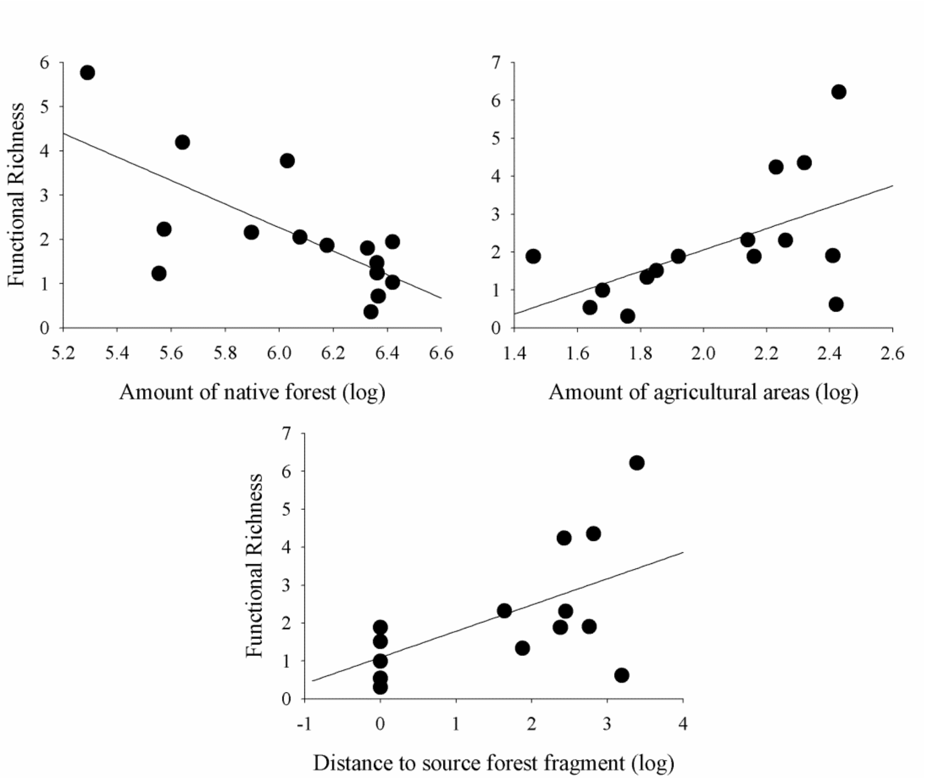

Figure 3 Best models results from Generalized Linear Mixed Models showing that the functional richness decrease with the amount of native forest in the fragments and in opposite way, increase in fragments surrounded by most agricultural matrices and in treatments far away from control forest on a Tableland Forest in southeastern Brazil. All the results were significant at $p \leq 0.02$.

Surprisingly the functional divergence was significantly lower within the interior than the edge of the control forest and in the linear remnants connected to the control forest, compared with the remnants connected far, and did not have significant differences with the other treatments (Figure 2). Furthermore, the functional evenness did not differ significantly from interior of control forest and the other habitat types (Figure 2). Both the fragment size and structural connectivity had no significant impacts on the functional diversity and functional evenness (Table D.1, Suplementary material).

The richness of omnivore species was significantly higher in the interior of the control forest than in unconnected linear remnants. The richness of large bodied species was higher in linear connected far than in near (Figure B.1, Suplementary material). For other functional attributes considering diet, locomotion and body size, the species richness did not differ significantly between the sampled treatments (no significant results).

The functional trait richness was not significantly influenced by fragment size and shape (Table D.2, Suplementary material). On the contrary, the structural connectivity had significant effects on this attribute attribute (Figure B.1, Table C.2, Supplementary material). The decrease of the amount of native forest fragments had a significant and positive effect on the insectivore/omnivore richness, large bodied species and terrestrial species. The terrestrial species richness was negatively influenced by the proximity to source fragments. Further, the amount of agricultural areas in the matrix surrounding the fragments negatively affected the arboreal species and positively affected the species with insectivore/omnivore diet. Otherwise, the amount of native forest in the matrix had a positive influence on the arboreal species and negative influence on the terrestrial species in the sampled treatments.

When we considered the small mammal abundance by functional trait, the model results showed that the omnivore abundance did not differ among the interior, control forest edge and linear remnants connected near. However, the omnivore abundance was higher in the control forest than in the linear remnants connected far and those unconnected (Figure C.1, Suplementary material). Furthermore the interior of the control forest had lower scansorial abundance than edges and unconnected remnants. Also, the interior harbors a higher abundance of terrestrial individuals than edges and linear connected far (Figure C.1, Suplementary material). The individuals in other functional traits did not differ significantly between the sampled treatments.

The fragments size had significant effect only on the omnivore abundance, which increased positively with fragment size, as shown by the best model selected (Table D.3 and Figure C.2, Suplementary material). Furthermore, the omnivore abundance and individuals with higher litter size were most abundant in fragments closest to the source fragments. On the other hand, the terrestrial abundance was highest in fragments farther away from control forest (Figure C.2, Suplementary material). Moreover, the omnivore abundance was highest in fragments closer to each other. The increase of native forest in the surrounding matrix positively influenced the arboreal abundance and negatively influenced the terrestrial abundance abundance (Figure C.2, Supplementary material). Furthermore, fragments with more agricultural areas around them harbor more insectivore/omnivore individuals (Figure C.2, Supplementary material). Yet, the insectivore omnivore and scansorial abundances were positively influenced by the reduction in native forest amount (Figure C.2, Suplementary material).

3.3 Is the functional diversity and small mammal functional traits influenced by the structural connection, structural features and spatial arrangement of the linear remnants?

The functional richness and functional divergence were significantly lower in linear remnants connected near than in linear connected far (Figure 4). However, both indices did not differ in these treatments in comparison with the unconnected linear remnants. The functional evenness did not differ significantly between the linear remnants (Figure 4).

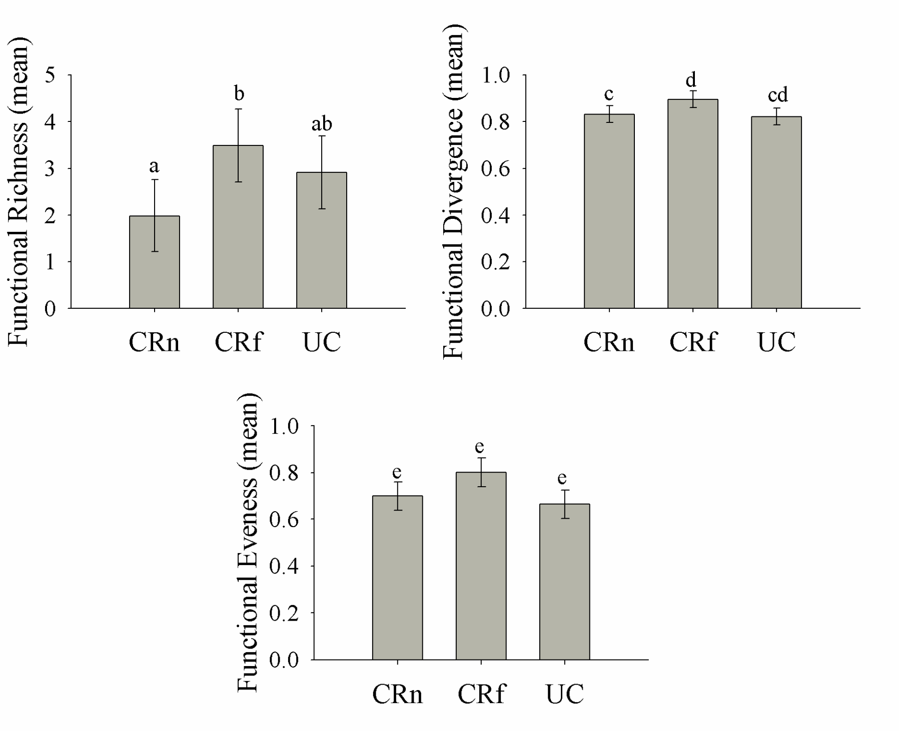

Figure 4 The graphs showing the influence of structural connection on the small mammal functional richness and functional divergence. Otherwise, no significant effect was verified on the functional evenness on a Tableland Forest in southeastern Brazil. Label: CRn=linear remnants connected near; CRf=linear remnants connected far and UC=linear remnants unconnected. Different letter in each graph denote significant differences at p≤0.04.

Our best model showed that the structural features of linear remnants had significant effects on the small mammal functional diversity (Figure 5). The functional richness increased with the increase of linear remnants width and in large linear remnants far. On the other hand, this index decreased with the size increase of unconnected linear remnants. The functional divergence was highest on the wide remnants connected far and the functional evenness rose with the increase of linear remnants size. Conversely, no effect of the spatial arrangement of linear remnants influenced the small mammal functional diversity.

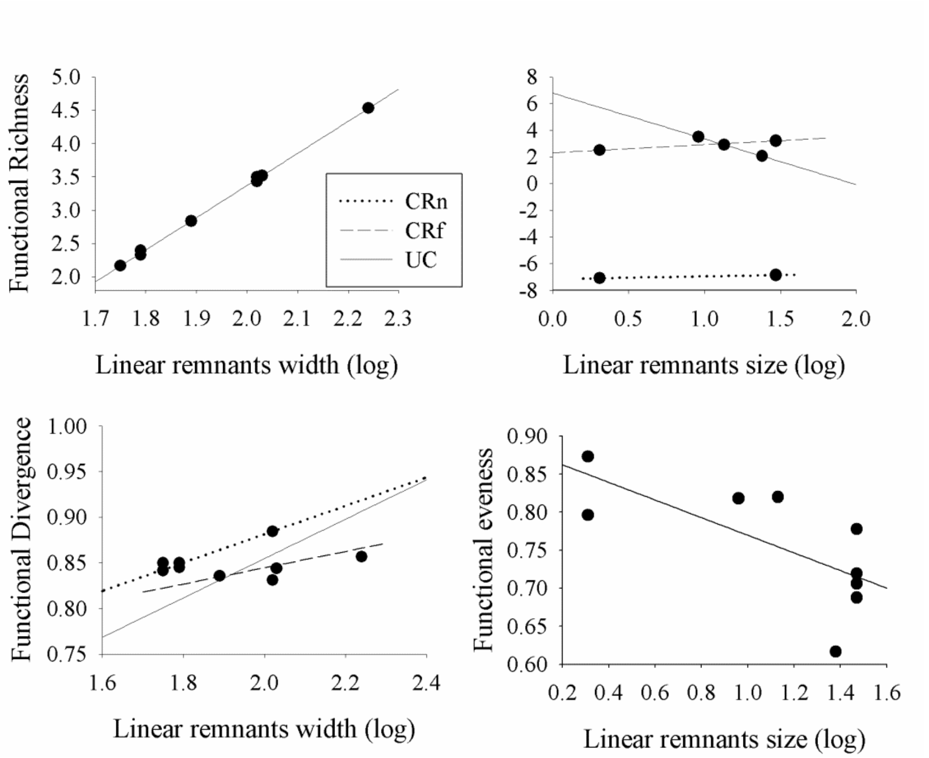

Figure 5 Best models results from Generalized Linear Mixed Models showing that the influence of structural features of linear remnants on small mammal functional richness and functional evenness on a Tableland Forest in southeastern Brazil. Label: CRn=linear remnants connected near; CRf=linear remnants connected far and UC=linear remnants unconnected. All the results were significant at p≤0.02.

The functional traits richness was significantly influenced by the linear remnant features (Figure E.1 and E.2, Table D.1, Supplementary material), unlike results found for the fragment structural features (previously described). The linear remnant width significantly and positively influenced the richness of insectivore/onmivore and arboreal species and negatively the frugivore/omnivore richness. Moreover, our best models showed that the interaction between width and connection had significant influences on these functional traits. Thus, the richness of insectivore/onmivore and small body size species was highest in the widest linear connected far. The arboreal species richness increased in linear remnants connected far and in the widest unconnected remnants, whilst the frugivore/omnivore richness was lower. The scansorial richness was lowest in the connected far with more width as well. Like the width, the remnants size had significant influences on small mammal functional traits (Figure D.1, Table E.1, Supplementary material). The increase of the linear remnant size influenced positively the richness of arboreal, small-bodied and omnivore species. Furthermore, the interaction between linear remnant size and connection had influence on the large-bodied species richness, which was highest in larger linear remnants connected far and lowest within the unconnected remnants.

In contrast to the strong influence of the structural characteristics of linear remnants, the composition of the matrix surrounding the linear remnants and the distance to other fragments influenced just the functional traits related to species locomotion and the richness of frugivore/granivores (Table E.1 and Figure D.3, Suplementary material). Thus the arboreal richness increased in linear remnants most close to the source fragments. The terrestrial richness was highest in linear remnants with the lowest amount of native forest and surrounded by a matrix with less amount of native forest. Also the structural connection had significant influence on the frugivore/granivore richness (Figure D.3, Suplementary material), which was higher in the unconnected linear remnants than in those connected far. The other functional traits did not have significant differences between the sampled treatments and were not influenced by the landscape effects (Table E.1, Suplementary material).

Conversely, the spatial arrangement of linear remnants influenced the abundance of all functional traits analyzed (Table E.2 and Figure E.1, Supplementary material). The abundance of arboreal individuals increased in linear remnants with the highest amount of native forest, surrounded by matrices with low amount of agricultural areas and high amount of native forest areas. Otherwise, the most deforested linear remnants with lowest amount of surrounding native forest harbor more terrestrial individuals and unexpectedly, more scansorial individuals. Also, the scansorial abundance increased in linear remnants far away from the control forest, whilst the abundance of individuals with large litter size rose in linear remnants closest to the control forest. The abundance of individuals with medium body size were lowest in linear remnants surrounded by most agricultural matrices. Otherwise and surprisingly, the abundance of individuals with frugivore/gramivore diet were highest in linear remnants surrounded by most agricultural matrices. The large-bodied individuals were most abundant in linear remnants more isolated from the neighboring fragments. The other attributes were not influenced by the linear remnant spatial arrangement.

The size of linear remnants had strong influence on the abundance of functional traits and was present in many of the best models selected (Table E.2 and Figure E.2, Supplementary material). The linear remnants size decrease positively affected the arboreal abundance and the individuals with medium body size and negatively affected the terrestrial abundance. Also, the individuals with small litter size were most abundant in the unconnected large remnants and within wide remnants. Furthermore, the arboreal abundance was highest in the linear remnants with more irregular shape.

4 DISCUSSION

4.1 The proliferation of an exotic species and its influence on the small mammal functional diversity and functional traits

It is already known that the availability of primary energy in the forest environments is largely dependent on plant productivity (EVANS et al., 2005). Also, areas with high productivity have more resource abundance and this influences the niche overlap, because different species can explore different resources, decreasing the niche overlap and breadth and consequently increasing the species co-existence (HUTCHINSON, 1957; MACARTHUR; LEVINS, 1967; EVANS et al., 2005). As the functional richness is strongly linked with the amount of available niche (MASON et al., 2005; SCHEUTER et al., 2010), we expected that in the interior of the control forest, which showed the highest productivity (see results from Chapter 3), species were playing more distinct roles than in other treatments (meaning high functional richness; see MASON et al., 2005; VILLÉGER; MASON; MOUILLOT, 2008; SCHEUTER et al., 2010). Furthermore, areas with higher productivity and resource abundance (and consequently with less restrictive conditions) are expected to support more species and individuals co-existing (HUTCHINSON, 1957; PAGLIA; FERNANDEZ; DE MARCO, 2006) and having more distinct functions, playing different ecological roles (meaning high functional evenness; see WELLNITZ; POFF, 2001; VILLÉGER; MASON; MOUILLOT, 2008; CADOTTE; CARSCADDEN; MIROTCHNICK, 2011). The functional divergence concept includes the overlap on the functionality of the most abundant species (see VILLÉGER; MASON; MOUILLOT, 2008). Thus, in areas where the most abundant species have similar ecological functions, the functional divergence is lower (see VILLÉGER; MASON; MOUILLOT, 2008) and consequently, the niche differentiation is smaller and the resource competition is higher (MASON et al., 2005; SCHEUTER et al., 2010). Moreover, these three indices are dependent on landscape context, becoming reduced or changing in more disturbed areas with more land use intensification, for the most diverse groups (FLYNN et al., 2009; VILLÉGER et al., 2010; BARRAGÁN et al., 2011; CADOTTE; CARSCADDEN; MIROTCHNICK, 2011; EDWARDS et al., 2013; LUCK; CARTER; SMALLBONE, 2013; MASON AND DE BELLO, 2013; MAGNAGO et al., 2014).

As the interior of primary forests is more productive (see resuls Chapter 3), are the most intact areas in our landscape context (large control forest) and as we know that the primary forest are the most important areas to maintain the tropical biodiversity (GIBSON et al., 2011), we expected higher functional diversity in this area (i.e., highest functional richness, functional evenness and functional divergence, see VILLÉGER; MASON; MOUILLOT, 2008) than in the other sampled habitat types. However, contrary to the expected results, we found that the functional richness and functional evenness did not differ significantly in the interior of the control forest and the other habitat types. Most important and surprising, the functional richness increased in fragments far away from the control forest, with the lowest amount of native forest and surrounded by the most agricultural matrices and was not influenced by the fragment size (see results). Furthermore, the functional divergence was significantly lower within the interior than the control forest edges and lower in the linear remnants connected near to this control forest compared with the linear remnants connected far from it.

Our results showed that the exotic mouse *M.musculus* was the most abundant species in the interior of the control forest (accounting for more than half of the captured individuals, 54.3%) and in the linear connected near (31.6% of captured individuals). Also, the abundance of this species was higher in these habitats than in unconnected linear remnants and in fragments closer to the control forest and with largeer sizes (see results). Thus, considering that the functional diversity index is directly influenced by the presence of exotic species in the natural ecosystems (MASON et al., 2005) which provide niche homogenization and biodiversity loss (MCKINNEY; LOCKWOOD, 1999; CUTHBERT; HILTON, 2004; OLDEN et al., 2004; GALETTI et al., 2009; MORRIS, 2010; FERREIRA et al., 2012; GIBSON et al., 2013), our results show that the high abundance of *M. musculus* could explain the high functional redundancy (low functional evenness and functional divergence) within the interior of the control forest, due to the high abundance of this species which plays the same functional role.

Moreover, the lower functional divergence in the interior of the control forest and in linear connected near than in edges of the control forest and linear connected far, indicate that the resource competition is higher within of interior and linear connected near than in the other habitats (see MASON et al., 2005; CADOTTE; CARSCADDEN; MIROTCHNICK, 2011). Also, it can be related to the higher abundance of *M.musculus*. In this way, we highlight the negative impact of an exotic species on functional diversity and show that even the largest and most intact forest can be impacted by the exotic species. Thus, we found a different result about the exotic species abundances in tropical forest areas. In general, exotic species seem to be more abundant in the smallest fragments and in the agricultural matrices or other disturbed areas and in most cases, not invading the largest and intact fragments (FELICIANO et al., 2002; MCKINNEY et al., 2006; UMETSU; PARDINI, 2007; GIBSON et al., 2013).

Also, besides functional richness not differing significantly among the habitat types, we found significantly more omnivore species in the interior of the control forest than in unconnected linear remnants, while frugivore/granivore species were captured just in the linear connected far and in the unconnected remnants. Once more, these results showed the opposite from what we expected. The food supply is determinant for species distribution, that species with more specialized diet having more restricted territory (GRINNEL, 1917). Also, the small mammal populations have highest densities in areas with the most food supplementation (PREVEDELLO et al., 2013). In this way, omnivore species, which are generalists in the use of food resource, are considered less sensitive to fragmentation effects, such as habitat loss and shifts in the available food resources. Otherwise, frugivore/granivore species are the most sensitive to these effects, because they are dependent on specific resources, being most common in medium and large fragments than in small ones (see MAYSON; MAY, 1976; CHIARELLO, 1999; RIBON; SIMON; MATTOS, 2003). In this way, beyond functional diversity loss, the species functional traits are reversed within the interior of the control forest and in the unconnected linear remnants.

Although the functional evenness did not differ significantly between the habitat types, our results showed that most individuals with omnivore diet occur in the control forest (both, edges and interior habitats) and in linear remnants connected near. Moreover, the omnivore abundance was higher in the control forest than in linear connected far and unconnected remnants. Contrary to what we thought, more of the individuals with scansorial locomotion were captured in the unconnected linear remnants than in the interior of the control forest. However, more of the terrestrial individuals were sampled in the interior than within edges and linear connected far. Nevertheless we expected that the interior of the control forest would be the habitat that harbored the most individuals sensitive to fragmentation effects (see metodology, Page 75), whereas the unconnected linear remnants would have a small mammal community with more individuals which have low fragmentation sensitivity, like most omnivores, terrestrials, large- bodied and individuals with a large litter size (see traits description, Page 75). Also, the lowest functional divergence, shown in the interior of the control forest and in linear remnants connected near, pointed to the highest functional redundancy of one functional trait over the others (see VILLÉGER; MASON; MOUILLOT, 2008; CADOTTE; CARSCADDEN; MIROTCHNICK, 2011). This result can be explained by the high abundance of omnivore individuals, in this case, by the high abundance of *M. musculus* within these habitats, causing a strong impact on the fragment function, via homogenization and simplification of the functional roles played by the community (MCKINNEY; LOCKWOOD, 1999; OLDEN et al., 2004).

Moreover, as we expected, changes in the matrix composition surrounding the fragments had significant effects on richness and abundance of functional traits. These results enhance our knowledge about the strong influence of the matrix composition in the fragmented landscapes (see LAURANCE, 1994; GASCON et al., 1999; LAURANCE et al., 2007; UMETSU; PARDINI, 2007; LAURANCE, 2008; FRANKLIN; LINDENMAYER, 2009; METZGER, 2010; WATLING et al., 2011; TSCHARNTKE et al., 2012). Also, our results showed that the loss of native forest in the matrix negatively affected the richness and abundance of arboreal species. Otherwise, fragments surrounded by more agricultural areas had the least arboreal species and more insectivore/omnivore species and individuals (see results). Thus, these results show that alterations in the matrix composition has an influence on the species and individual functional traits of the small mammal community. Most importantly our results show that more deforested matrices drive the replacement of functional traits and harbor species and individuals with little sensitivity to the fragmentation effects.

4.2 Are the small mammal functional diversity and functional traits influenced by the structural connection, structural features and spatial arrangement of linear remnants?

We found that the functional richness and functional divergence was lower in linear remnants connected near than in linear connected far. However the functional evenness did not differ significantly among the linear remnants (see results). Some studies have shown that the source fragment influences the species composition in the surrounding forest fragments (LOUZADA et al., 2010; HILL et al., 2011; MENEZES; FERNANDEZ, 2013). However, the stronger influences occur in habitats closest to the source fragment more than in remnants far away (see results from Chapter 1; RICKETTS et al., 2001; COOK et al., 2002; RICKETTS, 2004; BRUDVIG et al., 2009). This can be explained through the spillover effect, which can be understood as the movement of organisms from one habitat to another distinct habitat type (TSCHARNTKE et al., 2012). In fragmented landscapes, the high spillover effect happens from the source fragment (or in the large ones) to their boundaries (BRUDVIG et al., 2009; TSCHARNTKE et al., 2012; ESTAVILLO; PARDINI; ROCHA, 2013). Therefore, large the control forest has strong influences on the sorrounding environments, in our case, in the linear forest remnants or not. In this way, we believe that our results could be the consequence of the proximity of linear remnants to the control forest.

Furthermore, our results showed that the distance until the connection with the fragment source (near or far), influences a variety of functional traits of species and the ecological roles played by the species in the linear remnants. Moreover, the functional richness and species traits changed when we considered both the linear remnant structural characteristics (size and width) and the structural connection (i.e. connected or unconnected). For example, the functional richness increased in largest linear remnants far and to the contrary, in unconnected linear remnants with smaller size (see results). Therefore these results shows that we need to consider the connection distances from linear remnants until the fragment source and also, the presence of structural connection (connected or unconnected linear remnants) in future studies development within fragmented landscapes.

According to niche theory, larger areas show more niche space than the small ones (MacArthur, 1972). This assumption implies in more species and individuals occupying different niches and playing distinct ecological functions in the environment (that means higher functional richness, functional evenness and functional divergence, see MASON et al., 2005; VILLÉGER; MASON; MOUILLOT, 2008). In this way, as we expected, the functional diversity (functional richness, functional divergence and functional evenness) was positively influenced by the high amount of available habitat (larger and wider linear remnants).

Although the surrounding matrix composition of the linear remnants showed no significant influences on functional diversity, we found strong matrix influences on the functional trait richness and mostly, on the functional trait abundances (see results). Arboreal species, which have strong dependence of forest habitats and low interfragment movement rates through open areas or in less forested matrices (PIRES et al., 2002; LIRA et al., 2007;) increased in linear remnants closest to the control forest. Also, the abundance of arboreal individuals was highest in linear remnants with more native forest and surrounded by most forest matrices, decreasing in linear remnants surrounded by non-forest matrices (see results). The scansorial individuals, which use both the terrestrial and arboreal strata in the forest, were positively influenced by the increase of native forest and by the increase of forest in the matrices. However, some ecological groups, with low sensitivity to fragmented effects like terrestrial and large-bodied individuals (see PIRES et al., 2002; LIRA et al., 2007; PASSAMANI; FERNANDEZ 2011; FORERO-MEDINA; VIEIRA, 2009; PREVEDELLO; FORERO-MEDINA; VIEIRA, 2010), were positively influenced by the loss of native forest and by the isolation of linear remnants as well (see results). In this way, as discussed previously, our results showed the strong influence of the surrounding matrix composition to drive shifts in functional traits found in forest remnants (linear or not) in fragmented landscapes.

5 CONCLUSIONS AND IMPLICATIONS FOR CONSERVATION

Unexpectedly, we found that the functional richness and functional evenness did not differ significantly from the interior of the control forest to the other habitat types. Also, the functional divergence was lower in the interior of the control forest than edge and did not differ from the other treatments as well. Furthermore, our results indicate, at the first time, that the exotic species can have strong influence on functional diversity (functional richness, functional evenness and functional divergence) and on the functional traits found in the interior of large forests. Within the habitat types where *M.musculus* was most abundant, the variety of functional traits displayed by the species was lowest and the functional redundancy was highest. In this way, our results indicate that even the most large and intact forest can be under these negative effects provided by exotic species more than small remnants.

Our results were the first step for understanding the influences of exotic species on the functional diversity in tropical forests placed in fragmented landscapes. Thus, future studies should directly evaluate this relation to aid the management decisions for biodiversity conservation. This is true for the sampled control forest. Our study area, plus the Sooretama Biological Reserve forms an area with more than 40,000 ha, and one of the last large Atlantic Forest fragments in Brazil, which is composed mainly by isolated fragments smaller than 250 ha (RIBEIRO et al., 2009). Furthermore, this forest is very important for biodiversity conservation, from plant diversity (PEIXOTO; GENTRY, 1990;

PEIXOTO; SILVA, 1997) to the medium and large vertebrates, like birds and mammals, harboring large frugivores and top-predators (CHIARELLO, 1999; MARSDEN; WHIFFIN; GALETTI, 2001; SRBEK-ARAÚJO; CHIARELLO, 2006; GALETTI et al., 2009).This control forest harbors arboreal species which promote food resources, such as medium and large fruits and large seeds, for large frugivores (MAGNAGO et al., 2014). These findings indicate that strong impacts on this control forest can drive the loss of important biological and ecological functioning.

Moreover, our results showed that the structural characteristics of linear remnants and the structural connection (connected or not) have significant influences on the functional diversity within the linear remnants and on the functional traits of species and individuals. Also, we found that larger and wider linear remnants had more species playing different ecological roles (high functional richness) and low functional redundancy. These findings highlight the importance of these structural features to maintain high functional diversity in fragmented landscapes.

We also verified strong influences of the matrix composition on the functional traits of species and abundances. In general, more forested matrices harbor more species and individuals with functional traits considered most sensitive to the fragmentation effects. Otherwise, more deforested matrices harbor species and individuals with functional traits considered less sensitive to the fragmentation effects. Thus, over the long term, it is expected that the replacement of forest matrices by the agricultural matrices could drive the loss of functional groups and harbor a more impoverished community from a conservation point of view.

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7 SUPPLEMENTARY MATERIAL

7.1 Tables

Table A.1 Mean values (±SD) of independent variables used to characterize the structural and landscape variables of the sampled treatments on a Tableland Forest in southeastern Brazil. Label: interior of control forest (CFi), edge of control forest (CFe), linear remnants connected near (CRn), linear remnants connected far (CRf) and linear remnants unconnected (UC).

	Structural variables			**Landscape variables**			
Sampling treatments	**Size (ha)**	**Mean Width (m)**	**Shape (m)**	**Amount of agricultural areas (ha)**	**Amount of native forest in the matrix (m)**	**Distance to source fragment (m)**	**Mean distance to neighbors fragments (m)**
CFi	16,48±7,75	-	0.00149± 00007.8	29.4±18.3	1.9±0.9	-	-
CFe	16,48±7,75	-	0.00149± 00007.8	98.6±36.8	21.±37.3	-	-
CRn	22.8±14.27	75±18.58	0.038±0.00608	107.8±51.5	169±65.5	95.7±106.3	767.3±218.54
CRf	22.8±14.27	75±18.58	0.038±0.00608	138±51.6	139.3±74.8	347.9±156.2	815.8±122.17
UC	12.5±7.06	93±50.73	0.0365±0.0169	259.6±10	25.4±14.9	1,277±775	1,45±363.45

Table B.1 Description of categories used for small mammal classification into body size functional trait, as proposed by Rossi (2011).

Order	**Categories**	**Body size (gram)**
Marsupialia	Small	average weight untill 100g
	Medium	average weight between 100 and 500g
	Large	average weight larger than 500g
Rodentia	Small	average weight untill 50g
	Medium	average weight between 50 and 100g
	Large	average weight larger than 100g

Table C.1 List of small mammal species captured on a Tableland Forest in southeastern Brazil, with the abundance in each treatment and its classification into functional trait, considering the diet, locomotion, body size and the litter size (represented by the number of offspring). Label: interior of control forest (CFi), edge of control forest (CFe), linear remnants connected near (CRn), linear remnants connected far (CRf) and linear remnants unconnected (UC). Diet categories: Ins/Omn=insectivore/omnivore; Fr/Gra=frugivore/granivore; Fr/Se=furgivore/seed predator; Fr/Omn=frugivore/omnivore and Omn=omnivore. Locomotion categories: Ar=arboreal; Sc=scansorial; Te=terrestrial. Body size categories: S=small bodied; M=medium; L=large.

Species	**Sampled treatments**					**Functional traits**	
	CFi	**CFb**	**CRn**	**CRf**	**UC**	**Diet; Locomotion; Body size**	**Litter size**
Didelphidae							
Didelphis aurita	7	17	5	6	5	Omn; Sc; L	6.11
Marmosops incanus	2	4	3	2	15	Ins/Omn; Sc; S	4.76
Marmosa murina	0	2	3	12	3	Ins/Omn; Ar; S	8.4
Monodelphis americana	8	2	4	3	2	Ins/Omn; Te; S	8.84
Gracilinanus microtarsus	0	0	5	0	1	Ins/Omn; Sc; S	8.99
Caluromys philander	1	1	1	1	0	Fr/Omn; Ar; M	4.18
Marmosa paraguayana	0	2	0	0	0	Ins/Omn; Sc; L	5.56
Metachirus nudicaudatus	0	1	0	0	1	Ins/Omn; Te; M	3.87
Rodents							
Trinomys setosus	0	0	0	1	17	Frug/Gra; Te; L	1.83

“Table C.1, conclusion”

Nectomys squamipes	1	0	3	4	3	Frug/Omn; Te; L	4.5
Necromys lasiurus	0	0	1	0	3	Omn; Te; S	4.5
Blarinomys breviceps	1	0	1	0	1	Ins/Omn; Te; S	1.21
Rhipidomys mastacalis	0	0	0	2	0	Fr/Se; Ar; M	3.8
Akodon cursor	0	0	0	1	0	Ins/Omn; Te; S	4.2
Mus musculus	25	9	12	4	1	Omn; Te; S	5.54
Rattus rattus	1	0	0	1	0	Omn; Sc; L	5.88
Total	46 (15.33±7.57)	38 (12.66±4.51)	38 (12.66±3.21)	37 (12.33±6.66)	52 (17.33±5.86)		

Table D.1 Best models results from Generalized Linear Mixed Models to verify if the small mammal functional diversity index are influenced by the structural features of treatments and/or structural connectivity on a Tableland Forest in southeastern Brazil. Values inside the brackets shows coefficient estimates and standard errors, for each model.

	Structural features		**Structural connectivity**				
Functional Diversity	**Size**	**Shape**	**Distance to source fragment**	**Mean distance to neighbors fragments**	**Amount of native forest**	**Amount of agricultural areas**	**Amount of native forest around**
Functional richness	-0.47 (0.25)^{ns}	34.4 (23.71)^{ns}	0.69 (0.25)*	-	-2.66 (0.9)*	2.82 (1.09)*	-
Functional divergence	0.0003 (0.01)^{ns}	-0.27 (1.91)^{ns}	0.002 (0.02)^{ns}	-0.001 (0.01)^{ns}	-0.006 (0.05)^{ns}	0.03 (0.06)^{ns}	0.03(0.03)^{ns}
Functional evenness	0.007 (0.02)^{ns}	0.02 (1.30)^{ns}	-0.019 (0.02)^{ns}	-0.01 (0.02)^{ns}	0.09 (0.08)^{ns}	-0.07 (0.05)^{ns}	-

Significant results at * p≤ 0.05, ns = no significant result.

Table D.2 Best models results from Generalized Linear Mixed Models to verify the influence of fragments structural features and/or structural connectivity on the richness of small mammal functional traits on a Tableland Forest in southeastern Brazil. Values inside the brackets shows coefficient estimates and standard errors, for each model. Label: Frug/Omn=frugivore/omnivore; Ins/Omn=insectivore/omnivore and Omn=omnivore; A= arboreal; S= scansorial; T= terrestrial.

	Structural features of treatments		**Structural connectivity**				
Diet	**Size**	**Shape**	**Distance to source fragment**	**Mean distance to neighbors fragments**	**Amount of native forest**	**Amount of agricultural areas**	**Amount of native forest around**
Frug/ Omn	-0.098 (0.115)^{ns}	6.952 (11.148)^{ns}	0.087 (0.128)^{ns}	0.052 (0.118)^{ns}	0.027 (0.443)^{ns}	-0.391 (0.504)^{ns}	0.106 (0.241)^{ns}
Ins/Omn	0.256 (0.187)^{ns}	21.520 (17.615)^{ns}	-	-	-1.540 (0.704)*	1.995 (0.808)*	-
Omn	0.183 (0.094)^{ns}	-12.429 (9.377)^{ns}	-0.183 (0.111)^{ns}	-0.157 (0.101)^{ns}	0.513 (0.397)^{ns}	-0.598 (0.475)^{ns}	-
Locom.							
A	-0.149 (0.148)^{ns}	16.490 (14.160)^{ns}	-	-	-	-1.151 (0.408)**	0.953 (0.193)***
S	-0.063 (0.177)^{ns}	8.615 (16.61)^{ns}	-	-	-0.921 (0.641)^{ns}	1.4315 (0.717)^{ns}	-
T	-0.122 (0.163)^{ns}	4.978 (15.407)^{ns}	0.588 (0.151)**	-	-1.581 (0.54)**	-	-0.998 (0.297)**
Body							
Small	-0.207 (0.198)^{ns}	22.674 (18.168)^{ns}	0.286 (0.220)^{ns}	0.312 (0.19)^{ns}	-1.075 (0.789)^{ns}	0.1628 (0.897)^{ns}	-
Medium	-0.083 (0.117)^{ns}	3.518 (11.374)^{ns}	0.013 (0.130)^{ns}	-0.005 (0.120)^{ns}	0.117 (0.44)^{ns}	-0.049 (0.512)^{ns}	0.206 (0.237)^{ns}
Large	-0.087 (0.139)^{ns}	5.389 (13.162)^{ns}	0.230 (0.147)^{ns}	-	-1.022 (0.483)^{ns}	-	-

Significant results at * $p \leq 0.05$, ** $p \leq 0.01$ and ***$p \leq 0.001$; ns = no significant result.

Table D.3 Best models results from Generalized Linear Mixed Models to verify if the abundance of small mammal functional traits were influenced by the fragments structural features and/or structural connectivity on a Tableland Forest in southeastern Brazil. Values inside the brackets show coefficient estimates and standard errors, for each model. Label: Frug/Omn=frugivore/omnivore; Insect/Omn=insectivore/omnivore and Omn=omnivore. Significant results at * p≤ 0.05, ** p≤ 0.01 and ***p≤ 0.001; ns = no significant result.

	Structural features of treatments			**Structural connectivity**			
Diet	**Size**	**Shape**	**Distance to source fragment**	**Mean distance to neighbors fragments**	**Amount of native forest**	**Amount of agricultural areas**	**Amount of native forest around**
Frug/Omn	-0.38 (0.3)^{ns}	25.45 (27.9)^{ns}	0.42 (0.29)^{ns}	0.34 (0.29)^{ns}	-	-	-1.03 (0.87)^{ns}
Ins/Omn	-0.15 (0.09)^{ns}	10.16 (8.7)^{ns}	-	-	-	1.11 (0.42)**	-
Omn	0.32 (0.09)***	-	-0.35 (0.12)**	-0.30 (0.01)**	-0.94 (0.28)***	-	-
Locom.							
A	-0.41 (0.44)^{ns}	41.13 (41.16)^{ns}	-0.057 (0.287)*	-	3.3 (1.8)^{ns}	-	2 (0.82)*
S	-0.01 (0.1)^{ns}	-12.52 (9.12)^{ns}	-	-	-0.79 (0.3)^{ns}	0.86 (0.46)^{ns}	-
T	0.09 (0.13)^{ns}	-10.07 (12.14)^{ns}	0.245 (0.11)*	-	-	-	-0.6 (0.25)*
Body size							
Small	-0.02 (0.15)^{ns}	6.6 (13.49)^{ns}	0.03 (0.26)^{ns}	0.044 (0.146)^{ns}	0.26 (0.59)^{ns}	-0.29 (0.697)^{ns}	0.003 (0.32)^{ns}
Medium	-0.23 (0.3)^{ns}	11.2 (28.35)^{ns}	0.02 (0.36)^{ns}	-	0.83 (1.35)^{ns}	-0.85 (1.4)^{ns}	0.6 (0.65)^{ns}
Large	-0.04 (0.11)^{ns}	6.83 (10.2)^{ns}	0.14 (0.11)n^{s}	-	-0.65 (0.35)^{ns}	0.88 (0.47)^{ns}	-
Litter size	0.15 (0.17)^{ns}	-6.4 (17.2)^{ns}	-0.37 (0.17)*	-	1 (0.53)^{ns}	-	-

Table E.1 Best models results from Generalized Linear Mixed Models to verify if the functional trait richness of small mammal is influenced by the structural features of linear remnants on a Tableland Forest in southeastern Brazil. Values inside the brackets show coefficient estimates and standard errors, for each model. Labels: Frug/Omn=frugivore/omnivore; Insect/Omn=insectivore/omnivore; Frug/Gran=frugivore/gramnivore; Omn=omnivore.

Functional traits	**Structural features of linear remnants**					
Diet	**Size**	**Size*CRf**	**Size*UC**	**Width**	**Width*CRf**	**Width*UC**
Frug/Omn	0.1 (0.16)^{ns}	-	-	-3.76 (0.78)**	-3.93 (0.32)**	6.73 (1)***
Insect/Omn	0.57 (0.59)^{ns}	-	-	10.18 (3.5)*	-18.68 (4.38)*	-6.45 (4.5)^{ns}
Frug/Gran	-	-	-	-	-	-
Omn	1.06 (0.35)*	-	-	-3 (2.1)^{ns}	0.942 (2.595)^{ns}	3.26 (2.7)^{ns}
Locom.						
Arboreal	-1.8 (0.03)***	-	-	-4.15 (0.15)***	-3.92 (0.19)***	-7.87 (0.19)***
Scansorial	1.21 (0.79)^{ns}	-	-	1.44 (3.75)^{ns}	-7.85 (0.63)**	2.6 (4.76)^{ns}
Terrestrial	-0.33 (0.59)^{ns}	1.72 (0.77)^{ns}	2.12 (2.03)^{ns}	3.52 (1.64)^{ns}	-	-
Body size						
Small	1.4 (0.58)^{ns}	-	-	7.83 (3.49)^{ns}	-16.25 (4.35)**	-7.36 (4.46)^{ns}
Medium	-1.05 (0.11)^{ns}	-0.86 (0.00)^{ns}	-0.549 (0.4)^{ns}	1.43 (0.36)^{ns}	-	-
Large	0.54 (0.26)^{ns}	1.29 (0.34)**	-5.59 (0.88)***	-0.83 (0.72)^{ns}	-	-

Significant results at * p≤ 0.05, ** p≤ 0.01 and p≤ 0.001; ns = no significant result.

Table E.2 Best models results from generalized linear mixed models to verify if the abundance of small mammal in each functional trait was influenced by the structural features of linear remnants on a Tableland Forest in southeastern Brazil. Values inside the brackets show coefficient estimates and standard errors, for each model.

Functional traits	**Structural featrures of remnants**				
Diet	**Size**	Size*CRf	Size*UC	**Shape**	**Width**
Insect/Omn	-	-	-	-32.86 (25.45)^{ns}	-
Frug/Gran	0.23 (1.6)^{ns}	-	-		45.46 (748.9)^{ns}
Omn	0.83 (0.44)^{ns}	-	-	-	-
Locomotion					
Arboreal	-2.61 (0.81)**	-	-	256.5 (81.73)**	-
Terrestrial	1.06 (0.44)*	-	-	-	-
Body size					
Medium	-3.14 (1.38)*	-	-	-	-
Litter size	-0.02 (0.35)^{ns}	-0.13 (0.46)^{ns}	-6.14 (1.21)***	-	-2.88 (0.98)*

Significant results at * $p \leq 0.05$ and ** $p \leq 0.01$; ns = no significant result.

7.2 Figures

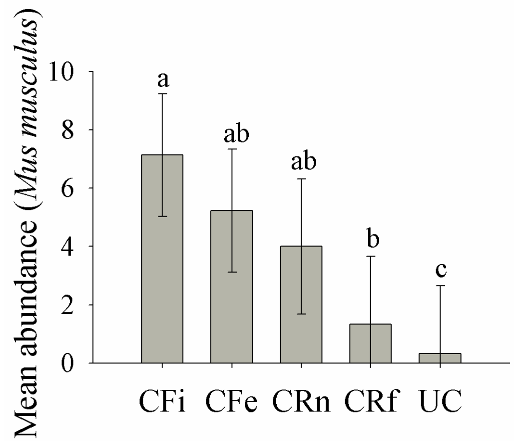

Figure A.1 The graph showing that the abundance of exotic species *Mus musculus* decrease significantly from the control forest and linear remnants connected near to the other treatments on a Tableland Forest in southeastern Brazil. Different letter denote significant differences at $p \leq 0.05$. Label: CFi = interior of control forest; CFe = edge of control forest; CRn = linear remnants connected near; CRf = linear remnants connected far; UC = linear remnants unconnected.

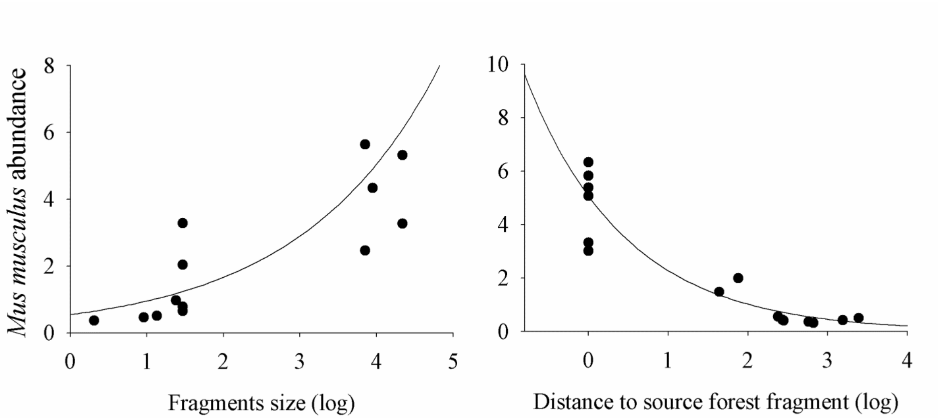

Figure A.2 Best models results from Generalized Linear Mixed Models showing that the *Mus musculus* abundance increase with the augmentation of fragments size (graph on the left) and is highest in control forest, decreasing in treatments far away from this control forest on a Tableland Forest in southeastern Brazil. All the results were significant at $p \leq 0.03$.

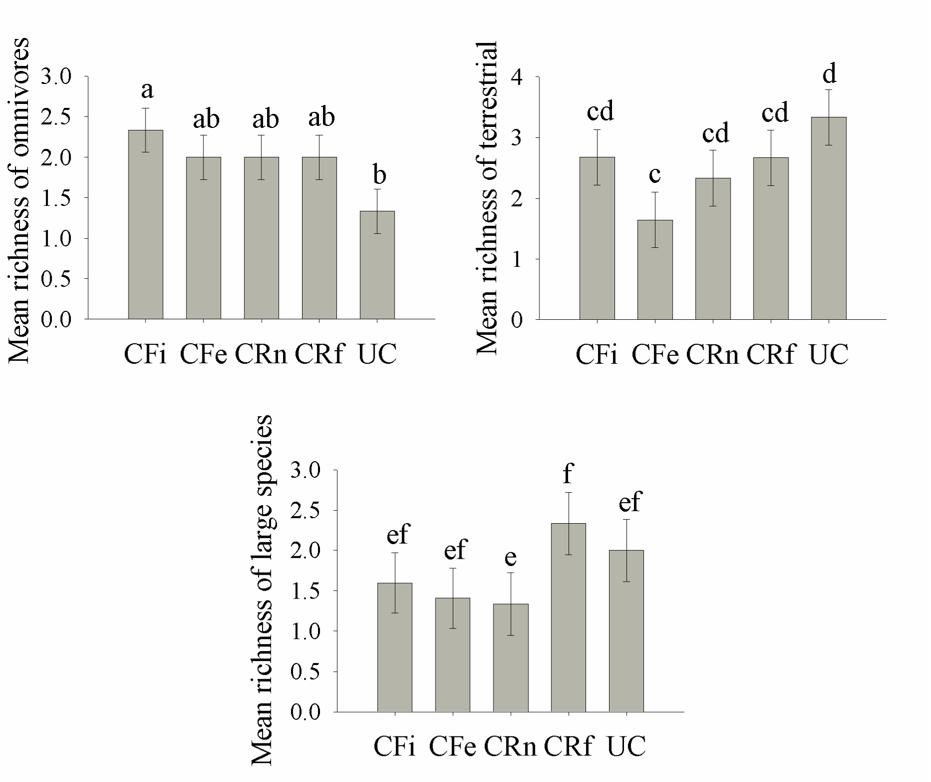

Figure B.1 The graphs showing that the small mammal functional traits are influenced by the landscape changes on the Tableland Forest in southeastern Brazil. Different letter denote significant differences at p≤0.02. Label: interior of control forest (CFi), edge of control forest (CFe), linear remnants connected near (CRn), linear remnants connected far (CRf) and linear remnants unconnected (UC).

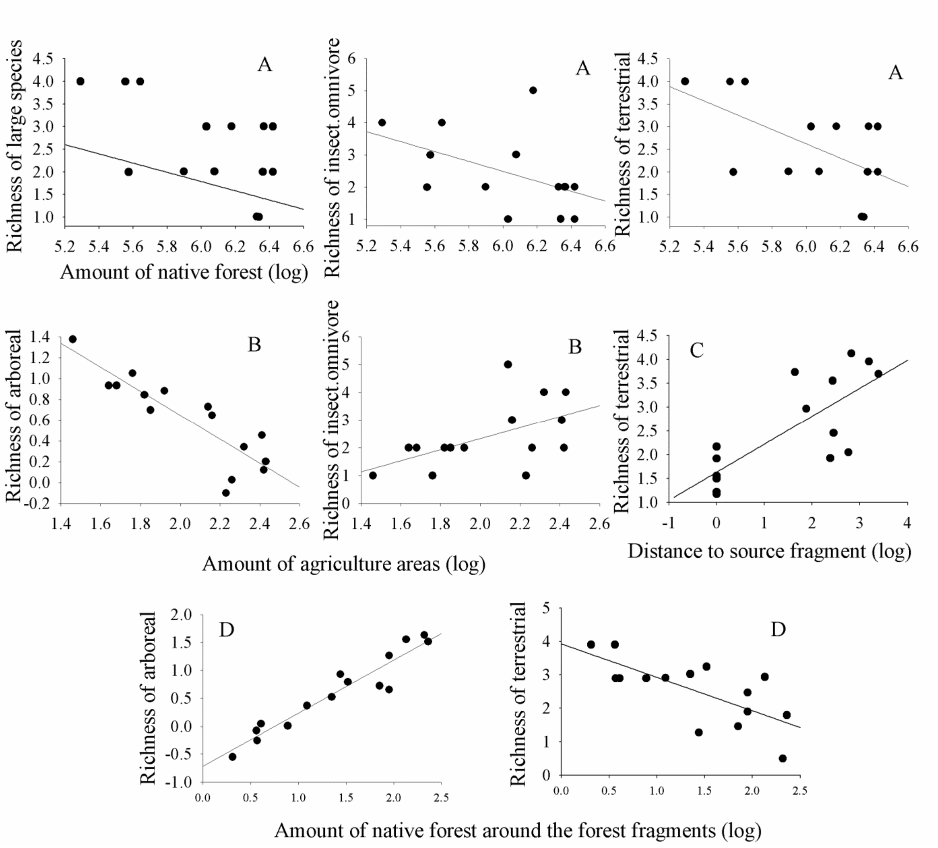

Figure B.2 Best models results from Generalized Linear Mixed Models. These graphs show that the amount of native forest in forest fragments have influence on the richness of large bodied species, insectivore omnivore and terrestrial species (A); forest fragments surrounded by most agricultural matrices have lowest arboreal species and highest insectivore omnivore richness (B); the distance influence to the source fragments on the terrestrial species (C) and the amount of native areas around the fragments influence differently the arboreal and terrestrial species inside the forest fragments, on a Tableland Forest in southeastern Brazil. All the results were significant at $p \leq 0.02$.

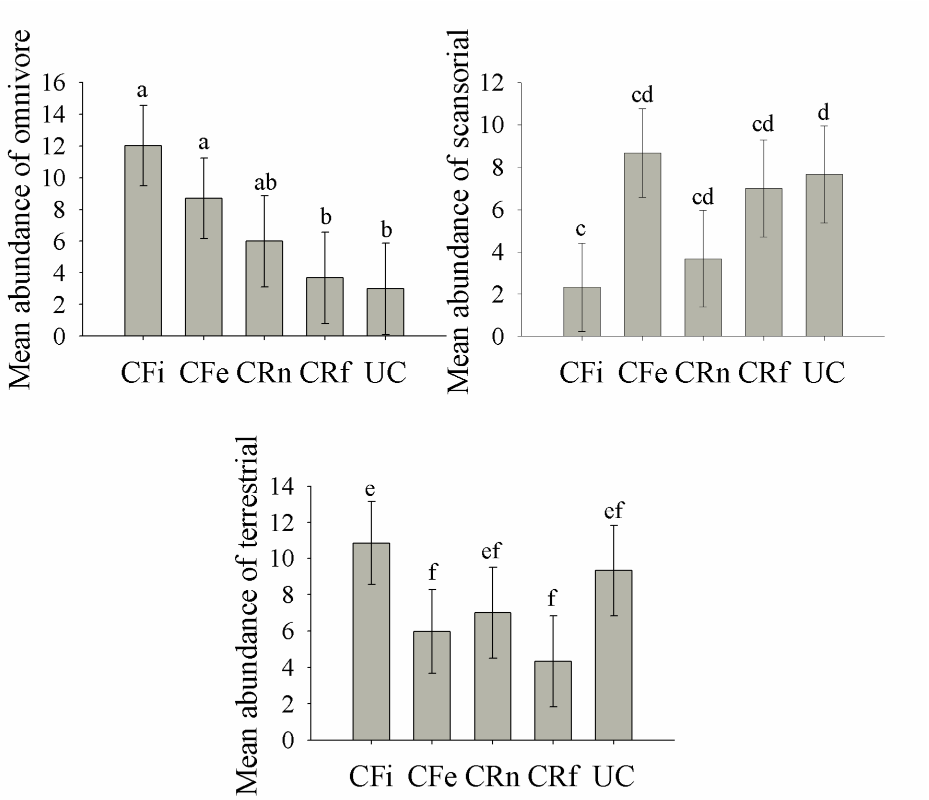

Figure C.1 The graphs showing the influence of habitat alteration on the functional traits abundances of small mammal on a Tableland Forest in southeastern Brazil. Different letter in each graph denote significant differences at p≤0.03. Label: interior of control forest (CFi), edge of control forest (CFe), linear remnants connected near (CRn), linear remnants connected far (CRf) and linear remnants unconnected (UC).

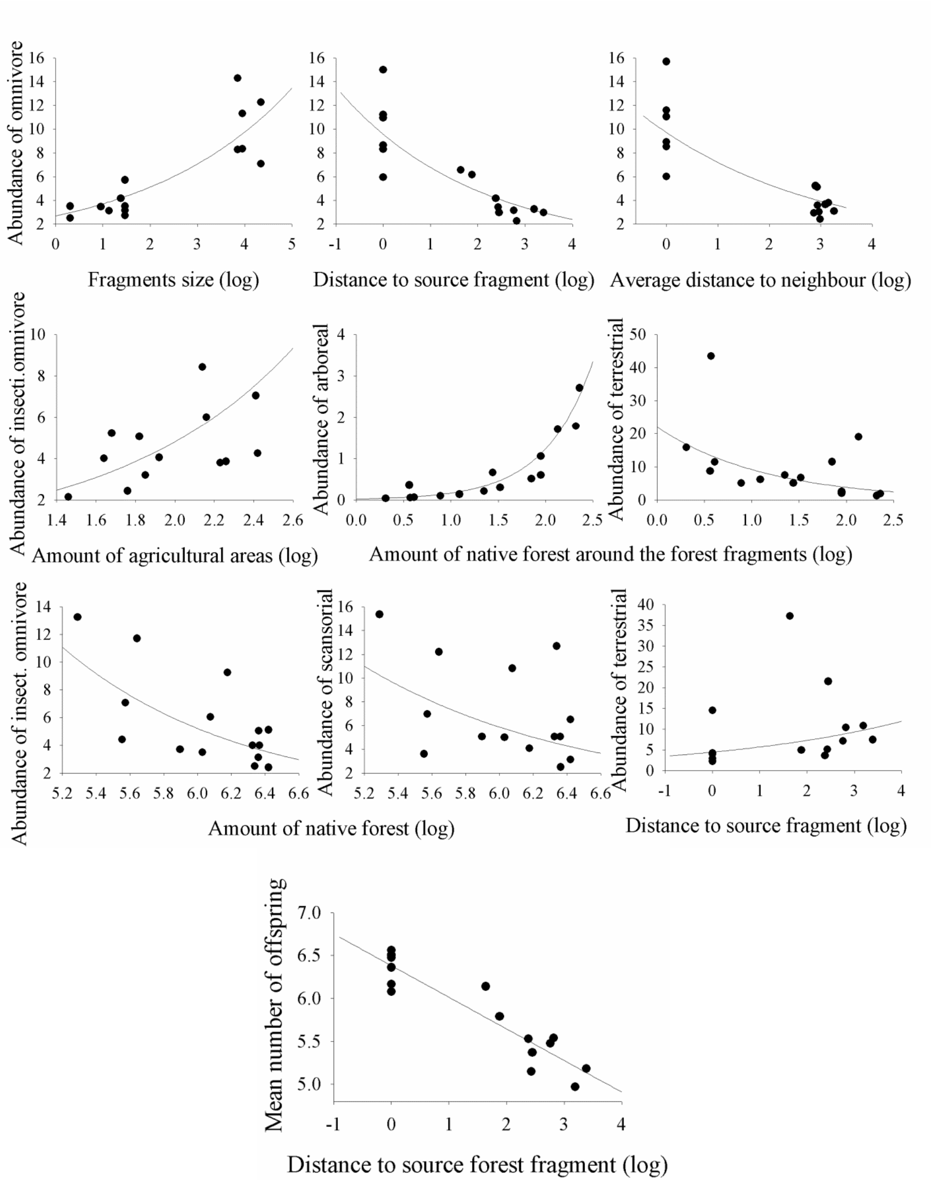

Figure C.2 Best models results from generalized mixed models showing that the omnivore abundance increased with the fragments size, in fragments most closest to the control forest and in fragments less isolated (the top tree first graphs); the other graphs show the influence of surrounding matrix around the fragments and of the distance until the source fragments on the abundance of small mammal functional traits on a Tableland Forest in southeastern Brazil. All results were significant at p≤0.02.

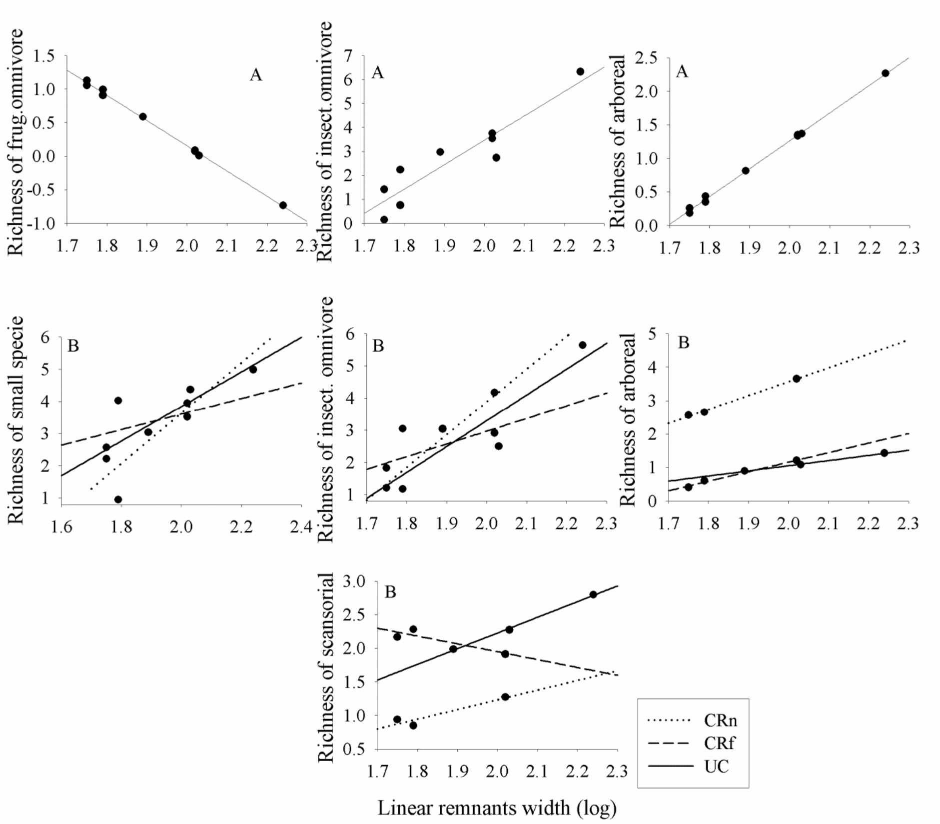

Figure D.1 Best models results from generalized mixed models showing the strong influence of linear remnants width on the richness of small mammal functional traits on a Tableland Forest in southeastern Brazil. (A) The influence of linear remnants width and (B) the width effects of each linear remnant on the richness of functional traits. All results were significant at p≤0.01. Label: linear remnants connected near (CRn), linear remnants connected far (CRf) and linear remnants unconnected (UC).

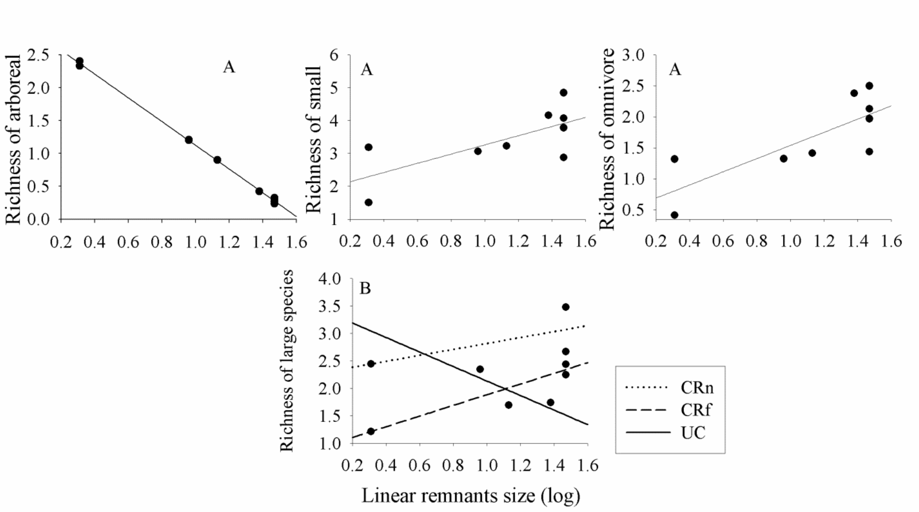

Figure D.2 Best models results from generalized mixed models showing the influence of linear remnants size on the functional traits richness of small mammal (A) and the size effect of each linear remnant on the large bodied size richness (B) on a Tableland Forest in southeastern Brazil. All results were significant at p≤0.02. Label: linear remnants connected near (CRn), linear remnants connected far (CRf) and linear remnants unconnected (UC).

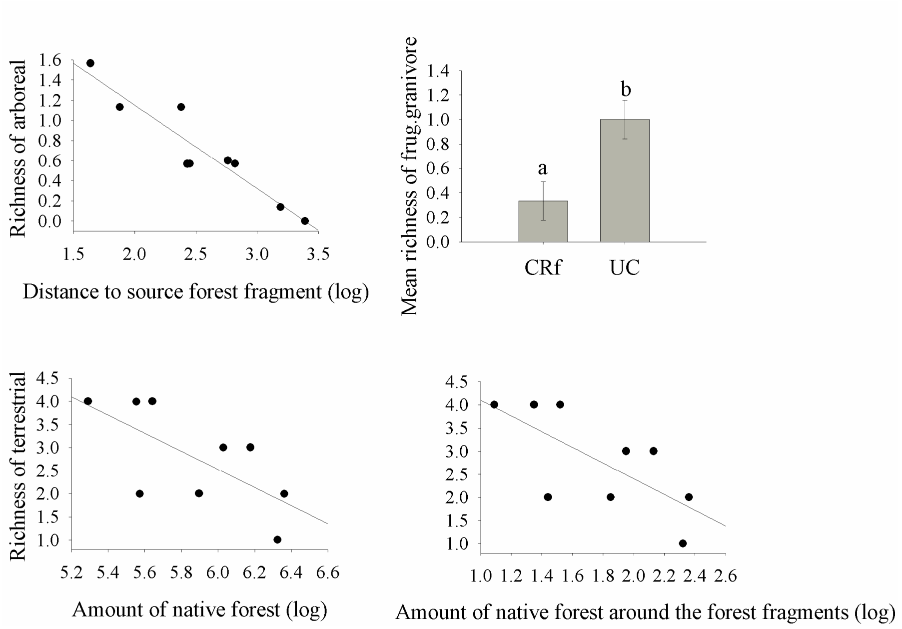

Figure D.3 Best models results from generalized mixed models showing the influence of structural connection and of the spatial arrangement of linear remnants on the functional traits richness of small mammal on a Tableland Forest in southeastern Brazil. All results were significant at p $p \leq 0.02$. Label: linear remnants connected far (CRf) and linear remnants unconnected (UC).

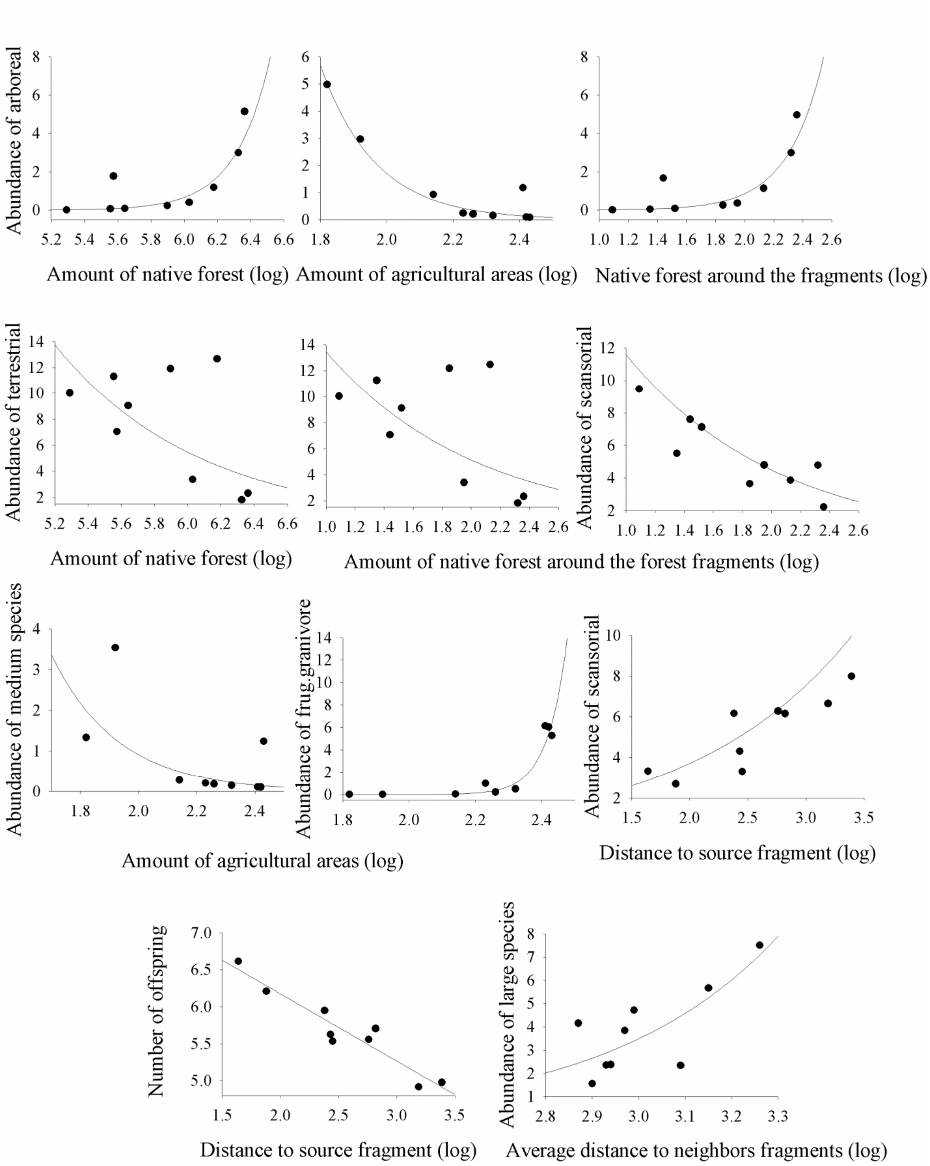

Figure E.1 Best models results from generalized mixed models showing the influence of spatial arrangement of linear remnants the functional traits abundances of small mammal on a Tableland Forest in southeastern Brazil. All results were significant at p≤0.02.

ARTIGO 3

Importance of connectivity for the distribution of tree species resources in a tropical fragmented landscape

ABSTRACT

The forest fragmentation and its associated effects drive drastic ecological losses in tropical forest. The plant dispersal mode and fruit/seed type and size are relevant predictors for assessing the fragmentation effects on biodiversity and drive important ecological process, as interaction between plant and animal disperser. However, the fragmentation effects on plant-animal dispersed is still poorly understood. In this way, we had two main goals. The first was to evaluate the importance of fragment characteristics and structural connectivity on the abundance of tree zoochoric species with different fruit/seed type and sizes in a fragmented landscape of Brazilian Atlantic Forest. The second was evaluate the effect of linear remnants, their configuration and spatial arrangement on those plant-tree traits to access their effectiveness for maintenance of plant-animal disperser interactions. Our study was carried out in five habitats: 1) control forest interior, 2) control forest edge, 3) linear remnants connected near to the control forest, 4) linear remnants connected far and 5) unconnected linear remnants. We sampled at a total of 25 sites. In each sampling unit we established one 10 x 30m plot and sampled every standing live individual tree with a diameter ≥4.8 cm. To evaluate the influence of each variable on abundance of zoochoric species and in each fruit type/size we constructed mixed models and to find the best models we used Akaike Information Criterion (AICc). Our results showed that zoochory was the main dispersion type even in fragmented landscapes, counting for 80.5% of samples and was effective for individual trees and species distribution in the landscape. Moreover both, linear remnants connected and/or the surrounding matrix, influenced the abundance of zoochoric individuals and the abundance of species with fleshy and non-fleshy fruits of different sizes, more than structural characteristics. This result shows the importance of structural connectivity and indicate that connected linear remnants and the surrounding matrix are promoting the ecological connectivity of fragmented landscapes, through plant-animal disperser interactions. Concluding, our results showed that the conservation of linear remnants needs to be encouraged, considering the surrounding matrix management, to maintain the continuity of animal-pant interactions in fragmented landscapes.

Key-words: Fragmentation. Isolation. Matrix permeability. Vegetation corridor. Conservation. Animal-plant interaction. Dispersion. Ecological connectivity.

1 INTRODUCTION

Biodiversity conservation in the human-dominated landscapes currently a challenge for ecology researchers around the tropics (LAURANCE, 1999; EWERS; DIDHAM, 2006; GARDNER et al., 2009; LOUZADA et al., 2010). Within the forest fragmentation context concerns about biodiversity and ecosystem services management appears on the top (LAURANCE, 1999; FAHRIG, 2003), since their effect can act on local scales, such as biodiversity extinctions, habitat alterations and changes in functional diversity (LAURANCE et al., 2002; FERNANDEZ, 2004; CRAMER; MESQUITA; WILLIAMSON, 2007; SANTOS et al., 2008; JORGE et al., 2013; MAGNAGO et al., 2014), as well as on large scales, like global climatic changes, which can lesd to complere alteration of wildlife functioning (LAURANCE et al., 1999; NASCIMENTO; LAURANCE, 2004; GARDNER e al., 2009; LAURANCE et al., 2011a).

The literature contains a lot of concepts, terms and procedures to study the fragmentation effects, which makes it important to follow a specific set of definitions to avoid the misunderstanding of the results and conclusions of these impacts (FAHRIG, 2003; EWERS; DIDHAM, 2006). Therefore we considered the habitat fragmentation as a process during which “a large habitat is transformed into a number of smaller patches of smaller total area, isolated from each other by a matrix of habitats unlike the original” (WILCOVE et al., 1986). Following this definition, we can distinguish four direct effects provided by habitat fragmentation: (I) subdivision of the remaining vegetation into fragments and consequent increase in their number (II) reduction in the total amount of the original vegetation, (III) decrease of forest remnants size and (IV) increase of isolation among these remnants (FAHRIG, 2003; BENNET; SAUNDERS, 2010).

Most fragment studies working with species and community responses to fragmented landscapes consider mainly the influence of five spatial attributes of forest remnants: (i) fragment size, (ii) fragment shape, (iii) edge effects, (iv) fragment isolation and (v) introduction of new forms of land-use to replace vegetation that was lost, or matrix structure (EWERS; DIDHAM, 2006; BENNET; AUNDERS, 2010). Moreover it is already known that the forest fragmentation process and its associated consequences leads to high loss of species and ecological processes in the highly diverse tropical forests of the planet (LAURANCE et al., 2002; FAHRIG, 2003; OLIVEIRA; GRILLO; TABARELLI, 2004; EWERS; DIDHAM, 2006; CRAMER; MESQUITA; WILLIAMSON, 2007; PERES; PALACIOS, 2007; SANTOS et al., 2008; JORGE et al., 2013; MAGNAGO et al., 2014) and seems to be more severe in human-dominated landscapes (LAURANCE et al., 2006).

Furthermore, the fragment size reduction, isolation increase and creation of non-forest matrices also promote negative influences on species abundance of frugivores (CHIARELLO, 1999; MARSDEN; WHIFFIN; GALETTI, 2001; RIBON; SIMON; MATTOS, 2003; UEZU; METZGER; VIELLIARD, 2005; GALETTI et al., 2006; PERES; PALACIOS, 2007; RODRÍGUEZ-CABAL; AIZEN; NOVARO, 2007; MARTENSEN; PIMENTEL; METZGER, 2008; MAGRACH; LARRINAGA; SANTAMARIA, 2012), mainly the large-bodied species (REDFORD, 1992; GALETTI; PIZO, 1996; CHIARELLO, 1999; PERES, 2000; CRAMER; MESQUITA; WILLIAMSON, 2007; JORGE et al., 2013; VIDAL; PIRES; GUIMARÃES, 2013). Since most of the trees in the Neotropics are dependent on animals for seed dispersion (HOWE; SMALLWOOD, 1982; FLEMING; BREITWISCH; WHITESIDES, 1987; JORDANO, 2000; ALMEIDA-NETO et al., 2008; FLEMING; KRESS, 2011), the presence and movement of frugivores through the matrix surrounding the remnants could be considered a very important process to connect plant populations across fragmented landscapes (VIDALE et al., 2013). Also, the capacity of dispersers to move across matrices, determine the persistence of plants by disperser animals in these landscapes (MCCONKEY et al., 2012).

Some studies have indirectly shown that edge creation and isolation among remnant patches negatively affect the plants dipersed by animals (see OLIVEIRA; GRILLO; TABARELLI, 2004; SANTOS et al., 2008; MAGNAGO et al., 2014) and that the dispersion of large seeds is drastically reduced in small fragments (LAURANCE et al., 2006a; CRAMER; MESQUITA; WILLIAMSON, 2007; MELO; LEMIRE; TABARELLI, 2007; SANTOS et al., 2008). Other studies directly tested the influence of fragment size (GALETTI et al., 2006) and structural connectivity, provided by vegetation corridors (TEWKSBURY et al., 2002; LEVEY et al., 2005; BRUDVIG et al., 2009) and by the matrix permeability (MAGRACH; LARRINAGA; SANTAMARIA, 2012) on plant-animal mutualistic relations. However the direct effects of fragmentation and the influence of structural connectivity on plant-disperser interaction is still poorly understood (MCCONKEY et al., 2012; HAGEN et al., 2012).

Therefore we evaluated the influence of patch characteristics and structural connectivity on abundance of individuals of zoochoric tree species, which are relevant predictors for assessing the fragmentation effects on plant-community and for describing the fauna resource interactions (HAGEN et al., 2012). More specifically, our main goal was to infer about the best spatial arrangement and configuration of linear remnants to maintain plant-animal dispersers in the human-dominated landscape. Our results will potentialy guide strategies for designing of linear remnants to ensure the ecological connectivity in fragmented landscapes.

2 MATERIAL AND METHODS

2.1 Study area

Our study was carried out in southeastern Brazil (19° 11 '52 "S and 40° 5' 29" W - 18° 54 '18 "S and 40° 5' 19 "W). The study area is located in one of the most important global hotspots (MYERS et al., 2000) in a keystone biodiversity area (PAESE et al., 2010). The landscape studied comprises a large forest of 46,000 ha belonging to the Companhia Vale S.A., a privately-owned company, and to the federal government (Reserva Biológica de Sooretama) surrounded by a matrix composed mainly of *Eucalyptus* spp., papaya and coffee plantations and pasture (PEIXOTO et al., 2008; ROLIM et al., 2005) and by forest fragments of different sizes, shapes, widths and degrees of isolation. This forest is the second largest reserve of Tableland Forest (PEIXOTO et al., 2008; PEIXOTO; SIMONELLI, 2007; SBREK-ARAÚJO; CHIARELLO, 2008) and the one of the largest forest remnants of the Atlantic Forest (RIBEIRO et al., 2009). Furthermore it is considered one of the 14 centers with the highest plant diversity in Brazil (PEIXOTO; GENTRY, 1990; PEIXOTO; SILVA, 1997), the second most important area for mammal conservation in the Brazilian Atlantic Forest (GALLETI et al., 2009) and a refuge for threatened bird species (MARSDEN; WHIFFIN; GALETTI, 2001; SRBEK-ARAÚJO; CHIARELLO, 2006) and mammals (CHIARELLO, 1999).

The forest in the region is classified as Lowland Rain Forest (IBGE, 1987) or Tertiary Tableland because of its occurrence on Cenozoic sediments from the Barreiras group, with altitudes ranging from 28 to 65 m (PEIXOTO et al., 2008). The lowland forest is characterized with trees up to 40 m tall, girths up to 400 cm and a sparse understory, with the dominance of the Fabaceae, Myrtaceae, Rubiaceae, Annonaceae, Sapotaceae and Bignoniaceae tree families

(JESUS; ROLIM, 2005; PEIXOTO et al., 2008; PEIXOTO; SIMONELLI, 2007).

2.2 Sampling design

We selected the large forest previously mentionated as control and we sampled the small mammals within five rainforest habitats or treatments: 1) interior of control forest, 2) edge of control forest, 3) unconnected linear forest remnants (termed “unconnected linear remnants”) and 4) linear forest remnants connected to the control forest (termed “connected linear remnants”). We separated the last treatment in two categories according to the distance until the structural connection as i) linear remnants connected near the control forest (placed after the edge), and ii) linear remnants connected far from the control forest, with the sample transect located along connected remnants and with a minimum distance of 400 m until the control forest. We chose these treatments considering the similarity in the composition of the surrounding matrix and a minimum distance of 400 m from the interior to the nearest edge of the control forest (range = 400 to 2,642 m) (see Figure1).

Fieldwork was conducted from April 2012 to May 2012. In each sampling unit we established one plot of 10 x 30 m, totaling 25 plots. Inside each plot, we sampled every standing live tree individual with a diameter ≥4.8 cm at breast height (1.3m above the ground) (DBH). We identified trees comparing with material references of collections of the CVRD Herbarium of the Vale and the VIES Herbarium of the Federal University of Espírito Santo, and with aid of taxonomic specialists.

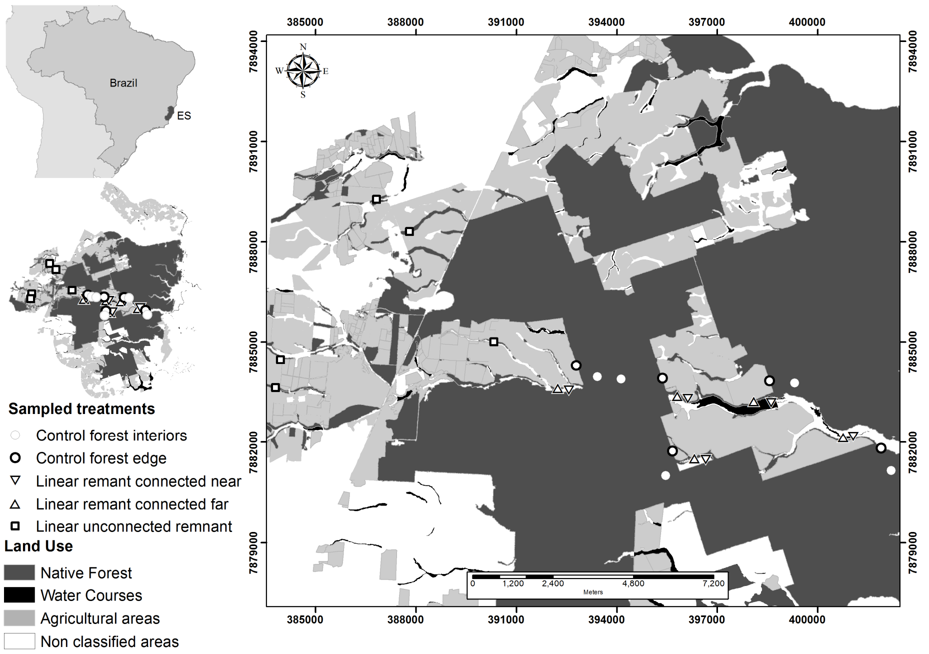

Figure 1 Study area and sampled treatments in a Tableland Forest in southeastern Brazil. To check the respective information about each treatment see the table A.1.

2.3 Land cover analysis and Independent variables

The metrics utilized to characterize the landscape changes and connectivity were extracted from a land cover map produced with an image with a high spatial resolution classification. We used an image with resolution of one meter, acquired in the year 2008. The orthorectified images and with atmospheric correction and visual evaluation of image registration, was obtained through the Vale Natural Reserve.

To classify the land cover we an used image based on multiscale segmentation The segmentation partitioned the image into groups of pixels spectrally similar and spatially adjacent (DESCLÉE; BOGAERT; DEFOURNY, 2006; DUVEILLER et al., 2008), using a "trial-and-error" attempt to find an fragmentation scale appropriate value. Once a successfully segmented image was obtained using 40 as a scale factor, we applied an object-based classification using Nearest Neighborhood (NN). We used 20 trained samples obtained in the field to apply the NN classification algorithm. The result is a class label for each of the segments in each class. A few wrongly-classified image objects were reassigned manually to the correct classes based on field knowledge and on visual interpretation of the image. Classification validation was obtained using 150 independent data sources as reference, randomly distributed over each class. User accuracy, producer accuracy, overall accuracy and kappa coefficient obtained high values, above 85%.

The resulting map was converted to vector format and we computed seven continuous variables using ArcGis (Table A.1, Supplementary material). For each sample treatment, we obtained their structural characteristics, such as size (hectare) and shape, using the ratio between area and perimeter according to Helzer and Jelinski (1999) and the mean width. For the mean width calculation, we obtained three widths for each treatment and considered the average among them. Also, to access the structural connectivity, we constructed a buffer with 2 km around each sampling treatment (total of 25). We quantified the amount of agricultural areas in the buffer (representing by coffee, *Eucalyptus* spp. and papaya plantations), the amount of native forest in the matrix and the amount of native forest of each treatment. Also, we measured the minimum distance between the sample treatment and the nearest source fragment and also, the mean distance to the neighbors nearest fragments. For this, we considered the four fragments nearest the sample treatment. We used these variables to characterize the spatial arrangement of linear remnants, as well. We chose these variables since they are key components to maintain species and ecological processes in fragmented landscapes, are essential to infer about the best spatial arrangement and to the evaluate the configuration of linear remnants for conservation in human-dominated landscape (see LAURANCE, 2004; PARDINI et al., 2005; EWERS; DIDHAM, 2006; HAWES et al., 2008; LEES; PERES, 2008; MARTENSEN; PIMENTEL; METZGER, 2008; BARLOW et al., 2010; MAGRACH; LARRINAGA; SANTAMARIA, 2011; ROCHA; PASSAMANI; LOUZADA, 2011; MARTENSEN et al., 2012; GARMENDIA et al., 2013).

We used two categorical variables to evaluate the effects of the structural connection among linear remnants and control forest (connected and unconnected), and the distance of the connection to the remnants (connected near and far). Furthermore, we used the width of linear remnants connected and unconnected as structural features. We considered the same variables used for evaluate the structural connectivity (described above) to characterize the spatial arrangement of linear remnants.

Thus, we verified the influence of habitat alteration, habitat features, structural connectivity and the effects of structural features and spatial arrangement of linear remnants on remnants on the abundance of tree species with seed dispersed by wildlife.

2.4 Dependent variables

We categorized the tree species according to the dispersion mode following Van Der Pijl (1982). We used two categories, zoochoric and non-zoochoric. A zoochoric tree produces diaspores surrounded by fleshy pulp, an arill or other features that are typically associated with dispersal by animals. The non-zoochoric trees have 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 dehiscence.

Moreover, we classified the sampled zoochoric trees into three categorical variables that are relevant to assess the role of trees as food resources: fruit type, fruit size and seed size (see HOWE; SMALLWOOD, 1982; VAN DER PIJL, 1982; FLEMING; BREITWISCH; WHITESIDES, 1987; TABARELLI; PERES 2002; MORAN; CATTERALL, 2010). We used these dependent variables because they are relevant predictors for assessing the fragmentation effects on the plant-community, are a proxy describing plant-animal interactions and have important consequences for the biodiversity of the entire area (ORIANS; DIRZO; CUSHMAN, 1996; HAGEN ET AL., 2012)

The detail classification for each category of tree zoochoric species follows:

(1) Fruit type: the fruits were categorized into (i) fleshy fruits (i.e., the pericarp can accumulate water and many organic compounds, see Coombe, 1976) and (ii) non-fleshy fruits.

(2) Fruit and seed size: we categorized the fruit and seed sizes into four categories according to Tabarelli and 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 very large (more than 3.0 cm).

2.5 Data analysis

As the sampling effort was equal for all treatments, each transect was considered a sampling unit or replicate. In this way, we obtained the zoochoric species abundances by the number of sampled tree individuals.

First, we constructed mixed models with all treatments together in R version 2.15.1 (R Development Core Team 2012) to evaluate the effects of fragment structural features and structural connectivity on the abundance of zoochoric species. After, the mixed models were constructed to evaluate the effect of these variables on the abundance of tree species with different fruit type (fleshy fruits and non-fleshy fruits) and with different fruit and seed size (small, medium, large and very large).

Secondarily, we constructed the mixed models, however considering just the connected/unconnected linear remnants in the analysis. Thus, we evaluate the influence of structural features and the spatial arrangement of the linear remnants on the same dependent variables. Within the models, we used an interaction with three levels (connected near, connected far and unconnected) to check the effects of linear remnants connection (connected and unconnected) and to assess the influences of the distance effect within connected remnants (connected near and far) between these levels separately. The analysis regarding to structural features and structural connectivity were run separately to verify the influence of each variable on the tree zoochoric abundance.

We constructed the mixed models using *glmmadmb* function from the *glmmADMB* package with Poisson family, once all data were countable (abundance). When these data showed overdispersion, we used the Negative Binomial family. The sites (each treatment) 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 variables included in the global model. However, to avoid multicollinearity between explanatory variables we not include in the same model the variables with autocorrelations (linear Pearson correlations large or equal to 0.6).

We selected the best model using an theoretical information approach based on the Akaike Information Criterion of Second Order, which is indicated for small sample sizes (AICc) and chose the models according with the lowest AICc value (BURNHAM; ANDERSON; HUYVAERT, 2011). The plausibility of alternative models was given by the differences in their AICc values in relation to the AICc of the most plausible model (ΔAICc). We considered as plausible models those with a value of ΔAICc<2. When the models showed the ΔAICc value <2 and the variable included on the models was significant (considering $p \leq 0.05$) we considered the variables in the models as an important result to induce the changes on abundance of tree zoochoric individuals. These analyses were performed in the R version 2.15.1 (R Development Core Team 2012).

3 RESULTS

3.1 Tree species characteristsics in the sampled landscape

We sampled a total of 1209 tree individuals of 262 species and two indeterminate species. Of these, 234 individuals of 56 species were non-zoochoric and 973 tree individuals were zoochoric, represented by 206 species (Table B.1, Suplementary material). We found 153 species and 633 individuals with fleshy fruits and 53 species and 340 individuals with non-fleshy fruits. The abundance of tree species with large fleshy fruits was highest (299 individuals) and the species with small fleshy fruits had the lowest abundance (43). Considering the tree species with non-fleshy fruits, the abundance was highest for species with medium fruits (109) and lowest for species with small fruits (54). The most abundant tree species had small seed size (371) and medium size (346), followed by the large (156) and very large size (100), as shown by the Anova results (F=18.01; $p<0.001$) and by the posteriori test (comparison between small and medium seed: p=0.95; small and large seed: $p<0.001$; small and very large seed: $p<0.001$; medium and large seed: $p<0.01$; medium and very large seed: $p<0.001$).

3.2 Impacts of landscape changes, habitat clearance and structural connectivity on zoochoric species components

The abundance of tree species with zoochoric dispersion in the interior of control forest was 214 (42.8±13.97), 205 (41±14.05) in the linear remnants connected near, the edges of the control forest harbor 195 individuals (39±7.52), 188 (37.6±15.65) in the linear remnants connected far and 177 (34.2±13.31) individuals in the unconnected linear remnants, 20% lower than the number found in the interior to the control forest (Table B.2, Supplementary material). Despite this, the models results did not show significant differences related to habitat alteration for the tree zoochoric species (Table B.1 and Figure A.1, Supplementary material). Also, ours best models showed that the fragment size, shape and the structural connectivity had no significant influence on the tree zoochoric abundances (Table C.1, Supplementary material).

When we consider the fruit type our results showed that the abundance of tree species with fleshy fruits was significantly lower within linear unconnected remnants than the interior, edge and linear connected near (Table B.2, Supplementary material). Otherwise, the number of tree individuals with non-fleshy fruits did not differ significantly between the sampled treatments (Table B.2, Supplementary material). The fragment features (fragments size and shape) and the structural connectivity had no significant influence on the abundance of tree individuals with fleshy fruits and non-fleshy fruits as showned by the results from selected best models (Table C.1, Supplementary material).

We found that the habitat alteration promoted shifts in the abundance of tree species depending on fruit size and type. The abundance of tree individuals with very large fleshy fruits was significantly higher in the interior of the control forest than in other treatments, except for the edge of the control forest (Table B.2, Supplementary material). However, the abundance of these individuals in the linear remnants connected near and unconnected did not differ from the edge of the control forest. The abundance of individuals with medium fleshy fruits in the interior of the control forest was significantly higher when compared with connected far and unconnected remnants, and did not differ for the connected near and edge of the control forest (Table B.2, Suplementary material). The abundance of species with medium fleshy fruits in the unconnected linear remnants differed significantly for all treatments, except for the connected far.

The abundance of tree species with large non-fleshy fruits did not differ significantly between the edge of the control forest, linear remnants connected far and unconnected linear remnants (Table B.2, Suplementary material). We found that the abundance of species with very large non-fleshy fruits was significantly lower in linear remnants connected far compared with the edge of the control forest, linear remnants near and unconnected (Table B.2, Suplementary material). Also, the abundance of these individuals was highest in unconnected linear remnants, however no significant difference was found.

The fragment features had effects on the abundance of tree species with medium and very large fleshy fruits, however, the other fruit types and sizes were not influenced by this variable (Table C.1, Suplementary material). The fragment size had a significantly positive influence on the abundance of tree species with medium fleshy fruits (GLMM; z=3.4, p<0.001, Figure 2). The fragment shape had a significantly negatively influence on the abundance of tree species with very large fleshy fruits (GLMM; z=-3.56, p<0.001, Figure 3). Also, the abundance of tree individuals with medium fleshy fruits was lower in fragments more distant from the source fragment and more distant from neighboring fragments (GLMM; z=-3.66, p<0.001, for distance to source fragment and GLMM; z=-3.62, p<0.001 for distance to neighboring fragments, Figure 2). The abundance of species with very large fleshy fruits declined with the increase of native forest in the matrix (GLMM; z=-3.27, p<0.01, Figure 5).

The abundance of tree species with non-fleshy fruits and different seed size was not significantly explained by the structural connectivity (Table C.1, Suplementary material).

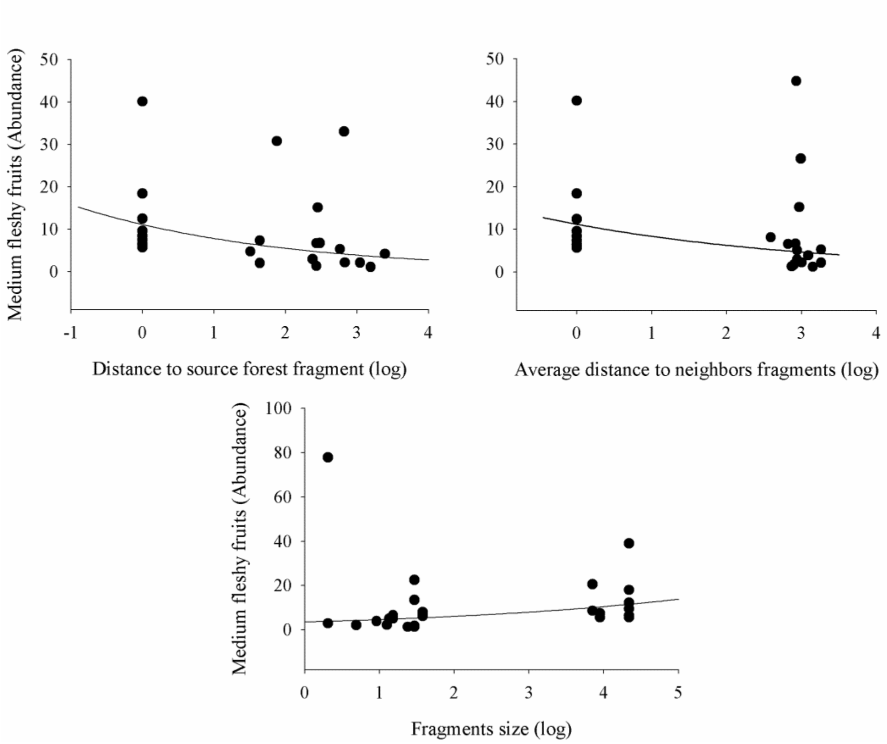

Figure 2 Best models results from Generalized linear mixed models showing the influences of structural connectivity (upper graphs) and fragments size on the abundance of tree zoochoric trees with medium fleshy fruits on a Tableland Forest in southeastern Brazil. All the results were significant at $p \leq 0.001$.

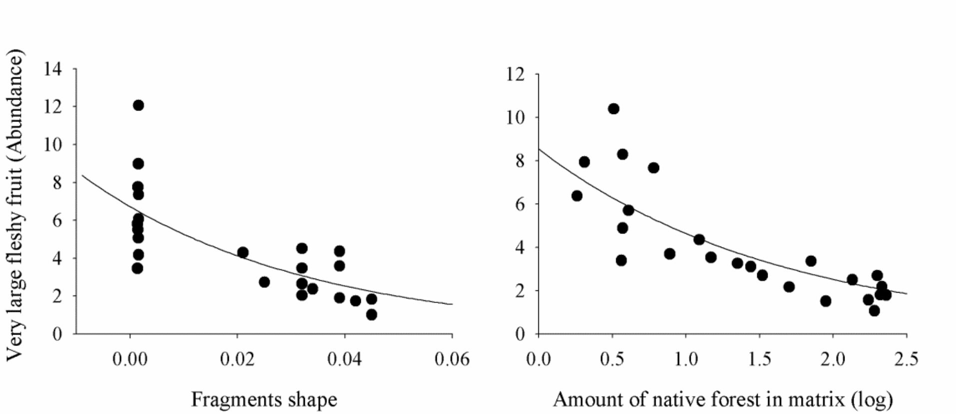

Figure 3 Best models results from Generalized linear mixed models showing the influences of fragments shape (on the left graph) and structural connectivity (on the right graph) on the abundance of tree zoochoric species with very large fleshy fruits on a Tableland Forest in southeastern Brazil. All the results were significant at $p<0.05$.

Ours results showed that, independent of seed size, the abundance of tree species was not influenced by the habitat type, not differing significantly among sampled treatments (Table B.2, Suplementary material). Only the abundance of tree species with very large seeds in linear remnants connected near was different from abundance in connected far (GLMM $z=2.2$ $p=0.03$).

The fragment shape had a significantly negative influence on the abundance of tree species with small seeds (GLMM $z=-1.96$ $p=0.05$, Figure B.1). The other species with medium, large and very large seeds were not influenced by the fragment features, as shown by the best models selected (Table C.1, Suplementary material). In the same way, the structural connectivity, represented by the matrix permeability and proximity to the other fragments, had no significant influence on the abundance of tree species of any seed size (Table F.1, Suplementary material).

3.3 Are the tree species which provide wildlife resource through fruits and seed affected by the features and spatial arrangement of the linear remnants?

Considering just the linear remnants (connected and unconnected), our results showed that the abundance of zoochoric species was negatively influenced by the linear remnants shape (GLMM; $z=-1.98$, $p=0.048$, Table H.1, Supplementary material). Otherwise, the abundance of zoochoric species was not influenced by the spatial arrangement of linear remnants (null model selected, Table D.1, Supplementary material).

The zoochoric tree abundance with fleshy and non-fleshy fruits was not influenced by the size, shape and width of the linear remnants (Table D.1, Supplementary material). Also, the abundance of individuals with fleshy and non-fleshy fruits was not influenced by the spatial arrangement of linear remnants (Table D.1, Supplementary material).

The structural connection and the distance of the connection, affected the abundance of tree species with small non-fleshy fruits. The abundance of these individuals was higher in linear remnants connected near than in linear remnants connected far (GLMM; $z= -3.62$, $p<0.01$) and unconnected linear remnants (GLMM; $z= -4.02$, $p<0.001$) and did not differ for the two last treatments (GLMM; $z= -1.78$, $p=0.075$). However, the spatial arrangement of linear remnants had no significant influences on the abundance of tree species with small non-fleshy fruits (Table D.1, Suplementary material). For other species with fleshy and non-fleshy fruits of different sizes, neither linear remnant features nor their spatial arrangement was significant (Table D.1, Suplementary material).

The abundance of tree species with small seed was significantly and negatively influenced by the linear remnants shape (GLMM $z= -2.25$ $p =0.03$,

Figure B.1, Supplementary material), while the other tree species with different seed sizes was not influenced by the linear remnant features (Table D.1, Supplementary material). Also, the spatial arrangement of the linear remnants did not influence the dispersion of zoochoric species with different seed sizes (Table D.1, Supplementary material).

4 DISCUSSION

4.1 Features of tree species with zoochoric dispersion in the sampled landscape and their responses to landscape changes, habitat clearance and structural connectivity

Our results showed that 80.5% of tree individuals were zoochoric, four times more than individuals with non-zoochoric dispersion. The abundance of individuals with fleshy fruits was also higher than tree individuals with non-fleshy. The zoochoric individuals were widely distributed over the entire landscape, not differing significantly between the sampled treatments. Moreover, the abundance of zoochoric species was not influenced by the structural characteristics of fragments and matrix composition. These results together show that zoochoric dispersion was the main dispersion type found in the sampled landscape. Our result corroborated with the studies in the Neotropics, showing that animals play a major role in seed dispersal (HOWE; SMALLWOOD, 1982; FLEMING; BREITWISCH; WHITESIDES, 1987; JORDANO, 2000; ALMEIDA-NETO et al., 2008; FLEMING; KRESS, 2011). Furthermore, our results demonstrate that zoochoric dispersion was efficiently even in fragmented landscapes.

The habitat alteration did not affect the zoochoric tree abundance with non-fleshy fruits (no significant results for the best models). However, the zoochoric tree abundance with fleshy fruits was significantly affected by the connection provided by the linear remnants, being higher in the control forest and within the connected linear remnant than in unconnected linear remnants (see results), showing that individuals with this particular trait are more sensitive to isolation effects than those with non-fleshy fruits (see MAGNAGO et al., 2014). Moreover, since frugivores are predominant in the same control forest sampled (CHIARELLO, 1999) these results indicate that the animal disperser can be moving through connected linear remnants. Because fragment isolation is a important factor which can restrict the interactions between plants and animal disperser (JORDANO et al., 2006; HAGEN e al., 2012; MAGRACH; LARRINAGA; SANTAMARIA, 2012), our results indicate that connected linear remnants could be working for the functional and ecological connectivity in the fragmented landscapes, as predicted by theoretical studies (Hagen et al., 2012) and observed by other studies as well (TEWKSBURY et al., 2002; LEVEY et al., 2005; BRUDVIG et al., 2009).

In this way, these findings show that the connectivity provided by linear remnants ensures that the tree species, which are dispersed by animals and also the interactions between plant and disperser are completely dependent on two major landscape conditions: (i) large fragments, that can act as source areas for plant and animal dispersers (CHIARELLO, 1999; BRUDVIG et al., 2009; MAGNAGO et al., 2014); and (ii) structural/functional connection between the linear remnants with a source forest fragment, which allows species movement between the connected linear remnants and ,aintains this ecological interaction (see TEWKSBURY et al., 2002; LEVEY et al., 2005; JORDANO et al., 2006; BRUDVIG et al., 2009; HAGEN e al., 2012; MAGRACH; LARRINAGA; SANTAMARIA, 2012).

Our results show that the resource type provided by tree species changes significantly depending on habitat type, fragment size and isolation. The zoochoric trees with medium fleshy fruits were more abundant in the control forest and within linear remnants connected near than in unconnected linear remnants. Otherwise the zoochoric trees with non-fleshy fruits were highest in unconnected linear remnants, linear connected far and control forest edges. The highest abundance of medium fleshy fruits in the control forest means that this forest type is providing more resource for the fauna, because fleshy fruits present greater quantities of nutrients and water in the pericarp than the non-fleshy fruits (COOMBE, 1976; VAN DER PIJL, 1982), allowing large-bodied frugivore fauna to exist (see VAN DER PIJL, 1982; CHIARELLO, 1999; ANDREAZZI; PIRES; FERNANDEZ, 2009). However the zoochoric non-fleshy fruits have other kinds of resources (e.g. arillate seeds, fleshy funicles and sarcotesta, see VAN DER PIJL, 1982), which generally are food resources for some birds and small mammal species (VAN DER PIJL, 1982; GALETTI; PIZO, 1996; ANDREAZZI; PIRES; FERNANDEZ, 2009), but are mostly consumed by invertebrates (FLEMING; KRESS, 2011). These results shows that each component of the landscape (i.e. our habitat type) provides different kinds of resources for fauna. Thus, we can expected shifts in the plant disperser species and in the interaction level between fauna disperser and different fruit type in each sampled habitat, which will impact the functional diversity of the forest fragments (see MAGNAGO et al., 2014).

4.2 Influences of the structural features and spatial arrangement of linear remnants on zoochoric tree species abundance

The abundance of tree species with zoochoric dispersion decreased significantly in linear remnants with more irregular shapes (see results), corroboring with the studies which showed that the amounts of area exposed to the edge effects promote changes in the species functional attributes (see

HELZER; ELINSKI, 1999; OLIVEIRA; GRILLO; TABARELLI, 2004; MELO; LEMIRE; TABARELLI, 2007; MAGNAGO et al., 2014). The edge effects could have negative consequences for disperser-plant interactions, either by driving larger loss of species/abundance of tree species in the edge habitat (LAURANCE et al., 1997; LAURANCE et al., 2002; OLIVEIRA; GRILLO, TABARELLI, 2004; LAURANCE et al., 2007; EWERS; DIDHAM, 2008; OLIVEIRA et al., 2008; SANTOS et al., 2008; MAGNAGO et al., 2014) or by causing abundance and richness impoverishment of keystone native dispersers in this habitat (CRAMER; MESQUITA; WILLIAMSON, 2007; PERES; PALACIOS, 2007).

No linear remnant spatial arrangement characteristic influenced the zoochoric individual abundance. The main group of vertebrate dispersers in the Tropics, medium/large mammals and birds (HOWE; SMALLWOOD, 1982; FLEMING et al., 1987; FLEMING; KRESS, 2011), is negatively influenced by the landscape changes in the same sampled landscape (CHIARELLO, 1999; MARSDEN; WHIFFIN; GALETTI, 2001). Large mammal frugivores had higher abundance in the control forest than in medium and small fragments (CHIARELLO, 1999) and few bird species use the eucalyptus plantation matrix that surrounds the forest fragments (MARSDEN; WHIFFIN; GALETTI, 2001). In this way, considering that isolation is an important factor which restricts the interactions between plant and animal dispersers (JORDANO et al., 2006; HAGEN et al., 2012; MAGRACH; LARRINAGA; SANTAMARIA, 2012), our findings indicate that the composition of the matrix surrounding the linear remnants permits, to a certain extent, the movement of animal dispersers. In fact, near the sampled landscape, the presence of one large-bodied frugivore, *Tapirus terrestris*, was recorded, using of eucalyptus plantation areas to reach forest fragments (CENTODUCATTE et al., 2011). Thus, the occasional displacement of frugivores through the matrix can be contributing to the dispersion of zoochoric trees in this fragmented landscape.

The movement throught the matrix by some large disperser species could also explain the absence of significant influence of spatial arrangement of linear remnants on the abundance of tree species with different fruits and seed size, because the larger dispersers are heavily affect by the forest fragment size and isolation (CHIARELLO, 1999; RIBON; SIMON; MATTOS, 2003; UEZU; METZGER; VIELLIARD, 2005; CRAMER; MESQUITA; WILLIAMSON, 2007; JORGE et al., 2013; VIDAL; PIRES; GUIMARÃES, 2013).

We have to consider that most of the tree species sampled in the studied landscape had small and medium seed sizes, followed by large and very large (see results). Smaller seed sizes can be dispersed by a huge variety of dispersers, from small bodied species to the larger ones (DONATTI et al., 2011). Also, the effects of landscape changes do not affect the abundance of these species, since most of these animal dispersers are not influenced by the habitat alteration and isolation effects and are able to use different habitats in the landscape, such as primary forest and matrix (GALETTI; PIZO, 1996; FARIA et al., 2007; MEDINA et al., 2007; PARDINI et al., 2009). This explains why the tree species with smaller seed size are more broadly distributed in the fragmented landscape.

5 CONCLUSION AND IMPLICATIONS FOR CONSERVATION

Our results showed that zoochory was the main dispersion type even in fragmented landscapes and was effective for the abundance and distribution of tree zoochoric species in the entire landscape. These findings reinforce the importance of frugivores as seed dispersers and for dispersion of plants. Furthermore, the higher abundance of trees species with bigger fruits in the sampled landscape indicates that the disperser fauna is composed mainly of species with large body size, since the body size of the disperser and the fruit size are positively correlated (DONATTI et al., 2011). Also, higher abundance of these frugivores, such as large/medium mammals and birds, is found in the control forest sampled in comparison with the small fragments and eucalyptus plantations (CHIARELLO, 1999; MARSDEN; WHIFFIN; GALETTI, 2001). As such, our results indicate that large frugivores are key components of tree dispersion in this fragmented landscape. The loss of keystone frugivores has drastic consequences for tropical forest dynamics (PERES; PALACIOS, 2007; FLEMING; KRESS, 2011) since it has a direct and negative influence on plant recruitment, species richness and diversity of plant species (REDFORD, 1992; GALETTI et al., 2006; CORLETT, 2011; BUENO et al., 2013; KURTEN, 2013). Over the long term, the absence of keystone frugivores can lead to the local extinction of tree species with large fruits and their replacement by species with small fruits (CARDOSO; TABARELLI, 2000), regional extinction of tree species (GALETTI et al., 2006) and, can also lead to evolutionary losses in important plant traits (GALETTI et al., 2013; GALETTI; DIRZO, 2013). Furthermore, these findings, together, highlight the conservation importance of this control forest, because large frugivores and top predators are locally extinct in most of the Atlantic Forest areas (CANALE et al., 2012; JORGE et al., 2013).

Moreover, we found that some traits of zoochoric trees, such as fruit type and size, found in the control forest were shared with the linear remnants connected to this control forest (for example, the abundance of fleshy fruits and medium fleshy fruits). Being that, we can conclude that the control forest is acting as a source of frugivores and they are moving thought these connected linear remnants (including the large body species) and consequently contributing to the functional and ecological connectivity on the fragmented landscapes, maintaining the plant-animal disperser relation.

Also, some results – such as zoochoric tree dispersion, abundance of species with very large fleshy fruits and large non-fleshy fruits, did not differ between linear remnants connected near and unconnected linear remnants, the absence of significant influence of spatial arrangement of linear remnants on the abundance of tree species with medium, large and very large seed size and others - indicate that the matrix should be permeable, because the larger dispersers are heavily affect by the forest fragment size and isolation (CHIARELLO, 1999; RIBON; SIMON; MATTOS, 2003; UEZU; METZGER; VIELLIARD, 2005; CRAMER; MESQUITA, WILLIAMSON, 2007; JORGE et al., 2013; VIDAL; PIRES; GUIMARÃES, 2013). However, the matrix permeability must be evaluated carefully. Besides recording one of the large frugivore species in the matrix of the eucalyptus plantation (CENTODUCATTE et al., 2011) most of the studies in the same landscape have shown that the mammals and bird species are negatively affected by the habitat loss and fragmentation (CHIARELLO, 1999; MARSDEN; WHIFFIN; GALETTI, 2001). However, these findings show that the occasional movement of frugivores throught the matrix must occur and contribute to dispersion of zoochoric trees in this fragmented landscape.

In conclusion, we can point out the importance of structural connectivity in this fragmented landscape, enabling the disperser species movement among forest fragments and connecting plant-animal populations across the fragmented landscape (see TEWKSBURY et al., 2002; LEVEY et al., 2005; BRUDVIG et al., 2009; MAGRACH; LARRINAGA; SANTAMARIA, 2012). Thus the conservation of linear remnants in fragmented landscapes needs be encouraged, considering the surrounding matrix management as well, to ensure the maintenance of plant-animal mutualistic interactions and consequently, the conservation of native biota in the entire landscape (see TSCHARNTKE et al., 2012). These conservation strategies should be implemented as soon as possible, because forest remnants are very small and isolated in the Brazilian Atlantic Forest, which has a negative impact on animal-plant interactions (TEWKSBURY et al., 2002; ANDREAZZI; PIRES; FERNANDEZ, 2009; BRUDVIG et al., 2009; HAGEN et al., 2012; MAGRACH; LARRINAGA; SANTAMARIA, 2012) and makes the remaining forest areas insufficient to maintain key ecological processes (SILVA; TABARELLI, 2001).

In deed, we cannot forget the high importance of primary and control forests for biodiversity conservation (CHIARELLO, 1999; MARSDEN; WHIFFIN; GALETTI, 2001; LOUZADA et al., 2010; GIBSON et al., 2011), and as a biodiversity source for other fragments in fragmented landscapes, thought spillover effects (see results from Chapter 1 and COOK et al., 2002; BRUDVIG et al., 2009). We are just searching for viable solutions to manage the strong impact of isolation in fragmented landscapes where these primary and control forest no longer exist or are relics of what is left, immersed in a heavily fragmented landscape, which consists predominantly of agricultural matrices and fragments with different sizes and shapes, with different degrees of isolation.

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7 SUPPLEMENTARY MATERIAL

7.1 Tables

Table A.1 Mean values (±SD) of independent variables used to characterize the structural and landscape variables of sampled treatments on a Tableland Forest in southeastern Brazil. Label: interior of control forest (CFi), edges of control forest (CFe), linear remnants connected near (CRn), linear remnants connected far (CRf) and linear remnants unconnected (UC).

	Structural variables			**Landscape variables**			
Sampling sites	**Size (ha)**	**Mean Width (m)**	**Shape (m)**	**Amount of native forest around (ha)**	**Amount of native forest (ha)**	**Distance to source fragment (m)**	**Mean distance to neighbors fragments (m)**
CFi	16,48±7,75	-	0.00149± 7.84	1.89±0.90	269.83±29.59	-	-
CFe	16,48±7,75	-	0.00149± 7.84	21.06±37.32	189.33±46.03	-	-
CRn	22.8±14.27	75.0±18.58	0.038±0.00608	168.95±65.49	177.59±63.21	95.75±106.3	767.30±218.54
CRf	22.8±14.27	75.0±18.58	0.038±0.00608	139.33±74.78	149.82±71.97	347.9±156.2	815.80±122.17
UC	12.5±7.06	93±50.73	0.0365±0.01689	25.44±14.91	34.68±13.13	1,277±775	1,452±363.45

Table B.1 List of zoochoric tree species sampled on a Tableland Forest in southeastern Brazil, with the abundance in each treatment and its classification into each attribute, considering the fruit type and size and the seed size. Label Label: CFi = interior of control forest; CFe = edge of control forest; CRn = linear remnants connected near; CRf = linear remnants connected far; UC = linear remnants unconnected. Lable: For fruit type: F=Fleshy fruit; NF=Non-fleshy fruit; For fruit and seed size: S=Small size; M=Medium size; L=Large size and VL=Very large size.

Zoochoric species	**Sampled treatments**					**Fruit type**		**Fruit size**				**Seed size**			
	CFi	**CFe**	**CRn**	**CRf**	**UC**	**F**	**NF**	**S**	**M**	**L**	**VL**	**S**	**M**	**L**	**VL**
Alchornea triplinervia (Spreng.) Müll.Arg.	1	-	-	-	-	-	X	X	-	-	-	X	-	-	-
Allophylus petiolulatus Radlk.	1	-	-	-	-	X	-	X	-	-	-	X	-	-	-
Amphirrhox longifolia (A.St.-Hil.) Spreng	1	-	-	1	3	X	-	-	-	X	-	-	X	-	-
Andira fraxinifolia Benth.	-	-	2	2	-	X	-	-	-	-	X	-	-	-	X
Aniba firmula (Nees & C. Mart.) Mez	-	-	1	-	-	X	-	-	X	-	-	-	X	-	-
Annona cacans Warm.	-	-	-	-	2	X	-	-	-	-	X	-	X	-	-
Annona dolabripetala Raddi	-	-	6	-	3	X	-	-	-	-	X	-	X	-	-
Astrocaryum aculeatissimum (Schott) Burret	7	5	4	3	7	X	-	-	-	-	X	-	-	-	X
Bactris ferruginea Burret	-	-	-	-	1	X	-	-	X	-	-	-	X	-	-
Beilschmiedia linharensis Sachiko Nishida & H.van der Werff	1	-	-	-	-	X	-	-	-	-	X	-	-	-	X
Bixa arborea Huber	1	6	-	-	23	-	X	-	-	-	X	X	-	-	-
Brosimum glaucum Taub.	3	5	1	2	2	X	-	-	X	-	-	-	X	-	-
Brosimum glaziovii Taub.	-	1	2	-	-	X	-	X	-	-	-	X	-	-	-
Byrsonima cacaophila W.R. Anderson	2	1	-	2	1	X	-	-	X	-	-	-	X	-	-
Byrsonima sericea DC.	-	-	-	1	-	X	-	X	-	-	-	X	-	-	-
Byrsonima stipulacea (Juss.) Nied.	-	2	1	1	-	X	-	-	X	-	-	-	X	-	-
Cabralea canjerana (Vell.) Mart. subsp.	-	-	-	-	1	-	X	-	-	X	-	-	X	-	-

canjerana

“Table B.1, continue”

Calophyllum brasiliense Cambess.	-	-	-	3	-	X	-	-	-	X	-	-	X	-	-
Calyptranthes lucida var. *polyantha* (Berg) C.D.Legrand	6	1	1	2	-	X	-	-	X	-	-	X	-	-	-
Calyptranthes sp.1	-	-	1	-	-	X	-	-	X	-	-	-	X	-	-
Campomanesia espiritosantensis Landrum	1	-	-	-	-	X	-	-	X	-	-	X	-	-	-
Campomanesia guazumifolia (Cambess.) O.Berg	-	-	-	-	1	X	-	-	-	X	-	-	X	-	-
Carpotroche brasiliensis (Raddi.) A. Gray	4	1	-	-	-	X	-	-	-	-	X	X	-	-	-
Caryodendron grandifolium Pax	3	-	-	-	-	X	-	-	-	-	X	-	-	X	-
Casearia arborea (L.C.Richard) Urban	3	-	-	-	1	-	X	-	X	-	-	X	-	-	-
Casearia commersoniana Cambess.	1	-	-	1	-	-	X	-	X	-	-	X	-	-	-
Casearia decandra Jacq.	-	1	-	-	-	-	X	-	X	-	-	-	X	-	-
Casearia javitensis H.B. & K.	2	-	-	-	-	-	X	-	X	-	-	X	-	-	-
Casearia sp. new	-	1	1	2	-	-	X	X	-	-	-	X	-	-	-
Casearia sp.2	-	1	-	-	-	-	X	X	-	-	-	X	-	-	-
Casearia ulmifolia Vahl. ex Vent.	-	3	-	-	-	-	X	X	-	-	-	X	-	-	-
Cecropia glaziovi Snethl.	-	1	-	-	3	X	-	X	-	-	-	X	-	-	-
Cecropia hololeuca Miq.	-	1	-	1	1	X	-	X	-	-	-	X	-	-	-
Cecropia pachystachya Trécul.	-	-	-	2	-	X	-	X	-	-	-	X	-	-	-
Chrysophyllum januariense Eichler	-	-	1	1	-	X	-	-	X	-	-	-	X	-	-
Chrysophyllum lucentifolium Cronquist	-	2	-	-	-	X	-	-	-	X	-	-	-	X	-
Citharexylum laetum Hiern.	-	1	-	-	-	X	-	-	X	-	-	X	-	-	-
Citronella paniculata (Mart.) Howard	1	-	-	-	1	X	-	-	-	X	-	-	X	-	-
Clarisia ilicifolia (Spreng.) Lanj. &	1	-	-	-	-	X	-	-	-	X	-	-	-	X	-

Rossb.															
Clarisia racemosa Ruiz & Pav.	1	-	1	2	-	X	-	-	-	X	-	-	-	X	-
Coccoloba tenuiflora Lindau	-	-	1	-	-	X	-	-	X	-	-	X	-	-	-
Coccoloba warmingii Meisn	-	1	-	-	-	X	-	-	X	-	-	X	-	-	-
Copaifera langsdorffii Desf.	1	-	-	1	-	X	-	-	-	-	X	-	-	X	-
Copaifera lucens Dwyer	-	-	-	-	1	X	-	-	-	-	X	-	-	X	-
Cordia acutifolia Fresen.	-	-	-	1	2	X	-	-	-	X	-	-	-	X	-

“Table B.1, continue”

Cordia ecalyculata Vell.	-	1	-	-	1	X	-	-	-	X	-	-	-	X	-
Cordia magnoliaefolia Cham.	-	-	-	-	3	X	-	-	-	X	-	-	X	-	-
Cordia sp.1	-	-	8	1	-	X	-	-	-	X	-	-	-	X	-
Cordia trichoclada DC.	-	1	1	-	1	X	-	-	-	X	-	-	X	-	-
Couepia schottii Fritsch	-	-	1	-	-	X	-	-	-	X	-	-	-	X	-
Crepidospermum atlanticum D.C. Daly	-	1	1	2	1	X	-	-	-	X	-	-	X	-	-
Cupania bracteosa Radlk.	-	1	-	-	-	-	X	-	-	X	-	-	X	-	-
Cupania cf. *scrobiculata* L.C. Rich.	4	4	-	1	3	-	X	-	-	X	-	-	X	-	-
Cupania rugosa Radlk.	-	1	-	-	-	-	X	-	-	X	-	-	X	-	-
Cupania sp.1	1	-	-	-	-	-	X	-	-	X	-	-	X	-	-
Cybianthus brasiliensis (Mez) G.Agostini	2	-	-	-	-	X	-	X	-	-	-	X	-	-	-
Drypetes sp.1	-	1	-	-	1	X	-	-	-	X	-	-	X	-	-
Duguetia chrysocarpa Maas	-	-	1	-	-	X	-	-	-	-	X	-	X	-	-
Ecclinusa ramiflora Mart.	3	3	1	1	1	X	-	-	-	-	X	-	-	X	-
Ephedranthus dimerus J.C. Lopes , Chatrou & Mello-Silva	1	-	-	-	-	X	-	-	-	X	-	-	-	X	-
Eschweilera ovata (Cambess.) Miers	1	2	1	3	-	-	X	-	-	X	-	-	-	X	-
Eugenia bahiensis DC	3	3	-	-	1	X	-	-	-	X	-	-	X	-	-
Eugenia batingabranca Sobral	2	1	-	-	-	X	-	-	-	X	-	-	X	-	-
Eugenia beaurepaireana (Kiaersk.) D.Legrand	-	-	-	-	1	X	-	-	-	X	-	-	X	-	-
Eugenia cf. *badia* O.Berg	-	2	-	-	-	X	-	-	X	-	-	-	X	-	-
Eugenia cf. *tinguyensis* Cambess.	3	2	1	1	-	X	-	-	X	-	-	-	X	-	-
Eugenia excelsa O.Berg	1	7	-	-	-	X	-	X	-	-	-	X	-	-	-
Eugenia fluminensis Berg	-	-	1	-	-	X	-	-	-	X	-	-	X	-	-

Eugenia involucrata DC.	1	1	-	-	-	X	-	-	-	X	-	X	-	-	-
Eugenia platyphylla O.Berg	10	7	-	-	-	X	-	-	X	-	-	X	-	-	-

"Table B.1, continue"

Eugenia plicatocostata O.Berg	-	1	-	-	-	X	-	-	X	-	-	-	X	-	-
Eugenia prasina O.Berg	2	1	2	-	1	X	-	-	X	-	-	-	X	-	-
Eugenia sp.new	1	-	-	-	-	X	-	-	-	X	-	-	-	X	-
Eugenia subterminalis DC.	4	1	3	-	-	X	-	-	X	-	-	X	-	-	-
Euterpe edulis Mart.	-	-	2	-	-	X	-	-	X	-	-	-	X	-	-
Exellodendron gracile (Kuhlmann) Prance	-	-	1	-	1	X	-	-	-	X	-	-	X	-	-
Ficus gomelleira Kunth & C.D. Bouché	1	-	-	1	1	X	-	-	X	-	-	X	-	-	-
Ficus mariae C.C. Berg, Emygdio & Carauta	1	-	-	-	-	X	-	-	X	-	-	X	-	-	-
Ficus nymphaeifolia Mill.	-	1	-	-	-	X	-	-	X	-	-	X	-	-	-
Ficus pulchella Schott	-	-	1	-	-	X	-	-	-	X	-	X	-	-	-
Geissospermum laeve (Vell.) Baill.	2	-	-	-	2	X	-	-	-	-	X	-	-	X	-
Glycydendron espiritosantense Kuhlm.	-	1	-	-	-	X	-	-	-	-	X	-	-	X	-
Guapira noxia (Netto) Lundell	3	-	1	-	-	X	-	-	X	-	-	-	X	-	-
Guapira opposita (Vell.) Reitz	-	5	-	1	1	X	-	-	X	-	-	-	X	-	-
Guapira venosa (Choisy) Lundell	2	-	-	-	1	X	-	-	X	-	-	-	X	-	-
Guarea aff. juglandiformis Pennington	-	1	-	-	-	X	-	-	-	X	-	-	X	-	-
Guarea penningtoniana Pinheiro	2	1	1	1	1	X	-	-	-	X	-	-	X	-	-
Guatteria australis A.St.-Hil.	-	1	-	-	-	X	-	-	X	-	-	-	X	-	-
Guatteria villosissima A. St.-Hil.	-	1	-	-	-	X	-	X	-	-	-	X	-	-	-
Guettarda angelica Mart. ex Müell. Arg.	-	1	-	-	-	X	-	-	X	-	-	X	-	-	-
Helicostylis tomentosa (Poep. et Endl.) Rusby	1	7	-	5	3	X	-	-	-	X	-	X	-	-	-
Hirtella hebeclada Moric. ex A. P. DC.	-	-	-	2	-	X	-	-	-	X	-	-	-	X	-
Hymenaea rubriflora Ducke var. rubriflora	-	-	-	-	1	-	X	-	-	-	X	-	-	X	-
Inga aff. cylindrica (Vell.) Mart.	-	1	1	-	-	-	X	-	-	-	X	-	X	-	-

Inga capitata Desv.	-	-	1	-	-	-	X	-	-	-	X	-	X	-	-
Inga flagelliformis (Vell.) Mart.	3	1	2	1	1	-	X	-	-	-	X	-	X	-	-
Inga lanceifolia Benth.	1	-	-	-	-	-	X	-	-	-	X	-	X	-	-

“Table B.1, continue”

Inga striata Benth.	-	-	2	1	3	-	X	-	-	-	X	-	X	-	-
Inga subnuda Salzm. ex Benth. subsp. subnuda	-	-	-	-	2	-	X	-	-	-	X	-	-	X	-
Joannesia princeps Vell.	2	3	13	1	13	-	X	-	-	-	X	-	-	-	X
Lacistema recurvum Schnizl.	3	-	-	-	1	-	X	X	-	-	-	X	-	-	-
Lecythis lanceolata Poir.	2	1	1	-	-	-	X	-	-	-	X	-	-	-	X
Lecythis lurida (Miers) S.A.Mori	1	-	-	-	1	-	X	-	-	-	X	-	-	-	X
Lecythis pisonis Cambess.	-	1	1	-	-	-	X	-	-	-	X	-	-	-	X
Licania belemii Prance	1	-	-	1	-	X	-	-	-	X	-	-	-	X	-
Licania heteromorpha Benth. var. heteromorpha	-	-	2	-	-	X	-	-	-	X	-	-	-	X	-
Licania kunthiana Hook.f.	2	1	2	2	3	X	-	-	-	X	-	-	-	X	-
Licaria bahiana Kutz	1	-	2	-	1	X	-	-	-	X	-	-	X	-	-
Manilkara elata (Allemão ex Miq.) Monach.	-	-	-	-	1	X	-	-	-	X	-	-	X	-	-
Manilkara salzmannii (A.DC.) H.J.Lam	-	-	-	-	2	X	-	-	-	X	-	-	X	-	-
Maprounea cf. guianensis Aubl.	-	-	2	-	-	-	X	X	-	-	-	X	-	-	-
Marlierea excoriata Mart.	1	1	-	-	-	X	-	-	X	-	-	X	-	-	-
Marlierea grandifolia O. Berg	-	-	-	2	-	X	-	-	X	-	-	-	X	-	-
Marlierea obversa Legrand.	2	-	-	-	-	X	-	-	X	-	-	-	X	-	-
Marlierea sucrei G.M. Barroso et Peixoto	1	-	-	-	-	X	-	-	X	-	-	-	X	-	-
Matayba guianensis Aubl.	-	-	1	-	1	-	X	-	X	-	-	X	-	-	-
Maytenus cestrifolia Reiss.	-	1	-	-	-	-	X	-	-	X	-	X	-	-	-
Maytenus multiflora Reiss.	-	-	-	-	1	-	X	-	-	X	-	X	-	-	-
Melanopsidium nigrum Colla	1	-	-	-	-	X	-	-	-	X	-	X	-	-	-
Miconia cf. cinnamomifolia (DC.) Naudin	-	-	3	4	-	X	-	X	-	-	-	X	-	-	-
Micropholis gardneriana (A.DC.) Pierre	-	-	1	-	-	X	-	-	X	-	-	-	X	-	-

Monilicarpa brasiliana (Banks ex DC.) Cornejo & Iltis	1	-	-	-	-	-	X	-	-	-	X	-	X	-	-

“Table B.1, continue”

Mouriri sp.1	-	-	1	-	-	X	-	-	-	X	-	-	-	X	-
Myrcia brasiliensis Kiaersk.	-	-	3	2	-	X	-	-	X	-	-	X	-	-	-
Myrcia fallax DC.	-	-	1	2	1	X	-	X	-	-	-	X	-	-	-
Myrcia follii G.M. Barroso & Peixoto	2	1	-	-	-	X	-	-	X	-	-	X	-	-	-
Myrcia lineata (Berg) G.M. Barroso	1	1	1	-	1	X	-	-	X	-	-	X	-	-	-
Myrcia rufipes DC.	1	-	-	-	-	X	-	-	X	-	-	X	-	-	-
Myrciaria aureana Mattos	1	-	-	-	-	X	-	-	-	X	-	X	-	-	-
Naucleopsis oblongifolia (Kuhlm.) Carauta	3	3	-	-	-	X	-	-	-	-	X	-	-	X	-
Neomitranthes langsdorffii (O.Berg) J.R. Mattos	-	-	1	-	-	X	-	X	-	-	-	X	-	-	-
Ocotea confertiflora (Meisn.) Mez	3	5	4	2	1	X	-	-	X	-	-	-	X	-	-
Ocotea divaricata (Nees & Mart.) Mez	-	-	1	-	-	X	-	-	-	X	-	-	-	X	-
Ocotea glauca (Nees & Mart.) Mez	1	-	-	-	-	X	-	-	X	-	-	-	X	-	-
Ocotea lancifolia (Schott) Mez	2	1	1	4	1	X	-	-	-	X	-	-	X	-	-
Ocotea odorifera (Vell.) Rohwer	1	-	-	-	-	X	-	-	-	X	-	-	X	-	-
Ocotea prolifera (Nees & Mart.) Mez	1	1	-	1	2	X	-	-	-	X	-	-	X	-	-
Ocotea pulchella (Nees) Mez.	-	-	1	1	-	X	-	-	X	-	-	X	-	-	-
Ocotea sp.1	-	-	3	-	-	X	-	-	X	-	-	X	-	-	-
Ormosia nitida Vogel	-	-	-	-	1	-	X	-	-	X	-	-	X	-	-
Oxandra reticulata Maas	1	-	-	1	-	X	-	-	X	-	-	-	X	-	-
Pachira stenopetala Casar.	-	1	-	-	-	-	X	-	-	-	X	-	X	-	-
Parinari parvifolia Sandw.	-	-	1	-	-	X	-	-	-	-	X	-	-	X	-
Pera furfuracea Müll.Arg.	-	-	9	5	-	-	X	-	X	-	-	-	X	-	-

Pera glabrata (Schott) Baill.	1	1	-	-	4	-	X	-	X	-	-	X	-	-	-
Picramnia ramiflora Planch.	1	-	-	-	-	X	-	-	-	X	-	-	-	X	-
Plinia renatiana G.M.Barroso & Peixoto	-	1	4	-	-	X	-	-	-	X	-	-	-	X	-

"Table B.1, continue”

Polyandrococos caudescens (Mart.) Barb. Rodr.	11	10	2	1	1	X	-	-	-	-	X	-	-	-	X
Posoqueria latifolia (Rudge) Roem & Schult.	-	1	-	-	-	X	-	-	-	-	X	-	X	-	-
Pouteria aff. bapeba T.D.Pennington	-	4	1	-	-	X	-	-	-	X	-	-	-	X	-
Pouteria aff. filipes Eyma	2	-	-	-	-	X	-	-	-	X	-	-	-	X	-
Pouteria bangii (Rusby) T.D.Pennington	1	2	-	2	-	X	-	-	-	X	-	-	-	X	-
Pouteria bullata (S.Moore) Baehni	-	1	-	-	-	X	-	-	-	X	-	-	-	X	-
Pouteria cuspidata (A.DC.) Baehni	-	-	-	1	-	X	-	-	-	X	-	-	X	-	-
Pouteria durlandii (Standl.) Baehni	1	-	-	-	-	X	-	-	-	X	-	-	-	X	-
Pouteria macrophylla (Lam) Eyma	-	-	-	-	1	X	-	-	-	X	-	-	-	X	-
Pouteria pachycalyx T.D. Penn.	1	-	-	-	-	X	-	-	-	X	-	-	-	X	-
Pouteria venosa subsp. *amazonica* T.D.Pennington	1	4	-	-	2	X	-	-	-	X	-	-	-	X	-
Pradosia lactescens (Vellozo) Radlk.	-	3	1	-	1	X	-	-	-	X	-	-	-	X	-
Protium brasiliense (Spreng.) Engl.	-	-	-	-	1	X	-	-	-	X	-	-	-	X	-
Protium glaziovii Swart	-	-	-	3	-	X	-	-	-	X	-	-	-	X	-
Protium heptaphyllum (Aubl.) Marchand.	1	2	7	1	-	X	-	-	-	X	-	-	X	-	-
Protium warmingianum Marchand	2	1	1	1	-	X	-	-	-	X	-	-	X	-	-
Pseudima frutescens (Aubl.) Radlk.	-	1	-	-	-	X	-	-	-	X	-	-	-	X	-
Pseudoxandra spiritus-sancti Maas	1	-	-	-	1	X	-	-	X	-	-	X	-	-	-
Psidium oblongatum O.Berg	-	1	-	1	1	X	-	-	-	X	-	X	-	-	-
Quararibea penduliflora (A.St.Hil.) K. Schum.	3	2	-	-	1	X	-	-	-	X	-	-	-	X	-
Randia armata D.C.	4	-	4	-	-	X	-	-	-	X	-	X	-	-	-
Rauvolfia capixabae I. Koch & Kin.-Gouv.	1	-	-	1	-	X	-	-	-	X	-	X	-	-	-
Rheedia gardneriana Triana & Planch.	-	-	2	1	-	X	-	-	X	-	-	X	-	-	-

Table B.1, continue”

Rinorea bahiensis (Moric.) Kuntze	12	9	9	2	3	-	X	X	-	-	-	X	-	-	-
Rouria sp.	1	-	-	-	-	-	X	X	-	-	-	X	-	-	-
Sapium glandulatum (Vell.) Pax.	-	-	-	1	-	-	X	X	-	-	-	X	-	-	-
Schefflera morototoni (Aubl.) Maguire, Steyermark & Frodin	1	-	-	-	3	X	-	X	-	-	-	X	-	-	-
Schoepfia brasiliensis A. DC.	1	-	-	-	-	X	-	-	X	-	-	X	-	-	-
Schoepfia obliquifolia Turcz.	2	-	1	-	2	X	-	-	-	X	-	-	-	X	-
Senefeldera multiflora Mart.	10	9	12	14	20	-	X	-	X	-	-	X	-	-	-
Simaruba amara Aubl.	-	-	3	2	-	X	-	-	-	X	-	-	X	-	-
Siparuna reginae (Tul.) A. DC.	-	-	-	-	3	X	-	-	-	X	-	X	-	-	-
Sloanea eichleri K. Schum.	-	1	1	2	-	-	X	-	-	X	-	-	X	-	-
Solanum pseudo-quina A.St.-Hil.	-	-	2	2	-	X	-	-	-	X	-	X	-	-	-
Solanum sooretamum Carvalho	-	1	1	5	1	X	-	-	X	-	-	X	-	-	-
Sorocea guilleminiana Gaudich.	9	2	2	-	-	X	-	-	X	-	-	X	-	-	-
Spondias macrocarpa Engl.	1	2	-	-	-	X	-	-	-	-	X	-	-	X	-
Stephanopodium blanchetianum Baill.	1	1	-	1	-	X	-	-	X	-	-	-	X	-	-
Sterculia speciosa Ducke	-	1	-	2	-	-	X	-	-	-	X	-	-	-	X
Swartzia apetala Raddi	2	-	-	-	-	-	X	-	-	-	X	-	-	X	-
Swartzia simplex var. *continentalis* Urban	-	2	-	-	-	-	X	-	-	-	X	-	-	X	-
Syagrus botryophora (Mart.) Mart.	-	1	-	-	-	X	-	-	-	-	X	-	-	X	-
Syzygium cumini (L.) Skeels	-	-	-	1	-	X	-	-	-	-	X	-	-	-	X
Tabernaemontana salzmanni A. DC.	1	-	-	-	1	X	-	-	-	-	X	-	X	-	-
Talisia intermedia Radlk.	-	1	-	-	-	X	-	-	-	X	-	-	-	X	-
Tapirira guianensis Aubl.	5	-	28	41	3	X	-	-	-	X	-	-	X	-	-
Thyrsodium spruceanum Benth.	-	1	-	1	-	X	-	-	-	X	-	-	-	X	-
Trichilia aff. *surumuensis* C.DC.	1	-	-	-	-	-	X	-	-	X	-	-	X	-	-
Trichilia casaretti C.DC.	1	1	-	-	-	-	X	-	-	X	-	-	X	-	-

Trichilia lepidota subsp. *schumanniana* (Harms) T.D.Pennington	2	4	3	4	-	-	X	-	X	-	-	-	X	-	-

“Table B.1, conclusion”

Vatairea heteroptera (Allem.) Ducke ex de Assis Iglesias	-	1	-	-	-	-	X	-	-	-	X	-	-	X	-
Vataireopsis araroba (Aguiar) Ducke	-	1	-	-	-	-	X	-	-	-	X	-	-	X	-
Virola gardneri (A.DC.) Warb.	2	3	4	3	2	-	X	-	-	X	-	-	-	X	-
Vismia aff. *martiana* Reichardt.	-	-	2	-	-	X	-	X	-	-	-	X	-	-	-
Vitex orinocensis Kunth	-	1	1	-	1	X	-	-	X	-	-	-	X	-	-
Xylopia ochrantha Mart.	-	1	-	-	-	-	X	-	-	X	-	X	-	-	-
Xylopia sericea A. St.-Hil.	-	-	3	16	3	-	X	-	-	X	-	X	-	-	-
Zanthoxylum acuminatum (Sw.) Sw.	-	1	-	-	1	-	X	X	-	-	-	X	-	-	-
Zollernia modesta A.M.de Carvalho & R.C.Barneby	-	-	-	2	-	X	-	-	-	X	-	-	X	-	-

Table B.2 Total abundance (mean ± SD) of tree zoochoric species for each attribute (fruit type and size and seed size) in the sampled treatments on a Tableland Forest in southeastern Brazil Label: CFi = interior of control forest; CFe = edge of control forest; CRn = linear remnants connected near; CRf = linear remnants connected far; UC = linear remnants unconnected. F=Fleshy fruit; NF=Non-fleshy fruit; For fruit and seed size: S=Small size; M=Medium size; L=Large size and VL=Very large size.

		Sampled treatments				
		CFi	**CFb**	**CRn**	**CRf**	**UC**
Fruit type	**F**	154 (30.8 ± 9.34) ^{a}	131 (26.2 ± 7.76) ^{a}	138 (27.6 ± 10.97) ^{a}	128 (25.6 ± 9.66) ^{ab}	82 (16.4 ± 10.76)^{b}
	NF	60 (12 ± 4.74) ^{a}	64 (12.8 ± 4.97) ^{a}	67 (13.4 ± 8.47) ^{a}	60 (12 ± 7.78) ^{a}	89 (17.8 ± 9.65) ^{a}
Fleshy fruit size	**S**	5 (1 ± 1) ^{a}	11 (2.2 ± 3.35) ^{a}	9 (1.8 ± 1.79) ^{a}	10 (2 ± 1.87) ^{a}	8 (1.6 ± 1.67) ^{a}
	M	63 (12.6 ± 5.41) ^{a}	46 (9.2 ± 4.09) ^{a}	35 (7 ± 4.47) ^{b}	26 (5.2 ± 4.09) ^{bc}	13 (2.6 ± 1.95) ^{c}
	L	49 (9.8 ± 4.02) ^{a}	47 (9.4 ± 5.41) ^{a}	69 (13.8 ± 5.81) ^{a}	91 (18.2 ± 10.94) ^{a}	43 (8.6 ± 8.05) ^{a}
	VL	37 (7.4 ± 4.39) ^{a}	27 (5.4 ± 2.07) ^{ab}	17 (3.4 ±2.51) ^{bc}	9 (1.8 ± 1.30) ^{c}	18 (3.6 ± 2.88) ^{bc}
Non-flesfy fruit size	**S**	17 (3.4 ± 2.79) ^{a}	15 (3 ± 2) ^{a}	12 (2.4 ± 3.21) ^{a}	5 (1 ± 1) ^{a}	5 (1 ± 1) ^{a}
	M	19 (3.8 ± 1.30) ^{a}	15 (3 ± 2.55) ^{a}	25 (5 ± 2.74) ^{a}	24 (4.8 ± 5.36) ^{a}	26 (5.2 ± 5.36) ^{a}
	L	11 (2.2 ± 0.84) ^{a}	15 (3 ± 1.87) ^{ab}	11 (2.2± 2.28) ^{a}	24 (4.8 ± 2.49) ^{b}	14 (2.8 ± 1.79) ^{a}
	VL	13 (2.6 ± 1.95) ^{ab}	19 (3.8 ± 1.92) ^{a}	21 (4.2± 4.76) ^{a}	5 (1 ± 1.22) ^{b}	44 (8.8 ± 12.32) ^{a}
Seed size	**S**	92 (18.4 ± 6.58) ^{a}	73 (14.6 ± 6.47) ^{a}	62 (12.4 ± 8.47) ^{a}	67 (13.4 ± 9.07) ^{a}	77 (15.4 ± 9.96) ^{a}
	M	61 (12.2 ± 5.45) ^{a}	58 (11.6 ± 6.19) ^{a}	90 (18 ± 14.71) ^{a}	88 (17.6 ± 7.09) ^{a}	49 (9.8 ± 6.38) ^{a}
	L	37 (7.4 ± 3.44) ^{a}	43 (8.6 ± 4.56) ^{a}	30 (6 ± 4.18) ^{a}	23 (4.6 ± 5.41) ^{a}	23 (4.6 ± 3.58) ^{a}
	VL	24 (4.8 ± 3.83) ^{ab}	21 (4.2 ± 2.17) ^{ab}	23 (4.6 ± 4.22) ^{a}	10 (2 ± 1.58) ^{b}	22 (4.4 ± 3.78) ^{ab}

Different letter in each line denote significative results at $p<0.05$, obtained throught pairwise comparisons.

Table C.1 Best models results from Generalized linear mixed models evaluating the influences of structural features and structural connectivity on abundance of tree species with zoochoric dispersion and with different fruit type, size and seed size on a Tableland Forest in southeastern Brazil. Values inside the brackets shows the coefficient estimates and standard errors for each selected model. All treatments were considered in these analysis.

	Structural features of treatments		**Structural connectivity**			
	Size	**Shape**	**Distance to source fragment**	**Mean distance to neighbors fragments**	**Amount of native forest**	**Amount of native forest around**
Abundance of zoochoric species	0.05 (0.05)^{ns}	-5.49 (3.63)^{ns}	-0.04 (0.06)^{ns}	-0.04 (0.05)^{ns}	-0.05 (0.1)^{ns}	0.03 (0.19)^{ns}
Fruit type						
Fleshy fruits	0.10 (0.12)^{ns}	-9.3 (0.06)^{ns}	-0.13 (0.06)^{ns}	-0.10 (0.13)^{ns}	-	0.4 (0.11)^{ns}
Non fleshy fruits	-0.03 (0.08)^{ns}	-0.21 (6.17)^{ns}	0.09 (0.08)^{ns}	-	-0.5 (0.26)^{ns}	
Abundance of species with different fleshy fruits size						
Small fruits	5.53 (11.07)^{ns}	-0.01 (0.15)^{ns}	0.06 (0.17)^{ns}	0.08 (0.02)^{ns}	-0.2 (0.58)^{ns}	0.148 (0.3)^{ns}
Medium fruits	0.27 (0.08)**	-	-0.35 (0.01)**	-0.29 (0.08)**	-	-
Large fruits	-0.04 (0.11)^{ns}	-0.65 (8.67)^{ns}	0.02 (0.11)^{ns}	0.06 (0.11)^{ns}	0.06 (0.38)^{ns}	0.15 (0.18)^{ns}
Very large fruits	0.13 (0.08)^{ns}	-24.4 (6.9)**	-	-	-	-0.61 (0.19)**
Abundance of species with different non-fleshy fruits size						
Small fruits	0.24 (0.14)^{ns}	-21.14 (11.45)^{ns}	-0.3 (0.17)^{ns}	-0.27 (0.14)^{ns}	-	-0.36 (0.3)^{ns}
Medium fruits	-0.08 (0.13)^{ns}	2.35 (10.07)^{ns}	0.94 (0.14)^{ns}	0.08 (0.13)^{ns}	-0.52 (0.46)^{ns}	-0.17 (0.3)^{ns}

“Table C.1, conclusion”

Large fruits	-0.006 (0.09)^{ns}	5.17 (6.68)^{ns}	0.09 (0.10)^{ns}	0.07 (0.09)^{ns}	-0.02 (0.37)^{ns}	0.15 (0.18)^{ns}
Very large fruits	0.03 (0.19)^{ns}	-0.29 (14.29)^{ns}	-	0.002 (0.19)^{ns}	-0.66 (0.64)^{ns}	0.01 (0.39)^{ns}
Abundance of species with different seed size						
Small seed	-	-10.9 (5.15)^{ns}	-0.05 (0.09)^{ns}	-0.08 (0.07)^{ns}	-0.2 (0.31)^{ns}	-0.21 (0.14)^{ns}
Medium seed	-0.02 (0.09)^{ns}	0.73 (7.02)^{ns}	-0.02 (0.1)^{ns}	0.04 (0.09)^{ns}	0.31 (0.35)^{ns}	0.24 (0.17)^{ns}
Large seed	0.15 (0.11)^{ns}	-10.28 (8.54)^{ns}	-0.22 (0.13)^{ns}	-0.19 (0.11)^{ns}	-	-0.29 (0.21)^{ns}
Very large seed	0.08 (0.11)^{ns}	-6.22 (8.89)^{ns}	-0.08 (0.13)^{ns}	-0.08 (0.12)^{ns}	-0.09 (0.45)^{ns}	-0.2 (0.23)^{ns}

Significant results at * $p \leq 0.05$, ** and $p \leq 0.001$; ns = not significant result

Table E.1 Best models results from Generalized linear mixed models evaluating the influences of structural features and spatial arrangement of linear remnants on the abundance of tree species with zoochoric dispersion and with different fruit type, size and seed size on a Tableland Forest in southeastern Brazil. Values inside the brackets shows coefficient estimates and standard errors for each model. Significant results at * p≤ 0.05, ** and p≤ 0.001; ns = not significant result.

	Structural features of linear remnants			**Spatial arrangement of linear remnants**			
	Size	**Shape**	**Width**	**Distance to source fragment**	**Mean distance to neighbors fragments**	**Amount of native forest**	**Amount of native forest around**
Abundance of zoochoric species	0.3 (0.3)^{ns}	-17.3 (8.75)*	-	-	-	-	-
Abundance of species with different fruit type							
Fleshy fruits	0.4 (0.3)^{ns}	-16.5 (13)^{ns}	-	-0.3 (0.2)^{ns}	-0.7 (0.8)^{ns}	-	-
Non fleshy fruits	-	-13.5 (16)^{ns}	-	0.2 (0.2)^{ns}	-	-0.4 (0.4)^{ns}	-0.3 (0.4)^{ns}
Abundance of species with different fleshy fruits size							
Small	-	-	2.1 (1.4)^{ns}	-	-	-	-
Medium	-	-	-1.9 (1.6)^{ns}	-	-	0.7 (0.6)^{ns}	0.6 (0.5)^{ns}
Large	0.7 (0.5)^{ns}	-24.3 (19)^{ns}	1.3 (1.2)^{ns}	-0.2 (0.3)^{ns}	-0.9 (1)^{ns}	-	-
Very large	-	-40.3 (23)^{ns}	1.93 (1.38)^{ns}	-	2 (1.26)^{ns}	-0.74 (0.42)^{ns}	-0.83 (0.53) ^{ns}

“Table E.1, conclusion”

“Table E.1, conclusion”

Abundance of species with different non-fleshy fruits size							
Small	-	-	-	-	-	-	-
Medium	-	-	-	-	-	-	$-0.34\ (0.65)^{ns}$
Large	-	-	-	-	-	-	-
Very large	-	-	-	-	-	-	-
Abundance of species with different seed size							
Small	-	-38 (16.9)*	-	$0.4\ (0.3)^{ns}$	-	$-0.6\ (0.4)^{ns}$	$-0.5\ (0.4)^{ns}$
Medium	$0.5\ (0.4)^{ns}$	-	$1.4\ (1.1)^{ns}$	$-0.3\ (0.2)^{ns}$	-	$0.5\ (0.5)^{ns}$	$0.4\ (0.4)^{ns}$
Large	$-0.6\ (0.6)^{ns}$	-	$-2.7\ (1.6)^{ns}$	-	-	-	-
Very large	-	-	-	-	-	-	-

7.2 Figures

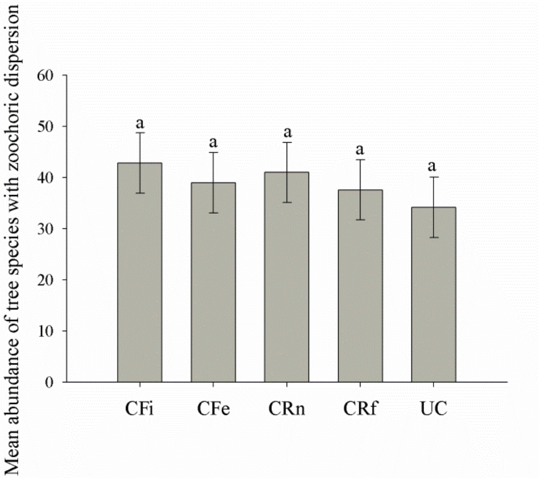
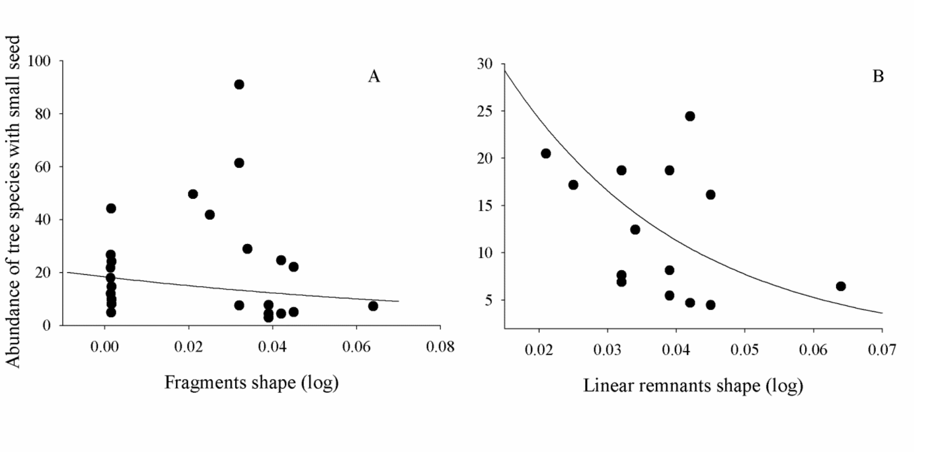


Figure A.1 The graph showing that the abundance of tree zoochoric species was not influenced by the habitat alteration on the Tableland Forest in southeastern Brazil. Equal letter denote no significant results at p≤0.05. Label: CFi = interior of control forest; CFe = edge of control forest; CRn = linear remnants connected near; CRf = linear remnants connected far; UC = linear remnants unconnected.



Figures B.1 Best models results from Generalized linear mixed models showing that the tree zoochoric with small seeds were influenced by the fragments shape (A) and linear remnants shape (B) on a Tableland Forest in southeastern Brazil. All the results were significant at p≤0.02