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Understanding the threshold of species loss by combining local knowledge with in-field data on mammal and bird species in the Cerrado Hotspot

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Entendendo o limiar da perda de espécies combinando conhecimento local com dados em campo de espécies de mamíferos e aves no Hotspot do Cerrado

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To Jose, to Martin. All my love.

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General abstract

Human-induced land use change is the most important threat affecting the survival of several species and reducing the provision of ecosystem services for local communities. Recent studies have shown that once the vegetation cover of the landscapes reaches certain levels, the number of species decline faster, which is called threshold of species loss. It is urgent to undertake such studies in human-modified landscapes in order to provide relevant information for decision making and conservation in private and public lands. Furthermore, it is highly relevant to include non-forested ecosystems in the scope, such as the Cerrado Hotspot, since those types of ecosystems have been often neglected for conservation in Brazil. This study approaches the threshold of species loss in human-modified landscapes in three different perspectives. First, it makes a review of empirical studies word-wide that use threshold of species loss approach with birds, and finds 31 papers published from 1994 to 2018, with 24 studies conducted at temperate latitudes and seven in tropical regions, remarking the increasing tendency of the studies and their potential application to conservation and restoration strategies of landscapes for bird conservation. Then, it performs an empirical research at the Serra da Bodoquena Plateau with 18 medium and large mammal and six bird species using data collected with camera traps. It focuses on the 9 mammals and 2 birds that negatively responded to native vegetation loss at 500 m buffer, resulting in an average threshold of 56% of native vegetation cover. When interpolating this value in modeled anthropogenic use conversion maps for 2030 and 2050 to project how the occupancy probability will change over time, the predicted annual loss found was 22.6 km^2 above the average threshold value, indicating that almost half of current area with values above the threshold will be below them by 2050. In addition, this study explores a more social component by investigating the local inhabitant's perception about mammal richness and

decline at property level. In order to understand differences in perceptions according to main economic activity, I performed interviews to 37 local inhabitants dedicated to agriculture production, cattle ranching and tourism in a Cerrado area of Mato Grosso do Sul state. Although I find no significant difference in the total richness perceived according to economic activity, there is a significant difference in the richness perception of open areas and forested species within touristic properties (t 0.0194, n=6), which suggest that this category of land owners, relying on economic activities directly related to biodiversity, have a better knowledge about their biodiversity and may be willing to protect larger tracts of forests within their properties.

Resumo geral

A mudança no uso da terra induzida pelos seres humanos é a ameaça mais importante que afeta a sobrevivência de várias espécies e reduz a provisão de serviços ecossistêmicos para as comunidades locais. Estudos recentes mostraram que, uma vez que a cobertura vegetal das paisagens atinge determinados níveis, o número de espécies diminui mais rapidamente, o que é chamado limiar de perda de espécies. É urgente realizar esses estudos em paisagens modificadas pelos seres humanos, a fim de fornecer informações relevantes para a tomada de decisões e conservação em terras públicas e privadas. Além disso, é altamente relevante incluir ecossistemas não florestais no escopo, como o Hotspot do Cerrado, uma vez que esses tipos de ecossistemas têm sido frequentemente negligenciados para conservação no Brasil. Este estudo aborda o limiar de perda de espécies em paisagens modificadas pelos seres humanos em duas perspectivas diferentes. Primeiro, faz uma revisão de estudos empíricos em todo o mundo que usam a abordagem do limiar de perda de espécies com aves e encontra 31 artigos publicados de 1994 a 2018, com 24 estudos realizados em latitudes temperadas e sete em regiões tropicais, observando a tendência crescente de os estudos e sua potencial aplicação a estratégias de conservação e restauração de paisagens para conservação de aves. Em seguida, realiza uma pesquisa empírica no planalto da Serra da Bodoquena com 18 mamíferos de médio e grande porte e seis espécies de aves, utilizando dados coletados com armadilhas fotográficas. Concentra-se nos 9 mamíferos e 2 aves que responderam negativamente à perda de vegetação nativa a 500 m de buffer, resultando em um limiar médio de 56% da cobertura vegetal nativa. Ao interpolar esse valor em mapas de conversão de uso antropogênico modelados para 2030 e 2050 para projetar como a probabilidade de ocupação mudará ao longo do tempo, a perda anual prevista foi 22,6 km2 acima do valor limite médio, indicando que quase metade da área atual com valores acima do limite estará abaixo deles em 2050. Além disso, este estudo explora um

componente mais social investigando a percepção do habitante local sobre a riqueza e o declínio de mamíferos no nível da propriedade. Para entender as diferenças de percepção de acordo com a principal atividade econômica, realizei entrevistas com 37 habitantes locais dedicados à produção agrícola, pecuária e turismo em uma área do Cerrado no estado de Mato Grosso do Sul. Embora não encontre diferença significativa na riqueza total percebida de acordo com a atividade econômica, há uma diferença significativa na percepção de riqueza de espécies de áreas abertas e de florestas dentro de propriedades turísticas (t 0,0194, n = 6), o que sugere que essa categoria de os proprietários de terras, contando com atividades econômicas relacionadas à biodiversidade, têm um melhor conhecimento sobre sua biodiversidade e podem estar dispostos a proteger grandes áreas de florestas em suas propriedades.

General Introduction

The Anthropocene is considered a new era in which humans are the dominant driver to climate and environmental change (Rockström et al. 2009). Native vegetation conversion to anthropogenic land use is a global phenomenon, driven by factors such as commodity production, forestry, agriculture and urbanization (Curtis et al 2018), and the Cerrado (Brazilian savannas) is not exempted from such trend. Furthermore, The Cerrado Biome is a biodiversity Hotspot (Myers et al. 2000), the second most extensive biome in South America (Sano et al. 2010) and has experienced an original cover loss of 46% (Klink & Machado 2005, Strassburg et al. 2017, Resende et al. 2019). Supported in governmental expansion and colonization programs, most of the Brazilian Cerrado is now an axis for agricultural development and cattle ranching, with the inherent land use transformation and native vegetation covers decline (Inocêncio & Calaça, 2010). Still, most of the conservation progress in Brazil, has been focused on forested ecosystems (Overbeck et al. 2015).

Native vegetation loss comes along with defaunation and the change in ecosystems dynamics at both, population and community levels (Pardini et al. 2010, Laurance et al. 2011). Numerous studies have shown that once the vegetation cover of the landscapes reaches certain levels, the number of animal species declines faster in response to shrinking patch size and increasing patch isolation (so-called threshold of species loss) (e.g. Andren 1994, Pardini et al. 2010, Estavillo et al. 2013, Hanski 2015). Estimating thresholds for given communities or populations in vulnerable landscapes aids to review current management and conservation strategies and have the potential to guide policy making (Andren 1994, Radford et al. 2005, Banks-Leite et al. 2014, Ochoa-Quintero et al. 2015). For example, Banks Leite and co-workers defined strategic areas for restauration and conservation of the Mata Atlantica that served as a basis for governmental conservation plans (Melo et al. 2018).

To successfully implement land use management strategies, it is ideal to take into account the way how the people leaving in the areas of interest interact with nature. According to Rajaram and Das (2010), understanding the way the ecological and social components interact in a settlement is the key to effectively manage the coupled human–natural system. Besides, Luzar and co-workers (2011) have documented the advantages of working with people subsisting from the forest in long-term biodiversity studies, acknowledging that local hunters develop ecological expertise with great ability to detect and identify animals. In fact, information from data collected through local knowledge has been already used to detect changes on species composition in forest fragments in the Amazon basin (Michalski & Peres 2005,) and also to estimate thresholds of species loss of medium and big size birds and mammals in the Amazon basin (Ochoa-Quintero et al. 2015).

This study aims to contribute to the understanding of the threshold of species loss concept, by reviewing worldwide literature of the subject, by implementing one empirical assessment in the Brazilian Cerrado, specifically at the Bodoquena plateau by using key bird and mammal species, using a novel approach to estimate both community and species thresholds, and finally by assessing local perceptions towards local fauna in order to understand the how local knowledge differ among economic activities. This study is part of the Long Term Ecological Research Planalto da Bodoquena (PELD Planalto da Bodoquena). The PELD is led by the Universidad Federal of Mato Grosso do Sul (UFMS) and is aiming to explore variations on ecological interactions along a forest gradient using a landscape scale approach.

This thesis is organized as follows. The first chapter, “A review of threshold responses of birds to landscape changes across the world” was published in 2018 in the Journal of Bird Ornithology (Melo et al. 2018). Based on a worldwide literature review, it analyses the geographic distribution of the studies, and compiles the current methodological trends of threshold estimation in this taxonomical group. Besides, it reviews which management recommendations are made based on the threshold estimations.

The second chapter “Disentangling changes in occupancy of mammals and birds across a land use cover gradient in the Cerrado hotspot” is going to be submitted to Conservation Biology. It uses information on occupancy probability across a land use gradient to understand species-specific responses to habitat change in the Brazilian Cerrado savannah, defining a percentage of native vegetation cover threshold. Analyses are based on 18 medium to large mammal and six bird species using data collected with camera traps. Then, it interpolates such threshold value in modelled anthropogenic use conversion maps for 2030 and 2050 and projects how the species’ occupancy probability will change over time.

The third chapter “Local perceptions of biodiversity according to land use cover in the Brazilian Cerrado” depicts a more social component investigating if local perceptions about mammals living inside the properties differ among landowners grouped according to the main economic activity. It collects information from 37 farmers using semi structured interviews.

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Chapter 1. A review of threshold responses of birds to landscape changes across the world

Abstract

Identifying the threshold of habitat cover beyond which species of birds are locally lost is useful for understanding the biological consequences of landscape changes. However, there is little consensus regarding the impact of landscape changes on the likelihood of species extinctions. We conducted a literature search using Scopus and ISI Web of Knowledge databases to identify studies where bird species were used to estimate threshold responses to landscape changes. We obtained a list of 31 papers published from 1994 to 2018, with 24 studies conducted at temperate latitudes and seven in tropical regions. Nineteen studies were based on species-level assessments, and investigators used a variety of response variables such as probability of detection and occurrence to detect threshold responses. Eight studies were based on communities, and species richness and abundance were primarily used to detect threshold responses. Four studies included both communities and species-level assessments. Methods used to identify threshold responses varied among studies, but most relied on either regression models to visually identify values from graphs or piecewise regression to estimate a specific threshold value. Although the limited number of studies and their variety of approaches and methods prevented a formal meta-analysis, we found that mean threshold responses in studies that reported either a range or a single threshold value were 27.9% at temperate latitudes (range = 1.3–90%; N = 11) and 33.6% at tropical latitudes (range = 20–50%; N = 7). Considering only studies where single threshold values were reported, the mean habitat cover threshold was 11% for studies conducted at temperate latitudes (N = 3) and 29.5% for studies in the tropics (N = 4). These crude estimates suggest that tropical species might be more susceptible to habitat loss

than temperate species. Although application of the threshold concept is still controversial, the number of studies using this approach is increasing because the results of such studies may have direct application to conservation strategies and restoration of landscapes for bird conservation.

Resumo

A identificação do limiar da cobertura do habitat além do qual as espécies de aves são localmente perdidas é útil para entender as consequências biológicas das mudanças na paisagem. No entanto, há pouco consenso sobre o impacto das mudanças da paisagem na probabilidade de extinção de espécies. Realizamos uma literatura pesquisa usando os bancos de dados Scopus e ISI Web of Knowledge para identificar estudos em que espécies de aves foram usadas para estimar respostas limiares a mudanças na paisagem. Obtivemos uma lista de 31 artigos publicados de 1994 a 2018, com 24 estudos realizados em latitudes temperadas e sete em regiões tropicais. Dezenove estudos foram com base em avaliações em nível de espécie, e os pesquisadores usaram uma variedade de variáveis de resposta, como probabilidade de detecção e ocorrência para detectar respostas de limiar. Oito estudos foram baseados em comunidades e espécies riqueza e abundância foram usadas principalmente para detectar respostas limiares. Quatro estudos incluíram ambos comunidades e avaliações em nível de espécie. Os métodos utilizados para identificar respostas limiares variaram entre os estudos, mas a maioria se baseou nos modelos de regressão para identificar visualmente os valores dos gráficos ou na regressão por partes para estimar um valor limite específico. Embora o número limitado de estudos e sua variedade de abordagens e métodos impediram uma meta-análise formal, descobrimos que respostas limiares médias em estudos que relataram um intervalo ou um valor limite único foram 27,9% em latitudes temperadas (intervalo = 1,3–90%; N = 11) e 33,6% em latitudes tropicais (variação = 20-50%; N = 7). Considerando apenas estudos em que valores-limite únicos foram relatados, o limiar médio de cobertura do habitat foi de 11% para estudos realizados em latitudes temperadas (N = 3) e 29,5% para estudos nos trópicos (N = 4). Essas estimativas brutas sugerem que espécies tropicais podem ser mais suscetível à perda de habitat do que as espécies temperadas. Embora a aplicação do conceito de limiar ainda seja

controverso, o número de estudos usando essa abordagem está aumentando, porque os resultados desses estudos podem ter aplicação direta a estratégias de conservação e restauração de paisagens para conservação de aves

Introduction

Human activities are impacting climate and ecosystems across the globe (Bennett & Ford 1997, Brook et al. 2008, Rockström et al. 2009, Sebastian-Gonzalez et al. 2015). Although habitat loss is recognized as having one of the most severe impact of human activity on biodiversity (Brook et al. 2008), little consensus exists on the extent to which such loss impacts species extinctions. The results of some studies show that biodiversity initially declines proportionally to the amount of suitable habitat in a landscape, but, when availability of habitat declines below a certain level, non-linear changes emerge in response to shrinking patch size and increasing patch isolation, the so-called threshold responses (Andren 1994, Pardini et al. 2010, Hanski 2011). These responses have been related to synergetic effects (Brook et al. 2008, Pardini et al. 2010, Swift & Hannon 2010), including the 'Allee effect', i.e., a positive relationship between components of individual fitness and numbers or density of conspecifics (Stephens et al. 1999).

Determining how species respond to landscape change, such as threshold responses, can improve our understanding of the underlying processes that cause species extinctions and can also be important in identifying conservation and restoration strategies at the landscape scale (Suding & Hobbs 2009). Lande (1987) defined extinction threshold as the "minimum proportion of suitable habitat distribution through a region that is necessary for population persistence", based on the demographic equilibrium model. Since then, a number of investigators have conducted studies with the aim of better understanding the patterns of species loss across a habitat gradient (e.g., Lawton et al. 1994, Hanski et al. 1996). Notably, Andren (1994) examined the factors influencing the abundance and distribution of bird and mammal species in landscapes with different degrees of habitat fragmentation and found a threshold of 30% of remaining habitat below which species tended to be more sensitive to habitat alterations. Following Lande's (1987) definition of extinction threshold and the work of Andren (1994), an increasing number of analyses within the context of thresholds have been used to suggest conservation actions (Banks-Leite et al. 2014, Lima & Mariano-Neto 2014, Ochoa-Quintero et al. 2015, Rodrigues et al. 2016). In addition, threshold values across different taxonomic groups and different regions have been used to identify the presence of warning signals in community metrics across landscape changes before major extinction events are expected to occur (de Oliveira et al. 2018). According to the State of the World's Birds, one of eight species (i.e., 13%) is threatened with extinction (BirdLife International 2013). Most early studies of bird conservation were conducted at local scales, focusing on habitat fragments. However, beginning in the 1990s, the landscape-scale approach started to appear in the scope of the conservation ecology of birds (Bennett & Ford 1997, Radford et al. 2005). Since then, a growing number of studies have gone beyond the fragment scale, aiming to better understand threats by identifying landscape characteristics such as the amount and configuration of habitat (e.g., Farigh 2003, Shanahan & Possingham 2009, Moura et al. 2013). Notably, an increasing number of studies are reporting thresholds in loss of bird species relative to the amount of habitat loss (e.g., Drinnan 2005, Suorsa et al. 2005, Ochoa-Quintero et al. 2015, Boesing et al. 2018). For a synthetic understanding of the responses of bird species to landscape changes, examining both the trends and geographical bias of studies investigating thresholds of species loss relative to habitat loss is necessary. We reviewed studies where bird species were used to estimate such putative habitat-cover thresholds. Our objectives were to: (1) present the methods, scale of analysis, and geographical distribution of empirical studies examining threshold responses of birds to landscape structure, (2) summarize their main results, including examining possible effects of latitude on threshold values, and (3) discuss the gaps in knowledge and future perspectives in light of bird conservation at the landscape level. In a time characterized by loss of native vegetation in many parts of the world (Hansen et al. 2013, Watson et al. 2016), understanding how loss of habitat impacts birds is increasingly relevant to their conservation.

Literature search.

We searched the literature using ISI Web of Knowledge and Scopus databases in March 2018. Given that the term "threshold' is widely used beyond the field of ecology, we used exact phrases such as deforestation threshold, species loss threshold, fragmentation threshold, extinction threshold, and habitat threshold, and also used related alternatives by changing the order of words (e.g., threshold of deforestation and threshold of species loss). We also included the synonyms tipping point and break*point. We restricted the search to agricultural, biological, and environmental sciences. An initial search retrieved 463 and 2550 records (Scopus and ISI Web of Science, respectively). Of those records, we selected only those that included bird species and only those dealing with environmental thresholds as defined in our introduction (i.e., species putative responses to declines in availability of native habitat and describing abrupt non-linear changes). In addition, to identify publications potential y missing from the databases, we checked the literature cited sections of all the selected papers

Studies examining responses of birds to landscape structure.

We identified 31 papers specifically designed to determine the percentage of suitable habitat at which species or communities abruptly decline at the landscape level (Table 1). Most studies were conducted after 2000, with a steady increase in the number of studies since then (Figure 1). Of the 31 studies, 24 were conducted in countries located at temperate latitudes, including the United States (10), Canada (five), Australia (five), Sweden (one), Finland (one), Sweden and Finland (one), and Tunisia (one). We found only seven studies that had been conducted in countries in the tropics, including Brazil (five), Ecuador (one), and Panama (one) (Figure 2). We can distinguish two different overarching levels to search for bird responses to landscape structure: species level (one or more species analyzed individually) and community level. In total, 19 studies were based on species (six based on a single species and 13 on multiple species;

Table 1). In most of these studies, investigators used either the probability of occurrence or occupancy as a response variable (e.g., Jansson & Angelstam 1999, Radford & Bennett 2004, Suorsa et al. 2005, Betts et al. 2010), or something similar such as frequency of occurrence or probability of extinction, absence, colonization, or persistence. Communities were used as the level of analysis in eight studies, with species richness used as a response variable in six of these studies (e.g. Bennett & Ford 1997, Rompre et al. 2009, Table 1). Exceptions to this main trend include studies where community integrity and phylogenetic integrity were used as response variables (Banks-Leite et al. 2014), and where taxonomic, functional, and phylogenetic diversity were used (Boesing et al. 2018). In addition, we found four studies focusing on both communities and single species individually, from which Morante- Filho et al. (2015), Richmond et al. (2015), and Becker et al. (2015) also used guilds as the level of analysis (Table 1).

Regardless of whether studies focused on species or communities, some investigators defined the most influential predictor variable by using a model-averaging process with an array of putative predictor variables (Burnham & Anderson 2002), whereas others used a single predefined predictor variable. In both cases, tree cover or percent cover of native vegetation were the most commonly used predictor variables (Table 1). In addition, Drinnan (2005) used the amount of tree cover, not a percentage, to find a threshold of 4 ha for an urban environment in Australia, and Suorsa et al. (2005) used timber volume as the predictor variable.

In general terms, methods used to identify threshold values have improved over time. Initially, thresholds were inferred arbitrarily from visual inspection of data in a graphic representation of, most often, vegetation cover as predictor of the response variable (e.g., species richness). In total, investigators visually inferred the results from a graphic representation in 15 papers. For example, Radford and Bennett (2004) used logistic regression models and hierarchical partitioning to infer a 5–25% woodland-cover threshold for White-browed Treecreepers (Climacteris affinis) in Australia. In the remaining 16 papers, investigators used different methods to statistically estimate threshold values rather than inferring them visually, e.g., Receiver-Operating Characteristic analysis (ROC), Akaike information criterion, and piecewise regressions. Beginning in 2007, based on Muggeo (2003), researchers started using piecewise regressions to estimate threshold values (e.g., Betts et al. 2007, Richmond et al. 2015, Boesing et al. 2018). Currently, this appears to be the most widely used method, with nine of the 16 studies that estimated threshold values using piecewise regressions. The piecewise regression method tests different starting points to find the most parsimonious breaking point (Muggeo 2003). As a result, a specific threshold value is obtained that provides a more objective estimate than if it is visually inferred from a graphic.

Another method used to estimate thresholds is TITAN (Threshold Indicator Taxa ANalysis), with indicator species used to integrate occurrence, abundance, and directionality of taxa responses to landscape changes (Baker and King 2010). To date, investigators have used this approach to identify threshold responses of birds in three studies (Suarez- Rubio et al. 2013, Becker et al. 2015, Suarez- Rubio & Lookingbill 2016). Additional methods have been used to estimate thresholds of habitat loss. For example, Yin et al. (2017) developed a method that identifies rapid changes in species distributions instead of a breaking point, but, to our knowledge, this method has not been used with birds.

Of the 31 reviewed papers, 18 reported either a range or a specific threshold of the percentage of habitat cover, or mixed results with threshold and non-threshold responses. To illustrate the latter case, Morante-Filho et al. (2015) found no response in the richness and abundance of the whole community, but a threshold response on diversity of forest- specialist, frugivorous, and insectivorous birds, and a positive effect on generalist birds. Zuckerberg and Porter (2010) tested for the presence of a threshold for 25 species of forest birds using the New York State Breeding Bird Atlas. They found that thresholds were a common, but not a pervasive, characteristic that defined species responses to changes in forest cover, with 22 species showing extinction threshold responses ranging from 24.4 to 88.2% forest cover. The remaining 13 papers reported an array of results, including thresholds based on habitat characteristics other than habitat percentage (e.g., Jansson & Angelstam 1999, Guenette & Villard 2005), multiple thresholds depending on scale species, and response variable (e.g., Suarez- Rubio et al. 2013, Becker et al. 2015, Suarez- Rubio & Lookingbill 2016), and no threshold response by one or more of the evaluated species. For example, Lindenmayer et al. (2005) reported no evidence of any kind of threshold. These authors examined responses of different species of birds and reptiles to native vegetation cover in Australia and argued that, because the choice of both response and predictor variables and the inherent ecological variability of species assemblages may influence the results, the predictive power and practical usefulness of the threshold concept are questionable.

Threshold responses from the 18 studies that reported either a range or a single threshold value ranged from ~ 1 to ~ 90% forest cover (Table 1). The different levels of analysis (species and communities) and the variety of methods used to collect and analyze data (i.e., threshold values inferred from graphs, piecewise regression, ROC, and so on) prevent a proper statistical comparison among studies. However, as a crude approximation, we note a possible trend between studies conducted in temperate regions and those in the tropics. Studies conducted in temperate regions reported lower values (mean = 27.9%, range = 1.3–90%; N = 11) than those conducted in the tropics (mean = 33.6%, range = 20–50%; N = 7). Comparison of the means (with a t-test using SPSS, version 22) revealed no significant difference between temperate and tropical regions (P = 0.20). Considering only the seven studies where single threshold values were reported, the same trend prevailed, but with a significant difference between regions (P = 0.015). The mean habitat cover threshold was 11% for studies conducted at temperate latitudes (N = 3) and 29.5% for studies in the tropics (N = 4; Figure 3). These crude estimates suggest that tropical species might be more susceptible to habitat loss than temperate species, and there may be more variation in responses of the species and communities in temperate regions as indicated by the wider range of threshold values than in tropical regions. Values from the tropics are more consistent with theoretical threshold studies that suggest a threshold of 30% of remaining habitat, below which species tend to be more sensitive to habitat alterations (Andren 1994).

Variation in threshold values, especially among studies conducted in temperate regions, may be due to differences in the conservation status of study areas, historical features of land-use changes, or the fact that more studies have been conducted in temperate regions. Variation may also be related to differences in the level of analysis (community or species) or the applied statistical analyses. In the literature, such variability has been acknowledged (Lindenmayer and Luck 2005, Ficetola & Denoël 2009, Estavillo et al. 2013, Boesing et al. 2018, Roque et al. 2018) as being the result of (1) statistical artefacts, (2) interrelated factors of nature, such as different species responses to landscape change, differences in habitat quality, and timing, intensity and extent of the change or, alternatively, (3) variability in the landscape matrix surrounding study areas. Thus, the uncontrolled variables inherent to the nature of this review may affect the results of our comparison between temperate and tropical regions.

Controversy still exists about the effect of the species selected for study and the community and population metrics used in studies of threshold responses. For example, Ochoa- Quintero et al. (2015) reported higher threshold values when including only threatened species, and Rodrigues et al. (2016) found that the threshold responses could drop from near 90% when using abundance to lower than 20% when using richness of aquatic macroinvertebrate communities. These facts highlight that our temperate versus tropical comparison was made only as a preliminary assessment. If the observed difference in threshold values between tropical and temperate regions is genuine, and not related to use of different methods, we suggest that it may be related to the historic climatic conditions that make species that evolved in more stable conditions (i.e., tropics) less resilient to change (Dalsgaard et al. 2011, Sandel et al. 2011). This would explain the observed higher and a narrower range of threshold values in the tropics, and the lower and wider range of thresholds in temperate regions with both resilient and non-resilient species. We hope this review stimulates tests of this proposed hypothesis, i.e., there is a synergetic effect between historical climate stability and habitat destruction on biodiversity.

Identifying threshold values can generate clear recommendations for habitat management and conservation. Although authors of most of the papers we reviewed made general recommendations, some authors made concrete recommendations concerning desirable habitat characteristics and the minimum amount of habitat to be preserved (e.g., Drinnan 2005, Radford et al. 2005, Martensen et al. 2012, Ochoa-Quintero et al. 2015). Notably, the findings and recommendations of Banks-Leite et al. (2014) have now been implemented as the official target for restoration in an environmental resolution legislated by the São Paulo State and a federal decree in Brazil. Banks-Leite et al. (2014) provided scientific evidence that led the Environment Secretariat of the State of São Paulo to prioritize reforestation projects in municipalities with less than 30% forest cover, and to define higher offsetting standards for entrepreneurs in municipalities with less than 30% forest cover. These new rules are part of Resolution SMA 7/2017. Also, these results were incorporated in the official map used by the Brazilian Environment Ministry to help support the Native Vegetation Protection Law (N_ 12.651/2012) and the National Policy for Native Vegetation Recovery (resolution n_8.972/2017), thus being instrumental in the development of an optimal restoration scenario (C. Banks-Leite, pers. comm.).

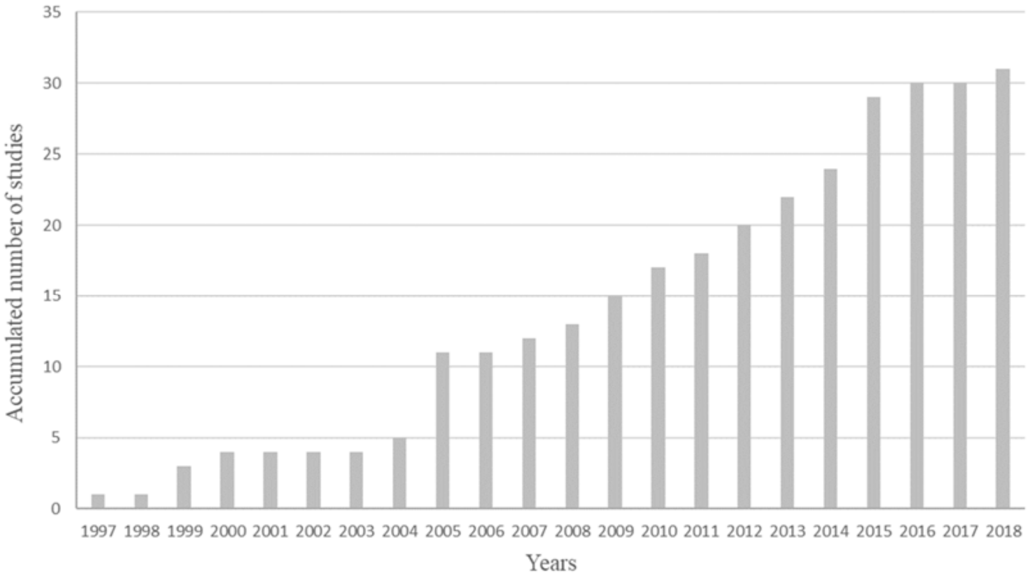


Figure 1. Accumulated number of studies per year focusing on birds to examine thresholds of species loss as a function of habitat loss.

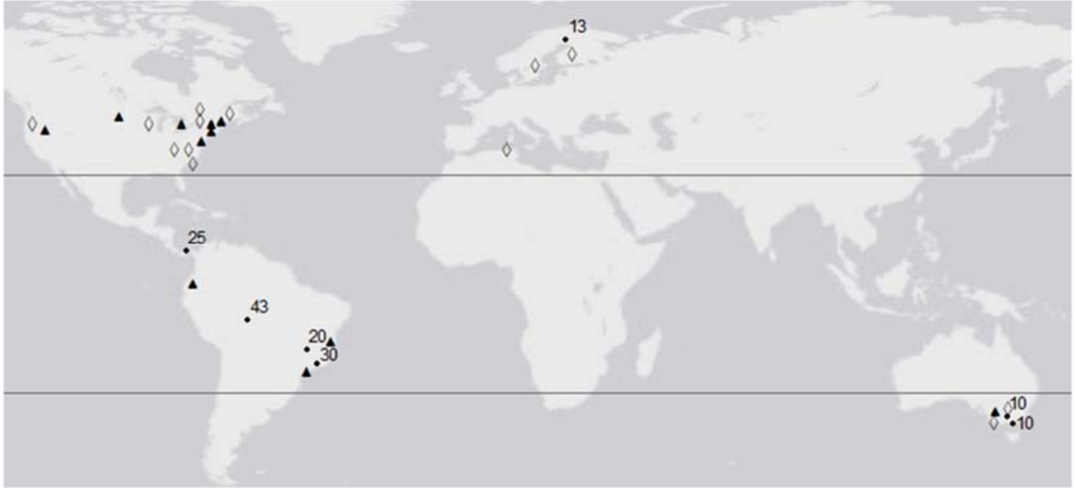


Figure 2. Location of 31 studies on thresholds of bird species loss as a function of habitat loss. The black circles n = 7 studies that reported a single threshold value of habitat cover percentage (value shown in the map); diamonds are the n = 11 studies that reported a range of habitat cover percentage; triangles are the remaining n = 13 studies that reported threshold in a different way (i.e. no-threshold response, tree density, canopy closure, etc.). When the studies did not report coordinates, the points were located as an approximation, based on the description of the study area. Grey lines indicate the tropics of Capricorn and Cancer.

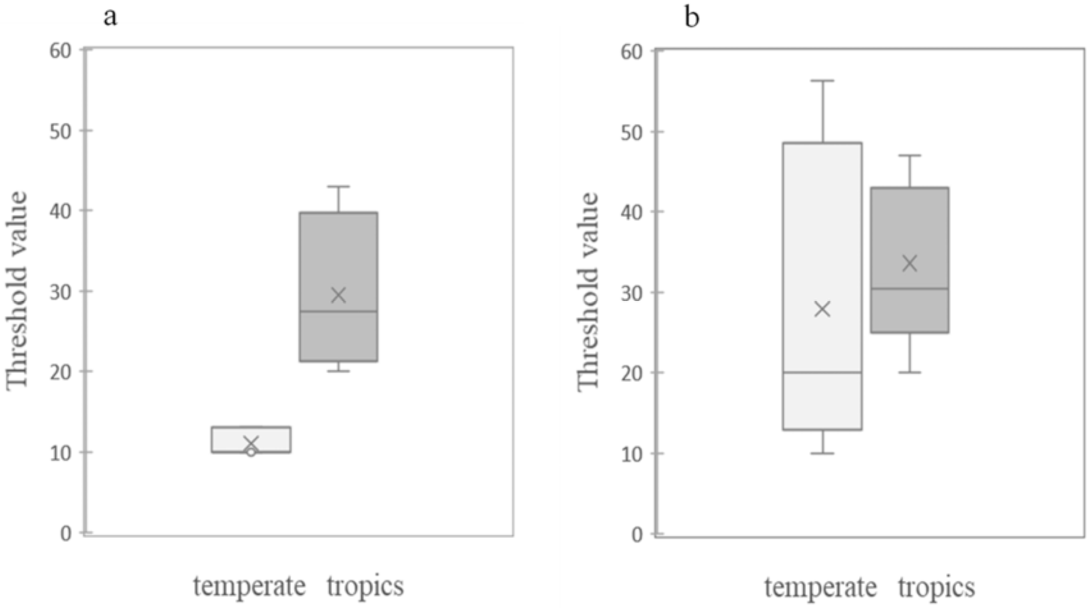
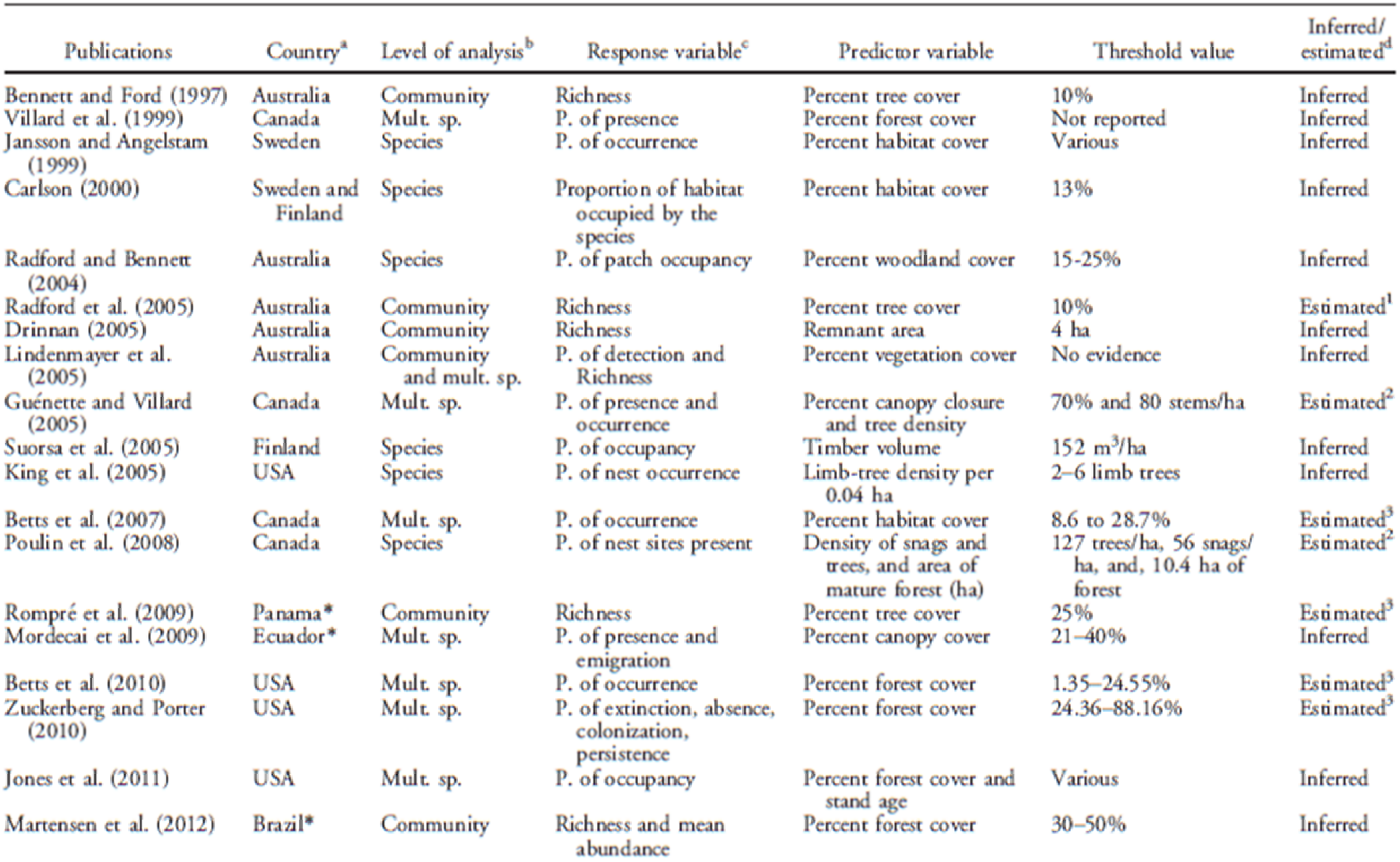


Figure 3. Comparison of the means from seven studies that reported single threshold values (a), and the 18 studies that reported either a range or a singular threshold values (b). The lines outside each box represent minimum and maximum values, lines within the boxes are the medians, and ‘x’ represents the mean of the threshold value for each region.

Table 1. List of studies examining bird threshold responses to landscape changes



Publications	Country[a]	Level of analysis[b]	Response variable[c]	Predictor variable	Threshold value	Inferred/ estimated[d]
Bennett and Ford (1997)	Australia	Community	Richness	Percent tree cover	10%	Inferred
Villard et al. (1999)	Canada	Mult. sp.	P. of presence	Percent forest cover	Not reported	Inferred
Jansson and Angelstam (1999)	Sweden	Species	P. of occurrence	Percent habitat cover	Various	Inferred
Carlson (2000)	Sweden and Finland	Species	Proportion of habitat occupied by the species	Percent habitat cover	13%	Inferred
Radford and Bennett (2004)	Australia	Species	P. of patch occupancy	Percent woodland cover	15-25%	Inferred
Radford et al. (2005)	Australia	Community	Richness	Percent tree cover	10%	Estimated[1]
Drinnan (2005)	Australia	Community	Richness	Remnant area	4 ha	Inferred
Lindenmayer et al. (2005)	Australia	Community and mult. sp.	P. of detection and Richness	Percent vegetation cover	No evidence	Inferred
Guénette and Villard (2005)	Canada	Mult. sp.	P. of presence and occurrence	Percent canopy closure and tree density	70% and 80 stems/ha	Estimated[2]
Suorsa et al. (2005)	Finland	Species	P. of occupancy	Timber volume	152 m^3/ha	Inferred
King et al. (2005)	USA	Species	P. of nest occurrence	Limb-tree density per 0.04 ha	2–6 limb trees	Inferred
Betts et al. (2007)	Canada	Mult. sp.	P. of occurrence	Percent habitat cover	8.6 to 28.7%	Estimated[3]
Poulin et al. (2008)	Canada	Species	P. of nest sites present	Density of snags and trees, and area of mature forest (ha)	127 trees/ha, 56 snags/ha, and, 10.4 ha of forest	Estimated[2]
Rompré et al. (2009)	Panama*	Community	Richness	Percent tree cover	25%	Estimated[3]
Mordecai et al. (2009)	Ecuador*	Mult. sp.	P. of presence and emigration	Percent canopy cover	21–40%	Inferred
Betts et al. (2010)	USA	Mult. sp.	P. of occurrence	Percent forest cover	1.35–24.55%	Estimated[3]
Zuckerberg and Porter (2010)	USA	Mult. sp.	P. of extinction, absence, colonization, persistence	Percent forest cover	24.36–88.16%	Estimated[3]
Jones et al. (2011)	USA	Mult. sp.	P. of occupancy	Percent forest cover and stand age	Various	Inferred
Martensen et al. (2012)	Brazil*	Community	Richness and mean abundance	Percent forest cover	30–50%	Inferred

Table 1. Continued

Publications	Country[a]	Level of analysis[b]	Response variable[c]	Predictor variable	Threshold value	Inferred/ estimated[d]
Cunningham and Johnson (2012)	USA	Mult. sp.	Frequency of occurrence	Percent tree cover	20–60%	Inferred
van der Hoek et al. (2013)	USA	Mult. sp.	P. of extinction and persistence	Percent forest cover	50.8–91.0%	Inferred
Suarez-Rubio et al. (2013)	USA	Mult. sp.	Abundance	Forest cover and percent forest fragments	Various	Estimated[4]
Banks-Leite et al. (2014)	Brazil*	Community	Community and phylogenetic integrity	Percent forest cover	30% overall	Estimated[3]
Touihri et al. (2014)	Tunisia	Mult. sp.	P. of detection	Tree and snag density	650 stems, 207 snags/ha	Inferred
Ochoa-Quintero et al. (2015)	Brazil*	Community	Richness	Percent tree cover	43%	Estimated[3]
Becker et al. (2015)	USA	Community, mult. sp., and guilds	Abundance and occurrence	Percent land cover	Various	Estimated[4]
Morante-Filho et al. (2015)	Brazil*	Community, mult. sp. and guilds	Richness and abundance	Percent forest cover	44–50%	Estimated[3]
Richmond et al. (2015)	Canada	Community, mult. sp. and guilds	Richness and occurrence	Percent forest cover	4–45%	Estimated[3]
Van der Hoek et al. (2015)	USA	Mult. sp.	P. of extinction	Percent forest cover	7–90%	Estimated[3]
Suarez-Rubio and Lookingbill (2016)	USA	Mult. sp.	Abundance	Percent compactness	Various	Estimated[4]
Boesing et al. (2018)	Brazil*	Community	Taxonomic, functional, and phylogenetic diversity	Percent forest cover	20% overall	Estimated[3]

[a] Countries with * correspond to studies conducted in tropical regions.
[b] Mult. sp. stands for multiple species analyzed individually.
[c] P. stands for probability.
[d] The procedure used to obtain the threshold value. Inferred is when the value is visually identified from the graphic, whereas estimated can be (1) using AIC to find best fit model, (2) using ROC, (3) using piecewise regression, or (4) using TITAN.

Gaps of knowledge and future perspectives.

Given the relevance for understanding species and the possibility of aiding conservation actions, habitat thresholds of bird species loss are clearly a subject attracting attention from researchers, conservationists, and decision-makers. Nonetheless, we found relatively few empirical studies on this topic. The need for a habitat gradient to identify thresholds may be a practical limitation, as might the time-consuming fieldwork requirements depending on the selected variable. The geographical distribution of threshold studies, with most conducted in North America (15), South America (six), and Australia (five), but with only a few conducted in Europe (three), Africa (one), and Central America (one), and the lack of studies in Asia, provide evidence of a strong geographical bias (Figure 2). Consequently, more threshold studies are needed in regions with few or no studies to date. However, we acknowledge that, because our search retrieved only papers in English, we may have missed records of studies conducted, for example, in Asia or Latin America. Amano et al. (2016) acknowledged that, although English is recognized as a global scientific language, ignoring non-English publications may cause biases in our understanding of study systems. Regarding the low number of studies in the tropics compared to temperate regions, we believe that it is crucial to conduct more (comparable) studies in the tropics to test if – and why – threshold values may differ between these regions.

Even though there is an increasing array of literature involving thresholds with different taxonomic groups (Figure 1), some controversies exist regarding their applicability, e.g., not all species in a community have the same responses (Lindenmayer et al. 2005, Estavillo et al. 2013), and the results of studies may not be transferable across regions, e.g., species may have different threshold responses in different locations (van der Hoek et al. 2013). Also, although the percentage of suitable habitat may be the most important and widely used variable, threshold responses may change with different landscape configurations such as the degree of fragmentation. Notably, some of these challenges have already been addressed (Jansson & Angelstam 1999, Villard et al. 1999, Suarez-Rubio et al. 2013, Van der Hoek et al. 2015). We argue that the applicability of the threshold concept, even though the number of studies is increasing, may benefit from more refined analyses that include the matrix (Boesing et al. 2018), and consider historical land use changes and other variables such as historical climate stability that help identify the mechanisms behind the responses.

Taken together, although the threshold concept is clearly relevant for understanding how current habitat destruction will impact patterns of biodiversity (Ratajczak et al. 2018), there are many challenges to overcome and the concept needs to be used with caution to make sound conservation or management recommendations (Lindenmayer & Luck 2005). However, our review suggests that tropical regions may have higher threshold responses than temperate regions, where the ranges of values is highly variable, indicating that tropical species may be more impacted by habitat alteration. Despite current limitations, identifying thresholds of species loss should improve our understanding of the consequences of landscape transformations on biodiversity and bird species. Further research is vital because agriculture expansion is one the most important threats to birds and biodiversity in general. We urge researchers to further develop the threshold concept and/or conduct large-scale studies using similar methods, allowing unbiased comparisons of species responses to landscape change. Such studies can be translated into conservation practices, given a clear understanding of the political and environmental contexts where they were conducted.

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Chapter 2. Disentangling changes in occupancy of mammals and birds across a land use cover gradient in the Cerrado hotspot

Abstract

Increasing food production for a growing human population while protecting biodiversity is one of the greatest challenges of the last years. Main focus of research on this subject have been devoted to identify species responses to land use change, including rely upon thresholds of species loss used to define to what extent land use changes affect biodiversity. Here we used information on occupancy probability across a land use gradient to understand species-specific responses to habitat change in the Brazilian Cerrado savannah. We analyzed 18 medium to large mammal and six bird species using data collected with camera traps. We found that 11 of these species (9 mammals; 2 birds) were negatively affected by land use change but the occupation probability had different threshold values, ranging from 15.15 to 61.35%, and averaging 56% of native vegetation cover. Importantly, we noticed that the trajectory of occupancy for such species is steeper above than below the threshold, which is the opposite to findings of threshold responses using richness instead of occupancy probability as the dependent variable. We modeled the anthropogenic use conversion for 2030 and 2050 and interpolated the average threshold value to project how the occupancy probability will change over time. According to our findings, the conversion rate per year would be 0.13%, representing an annual loss of 22.6 km^2 above the average threshold value. Finally, and highly relevant, almost half of current area with values above the threshold will be below them by 2050. Our findings not only change the perspective of species responses to land use change but also have implications on the applicability of the threshold concept for land use planning.

Resumo

Aumentar a produção de alimentos para uma população humana crescente e proteger a biodiversidade é um dos maiores desafios dos últimos anos. O foco principal da pesquisa sobre esse assunto foi dedicado à identificação das respostas das espécies às mudanças no uso da terra, incluindo os limiares de perda de espécies usados para definir em que medida as mudanças no uso da terra afetam a biodiversidade. Aqui, usamos informações sobre a probabilidade de ocupação em um gradiente de uso da terra para entender as respostas de espécies específicas à mudança de habitat na savana brasileira do Cerrado. Analisamos 18 mamíferos de médio a grande porte e seis espécies de aves usando dados coletados com armadilhas fotográficas. Constatamos que 11 dessas espécies (9 mamíferos; 2 aves) foram afetadas negativamente pela mudança no uso da terra, mas a probabilidade de ocupação apresentou diferentes limiares, variando de 15,15 a 61,35% e com média de 56% da cobertura vegetal nativa. É importante notar que a trajetória de ocupação de tais espécies é mais acentuada do que abaixo do limiar, o que é o oposto dos achados das respostas dos limiares usando riqueza em vez de probabilidade de ocupação como variável dependente. Modelamos a conversão de uso antropogênico para 2030 e 2050 e interpolamos o valor limite médio para projetar como a probabilidade de ocupação mudará ao longo do tempo. De acordo com nossas descobertas, a taxa de conversão por ano seria de 0,13%, representando uma perda anual de 22,6 km2 acima do valor limite médio. Finalmente e altamente relevante, quase metade da área atual com valores acima do limiar estará abaixo deles até 2050. Nossas descobertas não apenas mudam a perspectiva das respostas das espécies à mudança no uso da terra, mas também têm implicações na aplicabilidade do conceito de limiar para uso da terra planejamento.

Introduction

Increasing food production for supply a growing human population is transforming landscapes across the world (Green et al. 2005, Popp et al. 2017, Lanz et al. 2018). Human-induced land use change is the most important threat affecting the survival of several species as well as reducing the provision of ecosystem services for local communities (Brondizio et al. 2019). This is especially important in tropical regions where a net forest loss of 6.54×10^6 ha yr^{-1} was recorded between 2000 and 2010 (Kim et al. 2015). The Cerrado biodiversity hotspot is one of the most threatened in Brazil, with estimates of 40% of its native vegetation lost (Ferreira et al. 2012, Strassburg et al. 2017).

The biological consequences of native vegetation loss are undeniable, and different analyses based on empirical or modelled data have been used to explain such consequences (see Didham et al. 2012). Evidence has shown that as forest cover decreases there is a reduction in beta diversity (Laurance et al. 2002, Michalski & Peres 2005, Laurance et al. 2011). Other studies have reported that the dynamics of species' local extinctions are affected by dispersion and re-colonization from surrounding fragments (Pardini et al. 2010, Banks-Leite et al. 2014). At present, there is evidence about the correlated edge effects studies showing the detrimental consequences of fragmentation, historical disturbances, and habitat reduction on biodiversity (Orme et al. 2019, Pütkker et al. 2020). Moreover, a recent study showed that areas where aspects such as climate stability across the tropics make the species from this region highly susceptible to further native vegetation fragmentation (Betts et al. 2019).

In addition, there have been studies that focus on understanding the trajectory of species responses to native vegetation reduction particularly when such response is non-linear, the so-called threshold response (Radford et al. 2005, Banks-Leite et al. 2014, Ochoa-Quintero et al. 2015). The threshold of species loss is defined as the minimum proportion of suitable habitat necessary for population persistence (Lande 1987). In Brazil, thresholds have been estimated in the Atlantic Forest (Estavillo et al. 2013, Rigueira et al. 2013, Banks-Leite et al. 2014, Lima and Mariano-Neto 2014, Morante-Filho et al. 2015, Boesing et al. 2018, Santos et al. 2019), in the Amazon (Ochoa-Quintero et al. 2015, Brito et al. 2020), and in the Cerrado biomes (Muylaert et al. 2016, Rodrigues et al. 2016). Such studies were focused on volant and non-volant mammals, birds, Odonata, macroinvertebrates and plant species; and the estimated thresholds from different taxonomic groups ranging from 24% to 60% of habitat amount (see de Oliveira et al, 2018, Melo et al. 2018). Both theoretical and empirical studies have shown that species, populations and communities tend to decline proportionally to the reduction of the suitable habitat amount in a landscape at the beginning of the native vegetation loss, but, at certain point, they experience an abrupt decline attributed to shrinking patch size and increasing patch isolation (Andren 1994, Pardini et al. 2010, Hanski 2015). This has been called the threshold response, and has implications for conservation and management. Threshold response estimates have the potential to identify where particular actions of native vegetation management are most needed. For example, some authors consider that restoration efforts should be focused on the areas close and below the threshold level avoiding the rapid decline of deforestation when deforestation exceeds this value (Pardini et al., 2010, Ochoa-Quintero et al. 2015), while other authors have used these threshold responses to assess where to cost-efficiently invest on payments for environmental services across the Atlantic Forest biome (Banks-Leite et al. 2014). Hence, intermediate habitat amount would provide intermediate resilience landscapes, which are priority areas for restoration (Tambosi et al. 2014).

In this study, we used data on occupancy probability, based on camera trap sampling, across a native vegetation gradient to evaluate species and community responses to land use change such as decreasing native vegetation and expansion of cattle ranching and agriculture. We focused on forest dependent bird and mammal species in the Bodoquena Plateau, a key area of the central Brazilian Cerrado (savannah biome) that includes a gradient of native vegetation cover ranging from protected areas to agricultural lands. Our hypothesis is that populations and communities will have nonlinear responses to native vegetation loss and that responses might change according to each assessed species depending on their native vegetation requirement. We expect that by using occupancy probability instead of richness, we will obtain a more refined measure of the species' response to land use change as the data is analyzed by species. In addition, we used a land use cover model (Rosa et al. 2013) to assess expected conversion by 2030 and 2050, to project how the occupancy probability would change over time. This information will be useful to identify best avenues to balance conservation and production in one of the most threatened biomes of South America, given the current situation and political changes in Brazil.

Methods

Study area

The Cerrado's natural landscapes are a mosaic of grasslands, savannahs, woodlands and forests (Coutinho 1982), resulting in more open natural environments when compared to forested areas. Consequently, the species' dependence on forest areas may differ from those presented across forest dominated biomes (Enquist 2019). Besides and very importantly, the Cerrado has been rapidly transformed into agricultural lands (Klink & Machado 2005, Resende et al. 2019) and is a global biodiversity hotspot (Strassburg et al. 2017), which makes urgent looking for robust scientific knowledge for conservation in order to identify acceptable levels of transformation avoiding the biodiversity and environmental services collapse.

The study area is located in the east of the Cerrado, in the municipalities of Jardim, Bonito and Bodoquena, in Mato Grosso do Sul state (21°13'47"S, 56°30'45"W), in an area known as the

Serra da Bodoquena plateau (Figure 1). This area consists of the plateau, and together with the Pantanal wetland (lowlands) form the Upper Paraguay River Basin (UPRB). Our study took place across the Bodoquena plateau, an area of 18,000 km^2 devoted to the Long Term Ecological Research Project (Bodoquena LTER hereafter). The main native vegetation types in the study area are forest formations, wooded, steep (Chaco), and wet savanna (wet Chaco), forest savanna, and riparian vegetation (SOS Pantanal et al. 2017). In the region cattle ranching and agricultural plantations of soy beans and corns plus ecotourism are the main economic activities.

Camera trapping

Our sampling took place in 15 hexagons of 5000 ha (landscapes hereafter) that are part of a long term ecological research project in the region (Figure 1 b). They were selected to reflect a native vegetation cover gradient from 9.9 to 89.7% and a cattle ranching cover from 8.9 to 84.8%.

Our camera trapping sampling protocol adopted a systematic rather than random sampling design to represent the heterogeneity in the area across the native vegetation gradient. For the installation scheme, we draw a grid of 15 points in each landscape, 1800 meters apart, adjusting it when necessary to keep the number of cameras proportional to the native vegetation cover. It means a landscape with higher proportion of native vegetation cover held higher number of camera traps to better fit the dominant land use cover at landscape scale. In the field we adapted the scheme based on landowners permits, logistics, and terrain, resulting in a minimum distance of 390 m between cameras, maintaining the independence among sampling units. We deployed 189 Reconyx (HC500 HYPERFIRE) and Bushnell (Trophy Cam HD) camera traps from May 2016 to December 2017. There were 10 to 15 cameras in each landscape at a time, for a period of four to five weeks, adding a total sampling effort of 4928 camera trap sampling days. We processed all the photos using Wild.ID software (Fegraus et al. 2011), and identified all the terrestrial vertebrates based on regional guides for mammals (Hannibal et al. 2015) and birds (Gwynne et al. 2010).

Landscape variables

We used the land cover map of Landsat images with a 30 m resolution interpreted by the SOS Pantanal initiative from the year 2017 (SOS Pantanal et al. 2017). This initiative is monitoring land cover change in the Upper Paraguay River Basin since 1998. We then clipped the original extent of the map to fit the study area using ArcMap 10.3 and used two main classes of the SOS Pantanal categories: ‘natural’ (such as forests, forested savannas, arboreal savannas), and ‘anthropic’ (as cattle lands, urban, mining, agriculture). We grouped all types of natural areas into one class named ‘native vegetation’, and created a second class ‘cattle lands’, to represent the anthropic class in our study area. We calculated the percentage of natural vegetation cover and cattle land at two spatial scales: 500- and 1000-meters radius from each camera to use them as co-variables. We selected these two spatial scales based on studies that have acknowledge the importance of using multiple spatial scales when evaluating the effects of landscape on species occurrence (Gestich et al., 2019, Santos et al. 2019). Despite being correlated amongst each other (Appendix 1) we kept the four variables (natural vegetation and cattle lands, within 500 and 1000 m radius from camera locations) to determine the co-variable to which more species were sensitive to.

Data analyses

In order to determine which land use (natural or cattle, at both 500 and 1000 m radius) better predicted the occupancy of most species we model the occupancy (MacKenzi 2002) of the bird and mammal species registered by the camera traps. For that, we used the functions *occu* and *modSel* in the package *Unmarked* (Fiske & Chandler 2011) in the statistical language R, version 3.5.1 (R Core Team, 2018). We defined an event as one picture in 24 hours for each species, and absence when no pictures where registered in 24 hours. We built 13 models as follows: one null model with no co-variables; four models using the co-variables: native vegetation percentage cover at 500 m, and at 1000 m buffers, and cattle lands percentage cover at 500 m, and at 1000 m buffers; four models considering the covariates type of substrate where the camera was placed (i.e. open or forested) as detectability covariate; and, finally, four models including the percentage of vegetation at 500 m buffers, against each co-variable. We used the model selection procedure with the Akaike's Information Criterion (AIC) to compare the models, based on the assumption that models that had lower AIC value and delta AIC < 2 corresponded to those models that better explained the occupancy of a given species (Burnham & Anderson 2002). Since percentage of native vegetation cover at a 500 m buffer predicted the occupancy for the majority of species, we focused on such species and divided them in two groups: those who negatively responded to vegetation cover loss and those who responded positively. Using the function *predict* and the package *Segmented,* we calculated the inflection point of each species curve (Muggeo 2008), to determine where the abrupt change in occupancy and native vegetation cover took place (i.e. threshold). After that, we averaged the inflection points of all the negatively affected species to determine the community threshold. Additionally, we estimated threshold of species richness for forest dependent species using Piece-wise analysis (Muggeo 2003) for comparison (supplementary material).

Threshold value interpolation

In order to understand how current and future land use changes relate to changes in the occupancy, we created future land use maps and modeled the occupancy with the threshold values previously obtained. We produced the maps for three different years: 2017, since this was the year when the fieldwork ended, 2030 because this is the year for the UN Sustainable development goals (United Nations 2013), and 2050 for a longer time frame. For the 2017 case, we interpolated the community threshold result in a native vegetation cover raster of the whole study area (Bodoquena LTER area, figure 1. a). For that, we divided the raster in 1 km² cells in order to have a similar scale of analysis than the community threshold response (the total area of the 500 m buffer is 0.78 km² compared to the area 1 km² per cell).

For the years 2030 and 2050, we used a dynamic and spatially-explicit model that predicts the magnitude and location of future native vegetation loss, developed by Rosa et al. (2013, 2015) and used in a study conducted in the Upper Paraguay River Basin (Guerra et al. 2020). In the latter study, which partially coincides with our study area, the authors used a set of drivers of vegetation loss from the literature and modeled the probability of anthropogenic use conversion in each native vegetation cell, to obtain the probability of vegetation loss by 2030 and 2050 (see all model steps in Rosa et al. 2013, 2015, Guerra et al. 2020). In our study, we re-sampled the projected native vegetation maps for 2030 and 2050 produced by Guerra et al. (2020), specifically for our study area, with a resolution of 100 m x 100 m cells, assuming a business as usual scenario that continues with the current legislation established in the Native Vegetation Protection Law (NVPL, known as the "New Forest Code", Brazilian Federal Law 12,651 of 2012). Finally, we interpolated the community threshold results in 2030 and 2050 projection maps to predict the vegetation cover above and below the community threshold (i.e. the averaged community breaking points calculated from the occupancy probability models).

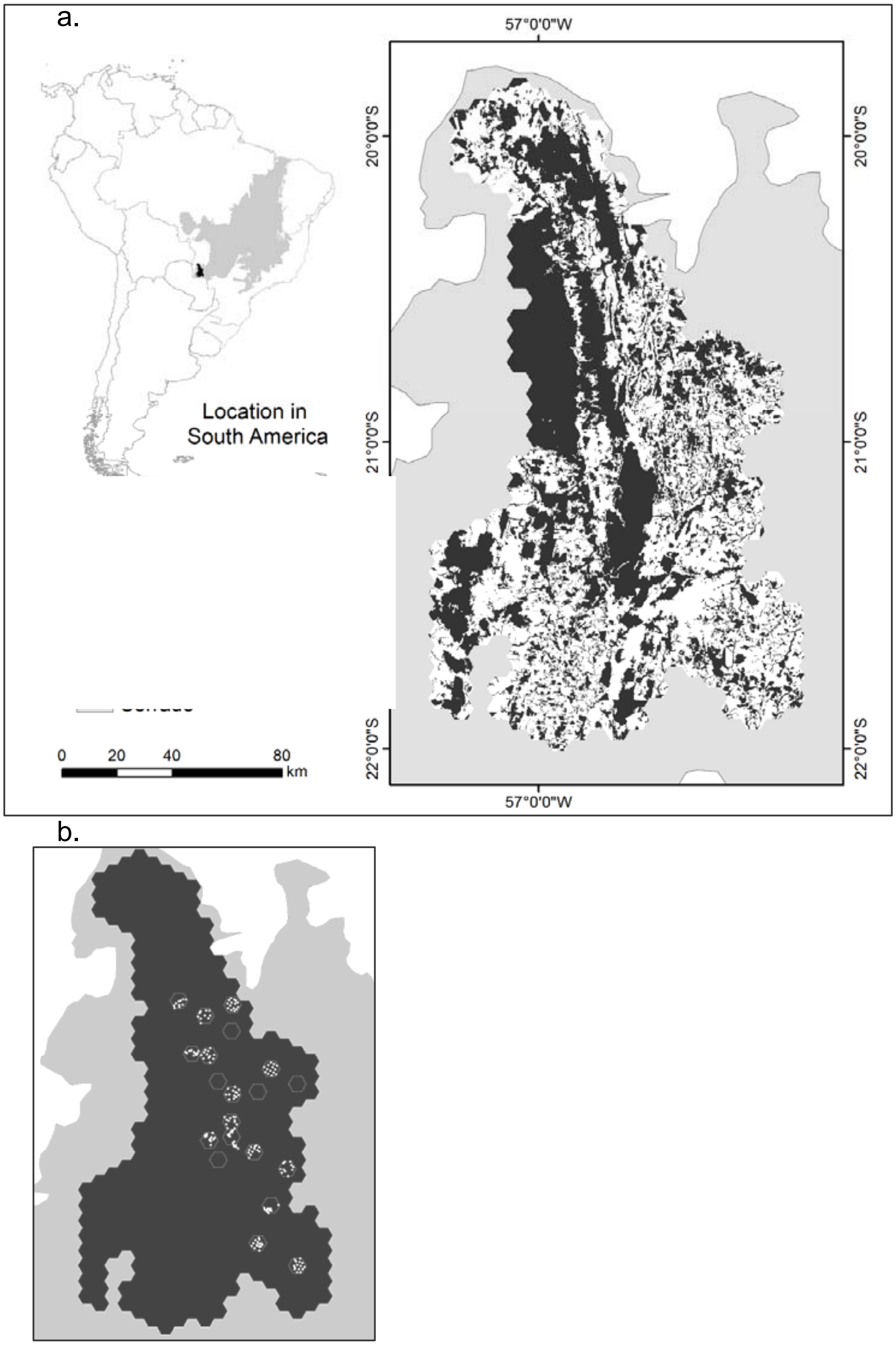

Figure 1. Study area. a. LTER location (zoomed in polygon). The Cerrado Biome is shown in gray, black areas represent native vegetation covers in 2017, and white areas are the anthropic land use cover; b. hexagons where camera trapping took place (white dots).

Results

From May 2016 to December 2017 the camera traps registered 61 species (31 birds and 30 mammals), for a total of 239,619 pictures of animals and 625 hours/camera (Appendix 2). Thirty species had 12 or more events that allowed us to calibrate the occupancy models. However, we excluded six species: the Puma (*Puma concolor*), Azara's capuchin (*Sapajus cay*), since their error covered almost the entire range of their occupancy probability; agile opossums (*Gracilinanus agilis*) and a rodent species (*Thrichomys pachyurus*) owing identification problems; the capybaras (*Hydrochoerus hydrochaeris)*, given that this species is associated with large water bodies; and Guira cuckoo (*Guira guira*) because it was only registered in the first 10% of the vegetation gradient. As a result, we focused our analyses in 24 species, from which 18 were mammals and six were birds.

Native vegetation cover at 500 m buffer was the covariable to which more species responded, 14 species (i.e. species that had more AIC values lower than 2) (Table 1). However, some other species also responded to other covariables: 11 species responded to native vegetation cover at 1000 m buffer, and 12 and 11 species responded to cattle land cover at 500 and 1000 meters buffers respectively (Appendix 3). In addition, for two species the null model was the one better explaining the occupancy variability. Based on the assumption that the more species responding to one covariable the better representation of the whole community, we selected the native vegetation cover at 500 m buffer for the forthcoming analyses.

From the 14 species that responded to native vegetation cover at 500 m buffer, three increased their occupancy probability as the native vegetation cover decreased: *Rhea americana*, *Cariama cristata,* and *Cerdocyon thous.* The occupancy probability of the 11 remaining species increased while the vegetation cover increased, that is, they were negatively affected by native vegetation loss (Figure 2). The threshold values of such negatively affected species ranged from

15.15% (*Pecari tajacu*) to 61.35% (*Cuniculus paca*) (Table 2). The averaged threshold values of the species that were negatively affected by native vegetation loss was 45.97% (S.E. 12.68), the slope of the occupancy probably line along the native vegetation gradient of these 11 species increased as the vegetation increased, and such slope was higher above the threshold value than below them (Figure 3). Similarly, the threshold of species loss using forest dependent species richness as the response value, estimated with the same data set and at the same scale was 52.57 (S.E. 11.02), using Muggeo (2008) piece-wise analysis (Appendix 4), and opposite to the findings of the averaged threshold value by species, the line trajectory was stepper below the threshold value than above it.

Threshold value interpolation

The results of the land cover change modeling that were used to interpolate the threshold value are presented in supplementary material (Appendix 5). After interpolating the 46% (rounded from 45,97 %) community threshold in the maps from 2017, 2030 and 2050, we found that the proportion of 1 km^2 cells, below or equal to 46% native vegetation cover was 56% of the total Bodoquena LTER area for 2017, 57.3% for 2030, and 60.3% for 2050 (Figure 4). This implies a 0.13% conversion rate per year, which represents the 22.6 km^2 of the Bodoquena LTER area annually and would mean that 746 km^2 that were above the threshold value in 2017, would be somewhere below 46% by 2050.

Table 1. Occupation analyses results for the 14 species whose occupancy probability was explained by the native vegetation cover percentage at 500 m buffer. Covariates include: native vegetation percentage at 500 m buffer (p_veg_500); native vegetation percentage at 1000 m buffer (p_veg_1000); cattle ranching cover percentage at 500 m buffer (p_pec_500); cattle ranching cover percentage at 1000 m buffer (p_pec_1000); camera located inside or outside native vegetation cover (substract); no covariables (.). psi = occupancy; p = detection; AICwt = Akaike weight.

	Δ AIC	**AICwt**
Red-legged seriema (*Cariama cristata)*		

p(substract)psi(p_veg_500)	0	6.50E-01
p(substract)psi(p_veg_1000)	1.25	3.50E-01
p(substract)psi(p_pec_500)	11.62	1.90E-03
p(substract)psi(p_pec_1000)	12.7	1.10E-03
p(p_veg_500)psi(p_veg_500)	32.96	4.50E-08
p(p_veg_500)psi(p_veg_1000)	34.37	2.20E-08
p(.)psi(p_veg_500)	38.41	3.00E-09
p(.)psi(p_veg_1000)	39.9	1.40E-09
p(p_veg_500)psi(p_pec_500)	45.36	9.20E-11
p(p_veg_500)psi(p_pec_1000)	46.62	4.90E-11
p(.)psi(p_pec_500)	51.2	4.90E-12
p(.)psi(p_pec_1000)	52.52	2.60E-12
p(.)psi(.)	58.85	1.10E-13
Greater rhea (*Rhea americana)*		
p(substract)psi(p_veg_500)	0	4.30E-01
p(substract)psi(p_veg_1000)	2.5	1.20E-01
p(p_veg_500)psi(p_veg_500)	3	9.50E-02
p(p_veg_500)psi(p_pec_1000)	3.34	8.00E-02
p(p_veg_500)psi(p_pec_500)	3.51	7.40E-02
p(p_veg_500)psi(p_veg_1000)	3.6	7.10E-02
p(substract)psi(p_pec_500)	3.92	6.00E-02
p(substract)psi(p_pec_1000)	4.29	5.00E-02
p(.)psi(p_veg_500)	6.4	1.70E-02
p(.)psi(p_veg_1000)	12.62	7.80E-04
p(.)psi(p_pec_500)	20.57	1.50E-05
p(.)psi(p_pec_1000)	22.75	4.90E-06
p(.)psi(.)	27.89	3.80E-07
Bare-faced curassow *(Crax fasciolata)*		
p(p_veg_500)psi(p_pec_500)	0	3.90E-01
p(p_veg_500)psi(p_veg_500)	0.84	2.50E-01
p(p_veg_500)psi(p_pec_1000)	1.51	1.80E-01
p(p_veg_500)psi(p_veg_1000)	1.6	1.70E-01
p(substract)psi(p_veg_500)	20.99	1.10E-05
p(.)psi(p_veg_500)	21.05	1.00E-05
p(substract)psi(p_pec_500)	21.29	9.30E-06
p(substract)psi(p_veg_1000)	23.01	3.90E-06
p(.)psi(p_pec_500)	23.18	3.60E-06
p(.)psi(p_veg_1000)	23.66	2.80E-06
p(substract)psi(p_pec_1000)	24.02	2.40E-06
p(.)psi(p_pec_1000)	26.47	6.90E-07
p(.)psi(.)	34.61	1.20E-08
White-tipped Dove (*Leptotila verreauxi)*		
p(substract)psi(p_pec_1000)	0	3.80E-01
p(substract)psi(p_pec_500)	0.91	2.40E-01
p(substract)psi(p_veg_500)	1.26	2.00E-01
p(substract)psi(p_veg_1000)	1.49	1.80E-01
p(p_veg_500)psi(p_pec_1000)	12.26	8.20E-04
p(p_veg_500)psi(p_veg_500)	12.44	7.50E-04
p(p_veg_500)psi(p_veg_1000)	13.13	5.30E-04
p(p_veg_500)psi(p_pec_500)	13.28	4.90E-04

p(.)psi(p_veg_500)	17.94	4.80E-05
p(.)psi(p_pec_1000)	18.24	4.10E-05
p(.)psi(p_veg_1000)	18.77	3.20E-05
p(.)psi(p_pec_500)	19.23	2.50E-05
p(.)psi(.)	24.35	1.90E-06
White-eared oposum *(Didelphis albiventris)*		
p(p_veg_500)psi(p_veg_500)	0	0.171
p(p_veg_500)psi(p_pec_500)	0.24	0.152
p(.)psi(p_pec_500)	0.69	0.122
p(substract)psi(p_pec_500)	1.4	0.085
p(.)psi(p_veg_500)	1.5	0.081
p(p_veg_500)psi(p_pec_1000)	1.78	0.07
p(.)psi(p_pec_1000)	1.91	0.066
p(substract)psi(p_veg_500)	1.92	0.066
p(.)psi(.)	2.43	0.051
p(substract)psi(p_pec_1000)	2.73	0.044
p(p_veg_500)psi(p_veg_1000)	3.06	0.037
p(.)psi(p_veg_1000)	3.3	0.033
p(substract)psi(p_veg_1000)	4.07	0.022
Southern tamandua *(Tamandua tetradactyla)*		
p(p_veg_500)psi(p_veg_500)	0	0.227
p(p_veg_500)psi(p_veg_1000)	1.16	0.127
p(p_veg_500)psi(p_pec_1000)	1.38	0.114
p(p_veg_500)psi(p_pec_500)	1.58	0.103
p(.)psi(.)	2.13	0.078
p(.)psi(p_pec_1000)	2.37	0.069
p(.)psi(p_veg_500)	2.49	0.065
p(.)psi(p_pec_500)	2.55	0.063
p(.)psi(p_veg_1000)	2.85	0.055
p(substract)psi(p_pec_1000)	4.22	0.028
p(substract)psi(p_pec_500)	4.4	0.025
p(substract)psi(p_veg_500)	4.41	0.025
p(substract)psi(p_veg_1000)	4.7	0.022
Tapeti (*Sylvilagus brasiliensis)*		
p(p_veg_500)psi(p_veg_500)	0	0.2993
p(substract)psi(p_veg_500)	0.83	0.1978
p(p_veg_500)psi(p_pec_500)	1.74	0.1252
p(substract)psi(p_pec_500)	1.87	0.1172
p(p_veg_500)psi(p_veg_1000)	3.28	0.0581
p(substract)psi(p_veg_1000)	3.57	0.0501
p(substract)psi(p_pec_1000)	3.78	0.0452
p(p_veg_500)psi(p_pec_1000)	3.84	0.044
p(.)psi(p_veg_500)	4.93	0.0254
p(.)psi(p_pec_500)	5.69	0.0174
p(.)psi(p_veg_1000)	7.41	0.0073
p(.)psi(p_pec_1000)	7.45	0.0072
p(.)psi(.)	7.94	0.0057
Crab-eating fox *(Cerdocyon thous)*		
p(.)psi(p_veg_500)	0	0.33611

p(p_veg_500)psi(p_veg_500)	0.051	0.3276
p(substract)psi(p_veg_500)	1.055	0.19831
p(p_veg_500)psi(p_veg_1000)	3.707	0.05266
p(.)psi(p_veg_1000)	3.937	0.04695
p(substract)psi(p_veg_1000)	5.126	0.0259
p(p_veg_500)psi(p_pec_500)	9.507	0.0029
p(p_veg_500)psi(p_pec_1000)	9.536	0.00286
p(.)psi(p_pec_500)	10.112	0.00214
p(.)psi(p_pec_1000)	10.211	0.00204
p(substract)psi(p_pec_500)	11.409	0.00112
p(substract)psi(p_pec_1000)	11.545	0.00105
p(.)psi(.)	13.6	0.00037
Leopardus sp.		
p(.)psi(p_pec_500)	0	0.17766
p(substract)psi(p_pec_500)	0.73	0.12363
p(.)psi(p_veg_500)	0.96	0.10975
p(.)psi(p_veg_1000)	1	0.10792
p(p_veg_500)psi(p_pec_500)	1.33	0.09158
p(p_veg_500)psi(p_veg_500)	1.34	0.09103
p(.)psi(p_pec_1000)	1.61	0.07934
p(p_veg_500)psi(p_veg_1000)	2.12	0.06159
p(p_veg_500)psi(p_pec_1000)	2.31	0.056
p(substract)psi(p_veg_500)	2.49	0.05122
p(substract)psi(p_veg_1000)	2.73	0.04534
p(.)psi(.)	7.27	0.00468
p(substract)psi(p_pec_1000)	12.88	0.00028
Tayra (*Eira barbara)*		
p(substract)psi(p_veg_500)	0	3.30E-01
p(substract)psi(p_pec_500)	0.043	3.20E-01
p(substract)psi(p_pec_1000)	0.298	2.80E-01
p(substract)psi(p_veg_1000)	6.197	1.50E-02
p(.)psi(p_veg_500)	6.674	1.20E-02
p(p_veg_500)psi(p_veg_500)	6.888	1.00E-02
p(p_veg_500)psi(p_veg_1000)	6.951	1.00E-02
p(p_veg_500)psi(p_pec_500)	7.111	9.40E-03
p(p_veg_500)psi(p_pec_1000)	7.276	8.60E-03
p(.)psi(p_pec_500)	9.831	2.40E-03
p(.)psi(p_pec_1000)	11.067	1.30E-03
p(.)psi(p_veg_1000)	14.194	2.70E-04
p(.)psi(.)	34.139	1.30E-08
Red brocket *(Mazama americana)*		
p(substract)psi(p_veg_500)	0	3.20E-01
p(substract)psi(p_veg_1000)	1.19	1.80E-01
p(.)psi(p_veg_500)	1.79	1.30E-01
p(substract)psi(p_pec_500)	2.74	8.20E-02
p(substract)psi(p_pec_1000)	3.03	7.10E-02
p(p_veg_500)psi(p_veg_500)	3.04	7.10E-02
p(.)psi(p_veg_1000)	3.53	5.60E-02
p(p_veg_500)psi(p_veg_1000)	4.36	3.70E-02

p(p_veg_500)psi(p_pec_500)	6.01	1.60E-02
p(p_veg_500)psi(p_pec_1000)	6.23	1.40E-02
p(.)psi(p_pec_500)	7.26	8.60E-03
p(.)psi(p_pec_1000)	7.67	7.00E-03
p(.)psi(.)	16.98	6.70E-05
Azara's agouti *(Dasyprocta azarae)*		
p(p_veg_500)psi(p_veg_1000)	0	5.20E-01
p(p_veg_500)psi(p_veg_500)	0.53	4.00E-01
p(substract)psi(p_veg_1000)	6.12	2.40E-02
p(substract)psi(p_veg_500)	6.74	1.80E-02
p(p_veg_500)psi(p_pec_1000)	7.87	1.00E-02
p(.)psi(p_veg_1000)	8.16	8.80E-03
p(p_veg_500)psi(p_pec_500)	8.42	7.70E-03
p(.)psi(p_veg_500)	8.56	7.20E-03
p(substract)psi(p_pec_1000)	11.6	1.60E-03
p(substract)psi(p_pec_500)	12.11	1.20E-03
p(.)psi(p_pec_1000)	16.93	1.10E-04
p(.)psi(p_pec_500)	17.41	8.60E-05
p(.)psi(.)	20.36	2.00E-05
Agouti (Cuniculus paca)		
p(substract)psi(p_veg_1000)	0	5.30E-01
p(substract)psi(p_veg_500)	0.58	4.00E-01
p(substract)psi(p_pec_1000)	5.26	3.90E-02
p(substract)psi(p_pec_500)	5.97	2.70E-02
p(.)psi(p_veg_1000)	31.38	8.20E-08
p(.)psi(p_veg_500)	32.05	5.90E-08
p(p_veg_500)psi(p_veg_1000)	33.35	3.10E-08
p(p_veg_500)psi(p_veg_500)	34.01	2.20E-08
p(.)psi(p_pec_1000)	35.82	8.90E-09
p(.)psi(p_pec_500)	36.51	6.30E-09
p(.)psi(.)	37.68	3.50E-09
p(p_veg_500)psi(p_pec_1000)	37.82	3.30E-09
p(p_veg_500)psi(p_pec_500)	38.51	2.30E-09
Collared peccary *(Pecari tajacu)*		
p(substract)psi(p_veg_500)	0	4.70E-01
p(substract)psi(p_pec_500)	0.14	4.40E-01
p(substract)psi(p_veg_1000)	5.54	2.90E-02
p(substract)psi(p_pec_1000)	5.97	2.40E-02
p(p_veg_500)psi(p_veg_500)	6.06	2.30E-02
p(.)psi(p_veg_500)	7.81	9.50E-03
p(p_veg_500)psi(p_pec_500)	9.98	3.20E-03
p(.)psi(p_pec_500)	10.43	2.60E-03
p(p_veg_500)psi(p_veg_1000)	13.77	4.80E-04
p(.)psi(p_veg_1000)	14.82	2.80E-04
p(p_veg_500)psi(p_pec_1000)	17.47	7.50E-05
p(.)psi(p_pec_1000)	17.58	7.10E-05
p(.)psi(.)	222.68	2.10E-49

Table 2. Threshold values of the fourteen species that responded to native vegetation cover at the 500 m buffer.

	Class	**Species**	**English name**	**Threshold value**	**Response to veg. Los**
1	*Aves*	*Cariama cristata*	Red-legged seriema	67.48	Positive
2	*Aves*	*Rhea americana*	Greater rhea	41.19	Positive
3	*Aves*	*Crax fasciolata*	Bare-faced curassow	43.03	Negative
4	*Aves*	*Leptotila verreauxi*	White-tipped Dove	53.42	Negative
5	*Mammalia*	*Didelphis albiventris*	White-eared oposum	56.33	Negative
6	*Mammalia*	*Tamandua tetradactyla*	Southern tamandua	38.39	Negative
7	*Mammalia*	*Sylvilagus brasiliensis*	Tapeti	54.24	Negative
8	*Mammalia*	*Cerdocyon thous*	Crab-eating fox	54.21	Positive
9	*Mammalia*	*Leopardus sp.*	Leopardus	38.2	Negative
10	*Mammalia*	*Eira barbara*	Tayra	43.91	Negative
11	*Mammalia*	*Pecari tajacu*	Collared peccary	15.15	Negative
12	*Mammalia*	*Mazama americana*	Red brocket	53.28	Negative
13	*Mammalia*	*Dasyprocta azarae*	Azara's agouti	48.41	Negative
14	*Mammalia*	*Cuniculus paca*	Agouti	61.35	Negative

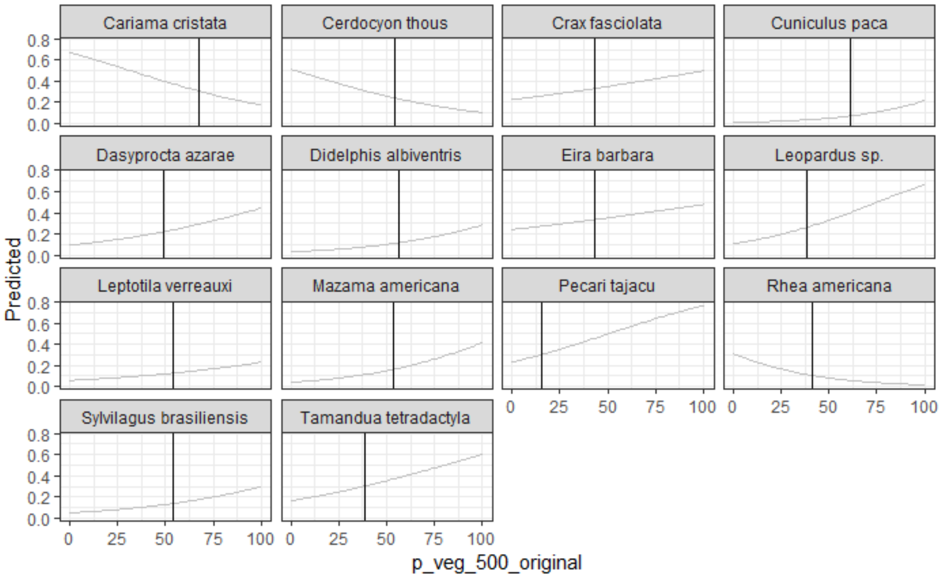

Figure 2. Occupancy models of the species that responded to native vegetation cover at 500 m buffer, and its threshold (black line)

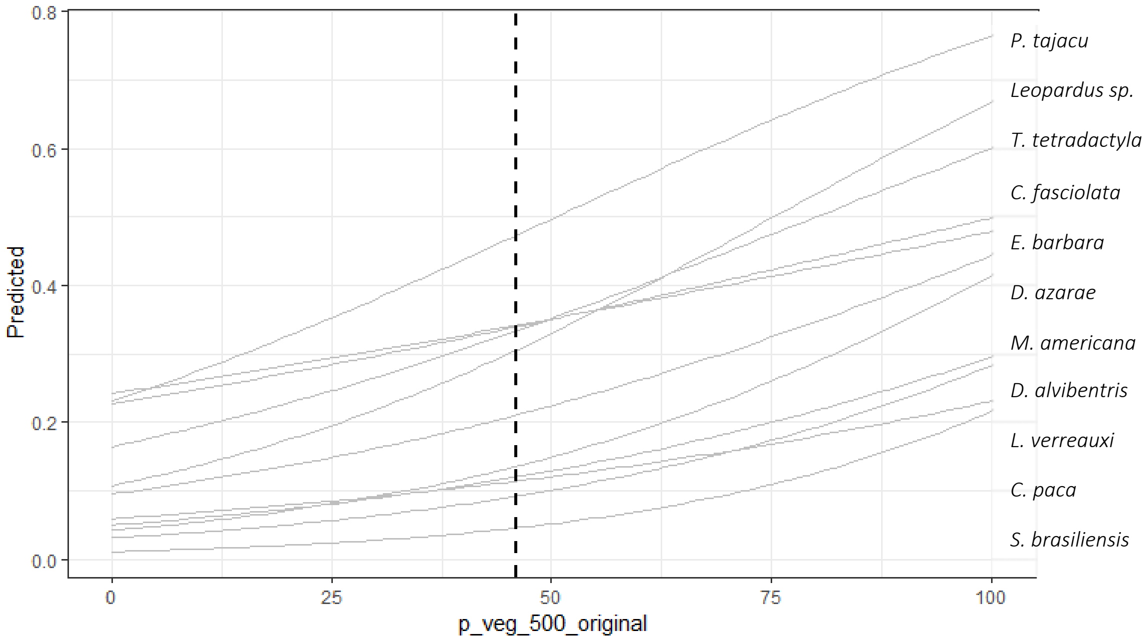

Figure 3. Occupation models with all the species that responded negatively to native vegetation loss and the averaged threshold value = 45.9%. black line.

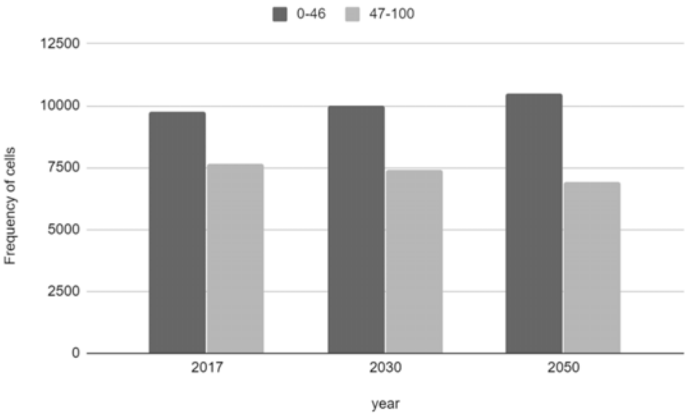

Figure 4. Changes in the frequency of 1 km^2 cells above and below the community threshold value in the Bodoquena LTER area.

Discussion

Even though the region where the study area is located comprised a vegetation mosaic of forested and open vegetation areas, native vegetation cover had an overarching influence over species occupancy variability across the assessed transformation gradient, which is consistent with most findings in analyses about species responses to land use cover transformation (Nagy-Reis et al. 2017, Regolin et al. 2017, Rocha et al. 2017, Boesing et al. 2018, Zimbres et al. 2018, Melo et al. 2018). Our results support the hypothesis that the occupancy probability of the species registered by the camera traps have a non-linear response to one or more landscape variables, mainly to native vegetation cover at 500 m buffer, but, since thresholds are species-specific, they occurred at different points across the land use change gradient.

Thresholds from species negatively affected by native vegetation loss ranged from 15.15% for Collared peccary (*Pecari tajacu*) to 61.35% for Paca (*Cuniculus paca*). However, apart from the Collared peccary, whose threshold value was the lowest, other species had a value higher than 38%, which is consistent with empirical studies of threshold responses at species level such as 35% of vegetation loss for the maned sloth (*Bradypus torquatus*) in the Brazilian

Atlantic Forest (Santos et al. 2019). Additionally, we obtained a community threshold of 45.97%, which is also consistent with findings, for vertebrates, in other Brazilian biomes. For instance, the community threshold for birds and mammals has been estimated as 30% to 50% in the Atlantic forest (Martensen et al. 2012, Banks Leite, 2014), 40% for the Amazon (Ochoa-Quintero et al. 2015), and 47% for bats in the Cerrado (Muylaert et al. 2016). Some authors have mentioned the downsides of using richness as dependent variable, since this high-level indicator may mask the species-specific responses to habitat conversion (de Oliveira et al. 2018, Van der Hoek et al. 2013, Lindenmayer & Luck 2005). By using a more refined indicator such as species' occupancy probability and selecting only those species that are negatively affected by native vegetation loss, we avoid masking the value with those species that are benefited with the increase of open areas. Besides, occupancy probability models take into account species detection and non-detection probabilities, which avoids the problem of implying that a species is absent from a site, because of the possible lack of detection on the sampling day (Mackenzie et al. 2002).

The species which occupancy probability decrease along with native vegetation cover constitute highly variable subset from birds such as *Crax faciolata* and *Leptotila verrauxi*, to medium to large mammals such as *T. tetradractyla*, *S. brasilliensis*, *Leopuardus* sp., *E. barbara* and *M. americana*. These species are known to be forest dependent species to some degree in other regions. On the opposite the species which occupancy increased as native vegetation decrease include the birds *C. cristata*, and *R. americana* and mammals such as *C. thous* species known to be associated to grasslands and open areas.

Remarkably, our results show that the occupancy probability tendency is more stable below the threshold value and increases steadily with native vegetation percentage cover above the threshold (Fig 2, Figure 3). This suggests that in areas with less vegetation the probability of occupancy is low, but this probability strongly decreases above the threshold value, meaning that populations are already vulnerable when they reach such value. This trend was consistent across the species where occupancy probability was negatively affected by land use change and may have different interpretations about species responses to land use change. Firstly, previous thresholds responses studies based on community measurements that showed strong decline below threshold value are probably masking the trajectory of the tendency line as explained before (as in Appendix 4). Secondly, the fact that the occupancy probability of each species is more stable below the threshold and increases as vegetation increases may indicate that once the threshold is crossed, the state of the population is already at risk, since its occupation probability was already decreasing before the threshold value was reached. Thirdly, and consequently, only those areas cells with high vegetating cover may maintain higher occupancy probabilities.

Previous thresholds studies showed that abrupt changes take place below threshold value which determine conservation or restoration priorities (Banks-Leite et al. 2014). Our results highlight the need to focus conservation efforts on areas with vegetation percentage above the threshold, protecting the landscapes with the higher vegetation cover, while restauration especially in areas above the threshold value where probability of occupancy is decreasing, may be the following priority. In addition, results of species decline above the threshold highlight the relevance of protected areas where occupancy for most species is high and may act as population source. Besides, our findings suggest that the area above the threshold may not guarantee that populations do not disappear, implying that caution is needed when using thresholds of species loss to advice policy making and land use planning.

Consequences of expected land use change on species occupancy

Land use change modeling shows that by 2030 and 2050 the native vegetation cover tends to decrease: almost half of current area with values above the threshold will be below them by 2050. Given that the threshold results suggest that having areas above the threshold does not guarantee that population are not declining, cells close to 100% vegetation are key to maintain in the landscape. This highlights the importance of protected areas such as Kadiwéu indigenous land and Bodoquena Natural Park, which comprises the majority of the 100% native vegetation cover cells nowadays and in modeled scenarios. Despite positive future projections for both indigenous lands and protected areas, caution should remain, protection of those areas depends on bills under way which may undermine the role of those areas to protect landscapes with high native vegetation cover.

Moreover, private protected areas are also crucial to maintain high native vegetation cover percentages. Current legislation is not enough to even secure preserved areas above the threshold, since according to the Native Vegetation Protection Law (NVPL, known as the "New Forest Code, Brazilian Federal Law 12,651 of 2012) in the Cerrado, 20% of the property need to be kept as reserve, and the averaged threshold for negatively affected species is close to 40%. This implies that, even if the current law is accomplished, the community of mammals and birds in this part of the Cerrado is still vulnerable. This highlights the importance of understanding species colonization and re-colonization processes on the interface of protected and non-protected areas, which may play role on rescuing species from local extinction in the region. Hence, actions beside command/control such as ecotourism, payments for environmental services or fiscal incentives, may act as a key incentive for conservation of this region.

During a time of discussions about the new deal of the Convention of Biological Diversity in 2020, these results support the need of further analyses to clarify the consequences of land use change over particular species in terms of their population viability. Our findings are crucial to identify the relevance of protected areas with low transformation and the relevance of retaining large tracks of forest across private properties in the region, these areas retain the higher occupancy probability of large birds and mammals, but also, the need to consider what is happening outside of those areas where human footprint is recognized as one of the largest threats affecting larger number of species globally (O`Bryan et al. 2019). In this context, it is relevant to generate time series to assessing occupation, extinction and colonization models and the role of protected areas in these processes (Ahumada et al. 2013, Ontiveros et al. 2019). Clarifying the interactions between protected and non-protected areas, including the discussion of conservation targets based on area (Wilson, 2016), is important to prevent populations reaching threshold points (nearly 50%) and consequently local extinctions. Losing birds and mammals such as the ones assessed in this study would mean that their functionality within the ecosystem, including their role in forest regeneration and seed dispersion, and their importance for ecotourism being attracted by large birds and mammals would be also lost, which would have negative implications for an already threatened biome such as the Cerrado.

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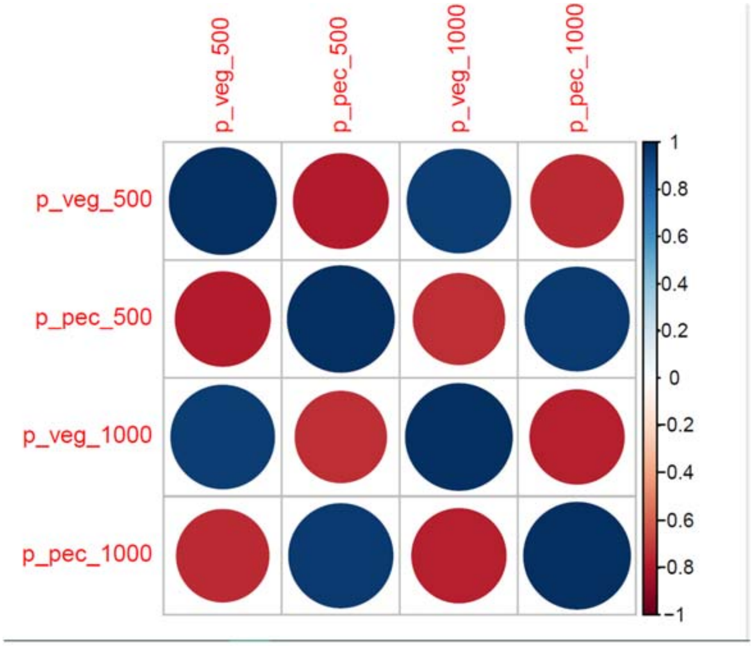
Supplementary Material

Appendix 1. Figure of correlation among the covariables. Native vegetation percentage at 500 m buffer (p_veg_500); native vegetation percentage at 1000 m buffer (p_veg_1000); cattle ranching cover percentage at 500 m buffer (p_pec_500); cattle ranching cover percentage at 1000 m buffer (p_pec_1000).



Appendix 2. Registered species from the camera traps, and their number of presence events. One presence event corresponds to a period of 24 hours per camera trap with at least one picture. Birds' scientific and English names are based on Gwynne et al. (2010). Mammals' scientific names follow Lopes et al. (2015), and their English names were taken from The IUCN Red List of Threatened Species™ website.

Class	**Species**	**English name**	**n events**
Aves	*Cariama cristata*	Red-legged seriema	355
Aves	*Rhea americana*	Greater rhea	39
Aves	*Crypturellus undulatus*	Undulated tinamou	12
Aves	*Crypturellus tataupa*	Tataupa tinamou	2
Aves	*Crypturellus parvirostris*	Small-billed tinamou	1
Aves	*Nothura maculosa*	Spotted nothura	1

Aves	*Ortalis canicollis*	Chaco chachalaca	13
Aves	*Penelope superciliaris*	Rusty-margined guan	1
Aves	*Pipile pipile*	Common piping guan	4
Aves	*Crax fasciolata*	Bare-faced curassow	108
Aves	*Bubulcus ibis*	Cattle egret	11
Aves	*Syrigma sibilatrix*	Whistling heron	4
Aves	*Theristicus caudatus*	Buff-necked ibis	10
Aves	*Aramus guarauna*	Limpkin	2
Aves	*Buteogallus meridionalis*	Savanna hawk	1
Aves	*Caracara plancus*	Southern crested caracara	3
Aves	*Aramides cajanea*	Gray-necked wood rail	7
Aves	*Vanellus chilensis*	Southern lapwing	4
Aves	*Leptotila verreauxi*	White-tipped Dove	79
Aves	*Guira guira*	Guira cuckoo	45
Aves	*Strix virgata*	Mottled owl	1
Aves	*Momotus momota*	Blue-crowned motmot	2
Aves	*Colaptes campestris*	Campo flicker	4
Aves	*Celeus flavescens*	Blond-crested woodpecker	2
Aves	*Campylorhamphus sp.*	Scythbill	1
Aves	*Furnarius Rufus*	Rufous hornero	4
Aves	*Pitangus sulphuratus*	Great kiskadee	1
Aves	*Cyanocorax chrysops*	Plush-crested jay	2
Aves	*Cyanocorax cyanomelas*	Purplish jay	1
Aves	*Tachyphonus Rufus*	Wite-lined tanager	1
Aves	*Coryphospingus cucullatus*	Red pileated finch	1
Mammalia	*Didelphis albiventris*	White-eared oposum	29
Mammalia	*Gracilinanus agilis*	Agile gracile oposum	12

Mammalia	*Dasypus novemcinctus*	Nine-banded armadillo	117
Mammalia	*Euphractus sexcinctus*	Yellow armadillo	70
Mammalia	*Cabassous unicinctus*	Southern naked-tailed armadillo	5
Mammalia	*Myrmecophaga tridactyla*	Giant anteater	79
Mammalia	*Tamandua tetradactyla*	Southern tamandua	50
Mammalia	*Sapajus cay*	Azara's capuchin	18
Mammalia	*Sylvilagus brasiliensis*	Tapeti	62
Mammalia	*Cerdocyon thous*	Crab-eating fox	179
Mammalia	*Chrysocyon brachyurus*	Maned Wolf	5
Mammalia	*Leopardus sp.*	Leopardus	34
Mammalia	*Puma concolor*	Puma	12
Mammalia	*Puma yagouaroundi*	Jaguarundi	7
Mammalia	*Panthera onca*	Jaguar	1
Mammalia	*Eira barbara*	Tayra	36
Mammalia	*Nasua nasua*	South American coati	61
Mammalia	*Procyon cancrivorus*	Crab-eating raccoon	26
Mammalia	*Tapirus terrestres*	Lowland tapir	230
Mammalia	*Pecari tajacu*	Collared peccary	232
Mammalia	*Tayassu pecari*	White-lipped peccary	25
Mammalia	*Mazama americana*	Red brocket	41
Mammalia	*Mazama gouazoubira*	Gray brocket	96
Mammalia	*Holochilus sp.*	Rodent	1
Mammalia	*Hylaeamys megacephalus*	Rodent	2
Mammalia	*Rhipidomys macrurus*	Rodent	2
Mammalia	*Hydrochoerus hydrochaeris*	Capybara	26
Mammalia	*Dasyprocta azarae*	Azara's agouti	191
Mammalia	*Cuniculus paca*	Agouti	28

Mammalia	*Thrichomys fosteri*	Rodent	16

Gwynne J.A., Ridgely R.S., Tudor G., & Argel M. 2010. Aves do Brasil. Vol. 1. Pantanal & Cerrado. Editora Horizonte

Lopes H.W., Duarte A.L., Santos C.C. 2015. Mamíferos não voladores do Pantanal e entorno. Campo Grande MS. Natureza em Foco. 224p

The IUCN Red List of Threatened Species (IUCN, accessed 15 October 2019); http://www.iucnredlist.org.

Appendix 3. bellow 2 AIC values of the four evaluated covariables for the occupancy models.

Species	**p_veg_500**	**p_pec_500**	**p_veg_1000**	**P_pec_1000**
Cariama cristata	0		1.25	
Rhea americana	0			
Crypturellus undulatus	0.36	1.41	1.56	1.76
Crax fasciolata	0.84	0	1.6	1.51
Leptotila verreauxi	1.26	0.91	1.49	0
Didelphis albiventris	0	0.24		1.78
Dasypus novemcinctus			1.11	0
Euphractus sexcinctus		0		
Myrmecophaga tridactyla			0	
Tamandua tetradactyla	0	1.58	1.16	1.38
Sylvilagus brasiliensis	0	1.74		
Cerdocyon thous	0			
Leopardus sp.	0.96	0	1	1.61
Eira barbara	0	0.04		0.29
Procyon cancrivorus				0
Tapirus terrestres		0		
Pecari tajacu	0	0.14		

Tayassu pecari		0.83		
Mazama americana	0		1.19	
Mazama gouazoubira		0		1.61
Dasyprocta azarae	0.53		0	
Cuniculus paca	0.58		0	
	14	**11**	**12**	**11**

Appendix 4. Threshold of species loss based on species richness and forest cover at 500 mt buffer from each camera trap, using Piece-wise analysis sensu Muggeo 2003.

We selected forest dependent species form the species registered in the camera trapping of the present study. We used generalized linear regression models (GLM) analysis in order to identify the influence of the variables (native vegetation cover at 500 and 1000 m buffers, anthropic cover at 500 and 1000 buffers) on the changes in species richness along the native vegetation gradient. The best model was identified using delta AIC according to (Burnham and Anderson 2002). Then, a model averaging procedure was performed to determine the contribution of each variable. As a result the native vegetation at 500 m buffer was identified as the most influencing, and we performed a Piece-Wise analysis using Segmented package (Muggeo 2008.) in the statistical language R, version 3.5.1 (R Core Team) run a piece-wise analysis based on this variable (Muggeo 2003) in order to find the potential thresholds associated with environmental gradient. The estimated threshold value was 52.57 ± 11.02.

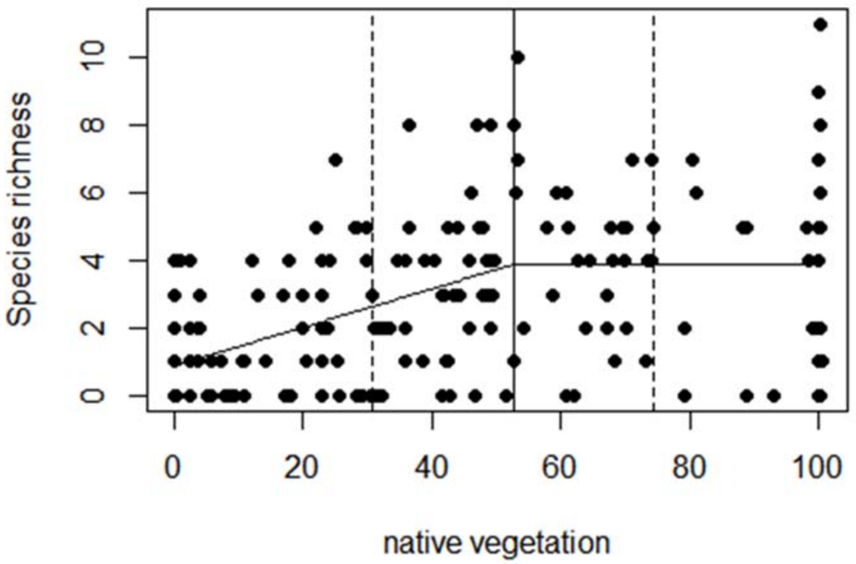


Figure. Threshold of species loss based on species richness and forest cover at 500 m buffer from each camera trap.

Burnham K & Anderson D. 2002. Model selection and multimodel inference: a practical information-theoretic approach, 2nd ed. Springer- Verlag, New York, NY.

Muggeo VM. 2008. Segmented: an R Package to Fit Regression Models with Broken-Line Relationships. R News **8(1)**: 20–25.

Appendix 5. Results of modelled native vegetation loss prediction maps for 2030 2050.

The results of the land use change modeling that were used to interpolate the threshold value presented above showed that roads and dry season length explained most native vegetation loss in the three analyzed periods (2010-2012, 2012-2014 and 2014-2016), while land cover and agricultural potential were important in only two periods (2010-2012 and 2012-2014). Altitude and cattle were important only in the 2014-2016 period. The presence of protected areas was the only variable reducing vegetation loss in all periods (see table below). As a result, vegetation cover will drop from 8,108 in 2016 to 7,907 km² of native vegetation (43.9 %) in 2030, and 7,575 km² (42,1 %) in 2050. Such values imply a native vegetation loss rate of 3.5% (± 0.1 95% C.I.) which corresponds to 265.0 km² (±8.3 95% C.I.) for 2030 and 7.9% (± 0.2 95% C.I.) which corresponds to 597.0 km² (± 18.9 95% C.I.) for 2050 (see maps below).

Table. Native vegetation loss drivers in the Bodoquena LTER area for each period.

Variables	2010	2012	2014
Land Cover	-0.576689	-0.121167	0
Distance to roads	-0.000017	-0.000019	-0.000026
Distance to cities	0	0	0
Dry season length	-0.993083	-0.981706	-0.323419
Elevation	0	0	-0.006655
Agricultural Potential	0.171168	0.248624	0
Distance to Rivers	0	0	0
Cattle	0	0	0.000013
Permanent Agriculture	0	0	0
Annual Crop Agriculture	0	0	0
Protected Areas	-1.977554	-1.982143	-1.975928

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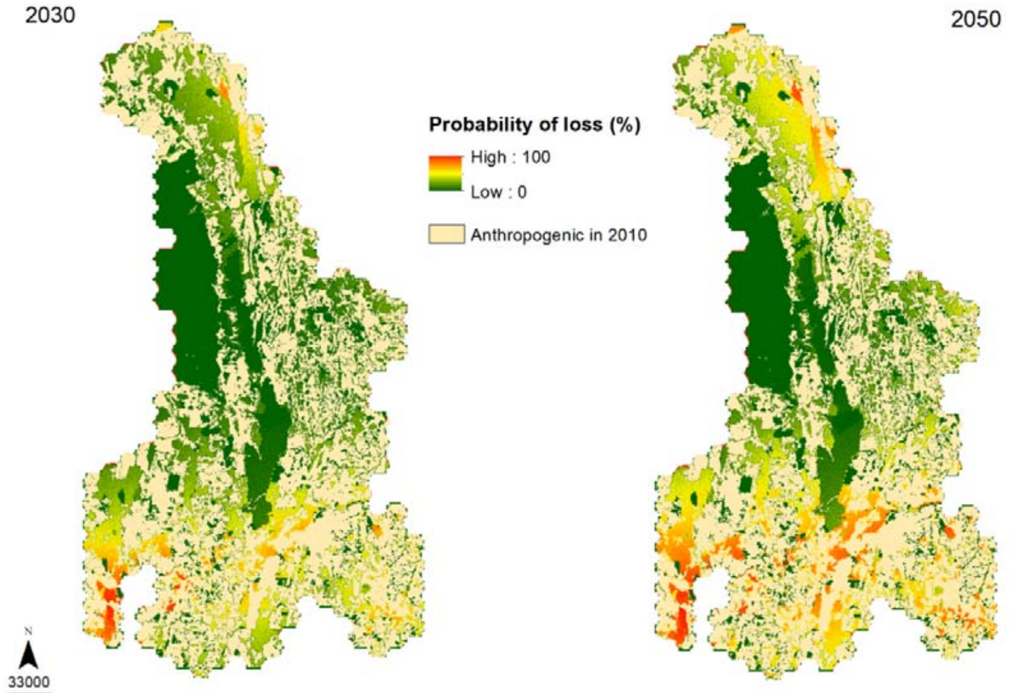

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Figure. Maps of the modelled native vegetation loss probability, adapted from Guerra et al (2020).

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Chapter 3. Local perceptions of biodiversity according to main economic activity in rural proprieties of the Cerrado Hotspot

Abstract

Land use transformation is one of the main drivers of biodiversity loss at global scale, and probably the main driver of biodiversity loss across the tropics. Several studies have shown the consequences of native vegetation loss and fragmentation over different taxonomic groups, reduction of phylogenetic diversity and decreasing of birds, mammals and environmental services. Although this evidence is increasing from analyses of different kinds, the local perception about biodiversity loss at a farm level is less conspicuous on the scientific literature and it is key for the implementation of applicable measurements aiming to reduce the impact of land use transformation over biodiversity components. We performed interviews to 37 local inhabitants dedicated to agriculture production, cattle ranching and tourism in a Cerrado area of Mato Gross do Sul state, to assess the knowledge about species richness, species dependence to native vegetation, and benefits or harms to household's main economic activity. Our results show that although there is not difference in the total richness perceived by the three interviewed groups, there is a significant difference in the richness perception of open areas species and forested species among tourist landowners (t 0.0194, n=6). Peccaries, tapirs, armadillos and felids were reported as prejudicial fauna for agriculture and cattle, and ant eaters, monkeys, agouties, and some bird species such as egrets and ibis were reported as beneficial. This information is crucial as local perception over biodiversity may be useful for implementing farm level strategies for biodiversity conservation, for this is the level at which management decisions across private lands are implemented.

Resumo

A transformação do uso da terra é um dos principais fatores de perda de biodiversidade em escala global e provavelmente o principal fator de perda de biodiversidade nos trópicos. Vários estudos demonstraram as consequências da perda e fragmentação da vegetação nativa sobre diferentes grupos taxonômicos, redução da diversidade filogenética e diminuição de aves, mamíferos e serviços ambientais. Embora essa evidência esteja aumentando a partir de análises de diferentes tipos, a percepção local sobre essas consequências no nível da propriedade é menos visível na literatura científica e é fundamental para a implementação de medidas aplicáveis com o objetivo de reduzir o impacto da transformação do uso da terra sobre os componentes da biodiversidade. . Realizamos entrevistas com 37 habitantes locais dedicados à produção agrícola, pecuária e turismo em uma área do Cerrado no estado de Mato Grosso do Sul, para avaliar o conhecimento sobre a riqueza de espécies, dependência de espécies da vegetação nativa e benefícios ou danos à principal atividade econômica do agregado familiar. Nossos resultados mostram que, embora não haja diferença significativa na riqueza total percebida pelos três grupos entrevistados, há uma diferença significativa na percepção de riqueza de espécies de áreas abertas e espécies florestais entre os proprietários turísticos (t 0,0194, n = 6). As queixadas, antas, tatus e felídeos foram relatados como fauna prejudicial para a agricultura e a pecuária, e tamanduás, macacos, tatus e algumas espécies de aves como garças e íbis foram relatadas como benéficas. Essas informações são cruciais, pois a percepção local sobre a biodiversidade pode ser útil para a implementação de estratégias no nível da fazenda, já que este é o nível no qual as decisões de gestão em terras privadas são implementadas.

Introduction

Since 1970´s the Brazilian Cerrado has experienced a rapid land use transformation, moved by the Government territorial development programs and economic dynamics, that has turned this region in an agro-industrial axis of the country (Inocêncio & Calaça 2010). Consequently, native ecosystems there have suffered significant alteration resulting in 46% of the original Cerrado biome lost (Klink & Machado 2005, Strassburg et al. 2017, Resende et al. 2019). At the same time, the Cerrado is considered a biodiversity hotspot and is remarkably important for the ecosystem services it provides (Strassburg et al. 2017). At present, rural areas across the Cerrado landscapes are formed by a mosaic of native vegetation and lands devoted to agriculture, cattle ranching, tourism, and a number of protected areas. As a result, in this biome, rural inhabitants live in a context of developing their economic activities surrounding of variable levels of biodiversity.

Considered that most of the Cerrado are rural private lands (Inocêncio & Calaça 2010), it is important to understand the dynamics of the relationship of people living with nature in these lands, since besides norms and institution, individuals, and the way how they perceive the world are the ultimate agents of decision making at farm level (Dinz et al. 2015). Hence, investigating how they perceived their own situation in regards their surrounding environment is essential to understand their actions and attitudes towards biodiversity. Understanding the way ecological and social components interact in a settlement (a village, town, city) is the key to manage effectively the coupled human–modified landscapes (Rajaram & Das 2010). In fact, medium to large birds and mammals are biodiversity groups widely integrated in the daily life of local farmers across the world, including association with environmental services and disservices (Lacher et al. 2019), but are also species disappearing from the local environment as previously discussed (Chapter 2).

Based on the fact that recognizing and dealing with the pluralities of stakeholders' perceptions is currently a key aspect of effective natural resource management, we investigated which is the perception and current knowledge of surrounding biodiversity and if the economic activity of the landowners influenced the way they perceive their environment. We assessed if different actors have distinctive perceptions about their environment, specifically about medium and large mammals, whereas they have different perceptions about the species richness, species dependence to native vegetation, and benefits or harms to household's main economic activity. This information is useful for policy implementation in one the most threatened biodiversity hotspot of the world, specifically, giving the importance of the role of the environmental services for people living in rural areas.

Methods

Study area

This study was carried out in South-west of Brazil, in a Cerrado region of the Mato Grosso do Sul State, part of the Serra da Bodoquena plateau, in the Paraguay River basin, (Figure 1). It comprises the municipalities of Bonito, Bodoquena and Jardim. There, main economic land uses are agricultural monocultures (e.g. soy and maize) and cattle production, and tourism (SOS Pantanal et al. 2017). Within this region we selected interviews sites based on the Long Term Ecological Research Planalto da Bodoquena (LTER hereafter), and the availability and willingness of inhabitants to take part of the study.

Interviews

We performed 37 semi-structured interviews to local inhabitants within the selected LTER landscapes between April and May 2018. These interviews recollected information of the inhabitant's perceptions about the medium and large mammal fauna occurring in their propriety and the possible benefits or prejudices they cause. The enquiry about presence/absence of medium and large size birds and mammals followed the methods used by Ochoa-Quintero and co-workers (2015), who used pictures to identify thresholds of species loss thresholds in the Brazilian Amazonia. Pictures of different species from other regions were included in order to tease apart false responses (Michalski & Peres 2005). Also, the interview included questions about ideas about conservation (such as the most efficient shape of the area for preserving the wild fauna, the most common and rare species and the benefits and problems of having biodiversity close by their properties (see details in appendix 1). Before each interview, we introduced as researchers from the Federal University Mato Grosso do Sul, and explained the consent and non-compulsory nature of their participation. The interviewee signed a consent form approved by the University's Etic Committee (see appendix 2).

Data analysis

Results from the interview were gathered in a database for analysis. We used descriptive statistic to analyze knowledge of local people about the species in their properties aiming to identify differences in local knowledge according to economic activities. Analyzes were performed using the statistical language R version 3.5.1 (R Core Team, 2018).

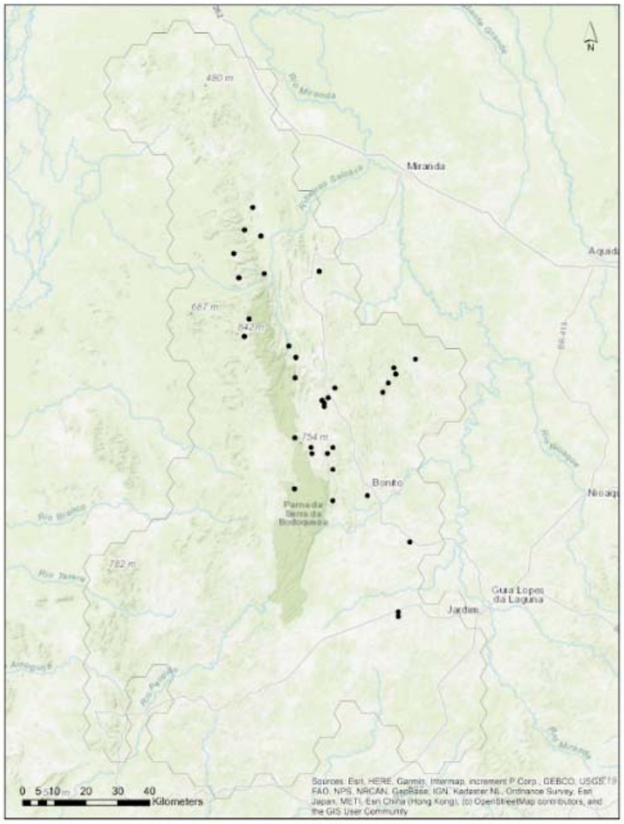

Figure 1. Map of the study area. black points are the interview points, gray line is the Bodoquena LTER study area.

Results

From the 37 interviews performed, 20 were in Bonito, 14 in Bodoquena and 2 in Jardim. The average time of permanence in the propriety was 14.45 years (SD 15.97). The economic use of the land of the 37 interviews performed was clustered in three categories: cattle ranching (n=18), agriculture (n= 13), and tourism (n=6), which is also related to the main land use cover across the study area (SOS Pantanal et al 2017).

In total, interviews reported 26 species of medium and large mammals, for a total of 520 reports (Figure 2, table 1). The most common reported species was the Lowland tapir (*Tapirus terrestis,* n=44), followed by the White-lipped peccary (*Tayassu pecari,* n=40), and the Collared pecary *(Pecari tajacu,* n=38). The less reported species was the Giant armadillo (*Priodontes maximus,* *n=1).* When compared the total richness perceived by category, we found no significant difference among the three categories (Figure 3). Landowners devoted to agriculture recognized an averaged richness of 10.15 ± 3.44 of large and medium sized mammals in their proprieties. Landowners devoted to cattle ranching recognized an averaged richness of 10.58 ± 2,65, and those devoted to tourism recognized, on average, 10.75 ± 2.76 species (Figure 3). There was no significant difference among those values, which means that landowners recognize a similar richness on their proprieties across the study area (table 2).

When inquiring for the different species found in open and forested areas, the richness varies according to the main economic activity as follows: 8.69 ± 1.84 in open areas and 11.61 ± 4.09 in forested areas for agriculture (n=13); 9.94 ± 2.73 in open areas and 11.22 ± 2.49 in forested areas for cattle ranching (n=18); and 9.00 ± 1.67 in open areas and 12.5 ± 2.59 in forested areas for tourism (n=6). Comparing such perceptions showed differences within landowners devoted to tourism (t 0.0194). Both Agriculture and Cattle ranching showed no differences of their perception about richness between open and forested areas (Figure 4).

According to their perception, when asking about which forest fragments configuration was better for the wild fauna in their propriety, they choose one big fragment (n= 28) over the other two options (small fragments, and connected small fragments see appendix 1). The reported beneficial species were monkeys (*Alouatta caraya* and *Sapajus cay*), tapirs (*Tapirus terrestris*) and agoutis (*Cuniculus paca*), especially among touristic proprieties, due to their importance for attracting tourists. Also, ant eaters (*Tamandua tetradactyla* and *Myrmecophaga tridactyla*) were reported as beneficial by cattle ranching land owners due to their ability to destroy termite nests. Birds such as egrets, ibis and seriemas were also considered as beneficial because their diet (insects or snakes). Two responses considered all animals beneficial and nine responses were neutral (nor beneficial nor damaging), most of the responses (13) referred not knowing the answer. The most prejudicial species were peccaries, armadillos, tapirs (for crops), and puma concolor and jaguar (for feeding on cattle). The Wild boar (*Sus scrofa)* was mentioned as introduced and increasing in number and territory area recently. Five responses were neutral and five did not know the answer.

Table 1. Registered species from the interviews. Mammals' scientific names are listed following Lopes et al. (2015), and their English names were taken The IUCN Red List of Threatened Species™ website

Species	**English name**	**n reports**
Priodontes maximus	Giant armadillo	1
Dasypus novemcinctus	Nine-banded armadillo	22
Euphractus sexcinctus	Yellow armadillo	27
Myrmecophaga tridactyla	Giant anteater	8
Tamandua tetradactyla	Southern tamandua	28
Alouatta caraya	Black-and-gold Howler Monkey	28
Sapajus cay	Azara's capuchin	37
Cerdocyon thous	Crab-eating fox	9
Chrysocyon brachyurus	Maned Wolf	4
Leopardus sp.	Leopardus	28
Puma concolor	Puma	31
Puma yagouaroundi	Jaguarundi	12
Panthera onca	Jaguar	15
Speothos venatucus	Bush dog	3
Eira barbara	Tayra	23

Nasua nasua	South American coati	8
Procyon cancrivorus	Crab-eating raccoon	7
Tapirus terrestres	Lowland tapir	44
Pecari tajacu	Collared peccary	38
Tayassu pecari	White-lipped peccary	40
Mazama americana	Red brocket	20
Mazama gouazoubira	Gray brocket	20
Hydrochoerus hydrochaeris	Capybara	9
Dasyprocta azarae	Azara's agouti	31
Cuniculus paca	Agouti	19
Sus scrofa	Wild boar	8

Lopes HW, Duarte AL, Santos CC. 2015. Mamíferos não voladores do Pantanal e entorno. Campo Grande MS. Natureza em Foco. 224p

The IUCN Red List of Threatened Species (IUCN, accessed 15 October 2019); http://www.iucnredlist.org.

Table 2. ANOVA comparing the perceived richness among the three categories of main economic land uses: agriculture, cattle ranching and tourism.

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Main economic activity	2	7.4	3.697	0.37	0.693
Residuals	34	333.7	9.991		

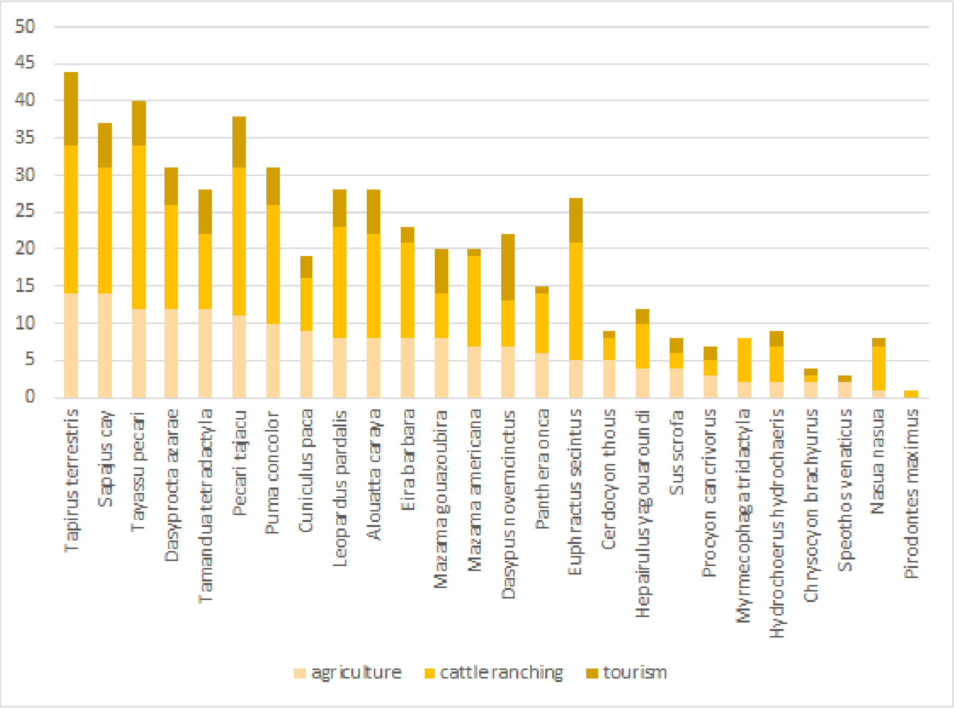

Figure 2. Frequency of species reported by main economic activity.

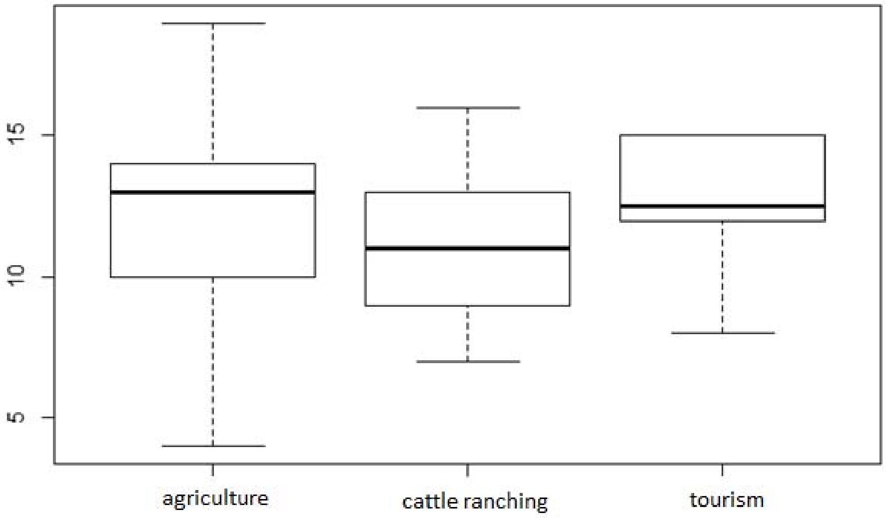

Figure 3. Total richness perceived by category of main economic activity of the propriety.

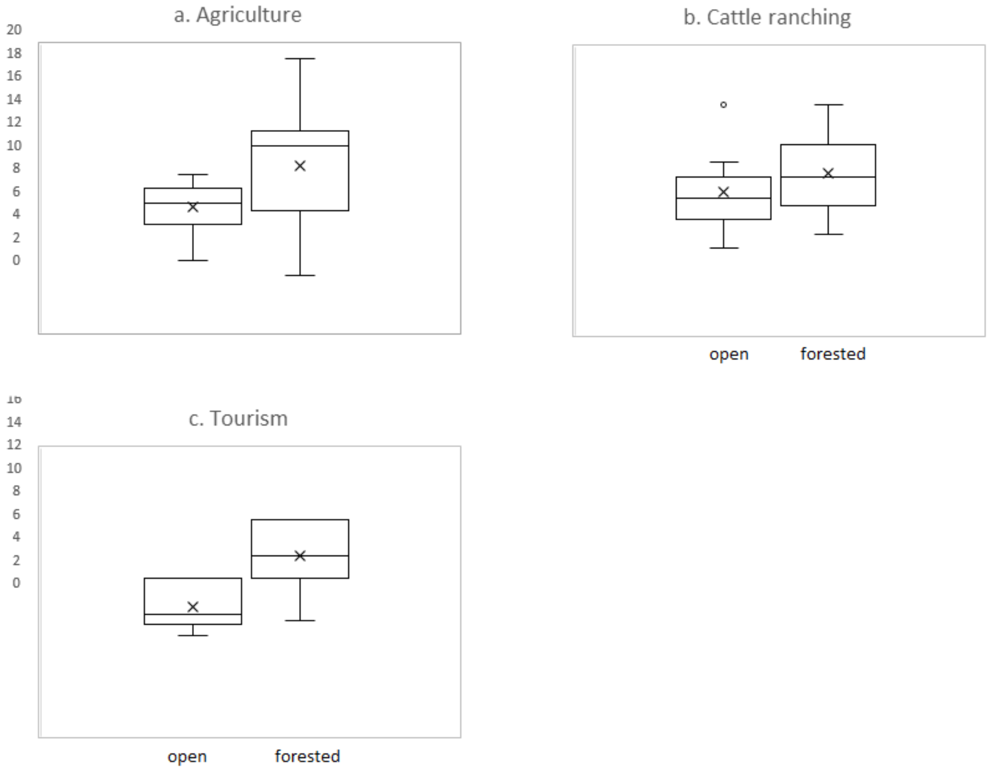

Figure 4 Mean differences of perceived richness compared by economic activity of the interviews, in the open and forested areas of their proprieties.

Discussion

Our results showed that local farmers are able to recognize a relative similar set of species independently to their main economic activities in their properties. This imply a good knowledge about the biodiversity across the human modified landscapes and the possibility to recognize their knowledge as tool for property level management aiming to reduce land use cover pressure over biodiversity (Gandiwa 2012, Camino et al. 2020). The fact that there is significant difference between perceived richness of open and forested areas of touristic proprieties may be related with the fact that this activity has a close relation with biodiversity, since tourism are attracted for wildlife in the region, and thus, landowners are better able to better recognize the fauna in their proprieties .

The fact that most of local land owners are able to recognize large body species such as the Lowland tapir (*Tapirus terrestris*) and the White lipped-peccary (*Tayassu pecari*) as common species across the study may have relevant signals about the conservation status of the region. These species are commonly affected by hunting (Cruz et al. 2014, Ferregueti et al. 2017) and deforestation pressure in different areas of their home range (Ochoa-Quintero et al. 2015). The presence of the Bodoquena national park and the presence of remnants of native vegetation cover across the study area may facilitate the maintenance of population of these species across the region. Nonetheless both the maintenance of native vegetation cover across the study area is threaten of disappearance in the coming years (Guerra et al. 2019, chapter 2) despite of the role of remaining vegetation for the protection the maintenance of environmental services for the whole Upper Pantanal River Basin (Guerra et al 2020).

The knowledge land owners have over biodiversity acknowledges, among other things, the dependence of much species to native vegetation cover. Indistinctly of main economic activities they recognize that there is a higher number of species (proxy of species richness) related to native vegetation, forest cover, that the number of species can survive in open areas. In addition, they also know that presence of rare species such as the Jaguar or the Giant armadillo are the ones that also disappear easily from human modified landscapes across the study area.

In addition, people identify the damages some species may incur in their lands according to production type. Nonetheless in some cases land owners recognize that damage is very low over production and do not consider any management strategy to control them, facilitating coexistence. But also, they know the benefits over agriculture management. For example, the presence of larger groups of ibis (*Theristicus caudatus*) over soy plantations imply the attack of caterpillars over the plants and the need to check and defining management strategies consequently. Another example is that some of the cattle ranching landowners recognized the presence of Tamanduas as a way to control termites in pasturelands of the region. This implies these species can used as central point for environmental services programs highlighting the relevance of them for improving management.

Land owners can also reflect on better strategies for conservation land management in their properties. Most land owners identified that best strategies for biodiversity protection is to maintain most of their legal reserve clustered in one forest block. Arguing that larger areas guarantee the possibility of larger areas for breeding success and also lower exposure to open areas which may threaten their survival. This is a relevant perception for properties land management because they not only recognize the importance of native vegetation for the survival of the species as previously mentioned, but also, the importance of having large forest tracks for their survival, as evidenced by Di Marco and colleagues (2019). This implies that proper implementation of strategies of farm management within the study area may reduce the impact of land transformation over biodiversity in the region (Negrões et al. 2018). Nonetheless, an economic analysis of the implementation of those strategies given farm size and commodities price, the location of legal reserve within farm for improving landscapes functionality, plus the need to assess environmental services benefits for agriculture and cattle ranching production in this region may play a relevant role for the protection of biodiversity in study area.

This study implies that strategies for land use planning may scale up from bottom up approaches which use a straying point the biodiversity knowledge land owners have about biodiversity in the study region. Both the knowledge and the potential benefits farmers perceive from biodiversity plus the income farms dedicated to ecotourism play role to get a better coexistence among biodiversity and production in the study region.

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Supplementary Material

Appendix 1 semi structured interviews forms

Nome:____
Data:____________________
Municipio:______________________
Proprietário: ____ Arrendatário _____ Funcionário ____
Há quanto tempo mora na propriedade ou com que frequência permanece nela: ____________
Qual é a principal atividade económica da propriedade _______________________

1. Quais 'animais grandes' existem em sua propriedade? (mostrar fotos das diferentes espécies) e classificar se são de áreas abertas ou mata

Área aberta	Mata

2. Desses animais:
    a. há alguns que beneficiam sua propriedade? Dê exemplos e explique por quê.

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    b. há alguns que prejudicam sua propriedade? Dê exemplos e explique por quê

----

    c. Há alguns que não beneficiam nem prejudicam? Dê exemplos e explique.

----

3. Quais são os mais comuns? Dê exemplos e explique por quê.

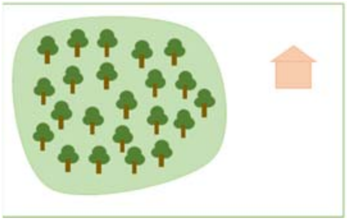
----

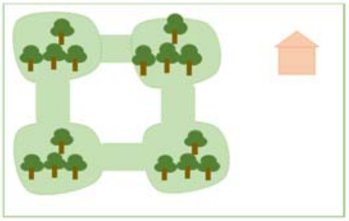
4. Quais são os mais escassos? Dê exemplos e explique por quê.

---O que acontece com os animais quando você transforma uma área de mata em uma área agropecuária?

Qual é o primeiro bicho que some? ____
    a. Há algum que depende do primeiro que some? _____________________________________
    b. Quando ele some, tem outros que somem com ele? _________________________________
    c. Quais? ____

5. Quanto de mata você acha que seria necessário para os animais que moram na sua propriedade existirem? ____
6. Em quais destas opções poderiam viver mais animais e como está dentro da sua propriedade? (mostrar desenhos de 3 opções)





Appendix 2. Consent form approved form the University's Etic Committee

TERMO DE CONSENTIMENTO LIVRE E ESCLARECIDO

Projeto: **Percepções de proprietários e moradores de áreas rurais do Planalto da Bodoquena (MS), sobre a fauna e a natureza em suas propriedades**

Os pesquisadores **Isabel Melo Vásquez** e **Fabio de Oliveira Roque,** da Universidade Federal do Mato Grosso do Sul – UFMS, solicitam sua colaboração para conceder entrevista sobre as ideias que você tem sobre os animais em sua fazenda. Com as informações, os pesquisadores pretendem entender quais são suas percepções sobre a natureza. A participação é voluntária e sua participação não acarretará em nenhuma despesa ou benefício. **Mesmo tendo autorizado, depois você terá o direito e a liberdade de retirar seu consentimento, independente do motivo e sem qualquer prejuízo para você**. A vantagem de participar deste estudo é somente de caráter científico e sua participação ajudará a ciência a entender melhor como o homem se relaciona o meio ambiente.

JUSTIFICATIVA, OBJETIVOS E PROCEDIMENTOS: As percepções dos moradores das áreas rurais são importantes pois é reconhecido que as propriedades privadas das áreas rurais podem contribuir ou não para a manutenção e o funcionamento dos serviços ecossistêmicos. O nosso objetivo é compreender as percepções de moradores da área rural dos municípios de Bonito, Bodoquena e Jardim em MS, sobre mamíferos em diferentes paisagens e sua relação com o meio ambiente. Com este propósito, realizaremos uma entrevista na qual você, caso aceite participar, registrará suas opiniões sobre os animais que moram em sua propriedade.

DESCONFORTOS, RISCOS E BENEFÍCIOS: Este projeto não apresenta riscos materiais ou para a saúde assim como benefícios para os sujeitos envolvidos. Pode haver uma exposição a riscos mínimos como constrangimento, cansaço ou desconforto pelo tempo das respostas ao questionário. Se isto ocorrer você poderá interromper as respostas e retomá-la posteriormente, se assim o desejar. Você também poderá parar a entrevista em qualquer momento e perguntar sobre o propósito das questões. Se quiser mais explicações sobre qualquer questão ou sobre o uso das informações do questionário ou do projeto, sinta-se à vontade para fazê-lo.

As entrevistas serão realizadas somente após o consentimento das pessoas entrevistadas. O Termo de Consentimento Livre e Esclarecido (TCLE) assegura que as informações obtidas serão utilizadas somente para fins científicos. Igualmente, este projeto não apresenta benefícios para os participantes. Os benefícios estão limitados à geração de conhecimento acadêmico e científico.

GARANTIA DE ESCLARECIMENTO, LIBERDADE DE RECUSA E GARANTIA DE SIGILO: As informações que você fornecer serão analisadas e os resultados serão divulgados somente para fins científicos. **Sua identidade será mantida em sigilo para sempre**. Você será esclarecido(a) sobre a pesquisa em qualquer aspecto que desejar, a qualquer tempo. Você é livre para recusar-se a participar, retirar seu consentimento ou interromper a participação a qualquer momento. A sua participação é voluntária e a recusa em participar não irá acarretar qualquer prejuízo ou perda de benefícios.

Os pesquisadores irão tratar a sua identidade com absoluto sigilo. Seu nome só será usado na assinatura deste documento, seu nome não aparecerá na entrevista e não será divulgado sem a sua permissão. Você não será identificado(a) em nenhuma publicação que possa resultar deste estudo. Este Termo de Consentimento Livre e Esclarecido será composto de duas vias, sendo uma via retida com o pesquisador responsável e outra via retida com você.

Consentimento Pós-Informação

Eu, ____ morador de _________________________ entendi o que a pesquisa vai fazer e aceito participar de livre e espontânea vontade. Por isso, dou meu consentimento para inclusão como participante da pesquisa e atesto que recebi uma cópia deste documento.

__________________________________ OU

Assinatura do entrevistado

Data ..../......../.........

Impressão do polegar

____

Nome do entrevistador

Contatos:

Aluna de doutorado: Isabel Melo Vasquez. Celular: (67) 9 8122-2767; e-mail: imelov@gmail.com
Nome do orientador: Fabio de Oliveira Roque. Telefone: (67) 3345-7807.

Universidade Federal do Mato Grosso do Sul. Campo Grande. Programa de Pós-Gradiação em Ecologia e Conservação. Secretaria do Programa: (67) 3573-7342

Comitê de Ética em Pesquisa com Seres Humanos da Universidade Federal de Mato
Grosso do Sul no telefone (67) 3345-7187, sito à Av. Costa e Silva, s/n - Cidade Universitária,
Campo Grande, MS.

General conclusion

Based on mathematical demographic equilibrium model, Lande (1987) defined extinction threshold as the minimum proportion of suitable habitat necessary for population persistence. Since then empirical studies that estimates this value have taken place (e.g., Lawton et al. 1994, Hanski et al. 1996). Although there is no one singular approach to estimate threshold of species loss, the variety of methods allows us to understand the species or community trajectories under a disturbance gradient. Furthermore, gaining deeper understanding in population or community trends in a landscape gradient, has a potential use for policy making and environmental management (Ochoa-Quintero 2015, Banks-Leite et al. 2014). Even that that the literature data was insufficient to perform a metanalysis, our prelaminar estimations suggest that tropical regions may have threshold value than temperate regions, which suggest that tropical species may be more impacted by habitat alteration. This was also recently concluded by Betts and coworkers (2019). This field of study of landscape ecology has a big potential given the undeniable threat due to habitat transformation and agriculture expansion for birds and biodiversity in general.

Given that one of the identified challenges in threshold studies was the fact that using community metrics, such as richness, may mask species trajectory in the landscape or habitat gradient (Melo et al 2018, de Olivieira et al. 2018, Van der Hoek et al. 2013, Lindenmayer & Luck 2005), we performed our empirical study using occupation models of singular species and selected the species that negatively responded to native vegetation loss, averaging the threshold value. When comparing these results with the threshold obtained from a community analysis (as in Muggeo 2003) we found that even that the values were similar (i.e. 45.97 % from occupation models and 52.57 % from richness), occupation models allowed us to interpret the specific species trajectories, giving us valuable information about the conservation state: that the occupancy probability tendency is more stable below the threshold value and increases steadily with native vegetation percentage cover above the threshold, suggesting that populations are already vulnerable when they reach such value.

Besides 45.9% of native vegetation cover indicates that current legislation is not enough to even secure preserved areas above the threshold, which mandates to preserve only 20% of native vegetation at Cerrado properties (New Forest Code, Brazilian Federal Law 12,651 of 2012). We conclude that it is urgent to take action aiming at reducing forest cover loss in the area, given that, according to our prediction, by 2050 almost half of the current areas with values above the threshold will be somewhere below them.

In the final chapter we identified that although there is not a difference in the species richness among the three main economic activities (agriculture, cattle ranging and tourism), there is a difference in the tourism category which recognized better the species from open and closed areas in their properties. This suggest that economic activities generated from activities directly associated to biodiversity, are important for improving local perception over biodiversity and probably to maintaining high levels of native vegetation cover within properties, which is crucial for the protection of medium to large birds and mammals, and for biodiversity in general, in a hot spot of biodiversity as the Brazilian Cerrado.

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