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Laboratório de Ecologia e Conservação - LECO

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INFLUÊNCIA DO USO DA TERRA NA DIVERSIDADE FUNCIONAL DE AVES, NA INSETIVORIA E NA PRODUÇÃO DE UVAS EM VINÍCOLAS NEOTROPICAIS

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Daniele Janina Moreno

Influência do uso da terra na diversidade funcional de aves, na insetivoria e na produção de uvas em vinícolas Neotropicais

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Certifico que a defesa realizou-se com a participação à distância do(s) membro(s) José Carlos Morante Filho e, depois das arguições e deliberações realizadas, o(s) participante(s) à distância está(ao) de acordo com o conteúdo do parecer da banca examinadora redigido neste relatório de defesa.

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A Daniel, Jane, Dani.

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“Além, muito além daquela serra, que ainda azula no horizonte, nasceu Iracema. Iracema, a virgem dos lábios de mel, que tinha os cabelos mais negros que a asa da graúna e mais longos que seu talhe de palmeira. O favo da jati não era doce como seu sorriso; nem a baunilha recendia no bosque como seu hálito perfumado. Mais rápida que a corça selvagem, a morena virgem corria o sertão e as matas do Ipu, onde campeava sua guerreira tribo, da grande nação tabajara. O pé grácil e nu, mal roçando, alisava apenas a verde pelúcia que vestia a terra com as primeiras águas.”

José de Alencar (Iracema-1865)

“Se eu vi mais longe, foi por estar de pé sobre ombros de gigantes.”

Isaac Newton

Resumo

A agricultura é um dos processos que mais alteram as paisagens naturais formando novas composições e configurações. Estas novas paisagens são compostas por fragmentos florestais e mosaicos de diferentes usos do solo permeadas pelas culturas agrícolas. Tais transformações nas paisagens trazem consequências na biodiversidade, não só em níveis taxonômicos, mas também em níveis funcionais, os quais podem ter implicações na provisão de serviços ecossistêmicos. Tendo tais conhecimentos, minha tese buscou elucidar as consequências dos diferentes usos da terra na diversidade taxonômica e funcional das aves em vinícolas no sudeste do Brasil, assim como na provisão de serviços ecossistêmicos executados por elas e por morcegos. Sendo assim, meu primeiro capítulo aborda a influência da paisagem em um gradiente de heterogeneidade e cobertura florestal sobre a diversidade das aves, e como posso inferir o modo em que se dá a estruturação das comunidades de aves por meio dos processos ecológicos. Os resultados mostram que áreas mais heterogêneas podem abrigar uma maior diversidade funcional de aves, mas não taxonômica. Filtros ambientais e similaridade limitante podem ser os processos que estruturam as comunidades em nível de diversidade (i.e. riqueza e abundância das espécies) dentro das vinícolas. No segundo capítulo, avaliei esse mesmo gradiente paisagístico no controle de pragas agrícolas e como isso afeta na produção das uvas. Por meio de exclusões dos predadores, confirmei que aves e morcegos auxiliam no controle de artrópodes das vinícolas, diminuem os danos foliares e resultam em uma maior produção agrícola, calculada em ~US$ 2300 por ha. Tais resultados podem auxiliar manejos das áreas de plantio, incentivando tomadores de decisões e agricultores na manutenção de áreas florestais e/ou na maior heterogeneidade dentro das vinícolas, beneficiando não só a biodiversidade, como as pessoas numa convivência mais vantajosa para ambos.

Palavras-chave: Conservação. Mata Atlântica. Agricultura. Serviços Ecossistêmicos. Controle de pragas.

Abstract

Agriculture is one of the main processes that change natural landscapes with new configurations. These new landscapes can be formed by forest fragments and mosaics of different land uses, such as agricultural crops. These transformations have consequences for biodiversity, not only at taxonomic *per se*, but also at functional diversity, which may have implications for ecosystem services provision. With this knowledge, my thesis aims to elucidate as different consequences of bird taxonomic and functional diversity in relation to different land uses in vineyards in southeastern Brazil. As well, the provision of ecosystem services performed by birds and bats in crops. Thus, I approach in my first chapter the influence of landscape on a gradient of heterogeneity and forest cover on bird diversity. My results show that heterogeneity landscapes explained the variation in three metrics of functional diversity, but it is not related to taxonomic diversity. Avian communities in vineyards landscapes suffer by environmental filter. However, in heterogeneous landscapes, these same communities can be structured by limiting similarity processes. In the second chapter, I used the same landscape gradient influence in pest control and how it affects crop yield. Using fishnets for predators exclusions, my results showed that birds and bats can control arthropods in vineyards, they can reduce leaf damage and result in higher agricultural production, estimated at ~USD 2.300 per ha. These results can be able to assist management agricultural areas by encouraging decision-makers and farmers to maintain forested areas or higher heterogeneity within the crops. Finally, these considerations can benefit biodiversity and people in a win-win process.

Keywords: Conservation. Atlantic forest. Agriculture. Ecosystem Services. Pest control.

SUMÁRIO

1. Introdução geral

1.1 Modificação das áreas naturais e a biodiversidade

A conversão de áreas naturais em áreas antrópicas é um fato que tem ocorrido há séculos (Foley 2005). Áreas urbanas, pastagem e, principalmente, áreas agrícolas têm substituído grandes extensões de habitats naturais (Foley et al. 2005). Segundo a Organização das Nações Unidas para a Alimentação e a Agricultura (FAO), a produção de alimentos ocupa hoje em torno de 12% da área do planeta, aliado às pastagens que ocupam mais 26%. Como consequência, temos que quase metade das florestas do mundo já foram perdidas, com decorrências na perda da biodiversidade (Myers et al. 2000, Ramankutty et al. 2008). As áreas tropicais podem ser um exemplo dessas alterações paisagísticas, pois tais áreas possuem a maior concentração de produção de alimentos no mundo, e ainda abrigam ambientes naturais, muitas vezes florestas tropicais, com a maior biodiversidade do planeta (Dirzo & Raven 2003; Gibson et al. 2011). De fato, grande parte das áreas naturais tropicais encontram-se reduzidas e altamente fragmentadas, servindo de refúgios para muitas espécies endêmicas e ameaçadas. Por causa disso, tais áreas acabam tornando-se *hotspots* mundais com grande importância ecológica e que merecem prioridades conservacionistas (Myers et al. 2000; Girardello et al. 2019). A modificação da paisagem, seja em nível configuracional ou composicional, juntamente com a intensificação agrícola, levam a uma alteração da biodiversidade em áreas antropizadas (Foley et al. 2005). Nesse sentido, houve a necessidade em se utilizar métricas da paisagem também são utilizadas para entender como a biodiversidade responde a essas novas configurações e composições, entre elas a heterogeneidade espacial e a cobertura florestal (Martensen, Pimentel & Metzger 2008; Fahrig et al. 2011). A heterogeneidade espacial pode ser caracterizada por uma paisagem em estilo mosaico, onde vários usos do solo compõem local (Fahrig & Nuttle 2005). A cobertura florestal nada mais é que a área total de floresta dentro da paisagem, podendo ser analisado sob vários pontos de vista, como a estrutura dos fragmentos, conectividade, variação nas escalas espaciais utilizadas, entre outros (e.g. Martensen, Pimentel & Metzger 2008; Carrara et al. 2015).

Estudos que relacionam a biodiversidade com o contexto paisagístico buscam entender como as características do meio podem moldar as comunidades ali existentes e, até mesmo, entender como a comunidade está estruturada em nível de diversidade de espécies (e.g. Hidasi-Neto, Barlow & Cianciaruso 2012). A possibilidade de entender a diversidade de uma comunidade em diferentes contextos paisagísticos pode auxiliar a mensurar o grau de degradação ou modificação de um ambiente, principalmente em áreas agrícolas (Holzschuh, Steffan-Dewenter & Tscharntke et al. 2010; Luck et al. 2015). Novas paisagens podem afetar as comunidades biológicas em nível de composição e estruturação, como na riqueza de espécies e/ou abundância dos indivíduos, afetando a diversidade, com consequências nas interações entre elas, funções ecológicas (Anjos 2004; Fahrig 2003). Com alterações nos grupos funcionais, normalmente as espécies com hábitos mais generalistas têm substituído as que possuem maiores exigências ambientais e são mais específicas ao meio (e.g. Sekercioglu et al. 2019), de modo que isso pode ser percebido, inclusive pelas funções ecológicas exercidas (Bianchi, Booij, & Tscharntke 2006). Dadas tais consequências, estudos que tragam mais informações sobre ecologia de comunidades e funções ecossistêmicas entre ambientes naturais e alterados são imprencindíveis (Bommarco, Kleijn, & Potts 2013).

A heterogeneidade dentro de uma paisagem traz mais habitats diferenciados e a possibilidade de abrigar mais espécies, as quais ocupam os vários nichos disponíveis (MacArthur and MacArthur 1961), ao passo que uma paisagem mais homogênea - antrópica de matriz agrícola - tende a ter uma diversidade menor de habitats gerando comunidades biológicas mais simplificadas (Fahrig et al. 2011; Sekercioglu et al. 2019). Em relação aos ambientes mais florestados, a floresta em sua estrutura mais complexa e madura é capaz de, também com maior número de nichos ecológicos disponíveis, abrigar espécies mais dependentes da cobertura florestal (Morante-Filho et al. 2015). Algumas espécies de aves, por exemplo, tem uma capacidade de dispersão limitada e/ou necessidades reprodutivas ligadas às condições florestais mais conservadas (e.g. Hinam & Clair 2008). Desta forma, é de se esperar que comunidades mais complexas e diversas em ambientes florestais tropicais varie de acordo com as caraterísticas do meio, sejam eles mais florestados ou não. Logo, áreas agrícolas geralmente abrigam espécies mais adaptadas a locais abertos e a uma vegetação menos estruturada, além de terem comunidades alteradas, como a riqueza de espécies modificada e a proporção de indivíduos também - os mais raros podem se tornar mais abundantes, e vice-versa (Sekercioglu 2012).

Dados com ecologia de comunidades mostram que pesquisas abordando somente a componentes convencionais (e.g. riqueza, abundância, diversidade taxonômica) podem não ser suficientes para se entender a resiliência de uma paisagem com distúrbios antrópicos (Cadotte et al. 2011). Nesse caso, o interesse em outras métricas tem tido destaque dentro das pesquisas mais recentes, como o uso da diversidade funcional (DF; Cianciaruso et al. 2009). A DF nada mais é que a variação nos traços ecológicos de uma comunidade, podendo ser representado por um valor numérico, o qual mostra a variação das características das espécies na comunidade e como isso reflete no funcionamento da comunidade (Tilman 2001). A métrica leva em conta as características funcionais, que são quaisquer características morfológica, fisiológica, fenológica ou comportamental (Luck et al., 2012; Tilman 2001; Petchey & Gaston 2006) em nível individual. De fato, trabalhos mostram que a DF varia em paisagens modificadas, principalmente pela avaliação das funções ecossistêmicas, que estão diretamente ligadas às características mensuradas nos indivíduos (Luck et al. 2015; Barbaro et al. 2017). A diminuição da DF em uma comunidade pode se dar pela perda de características funcionais efetivas encontradas em determinadas espécies (e.g. Bovo et al. 2018; Tscharntke et al. 2008). Para isso, índices complementares de diversidade funcional são utilizados (Mouchet et al. 2010), os quais abrangem toda a ocupação do nicho funcional pelos indivíduos. As métricas mais comuns e que conseguem avaliar toda a extensão de ocupação das espécies dentro do espaço funcional são: riqueza funcional (FRic), divergência funcional (FDiv) e equitabilidade funcional (FEve). Como são métricas independentes, são usadas de maneira complementar (Mouchet et al. 2010).

A análise de FRic indica o quanto os atributos funcionais estão espalhados pelo espaço do nicho. Valores dos atributos funcionais dos indivíduos são necessários para mensurar essa distribuição. Logo, baixos valores mostrariam que alguns recursos ainda estão potencialmente disponíveis dentro do espaço para serem usados, e isso reduziria a produtividade ecológica do meio (Mason et al. 2005; Villéger et al. 2008). No caso de FEve, o índice mostra como estão distribuídos os atributos funcionais de cada espécie dentro da comunidade, se estas espécies estão mais homogêneas entre as espécies ou não. É uma métrica dependente da abundância dos indivíduos e independente da riqueza, logo, valores de FEve mostram o quanto a distribuição da abundância das características funcionais são mais ou menos parcimoniosas. Como que abundância das espécies reflete na utilização dos recursos dentro do espaço funcional disponível; comunidades com maiores valores de FEve mostram maior capacidade de utilizar efetivamente todos os recursos disponíveis, contrariamente, valores baixos mostram lacunas em alguns nichos que não possuem espécies caracterizadas para ocupá-los. Consequentemente, isso demonstra que a produtividade do meio também cai. Ambas, FEve e FDiv, são dependentes da abundância, e não da riqueza e, ainda, FDiv indica o grau de diferenciação de nicho e a possível análise da competição por recursos disponíveis no meio. Em virtude disso, altos valores indicam um maior distanciamento entre as espécies graças ao maior número de características, e assim, maior eficiência no uso desses recursos, aumentando a eficiência das funções ecológicas. As definições das métricas e explanações foram baseadas em Mason et al. (2005) e Mouchet et al. (2010).

Compreender o papel da diversidade funcional é válido para trabalharmos com as funções ecológicas providas pelos organismos no meio ambiente, seja em áreas naturais ou antropizadas. Os animais interagem com a paisagem utilizando seus recursos e, inevitavelmente, acabam exercendo suas funções ecológicas, com benefícios ao ser humano. Tais funções tornam-se serviços ecossistêmicos, sendo definidos como “as condições e processos através dos quais os ecossistemas naturais e as espécies que os compõem, sustentam e mantêm a vida humana” (Daily 1997). Por isso, estudar serviços ecossistêmicos dentro de áreas agrícolas pode ser um dos passos importantes para considerar a conservação biológica nessas novas paisagens manejadas pelo homem (Wenny et al. 2011; Tscharntke et al. 2005).

1.2 Aves e morcegos: grandes provedores de serviços ecossistêmicos

Quase um terço das espécies de aves ocasionalmente frequentam os ambientes agrícolas para alimentação e até mesmo para a reprodução (Sekercioglu et al. 2004). Juntamente com as aves, os morcegos fazem parte da fauna em agriculturas por todo o mundo (e.g. Maas et al. 2013; Kelly et al. 2016). Ambos os grupos têm espécies que voam e se dispersam a longas distâncias, possuindo as mais variadas características ecológicas, dieta, hábitos reprodutivos e comportamentos (Boyles et al. 2013; Whelan et al. 2016). Por causa disso, são muito utilizados em estudos de comunidades em agroecossistemas, pois esses animais acabam exercendo funções importantes como a insetivoria, polinização, dispersão de sementes e ciclagem de nutrientes (Kunz et al. 2011; Sekercioglu & Buechley 2016).

Um quinto da produção agrícola no mundo é impactada por insetos herbívoros, gerando um prejuízo de bilhões de dólares (Bonning & Chougule 2014). Ainda, o uso de milhões de toneladas de inseticidas anuamente prejudica o meio ambientes e traz problemas à saúde humana pela contaminação do solo, da água e do alimento, sobretudo no Brasil (Castilhos et al. 2019). Além disso, a eliminação de espécies provedoras de funções, como os polinizadores, traria mais prejuízos à cultura (Brittain & Potts 2011). Por isso, os serviços ecossistêmicos de regulação de pragas agrícolas merecem estudos e esforços para a geração de alimentos com o menor prejuízo possível e aliado à conservação da biodiversidade (Turner et al. 2013; Mitchell et al. 2015). Indubitavelmente, aves e morcegos são ótimos provedores desse serviço. Foi demonstrado que ambos são capazes de controlar insetos-praga em várias culturas, incluindo nas uvas, além de terem papel importante no aumento da produção agrícola (Koh 2008; Maas et al. 2013; Librán-Embid, Coster & Metzger 2017; Barbaro et al. 2017; Baroja et al. 2019). As aves consomem em média 500 milhões de toneladas de insetos por ano (Nyffeler, Sekercioglu & Whelan 2018) e os morcegos chegam a consumir até 25% da sua massa corpórea em insetos por noite (Coutts, Fenton & Glen 1973). Com isso, entre economias e lucros no âmbito agrícola, os valores giram em torno de bilhões de dólares por ano (Cleveland et al. 2006; Whelan, Wenny & Sekercioglu 2015).

As características da paisagem entra como um fator importantíssimo para o serviço de controle de pragas, visto que os predadores são totalmente dependentes de sua configuração e composição (Tscharntke et al. 2005; Bianchi et al. 2008; Gagic et al. 2015), mas os efeitos da estrutura da paisagem sobre o controle de pragas ainda são pouco conhecidos (Boesing et al. 2017). Quando feitos nas regiões tropicais, esses estudos focam principalmente em café e cacau (e.g. Karp et al. 2013; Librán-Embid & Metzger 2017). Mais pesquisas precisam ser feitas nessas culturas economicamente tão importantes em níveis mundiais, mas também há uma necessidade de aumentar o foco dos estudos em diferentes culturas e sistemas, principalmente em áreas Neotropicais (Boesing et al. 2017, Morante-Filho & Faria 2017).

1.3 A Mata Atlântica e a vinicultura no Brasil

A Mata Atlântica Brasileira é um dos casos que mais se adequa ao contexto aqui abordado; é um Bioma intensamente afetado pelo desmatamento e substuição pela agricultura desde o sec XVI. Ocupava anteriormente praticamente toda a costa brasileira com mais de um milhão de km² e áreas no Paraguai e Argentina (SOS Mata Atlântica 2019) e, atualmente, está restrita a não mais do que 17% da cobertura original, incluindo remanescentes e florestas secundárias (Turner e Corlett 1996; Ribeiro et al. 2009). Fato este que levou muitas espécies à algum grau de ameaça de extinção, principalmente as endêmicas, tornando o bioma um dos hotspots mundiais para a conservação (Myers et al. 2000). Cerca de 70% da população brasileira vive nas áreas de Mata Atlântica, aliado a isso, culturas de cana, café, soja, extração de madeira entram nos motivos do desmatamento e fragmentação. Entender os efeitos negativos da antropização sobre esse bioma e buscar possíveis soluções conservacionistas, afeta não só o meio ambiente em si, mas também a economia e saúde do ser humano para as futuras gerações (Tabarelli et al. 2012).

O Brasil ocupa a 19ª posição mundial na produção de uvas, com uma média anual de 1,5 milhões de toneladas e faturamento de R$ 3,9 bilhões, aproximadamente. A produção de uvas ocorre em toda a extensão brasileira, com aproximadamente 73 mil ha, desde a região nordestina, onde iniciou-se o cultivo nos anos iniciais de 1500, até o extremo sul do país (Embrapa 2018). A maioria das regiões produtoras encontram-se em domínio da Mata Atlântica. Dentre os estados produtores, Pernambuco, Rio Grande do Sul e São Paulo se destacam pela extensão territorial e produtividade em toneladas produzidas; sendo este último, produtor de 130 mil ton/ano (Camargo et al. 2011). Dos vários tipos de uvas cultivadas no país, as variedades de Niagara (*Vitis* spp.) predominam, inclusive dentro do estado de São Paulo (Embrapa 2018; Mello 2013). Do total produzido, metade é destinado ao consumo *in natura* (“uvas de mesa”) e a outra metade na forma processada, dando origem a sucos, vinhos, doces e derivados (Camargo et al. 2011). Para muitos municípios paulistas, a produção da uva é feita por pequenos produtores rurais e com um manejo familiar e tradicional. Normalmente, as videiras têm duas fases em sua produção – que se inicia na poda e termina na colheita, mas a época com que isso é feito difere de região pra região dependendo das condições climáticas e também, do mercado consumidor. Cada época do ano reflete a poda e a colheita da uva em um momento diferente no país devido à grande variação latitudinal. Além de ser uma forma de evitar concorrência no mercado econômico, ficando o estado de São Paulo com as podas em torno dos meses de junho – setembro e a colheita variando de novembro – abril (Embrapa; observação).

Sabido da importância que estudos em meios agrícolas têm para um melhor manejo e conciliação da conservação da biodiversidade e da produção, nossas questões basearam-se em entender melhor os sistemas agrícolas neotropicais que ainda carecem de informação em vários sentidos. No caso das vinícolas, poucas são as abordagens nesse cultivo fora de ambientes temperados (Barbaro et al. 2014, 2017, but see Steel et al. 2017; Luck et al. 2013).

1.4 Capítulos da tese

No primeiro capítulo, busquei responder como a comunidade de aves e sua diversidade é afetada pelas diferentes composições da paisagem. Levei em consideração que a cobertura florestal e a heterogeneidade ambiental são importantes fatores da paisagem que afetam as diversidades taxonômica e funcional das aves. No segundo capítulo, busquei entender como as características das paisagens na qual as vinícolas estão inseridas influenciam no controle de pragas agrícolas realizado por aves e morcegos. Ainda, qual o papel destes grupos no controle de artrópodes nas videiras, como isso reflete na produção de frutos e pode impactar os valores monetários vinculados.

2. Área de estudo

2.1 Região de estudo

O estudo foi realizado no município de São Miguel Arcanjo no estado de São Paulo, Brasil (24°00’ S, 48°01’ O, Fig. 1). O clima é classificado como mesotérmico úmido (Cfa) com chuvas anuais variando de 1.700 a 2.000 mm, cuja média de temperatura varia de 15 a 19°C. A elevação da área fica entre 720 e 850 m acima do nível do mar (Fundação Florestal 2008). Com todas essas características geográficas, a área é favorecida pela capacidade na produtividade agrícola. A economia local baseia-se no setor agrícola, como batata, soja, trigo e variedades frutíferas, como pêssegos, maracujás, caquis, mas os variados tipos de uva são o principal cultivo. A área das vinícolas encontram-se na zona rural de aproximadamente 40 mil ha e, dentre as uvas cultivadas, Itália, Rubi e Niágara destacam-se em produtividade (Fig 2). No entanto, o tipo Niagara tem sido significativamente mais cultivadas nas propriedades rurais, principalmente devido ao menor custo de plantio. A produção de uvas Niagara são-miguelense corresponde em torno de 40 % da produção do estado de São Paulo (IBGE 2018; Fig. 3).

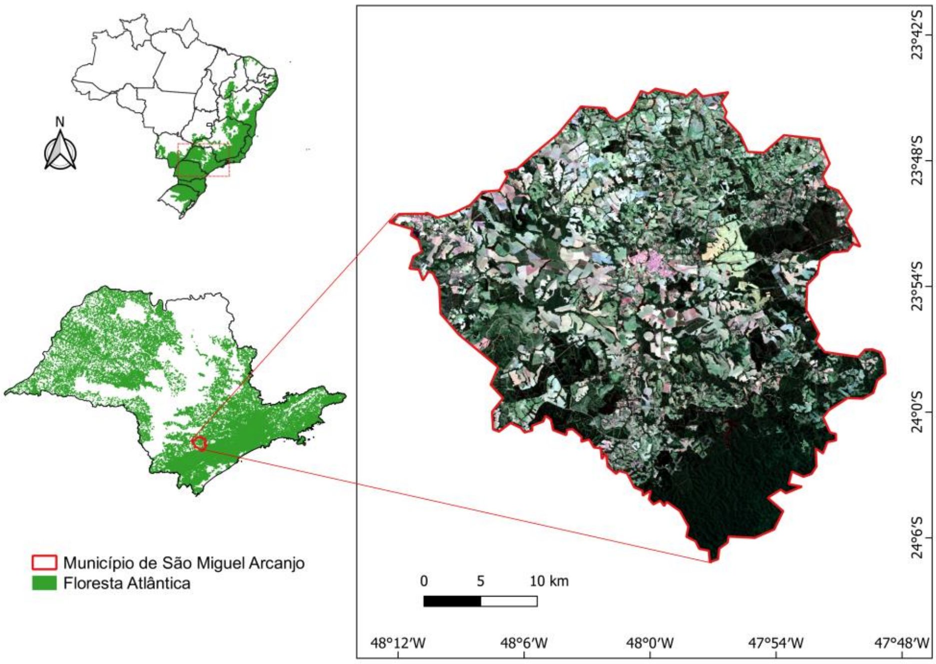

Fig. 1. Município de São Miguel Arcanjo em SP/Brasil, conhecido como “A capital das uvas finas”, cujas vinícolas são responsáveis por 40% da produção de uvas Niagara no estado de São Paulo (Prefeitura Municipal de São Miguel Arcanjo).

Fig. 2. Vinícolas no período pré-colheita que, no município, ocorre entre novembro e abril, aproximadamente. As paisagens das vinícolas são compostas por diferentes matrizes, como fragmentos de floresta, outros tipos de cultura e/ou áreas de pasto.

Fig. 3. Uvas do tipo Niagara (*Vitis* sp.) em São Miguel Arcanjo, cuja produção anual chega a ser em torno de 70,4 mil toneladas (Prefeitura Municipal de São Miguel Arcanjo).

2.2 Importância ecológica e social da região

São Miguel Arcanjo possui muitos atrativos naturais, e o principal deles é que parte do *continuum* Florestal da Serra do Paranapiacaba faz parte do município (Brocardo et al. 2012), ou seja, o local abriga um trecho do maior remanescente de Mata Atlântica existente. Frente a isso, parques e reservas naturais existem, como o Parque do Zizo, Parque da Onça Parda e o maior deles, o Parque Estadual Carlos Botelho (PECB). Com quase 38 mil ha de Mata Atlântica (Fundação Florestal 2008), o PECB possui uma fauna exuberante e rica, incluindo a de aves (Antunes et al. 2013; Galetti et al. 2009). Com 370 espécies de aves catalogadas (Antunes et al. 2006, 2013), 7,5% (n=25) delas estão sob alguma categoria de ameaça de extinção para o estado de São Paulo (Silveira & Uezu 2011). Diante disso, o PECB é considerado uma Área Importante para a Conservação das Aves (IBA - Important Bird Areas, em inglês), já que 23 espécies de aves são de distribuição restrita e 121 são endêmicas (Bencke et al. 2006). Com relação à fauna de morcegos, estudos básicos sobre a composição das espécies e distribuição foram feitos recentemente (Cláudio 2018a). Com 34 espécies catalogadas, o PECB teve sua primeira lista de quirópteros redigida com uma riqueza surpreendente. Informações inéditas sobre ocorrência das espécies no estado de São Paulo e seus dados ecológicos, entre outros pontos, começam a surgir (Cláudio 2018a, 2018b). Com isso, dados ecológicos e funcionais das espécies que ocorrem na região começam a ser mais bem conhecidos, além de estudos com os importantes serviços ecossistêmicos que esses animais provêm (Kunz et al. 2011).

O conhecimento da biodiversidade de um local auxilia na conservação da mesma, assim como, classificações em categorias de ameaça, endemismos, entre outros, propiciam vantagens não só a um grupo-alvo, mas também aos demais membros da fauna e flora do local, como plantas e mamíferos (Bencke et al. 2006). Uma área florestal nesse estágio de conservação consegue abrigar uma fauna rica e capaz de suprir às áreas entorno com muitas espécies, como a zona rural, por exemplo. Uma rica biodiversidade é capaz de realizar funções ecológicas, como abordado na seção anterior. Logo, áreas naturais são importantes para o lazer da população, conscientização ecológica e inclusão dos moradores nas atividades econômicas do município, como através do turismo ecológico. Além disso, as áreas agrícolas no entorno também são capazes de gerar renda, não só pela produtividade de alimentos, mas também pelo turismo rural, restaurantes, hospedagens, festas culturais entre outros.

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CAPÍTULO 1

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Landscape heterogeneity increases bird functional diversity in Neotropical Vineyards

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Abstract

Context: Land modification affects biodiversity in agricultural landscapes. Analyze changes in community structure by functional diversity in agriculture with different landscape metrics is necessary to link with the consequences in ecological functions.

Objectives: We aim to understand how bird taxonomic and functional diversity responds to heterogeneity and forest cover in neotropical vineyards.

Methods: We analyzed 19 vineyards landscapes in southeastern Brazil, following a gradient of forest cover and heterogeneity according to different land-uses. Bird richness, abundance, Shannon index and functional diversity (*functional richness, functional evenness, and functional divergence)* were calculated based on species traits. We used generalized linear models to test for the interacting effects of landscapes and bird assembles.

Results: Taxonomic diversity did not relate to any landscape metrics. On the other hand, heterogeneity landscapes explained the variation in three metrics of functional diversity – FEve, SESFRic, and SESFEve. Avian communities in vineyards landscapes suffer by environmental filter. However, in heterogeneous landscapes, these same communities can be structured by limiting similarity processes.

Conclusions: We highlight the importance of multi-metric approaches in biological communities and how the process of landscape homogenization can lead to the loss of ecological functions, but not species. These results can support public policies to reconcile agricultural production and biodiversity conservation.

Keywords: communities, agricultural landscapes, Atlantic Forest, biodiversity, hotspot, ecosystem function

Introduction

The conversion of natural habitats into human-made landscapes has increased over time (Foley et al. 2005). Pristine ecosystems with native vegetation have been increasingly replaced by several land-uses (Foley et al. 2005; Myers et al. 2000). In fact, there are estimates that ~ 40% of all the world's native landscapes have been converted to other types of land use (Ramankutty and Foley 1999) and several previous large-continuous native areas were changed into small isolated patches of vegetation (Fahrig et al. 2003; Haddad et al. 2015). Agricultural expansion has been a major cause of landscape changes mainly in tropical areas, where nearly 80% of agricultural lands have already replaced forests (Foley et al. 2011; Gibbs et al. 2010). This is because, human demands for food are also increasing, thus creating intensive use landscapes dominated by different agricultural systems (Foley et al. 2005; Tilman et al. 2002).

Landscape shifts, mostly conversion to intensive monocultures, have severe consequences for biodiversity as declines in population densities and/or extinction of local species, mainly due to native habitat fragmentation and/or loss (Hanski 2005; Tabarelli et al. 2004, Fahrig 2003). However, the impact on biodiversity may vary depending on both land use and landscape characteristics. In fact, there is evidence that maintaining biodiversity in agricultural environments depends on the degree of habitat preservation (e.g., forest cover percentage) and/or landscape complexity and environmental heterogeneity in agroecosystems (Fahrig 2001; Fahrig et al. 2011). This is because environments with lower land-use intensity and greater forest cover may preserve a set of the original habitats. Similarly, heterogeneous landscapes may provide more habitats, niche and/or complementary resources and thus increasing biodiversity (Fahrig et al. 2011). In this context, to understand how biodiversity is resilient is a central matter (e.g. Sekerciglu et al. 2019).

Traditional diversity metrics (i.e. taxonomic) that consider only the number of species and their abundance are often used to examine the relationship between landscape features and biodiversity (e.g. Anjos 2004; Maldonado-Coelho and Marini 2004). However, these metrics may be poorly predictive of the structure and functioning of a community, and the effects of anthropization as well (Cianciaruso et al. 2009; Díaz and Cabido 2001). By assuming that all species are ecologically equivalent, taxonomic metrics disregard conservation values, the ecological role performed by each species, and their sensitivity to environmental impacts (Díaz and Cabido 2001; Mouchet et al. 2010). Currently, functional diversity has emerged as an alternative that complements traditional metrics of diversity (Diáz and Cabido 2001; Gagic et al. 2015). By describing the variation of species within a community and the ecological functions they perform (Tilman 2001), it can more accurately reflect the consequences of land-use changes on biodiversity (Flynn et al. 2009). Yet, functional diversity can be represented by several metrics that should be used complementary, given the complexity of this approach and because only a single index cannot satisfactorily describe the diversity of functions across the community (Mouchet et al. 2010). The main metrics to describe functional diversity that should be considered are functional richness (FRic), functional evenness (FEve) and functional divergence (FDiv). Together, these metrics are able to evaluate the diversity of functions, as well as indicate the structuring processes of these communities, for example, limiting similarity, niche filtering, dispersal limitation and neutral processes (Mouchet et al. 2010).

Birds have high taxonomic diversity, interactions, and ecological niches and functions in agricultural landscapes (Anjos et al. 2015; Sekercioglu 2006; Whelan et al. 2015). They are relatively easy to track and have several measurable characteristics linked to major ecological or trophic processes (Bregman et al. 2016). They also respond quickly and by varied ways to environmental changes (Anjos et al. 2011; Sekercioglu et al. 2019). Therefore, birds are a useful group for assessing the impacts of agroecosystems on functional diversity. Indeed, birds can be used as outstanding models for understanding the overall community integrity in agricultural systems (Alexandrino et al. 2016; Piratelli et al. 2008) by monitoring their ecological functions (Sekercioglu et al. 2016; Whelan et al. 2008). Near a third of all bird species in the world use agroecosystems and ~3% may be considered "specialized" in agriculture and open areas (Piratelli et al. 2018; Sekercioglu et al. 2007; Sekercioglu 2012). On the other hand, Neotropical birds have a strong dependence on landscape features (Banks-Leite et al. 2010) and can be directly affected by agricultural (Sekercioglu et al. 2019). Several studies on bird communities in crop fields revealed that diversity may change with increasing land-use intensification (Almeida et al. 2016; Endenburg et al. 2019; Flynn et al. 2009), but sometimes the taxonomic and functional bird diversity can be affected not necessarily in the same direction (Lee and Martin 2017; Rocha et al. 2019) with increasing land-use intensification. Thus, as the impacts of land-use on avian diversity may be context-specific, it is essential to consider each situation (Luck et al. 2015).

Studies of functional bird diversity in Neotropical crop fields have received more attention only in recent years (e.g., coffee, Martínez-Salinas et al. 2016; cacao, Rocha et al. 2019) and they are still scarce (Boesing et al. 2017). Therefore, considering that the impact of agroecosystems on avian functional diversity depends on the land-use type, studies with this approach in other types of culture are imperative (Morante-Filho and Faria 2017; Sekercioglu

2012). Birds have been studied in vineyards with many objectives (e.g. Assandri et al. 2017; Duarte et al. 2014; Kross et al. 2012), including landscape effects on bird communities and diversity (e.g. Barbaro et al. 2017; Luck et. al. 2015). In Brazil, most of the vineyards are in the Brazilian Atlantic Forest Biome (Embrapa 2018), a hotspot for conservation, due to its biodiversity and a high degree of fragmentation (Myers et al. 2000). This biome has rich bird biodiversity and endemism, but also a remarkable history of degradation and human land-use (Morelatto and Haddad 2000; Ribeiro et al. 2009). Knowledge about the consequences of the deforestation in fauna can assist in conservation projects in fragile ecosystems.

Here we analyze how landscape heterogeneity and native forest cover affect bird assembles in Neotropical vineyards. Specifically, we assessed how the functional diversity of bird assembles changes along a gradient of heterogeneity in a landscape-scale and of Atlantic Forest cover in southeastern Brazil. We also tested whether the avian FD varies in the same way as the taxonomic metrics do. Finally, we compared observed patterns of functional diversity with expected patterns of random communities to test the role of environmental characteristics and assembly processes on functional structures of communities. We expected that functional richness and taxonomic metrics would be higher in landscapes that are more heterogeneous and/or with a larger percentage of forest cover. Although bird richness and FRic are expected to increase following the landscape gradient (i.e. heterogeneity or forest cover), this pattern may not be the same for FDiv and FEve. If new bird species, having different characteristics are added to an ecological community, FDiv tends to be higher, while the FEve values tend to decrease, depending on the abundance. We also expect communities in more homogeneous landscapes to be structured by niche filtering, while communities in more heterogeneous landscapes are structured by limiting similarity.

Material and Methods

Study area

The study was carried out in the region of São Miguel Arcanjo, a countryside area of ~40.000 ha in the state of São Paulo, Brazil (24°00’ S, 48°01’ W, Figure 1), imbibed in the Brazilian Atlantic Forest (Morelatto and Haddad 2000; Ribeiro et al. 2009). São Miguel Arcanjo is an important area for fruit production in Brazil, with a prevail of grapevines by smallholders, mainly *Vitis* spp. Type (IBGE 2018). The agroecosystem consists of a landscape with wide heterogeneity in land use, including remnants of natural and semi-natural forest and vineyards (Figure 1, Figure 2); these zones are closed to the largest remaining of the Atlantic Forest in Brazil (Brocardo et al. 2012; Figure 1). The climate is mesothermic humid (Cfa) with annual rainfall ranging from 1.700 to 2.000 mm and the average temperature, from 15°C to 19°C. The elevation varies between 720 and 850 m a.s.l., (Fundação Florestal 2008).

Landscape metrics and selecting vineyards

We selected areas using high-resolution images (ArcGIS 10.3 basemap imagery, DigitalGlobe satellites 2016, 0.5 m^2, 1:5000 scale). We manually outlined seven land-use classes (remnants of Atlantic forest, regenerating forests, *Eucalyptus* plantations, grape plantations, other agricultures, open areas, urban areas; Figure 1) using ArcGIS (ESRI) for the study area. The data was validated in the field in 2016 and 2017, and all interpretation errors were corrected. In each polygon characterized as grape plantation, we established a point in the center of the vineyards with a 1000 m buffer radius (Boscolo and Metzger 2009; García et al. 2018) to capture information about the landscape gradient according the proportion of natural habitats and heterogeneity (number of elements that structure the landscape). Previous analyses with other buffer radii (100, 250, 500 and 750 m) were tested, but 1000 m buffer was the best scale for landscapes and bird community metrics. We selected 19 sampling points (Figure 1) using FRAGSTATS v4.2.1 (McGarigal and Ene 2015), where forest cover percentage was 18-55% and environmental heterogeneity by Shannon´s diversity index (SHDI) ranged from 0.95-1.78. Forest cover and SHDI were not correlated (Spearman's $\rho = -0.33$, $p = 0.16$).

Bird Surveys

Bird assemblies were surveyed during the grape harvest period (January to April 2018). In each studied site, we settled four points with a minimum distance of 200 m between then and from the point center (Figure S1). We sampled bird communities using 50 m fixed-point counts for 10 min (Buckland 2006; Maas 2015; Van Bael et al. 2008). These censuses were carried out in four points per area/day during four days in the mornings (between 06 am and 10 am), ending 160 min of bird census per area in the total samples. All birds visually and acoustically detected were included, except for those individuals flying over. We rely on the South American Classification Committee (Remsen et al. 2019) for the scientific nomenclature.

Functional Diversity

Species richness and abundance were calculated from our survey data and we used the Shannon index. We used functional traits (i.e. any morphological, physiological, behavioral characteristic of an individual that may indicate the ecological functional diversity; Luck et al. 2012) to characterize the functional structure (i.e. the distribution of species and their abundance in the space they occupy; Villéger et al. 2008). Specifically, we compute bird functional diversity through a matrix of functionally important traits related to the diet and life history of birds, how they use resources in the environment and their abundance. We chose three categorical traits related to foraging (e.g. diet items), one categorical trait related to migratory status, one categorical trait related to strategy of reproduction and one related to activity period, one continuous trait may represent the use of resources by birds (body mass) (see Table 1 for more information on the used traits). These traits were based on similar papers (Barbaro et al. 2017; Luck et al. 2012, 2015) and information was based on Wilman et al. (2014), Del Hoyo et al. (2018) and BirdLife International (2019). These characteristics usually are analyzed in studies aiming to detect species responses to environmental changes (e.g. reproductive strategy), the ecological role in vineyards (e.g. migration) and how this contributes to ecosystem functions, like pest control and seed dispersal (e.g. foraging strategy) (Barbaro et al. 2017; Luck et al. 2012; Philpott et al. 2009; Sekercioglu 2012; Whelan et al. 2008). Pairwise correlations between functional metrics were checked before further analyses.

We created a distance matrix from the traits matrix using the Gower distance (Gower 1971) and calculated the functional diversity indexes using the FD package (Laliberté and Legendre 2010; Laliberté et al. 2015) in R v.3.5.1 (R Core Team 2018). Complementary indexes better represent a community by capturing its entire functional structure (Barbaro et al. 2017): functional richness (FRic), functional evenness (FEve), functional divergence (FDiv) (Laliberté and Legendre 2010; Mason et al. 2005; Petchey and Gaston 2002; Villéger et al. 2008). FRic is expressed as the convex hull volume of the functional trait space summarized by a principal coordinates analysis (Laliberté, Legendre and Shipley 2015). FEve is based on a minimum spanning tree measuring the regularity of trait abundance distribution within the functional space, while FDiv measures trait abundance distribution within this volume and increases with extreme trait values (Mason et al. 2005; Villéger et al. 2008; Laliberté and Legendre 2010). As some metrics of FD (e.g. FRic) can be influenced by species richness. Thus, we removed the effects of species richness on these metrics by means of standardized effect sizes (SES) (Mason et al. 2013). We used a simulation approach to create a null model with expected values at random. We kept the number of species constant and randomized the abundances among species to generate 1,000 random communities each site (Mason et al. 2013). Next, we calculate the FD metrics for each community and used the means and standard deviations to compute the standard effect size (SES) for each metric from SES (observed values - mean of expected values) / standard deviation of expected values (Gotelli and McCabe 2002). Negative values of SES indicate that observed metrics are less than expected by chance and suggest stronger environmental filtering and greater similarity in the community. On the other hand, positive SES values indicate that the observed metrics are higher than expected by chance and suggest greater niche complementarity and a lower similarity in the community (Petchey and Gaston 2007)

We performed generalized linear models with Gaussian error distribution to analyze the effects of forest cover and landscape heterogeneity on taxonomic diversity metrics and standardized effects sizes for FD metrics. We evaluated the significance of each variable by comparing models assessing the goodness of fit by likelihood-ratio test (Quinn and Keouh 2002).

We tested the relationships between taxonomic diversity metrics with functional diversity metrics using Pearson correlations. We compared the observed values of functional diversity with the mean expected values (mean FD of 999 randomizations) using the Wilcoxon test for paired samples.

Results

Bird diversity and landscape characteristics

We recorded 10,438 contacts from 149 bird species across all vineyards (Table S1). Bird abundance varied from 231 to 802 (549.4 ± 144.1, average ± SD) individuals and richness from 37 to 77 (52.1 ± 10.4) species. The most common species were *Sicalis flaveola*, *Zonotrichia capensis*, *Troglodytes musculus* and *Pitangus sulphuratus* (Table S1).

Landscape metrics did not influence taxonomic bird metrics as species richness, abundance, and Shannon Index ($p > 0.05$; Table S2). Related to the functional diversity metrics, while FRic, FDiv and SESFDiv (standard effect size of functional divergence) did not respond significantly either to landscape heterogeneity or to forest cover (Table S2). FEve, SESFRic (standard effect size of functional richness), SESFEve (standard effect size of functional evenness) responded positively to landscape heterogeneity (Table 2, Figure 3). Increases of each unit in the heterogeneity landscape increased FEve by 0.1 times, SESFRic by about 2 times and SESFEve by 2.5 times (Table 2, Figure 3).

Relationships between functional and taxonomic diversity

As expected, there was a strong positive correlation between species richness and FRic (Table 3). Species richness was also positively correlated with FDiv, SESFDiv, and SESFEve, but not with SESFRic. Shannon index was positively correlated only with FRic and SESFDiv. Markedly, there was a strong positive correlation between FDiv and FEve and their standardized values (SES), but we did not find a correlation between FRic and SESFRic. On the other hand, FRic was positively correlated with FEve, SESFDiv, and SESFEve (Table 3).

Differences from expected functional diversity indices

Expected FRic values (mean FRic of 999 randomizations) increased linearly with species richness in the sampled communities ($r^2 = 0.99$, $p < 0.001$, $\beta = 0.0082$, Figure 4), while expected values of FDiv (mean FDiv of 999 randomizations) and expected FEve values (mean FEve of 999 randomizations) did not vary with increasing species richness ($p > 0.05$, Figure 4).

In all communities, the FRic values were significantly lower than the calculated values for the corresponding simulated communities (i.e., the observed FRic values were below the lower limit of the confidence interval of FRic values of the simulated community FRic values, Figure S2). When including all communities, the observed FRic values were 1.6 times lower than the simulated community averages (Wilcoxon Paired test: $V = 190$, $p < 0.001$, Figure S2). On the other hand, less heterogeneous landscapes showed greater differences between expected and observed FRic values (more negative SESFRic values, Figure 3). In 18 of the 19 sampled communities, the observed FDiv was significantly higher than calculated values for the corresponding simulated communities and in one, the observed value was significantly lower (Figure S2). Considering all communities, the values of FDiv were 1.1 times higher than the simulated community means ($V = 1$, $p < 0.001$, Figure S2). There was no pattern in the differences between expected and observed values regarding heterogeneity (Figure S3). In all communities, the observed FEve differed significantly from the calculated values for simulated communities. Of these communities, in six, the observed values were lower than expected and in 13, the observed values were higher (Figure S2). Hence, considering all communities, the observed FEve values did not differ from the means of the simulated communities ($V = 65$, $p = 0.241$, Figure S2). On the other hand, less heterogeneous landscapes showed greater differences between expected and observed FEve values (more negative SESFEve values, Figure 3).

Discussion

Our results show that increased landscape heterogeneity may enhance bird functional diversity in Neotropical vineyards when considering at least three functional components (evenness and the standardized effect size of functional richness and evenness). This indicates that a matrix with more types of land use in agricultural landscapes is important to maintain bird functional diversity and our first hypothesis was partially confirmed. On the other hand, we did not find a relationship between landscape heterogeneity and/or forest cover with taxonomic diversity, suggesting that taxonomic metrics may be uncoupled of functional ones. Although these results are different from what we expected, they underscore the importance of a complementary approach for studying community ecology, mainly for better management and conservation of biodiversity in agricultural landscapes. Finally, we have shown that different mechanisms may be involved in the structuring of the studied communities. Although FRic reflects the environmental filters driving bird communities in agricultural landscapes, it is possible that this effect may be diluted in more heterogeneous landscapes, where according to the results shown by FEve, the similarity limitation process can structure the communities.

Bird diversity in vineyards landscapes

Land-use changes due to agriculture are known to cause biodiversity losses mainly in the tropics (Foley et al. 2005; Sekercioglu et al. 2019). Yet, many studies have shown that forest cover (Barros et al. 2019; Banks-Leite et al. 2014) and landscape heterogeneity (Carrara et al. 2015; Lee and Martin 2017) can somewhat mitigate this impact and allow species to persist in agrosystems and human-modified landscapes. Our results do not support the prediction on forest cover. This is because both taxonomic and functional diversity are not related to the percentage of forest cover. This means that landscapes with low forest cover may have a similar average diversity of landscapes with high forest cover. Although this result is surprising, it is possible that fragment quality and connectivity are more important than the percentage of forest cover and/or fragment area (Martensen et al. 2008), as already tested for two understory insectivorous species (*Basileuterus leucoblepharus* and *Pyriglena leucoptera*) (Uezu et al. 2008). In our study, some southernmost sampling areas have more structured forest fragments (D.J. Moreno, pers. obs.) and connected to a large forest continuum, Carlos Botelho State Park (Figure 1). However, these sampling areas have an average forest cover 1.2 times lower than the northern areas, which have more isolated and less structured fragments. Thus, although we have not tested the effects of forest connectivity in our study, it is likely that it may have diluted the effect of forest cover variation on bird diversity in vineyards.

We did not find a link between taxonomic diversity and landscape heterogeneity. According to the hypotheses of habitat heterogeneity (MacArthur and MacArthur 1961), heterogeneous landscapes may offer more niches and complementary resources than homogeneous landscapes, and therefore may harbor more species, individuals, and functions (Benton, Vickery and Wilson 2003). However, although this hypothesis has broad empirical support in previous avian studies (e.g. Barbaro et al. 2017, Lee and Martin 2017), our taxonomic diversity data did not support that. We also did not find any relation between landscape heterogeneity and functional richness. This is not surprising since FRic is highly dependent on species richness and it is known that with more species added, the volume of feature space occupied in the community increases (Mason et al. 2005; Villéger et al. 2008). Thus, as species richness and FRic are correlated, the same pattern would be expected in relation to heterogeneity. On the other hand, SESFRic is positively related to heterogeneity. This means that the occupation of multidimensional volume, regardless of taxonomic richness, increases in more heterogeneous landscapes. Thus, niche occupation in the community may increase due to landscape characteristics and not necessarily the number of species (but see our discussion in the next section).

In this study, FEve and its standardized effect size also increased with landscape heterogeneity. In general, FEve indicates how uniform is the distribution of the abundance of functional characteristics in assemblies and may indicate efficiency in resource use (Mason et al. 2005; Laliberté and Legendre 2010). For example, the value increases when the variation in abundance of species characteristics is very homogeneous, indicating an efficient use of available resources for species (Mason et al. 2005). On the other hand, FEve value decreases when the variation in the abundance of species characteristics is not very homogeneous, with some characteristics being overrepresented in the community, while others are rare. In a way, FEve can be compared to species evenness (Pielou evenness) and thus we make an analogy between species and characteristics. Just as it is predicted that in communities with low species uniformity, the many rare species (Jost 2010) would be more subject to local extinction resulting from demographic and environmental stochastic (Simberloff 1986; Caughley 1994). In communities with low FEve value, the underrepresented characteristics may also be more likely to disappear from the community. In our study, areas with homogeneous landscapes had a lower FEve value and therefore had a higher risk of losing functions. As heterogeneity increases, uniformity in feature distribution increases and this leads to increased uniformity in the occupancy of the trait space.

Finally, the values of FDiv and SESFDiv are not related to landscape heterogeneity. The FDiv quantifies how the most abundant species are distributed within the functional space volume (Mason et al. 2005; Villéger et al. 2008). In this sense, the FDiv may decrease as the functional characteristics of the most abundant species are near the center of the characteristic space or increase as they are near the ends (Mason et al. 2005; Karadimou et al. 2016). In our study, the dominant species and their abundance did not differ significantly among the sampled areas and, for this reason, we may not have found a pattern with heterogeneity.

In summary, our data show some inconsistencies between taxonomic and functional diversity, suggesting that bird functional diversity is not necessarily high in communities with high species richness. This dissonance between the two types of diversity measurement may be a result of long historical processes and ecological mechanisms that differently affect species composition and functions in biological communities. Thus, this study shows that - although landscape homogenization does not necessarily lead to species losses - this may lead to a simplification of bird communities. Considering the occurrence of increasingly human-modified landscapes, it is essential to complement each approach to better manage and conserv biodiversity.

Expected differences in functional diversity indices

In our study, the expected values of FRic increased linearly with species richness, however, the expected values of FEve and FDiv did not vary with increasing species richness (Figure 4). FRic was expected to increase with species richness since regardless of the community assembly rule, increased richness leads to increased convex hull volume (Mouchet et al. 2010). However, FEve and FDiv quantify functional units regardless of the number of functional niche dimensions, and for this reason, these measures are few sensitive to species richness (Villéger et al. 2008, Mouchet et al. 2010).

In all areas, FRic values were significantly lower than randomly expected values. Because of this, we can predict that environmental filtering is structuring these communities. This means that species occurring in these communities share more similarities than might be expected, as environmental conditions act as a filter, allowing only the establishment of a set of characteristics. Land-use changes largely due to disturbances created by agriculture are known to influence environmental filtering and loss of functional diversity (Flynn et al.

2009). Therefore, the pattern here observed seems adjusted with these previous findings. In fact, in our study, considering all areas, the observed FRic values were 1.6 times lower than the simulated community averages. This indicates a significant loss of functions in these communities and therefore increased functional redundancy among species (Mason et al. 2005, but see De Coster, Banks-Leite and Metzger 2015). On the other hand, less heterogeneous landscapes showed greater differences between expected and observed FRic values (more negative SESFRic values, Figure 3), indicating that the degree of landscape homogenization may cause different patterns on the loss of functional diversity. In this case, simplified landscapes may have higher rates of losses of functions than more heterogeneous landscapes.

This interpretation is confirmed by the pattern shown by SESFEve in relation to heterogeneity, where less heterogeneous landscapes showed greater differences between expected and observed FEve values (more negative values of SESFEve, Figure 3). However, the observed FEve values seem very idiosyncratic. In heterogeneous landscapes, the observed values were higher than expected, suggesting that these communities may be being structured by similarity limitation (Mouchet et al. 2010). This means that birds in these areas must have a lower similarity (Funk et al. 2008). In addition, due to the greater diversity of characteristics, these bird communities may be more resilient to environmental change (Luck et al. 2013). In turn, some more homogeneous or with medium heterogeneity landscapes presented lower than expected FEve values, once again suggesting the role of an environmental filter in the structuring of these communities. Contrary to our expectations and the pattern presented so far, the observed values for FDiv were higher than expected for our communities. On the other hand, it did not follow any direction towards heterogeneity. This is because although the most abundant species are constant across areas, they have extreme characteristic values, thus inflating the observed FDiv values.

Our work aims to provide more information on how landscape can affect and structure bird communities in agricultural areas in Neotropical regions. The focus has a conservation bias and seeks improvements in landscape composition as an important factor in maintaining bird communities capable of performing ecosystem functions in agricultural areas. In this sense, encourage smallholders and decision-makers to maintain and increase heterogeneous and well-structured areas throughout planting (e.g. Maas et al. 2018) can be of great value for more satisfactory conservation results and improvements in biological communities essential to ecosystem functions.

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Authors' contributions

Conceptualization, methodology, formal analysis: Daniele Janina Moreno, Milton Cezar Ribeiro and Augusto João Piratelli; Investigation, writing – original draft preparation: Daniele Janina Moreno; Writing – review and editing: Daniele Janina Moreno, Milton Cezar Ribeiro and Augusto João Piratelli; Funding acquisition: Daniele Janina Moreno; Supervision: Milton Cezar Ribeiro and Augusto João Piratelli. All authors read and approved the final manuscript.

Conflict of Interest

The authors declare that they have no conflict of interest.

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Table 1. Traits used to calculate the functional diversity of birds in vineyards. Binary variable is 0 or 1.

Trait type	Trait	Categories	Explanation
Ability of resources use	Mean body-mass (g)	Continuous	It is related to other characteristics associated with environment use, foraging and resource utilization
Diet	Invertebrates	Fuzzy	Associated with environment use, foraging, and resource utilization. Infers the ecosystem function that a bird can play
	Vertebrates (amphibians, reptiles)	Fuzzy	
	Vertebrates (mammals, birds)	Fuzzy	
	Vertebrates (Unknown)	Fuzzy	
	Fish	Fuzzy	
	Scavenger	Fuzzy	
	Fruits	Fuzzy	
	Nectar	Fuzzy	
	Seeds	Fuzzy	
	Plants (others)	Fuzzy	
Foraging stratum	Water (below)	Fuzzy	Species resource use site determines environments where birds can stay or use
	Water (around)	Fuzzy	
	Ground	Fuzzy	
	Understory	Fuzzy	
	Mid-high	Fuzzy	
	Canopy	Fuzzy	
	Aerial	Fuzzy	

Foraging method	Pursuit	Binary	The behavior of foraging food resources reflects the resilience of a species in altered areas
	Gleaning	Binary	
	Pouncing	Binary	
	Grazing	Binary	
	Scavenging	Binary	
	Probing	Binary	
Reproductive strategy (local of the nest)	Ground	Binary	More specialized nesting birds may be more affected by changes in the environment and loose suitable breeding sites; Generalist species may benefit from human-made sites
	Cavity on the ground	Binary	
	Tree cavity	Binary	
	Vegetation	Binary	
	Water vegetation	Binary	
	Artifacts	Binary	
Migratory status	Migrant	Binary	Seasonal occurrence and or variation in the abundance may modify the rates of ecosystem services pest control (Van Bael et al. 2008)
	Resident	Binary	
	Altitudinal migrant	Binary	
Activity period	Diurnal	Binary	How birds use resources and it can reflect a type of food consumed
	Nocturnal	Binary	

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Table 2. Summary of parameters for each dependent variable of functional diversity explained by landscape heterogeneity. FRic: functional richness; FEve: functional evenness; FDiv: functional divergence; SES: standard effect size.

	B	**SE**	**T**	**p**
FEve				
Intercept	0.447	0.063	7.089	< 0.0001
Landscape heterogeneity	0.107	0.040	2.666	0.0163
SESFRic				
Intercept	-4.896	1.065	-4.595	0.0003
Landscape heterogeneity	1.918	0.677	2.833	0.0115
SESFEve				
Intercept	-3.664	1.603	-2.285	0.0354
Landscape heterogeneity	2.503	1.019	2.456	0.0251

Table 3. Correlation coefficients between taxonomic diversity metrics and functional diversity

	FRic	**FDiv**	**FEve**	**SESFRic**	**SESFDiv**	**SESFEve**
Species richness	0.898 ***	0.496 *	0.36	-0.274	0.667 **	0.561*
Shannon index	0.757 **	0.233	0.299	-0.364	0.476 *	0.288
FRic		0.471	0.473 *	0.161	0.638 **	0.646 **
FDiv			0.051	-0.070	0.966 ***	0.541 *
FEve				0.268	0.127	0.727 ***
SESFRic					-0.110	0.239
SESFDiv						0.545 *

metrics. FRic: functional richness; FEve: functional evenness; FDiv: functional divergence; SES: standard effect size.

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Figure legends

Figure 1. Map with the locations of sampled vineyards in an area of 40,000 ha in São Miguel Arcanjo, Southeastern Brazil. Black spots and circles indicate 19 experimental plots and surrounding landscapes buffers. Different colors represent different land use classes.

Figure 2. Vineyards in São Miguel Arcanjo, one of the most vines plantations areas in southeastern Brazil. Colored arrows represent different land use: Blue: open area; Green: Forest fragment; Red: *Eucalyptus*; Purple: Vineyards. Photo: Daniele J. Moreno.

Figure 3. Landscapes heterogeneity modulates bird Functional Evenness (FEve) and Standardized Effect Sizes on Functional Richness (SESFRic) and Functional Evenness (SESFEve).

Figure 4. Avian functional diversity indices vs. species richness observed in vineyards (black circles) and calculated for correspondent simulated communities (white circles).

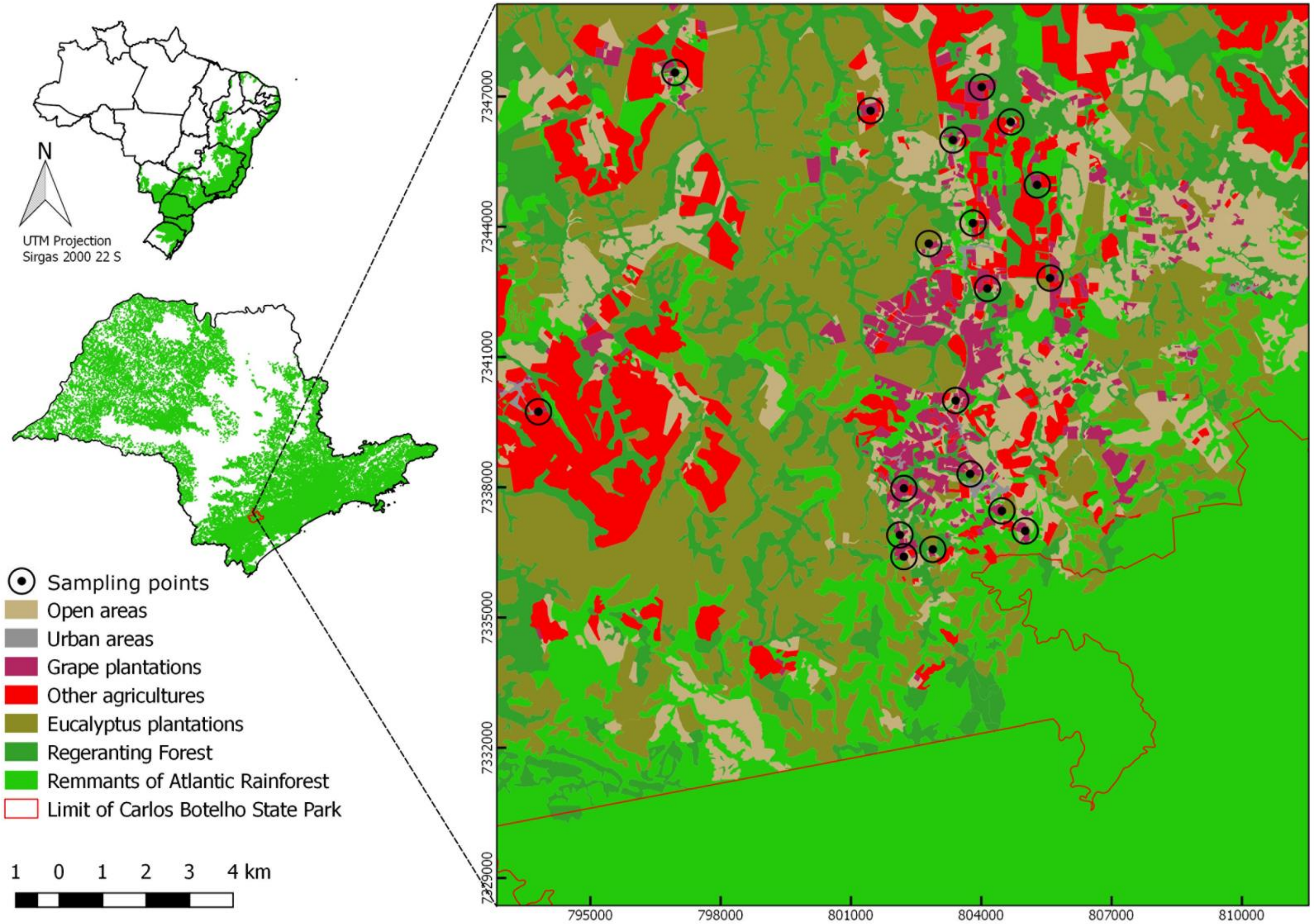
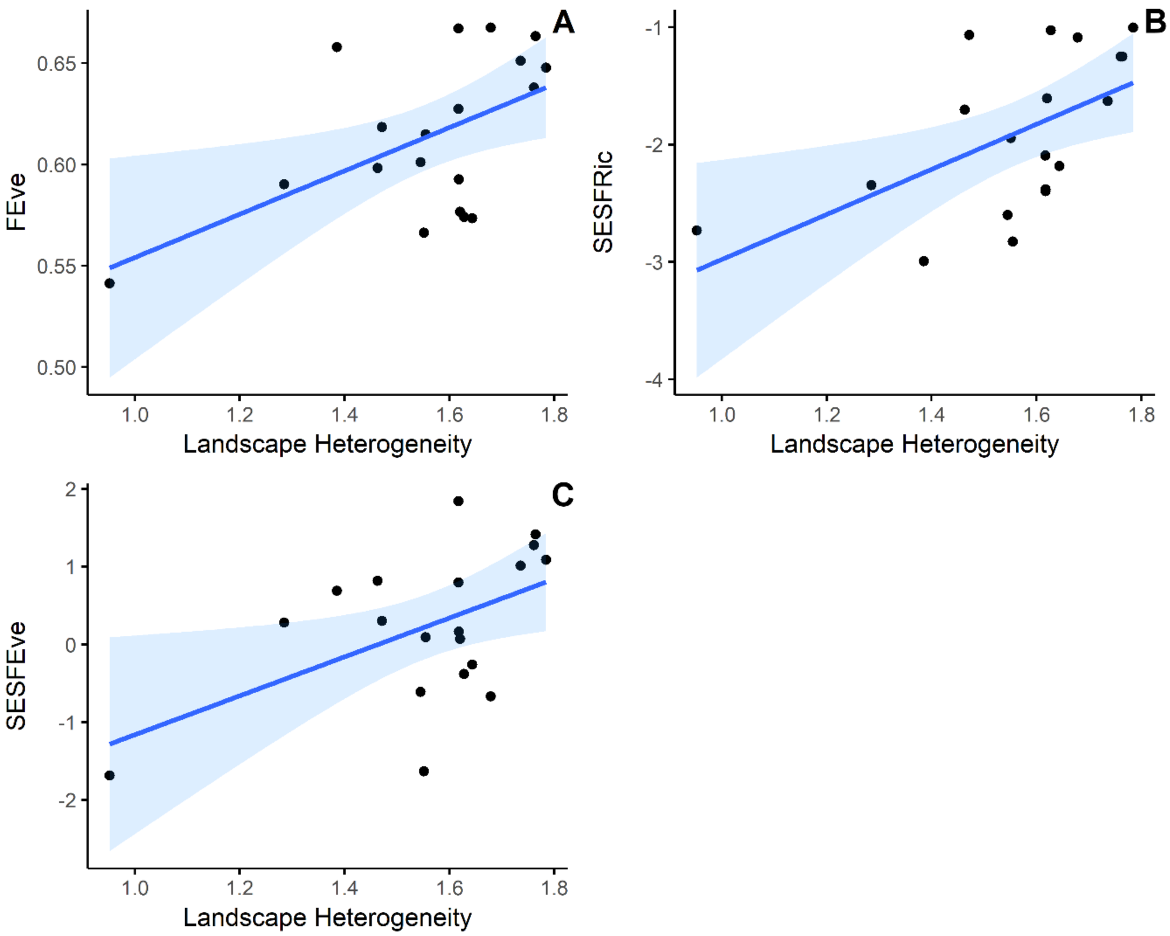
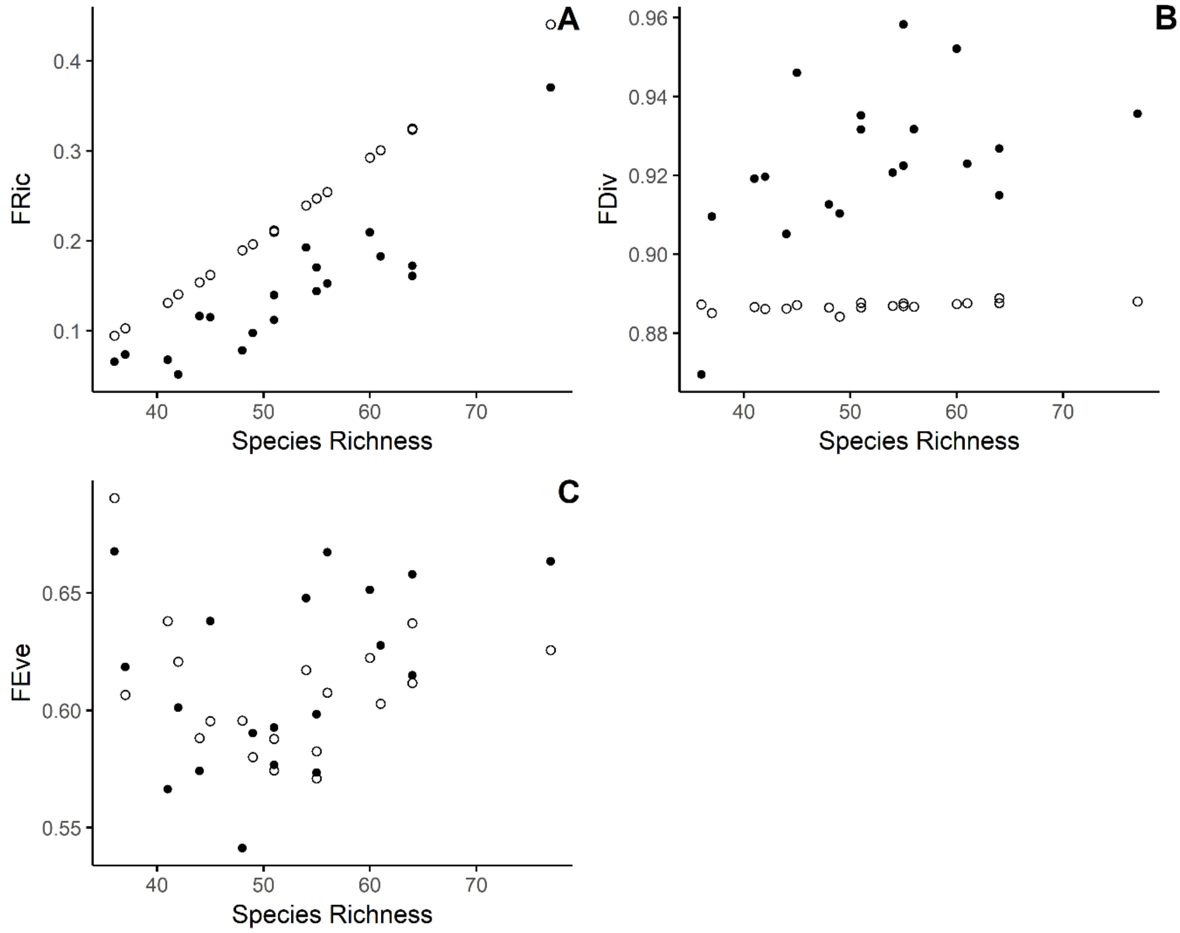

N
UTM Projection
Sirgas 2000 22 S
Sampling points
Open areas
Urban areas
Grape plantations
Other agricultures
Eucalyptus plantations
Regeranting Forest
Remmants of Atlantic Rainforest
Limit of Carlos Botelho State Park
1 0 1 2 3 4 km
7347000
7344000
7341000
7338000
7335000
7332000
7329000
795000
798000
801000
804000
807000
810000

Figure 1.

Figure 2.



707
708 Figure 3.


A
FRic
0.4
0.3
0.2
0.1
40
50
60
70
Species Richness
B
FDiv
0.96
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0.92
0.90
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Species Richness
C
FEve
0.65
0.60
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50
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70
Species Richness

Figure 4.

Supplementary Materials for

Landscape heterogeneity increases bird functional diversity in Neotropical Vineyards

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Additional data Table S1. Bird species in vineyards areas with their respective occurrence and abundance.

Bird species	**Bird abundance in each area**																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Amazilia_fimbriata	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0
Amazilia_lactea	2	1	2	0	2	1	0	6	10	34	7	4	1	1	0	0	1	6	1
Amazilia_versicolor	0	0	0	0	0	0	0	0	2	0	7	0	1	0	0	0	0	3	0
Ammodramus_humeralis	4	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anthracothorax_nigricollis	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anthus_lutescens	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aramides_cajanea	0	3	3	0	1	2	0	0	0	0	1	0	3	10	0	0	0	2	3
Aramides_saracura	0	0	0	0	0	0	0	1	0	0	2	0	0	0	0	0	3	0	0
Athene_cunicularia	0	5	0	0	12	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Attila_phoenicurus	0	0	0	0	0	0	1	0	0	0	0	3	0	0	0	0	0	0	0
Attila_rufus	0	1	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Automolus_leucophthalmus	0	2	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Basileuterus_culicivorus	1	7	6	3	6	3	4	8	10	1	6	7	18	2	0	18	0	17	5

Bird species	Bird abundance in each area																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Buteo_brachyurus	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cacicus_cela	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	4
Cacicus_chrysopterus	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Campephilus_robustus	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Camptostoma_obsoletum	0	9	0	0	0	5	3	5	6	2	4	4	0	0	0	0	0	0	1
Caracara_plancus	0	0	4	4	2	0	0	2	0	1	0	4	6	0	4	0	0	0	0
Cariama_cristata	0	0	0	0	0	2	0	0	0	2	0	0	1	0	5	0	4	0	3
Celeus_flavescens	0	1	0	0	0	0	1	4	0	0	0	4	0	2	0	0	0	2	4
Certhiaxis_cinnamomeus	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0
Chiroxiphia_caudata	1	0	0	0	1	3	0	0	0	0	1	0	0	0	0	0	0	10	0
Chloroceryle_americana	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Chlorostilbon_lucidus	3	1	4	2	3	1	0	4	10	17	18	3	1	2	0	1	3	6	3
Chrysomus_ruficapillus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	25	0	0
Coereba_flaveola	0	1	3	0	17	9	19	2	25	3	14	10	5	0	0	3	0	2	0

Bird species	Bird abundance in each area																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Colaptes_campestris	2	4	8	2	7	4	0	6	1	4	16	7	19	4	17	0	7	8	12
Colaptes_melanochloros	0	0	0	0	2	0	0	0	0	0	0	0	0	1	0	0	0	0	4
Colonia_colonus	0	2	0	0	0	5	0	0	1	1	0	0	1	0	0	0	0	0	0
Columbina_picui	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Columbina_talpacoti	11	25	5	4	2	55	13	27	31	8	13	37	5	21	3	4	31	0	24
Conirostrum_speciosum	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0
Conopophaga_lineata	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0
Cranioleuca_pallida	0	1	0	0	2	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Cranioleuca_vulpina	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0
Crotophaga_ani	0	11	2	0	3	8	0	0	4	19	0	3	15	0	16	1	15	0	1
Crypturellus_sp	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	1	1	0
Cyanocorax_cristatellus	0	0	2	0	0	0	0	1	0	0	14	0	0	0	0	0	0	0	0
Cyclarhis_gujanensis	0	14	2	5	11	17	16	15	0	0	17	22	9	6	7	4	6	4	3
Dacnis_cayana	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Bird species	Bird abundance in each area																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Dryocopus_lineatus	0	1	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Elaenia_flavogaster	0	3	0	0	0	11	0	5	14	12	0	4	1	0	3	0	0	1	0
Elanus_leucurus	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Emberizoides_herbicola	0	0	0	0	0	0	0	0	0	3	0	0	0	0	1	0	0	0	0
Empidonomus_varius	0	3	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Eupetomena_macroura	0	2	1	0	2	0	1	2	2	3	4	0	1	3	0	0	3	0	4
Euphonia_chlorotica	0	0	0	2	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Euphonia_pectoralis	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0
Euphonia_violacea	1	0	0	0	0	0	0	0	4	0	1	10	0	0	0	0	0	0	0
Falco_femoralis	0	0	0	1	0	0	0	1	1	0	0	0	0	0	1	0	2	0	0
Florisuga_fusca	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	2	0
Fluvicola_nengeta	0	4	0	0	0	7	1	2	5	10	4	2	1	8	0	0	0	2	4
Forpus_xanthopterygius	0	36	0	0	36	0	0	0	22	0	0	0	0	0	0	0	0	0	15
Furnarius_rufus	12	21	34	0	32	6	19	27	25	27	29	5	33	45	5	16	26	36	33

Bird species	Bird abundance in each area																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Geothlypis_aequinoctialis	0	0	0	0	0	0	0	0	0	0	2	0	0	3	0	0	2	0	0
Gnorimopsar_chopi	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0
Gubernetes_yetapa	0	0	0	0	0	0	0	0	9	1	0	0	0	0	0	0	0	0	0
Guira_guira	5	2	0	0	2	12	3	0	1	5	22	2	7	6	21	5	23	1	11
Haplospiza_unicolor	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Hirundinea_ferruginea	1	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0
Icterus_pyrrhopterus	0	0	0	0	0	0	0	0	0	0	12	1	0	0	11	0	2	0	0
Lanio_melanops	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lathrotriccus_euleri	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Leptopogon_amaurocephalus	0	2	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Leptotila_rufaxilla	0	1	0	2	0	2	0	2	0	4	2	2	0	0	0	0	1	2	1
Leptotila_verreauxi	0	4	0	0	0	6	1	2	0	0	0	5	0	0	0	2	4	0	0
Leucochloris_albicollis	0	0	0	0	0	0	1	0	1	0	3	0	0	0	0	0	0	0	0
Machetornis_rixosa	2	0	0	0	0	4	0	0	0	0	1	0	0	0	0	0	0	0	0

Bird species	Bird abundance in each area																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Megarynchus_pitangua	0	8	0	5	3	3	2	6	11	1	1	2	1	5	3	5	0	3	3
Melanerpes_candidus	0	5	5	5	2	0	6	3	6	0	0	0	4	3	3	12	0	0	4
Milvago_chimachima	0	0	2	1	0	2	0	0	4	1	2	1	1	1	1	0	0	1	3
Mimus_saturninus	0	0	3	0	0	7	0	0	0	7	7	0	7	0	5	2	0	0	9
Molothrus_bonariensis	4	12	0	0	10	0	1	2	7	0	1	0	0	0	0	3	11	55	0
Myiarchus_ferox	0	1	6	0	0	0	3	1	9	0	0	1	0	0	0	0	1	0	0
Myiarchus_swainsoni	2	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Myiodynastes_maculatus	2	3	0	4	1	8	1	0	5	0	0	1	0	2	0	4	0	1	4
Myiothlypis_leucoblephara	0	4	0	0	5	1	0	1	1	0	1	0	3	0	0	0	0	0	0
Myiozetetes_similis	1	1	0	0	0	0	2	0	17	0	0	0	1	0	0	0	2	1	0
Nothura_maculosa	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Odontophorus_capueira	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Orchesticus_abeillei	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pachyramphus_sp	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0

Bird species	**Bird abundance in each area**																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Passer_domesticus	3	0	0	1	8	0	0	0	0	42	0	0	9	12	0	9	0	0	5
Patagioenas_cayannensis	0	0	0	1	0	0	0	0	3	0	0	0	0	0	0	0	0	1	1
Patagioenas_picazuro	16	13	21	20	20	43	11	14	65	24	38	27	36	10	7	10	22	23	48
Penelope_obscura	0	1	0	0	0	0	0	1	0	0	0	8	1	0	0	0	0	0	0
Phaethornis_pretrei	1	2	0	0	0	1	0	0	0	0	1	3	0	0	0	0	1	1	0
Picumnus_temminckii	0	4	0	1	0	2	0	2	12	1	0	0	5	0	0	5	3	1	1
Pionus_maximiliani	12	5	5	0	66	36	12	0	0	0	5	4	6	2	1	3	7	19	36
Pitangus_sulphuratus	10	32	36	10	30	31	30	43	54	27	50	27	35	36	43	19	57	38	11
Poecilotriccus_plumbeiceps	1	1	0	0	0	0	0	1	4	0	1	1	0	0	0	1	0	1	0
Procnias_nudicollis	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Progne_chalybea	0	0	0	0	0	0	0	0	0	0	0	0	0	21	0	0	0	0	4
Psarocolius_decumanus	0	0	0	0	0	5	0	1	4	0	0	1	0	0	0	0	0	0	0
Pseudoleistes_guirahuro	0	5	0	0	0	0	0	0	20	0	0	0	0	0	0	0	10	0	0
Psittacara_leucophthalmus	0	17	0	0	0	0	0	0	11	23	0	0	1	0	21	0	0	0	0

Bird species	Bird abundance in each area																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Pygochelidon_cyanoleuca	22	50	8	4	26	10	3	72	48	0	10	30	3	6	41	41	49	6	11
Pyriglena_leucoptera	0	3	0	0	0	1	0	0	0	0	1	6	2	0	0	0	0	0	0
Pyroderus_scutatus	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1
Pyrrhura_frontalis	0	35	0	0	5	21	0	0	0	0	0	0	0	0	0	0	0	2	32
Ramphastos_dicolorus	0	0	0	0	0	0	0	1	2	0	0	8	2	0	0	0	0	5	3
Ramphastos_toco	0	1	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Ramphocelus_carbo	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0
Rupornis_magnirostris	0	5	9	2	0	14	2	5	1	2	0	3	1	0	16	0	4	1	0
Satrapa_icterophrys	0	0	0	0	0	0	1	0	0	2	0	0	0	0	0	0	0	0	0
Serpophaga_subcristata	0	0	1	0	0	4	0	2	0	1	0	0	0	0	1	0	0	0	3
Setophaga_pitiayumi	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0
Sicalis_flaveola	23	20	94	3	14	36	58	118	11	31	46	28	67	28	37	14	106	46	75
Sicalis_luteola	6	8	0	4	1	0	2	0	0	0	0	0	10	0	0	10	0	0	1
Sittasomus_griseicapillus	0	0	2	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0

Bird species	Bird abundance in each area																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Spinus_magellanicus	0	5	0	0	10	0	4	0	0	7	0	0	5	1	0	0	0	11	9
Spizaetus_tyrannus	0	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Sporophila_caerulescens	35	21	29	24	48	5	51	65	4	4	23	24	21	43	3	26	19	29	53
Sporophila_lineola	0	28	0	0	1	14	7	5	1	0	0	11	0	1	0	28	8	0	0
Stelgidopteryx_ruficollis	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	36	0	0	0
Streptoprocne_zonaris	0	0	0	10	0	0	0	0	0	0	0	10	0	0	0	12	0	0	0
Synallaxis_frontalis	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synallaxis_spixi	0	3	1	0	3	0	1	2	2	0	0	6	4	0	0	0	0	2	0
Tachycineta_leucorrhoa	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	14	0	0	0
Tachyphonus_coronatus	3	6	0	0	2	3	0	6	4	3	6	1	6	0	0	0	0	0	0
Tangara_cayana	0	0	2	0	2	0	0	1	4	0	0	0	0	0	0	0	0	0	3
Tangara_palmarum	0	3	4	0	0	0	2	0	4	0	7	0	0	0	0	7	0	0	0
Tangara_sayaca	3	18	18	4	40	20	16	46	33	43	25	27	29	23	19	43	9	3	9
Tapera_naevia	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0

Bird species	Bird abundance in each area																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Tersina_viridis	7	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0
Thalurania_glaucopis	0	1	0	0	0	1	0	0	3	1	7	0	0	0	0	0	1	0	0
Thamnophilus_caerulescens	0	5	3	5	0	11	1	9	4	0	2	18	4	0	0	0	1	2	1
Thamnophilus_doliatus	0	6	0	0	5	3	0	2	1	0	0	0	0	1	0	0	0	2	2
Theristicus_caudatus	0	2	0	0	1	2	5	0	23	2	3	0	6	0	0	0	18	0	0
Todirostrum_cinereum	0	1	3	1	3	3	0	6	3	3	0	4	8	1	1	0	2	0	3
Todirostrum_poliocephalum	0	0	1	0	0	3	0	0	0	0	0	0	0	0	0	0	0	1	0
Tolmomyias_sulphurescens	1	3	0	0	0	13	3	2	0	0	1	5	7	1	3	1	0	0	1
Troglodytes_musculus	17	40	38	17	17	14	27	52	21	14	53	47	33	28	7	67	67	33	41
Trogon_rufus	0	1	0	0	0	0	0	0	0	0	0	2	3	0	0	0	0	0	0
Trogon_sp	0	0	0	0	0	0	1	0	0	0	0	3	0	0	0	0	2	0	0
Trogon_surrucura	0	0	0	0	0	4	1	0	0	0	0	0	0	0	0	0	1	0	0
Turdus_albicollis	0	0	0	0	0	0	0	0	10	0	0	0	2	0	0	0	0	0	0
Turdus_amaurochalinus	2	12	14	1	6	5	18	26	23	4	34	20	16	9	5	7	6	17	12

Bird species	Bird abundance in each area																		
	d03	d04	d06	d09	d10	d12	d16	d18	d20	d21	d29	d31	d40	d42	d44	d49	d50	d51	d52
Turdus_leucomelas	11	7	18	8	6	1	13	32	30	6	12	2	18	10	3	5	11	0	5
Turdus_rufiventris	0	7	5	0	0	1	3	5	14	0	7	1	7	0	0	4	3	0	0
Tyrannus_melancholicus	9	3	19	0	2	19	16	4	13	2	7	8	0	34	3	14	8	2	3
Tyrannus_savana	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0	0	1	2
Vanellus_chilensis	0	8	20	6	3	0	16	0	21	6	35	2	21	5	27	21	4	10	39
Vireo_chivi	0	0	0	0	2	0	0	0	0	0	3	0	0	0	0	0	0	0	0
Volatinia_jacarina	13	12	52	24	8	9	13	60	3	1	0	68	8	7	0	14	21	12	8
Xolmis_velatus	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Zenaida_auriculata	29	11	21	4	4	44	7	16	24	12	14	33	14	12	4	16	20	9	29
Zonotrichia_capensis	24	17	60	35	35	22	41	66	31	37	48	26	53	23	35	31	23	31	53

Table S2. Model selection to explain avian diversity metrics in relation to forest cover (Cov) and/or landscape heterogeneity (Het). The best model with each diversity metric is underlined.

Variable under test	χ^2	df	p
Species richness			
Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity	99.174	3	0.8473
Abundance			
Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity	69041	3	0.3341
Shannon index			
Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity	0.024	3	0.9158
FRic			
Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity	0.018	3	0.3369
FDiv			
Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity	0.001	3	0.6311
FEve			
Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity	0.010	3	0.0329
Forest cover + Landscape heterogeneity	1.3×10^{-5}	1	0.9125
Forest cover	0.009	1	0.0046
<u>Landscape heterogeneity</u>	0.0020	1	0.1743
SESFRic			

Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity	2.777	3	0.0462
Forest cover + Landscape heterogeneity	0.161	1	0.4959
Forest cover	2.486	1	0.0065
Landscape heterogeneity	0.053	1	0.6898
SESFDiv			
Forest cover + Landscape heterogeneity + Forest cover*Landscape heterogeneity	0.612	1	0.7053
SESFEve			
Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity	6.422	3	0.0223
Forest cover + Landscape heterogeneity	2.010	1	0.0860
Forest cover	4.409	1	0.0164
Landscape heterogeneity	0.050	1	0.7986

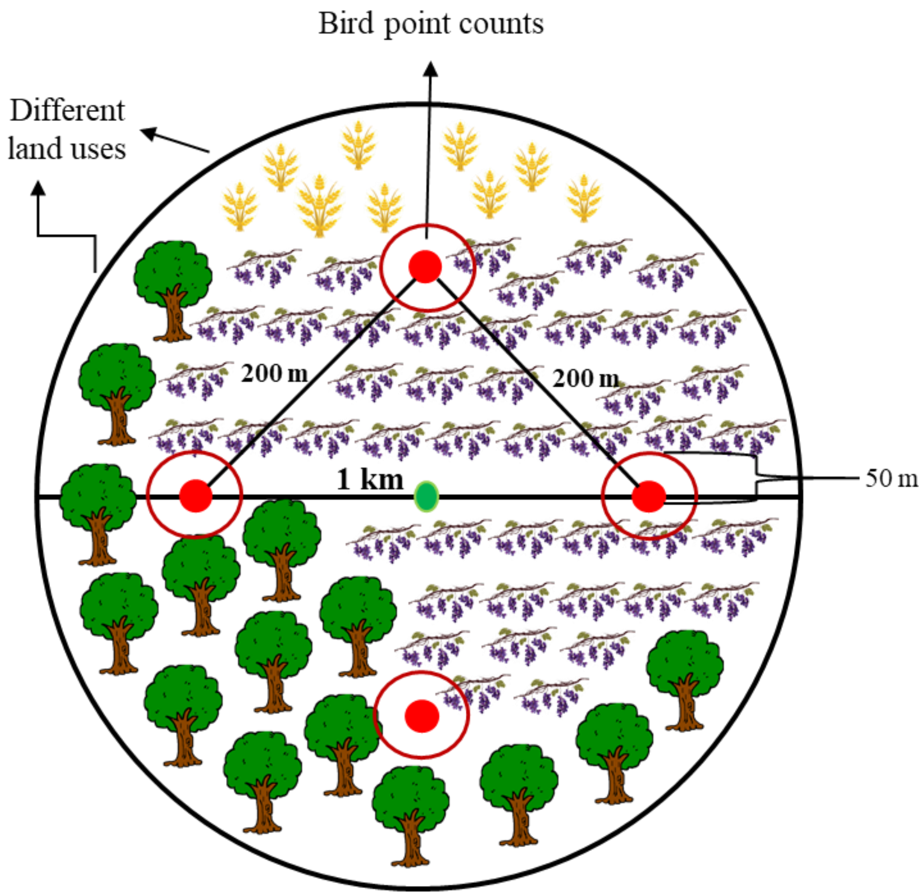

Figure S1. Bird point counts scheme: bird assemblies were surveyed during the grape harvest period. In each studied site, we settled four points with a minimum distance of 200 m between then (red dots) and from the point center (green dot). We sampled bird species using four 50 m fixed-point counts (red circles) for 10 min/each one. Drawings are for illustrative purposes only and do not reflect the correct scale.

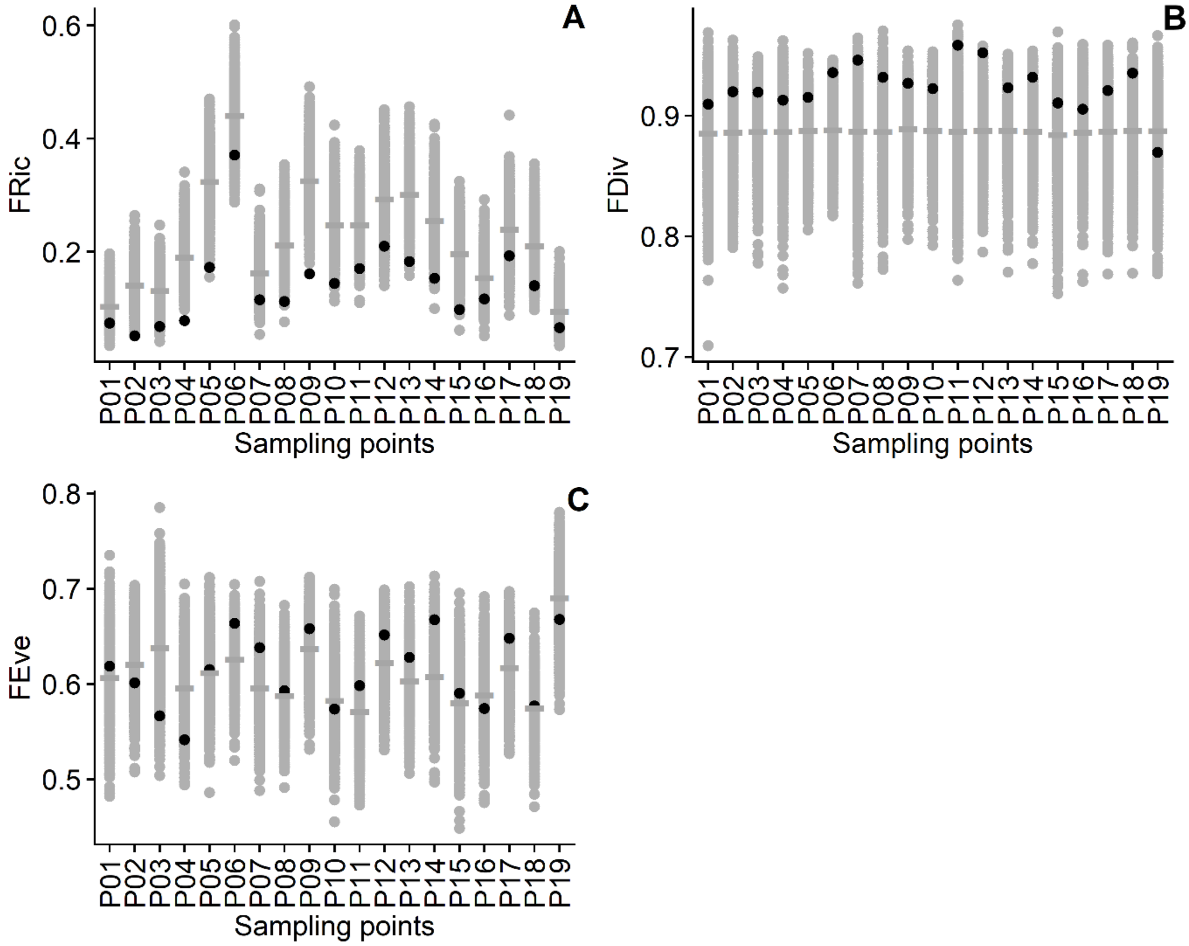

Figure S2. Comparison of observed functional diversity (black circles) and expected values resulting from 999-fold randomized communities (gray circles). Short dashes are mean values of expected avian diversity. A: Functional Richness (FRIc); B: Functional Divergence (FDiv); Functional Evenness (FEve).

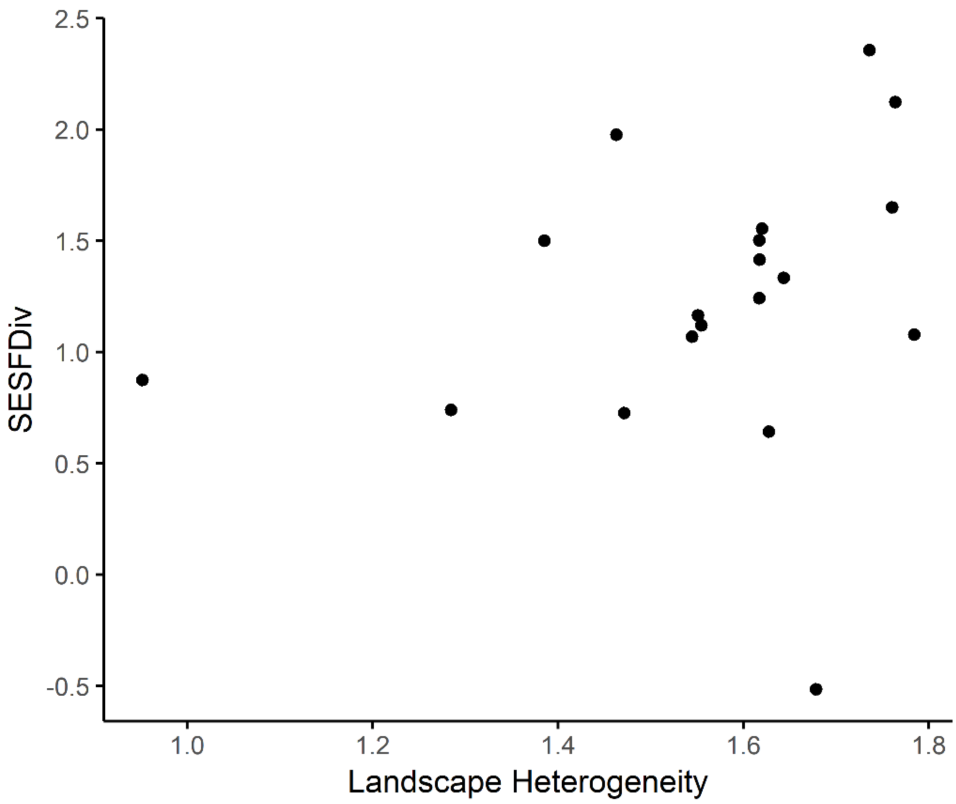

Figure S3. Relation between landscapes heterogeneity and Standardized Effect Sizes on Functional Divergence (SESFDiv).

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CAPÍTULO 2

Forest cover and landscape heterogeneity improve bird and bat pest control and increase vineyard crop yield

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FOREST COVER AND LANDSCAPE HETEROGENEITY IMPROVE BIRD AND BAT PEST CONTROL AND INCREASE VINEYARD CROP YIELD

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Abstract

Context: Modification in natural landscapes cause negative impacts in ecological functions provide by biodiversity, mainly in agricultural landscapes.

Objectives: Understand the influence of landscape composition on the trophic cascade effect of arthropod predation by birds/bats in neotropical vineyards and the consequences for crop yield.

Methods: We did experiments in 21 Brazilian vineyards (1 km radius) in a gradient of landscape heterogeneity and forest cover. Birds/bats were excluded from vines and compared them with open control. We estimated arthropod predation using plasticine caterpillars and arthropod abundance to plant damage using leaf herbivory and crop yield and the economic gain and losses in each area.

Results: Birds/bats in vineyards contributed to arthropod control, decrease herbivory percentage and increased crop yield by 19%. In addition, heterogeneity landscapes and forest cover modulated these results.

Conclusions: Our results suggest that bird and bat pest control depends on landscape metrics, such as forest percentage and heterogeneity. We highlight the importance of farmland management to preserve birds/bats to provide ecosystem services and increase economic benefits.

Keywords: biological control, ecosystem services, grape production, insectivorous birds, pest control, plasticine caterpillars.

Introduction

Ecologists and farmers often have contrasting opinions on the relevance of natural habitats (Tscharntke et al. 2016). While some people view landscapes as potentially cultivable, conservationists look at pristine areas as refuges for biodiversity and conservation, supporting ecological functions and services (Bianchi et al. 2006; Mace et al. 2012; Tscharntke et al. 2016; Maas et al. 2018). Considering that ecosystem services are human benefits of the ecological functions (Constanza et al. 1997; MEA 2005), they have been used to convince people of the importance of the environment and its conservation, and the scientific interest on these mechanisms has been increasing (McDonough et al. 2017), either in native areas or in agroecosystems (e.g. Boesing et al. 2017). Pollination, seed dispersal, nutrient cycling, and pest control are central benefits to agroecosystems, and this last is visible to farmers (see Maas et al. 2013). This service reflects in a reduction of leaf damage and plant mortality, contributing to improve agricultural crop yield and monetary income (Johnson et al. 2010; Mols & Visser 2002; Philpott, et al. 2009). Yet, ecological functions are dependent of biodiversity, and this is a liable argument for convincing farmers to manage areas for maintaining natural or semi-natural habitats, as an alternative to completely replace or reducing the use of pesticides in agriculture (Clough et al. 2011; Bennett et al. 2015 Maas et al. 2018; Girardello et al. 2019).

Birds and bats provide many important ecosystem services such as the suppression of arthropods, seed dispersal and pollination (Whelan, Wenny & Marquis, 2008; Boyles et al. 2013). Their diversity of foraging strategies and morphological characteristics enable them to provide the top-down control in both tropical forests and agroecosystems (Maas et al. 2015; Faria et al. 2006; Mäntylä, Klemola & Laaksonen 2011). More than 50% of species of birds and 70% of bats predominantly feed on invertebrates (Simmons, 2008; Whelan, Sekercioglu & Wenny 2015; Nyffeler, Şekercioğlu, & Whelan 2018) alternating the period of foraging. Yet, bird and bat assemblages are sensible to landscape structures and the intensity of farming

(Faria et al. 2006; Martensen et al. 2008), and these landscapes configurations are so important for maintenance richness and abundance species (Faria et al. 2006; Barbaro et al. 2017; Monck-Whipp et al. 2018; Sekercioglu et al. 2019). In fact, recent studies have shown that the prevalence of agricultural areas over natural habitats affects bird communities, which is mostly composed of less specialized, more widespread species, with functional groups in altered proportions (Sekercioglu 2012; Morante-Filho et al. 2015). As for birds, native vegetation and habitat heterogeneity can increase bat diversity and species richness and feeding activity as well (Sekercioglu 2012; Rodríguez-San Pedro et al. 2019). On the other hand, with less integrity assembles ecological functions provided by bats, like pest control, decay (Tscharntke et al. 2012; Dainese et al. 2019).

Typically, agricultural areas can be either heterogeneous or extremely simplified landscapes, depending on the set of different land uses (Fahrig et al. 2011). In turn, landscape homogenization can lead to disruption in pest control, due to failure in the top-down effect (Rusch et al. 2016; Dainese et al. 2019). However, the consequences vary according to landscape context, interactions among animals and local management (Maas 2018).

Over a mosaic composition, forest remnants can be the unique refuge for biodiversity (Sekercioglu et al. 2002). Considering their central relevance in providing ecosystem services, natural areas can be critical to agriculture landscapes (Tscharntke et al. 2016). For this reason, studies including landscape context have been increasing, mainly in tropical areas (Boesing et al. 2017; Morante-Filho & Faria 2017). Yet the effects of the landscape structure on the provision of biological pest control has been poorly known, mainly in the context of the agriculture productivity (Chaplin-Kramer, O'Rourke, Blitzer, & Kremen, 2011; Maas et al. 2013). Thus, landscape perspective is needed to understand the effects of agriculture land use on biodiversity and ecological process, such as arthropod predation (Tscharntke et al. 2005; Turner et al. 2013).

Studies focusing on the effects of land use on biodiversity and ecological process have been developed in the Neotropics mostly in coffee and cacao plantations (e.g. Wenny et al. 2011; Perfecto et al. 2004; Maas, Tscharntke, Saleh, Dwi Putra, & Clough, 2015; Menezes, Cazetta, Morante-Filho, & Faria, 2016). Yet, there are gaps in the knowledge on the ecosystem services played by both birds/bats in agroecosystems using empirical data (Tscharntke et al., 2008; Librán, De Coster, & Metzger, 2017), mainly in monocultures (but see Koh 2014). Links among landscape composition, the structure of birds/bats communities and mediating arthropod control are poorly known (Boesing, Nichols, & Metzger 2017; Morante-Filho & Faria 2017).

Brazil has nearly 78,000 ha of vineyards, whose production reached almost one million tons in 2016 (Mello et al. 2017). All these crops are predominantly in the Atlantic Forest (Embrapa 2017), one of the most threatened biomes by deforestation and fragmentation in the world (Myers et al. 2000). Ecological research in vineyards have been carried out worldwide (e.g. Assandri, Bogliani, Pedrini, & Brambilla, 2017a; Barbaro et al. 2017; Luck, Hunt, & Carter, 2015; Steel et al. 2017; Jedlicka, Letourneau & Cornelisse 2014, Rodríguez-San Pedro et al. 2019), but there are no previous studies managing the exclusion of top predators, and considering the landscape context and consequences for arthropod control and productivity.

Here we analyze whether the landscape composition (heterogeneity and forest cover) mediate the cascade effect of arthropod predation played by birds-and bats in Neotropical vineyards. We expect that (1) the absence of birds/bats increases arthropod abundance; (2) increasing in the population of herbivorous arthropods results in increased leaf herbivory; (3) this failure in the cascade effect results in losses to grape production and in the household incoming for failure of ecosystem services.

Specifically, we limited the access of birds/bats to vineyards and verify the effect on arthropod abundance, leaf damage, and grape crop yield, across a gradient of landscape heterogeneity and forest cover. We expect that the service provided by birds/bats in pest control and leaf damage have positive results in crop production in open control vineyards than in birds/bats exclosures. Even more, these results can be correlated with more heterogeneous areas and/or with greater forest cover.

Material and Methods

Study sites

The study was conducted in São Miguel Arcanjo (24°00’ S, 48°01’ W; Fig. 1), one of the largest grape productions in the state of São Paulo, Brazil. The area has ~40.000 ha with predominating *Vittis* spp. (Mello et al. 2017; IBGE 2018), imbibed in crops of soy, passion fruit (*Passiflora edulis*), persimmon orchards (*Diospyros* spp.), pasturelands and remnants of native Atlantic Forest (Fig. 1).

Landscape selection

We performed our experiments during two grapes harvests between 2018 and 2019 in 21 properties. To select these areas, we used images from DigitalGlobe (2016, spatial resolution of 0.5 m, 1:5000 scale). Thus, we delimited seven land-use types (remnants of Atlantic forest, regenerating forests, *Eucalyptus* plantations, grape plantations, other agricultures, open areas, urban areas; Fig. 1) with ArcGIS v 10.3 (ESRI). Information on land-uses was validated in the field and all interpretation errors were corrected. In each area with grape plantation, we established a central point with a 1000 m buffer radius (like Boscolo & Metzger, 2009; García, Miñarro, & Martínez-Sastre, 2018). After compared this scale with previous analyzes with other measures, we chosen this as the best value. We selected the 21 sampling points (Fig. 1) using FRAGSTATS v4.2.1(McGarigal & Ene 2015) following a gradient of forest cover and heterogeneity. Forest cover percentage ranged 18-55% and environmental heterogeneity - by Shannon´s diversity index (SHDI) - ranged from 0.95-1.78.

Forest cover and SHDI were not correlated (Spearman's $\rho = 0.039$, $p = 0.89$).

Predation on artificial caterpillars

To estimate arthropod consumption by birds, we designed a field experiment using green plasticine caterpillars to mimic natural foliage lepidopteran larvae (Howe, Lövei, & Nachman, 2009; Nurdiansyah et al. 2016; Peisley, Saunders, & Luck, 2016). Predation experiments were conducted during the harvesting period between January - April 2018, in 19 vineyards. All artificial caterpillars (30 mm length x 5 mm diameter) were made with green nontoxic modeling clay representing some lepidopteran larvae species that occur in vineyards (e.g. *Eumorpha vitis* and another smaller species; Haji et al. 2001; D. J. Moreno, pers. obs.). We fixed 570 caterpillars (30 per area), with nontoxic scholar glue in different parts of each vine (leaves, trunk, and fruits), aiming to analyze the effects of different foraging groups (e.g. trunk gleaning, leaf gleaning). Caterpillars were separated from others by at least 2 m (Fig. S1) We checked predation marks on all models in the early morning (nearly 06 a.m.) and in the evening (nearly sunset 5 p.m.) for estimate diurnal and nocturnal predation activities (Ferrante et al. 2017). When a caterpillar presented predation mark done by bird or bat, it was removed and not replaced. After six days, we collected all plasticine models in vineyards and quantified predation marks following papers that approach this method (e.g. Low, Sam, McArthur, Posa, & Hochuli, 2014; Nurdiansyaha, Denmeada, Clough, Wiegandc, & Tscharntkea; Howe, Lövei, & Nachman; Fig. S2).

Vineyards exclusion and treatments

We carried out experiments with bird/bat exclusion in vineyards in 14 areas along all the cycle of the vineyards since pruning branches to crop production and sales (September 2018 to April 2019; Fig. S1). In each area, we settled one 12m-experimental unit per treatment, which consisted of grape row crop with 10 plants each, resulting in 28 experimental units and 280 vines plants. They consisted of (A) no exclusion (i.e. open control; vines with no exclosure and bird and bat access all the time) and (B) birds/bats exclusion (Fig. S2). The distance between the treatments was at least 4 m and the size of the structure was made according to the size of the plants (see more information about the use of nets in Maas et al. 2019). Birds/bats exclusions were made by wood and bamboo, covered with transparent fishing nets (15 m long x 6 m width) made with 0.4 mm nylon monofilament with 35 mm mesh size. These nets allowed arthropods access, but excluded birds/bats. In the same period of study in exclusion treatments, we also performed in open control, where the length of the vine was equal, but without any kind of interference (Fig. S2).

Arthropod sampling

Arthropod communities were sampled in 14 areas using two methods, a diurnal line-transect census, and the *branch clipping* technique. For the first, we searched for arthropods in all part of the vines in each treatment for 20 min per area. We documented all data with digital photographs and described some arthropod morphological characteristics (e.g. color, morphological type), place on a grapevine (leaves, trunk or fruits) and their activity (e.g. foraging; see Maas, Clough, & Tscharntke, 2013). We also sampled arthropods using the *branch clipping* technique (see Cooper and Whitmore 1990). We collected one branch of per treatment on the day of the harvest using 30 cm black plastic bags. The branch was inserted into the bags, shaken and pruned with all the insects there. After that, the bags were frozen, and the insects were screened in the laboratory and fixed in 70 % alcohol. We aimed to identify arthropods with the highest taxonomic resolution as possible following Ruppert et al. 2005, Triplehorn & Johnson 2011, Haji 2001.

Herbivory

To estimate leaf damage, we randomly collected 30 leaves from each treatment in vineyards along the harvesting, totalizing 840 leaves in 14 areas. Leaves were picked up in all sections of a plant (top to bottom), including young and old leaves. In the laboratory, we calculated leaf damage (in %) done by insects only. Using ImageJ (Rasband W., 2003), we measured the total area of the leave and calculated the herbivory.

Fruit production and valuation

We evaluated the grape production in 12 areas; two experimental areas were discarded due to grape theft or fungal infestation. We estimated vineyard production and monetary value using the kilogram of fruits produced in each treatment row crop (i.e. the local where were our experiments) and the value of grape kilogram sold; then it was extrapolated to 1 ha, considering that the area is occupied for ~400 grapes row crops (~4,000 grapes plants). We obtained vineyards production prices by interviews with farmers, considering then the productive value obtained at the first point of sale (Primack 2014). As monetary values were originally available in Brazilian Reais, we converted to an approximate value for the US Dollars using the quotation of the day.

Data Analyses

To analyze the effects of forest cover and landscape heterogeneity on predation of artificial caterpillars, we performed generalized linear models with binomial error distribution. We evaluated the significance of each variable by comparing models assessing the goodness of fit by likelihood-ratio test (Quinn and Keouh 2002).

We compared insect abundance, herbivory percentage and grape yield between exclusion and open control treatments using paired t-tests. To determine the relationship between the effect size of bird/bat exclusion and landscape metrics (i.e. landscape heterogeneity and forest cover), we fitted a linear regression model with the effect size of bird/bat exclusion as the response variable. The effect size of bird/bat exclusion was given as the ration between the values obtained in exclusion treatment and open control (arthropod abundance and herbivory) or between the values obtained in exclusion treatment and open control. To satisfy the assumptions of normality, we did log-transformations on arthropod abundance, herbivory percentage, and their variables of effect size prior to the analyses. All statistical tests were performed using R (R Development Core Team 2018).

Results

Caterpillar predation

Pecking marks by birds were present in 13.26 ± 14.94% of the artificial caterpillars (Fig. S2). We have caterpillars attacked by all the other groups together (e.g. snails, ants, mammals) and some lost ones, representing 19.65 ± 10.18% and 3.15 ± 4.78%, respectively. Only one caterpillar had bat marks.

Caterpillar predation by birds varied between areas due to the landscape heterogeneity, forest cover and the interaction between these two variables (Table S1). The predation of caterpillars was higher in landscapes with a high percentage of forest cover but with low heterogeneity (Table 1). On the other hand, predation was also higher in landscapes more heterogeneous but with low forest cover (Table 1; Fig. 2).

Effects of exclusion on arthropod abundance

We found no difference in arthropod abundance between bird/bat exclusion treatment (148.79 ± 230.80; mean ± SD) *vs.* open control treatment (176.79 ± 197.89) (t paired = 0.23, df = 13, p = 0.59, Fig. 3A). On the other hand, the effect size of bird/bat exclusion varied significantly as a function of forest cover, landscape heterogeneity, and the interaction between these landscape metrics (Table S2). The effect size of the bird/bat exclusion on arthropod abundance was higher (i.e. greater abundance in exclusion treatment than control treatment) in landscapes with a high proportion of forest cover but with low heterogeneity (Table 2, Fig. 3B). The effect size was also higher in landscapes more heterogeneous but with low forest cover (Table 2, Fig. 3B).

Herbivory

A total of 357 (42.5%) of the sampled leaves had damage caused by herbivorous insects. We found a positive correlation between insect abundance and percentage of leaf damage in the control treatments ($r = 0.64$, $n = 14$, $p = 0.014$), but this correlation was not found in the exclusion treatments ($r = -0.15$, $n = 14$, $p = 0.60$).

The herbivory in open control treatment (0.70 ± 0.69%) was on average 2.6 lower than in birds/bats exclusion treatment (1.83 ± 0.86%) (paired t-test = 5.83, df = 13, $p < 0.0001$, Fig. 4A). The effect size of bird/bat exclusion varied significantly as a function of forest cover, landscape heterogeneity, and the interaction between these landscape metrics (Table S2). The effect size of the bird/bat exclusion on herbivory was higher in landscapes with a high proportion of forest cover but with low heterogeneity (Table 2, Fig. 4B). The effect size was also higher in landscapes more heterogeneous but with low forest cover (Table 2, Fig. 4B).

Fruit production and valuation

We found a negative correlation between arthropod abundance and grape production ($r = -0.59$, $n = 12$, $p = 0.043$) and between herbivory and grape production ($r = -0.66$, $n = 12$, $p = 0.019$) in the control treatments. However, these correlations were not found in the exclusion treatment (arthropod abundance *vs*. grape production: $r = -0.32$, $n = 12$, $p = 0.32$; herbivory *vs.* grape production: $r = -0.023$, $n = 12$, $p = 0.94$).

The grape production (by vineyard row crop) in open control treatment (48.0 ± 17.6 kg) was on average 1.2 times greater than in bird/bat exclusion treatment (38.7 ± 11.6 kg) (paired t-test = 2.254, df = 11, $p = 0.023$, Fig. 5A). In the same way, the effect size of bird and bat exclusion varied significantly as a function of forest cover, landscape heterogeneity, and the interaction between these landscape metrics (Table S2). The effect size of the bird and bat exclusion on herbivory was higher (i.e. greater grape production in control treatment than exclusion treatment) in landscapes with a high proportion of forest cover but with low heterogeneity (Table 2, Fig. 5B). The effect size was also higher in landscapes more heterogeneous but with low forest cover (Table 2, Fig. 5B).

Our results indicate that bird and bat arthropod control can increase on average 9.3 kg of grapes per row crop of vines. Considering that one hectare has an average of 400 row crop (area of a row crop ~25 m²), it is possible to estimate that the ecosystem services played by birds/bats could increase the grape harvest by 3,720 kg/ha. Given that a kilogram of grapes is sold for R$ 2,5 at the first selling point (~ USD 0.60) it is possible to estimate that the value of the pest control service by birds/bats is R$ 9,300.00 per ha (~ USD 2,229.89 per ha). When considering the landscape metrics, this service may increase in areas with high forest cover or high heterogeneity (Table 2). For example, in landscapes whose forest cover is 48% or heterogeneity is 1.78, the difference between grape production where birds/bats have access (open control) and where they are excluded may be the double (Fig. 5B). Therefore, in these areas, the values of ecosystem services of pest control performed by birds/bats may be even higher.

Discussion

Our study provides the first empirical evidence of the role of birds/bats in vineyards, contributing to arthropod control and increaseing crop yield. We also show that landscape heterogeneity and forest cover can maximize the pest control in grapes, decreasing leaf herbivory and contributing to agricultural production. To farmers, the loss of pest control service provided by birds/bats to vines productivity costs ~USD 2,230 per hectare.

Pest control by birds and bats

With these multidimensional analyzes, we for the first time have valued financially the ecosystem services delivered from the arthropods control played by birds/bats on Neotropical vineyards. Our results appear consistent across experiments (significant correlations between experiments) and support our hypothesis that the trophic cascade by birds/bats can decrease the abundance of herbivorous arthropods (although we found no difference between treatments), decreasing the herbivory percentage with consequences in higher grape yield. These results are in accordance to previous studies highlighting similar effects in other cultures (e.g. Maas, Clough & Tscharntke 2013; Librán-Embid, De Coster and Metzger 2017).

Results from predation experiments on artificial caterpillars suggest that birds may play a more significant role in insectivory than bats. This is because we found more caterpillars with pecking marks than with bat bite marks. One possible explanation may be related to the differential foraging behavior of both groups and prey location, although we have settled the caterpillars in the most varied places of the plants. While some birds forage in all parts of the plant, such as tree trunks and leaves (Holmes and Schultz 1988), bats can use echolocation to detect immobile prey, being more efficient in more external parts of the plants (Kalko and Schnitzler 1993). This is because, some bats species can avoid overlap of echoes from potential prey and obstacles (Kalko and Schnitzler 1993). In addition, it is possible that the use of artificial caterpillars is not a good predictor of bat arthropod predation (Seifert et al. 2016), although it is a recognized method for the efficiency in evaluating arthropod predation by birds (Howe, Gabor & Nachman 2009). On the other hand, although our results show that birds may have preyed on caterpillars more than bats, these last are known to be effective consumers of arthropods in vineyards, playing a key role in pest control (Baroja et al. 2019). Our results are limited in attributing the specific importance of birds/bats in this service.

Contrary to our expectations, we found no difference in the abundance of arthropods between treatments. This may have occurred because the structure of exclosures can attract spiders disproportionately (Maas et al. 2018), which can also prey on some arthropods within the exclusions and thus dilute the result between the two treatments (Karp & Daily 2014; Gras et al. 2016). On the other hand, in the control treatment, we found a positive correlation between arthropod abundance and herbivory, but this relationship does not occur in the exclusion treatment. Other studies have also shown that leaf damage was not correlated to insect abundance (e.g. Maas, Clough & Tscharntke 2013), reflecting that not all arthropod species within the screen may act as herbivores. In addition, we recognize that our arthropod survey protocol may have been limited. For instance, we did not collect at night when some herbivores such as Orthoptera and leaf-cutter-ants are more active (Maas, Clough & Tscharntke 2013).

Birds/bats exclosures in vineyards resulted in a percentage of average herbivory more than twice that in control treatment. Although we found no difference in the number of arthropods beteween our study, the increase of herbivory in exclosures suggest that birds/bats may limit the activity of arthropods by reducing their abundance. To our knowledge, there are no estimates of arthropod leaf damage in vineyards. However, our results are similar to other experiments in some crops such as coffee and cocoa (Cassano et al. 2016; Morrison & Lindell 2012; Van Bael & Brawn 2005; Maas et al. 2016). Leaf damage can generally affect plant reproductive capacity (e.g. fruit yield; Marquis 1984; Blue et al. 2015). Here we find a negative correlation between herbivory and grape production in control treatments, indicating that herbivory can have a significant impact on yield. In fact, grape production was also significantly reduced excluding birds/bats, where it decreased by 19%. Some species of birds/bats are known to forage upon vineyards, damaging and eating or damaging the grapes (Somers & Morris 2002; D. J. Moreno pers. obs.). However, our results show an income of almost USD 2,230 per ha as a possible effect of arthropod control. Indeed, few studies have estimated the crop yield and the economic impact of arthropod control by birds/bats before. There are results for apples (Mols & Visser 2002), coffee (Librán, De Coster and Metzger 2017), and cacao (Maas, Clough & Tscharntke 2013) increasing yield, but this is the first results for vineyards, considering landscape composition.

Landscapes effects

Our study also shows that caterpillar predation by birds, the effect size of bird and bat exclusion on arthropod abundance, herbivory and grape production, have varied depending on the landscape. Therefore, our results also have important implications for vineyard management, as the landscape can affect pest control by birds/bats and crop production in vineyards. Caterpillar predation by birds was higher in landscapes with higher heterogeneity or higher forest cover, suggesting a more important provision of pest control services in these landscapes. This is because more heterogeneous landscapes can provide a wider range of habitats and resources, while areas with higher forest cover can provide more preserved habitats (Fahrig 2001; Fahrig et al. 2011). Thus, these landscapes can shelter a diversity of arthropod consumers, thus providing a more efficient pest control service (Sekercioglu 2012; Assandri et al. 2017a). Indeed, many studies highlight the importance of heterogeneity in bird/bat diversity, as well as the provision of ecosystem services in vineyards (Moreno et al. in prep.; Barbaro et al. 2017; Rodriguéz-San Pedro et al. 2019) or other crop types (Monck-Whipp et al. 2018; Liden et al. 2019). In a study on the functional diversity of birds in these same vines, Moreno et al. (in prep.) showed that functional diversity can increase in more heterogeneous landscapes. Similarly, Barbaro et al. (2017) showed the predation of artificial caterpillars in French vineyards may be higher in more heterogeneous landscapes when functional uniformity is also high. Similar patterns have been found in other cultures showing the role of heterogeneity in pest control in (Kellermann et al. 2008). On the other hand, regarding the importance of forest cover, Librán-Embid, De Coster and Metzger (2017) also showed that the loss of coffee leaves may be lower in landscapes with greater local forest cover and the fruit set in the bird and bat exclusion treatments also followed a similar pattern.

We found that heterogeneity had a smaller effect size on the proportion of caterpillars predation than that provided by forest cover, indicating that it is more advantageous for smallholders to maintain larger sets of natural forest cover instead of more heterogeneous landscapes, for efficient pest control. Interestingly, this pattern is true even in analyzes of arthropod abundance and herbivory percentage, which are also included in bats, suggesting that the role of bats in pest control may also be influenced by landscape. Thus, the effect size of bird and bat exclusion on arthropod and herbivore abundance (i.e., ratio of arthropod abundance/herbivory percentage between exclusion and control) also increases in landscapes with higher forest cover and more attenuated way in landscapes with high heterogeneity. On the other hand, this interpretation must be made with caution. The effect size of bird and bat exclusion on grape production was very similar in landscapes with high coverage and high heterogeneity, indicating that these two landscapes metrics may play similar roles in bird and bat performance on grape production.

On average, the value of the bird and bat pest control service is around USD 2300 per ha. However, this value may vary depending on the landscape characteristics. For example, the size of the effect of bird and bat exclusion on grape production ranged from 0.77 to 2.33 being larger in highly heterogeneous or high forested landscapes and smaller in low heterogeneity and low forested landscapes (Fig. 5B). Effect size values greater than zero means that grape yield in the control treatment was greater than in the exclusion treatment, while effect size values over than zero mean the opposite. Thus, birds/bats may become ineffective in controlling pest populations and/or rendering a disservice (e.g. over-consumption of grapes) in more homogeneous landscapes with low forest cover. This is because homogenization of deforested landscapes is likely to lead to a change in species composition or species functions (De Coster, Banks-Leite & Metzger 2015; Morante-Filho et al. 2015), or the scarcity of resources in these areas leads species to seek alternative sources, such as Tobin (1984). On the other hand, birds/bats can become more efficient pest controlers in heterogeneous landscapes or with greater forest cover. Thus, monetary values for services may be even higher in these landscapes.

Our findings show the importance of habitat heterogeneity and/or remnant habitat patches for birds/bats in anthropogenic landscapes (Kelly et al. 2016; Sekercioglu et al. 2002; Bereczki et al. 2014). In addition to higher species richness, vineyard bat activity and specialists forest birds are higher when there are native vegetation forest structures, trees and shrubs and greater habitat diversity surrounding the crops (Rodriguéz-San Pedro et al. 2018, 2019; Sekercioglu 2019). Therefore, considering this issue of vineyard landscape may favor foraging in order to obtain greater efficiency of these animals in pest control. Our results also suggest the negative effects of agricultural intensification on pest control provided by birds/bats in Brazilian vineyards.

Applications

Our results can lead to better vineyard management, benefiting both biodiversity and farmers, as green products may be more interesting in the consumer market today (see Wilson and Daane 2017). Consideration of landscape composition in agriculture should be incorporated into vineyards by farmers and other stakeholders, and in this sense, economic valuation of biodiversity can be an interesting tool for convincing and raising environmental awareness, thus maximizing the benefits derived from biodiversity with conservation practices in a threatened hotspot.

Authors' contributions

D.J.M., M.C.R and A.J.P conceived the idea and designed the studies; D.J.M. installed the experiments and collected the data; D. J. M., M.C.R and A.J.P. analyzed the data and wrote the manuscript.

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Table 1. Summary of parameters of the model of artificial caterpillars preyed on by birds explained by the proportion of forest cover and landscape heterogeneity.

	β	**SE**	**z**	**p**
Intercept	-6.088	2.947	-2.065	0.039
Forest cover	27.017	9.941	2.718	0.007
Landscape heterogeneity	2.078	2.065.	1.007	0.314
Forest cover x Landscape heterogeneity	-15.908	6.984	-2.278	0.022

Table 2. Summary of parameters of the models to explain the effect size of bird/bat exclusion on arthropod abundance, herbivory and grape production in relation to proportion of forest cover and landscape heterogeneity.

	B	**SE**	**t**	**p**
Effect size of bird/bat exclusion on arthropod abundance				
Intercept	-26.642	7.964	-3.345	0.0074
Forest cover	112.743	35.107	3.211	0.0093
Landscape heterogeneity	15.411	5.165	2.984	0.0137
Forest cover x Landscape heterogeneity	-66.440	22.388	-2.968	0.0141
Effect size of bird/bat exclusion on herbivory				
Intercept	-31.502	12.070	-2.610	0.0260
Forest cover	142.460	51.428	2.770	0.0198
Landscape heterogeneity	20.166	7.712	2.615	0.0258
Forest cover x Landscape heterogeneity	-85.635	32.327	-2.649	0.0244
Effect size of bird/bat exclusion on grape production				
Intercept	-23.520	2.754	-8.541	< 0.0001
Forest cover	63.607	7.559	8.414	< 0.0001
Landscape heterogeneity	14.921	1.665	8.963	< 0.0001
Forest cover x Landscape heterogeneity	-38.413	4.642	-8.276	< 0.0001

Figure legends

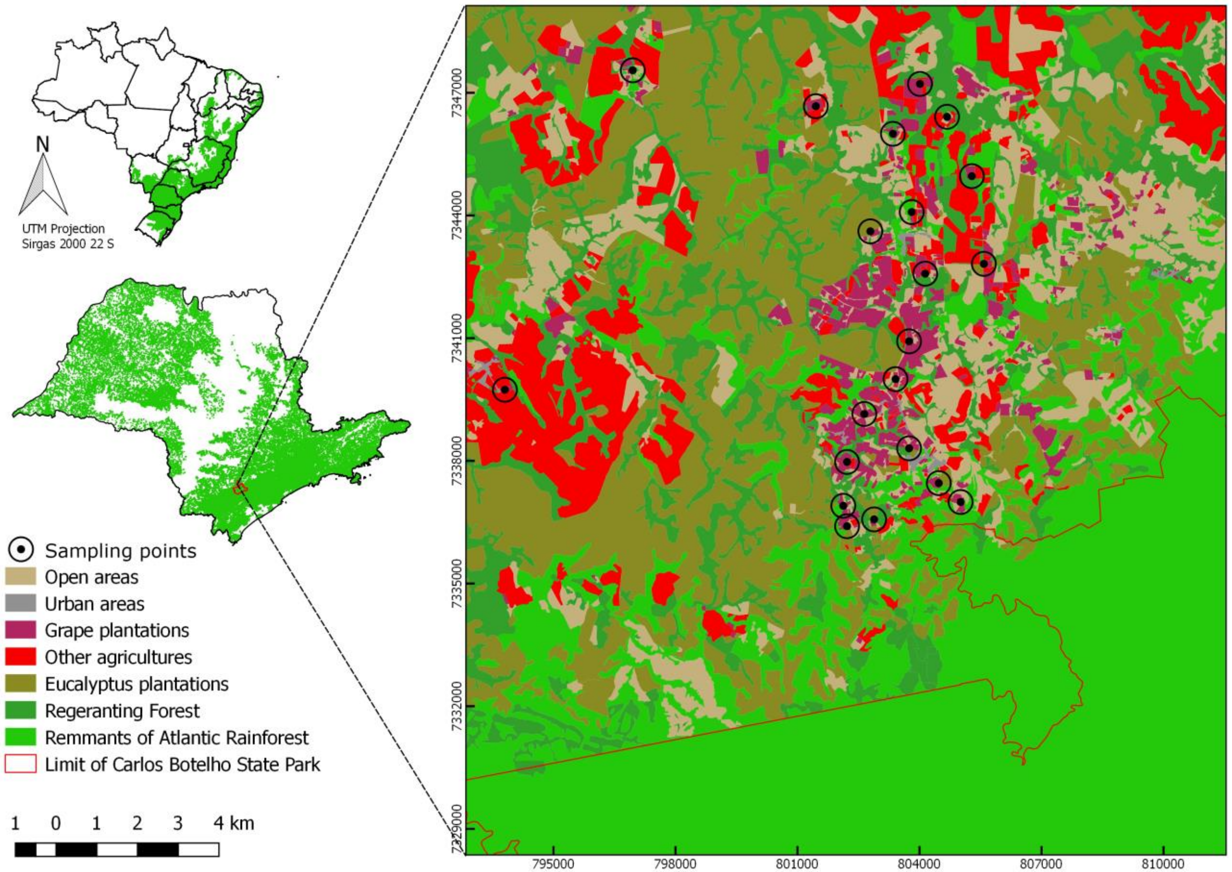
Fig. 1. Map with the locations of sampled vineyards in Southeast Brazil. Black circles indicate 21 surrounding landscapes buffers with 19 experimental caterpillar plots and exclusions experiments. Different colors in map represent different land use classes.

Fig. 2. Effects of landscape heterogeneity and forest cover on bird predation over plasticine models caterpillar in vineyards.

Fig. 3. Effects of bird/bat exclusion experiments on arthropod abundance in Neotropical vineyards (A). Relationship of landscape heterogeneity and forest cover on effect size of bird/ bats exclusion on arthropod abundance (B). Effect size was calculated as the ratio of the arthropod abundance in the bird/bat-exclosure treatment against the paired control treatment.

Fig. 4. Effects of birds/bats exclusion and open control on herbivory percentage in Neotropical vineyards (A). Relationship of landscape heterogeneity and forest cover on effect size of bird/bats exclusion on herbivory percentage (B). Effect size is calculated as the ratio of the herbivory percentage in the birds/bats-exclosures treatment to that in the paired control treatment.

Fig. 5. Effects of birds/ bats exclusion and open control on grape production in neotropical vineyards (A). Relationship of landscape heterogeneity and forest cover on effect size of bird and bats exclusion on grape production (B). Effect size is calculated as the ratio of the grape production in the control treatment to that in the paired bird/bat-exclosure treatment.


N
UTM Projection
Sirgas 2000 22 S
Sampling points
Open areas
Urban areas
Grape plantations
Other agricultures
Eucalyptus plantations
Regeranting Forest
Remmants of Atlantic Rainforest
Limit of Carlos Botelho State Park
1 0 1 2 3 4 km
7347000
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795000
798000
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804000
807000
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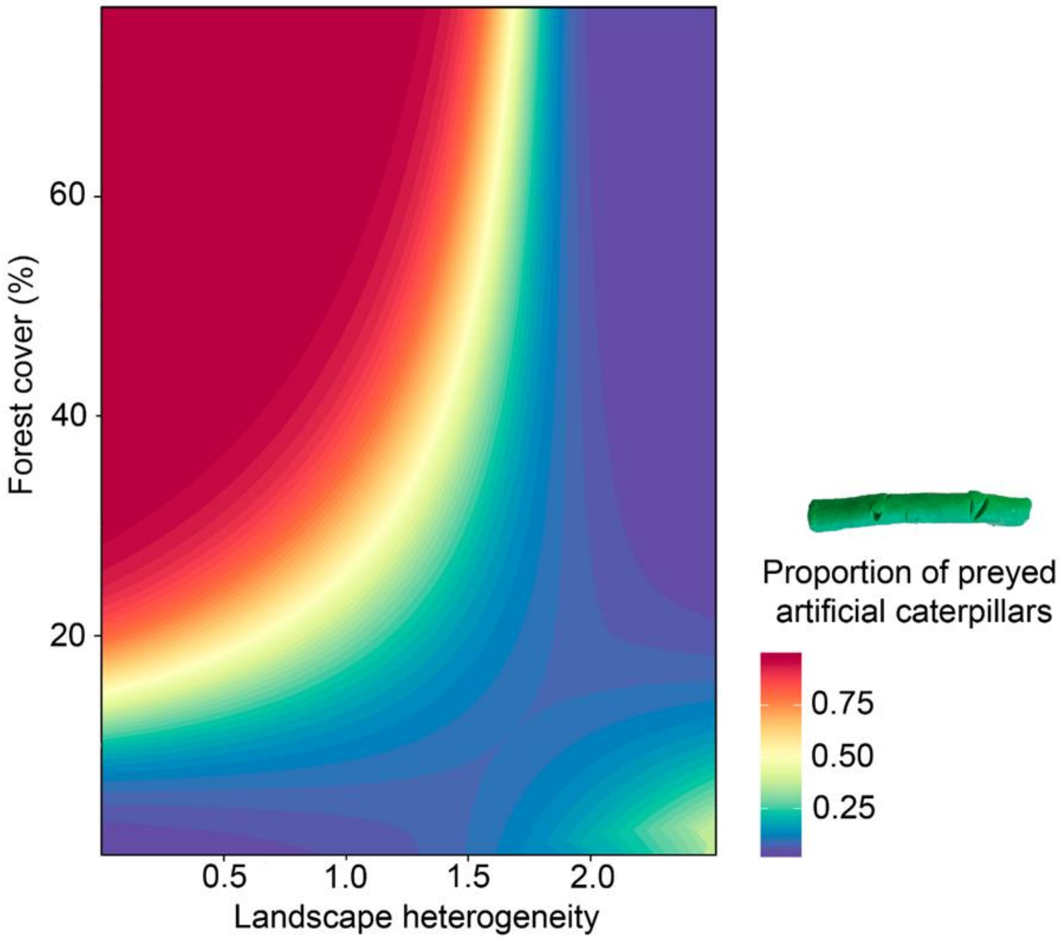
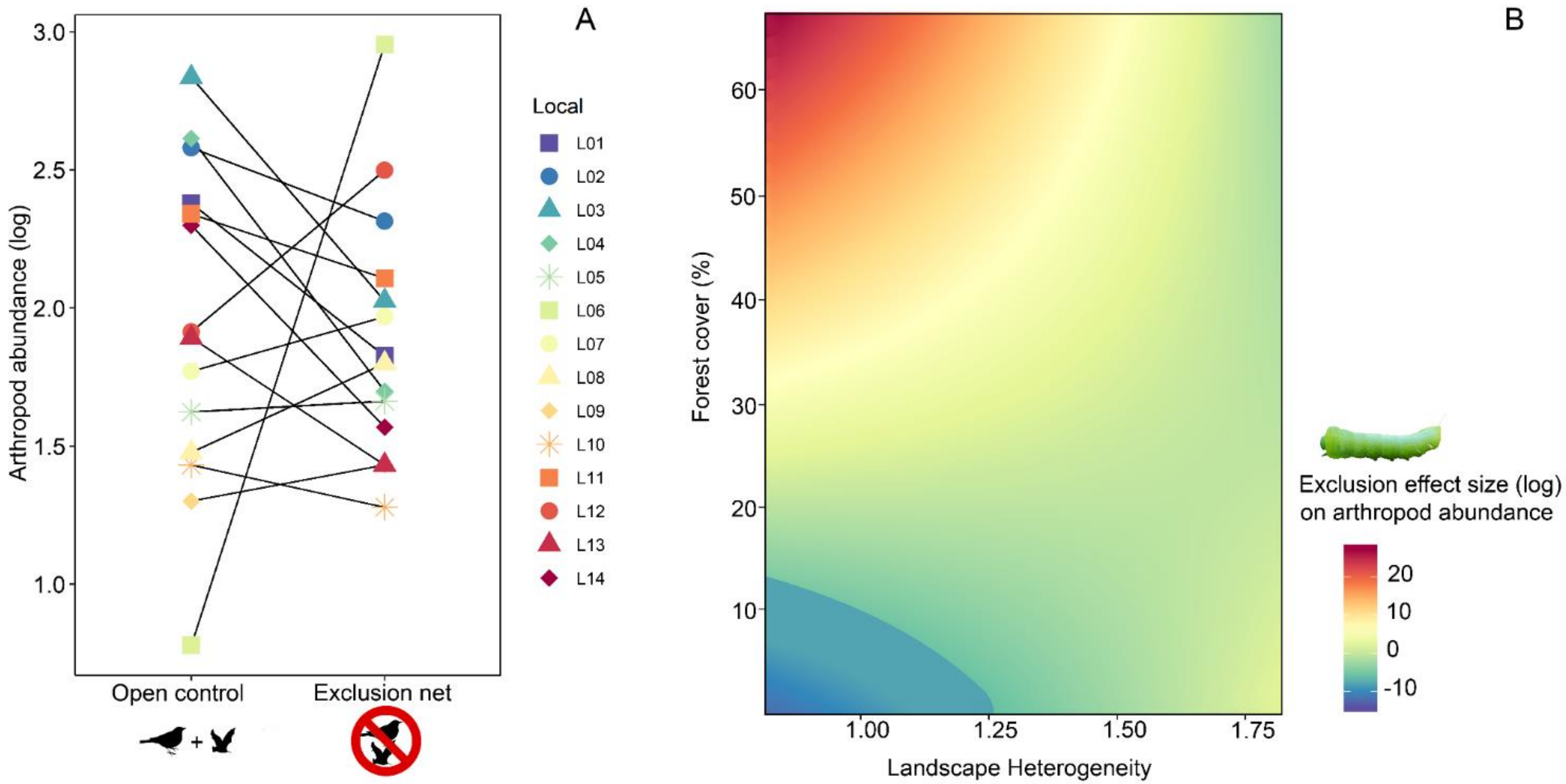
Fig. 1.


60
40
20
Forest cover (%)
0.5
1.0
1.5
2.0
Landscape heterogeneity
Proportion of preyed artificial caterpillars
0.75
0.50
0.25

774

775 Fig. 2.



776

777 Fig. 3.

778

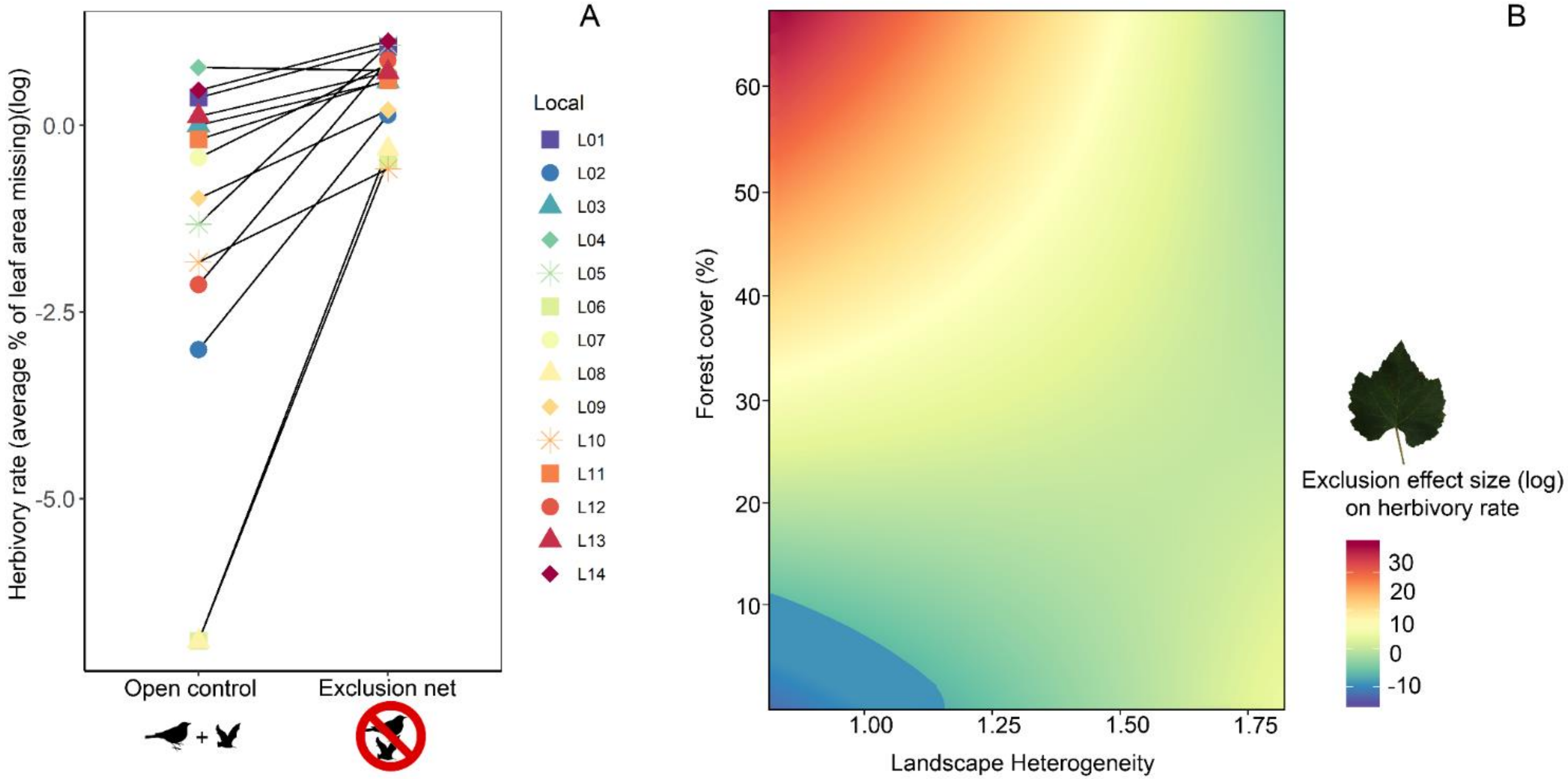
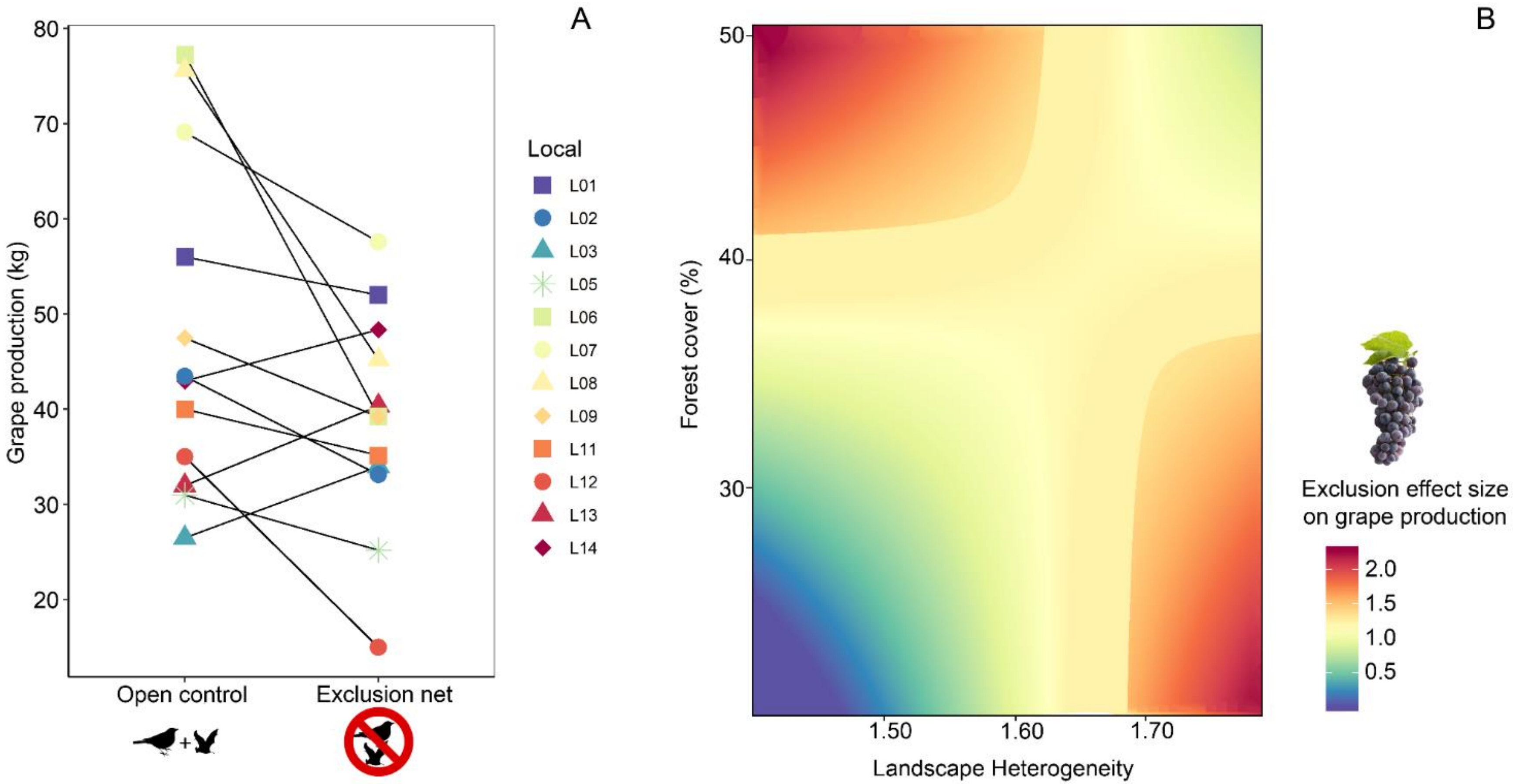

A
Herbivory rate (average % of leaf area missing)(log)
0.0
-2.5
-5.0
Open control
Exclusion net
Local
L01
L02
L03
L04
L05
L06
L07
L08
L09
L10
L11
L12
L13
L14
B
Forest cover (%)
60
50
40
30
20
10
1.00
1.25
1.50
1.75
Landscape Heterogeneity
Exclusion effect size (log)
on herbivory rate
30
20
10
0
-10

Fig. 4.


A
80
70
60
50
40
30
20
Grape production (kg)
Open control
Exclusion net
Local
L01
L02
L03
L05
L06
L07
L08
L09
L11
L12
L13
L14
B
50
40
30
Forest cover (%)
1.50
1.60
1.70
Landscape Heterogeneity
Exclusion effect size
on grape production
2.0
1.5
1.0
0.5

781

782 Fig. 5.

Supplementary Materials for

Forest cover and landscape heterogeneity improve bird and bat pest control and increase vineyard crop yield

Daniele J. Moreno*, Milton C. Ribeiro, Augusto J. Piratelli

*Corresponding author: e-mail: da_moreno88@hotmail.com

Table S1. Model selection to explain plasticine caterpillar models preyed on by birds and caterpillars preyed on by other groups in relation to forest cover and landscape heterogeneity. The best model is underlined.

Tested variables	**χ^2**	**df**	**P**
Caterpillars preyed on by birds			
<u>Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity</u>	32.453	3	< 0.0001
Forest cover + Landscape heterogeneity	5.399	1	0.020
Caterpillars preyed on by other groups			
Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity	5.002	3	0.1716

Table S2. Model selection to explain the effect size of birds/bats exclusion on arthropod abundance, herbivory and grape production in relation to forest cover and landscape heterogeneity. The best model is underlined.

Tested variables	χ^2	df	P
Effect size of bird/bat exclusion on arthropod abundance			
<u>Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity</u>	26.757	3	0.0005
Forest cover + Landscape heterogeneity	13.312	1	0.0030
Effect size of bird/bat exclusion on herbivory			
<u>Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity</u>	39.732	3	0.0279
Forest cover + Landscape heterogeneity	30.615	1	0.0081
Effect size of bird/bat exclusion on grape production			
<u>Forest cover + Landscape heterogeneity + Forest cover x Landscape heterogeneity</u>	2.231	3	< 0.0001
Forest cover + Landscape heterogeneity	1.872	1	< 0.0001

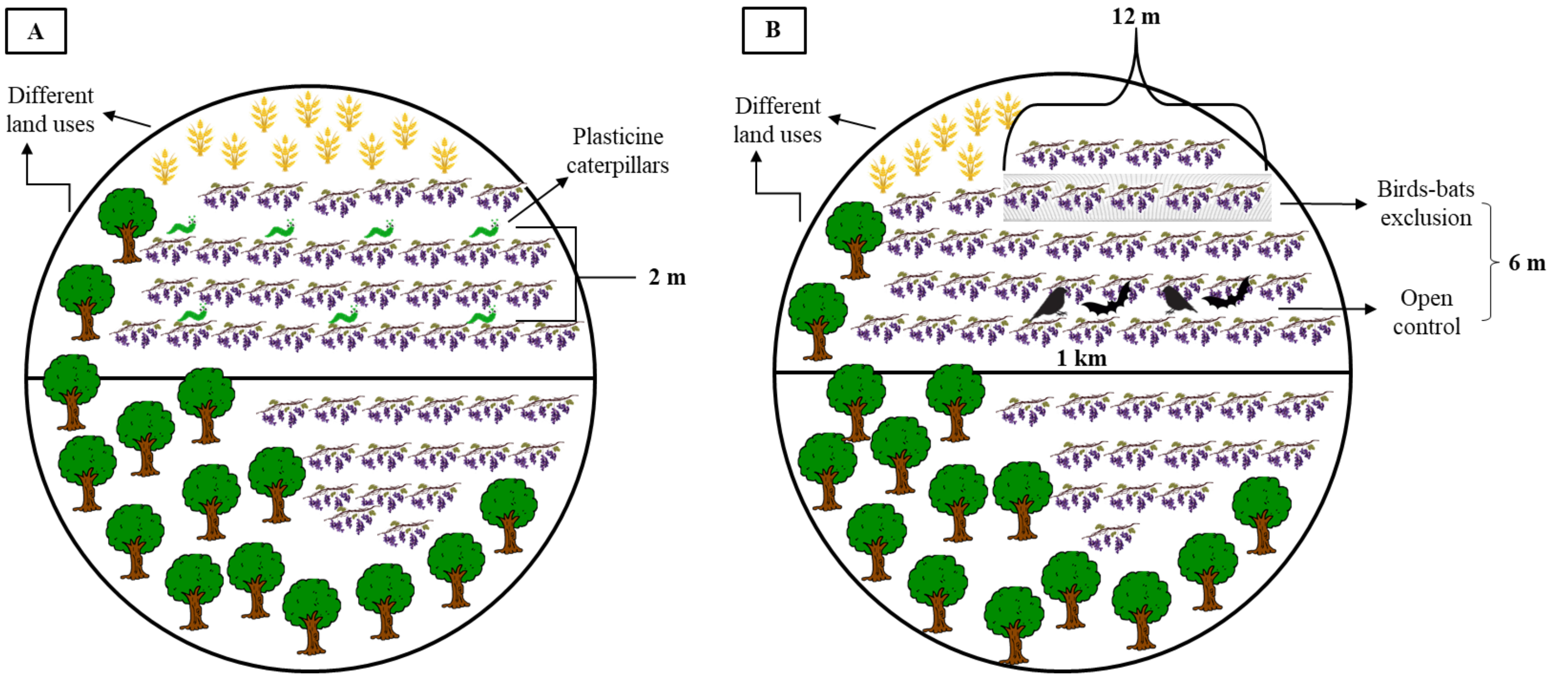


Fig. S1. Field sampling scheme. A: Testing predation in different land uses with plasticine caterpillars. Each caterpillar was placed at least 2 m between each other in the vineyards. B: Treatments to measure the influence of birds and bats on arthropod communities. Birds-bats exclusion with fish nets was at least 6 m from ant exclusion with insect glue and more than 6 m from open control. The diameter of the buffer was 1 km. Drawings are for illustrative purposes only and do not reflect the correct scale.

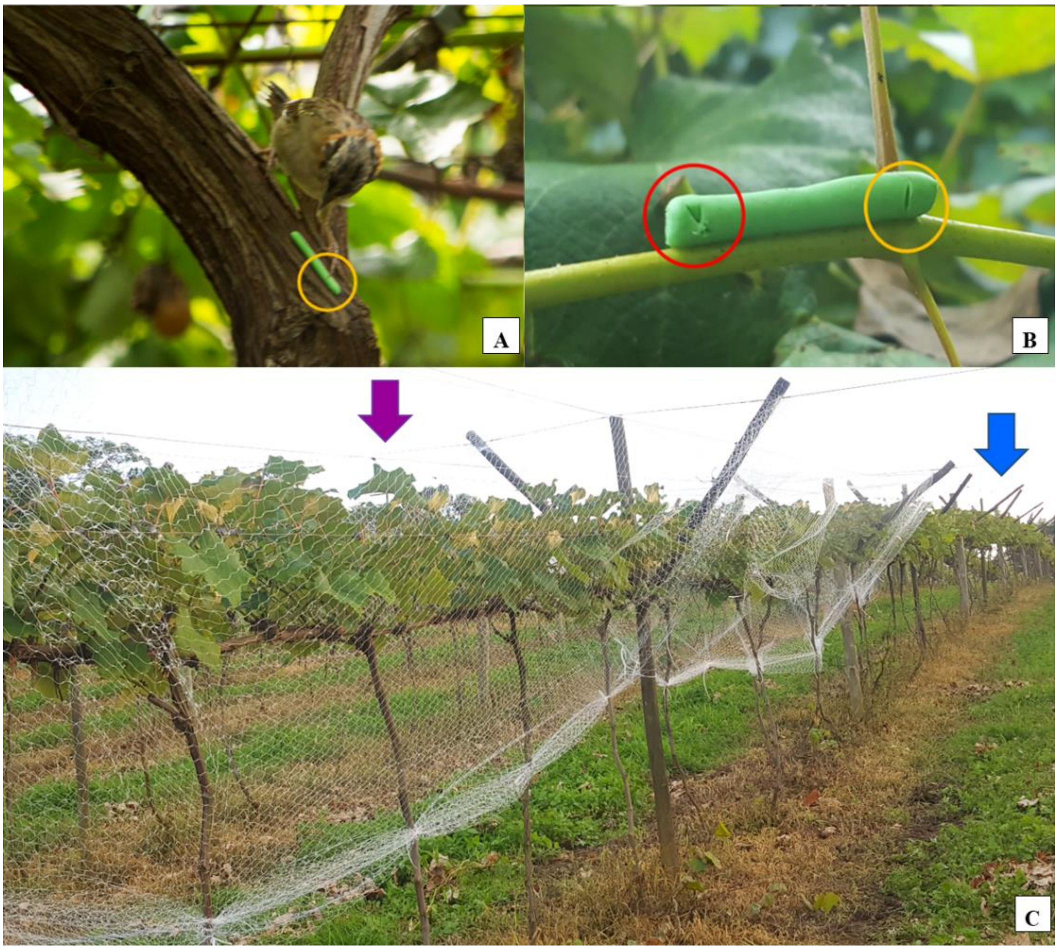
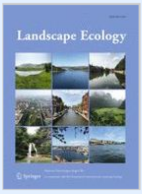


Fig. S2. A: Rufous-collared Sparrow *Zonotrichia capensis*, one of the most frequent bird in vineyards, picking a plasticine caterpillar. B: red circle shows bird beak mark; yellow circles show bird claws marks. C: Birds-bats exclusions made with nylon nets (purple narrow) and open control (blue narrow).

Normas dos periódicos



Landscape Ecology

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The following types of papers are published:

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Prof. Dan Kashian

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Manuscript Submission

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State the location, design, and major method(s) and procedures of your study. Do not include details on equipment, statistics, and modeling methods. This section should be 2 to 3 sentences.

Results.

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Conclusions.

State the significance and implications of your key findings in a broader context. Also, does your study lead to any recommendations relevant to landscape management and planning? This section should be 2 to 4 sentences.

All the elements under each heading described above must be covered. The total length of the structured abstract should not exceed 250 words. The language used in the abstract should be concise and precise. The use of adjectives is generally discouraged unless it is necessary.

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Always use footnotes instead of endnotes.

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Citation

Cite references in the text by name and year in parentheses. Some examples:

Negotiation research spans many disciplines (Thompson 1990).

This result was later contradicted by Becker and Seligman (1996).

This effect has been widely studied (Abbott 1991; Barakat et al. 1995a, b; Kelso and Smith 1998; Medvec et al. 1999, 2000).

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Reference list entries should be alphabetized by the last names of the first author of each work. Order multiauthor
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Two authors: (Mathes & Severa, 2004) and Mathes and Severa (2004)

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In Reference List:

Journal article

Example of reference with 2 to 7 authors

Beers, S. R. , & De Bellis, M. D. (2002). Neuropsychological function in children with maltreatment-related posttraumatic stress disorder. The American Journal of Psychiatry, 159, 483–486. doi:10.1176/appi.ajp.159.3.483

Ramus, F., Rosen, S., Dakin, S. C., Day, B. L., Castellote, J. M., White, S., & Frith, U. (2003). Theories of developmental dyslexia: Insights from a multiple case study of dyslexic adults. Brain, 126(4), 841–865. doi: 10.1093/brain/awg076

Example of reference with more than 7 authors

Rutter, M., Caspi, A., Fergusson, D., Horwood, L. J., Goodman, R., Maughan, B., … Carroll, J. (2004). Sex di□erences in developmental reading disability: New _ndings from 4 epidemiological studies. Journal of the American Medical Association, 291(16), 2007–2012. doi: 10.1001/jama.291.16.2007

Book edition

Bradley-Johnson, S. (1994). Psychoeducational assessment of students who are visually impaired or blind: Infancy through high school (2nd ed.). Austin, TX: Pro-ed.

Edited book

Hawkley, L. C., Preacher, K. J., & Cacioppo, J. T. (2007). Multilevel modeling of social interactions and mood in

lonely and socially connected individuals: The MacArthur social neuroscience studies. In A. D. Ong & M. Van Dulmen (Eds.), *Oxford handbook of methods in positive psychology* (pp. 559–575). New York, NY: Oxford University

Press.

Data sets

For any data with a unique identi_er the format should be as follows:

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Olden, J. (2015). Integrating landscape connectivity and invasion vulnerability to guide o□ensive and defensive invasive species management. *_gshare*. https://dx.doi.org/10.6084/m9._gshare.1285847.v2

Citations from web pages:

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