Universidade
Estadual de Londrina

TATIANA MOTTA TAVARES

OS EFEITOS DA PERDA DO HABITAT FLORESTAL SOBRE A RIQUEZA DE ANFÍBIOS

Londrina
2020

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Tese apresentada ao Programa de Pós-graduação em Ciências Biológicas da Universidade Estadual de Londrina - UEL, como requisito parcial para a obtenção do título de Doutor.

Orientador: Prof. Dr. Luiz dos Anjos
Co-orientador: Prof. Dr. Carlos Frederico Duarte da Rocha

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“Take a moment to think of just
flexibility, love, and trust”

Steven Universe

MOTTA-TAVARES, Tatiana. **Os efeitos da perda do habitat florestal sobre a riqueza de anfíbios.** 2020. 119 f. Tese (Doutorado em Ciências Biológicas) – Universidade Estadual de Londrina, Londrina, 2020.

RESUMO

A perda do habitat natural é apontada como uma das causas da atual crise da perda de biodiversidade. A perda do habitat natural impacta os anfíbios devido principalmente às limitações fisiológicas deste grupo. As florestas são um importante habitat para os anfíbios, por fornecerem os habitats necessários para a sobrevivência e reprodução das espécies. Existem duas formas de avaliar os efeitos da perda do habitat sobre a fauna: a relação espécie-área, que considera o remanescente como unidade natural, e o montante de habitat, que considera o montante proporcional do habitat disponível na paisagem. O objetivo geral desta tese foi avaliar os efeitos da perda do habitat florestal sobre a riqueza de anfíbios, se utilizando de ambas as abordagens. O objetivo do primeiro capítulo foi de avaliar a relação espécie-área de anfíbios em remanescentes florestais. Para isso, foi realizada uma meta-análise em nível global, para verificar se houve influência do tamanho do remanescente florestal sobre a riqueza dos anfíbios residentes, além da influência de variáveis ambientais e de amostragem sobre esta relação. No segundo capítulo, foi avaliada a relação entre o montante de habitat florestal sobre a riqueza de anfíbios de várzeas, em uma paisagem fragmentada do norte do Paraná. Neste, avaliamos o efeito de quatro variáveis da paisagem (entre eles, o montante de floresta) sobre a riqueza de anfíbios de várzeas nos municípios de Londrina, Ibiporã e Jataizinho. O primeiro capítulo confirmou que existe uma relação positiva entre o tamanho do remanescente florestal e a riqueza de anfíbios. Esta relação foi parcialmente afetada pelo tipo de matriz antrópica que cerca o habitat florestal. O segundo capítulo detectou uma forte influência da paisagem sobre a riqueza de anfíbios nas zonas de várzea, porém não foi possível discriminar se isso foi resultado de relação causal direta com o montante de habitat florestal ou efeito do domínio de matriz agrícola na região amostrada. Como conclusão geral desta tese, é evidente o efeito negativo da perda de habitat sobre a riqueza dos anfíbios, mas que é importante considerar a matriz dominante que compõem a paisagem. Fica claro que as atuais leis ambientais brasileiras não são adequadas para a proteção dos anfíbios, uma vez que não consideram a natureza das matrizes que cercam as florestas e os arredores dos sítios reprodutivos.

Palavras-chave: Floresta Atlântica. Floresta estacional. Semi-decidual. Matriz agrícola. Montante de hábitat. Relação espécie-área.

MOTTA-TAVARES, Tatiana. **The effects of forest habitat loss on the amphibian's richness.** 2020. 119 p. Thesis (Doctorate degree in Biological Sciences) – Universidade Estadual de Londrina, Londrina, 2020.

ABSTRACT

Habitat loss is considered to one of the main causes of the current crisis in the loss of biodiversity. The loss of natural habitat impacts amphibians mainly due to the group's physiological limitations of this group. Forests are important for amphibians, as they provide the necessary habitats for the species' survival and reproduction. There are two ways of evaluating the effects of habitat loss on fauna: the species-area relationship, which considers the remnant as a natural unit, and the habitat amount, which considers the proportional amount of the habitat available in the landscape. The general objective of this thesis was to evaluate the effects of forest habitat loss on the amphibian richness using both approaches. The objective of the first chapter was to evaluate the species-area relationship of amphibians in forest remnants. To do this, a meta-analysis was developed at the global level, to verify if there was influence of the size of the forest remnant on amphibian species richness, and also the influence of environmental and sampling variables on this relationship. In the second chapter, the relationship between the amount of forest habitat and the richness of floodplain amphibians was evaluated in a fragmented landscape of North Paraná. Here, we evaluated the effect of four landscape variables (among them, the amount of forest) on the richness of floodplain amphibians in the municipalities of Londrina, Ibiporã and Jataizinho. The first chapter confirmed that there is a positive relationship between the size of the forest remnant and amphibian species richness. This relationship is partly influenced by the type of anthropogenic matrix that surrounded the forest. The second chapter detected a strong influence of the landscape on the richness of amphibians in the floodplain areas, however it was not possible to discriminate whether this is an directly influence of the amount of forest habitat or the dominance of the agricultural matrix in the region. As a general conclusion of this thesis, its evident the negative effect of forest habitat loss on amphibian species' richness, but that it is important to consider the dominant matrix in the landscape. Its also clear that the currents Brazilian environmental laws are not be suitable for the protection of amphibians, since it does not consider the matrix that surrounds the forests and the reproductive sites.

Key-words: Agriculture matrix. Atlantic rainforest. Habitat amount. Semideciduous forest. Species-area relationship.

LISTA DE ILUSTRAÇÕES

SUMÁRIO

1. INTRODUÇÃO GERAL

A perda do habitat natural é apontada como uma das principais causas da atual crise da biodiversidade (CARDINALE et al., 2012; HADDAD et al., 2015). As florestas estão cada vez menores, mais fragmentadas e isoladas, o que leva à redução e isolamento do habitat necessário para a sobrevivência das espécies (HADDAD et al., 2015). As projeções indicam uma constante queda na riqueza de espécies e das populações se medidas protetivas específicas não forem tomadas (NEWBOLD et al., 2015; POWERS; JETZ, 2019). Os anfíbios apresentam mais de 31,5% das espécies com risco de extinção (IUCN, 2019), sendo este o grupo de vertebrados mais impactados pela perda do habitat natural (CUSHMAN, 2006; PFEIFER et al., 2017; POWERS; JETZ, 2019; STUART et al., 2004).

PORQUE OS ANFÍBIOS SOFREM COM A PERDA DO HABITAT?

A perda do habitat impacta os anfíbios devido principalmente às limitações fisiológicas do grupo (CUSHMAN, 2006; STUART et al., 2004). Anfíbios possuem respiração cutânea e ovos gelatinosos, o que os deixa dependentes de ambientes úmidos (VITT; CALDWELL, 2013; WATLING; BRAGA, 2015). A maioria das espécies também possui estágio larval, o que deixa tais espécies dependentes de corpos d’água e ainda mais vulneráveis à fragmentação do habitat (ALMEIDA-GOMES; ROCHA, 2015; BECKER et al., 2007, 2010a; VITT; CALDWELL, 2013). As florestas fornecem uma grande variedade de corpos d’água em seu interior (ex. poças, riachos, bromélias-tanque) e a umidade ambiental necessária, como também fornecem poleiros de vocalização (e.g. galhos, arbustos), alimento (em maioria insetos) e abrigos contra predadores (e.g. troncos caídos, cavidades nas árvores, cobertura de vegetação e serapilheira). Quando comparados com a matriz que os cerca, os remanescentes florestais possuem maior riqueza de espécies e de modos reprodutivos, incluindo modos reprodutivos mais especializados (ALMEIDA-GOMES; ROCHA, 2014, 2015; THOMPSON; NOWAKOWSKI; DONNELLY, 2016). A redução do habitat florestal afeta de forma negativa a abundância e a riqueza dos anfíbios, mesmo as espécies sem estágio larval (desenvolvimento direto, ex. Grupo Terrana) ou cujo o habitat reprodutivo se situa na borda ou fora da floresta (ALMEIDA-GOMES et al., 2016; ALMEIDA-GOMES; ROCHA, 2015; BECKER et al., 2007, 2010b; CUSHMAN, 2006; GALLANT et al., 2007; QUESNELLE; LINDSAY; FAHRIG, 2015; ZIMMERMAN; BIERREGAARD, 1986).

A perda de números populacionais ou de espécies como consequência da fragmentação do habitat natural, somado com a relativa baixa mobilidade dos anfíbios pode resultar no isolamento das populações, dificultando as migrações e sujeitando as populações locais à estocasticidade populacional e ambiental, além da deriva genética (CAYUELA et al., 2020; CUSHMAN, 2006; NOWAKOWSKI et al., 2017; STUART et al., 2004). O indivíduo que tenta atravessar a matriz antrópica fica sujeito à predação e à dessecação no ambiente da matriz, elevando o risco de mortalidade (BECKER et al., 2010a, 2010b; WATLING; BRAGA, 2015). A matriz antrópica pode afetar a composição das espécies encontradas dentro dos remanescentes (DIXO; METZGER, 2010; PRUGH et al., 2008; RICKETTS, 2001). Isso possivelmente se dá devido aos diferentes níveis de tolerância ao habitat matriz entre espécies (DIXO; METZGER, 2010; NOWAKOWSKI et al., 2017; PRUGH et al., 2008; WATLING; BRAGA, 2015). As espécies consideradas “generalistas” tendem a serem maiores, mais móveis, mais resistentes à dessecação e menos exigentes com o habitat reprodutivo, o que permite estas se dispersarem no ambiente da matriz (CAYUELA et al., 2020; NOWAKOWSKI et al., 2017; WATLING; BRAGA, 2015). As espécies consideradas “especialistas” tendem à maior grau de especialização no seu modo reprodutivo (ex. bromelígenas *sensu* PEIXOTO, 1995; espécies que possuem todo o ciclo de vida associado às bromélias). Caso consigam atravessar o habitat matriz, a probabilidade das destas espécies especialistas de se estabelecer no novo remanescente pode ser baixa, principalmente se este não fornecer o habitat necessário para a reprodução da espécie (BECKER et al., 2010a, 2010b).

COMO PODEMOS AVALIAR OS EFEITOS DA PERDA DO HABITAT SOBRE A RIQUEZA DE ESPÉCIES?

Para avaliarmos o efeito da perda do habitat sobre a riqueza das espécies de anfíbios, primeiro precisamos definir como quantificar esse habitat e testar o efeito da redução do mesmo. Atualmente, os pesquisadores se utilizam de duas abordagens: O tamanho do remanescente, a partir da teoria da biogeografia de ilhas (MACARTHUR; WILSON, 1967) e o montante de habitat, derivado da hipótese do montante de habitat (FAHRIG, 2013). Também é necessário avaliar a escala “ótima” da paisagem, uma vez que o efeito da paisagem pode ser diferente de acordo com tamanho da mesma (HOLLAND; BERT; FAHRIG, 2004).

Darlington (1957) em seu compilado de trabalhos “Zoogeografia” compilou pela primeira vez o conceito da relação espécie-área, aonde a riqueza de espécies em um dado local está diretamente relacionada com a área do habitat disponível. MacArthur e Wilson (1967) incorporam a relação espécie-área à Teoria da Biogeografia de Ilhas, na qual a riqueza de espécies encontradas em ilhas oceânicas dependeria do balanço da relação entre o tamanho da ilha e a distância da mesma em relação ao continente. Diamond (1975) adaptou a Teoria da Biogeografia de Ilhas para a conservação de florestas, aonde as “ilhas” representariam os remanescentes florestais enquanto o “oceano” seria a matriz de origem antrópica que circunda o que sobrou da floresta. O autor sugeriu um “Guia para as reservas naturais”, no qual o tamanho do remanescente seria um fator chave para a manutenção das espécies em remanescentes florestais. Quanto maior a área do remanescente, maior seria a capacidade do mesmo em reter as populações das espécies naturais.

Fahrig (2013) desafiou a ideia do remanescente como uma unidade natural para respostas biológicas, propondo a hipótese do montante de habitat. A teoria da biogeografia de ilhas supõe que os remanescentes de habitat seriam unidades naturais, aonde os limites do habitat contêm e limitam as populações dentro delas. Entretanto, a matriz que circunda os remanescentes possui diferentes graus de tolerância entre espécies (DIXO; METZGER, 2010; PREVEDELLO; VIEIRA, 2010; PRUGH et al., 2008; RICKETTS, 2001). Os indivíduos que habitam um dado local não estão limitados às “ilhas” de habitat em um “oceano” inóspito. O tamanho do remanescente e o isolamento do mesmo seriam consequências diretas do montante do habitat disponível. Uma resposta biológica (ex. variedade genética, abundância, riqueza de espécies) seria avaliada pelo montante de habitat disponível na paisagem ao redor em substituição às análises com a área e o isolamento do remanescente. Segundo a autora, a hipótese do montante de habitat é uma simplificação, através da transformação de duas variáveis da paisagem em uma. Isso permitiria que as variáveis biológicas sejam avaliadas considerando o local amostrado, ao invés de considerar remanescentes inteiros.

O efeito da perda do habitat pode ser detectado em diferentes escalas, de acordo com o organismo e a variável biológica testadas (MARTIN, 2018; MIGUET et al., 2016). O “efeito de escala” propõem que a força da relação de uma variável biológica com uma variável da paisagem (ex. montante de floresta, densidade de rodovias) está diretamente relacionada com a escala da paisagem considerada (HOLLAND; BERT; FAHRIG, 2004; JACKSON; FAHRIG, 2012, 2015; MARTIN, 2018; MARTIN; FAHRIG, 2012; MIGUET et al., 2016). A escala na qual a relação da variável resposta com a variável ambiental é a mais forte (ex. o modelo linear que apresentou o maior valor de R^2, ou o modelo mais ajustado a partir do critério de Akaike) é chamada de “escala de efeito” (JACKSON; FAHRIG, 2012; MARTIN; FAHRIG, 2012).

OS ANFÍBIOS DO NORTE DO PARANÁ

As comunidades dos anfíbios da região norte do Paraná só começaram a serem estudadas no final da década de 1990 (BERNARDE; ANJOS, 1999; MACHADO et al., 1999; MACHADO; BERNARDE, 2002). A região possui cerca de 40 espécies de anfíbios detectadas até o momento, oriundas de 11 famílias, todas da Ordem Anura (FIGUEIREDO et al., 2019; ODA et al., 2016; SANTOS-PEREIRA; POMBAL; ROCHA, 2018). Mais recente ainda são os estudos que procuraram avaliar os efeitos da perda do habitat e do ambiente agrícola sobre as comunidades de anfíbios da região (FIGUEIREDO et al., 2019; ODA et al., 2016; presente tese).

A região norte do Paraná possui predomínio da Mata Atlântica, na forma de Floresta Estacional Semidecidual (IBGE, 2012). Esta floresta é caracterizada pela relativa baixa incidência de chuvas, em comparação com a Floresta Atlântica Ombrófila Densa e a Floresta Atlântica Ombrófila Mista, encontradas em outras regiões do estado (BERGAMIN; MÜLLER; MELLO, 2012; OLIVEIRA-FILHO; FONTES, 2000). As florestas ombrófilas do estado do Paraná apresentam pluviosidade bem distribuída ao longo do ano, com uma incidência entre 2000 e 3000 mm (BERGAMIN; MÜLLER; MELLO, 2012; OLIVEIRA-FILHO; FONTES, 2000). Já a Floresta Atlântica Estacional Semidecidual apresenta um período de estiagem que pode durar entre 40 a 160 dias do ano, com pluviosidade entre 1500 e 2000 mm durante a estação chuvosa, entre outubro e fevereiro (BERGAMIN; MÜLLER; MELLO, 2012; OLIVEIRA-FILHO; FONTES, 2000). A riqueza e abundância das espécies de anfíbios apresenta forte correlação com a pluviosidade local (ex. TOFT, 1980; VASCONCELLOS; COLLI, 2009; VASCONCELOS et al., 2011). Por consequência, florestas mais secas apresentariam uma menor diversidade de espécies e menores densidades de indivíduos.

A Floresta Estacional Semidecidual foi a que mais sofreu com a perda do habitat florestal no estado. Restam cerca de 7% das florestas originais, aonde a maioria dos remanescentes não são maiores do que 10 ha (SOS MATA ATLÂNTICA; IBGE, 2014). O maior remanescente da região norte do Paraná é o Parque Estadual Mata dos Godoy, com 675 ha de floresta contínua. O desmatamento da região começou em menor escala por bandeirantes e posseiros no século XVII, mas este se intensificou nos séculos XIX e XX em decorrência da exploração madeireira e do plantio do café na região (MARTINS, 1995; SOARES; MEDRI, 2002). Hoje, a matriz antrópica no norte do Paraná predominam monoculturas de soja e milho (IBGE, 2012).

Os anfíbios do norte do Paraná vivem em um ambiente naturalmente mais seco, fragmentado e isolado. A floresta é um ambiente essencial para os anfíbios locais, uma vez que ela permite o acesso das espécies à ambientes reprodutivos (FIGUEIREDO et al., 2019; ODA et al., 2016), como as várzeas. As várzeas possuem um regime de inundação temporário associada à vazão dos rios, aonde os sedimentos orgânicos se acumulam durante a estação chuvosa. Elas são pontos reprodutivos essenciais para os anfíbios, permitindo a atividade reprodutiva mesmo quando inseridas em uma paisagem perturbada (HOLGERSON et al., 2019; TOCKNER; STANFORD, 2002).

OBJETIVOS

O objetivo geral da presente tese foi avaliar os efeitos da perda do habitat florestal sobre a riqueza de anfíbios. Para isso, a tese foi composta por dois capítulos, cada um com uma abordagem distinta para se avaliar o efeito da perda do habitat florestal sobre os anfíbios.

O primeiro capítulo avaliou a relação espécies-área de anfíbios em remanescentes florestais (DIAMOND, 1975; MACARTHUR; WILSON, 1967). Para isso, desenvolvemos uma meta-análise global, aonde testamos a influência do tamanho do remanescente florestal sobre a riqueza dos anfíbios residentes. Também avaliamos se fatores locais e intrínsecos aos estudos podem afetar a relação espécie-área de anfíbios encontrados nos remanescentes.

O segundo capítulo avaliou as consequências da perda do habitat através do montante de habitat florestal (FAHRIG, 2013). Nós avaliamos a relação entre o montante de floresta natural com a riqueza de anfíbios de várzeas em uma paisagem rural do norte do Paraná. Para atingir este objetivo, também avaliamos a escala de efeito dessa relação. Também testamos o efeito do montante de cobertura do habitat agrícola, densidade de borda de floresta e a heterogeneidade de coberturas de solo nos arredores das várzeas.

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2. Capítulo 1

O tipo de matriz influencia na relação entre o tamanho do remanescente com a riqueza de anfíbios: Uma meta-análise global.

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The influence of matrix type in the relationship between patch size and Amphibia richness: a global meta- analysis

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Abstract

The species-area relationship (SAR) is one of the main topics debated in ecology. Amphibians are highly sensitive to habitat loss and fragmentation, due to their traits and reproductive requirements. In the present study, we developed a meta-analysis to address a mean effect size to SAR regarding forest amphibians in man-made fragmented landscapes, and tested the influence of seven in-study and landscape variables that may affect this relationship globally. We conducted a search using the Preferred Reporting Items for Systematic Reviews and Meta-Analysis (PRISMA) protocol and calculated an effect size for each one of the 28 studies elected by our criteria. There is a positive relationship between amphibia richness and patch area, with a mean effect size of 0.68 and high heterogeneity among studies. The meta-regressions showed that only the matrix type surrounding the patches affected the variance between studies, explaining 46%. Clear cut was the matrix with the highest effect size, followed by silviculture, but both had a low representation of studies. Agriculture, grazing and urban matrices had a significant influence on patch area and the species richness relationship, but with distinct mean effects sizes and variances between-studies. The matrix type around the remnants may substantially affect results and interpretation of the SAR, possibly due to the varying capacity of amphibia species to cross and inhabit each matrix type. If possible, SAR should be compared within a context of multiple landscapes, and not only with a patch-focused and local approach.

Key words: Fragmentation, habitat loss, heterogeneity, landscape ecology, meta-regression, species-area relationship.

1. Introduction

The species-area relationship (SAR) was described in the first years of the 20th century (Arrhenius, 1921) and since then has been one of the main topics in ecology (Preston, 1948; Darlington, 1957). The SAR suggests that communities with reduced area will have reduced species richness. This relationship has been screened and tested extensively in the field of conservation biology with forest patches surrounded by a human-made matrix (MacArthur & Wilson, 1967; Diamond, 1975; Wilson & Willis, 1975; Zimmerman & Bierregaard, 1986). Despite the non-natural surroundings, these patches are not completely isolated from each other as in an impenetrable matrix (Laurance, 2008). The matrix surrounding patches has a degree of permeability to certain species, and can therefore directly affect species composition inside patches (Ricketts, 2001; Prugh, Hodges, Sinclair, & Brashares, 2008; Dixo & Metzger, 2010; Koh & Ghazoul, 2010; Lindsay et al., 2013; Thompson, Nowakowski, & Donnelly, 2016). Many other aspects interfere in the species composition inside the forest remnants: geographical location (e.g., peninsula effect), time since the fragmentation event, local disturbance history (e.g., fire patterns), environmental synergism, remnant shape, habitat heterogeneity inside the patches, species traits, presence of roads and even climate (Boecklen & Gotelli, 1984; Swetnam, Allen, & Betancourt, 1999; Lindell et al., 2007; Laurance, 2010; Rytwinski & Fahrig, 2012; Battisti, 2014; Keinath et al., 2017).

Watling and Donelly (2006) quantified several aspects of the relationships between species richness and area together with isolation in a meta-analysis regarding multi-taxa populations inside patches of natural habitat (not restricted to forest remnants). In their review, 118 studies tested the SAR, where 91% of studies showed a significant and positive influence of area in species richness, 1% had a significant and negative influence and 8% had no significant results. They found that area explained more variance in species richness than isolation, and that it was more strongly correlated with species differences between patches. In contrast, another meta-analysis by Prugh et al., (2008) reported that area and isolation were, surprisingly, poor predictors of species occupancy when analyzed individually or combined (global meta-analysis, n = 55 855 occupancy records). Area predictors of species occupation were strongly influenced by matrix type, and could vary according to the species' taxonomic group and traits (Prugh et al., 2008).

Amphibians are the most threatened vertebrate species worldwide, with 31.5% of species categorized as vulnerable, endangered or critically endangered (IUCN, 2019). In comparison with others vertebrate taxa, amphibians are impacted more negatively by habitat loss and fragmentation due to their relatively low movement capacity, physiological traits and habitat requirements (Stuart et al., 2004; Cushman, 2006). The majority of amphibians species have a larval stage, which makes them dependent on water bodies (e.g. ponds, streams, tank bromeliads) and highly sensitive to habitat fragmentation (Becker, Fonseca, Haddad, Batista, & Prado, 2007). Amphibian species with direct development (for example those from the so called group Terrarana, with 989 species) are not directly dependent on water bodies. They do however need a minimum environmental humidity due to their sensitive skin and gelatinous eggs, a requirement that a forested environment fulfills (Vitt & Caldwell, 2013; Watling & Braga, 2015). Terrestrially developing species are much more sensitive to habitat loss than habitat disruption (Becker et al., 2007). A vote-counting study conducted by Almeida-Gomes et al., (2016b) found that 11 out of 14 studies (considering only the studies which tested the SAR for amphibians) showed a significant positive effect of the forest patch area on species richness, with just a few studies discovering a null effect (e.g., Hillers et al., 2008) and none reporting a negative effect. For amphibians inhabiting wetlands, a meta-analysis performed by Quesnelle et al. (2015) demonstrated that the amount of surrounding forest was a more important predictor for species abundance than the amount of wetland itself, or the matrix composition (d.f. = 204). Habitat loss and fragmentation lead to reduce amphibian reproduction, due to the reduction in the availability of reproductive habitats (Zimmerman & Bierregaard, 1986; Becker et al., 2007; Almeida-Gomes & Rocha, 2015; Almeida-Gomes et al., 2016b).

Habitat loss is one of the greatest threats to amphibian species from forested habitats (Stuart et al., 2004; Cushman, 2006; Van Sluys et al., 2009; Pfeifer et al., 2017). With the constant reduction of forest habitats and the isolation of populations into small remnants (Haddad et al., 2015), we need a deeper understanding of the mechanisms behind the SAR in forest-sensitive species. In the present study, we developed a meta-analysis to (1) determine the mean effect size to SAR regarding amphibians from forested areas within man-made fragmented landscapes worldwide, and (2) test in-study and landscapes variables that may affect this relationship globally.

2. Material and methods

2.1 Literature search and eligibility criteria

We recorded the number of studies identified and the number of studies included or excluded in each step according to the Preferred Reporting Items for Systematic Reviews and Meta-Analysis Statement (PRISMA Statement; Moher et al., 2009). We searched the databases ISI Web of Science and Scopus until November 2018, with a combination of keywords (*amphibi** OR *anur**) AND (*fragmentation* OR "*patch size*"), and no year restriction ("identification" step). Despite having used English keywords, we did not exclude studies from other languages if the statistical reports were intelligible. We also searched in the cited literature section of the studies captured in our search to refine it and find relevant papers not captured in the original database search. Each study had its title and abstract screened ("screening" step). The study proceeded into reading of the full text when we understood that it was related to habitat loss and/or fragmentation and amphibian communities. Studies were not included in the analyses if they were not on amphibians or habitat loss.

Next, we applied our eligibility criteria ("eligibility" step, as follows). We excluded the following types of studies: data from naturally fragmented areas (e.g., natural islands formed as a consequence of Pleistocene's climate changes), data from non-forested habitats (e.g., South America grassy plains "Pampas"), data from reforested patches or patches with selective cutting, systematic reviews, meta-analysis, computer models and simulations. Despite the exclusion, we screened their "cited literature" section as well. If two or more studies had a common data set (e.g., Watling & Donnelly, 2008; Watling et al., 2009), we only considered the study with the highest number of patches, or the one with the necessary data available to extract the correlation coefficient. As we focused our search on the relationship between patch size and amphibian species richness, we did not use data from studies that measured occupancy, means and variance between groups (e.g., richness in forested habitat vs. non-forested habitats). We included articles which studied the amphibia communities inside forest remnants, as long as they fulfilled the previous eligibility criteria and the authors made the data available as a result, either as supplementary material or by direct contact with the corresponding author. We used this available information to calculate the correlation coefficient with correlation analysis (explanation below).

2.2 Data extraction and meta-analysis

We calculated an effect size for each study eligible for meta-analysis. As our question is based on the relationship between two continuous variables (patch size and amphibian richness), we calculated the effect size based on correlation coefficients from correlation and regression analysis (Borenstein, Hedges, Higgins, & Rothstein, 2009; Rosenberg, Rothstein, & Gurevitch, 2013). To calculate the effect size and its variance between-studies, we extracted the correlation coefficient and the degree of freedom from the respective analyses. We extracted the Pearson's *r* or the Spearman's *rho* values by direct citation in the text, tables or figures, or by removing the square root from the R^2 value. We did not use slope coefficients, AIC values or partial R^2 values from multiple regression in our meta-analysis. Instead, we accessed the raw available data and calculated the Pearson's *r* or the Spearman's *rho* (description below). Then, we converted all the *r* and *rho* values to Fischer's *z*-scale, in order to normalize the correlation coefficient distribution and have an independent variance between studies (Borenstein et al., 2009; Rosenberg et al., 2013). We calculated the effect size and reported the present meta-analysis in Fischer's *z*-scale.

Some of the selected papers did not test the relationship between patch size and amphibian species richness, or did a correlation / regression analysis but reported neither the *r* / *rho* nor the R^2 value. If the raw data was available, we included these studies in the meta-analysis by calculating the Pearson's *r* or Spearman's *rho*. If the raw data was not available, we contacted the corresponding author asking for the raw data. We first tested the data for normality, homoscedasticity and linearity, and if they fulfilled all those prerequisites, performed a Pearson correlation analysis between a log-transformed patch size and amphibia richness. If these prerequisites were not fulfilled, we performed a Spearman ranking correlation (non-parametric) instead. We did not consider studies with less than five patches, because it was not possible to estimate sampling variance with a correlation for four or less samples (Viechtbauer, 2010).

We applied a random-effect (RE) model to calculate the general effect size among all the studies and to explore possible sources of heterogeneity between studies. We opted to proceed with a random-effect model instead of fixed-effect model because: 1 - The fixed-effect model assumes that the true effect size is the same between all studies and the sampling error is zero (Borenstein et al., 2009; Gurevitch, Koricheva, Nakagawa, & Stewart, 2018), which we cannot assume in the present meta-analysis; 2 - heterogeneity is expected between ecological studies (Senior et al., 2016; Gurevitch et al., 2018); and 3 - the studies were not homogenous between them (Q test for heterogeneity: $Q = 93.22$, d.f. = 27, $p < 0.0001$). Such heterogeneity was inspected with meta-regressions (Borenstein et al., 2009; Rosenberg, 2013). We tested the influence of moderators in the variance between studies with regression models (Borenstein et al., 2009; Rosenberg, 2013). We applied individual meta-regression models for each moderator and a global RE model, which includes all the moderators. If a categorical moderator was significant, we also applied individual RE models for each category as a method to track and understand part of the between-studies variance (Borenstein et al., 2009; Rosenberg, 2013; Gurevitch et al., 2018).

We use Cochran's Q statistics (Cochran, 1954; Borenstein et al., 2009) to evaluate and test the variance among k studies. The Q value represents the observed variation to the within-study error. The ^{2} and values are the between-studies variance and its standard definition, respectively. The I^2 is the proportion of the observed variance between studies, accessed by the formula $[(Q - \text{d.f.}) / Q] * 100$. The I^2 ranges from 0 to 100%, measuring the degree of inconsistency across studies, where higher levels of heterogeneity are represented by a high I^2 value (Higgins & Thompson, 2002). In meta-regression models, the total amount of heterogeneity (Q_{T}, from Q_{Total}) can be partitioned into heterogeneity from the structural model Q_{M} (from Q_{Model}) and the unexplained heterogeneity Q_{E} (from Q_{Erro}), where $Q_{\mathrm{T}} = Q_{\mathrm{M}} + Q_{\mathrm{E}}$. In simple models (without moderators), there is no explanatory variable, so $Q_{\mathrm{M}} = 0$ and $Q_{\mathrm{T}} = Q_{\mathrm{E}}$. To test the presence of heterogeneity between studies, we used Q tests for heterogeneity, which consist in a chi-distribution with k-1 degrees of freedom and tested the hypothesis of homogeneity between studies (Cochran, 1954; Borenstein et al., 2009). A significant Q test means heterogeneity in the variance between-studies. In the meta-regression model summaries, the R^2 value represents the amount of I^2 explained by the model (Borenstein et al., 2009).

We tested the influence of seven moderators in the relationship between patch size and amphibian species richness: Number of patches considered in correlations, patch size range, survey method, latitude, global biome (*sensu* Lomolino et al., 2016), the total amphibian species richness in the respective landscape and dominant surrounding matrix (Table 1). To evaluate the possible influence of the survey method, we categorized the methods based on the nature of the search for amphibians. We defined any kind of active detection for amphibians by the researchers, such as visual and auditory encounter surveys, transects and plots, as "active" (Heyer, Donnelly, McDiarmid, Hayek, & Foster, 1994); any kind of detection based in fixed points, such as pitfall traps, net trapping and audio recorders, as "passive"; and studies that used both types of methods without distinguished results (e.g. visual encounter survey combined with pitfall traps) as "active-and-passive". We tested the possible influence of the landscape distance from Equator line with the latitude coordinates given by the studies. If the coordinate was not provided in the text, we used the median latitudinal coordinate from the described municipality or region. We converted the given coordinates to decimal in WGS84 datum, and converted negative values to positive. We assumed the dominant surrounding matrix as the predominant land use as defined by the authors. If the study did not specify the land use in the surroundings, we consulted other contemporary studies that were performed in the same locality and /or local Governments official websites. We categorized the land use similarly to Thompson et al., (2016), as described in Table 1. We could not appoint the time of fragmentation as a moderator because few eligible studies could address the precise moment of fragmentation in their respective landscape.

Table 1 – Variables moderators insert in the present meta-analysis

Moderator variable	Variable type	Variable description
Number of patches	Continuous	Total number of patches considered in the regression analysis.
Patch area range	Continuous	The difference between the largest and the smallest patch.
Sample method	Categorical	Survey method used to detect species: *Active* - Visual and/or auditive encounter surveys, transects, plots, any survey based in active search for individuals. *Passive* - Pitfall traps, net trapping, audio recorders, any survey based in a fixed point. *Active-and-passive* - The use of both methods (e.g., visual encounter surveys and pitfall).
Latitude	Continuous	Distance from the Equator (decimal, datum WGS84).
Biome	Categorical	Biome zone according to Lomolino et al., (2016).
Total amphibian richness	Continuous	Total number of species recorded in the landscape.
Matrix type	Categorical	Dominant matrix surrounding the patches, according to the study's authors' description: *Agriculture* - Crop plantations; *Clear cut* - Clear cut followed by abandonment; *Flooding* – Patches as artificial islands, as a direct consequence of artificial dams; *Grazing* - Pasture land for animal farm; *Silviculture* - Cacao, shadow coffee, *Eucalyptus, Pinus* and/or rubber plantations; *Urban* - Land occupation for human residential use.

We tested publication bias with Rosenberg, Rosenthal and Orwin fail-safe N tests (Rosenthal, 1979; Orwin, 1983; Rosenberg, 2005) and regression tests for funnel plot asymmetry (Egger, Smith, Schneider, & Minder, 1997) for each RE model. We considered a test as significant when $p < 0.05$. All the analyses were made in R version 3.5.2 (R Core Team, 2019). The correlation analyses were made with the *cor* function (*stats* package; R Core Team, 2018) and the meta-analysis were made with functions from the *metafor* package (Viechtbauer, 2010).

3. Results

We found a total of 1 333 matches (excluding duplicates) with the selected key-words, but only 441 studies proceeded after the screening step (Figure 1). After the eligibility and inclusion steps, 28 studies were selected for the meta-analysis (Table 2). We collected the correlation coefficient directly from 11 studies, and calculated it independently from the others 17. These 28 studies were carried out in 13 different countries: South (8) and North (5) America, Asia (5), Africa (4), Central America (2), Australia (2) and Europe (2).

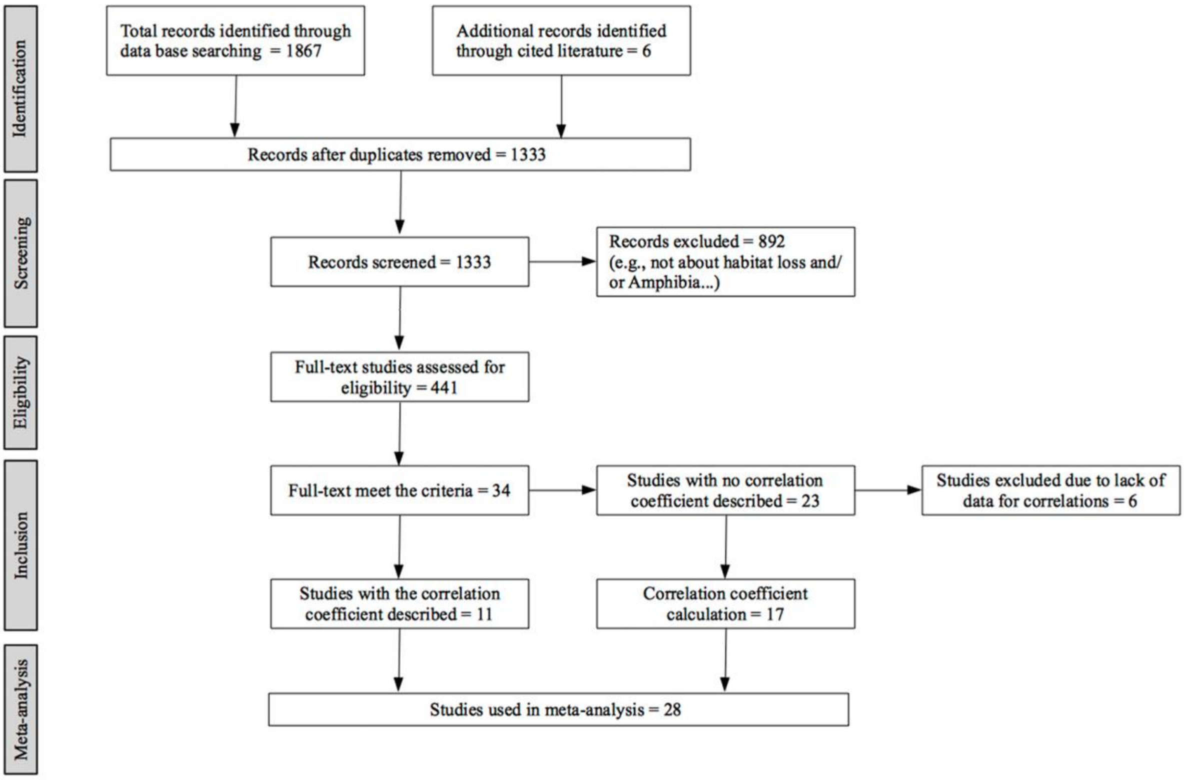

Figure 1 – Flow diagram indicating the number of studies considered and excluded at each step, as suggested by the Preferred Reporting Items for Systematic Reviews and Meta-Analysis Statement (PRISMA Statement). Each gray box indicates the respective step.

Table 2 – Summary of studies used in meta-analysis. Studies with a "*" had their correlation coefficients (*r*, *rho* or $\sqrt{R^2}$) directly extracted.

Study	**Number of patches**	**Patch range (ha)**	**Matrix type**	***Total amphibia richness***	**Latitude**	**Biome**	**Method**	**Country**
Li et al., 2018*	16	170 (8 to 178)	Urban	6	31.2222	Temperate subtropical	active	China
Wang et al., 2018	23	1288.61 (0.59 to 1289.2)	Flooding	9	29.6055	Temperate subtropical	active	China
da Silva & Rossa-Feres, 2017*	18	3308 (59 to 3367)	Agriculture	36	19.5666	Tropical rainforest	active-and-passive	Brazil
Dayananda et al., 2017	18	13871.16 (1.71 to 13872)	Silviculture	25	21.9166	Tropical rainforest	active	China
Ferrante et al., 2017	16	208.74 (5.14 to 213.88)	Agriculture	21	-21.4256	Tropical rainforest	active	Brazil
Ferreira et al., 2017	11	168.4 (1.8 to 170.2)	Urban	17	-20.4427	Tropical savanna	active-and-passive	Brazil
Almeida-Gomes et al., 2016b	21	617.15 (1.87 to 619.02)	Grazing	50	-22.4	Tropical rainforest	active	Brazil
MacGregor-Fors et al., 2016	6	92.3 (3.2 to 95.5)	Urban	10	19.5312	Tropical rainforest	active	Mexico
Russildi et al., 2016	9	89 (3 to 92)	Agriculture	22	16.35	Temperate subtropical	active	Mexico
Lima et al., 2015*	10	2137 (3 to 2140)	Flooding	32	-3.8316	Tropical rainforest	active-and-passive	Brazil

Riemann et al., 2015	9	14.5 (2 to 16.5)	Agriculture	40	-21.2166	Tropical rainforest	active	Madagascar
Lion et al., 2014*	23	25.7 (1.7 to 27.4)	Agriculture	14	-6.1089	Tropical rainforest	passive	Brazil
Cabrera-Guzmán & Reynoso, 2012	6	16 (1.4 to 17.4)	Grazing	10	18.6	Temperate subtropical	active	Mexico
Hilje & Mitchell Aide, 2012	15	10195.2 (2.4 to 10197.6)	Agriculture	29	10.4333	Tropical rainforest	active	Costa Rica
Bickford et al., 2010	12	924 (11 to 935)	Urban	20	1.3516	Tropical rainforest	active	Singapore
Vignoli et al., 2009*	62	1599 (1 to 1600)	Urban	15	41.8919	Temperate subtropical	active	Italy
Hillers et al., 2008*	16	46.5 (1.5 to 48)	Agriculture	19	5.75	Tropical rainforest	active	Ivory Coast
Watling & Donnelly, 2008	24	7.82 (0.64 to 8.46)	Grazing	13	-14.75	Tropical savanna	passive	Bolivia
Rakotondravony, 2007*	11	4103 (47 to 4150)	Grazing	15	-13.2011	Tropical rainforest	active-and-passive	Madagascar
Bell & Donnelly, 2006*	9	5.45 (1.43 to 6.88)	Grazing	41	10.4333	Tropical rainforest	active	Costa Rica
Drinnan, 2005*	19	171.36 (0.24 to 171.6)	Urban	8	-34.05	Temperate subtropical	active	Australia
Alcala et al., 2004	10	117 (5 to 122)	Agriculture	11	10	Tropical rainforest	active	Philippines
Pineda & Halffter, 2004	10	111 (11 to 122)	Silviculture	21	19.5666	Tropical rainforest	active	Mexico

Silva et al., 2003	11	39.7 (3.1 to 42.8)	Agriculture	9	46.4166	Tundra / Boreal / Alpine	active-and-passive	Canada
How & Dell, 2000	34	336.90 (1.1 to 338)	Urban	15	-31.9522	Temperate subtropical	passive	Australia
Vallan, 2000*	7	135.8 (0.16 to 136)	Grazing	25	-18.161	Tropical rainforest	active	Madagascar
Dickman, 1987	8	7.17 (0.23 to 7.40)	Urban	5	51.752	Temperate subtropical	active	England
Zimmerman & Bierregaard, 1986*	7	499 (1 to 500)	Clear cut	32	-3.1071	Tropical rainforest	active	Brazil

3.1 Effect size and publication bias

There was a highly significant positive overall influence of patch size on amphibian species richness, with a mean effect size of 0.68 (Random-effect model; $k = 28$; 95% CI: 0.48 – 0.88; $p < 0.0001$). In fact, 26 (92.8%) of the effect sizes were positive and only two (7.2%) were negative (Figure 2). However, there was significantly high heterogeneity among studies, with an I^2 value over 70% (RE model, Table 3). Despite the majority of landscapes coming from the American continent (15) and from tropical rainforests (17), we did not detect publication bias among studies (Table A for the N-fail safe tests), with a symmetrical distribution of the effect sizes around the mean effect size (Test for funnel plot asymmetry: $z = 1.23$, $p = 0.22$; Figure B and Table B).

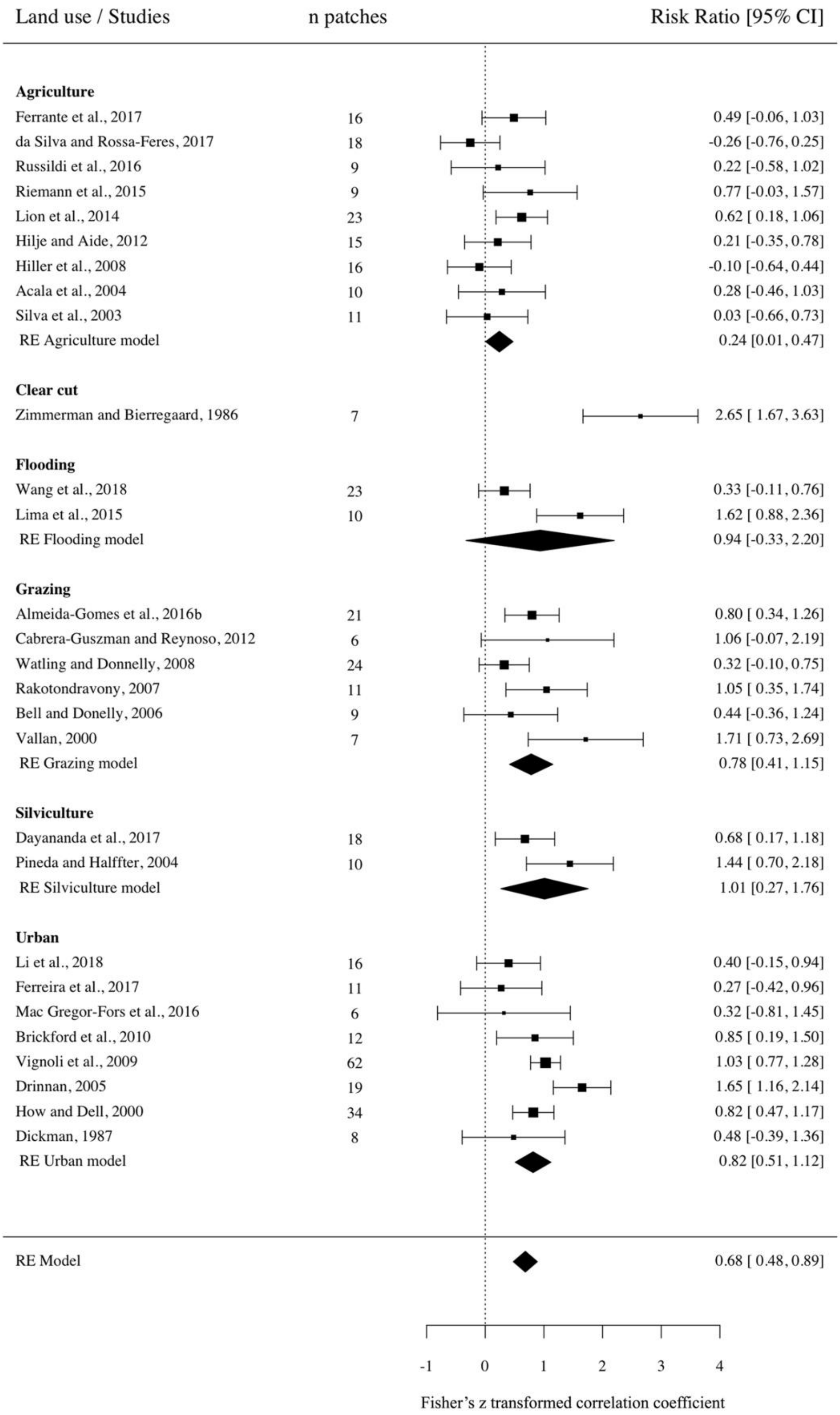

Figure 2 – Forest plot representing the individual effect sizes from each study (squares) and the mean effect size (diamond shape) from each Random Effect (RE) model. Error bars represent the 95% confidence interval. The dashed line represents the null effect.

Table 3 – Test for residual heterogeneity and test of moderators among the random effect (RE) models. $\tau^2 \pm \tau$ = between-studies variance ± its standard definition; Q_E = unexplained heterogeneity; I^2 = the proportion of the observed variance between-studies. Q_M = moderator heterogeneity. In a model without moderators (RE model, first line), Q_E represents the total heterogeneity (Q_T). R^2 = the amount of I^2 explained by the model. The "*" means a significant *p* value.

Moderator	**Test for residual heterogeneity**					**Test of moderators**			
	$\tau^2 \pm \tau$	Q_E	d.f.	*p* value	I^2 (%)	Q_M	d.f.	*p* value	R^2 (%)
No moderators	0.1979 ± 0.0873	93.22	27	< 0.0001*	71.04	-	-	-	-
All variables	0.1719 ± 0.1085	37.07	14	0.0007*	62.24	21.26	13	0.0680	13.10
Number of patches	0.2121 ± 0.0911	87.50	26	< 0.0001*	70.29	< 0.01	1	0.9466	< 0.01
Patch size range	0.2065 ± 0.0926	92.15	26	< 0.0001*	71.78	0.24	1	0.6257	< 0.01
Sample method	0.2099 ± 0.0953	87.55	25	< 0.0001*	71.44	0.88	2	0.6427	< 0.01
Latitude	0.2204 ± 0.4695	93.03	26	< 0.0001*	72.05	1.19	1	0.2751	< 0.01
Biome	0.1996 ± 0.0928	80.77	24	< 0.0001*	70.29	2.65	3	0.4479	< 0.01
Total amphibia Richness	0.2104 ± 0.0937	93.14	26	< 0.0001*	72.08	0.26	1	0.6106	< 0.01
Matrix type	0.1070 ± 0.0623	49.95	22	0.0006*	55.95	21.48	5	0.0007*	45.90

3.2 Meta-regressions

The individual meta-regression analysis indicates that only the matrix type surrounding the patches could explain part of the variance between studies (Table 3). By adding the matrix type as a moderator, the I^2 drops from 71% to 56%. Also, matrix type could explain 45.9% of the inconsistency variance between studies. All other moderators explained less than 0.01% of the variance between studies when tested in individual models, with I^2 values above 70%. A global model reduces the I^2 to 62.2% and explains 13.1% of the inconsistent variance.

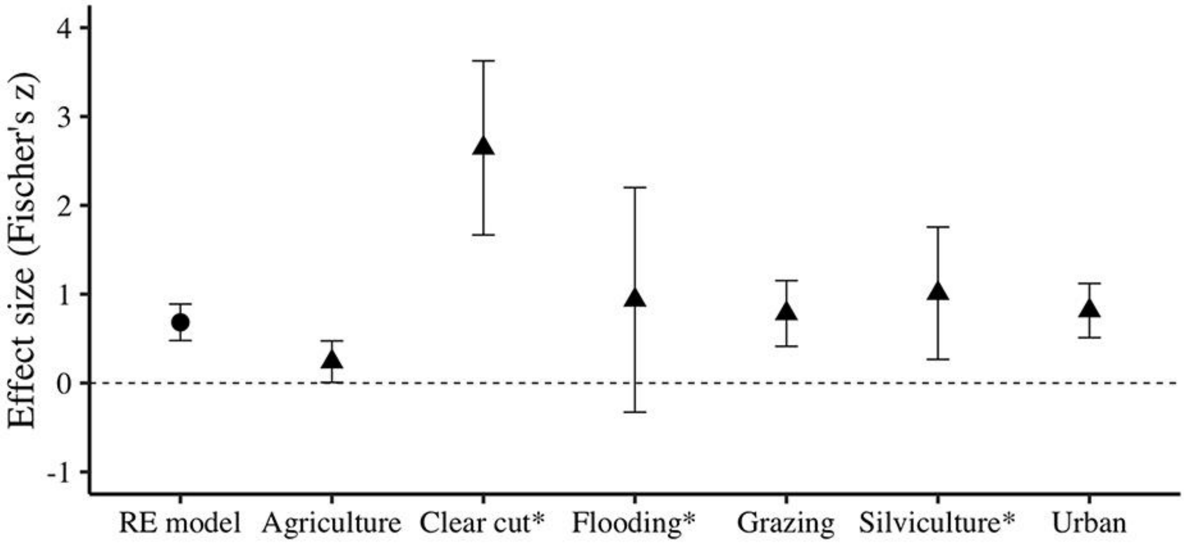

Figure 3 – Mean effect size for random effect models from each matrix type (triangles). RE model (circle) represents the complete random effect model, with all the studies. The dashed line represents the null effect. The error bars represent the 95% confidence interval. The "*" means a matrix type with less than three studies.

The influence of matrix type on the between-studies variance is notable when we aggregated each study according to matrix type (Figure 2). The most representative matrix type was agriculture (nine, 32.1%), followed by urban (eight, 28.6%) and grazing (six, 21.4%). After partition, the model representing only studies with agriculture or grazing matrices reduces its I^2 to below 50% (Table 4). Clear-cut was the matrix with the highest effect size (effect size = 2.65, 95% CI: 1.65, 3.63; Figure 2 and 3) followed by silviculture (mean effect size = 1.01, 95% CI:0.27, 1.76). The lowest mean effect size was found in the agriculture-only model (= 0.24, 95% CI: 0.01, 0.47), followed by the grazing-only model (= 0.78, 95% CI: 0.41, 1.15). With the exception of the flooding-only model, the models' confidence intervals did not cross the null effect line (when Fischer's z-value is equal to 0; Figures 2 and 3), indicating that the significant positive relationship between patch size and amphibian richness was maintained, but with a distinct degree of influence according to matrix type (Table 4). We did not detect publication bias when we separated the collection of studies according to the matrix type (Figure B and Table B; we could not evaluate publication bias in the categories with less than three studies).

Table 4 – Individual random-effect models for each matrix type. $\tau^2 \pm \tau$ = between-studies variance ± standard definition; Q_T = the total amount of heterogeneity; *k* = total amount of studies considered; I^2 = the proportion of the observed variance between studies; *z*-value = Fischer's *z* value. The "*" means a significant *p* value.

Matrix type	**Test for heterogeneity**					**Model summary**		
	$\tau^2 \pm \tau$	Q_T	*k*	*p*-value	I^2 (%)	Estimate ± SE	*z*-value	*p*-value
Agriculture	0.0338 ± 0.0632	10.94	9	0.2049	26.90	0.24 ± 0.12	2.02	0.0433*
Clear cut	-	0	1	1	-	2.65	5.29	< 0.0001*
Flooding	0.7382 ± 1.1803	8.65	2	0.0033	88.45	0.94 ± 0.64	1.45	0.1467
Grazing	0.0891 ± 0.1315	9.02	6	0.1081	44.59	0.78 ± 0.19	4.15	< 0.0001*
Silviculture	0.1889 ± 0.4153	2.80	2	0.0941	64.32	1.01 ± 0.38	2.66	0.0078*
Urban	0.1057 ± 0.1026	18.52	8	0.0098	62.21	0.82 ± 0.15	5.24	< 0.0001*

4. Discussion

A meta-analysis as used in the present study has two main objectives: to detect and describe ecological patterns pointing out the central tendency and the heterogeneity of mean effect across studies (Harrison, 2011; Gurevitch et al., 2018). The interpretation of the current meta-analysis in our study indicated a significative and positive relationship between forest patch size and richness of amphibians. We detected the heterogeneity in the selected set of data and found that matrix type was the main source of such heterogeneity among the tested moderators.

Previous similar meta-analyses explored the habitat amount with amphibian species richness and abundance (e.g., Bender et al., 1998; Watling & Donnelly, 2006; Quesnelle et al., 2015). Our study focused on the association between forest amphibians and forest habitat availability. The biological response of species to habitat area varies according to the taxa, where species traits played a major role in the species' capacity to successfully access and occupy the patch (Bender et al., 1998; Watling & Donnelly, 2006; Prugh et al., 2008; Becker, Fonseca, Haddad, & Prado, 2010; Keinath et al., 2017). Forest is a major variable in a landscape for amphibians, even when they use and reproduce in habitats outside the forest patches (Quesnelle et al., 2015; Almeida-Gomes et al., 2016a).

4.1 Patch size influences amphibia richness

The positive relationship between habitat area and species richness was conceived and established before the Island Biogeography theory itself (Arrhenius, 1921; Darlington, 1957; MacArthur & Wilson, 1967). It also came to be supported recently by field and meta-analytical studies regarding habitat area (e.g., Watling & Donnelly, 2006; Prugh et al., 2008). The development of studies investigating the SAR has resulted in three main current hypotheses that explain the relationship between species richness and patch area. (1) The population hypothesis suggests that species can maintain larger populations in larger areas, and are less likely to become locally extinct in comparison to small populations from smaller areas (Shaffer, 1981). (2) The passive sampling hypothesis describes this relationship as a function of probability: a bigger patch in a fragmented landscape has a higher chance to be reached by individuals through dispersion (Coleman, Mares, Willig, & Hsieh, 1982). (3) Alternatively, the habitat heterogeneity hypothesis proposes that larger areas have larger species richness due to higher availability of variable types of habitats (Boecklen & Gotelli, 1984; Ricklefs & Lovette, 1999; Tews et al., 2004). The habitat heterogeneity hypothesis is especially reasonable for amphibians, once their species richness had already been related to habitat heterogeneity, possibly due to the variety of reproductive modes (Keller et al., 2009; Vasconcelos et al., 2009; Bickford et al., 2010; Becker et al., 2010; Souza et al., 2014; Almeida-Gomes et al., 2016b; Araújo et al., 2018; Li et al., 2018; Figueiredo et al., 2019). Forest-specialist species (stream-dwelling, directly-development leaf-litter and some bromeligenous species) had a positive response to forest patch size, probably because there are more reproductive habitats available inside larger patches (Bender et al., 1998; Becker et al., 2007; Almeida-Gomes et al., 2014; Almeida-Gomes et al., 2016a; Almeida-Gomes et al., 2016b). On the other hand, generalist species can be more strongly associated with the amount of remaining edge (Bender et al., 1998; Ferrante et al., 2017; Ferreira et al., 2017; Figueiredo et al., 2019). The influence of matrix type only contributes to this discussion, because amphibian species' capacity to disperse across a human-made matrix is also related to each particular species-specific traits, such as body size, dispersal capacity, desiccation tolerance and habitat use (Fonseca et al., 2013; Watling & Braga, 2015; Thompson et al., 2016; Nowakowski, Thompson, Donnelly, & Todd, 2017).

4.2 (Trying to) Interpreting variance between-studies

Ecological studies inherently have a high amount of variability among them (Senior et al., 2016), which is one of the main criticisms of the use of meta-analysis in this field (Whittaker, 2010). Ecological and evolutionary meta-analytic studies had a mean I^2 between 83% and 92%, and less than 5% had an $I^2 = 0$ (Senior et al., 2016). However, detecting, labeling and explaining such sources of heterogeneity is one of the central questions when applying a meta-analysis to ecological and evolutionary studies (Senior et al., 2016; Gurevitch et al., 2018). In the present study, the I^2 was 71% in the RE model, which drops to 55% when we added the matrix type as a moderator. Close to 46% (R^2) of the 55.9% model heterogeneity (I^2) was explained by this moderator. This means that 25.7% of the total variance could be explained just by the matrix in the surrounding remmants. Others variables not explored in the present analysis could be important moderators to explain additional portions of the heterogeinety, such as isolation, time since fragmentation and availability of water bodies inside the remanescent (Watling & Donnelly, 2006; Prugh et al., 2008; Bickford et al., 2010; Chen & Peng, 2017; Semlitsch, Walls, Barichivich, & O'Donnell, 2017). We could not appoint them as a moderator in the present study due to the lack of information and the complexity of each topic. For example, only the flooding matrix studies (Lima et al., 2015; Wang et al., 2018) could address the exact moment of the fragmentation event, and few others tested the influence of water bodies inside the habitat remnants (e.g., Bickford et al., 2010; Li et al., 2018). We believe that isolation and extinction debt are complex topics that deserve their own reviews and meta-analyses, with a literature search and screening focused in exploring these points.

The low explanation from in-study and local variables (with exception of matrix, Table 3), indicates a possible global pattern from the SAR in amphibians in forested habitats. Assembling all of the variables in a global model explained 33% less than a model with only the matrix variable. We summarize which moderators would have influence on SAR in the present meta-analysis in a final table (Table 5). Almeida-Gomes et al., (2016b) suggested a possible influence of the sampling method (e.g., pitfall or plots) and sample design on the range of responses between previous studies, some of which are considered in the present meta-analysis (Hillers et al., 2008; Watling & Donnelly, 2008; Lion, Garda, Fonseca, & Loyola, 2014). The limited number of sampling units in fact provide a weaker response of amphibian richness to patch size in some studies (a small number of patches resulted in bigger errors in effect sizes; Figure 2). Also, different sampling methods may result in obtaining distinct species compositions, a possibility that we did not explore in the present meta-analysis. However, neither method nor sample size influenced the meta-regression models (Table 3), indicating a lack of influence in the SAR regarding amphibians in forest habitats. Drakare et al., (2006) found a significant influence of latitude to SAR, with a reduction of z values from tropical to temperate regions. However, this tendency may vary according to the taxa as well, since we did not find this influence in our analysis. We did not detect publication bias among studies (Appencies A, B and C), so it is unlikely that this null result is due to the majority of studies being placed in tropical or sub-tropical regions. Our understanding of this collection of results is that the landscape had a greater influence on the SAR regarding amphibians in forested habitats than methodological and geographical aspects of each study.

Table 5 – A summary of the intrinsic and local moderators and why they would influence species-area relationship (SAR) for amphibians in forest patches in the present meta-analysis.

Moderator	**Nature**	**Explanation**	**Influence on SAR**
Number of patches	Intrinsic to the study	A lower number of sampling units would result in a weak response to SAR (Almeida-Gomes et al., 2016b).	No
Patch area range	Intrinsic to the study	A large range of fragment sizes would result in a more robust response in the SAR (Almeida-Gomes et al., 2016b).	No
Sample method	Intrinsic to the study	Sample method would survey only a group of amphibians (e.g., pitfall traps surveys mostly leaf-litter amphibians; e.g., Siqueira et al., 2011), affecting the number of species detected in the patches (Almeida-Gomes et al., 2016b)	No
Latitude	Local variable	Regional species richness increases from the poles to the equator, so the SAR effect size should show a latitudinal decline (Hillebrand, 2004; Drakare et al., 2006).	No
Biome	Local variable	There is a sample bias, were amphibians from temperate forest were more surveyed than tropical and subtropical forests (Gardner, Barlow, & Peres, 2007); and / or	No

		The biodiversity of the Amphibia taxon is globally determined by the mean temperature, rainfall and its periodicity, where tropical forest holds most of the amphibia species (Vitt & Caldwell, 2013).	
Total amphibian richness	Local variable	A lower local richness would reduce the variance in the correlation, making harder to detect a correlation if exist.	No
Matrix type	Intrinsic to the study	Species' tolerance to cross a human-made matrix and reach the forest patch is related to particular species-specific traits (James I. Watling & Braga, 2015; Thompson et al., 2016; Nowakowski et al., 2017).	Yes

4.3 Matrix is an important factor to consider in SAR studies

Human altered habitats are in a state of constant modification: At the moment of deforestation, the very first matrix type is probably a clear-cut, and may be followed by agriculture, grazing or urban matrices (Foley et al., 2005). After a long time of agriculture, exhausted soils may be converted for grazing or to urban matrix, or simply be abandoned. Our results only represent a moment of the patch lifetime since fragmentation, but elucidate part of this complex relationship between area and species occurrence.

Increasing human activity results in a large conversion of original habitat to human-made landscapes, and among all the progress there are pieces of forest remaining. Amphibian individuals can be negatively affected by human interference at the landscape and local scales. At the landscape scale, the species need to cross different kinds of environments in order to access a habitat remnant. This usually expose them to unfavorable conditions such as desiccation and/or predation (Becker et al., 2010; Watling & Braga, 2015). At the local scale, even if the amphibian reaches its goal, the new patch may not have water resources available for the species reproduction, i,e. to perform its specific reproductive mode (“dry” fragments; Becker et al., 2010). The remaining habitat may present a history of human-made disturbances (e.g., intentional fire, logging), which can remove part of the habitat required for amphibians to survive (e.g., leaf-litter for direct-developing species, epiphytic bromeliads for bromeligenous species), reducing habitat suitability for most species (Pilliod, Bury, Hyde, Pearl, & Corn, 2003; Gardner et al., 2007; Burivalova, Şekercioğlu, & Koh, 2014). Forest-specialist species may leave the patch if it does not have the necessary habitat to survive and reproduce, which may result in a higher mortality during migration through matrix (Becker et al., 2010). In the present study, matrix type explained close to 25% of the total variance in the respective model, and its effect size varied according to matrix type. There is variability among the capacity of these species to cross human-made environments, due to mobility and physiological tolerance (Gascon et al., 1999; Dixo & Metzger, 2010; Nowakowski, Veiman-Echeverria, Kurz, & Donnelly, 2015; Watling & Braga, 2015). Generalist species are usually more adapted to open habitats and inhabit the remnant’s edge, while forest-specialists avoid the environment outside the remnant usually preferring more central portions (Becker et al., 2010; Dixo & Metzger, 2010; Thompson et al., 2016; Pfeifer et al., 2017). This gradient of habitat tolerance could explain the divergent effect-sizes between matrix types.

In contrast, clear-cut and silviculture represents the most influential matrix types in our meta-analysis (Table 4). But they were also the least represented ($k < 3$). Clear-cut matrix was represented only by the Zimmerman and Bierregaard (1986) study, which came from the

Biological Dynamics of Forest Fragments Project (INPE-WWF), a long term project in Brazil's Amazon forest specially designed to test the species-area relationship with forest patches. The study's high correlation ($r = 0.99$) was attributed to the availability of breeding habitats inside the larger remnants, where half of the frog species were habitat specialists and another half were less demanding about reproductive habitat. Silviculture was considered to have some conservation value, so long as they preserve part of the original forest or at least kept a relatively hospitable environment for some species to explore and disperse through. However, this is not the case among types and taxa (e.g., Volpato et al., 2010; Kellner & Swihart, 2014). Silviculture had no distinct effect on the herpetofauna richness in comparison with natural habitats (Thompson et al., 2016). Despite our results showing a significant positive effect for clear-cut and silviculture matrices, more studies regarding their effects in a SAR context are needed to confirm these tendencies. Similarly, flooding matrix may not have been significant due to the low number of studies found ($k = 2$).

Agricultural landscapes proved to have a significant and homogeneous effect on the SAR ($I^2 = 26.9\%$), but with a lower effect size (0.24) in comparison to grazing and urban matrices (0.78 and 0.82, respectively). Watling and Donelly (2006) also tested the influence of the surrounding matrices in SAR, and reached a similar result when considering multiple taxa (invertebrates, birds, herps and mammals, $k = 118$). In their study, agriculture matrix (crops plus grazing) had the lowest z slope (= 0.18). Agricultural matrices may provide low habitat heterogeneity for amphibians (e.g., Souza et al., 2014; Figueiredo et al., 2019). Furthermore, agrosystems are usually subject to periodic exposure to pesticides and fertilizers, which may cause deleterious effects such as deformations, decrease reproduction and increase mortality (Mann, Hyne, Choung, & Wilson, 2009). Even protected areas are subject to these chemical substances, despite buffer zones (García-Muñoz, Guerrero, Arechaga, & Parra, 2019; Sawatzky & Fahrig, 2019). Both agriculture and grazing matrices were previously associated with high species turn-over, with greater substitution of specialist species to generalist species when compared to natural habitats (Gascon et al., 1999; Thompson et al., 2016). Although, the abundance and richness of generalist amphibian species are greater in grazing than in agriculture systems (Thompson et al., 2016). Grazing systems consist predominantly of maintained grassy plants, but in contrast to crop monocultures system, allow the occurrence of scattered trees and ponds, acting as refugia and support for farm animals. Trees and ponds allow some degree of connectivity to less tolerant species across open environments (Ribeiro et al., 2011; Prevedello, Almeida-Gomes, & Lindenmayer, 2018; Jeliazkov et al., 2019) as they provide cover and humidity refugia.

Urban matrices had a mean effect size similar to grazing matrices, but demonstrated a higher heterogeneity among studies in comparison to agricultural and grazing systems (Figure 2, Table 4). The major characteristic of human occupation is the high human density, associated with physical and chemical alterations inside and around patches. Most studies determined a negative effect of urbanization on amphibian richness, abundance, occupancy and community structure (as reviewed by Hamer & McDonnell, 2008). Urbanization modifies the amphibians' breeding habitats and hydroperiod by the construction of dams, artificial channels or impermeable surfaces (Hamer & McDonnell, 2008). The habitat may also reduce its suitability to amphibian reproduction due to water and noise pollutions (Bee & Swanson, 2007; Sievers, Hale, Parris, & Swearer, 2018; Sievers, Hale, Swearer, & Parris, 2019), which enable only highly tolerant species to utilize some of the urban breeding sites (Menin, Melo, Gordo, Hattori, & Sant Anna, 2019). Moreover, the high road density have a significant negative effect on amphibians, by limiting species mobility and reproduction (Fahrig, Pedlar, Pope, Taylor, & Wegner, 1995; Mazerolle, 2004). Nevertheless, the occurrence of species in urban areas was especially related to the availability and heterogeneity of breeding habitats, sometimes more than patch size and shape (Hamer & McDonnell, 2008; Li et al., 2018). Both grazing and urban systems, despite their ecological differences, lead to the replacement of specialist species by highly tolerant generalist species in the context of habitat loss, which may explain their similar effect sizes. The high structure variability between urban patches may result in miscellaneous effect sizes regarding the SAR, and a higher heteroscedasticity between studies as consequence.

5. Conclusions

Since the implementation of Island Biogeography theory in conservation ecology (Diamond, 1975), the theory is one of the main topics debated in this respective area. Yet, thinking in a global and general model focused only in patch size and isolation may be a "caricature of reality"(Laurance, 2008), because matrix type surrounding the patches also has a large amount of influence in the species richness and composition (Ricketts, 2001; Laurance, 2008, 2010; Dixo & Metzger, 2010). Our data integrate this fact, by suggesting that forest patch size have distinct degrees of influence in amphibian richness across multiple landscapes worldwide. One quarter of the between-studies variance can be attribute to the distinct matrix surroundings alone.

Protective laws and guidelines to create preserved areas may be strongly based on the "natural reserve design guidelines", suggested by Diamond (1975), but they usually ignore the landscape context (Fahrig, 2019). Species did not restrict themselves into the patches, and biodiversity threats do not respect the protected-area boundaries (Laurance et al., 2012; Pfeifer et al., 2017; Sawatzky & Fahrig, 2019). Creating and restoring protected areas is a challenge for decision-makers, particularly in developing and forested countries, because of conflicts with important social and economic issues (Bergallo et al., 2009; Bustamante et al., 2019). A possible alternative to this problem is the creation of laws for biodiversity-based land management (Foley et al., 2005; Kremen & Merenlender, 2018), where the land use surrounding protected areas may be managed as well.

Before any discussion regarding the divergent responses of fauna to patch size, we suggest an investigation on the local landscape first, followed by a discussion of how this could affect different local taxa. We are not proposing the adoption of one particular kind of matrix habitat for management purposes, but remarking that they affect the SAR and need to be considered in future landscape ecology studies. There is no doubt that patch size is an important driver for species occupancy and abundance, but it needs to be interpreted in a specific landscape context. If possible, patch size should be compared within a context of multiple landscapes, and not only in a patch-focused and local approach. The scale effects of patch size, habitat amount and especially fragmentation *per se,* were intensely discussed between Fahrig, 2017; Fletcher et al., 2018; Fahrig et al., 2019; and Miller-Rushing et al., 2019.

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Appendices

Table A – Fail N Safe analysis results, to detect possible bias among studies (*k* = 28). NA = Not applicable.

Fail-safe N approach	**Fail-safe N**	**Average effect size**	***p* value**
Rosenthal's	1499	NA	< 0.0001
Orwin's	28	0.7207 (target = 0.3664)	NA
Rosenberg's	1209	0.6893	< 0.0001

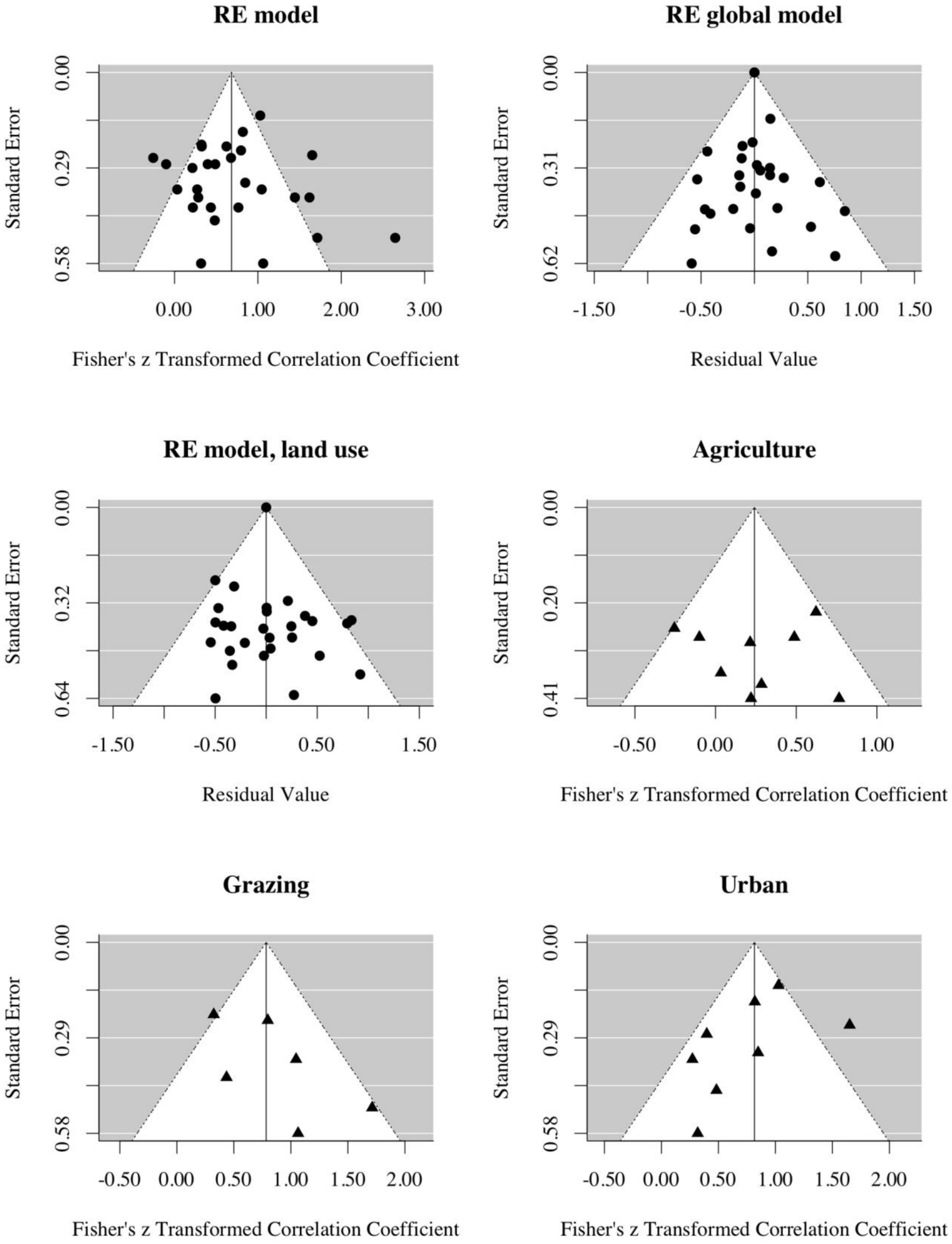

Figure A – Funnel plots to detect possible data bias for each main random effect (RE) model. We could not evaluate publication bias in the categories with less than three studies.

Table B – Regression test results for funnel plot asymmetry, with standard error as predictor. A non-significant (*p* > 0.05) result indicates that the funnel plot is symmetric, probably with no publication bias. We could not evaluate publication bias in the categories with less than three studies.

Model	**Description**	***z*-value**	***p*-value**
RE model	Random effect model, without moderators.	1.23	0.2190
Global model	Random effect model with all variables as moderators.	0.18	0.8530
Matrix type	Random effect model with matrix type as moderator.	0.52	0.6029
Agriculture matrix	Random effect model only with studies which agriculture is the predominant matrix surroundings the patches.	0.23	0.8183
Grazing matrix	Random effect model only with studies which pastures are the predominant matrix surroundings the patches.	1.68	0.0932
Urban matrix	Random effect model only with studies which urban areas are the predominant matrix surroundings the patches.	-1.44	0.1486

3. Capítulo 2

Anfíbios de várzeas respondem à composição da paisagem no Norte do Paraná, Brasil

Artigo a ser submetido ao periódico *Biotropica*

Floodplain amphibians' response to landscape composition in North Paraná, Brazil.

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ABSTRACT

The expansion of agriculture at the cost of natural forests loss is reducing biodiversity worldwide. Amphibians are the vertebrate group most affected by this habitat loss, due its physiological restrictions. Our main goal was test the relationship between amphibian species richness decline in tropical floodplains with forest conversion to farmland in the surroundings of their reproductive sites. We sampled amphibian communities in 10 floodplains from a highly deforested landscape inserted in the Semideciduous Atlantic Rainforest. We tested the influence of forest and farming habitats amounts, forest edge density and landscape diversity on amphibia total and forest-specialist richness, and evaluated the scale of effect. We first created a composite landscape context variable by performing principal component analyses at 18 spatial scales, and then extracted the values from the first principal component. For each scale, we tested the influence of the first component on all species and forest-specialist species richness with linear models, and selected the model with the lowest Akaike criteria. Both the total and forest-specialist richness had a scale effect at the max scale of 2,000m. Forest-specialist richness also had nested models at shorter scales. Both richness responded negatively to the first principal component, where most of the landscape variation were explained by farming amount, followed by forest amount. Farming and forest amount had respectively a negative and positive effect over both total and forest-specialist species richness. Forest amount was probably more influent for forest-specialist species, because all of the nested models had a stronger influence of the forest habitat.

Key-words: Anura, Atlantic rainforest, Edge density, Habitat amount, Landscape heterogeneity, Scale of effect, Semideciduous forest, Várzeas.

1. INTRODUCTION

Agriculture is in constant expansion worldwide, presumably due to the increasing demand in food production (Godfray *et al.* 2010). This expansion comes at cost to natural forests. Some estimates suggested that 80 % of the worldwide deforestation is driven by agriculture expansion alone (Kissinger *et al.* 2012). Historical changes in land-use have strongly reduced biodiversity, being one of the main causes of the current biodiversity crisis (Cardinale *et al.* 2012, Haddad *et al.* 2015, Newbold *et al.* 2015, Powers & Jetz 2019).

Amphibians are the vertebrate group most affected by habitat loss (Stuart *et al.* 2004, Cushman 2006, Pfeifer *et al.* 2017, Powers & Jetz 2019). The conversion of forest into agriculture habitats reduces the abundance and species richness of amphibians, even for species were the reproductive habitat is located at the forest edge or in an open area habitat (Zimmerman & Bierregaard 1986, Cushman 2006, Becker *et al.* 2007, 2010, Gallant *et al.* 2007, Quesnelle *et al.* 2015, Ribeiro *et al.* 2018). Species that are forest-habitat specialist should be more affected by forest loss than open-area specialists, due to the group's high habitat specialization, were the forest offers the necessary habitat for its survival and reproduction (Becker *et al.* 2010, Almeida-Gomes & Rocha 2015, Almeida-Gomes *et al.* 2016, Nowakowski *et al.* 2017, Pfeifer *et al.* 2017).

The process of habitat loss can lead (but not necessary) to the creation of forest edge-habitat due to the fragmentation process (Fahrig 1997). The abrupt removal of the forest instantly changes microclimate variables such as wind and light incidence, which can create disturbances could be found as far as 400 meters inside the forest habitat (Laurance 2010). Some amphibians that are specialized in open-area habitat could benefit from the expansion of forest-edge environment (e.g., Oda *et al.* 2016, Figueiredo *et al.* 2019). However, for most of the species, the increase in forest-edge habitat through fragmentation reduces amphibians abundance and occupancy, especially for forest-core habitat specialists (according to a global meta-analysis performed by Pfeifer *et al.* 2017, d.f. = 103).

Agriculture is also increasing its intensity. Monocultures of commodities (e.g., soy, corn) reduces crop's landscape diversity, which causes severe effects on local biodiversity (Benton *et al.* 2003, Purtauf *et al.* 2005). Agriculture matrix provide low diversity of habitats (compositional heterogeneity; Fahrig *et al.* 2011) for amphibians reproduction (Silva *et al.* 2011, Oda *et al.* 2016). In theory, a high diversity of landscape cover types (configurational heterogeneity; Fahrig *et al.* 2011) should provide more prey resources for amphibians in the matrix (Collins & Fahrig 2017, Li *et al.* 2018). However, Li *et al.* (2018) found positive effects of cover-type diversity in the abundance of amphibians in a urban landscape. However, Collins and Fahrig (2017) couldn't support it for North American frogs in an agricultural landscape. Apparently, amphibians response to configurational heterogeneity varies according to the species life-history, where particular land-use types could allow the occurrence of specific habitat specialists species (Collins & Fahrig 2017, Brüning *et al.* 2018, Li *et al.* 2018).

To access the effect of a landscape variable (e.g., forest amount) over a biological variable (e.g., species richness), we need to evaluate the scale in which the relationship is strongest – known as the "scale of effect" (Jackson & Fahrig 2012, Martin & Fahrig 2012). The scale of effect suggest that the strength of a relationship between a biological variable (e.g., fecundity, abundance, species richness) with a landscape variable (e.g., habitat amount, road density) is related with the studied scale (Holland *et al.* 2004, Jackson & Fahrig 2012, 2015, Martin & Fahrig 2012, Miguet *et al.* 2016, Martin 2018). It is important to address the scale of effect, because it varies according to the type of biological response and the studied taxon (Miguet *et al.* 2016, Martin 2018, Moraga *et al.* 2019). The scale of effect may also vary according to the species degree of habitat specialization, as specialist species supposed respond at a closer range than generalist species (Miguet *et al.* 2016). However, there's little empirical evidence supporting it (Chaplin-Kramer *et al.* 2013), including a report that might be the opposite response for amphibians (Almeida-Gomes *et al.* 2019).

Objectives

Our main goal was to test the prediction that amphibian species richness in tropical floodplains declines with forest conversion to farmland surrounding the floodplains. Specifically, we predicted that the amount of crops and pasture areas should have a negative relationship in species richness considering all the species and forest-specialist only, while forest cover should show positive relationships. We also tested the possible influences of forest edge density and landscape diversity on species richness, predicting that edge density would reduce richness and landscape diversity would increase richness. We expected these landscape variables to influence amphibian richness within 300 meters, based on previous tropical rainforest studies (Ribeiro *et al.* 2018, Almeida-Gomes *et al.* 2019).

2. METHODS

Overview

To achieve our objectives, we selected 10 floodplains from a highly degraded landscape in the Atlantic Rainforest domain (North Paraná region; Figure S1) and registered their amphibian composition during two consecutive rainy seasons. We selected floodplain areas to minimize structural variation between distinct reproduction sites (e.g., permanent ponds vs. floodplains vs. streams). Also, floodplains had a regular cycle of flooding every rainy season, what make those reliable reproduction sites every year (e.g., compare to temporary pounds).

We selected the surveyed floodplains along a gradient in the amount of forest cover in their surroundings.

We evaluated the landscape surroundings the floodplains in 18 nested scales. Due to low sample size (10 floodplains), the number of landscape variables to be tested (four), and the correlations among them (see Results), we created a composite landscape context variable by performing principal components analyses at each spatial scale, and then extracted the values from the first principal component. We then tested the relationship between the first principal component and amphibian richness with linear models. We performed separate models for all amphibian richness and for forest-specialist amphibian richness as dependent variables. We assessed the scale of effect as the scale with the best model using the Akaike criterion. Then, we interpreted the results from the models at the scale of effect based on the loadings of the landscape variables on the first principal component.

Study region and species

We conducted the study in a fragmented landscape around the municipalities of Londrina, Ibiporã and Jataizínho, in the northern region of Paraná State, South Brazil. The predominant native vegetation is the Seasonal Semideciduous forest, in the Atlantic Forest biome (IBGE 2012). This forest is characterized by the relatively low rainfall in comparison with the other two types of Atlantic forests found in Paraná (annual precipitation of 1500 mm; IAPAR 2017) and the partial loss of leaves from native trees during winter (Roderjan *et al.* 2002, IBGE 2012). The area was severely deforested during the 19th and 20th centuries due to intensive logging and agriculture expansion for coffee plantations (Martins 1995, Soares & Medri 2002). In 2017, the area in the studied municipalities was predominantly composed by agriculture and pasture, representing 79.1%, while forest represented only 15.0% and urban settlement 5.0% (MapBiomas Project 2019). The local agriculture matrix is mostly composed by monocultures of soy or corn (IBGE 2012; confirmed with pers. obs.). The climate is subtropical humid, with a mean temperature of 22˚C, 28˚C in the warm season and 17˚C in the cold season (IAPAR 2017).

The local amphibian fauna has been studied only recently (Bernarde & Anjos 1999, Machado *et al.* 1999, Machado & Bernarde 2002, Figueiredo *et al.* 2019). To our knowledge, only amphibians from the Anura order have been detected in the North Paraná’s region (Santos-Pereira *et al.* 2018). The majority of the species belong to the Hylidae and Leptodactylidae families (Bernarde & Anjos 1999, Machado *et al.* 1999, Figueiredo *et al.* 2019).

Sample sites and data survey

Data were collected from 10 different floodplains in North Paraná. Specifically, these floodplains are called “várzeas”, temporarily flooding areas associated with rivers, where sediment and organic material accumulates and fill during the rainy season. These are breeding zones for terrestrial organisms with aquatic larvae (Tockner & Stanford 2002), such as most anurans.

We recorded the amphibian species occurrence between October and February of the 2016-2017 and 2017-2018 rainy seasons, which was the breeding season for most of the species (Bernarde & Anjos 1999, Machado & Bernarde 2002), and the maximum extension of the flooding zone. We used time-limited visual and auditory active surveys (Crump & Scoot Jr. 1992), both on the floodplain margins and within the floodplains whenever possible. Two to three researchers made four visits to each floodplain, with a sampling effort of one hour for each visit (40 hours total sampling), after sunset and before midnight. There was at least a one-week difference between field samples at a given site. Data were collected under the authorizations of the Chico Mendes Institute for Biodiversity Conservation (SISBIO 53132), the Paraná Environmental Institute (IAP 09.16), the Animal Ethics Council of the State University of Londrina (CEUA 139.2017) and approval from all landowners when sample sites were located in private land. We collected sample species for record, and deposited the individuals in the Zoological Museum of the State University of Londrina.

Habitat associations

We classified the amphibians according to their habitat use from Machado & Bernarde (2002) and Figueiredo *et al*. (2019). We considered a species as a "forest-specialist" if it was found only inside and/or at the forest edge; as an "open-area specialist" if it was only located in the open-areas or in the open-areas and at the forest edge; and as a "true generalist" if it was found in forest interior, forest edge and open area habitats. Most species detected inside the floodplains were open-area specialists (Results section) and, as a consequence, were highly related to overrall amphibian richness (Spearman's $\rho = 0.81$). Therefore, we opted to do two analyses, including all species and a subset considering only the forest-specialist species, due to their conservation relevance. Our initial prediction was that forest-specialist species would be more related to forest amount in comparison with the total amphibian richness, due to the species strong association with forest habitat.

Landscape metrics

We measured the following landscape information surrounding the floodplains: forest amount (%), farming amount (%), forest edge density (meters of edge / landscape area in meters^2) and landscape diversity, which we measured as Shannon landcover diversity. Land use information were extracted from the MapBiomas Project (2019), based in the 2017's year database. The MapBiomas Project is a multi-institutional initiative to generate annual land use cover maps for Brazil, based on automatic classification processes applied to satellite images.

With the current raster we classified most of the farming cover as ‘annual and perennial crops cover’, represented by soy or corn plantations (pers. obs.), or ‘mosaic of agriculture and pasture’, when the automatic classification couldn’t distinguish a pixel as crop or pasture. We grouped those pixels with others less represented by farming cover types (e.g., pasture) as a single cover type called ‘farming’. We expressed the landscape diversity as Shannon landcover diversity (Shannon & Weaver 1949) of the cover types pixels inside the surrounding landscape area. We considered the following cover types in the calculation of landscape diversity: forest, farming, urban and water. We did not detect any forest plantations of *Eucalyptus* and *Pinus* in our landscapes. We tested the correlations between landscape variables using Spearman’s correlation ρ, due to the non-normal distributions of all landscape variables.

Landscape composition surrounding floodplains

We extracted the landscape composition surrounding each of the surveyed floodplains within 18 different distances from the floodplain central point. We collected landscape information from 300 to 2000 meters radii, with 100 meters distance between scales. We choose the 300- and 2000-meters limits due to our data limitations. The minimal scale was 10 times the pixel size of our landscape data (30 meters). The maximum spatial extent was the limit to avoid spatial overlap greater than 50% between the landscapes surrounding any two sites.

Statistical analysis

We performed a principal component analysis (Pearson 1901) with the four landscape variables at each landscape scale. We tested the influence of the first principal component on species richness, at each scale, by performing generalized linear models with a Poisson distribution and log link (Zuur *et al.* 2009). Because of the log link, the intercept and estimate values from the models were expressed in log values. We determined the best landscape scale by comparing the values from the model Akaike Information Criterion with small-sample correction (AICc; Hurvich & Tsai 1989, Burnham & Anderson 2004). We considered the best model the one with the lowest AICc value (Burnham & Anderson 2004). Additionally, we calculated a *pseudo* R^2 value and a dispersion parameter for each model (Dobson 2002, Zuur *et al.* 2009). The ideal dispersion parameter value should be around one, but due to our small sample size we expected it to be less than one (Zuur *et al.* 2009).

All the analyses were made with R software version 3.6.1 (R Core Team 2019). We processed the landscape data with functions from *raster* (Hijmans 2019), *rgeos* (Bivand & Rundel 2019) and *rgdal* (Bivand *et al.* 2019) packages. Land cover amounts and the forest edge density were extracted and calculated with the *spatialEco* package (Evans 2019). Shannon diversity for landscape diversity and the principal component analyses were made with the *vegan* package (Oksanen *et al.* 2019). We created the generalized linear models with the s*tats* package (R Core Team 2019) and calculated the AICc values and made the model selection with the *MuMIn* package (Bartoń 2019).

3. RESULTS

We detected a total of 23 anuran species from eight families (Table S1), in an average of 7.5 species (range: 4 – 13) per floodplain. The most common species was *Dendropsophus nanus*, which was present in 80% of the surveyed floodplains, while *Boana prasina*, *Crossodactylus schmidti* and *Vitreorana uranoscopa* were detected only in one floodplain each

(Table S1). We categorized six of the species as forest specialists, 15 as open area specialists and two of them as true generalists (Table S2).

The four landscape variables we considered in our study were inter-correlated, sometimes very highly, depending on the landscape extent (Table S3). The variable loadings on the first principal component also varied with landscape extent (Figure S2). Farming amount loaded positively onto the first principal component and had the highest loading value at most scales. Forest amount, edge density and landscape diversity loaded negatively onto the first principal component. Forest amount usually had a similar loading value to farming amount, due to their consistently strong, negative correlation (Table S3). At very small spatial extents landscape diversity did not load strongly onto the first axis, due to low variation across sites in landscape diversity at small spatial extents.

The principal components analysis in our study indicated that the first principal component at the scale of effect (2000 meters, next paragraph) corresponded to 50.1% of the variance from the landscape variables. Farming and forest coverages had the highest loadings values (0.69 and -0.62 respectively), while edge density and landscape diversity had the lowest ones (-0.23 and -0.27 respectively). These last two variables were more related to the second principal component, which corresponded to 39.4% of the variance (Figure S3).

The first principal component was most strongly related to both total amphibian species richness and forest-specialist richness within 2000 meters (Figure 1, Tables S4 and S5). As this was also the largest spatial extent scale we tested, this indicates that 2000 m is a minimum estimate of the spatial extent for landscape context effects on these species. The true 'scale of effect' (Jackson & Fahrig, 2012; Martin & Fahrig, 2012) could be larger than 2000 m.

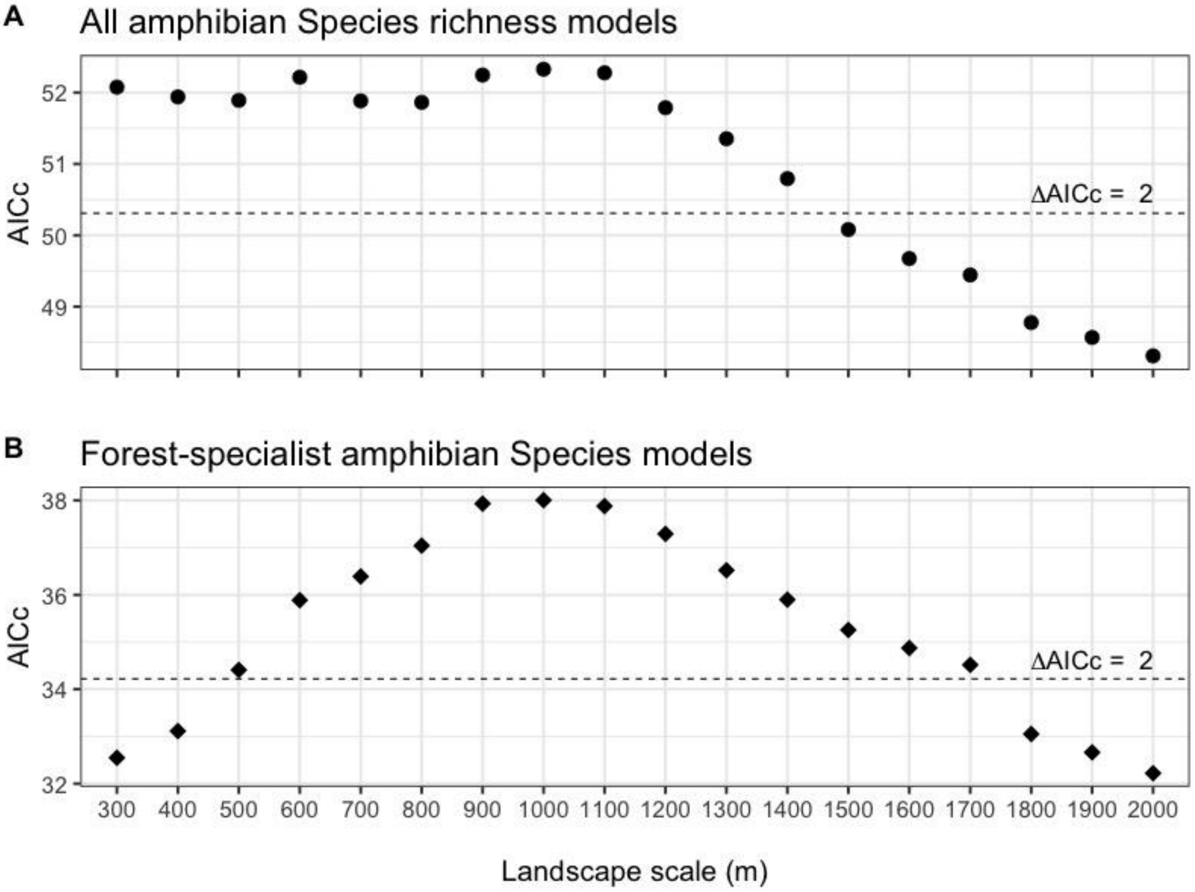

Figure 1 – Estimating the scale of effect of landscape variables on total amphibian richness (A) and forest-specialist amphibians richness (B). For each scale, a Poisson-distribution model tested the relationship between the first principal component from a PCA combining four landscape variables (farming cover, forest cover, edge density and landscape diversity) and the respective amphibian species richness response. The ‘scale of effect’ is the scale where the model had the lowest Second-order Akaike information criterion values (AICc).

At this spatial extent, the first principal component was negatively related to both total amphibian richness and forest-specialist richness (Table S6, Figure 2). However, for forest specialist richness the relationship was strong at both 2000 m and at the minimum spatial extent measured, 300 m (Figure 2B, Table S5).

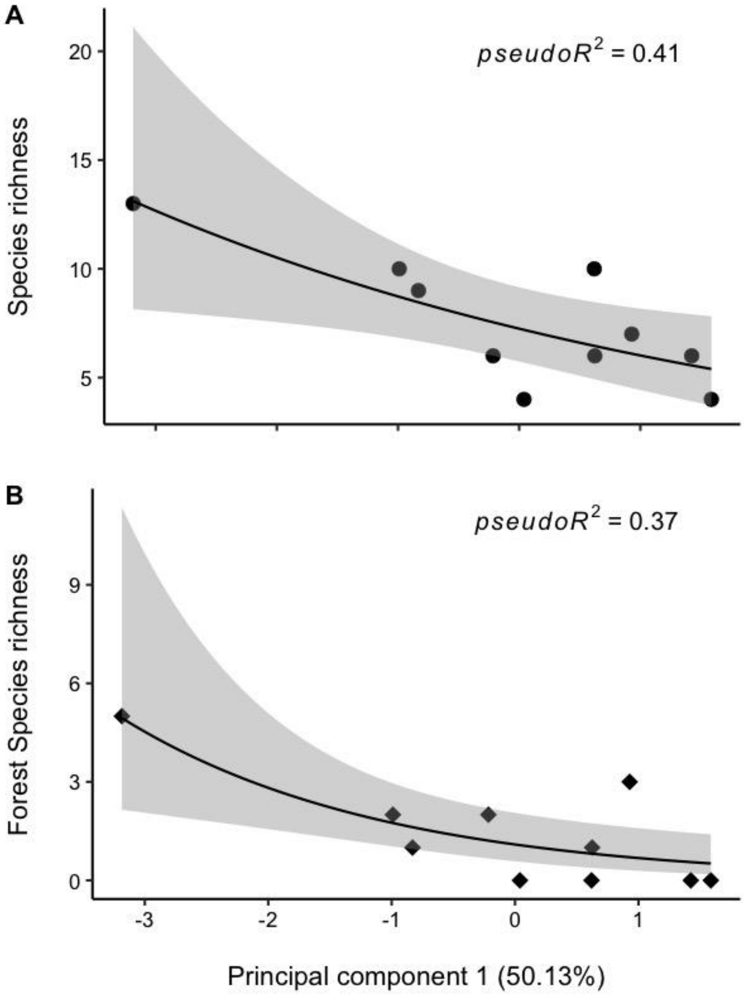

Figure 2 – Relationship between total amphibian species richness (A) and forest-specialist amphibian species richness (B) and the first principal component combining four landscape variables within 2000 meters of each of 10 floodplains. Principal component 1 is a positive function of farming amount and a negative function of forest amount, edge density, and landscape diversity (Fig S3). The line represents the model prediction (Table S6) and the gray area is the 95% confidence interval.

4. DISCUSSION

At the scale of effect, both our models indicate a negative relationship between all-species and forest-specialist species richness with the first principal component (Figure 2). It may demonstrate, respectively, a negative influence from farming amount and a positive influence from forest cover, edge density and landscape diversity over both tested sets of amphibian species richness. It was consistent withour predictions regarding landscape configuration, except for edge density. We expected a negative relationship of edge density with both sets of amphibians richness. Surprisingly, the scale of effect demonstrates to be way larger than expected, at least 2000 meters (Figure 1).

The present study is one of the few (see Oda *et al.* 2016 and Figueiredo *et al.* 2019), performed in the northern region of Paraná seeking to understand the effects of landscape configuration on the presence of amphibian species. Our findings were similar to previous habitat amount studies regarding amphibian species (Eigenbrod *et al.* 2008, Collins & Fahrig 2017, Ribeiro *et al.* 2018). We reinforce that the reduction of forest habitat availability by increasing the amount of farming has severe consequences on amphibian species in reproductive sites.

Habitat amount influence species richness

The influence from the first principal component over both species richness (total and forest-specialist) is a possible consequence of the major amount of variation explained by farming and forest amounts (0.69 and -0.62, respectively; Figure S2), in comparison with forest-edge density (-0.23) and landscape diversity (-0.27) at the scale of effect. We are not able to distinguish if this is an isolated effect from farming or forest amount, because both variables were highly correlated (Spearman’s ρ = - 0.88). The northern region of Paraná has severely suffered along time with deforestation due to agriculture, with only 7% of the original forest left (SOS Mata Atlântica & IBGE 2014). Natural forests were mainly substituted with coffee, followed by soy or corn plantations in the last century.

Habitat loss is one of the main consequences of fauna extinctions worldwide, especially in the case of amphibian species (Stuart *et al.* 2004, Cushman 2006, Powers & Jetz 2019). Due to the amphibian’s physiological limitations (permeable skin and gelatinous eggs), they depends on humid environments to live and reproduce (Vitt & Caldwell 2013). The forest habitat offers the necessary humidity and access to reproduction sites (e.g., bromeliads, leaf litter, ponds, streams, phytolimnos). In fact, several studies consider that forest amount increases amphibians richness (Watling & Donnelly 2006, Eigenbrod *et al.* 2008, Prugh *et al.* 2008, Almeida-Gomes *et al.* 2016, Collins & Fahrig 2017) probably due to a higher availability of reproduction habitats (Boecklen & Gotelli 1984, Ricklefs & Lovette 1999, Tews *et al.* 2004). The richness of forest-specialist amphibian is even more affected by the size of the forest patch in comparison with open-area and generalist species (Almeida-Gomes *et al.* 2016). However, the size of the forest patch alone does not explain species occurrence inside of it (Boecklen & Gotelli 1984, Laurance 2010, Keinath *et al.* 2017). The human-made matrix is an important predictor of species occurrence inside the patch for several taxa, due to the matrix-tolerance differences between species (Ricklefs & Lovette 1999, Prugh *et al.* 2008, Dixo & Metzger 2010, Prevedello & Vieira 2010, Thompson *et al.* 2016, Keinath *et al.* 2017). For example, matrix type is a very important factor determining the occurrence of insectivorous birds in small fragments (Antongiovanni & Metzger 2005). The effect of forest patch size on the richness of carabid beetles was depended on the level of urbanization surroundings the patches, due to species distinct affinity to the urban habitat (Gaublomme *et al.* 2008). For amphibians, the relationship between amphibians richness and patch size is partially explained by the matrix type surrounding the remain (Motta-Tavares *et al*. *in press*, first chapter of this thesis).

Fahrig (2013) proposed the habitat amount hypothesis as a potential mechanism of species loss due to habitat loss, instead of a patch-centered interpretation. The author challenges the distinct effects of habitat patch size and isolation on species richness, arguing that this increase is due to a sample area effect. In areas with a continuous habitat, larger sample areas will contain more individuals, and as a consequence, more species. So, instead of evaluating the size and isolations of a patch, Fahrig (2013) suggest to evaluate the amount of forest surroundings the sample area. For wetland species, forest amount was an important predictor of species abundance, including amphibians (Quesnelle *et al.* 2015 meta-analysis, d.f. = 204). Two previous studies performed in the Atlantic rainforest found similar results regarding species richness. Ribeiro *et al*. (2018) found a positive effect of forest amount and a negative effect of agriculture over amphibian occupancy in southern São Paulo State. Similarly, Almeida-Gomes *et al.* (2019) found a positive relationship between forest cover and both species richness and functional diversity of amphibian in Rio de Janeiro montane forests.

Scale of effect

While our results agree with Ribeiro *et al*. (2018) and Almeida-Gomes *et al.* (2019) regarding the positive effect of the Atlantic rainforest cover, we found distinct results in terms the scale of effect. In the present study, the scale of effect is at a minimal of 2000 meters scale, with a tendency of both group of models to reduce its AICc values with distance (Figure 1).

As far as we could gather from the scientific literature, there were two studies performed in the Atlantic rainforest that actually tested the scale of effect of forest amount over amphibian richness. In Ribeiro *et al*. (2018), according to their Material and Methods section, they only tested the 200 and 500 meters distance scales, and found out that 200 meters presented stronger relationships between their response variables (species occupancies) and forest cover. On the other hand, Almeida-Gomes *et al.* (2019) applied a multi-scale analysis (Holland *et al.* 2004, Miguet *et al.* 2016) and found out that the best explanation when considering all-species richness was at 200 meters distance and 1400 meters when considering only forest-specialist species. The main difference in our methods in relation to those studies was that we summarized the landscape variables of our study into a single dimension, while Ribeiro *et al*. (2018) and Almeida-Gomes *et al.* (2019) tested only for forest-amount and/or agriculture. The first principal components in our analyses are mostly influenced by farming amount, while forest amount loading varies according to the scale (Figure S2). For forest-specialist species, the scale of effect was probably a consequence of the forest amount, despite the major influence from farming amount. Two of the models with ΔAICc < 2 were at 300 and 400 meters scale, the scales with the highest forest amount loadings after 1800, 1900 and 2000 scales. Also, the highest AICc models were at the minimal influence of forest-amount.

Miguet *et al*. (2016) review proposed 14 possible predictions to explain what determined the scale of effect in landscape studies. Among those, the mobility-related predictions suggested that species local movement and dispersal distance may have influence on the scale of effect between a landscape variable and a biological response (e.g., abundance, occurrence). Generally speaking, the scale of effect would be larger for more mobile species, because they interact with the environment in a larger scale (Predictions # 1-7). Ribeiro *et al*. (2018) initially used the 200 meters scale based on the range average habitat use by amphibians (Semlitsch & Bodie 2003). However, we neither confirm nor deny that the present scale of effect was driven by the species movements due to lack of studies for local species (e.g., *Leptodactylus labyrinthicus* - Tozetti & Toledo 2005; and *L. latrans* - Henrique & Grant 2019).

Another possible explanation for the differences between scale of effect is the possible differences in species density between localities. Miguet *et al*. (2016) suggested demography-related predictions, where higher-density populations may have a smaller scale of effect in comparison with lower density populations (prediction #8). Despite Almeida-Gomes *et al.* (2019) study was also performed in an Atlantic rainforest, the study was performed in a Dense Atlantic rainforest locality, while our research was performed in a Semi-decidual Atlantic rainforest. The main natural differences between these two types of Atlantic rainforest is the amount and distributions of rainfall (Oliveira-Filho & Fontes 2000): while the Dense Atlantic rainforest has more rainfall (between 2000 and 3600 mm), with a comparatively more uniform distributions along the year, the Semi-deciduous forest receive less rain (between 1500 and 2000 mm) and has a marked dry season that can last between 40 and 160 days. Amphibians richness and abundance are very sensitive to rainfall, even at a fine-scale (Toft 1980, Lion *et al.* 2019, Pereira-Ribeiro *et al.* 2020). As a consequence, the richness in the Semi-decidual Atlantic rainforest is considerably lower (111 species in Garcia *et al.* 2007, cited as “Floresta Atlântica de Interior”) in comparison with the Dense Atlantic rainforest (165 species in Garcia *et al.* 2007, cited as “Floresta Atlântica Serra do Mar”). We did not find a study comparing amphibian densities between the Atlantic rainforest types. However, assuming that this difference exists in terms of species richness, we extrapolate that this difference in amphibian densities may also exists between Semi-deciduous and Dense rainforests.

Distinct landscape context and distinct species traits may also result in different scales of effect among landscapes (Miguet *et al.* 2016 – Prediction # 14). Another difference between the current study and that of Almeida-Gomes et al. 2019 may be the degree of deforestation between localities. Cachoeiras de Macacu municipality has one of the largest Atlantic rainforests remnant, with over 60,000 ha. The remamnt in North Paraná (Mata dos Godoys State Park) has 675 ha, with many other with less than 10 ha (SOS Mata Atlântica & IBGE 2014).

Neither Miguet *et al*. (2016) nor us found a study that tested for regional variation in the scale of effect, but it is a plausible explanation to the considerable differences between the scale of effect evaluated by Almeida-Gomes *et al.* (2019) and in our study.

Considerations and caveats

Due to the low sample size (10 floodplains), we summarized the four variables of our study into a single one, which retained 50% of the landscape variance. Our study demonstrated that habitat amount (forest and farming coverages) had a bigger role in influencing species richness inside floodplains. But it does not mean that edge density or landscape diversity had no influence at all because they had lower loadings values. A possible explanation for the lower loadings from edge density and landscape diversity that our sample size was unable to capture enough variance for these variables. Also, the dominance of farming cover at larger scales may dilute the influence of these variables at larger scales. Edge density and landscape diversity may have a major influence over species richness with larger sample sizes, or in a less deforested landscape.

To properly evaluate the scale effect of habitat loss over amphibian species in North Paraná, the analysis should consider distances over 2000 meters from the sample site. However, our sampling was unable to evaluate larger scales without to avoiding overlap between localities, which would lead to a spatial-correlation between sites.

Conclusions

Despite the small sample size, our results demonstrated that, in fact, the amount of forest and farming coverages had a major influence over all amphibian species and forest-specialist in floodplains. For all-species richness, we were unable to affirm that species richness effectively increased with forest cover or decrease with farming cover due to their high correlation. However, we can speculate that forest-specialist species were positively related to forest amount, despite the major influence of farming amount, because the best models were the ones with the highest influence of forest amount in the first principal component.

Researchers should consider testing the scale of effect over 2000 meters distance, even with species with relatively lower mobility. It is uncommon for landscape studies focused on amphibians to use scales over 2 km distance (Jackson & Fahrig 2015), where researches argue the lack of mobility of amphibian species (Jackson & Fahrig 2015; e.g., Ribeiro *et al.* 2018). Amphibians may respond more to the landscape than previously expected. Also, if Miguet *et al*. (2016) predictions were correct, the scale of effect may vary according to the local species density and / or the strength of the relationship between species traits and the environment surroundings.

Our study demonstrates that landscape management may not consider the appropriate scale to protect species reproduction sites. For example, the Brazilian forest code law (Law number 12.651/2012) stablished that várzeas are permanent protected zones and natural water coursed must have a protection buffer zone. However, the law set a minimal buffer of only 30 meters in smaller water courses, up to 500 meters for larger rivers. These values are clearly far from the scale of effect (2000 meters) of our study. Our findings demonstrate that the substitution of forest habitat by farming habitat in the surroundings of the floodplain may result in a severe reduction in species richness. Not only the natural vegetations surrounding the reproduction site should be protected, but the management plan should also consider the scale of influence of the human-made matrix on the occurrence of species inside.

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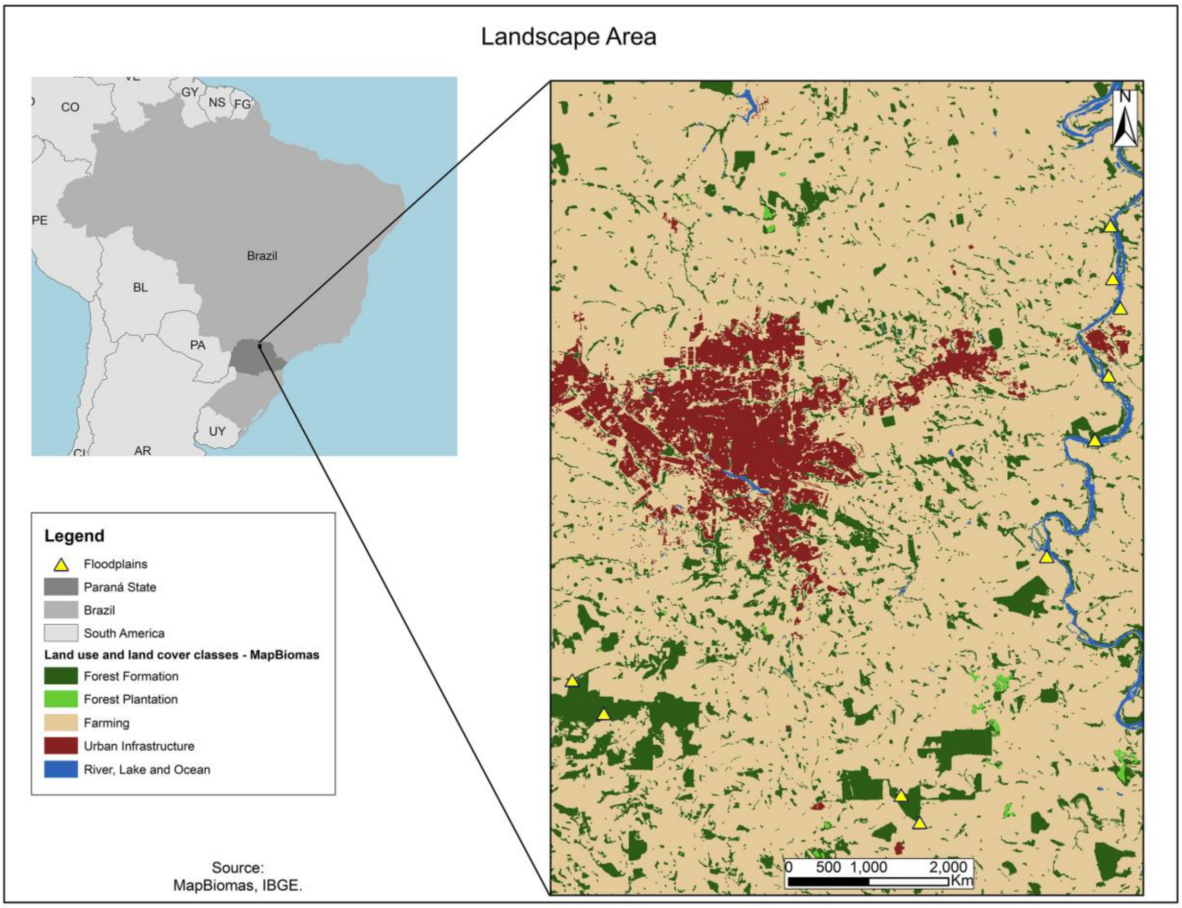
SUPPORT INFORMATION



Source:
MapBiomas, IBGE.

Figure S1 – Landscape area and distribution of floodplains associated to rivers (“várzeas”) sampled for amphibian in north Paraná region, south Brazil.

Table S1 – Amphibian species occurrence in each surveyed floodplain from North Paraná, between the rainy seasons of 2016 and 2018. + = Species occurence; • = Species detected in the surroundings but not within the floodplain habitat. Code names: MGN = Mata dos Godoys State Park, North region; MGS = Mata dos Godoys State Park, South region; PaN = Paiquerê remain, North region; PaS = Paiquerê remaint, South region; DoF = Doralice farm; ViF = Vitória farm; OdP = Odair's place; RaP = Rafaela's place; SLR = São Luís resort; CiP = Cido's place.

Family	Species	Floodplain										Occurence
		MGN	**MGS**	**PaN**	**PaS**	**DoF**	**ViF**	**OdP**	**RaP**	**SLR**	**CiP**	
Bufonidae	*Rhinella ornata*	•	•					+	+		+	3
	Rhinella diptycha		+			•	•	•	•		+	2
Centrolenidae	*Vitreorana uranoscopa*		+									1
Hylidae	*Aplastodiscus perviridis*		+		+							2
	Boana albopunctata	+	+		+		+					4
	Boana faber		+	+	+	•	+			+		5
	Boana prasina				+							1
	Boana raniceps							+	+	+	+	4
	Dendropsophus minutus		+	+		+	+			+		5
	Dendropsophus nanus		+	+		+	+	+	+	+	+	8
	Scinax fuscovarius			+		+		•				2
	Scinax perereca	+		+								2

	Trachycephalus typhonius		+			+						2
Hylodidae	*Crossodactylus schmidti*		+									1
Leptodactylidae	*Leptodactylus fuscus*		+	+		+	+	+	•		+	6
	Leptodactylus labyrinthicus	+	+	+	+							4
	Leptodactylus latrans					+	+					2
	Leptodactylus mystacinus	+	+			•	+					3
	Leptodactylus podicipinus			+		+	+	+	+	•	+	6
	Physalaemus cuvieri	+		+	+	+	+					5
Microhylidae	*Elachistocleis bicolor*					+	+	•				2
Odontophrynidae	*Proceratophrys avelinoi*	+	+		+							3
Phyllomedusidae	*Phyllomedusa tetraploidea*	•				+		+				2
Species richness	**23**	**6**	**13**	**9**	**7**	**10**	**10**	**6**	**4**	**4**	**6**	

Table S2 – Habitat use of each amphibian species, according to Machado & Bernardes (2002) and Figueiredo *et al*. (2019) and habitat specialization based on the habitat use. + = The species was detected in the respective habitat.

Family	**Species**	**Habitat use**			**Habitat specialization**
		Forest interior	Forest edge	Open areas	
Bufonidae	*Rhinella ornata*	+	+	+	True generalist
	Rhinella diptycha		+	+	Open area
Centrolenidae	*Vitreorana uranoscopa*	+			Forest
Hylidae	*Aplastodiscus perviridis*	+	+		Forest
	Boana albopunctata		+	+	Open area
	Boana faber		+	+	Open area
	Boana prasina		+	+	Open area
	Boana raniceps		+	+	Open area
	Dendropsophus minutus		+	+	Open area
	Dendropsophus nanu		+	+	Open area
	Scinax fuscovarius		+	+	Open area
	Scinax perereca		+		Forest
	Trachycephalus typhonius		+		Forest
Hylodidae	*Crossodactylus schmidti*	+			Forest
Leptodactylidae	*Leptodactylus fuscus*		+	+	Open area
	Leptodactylus labyrinthicus		+	+	Open area
	Leptodactylus latrans		+	+	Open area
	Leptodactylus mystacinus		+	+	Open area
	Leptodactylus podicipinus		+	+	Open area
	Physalaemus cuvieri		+	+	Open area
Microhylidae	*Elachistocleis bicolor*		+	+	Open area
Odontophrynidae	*Proceratophrys avelinoi*	+	+		Forest
Phyllomedusidae	*Phyllomedusa tetraploidea*	+	+	+	True generalist

Table S3 – Spearman correlations *ρ* between landscape variables within different landscape extents around each of 10 floodplains.

Scale (meters)		Edge density	Farming amount	Forest amount	Landscape diversity
300	Edge density	1			
	Farming amount	-0.81	1		
	Forest amount	0.94	-0.87	1	
	Landscape diversity	0.55	-0.19	0.31	1
1000	Edge density	1			
	Farming amount	-0.53	1		
	Forest amount	0.21	-0.82	1	
	Landscape diversity	0.3	-0.79	0.9	1
1500	Edge density	1			
	Farming amount	-0.52	1		
	Forest amount	0.15	-0.83	1	
	Landscape diversity	0.16	-0.84	0.98	1
2000	Edge density	1			
	Farming amount	-0.18	1		
	Forest amount	-0.07	-0.88	1	
	Landscape diversity	0.24	-0.94	0.83	1

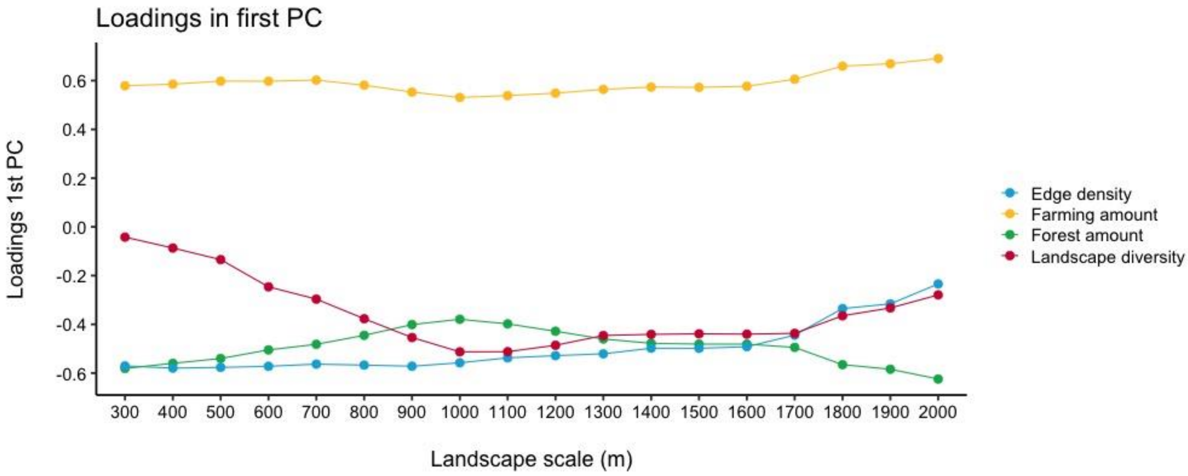


Figure S2 – Loading of each of the four landscape variables on the first principal component axis (1st PC). A separate principal component analysis was calculated at each spatial extent surrounding floodplains (N = 10).

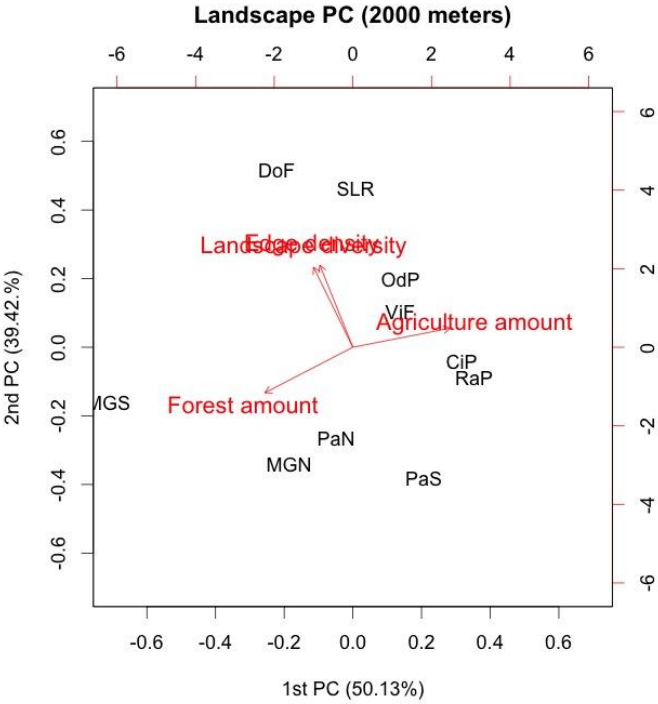


Figure S3 – Principal component analysis plot between the first (1st PC) and second components (2nd PC), considering the landscape variables at a landscape scale of 2000 meters. Amount of variance explained by each component in parentheses. Red arrows represent the loadings vectors for each landscape variable. Code names indicate the particular sample flood plain around which the landscape variables were measured: MGN = Mata dos Godoys State Park, North region; MGS = Mata dos Godoys State Park, South region; PaN = Paiquerê remain, North region; PaS = Paiquerê remaint, South region; DoF = Doralice farm; ViF = Vitória farm; OdP = Odair's place; RaP = Rafaela's place; SLR = São Luís resort; CiP = Cido's place.

Table S4 – Model selection based in the second order Akaike criteria (AICc), to evaluate the scale of effect of landscape variables over all amphibian species richness. Each model tested the influence of the first principal component from the respective landscape scale on total amphibian species richness. As models assumed a Poisson distribution, the intercept and estimate values are on a log scale. df = degree of freedom; ΔAICc = difference in AICc between the best model and the current model. Weight AICc = AICc model weight.

Landscape-scale	Intercept	Estimate	df	AICc	ΔAICc	Weight AICc	*pseudoR*^{2}	Dispertion parameter
2000	1.9809	-0.1860	2	48.31	0.00	0.16	0.42	0.71
1900	1.9821	-0.1822	2	48.57	0.26	0.14	0.37	0.75
1800	1.9830	-0.1807	2	48.78	0.47	0.13	0.30	0.84
1700	1.9865	-0.1683	2	49.44	1.13	0.09	0.44	0.69
1600	1.9874	-0.1620	2	49.67	1.36	0.08	0.25	0.91
1500	1.9898	-0.1562	2	50.08	1.77	0.07	0.20	0.97
1400	1.9943	-0.1431	2	50.79	2.48	0.05	0.50	0.62
1300	1.9980	-0.1296	2	51.35	3.04	0.04	0.55	0.56
1200	2.0009	-0.1189	2	51.79	3.48	0.03	0.53	0.59
800	2.0016	-0.1148	2	51.86	3.55	0.03	0.16	1.03
700	2.0018	-0.1132	2	51.88	3.58	0.03	0.15	1.05
500	2.0018	-0.1066	2	51.89	3.58	0.03	0.16	1.04
400	2.0021	-0.1040	2	51.94	3.63	0.03	0.20	0.99
300	2.0030	-0.0988	2	52.08	3.77	0.02	0.20	1.00
600	2.0040	-0.0994	2	52.22	3.91	0.02	0.16	1.04
900	2.0042	-0.1052	2	52.25	3.94	0.02	0.19	1.01
1100	2.0043	-0.1051	2	52.28	3.97	0.02	0.19	1.02
1000	2.0047	-0.1030	2	52.33	4.02	0.02	0.18	1.04

Table S5 – Model selection based in the second order Akaike criteria (AICc), to evaluate the scale of effect of landscape variables over forest-specialist amphibian species richness. Each model tested the influence of the first principal component from the respective landscape scale on forest-specialist amphibians species richness. As models assumed a Poisson distribution, the intercept and estimate values are on a log scale. df = degree of freedom; ΔAICc = difference in AICc between the best model and the current model. Weight AICc = AICc model weight.

Landscape-scale	Intercept	Estimate	df	AICc	ΔAICc	Weight AICc	*pseudoR*^{2}	Dispertion parameter
2000	0.0892	-0.4736	2	32.22	0.00	0.17	0.38	2.05
300	0.0159	-0.5355	2	32.55	0.33	0.14	0.20	2.14
1900	0.0969	-0.4690	2	32.66	0.44	0.13	0.17	2.11
1800	0.1019	-0.4716	2	33.05	0.83	0.11	0.16	2.16
400	0.0377	-0.5243	2	33.11	0.89	0.11	0.23	2.22
500	0.1039	-0.4616	2	34.41	2.19	0.06	0.28	2.03
1700	0.1397	-0.4348	2	34.52	2.30	0.05	0.40	1.57
1600	0.1437	-0.4275	2	34.87	2.65	0.04	0.43	1.41
1500	0.1513	-0.4274	2	35.26	3.04	0.04	0.38	1.71
600	0.1682	-0.3981	2	35.89	3.67	0.03	0.30	1.93
1400	0.1706	-0.4134	2	35.90	3.68	0.03	0.26	2.12
700	0.1894	-0.3850	2	36.39	4.17	0.02	0.20	2.21
1300	0.1923	-0.3874	2	36.52	4.30	0.02	0.12	2.09
800	0.2164	-0.3497	2	37.04	4.82	0.01	0.12	2.06
1200	0.2220	-0.3485	2	37.29	5.07	0.01	0.12	2.09
1100	0.2460	-0.3134	2	37.88	5.66	0.01	0.23	2.11
900	0.2504	-0.3017	2	37.93	5.71	0.01	0.31	2.12
1000	0.2530	-0.2986	2	38.01	5.78	0.01	0.41	1.80

Table S6 – Generalized linear models summaries, tested the influence of landscape context variables within 2000 m of 10 floodplains on total species richness and forest-specialist species richness of amphibians. Four landscape context variables - forest amount, farming amount, edge density, and landscape diversity - were combined into a single variable using a principal component analysis. As models assumed a Poisson distribution, the estimates and standard errors are shown on a log scale. PC = Principal component; AICc = Second order Akaike value; Weight AICc = AICc model weight.

Dependent variable	**Models summaries**								
		Estimate	Standard erro	*z*-value	*p*-value	AICc	Weight AICc	$PseudoR^2$	Dispersion parameter
Species total richness	(Intercept)	1.9809	0.1191	16.6355	< 0.0001				
	1st PC (2000m)	-0.1860	0.0762	-2.4401	0.0147	48.31	0.1610	0.4155	0.7075
Forest-specialist Species richness	(Intercept)	0.0892	0.3224	0.2768	0.7819				
	1st PC (2000m)	-0.4736	0.1602	-2.9559	0.0031	32.22	0.1670	0.3780	2.0466

4. CONCLUSÃO GERAL

Os resultados apresentados em ambos os capítulos da presente tese demostraram que a diminuição do habitat florestal de fato diminui a riqueza de espécies encontradas, independente da abordagem utilizada ("relação espécie-área" ou "montante de habitat"). Entretanto, eles também demonstram que é essencial que os estudos focados na conservação dos anfíbios não considerem apenas quanto de ambiente florestal disponível as espécies possuem, mas também qual o tipo de ambiente antrópico que cerca o remanescente.

No primeiro capítulo, os resultados mostraram que o tamanho do remanescente influenciou a riqueza de anfíbios, mas a matriz que o cerca pode influenciar o tamanho do efeito desta relação. O tipo de matriz explicou isoladamente cerca de um-quarto de toda a variação entre os estudos. Dentre estas, a matriz agrícola foi a que apresentou o menor tamanho de efeito e a maior homogeneidade entre estudos.

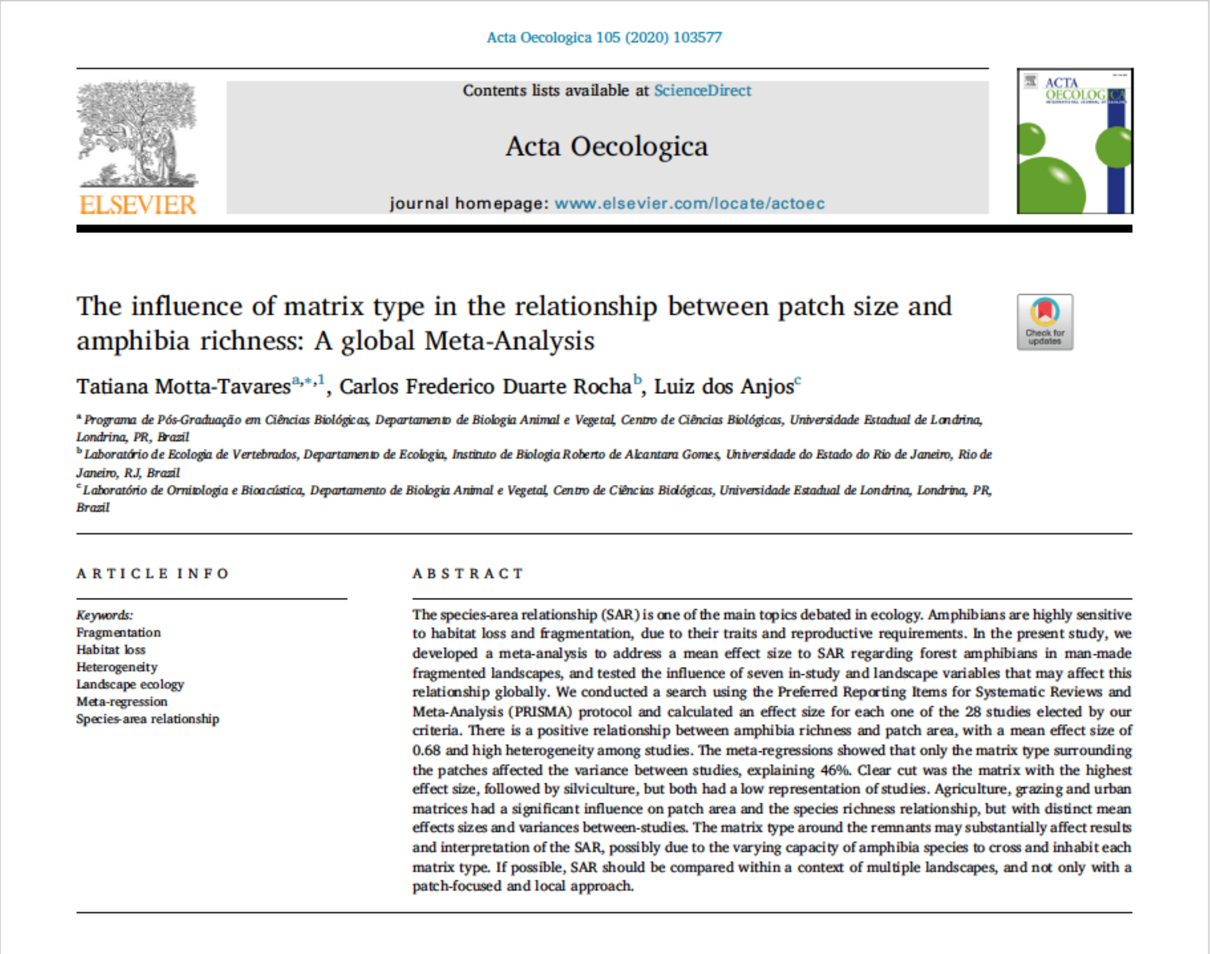
No segundo capítulo, a substituição do montante de floresta pelo montante de agricultura de fato explicou a diminuição da riqueza das espécies locais. A escala de efeito entre as variáveis da paisagem e a riqueza de anfíbios do norte do Paraná é cerca de 10 vezes maior do que se havia detectado em outros estudos na Mata Atlântica. A distância de efeito do montante de habitat (florestal ou agrícola) pode passar dos 2 km a partir do local da amostragem.

Os resultados mostram que as atuais leis ambientais no Brasil não contemplam a real necessidade de extensão para a real conservação dos anfíbios, por considerarem muito pouco os tipos de matriz que compõem os arredores das florestas. A lei nº 9.985/2000 (lei que rege o Sistema Nacional de Unidades de Conservação da Natureza) estabelece uma "zona de amortecimento" nos arredores de unidades de conservação, limitando os tipos de atividades que podem ser realizadas. Entretanto, a lei não estabelece um limite claro da zona de amortecimento, que deve ser estabelecido pelos gestores ambientais envolvidos com a Unidade de Conservação via lei especifica. A lei também não discrimina quais os tipos de atividades que podem ser realizadas no entorno das unidades de conservação, cabendo aos orgãos gestores das unidades definir o que pode ser feito ou não a partir do processo de licenciamento ambiental do proprietário das terras (Resolução CONAMA nº 428, de 17 de dezembro de 2010). Além disso, a lei não inclui os remanescentes florestais situados em propriedades privadas (Reserva Legal). Já a lei nº 12.651/2012 (“Novo Código florestal”) estabelece que várzeas são zonas de proteção integral, porém estabelece que a zona de proteção nos arredores dos corpos d’água seja limitada entre 30 até 500 metros da borda do curso. A distância máxima de proteção é 25% da distância mínima dos efeitos da paisagem sobre as populações de anfíbios nas várzeas do norte do Paraná.

Considerando que o foco de um gestor ambiental seja a proteção e conservação de anfíbios, sugerimos ser essencial a proteção do máximo de habitat florestal possível nos arredores dos sítios reprodutivos. É essencial também a gestão da paisagem nos arredores das florestas, uma vez que o tipo de matriz afeta as espécies encontradas na floresta e em pontos reprodutivos. Este manejo pode se dar a partir da elaboração de leis específicas, que prezem a conservação da biodiversidade local como um todo, e não apenas no que é encontrado no interior dos remanescentes.

ANEXOS

ANEXO A
Página de rosto do manuscrito aceito no periódico *Acta Oecologica*



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The influence of matrix type in the relationship between patch size and amphibia richness: A global Meta-Analysis

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ARTICLE INFO

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Landscape ecology
Meta-regression
Species-area relationship

ABSTRACT

The species-area relationship (SAR) is one of the main topics debated in ecology. Amphibians are highly sensitive to habitat loss and fragmentation, due to their traits and reproductive requirements. In the present study, we developed a meta-analysis to address a mean effect size to SAR regarding forest amphibians in man-made fragmented landscapes, and tested the influence of seven in-study and landscape variables that may affect this relationship globally. We conducted a search using the Preferred Reporting Items for Systematic Reviews and Meta-Analysis (PRISMA) protocol and calculated an effect size for each one of the 28 studies elected by our criteria. There is a positive relationship between amphibia richness and patch area, with a mean effect size of 0.68 and high heterogeneity among studies. The meta-regressions showed that only the matrix type surrounding the patches affected the variance between studies, explaining 46%. Clear cut was the matrix with the highest effect size, followed by silviculture, but both had a low representation of studies. Agriculture, grazing and urban matrices had a significant influence on patch area and the species richness relationship, but with distinct mean effects sizes and variances between-studies. The matrix type around the remnants may substantially affect results and interpretation of the SAR, possibly due to the varying capacity of amphibia species to cross and inhabit each matrix type. If possible, SAR should be compared within a context of multiple landscapes, and not only with a patch-focused and local approach.

ANEXO B

Normas de formatação do periódico *Acta Oecologica*

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Subdivision - numbered sections

Divide your article into clearly defined and numbered sections. Subsections should be numbered 1.1 (then 1.1.1, 1.1.2, ...), 1.2, etc. (the abstract is not included in section numbering). Use this numbering also for internal cross-referencing: do not just refer to 'the text'. Any subsection may be given a brief heading. Each heading should appear on its own separate line.

Introduction

State the objectives of the work and provide an adequate background, avoiding a detailed literature survey or a summary of the results.

Material and methods

Provide sufficient details to allow the work to be reproduced by an independent researcher. Methods that are already published should be summarized, and indicated by a reference. If quoting directly from a previously published method, use quotation marks and also cite the source. Any modifications to existing methods should also be described.

Theory/calculation

A Theory section should extend, not repeat, the background to the article already dealt with in the Introduction and lay the foundation for further work. In contrast, a Calculation section represents a practical development from a theoretical basis.

Results

Results should be clear and concise.

Discussion

This should explore the significance of the results of the work, not repeat them. A combined Results and Discussion section is often appropriate. Avoid extensive citations and discussion of published literature.

Conclusions

The main conclusions of the study may be presented in a short Conclusions section, which may stand alone or form a subsection of a Discussion or Results and Discussion section.

Appendices

If there is more than one appendix, they should be identified as A, B, etc. Formulae and equations in appendices should be given separate numbering: Eq. (A.1), Eq. (A.2), etc.; in a subsequent appendix, Eq. (B.1) and so on. Similarly for tables and figures: Table A.1; Fig. A.1, etc.

Essential title page information

- ***Title.*** Concise and informative. Titles are often used in information-retrieval systems. Avoid abbreviations and formulae where possible.

- ***Author names and affiliations.*** Please clearly indicate the given name(s) and family name(s) of each author and check that all names are accurately spelled. You can add your name between parentheses in your own script behind the English transliteration. Present the authors' affiliation addresses (where the actual work was done) below the names. Indicate all affiliations with a lower-case superscript letter immediately after the author's name and in front of the appropriate address. Provide the full postal address of each affiliation, including the country name and, if available, the e-mail address of each author.

- ***Corresponding author.*** Clearly indicate who will handle correspondence at all stages of refereeing and publication, also post-publication. This responsibility includes answering any future queries about Methodology and Materials. **Ensure that the e-mail address is given and that contact details are kept up to date by the corresponding author.**

- ***Present/permanent address.*** If an author has moved since the work described in the article was done, or was visiting at the time, a 'Present address' (or 'Permanent address') may be indicated as a footnote to that author's name. The address at which the author actually did the work must be retained as the main, affiliation address. Superscript Arabic numerals are used for such footnotes.

Keywords

Immediately after the abstract, provide a maximum of 6 keywords, using American spelling and avoiding general and plural terms and multiple concepts (avoid, for example, 'and', 'of'). Be sparing with abbreviations: only abbreviations firmly established in the field may be eligible. These keywords will be used for indexing purposes.

Abbreviations

Define abbreviations that are not standard in this field in a footnote to be placed on the first page of the article. Such abbreviations that are unavoidable in the abstract must be defined at their first mention there, as well as in the footnote. Ensure consistency of abbreviations throughout the article.

Acknowledgements

Collate acknowledgements in a separate section at the end of the article before the references and do not, therefore, include them on the title page, as a footnote to the title or otherwise. List here those individuals who provided help during the research (e.g., providing language help, writing assistance or proof reading the article, etc.).

Formatting of funding sources

List funding sources in this standard way to facilitate compliance to funder's requirements:

Funding: This work was supported by the National Institutes of Health [grant numbers xxxx, yyyy]; the Bill & Melinda Gates Foundation, Seattle, WA [grant number zzzz]; and the United States Institutes of Peace [grant number aaaa].

It is not necessary to include detailed descriptions on the program or type of grants and awards. When funding is from a block grant or other resources available to a university, college, or other research institution, submit the name of the institute or organization that provided the funding.

If no funding has been provided for the research, please include the following sentence:

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

Artwork

Electronic artwork

General points

- Make sure you use uniform lettering and sizing of your original artwork.
- Preferred fonts: Arial (or Helvetica), Times New Roman (or Times), Symbol, Courier.
- Number the illustrations according to their sequence in the text.
- Use a logical naming convention for your artwork files.
- Indicate per figure if it is a single, 1.5 or 2-column fitting image.
- For Word submissions only, you may still provide figures and their captions, and tables within a single file at the revision stage.
- Please note that individual figure files larger than 10 MB must be provided in separate source files.

A detailed guide on electronic artwork is available.

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Regardless of the application used, when your electronic artwork is finalized, please 'save as' or convert the images to one of the following formats (note the resolution requirements for line drawings, halftones, and line/halftone combinations given below):

EPS (or PDF): Vector drawings. Embed the font or save the text as 'graphics'.

TIFF (or JPG): Color or grayscale photographs (halftones): always use a minimum of 300 dpi.

TIFF (or JPG): Bitmapped line drawings: use a minimum of 1000 dpi.

TIFF (or JPG): Combinations bitmapped line/half-tone (color or grayscale): a minimum of 500 dpi is required.

Please do not:

- Supply files that are optimized for screen use (e.g., GIF, BMP, PICT, WPG); the resolution is too low.
- Supply files that are too low in resolution.
- Submit graphics that are disproportionately large for the content.

Color artwork

Please make sure that artwork files are in an acceptable format (TIFF (or JPEG), EPS (or PDF), or MS Office files) and with the correct resolution. If, together with your accepted article, you submit usable color figures then Elsevier will ensure, at no additional charge, that these figures will appear in color online (e.g., ScienceDirect and other sites) regardless of whether or not these illustrations are reproduced in color in the printed version. **For color reproduction in print, you will receive information regarding the costs from Elsevier after receipt of your accepted article**. Please indicate your preference for color: in print or online only. Further information on the preparation of electronic artwork.

Figure captions

Ensure that each illustration has a caption. A caption should comprise a brief title (**not**on the figure itself) and a description of the illustration. Keep text in the illustrations themselves to a minimum but explain all symbols and abbreviations used.

Tables

Please submit tables as editable text and not as images. Tables can be placed either next to the relevant text in the article, or on separate page(s) at the end. Number tables consecutively in accordance with their appearance in the text and place any table notes below the table body. Be sparing in the use of tables and ensure that the data presented in them do not duplicate results described elsewhere in the article. Please avoid using vertical rules and shading in table cells.

References

Citation in text

Please ensure that every reference cited in the text is also present in the reference list (and vice versa). Any references cited in the abstract must be given in full. Unpublished results and personal communications are not recommended in the reference list, but may be mentioned in the text. If these references are included in the reference list they should follow the standard reference style of the journal and should include a substitution of the publication date with either 'Unpublished results' or 'Personal communication'. Citation of a reference as 'in press' implies that the item has been accepted for publication.

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Increased discoverability of research and high quality peer review are ensured by online links to the sources cited. In order to allow us to create links to abstracting and indexing services, such as Scopus, CrossRef and PubMed, please ensure that data provided in the references are correct. Please note that incorrect surnames, journal/book titles, publication year and pagination may prevent link creation. When copying references, please be careful as they may already contain errors. Use of the DOI is highly encouraged.

A DOI is guaranteed never to change, so you can use it as a permanent link to any electronic article. An example of a citation using DOI for an article not yet in an issue is: VanDecar J.C., Russo R.M., James D.E., Ambeh W.B., Franke M. (2003). Aseismic continuation of the Lesser Antilles slab beneath northeastern Venezuela. Journal of Geophysical Research, https://doi.org/10.1029/2001JB000884. Please note the format of such citations should be in the same style as all other references in the paper.

Web references

As a minimum, the full URL should be given and the date when the reference was last accessed. Any further information, if known (DOI, author names, dates, reference to a source publication, etc.), should also be given. Web references can be listed separately (e.g., after the reference list) under a different heading if desired, or can be included in the reference list.

ANEXO c

Normas de formatação do periódico *Biotropica*

Disponível no endereço:

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1 Assembling your Manuscript:

Assemble manuscripts in this order:

1. Title page
2. Abstract (s)
3. Keywords
4. Text
5. Tables
6. Figure legends
7. Figures
8. Acknowledgments
9. Disclosure Statements
10. References
11. Supplementary Information (to be supplied as separate files)

1. TITLE PAGE

Running Heads: The authors' family name should be included as left and right running heads. It is set in small caps. The format is as follows:

LRH and RRH: YAZ and PEIGH

(may not exceed 50 characters, two or more authors use YAZ et al.)

Title: No more than 12 words (usually), flush left, near the middle of the page. Use Bold Type.

Where species names are given in the title, it should be clear to general readers what type(s) of organism(s) are being referred to, either by using Family appellation or common name:

'Invasion of African savanna woodlands by the Jellyfish tree *Medusagyne oppositifolia*', **OR** 'Invasion of African savanna woodlands by *Medusagyne oppositifolia* (Medusagynaceae)'

Titles that include a **geographic locality** should make sure that this is clear to the general reader:

'Effect of habitat fragmentation on pollination networks on Flores, Indonesia', **NOT**

'Effect of habitat fragmentation and pollination networks on Flores'.

Authors: Below title, include the author(s) full name(s), affiliation(s), and unabbreviated complete address(es). Use superscript number(s) following author(s) name(s) to indicate current location(s) if different than above. In multi-authored papers, additional footnote superscripts may be used to indicate the corresponding author and e-mail address. Although geographical place names should use the English spelling in the text (e.g., Zurich, Florence, Brazil), authors may use their preferred spelling when listing their affiliation (e.g., Zürich, Firenze, Brasil).

Submission and Acceptance Dates: At the bottom of the title page every article must include: Received: ______; Revised:____(optional); Accepted:______. (*Biotropica* will fill in the dates.)

2. ABSTRACT PAGE

Abstracts have maximum of 250 words for papers and reviews and 50 words for Insights. There is no abstract for Commentary papers.

The Abstract should include brief statements about the intent or purpose, materials and methods, results, and significance of findings. Abstract can be given as multiple paragraphs (with subheadings such as Aim, Methods, Results, and Conclusion) or as a single paragraph. Do not use abbreviations in the abstract.

Authors are strongly encouraged to provide a second abstract in the language relevant to the country in which the research was conducted. The second abstract will be published in the online versions of the article. This second abstract should follow the first abstract.

3. KEYWORDS

Provide up to eight keywords after the abstract, separated by a comma (,). Keywords should be in English (with the exception of taxonomic information) and listed alphabetically.

Include the location of the study as a key word if it is not already mentioned in the title (see example below). Key words should *not* repeat words used in the title. Avoid words that are too broad or too specific. (e.g., keywords: Melastomataceae, *Miconia argentea*, Panama, seed dispersal, tropical wet forest).

4. TEXT

Headings

Main headings are 1. INTRODUCTION, 2. METHODS, 3. RESULTS, and 4. DISCUSSION in bold, capital letters, numbered, and flush left.

Indent all but the first paragraph of each section.

Leave one blank between main heading and text.

Second level headings should be in Initial caps, bold, numbered, and flush left. (e.g., **2. Inventory technique.**)

First three headings are numbered and fourth and fifth order headings are unnumbered.

Insights submissions do not use any subject headings.

When using previously published data in analyses please cite both the data archive(s) and the original manuscript(s) for which they were collected in the text: “We used previously archived data (Bruna et al., 2011a,b) in our simulations.”, where a is the data archive and b is the publication. Be sure both citations are included in the literature cited.

Do not use footnotes in the main text.

Refer to figures as ‘Figure 1’, and tables as ‘Table 1’. Reference to online Supporting Information is referred to as ‘Figure S1’ or ‘Table S1’.

Units, Abbreviations, and style

Abbreviations: year(s), month(s), week(s), day(s), hr, min, s, km, cm, mm, ha, kg, g, L, g/m2

Units: Use solidus style for simple units (e.g., m/s) and follow negative indices style for compound units (e.g., $\mathrm{nmol} \cdot \mathrm{hr}^{-1} \cdot \mathrm{mg}^{-1}$)

Write out other abbreviations the first time they are used in the text and abbreviate thereafter: "El Niño Southern Oscillation (ENSO) . . ."

Numbers: Write out one to 9 unless a measurement or in combination with other numbers: four trees, 6 mm, 35 sites, 7 year, 10 × 5 m, 7 m, ± *SE,* 5 bees and 12 wasps).

Use a comma as a separator in numbers with four or more digits: 1,000 vs. 10,000

Decimals: 0.13 (leading zero and points, never commas)

Temperature: 21°C (no space after the degree symbol)

Use dashes to indicate a set location of a given size (*e.g.,* 1-ha plot).

Spell out 'percent' when used at the beginning of a sentence and use symbols when used in number combinations (e.g., "there was a 5% increase…", "plants were grown at high light levels (20%)…", 95% CI.)

Statistical abbreviations:

- Use italics for *P, N, t, F, R2, r, G, U, N,* χ^2 (italics, superscripts non-italics)
- Use italic for: *df, SD, SE, SEM*
- Use roman for CI, two-way ANOVA, ns

Dates: 10 December 1997

Times: 0930 h, 2130 h

Latitude and Longitude: 10°34′21″ N, 14°26′12″ W

Above sea level: a.s.l.

Regions: SE Asia, UK, USA (no periods)

Geographical place names should use the English spelling in the text (Zurich, Florence, Brazil), but authors may use their preferred spelling when listing their affiliation (Zürich, Firenze, Brasil).

Lists: Follow the style**…** (a)… ; (b)…; and (c)…: "The aims of the study were to: (a) evaluate pollination success in *Medusagyne oppositifolia*; (b) quantify gene flow between populations; and (c) score seed set."

5. TABLES

While Biotropica does have word limits that differ by manuscript category, there are not have strict limits on the number of tables and/or figures. However, printed manuscripts rarely exceed 32 pages in length, and we encourage authors to submit only necessary tables and figures. Additional information, figures, and tables should appear in the Supporting Information."

Each table must start on a separate page

Number tables with Arabic numerals followed by a period. Capitalize ‘Table’ (e.g., Table 1, Table 2, etc.).

Indicate footnotes by lowercase superscript letters

Do not use vertical lines in tables.

6. FIGURE LEGENDS (Continue page numbering)

Type figure legends in paragraph form, starting with ‘Figure’ and number.

Do not include symbols (lines, dots, triangles, etc.) in figure legends; either label them in the figure or refer to them by name in the legend.

Label multiple plots/images within one figure as a, b, c etc., and please ensure the panels of each plot include these labels and are referred to in the legend (e.g., Figure 1 Fitness of *Medusagyne oppositifolia*as indicated by (a) seed set and (b) seed viability’, making sure to include the labels in the relevant plot.)

7. FIGURES

ATBC members can publish graphs and other figures of results in color at no additional charge. Please make sure these figures are accessible by following our **Figures Guidelines**.

Please consult Wiley Author Services’ **figures and illustrations guide** (PDF) for more detailed information about submitting electronic artwork. Authors are encouraged to utilize online Supporting Information for tables and figures that do not have central importance to the manuscript.

All figures and photographs are referred to as ‘Figures’ in the text.

If it is not possible to submit figures embedded within the text file, then submission as *.pdf, *.tif or *.eps files is permissible.

Native file formats (Excel, DeltaGraph, SigmaPlot, etc.) cannot be used in production. When your manuscript is accepted for publication, for production purposes authors will be asked upon acceptance of their papers to submit:

- Line artwork (vector graphics) as *.eps, with a resolution of > 300 dpi at final print size
- Bitmap files (halftones or photographs) as *.tif or *.eps, with a resolution of >300 dpi at final size

Final figures will be reduced. To ensure all text will be legible when reduced to the appropriate size use large legends and font sizes. We recommend using Arial for labels within figures without bolding text.

Do not use negative exponents in figures, including axis labels.

Each plot/image grouped in a figure or plate requires a label (*e.g.*, a, b). Use lower case letters on grouped figures, and in text references.

8. ACKNOWLEDGEMENTS

Authors are encouraged to acknowledge funding, general supervision of the research group, or general support, in addition to any writing assistance, technical editing, language editing, and proofreading provided outside of the typical production process.

10. REFERENCES (follow APA Style Manual, 6th Edition)

We strongly recommend using reference management software such as Zotero or Endnote to simplify building the literature cited and to minimize mistakes.

- Citations of manuscripts as ‘in ’ or ‘submitted’ or ‘in progress’ are acceptable – similar to articles published, ‘in press’, or that have been deposited in pre-print archives (include DOI). Articles or book chapters cited as ‘In press’ must be accepted for publication; please include the journal or publisher.

Verify all entries against original sources, especially journal titles, accents, diacritical marks, and spelling in languages other than English.

When using data archives in the paper, cite both the data archive and the original manuscript using the following format:

Bruna, E. M., Izzo, T. J., Inouye, B. D., Uriarte, M., & Vasconcelos, H. L. (2011a). Data from: Asymmetric dispersal and colonization success of Amazonian plant-ants queens. Dryad Digital Repository. **http://dx.doi.org/10.5061/dryad.h6t7g**

Bruna, E. M., Izzo, T. J., Inouye, B. D., Uriarte, M., & Vasconcelos, H. L. (2011b). Asymmetric dispersal and colonization success of Amazonian plant-ants queens. PLoS ONE 6, e22937.

When using datasets, use:

[dataset]Authors; Year; Dataset title; Data repository or archive; Version (if any); Persistent identifier (e.g. DOI)

The term [Dataset] will be removed before publication.

Cite references in alphabetical order by first author's surname. References by a single author precede multi-authored works by the same senior author, regardless of date.

List works by the same author chronologically, beginning with the earliest date of publication.

Insert a period and space after each initial of an author's name; example: Yaz, A. B., & Azy, B. (1980).

Authors Names should be in Title Case and every reference should spell out author names.

Use journal name in expanded form. To find a journal, please search the NIH's **list**.

Double-space all citations with a hanging indent of 0.5 inch.

Leave a space between the volume number and page numbers and include issue numbers (if available). 27, 3–12

Article inbooks: Azy, A. B. (1982). Title of book chapter. In: G. Yaz (Ed.). *Book title* (pp. 24–36). Boca Raton, FL: CRC Press.

For theses and dissertations: 'PhD Dissertation' and 'MSc Dissertation'.
Author, A. A. (2003). *Title of doctoral dissertation or master's thesis* (Doctoral dissertation or master's thesis). Retrieved from Name of database. (Accession or Order No.)

2 SUPPLEMENTARY INFORMATION

Supporting Information (SI) accompanies the online version of a manuscript and will be fully accessible to everyone with electronic access to *Biotropica*. We urge authors to make use of the SI section to add context or additional information that do not have central relevance to the manuscript. **All such material must be cited in the text of the printed manuscript.**

We ask authors to place maps of field sites and figures and tables as online Supporting Information unless they also present the results of analyses. The SI can also be used for species lists, detailed technical methods, photographs, mathematical equations and models, or additional references from which data for figures or tables have been derived (*e.g.*, in a review paper).