Programa de Pós-Graduação em
ECOLOGIA E EVOLUÇÃO
ICB - UFG

Universidade Federal de Goiás

Instituto de Ciências Biológicas

PPG em Ecologia e Evolução

UFG

A CONSERVAÇÃO DA DIVERSIDADE DE ANUROS NO CERRADO BRASILEIRO

Luciana Signorelli Faria Lima

Goiânia – 2014

UNIVERSIDADE FEDERAL DE GOIÁS

INSTITUTO DE CIÊNCIAS BIOLÓGICAS

PPG EM ECOLOGIA E EVOLUÇÃO

LUCIANA SIGNORELLI FARIA LIMA

TESE DE DOUTORADO

A CONSERVAÇÃO DA DIVERSIDADE DE ANUROS NO CERRADO BRASILEIRO

ORIENTADOR: PROF. DR. ROGÉRIO PEREIRA BASTOS

CO-ORIENTADOR: PROF. DR. PAULO DE MARCO JR.

ORIENTADOR NO EXTERIOR: DRA. KIMBERLY WITH

Goiânia ï Goiás

Junho de 2014

Programa de Pós-Graduação em
ECOLOGIA E EVOLUÇÃO
ICB - UFG

UNIVERSIDADE FEDERAL DE GOIÁS

INSTITUTO DE CIÊNCIAS BIOLÓGICAS

PPG EM ECOLOGIA E EVOLUÇÃO

UFG

LUCIANA SIGNORELLI FARIA LIMA

TESE DE DOUTORADO

A CONSERVAÇÃO DA DIVERSIDADE DE ANUROS NO CERRADO BRASILEIRO

ORIENTADOR: PROF. DR. ROGÉRIO PEREIRA BASTOS

CO-ORIENTADOR: PROF. DR. PAULO DE MARCO JR.

ORIENTADOR NO EXTERIOR: DRA. KIMBERLY WITH

TESE APRESENTADAÀ UNIVERSIDADE FEDERAL DE GOIÁS, COMO PARTE DAS EXIGÊNCIAS DO PROGRAMA DE PÓS-GRADUAÇÃO EM ECOLOGIA E EVOLUÇÃO PARA OBTENÇÃO DO TÍTULO DE DOUTOR EM ECOLOGIA E EVOLUÇÃO.

Goiânia ï Goiás

Junho de 2014

Dados Internacionais de Catalogação na Publicação (CIP)
GPT/BC/UFG

Lima, Luciana Signorelli Faria.
L732c A Conservação da diversidade de anuros no cerrado brasileiro [manuscrito]/ Luciana Signorelli Faria Lima. - 2014.
157 f. : figs, tabs.

Orientador: Prof. Dr. Rogério Pereira Bastos.
Tese (Doutorado) – Universidade Federal de Goiás, Instituto de Ciência Biológicas, 2014.
Bibliografia.

1. Anuros – Cerrado 2. Aspectos ecológicos 3. Bioma Cerrado. I. Título.

CDU: 597.8(81)

Luciana Signorelli Faria Lima

A CONSERVAÇÃO DA DIVERSIDADE DE ANUROS NO CERRADO BRASILEIRO

Dr Fausto Nomura
Universidade Federal de Goiás

Dr Natan Medeiros Maciel
Universidade Federal de Goiás

Dr Arthur Angelo Bispo de Oliveira
Universidade Federal de Goiás

Dr Franco L. Souza
Universidade Federal de Mato Grosso do Sul

Dra. Márcio José da Silveira
Universidade Estadual de Maringá(Suplente)

Dr. Daniel de Brito C. da Silva
Universidade Federal do Goiás
(Suplente)

Dr. Mário Almeida Neto
Universidade Federal de Goiás
(Suplente)

Prof. Dr. Rogério Pereira Bastos
Universidade Federal de Goiás
(Orientador)

*ñIt is not the critic who counts; not the man who points out how the strong man stumbles, or where the doer of deeds could have done them better. The credit belongs to the man who is actually in the arena, whose face is marred by dust and sweat and blood, who strives valiantly; who errs and comes short again and again; because there is not effort without error and shortcomings; but who does actually strive to do the deed; who knows the great enthusiasm, the great devotion, who spends himself in a worthy cause, who at the best knows in the end the triumph of high achievement and who at the worst, if he fails, at least he fails while daring greatly. So that his place shall never be with those cold and timid souls who know neither victory nor defeat.***ò**

Theodore Roosevelt

AGRADECIMENTOS

Inúmeras foram as pessoas e instituições que contribuíram de alguma forma para a conclusão desta tese de doutorado. Por quase dez anos tive a honra e o privilégio de ter sido orientada pelo professor Rogério Bastos. Sou grata pela sua orientação, paciência, estrutura, equipamentos fornecidos, oportunidades de trabalho, incentivo e, até mesmo, pelas broncas! Sei que todas as broncas foram para o meu bem ou porque eu estava perdendo o foco ao tentar abraçar o mundo. Devo a ele a minha formação como pesquisadora e como herpetóloga. Sempre preocupado com o meu futuro, minha carreira acadêmica e meu bem estar. Obrigada por todas as vezes que você me chamou em sua sala para conversar, pois sabia que as coisas não estavam indo bem. Sei que ainda tenho que crescer e amadurecer muito. Sei também sobre um dos meus principais defeitos e da sua preocupação sobre como eu irei resolvê-los. Só tenho a agradecer a você!

Agradeço ao professor Paulo De Marco Junior pelas oportunidades de trabalho, pela orientação e por ter acreditado em meu trabalho. Apesar das minhas dificuldades e do desespero, você sempre me incentivou e me estimulou a crescer e a buscar novos horizontes. É um exemplo de pesquisador e de pessoa. Só tenho a agradecer a oportunidade de trabalhar com você e com seu grupo de pesquisa. Na verdade, é uma honra e um privilégio ser um dos membros dele.

Agradeço aos professores do Programa de Pós-Graduação em Ecologia e Evolução da Universidade Federal de Goiás pelos ensinamentos. São doutores de alta qualidade, que são responsáveis por boa parte dos meus conhecimentos atuais.

Agradeço também ao PROCAD (Programa de Intercambio entre Universidades) que me permitiu conhecer outra instituição de Ensino e um novo grupo de pesquisa, algo que foi fundamental para meu processo de formação e maturidade pessoal e profissional. Foram seis meses que passei em Maringá, tendo o privilégio de frequentar o laboratório do professor Sidinei Magela Lá pude discutir sobre trabalhos científicos, projetos e metodologia científica, além de participar ativamente de coletas de campo e das atividades do laboratório de Macrófita Aquáticas. Além disso, agradeço a hospitalidade do professor Nei, daprofessora Claudinha, da técnica Maria Do Carmo, do professor Felipe e de inúmeros funcionários diretos ou indiretos do Nupélia (Núcleo de Pesquisas em Limnologia, Ictiologia e Aquicultura), que fizeram da minha estadia muito mais agradável e tranquila. Agradeço aos colegas do Laboratório de Macrófitas Aquáticas pelo carinho, pela amizade e pelas inúmeras discussões. Agradeço à Helo, à Cris, ao João, ao Marcinho, ao Du, ao Roger, ao Fabrício Oda e à Thaísa, pelo carinho e pelas inúmeras conversas produtivas (e improdutivas, rs). São grandes amigos e que sempre serão lembrados com muito carinho.

Aos professores Fausto e Denise por terem me convidado para participar do projeto ñGirinos do Brasilò(Edital SISBIOTA, Processos FAPESP 2010/52321-7 e CNPq 563075/2010-4), que financiou parte das viagens de campo e pelo qual fui bolsista. Ao professor Paulo De Marco por ter me convidado para participar do projeto ñO Cerrado em pedaços: múltiplas respostas de componentes da biodiversidade à fragmentação e a perda de habitatsò(Fundação ñO Boticárioòde Proteção à Natureza, Processo 0880_201020110), que também disponibilizou recursos financeiros que possibilitou as atividades de campo. Ao Programa de Doutorado Sanduiche no Exterior (BEX 18618/12-0) pela bolsa concedida e pela oportunidade de trabalhar em uma instituição de renome internacional (*Kansas State University*) me concedendo a oportunidade de trabalhar com a Dra. Kimberly With, pesquisadora de renome na área de Ecologia da Paisagem.

Agradeço a Dra. Kimberly With por ter me ñadotadoò M*y American mother hen*, como ela mesma diz. A Kim me mostrou que os Americanos estão longe de serem pessoas frias e individualistas. Pelo contrário, desde que a conheci a Kim sempre esteve presente. Antes mesmo de conhecê-la pessoalmente, ela já dizia que faria tudo para tornar minha mudança o menos drástica e traumática possível. E assim ela a fez! A Kim me deu estrutura profissional e ofereceu toda a infra estrutura necessária para o desenvolvimento da tese. Além disso, fizemos inúmeras reuniões e discussões a respeito do nosso trabalho. Me mostrou um pouquinho da cultura americana e me fez sentir em casa inúmeras vezes. É um exemplo de profissional, de mãe e de mulher.

Agradeço aosprofessores Dr. Douglas Goodin (professor e diretor do Remote Sensing Research Laboratory da K-State University) e Dr Marcellus Caldas (professor do departamento de Geografia da K-State University) pela orientação em relação às análises de imagens de satélite e por terem me proporcionado espaço físico e equipamento com os programas ENVI e ArcGIS, necessários para classificação de imagens de satélite.

Agradeço ao professor Arthur e a Carol Costa por terem me disponibilizado imagens de satélite e os mapas de remanescentes de Cerrado. Obrigada por todas as nossas conversas, que foram extremamente produtivas e esclarecedoras, e pela orientação em relação a conceitos de Ecologia da Paisagem e em relação ao uso de alguns softwares.

Agradeço aos meus amigos e minha família americana Marina, Gabriella, Ana e Erick. Apesar de pertencermos a áreas diferentes, sempre contribuíram com sugestões quanto a escrita, estrutura de texto, sem contar no apoio moral para continuar e finalizar o trabalho.

Agradeço também aos amigos do Theory, Metacommunity and Landscape Ecology Lab, Poli, Pedro, Renas, Flavinha, Fran, Arthur, Eduardo, Carol, Fabão, Andressa, Andrezza, Camila, Fernanda, Karina, Camila, Cadu, Mirian, Paulinho, Thiago, Carol Costa, Albert, André, Klein, Carlos, Zander, Bruno, Ricardo, Flaviana, pelo companheirismo, sugestões e brincadeiras (em que muitas vezes apelei, confesso! rs).Agradeço em especial ao grande amigo Denis, que colaborou ativamente para o desenvolvimento da tese. Sempre com comentários e sugestões muito pertinentes, tendo paciência para discutir teorias, análises e possíveis trabalhos. Tenho certeza de que muitos trabalhos sairão desta parceria.

Agradeço a todos os integrantes do Laboratório de Herpetologia e Comportamento Animal (UFG) pelo auxílio em campo. Em especial gostaria de agradecer aos professores Natan e Fausto pelos comentários, sugestões,conselhos e ajuda em campo; e aos amigos Fernandinha, Muryllo (meu braço direito e esquerdo), Vinny, Pri Gambale e Renan, pelo companheirismo, sugestões, colaborações e amizade.

Agradeço a*Teacher* Ana Paula, que sempre me ajudou e incentivou. Esteve do meu lado desde o início do sonho do doutorado sanduíche e me convenceu de que os Estados Unidos seria uma das melhores opções para mim, tanto pela língua inglesa, quanto pela cultura. A Teacher me ajudou a construir cada pedacinho desse sonho. Como sempre disse, tive muito mais do que uma aula de Inglês. Sempre auto astral e otimista, me incentivando e me fazendo acreditar que sou capaz sim. Sou eternamente grata à minha *Teacher*, por ter sido professora, orientadora e amiga.

Aos amigos Ciências Sem fronteiras: Poli, Pedro e Renas. Já conquistamos a América. Obrigada por todo apoio, compreensão e amizade. Obrigada pelas leituras de textos, e-mails e sugestões. Foram inúmeras manhãs, tardes e/ou noites que passamos juntos no lab para fecharmos as teses, dissertações ou projetos. Sonhamos juntos e juntos estamos concretizando nossos sonhos.

E, para finalizar, gostaria de agradecer à minha família. Eles são a minha base e minha referência. Sempre estiveram do meu lado, literalmente me aguentando nos momentos de maior nervosismo e vibrando nos momentos de maior felicidade. Como sempre me disseram: ña maior riqueza que podemos dar a vocês são os estudos e um educação de qualidadeò. A conclusão dessa tese nada mais é que uma consequência da educação que tive.

RESUMO

As atividades de uso da terra têm transformado grandes áreas naturais em áreas de pastagens ou agricultáveis. Este processo se tornou um problema mundial e é tido como um dos principais responsáveis pelo declínio de espécies de diferentes grupos taxonômicos. Dentre os vertebrados, os anfíbios são um dos grupos mais ameaçados, sendo que as espécies com larvas aquáticas são as mais susceptíveis. Tais populações seguem uma dinâmica de metacomunidades e podem estruturar-se de acordo com uma combinação de processos, conhecidos como dinâmica de manchas, dinâmica de ordenação de espécies, efeito de massa e dinâmica neutra. No entanto, a escassez de informações a respeito do padrão de riqueza e ocupação das espécies de anuros em escala local e regional é um problema para a conservação da diversidade dos mesmos. Tendo como principal objetivo fornecer informações relevantes para a conservação de anuros no Cerrado brasileiro, abordei questões relacionadas aos padrões em escala local e regional, bem como modelos de ocupação de algumas espécies de anfíbios com o intuito de identificar fatores que estão direcionando a riqueza e ocupação das espécies de anuros que se reproduzem em poças no Cerrado. Para isso, coletei dados no estado de Goiás, único estado brasileiro totalmente inserido no bioma Cerrado, e que segue a mesma tendência do restante do bioma em relação a perda de habitat devido ao avanço das fronteiras agrícolas. Com o intuito de cobrir lacunas de inventários para a região, no Capítulo I apresento a primeira lista oficial de espécies de anuros para o estado de Goiás. No Capítulo II, abordo como fatores locais e da paisagem determinam a diversidade local e regional de anuros. Busquei explorar os efeitos da área, heterogeneidade e complexidade de hábitats local e da paisagem e da produtividade sobre a diversidade alfa e beta de anuros. No Capítulo III¸ abordo um dos modelos mais clássicos e controversos para conservação das espécies, que é conhecido como: ñmuitas pequenas ou uma única grandeò ou ñSLOSSò (*single large or several small*). Este modelo deve ser especialmente considerado quando o objetivo é preservar o maior número de espécies de anuros associados a poças. E, por fim, no Capítulo IV, construí modelos de ocupação para ter acesso aos efeitos da quantidade de remanescentes de habitas, isolamento entre remanescentes de Cerrado, bem como características locais sobre a ocorrência de algumas espécies de anfíbios típicas do Cerrado Brasileiro.

Palavras-chave: Cerrado, anfíbios, poças, área, heterogeneidade, SLOSS, conservação, modelos de ocupação.

ABSTRACT

Land use activities have been frequently transforming broad native areas into pastures or plantations. This process turned out to be a global problem and is known as one major responsible for declinesin various taxonomical groups. Frogs are one of the most threated groups among vertebrates, from which species with aquatic larvae are more susceptible. Such populations follow metacommunity dynamics and can be structured in function of combined processes, such as patch dynamics, species ordination dynamics, mass effect and neutral dynamics. Nevertheless, the lack of knowledge with respect to anuran species occupation and richness patterns in local and regional scale poses as a threat to their conservation. The aim of my work is to provide relevant information to the conservation of anurans in the Brazilian Cerrado. I investigated regional and local scale patterns and identified factors related to richness and occupation of anuran species that breed in Cerrado ponds. For that, I have collected data in the state of Goiás, which is the only Brazilian state totally inserted in the Cerrado biome and that follows that same tendency of habitat loss as the whole biome (due to agriculture expansion). In Chapter I present the first official list for the whole state of Goiás, with the objective to cover inventory gaps. In Chapter II, I explore local and landscape factors that determine local and regional diversities of anurans. I also assess the effects of area, heterogeneity, productivity and local and landscape habitat complexity over alpha and beta diversities of anurans. In Chapter III, I approach one of the most classic and controversial models for the conservation of the species, known as "SLOSS" (single large or several small). This model should be considered especially when the goal is to preserve as many frog species associated with ponds.Finally, in Chapter IV, I built occupation models to assess the effects of amount of remaining habitats, isolation between remnants as well as local characteristics on the occurrence of some species of amphibians typical Brazilian Cerrado.

Keywords: Cerrado, amphibians, puddles, area, heterogeneity, SLOSS, conservation, occupancy models.

SUMÁRIO

APRESENTAÇÃO

A presente tese, intitulada ñA Conservação da Diversidade de Anuros no Cerrado Brasileiroò, está apresentada sob a forma de quatro capítulos, sendo que cada capítulo corresponde a um artigo científico. Os dados utilizados para elaboração dessa tese foram coletados por especialistas do Laboratório de Herpetologia e Comportamento Animal da Universidade Federal de Goiás. As coletas de campo foram financiadas pelos projetos ñGirinos do Brasilò (Edital SISBIOTA, Processos FAPESP 2010/52321-7 e CNPq 563075/2010-4), ñO Cerrado em pedaços: múltiplas respostas de componentes da biodiversidade à fragmentação e a perda de habitatsò (Fundação Grupo Boticário de Proteção à Natureza, Processo 0880_201020110). Uma seção intitulada ñIntrodução Geralò apresenta as questões gerais e conceitos ecológicos que motivaram a elaboração desta tese, bem como uma breve apresentação das principais hipóteses testadas. O capítulo I, intitulado ñDiversity of amphibian anurans from state of Goiás, core of Brazilian Cerradoò será submetido para apreciação na revista ñZookeysò. O capítulo II, intitulado ñMore Individual Hypothesis and Habitat Heterogeneity Drive Amphibian Ponds Metacommunitiesò será submetido para a revista ñPlos oneò. O capítulo III é intitulado: ñAnuran Conservation in Brazilian Cerrado: single large or several small ponds, which is better?ò será submetido para a revista ñBiological Conservationò. O Capítulo IV, intitulado ñFactors Affecting Landscape Occupancy for Anurans Across a Disturbance Gradient in the Brazilian Cerradoò será submetido para apreciação na revista ñLandscape Ecologyò. Após a apresentação dos capítulos, uma seção intitulada ñConclusões Geraisò apresenta as principais conclusões da tese e implicações para a conservação da diversidade de Anuros no Cerrado Brasileiro.

INTRODUÇÃO GERAL

Atividades de uso da terra tornaram-se um problema mundial a partir do momento em que grandes áreas de paisagens naturais foram transformadas em áreas antrópicas e/ou agricultáveis [1], dando lugar a mosaicos com diferentes tipos de uso do solo [2]. Esse processo, também conhecido como fragmentação de habitat, está relacionada à quatro padrões principais: a perda de habitat, o aumento no número de manchas, a redução no tamanho das manchas e o aumento no isolamento entre manchas. Em termos de biodiversidade, os efeitos da fragmentação são variados e ocorrem em diferentes níveis de organização biológica [3]. É uma das principais causas do declínio de muitas espécies [4,5], resultando em mudanças no funcionamento do ecossistema [6].

Dentre os vertebrados, os anfíbios são um dos grupos mais ameaçados pela fragmentação e pela perda de conectividade de habitats [4],[7],[8], sendo os efeitos negativos mais intensos em espécies com larvas aquáticas, que dependem tanto de habitats terrestres como de hábitats aquáticos para completar seu ciclo de vida [4]. Isso ocorre porque a fragmentação implica na perda de habitat e desconexão entre sítios reprodutivos e remanescentes de vegetação nativa, utilizada por adultos e jovens como refúgio e fonte de recurso [8]. Esse isolamento dificulta o processo de dispersão, favorece a redução da diversidade genética, aumenta o risco de extinções locais [5], altera a dinâmica das espécies [9] e reduz a riqueza e abundância de espécies de anfíbios, uma vez que reduz a qualidade e estrutura dos habitats [10],[11]

Como as populações de anfíbios que utilizam poças como locais para oviposição estão inseridas em ilhas de água imersas em uma matriz de solo [12],[13], elas estão sendo provavelmente regidas por uma dinâmica de metacomunidades. A visão moderna dos modelos tratados inicialmente pela teoria de biogeografia de ilhas, e aplicados à dinâmica de metacomunidades de óilhasôde hábitats continentais [14]ï [16], tem gerado uma grande quantidade de informações de importância central para gerenciamento e conservação da biodiversidade de paisagens fragmentadas [17]. As metacomunidades podem estruturar-se de acordo com uma combinação de processos, nominalmente a dinâmica de manchas, dinâmica de ordenação de espécies, efeito de massa e dinâmica neutra (ver [12], que refletem a força de fatores relacionados ao nicho Hutchinsoniano (fatores bióticos e abióticos locais) e à dispersão das espécies dentro de uma região [18].

As discussões sobre o efeito da área das ilhas ou manchas de habitats, bem como efeitos secundários que afetam a quantidade de energia e disponibilidade de habitat sobre a riqueza de espécies ainda estão pouco claras [19]ï [22]. Estes efeitos têm sido ainda menos explorados quando se trata da mudança na composição de espécies (diversidade beta) [23], especialmente para comunidades de poças [24]. A diversidade beta é altamente dependente da diversidade nas escalas local e regional (alfa e gama), e consequentemente varia de acordo com processos que afetam as demais escalas [25]. Além disso, a diversidade beta é influenciada pelo potencial de dispersão das espécies que compõem as comunidades locais [26].

Considerando as espécies de anfíbios que ocorrem no Cerrado, espécies que se reproduzem em poças representam uma porção considerável da diversidade de anuros do bioma. Na realidade, apesar de pequenas, as poças podem apresentar uma parcela representativa da biodiversidade de uma dada paisagem [27],[28]. Em diversas partes do mundo há uma necessidade de conservar as poças, pois estas estão sendo destruídas para dar espaço à agricultura, colocando diversas espécies em risco de extinção [29],[30]. No Brasil, especialmente no Cerrado brasileiro, temos uma situação inversa no que diz respeito ao número de poças, que está aumentando a partir do represamento de riachos e veredas [31]. No entanto, a escassez de informações a respeito da organização das comunidades pode impedir a nossa habilidade de predizer a resposta a ameaças maiores, tais como alterações no uso da terra ou a introdução de espécies invasoras [31]. Além disso, não sabemos qual a real importância das poças para a estruturação da biodiversidade no Cerrado, o que se faz essencial para conservar a biodiversidade de áreas agriculturáveis.

A escassez de informações a respeito do padrão de riqueza e ocupação das espécies de anuros em escala local e regional ainda pode ser um problema para a conservação da diversidade dos mesmos [32]ï [34]. Há um grande número de espécies com pouca informação a respeito do habitat, modo de vida, ocorrência e ameaças, corroborado pelo grande número de espécies deficientes de dados segundo a lista vermelha da IUCN. No bioma Cerrado, em particular, o maior conhecimento da anurofauna está localizado em regiões onde há maior concentração da população humana [35],[36], sendo que a maioria dos inventários estão restritos a região central e sudeste do bioma [32]ï [34]. Além disso, ainda há uma porção significativa do bioma que permanece não amostrada ou subamostrada [35].

Tendo como principal objetivo fornecer informações relevantes para a conservação de anuros no Cerrado brasileiro, esta tese foi construída de forma a abordar padrões em escala local e regional, bem como modelos de ocupação de algumas espécies de anfíbios com o intuito de identificar fatores que estão direcionando a riqueza e ocupação das espécies de anuros que se reproduzem em poças no Cerrado. Para isso, os dados foram coletados no estado de Goiás, único estado brasileiro totalmente inserido no bioma Cerrado, e que segue a mesma tendência do restante do bioma em relação a perda de habitat devido ao avanço das fronteiras agrícolas.

Considerando-se a escassez de informações sobre a ocorrência das espécies e com o intuito de cobrir lacunas de inventários na região, no ***Capítulo I*** apresento a primeira lista oficial de espécies registradas para o estado de Goiás. No ***Capítulo II***, procurei contribuir para a compreensão isolada de fatores locais e da paisagem sobre a diversidade local e regional de anuros de poças do Cerrado. Busquei explorar os efeitos da área, heterogeneidade e complexidade de hábitats local e da paisagem e da produtividade sobre a diversidade alfa e beta de anuros. Mais especificamente, abordo questões relacionadas a espécie área (proposta inicialmente por Arrhenius, [37]) e espécie energia (proposta por Wright [19]). No ***Capítulo III***¸ abordei um dos modelos mais clássicos e controversos para conservação das espécies, que é conhecido como: ñmuitas pequenas ou uma única grandeò (*several small or single large* ***ï*** SLOSS, proposto inicialmente por Diamond [38]). Este modelo deve ser levado em consideração quando o objetivo é preservar o maior número de espécies que ocorrem em poças, sendo que o tamanho das poças são as manchas de habitat em questão. E, por fim, no **Capítulo IV,** construí modelos de ocupação para ter acesso aos efeitos da quantidade de remanescentes de habitas, isolamento entre remanescentes de Cerrado, bem como características locais sobre a ocorrência de algumas espécies de anfíbios típicas do Cerado Brasileiro.

REFERÊNCIA BIBLIOGRÁFICA

1. Foley J a, Defries R, Asner GP, Barford C, Bonan G, et al. (2005) Global consequences of land use. Science (80-) 309: 570ï 574. Available: http://www.ncbi.nlm.nih.gov/pubmed/16040698. Accessed 21 May 2013.

2. Bennett AF, Radford JQ, Haslem A (2006) Properties of land mosaics: Implications for nature conservation in agricultural environments. Biol Conserv 133: 250ï 264. Available: http://linkinghub.elsevier.com/retrieve/pii/S0006320706002850. Accessed 23 May 2013.

3. Fischer J, Lindenmayer DB (2007) Landscape modification and habitat fragmentation: a synthesis. Glob Ecol Biogeogr 16: 265ï 280. doi:10.1111/j.1466-8238.2006.00287.x.

4. Becker CG, Fonseca CR, Haddad CFB, Batista RF, Prado PI (2007) Habitat split and the global decline of amphibians. Science 318: 1775ï 1777. Available: http://www.ncbi.nlm.nih.gov/pubmed/18079402. Accessed 24 May 2013.

5. Cushman S a. (2006) Effects of habitat loss and fragmentation on amphibians: A review and prospectus. Biol Conserv 128: 231ï 240. Available: http://linkinghub.elsevier.com/retrieve/pii/S0006320705003940. Accessed 22 May 2013.

6. Hooper DU, Adair EC, Cardinale BJ, Byrnes JEK, Hungate B a, et al. (2012) A global synthesis reveals biodiversity loss as a major driver of ecosystem change. Nature 486: 105ï 108. Available: http://www.ncbi.nlm.nih.gov/pubmed/22678289. Accessed 21 May 2013.

7. Silvano DL, Segalla M V. (2005) Conservation of Brazilian Amphibians. Conserv Biol 19: 653ï 658. Available: http://doi.wiley.com/10.1111/j.1523-1739.2005.00681.x.

8. Becker CG, Fonseca CR, Haddad CFB, Prado PI (2010) Habitat split as a cause of local population declines of amphibians with aquatic larvae. Conserv Biol 24: 287ï 294. Available: http://www.ncbi.nlm.nih.gov/pubmed/19758391. Accessed 12 June 2013.

9. Ernst R, Linsenmair KE, Rödel M-O (2006) Diversity erosion beyond the species level: Dramatic loss of functional diversity after selective logging in two tropical amphibian communities. Biol Conserv 133: 143ï 155. Available: http://linkinghub.elsevier.com/retrieve/pii/S0006320706002400. Accessed 22 May 2013.

10. Fahrig L (2003) EFFECTS OF HABITAT FRAGMENTATION ON BIODIVERSITY. Annu Rev Ecol Evol Syst 34: 487ï 515. Available: http://www.annualreviews.org/doi/abs/10.1146/annurev.ecolsys.34.011802.132419. Accessed 29 April 2014.

11. Ribeiro R, Carretero M a., Sillero N, Alarcos G, Ortiz-Santaliestra M, et al. (2011) The pond network: can structural connectivity reflect on (amphibian) biodiversity patterns? Landsc Ecol 26: 673ï 682. Available: http://link.springer.com/10.1007/s10980-011-9592-4. Accessed 12 June 2013.

12. Leibold MA, Holyoak M, Mouquet N, Amarasekare P, Chase JM, et al. (2004) The metacommunity concept: a framework for multi-scale community ecology. Ecol Lett 7: 601ï 613. Available: http://doi.wiley.com/10.1111/j.1461-0248.2004.00608.x. Accessed 21 May 2013.

13. De Meester L, Declerck S, Stoks R, Louette G, Van De Meutter F, et al. (2005) Ponds and pools as model systems in conservation biology, ecology and evolutionary biology. Aquat Conserv Mar Freshw Ecosyst 15: 715ï 725. Available: http://doi.wiley.com/10.1002/aqc.748. Accessed 22 May 2013.

14. Holt RD (1984) Spatial heterogeneity, indirect interactions, and the coexistence of prey species. Am Nat 124: 377ï 406.

15. Hanski I (1991) Metapopulation dynamics: brief history and conceptual domain. Biol J Linn Soc 42: 3ï 16.

16. Lichstein JW, Dushoff J, Levin S a, Pacala SW (2007) Intraspecific variation and species coexistence. Am Nat 170: 807ï 818. Available: http://www.ncbi.nlm.nih.gov/pubmed/18171164. Accessed 28 April 2014.

17. Forman RTT (1995) Some general principles of landscape and regional ecology. Landsc Ecol 10: 133ï 142. Available: http://link.springer.com/10.1007/BF00133027. Accessed 17 January 2014.

18. Leibold M, Geddes P (2005) El concepto de nicho en las metacomunidades. Ecol austral: 117ï 129. Available: http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:El+concepto+de+nicho+en+las+metacomunidades#0. Accessed 19 March 2014.

19. Wright D (1983) Species-energy theory: an extension of species-area theory. Oikos 41: 496ï 506.

20. Oertli B, Joye DA, Castella E, Juge R, Cambin D, et al. (2002) Does size matter? The relationship between pond area and biodiversity. Biol Conserv 104: 59ï 70. Available: http://linkinghub.elsevier.com/retrieve/pii/S0006320701001549.

21. Scheffer M, van Geest GJ, Zimmer K, Jeppesen E, Søndergaard M, et al. (2006) Small habitat size and isolation can promote species richness: second-order effects on biodiversity in shallow lakes and ponds. Oikos 112: 227ï 231. Available: http://onlinelibrary.wiley.com/doi/10.1111/j.0030-1299.2006.14145.x/full. Accessed 27 March 2014.

22. Kolasa J, Manne LL, Pandit SN (2012) Speciesï area relationships arise from interaction of habitat heterogeneity and species pool. Hydrobiologia 685: 135ï 144. Available: http://link.springer.com/10.1007/s10750-011-0846-6. Accessed 24 March 2014.

23. Chase JM, Kraft NJB, Smith KG, Vellend M, Inouye BD (2011) Using null models to disentangle variation in community dissimilarity from variation in Ŭ-diversity. Ecosphere 2: art24. Available: http://www.esajournals.org/doi/abs/10.1890/ES10-00117.1. Accessed 22 May 2013.

24. Werner EE, Skelly DK, Relyea R a., Yurewicz KL (2007) Amphibian species richness across environmental gradients. Oikos 116: 1697ï 1712. Available: http://www.blackwell-synergy.com/doi/abs/10.1111/j.2007.0030-1299.15935.x. Accessed 24 May 2013.

25. Anderson MJ, Crist TO, Chase JM, Vellend M, Inouye BD, et al. (2011) Navigating the multiple meanings of β diversity: a roadmap for the practicing ecologist. Ecol Lett 14: 19ï 28. Available: http://www.ncbi.nlm.nih.gov/pubmed/21070562. Accessed 22 May 2013.

26. Meutter F Van de, Meester L De, Stoks R (2007) METACOMMUNITY STRUCTURE OF POND MACROINVERTEBRATES: EFFECTS OF DISPERSAL MODE AND GENERATION TIME. Ecology 88: 1687ï 1695. Available: http://www.esajournals.org/doi/abs/10.1890/06-0333.1. Accessed 24 January 2014.

27. Biggs J, Corfield A, Walker D, Whitfield M, Williams P (1994) New approaches to the management of ponds. Br Wildl 5: 273ï 287.

28. Williams P (2004) Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. Biol Conserv 115: 329ï 341. Available: http://linkinghub.elsevier.com/retrieve/pii/S0006320703001538. Accessed 20 March 2014.

29. Oertli B, Biggs J, Céréghino R, Grillas P, Joly P, et al. (2005) Conservation and monitoring of pond biodiversity: introduction. Aquat Conserv Mar Freshw Ecosyst 15: 535ï 540. Available: http://doi.wiley.com/10.1002/aqc.752. Accessed 22 November 2013.

30. Ruggiero A, Céréghino R, Figuerola J, Marty P, Angélibert S (2008) Farm ponds make a contribution to the biodiversity of aquatic insects in a French agricultural landscape. C R Biol 331: 298ï 308. Available: http://www.ncbi.nlm.nih.gov/pubmed/18355753. Accessed 24 March 2014.

31. De Marco P, Nogueira DS, Correa CC, Vieira TB, Silva KD, et al. (2014) Patterns in the organization of Cerrado pond biodiversity in Brazilian pasture landscapes. Hydrobiologia 723: 87ï 101. Available: http://link.springer.com/10.1007/s10750-013-1695-2. Accessed 26 February 2014.

32. Diniz-Filho JAF, Bini LM, Pinto MP, Rangel TFLVB, Carvalho P, et al. (2006) Conservation biogeography of anurans in Brazilian Cerrado. Biodivers Conserv 16: 997ï 1008. Available: http://link.springer.com/10.1007/s10531-006-9010-4. Accessed 22 January 2014.

33. Diniz-Filho JAF, Bini LM, Vieira CM, Blamires D, Terribile LC, et al. (2008) Spatial patterns of terrestrial vertebrate species richness in the Brazilian Cerrado. Zool Stud 47: 146ï 157. Available: http://www.sectec.go.gov.br/portal/wp-content/uploads/2010/08/25.Diniz-Filho.pdf. Accessed 3 May 2014.

34. Valdujo PH, Silvano DL, Colli G, Martins M (2012) Anuran Species Composition and Distribution Patterns in Brazilian Cerrado, a Neotropical Hotspot. South Am J Herpetol 7: 63ï 78. Available: http://www.bioone.org/doi/abs/10.2994/057.007.0209. Accessed 10 July 2013.

35. Diniz-Filho JAF, Bastos RP, Rangel TFLVB, Bini LM, Carvalho P, et al. (2005) Macroecological correlates and spatial patterns of anuran description dates in the Brazilian Cerrado. Glob Ecol Biogeogr 14: 469ï 477. Available: http://doi.wiley.com/10.1111/j.1466-822X.2005.00165.x. Accessed 22 November 2013.

36. Diniz-Filho JAF, Bini LM, Pinto MP, Rangel TFLVB, Carvalho P, et al. (2006) Anuran species richness, complementarity and conservation conflicts in Brazilian Cerrado. Acta Oecologica 29: 9ï 15. Available: http://linkinghub.elsevier.com/retrieve/pii/S1146609X05000846. Accessed 23 January 2014.

37. Arrhenius O (1921) Species and area. J Ecol 9: 95ï 99. Available: http://www.jstor.org/stable/10.2307/2255763. Accessed 21 November 2013.

38. Diamond J (1975) The island dilemma: lessons of modern biogeographic studies for the design of natural reserves. Biol Conserv: 129ï 146. Available: http://www.sciencedirect.com/science/article/pii/000632077590052X. Accessed 23 April 2014.

CAPÍTULO I - Diversity of amphibian anurans from state of Goiás, core of Brazilian Cerrado

---

Luciana Signorelli[1]*, Vinicius Guerra Batista[2], Renan Nunes Costa[3]& Rogério Pereira Bastos[4]

[1]Programa de Pós-Graduação em Ecologia e Evolução, Instituto de Ciências Biológicas, Universidade Federal de Goiás, Campus Samambaia, 74001-970, Cx. Postal 131, Goiânia, GO, Brasil.

[2]Programa de Pós-Graduação em Ecologia de Ambientes Aquáticos Continentais, Universidade Estadual de Maringá, NUPELIA - Núcleo de Pesquisas em Limnologia, Ictiologia e Aquicultura, Bloco G-90, Av. Colombo, 5790, CEP 87020-900. Maringá, PR, Brasil.

[3]Programa de Pós-Graduação em Ecologia e Conservação da Biodiversidade, Universidade Estadual de Santa Cruz, Campus Soane Nazaré de Andrade, Rodovia Jorge Amado, km 16, 45662-900, Ilhéus, BA, Brasil.

[4]Laboratório de Herpetologia e Comportamento Animal, Departamento de Ecologia, Universidade Federal de Goiás, 74.001-970 Goiânia, Goiás, Brasil.

*Corresponding author: lfl.signorelli@gmail.com

Running title: Anuran amphibians of state of Goiás

Formatted according to *Zookeys.*

ABSTRACT

Considered the most biodiverse savanna in the world, Cerrado has been highlighted as the largest agricultural frontier in Brazil. The state of Goiás follows the same pattern then the rest of biome and, consequently, its diversity has being threatened by intense activity land use. This process is threatened species group as amphibians that are sensible to habitat loss. Based on field samples and literature records we presented the first list of anuran species from the central region of the Cerrado Biome, more precisely in the state of Goiás, Brazil. Information about reproductive modes and conservation status of species were also presented. In the field, we recorded 58 species of frogs represented by 19 genera, seven families and nine reproductive modes. Eight species were not assessed for conservation status and four are considered data deficient. None of the species is considered endangered. With literature data, we raised 92 frog species represented by 30 genera, 11 families and 12 reproductive modes. Most species recorded use lentic water bodies and lay their eggs directly or indirectly in the water. This list contributes positively to the knowledge of the occurrence of species of anurans from central Cerrado and can encourage the development of strategies for the conservation of this group in this highly threatened biome.

KEY-WORDS. Species diversity, reproductive modes, conservation status.

RESUMO

Considerada a savana mais biodiversa do mundo, o Cerrado tem sido destacado como a maior fronteira agrícola do Brasil. O estado de Goiás segue o mesmo padrão do restante do bioma e, consequentemente, sua diversidade tem sido ameaçada pela intensa atividade de uso do solo. Este processo está ameaçando grupos de espécies como os anfíbios, que são sensíveis à perda de habitat. Com base em amostras de campo e registros na literatura, apresentamos aqui a primeira lista de espécies de anuros para o estado de Goiás, Brasil, com informações sobre os modos reprodutivos e o status de conservação das espécies. Em campo, registramos 58 espécies de anuros representadas por 19 gêneros, sete famílias e nove modos reprodutivos. Oito espécies não foram avaliadas quanto ao status de conservação e quatro são consideradas deficientes de dados. Nenhuma das espécies amostradas é considerada em perigo. Com dados da literatura, levantamos 92 espécies de anuros representadas por 30 gêneros, 11 famílias e 12 modos reprodutivos. A maior parte das espécies registradas utilizam corpos d'água lênticos e depositam seus ovos direta ou indiretamente na água. A presente lista contribui positivamente com o conhecimento da ocorrência das espécies de anuros no cerrado goiano e pode favorecer a criação de estratégias para a conservação deste grupo e, consequentemente, deste bioma altamente ameaçado.

PALAVRAS-CHAVE. Diversidade de espécies, modos reprodutivos, status de conservação.

INTRODUCTION

The intense economic activities have being considered a world problem from the moment in which larger native areas have being converted into human dominated lands (Foley et al. 2005, Schiesari and Grillitsch 2011), being a mosaic with different kinds of land use activities (Bennett et al. 2006). This process has as major consequence reduction in habitat amount, and isolation between remnants patches (Fahrig 2003). In terms of biodiversity, the effects of habitat loss and lack or reduction of habitat connectivity are vast and can be occur in different levels of biological organization (With 1997, Fahrig 2003, Fischer and Lindenmayer 2007). Moreover, these are the main cause of species decline in the world (Cushman 2006, Becker et al. 2007), altering key process in the ecosystem (Hooper et al. 2012).

The Cerrado biome, richest savanna ecosystem in the world (Diniz-Filho et al. 2009), follow the same world tendency. The accelerated conversion process of natural landscapes (Klink and Moreira 2002, Klink and Machado 2005) into soya, maize and sugar cane plantation, as well an extensive cattle raising (Carvalho et al. 2009), started especially after the advance of the agricultural frontier, from the 1960s to the 1980s, a set of government incentives to stimulate the advance of cattle ranches and agriculturally used area in the Central Plateau of Brazil (Pufal et al. 2000). Currently, only about 20% of the Cerrado remains undisturbed, and only about 1.2% of it is protected (Mittermeier et al. 2004), despite being recognized as a global biodiversity hotspot (Myers et al. 2000). The Cerrado areas in the state of Goiás follow the same pattern of the rest of the biome, losing large portions of native areas by pastures or plantations (Carvalho et al. 2009). Currently, 1,169.368 km² (0.34% of state territory) of areas on the state of Goiás belong to integral protection conservation units.

Intensive human occupation process is linked to the rapid loss of biodiversity (Diniz-Filho et al. 2005). Among vertebrates, amphibians are the most threatened group by habitat and connectivity loss (Houlahan and Findlay 2003, Bowne and Bowers 2004, Silvano and Segalla 2005), being those effects more intense in species with aquatic larvae, which depends of both terrestrial and aquatic habitats to complete their life cycle (Becker et al. 2007, Becker et al. 2010). In the Cerrado biome there were recorded 209 anuran species, in which 108 is being endemic to the Cerrado biome (Valdujo et al. 2012). However, these numbers probably does not reflect the actual anuran species richness in the Cerrado biome (Valdujo et al. 2012).

Considering the lack of detailed data on species distribution for anuran in the Cerrado region, our main objective was to provide the first checklist of the amphibians for the state of Goiás, the only Brazilian state thoroughly within the Cerrado biome. We presented results of anuran surveys conducted in the state and we summarized records from data available in the literature. Moreover we provide information about reproductive modes and status conservation in accordance with IUCN.

MATERIAL AND METHODS

Study area

This study was conducted in state of Goiás, located in the core area of Cerrado biome. This state comprise around 349,000 Km^2, representing an area larger than many countries in the Latin American (IBGE 2014). To surveys landscapes with most diverse pattern of land use, we divided the state into a grid of 478 cells of 0.25 latitude by 0.25 longitude and we calculated the proportion of Cerrado remnants (PLAND) and isolation between remnants (ENN_MN) for each cells. For this, state of Goiás grid was overlaid on a raster file containing land-cover information obtained from 2001 and 2002 Landsat ETM+ satellite images. The PLAND and ENN_MN were calculated using Fragstats 3.3 (McGarigal and Marks 1995). We selected 18 landscapes with different levels of land use activity for anuran surveys.

Moreover, we realized some extra surveys with the objective to cover gaps in our sample design. Sites surveyed do not include conservation unities from state of Goiás (Figure 1).

Data collection

We surveyed a total of 146 sites for adults frogs across 18 landscapes distributed at the state of Goiás (Table 1). Fieldworks were conducted during the rainy season (October-March), from 2010 to 2013. During surveys, observers spent 1 h at each site between the hours of 1900 and 2400 to assess the presence and abundance of species. Adult frogs were surveyed via a combination of both acoustic and visual means (Rödel and Ernst 2004) while walking slowly around the pond and systematically searching or listening for adult frogs (Heyer et al. 1994). This method of survey is sufficient to detect sites where a species is present even with few visits (Pellet and Schmidt 2005). In our case, sites were surveyed a single time. Some adults were collected to confirm identification, and all specimens collected were euthanized by injecting an overdose of 2% lidocaine parenterally, minimizing pain and distress to the animal. Lethal injectable agents are rapid and reliable methods for performing euthanasia, being accepted by the American Veterinary Medical Association (Leary et al. 2013) and by the Brazilian National Council on the Control of Animal Experiments (CONCEA 2013). Thus, posteriorly of euthanasia, specimens were fixed in 10% formalin and preserved in 70% alcohol (Heyer et al. 1994). All adults collected were deposited at the Zoological Collection of the Universidade Federal de Goiás (ZUFG).

As our main objective is to provide a complete list of species recorded in the state of Goiás, we performed a bibliographic data compilation based on species lists, distribution notes, and descriptions papers from this Brazilian region. To avoid misidentification issues, we unconsidered species with questionable classification (e.g. sp., aff., or group). Reproductive modes of the amphibian species were determined following Haddad and Prado (2005), Wells

(2007), and Haddad et al. (2013) classification criteria. Additional information on the reproductive modes of some species were obtained by observations in the field and bibliography literature (e.g. Silva and Giaretta 2009, Bitar et al. 2012, Silva et al. 2012).

RESULTS

Species recorded

In our samples, we recorded 58 amphibian species from seven families and 19 genus during ponds surveys (Table 2, Figure 2, 3, 4 and 5). The most representative family was Hylidae (30 species), followed by Leptodactylidae (18 species), Bufonidae (three species), Odontophrynidae (two species), Microhylidae (three species), and finally by Dendrobatidae and Craugastoridae (both with one species). From the total, 31 species (53.44%) recorded are endemic, 56 (96.55%) are typical, and two (3.44%) present marginal distribution to the Cerrado biome (Table 2).

The most common species among the 146 sampled points were *Dendropsophus minutus* (70 points)*, Dendropsophus jimi* (42 points), *Dendropsophus rubicundulus* (41 points), *Hypsiboas albopunctatus* (70 points)*, Leptodactyçus latrans* (44 points), *Physalaemus cuvieri* (90 points), *Scinax fuscomarginatus* (65 points) and *Scinax fuscovarius* (41 points). Six species were found in a single point (*Bokermannohyla sapiranga*, *Hypsiboas phaeopleurus*, *Hypsiboas punctatus*, *Physalaemus marmoratus*, *Rhinella mirandaribeiroi* and *Scinax squalirostris*). Eight species were not formally assessed for the IUCN conservation status and four are considered data deficient (Table 1). None of recorded species is considered endangered.

Considering data obtained from literature, we recorded a total of 92 anurans species to the state of Goiás (Table 3). Among these species, 52 are endemics (56.52%), 19 (20.65%)

have widespread distribution, and 12 (13.04%) are known to be marginally distributed in the Cerrado biome. The most representative family was Hylidae (38 species), followed by Leptodactylidae (27 species), Bufonidae (seven species), Odontophrynidae (seven species), Microhylidae (four species), Craugastoridae (three species), Dendrobatidae (two species) and finally Aromobatidae, Brachycephalidae, Pipidae and Ranidae (with one species, respectively). Here, 16 species were not formally assessed for the IUCN conservation status and 10 species are considered data deficient (Table 3). The other species are considered least concerned and none is considered endangered. Thus, our field samples represent 63.04% of anuran species recorded in the compilation of bibliographic data in the state of Goiás.

Reproductive modes

Considering just species sampled here, we found nine reproductive modes observed for 58 anurans species recorded (Table 2). Leptodactylidae and Hylidae were the families with most diverse number of reproductive modes recorded here (four modes), followed by Bufonidae (two modes), Craugastoridae (one mode), Dendrobatidae (one mode), Microhylidae (one mode), and Odontophrynidae (one mode). The most part of species (N=48) deposit their eggs directly or indirectly in the ponds (e.g. mode 24, in which eggs are arboreals and tadpoles drop in the water), two species deposit their eggs in lotic water, and three species deposit their eggs in both lentic and lotic water. Just one species recorded here present direct development.

Twelve reproductive modes were observed for 92 anuran species previously recorded in the state of Goiás (Table 3). Leptodactylidae was the family with most diverse number of reproductive modes recorded (five modes), followed by Hylidae (three modes), Bufonidae (two modes), Odontophrynidae (two modes), Brachycephalidae (one mode), Craugastoridae (one mode), Dendrobatidae (one mode), and Microhylidae (one mode). The most part (N=66) of species deposit their eggs directly or indirectly in the ponds, two species deposit their eggs in lotic water, and twelve deposit in both lentic and lotic water. Four species recorded here present direct development (reproductive mode 23).

DISCUSSION

We are providing in this work the most complete list of anuran species to the state of Goiás, Brazil.We recorded a total of 92 anuran species to the state, including 58 species recorded through our surveys. This species richness includes a set of species registered previously in studies conducted in the state of Goiás (e.g. Oda et al. 2009, Vaz-Silva et al. 2007, Morais et al. 2011, Morais et al. 2012, Nomura et al. 2012, Santos et al. 2014), and is representative of the Cerrado biome, which presents a total of 209 species (Valdujo et al. 2012). Hylidae and Leptodactylidae were the most specious family. This pattern is in agreement with other works realized in the Neotropical region (e.g. Morais et al. 2011, Piatti et al. 2012, Santos et al. 2014), and is expected since both families have their greatest diversity in the Neotropics (Duellman 1999). Moreover, differently from other authors (e.g. Oda et al. 2009, Morais et al. 2011, Santos et al. 2014), we are not considering species with doubtful identification and species that are currently being described. Thus, the total number of species reported here probably does not reflect the actual richness of Goiás anurans species, and this number will increase even more in the next years due description of new species (as suggested by Diniz-Filho et al. 2005 and Valdujo et al. 2012).

The diversity of reproductive modes recorded in this study (twelve and nine) is considered high to Cerrado biome. In previous studies were registered between four to seven reproductive modes (e.g. Bastos et al. 2003, Toledo et al. 2003, Eterovick and Sazima 2004). We found that most part of species present indirect development, in which a minority deposits their eggs in lotic water bodies in areas of open vegetation, as expected to anuran species that occur in the Cerrado biome. Some species (e.g. *Hypsiboas albopunctatus, Leptodactylus fuscus*, *Physalaemus cuvieri, Rhinella schneideri e Scinax fuscovarius*) have been colonized successfully areas with high degree of anthropization, without major specificity of reproductive sites, being known as habitat-generalist (Brasileiro et al. 2005, Silva and Rossa-Feres 2007). In contrast, species such as *Bokermannohyla sapiranga* demonstrate specificity of breeding habitat and are known to occur in streams inserted in gallery forest (Brandão et al. 2012).

Barycholos ternetzi was the only species with direct development (mode 23) registered in our surveys, and *B. ternetzi*, *Ischnocnema juipoca*, *Oreobates remotus*, and *Pristimantis ventrigranulosus* are the species with direct development considering the state of Goiás. There are few species in the Cerrado biome with direct development, especially when we compared with forestal biomes like as Atlantic or Amazonian Forest. This pattern is expected since Cerrado is considered a tropical savanna, with a considered proportion of open areas. Moreover, this condition seems to be associate to hot weather and low air humidity inside forest areas of Cerrado, where normally these species occur (Colli et al. 2002).

Occurrence of species that may be found over other phytogeography areas, contributed to the species pool observed here. Anuran species richness in the Brazilian Cerrado has been explained as a consequence of Cerrado habitat environmental heterogeneity (Colli et al. 2002, Nogueira et al. 2009), and by contact with Amazonia Forest, Atlantic Forest, Caatinga, and Chaco, with an overlapping of biogeographic histories (Valdujo et al. 2012). Added to high anuran species richness, a considerable endemism rate emphasizes the importance of studies that can contribute to knowledge of the Cerrado diversity (Valdujo et al. 2012). However, many areas in the state of Goiás and Brazilian Cerrado should yet to be inventoried, and there are several localities where surveys should be reinforced, which suggest that anurofauna in the Cerrado biome are still underestimated (Diniz-Filho et al. 2005, Silvano and Segalla 2005, Valdujo et al. 2012). It is possible that many species that were not formally described by science have been extinct (Whittakker et al. 2005). As predicted by Diniz-Filho et al. (2005), several species with small body size were not known by scientific community due lack knowledge from the region of interest. This pattern can be corroborated by recently species descriptions (e.g. *Scinax pusillus* ï Pombal et al. 2011, *Ameerega berohoka* ï Vaz-Silva and Maciel 2011; *Adenomera cotuba* and *A. juikitam* ï Carvalho and Giaretta 2013), and because of this many those species are data deficient or were not evaluated following IUCN criteria. There are probably more species to be descripted and to be discovered by science. Thus, the species number of anurans in the state of Goiás and consequently in the Brazilian Cerrado will increase even more during the next years.

As is known, habitat loss and intensification of agriculture activities have negatively impacted species diversity and abundance of amphibians (Stuart et al. 2004, Becker et al. 2007). The Cerrado biome still has high expectation of intensifying their agricultural frontier, and it has been highly threatened by increased habitat loss (Klink and Moreira 2002, Schiesari and Grillistch 2011). Thus, it becomes urgent discussions and implementation of strategies that maximize efforts to conserve amphibians in this biome and in the other parts of the world (Young et al. 2001, Silvano and Segalla 2005, Diniz-Filho et al. 2009). Therefore, studies like this are important because provide information about occurrence of species in different regions in the state of Goiás, and consequently in the Cerrado biome. A systematization and description of species distributions patterns is the first step to understand the mechanism which are driving anuran communities (Valdujo et al. 2012).

LITERATURE CITED

Andrade SP, Vaz-Silva W (2012) First state record and distribution extension of *Pleurodema diplolister* (Peters 1870) (Anura: Leiuperidae) from state of Goiás, Brazil. CheckList 8(1): 149-151.

Andrade SP, Vaz-Silva W, Victor-Júnior EP, Santos DL, Maciel NM (2012) Distribution extension, new state record and geographic distribution map of *Oreobates remotus* Teixeira-Jr, Amaro, Recorder, Sena and Rodrigues, 2012 (Amphibia, Anura, Strabomantidae) in Central Brazil. Herpetology Notes 5: 275-276.

Bastos RP, Motta JAO, Lima LP, Guimarães LD (2003) Anfíbios da Floresta Nacional de Silvânia, estado de Goiás. Stylo gráfica e editora, Goiânia. 82pp.

Becker CG, Fonseca CFR, Haddad CFB, Batista RF, Prado PI (2007) Habitat split and the global decline of amphibians. Science 318(5857): 1775-1777.

Becker CG, Loyola RD, Haddad CFB, Zamudio KR (2010) Integrating species life-history traits and patterns of deforestation in amphibian conservation planning. Diversity and Distributions 16(1): 10-19.

Bennett AF, Radford JQ, Haslem A (2006) Properties of land mosaics: implications for nature conservation in agricultural environments. Biological Conservation 133(2): 250-264.

Bitar YOC, Pinheiro LPC, Abe OS, Santos-Costa MC (2012) Species composition and reproductive models of anurans from a transitional Amazonian forest, Brazil. Zoologia 29(1): 19-26.

Borges FJA, Juliano RF (2007) Distribuição espacial e temporal de uma comunidade de anuros do município de morrinhos, Goiás, Brasil (Amphibia: Anura). Neotropical Biology and Conservation 2(1):21-27.

Bowne DR, Bowers MA (2004) Interpatch movements in spatially structured populations: a literature review. Landscape Ecology 19(1): 1-20.

Brandão RA, Magalhães RF, Garda AA, Campos LA, Sebben A, Maciel NM (2012) A new species of Bokermannohyla (Anura: Hylidae) from highlands of Central Brazil. Zootaxa 3527: 28-42.

Brandão RA, Caramaschi U, Vaz-Silva W, Campos LA (2013) Three new species of Proceratophrys Miranda-Ribeiro 1920 from Brazilian Cerrado (Anura, Odontophrynidae). Zootaxa 3750 (4): 321-347.

Brasileiro CA, Sawaya RJ, Kiefer MC, Martins M (2005) Amphibians of an open cerrado fragment in southeastern Brazil. Biota Neotropica 5(2): 1-17.

Campos FS, Vaz-Silva W (2010) Distribuição espacial e temporal da anurofauna em diferentes ambientes no município de Hidrolândia, GO, Brasil Central. Neotropical Biology and Conservation 5(3): 179-187.

Carvalho FMV, De Marco JrP, Ferreira LG (2009) The Cerrado into-pieces: habitat fragmentation as a function of landscape use in the savannas of central Brazil. Biological Conservation 142(7): 1392-1403.

Carvalho TR, Giaretta AA (2013) Bioacoustics reveals two new syntopic species of *Adenomera steindachner* (Anura: Leptodactylidae: Leptodactylinae) in the Cerrado of central Brazil. Zootaxa 3731: 533-551.

Cintra CED, Silva HLR, Silva NJJr (2009) Herpetofauna, Santa Edwiges I and II hydroelectric power plants, state of Goiás, Brazil. CheckList 5(3): 570-576.

Colli GR, Bastos RP, Araújo AFB (2002) The character and dynamics of the Cerrado herpetofauna. In: Oliveira PS, Marquis RJ (eds) The Cerrados of Brazil: ecology and natural history of a neotropical savanna. New York: Columbia University Press. 223-241.

CONCEA (2013) Diretrizes da prática de eutanásia do CONCEA. Ministério da Ciência, Tecnologia e Inovação.

Cushman SA (2006) Effects of habitat loss and fragmentation on amphibians: a review and prospectus. Biological Conservation 128(2): 231-240.

Diniz-Filho JAF, Bastos RP, Rangel TFLVB, Bini LM, Carvalho P, Silva RJ (2005) Macroecological correlates and spatial patterns of anuran description dates in the Brazilian Cerrado. Global Ecology and Biogeography Letters, Oxford 14(4): 469-477.

Diniz-Filho JAF, Bini LM, Oliveira G, Barreto BS, Silva MMFP, Terrible L, Rangel TFLV, Pinto MP, Sousa NPR, Vieira LCG, Melo AS, De Marco Jr. P, Blamires D, Bastos RP, Carvalho P, Ferreira LG, Telles MPC, Rodrigues FM, Silva DM, Silva Junior NJ, Soares TN (2009) Macroecologia, biogeografia e áreas prioritárias para conservação no cerrado. Oecologia Brasileira 13: 470-497.

Duellman WE (1999) Patterns of distribution of amphibians: a global perspective. The John Hopkins University Press, Baltimore. 648pp.

Eterovick PC, Sazima I (2004) Anfíbios da Serra do Cipó. Belo Horizonte (MG, Brazil): Ed. PUC Minas. 151pp.

Fahrig L (2003) Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution and Systematics 34: 487-515.

Fischer J, Lindenmayer DB (2007) Landscape modification and habitat fragmentation: a synthesis. Global Ecology and Biogeography 16: 265-280.

Foley JA, Defries R, Asner GP, Barford C, Bonan G, Carpenter SR, Chapin FS, Coe MT, Daily GC, Gibbs HK, Helkowski JH, Holloway T, Howard EA, Kucharik CJ, Monfreda C, Patz JA, Prentice IC, Ramankutty N, Snyder PK (2005) Global consequences of land use. Science 309 (5734): 570-744.

Haddad CFB, Prado CPA (2005) Reproductive modes in frogs and their unexpected diversity in the Atlantic Forest of Brazil. Bioscience 55: 207-217.

Haddad CFB, Toledo LF, Prado CPA, Loebmann D, Gasparini JL, Sazima I (2013) Guia dos Anfíbios da Mata Atlântica: Diversidade e Biologia. São Paulo: Editora Anolis Books. 544pp.

Heyer WR, Donnelly MA, McDiarmid RW, Hayek LC, Foster MS (1994) Measuring and monitoring biological diversity - Standard methods for Amphibians. Smithsonian Institution Press, Washington. 364pp.

Hooper DU, Adair EC, Cardinale BJ, Byrnes JEK, Hungate BA, Matulich KL, Gonzalez A, Duffy JE, Gamfeldt L, O'Connor MI (2012) A global synthesis reveals biodiversity loss as a major driver of ecosystem change. Nature 486: 105-129.

Houlahan JE, Findlay CS (2003) The effects of adjacent land use on wetland amphibian species richness and community composition. Canadian Journal of Fisheries and Aquatic Sciences 60: 1078-1094.

IBGE 2014. http://www.ibge.gov.br/estadosat/perfil.php?sigla=gohttp://www.ibge.gov.br/estadosat/perfil.php?sigla=go Acessed in 15/04/2014.

Klink CA, Moreira AG (2002) Past and current human occupation and land-use. In: Oliveira PS, Marquis RJ (eds.) The Cerrados of Brazil: Ecology and Natural History of a Neotropical Savanna. Columbia University Press, New York, 69-88.

Klink CA, Machado RB (2005) Conservation of the Brazilian cerrado. Conservation Biology 19(3): 707-713.

Kopp K, Signorelli L, Bastos RP (2010) Distribuição temporal e diversidade de modos reprodutivos de anfíbios anuros no Parque Nacional das Emas e entorno, estado de Goiás, Brasil. Iheringia 100(3): 192-200.

Leary S, Underwood W, Anthony R, Cartner S, Corey D, Grandin T, Greenacre C, Gwaltney-Brant S, McCrackin MA, Meyer R, Miller D, Shearer J, Yanong R (2013). AVMA Guidelines for the Euthanasia of Animals: 2013 Edition. Version 2013.0.1. American Veterinary Medical Association.

McGarigal K, Marks BJ (1995) FRAGSTATS: Spatial pattern analysis program for quantifying landscape structure, General Technical Report, USDA Forest Service Pacific Northwest Research Station.

Mello M, Fava F, Pinto HBA, Bastos RP, Nomura F (2013) Diversidade de Anuros (Amphibia) da reserva extrativista Lago do Cedro e seu entorno, Aruanã, Goiás. Biota Neotropica 13(2): 1-13.

Mittermeier RA, Gil PR, Hoffmann M, Pilgrin J, Brooks T, Mittermeier CG, Lamoreux J, Fonseca GAB (2004) Hotspots revisited. Earthôs biologically richest and most endangered terrestrial ecoregions. Sierra Madre, Cemex, 392 pp.

Morais AR, Signorelli L, Gambale PG, Kopp K, Nomura F, Guimarães LD, Vaz-Silva W, Ramos J, Bastos RP (2011) Anuran amphibians associated to water bodies in Southwest of Goiás State (Brazil). Biota Neotropica 11(3): 355-363.

Morais AR, Bastos RP, Vieira R, Signorelli L (2012) Herpetofauna da Floresta Nacional de Silvânia, um remanescente de Cerrado no Brasil Central. Neotropical Biology and Conservation 7(2): 114-121.

Myers N, Mittermeier RA, Mittermeier CG, Da Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403: 853-858.

Nóbrega CC, De Marco Jr. P (2011) Unprotecting the rare species: a niche-based gap analysis for odonates in a core Cerrado area. Diversity and Distributions 17:491-505.

Nogueira C, Colli GR, Martins M (2009) Local richness and distribution of the lizard fauna in natural habitat mosaics of the Brazilian Cerrado. Austral Ecology 34: 83-96.

Nomura F, Maciel NM, Pereira EB, Bastos RP (2012) Diversidade de anuros (Amphibia) em áreas recuperadas de atividade mineradora e de plantio de *Eucalyptus urophyla*, no Brasil Central. Bioscience Journal 28(2): 312-324.

Oda FH, Bastos RP, Sá Lima, MAC (2009) Taxocenose de anfíbios anuros no Cerrado do Alto Tocantins, Niquelândia, Estado de Goiás: diversidade, distribuição local e sazonalidade. Biota Neotropica 9: 219-232.

Pellet J, Schmidt BR (2005) Monitoring distributions using call surveys: estimating site occupancy, detection probabilities and inferring absence. Biological Conservation 123: 27-35.

Piatti L, Amaro PMO, Araújo JFJ, Sanches VQ A, Bernarde PS (2012) Anurans from a disturbed area in Jarú, Rondônia, Brazil. Check List 8(1): 83-87.

Pombal JPJr., Bilate M, Gambale PG, Signorelli L, Bastos RP (2011) A new miniature treefrog of the *Scinax ruber* clade from the cerrado of central Brazil (Anura: Hylidae). Herpetologica 67: 288-299.

Pombal JPJr, Menezes VA, Fontes AF, Nunes I, Rocha CFD, Van-Sluys M (2012) A second species of the casque-headed frog genus *Corythomantis* (Anura:Hylidae) from northeastern Brazil, the distribution of C. greening, and comments on the genus. Boletim do Museu Nacional, Nova Série, Zoologia 530: 1-14.

Pufal DVL, Buschbacher R, Garcia MA (2000) Brazil: Cerrado. In: Wood A, Stedman-Edwards P, Mang J (eds.) The root causes of biodiversity loss. World Wildlife Fund and Earthscan Publications, Sterling, Virginia, 95-125.

Pugliese A, Baêta D, Pombal JPJr (2009) A new species of *Scinax* (Anura: Hylidae) from rocky montane fields in southeastern and central Brazil. Zootaxa 2269: 53ï 64.

Rödel MO, Ernst R (2004) Measuring and monitoring amphibian diversity in tropical forests. I. An evaluation of methods with recommendations for standardization. Ecotropica 10: 1-14.

Santos DL, Andrade SP, Victor-Jr. EP, Vaz-Silva W (2014) Amphibians and reptiles from southeastern Goiás, Central Brazil. Check List 10(1): 131-148.

Schiesari L, Grillitsch B (2011) Pesticides meet megadiversity in the expansion of biofuel crops. Frontiers in Ecology and the Environment.9(4):215-221.

Silva FR, Rossa-Feres DC (2007) Uso de fragmentos por anuros (Amphibia) de área aberta na região noroeste do Estado de São Paulo. Biota Neotropica. 7(2):141-147.

Silva WR, Giaretta AA (2009) On the natural history of *Leptodactylus syphax* with comments on the evolution of reproductive features in the *L. pentadactylus* species group (Anura, Leptodactylidae). Journal of Natural History 43(3): 191-203.

Silva FR, Almeida-Neto M, Prado VHM, Haddad CFB, Rossa-Feres DC (2012) Humidity levels drive reproductive modes and phylogenetic diversity of amphibians in the Brazilian Atlantic Forest. Journal of Biogeography 39: 1720-1732.

Silvano DL, Segalla MV (2005) Conservation of Brazilian amphibians. Conservation Biology. 19: 653-658.

Stuart SN, Chanson JS, Cox N, Young BE, Rodrigues ASL, Fischman DL, Waller RW (2004) Status and trends of amphibian declines and extinctions worldwide. Science 306(5702): 1783-1786.

Toledo LF, Zina J, Haddad CFB (2003) Distribuição espacial e temporal de uma comunidade de anfíbios anuros do Município de Rio Claro, São Paulo, Brasil. Holos Environment 3: 136-149.

Valdujo PH, Silvano DL, Colli G, Martins M (2012) Anuran species composition and distribution patterns in Brazilian Cerrado, a Neotropical Hotspot. South American Journal of Herpetology 7(2): 63-78.

Vaz-Silva W, Guedes AG, Silva PLA, Gontijo FF, Barbosa RS, Aloisio GR, Oliveira FCG (2007) Herpetofauna, Espora Hydroelectric Power Plant, state of Goiás, Brazil. Check List 3: 338-345.

Vaz-Silva W, Maciel NM (2011) A new cryptic species of Ameerega (Anura: Dendrobatidae) from Brazilian Cerrado. Zootaxa 2826: 57-68.

Vaz-Silva W, Valdujo PH, Pombal JPJr (2012) New species of the Rhinella crucifer group (Anura, Bufonidae) from the Brazilian Cerrado. Zootaxa 3265: 57ï 65.

Wells KD (2007) The ecology and behavior of amphibians. Chicago, The University of Chicago Press. 1400pp.

With K (1997) The application of neutral landscape models in conservation biology. Conservation Biology 11(5): 1069-1080.

Whittaker RJ, Araujo MB, Jepson P, Ladle RJ, Watson JEM, Willis KJ (2005) Conservation biogeography: assessment and prospect. Diversity and Distributions 11: 3-23.

Young BE, Lips KR, Reaser JK, Ibáñez R, Salas AW, Cedeño JR, Coloma LA, Ron S, La Marca E, Meyer JR, Muñoz A, Bolaños F, Chaves G, Romos D (2001) Population declines and priorities for amphibian conservation in Latin America. Conservation Biology 15(5): 1213-1223.

Table 1. Geographical location of 146 water bodies surveyed in the state of Goiás, Brazil.

Ladscape	Pond code	Latitude	Longitude
Landscape 1 (L1)	Site 1	-46.84666	-13.69103
	Site 2	-46.84300	-13.70849
	Site 3	-46.82911	-13.70437
	Site 4	-46.82630	-13.68038
	Site 5	-46.87074	-13.61987
	Site 6	-46.96302	-13.69521
Landscape 2 (L2)	Site 7	-46.87223	-13.84411
	Site 8	-46.90405	-13.75715
	Site 9	-46.82923	-13.80382
	Site 10	-46.78417	-13.86663
	Site 11	-46.80974	-13.83683
	Site 12	-46.85291	-13.78571
	Site 13	-46.88168	-13.93844
Landscape 3 (L3)	Site 14	-47.53414	-14.28447
	Site 15	-47.51892	-14.49711
	Site 16	-47.57891	-14.47548
	Site 17	-47.63063	-14.28833
	Site 18	-47.52320	-14.41306
	Site 19	-47.51535	-14.32392
	Site 20	-47.60861	-14.35013
	Site 21	-47.58993	-14.35087
	Site 22	-47.64447	-14.31230
Landscape 4 (L4)	Site 23	-47.56330	-14.73322
	Site 24	-47.55355	-14.72789
	Site 25	-47.53941	-14.61175
	Site 26	-47.52669	-14.65993
	Site 27	-47.64401	-14.58648
Landscape 5 (L5)	Site 28	-50.98154	-14.87546
	Site 29	-50.96453	-14.76405
	Site 30	-50.97963	-14.85642
	Site 31	-50.98027	-14.88871
	Site 32	-50.96459	-14.76400
	Site 33	-50.97731	-14.81927
	Site 34	-50.91181	-14.69426
	Site 35	-50.83342	-14.96622
	Site 36	-50.78246	-14.92639
	Site 37	-50.77708	-14.85250
Landscape 6 (L6)	Site 38	-51.11599	-15.33093
	Site 39	-51.12589	-15.36272
	Site 40	-51.07017	-15.45616
	Site 41	-51.06906	-15.38487
Landscape 7 (L7)	Site 42	-47.65840	-16.21173
	Site 43	-47.67249	-16.12817
	Site 44	-47.70883	-16.09888
	Site 45	-47.72364	-16.10909
	Site 46	-47.67822	-16.10908
	Site 47	-47.68064	-16.05431
	Site 48	-47.60990	-16.12602

Ladscape	Pond code	Latitude	Longitude
Landscape 8 (L8)	Site 49	-48.37849	-16.39842
	Site 50	-48.40567	-16.41786
	Site 51	-48.47326	-16.47452
	Site 52	-48.47780	-16.43202
	Site 53	-48.44280	-16.46549
	Site 54	-48.43357	-16.44765
Landscape 9 (L9)	Site 55	-51.10805	-16.80075
	Site 56	-51.13783	-16.76715
	Site 57	-51.03234	-16.77583
	Site 58	-51.06884	-16.80673
	Site 59	-51.09909	-16.88769
	Site 60	-51.13158	-16.85313
Landscape 10 (L10)	Site 61	-47.44696	-16.78539
	Site 62	-47.35887	-16.86050
	Site 63	-47.26977	-16.90803
	Site 64	-47.30827	-16.86667
	Site 65	-47.25495	-16.89093
	Site 66	-47.44366	-16.83297
	Site 67	-47.42547	-16.79221
	Site 68	-47.41719	-16.82065
Landscape 11 (L11)	Site 69	-51.46745	-17.24056
	Site 70	-51.36153	-17.19265
	Site 71	-51.86924	-17.33796
	Site 72	-51.89152	-17.28975
	Site 73	-51.44235	-17.21572
	Site 74	-51.46844	-17.21082
Landscape 12 (L12)	Site 75	-49.36240	-17.40496
	Site 76	-49.34385	-17.43270
	Site 77	-49.36326	-17.47468
	Site 78	-49.27620	-17.49405
	Site 79	-49.34764	-17.42079
	Site 80	-49.35492	-17.42575
	Site 81	-49.32304	-17.36864
	Site 82	-49.37065	-17.42583
	Site 83	-49.47584	-17.43700
Landscape 13 (L13)	Site 84	-48.35661	-17.40155
	Site 85	-48.40864	-17.44053
	Site 86	-48.38332	-17.40376
	Site 87	-48.39866	-17.41829
	Site 88	-48.40158	-17.42190
	Site 89	-48.38397	-17.45221
	Site 90	-48.45634	-17.32725
	Site 91	-48.43857	-17.32291
	Site 92	-48.30927	-17.30275
Landscape 14 (L14)	Site 93	-51.65510	-17.71865
	Site 94	-51.56791	-17.68387
	Site 95	-51.62569	-17.71140
	Site 96	-51.63255	-17.73722
	Site 97	-51.52722	-17.57358
	Site 98	-51.53880	-17.54919
	Site 99	-51.53881	-17.54919
	Site 100	-51.70738	-17.67462
	Site 101	-51.69717	-17.62131

Ladscape	Pond code	Latitude	Longitude
Landscape 15 (L15)	Site 102	-52.66128	-18.04371
	Site 103	-52.61319	-18.23584
	Site 104	-52.60674	-18.23556
	Site 105	-52.72495	-18.00507
	Site 106	-52.57946	-18.23293
Landscape 16 (L16)	Site 107	-51.14320	-18.23985
	Site 108	-51.03950	-18.11085
	Site 109	-51.12025	-18.10498
	Site 110	-51.11956	-18.20267
	Site 111	-51.15503	-18.15154
	Site 112	-51.09069	-18.22123
	Site 113	-51.07094	-18.19755
	Site 114	-51.08348	-18.17433
	Site 115	-51.05573	-18.12625
Landscape 17 (L17)	Site 116	-52.00917	-18.35927
	Site 117	-52.18723	-18.42779
	Site 118	-52.08289	-18.45008
	Site 119	-52.07329	-18.45659
	Site 120	-52.23487	-18.46953
	Site 121	-52.24154	-18.45776
	Site 122	-52.10810	-18.37642
Landscape 18 (L18)	Site 123	-50.92379	-18.70928
	Site 124	-50.97166	-18.67270
	Site 125	-50.88402	-18.73352
	Site 126	-50.87427	-18.72044
	Site 127	-50.90222	-18.73140
	Site 128	-50.91433	-18.65751
	Site 129	-50.93675	-18.64684
	Site 130	-50.94501	-18.59052
Extra site 1 (ES1)	Site 131	-15.86986	-50.91033
	Site 132	-15.83236	-50.90014
	Site 133	-15.84336	-50.90578
	Site 134	-15.86664	-50.76725
	Site 135	-15.85131	-50.76406
Extra site 2 (ES2)	Site 136	-50.07061	-15.99511
	Site 137	-50.07219	-15.99911
	Site 138	-50.15881	-15.89011
	Site 139	-50.13923	-15.89900
Extra site 3 (ES3)	Site 140	-13.25683	-50.13661
	Site 141	-13.26669	-50.12025
	Site 142	-13.26736	-50.12089
	Site 143	-13.26494	-50.11031
	Site 144	-13.32250	-50.27639
	Site 145	-13.30158	-50.28533
	Site 146	-13.29644	-50.24097

Table 2. Species list of anurans sampled in state of Goiás, Brazil. Abbreviations: A = Association degree with the Cerrado, D = distribution pattern, E = Cerrado endemic, O = species that occur in open domains, M = marginal species, S = species with meridional distribution that occur in Cerrado, T = typical species, W = widely distributed, AM = species occurring in both Amazonia and Cerrado, AT = species occurring both in Atlantic Forest and Cerrado, CA = species occurring both in Caatinga and Cerrado, , RM = Reproductive Mode (*sensu*Haddad and Prado, 2005; Wells, 2007; Haddad et al., 2013), RL = Red List (IUCN 2014), NL = No Listed, LC = Least Concern, DD = Data Deficient.

TAXON	LANDSCAPE	A	D	RM	RL
AMPHIBIA					
ANURA					
Bufonidae					
Rhinella mirandaribeiroi (Gallardo, 1965)	L3	T	E	1	NL
Rhinella rubescens (Lutz, 1925)	L8	T	E	1	LC
Rhinella schneideri (Werner, 1894)	L1,L2,L5,L7 ï L10, L12 ï L15, L17, ES1, ES2	T	W	1 ï 2	LC
Craugastoridae					
Barycholos ternetzi (Miranda-Ribeiro, 1937)	L4, L7, L8, L13, L16, ES3	T	E	23	LC
Dendrobatidae					
Ameerega flavopicta (Lutz, 1925)	213, 272	T	E	20	LC
Hylidae					
Bokermannohyla pseudopseudis (Miranda-Ribeiro, 1937)	L3	T	E	2	LC
Bokermannohyla sapiranga Brandão, Magalhães, Garda, Campos, Sebben, and Maciel, 2012	L8	T	E	2	NL
Dendropsophus cruzi (Pombal and Bastos, 1998)	L8, L9, L11 ï L14, L16 ï L18, ES1 ï ES3	T	E	1	LC
Dendropsophus jimi (Napoli and Caramaschi, 1999)	L5 ï L8, L10 ï L12, L14 ï L18	T	E	1	LC
Dendropsophus minutus (Peters, 1872)	L1 ï L8, ES1 ï ES3	T	W	1	LC

TAXON	LANDSCAPE	A	D	RM	RL
Dendropsophus melanargyreus (Cope, 1887)	641	T	W	1	LC
Dendropsophus nanus (Boulenger, 1889)	L1, L2, L5 ï L7, L9, L12, L16, L18, ES1 ï ES3	T	W	1	LC
Dendropsophus rubicundulus (Reinhardt and Lütken, 1862)	L1 ï L6, L8, L9, L11, L13 ï L18, ES1 ï ES3	T	E	1	LC
Dendropsophus soaresi (Caramaschi and Jim, 1983)	L3, L8, L16, ES3	M	CA	1	LC
Hypsiboas albopunctatus (Spix, 1824)	L3 ï L5, L7 ï L18, ES1, ES3	T	W	1	LC
Hypsiboas crepitans (Wied-Neuwied, 1824)	L1, L2	T	W	4	LC
Hypsiboas goianus (Lutz, 1968)	L8, L13	T	E	1	LC
Hypsiboas lundii (Burmeister, 1856)	L3 ï L5, L7, L8, L10 ï L14, L17, L18	T	E	4	LC
Hypsiboas paranaiba Carvalho and Giaretta, 2010	L12, L14, L16, L18, ES3	T	E	1	NL
Hypsiboas phaeopleura (Caramaschi and Cruz, 2000)	L3	T	E	1	DD
Hypsiboas punctatus (Schneider, 1799)	L5	T	W	1 ï 2	LC
Hypsiboas raniceps (Cope, 1862)	L1, L2, L5, L6, L9, L11, L12, L14, L18, ES1-ES3	T	W	1	LC
Lysapsus caraya Gallardo, 1964	L5, ES1, ES3	T	E	1	LC
Phyllomedusa azurea Cope, 1862	L1 ï L14, L6 ï L9, L11 ï L14, L16, L17, ES2, ES3	T	E	24	DD
Phyllomedusa oreades Brandão, 2002	L3	T	E	24	DD
Pseudis bolbodactyla Lutz, 1925	L5, L6, L9, L11 ï L13, L16, L18, ES1, ES2	T	AT	1	LC
Scinax centralis Pombal and Bastos, 1996	L13	T	E	1	LC
Scinax constrictus Lima, Bastos, and Giaretta, 2005	L5, L12, L18, ES1 ï ES3	T	E	1	LC
Scinax fuscomarginatus (Lutz, 1925)	L3 ï L12, L14, L16 ï L18, ES1 ï ES3	T	W	1	LC
Scinax fuscovarius (Lutz, 1925)	L1 ï L4, L9 ï L12, L14 ï L17, ES1 ï ES3	T	W	1	LC
Scinax pusillus Pombal, Bilate, Gambale,	L5, L10, L11, L14, L15, L17	T	E	1	NL

TAXON	LANDSCAPE	A	D	RM	RL
Signorelli, and Bastos, 2011					
Scinax rogerioi Pugliese, Baêta, and Pombal, 2009	L3, L10	T	E	1	NL
Scinax similis (Cochran, 1952)	L3, L4, L16, L18, ES1 ï ES3	M	AT	1	LC
Scinax squalirostris (Lutz, 1925)	L3	T	S	1	LC
Trachycephalus typhonius (Linnaeus, 1758)	L8, L9, L13	T	W	1	LC
Leptodactylidae					
Adenomera hylaedactyla (Cope, 1868)	L11, L14, L17	T	AM	32	LC
Adenomera saci(Bokermann, 1956)	L9, L16, L18	T	E	32	LC
Leptodactylus furnarius Sazima and Bokermann, 1978	L3, L9, L16	T	E	30	LC
Leptodactylus fuscus (Schneider, 1799)	L1, L2, L6, L7, L9 ï L11, L14 ï L18, ES1 ï ES3	T	W	11 ï 30	LC
Leptodactylus labyrinthicus (Spix, 1824)	L1 ï L12, L14 ï L18, ES1 ï ES3	T	W	11	LC
Leptodactylus latrans (Steffen, 1815)	L1, L2, L5 ï L15, L17, L18, ES1 ï ES3	T	W	11 ï 30	LC
Leptodactylus leptodactyloides (Andersson, 1945)*	L12, L17	T	W	11	LC
Leptodactylus mystaceus (Spix, 1824)	L7, L9	T	W	30	LC
Leptodactylus mystacinus (Burmeister, 1861)	L7, L10, L12	T	AT	30	LC
Leptodactylus podicipinus (Cope, 1862)	L1, L2, L5, L9, L11 ï L15, L17, L18, ES1, ES3	T	O	11	LC
Leptodactylus pustulatus (Peters, 1870)	L5, L6, ES3	T	E	11	LC
Leptodactylus sertanejo Giaretta and Costa, 2007	L3	T	E	11	LC
Physalaemus centralis Bokermann, 1962	L2, L5 ï L7, L9, L11, L12, L14 ï L18, ES1 ï ES3	T	E	11	LC
Physalaemus cuvieri Fitzinger, 1826	L1 ï L4, L7 ï L18, ES1 ï ES3	T	W	11	LC
Physalaemus marmoratus (Reinhardt	L11, L16	T	E	11	LC

TAXON	LANDSCAPE	A	D	RM	RL
and Lütken, 1862)					
Physalaemus nattereri (Steindachner, 1863)	L1, L3, L6 ï L10, L15, L17, ES1 ï ES3	T	E	11	LC
Pseudopaludicola saltica (Cope, 1887)	L3, L4	T	E	1	LC
Pseudopaludicola mystacalis (Cope, 1887)	ES2	T	W	1	LC
Microhylidae					
Chiasmocleis albopunctata (Boettger, 1885)	L3	T	E	1	LC
Dermatonotus muelleri (Boettger, 1885)	L1, ES3	T	O	1	LC
Elachistocleis cesarii (Miranda-Ribeiro, 1920)	L2 ï L12, L14, L16 ï L18, ES1 ï ES3	T	W	1	NL
Odontophrynidae					
Odontophrynus cultripes Reinhardt and Lütken, 1862	L1, L3, L12, L13	T	E	1	LC
Odontophrynus salvatori Caramaschi, 1996	L3	T	E	1	DD

Table 3. Species list of anurans sampled in the state of Goiás according to scientific literature. Abbreviations: A = Association degree with the Cerrado, E = Cerrado endemic, O = species that occur in open domains, W = widely distributed, S = species with meridional distribution that occur in Cerrado, AM = species occurring in both Amazonia and Cerrado, AT = species occurring both in Atlantic Forest and Cerrado, CA = species occurring both in Caatinga and Cerrado, , RM = Reproductive Mode (*sensu*Haddad and Prado, 2005; Wells, 2007; Haddad et al., 2013), RL = Red List (IUCN 2014), NL = No Listed, LC = Least Concern, DD = Data Deficient.

TAXA	A	RM	RL	REFERENCE
AMPHIBIA				
ANURA				
Aromobatidae				
Allobates goianus (Bokermann, 1975)	E	20	DD	Bastos et al. 2003, Valdujo et al. 2012
Bufonidae				
Rhaebo guttatus (Schneider, 1799)	AM	1	LC	Valdujo et al. 2012
Rhinella cerradensis Maciel, Brandão, Campos, and Sebben, 2007	E	1*	DD	Valdujo et al. 2012, Santos et al. 2014
Rhinella inopina Vaz-Silva, Valdujo, and Pombal, 2012	E	1*	NL	Vaz-Silva et al. 2012
Rhinella mirandaribeiroi (Gallardo, 1965)	E	1	LC	Cintra et al. 2009, Vaz-Silva et al. 2007, Morais et al. 2011, Valdujo et al. 2012, Mello et al. 2013
Rhinella ocellata (Günther, 1858)	E	1*	LC	Borges and Juliano 2007, Vaz-Silva et al. 2007, Santos et al. 2014
Rhinella rubescens (Lutz, 1925)	E	1	LC	Bastos et al. 2003
Rhinella schneideri (Werner, 1894)	W	1 ï 2	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Campos and Vaz-Silva 2010, Oda et al. 2009, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014
Brachycephalidae				
Ischnocnema juipoca (Sazima and Cardoso, 1978)	S	23	LC	Bastos et al. 2003
Craugastoridae				

Barycholos ternetzi (Miranda-Ribeiro, 1937)	E	23	LC	Bastos et al. 2003, Borges and Juliano 2007, Cintra et al. 2009, Oda et al. 2009, Campos and Vaz-Silva 2010, Nomura et al. 2012, Santos et al. 2014
Oreobates remotus Teixeira, Amaro, Recoder, Sena, and Rodrigues, 2012	E	23	NL	Andrade et al. 2012
Pristimantis ventrigranulosus Maciel, Vaz-Silva, Oliveira, and Padial, 2012	E	23	NL	Valdujo et al. 2012
Dendrobatidae				
Ameerega berohoka Vaz-Silva and Maciel, 2011	E	20	NL	Vaz-Silva and Maciel 2011
Ameerega flavopicta (Lutz, 1925)	E	20	LC	Oda et al. 2009, Santos et al. 2014
Hylidae				
Aplastodiscus perviridis Lutz, 1950	AT	5	LC	Bastos et al. 2003
Bokermannohyla pseudopseudis (Miranda-Ribeiro, 1937)	E	2	LC	Valdujo et al. 2012
Bokermannohyla sapiranga Brandão, Magalhães, Garda, Campos, Sebben, and Maciel, 2012	E	2	NL	Brandão et al. 2012
Corythomantis greeningi Boulenger, 1896	CA	1	LC	Pombal Jr. et al. 2012
Dendropsophus anataliasiasi (Bokermann, 1972)	E	1	LC	Cintra et al. 2009
Dendropsophus cruzi (Pombal and Bastos, 1998)	E	1	LC	Bastos et al. 2003, Vaz-Silva et al. 2007, Oda et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014
Dendropsophus jimi (Napoli and Caramaschi, 1999)	E	1	LC	Borges and Juliano 2007, Vaz-Silva et al. 2007, Kopp et al. 2010, Morais et al. 2011
Dendropsophus melanargyreus (Cope, 1887)	W	1	LC	Nomura et al. 2012, Santos et al. 2014
Dendropsophus minutus (Peters 1872)	W	1	LC	Bastos et al. 2003; Borges and Juliano 2007, Vaz-Silva et al. 2007; Cintra et al. 2009, Oda et al. 2009; Campos and Vaz-Silva 2010; Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014

Dendropsophus nanus (Boulenger, 1889)	W	1	LC	Vaz-Silva et al. 2007, Cintra et al. 2009, Campos and Vaz-Silva 2010, Morais et al. 2011, Nomura et al. 2012, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014
Dendropsophus rubicundulus (Reinhardt and Lütken, 1862)	E	1	LC	Bastos et al. 2003, Vaz-Silva et al. 2007, Campos and Vaz-Silva 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014
Dendropsophus soaresi (Caramaschi and Jim, 1983)	CA	1	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Oda et al. 2009, Morais et al. 2011, Santos et al. 2014
Hypsiboas albopunctatus (Spix, 1824)	W	1	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Oda et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014
Hypsiboas crepitans (Wied-Neuwied, 1824)	W	4	LC	Cintra et al. 2009
Hypsiboas ericae (Caramaschi and Cruz, 2000)	E	1	DD	Valdujo et al. 2012
Hypsiboas goianus (Lutz, 1968)	E	1 ï 2	LC	Bastos et al. 2003, Valdujo et al. 2012
Hypsiboas lundii (Burmeisteri, 1856)	E	4	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Oda et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014
Hypsiboas paranaiba Carvalho and Giaretta, 2010	E	1	NL	Vaz-Silva et al. 2007, Oda et al. 2009, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014
Hypsiboas phaeopleura (Caramaschi and Cruz, 2000)	E	1	DD	Valdujo et al. 2012
Hypsiboas punctatus (Schneider, 1799)	W	1 ï 2	LC	Mello et al. 2013

Hypsiboas raniceps (Cope, 1862)	W	1	LC	Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Oda et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013
Lysapsus caraya Gallardo, 1964	E	1	LC	Valdujo et al. 2012, Mello et al. 2013
Phyllomedusa azurea Cope, 1862	E	24	DD	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Oda et al. 2009, Campos and Vaz-Silva 2010, Morais et al. 2011, Nomura et al. 2012, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014
Phyllomedusa nordestina Caramaschi, 2006	CA	24	DD	Valdujo et al. 2012
Phyllomedusa oreades Brandão, 2002	E	24	DD	Valdujo et al. 2012
Pseudis bolbodactyla Lutz, 1925	AT	1	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Campos and Vaz-Silva 2010, Morais et al. 2011, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014
Scinax centralis Pombal and Bastos, 1996	E	1 ï 2	LC	Bastos et al. 2003, Campos and Vaz-Silva 2010, Nomura et al. 2012, Valdujo et al. 2012
Scinax constrictus Lima, Bastos and Giaretta, 2004	E	1	LC	Campos and Vaz-Silva 2010, Morais et al. 2011, Valdujo et al. 2012, Mello et al. 2013
Scinax fuscomarginatus (Lutz, 1925)	W	1	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Oda et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014
Scinax fuscovarius (Lutz, 1925)	W	1	LC	Vaz-Silva et al. 2007, Cintra et al. 2009, Oda et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014
Scinax pusillus Pombal, Bilate, Gambale, Signorelli, and Bastos, 2011	E	1	NL	Morais et al. 2011, Valdujo et al. 2012

Scinax rogerioi Pugliese, Baêta, and Pombal, 2009	E	1	NL	Pugliese et al. 2009
Scinax similis (Cochran, 1952)	AT	1	LC	Cintra et al. 2009, Nomura et al. 2012
Scinax skaios Pombal, Carvalho, Canelas, and Bastos, 2010	E	1	NL	Valdujo et al. 2012
Scinax squalirostris (Lutz, 1925)	S	1	LC	Valdujo et al. 2012
Scinax x-signatus (Spix, 1824)	W	1	LC	Vaz-Silva et al. 2007, Campos and Vaz-Silva 2010, Morais et al. 2011, Santos et al. 2014
Trachycephalus mambaiensis Cintra, Silva, Silva, Garcia, and Zaher, 2009	E	1	NL	Cintra et al. 2009
Trachycephalus typhonius (Linnaeus, 1758)	W	1	LC	Borges and Juliano 2007, Vaz-Silva et al. 2007, Nomura et al. 2012, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014
Leptodactylidae				
Adenomera cotuba Carvalho and Giaretta, 2013	E	32	NL	Carvalho and Giaretta 2013
Adenomera hylaedactyla (Cope, 1868)	AM	32	LC	Bastos et al. 2003, Kopp et al. 2010, Morais et al. 2011, Santos et al. 2014
Adenomera juikitam Carvalho and Giaretta, 2013	E	32	NL	Carvalho and Giaretta 2013
Adenomera saci (Bokermann, 1956)	E	32	LC	Oda et al. 2009, Kopp et al. 2010, Valdujo et al. 2012
Leptodactylus furnarius Sazima and Bokermann, 1978	E	30	LC	Cintra et al. 2009, Kopp et al. 2010, Morais et al. 2011, Valdujo et al. 2012
Leptodactylus fuscus (Schneider, 1799)	W	11 ï 30	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Campos and Vaz-Silva 2010, Oda et al. 2009, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014
Leptodactylus labyrinthicus (Spix, 1824)	W	11	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Oda et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014

Leptodactylus latrans (Steffen, 1815)	W	11 ï 30	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Oda et al. 2009, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013; Santos et al. 2014
Leptodactylus leptodactyloides (Andersson, 1945)	AM	11	LC	Nomura et al. 2012
Leptodactylus mystaceus (Spix, 1824)	W	30	LC	Borges and Juliano 2007, Oda et al. 2009, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014
Leptodactylus mystacinus (Burmeister, 1861)	AT	30	LC	Borges and Juliano 2007, Cintra et al. 2009, Oda et al. 2009, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014
Leptodactylus petersii (Steindachner, 1864)	AM	11	LC	Valdujo et al. 2012
Leptodactylus podicipinus (Cope, 1862)	O	11	LC	Borges and Juliano 2007, Vaz-Silva et al. 2007, Kopp et al. 2010, Morais et al. 2011, Mello et al. 2013, Santos et al. 2014
Leptodactylus pustulatus (Peters, 1870)	E	11	LC	Mello et al. 2013
Leptodactylus sertanejo Giaretta and Costa, 2007	E	11	LC	Kopp et al. 2010, Mello et al. 2013
Leptodactylus syphax Bokermann, 1969	O	11	LC	Oda et al. 2009, Campos and Vaz-Silva 2010, Morais et al. 2011, Nomura et al. 2012, Santos et al. 2014
Leptodactylus tapiti Sazima and Bokermann, 1978	E	30	DD	Valdujo et al. 2012
Leptodactylus vastus Lutz, 1930	CA	11 ï 13	LC	Valdujo et al. 2012
Physalaemus centralis Bokermann, 1962	E	11	LC	Bastos et al. 2003, Vaz-Silva et al. 2007, Cintra et al. 2009, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014
Physalaemus cuvieri Fitzinger, 1826	W	11	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007; Cintra et al. 2009, Oda et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014

Physalaemus marmoratus (Reinhardt and Lütken, 1862)	E	11	LC	Vaz-Silva et al. 2007, Mello et al. 2013
Physalaemus nattereri (Steindachner, 1863)	E	11	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Nomura et al. 2012, Mello et al. 2013, Santos et al. 2014
Pleurodema diplolister (Peters, 1870)	CA	11	LC	Andrade and Vaz-Silva 2012
Pseudopaludicola falcipes (Hensel, 1867)	S	1	LC	Vaz-Silva et al. 2007, Cintra et al. 2009, Morais et al. 2011
Pseudopaludicola mystacalis (Cope, 1887)	W	1	LC	Cintra et al. 2009, Kopp et al. 2010, Santos et al. 2014
Pseudopaludicola saltica (Cope, 1887)	E	1	LC	Kopp et al. 2010, Morais et al. 2011, Santos et al. 2014
Pseudopaludicola ternetzi Miranda-Ribeiro, 1937	E	1	LC	Valdujo et al. 2012
Microhylidae				
Chiasmocleis albopunctata (Boettger, 1885)	E	1	LC	Bastos et al. 2003, Borges and Juliano 2007, Vaz-Silva et al. 2007, Oda et al. 2009, Nomura et al. 2012, Valdujo et al. 2012, Mello et al. 2013, Santos et al. 2014
Chiasmocleis centralis Bokermann, 1952	E	1	DD	Valdujo et al. 2012
Dermatonotus muelleri (Boettger, 1885)	O	1	LC	Cintra et al. 2009
Elachistocleis cesarii(Miranda-Ribeiro, 1920)	W	1	NL	Borges and Juliano 2007, Vaz-Silva et al. 2007, Cintra et al. 2009, Oda et al. 2009, Campos and Vaz-Silva 2010, Kopp et al. 2010, Morais et al. 2011, Mello et al. 2013, Nomura et al. 2012, Santos et al. 2014
Odontophrynidae				
Odontophrynus cultripes Reinhardt and Lütken, 1862	E	1	LC	Bastos et al. 2003, Santos et al. 2014
Odontophrynus salvatori Caramaschi, 1996	E	1	DD	Valdujo et al. 2012

Proceratophrys bagnoi Brandão, Caramaschi, Vaz-Silva, and Campos, 2013	E	1 ï 2*	NL	Brandão et al. 2013
Proceratophrys cristiceps (Müller, 1883)	E	1 ï 2	LC	Cintra et al. 2009, Oda et al. 2009
Proceratophrys dibernardoi Brandão, Caramaschi, Vaz-Silva, and Campos, 2013	E	1 ï 2*	NL	Brandão et al. 2013
Proceratophrys goyanus (Miranda-Ribeiro, 1937)	E	1 ï 2	LC	Bastos et al. 2003, Oda et al. 2009, Nomura et al. 2012, Valdujo et al. 2012, Santos et al. 2014
Proceratophrys vielliardi Martins and Giaretta, 2011	E	1 ï 2	NL	Valdujo et al. 2012
Pipidae				
Pipa pipa (Linnaeus, 1758)	AM	16	LC	Vaz-Silva and Andrade 2009, Valdujo et al. 2012
Ranidae				
Lithobates palmipes (Spix, 1824)	AM	1	LC	Valdujo et al. 2012

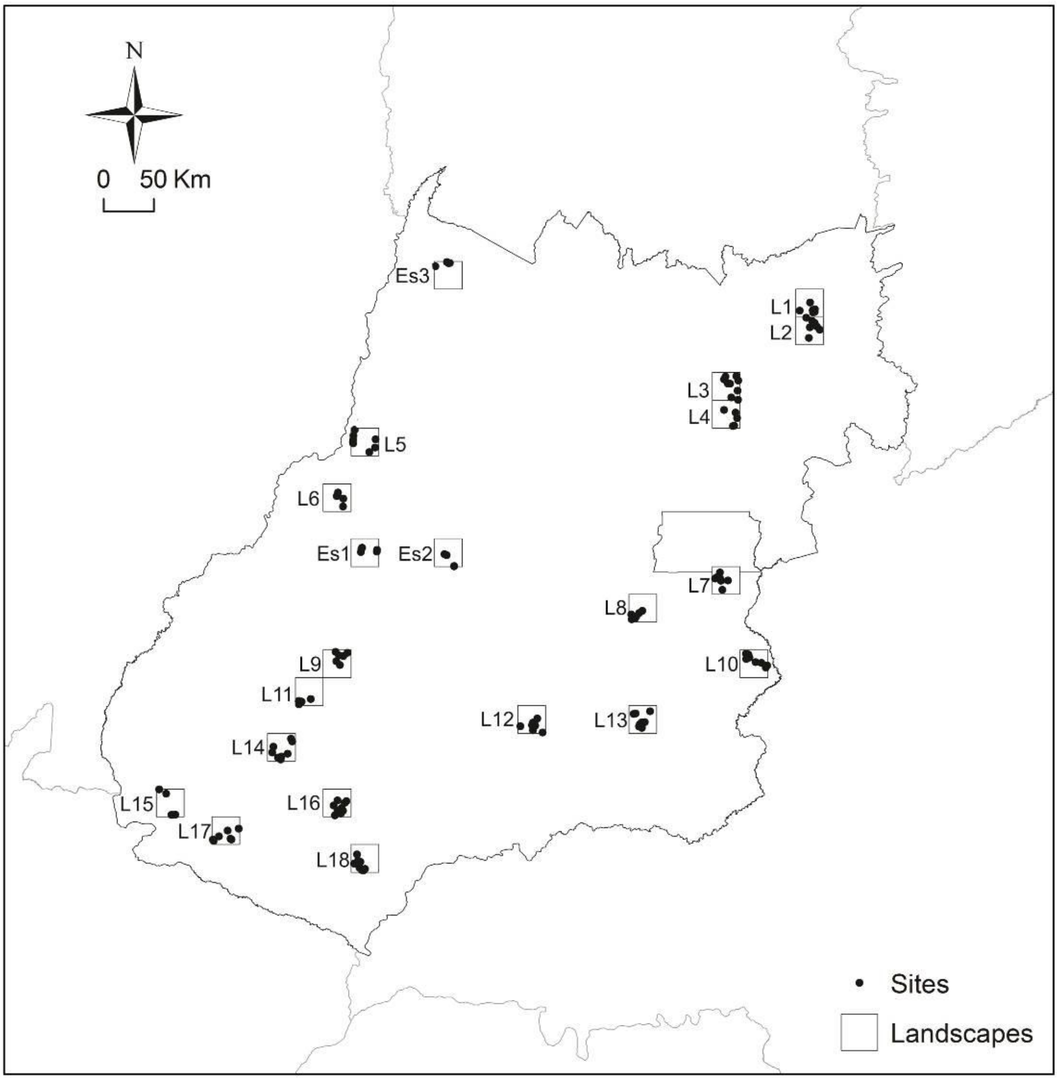


Figure 1. Landscapes and sites surveyed in the state of Goiás, Brazil.

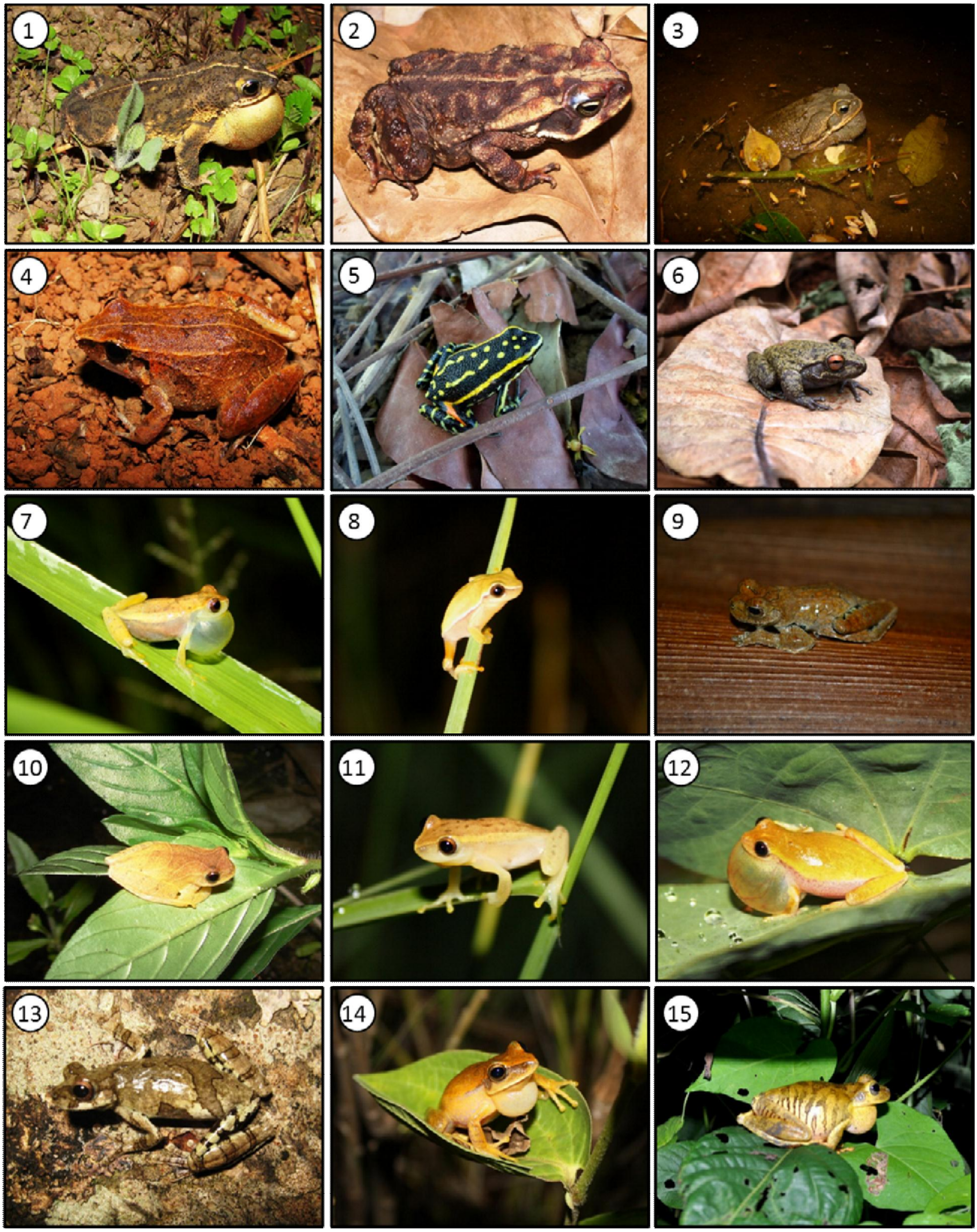


Figure 2. Species recorded in the state of Goiás: 1- *Rhinella mirandaribeiroi*, 2 - *R. rubescens*, 3 ï *R. schneideri*, 4- *Barycholos ternetzi,* 5- *Ameerega flavopicta*, 6- *Bokermannohyla sapiranga,* 7 - *Dendropsophus cruzi*, 8- *D. jimi*, 9- *D. melanargyreus*, 10 - *D. minutus*, 11 - *D. nanus*, 12 - *D. rubicundulus*, 13 ï *D. soaresi*, 14- *Hypsiboas albopunctatus*, 15- *H. creptans*.

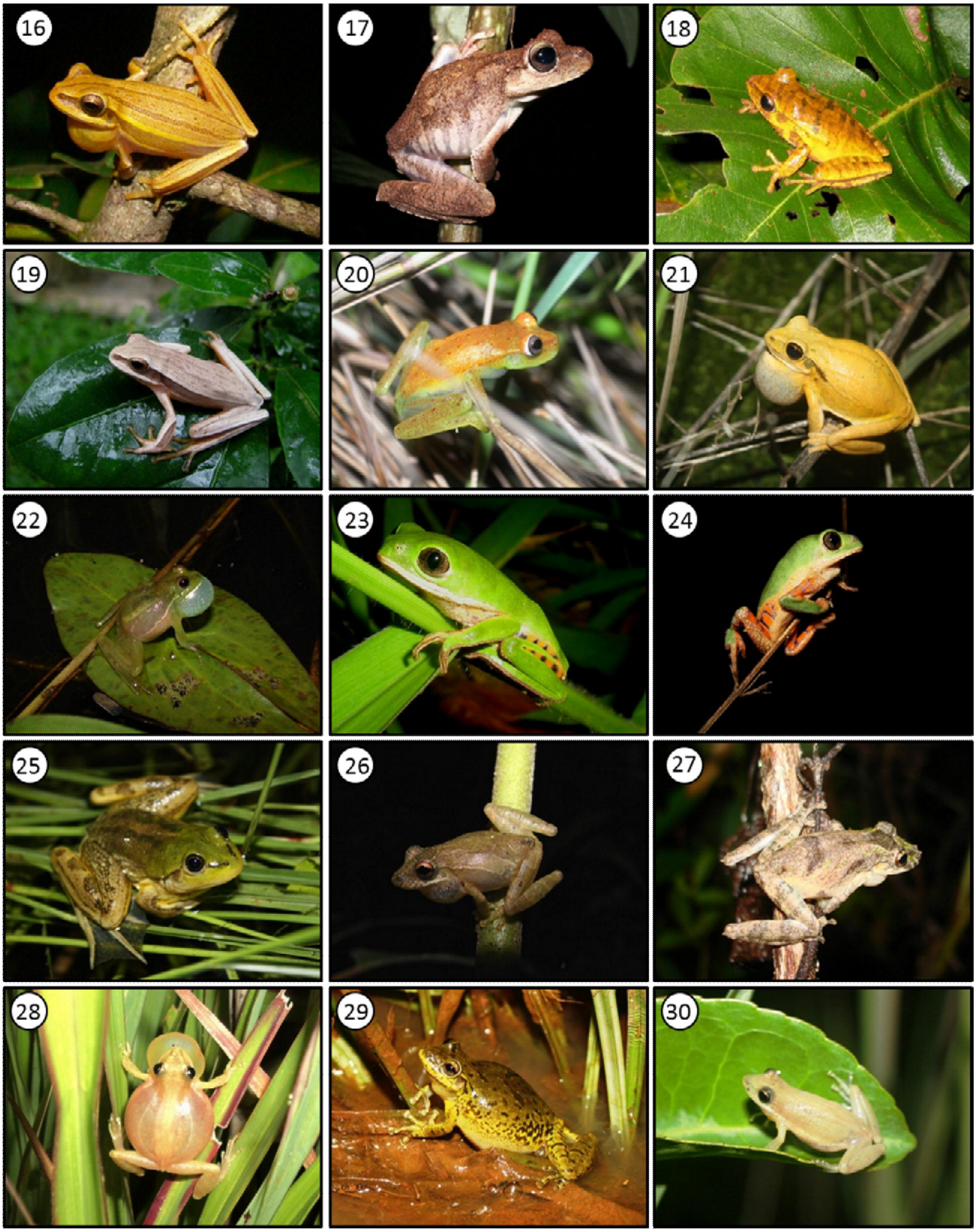


Figure 3. Species recorded in the state of Goiás: 16- *Hypsiboas goianus*, 17 - *H. lundii*, 18 - *H. paranaiba*, 19 - *H. phaeopleura,* 20- *H. punctatus*, 21 - *H. raniceps,* 22 - *Lysapsus caraya*, 23 - *Phyllomedusa auzurea*, 24 - *P. oreades*, 25 - *Pseudis bolbodactyla*, 26 - *Scinax centralis*, 27 - *Scinax constrictus*, 28 - *S. fuscomarginatus*, 29 - *S. fuscovarius*, 30- *S. pusillus*.

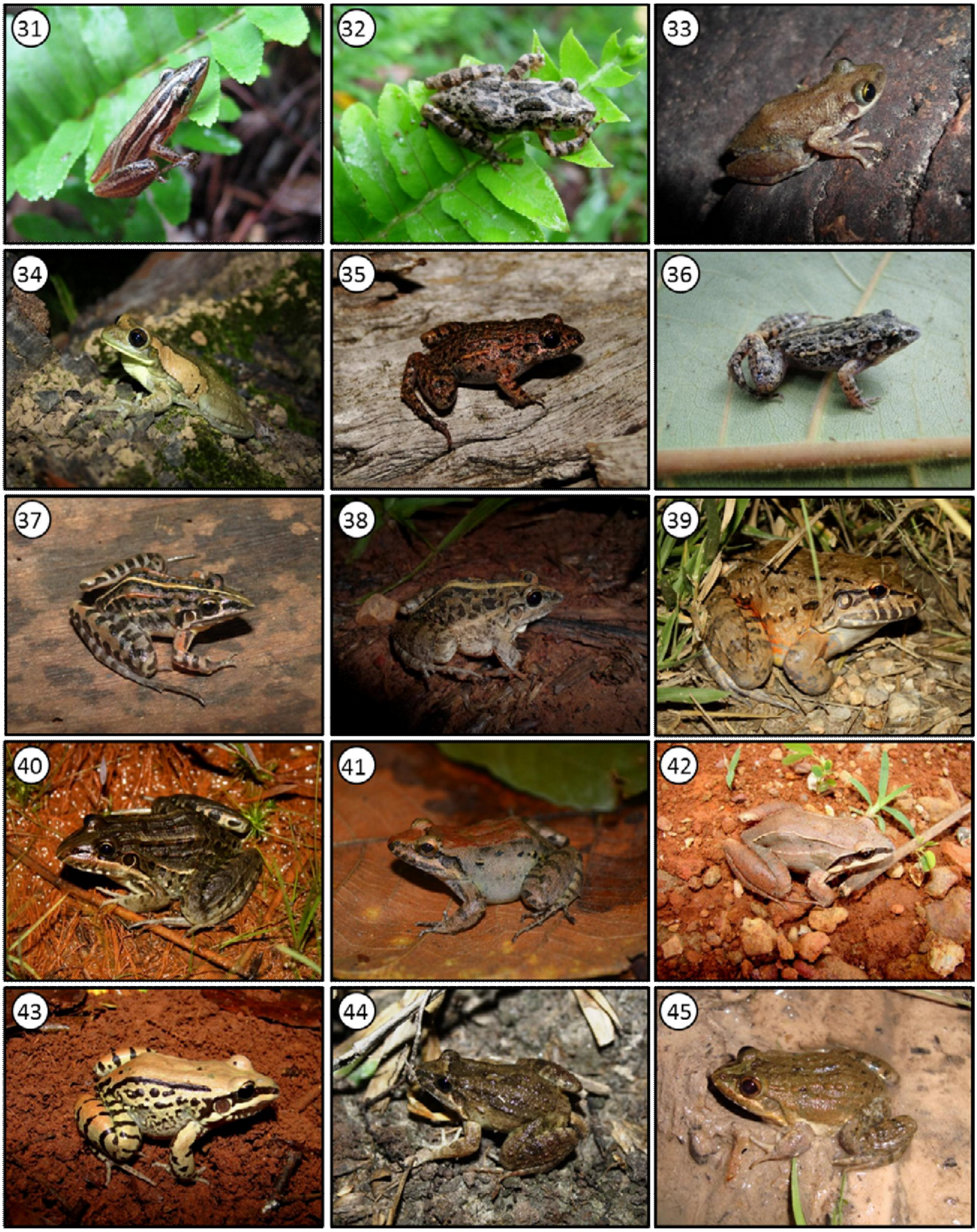


Figure 4. Species recorded in the state of Goiás: 31- *Scinax squalirostris,* 32 - *S. rogerioi*, 33 - *S. similis*, 34 - *Trachycephalus typhonius,* 35 - *Adenomera hylaedactyla*, 36 - *A. saci,* 37 - *Leptodactylus furnarius*, 38 - *L. fuscus*, 39 - *L. labyrinthicus*, 40 - *L. latrans*, 41 - *L. leptodactyloides*, 42 - *L. mystaceus*, 43 - *L. mystacinus*, 44 - *L. podicipinus*, 45- *L. pustulatus*.

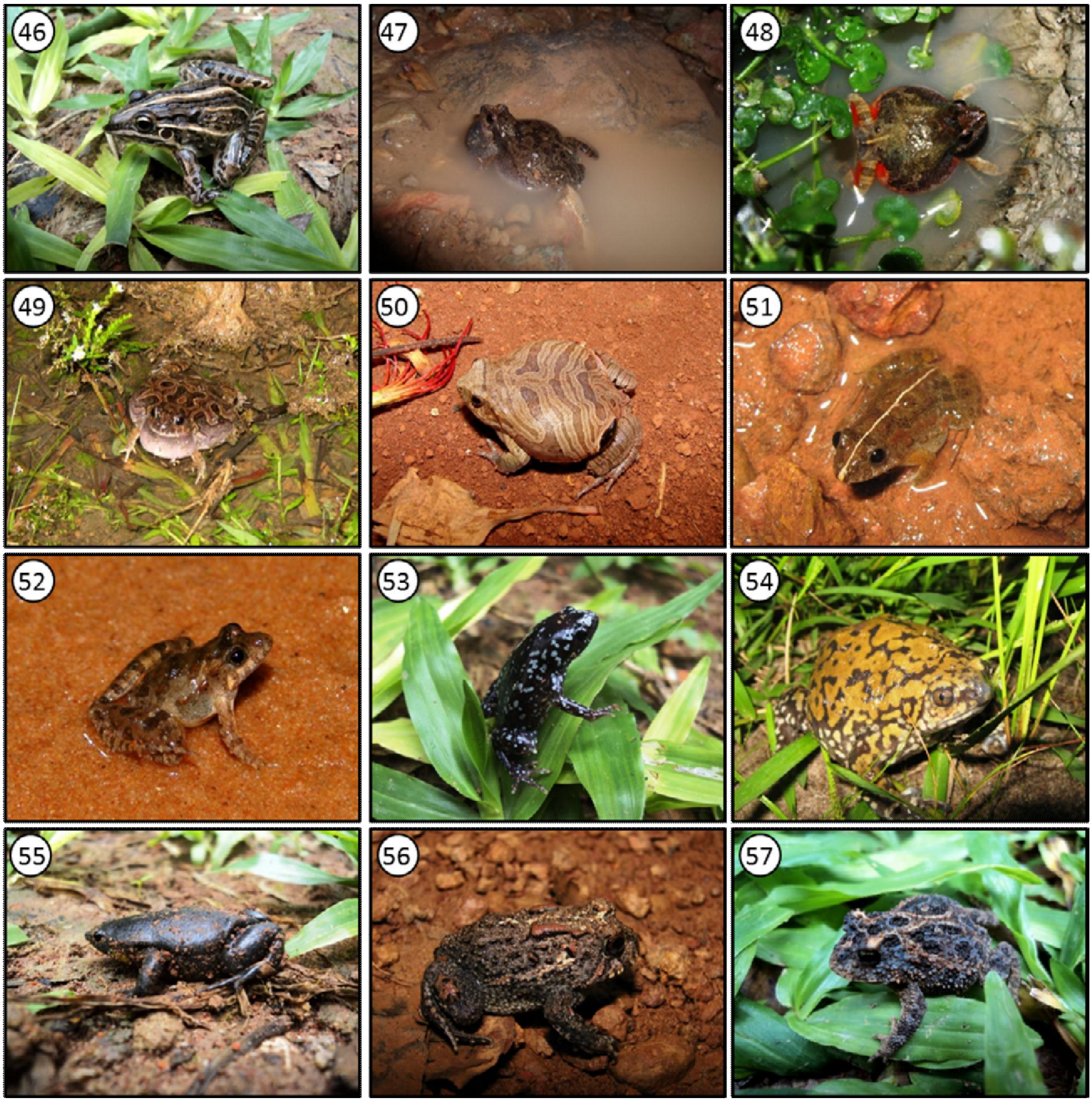


Figure 5. Species recorded in the state of Goiás: 46- *Leptodactylus sertanejo*, 47 - *Physalaemus centralis*, 48 - *P. cuvieri*, 49 - *P. marmoratus,* 50 - *P. nattereri*, 51 - *Pseudopaludicola mystacallis,* 52 - *P. saltica*, 53- *Chiasmocleis albopunctata*, 54 - *Dermatonotus muelleri*, 55 - *Elachistocleis cesarii*, 56 - *Odontophrynus cultripes*, 57 - *O. salvatori*

CAPÍTULO II - More individual hypothesis and habitat heterogeneity drive amphibian ponds metacommunities

---

Luciana Signorelli [1]; Denis S. Nogueira [1;] Caroline C. Corrêa [1]; Rogério P. Bastos[2]; Paulo De Marco Júnior[3]

[1] Programa de Pós Graduação em Ecologia e Evolução, Departamento de Ecologia, Universidade Federal de Goiás, 74.001-970 Goiânia, Goiás, Brasil.

[2] Laboratório de Herpetologia e Comportamento Animal, Departamento de Ecologia, Universidade Federal de Goiás, 74.001-970 Goiânia, Goiás, Brasil

[3]Laboratório de Ecologia Teórica e Síntese, Departamento de Ecologia, Universidade Federal de Goiás, 74.001-970 Goiânia, Goiás, Brasil.

Corresponding author: lfl.signorelli@gmail.com

Formatted according to *PLOS One*

Abstract

We investigated the area, energy and heterogeneity effects on anuran species richness and beta diversity in ponds from Brazilian Cerrado. We were able to test some of the most claimed mechanism argued to explain diversity patterns of anurans in ecosystems with relatively discrete boundaries, such ponds: the species-area relationships, species-energy relationships and heterogeneity-diversity relationships. In addition, we attempt to clarify if these predictors explain beta diversity in order to account for the main drivers of the anuran metacommunities. Alpha and beta diversity were driven by different mechanisms acting in distinct scales: While alpha diversity was explained by productivity and local heterogeneity mainly, beta diversity was explained by only heterogeneity and complexity in local and in the landscape scale (a buffer around the ponds). We discuss the effect of productivity in the context of theoretical expectation of hypothesis of more individuals, while heterogeneity influence upon anurans under lights of metacommunity niche theory (*e.g.*, sorting species). These results highlights the importance of different processes driving anuran regional diversity pattern in the hotspots Brazilian Cerrado, and enable us to discuss how improve the understanding of conservation strategies from this threatened group.

Key-words: Alpha diversity, beta diversity, species-area relationship, species-energy, Anura, pondsmetacommunities

Introduction

The intriguing patterns of species diversity and composition has generated a set of theoretical models in metapopulation and metacommunity ecology since the middle 1960s [1]ï [3]. The species-area relationship (SAR), one attractive model based on the simply initial prediction proposed by Arrhenius [4] that the number of species increases continuously with the area. These patterns was incorporated by the Island Biogeography Theory (IBT) [3] providing a meaningful process to explain SAR. In agreement with IBT, the balance between immigration of new species and extinction of resident species are both dependent on the area and the isolation of the island. Thus, the total number of species present on islands are constant due temporal turnover, with islands near to the source pool expected to have more species at equilibrium, because of rescue effect of the immigration rate minimizing the local species extinction [5]**.**

As an alternative to IBT,Wright [1] proposed the species-energy theory (SET), which suggests that population sizes are principally affected by available energy, measured by the total amount of available resource production on an island. Differences in available energy between islands can distort the species-area relationship, suggesting that area alone cannot be a precise predictor of species diversity. Increases in available energy, similar to increases in area, should result in proportional increases in the total number of individuals [1]. In animal communities, for instance, the positive correlations between species richness and productivity are attributed to the ñMore Individuals Hypothesisò(MIH) [6], in which species richness in a given area is limited by the productivity and ability of such sites to support large populations of each species. Thus, positive correlations among total number of individuals (a proxy to productivity), and richness are sufficient to corroborate both the more individuals hypothesis and species-energy theory [6].

Both theories, TBI and SET, have been focused only in stochastic effects of energy and area on species richness, non-considering the importance of habitat heterogeneity. The habitat heterogeneity is an important component that could be associated with species-area relationship, since habitat availability tends to increase as the area increases [7], [8]. In this case, larger areas probably would have different kinds of habitats [7], it would expected to hold species with different requirements allowing more species to coexist [9], [10]. According to SET, area is a variable that could be correlated to the heterogeneity of resources, being an indirect available energy measure [1]. Larger areas could provide a greater portion of available habitat and a greater amount of resources [7], consequently, a greater number of individuals and species [1].

The area, energy and habitat quality effects on species richness vary widely among taxonomic groups [1],[8],[11],[12], however, such effects have been less explored regarding change in species composition (beta diversity) [13]. Otherwise ponds communities are bounded by terrestrial habitat, and may function like islands of habitat inserted in a terrestrial matrix [14],[15]. The degree which this terrestrial habitats affect the communities will differ between species [14]. Anurans populations, for example, which use ponds as sites for ovipostion, are inserted in a discret habitat patch wich are finely shaped on the environment around [16]. Most of them have limited vagility and some ecological, physiological and behavioral characteristics that restrict their dispersion ability [17]ï [19]. Due to their permeable skin, anurans should select microhabitats that provide moisture for water uptake through the skin, which limits the migration distance [19]. Furthermore, most anurans species show little resistance to evaporative water loss from the body [18],[20]. For these reasons amphibians are good models for the study of some driving mechanisms in the metacommunity dynamics.

Herein, our study encompass both local and landscape factors driving alpha and beta diversity of anuran metacommunities on breeding ponds. Firstly, we investigated the area effects on anuran species richness, and our prediction, according to TBI, is that (i) alpha diversity increases with pond sizes due the effect that passive sampling of rare species insofar more individuals are sampled. Given that the total number of individuals in a community may reflect the increase on local productivity [1] and this effect is not necessarily related with pond size, our prediction is that (ii) alpha diversity increases with the number of individuals (as a proxy for productivity) in local communities. Considering that environmental heterogeneity can explain the variation on richness because of more habitats enable more distinct species to co-occur locally, regardless of the area and productivity effect, we predict that (iii) alpha diversity increases with habitat heterogeneity, and this effect is potentially higher considering landscape heterogeneity and complexity around the ponds. Finally, we explore possible relations that could emerge from the landscape scale and will be more related to the spatial organization of both ponds and vegetation remnants in this area. We test the effect of isolation, local and landscape complexity and heterogeneity, and abundance on beta diversity among the ponds. We predict that iv) amphibians beta diversity would increase along with isolation due to dispersive limitation; v) increase along with local and landscape complexity and heterogeneity due to niche availability, and vi) increase along with abundance as an indicative of positive effect of local productivity within the ponds.

Methods

We obtained the dataset used in this study from the database of the Animal Behavior and Herpetology Lab, from the Federal University of Goiás, Brazil. We compiled data from 39 water bodies distributed along the southeastern region of state of Goiás, Brazil (Figure 1). As a criterion of selection we used samples that followed the same protocol and whose water bodies presented different areas. Search for anurans lasted 1h per site and was made along the perimeter of breeding ponds with combination of both visual and acoustic encounters [30]. Ponds were sampled from 1800 until 2400, and were surveyed once during the rainy season (October to March) between 2007 and 2011.

Using the geographic coordinates of each pond, we measured the pond perimeter and the distance from the nearest breeding pond by recent high-resolution aerial photographs of the region (available from Google Earth). We prefer to use perimeter instead of total area because in most of anuran species use the edges of ponds to reproduce [27]. The just one exception registered in our case was *Pseudis bolbodactyla*, which occur in aquatic or semi aquatic habitats.

Habitat diversity was represented by local heterogeneity and landscape heterogeneity and complexity. The local heterogeneity was measured in the field, using a protocol of physical environmental heterogeneity (modified from Nessimian *et al.*[31]). This protocol comprises five variables that describe the environmental conditions. Each item is composed of three to five alternatives arranged to represent more heterogeneous systems. The following variables were analyzed: depth up to two meters from the edge: (1) shallow (30 cm), (2) intermediate (31-50 cm), (3) deep (greater than 51 cm); b) vegetation in the interior of the pond (VI): (1) no vegetation, (2) one type, (3) two types, (4) three or more types; c) vegetation in the edge (VE): (1) no vegetation, (2) one type, (3) two types, and (4) three or more types; d) vegetation cover on the pond surface (VC), visually estimated: (1) no vegetation, (2) up to 25%, (3) 26-50%,(4) 51-75%, and (5) 76-100%; e) profile of the pond edge (PE; slope, flat and gully): (1) one type, (2) two types, and (3) three types. The local environmental heterogeneity data was transformed to reduce the effect of categorical variables, so that each item would have equal weights in the analysis, the observed values were standardized by dividing by the maximum value possible for the item, defined as:

$$p_i = \frac{a_o}{a_m} \qquad \text{Eq. 1}$$

Where *pi* is the environmental variable, a_o is the observed values and a_m is the maximum value possible for the item.

To reduce the dimensionality relative to the local heterogeneity data set we carried on a principal component analysis (PCA). Our objective was to obtain a small number of axes that would explain most of the total environmental heterogeneity variation among ponds. We selected axes with eigenvalues greater than the average value of all eigenvalues, using Kaiser-Guttman criterion [32], [33].Moreover, we discriminated ponds inserted in agriculture matrix and pasture matrix according to field inspection and used this category to classify ponds in relation to heterogeneity PCA ordination.

The landscape heterogeneity and complexity were calculated using the normalized difference vegetation index (NDVI), which provides information on vegetation distribution and dynamics [33]. Various studies have shown that NDVI is a useful tool for investigating richness, distribution, abundance or life history traits from insects [35], birds [36] and mammals [37]. We can use NDVI to characterize heterogeneity and complexity of vegetation in the landscape. The landscape complexity is represented by average of vegetation index and is related to vertical systems differentiation. Heterogeneity is represented by the standard deviation of the vegetation index, representing the spatial stratification.

The NDVI was calculated from the buffers of 60m around ponds edges, so that ponds area does not be included. The images were obtained from the Landsat 5TM. The images were selected according to the sampling date, trying to approximate the most of same rain period in which samples were collected. Differently the local heterogeneity measure, landscape heterogeneity and complexity reflect the horizontal and vertical structure of vegetation, respectively.

Species richness: species-area relationship, species-abundance and habitat heterogeneity

The hypothesis of species-area, energy, local and landscape environmental were tested by multiple regression analyses. It was possible because we did not found significative relationships among the predictors, with all regression present high tolerance level (see table 2). To demonstrate that our results were not in function of problems associated with sampling methods, we used individual-based rarefaction [38] and we repeated the analyses substituting observed richness for rarefied richness. Besides, we did a simple regression between the perimeter (predictor variable) and number of individuals (response variable) to verify if the number of individuals will increase with the perimeter of ponds. Finally, we tested if local and landscape heterogeneity and landscape complexity were influenced by perimeter of ponds, to account for colinearity between predictors with simple regression. The abundance data were transformed using base-10 logarithms to data normalization [39].

Beta diversity: abundance, local environmental heterogeneity and landscape heterogeneity and complexity

To evaluate if the difference on species composition among anurans assemblage can be explained by the environmental predictors of ponds, we used beta diversity estimates based on pairwise dissimilarity index [40]. In order to specifically to observe patterns of species substitution, we calculated the dissimilarity by a probabilistic index based on frequency of species occurrence, namely Raup-Crick dissimilarity (see details in Chase *et al.*[13]). The Raup-Crick dissimilarity was calculated maintaining the column marginal frequencies as probabilities, with 10000 randomizations. We used the Raup Crick dissimilarity matrix to calculate multivariate regression analysis on distance matrices [41]. The probabilistic dissimilarity matrix generated by Raup-Crick index was previously transformed (square root) to control the formation of negative eigenvalues [33]. This procedure satisfactorily controlled the negative eigenvalues of our data. For this, we used vegan package of R software [42].

Results

We registered 36 species of anurans in 39 sampling sites, located in the southeastern region of state of Goiás, Brazil. Species are distributed among seven families, being Hylidae the most representative, following by Leptodactylidae, Leiuperidae, Bufonidae and Cycloramphidae, respectively (reported in Table S1 in File S1). The other families were represented by one species. Just four of the 36 species occurred in more than 50% of sampling ponds (*Dendropsophus minutus*, *Hypsiboas albopunctatus*, *Scinax fuscomarginatus* e *Physalaemus cuvieri*).

Local heterogeneity

Ordination of local heterogeneity variables reduced data dimensionality into two main axes that accounts for 63.57% of the total variation. The first principal component accounts for 42.35% with the most important variables being: vegetation in the interior of the pond, vegetation cover on the pond surface and vegetation in the edge (Table 1, Fig. 2). The second principal component accounts for 21.04% with the most important variables being: depth, vegetation in the edge and profile of the pond edge (Table 1). The local heterogeneity measured by the principal components of PCA reflects the local heterogeneity which indicates the presence of a kind of vertical or horizontal structure, not only in the edges but also in the interior of the ponds. Moreover, there is no difference between ponds in agricultural or pasture matrix (Fig. 2).

Alpha diversity: species-area relationship, species-abundance and habitat heterogeneity

Alpha diversity of breeding ponds anurans assemblage is not related with area and/or with landscape variables, rejecting the hypotheses that alpha diversity increases in larger areas and with landscape heterogeneity or complexity (Table 2). Our results corroborate the hypothesis that alpha diversity increases with the number of individuals in local communities (Figure 3A), but the number of individuals is not related to the size of the sampled area ($R^2=0.062$; $p=0.126$). Supporting the more individuals hypothesis the relationships found when we used richness controlled by individual-based rarefaction, practically not differ from the results with observed richness, giving a control for passive sampling effects of the effect of bigger populations (Figure 3B; Table 3).

Beta diversity: abundance, local environmental heterogeneity and landscape heterogeneity and complexity

The isolation between sampling ponds and the number of individuals were not important to determined beta diversity on anuran assemblages. When we evaluated the effect of local heterogeneity, we found that the second principal component of PCA and the interaction between the first and second principal components of PCA were correlated with beta diversity. The landscape complexity and the interaction between landscape complexity and heterogeneity were related with beta diversity (Table 3).

Discussion

Anuran diversity in breeding ponds was consistently related to the number of individuals and to heterogeneity within and around ponds. The theory proposed by Wright [1] recognizes available energy as main factor that determines species richness. Like other authors [43],[44], we do not directly use available energy within the system, but the evidence of how much energy is available: the total number of individuals inhabiting ponds. Our results support the More Individual Hypothesis [6] and indicate the importance of productivity to maintenance of amphibians local diversity within a region. It suggests that high species richness is a consequence of sites carrying support to large populations and, moreover, indicated that each Anuran species should be able to select sites according to local characteristics indicative of available energy. Anurans assemblage inproductive sites probably translated it into more individuals of each species by having bigger breeds, which plausibly decrease their chance to become extinct locally, and in consequence holds more distinct species together, according to theoretical expectations [1].

The species-area relationship has been used as a resource that summarizes diversity patterns for a large number of taxa [21], but have limitations when applied to ponds [11],[12],[22]. Some groups of organisms, like as fishes [23], aquatic Coleoptera, Odonata and Gastropoda, have a good fit to species-area relationship [11], however, these are not the case for Amphibians [12], for which this relation can be inexistent [11],[24]ï [27] or negative [23],[28]. These departures from expected patterns can be related to predation pressure by fish which tends to be higher in larger ponds [23],[29], specially man-made ponds [12].

Effect of habitat heterogeneity on alpha and beta diversity was consistently evidenced by our analysis, and we attributed this to divergent strategies of habitat choice among species enabling local habitats holds not only more species but also some species with specific requirements, as predicted by the niche theory [45]. We have found a good fit to a linear relation among habitat heterogeneity and local diversity. However, some authors have discussed that area plus habitat heterogeneity could have an unimodal rather than a positive linear effect on local species richness,because increasing heterogeneity increases the potential number of species that could co-exist whileshould reduce the amount of suitable area increasing the likelihood of stochastic extinction [46]. Evidences of other observational studies suggests that patterns evidenced by these authors should rely upon Amphibians assemblages inhabiting ponds, because larger ponds generally have lower species diversity [11], but this expectances was not appropriately tested yet. Two another claims argued in relation to the absence of area effect for amphibians are related to the presence of high density of predator fishes and greater depth in larger ponds [23] . Despite this, we found results that support the species ï heterogeneity relationship, as expected by Niche Theory, as well as for productivity, moreover, this last one is predicted to arguably generate patterns similar to SAR, acting as a neutral processes upon diversity patterns, as predicted by Species-Energy Theory [1],[47],[48].

In turn, anuran beta diversity was shaped by factors related to habitat heterogeneity and complexity at local and landscape scale only. Despite their validity to explain local diversity, more individual hypothesis was not valid to explain the variation on anuran species composition, once the abundance was not related with beta diversity. Moreover, like species richness, local heterogeneity also influenced anuran beta diversity pattern in our study. We know that amphibians choice for vocalization sites are influenced by terrestrial habitats quality around the breeding sites as well as by characteristics of water bodies [49]. Terrestrials habitats, that surround the ponds, are used for amphibians hibernation and feeding, being important to population persistence [49]. Therefore, local and landscape characterization provide clues environmental quality and about factors that are directly influencing on anurans community.

In addition, we do not support the effect of ponds isolation on anuran alpha and beta diversity, despite isolation between ponds has been registered by other works as an important determinant to pond use and population viability by amphibians (e.g. [16],[50]). Most part of our sampled ponds were connected by permeable or semi permeable environment, like vegetation corridors, gallery forest or temporary ponds, which could permit species dispersion, but these factors were not analyzed here and needs to be evaluated with care further. We do not discard isolation that effects can exert strong effect in more wide spatial scales, once that we analyze a relatively small region of the Cerrado biome covering about 56,111 km^{2}on the Southeastern of state of Goiás, which could not be the sufficient to enable us cover biogeographical barriers limiting dispersal for amphibians.Despite landscape variables did not influence local species richness, these are the most important factor responsible by variation on species composition between anuran assemblages, contributing for regional species diversity [51].

There are at least four metacommunity models which claims for gradients that account for the balance among three main mechanism in modern community ecology: the species-sorting dynamics, mass effect dynamics, patch dynamics and neutral model [14],[52]. These mechanisms are environmental filtering, dispersive limitation and interspecific interactions among species that co-occur in the metacommunity [53]. Our results provide information about the processes that are generating local diversity as well as dissimilarity within studied amphibian metacommunity, and suggest that factors related to spatial heterogeneity among ponds and to local niche partitioning are most important then stochastic processes, like dispersive limitation and area effect, contrary to the expectance of IBT. In this way, the most probable mechanism driving anuran regional diversity is species-sorting dynamics. We believe that mass effects are not related with anurans assemblage, due to physiological and behaviors limitations that restrict your dispersive potential [17],[54]. Moreover, despite the most part of anuran species recorded in this work being generalist, we found low number of species that occurred in more than 50% of sampling ponds, suggesting that the species in this work presented low potential dispersive. For instance, the four species that were widely distributed were also the most abundant locally (*Dendropsophus minutus*, *Hypsiboas albopunctatus*, *Scinax fuscomarginatus*, *Physalaemus cuvieri*). They have great dispersal ability and are probably resistant to habitat fragmentation [55].

Local anuran diversity in turn, is structured by a mixture of local factors related to heterogeneity and productivity, which relates to both deterministic and stochastic processes, as emphasized by some authors [56]. Our study added evidences that match to the assumption of both niche theory, by the effect of habitat heterogeneity control [25],[26],[57],[58], and to the hypothesis of more individuals, a proxy to productivity [1],[6], as importantfactors acting upon anuran metacommunities. We reinforce the idea that a good research aim is to pursuit for patterns that couple with niche theory and neutral theory like species-area-energy theories in order to advances metacommunity models [45],[46],[48].

Acknowledgments

We are tankful to Animal Behavior ad Herpetology Lab, from the Federal University of Goiás, Brazil, by the logistic support. LS acknowledge fellowship by Coordenação de Aperfeiçoamento de Pessoal de Nível (CAPES). Financial support was provided by the Ministério da Ciência, Tecnologia e Inovação/Conselho Nacional de Desenvolvimento Científico e Tecnológico (CNPq 563075/2010-4) and Fundação de Amparo à Pesquisa do Estado de São Paulo (FAPESP 2010/52321-7).

References

1. Wright D (1983) Species-energy theory: an extension of species-area theory. Oikos 41: 496ï 506.

2. Maurer BA, McGill BJ (2004) Neutral and non-neutral macroecology. Basic Appl Ecol 5: 413ï 422. Available: http://linkinghub.elsevier.com/retrieve/pii/S1439179104000581. Accessed 7 November 2013.

3. MacArthur RH, Wilson EO (1967) The theory of island biogeography. MacArthur RH, editor New Jersey: Princeton. Available: http://books.google.com/books?hl=en&lr=&id=a10cdkywhVgC&oi=fnd&pg=PR7&dq=The+theory+of+Island+Biogeography&ots=RfcXBzPdJK&sig=zCdknM0uWws7lowKnpvghqGkIxI. Accessed 19 February 2014.

4. Arrhenius O (1921) Species and area. J Ecol 9: 95ï 99. Available: http://www.jstor.org/stable/10.2307/2255763. Accessed 21 November 2013.

5. Brown JH, Kodric-Brown A (1977) Turnover rates in insular biogeography: effect of immigration on extinction. Ecology 58: 445ï 449. Available: http://www.esajournals.org/doi/abs/10.2307/1935620. Accessed 27 March 2014.

6. Srivastava DS, Lawton JH (1998) Why more productive sites have more species: an experimental test of theory using tree-hole communities. Am Nat 152: 510ï 529. Available: http://www.jstor.org/stable/10.1086/286187. Accessed 26 February 2014.

7. Kohn DD, Walsh DM (1994) Plant Species Richness-The Effect of Island Size and Habitat Diversity. J Ecol 82: 367. Available: http://www.jstor.org/stable/10.2307/2261304. Accessed 27 March 2014.

8. Kolasa J, Manne LL, Pandit SN (2012) Speciesï area relationships arise from interaction of habitat heterogeneity and species pool. Hydrobiologia 685: 135ï 144. Available: http://link.springer.com/10.1007/s10750-011-0846-6. Accessed 24 March 2014.

9. Haddad CFB, Prado CPA (2005) Reproductive Modes in Frogs and Their Unexpected Diversity in the Atlantic Forest of Brazil. Bioscience 55: 207ï 217.

10. Mouquet N, E. Miller T, Daufresne T, M. Kneitel J (2006) Consequences of varying regional heterogeneity in source-sink metacommunities. Oikos 113: 481ï 488. Available: http://onlinelibrary.wiley.com/doi/10.1111/j.2006.0030-1299.14582.x/full. Accessed 27 March 2014.

11. Oertli B, Joye DA, Castella E, Juge R, Cambin D, et al. (2002) Does size matter? The relationship between pond area and biodiversity. Biol Conserv 104: 59ï 70. Available: http://linkinghub.elsevier.com/retrieve/pii/S0006320701001549.

12. De Marco P, Nogueira DS, Correa CC, Vieira TB, Silva KD, et al. (2014) Patterns in the organization of Cerrado pond biodiversity in Brazilian pasture landscapes. Hydrobiologia 723: 87ï 101. Available: http://link.springer.com/10.1007/s10750-013-1695-2. Accessed 26 February 2014.

13. Chase JM, Kraft NJB, Smith KG, Vellend M, Inouye BD (2011) Using null models to disentangle variation in community dissimilarity from variation in Ŭ-diversity. Ecosphere 2: art24. Available: http://www.esajournals.org/doi/abs/10.1890/ES10-00117.1. Accessed 22 May 2013.

14. Leibold MA, Holyoak M, Mouquet N, Amarasekare P, Chase JM, et al. (2004) The metacommunity concept: a framework for multi-scale community ecology. Ecol Lett 7: 601ï 613. Available: http://doi.wiley.com/10.1111/j.1461-0248.2004.00608.x. Accessed 21 May 2013.

15. Leibold M, Geddes P (2005) El concepto de nicho en las metacomunidades. Ecol austral: 117ï 129. Available: http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:El+concepto+de+nicho+en+las+metacomunidades#0. Accessed 19 March 2014.

16. Marsh DM, Trenham PC (2001) Metapopulation Dynamics and Amphibian Conservation. Conserv Biol 15: 40ï 49. Available: http://doi.wiley.com/10.1046/j.1523-1739.2001.00129.x.

17. Lemckert F (2004) Variations in anuran movements and habitat use: Implications for conservation. Appl Herpetol 1: 165ï 181. Available: http://booksandjournals.brillonline.com/content/10.1163/157075403323012179.

18. Mazerolle MJ, Desrochers A (2005) Landscape resistance to frog movements. 464: 455ï 464. doi:10.1139/Z05-032.

19. Titon B, Navas CA, Jim J, Gomes FR (2010) Water balance and locomotor performance in three species of neotropical toads that differ in geographical distribution. Comp Biochem Physiol A Mol Integr Physiol 156: 129ï 135. Available: http://www.ncbi.nlm.nih.gov/pubmed/20096361. Accessed 29 May 2013.

20. Shoemaker VH, Balding D, Ruibal R, McClanahan LL (1972) Uricotelism and low evaporative water loss in a South American frog. Science (80-) 175: 1018ï 1020. Available: http://www.ncbi.nlm.nih.gov/pubmed/5009394.

21. Ricklefs RE, Lovette IJ (1999) The roles of island area per se and habitat diversity in the species-area relationships of four Lesser Antillean faunal groups. J Anim Ecol 68: 1142ï 1160. Available: http://doi.wiley.com/10.1046/j.1365-2656.1999.00358.x.

22. Francesco Ficetola G, De Bernardi F (2004) Amphibians in a human-dominated landscape: the community structure is related to habitat features and isolation. Biol Conserv 119: 219ï 230. Available: http://linkinghub.elsevier.com/retrieve/pii/S0006320703004464. Accessed 24 May 2013.

23. Scheffer M, van Geest GJ, Zimmer K, Jeppesen E, Søndergaard M, et al. (2006) Small habitat size and isolation can promote species richness: second-order effects on biodiversity in shallow lakes and ponds. Oikos 112: 227ï 231. Available: http://onlinelibrary.wiley.com/doi/10.1111/j.0030-1299.2006.14145.x/full. Accessed 27 March 2014.

24. Snodgrass JW, Komoroski MJ, Bryan Jr AL, Burger J (2000) Relationships among Isolated Wetland Size , Hydroperiod , and Amphibian Species Richness;: Implications for Wetland Regulations. 14: 414ï 419.

25. Vasconcelos TS, Santos TG, Rossa-Feres DC, Haddad CFB (2009) Influence of the environmental heterogeneity of breeding ponds on anuran assemblages from southeastern Brazil. Can J Zool 87: 699ï 707. Available: http://www.nrcresearchpress.com/doi/abs/10.1139/Z09-058. Accessed 22 May 2013.

26. Silva FR, Candeira CP, Cerqueira Rossa-Feres D (2012) Dependence of anuran diversity on environmental descriptors in farmland ponds. Biodivers Conserv 21: 1411ï 1424. Available: http://link.springer.com/article/10.1007/s10531-012-0252-z. Accessed 22 May 2013.

27. Hartel T, Moga CI, David a., Coroiu I (2009) Species richness ð pond area relationships of amphibians and birds in two Natura 2000 protected areas of Romania. Community Ecol 10: 159ï 164. Available: http://www.akademiai.com/openurl.asp?genre=article&id=doi:10.1556/ComEc.10.2009.2.4. Accessed 12 June 2013.

28. Afonso LG, Eterovick PC (2007) Microhabitat choice and differential use by anurans in forest streams in southeastern Brazil. J Nat Hist 41: 937ï 948. Available: http://www.tandfonline.com/doi/abs/10.1080/00222930701309544. Accessed 22 May 2013.

29. Werner EE, Yurewicz KL, Skelly DK, Relyea R a. (2007) Turnover in an amphibian metacommunity: the role of local and regional factors. Oikos 116: 1713ï 1725. Available: http://doi.wiley.com/10.1111/j.2007.0030-1299.16039.x. Accessed 24 May 2013.

30. Heyer WR, Donnelly MA, McDiarmid RW, Hayek LC, Foster MS (1994) Measuring and monitoring biological diversity: standard methods for amphibians. Washington: Smithsonian Institution Press. Available: http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:Standard+Techniques+for+Inventory+and+Monitoring#1. Accessed 5 May 2014.

31. Nessimian JL, Venticinque EM, Zuanon J, Marco P, Gordo M, et al. (2008) Land use, habitat integrity, and aquatic insect assemblages in Central Amazonian streams. Hydrobiologia 614: 117ï 131. Available: http://link.springer.com/10.1007/s10750-008-9441-x. Accessed 8 November 2013.

32. Guttman L (1954) Some necessary conditions for common factor analysis. Psychometrika 19: 149ï 161.

33. Legendre P, Legendre L (2012) Numerical Ecology. Third edit. Amsterdam, The Netherlands: Elsevier.

34. Pettorelli N, Ryan S, Mueller T, Bunnefeld N, Jedrzejewska B, et al. (2011) The Normalized Difference Vegetation Index (NDVI): unforeseen successes in animal ecology. Clim Res 46: 15ï 27. Available: http://www.int-res.com/abstracts/cr/v46/n1/p15-27/. Accessed 22 May 2013.

35. Rueda LM, Brown TL, Kim HC, Chong S-T, Klein T a, et al. (2010) Species composition, larval habitats, seasonal occurrence and distribution of potential malaria vectors and associated species of Anopheles (Diptera: Culicidae) from the Republic of Korea. Malar J 9: 55. Available: http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=2834702&tool=pmcentrez&rendertype=abstract.

36. Koh C-N, Lee P-F, Lin R-S (2006) Bird species richness patterns of northern Taiwan: primary productivity, human population density, and habitat heterogeneity. Divers <html_ent glyph=ñ@amp;òascii=ñ&ò/> Distrib 12: 546ï 554. Available: http://doi.wiley.com/10.1111/j.1366-9516.2006.00238.x. Accessed 3 July 2013.

37. Garel M, Gaillard J-M, Jullien J-M, Dubray D, Maillard D, et al. (2011) Population abundance and early spring conditions determine variation in body mass of juvenile chamois. J Mammal 92: 1112ï 1117. Available: http://www.bioone.org/doi/abs/10.1644/10-MAMM-A-056.1. Accessed 3 July 2013.

38. Gotelli NJ, Colwell RK (2001) Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. Ecol Lett 4: 379ï 391. Available: http://onlinelibrary.wiley.com/doi/10.1046/j.1461-0248.2001.00230.x/full. Accessed 8 November 2013.

39. Zar JH (2010) Biostatistical Analysis. Fifth Edit. New Jersey: Prentice Hall.

40. Anderson MJ, Crist TO, Chase JM, Vellend M, Inouye BD, et al. (2011) Navigating the multiple meanings of b diversity: a roadmap for the practicing ecologist. Ecol Lett 14: 19ï 28. Available: http://www.ncbi.nlm.nih.gov/pubmed/21070562. Accessed 22 May 2013.

41. Zapala M a, Schork NJ (2006) Multivariate regression analysis of distance matrices for testing associations between gene expression patterns and related variables. Proc Natl Acad Sci U S A 103: 19430ï 19435. Available: http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=1748243&tool=pmcentrez&rendertype=abstract.

42. Oksanen J, Blanchet FG, Kindt R, Legendre P, B, OõHara R, et al. (2012) vegan: Community Ecology Package (R package version 1.17-6).

43. Honkanen M, Roberge J-M, Rajasärkkä A, Mönkkönen M (2010) Disentangling the effects of area, energy and habitat heterogeneity on boreal forest bird species richness in protected areas. Glob Ecol Biogeogr 19: 61ï 71. Available: http://doi.wiley.com/10.1111/j.1466-8238.2009.00491.x. Accessed 30 May 2013.

44. Jonsson M, Englund G, Wardle D a. (2011) Direct and indirect effects of area, energy and habitat heterogeneity on breeding bird communities. J Biogeogr 38: 1186ï 1196. Available: http://doi.wiley.com/10.1111/j.1365-2699.2010.02470.x. Accessed 22 May 2013.

45. Allouche O, Kalyuzhny M, Moreno-Rueda G, Pizarro M, Kadmon R (2012) Area-heterogeneity tradeoff and the diversity of ecological communities. Proc Natl Acad Sci U S A 109: 17495ï 17500. Available: http://www.pnas.org/content/109/43/17495.short. Accessed 23 May 2013.

46. Kadmon R, Allouche O (2007) Integrating the effects of area, isolation, and habitat heterogeneity on species diversity: a unification of island biogeography and niche theory. Am Nat 170: 443ï 454. Available: http://www.ncbi.nlm.nih.gov/pubmed/17879194. Accessed 24 May 2013.

47. Hurlbert AH (2004) Species-energy relationships and habitat complexity in bird communities. Ecol Lett 7: 714ï 720. doi:10.1111/j.1461-0248.2004.00630.x.

48. Storch D, Evans KL, Gaston KJ (2005) The species-area-energy relationship. Ecol Lett 8: 487ï 492. Available: http://www.ncbi.nlm.nih.gov/pubmed/21352452. Accessed 3 July 2013.

49. Semlitsch RD, Bodie JR (2003) Biological Criteria for Buffer Zones around Wetlands and Riparian Habitats for Amphibians and Reptiles. Conserv Biol 17: 1219ï 1228. Available: http://doi.wiley.com/10.1046/j.1523-1739.2003.02177.x.

50. Blaustein AR, Wake DB, Sousa WP (1994) Amphibian Declines: Judging Stability, Persistence, and Susceptibility of Populations to Local and Global Extinctions. Conserv Biol 8: 60ï 71. Available: http://doi.wiley.com/10.1046/j.1523-1739.1994.08010060.x.

51. Gómez-Rodríguez C, Díaz-Paniagua C, Bustamante J, Serrano L, Portheault A (2010) Relative importance of dynamic and static environmental variables as predictors of amphibian diversity patterns. Acta Oecologica 36: 650ï 658. Available: http://linkinghub.elsevier.com/retrieve/pii/S1146609X10001128. Accessed 12 June 2013.

52. Winegardner AK, Jones BK, Ng ISY, Siqueira T, Cottenie K (2012) The terminology of metacommunity ecology. Trends Ecol Evol 27: 253ï 254. Available: http://www.ncbi.nlm.nih.gov/pubmed/22325446. Accessed 22 May 2013.

53. Holyoak M, Leibold MA, Holt RD (2005) Metacommunities: Spatial Dynamics and Ecological Communities. Chicago and London: University of Chicago Press.

54. Becker CG, Fonseca CR, Haddad CFB, Prado PI (2010) Habitat split as a cause of local population declines of amphibians with aquatic larvae. Conserv Biol 24: 287ï 294. Available: http://www.ncbi.nlm.nih.gov/pubmed/19758391. Accessed 12 June 2013.

55. IUCN (2013) The IUCN Red List of Threatened Species. Version 2013.2. Available?http://www.iucnredlist.org. Acessed 15 May 2014.

56. Myers JA, Chase JM, Jiménez I, Jørgensen PM, Araujo-Murakami A, et al. (2013) Beta-diversity in temperate and tropical forests reflects dissimilar mechanisms of community assembly. Ecol Lett 16: 151ï 157. Available: http://www.ncbi.nlm.nih.gov/pubmed/23113954. Accessed 21 January 2014.

57. Pedruski MT, Arnott SE (2011) The effects of habitat connectivity and regional heterogeneity on artificial pond metacommunities. Oecologia 166: 221ï 228. Available: http://www.ncbi.nlm.nih.gov/pubmed/20976605. Accessed 8 June 2013.

58. Richter-Boix A (2005) Pond breeding amphibian metacommunity structure: the importance of trade-offs. importance tradeoffs metacommunity Struct: 135ï 160.

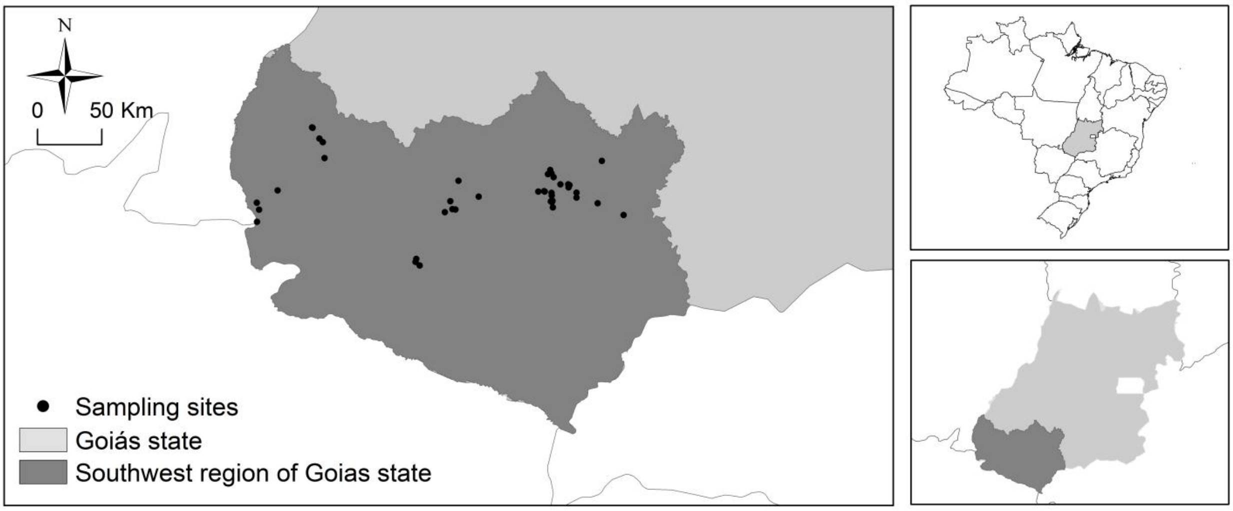


Figure 1. Geographical localization of the 39 sampling sites, located in the southeastern region of state of Goiás, Brazil.

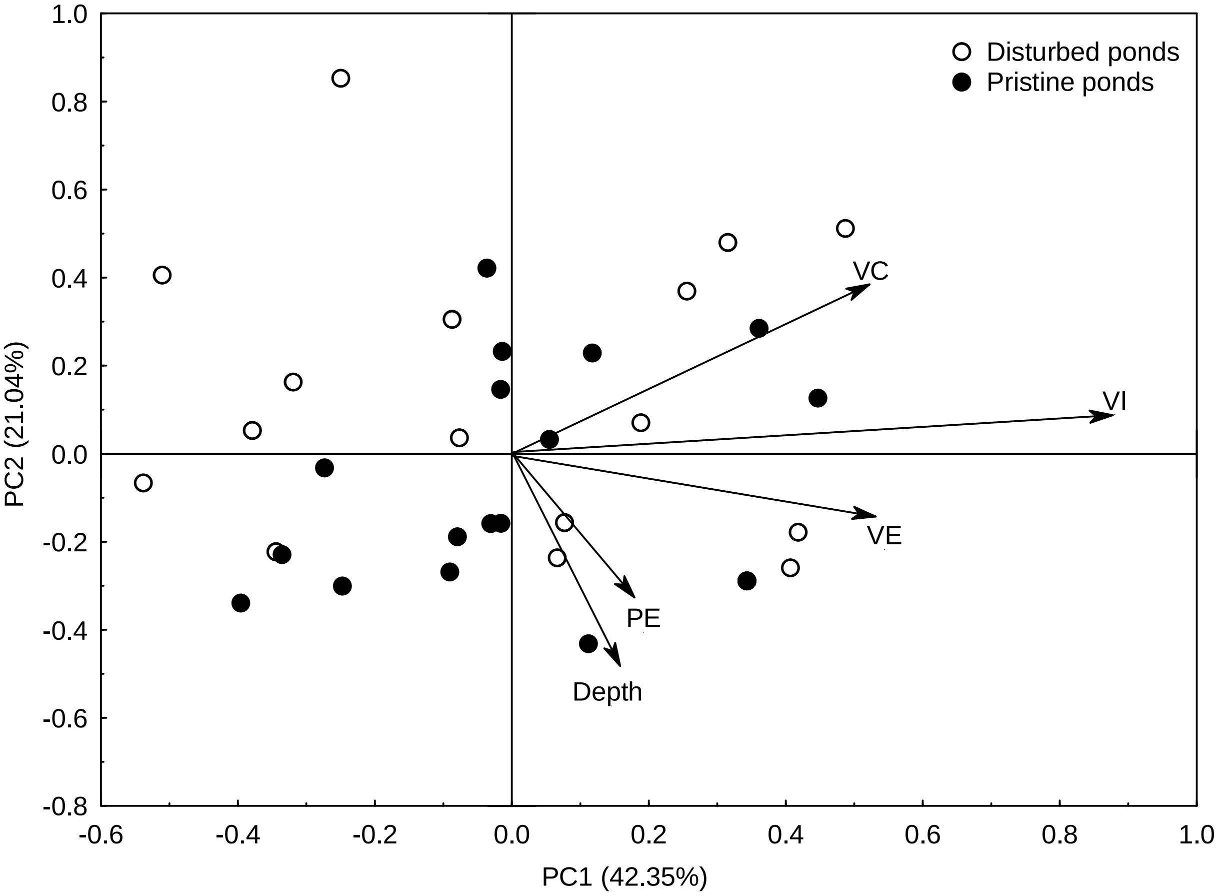


Figure 2. Sites ordination (PCA) according to environmental variables used as proxy to local heterogeneity.

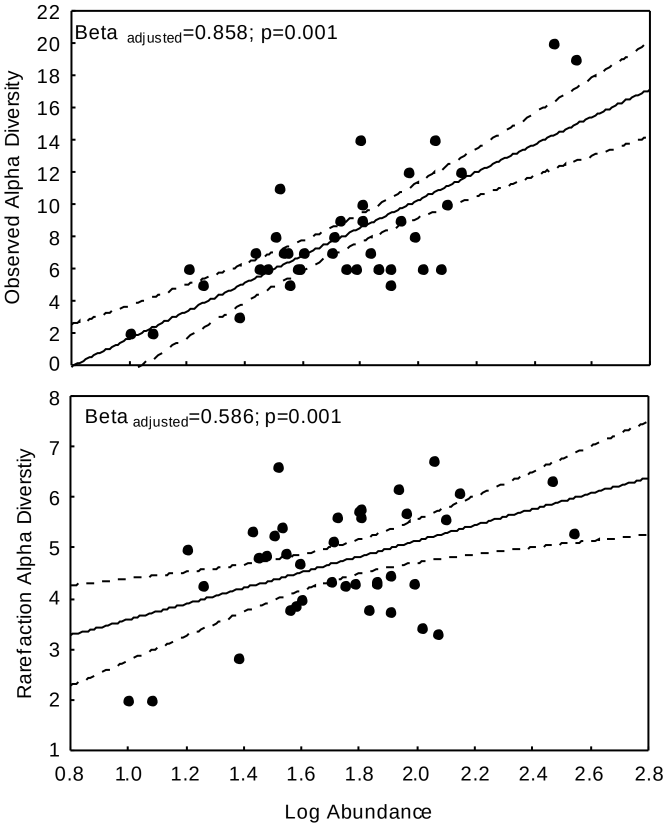


Figure 3. Scatterplots between log abundance and species richness (A) and rarefied species richness (B), obtained by multiple regression analysis. Dashed line is confidence interval and dotted line is prediction intervals.

Table1. Correlations between local environmental variables and the principal axes of PCA. CP1, first principal component; CP2, second principal component; * correlations coefficients whose magnitude are greater than 0.5.

Environmetal variables	**CP1**	**CP2**
Depth	0.188	-0.742*
Vegetation in the interior of the pond	0.924*	0.093
Vegetation cover on the pond surface	0.689*	-0.275
Vegetation in the edge	0.251	-0.529*
Profile of the pond edge	0.685*	0.501*
Eigenvalue	0.122	0.060
Proportion explained	42.35%	21.04%

Table 2. Multiple regression between observed species richness and following predictor variable: perimeter, isolation, number of individuals, local heterogeneity, landscape heterogeneity and landscape complexity ($R^2_{adjusted}$=0.662; $F_{(7,31)}$=9.059; pÒ0.001). We have: SE ï standard error of Beta. *p<0.05

Predictors	**ɓ $_{adjusted}$**	**SE ofɓ**	**t(33)**	***P***	**Tolerance**
Perimeter	0.026	0.133	0.197	0.845	0.602
Isolation	-0.019	0.129	-0.150	0.882	0.639
Number of individuals $_{log}$	**0.858**	**0.118**	**7.253**	**0.000***	**0.757**
Local heterogeneity 1	-0.197	0.126	-1.569	0.127	0.671
Local heterogeneity 2	0.251	0.132	1.903	0.066	0.609
Landscape Heterogeneity	-0.135	0.129	-1.044	0.304	0.638
Landscape Complexity	-0.158	0.146	-1.077	0.290	0.495

Table 3. Multiple regression between rarefied species richnessand following predictor variable: perimeter, isolation, number of individuals, local heterogeneity, landscape heterogeneity and landscape complexity ($R^2_{adjusted}=0.662$; $F_{(7,31)}=9.059$; $p\leq 0.001$). We have: SE – standard error of Beta. *p<0.05

Predictors	**β adjusted**	**SE of β**	**t(33)**	***P***	**Tolerance**
Perimeter	-0.138	0.176	-0.786	*0.438*	0.602
Isolation	-0.064	0.171	-0.377	*0.709*	0.639
Number of individuals log	**0.586**	**0.157**	**3.735**	***0.001***	**0.757**
Local heterogeneity 1	-0.068	0.167	-0.410	*0.685*	0.671
Local heterogeneity 2	**0.403**	**0.175**	**2.303**	***0.028***	**0.609**
Landscape Heterogeneity	0.001	0.171	0.004	*0.997*	0.638
Landscape Complexity	-0.069	0.194	-0.356	*0.725*	0.495

Table 4. Results of multivariate regression between ponds isolation, abundance, local environmental heterogeneity and landscape environmental heterogeneity and complexity on the anuran beta diversity (raup-crick dissimilarity). * $p<0.05$

		F	**R^2**	**P**
Isolation				
	Isolation	0.237	0.006	0.907
Abundance				
	Abundance	-0.032	-0.001	0.972
Local environmental **heterogeneity**				
	Local heterogeneity 1	1.891	0.045	0.102
	Local heterogeneity 2	**2.342**	**0.055**	**0.040***
	Local heterogeneity 1 x 2	**2.964**	**0.070**	**0.009***
Landscape				
	Landscape complexity	**3.579**	**0.084**	**0.003***
	Landscape heterogeneity	0.863	0.020	0.540
	Landscape complexity x heterogeneity	**3.119**	**0.073**	**0.005***

Supporting Information

File S1.

Table S1, Anuran species recorded in the southwest region from the state of Goiás, with their respective occurrence frequency.

Family	**Species**	**Frequency (%)**
Aromobatidae		
	Eupemphix nattereri	10.00
Bufonidae		
	Rhinella mirandaribeiroi	7.50
	Rhinella schneideri	22.50
Cycloramphidae		
	Odontophrynus salvatori	2.50
	Odontophrynus sp.	2.50
Hylidae		
	Dendropsophus cruzi	45.00
	Dendropsophus jimi	17.50
	Dendropsophus minutus	52.50
	Dendropsophus nanus	25.00
	Dendropsophus rubicundulus	40.00
	Dendropsophus soaresi	2.50
	Hypsiboas albopunctatus	52.50
	Hypsiboas lundii	10.00
	Hypsiboas raniceps	15.00
	Hypsiboas paranaíba	20.00
	Phyllomedusa azurea	22.50
	Pseudis bolbodactyla	15.00
	Scinax constrictus	20.00
	Scinax fuscomarginatus	75.00
	Scinax gr. *ruber*	32.50

	Scinax pusillus	7.50
	Scinax similis	5.00
Leiuperidae		
	Physalaemus centralis	20.00
	Physalaemus cuvieri	57.50
	Physalaemus fuscomaculatus	2.50
	Pseudopaludicola falcipes	32.50
	Pseudopaludicola saltica	15.00
	Leptodactylus hylaedactylus	52.50
	Leptodactylus furnarius	10.00
	Leptodactylus fuscus	15.00
	Leptodactylus labyrinthicus	20.00
	Leptodactylus latrans	22.50
	Leptodactylus mystacinus	7.50
	Leptodactylus podicipinus	27.50
	Leptodactylus sp.	32.50
Microhylidae		
	Elachistocleis sp.	27.50

CAPÍTULO III - Anuran Conservation in Brazilian Cerrado: single large or several small ponds, which is better?

Luciana Signorelli[a*], Denis Silva Nogueira[a], Rogério Pereira Bastos[b], Paulo De Marco Junior[c]

[a] *Programa de Pós Graduação em Ecologia e Evolução, Departamento de Ecologia, Universidade Federal de Goiás, 74.001-970 Goiânia, Goiás, Brasil.*

[b] *Laboratório de Herpetologia e Comportamento Animal, Departamento de Ecologia, Universidade Federal de Goiás, 74.001-970 Goiânia, Goiás, Brasil.*

[c] *Laboratório de Ecologia Teórica e Síntese, Departamento de Ecologia, Universidade Federal de Goiás, 74.001-970 Goiânia, Goiás, Brasil.*

Corrersponding author: lfl.signorelli@gmail.com

Formatted according to *Biological Conservation*

ABSTRACT

The debate over several small or single large (SLOSS) seems to be relevant to the biodiversity conservation planning in general. It can be especially useful for conservation of amphibians associated to ponds. Considering this, the aim of our study was find good ways to conserve anuran diversity even increasing human pressures at a regional perspective. Thus we intend to test if small ponds hosts higher anuran species than single large pond, and if accumulating ponds with higher habitat heterogeneity retain more species than accumulating most homogeneous ponds. To test these hypotheses, we compared sample-based species accumulation curves both ranking ponds from large-to-small and from small-to-large or low-to-high and high-to-low heterogeneity, each time calculating the cumulative number of species. We found that an equal area partitioned in various small ponds retain a larger part of anuran diversity than the same area in a single large ponds. Considering habitat heterogeneity, we did not find a clear relationship in respect of which areas aggregate greater number of species, if are several homogeneous ponds or single heterogeneous ponds, rejecting our hypothesis about habitat heterogeneity. We can conclude that the ultimate factor determining higher species richness is the relative important of between-pond spatial relationships, and that preserve several small ponds, even in an agricultural landscape, is a good alternative to management of amphibian's diversity.

Keywords: SLOSS, cumulative curves, amphibian, ponds, regional diversity, conservation.

1. Introduction

One of the most popular and controversial conservation debate is the ñSingle large or several smallò, denoted by its acronym SLOSS. This debate started in the mid-1970s when Diamond (1975) proposed six rules of reserve design, including the idea that single large land reserve would be better in order to maintain the regional species diversity than several small lands reserves. If small habitat patches contain species that are a subset of species present in a large habitat patch, then conserve single large habitat patch will be better than conserve several small habitat patches (Simberloff and Abele, 1976). Otherwise, if the main objective is to maximize the number of species currently occurring in a given landscape, and maximize time to extinction, thus several smalls are preferable as conservational proposal (Ovaskainen, 2002). Advances in the theoretical approaches in landscape ecology and in metapopulation and metacommunity ecology have shown potential to contribute for this debate by adding elemental components of neutral and niche theories like dispersal limitation and habitat heterogeneity (Allouche et al., 2012; Economo, 2011; Mouquet and Loreau, 2003).

Although SLOSS has been highly criticized because of the oversimplification of species diversity dynamics (Wu, 2008), the concept still seems to be relevant to the biodiversity conservation planning in general. This is especially important when the purpose is select sites that host the highest species richness in a fragmented landscape (Fattorini, 2010). Habitat loss, fragmentation and loss of connectivity are being attributed as major causes to species decline in the world, and these factors are an important determinant of regional species richness (Foley et al., 2005; Krauss et al., 2010). Thus, it is essential to search how to combine species conservation and human economic activities. In this way, some questions were very well established by Wu (2008), as: ñhow can biodiversity be conserved with ever increasing human pressure on the natural environment?ò or ñHow should humans and their activities be viewed and treated in planning and managing natural resources for conserving biodiversity?ò

The debate over several small or single large could be useful for conservation of the biodiversity associated to ponds. Ponds make an important contribution to the regional biodiversity than any other aquatic habitat (Biggs et al., 2005; Davies et al., 2008a; Williams, 2003), even in intensely farmed landscapes (Sayer et al., 2012). In some landscapes dominated by agriculture, some ponds are disappearing, generating an increase concern regarding the ponds conservation (Oertli et al., 2005; Ruggiero et al., 2008). However, in the Brazilian Cerrado we have an opposite situation considering the number of ponds, which are increasing due to the stream damming for economic purposes (De Marco et al., 2014). Ponds can provide water supply (as agricultural irrigation), hydrological regulation, and fish production to local society (Oertli et al., 2005). Like other parts of the world, the Cerrado ponds hold a high biodiversity at regional scale, with higher contribution of beta diversity component (De Marco et al., 2014; unpublished Signorelli et al., 2014). Nevertheless, the few studies that have accounted for the processes driving Cerrado ponds biodiversity until now suggested a smaller importance of area and larger effect of habitat heterogeneity ï at least for amphibians and water beetles on a local perspective (De Marco et al., 2014; unpublished Signorelli et al., 2014). On a regional perspective, the importance of area or heterogeneity has never been investigated despite the relevance of that for Cerrado ponds biodiversity conservation and landscape management.

Small to large and large to small cumulative curves may be considered a good approach to assess which is better for biodiversity associated to ponds inserted in farmed landscapes: several small or single large (Fattorini, 2010; Quinn and Harrison, 1988). In this approach, the species area curve is ranked in the order to ascend or descend cumulatively according to area (Quinn and Harrison, 1988) or any other relevant environmental predictor (Kolasa et al., 2012). The curve positioned above of another curve accumulates more species in a smaller amount of area, and may be considered a clue to select which areas should be prioritized for species conservation (Fattorini, 2010; Quinn and Harrison, 1988). For instance, a set of small size ponds has more species of aquatic species, such as aquatic plants, amphibian and some macroinvertebrates, than single large ponds of same size (see Martínez-Sanz et al., 2012a; Oertli et al., 2002).

Especially for amphibians that use lentic water bodies to complete their life cycles, several small ponds is probably the best choice, since this is negatively associated with richness of predatory fish (Hamer and Parris, 2013, 2011; Hecnar and M'Closkey, 1997). In addition, some species prefer ponds colonized by emergent vegetation, with some vertical structure and with flat edge profiles, which provides food, and shelter (Hartel et al., 2009; Scherer et al., 2012). The SLOSS strategy used here combine information of area, perimeter and habitat heterogeneity to predict the best choice to conserve amphibians' regional biodiversity in agricultural landscapes of Brazilian Cerrado. These characteristics make amphibians good models to apply the SLOSS strategy proposed here, which could be extended to any other organisms that have similar affinities in relation to habitat selection, making amphibians a candidate to ponds biodiversity surrogate. The aim of our study was suggest alternatives to conserve anuran diversity even increasing human pressures at a regional perspective. Thus, first of all we intend to test the hypothesis that several small ponds host higher anuran species than single large pond. Subsequently, we intend to test the hypothesis regarding habitat heterogeneity, if accumulating ponds with higher habitat heterogeneity retain more species than accumulating most homogeneous ponds, assuming that heterogeneous habitats can support more species than homogenous habitats, as proposed by niche theory.

2. Materials and methods

2.1. Study area and study site selection

The surveys occurred on the core of Cerrado biome, more specifically on the state of Goiás, Brazil. We sampled a total of 127 ponds, which were inserted in landscapes with different degrees of fragmentation (Fig. 1).

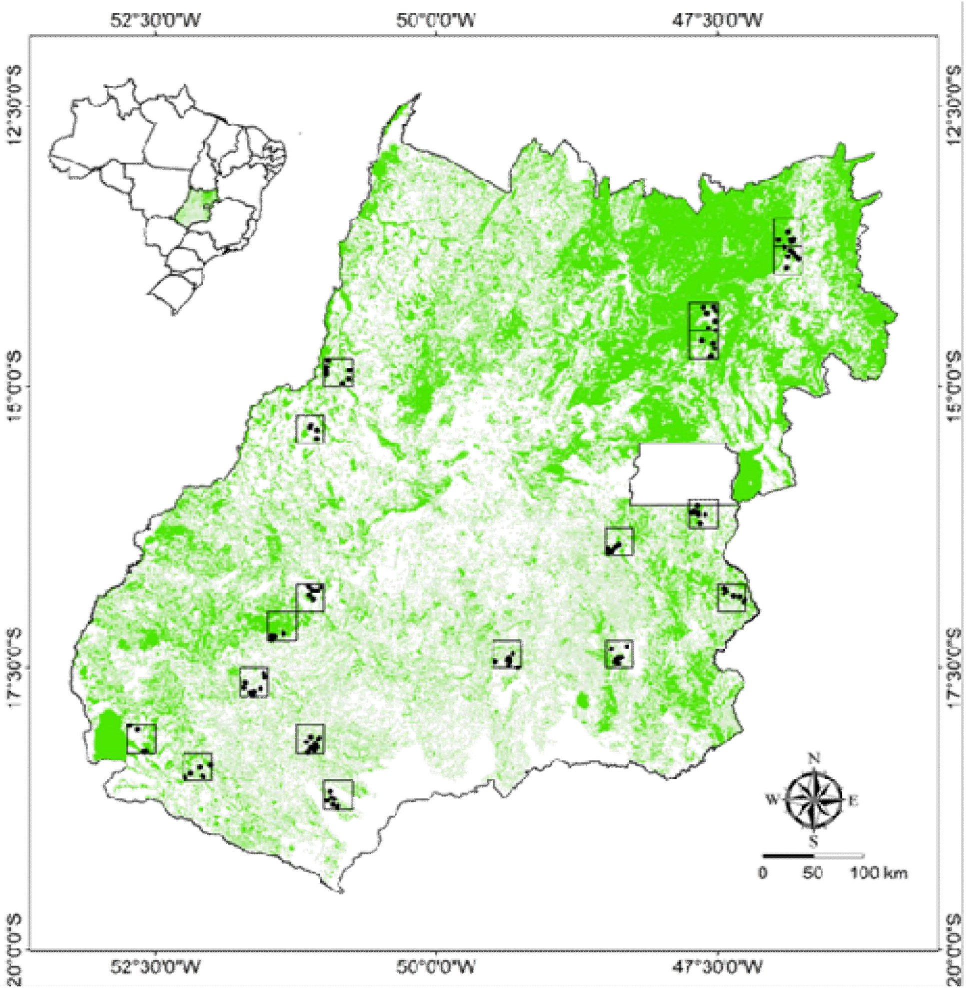


Fig. 1. Study sites and landscapes surveyed in the state of Goiás, core of Brazil.

2.2. Surveys

We surveyed ponds for adult frogs during the rainy season (October-March), from 2010 to 2013. To assess the presence of individual species, observers spent 1 h at each pond between the hours of 1900 and 2400. Adult frogs were surveyed, via a combination of both acoustic and visual means (Rödel and Ernst, 2004) while walking slowly around the pond and systematically searching or listening for calling males (Heyer et al., 1994). A few adults were collected to confirm identification. All specimens collected were euthanized before preservation through an overdose of the anesthetic 2% lidocaine through parenteral injection. The use of injectable euthanasia agents is one of the most rapid and reliable methods of performing euthanasia, minimizing pain and distress to the animal. Those methods are acceptable by international (AVMA - Leary et al., 2013) and Brazilian government institution (National Council on the Control of Animal Experiments - CONCEA, 2013). The animals euthanized were then fixed in formalin 10% and preserved in alcohol 70% (Heyer et al., 1994).They are actually deposited in the Zoological Collection (Coleção Zoológica) of the Universidade Federal de Goiás (ZUFG).

2.3. Habitat classification

We selected some characteristics expected to be important for habitat selection by anuran species, being related to their life history and to represent the environment in the Cerrado ponds. Then, we developed a quantitative heterogeneity index derived from a set of qualitative and semi-quantitative habitat measures of ponds characteristics, which were proportional to the pond area. The index includes seven variables that are expected to influence site suitability for anurans: (1) profile of the pond edge, (2) substrate of the water body, (3) vegetation on the pond surface, (4) vegetation in the edge of the pond, (5) water color, (6) type of land use within 500 m of the pond, and (7) position of the pond considering the remnants of vegetation (Supplementary Table 1).

We weighted each of these variables by theperceived importance of each to anurans, based on our experience and understanding of the natural history of these species.(Supplementary Table A.1).

In order to allow for the weighted importance of the measured characteristics around the pond, we multiple each observed value by the score of the variable which represented our perception of the importance of that predictor to anurans. For each class of semi-quantitative (ordinal) environmental characteristics discussed above, the scores ranged from 1 to maximal number of questions in that class. We give more weight for items that we think were more important to local anuran diversity. The score were extremely based on our previous knowledge about natural history of the Cerrado anurans. Doing so, as we have quantified in the field the proportion of that class of habitat characteristics around the ponds, we obtained a weighted measure of each variable proportional to the pond area.

Then, we summed the alternatives within each weighted class of habitat characteristics. Finally, we divided each value of these predictors by the maximum value of that category in order to obtain a standardize measure between 0 and 1, and then summed over all seven classes of variables to derive the heterogeneity proportionally weighted (area) index (Eq. 1).

$$\times\ (\times\ (qi*score)/max\ (qi)) \qquad \text{Eq. 1}$$

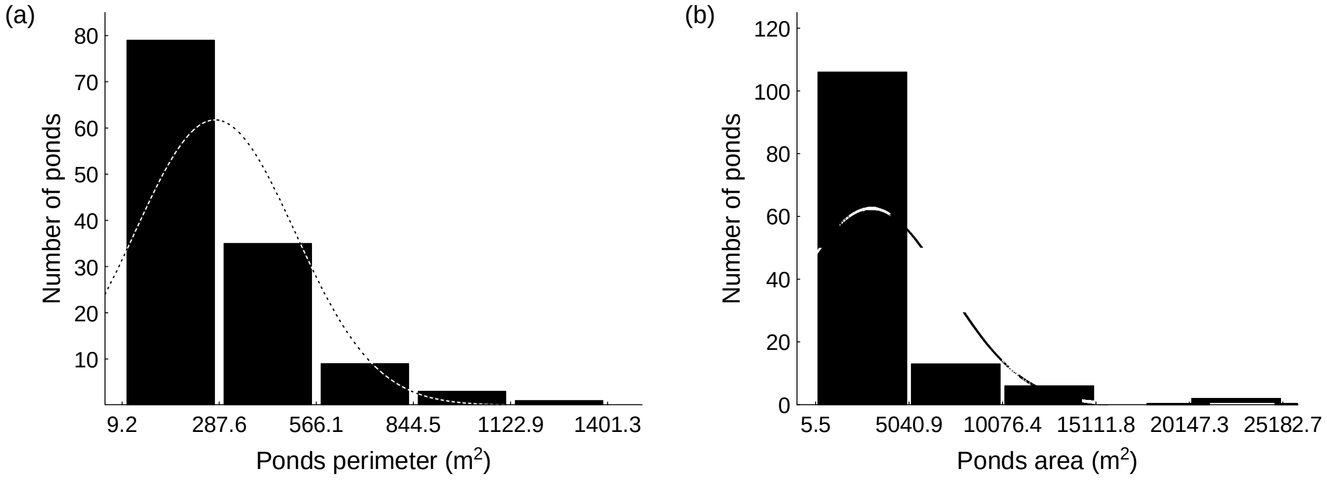
The index presented here enable one to measure more appropriately the importance of environmental characteristics and jointly the importance of the amount of such characteristics in a local site. In addition, it could be weighted by the own area of the pond providing a better approximation of the anuran perception of the pond habitats. The advantage of the present index is their plasticity in relation to the incorporation of other qualitative, semi-quantitative, and quantitative measures of local and landscape environmental features. In addition, one can test for the effects of the class of the variables that compose the index separately by using each one of the specific class of environmental predictors.

2.4. Species accumulation curves

To evaluate if larger ponds had higher species richness than smaller ponds we compared sample-based species accumulation (see Gotelli and Colwell, 2001) both ranking ponds from large to small and from small to large, each time calculating the cumulative number of species (Quinn and Harrison, 1988). The same procedure were done considering perimeter instead area as a better surrogate for available habitat for amphibians as suggested in other studies (unpublished Signorelli et al. 2014). To evaluate the effect of habitat heterogeneity on the rate of species accumulation, we ranked habitat heterogeneity from the most heterogeneity to most homogeneous ponds and from the most homogeneous to most heterogeneous ponds, in order to provide strict linkage with the approach used with area and perimeter for the regional landscape. In this case, one would expect that accumulating areas with higher habitat heterogeneity would enable a higher number of species more quickly than accumulating homogeneous ones, giving steeper z-values for regional SACs (as proposed by Kolasa et al., 2012). Nevertheless, new theoretical insights that non-linear effect of the covariance between heterogeneity and area locally generate unimodal patterns of species richness (Allouche et al., 2012) could be detrimental explanation for the lower initial species richness accumulated by adding highly heterogeneous habitat first (Kolasa et al., 2012), because of limits for the population size imposed by restrictions of area increasing heterogeneity (Allouche and Kadmon, 2009; Allouche et al., 2012; Kadmon and Allouche, 2007). To each of species accumulation curve, we fitted the power function $S=cA^z$, where S is the richness, A is the area (or perimeter and habitat heterogeneity in our case), c is a constant, and z is the logarithmic rate at which number of species increases with sampled area (see Arrhenius, 1921; Connor and McCoy, 2001; Kolasa et al., 2012). More specifically, z gives the slope of the increasing number of species with increasing of sampled area, perimeter or habitat heterogeneity (see Arrhenius, 1921; Kolasa et al., 2012). Theregression analysis was done according to the same settings used by Dengler (2009) with the non-linear model estimation of statistica 7.1 (StatSoft, Inc., 2005): least squares regression model; method of estimation: quasi-Newton; criterion ofconvergence: 0.0001 (Dengler, 2009).

3. Results

For the 127 ponds investigated, we recorded a total of 57 amphibiansôspecies belonging to 16 genera and five families (see Appendix S1). Ponds surveyed have an accumulation perimeter of 34.789 km and area of 37.934 ha. Perimeter had an average size of 273.93 (±227.53 SD) m and ranged from 9.22 to 1401.30 m (Fig.2a). Area ranged had an average size of 2986.96 (±4058.10 SD) m^2 and ranged from 5.50 to 25182.69 m^2 (Fig.2b). Considering habitat heterogeneity, ponds were characterized to have predominantly an intermediary heterogeneity index, in which an average of 0.491 (± 0.097 SD) and rang from 0.312 to 0.822 (Fig. 2c).



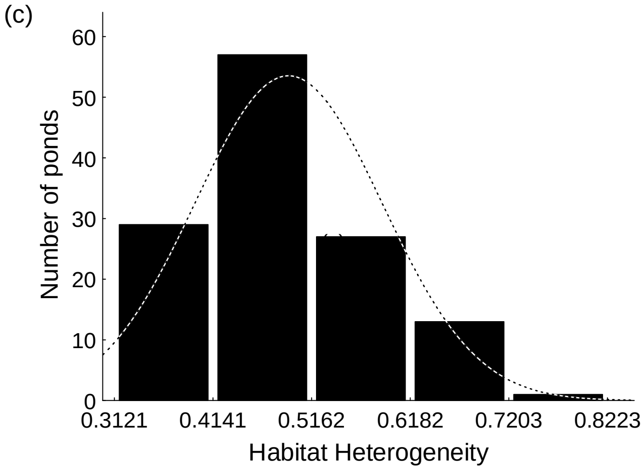
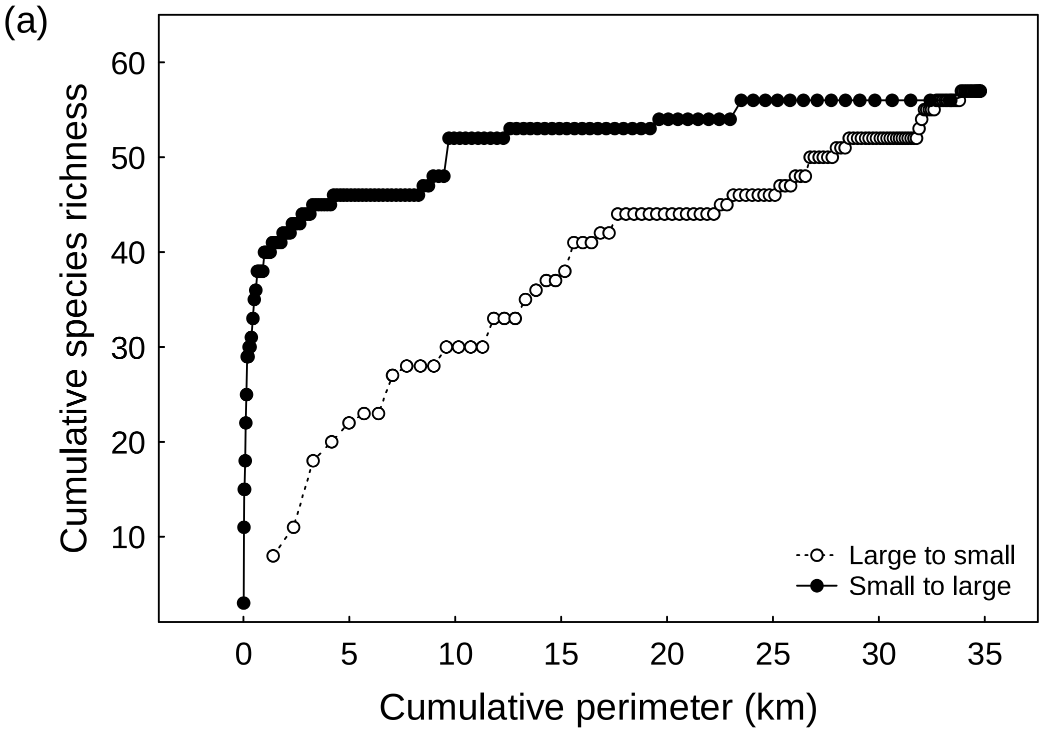


Fig. 2. Frequency distribution for ponds (N=127) according to (a) perimeter (D= 0.147, *P*< 0.010, by Kolmogorov-Smirnoff test), (b) area (D= 0.232, *P*< 0.010, by Kolmogorov-Smirnoff test), and (c) habitat heterogeneity (D=0.100, p < 0.200, by Kolmogorov-Smirnoff test).

A consistent pattern of species accumulation curve was found considering perimeter and area of sampled ponds: conserve several smaller areas/perimeter protect the larger amount of the total species richness in this landscape. This means that cumulative large-to-small curves lie consistently below to the small-to-large curves (see Fig.3a-b). The largest pond has an area and perimeter of about 2.518 ha and 0.582 km respectively and includes 14 species. With a similar area (~2.500 ha) by accumulating several small ponds we can achieve 39 species, and there are necessity of only three small ponds with a total area of about 0.002 ha to achieve ~15 species as the largest pond. Several small accumulation area of 12.889 ha may either support 56 species, (more specifically, 99 ponds), about 98.24% of all anuran species registered while when the same area is composed by eight large ponds only 28 species are reached, about 49.12% of all anuran species registered. In similar way, accumulation area of 9.702 ha may either support 52 species, about 91.23% of all anuran species registered, when this area is composed by 75 small ponds or 30 species, about 52.63% of all anuran species registered, when this area is composed by 12 large ponds.

Cumulative species number considering ponds ranked from both high to low and low to high habitat heterogeneity are quite similar (Fig. 3c). However, when we ranked habitat heterogeneity from low to high, the number of species is higher than when we ranked ponds from high to low, up intermediary levels of heterogeneity. But after this threshold (habitat heterogeneity accumulated of 20) the total number of species become higher in the curves high to low, and later it becomes similar.

Regional accumulative species curves showed good non-linear fit to our predictors perimeter, area and habitat heterogeneity. In the relationship between accumulated pond perimeter or area and accumulated species richness, small-to-large always presented higher total variance explained than large-to-small. The fitted z-values for pond perimeter and area considering the curve small-to-large were 0.139 and 0.103, respectively (Table 1). The fitted z-values for pond perimeter and area considering the curve large-to-small were 0.499 and 0.705, respectively (Table 1). Cumulative species curves based on habitat heterogeneity had higher fitted z-values for curves high-to-low than curves low-to-high (Table 1).



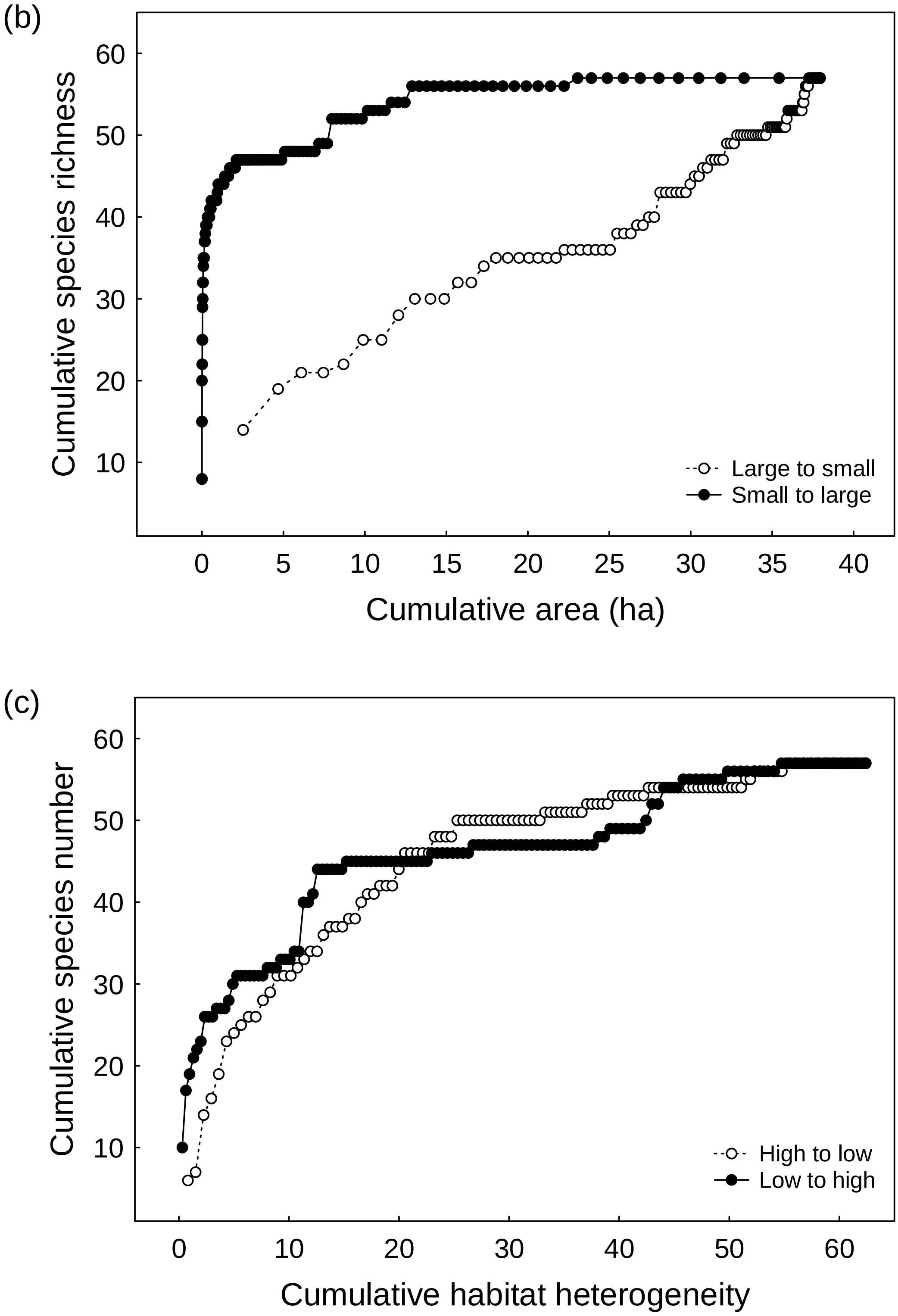


Fig. 3. Species accumulation curve and species area relationship of anurans in Cerrado ponds over perimeter, area, and habitat heterogeneity either beginning with the largest pond and adding patches in order of decreasing size, and beginning with the smallest patch and adding patches in order of increasing size. Figures show (a) cumulative perimeter and

cumulative species richness, (b) cumulative area and cumulative species richness, and (c) cumulative habitat heterogeneity and cumulative species richness.

Table 1

Fitted species accumulation curves from the smallest to the largest and from the largest to the smallest, considering area and perimeter, and from the lowest to highest and from the highest to lower habitat heterogeneity of ponds surveyed on the state of Goiás, Brazil.

Predictor	Ordination	R^2	Parameter	Estimated	SE	t	P level
Area							
	Large to small	0.974	intercept	0.006	0.001	4.062	<0.001
			z-value	0.705	0.019	36.425	<0.001
	Small to large	0.969	intercept	16.099	0.485	33.172	<0.001
			z-value	0.103	0.003	37.175	<0.001
Perimeter							
	Large to small	0.993	intercept	0.306	0.022	14.151	<0.001
			z-value	0.499	0.007	72.161	<0.001
	Small to large	0.962	intercept	13.880	0.531	26.156	<0.001
			z-value	0.139	0.004	33.028	<0.001
Heterogeneity							
	High to low	0.973	intercept	15.513	0.497	31.187	<0.001
			z-value	0.327	0.009	37.381	<0.001
	Low to high	0.977	intercept	20.106	0.416	48.341	<0.001
			z-value	0.255	0.006	42.485	<0.001

4. Discussion

We found that an equal area partitioned in various small ponds retain a larger part of anuran diversity than the same area in a single large ponds. However, when we considered habitat heterogeneity, we did not find a clear relationship in respect of which areas aggregate greater number of species, if are several homogeneous ponds or single heterogeneous ponds, rejecting our hypothesis about habitat heterogeneity. Our results also support the claim that, when the intention is to conserve higher species richness, pond area/perimeter should be evaluated prior to habitat heterogeneity. This is an important finding, since habitat heterogeneity is a variable more difficult to access than pond size, from which we could access free through satellite images (as example, Google Earth). Otherwise, these results may highlight that the ultimate factor determining these results are the relative importance of between-pond spatial relationships (generating higher compositional dissimilarity among small ponds).

We have found evidences that several small ponds contain more anuran species than a comparable area (and perimeter) of few large ponds, as also observed by Oertli and collaborators (2002). Besides amphibians, this relationship seems to be true to other group of aquatic organisms associated with ponds, such as aquatic plants, Gastropoda, Coleoptera, Odonata and other macroinvertebrates (Martínez-Sanz et al., 2012a; Oertli et al., 2002). This result does not necessary imply that area itself is the best predictor to species richness on local or regional scale, but accumulating several small ponds is the better choice to conserve amphibians regional diversity. Small ponds can have a similar species richness than large water bodies, but unique species composition, even within small spatial scales (Hamerlík et al., 2013; Kiflawi et al., 2003; Scheffer et al., 2006), which may be more important to regional diversity.

The fact that many small ponds accumulate higher species richness than a single large pond implies that species composition of small ponds is more diverse among them, than large ponds.There is a set of mechanisms that could be attributed to this high beta diversity on small ponds, such as spatial isolation, stochastic events, and environmental characteristics (Hamerlík et al., 2013; Scheffer et al., 2006). Spatial isolation of ponds, for example, can provide distinct communities due low connectivity, which is a barrier to movement dispersions (Hamerlík et al., 2013; Scheffer et al., 2006). Thus, the greater community heterogeneity may be a result of stochastic events acting on the colonization process (Oertli et al., 2002; Scheffer et al., 2006; Williams, 2003). Small ponds can vary extensively in physical characteristics over a small spatial scale (Kiflawi et al., 2003), and they may reflect specific micro-sites conditions provided by environmental characteristics (Hamerlík et al., 2013; Scheffer et al., 2006). For example, small ponds were also more variable in terms of substrate heterogeneity than larger water bodies (Hamerlík et al., 2013), which is highly associated with diversity of some organism, as macroinvertebrates or aquatic plants (Suurkuukka et al., 2012). Moreover, smaller ponds have more variability consider physicochemical conditions than large ponds, and they are more subject to multiple influences on a variety of scales (Davies et al., 2008a). Due their small catchment areas, ponds in close proximity are enable to have quite different catchment characteristics (Davies et al., 2008a; Davies et al., 2008b; Williams, 2003). All this factors make ponds substantially more productivity and biologically active than larger ponds or lakes (Downing, 2010), being an important contribution to regional diversity.

Habitat heterogeneity is known to be one of the major drivers of species diversity (Martinez-Sanz et al., 2012b; Suurkuukka et al., 2012), and because of this we should expected that increasing the number of ponds we are increasing the habitat heterogeneity, and consequently we are increasing the species richness, as proposed by niche theory (Martínez-Sanz et al., 2012b). However, we have found that habitat heterogeneity is not the best predictor to be considered when our proposal is to accumulate the higher number of species. But we cannot discard their importance to amphibian's diversity. It is known that habitat heterogeneity is an important factor driving species composition and diversity at local scale (e.g. (R. A. Silva et al., 2011; Vasconcelos et al., 2009), being important for both local and regional diversity (unpublished Signorelli et al 2014).

Most ponds we have studied here were originally created and maintained as drinking ponds for cattle or as source of water to crop irrigation. Ponds created to agricultural purposes are common on Brazilian Cerrado, and this number tends to increase according to economic activities and their respective dependencies on water resources (De Marco et al., 2014). Some authors have suggested the construction of ponds, properly managed, to sustain amphibiansôpopulation in agricultural landscapes (e.g. Peltzer et al., 2006; Ruggiero et al., 2008; F. R. Silva et al., 2011). Even created to economical purposes, they make a positive contribution to the maintenance of aquatic habitat (Ruggiero et al., 2008). These ponds can be used as stepping-stones to reduce inter breed sites distances, being highly important for successful and maintenance of amphibian populations (Semlitsch, 2002). Moreover, small ponds inserted in an agricultural landscape represent islands of aquatic biodiversity, in which provide habitat for aquatic and semi-aquatic group of organisms surrounded by otherwise species-poor environment (Declerck et al., 2006; Ruggiero et al., 2008; Sayer et al., 2012).

Small ponds are considered to be of high conservation value (Oertli et al., 2002), and they contribute significantly to regional biodiversity, since they can support heterogeneous communities of aquatic organisms (De Marco et al., 2014; Oertli et al., 2002). This implies that they should have higher priority in conservation concern and landscape management (Hamer and McDonnell, 2008), and the most parsimonious way is reducing conflicts between biodiversity conservation and human development. However, the management and conservation efforts have been focused especially upon large water bodies and ponds have been overlooked. Indeed, the Brazilian Cerrado ponds are neglected habitats despite their importance for biodiversity and maintenance of economic activities (De Marco et al. 2014). Some actions can be easily adopted for maintenance of pond integrity, as example the restriction of access for cattle to a limited section of ponds, which may reduce the impact of trampling and may provide major diversity (Declerck et al., 2006). Thus, we suggested that small, isolated and properly management ponds as a strategic tool enabling biodiversity conservation of Brazilian Cerrado.

Acknowledgements

We thank Animal Behavior ad Herpetology Lab, from the Federal University of Goiás, Brazil, for the logistic support. LS and DSN acknowledge fellowship by Coordenação de Aperfeiçoamento de Pessoal de Nível (CAPES). Financial support was provided by the Ministério da Ciência, Tecnologia e Inovação/Conselho Nacional de Desenvolvimento Científico e Tecnológico (CNPq 563075/2010-4) and Fundação de Amparo à Pesquisa do Estado de São Paulo (FAPESP 2010/52321-7).

References

Allouche, O., Kadmon, R., 2009. A general framework for neutral models of community dynamics. Ecol. Lett. 12, 1287ï 97. doi:10.1111/j.1461-0248.2009.01379.x

Allouche, O., Kalyuzhny, M., Moreno-Rueda, G., Pizarro, M., Kadmon, R., 2012. Area-heterogeneity tradeoff and the diversity of ecological communities. Proc. Natl. Acad. Sci. U. S. A. 109, 17495ï 500. doi:10.1073/pnas.1208652109

Arrhenius, O., 1921. Species and area. J. Ecol. 9, 95ï 99.

Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Weatherby, A., 2005. 15 years of pond assessment in Britain: results and lessons learned from the work of Pond Conservation. Aquat. Conserv. Mar. Freshw. Ecosyst. 15, 693ï 714. doi:10.1002/aqc.745

CONCEA, 2013. Diretrizes da prática de eutanásia do CONCEA Brasília/DF.

Connor, E., McCoy, E., 2001. Species-area relationships. Encycl. Biodivers. 5, 397ï 411.

Davies, B., Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Sear, D., Bray, S., Maund, S., 2008a. Comparative biodiversity of aquatic habitats in the European agricultural landscape. Agric. Ecosyst. Environ. 125, 1ï 8. doi:10.1016/j.agee.2007.10.006

Davies, B.R., Biggs, J., Williams, P.J., Lee, J.T., Thompson, S., 2008b. A comparison of the catchment sizes of rivers, streams, ponds, ditches and lakes: implications for protecting aquatic biodiversity in an agricultural landscape. Hydrobiologia 597, 7ï 17. doi:10.1007/s10750-007-9227-6

De Marco, P., Nogueira, D.S., Correa, C.C., Vieira, T.B., Silva, K.D., Pinto, N.S., Bichsel, D., Hirota, A.S.V., Vieira, R.R.S., Carneiro, F.M., Oliveira, A. a. B., Carvalho, P., Bastos, R.P., Ilg, C., Oertli, B., 2014. Patterns in the organization of Cerrado pond biodiversity in Brazilian pasture landscapes. Hydrobiologia 723, 87ï 101. doi:10.1007/s10750-013-1695-2

Declerck, S., De Bie, T., Ercken, D., Hampel, H., Schrijvers, S., Van Wichelen, J., Gillard, V., Mandiki, R., Losson, B., Bauwens, D., Keijers, S., Vyverman, W., Goddeeris, B., De meester, L., Brendonck, L., Martens, K., 2006. Ecological characteristics of small farmland ponds: Associations with land use practices at multiple spatial scales. Biol. Conserv. 131, 523ï 532. doi:10.1016/j.biocon.2006.02.024

Dengler, J., 2009. Which function describes the species-area relationship best? A review and empirical evaluation. J. Biogeogr. 36, 728ï 744. doi:10.1111/j.1365-2699.2008.02038.x

Diamond, J., 1975. The island dilemma: lessons of modern biogeographic studies for the design of natural reserves. Biol. Conserv. 129ï 146.

Downing, J., 2010. Emerging global role of small lakes and ponds: little things mean a lot. Limnetica 29, 9ï 24.

Economo, E.P., 2011. Biodiversity conservation in metacommunity networks: linking pattern and persistence. Am. Nat. 177, E167ï 80. doi:10.1086/659946

Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. Annu. Rev. Ecol. Evol. Syst. 34, 487ï 515. doi:10.1146/132419

Fattorini, S., 2010. The use of cumulative area curves in biological conservation: A cautionary note. Acta Oecologica 36, 255ï 258. doi:10.1016/j.actao.2009.11.005

Foley, J. a, Defries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E. a, Kucharik, C.J., Monfreda, C., Patz, J. a, Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science. 309, 570ï 4. doi:10.1126/science.1111772

Gotelli, N.J., Colwell, R.K., 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. Ecol. Lett. 4, 379ï 391. doi:10.1046/j.1461-0248.2001.00230.x

Hamer, A.J., McDonnell, M.J., 2008. Amphibian ecology and conservation in the urbanising world: A review. Biol. Conserv. 141, 2432ï 2449. doi:10.1016/j.biocon.2008.07.020

Hamer, A.J., Parris, K.M., 2011. Local and landscape determinants of amphibian communities in urban ponds. Ecol. Appl. 21, 378ï 90.

Hamer, A.J., Parris, K.M., 2013. Predation Modifies Larval Amphibian Communities in Urban Wetlands. Wetlands 33, 641ï 652. doi:10.1007/s13157-013-0420-2

Hamerlík, L., Svitok, M., Novikmec, M., Oļ adlík, M., Bituġík, P., 2013. Local, among-site, and regional diversity patterns of benthic macroinvertebrates in high altitude waterbodies: do ponds differ from lakes? Hydrobiologia 723, 41ï 52. doi:10.1007/s10750-013-1621-7

Hartel, T., Moga, C.I., David, a., Coroiu, I., 2009. Species richness ð pond area relationships of amphibians and birds in two Natura 2000 protected areas of Romania. Community Ecol. 10, 159ï 164. doi:10.1556/ComEc.10.2009.2.4

Hecnar, S., MôCloskey, R., 1997. Patterns of nestedness and species association in a pond-dwelling amphibian fauna. Oikos 80, 371ï 381.

Heyer, W.R., Donnelly, M.A., McDiarmid, R.W., Hayek, L.C., Foster, M.S., 1994. Measuring and monitoring biological diversity: standard methods for amphibians. Smithsonian Institution Press, Washington.

Kadmon, R., Allouche, O., 2007. Integrating the effects of area, isolation, and habitat heterogeneity on species diversity: a unification of island biogeography and niche theory. Am. Nat. 170, 443ï 54. doi:10.1086/519853

Kiflawi, M., Eitam, a., Blaustein, L., 2003. The relative impact of local and regional processes on macro-invertebrate species richness in temporary pools. J. Anim. Ecol. 72, 447ï 452. doi:10.1046/j.1365-2656.2003.00712.x

Kolasa, J., Manne, L.L., Pandit, S.N., 2012. Speciesï area relationships arise from interaction of habitat heterogeneity and species pool. Hydrobiologia 685, 135ï 144. doi:10.1007/s10750-011-0846-6

Krauss, J., Bommarco, R., Guardiola, M., Heikkinen, R.K., Helm, A., Kuussaari, M., Lindborg, R., Ockinger, E., Pärtel, M., Pino, J., Pöyry, J., Raatikainen, K.M., Sang, A., Stefanescu, C., Teder, T., Zobel, M., Steffan-Dewenter, I., 2010. Habitat fragmentation causes immediate and time-delayed biodiversity loss at different trophic levels. Ecol. Lett. 13, 597ï 605. doi:10.1111/j.1461-0248.2010.01457.x

Leary, S., Underwood, W., Anthony, R., Cartner, S., Corey, D., Grandin, T., Greenacre, C., Gwaltney-brant, S., McCrackin, M.A., Meyer, R., Miller, D., Shearer, J., Yanong, R., Lilly, E., Clinic, A.V., Walla, W., Collins, F., Polytechnic, V., State, M., Golab, G.C., Division, A.W., Patterson-kane, E., Scientist, A.W., 2013. AVMA Guidelines for the Euthanasia of Animals;: 2013 Edition.

Martínez-Sanz, C., Cenzano, C.S.S., Fernández-Aláez, M., García-Criado, F., 2012a. Relative contribution of small mountain ponds to regional richness of littoral macroinvertebrates and the implications for conservation. Aquat. Conserv. Mar. Freshw. Ecosyst. 22, 155ï 164. doi:10.1002/aqc.2227

Martinez-Sanz, C., Fernandez-Alaez, C., Garcia-Criado, F., 2012b. Richness of littoral macroinvertebrate communities in mountain ponds from NW Spain: what factors does it depend on? J. Limnol. 71, 154ï 163. doi:10.4081/jlimnol.2012.e16

McGarigal, K., Cushman, S.A., Neel, M.C. Ene, E., 2002. FRAGSTATS v3: Spatial Pattern Analysis Program for Categorical Maps. Computer software program produced by the authors at the University of Massachusetts, Amherst. Available at the following web site: http://www.umass.edu/landeco/research/fragstats/fragstats.html

Mouquet, N., Loreau, M., 2003. Community patterns in source-sink metacommunities. Am. Nat. 162, 544ï 57. doi:10.1086/378857

Oertli, B., Biggs, J., Céréghino, R., Grillas, P., Joly, P., Lachavanne, J.-B., 2005. Conservation and monitoring of pond biodiversity: introduction. Aquat. Conserv. Mar. Freshw. Ecosyst. 15, 535ï 540. doi:10.1002/aqc.752

Oertli, B., Joye, D.A., Castella, E., Juge, R., Cambin, D., Lachavanne, J.-B., 2002. Does size matter? The relationship between pond area and biodiversity. Biol. Conserv. 104, 59ï 70. doi:10.1016/S0006-3207(01)00154-9

Ovaskainen, O., 2002. Long-term persistence of species and the SLOSS problem. J. Theor. Biol. 419ï 433.

Peltzer, P.M., Lajmanovich, R.C., Attademo, A.M., Beltzer, A.H., 2006. Diversity of Anurans Across Agricultural Ponds in Argentina. Biodivers. Conserv. 15, 3499ï 3513. doi:10.1007/s10531-004-2940-9

Quinn, J.F., Harrison, S.P., 1988. Effects of habitat fragmentation and isolation on species richness: evidence from biogeographic patterns. Oecologia 75, 132ï 140. doi:10.1007/BF00378826

Rödel, M.-O., Ernst, R., 2004. Measuring and monitoring amphibian diversity in tropical forests. I. An evaluation of methods with recommendations for standardization. Ecotropica 10, 1ï 14.

Ruggiero, A., Céréghino, R., Figuerola, J., Marty, P., Angélibert, S., 2008. Farm ponds make a contribution to the biodiversity of aquatic insects in a French agricultural landscape. C. R. Biol. 331, 298ï 308. doi:10.1016/j.crvi.2008.01.009

Sayer, C., Andrews, K., Shilland, E., Edmonds, N., Edmonds-Brown, R., Patmore, I., Emson, D., Axmacher, J., 2012. The role of pond management for biodiversity conservation in an agricultural landscape. Aquat. Conserv. Mar. Freshw. Ecosyst. 22, 626ï 638. doi:10.1002/aqc.2254

Scheffer, M., van Geest, G.J., Zimmer, K., Jeppesen, E., Søndergaard, M., Butler, M.G., Hanson, M.A., Declerck, S., De Meester, L., 2006. Small habitat size and isolation can promote species richness: second-order effects on biodiversity in shallow lakes and ponds. Oikos 112, 227ï 231. doi:10.1111/j.0030-1299.2006.14145.x

Scherer, R.D., Muths, E., Noon, B.R., 2012. The importance of local and landscape-scale processes to the occupancy of wetlands by pond-breeding amphibians. Popul. Ecol. 54, 487ï 498. doi:10.1007/s10144-012-0324-7

Signorelli, L., Nogueira, D.S., Corrêa, C.C., Bastos, R.P., De Marco, P, 2014. More individual hypothesis and habitat heterogeneity drive amphibian ponds metacommunities. Unpublished.

Semlitsch, R.D., 2002. Critical Elements for Biologically Based Recovery Plans of Aquatic-Breeding Amphibians. Conserv. Biol. 16, 619ï 629. doi:10.1046/j.1523-1739.2002.00512.x

Silva, F.R., Oliveira, T. A. L., Gibbs, J.P., Rossa-Feres, D.C., 2011. An experimental assessment of landscape configuration effects on frog and toad abundance and diversity in tropical agro-savannah landscapes of southeastern Brazil. Landsc. Ecol. 27, 87ï 96. doi:10.1007/s10980-011-9670-7

Silva, R.A., Martins, I.A., Rossa-Feres, D.D.C., 2011. Environmental heterogeneity: Anuran diversity in homogeneous environments. Zoologia 28, 610ï 618. doi:10.1590/S1984-46702011000500009

Simberloff, D., Abele, L., 1976. Island biogeography theory and conservation practice. Science. 191, 285ï 286.

StatSoft, INC. 2005. Statistica (data analysis software system), version 7.1. www.statsoft.com.

Suurkuukka, H., Meissner, K.K., Muotka, T., 2012. Species turnover in lake littorals: spatial and temporal variation of benthic macroinvertebrate diversity and community composition. Divers. Distrib. 18, 931ï 941. doi:10.1111/j.1472-4642.2012.00889.x

Vasconcelos, T.S., Santos, T.G., Rossa-Feres, D.C., Haddad, C.F., 2009. Influence of the environmental heterogeneity of breeding ponds on anuran assemblages from southeastern Brazil. Can. J. Zool. 87, 699ï 707. doi:10.1139/Z09-058

Williams, P., 2003. Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. Biol. Conserv. 115, 329ï 341. doi:10.1016/S0006-3207(03)00153-8

Wu, J., 2008. Changing perspectives on biodiversity conservation: from species protection to regional sustainability. Biodivers. Sci. 16, 205ï 213. doi:10.3724/SP.J.1003.2008.08037

Supplementary Online Materials for: ñANURAN CONSERVATION IN BRAZILIAN CERRADO: SINGLE LARGE OR SEVERAL SMALL PONDS, WHICH IS BETTER?ò

2. Material and methods

Table A.1.

Characteristic of ponds	Condition	Score
Position	Open area	1
	External edge of the forest	2
	Internal edge of the forest	3
	Inside the forest	4
Edge	Ravine	2
	Flat	3
	Slope	4
	Excavated	1
Substratum	Rock	1
	Stone	2
	Gravel	3
	Sand	4
	Clay	6
	Lama	6
	Leaf	5
Vegetation types inside the pond	No vegetation	1
	Submerged	4
	Floating	7
	Herbaceous	6
	Shrubby	5
	Sparse Trees	2
	Cattail	3
Vegetation types in the margin	No vegetation	1
	Erect herbaceous	5
	Creeping herbaceous	4
	Shrubby	6
	Sparse tree	2
	Cattail	3
Land use	Short-term crop rotation	1
	Long-term crop rotation	2
	Capoeira	4
	Forest	5
	Pasture	3
Watercolor	White	1
	Black	2
	Clear	2

3. Results

Table B.1. Species composition of amphibians registered in the present study and the occurrence number of study sites where they were observed.

Species	Dist	Occurrence
Bufonidae		
Rhinella mirandaribeiroi (Gallardo, 1965)	E	1
Rhinella rubescens	E	1
Rhinella schneideri	W	22
Hylidae		
Bokermannohyla sapiranga Brandão, Magalhães, Garda, Campos, Sebben, and Maciel, 2012	E	1
Dendropsophus cruzi (Pombal and Bastos, 1998)	E	36
Dendropsophus jimi (Napoli and Caramaschi, 1999)	E	42
Dendropsophus minutus (Peters, 1872)	W	70
Dendropsophus nanus (Boulenger, 1889)	W	39
Dendropsophus rubicundulus (Reinhardt and Lütken, 1862)	E	41
Dendropsophus soaresi (Caramaschi and Jim, 1983)	CA	3
Dendropsophus sp.	W	2
Hypsiboas albopunctatus (Spix, 1824)	W	70
Hypsiboas crepitans (Wied-Neuwied, 1824)	W	2
Hypsiboas goianus (Lutz, 1968)	E	3
Hypsiboas lundii (Burmeister, 1856)	E	31
Hypsiboas paranaiba Carvalho and Giaretta, 2010	E	10
Hypsiboas phaeopleura (Caramaschi and Cruz, 2000)	E	1
Hypsiboas punctatus (Schneider, 1799)	W	1
Hypsiboas raniceps (Cope, 1862)	W	28
Hypsiboas sp.	E	7
Lysapsus caraya Gallardo, 1964	E	5
Phyllomedusa azurea Cope, 1862	E	33
Phyllomedusa oreades Brandão, 2002	E	3
Pseudis bolbodactyla Lutz, 1925	AT	25
Scinax centralis Pombal and Bastos, 1996	E	2
Scinax constrictus Lima, Bastos, and Giaretta, 2005	E	9
Scinax fuscomarginatus (Lutz, 1925)	W	65
Scinax fuscovarius (Lutz, 1925)	W	41
Scinax pusillus Pombal, Bilate, Gambale, Signorelli, and Bastos, 2011	E	18
Scinax rogerioi Pugliese, Baêta, and Pombal, 2009	E	2
Scinax similis (Cochran, 1952)	AT	7
Scinax squalirostris (Lutz, 1925)	S	1
Trachycephalus typhonius (Linnaeus, 1758)	W	2

Leptodactylidae		
Adenomera saci (Bokermann, 1956)	E	3
Leptodactylus furnarius Sazima and Bokermann, 1978	E	6
Leptodactylus fuscus (Schneider, 1799)	W	34
Leptodactylus latrans (Steffen, 1815)	W	44
Leptodactylus labyrinthicus (Spix, 1824)	W	38
Leptodactylus leptodactyloides (Andersson, 1945)	W	6
Leptodactylus mystaceus (Spix, 1824)	W	2
Leptodactylus mystacinus (Burmeister, 1861)	AT	2
Leptodactylus podicipinus (Cope, 1862)	O	21
Leptodactylus pustulatus (Peters, 1870)	E	6
Leptodactylus sertanejo Giaretta and Costa, 2007	E	2
Physalaemus centralis Bokermann, 1962	E	25
Physalaemus cuvieri Fitzinger, 1826	W	90
Physalaemus sp.	S	2
Physalaemus marmoratus (Reinhardt and Lütken, 1862)	E	1
Physalaemus nattereri (Steindachner, 1863)	E	15
Pseudopaludicola saltica (Cope, 1887)	E	4
Pseudopaludicola sp.	E	28
Microhylidae		
Chiasmocleis albopunctata (Boettger, 1885)	E	2
Dermatonotus muelleri (Boettger, 1885)	O	3
Elachistocleis cesarii (Miranda-Ribeiro, 1920)	W	39
Odontophrynidae		
Odontophrynus sp.	S	2
Odontophrynus cultripes Reinhardt and Lütken, 1862	E	5
Odontophrynus salvatori Caramaschi, 1996	E	3

CAPÍTULO IV - Factors Affecting Landscape Occupancy for Anurans Across a Disturbance Gradient in the Brazilian Cerrado

---

Luciana Signorelli, Rogério P. Bastos, Paulo De Marco Jr,Kimberly A. With

L. Signorelli (Corresponding author)
Programa de Pós Graduação em Ecologia e Evolução, Departamento de Ecologia,
Universidade Federal de Goiás,74.001-970
Goiânia, GO, Brasil
e-mail: lfl.signorelli@gmail.com

R.P. Bastos ÅP. De Marco Junior
Departamento de Ecologia, Universidade Federal de Goiás, 74.001-970
Goiânia, GO, Brasil.

K. A. With
Laboratory for Landscape and Conservation Ecology, 2 Bushnell Hall, Division of
Biology, Kansas State University, 66505
Manhattan, KS, USA

Formatted according to *Landscape Ecology*

ABSTRACT

Although local-site variables are expected to be important in determining species occurrence, this may still be mediated by the broader landscape and management context in which the site occurs. As amphibians are good indicators of environmental disturbance, we sought to uncoverthe relationship between landscape structure and local environmental characteristic over occurrence of several frog species along a disturbance gradient in the Brazilian Cerrado. We adopted a multi-model information theoretic approach in which we related each speciesô probability of landscape occupancy to several measures indicative of broader-scale disturbance (i.e., the proportion of native habitat remaining on the landscape and the mean isolation among habitat remnants), after first correcting for species detection bias. In general, we found that the occupancy of the various frogs studied here was sensitive to both the proportion of Cerrado habitat remaining on the landscape,as well as the isolation among these remnants, and this response will depend of species.

Keywords:Anuran, Occupancy models; detection bias, landscape, local environmental.

INTRODUCTION

Site occupancy is dependent on a number of factors that operate across a range of scales (Blevins et al. 2011). Although local site variables are expected to be important in determining the likelihood of site occupancy, this may still be mediated by the broader landscape context in which the site occurs. For example, agriculture and range-management practices are typically implemented at broad landscape or regional scales, and can thus alter the suitability of whatever native habitat remains within its borders (Carvalho et al. 2009). This may be caused by a reduction in speciesô preferred habitat availability, owing to a decrease in the amount and/or increasing isolation of habitat remnants, and through more subtle changes that alter ecological flows or functional linkages among habitat remnants (Cushman 2006; Becker et al. 2007). Land-use changes may alter the permeability or resistance of the matrix to movement, thereby exacerbating isolation-by-distance effects on dispersal, colonization success and gene flow among habitat remnants (Chetkiewicz et al. 2006; Gilbert-Norton et al. 2010; Beier et al. 2011). Land-use change may also introduce new or different stressors that decrease habitat suitability (e.g., increased competition or predation, food availability), which can then lead to lower abundance or population densities, higher mortality rates, and lower reproductive success within habitat remnants (Becker et al. 2010; Baudron and Giller 2014). Subsequently, site avoidance and local extinctions may reduce the occurrence of the species across the landscape (Nilsson et al. 2008; Becker et al. 2010). Thus, simply mapping the availability of suitable habitat is unlikely to be sufficient for evaluating a speciesô probable occurrence or distribution across the landscape, unless due consideration is given to the broader landscape context in which that habitat occurs (Scherer et al 2012).

Landscape context is well known to influence aquatic habitats (Naiman et al. 1993). Agricultural land use may alter allochthonous inputs and nutrient loads into these systems; some nutrient inputs may be increased as a consequence of run-off (*e.g.*, nitrogen and phosphorus from fertilizers), while others may be substantially reduced (*e.g.*, dead organic matter from leaf litter or terrestrial insects that subsidize aquatic food webs), whereas new threats (e.g., chemical pollutants from herbicides and pesticides or the introduction of predatory species, such as fish) may render entire systems unsuitable and uninhabitable for a given species (Knapp et al. 2003; Foley et al. 2005; Baudron and Giller 2014). Even if the apparent availability of habitat is unaltered by land use (thewetlands are still present), the suitability of these habitats may have been compromised. Furthermore, some wetland species, such as amphibians, have different habitat requirements during different stages of their life cycle (tadpoles vs. adults) or at different times of the year (breeding vs. non-breeding season), which necessitates that wetlands are also functionally connected to other upland habitats (Pope et al. 2000). Thus, agricultural land use poses a double jeopardy for amphibians, by impacting the suitability of both their wetland and terrestrial habitats, as well as the possibility that the altered land-use matrix may disrupt the functional connectivity (dispersal and gene flow) among these habitats (Becker et al. 2007; Becker et al. 2010).

In this study, we considered the relationship between landscape context and site occupancy for several anuran species (Anura: Hylidae) in the Brazilian Cerrado. The Cerrado is a vast tropical savanna that covers about a quarter of the land area in Brazil (~2 million km^2), making it the country's second largest ecoregion after the Amazon rainforest (Ratter et al. 1997). The Cerrado is characterized by a variety of vegetation types, ranging from open grasslands to dense woodlands. Among the major habitat types within the Cerrado are *campo limpo* (“clean field”), *campo sujo* (“dirty field”), *campo cerrado* (“closed field”), cerrado *senso stricto (woody savanna), cerradão (forest-like savanna)*. Gallery forests line the streams and rivers, and a variety of wetland habitats (temporary and permanent ponds, puddles, and swampy areas) occur throughout the Cerrado (Brasileiro et al. 2005). As a consequence of this habitat diversity, the Cerrado supports a high level of biological diversity, including a rich herpetofauna that includes 209 species of frogs (Valdujo et al. 2012). Differently of grasslands and savannahs all over the world, the Cerrado is being used for agricultural production, including cattle ranching, which has intensified in recent decades owing to government incentives (Ratter et al. 1997; Carvalho et al. 2009; Sano et al. 2010). Currently, only about 20% of the Cerrado remains undisturbed by agricultural land use, and only about 1.2% of it is protected (Mittermeier et al. 2000), despite being recognized as a global biodiversity hotspot (Myers et al. 2000).

We employ occupancy models within a multimodel information-theoretic framework to assess the relative effects of habitat amount, isolation between habitat remnants, and local environmental characteristics on the occurrence of several target frog species in the Brazilian Cerrado. Our habitat-amount hypothesis predicts that a speciesô probability of occurrence within a given landscape should increase with the total amount of native habitat in the landscape. Because we expect an inverse relationship between the amount of native habitat and agricultural land use, this enables us to assess landscape effects on species occurrences across a disturbance gradient. Habitat amount is not the only driver of species occurrence, however (Fahrig 2013). Habitat-isolation effects are also important, especially since many amphibians are critically dependent on terrestrial habitats in close proximity to their aquatic breeding sites. Amphibians may spend a majority of their adult lives in upland habitats or utilize these during dispersal (Cushman 2006). Thus, the increased isolation of native habitat remnants is expected to disrupt connectivity among frog populations and their critical habitats, and should therefore jeopardize the occurrence of species within the broader landscape. Finally, we cannot discount the role that local environmental factors play on the occurrence of anurans surveyed within their breeding habitat (Werner et al. 2007), especially given the physiological, ecological and behavioral constraints that restrict dispersal ability in this group (Lemckert 2004; Mazerolle and Desrochers 2005; Titon et al. 2010). Thus, local characteristics of the breeding habitat (such as amount of emergent vegetation or vegetation in the edge of the pond, profile of the pond edge) may well be a more important determinant of species occurrence than landscapes variables. Then again, species occurrence is likely to involve some combination of local and landscape effects, which we also consider within our modeling framework.

Methods

Selection of Study Landscapes and Wetlands

Our objective was to identify landscapes that represented a gradient in habitat disturbance, which is expected to reflect the degree of agricultural land use in the Cerrado. We therefore divided the central Brazilian state of Goiás into a grid of 478 cells of 0.25 latitude by 0.25 longitude (~ 25 km x 25 km). This grid was superimposed on a raster file containing land-cover information obtained from 2001 and 2002 Landsat ETM+ satellite images. For each grid cell (i.e., landscape = ~625 km^2), we calculated the overall percentage of native habitats (PLAND; including grassland, shrublands, forests, and secondary growth) and the mean Euclidean nearest-neighbor distance between native habitat remnants (ENN_MN) using Fragstats 3.3 (McGarigal and Marks, 1995). Together, these two metrics provide a measure of native habitat loss: landscapes that have been heavily disturbed and converted to agricultural land use are expected to have little native habitat remaining, with a high degree of isolation among native habitat remnants (i.e., low PLAND and a high ENN_MN).

We performed a simple regression analysis between PLAND and ENN_MN for all of our defined landscapes that were located entirely within the state of Goiás. From this relationship, we created a gradient of landscape disturbance (among 478 landscapes that varied in PLAND and ENN_MN; Figure 1A). We then pre-selected 24 landscapes (Figure 1B) that encompassed the entire disturbance gradient by selecting sites that lay along the regression line. In our final selection of study landscapes, we additionally considered the following criteria: 1) landscapes must contain agricultural land covers (cropland and/or pasture); and, 2) landscapes must contain >5% native habitat. Of the 24 pre-selected landscapes, 18 landscapes met all of our criteria and were therefore ultimately surveyed for this study (Figure 2).

We used aerial photographs and field reconnaissance to identify which wetlands (ponds) would be sampled within a given landscape. We selected ponds non-randomly to ensure that a mix of sites were surveyed within either a predominantly agricultural (disturbed) or native habitat context in each landscape. To ensure independence among sites, we selected ponds that were at least 1 km apart, as this distance exceeds the maximum annual dispersal for most species of frogs (Alex Smith and M. Green 2005). Thus, we ultimately surveyed four to nine ponds in each landscape (7.5 ± 1.26 ponds/landscape), for a total of 127 ponds across all 18 landscapes.

Anuran Survey Methods

Between 2010 and 2013, we surveyed ponds for adult frogs across all 18 landscapes during the rainy season (October-March). Each pond was visited once during the study, given the long distances and travel times between landscapes. Up to four trained observers conducted frog surveys in a given year, with one observer (L. Signorelli) participating in surveys during all three years. Each observer was responsible for surveying all the ponds in a given landscape. To assess the presence of individual species, observers spent 1 h at each pond between the hours of 1900 and 2400. Adult frogs were surveyed via a combination of acoustic and visual means (Rödel and Ernst 2004) while walking slowly around the pond and systematically searching or listening for calling males (Heyer et al. 1994). Although these methods are generally sufficient for detecting frog species (Pellet and Schmidt 2005), we additionally explore factors that could influence species detection bias as part of our occupancy modeling (see *Analysis of Species Detection Bias and Landscape-Occupancy Modeling*). A few adults were collected to confirm identification. All specimens collected were euthanized by injecting an overdose of 2% lidocaine parenterally. The use of lethal injectable agents is one of the most rapid and reliable methods for performing euthanasia, minimizing pain and distress to the animal, and has been accepted by the American Veterinary Medicine Association (Leary et al. 2013) and by the Brazilian National Council on the Control of Animal Experiments (CONCEA 2013). The specimens were then fixed in 10% formalin and preserved in 70% alcohol (Heyer et al. 1994), and have been deposited in the Zoological Collection (Coleção Zoológica) of the Universidade Federal de Goiás (ZUFG).

Local-Site Variable:

Environmental Integrity Index

We characterized the local environment of ponds using a composite measure, which we refer to as the Environmental Integrity (EI) Index. These index was created as proposed by Signorelli et al (2014), and it includes seven variables that are expected to influence site suitability for anurans: (1) profile of the pond edge, (2) substrate of the water body, (3) vegetation on the pond surface, (4) vegetation in the edge of the pond, (5) water color, (6) type of land use within 500 m of the pond, and (7) position of the pond considering to the nearest remnants of vegetation. This index thus subsumes a great deal of qualitative, semi-quantitative, and quantitative information about local environmental features into a single, easily interpretable measure. One can also test for the relative importance of each variable on the index separately for a given species, however.

Analysis of Species Detection Bias and Landscape-Occupancy Modeling

Our survey methods were designed to maximize detections of any anurans that might be present in these landscapes: 1) surveys were conducted during the rainy season, a time of maximum activity and reproduction for most anurans in this system; 2) we restricted our surveys to the time of day when adult males could be expected to be calling most actively; 3) surveys were performed only during favorable , weather conditions (i.e., no rain or wind); 4) we spent a large amount of time (1 h) at each pond in an effort to exhaustively survey the site; and, 5) observers were rigorously trained in identification and survey methods.

Nevertheless, a number of factors could still influence the likelihood of detecting a species, such as the innate ability of observers to detect different frog species. We therefore analyzed detection bias for each species using a multimodel information-theoretic approach, by including information on the individual observer, amount of rainfall that occurred on the day of the survey (taken from the meteorological station nearest the surveyed landscape), the air temperature and humidity at the time of the survey, on the likelihood of detecting the focal species (Table 1). A set of candidate models was constructed to investigate the effects of each covariate individually and in combination (two-, three-, and four-variable additive models) on detection probabilities, resulting in a total of 16 different detection models for each species (a constant detection model, four single-variable models, six two-variable models, four three-variable models, and a global model with all variables). Posteriorly, a set of candidate models was constructed to evaluate the relative effects of habitat amount, isolation between remnants, and local site characteristics (EI index) on landscape occupancy, resulting in a total of 21 candidate models for each species (see Table 2). To reduce collinearity, we analyzed the correlations among all variables used in occupancy models. Only the coefficient of variation for the pond perimeter was significantly correlated with both the mean integrity index and the mean isolation between native habitat remnants (Table 3); we therefore dropped this covariate from our candidate model set. These set of occupancy models were then paired to the best model for detection probability.

Although ponds were surveyed only a single time, we can still analyze detection bias and occupancy by treating each landscape as a sampling block and individual ponds as the replicated surveys within each landscape (Kendall and White 2009). Thus, a detection history for a given landscape is made up of the four to nine pond-surveys in which the species was either detected or not, giving a record of presence-absence across space rather than time (Kendall and White 2009). In other words, we are making a space-for-time substitution in this analysis, and are therefore assessing detection bias, and ultimately occupancy, at the scale of the entire landscape, rather than at the scale of individual ponds. This is appropriate given that species detections are assumed to be independent across sites, but is also consistent with our main objective of uncovering the factors that determine the distribution of species at the landscape scale, and not just within individual ponds.

Nevertheless, we acknowledge that there is still a potential for bias in our occupancy estimator because we were unable to sample every pond in a given landscape (exhaustive sampling), nor did we sample sites completely at random (sampling with replacement, such that each site has a possibility of being selected again at random and therefore visited again) as advocated by Kendall and White (2009) for single surveys. Our survey thus qualifies as ósampling without replacementô(because locations were selected and visited only once), which would require information on the availability of habitat for each species in order to mitigate bias. Unfortunately, habitat availability is not so easily assessed, given the diverse range of wetland habitats in which these species occurred. We note, therefore, that while bias cannot be totally eliminated in this (or any) study, we are nevertheless adjusting our occupancy estimator as needed for a number of major factors that contributed to detection bias for each species (i.e., based on the results of our detection-probability analysis). Furthermore, we restricted our site-occupancy modeling to just those species that exhibited an overall naïve occupancy of at least 20% (see below), as bias was shown to be reduced above this threshold in a simulated survey in which sampling was done without replacement (Kendall and White 2009).

Finally, detection probability and site occupancy can be robustly estimated using a single survey provided (i) the probabilities of detection and occupancy are both dependent on covariates and (ii) the set of covariates that affect detection differ by at least one variable from the set of covariates that affect occupancy (Lele et al., 2012). We meet those criteria for all but three of the seven species we focus on here (*P. azurea, E. cesarii*, and *L. fuscus* had a constant probability of detection, and thus detection of these species was not dependent on any of the covariates in our candidate model set; see Results).

Model selection and goodness-of-fit

We determined the goodness-of-fit of the global model, as proposed by MacKenzie and Bailey (2004), for both the detection and occupancy models for each species. In this technique, we calculate a Pearson Chi-square statistic and use a bootstrap procedure to determine whether the observed statistic is unusually large. If the global model is determined to be a poor fit to the data, we can use an overdispersion parameter ($\hat{c}$) to inflate standard errors and to adjust model-selection procedures (MacKenzie and Bailey 2004; MacKenzie et al. 2006). The global model is considered to be a good fit to the data (i.e, no overdispersion) when $\hat{c} \sim 1$. A lack of model fit is given either by $\hat{c} > 1$, when there is more variation in the observed data than expected by the model (overdispersion), or by $\hat{c} < 1$, when there is less variation than expected (MacKenzie and Bailey 2004). To approximate the distribution of the test statistic, we bootstrapped the data 10,000 times. As we had some situations in which $\hat{c} > 1$, it was necessary to adjust standard errors by multiplying them by $\sqrt{\hat{c}}$ (see Results). In those cases, we based our model selection on AIC scores corrected for overdispersion and small sample size corrected by the number of sample sites ($QAIC_{c,}$Burnham and Anderson 2002).

Target Species

Of the 57 frog species we identified in these Cerrado wetlands over the course of the study (see unpublished Signorelli et al. 2014), our analysis here is centered on seven species, each of which has an overall apparent (naïve) occupancy of 20-85% within landscapes. As discussed previously, occupancy rates <20% are prone to greater bias in the occupancy estimator, whereas occupancy rates >85% suggest a widely distributed and ubiquitous species whose site occupancy is unlikely to be affected by the landscape covariates we seek to uncover here. We therefore focused on the following seven species: *Dendropsophus cruzi*, *D. nanus*, *D. rubicundulus*, *Hypsiboas albopunctatus*, *Phyllomedusa azurea, Elachistocleis cesarii,* and *Leptodactylus fuscus.*

*Dendropsophus cruzi*is a diminutive tree frog (male snout-vent length, SVL: 16.3-19.4 mm) that is found in a wide range of habitats throughout the Cerrado of central Brazil (Pombal Jr. and Bastos 1998;Bastos et al 2003). Males are found calling from vegetation at pond margins in both natural and disturbed environments (Pombal Jr. and Bastos 1998; Vaz-Silva et al. 2007).*Dendropsophus nanus* is a small anuran that is common in large, flooded areas where males call from grasses and shrubs (Rossa-Feres and Jim 2001). *Dendropsophus rubicundulus* is also a small tree frog (male SVL: 18.0 - 23.4 mm; Bastos et al. 2003); males normally call form herbaceous vegetation, on branches or leaves near temporary and permanent ponds (Bastos et al. 2003). The white-spotted tree frog, *Hypsiboas albopunctatus* is of moderate size (male SVL: 53.1 ± 7.2 (SD) mm; Ribeiro et al. 2005) and is considered to be shrub-dwelling or arboreal, although it can be found vocalizing on the ground. It also has a widespread distribution, occurring in a variety of habitats (Guimarães et al. 2011). *Phyllomedusa azurea* is another medium-sized tree frog in the *Phyllomedusa hypocondrialis* group (male SVL: 31.2- 43.3 mm; Caramaschi 2006). This species also has a protracted breeding season (Prado et al. 2005; Rodrigues et al. 2007). Males vocalize on leaves, twigs, and branches of trees, in addition to the leaves of shrubs and grasses (Costa et al. 2010). It is largely arboreal, as evidenced by its unique morphological adaptation in which the first toe of the forelimb, and the first and second toes of the hindlimb, are opposable (Caramaschi 2006). *Elachistocleis cesarii* is a small, widespread species that has short legs and is fossorial (male SVL: 22.6-22.7 mm; Toledo et al. 2010). Males typically vocalize with their hindlegs in the water, while holding the emergent vegetation with the forelimbs. This species has a long breeding season, but can behave like an explosive breeder after heavy rains (Toledo et al. 2010). *Leptodactylus fuscus* is a moderate-sized terrestrial frog (male SVL: 42.8 ± 4.0 (SD) mm; Heyer 1978) that is found in a wide range of habitats, including open areas, savannahs, grasslands, degraded forests and urban habitats. Males build subterranean chambers in the dry margins of ponds, where mating occurs (Lucas et al. 2008). Finally, the butter frog, *Leptodactylus latrans*, is a nocturnal frog that is occasionally seen in the daytime (Heyer et al. 1990); this is the largest of the species we focus on here (male SVL: 92-120 mm; Gallardo, 1964). The butter frog can be found at the margin of ponds or small lakes, but can easily walk over land.

Results

Occurrence of amphibians at a landscape scale can be determined by either landscape or local variables, and these relationships will vary between species (Table 4). In general, species of the family Hylidae seems to be sensitive to landscape variables, in way that most of them were sensitive by habitat area (*Dendropsophus nanus*), habitat configuration (*D. cruzi* and *Phyllomedusa azurea*) or both habitat area and configuration (*D. rubicundulus*). Two species, *D. nanus* and *D. rubicundulus*, are sensitive to both landscape variables and local-site variables. Only *D. rubicundulus* seems to be sensitive to the disturbance gradient. *Elachistocleis cesarii* and *Hypsiboas albopunctatus* were not influenced by either landscape or local-site variables.

Dendropsophus cruzi

This species was recorded in 9 of 18 landscapes surveyed, and the naïve landscape occupancy for this species is 0.50. Because there was a lack of fit in the global detection model (χ^2 = 1237.74, p = 0.176, ⍰ = 1.384), QAICc was used for model selection and standard errors were inflated by a factor of √⍰ = 1.176. In the analysis of detection bias, the summed model weights for the four covariates are 64.57% for observer, 30.14% for rainfall, and 25.52% for both air humidity and temperature. Observer was identified as important covariate in modelling detection bias (Table 5), being present in two of three selected models. Rainfall also appears to have had an important, but more minor, effect on detection, given that this covariate was ranked third among the three top-ranked detection models (along with observer), but it appeared after the null model. Because of the major importance of the observer in the detection probability of *Dendropsophus cruzi*, we used this top-ranked model to adjust for detection bias in our candidate set of occupancy models.

As in the detection models, there was a lack of fit in the global occupancy model ($\chi^2 = 1346.80$, $p = 0.043$, $\hat{c} = 1.853$), and thus $QAIC_c$ was used for model selection and standard errors were inflated by a factor of $\sqrt{\hat{c}} = 1.361$. The summed model weights for the five covariates are: 60.80% for isolation between Cerrado remnants (ENN_MN), 34.10% for proportion of native habitat (PLAND), 24.60% for the mean EI index, 23.80% for the mean pond perimeter, and 16.10% for the coefficient of variation of the EI index. Hence, the isolation among native-habitat remnants is most important to the occurrence of *D. cruzi*, and this covariate alone appears in the top-ranked model (Table 6). The probability of landscape occupancy by *D. cruzi* tends toincrease in landscapes that have less isolation between Cerrado remnants (β_{ENN}=-1.627, 95% CI ± 2.501). All of the other top-ranked models (QAICc<2), which included the model with constant occupancy, have a similar weight, and thus we cannot assign particular importance to one or more of these covariates.   However, landscape variables (ENN_MN, pland) were present in four of the five top-ranked models, with the mean isolation of Cerrado remnants present in all of these (the constant model), whereas local-site variables (per_mean, ind_mean) were present in only two of the top-ranked models. Thus, landscapes covariates appear to be more important than local-site variables in determining the occurrence of *D. cruzi*.

Dendropsophus nanus

This species was recorded in 9 of 18 landscapes surveyed, and the naïve landscape occupancy is 0.50. The global detection model provided a good fit to the data (χ^2 = 779.948, p = 0.247, ⍰ = 1.007), so it was not necessary to adjust the standard error. In the analysis of detection bias, the summed model weights for the four covariates were 99.40% for observer, 37.30% for humidity, 29.30% for air temperature, and 24.90% for rainfall. The top-ranked model highlighted observer bias as being important in the detection of *D. nanus*, and this model was almost twice as likely as any of the other candidate models to influence detection (w_1/w_2= 0.308/0.169 = 1.822; Table 5). Moreover, the differences in negative log likelihood values of the second and third models compared to the top model were 1.00 and 0.83, respectively, with an additional parameter in both models. Thus, the increment in the fit of the other models is small, suggesting that there was little or no evidence of a relationship between detection probability and other covariates for *D. nanus*. Hence, observer bias is important in the detection of *D. nanus*, and is more plausible as any of the other candidate models for detection.

For the occupancy models, there is less variation in the observed data than expected by the global occupancy model (χ^2 = 550.87, p = 0.573, ⍰ = 0.698), so it was not necessary adjust the standard error. The summed model weights for the five covariates are: 82.00% for the coefficient of variation for the EI index, 59.20% for proportion of Cerrado habitat on the landscape, 30.30% for the mean isolation of Cerrado remnants, 10.20% for the mean pond perimeter, and 5.60% for the mean EI index. The coefficient of variation for the EI and the proportion of Cerrado remnants are in the top-ranked model, and thus are important in determining the landscape occupancy of *D. nanus* (Table 6). The occurrence of this species tends to increase on landscapes with less Cerrado remnants in which ponds surveyed had similar environmental characteristics (ɓ$_{II\_cv}$=-4.174, 95% CI ± 5.538; ɓ$_{PLAND}$=-1.690, 95% CI ± 2.512). There are two other models that are equally plausible (æAICc < 2) in explaining the landscape occupancy by *D. nanus*, but both include the coefficient of variation of the EI index of ponds, thereby reinforcing the importance of this covariate. In spite of these other two models in the top tier, the first-ranked model alone accounted for 36.80% of weight, which is the same as the second and third models combined. Thus, both local-site and landscape variables are important in predicting landscape occupancy for *D. nanus*.

Dendropsophus rubicundulus

This species was recorded in 15 of 18 landscapes surveyed, and the naïve landscape occupancy for this species is 0.833. There is less variation in the observed data than expected by the global detection model ($□^2$ = 417.39, p = 0.783, ⍰ = 0.504). In the analysis of detection bias, the summed model weights for the four covariates were 72.20% for observer, 61.10% for air temperature; 46.60% for rainfall and 33.50% for air humidity. Observer and air temperature were identified as important covariates in modelling detection bias (Table 5), and were present in the first two of the three selected models. The detection bias for *D. rubicundulus* improve with increased air temperature ($ƀ_{temp}$=0.678, 95% CI ± 0.731). Although the two top-ranked detection models had similar weight, there was no substantial improvement in the fit of the second model with the additional variable. The difference in negative log-likelihood values between the top model and the second was 1.830. Thus, our occupancy models were paired with the top detection model that included air temperature and observer as covariates.

As in the detection model, the global occupancy model is underdispersed (χ^2 = 395.51, p = 0.602, ⍰ = 0.898). The summed model weights for the five covariates are: 86.3% for the coefficient of variation of the EI index, 79.1% for the mean isolation among Cerrado remnants, 75.0% for the proportion of Cerrado remnants, 9.6% for the mean pond perimeter, and, 9.3% for the mean EI index. Only one landscape occupancy model was ranked highly and received 55% of the support; the occurrence of *D.*

*rubicundulus*was influenced by the proportion of Cerrado remnants and the mean isolation among those remnants on the landscape, as well as the variability in the EI index of ponds (Table 6). Thus, a combination of landscape and local-site variables was important in predicting the occurrence of *D. rubincundulus* at a landscape scale.

Hypsiboas albopunctatus

This species was recorded in 15 of 18 landscapes surveyed, and the naïve landscape occupancy for this species is 0.833. There is less variation in the observed data than expected by the global detection model (detection: $\chi^2 = 675.21$, $p = 0.418$, ⍰ = 0.873). The summed model weights for the four covariates were 100% for air temperature, 66.70% for air humidity, 43.40% for rainfall, and 25.90% for observer. Air temperature was clearly an important covariate in modelling detection bias, and along with air humidity, it was in the top-ranked model (Table 5). Detections of *H. albopunctatus* increased with increasing air humidity ($\beta_{hum}=0.847$, 95% CI ± 1.082) and decreasing air temperature ($\beta_{temp}=-2.260$, 95% CI ± 1.611). Beside this top model, there were two other models that met the selection criteria, and air temperature was present in all of them and air humidity was present in two of them. Thus, both variables are important in controlling detection bias for *H. albopunctatus*, and thus we include the first model in our candidate set of occupancy models.

As in the detection model, there is less variation in the observed data than expected by the global occupancy model ($\chi^2 = 636.12$, $p = 0.560$, ⍰ = 0.802). The summed model weights for the five covariates were 32.60% for proportion of Cerrado remnants, 30.50% for the mean EI index, 30.30% for the mean isolation between remnants, 20.10% for the mean pond perimeter, and 18.30% for the coefficient of variation for the EI index. The constant model (without covariates) was ranked as the best model among the candidate set (Table 6). Four other models, each with one covariate , also met our model-selection criteria, but as these models were 1.6-2.5x less likely than the top model, we conclude that landscape occupancy by *H. albopunctatus* is a constant and is unaffected by these other covariates.

Phyllomedusa azurea

This species was recorded in 9 of 18 landscapes surveyed, and the naïve landscape occupancy for this species is 0.5. There was a lack of fit in the global detection model (χ^2 = 1036.46, p = 0.243, ▯ = 1.138); thus QAIC$_c$ was used for model selection and standards errors were inflated by a factor of √▯ =1.067. The summed model weights for the four covariates were 37.43% for air humidity, 28.17% for rainfall, 27.61% for air temperature, and 9.6% for observer. The constant detection model was ranked as the best model; three other models, each with one covariate, met our model-selection criteria, but were 1.6-2.4 less-likely than the top model (Table 5). Thus, we assumed a constant probability of detection in our occupancy models.

For *Phyllomedusa azurea*, there was a lack of fit in the global occupancy model (χ^2 = 1001.77, p = 0.260, ▯ = 1.076), and thus QAICc was used for model selection and standard errors were inflated by a factor of √▯ = 1.037. The summed model weights for the five covariates were 57.60% for mean isolation between remnants, 42.50% for proportion of Cerrado remnants, 34.90% for the mean EI index, 23.50% for mean pond perimeter, and 18.50% for the coefficient of variation of the EI index. The top-ranked occupancy model indicates that landscape occupancy by *P. azurea* is influenced by the mean isolation of Cerrado remnants (Table 6). In this case, landscape occupancy by *P. azurea* actually appears to increase with increasing isolation of habitat remnants (ϐ = 1.968, 95% CI ± 3.234). Five other models met our model-selection criteria, but had 1.5-2.6x less support than the top-ranked model. Thus, we conclude that landscape occupancy by *P. azurea* is positively influenced by increasing isolation of native habitat (i.e., a landscape-scale variable).

Elachistocleis cesarii

This species was recorded in 15 of 18 landscapes surveyed, and the naïve landscape occupancy for this species is 0.833. The global detection model provided a good fit to the data (χ^2 = 843.28, p = 0.264, □= 0.999).The summed model weights for the four covariates were 39.64% for humidity, 31.91% for rainfall, 27.47% for air temperature, and 22.41% for observer. Despite the apparent importance of humidity in the detection of this species, the top-ranked and most parsimonious model was the constant-detection model (Table 5). We thus included this model in our candidate set of occupancy models.

The global occupancy model provided an adequate description of the data, in which □was approximately 1 (χ^2 = 736.43, p = 0.525, □= 0.832). The summed model weights for the five covariates were 34.20% for the mean isolation between remnants, 31.90% for proportion of Cerrado remnants, 26.80% for the mean pond perimeter, 21.60% for the mean EI index and 15.10% for the coefficient of variation in the EI index. The only model that met our model-selection criteria was the constant model, which indicates that landscape occupancy by *E. cesarii* was not influenced by any of the variables we analyzed in this study (Table 6).

Leptodactylus fuscus

This species was recorded in 12 of 18 landscapes surveyed, and the naïve landscape occupancy for this species is 0.667. There was less variation in the observed data than expected by the global detection model (χ^2 = 472.50, p = 0.850, □= 0.507). The summed model weights for the four covariates were 49.81% for rainfall, 34.11% for observer, 28.91% for air temperature, and 26.22% for humidity. The top-ranked model was the constant-detection model, although the model with rainfall also had good support; the other three models that met our model-selection criteria were 1.6-2.4x less likely than the constant model (Table 5). Although the model with rainfall was parsimonious with the constant model, we went with the constant-detection model in our occupancy models owing to its higher ranking and fewer numbers of parameters.

For the occupancy models, there was less variation in the observed data than expected by the global detection model ($\chi^2 = 549.851$, $p = 0.810$, [illegible] = 0.609). The summed model weights for the five covariates were 49.90% for mean isolation between remnants, 36.00% for the coefficient of variation for the EI index, 28.10% for the mean pond perimeter, 26.90% for proportion of Cerrado remnants on the landscape, and 17.00% for the mean EI index. The mean isolation between Cerrado remnants is important in predicting landscape occupancy for *L. fuscus*, given that it occurs in three of the five models that met our model-selection criteria (Table 6). In this case, the probability of occupancy declines with increasing isolation of habitat remnants ($\beta = -0.827$,95% CI ± 1.194). Nevertheless, the top-ranked model suggests that variability in the ecological integrity of ponds across the landscape is ultimately more important to the occurrence of *L. fuscus* at the landscape scale. The probability of occurrence for *L. fuscus* is higher in landscapes which have ponds of similar environmental quality (i.e., a low coefficient of variation). All of the other models received less than 1.7-1.9x the support of this top-ranked model. We therefore conclude that landscape occupancy for *L. fuscus* is best explained by local-site variables related to pond quality.

Discussion

Occurrence of anurans at a landscape scale can be determined by both landscape and local variables, and these relationships will vary between species. Our results support that landscape variables involving the amount and configuration of Cerrado habitat remnants were generally more important to the occurrence of our focal species than the index of local-site variables used to describe the pond breeding habitat. Landscape variables alone (either habitat area or isolation) were important to the occurrence of two anurans (*Dendropsophus cruzi*and *Phyllomedusa azurea*). Only one species had its occurrence best explained by local-site rather than landscape variables (*Leptodactylus fuscus*),whereas both landscape and local-site variables were important to the occurrence of*Dendropsophus rubicundulus*. Processes operating at a landscape scale may thus be more important in influencing patterns of landscape occupancy within these pond-breeding amphibians of the Brazilian Cerrado. Given that the amount and isolation of native-habitat remnants are inversely related to the intensity of land use, our study implicate grazing and agricultural practices as the major anthropogenic disturbances that affect the occurrence of pond-breeding anurans at a landscape scale.

Landscape configuration, as assayed by the average isolation of native-habitat remnants, was featured in the top-ranked occupancy models for three species: *Dendropsophus cruzi*, *D. rubicundulus,* and *Phyllomedusa azurea*. All three species are endemic to the Cerrado (Valdujo et al. 2012), but are influenced differently by landscape configuration. For instance, the occurrence of the two *Dendropsophus* species (*D. cruzi* and *D. rubicundulus*) was negatively affected by the isolation between habitat remnants. Most amphibians engage in seasonal movements between upland habitats and breeding ponds, and thus are likely to suffer lower dispersal success or greater mortality during dispersal when the landscape is highly altered by different land uses, such as those involving agricultural and grazing practices (Marsh and Trenham 2001). The third species, *P. azurea*, had a positive association with habitat isolation, which runs counter to expectations for most anurans, but especially for an arboreal frog. Species of the genus *Phyllomedusa* are well adapted to extreme conditions, however, which may enable them to tolerate the generally drier conditions expected in the more-open areas and longer distances between native-habitat remnants that result from agricultural conversion of the landscape. For example, this genus is commonly referred to as ñwaterproof frogsò(Faivovich et al. 2009), owing to the waxy cutaneous secretion that they wipe over their bodies and which helps to minimize evaporative water loss (Toledo and Jared 1993; Faivovich et al. 2009). Moreover, all species of this genus have skin secretions that contain bioactive peptides with antimicrobrial properties, as well as noxious or toxic compounds that provide a deterrent to predators (Calderon et al. 2011), and which together might offer protection to individuals that are forced to disperse across greater distances between habitat fragments. Unfortunately, there is no information on the dispersal behavior or landscape resistance to movement in Phyllomedusinaes. We do know, however, that *P. azurea* can be found in open areas within native Cerrado habitats as well as in highly disturbed environments elsewhere (e.g., (Freitas et al. 2008; Nomura et al. 2012; Bruschi et al. 2013), which again reinforces the results of our occupancy model analysis. More research is clearly needed to determine how or why landscape occupancy by *P. azurea* increases with increasing isolation of native-habitat remnants.

Beyond landscape configuration, habitat-area effects were evident for two species. *D. nanus* and *D. rubicundulus*, although perhaps not in the way we had expected, given that both species were negatively associated with the amount of Cerrado habitat on the landscape. Although *D. rubicundulus* is considered to be endemic of Brazilian savanna, it tends to occur in open habitats within this biome. Moreover, the number of small ponds may actually increase in landscapes with greater human development and agricultural land use, which they are mostly related to cattle raising (De Marco et al. 2014). Ironically, then, agricultural land use may be providing more breeding sites for some anuran species in the Cerrado. For other species, the amount of native habitat per se may not be important, but if there are fragments of native vegetation, independent of their size, it could provide more suitable conditions to their persistence, as shelter, foraging areas, besides facilitating dispersion movements (Silva and Rossa-Feres 2007)(Silva and Rossa-Feres 2011).

Local-site variables were important to the landscape occurrence of three species: *Leptodactylus fuscus*, *D. nanus*, and *D. rubicundulus*. Among these species, the occurrence of *D. rubicundulus* was positively associated with high spatial heterogeneity at the landscape scale (large CV of the local pond-habitat variables). This species is widely distributed throughout the Cerrado (Napoli and Caramaschi 1999; Annunziata et al. 2007; Silva et al. 2011a), and it can occur in landscapes with certain variability in environmental integrity among ponds. On the other hand, *D. nanus* and *L. fuscus* are also widely distributed throughout Brazil (Frost 2014), and are often found within open habitats and highly disturbed environments (Wynn and Heyer 2001). In the case of these two species, landscape occupancy decreased with variability in the local-site index, which may simply reflect that the pond environment is more uniform in open or disturbed landscapes in which these species have high occurrence. In spite of this, the quality of breeding ponds is generally important for anurans(e.g.(Vasconcelos et al. 2009; Silva et al. 2011b; Silva et al. 2012)), and drives both local and regional patterns of species richness in the Brazilian Cerrado (unpublished Signorelli et al. 2014). Still, the local-site index related to pond habitat was only important to the occurrence of three of our seven target species, although it was the most important covariate explaining landscape occupancy in *L. fuscus*.

The occurrence of two species, *Hypsiboas albopunctatus* and *Elachistocleis cesarii*, was independent of either local-site or landscape covariates. Among the Hylidae analyzed here, *H. albopunctatus* is considered a moderate-sized tree frog, and probably has better dispersal ability than species in the genus *Dendropsophus* or *Phyllomedusa*. In frogs, larger species can jump longer distances (Gomes et al. 2009), which can favor long-distance movement over short time periods. Large-bodied species also have proportionately less surface area, which reduces evaporative water loss and extends the amount of time they can spend away from water sources (Tracy et al. 2010). *Hypsiboas albopuntatus* and *Elachistocleis cesarii* are a generalist and can be found in a wide range of habitats, from pristine to highly environment (e.g. (Thomé and Brasileiro 2007; Muniz et al. 2008; Valdujo et al. 2012)(Muniz et al. 2008; Kopp et al. 2010; Guimarães et al. 2011; Silva et al. 2011b). *Hypsiboas albopunctatus* can use both lentic and lotic water bodies to breed (see (Thomé and Brasileiro 2007; Muniz et al. 2008; Kopp et al. 2010), and *Elachistocleis cesarii* use permanent ponds or shallow temporary ponds, even those created after rains or in rural gardens and pastureland ((Thomé and Brasileiro 2007); Rodrigues et al. 2010). Thus, it stands to reason that this species would be found most anywhere where there is water, which is likely why its occurrence was not associated with any of the local-site or landscape variables in our occupancy model analysis.

Model Concerns and Caveats

Various sources of error can cause bias in estimates of species distribution. It means that the probability of detection is less than 1, and that species can be non-detected in a site in which this species is present (MacKenzie et al. 2002; Royle 2006). There are many and varied of factors that could influence the detectability of species, and sometimes it may not be possible to identify or at least control all these sources of errors, even with a careful field design (Royle 2006). Thus, it is important to consider that exists a heterogeneity detection probability and control this detection bias in our occupancy models. Knowledge of the factors that affect detection probability can help to design efficient surveys (Pellet and Schmidt 2005).Thus, we discuss here methodological aspects that could be influencing in the detection capability, hoping the use of this information may help to control or minimize these errors in other situations.

In this study, three of seven species analyzed did not have their detectability skewed for any of considered covariates. However, two species had their detectability influenced mainly by environmental covariates, and three species by observer bias. The temperature was expected to influence the detectability of anurans, since they are ectothermic animals, and their physiology and behavior may be directly and indirectly affected by air temperature (Wells 2007). Environmental temperature influences rates of energy use and assimilation as well as performance in gathering resources and interacting with other organisms (Bennett 1990). The relationship between temperature and detection probability can vary between species, and it is important to determine ideal conditions for surveys (Pellet and Schmidt 2005).

Besides air temperature, humidity also influences the detectability of some species (*Dendropsophus nanus, Hypsiboas albopunctatus, Phyllomedusa azurea, Elachistocleis cesarii*), although only one species actually had humidity in the top-ranked detection model (*H. albopunctatus*). Terrestrial species of amphibians may dehydrate when exposed to low humidity (Dabés et al. 2012), and some species decrease their activity during nights with low air humidity (Van Sluys et al. 2012). Terrestrial amphibians are liable to dehydration, and thus the interaction with biotic and abiotic factors in the environment are important to absorb water (Dabés et al. 2012). In *Elachistocleis cesarii*, the most important model was those without covariates, but the model with humidity as a covariate was selected as the second-best model, which is consistent with the natural history of this species. Despite it being a prolonged breeder during the wet and hot season of the year, it may behave like an explosive breeder (sensu (Wells 2007)) after heavy rains and in nights with high humidity (Toledo et al. 2010). Observer bias was the variable that most influenced the detectability of species. Although all observers received training before the surveys, which has been shown to reduce inter-observer variability (Sewell et al. 2010), observers may still fail to detect species present or may incorrectly identify calls of species (Lotz and Allen 2007). In this study, observer bias was mainly detected in species of the genus *Dendropsophus*. Species of this genus are characterized by their small size and lek mating system during the breeding season, which results in many males vocalizing in a common area. Normally, species that present this behavior are easily identified by observer. However, when there are two or more closely related species that also emit similar calls, the lek behavior can generate some confusion among collectors. Thus, the most-obvious ways to minimize observer bias in future studies would be to either use a single highly trained observer across all sites and years of the study (which is generally not feasible); invest in longer and more-intensive training workshops (which would be a luxury for most studies) ; or, as we did here, correct for detection bias within the occupancy modeling framework, as we have done in this study (Lotz and Allen 2007).

Table 1. Covariates used to assess detection probability and landscape occupancy of frogs in the Brazilian Cerrado.

Covariates		Variable type	Description
Detection			
Constant	.	None	Detection assumed to be constant
Rainfall	Rain	Continuous	Precipitation (measured in millimeter) on the day of survey
Temperature (°C)	Temp	Continuous	Air temperature in the end of survey
Humidity (%)	Hum	Continuous	Air humidity in the end of survey
Observer	Obs	Categorical	Individual observer that performed the survey
Occupancy			
Constant	.	None	Occupancy assumed to be constant
Percentage of remnants	Pland	Continuous	Proportion of the landscape occupied by Cerrado remnants
Isolation	ENN_MN	Continuous	Euclidean Nearest-Neighbor Distance; distance from patch of remnant ij to nearest neighboring patch of the same type, based on patch edge-to-edge distance
Integrity Index Mean	Ind_mean	Continuous	Mean of quantitative integrity index derived from a set of qualitative and semi-quantitative habitat measures of ponds characteristics, which were proportional to the pond area
Integrity Index CV	Ind_cv	Continuous	Integrity index coefficient of variation (CV). It was calculated as follows: CV= SD/
Pond perimeter Mean	Per_mean	Continuous	Mean of ponds surveyed perimeter in each landscape
Pond perimeter CV	Per_cv	Continuous	CV of ponds surveyed perimeter in

each landscape. It was calculated as mentioned above

Table 2. Set of candidate occupancy models to test hypotheses about the effects of habitat amount, isolation, disturbance gradient (interaction between habitat amount and isolation),and local environmental factors related to breeding ponds on landscape occupancy of anurans in the Brazilian Cerrado.

Hypotheses	Models
Habitat amount hypothesis (HAH)	
	Pland
	Pland + ind_mean + per_mean
	Pland + per_mean
	Pland + ind_mean
	Pland + ind_cv
Isolation hypothesis (IH)	
	ENN_MN
	ENN_MN + ind_mean + per_mean
	ENN_MN + per_mean
	ENN_MN + ind_mean
	ENN_MN + ind_cv
Interaction between HAH and IH (HAIH)	
	Pland + ENN_MN
	Pland + ENN_MN + ind_mean + per_mean
	Pland + ENN_MN + per_mean
	Pland + ENN_MN + ind_mean
	Pland + ENN_MN + ind_cv
Local environmental hypothesis (LEH)	
	ind_mean + per_mean
	per_mean
	ind_mean
	ind_cv
Null hyposthesis (NH)	
	no covariates
Global model	
	Pland + ENN_MN + ind_mean + per_mean + ind_cv

Table 3. Pearson correlation coefficient between the different possible covariates considered for use in the occupancy models.

	PLAND	ENN_MN	per_mean	ind_mean	per_CV	ind_CV
PLAND	1.000					
ENN_MN	-0.290	1.000				
per_mean	0.350	-0.150	1.000			
ind_mean	-0.200	0.330	-0.140	1.000		
per_CV	-0.250	**0.530**	-0.040	**0.480**	1.000	
ind_CV	-0.060	0.360	0.110	0.130	0.250	1.000

P< 0.05 are in bold

Table 4. Speciesôresponses at a landscape scale considering both landscape and local predictors, resulting from occupancy models.

Species	Null model	Local-site variable	Landscape variables	
	Constant	EI index (ind_cv)	Habitat area (PLAND)	Habitat configuration (ENN_MN)
Hypsiboas albopunctatus	x			
Elachistocleis cesarii	x			
Leptodactylus fuscus		x		
Dendropsophus nanus		x	x	
Dendropsophus rubicundulus		x	x	x
Dendropsophus cruzi				x
Phyllomedusa azurea				x

Table 5. Model-selection results for the probability of detection for seven frog species in the Brazilian Cerrado. Only the top-ranked models (æAIC_c or æQAIC_c as appropriate) are shown here. See Table 1 for an explanation of the model covariates.

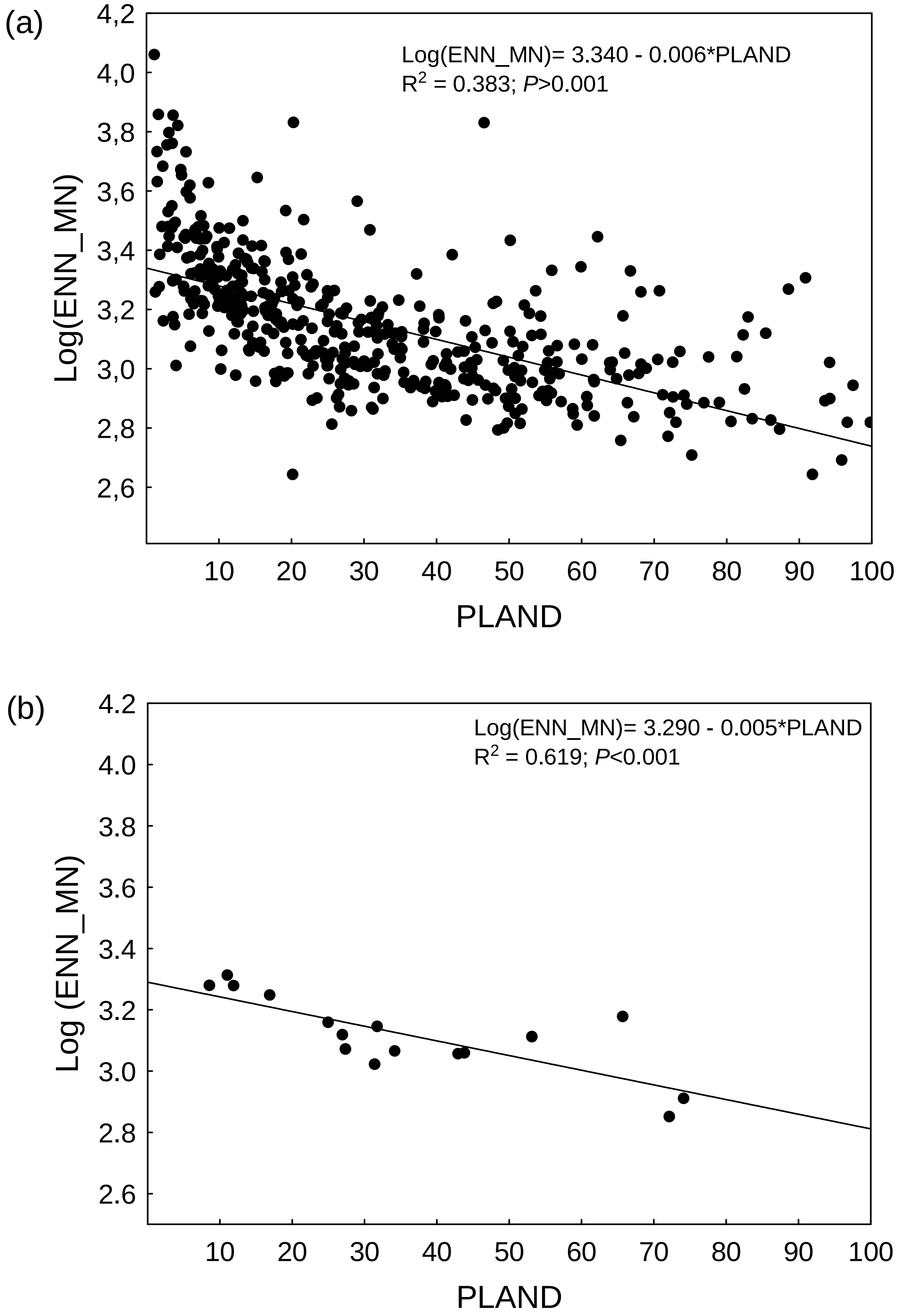
	Model	QAICc/AICc	æQAICc/æAICc	w	K	-2l
Dendropsophus cruzi						
	p(obs)psi(.)	87.880	0.000	0.244	5	107.120
	p(.)psi(.)	88.960	1.080	0.142	2	117.480
	p(rain+obs)psi(.)	89.320	1.440	0.119	6	106.070
D. nanus						
	p(obs)psi(.)	102.780	0.000	0.308	5	92.280
	p(hum+obs)psi(.)	103.980	1.200	0.169	6	91.280
	p(rain+obs)psi(.)	104.150	1.370	0.155	6	91.450
D. rubicundulus						
	p(temp+obs)psi(.)	153.530	0.000	0.198	6	140.830
	p(temp+rain+obs)psi(.)	153.940	0.410	0.161	7	139.000
	p(obs)psi(.)	155.170	1.640	0.087	5	144.670
Hypsiboas albopunctatus						
	p(temp+hum)psi(.)	150.650	0.000	0.280	4	142.320
	p(temp+hum+rain)psi(.)	151.180	0.530	0.214	5	140.680
	p(temp)psi(.)	151.710	1.060	0.165	3	145.510
Phyllomedusa azurea						
	psi(.)p(.)	131.260	0.000	0.289	2	144.760
	p(hum)psi(.)	132.140	0.880	0.186	3	143.380
	p(temp)psi(.)	133.020	1.760	0.120	3	144.380
	p(rain)psi(.)	133.030	1.770	0.119	3	144.390
Elachistocleis cesarii						
	psi(.)p(.)	158.930	0.000	0.199	2	154.830
	p(hum)psi(.)	159.110	0.180	0.182	3	152.910
	p(rain)psi(.)	159.740	0.810	0.133	3	153.540
	p(obs)psi(.)	160.780	1.850	0.079	5	150.280
	p(temp)psi(.)	160.910	1.980	0.074	3	154.710
Leptodactylus fuscus						
	psi(.)p(.)	138.450	0.000	0.181	2	134.350
	p(rain)psi(.)	138.710	0.260	0.159	3	132.510
	p(rain+obs)psi(.)	139.460	1.010	0.109	6	126.760
	p(temp)psi(.)	140.070	1.620	0.081	3	133.870
	p(obs)psi(.)	140.180	1.730	0.076	5	129.680

Table 6. Model-selection results for the probability of landscape occupancy by seven frog species in the Brazilian Cerrado. Only the top-ranked models (æAIC$_c$ or æQAIC$_c$ as appropriate) are shown here. See Table 1 for an explanation of the model covariates.

Model	Hypothesis	QAICc/ AICc	æQAICc/ æAICc	w	K	-2l
Dendropsophus cruzi						
psi(ENN_MN)	IH	67.690	0.000	0.159	6	101.870
psi(.)	NH	68.320	0.630	0.116	5	107.120
psi(pland+ENN_MN)	HAIH	68.620	0.930	0.100	7	99.450
psi(ENN_MN+per_mean)	IH	69.120	1.430	0.078	7	100.370
psi(ENN_MN+ind_mean)	IH	69.670	1.980	0.059	7	101.390
D. nanus						
psi(pland+cv_ind)	HAH	97.670	0.000	0.368	7	82.730
psi(cv_ind)	LEH	98.730	1.060	0.217	6	86.030
psi(pland+ENN_MN+cv_ind)	HAIH	99.470	1.800	0.150	8	82.250
D. rubicundulus						
psi(pland+ENN_MN+ind_cv)	HAIH	146.980	0.000	0.550	9	127.440
Hypsiboas albopunctatus						
psi(.)	NH	150.650	0.000	0.183	4	142.320
psi(ind_mean)	LEH	151.650	1.000	0.111	5	141.150
psi(ind_cv)	LEH	152.320	1.670	0.079	5	141.820
psi(pland)	HAH	152.380	1.730	0.077	5	141.880
psi(ENN_MN)	IH	152.470	1.820	0.074	5	141.970
Phyllomedusa azurea						
psi(ENN_MN)	IH	137.900	0.000	0.141	3	141.660
psi(.)	NH	138.680	0.780	0.096	2	144.760
psi(pland+ENN_MN+ind_mean)	HAIH	139.140	1.240	0.076	5	138.370
psi(pland+per_mean+ind_mean)	HAH	139.140	1.240	0.076	5	138.370
psi(ENN_MN+ind_mean)	IH	139.170	1.270	0.075	4	140.730
psi(ENN_MN+ind_cv)	IH	139.840	1.940	0.054	4	141.450
Elachistocleis cesarii						
psi(.)p(.)	NH	158.930	0.000	0.229	2	154.830
Leptodactylus fuscus						
psi(ind_cv)	LEH	137.270	0.000	0.169	3	131.070
psi(ENN_MN)	IH	138.320	1.050	0.100	3	132.120
psi(ENN_MN+per_mean)	IH	138.350	1.080	0.098	4	130.020
psi(.)	NH	138.450	1.180	0.094	2	134.350
psi(ENN_MN+ind_cv)	IH	138.500	1.230	0.091	4	130.170

Figure 1 ï Linear relationship between two landscape metrics used to characterize a landscape-disturbance gradient within the Brazilian Cerrado): PLAND, the proportion of native habitat in the landscape (a measure of habitat loss) and EMN_MN, the average nearest-neighbor distance between native-habitat remnants (a measure of habitat isolation). A) All 478 landscapes (25 km x 25 km-cells) within the state of Goiás; B) the 24 landscapes pre-selected in this study.

Figure 1


(a)
Log(ENN_MN)= 3.340 - 0.006*PLAND
R2 = 0.383; P>0.001
4,2
4,0
3,8
3,6
3,4
3,2
3,0
2,8
2,6
Log(ENN_MN)
10
20
30
40
50
60
70
80
90
100
PLAND
(b)
Log(ENN_MN)= 3.290 - 0.005*PLAND
R2 = 0.619; P<0.001
4.2
4.0
3.8
3.6
3.4
3.2
3.0
2.8
2.6
Log (ENN_MN)
10
20
30
40
50
60
70
80
90
100
PLAND

Literature Cited

Alex Smith M, M. Green D (2005) Dispersal and the metapopulation paradigm in amphibian ecology and conservation: are all amphibian populations metapopulations? Ecography (Cop) 28:110ï 128. doi: 10.1111/j.0906-7590.2005.04042.x

Annunziata BB, Fontenele WM, Castro IS, et al. (2007) Amphibia , Anura , Hylidae , Dendropsophus rubicundulus;: Distribution extension. Check List 3:242ï 243.

Bastos RP, Motta JA, Lima LP, Guimaraes LD (2003). Anfíbios da Floresta Nacional de Silvânia, Estado de Goiás. Stylo Gráfica e Editora, Goiânia.

Baudron F, Giller KE (2014) Agriculture and nature: Trouble and strife? Biol Conserv 170:232ï 245. doi: 10.1016/j.biocon.2013.12.009

Becker CG, Fonseca CR, Haddad CFB, et al. (2007) Habitat split and the global decline of amphibians. Science (80-) 318:1775ï 7. doi: 10.1126/science.1149374

Becker CG, Fonseca CR, Haddad CFB, Prado PI (2010) Habitat split as a cause of local population declines of amphibians with aquatic larvae. Conserv Biol 24:287ï 94. doi: 10.1111/j.1523-1739.2009.01324.x

Beier P, Spencer W, Baldwin RF, McRae BH (2011) Toward best practices for developing regional connectivity maps. Conserv Biol 25:879ï 92. doi: 10.1111/j.1523-1739.2011.01716.x

Bennett A (1990) Thermal dependence of locomotor capacity. Am J Physiol 259:R253ï R258.

Blevins E, Wisely SM, With KA (2011) Historical processes and landscape context influence genetic structure in peripheral populations of the collared lizard (Crotaphytus collaris). Landsc Ecol 26:1125ï 1136. doi: 10.1007/s10980-011-9631-1

Brasileiro CA, SawayaRJ, KieferMC, Martins M (2005) Amphibians of an open Cerrado fragment in southeastern Brazil. Biota Neotropica 5, 2. doi:10.1590/S1676-06032005000300006

Bruschi DP, Busin CS, Toledo LF, et al. (2013) Evaluation of the taxonomic status of populations assigned to Phyllomedusa hypochondrialis (Anura, Hylidae, Phyllomedusinae) based on molecular, chromosomal, and morphological approach. BMC Genet 14:70. doi: 10.1186/1471-2156-14-70

Burnham K, Anderson D (2002) Model selection and multi-model inference: a practical information-theoretic approach, Second edi. Technometrics 45:488.

Calderon L de A, Silva ADAE, Ciancaglini P, Stábeli RG (2011) Antimicrobial peptides from Phyllomedusa frogs: from biomolecular diversity to potential nanotechnologic medical applications. Amino Acids 40:29ï 49. doi: 10.1007/s00726-010-0622-3

Caramaschi U (2006) Redefinição do grupo de Phyllomedusa hypochondrialis, com redescrição de P. megacephala (Miranda-Ribeiro, 1926), revalidação de P. azurea Cope, 1862 e descrição de uma nova espécie (Amphibia, Anura, Hylidae). Arq do Mus Nac 64:159ï 179.

Carvalho FMV, De Marco P, Ferreira LG (2009) The Cerrado into-pieces: Habitat fragmentation as a function of landscape use in the savannas of central Brazil. Biol Conserv 142:1392ï 1403. doi: 10.1016/j.biocon.2009.01.031

Chetkiewicz C-LB, St. Clair CC, Boyce MS (2006) Corridors for Conservation: Integrating Pattern and Process. Annu Rev Ecol Evol Syst 37:317ï 342. doi: 10.1146/annurev.ecolsys.37.091305.110050

Costa T, Guimarães L, Bastos R (2010) Territorial and mating behavior in Phyllomedusa azurea (Anura: Hylidae) at a temporary pond in west-central Brazil. Phyllomedusa 9:99ï 108.

CONCEA, 2013. Diretrizes da prática de eutanásia do CONCEA. Ministério da Ciência, Tecnologia e Inovação.

Cushman SA (2006) Effects of habitat loss and fragmentation on amphibians: A review and prospectus. Biol Conserv 128:231ï 240. doi: 10.1016/j.biocon.2005.09.031

Dabés L, Bonfim V, Napoli M, Klein W (2012) Water Balance and Spatial Distribution of an Anuran Community from Brazil. Herpetologica 68:443ï 455.

De Marco P, Nogueira DS, Correa CC, et al. (2014) Patterns in the organization of Cerrado pond biodiversity in Brazilian pasture landscapes. Hydrobiologia 723:87ï 101. doi: 10.1007/s10750-013-1695-2

Fahrig L (2013) Rethinking patch size and isolation effects: the habitat amount hypothesis. J Biogeogr 40:1649ï 1663. doi: 10.1111/jbi.12130

Faivovich J, Haddad CFB, Baêta D, et al. (2009) The phylogenetic relationships of the charismatic poster frogs, Phyllomedusinae (Anura, Hylidae). Cladistics 26:227ï 261. doi: 10.1111/j.1096-0031.2009.00287.x

Foley JA, Defries R, Asner GP, et al. (2005) Global consequences of land use. Science (80-) 309:570ï 4. doi: 10.1126/science.1111772

Freitas EB de, De-Carvalho CB, Faria RG, et al. (2008) Nicho ecológico e aspectos da história natural de Phyllomedusa azurea (Anura: Hylidae, Phyllomedusinae) no Cerrado do Brasil Central. Biota Neotrop 8:2ï 11.

Frost DR (2014) Amphibian Species of the World: an Online Reference. Version 6.0. Electronic Database accessible at http://research.amnh.org/herpetology/amphibia/index.html. American Museum of Natural History, New York, USA. (accessed March 2014)

Gallardo JM (1964) Consideraciones sobre *Leptodactylus ocellatus* (L.) (Amphibia, Anura) y especies aliadas. Physis 24: 373ï 384.

Gilbert-Norton L, Wilson R, Stevens JR, Beard KH (2010) A meta-analytic review of corridor effectiveness. Conserv Biol 24:660ï 8. doi: 10.1111/j.1523-1739.2010.01450.x

Gomes FR, Rezende EL, Grizante MB, Navas C a. (2009) The evolution of jumping performance in anurans: morphological correlates and ecological implications. J Evol Biol 22:1088ï 1097. doi: 10.1111/j.1420-9101.2009.01718.x

Guimarães TCS, Figueiredo GB de, Mesquita DO, Vasconcellos MM (2011) Ecology of Hypsiboas albopunctatus (Anura: Hylidae) in a Neotropical Savanna. J Herpetol 45:244ï 250.

Heyer W (1978) Systematics of the fuscus group of the frog genus Leptodactylus (Amphibia, Leptodactylidae). Nat Hist Museum Los Angeles Cty Sci Bull 29:1ï 85.

Heyer W, Rand A, Cruz CAG, et al. (1990) Frogs of Boracéia. Arq Zool (Museu Zool da Univ São Paulo) 31:231ï 410.

Heyer WR, Donnelly MA, McDiarmid RW, et al. (1994) Measuring and monitoring biological diversity: standard methods for amphibians. 364.

Knapp R, Matthews K, Preisler HK, Jellison R (2003) Developing probabilistic models to predict amphibian site occupancy in a patchy landscape. Ecol Appl 13:1069ï 1082. doi: 10.1890/1051-0761

Kendall WL, White GC (2009) A cautionary note on substituting spatial subunits for repeated temporal sampling in studies of site occupancy. J Appl Ecol 1182ï 1188. doi: 10.1111/j.1365-2664.2009.01732.x

Kopp K, Signorelli L, Bastos R (2010) Distribuição temporal e diversidade de modos reprodutivos de anfíbios anuros no Parque Nacional das Emas e entorno, estado de Goiás, Brasil. Iheringia (Sér Zool) 100:192ï 200.

Leary S, Underwood W, Anthony R, et al. (2013) AVMA Guidelines for the Euthanasia of Animals;: 2013 Edition.

Lele SR, Moreno M, Bayne E (2012) Dealing with detection error in site occupancy surveys: what can we do with a single survey? J Plant Ecol 5: 22-31. doi: 10.1093/jpe/rtr042

Lemckert F (2004) Variations in anuran movements and habitat use: Implications for conservation. Appl Herpetol 1:165ï 181. doi: 10.1163/157075403323012179

Lotz A, Allen CR (2007) Observer Bias in Anuran Call Surveys. J Wildl Manage 71:675ï 679. doi: 10.2193/2005-759

Lucas EM, Brasileiro C a., Oyamaguchi HM, Martins M (2008) The reproductive ecology of Leptodactylus fuscus (Anura, Leptodactylidae): new data from natural temporary ponds in the Brazilian Cerrado and a review throughout its distribution. J Nat Hist 42:2305ï 2320. doi: 10.1080/00222930802254698

MacKenzie D, Nichols J, Lachman G, et al. (2002) Estimating site occupancy rates when detection probabilities are less than one. Ecology 83:2248ï 2255. doi: 10.1890/0012-9658

MacKenzie DI, Bailey LL (2004) Assessing the fit of site-occupancy models. J Agric Biol Environ Stat 9:300ï 318. doi: 10.1198/108571104X3361

MacKenzie DL, Nichols JD, Royle JA, et al. (2006) Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence. Auk 344.

McGarigal K, Marks BJ (1995) FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. USDA For. Serv. Gen. Tech. Rep.

Marsh DM, Trenham PC (2001) Metapopulation Dynamics and Amphibian Conservation. Conserv Biol 15:40ï 49. doi: 10.1046/j.1523-1739.2001.00129.x

Mazerolle MJ, Desrochers A (2005) Landscape resistance to frog movements. 464:455ï 464. doi: 10.1139/Z05-032

Myers N., Mittermeier RA, MittermeierCG, da FonsecaGAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403: 853-858. doi:10.1038/35002501.

Mittermeier, R.A., Myers, N., Mittermeier, C.G. 2000. Hotspots: Earth´s biologically richest and most endangered terrestrial ecorregions. Mexico City: CEMEX.

Muniz KPR, Giaretta AA, Silva WR, Facure KG (2008) Auto-ecologia de Hypsiboas albopunctatus (Anura , Hylidae) em área de Cerrado no sudeste do Brasil. Iheringia 98:254ï 259.

Naiman R, Decamps H, Pollock M (1993) The role of riparian corridors in maintaining regional biodiversity. Ecol Appl 3:209ï 212.

Napoli M, Caramaschi U (1999) Geographic variation of Hyla rubicundula and Hyla anataliasiasi, with the description of a new species (Anura, Hylidae). Alytes 16:165ï 175.

Nilsson SG, Franzén M, Jösson E (2008) Long-term land-use changes and extinction of specialised butterflies. Insect Conserv Divers 1:197ï 207. doi: 10.1111/j.1752-4598.2008.00027.x

Nomura F, Maciel NM, Pereira EB (2012) Diversidade de anuros (Amphibia) em áreas recuperadas de atividade mineradora e de plantio de Eucalyptus urophyla, no Brasil Central. Biosci J 28:312ï 324.

Pellet J, Schmidt B (2005) Monitoring distributions using call surveys: estimating site occupancy, detection probabilities and inferring absence. Biol Conserv 123:27ï 35. doi: 10.1016/j.biocon.2004.10.005

Pombal Jr. JP, Bastos RP (1998) Nova espécie de Hyla Laurenti, 1768 do Centro-Oeste Brasileiro e a posição taxonômica de H. microcephala meridiana B. Lutz, 1952 (Anura, Hylidae). Bol do Mus Nac 390:1ï 14.

Pope S, Fahrig L, Merriam H (2000) Landscape complementation and metapopulation effects on leopard frog populations. Ecology 81:2498ï 2508.

Prado CP de A, Uetanabaro M, Haddad CFB (2005) Breeding activity patterns, reproductive modes, and habitat use by anurans (Amphibia) in a seasonal environment in the Pantanal, Brazil. Amphibia-Reptilia 26:211ï 221. doi: 10.1163/1568538054253375

Ratter J, Ribeiro J, Bridgewater S (1997) The Brazilian cerrado vegetation and threats to its biodiversity. Ann Bot 80:223ï 230.

Ribeiro RDS, Egito GTBT Do, Haddad CFB (2005) Chave de identificação: anfíbios anuros da vertente de Jundiaí da Serra do Japi, Estado de São Paulo. Biota Neotrop 5:1ï 15. doi: 10.1590/S1676-06032005000300017

Rödel M-O, Ernst R (2004) Measuring and monitoring amphibian diversity in tropical forests. I. An evaluation of methods with recommendations for standardization. Ecotropica 10:1ï 14.

Rodrigues MT, Acosta-Galvis A, Lavilla E, Solís F, Ibáñez R, Jaramillo C Fuenmayor Q, Hardy J (2010) *Elachistocleis ovalis*. In: IUCN 2013. IUCN Red List of Threatened Species. Version 2013.2. www.iucnredlist.org (accessed May 2014).Rodrigues DJ, Uetanabaro M, Lopes FS (2007) Breeding biology of Phyllomedusa azurea Cope, 1862 and P. sauvagii Boulenger, 1882 (Anura) from the Cerrado, Central Brazil. J Nat Hist 41:1841ï 1851. doi: 10.1080/00222930701507220

Rossa-Feres D, Jim J (2001) Similaridade do sítio de vocalização em uma comunidade de anfíbios anuros na região noroeste do Estado de São Paulo, Brasil. Rev Bras Zool 18:439ï 454.

Royle JA (2006) Site occupancy models with heterogeneous detection probabilities. Biometrics 62:97ï 102. doi: 10.1111/j.1541-0420.2005.00439.x

Sano E, Rosa R, Brito J, Ferreira L (2010) Land cover mapping of the tropical savanna region in Brazil. Environ Monit é 166:113ï 24. doi: 10.1007/s10661-009-0988-4

Scherer RD, Muths E, Noon BR (2012) The importance of local and landscape-scale processes to the occupancy of wetlands by pond-breeding amphibians. Popul Ecol 54:487ï 498. doi: 10.1007/s10144-012-0324-7

Sewell D, Beebee TJC, Griffiths R A (2010) Optimising biodiversity assessments by volunteers: The application of occupancy modelling to large-scale amphibian surveys. Biol Conserv 143:2102ï 2110. doi: 10.1016/j.biocon.2010.05.019

Signorelli L, Nogueira DS, Bastos RP, De Marco P (2014) Anuran Conservation in Brazilian Cerrado: single large or severall small ponds, which is better?Unpublished

Signorelli L, Nogueira DS, Corrêa CC, Bastos RP, De Marco P (2014) More individual hypothesis and habitat heterogeneity drive amphibian ponds metacommunities. Unpublished.

Silva FR, Candeira CP, Cerqueira Rossa-Feres D (2012) Dependence of anuran diversity on environmental descriptors in farmland ponds. Biodivers Conserv 21:1411ï 1424. doi: 10.1007/s10531-012-0252-z

Silva FR da, Rossa-Feres D de C (2007) Uso de fragmentos florestais por anuros (Amphibia) de área aberta na região noroeste do Estado de São Paulo. Biota Neotrop 7:141ï 147. doi: 10.1590/S1676-06032007000200016

Silva FR, Rossa-Feres DDC (2011) Influence of terrestrial habitat isolation on the diversity and temporal distribution of anurans in an agricultural landscape. J Trop Ecol 27:327ï 331. doi: 10.1017/S0266467410000842

Silva G da, Vieira J, Alves M (2011a) Amphibia, Anura, Hylidae, Dendropsophus rubicundulus (Reinhardt and Lütken, 1862): Distribution extension in northeastern Brazil. Check List 7:843ï 844.

Silva RA, Martins IA, Rossa-Feres DDC (2011b) Environmental heterogeneity: Anuran diversity in homogeneous environments. Zoologia 28:610ï 618. doi: 10.1590/S1984-46702011000500009

Van Sluys M, Marra R V., Boquimpani-Freitas L, Rocha CFD (2012) Environmental Factors Affecting Calling Behavior of Sympatric Frog Species at an Atlantic Rain Forest Area, Southeastern Brazil. J Herpetol 46:41ï 46. doi: 10.1670/10-178

Thomé MTC, Brasileiro CA (2007) Dimorfismo sexual, uso do ambiente e abundância sazonal de Elachistocleis cf. ovalis (Anura: Microhylidae) em um remanescente de Cerrado no estado de São Paulo, sudeste do Brasil. Biota Neotrop 7:27ï 33. doi: 10.1590/S1676-06032007000100003

Titon B, Navas CA, Jim J, Gomes FR (2010) Water balance and locomotor performance in three species of neotropical toads that differ in geographical distribution. Comp Biochem Physiol A Mol Integr Physiol 156:129ï 35. doi: 10.1016/j.cbpa.2010.01.009

Toledo L, Loebmann D, Haddad C (2010) Revalidation and redescription of Elachistocleis cesarii (Miranda-Ribeiro, 1920)(Anura: Microhylidae). Zootaxa 2418:50ï 60.

Toledo R, Jared C (1993) Cutaneous adaptations to water balance in amphibians. Comp Biochem Physiol Part A Physiol 105:593ï 608. doi: 10.1016/0300-9629(93)90259-7

Tracy CR, Christian K a, Tracy CR (2010) Not just small, wet, and cold: effects of body size and skin resistance on thermoregulation and arboreality of frogs. Ecology 91:1477ï 84.

Valdujo PH, Silvano DL, Colli G, Martins M (2012) Anuran Species Composition and Distribution Patterns in Brazilian Cerrado, a Neotropical Hotspot. South Am J Herpetol 7:63ï 78. doi: 10.2994/057.007.0209

Vasconcelos TS, Santos TG, Rossa-Feres DC, Haddad CFB (2009) Influence of the environmental heterogeneity of breeding ponds on anuran assemblages from southeastern Brazil. Can J Zool 87:699ï 707. doi: 10.1139/Z09-058

Vaz-Silva W, Guedes A, Azevedo-Silva P (2007) Herpetofauna, Espora hydroelectric power plant, state of Goiás, Brazil. Check List 3:338ï 345.

Wells KD (2007) The Ecology and Behavior of Amphibians. 1400.

Werner EE, Yurewicz KL, Skelly DK, Relyea R a. (2007) Turnover in an amphibian metacommunity: the role of local and regional factors. Oikos 116:1713ï 1725. doi: 10.1111/j.2007.0030-1299.16039.x

CONCLUSÕES GERAIS

O aumento das atividades humanas tem sido uma das maiores ameaças à biodiversidade. A perda de habitat, o isolamento de remanescentes, a alteração do fluxo gênico e dos movimentos de dispersão, são exemplos de fatores que estão diretamente relacionados a alteração da configuração de uma paisagem em função da substituição de áreas nativas por áreas agricultáveis. Esse avanço das fronteiras agrícolas dificilmente é reversível, então é necessário buscar alternativas para conservar a diversidade mesmo com o aumento das atividades humanas. No entanto, a falta de conhecimento a respeito da diversidade de espécies e dos fatores que a governam em escala local e regional pode ser um problema, uma vez que reduz nosso poder preditivo. Então, com a intenção de preservar a riqueza de espécies de anuros associados a poças no Cerrado brasileiro, tentamos cobrir lacunas de amostragem e criamos a primeira lista de espécies para o estado de Goiás. No total, foram registradas 92 espécies para o estado sendo que 58 foram registradas em nossas amostragens. Dentre as 92 espécies, estão inclusas as espécies de desenvolvimento direto e as que se reproduzem em corpos dágua lótica. Este número engloba cerca de metade das espécies registradas para o bioma e, provavelmente, deve aumentar uma vez que há o conhecimento de espécies que estão sendo descritas e de espécies não identificadas, com taxonomia duvidosa.

Além disso, nosso trabalho contêm informações importantes sobre os processos que estão estruturando as assembleias de anfíbios que se reproduzem em poças. A relação espécie-área não expressa de maneira determinista o verdadeiro mecanismo que gera a riqueza de espécies, dado que a área não é um bom preditor da quantidade de energia no sistema. A energia do sistema e a heterogeneidade da paisagem são os fatores que estão estruturando as assembleias de anuros. E, seguindo o efeito de prioridade, a cada estação reprodutiva diferentes espécies podem ser pioneiras na colonização das poças e inibir o estabelecimento de espécies subsequentes. Este processo deve oscilar ao longo da escala temporal, o que tornaria a dinâmica da comunidade altamente variável em função da qualidade das manchas de habitats e pelo potencial dispersivo das espécies. Então, esperamos que poças com baixa qualidade de habitat apresentem pouca energia disponível e baixa riqueza de espécies, sendo dominadas pela espécie pioneira. Enquanto que poças com alta qualidade de habitat, os nichos estão equitativamente particionados, tendo alta energia disponível no sistema e apresentando pouca ou nenhuma dominância de uma determinada espécie,

Apesar de a relação espécie área não ser um bom preditor da riqueza de espécies de anuros, nós encontramos que várias pequenas poças retêm mais espécies do que uma única poça grande de mesma área. Isso pode acontecer, pois poças pequenas podem ter riqueza de espécies similar, mas com composição de espécies diferentes. Além disso, encontramos que é mais eficiente considerarmos a área das poças ao invés da heterogeneidade de habitats. As poças pequenas apresentam alto valor de conservação até mesmo em ambientes dominados pela agricultura. Além de serem importantes para a manutenção das atividades econômicas, as poças são trampolins ecológicos, sendo importantes para a manutenção das populações viáveis de anuros. Então, a criação de poças pequenas, isoladas e devidamente manejadas, deve ser considerada como uma estratégia para manter a diversidade de espécies de anuros em uma região, mesmo ela sendo altamente impactada.

E, para finalizar, encontramos que fatores da paisagem e variáveis locais são ambos importantes para determinar a ocorrência das espécies de anuros. No entanto, esta relação pode mudar de acordo com as espécies e de como as mesmas percebem a paisagem. Dado que a porção de habitat e o isolamento entre remanescentes são inversamente relacionados as atividades de uso de solo, nosso estudo implicam que pastagens e práticas agrícolas são os maiores fatores antrópicos que estariam afetando na ocorrência de espécies de anuros na escala da paisagem.