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MODELAGEM DE PRESSÕES, AMEAÇAS E OPORTUNIDADES PARA A CONSERVAÇÃO DE MORCEGOS NO BRASIL

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Tese apresentada ao Programa de Pós-Graduação em Biologia Animal da Universidade Federal de Pernambuco, como requisito parcial para a obtenção do título de Doutora em Biologia Animal.

Area de concentração: Conservação Animal

Orientador: Prof. Dr Enrico Barnard
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Aprovada em: 27/02/2018.

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RESUMO

A efetiva conservação e manejo das espécies requer conhecimentos detalhados sobre sua distribuição, pressões e ameaças. O Brasil é um país com dimensões continentais com a segunda maior riqueza de espécies de morcegos no mundo, mas com quase 60% de seu território sem um único registro do grupo, contrastando com altas taxas de perda de habitat. Portanto, avaliar seus padrões de distribuição e riqueza, assim como suas pressões e ameaças, é uma necessidade para sua conservação. Para preencher essas lacunas do conhecimento compilamos os registros georeferenciados e usamos modelos de distribuição de 132 espécies para gerar mapas de riqueza para todas as espécies, ameaçadas, e endêmicas do Brasil. Usamos Sistemas de Informações Geográficas para avaliar a magnitude das diferentes pressões e ameaças relacionadas à mineração, desmatamento, parques eólicos em Unidades de Proteção Integral (UPI), cavernas, remanescentes de vegetação no Brasil e sobre os mapas de riqueza. Finalmente, para determinar áreas prioritárias para a conservação, escolhemos 81 espécies identificadas como prioritárias, e realizamos uma análise de lacunas para avaliar a representatividade das UPI na proteção de morcegos. Utilizamos o software MARXAN, para identificar potenciais áreas de conservação complementares. Considerando células de $25km^2$, os resultados indicaram que 90% do território encontra-se sem registros de morcegos, e que existe um alto viés tanto espacial (~75% dos registros só na Mata Atlântica e Amazônia) como taxonômico (72% dos registros pertencentes a família Phyllostomidae). Os modelos gerados predizem que a riqueza de espécies de morcegos no Brasil varia entre 6 e 112 espécies/$25km^2$, e ressaltam uma riqueza potencial maior que a registrada atualmente para Caatinga, Pantanal e Pampa. Encontramos que 39% das UPI estão sob influência de mineração, desmatamento e/ou parques eólicos. Se todos os projetos de mineração potenciais fossem aprovados, afetariam quase 70% das UPI. Mais de 50% das cavernas potenciais como refúgios de morcegos estão sob pressão pela mineração e desmatamento de seus arredores. Se todos os projetos de mineração em potencial fossem executados, mais de 80% das cavernas estariam ameaçadas. A Caatinga e a Mata Atlântica são os biomas mais afetados pela mineração e desmatamento e preocupantemente os que apresentam a maioria das espécies endêmicas e ameaçadas. Atualmente, 90% das espécies de morcegos no Brasil têm menos de 10% de sua distribuição em UPI. Uma expansão de 19% no atual sistema de áreas protegidas poderia melhorar significativamente a proteção de morcegos brasileiros. Esta possível expansão ocorreria principalmente na Amazônia (57%), Cerrado (21%) e Caatinga (11%). O cenário de conservação para a biodiversidade brasileira - e para os morcegos em particular - pode se agravar se as taxas de desmatamento aumentarem como resultado do novo Código Florestal, ou devido ao recente enfraquecimento das legislações ambientais. Nossa análise sugere a necessidade de reavaliar o estado de conservação de algumas espécies de morcegos no Brasil, e uma necessidade urgente de melhorias na regulamentação de mineração, agronegócios e parques eólicos no Brasil, bem como expandir o sistema atual de UPI a fim de garantir a proteção de morcegos e os serviços ecossistêmicos que eles prestam.

Palavras chave: Desmatamento. Marxan. MaxEnt. Mineração. Modelos de Distribuição de Espécies. Sistemas de informação Geográfica.

ABSTRACT

The effective conservation and management of species requires detailed knowledge of its distribution, and on the pressures and threats affecting them. Brazil is a continental-size country, holding the second highest bat species richness in the world, but nearly 60% of its area has not a single bat record, in spite of high rates of habitat loss. So, assessing their species distributions, richness, pressures and threats is a necessity for their conservation. Using species distribution modelling, here we assessed the potential distribution of 132 species of bats in Brazil. We produced maps of species richness for all evaluated species, for the threatened ones, and for those endemic to Brazil. We use Geographic Information Systems to evaluate the magnitude of different pressures and threats from mining, deforestation, and wind power generation on Strictly Protected Areas (SPA), caves, on the remaining vegetation in Brazil and on the species richness map. Finally, based on 81 species identified as having high conservation priority, we conducted a gap analysis to evaluate the representativeness of the SPA for the protection of bat species. We also used MARXAN software to identify potential complementary conservation areas. Considering grid cells of 25 km^2, we identified that 90% of the Brazilian territory has not a single bat record, and there is a high spatial bias - with ~ 75% of the records restricted to the Atlantic Forest and Amazonia-, and taxonomic bias - with 72% of the records belonging to the Phyllostomidae family. Our models suggest that the bat species richness in Brazil varies between 6 and 112 species/25 km^2 (modal 73-80 species/25 km^2), and highlight a higher potential species richness than the one currently recorded for Caatinga, Pantanal and Pampa. We found that 39% of SPA are currently affected by mining, deforestation and wind farms. If all potential mining projects would be active they would affect almost 70% of the SPA. More than 50% of the caves considered as potential roosting areas for bats are under pressure by mining and/or deforestation of their surroundings. If all possible mining projects would be executed more than 80% of potential roosting caves would be threatened. The Caatinga and the Atlantic Forest biomes are the most affected by mining, road infraestructure and deforestation, and alarmingly those harbor most of the endemic and endangered species. Currently, 90% percent of the bat species in Brazil have less than 10% of their distribution within SPA. A 19% expansion in the current protected areas system could significantly improve the protection of Brazilian bats, with most of such expansion in Amazonia (57%), Cerrado (21%) and Caatinga (11%). The conservation scenario for the Brazilian biodiversity – and for bats in particular – may deteriorate with an increase of the deforestation rates increase due to the new Forest Code, or due to the recent weakening of the national environmental legislation. We suggest the necessity to re-evaluate the conservation status of some species with restricted distributions. We also urge for an improvement of the environmental regulations of mining, agribusiness, and wind energy sectors in Brazil, as well as for an expansion of the current SPA system to guarantee the protection of bats and the ecosystem services they provide.

Keywords: Deforestation. Geographic Information Systems. Marxan. MaxEnt. Mining. Species Distribution Models.

SUMÁRIO

1 INTRODUÇÃO GERAL

1.1. A ORDEM CHIROPTERA

Os representantes da Ordem Chiroptera, cujo nome deriva do grego *cheir* (mão) e *pteron* (asa), conhecidos popularmente como morcegos, são os únicos mamíferos capazes de realizar um voo verdadeiro graças a suas mãos altamente modificadas e estruturas especializadas (PERACCHI et al., 2006). Possivelmente esta alta capacidade de dispersão e adaptabilidade lhes permitiu uma alta radiação adaptativa que os converteu no segundo grupo mais rico em espécies (só perdendo para os roedores), mas provavelmente o mais diverso dentro da mastofauna (ALTRINGHAM, 2011). Com mais de 1300 espécies, distribuídas em 18 famílias e 202 gêneros, os morcegos representam quase um quarto dos mamíferos do mundo (FENTON; SIMMONS, 2014).

Esta elevada riqueza de espécies reflete-se também em uma alta diversidade na forma como estes animais exploram os recursos alimentares e ocupam os hábitats (KALKO et al., 1996). Morcegos estão diretamente envolvidos em vários processos ecológicos: são polinizadores e dispersores de sementes, incluindo várias espécies de interesse comercial (e.g. ALLEN-WARDELL et al., 1998; GORCHOV et al., 1995; LOBOVA et al., 2009), são predadores e presas (e.g. BELWOOD; MORRIS, 1987; FENTON et al., 1992), controlam populações de insetos, incluindo pragas agrícolas (e.g. AGUIAR; ANTONINI, 2008; BOYLES et al., 2011; CLEVELAND et al., 2006; REISKIND; WUND, 2009), e também são vetores de doenças, como a raiva, com significativo impacto sobre rebanhos animais (AGUIAR et al., 2010; SCHNEIDER et al., 2009).

O amplo leque de interações ecológicas envolvendo os morcegos está diretamente relacionado às múltiplas adaptações dos morcegos para a interação com o ambiente que os cerca, e estes animais mostram-se sensíveis à perda e degradação de habitats (e.g. OCHOA, 2000). Morcegos podem ser usados como indicadores do estado de conservação de alguns ecossistemas, particularmente onde as comunidades mostram um alto grau de complexidade taxonômica e ecológica (CASTRO-ARELLANO et al., 2007; CUNTO; BERNARD, 2012; FENTON et al., 1992; MEDELLÍN et al., 2000; OCHOA, 2000; WILSON, et al., 1996). Assim, por participarem diretamente de processos ecológicos associados aos ecossistemas, e por representarem uma parcela significativa da mastofauna brasileira, morcegos – e os serviços ambientais por eles prestados – também sofrerão os impactos decorrentes da forte alteração da paisagem natural experimentada no Brasil.

1.2. MORCEGOS NO BRASIL E SUAS LACUNAS DE DISTRIBUIÇÃO

Com ao menos 180 espécies de morcegos, distribuídas em 9 famílias e 68 gêneros (FEIJÓ et al., 2015; MORATELLI; DIAS, 2015; NOGUEIRA et al., 2014), o Brasil é o segundo país com a maior riqueza de espécies (BERNARD et al., 2011) abrangendo cerca de 14% da riqueza mundial para o grupo (FENTON; SIMMONS, 2014). O país abriga 11 espécies endêmicas (MORATELLI; DIAS, 2015; NOGUEIRA et al., 2014), e sete espécies nacionalmente ameaçadas (MMA, 2016). *Lonchophylla bokermani* é ainda sinalizada como Vulnerável pela União Internacional para a Conservação da Natureza (IUCN, 2016). Além disso, atualmente estudos genéticos estão mostrando que várias das "espécies" brasileiras são de fato complexos de espécies e novas adições podem ser observadas no futuro próximo (THOISY et al., 2014).

Apesar de sua alta riqueza de espécies, e dos progressos significativos na descrição de novas espécies e gêneros a partir dos anos 2000 (e.g. FEIJÓ et al., 2015; GREGORIN; DIETCFIELD, 2005; GREGORIN et al., 2006; MORATELLI; DIAS, 2015; MORATELLI et al., 2011; TADDEI; LIM, 2010), assim como a ampliação de distribuição de várias outras no país (e.g. DA SILVA et al., 2015; DALPONTE; AGUIAR, 2009; GREGORIN; LOUREIRO, 2011; GREGORIN et al., 2015; LOUZADA et al., 2015; NOGUEIRA et al., 2008; PIMENTA et al., 2010), o Brasil continua a ser um país amplamente subamostrado, com entre 60 e 90% do território sem um único registro formal de morcegos (BERNARD et al., 2011; M. DELGADO; L. AGUIAR; R. MACHADO and E. BERNARD, em preparação).

Adicionalmente, além da escassez da informação de ocorrências e padrões de distribuição das espécies, a informação existente é altamente heterogênea e enviesada (VARZINCZAK et al., 2016). Em termos de porcentagem, a Mata Atlântica é o bioma melhor amostrado, com registros em cerca de 80% de sua extensão, enquanto a Amazônia é o menos amostrado, com 23% do seu território estudado (BERNARD et al., 2011). Já em termos de riqueza de espécies, Bernard, et al. (2011) apontaram que o número médio de espécies de morcegos por 3.000 km^2

varia de 4,8 no Pampa, a 13,7 na Mata Atlântica, valores muito baixos considerando-se que no Brasil se conhecem pelo menos 180 espécies.

Essa situação suscita preocupações considerando que diversas lacunas de conhecimento para morcegos no país sobrepõem-se com frentes de desmatamento e expansão do agronegócio (BERNARD et al., 2012; BERNARD et al., 2014). Sendo assim, estamos frente a um risco de não chegar a conhecer parte da quiropterafauna do país antes de que ela seja impactada. Assim como antes de que possamos aplicar um manejo e conservação efetivos para as espécies vulneráveis, especialmente considerando que o manejo e conservação de uma determinada espécie requerem informações básicas sobre sua história natural, seus padrões de distribuição, o uso do habitat, e as pressões e ameaças que experimentam (e.g. MARGULES; PRESSEY, 2000).

1.3. PRESSÕES E AMEAÇAS PARA A CONSERVAÇÃO DE MORCEGOS DO BRASIL

As taxas de perda, conversão e degradação dos ecossistemas brasileiros são bastante elevadas: 93% da Mata Atlântica, 80% do Cerrado, 60% da Caatinga, 54% do Pampa, 40% do Pantanal, e 23% da Amazônia já foram significativamente alterados pela ação humana (MMA, 2013). A extensão destas alterações ameaça a sobrevivência de várias espécies no país (MONTEIRO et al., 2008), incluindo morcegos. A Lista da Fauna Brasileira Ameaçada de Extinção aponta sete espécies de morcegos como ameaçadas no país (MMA, 2016). Entretanto, um número maior de espécies pode estar em risco real de extinção (BERNARD et al., 2013), mas a correta classificação da magnitude do número de espécies em risco real é dificultada pela falta de dados básicos sobre os morcegos brasileiros (BERNARD et al., 2011).

Os morcegos brasileiros experimentam pressões e ameaças de diferentes formas, origens e complexidades, que incluem, entre outras, a redução do status de proteção das cavernas brasileiras, as alterações recentes no Código Florestal, a existência de uma indústria desregulamentada de “extermínio de pragas”, a expansão do rebanho bovino e da área plantada no país, e os impactos diretos e indiretos da geração de energia elétrica (BERNARD et al., 2012). Entre estas, a perda da proteção da maioria das cavernas brasileiras pelo Decreto 6640 (Brasil - Presidência da República, 2008), assim como a flexibilização do desmatamento como resultado das alterações no Código Florestal feitas em 2011, e a perda de habitat por agronegócio podem ser consideradas como as mais preocupantes em termos de conservação. Todas resultam na perda direta, e na deterioração de habitats e seus abrigos, tanto em cavernas como na vegetação remanescente. A perda de remanescentes ocasiona uma redução da oferta de abrigos e de alimento na comunidade global, em especial para espécies dependentes de cobertura vegetal ou sensíveis à fragmentação (BERNARD et al., 2012; MEYER et al., 2008), enquanto que a destruição de cavernas pode extinguir localmente aquelas subpopulações de espécies de morcegos dependentes destes ambientes (AGUIAR et al., 2006).

No entanto, outros tópicos como os impactos diretos e indiretos da geração de energia elétrica poderiam ser tão graves quanto os primeiros, só que, até o momento, são desconhecidos no Brasil, tanto quantitativa como qualitativamente, a magnitude de tais efeitos. Um exemplo são os empreendimentos eólicos, que têm sido apontados atualmente como a maior causa de múltiplos eventos de mortandade de morcegos na Europa e o segundo na América do Norte, enquanto seus efeitos permanecem pouco conhecidos na América do Sul (O'SHEA et al., 2016; BARROS et al., 2015). Entender como estas pressões e ameaças afetam os morcegos é vital para a conservação destes animais e dos serviços ambientais que eles prestam no Brasil, e também para a reversão de um cenário que se mostra pessimista em relação ao futuro dos ecossistemas naturais brasileiros e de várias espécies em risco eminente de extinção.

1.4. APLICAÇÕES PRÁTICAS DE FERRAMENTAS COMPUTACIONAIS PARA CONSERVAÇÃO DE MORCEGOS

A Ciência tem buscado diferentes formas de lidar com as carências e vieses dos dados geográficos, as pressões e ameaças que recaem sobre as espécies e o que resta dos ecossistemas naturais do planeta (RODRÍGUES et al., 2004). O desenvolvimento de softwares e técnicas de modelagem computacional vem permitindo um salto quali-quantitativo na análise de cenários, predição da distribuição, predição de impactos e consequências, e na proposição de ações conservacionistas proativas (SARKAR et al., 2006). O Planejamento Sistemático para a Conservação – PSC (MARGULES; PRESSEY, 2000) é uma destas técnicas. O PSC utiliza-se de softwares de Sistemas de Informações Geográficas (SIG) e de aplicação sistematizada de algoritmos de otimização para a produção de cenários úteis para a tomada de decisões na conservação da biodiversidade como, por exemplo, o software Marxan (Decision Support System or Software Marxan-SSD; BALL et al., 2009; MARGULES; PRESSEY, 2000).

O uso de técnicas associando SIG, PSC e Modelos de Distribuição de Espécies (MDE) vem sendo amplamente difundido como uma importante ferramenta de conservação, adotada tanto por pesquisadores quanto agências governamentais ambientais (BRUM et al., 2013; CAVIGLIA-HARRIS, 2005; HIGGINS et al., 2005; MARGULES et al., 2002; MMA, 2002). As implicações de conservação destas técnicas são particularmente valiosas em países ou áreas mal amostradas, extensas e biologicamente diversas, com recursos limitados (COOPER-BOHANNON et al., 2016; HERNANDEZ et al., 2008). Tal é o caso do Brasil, com uma dimensão continental e altas riqueza de espécies. No país, estas técnicas vêm sendo utilizadas com resultados teóricos e práticos bastante interessantes e que têm sido úteis na definição de áreas prioritárias, zoneamento ambiental, areas protegidas e tomada de decisões (DOBROVOLSKI, et al., 2013; MACHADO et al., 2009; MMA, 2002; SILVA et al., 2008; ZIMBRES et al., 2012).

1.4.1 Modelos de distribuição de espécies e MaxEnt

Mapas de distribuição geográfica são essenciais independentemente do tipo de abordagem de conservação (por exemplo, PSC para áreas importantes para aves (Important Bird Areas, areas of zero extinction), ou áreas-chave para biodiversidade, áreas com maior endemismo ou espécies com distribuição muito restrita (KOCHA et al., 2017; MYERS et al., 2000; ORME et al., 2005). Porém, como já mencionado, para o Brasil os dados de distribuição de morcegos (bem como para outros grupos) são escassos e incompletos ou espacialmente tendenciosos (BERNARD et al., 2011; WILSON et al., 2005).

Comumente esses dados de distribuição são encontrados na forma de pontos de ocorrência e/ou "Extensão de Ocorrência – EOO" (IUCN, 2014). Os pontos de ocorrência geralmente subestimam a distribuição das espécies (os chamados erros de omissão), pois as localidades estudadas são geralmente aquelas facilmente acessadas pelos pesquisadores (BRITO et al., 2009; FERNÁNDEZ; NAKAMURA, 2015; RONDININI et al., 2006). Por outro lado, os mapas de EOO por serem polígonos com o intervalo plausível da espécie, muitas vezes construídos por pontos de interpolação ou desenhando um polígono que inclui os pontos mais externos (técnica de polígono convexo mínimo – MCP), geralmente falham ao representar a distribuição das espécies incluindo áreas inadequadas (i.e., áreas onde a espécie está ausente – ELITH; LEATHWICK, 2009; RODRIGUES, 2011). As distribuições estimadas usando esta metodologia geralmente resultam em altos níveis de erros de falsa presença (os chamados erros de comissão) e, assim, as espécies são assumidas como protegidas em locais onde na verdade não ocorrem (FOURCADE, 2016; GASTON; RODRIGUES, 2003; RONDININI et al., 2005).

As técnicas de modelagem, conhecidas como MDE emergiram para superar estes problemas de escassez de dados geográficos, seus vieses e erros da representação da distribuição de espécies por EOO ou MCP (ELITH; LEATHWICK, 2009; PETERSON, 2001). Idealmente, um MDE deve reduzir tanto os erros de omissão (falsa ausência) quanto os de comissão (falsa presença) nas distribuições das espécies (RONDININI et al., 2006). Porem também apresenta limitações, entre estas se incluim: não consideram interações biológicas, fatores históricos e biogeográficos (Elith e Leathwick, 2009; Wisz, et al., 2013). Adicionalmente, os dados do museu têm limitações e bieses, geralmente devido a coleções não padronizadas e não sistemáticas, que priorizam freqüentemente algumas áreas, tipos de habitats e espécies (Elith e Leathwick, 2009; Oliveira, et al., 2016). Esses bieses produzirão limitações para os modelos de distribuião baseados nesses dados (Elith e Leathwick, 2009; Oliveira, et al., 2016)

O MaxEnt (PHILLIPS et al., 2006) é um software cujo nome vem de seu enfoque de máxima entropia (Maximum Entropy Species Distribution Modelling), e está entre os softwares de MDE mais utilizados (MEROW et al., 2013). Ele usa dados ambientais de locais de presença conhecida e de outros pontos aleatórios dentro da área de estudo para caracterizar o "fundo" conhecido como Background. A partir destas informações incompletas, ele faz inferências das regiões de maior e menor adequabilidade ambiental para a espécie dentro de uma área (por isso também é conhecido como modelo de nicho baseado em presença ou de presença-fundo) (PHILLIPS et al., 2006; PHILLIPS; DUDÍK, 2008).

O MaxEnt possui uma série de vantagens, e dentro destas, Phillips et al. (2006) mencionam: (1) requer apenas dados de presença (dados de ausência verdadeiras são extremamente difíceis para fauna), e informações ambientais para toda a área de estudo (facilmente disponíveis on-line); (2) pode se utilizar variáveis ambientais tanto contínuas como categóricas; (3) é um algoritmo determinista, ou seja sempre garante a convergência para a ótima distribuição de probabilidade (máxima entropia); (4) a distribuição de probabilidade máxima tem uma concisa definição matemática, e, portanto, é passível de análise; (5) pode evitar modelos super ajustados usando β-regularização; (6) seu resultado ou output é um mapa com valores contínuos de adequabilidade, e pode ter uma flexibilidade e fina distinção entre a adequação das diferentes áreas modeladas; e (7) possui um bom desempenho quando a quantidade de dados de treinamento é limitada devido a sua abordagem generativa, em vez de discriminatória.

Todas estas vantagens têm resultado em um bom desempenho para um grande número de espécies e uma alta precisão preditiva em relação a outros métodos (ELITH et al., 2006) inclusive quando o tamanho amostral é pequeno (WISZ et al., 2008). Estas características, junto a uma fácil acessibilidade e uma fácil manipulação (MEROW et al., 2013), têm convertido o MaxEnt no software mais usado a partir dos anos 2000 para modelar a distribuição de morcegos, e com uma tendência no uso significativamente crescente (RAZGOUR et al., 2016).

1.4.2 Planejamento Sistemático para conservação e Marxan

Dentro dos softwares de PSC, o Marxan é um dos mais comuns e utilizados (SMITH et al., 2009; HEATHER; JEFF 2015). Ele foi criado primariamente como outros softwares de suporte de decisões, para apoiar na tomada de decisões no design de um sistema de reservas (BALL et al., 2009). Para isto, o Marxan seleciona áreas de planejamento que atinjam o menor custo/área possível e as metas de alvos conservacionistas pré-selecionados, permitindo maior ou menor ênfase no agrupamento especial de unidades de planejamento (MCDONNELL et al., 2002). Nesse sentido, o objetivo é minimizar custos e, portanto, a representação da biodiversidade deveria ser a mínima possível que garanta sua viabilidade, dado que ela entra como uma limitação (POSSINGHAM et al., 2000).

Este sistema permite, entre outros, integrar informações sobre a ocorrência e distribuição da biodiversidade, o grau de ameaça ou endemismo por ela experimentado, a presença e magnitude de vetores de pressão e ameaça, e a existência de oportunidades conservacionistas. Esta integração permite outra série de benefícios como avaliar a efetividade de um sistema existente de áreas protegidas, identificar lacunas neste sistema, identificar as áreas que cumprem os objetivos de conservação com um custo mínimo, avaliar quão bem cada opção atende aos objetivos de conservação e socioeconômicos, desenvolver planos de gerenciamento/zoneamento para as áreas selecionados, ou ainda apontar áreas prioritárias ou insubstituíveis para a conservação da biodiversidade (PRESSEY et al., 2007; RODRÍGUES et al., 2004; STEWART et al., 2003).

Outras das vantagens do Marxan é que ele permite simular cenários baseados em restrições e oportunidades, tais como a existência de áreas protegidas, os custos de aquisição de novas áreas ou a existência de usos conflitantes (MARGULES; PRESSEY, 2000). Finalmente, o algoritmo usado pelo Marxan – simulated annealing - proporciona bons resultados rapidamente, é muito flexível e trabalha muito bem com problemas em escalas muito diferentes (BALL et al., 2009). O algoritmo provou ser relativamente rápido, simples e robusto a mudanças no tamanho e tipo de problema (BALL et al., 2009).

Porem Marxan também apresenta limitações que são divididas em analíticas e operacionais (ARDRON et al., 2010). Entre as analíticas se mencionam: Marxan não consegue integrar facilmente dados estocásticos ou temporalmente dinâmicos; Dentro de Marxan, apenas uma única superfície de "custo" pode ser empregada o que sinifica que se o usuário quer incluir diferentes tipos de custos (por exemplo, custo de aquisição de terra e custo de oportunidade), esses custos devem ser combinados fora do Marxan e depois incluídos como uma única superfície de custo; E finalmente, Marxan só pode lidar com problemas binários ou duas zonas de planejamento, por exemplo, uma unidade de planejamento está dentro ou fora da reserva (ARDRON et al., 2010).

Dentre as limitações operacionais de Marxan são mencionadas (ARDRON et al., 2010): Como qualquer tipo de ferramenta de suporte, a qualidade de as soluções são um reflexo da qualidade dos dados que são utilizados. Marxan pode, como outras ferramentas, ser mal utilizado e suas saídas mal interpretadas. Enquanto o uso de Marxan como uma ferramenta de apoio à decisão pode facilitar o engajamento das partes interessadas, não é uma bala mágica para participação e aceitação do processo de planejamento. Marxan não mitiga questões contextuais, ou conflitos políticos pré-existentes. Finalmente, preparar os conjuntos de dados e arquivos de entrada de Marxan, bem como aprender seu uso adequado e terminologia contra-intuitiva, leva tempo (ARDRON et al., 2010).

No caso de morcegos, o Marxan tem sido utilizado principalmente para desenhar efetivos esquemas de monitoramento, encontrados na literatura como multi-species monitoring networks, e assim detectar possíveis alterações produzidas por mudanças climáticas (AMORIM et al., 2014; REBELO et al., 2012). O Marxan já foi utilizado para identificar os melhores pontos e conjuntos de estações de monitoramento para rastrear mudanças nas distribuições de múltiplas espécies de morcegos (AMORIM et al., 2014; REBELO et al., 2012). Já no Brasil, recentemente foi usado o Marxan para determinação de áreas prioritárias para conservação de morcegos do Cerrado (SILVA et al., 2017)

Pelo exposto até aqui, esta tese utilizará ferramentas de MDE e PSC para: 1) gerar modelos de distribuição potencial e mapas de riqueza para espécies de morcegos no Brasil (Capítulo 1), 2) identificar pressões e ameaças que recaem sobre a conservação dos morcegos no Brasil, e 3) determinar as áreas com maior concentração destas pressões e ameaças (ambos no Capítulo 2), 4) estabelecer cenários conservacionistas que contemplem áreas prioritárias para a conservação de morcegos no país, e 5) avaliar a efetividade do atual sistema de áreas protegidas brasileiro para a conservação de morcegos (ambos no Capítulo 3). Os capítulos aqui apresentados já se encontram em formato de artigo científico, a fim de agilizar suas publicações.

2 OBJETIVOS

2.1 OBJETIVO GERAL

Analisar, por meio de modelagem computacional, cenários que consideram o status do conhecimento, as pressões, ameaças e oportunidades para a conservação de morcegos no Brasil.

2.2. OBJETIVOS ESPECÍFICOS

2.2.1. Sintetizar o estado-da-arte da informação e gerar modelos de distribuição e mapeamento da riqueza, endemismo e grau de ameaça para espécies de morcegos no Brasil;

2.2.2. Identificar e espacializar a distribuição de pressões e ameaças à conservação destas espécies no país;

2.2.3. Acessar, por meio de uma análise de lacunas, a eficiência de representatividade do Sistema Nacional de Unidades de Conservação (SNUC) para a proteção destas espécies;

2.2.4. Acessar, por meio de uma análise de lacunas, a eficiência de representatividade das Áreas Prioritárias para a Conservação, Uso Sustentável e Repartição de Benefícios da Biodiversidade Brasileira, já identificadas pelo Ministério do Meio Ambiente para a proteção de espécies de morcegos no Brasil;

2.2.5. Identificar e espacializar oportunidades para a conservação destas espécies;

2.2.6. Identificar áreas prioritárias para a conservação dos morcegos brasileiros, com a elaboração de cenários conservacionistas baseados em informações sobre o estado atual, custos e oportunidades.

3 SPECIES DISTRIBUTION, RICHNESS, AND ENDEMISM PATTERNS FOR BATS IN BRAZIL

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ABSTRACT

The effective conservation and management of any given species requires detailed knowledge on its distribution. This is more problematic for highly mobile, speciose and ecologically-rich groups, like bats. Brazil is a continental-sized, bat species-rich country but with nearly 60% of its area without a single bat record. Assessing the species distribution, richness and endemism patterns for bat species in Brazil is a scientific challenge. Using species distribution modelling, here we assessed the potential distribution of 132 species of bats in Brazil. We produced three different maps of species richness for 1) all species evaluated, 2) for threatened species, 3) and for species endemic to Brazil. Based on a database of more than 9,500 records, we observed that 48% of the available occurrence data are located in the Atlantic Forest, followed by Amazonia (23%) and Cerrado (15%). Amazonia harbors the highest bat species richness in Brazil (77% of the species), followed by the Atlantic Forest and Cerrado (67% each), and the Caatinga (54%). Bat species richness in 5×5 km grid cells varied between 6 and 112 species (modal 73-80 species/25 km^2), and was highest in the northern coast of the Atlantic Forest, and Northeastern´s Caatinga. Our models suggest that Pantanal and Pampa are clearly under sampled, with less than 50% of their expected richness already assessed. Endemic and endangered bat species are distributed in the Atlantic Forest, Cerrado and Caatinga, the most threatened biomes in Brazil. In addition to the seven officially threatened bat species in Brazil, the situation for other species is worrisome, with the necessity to reassess their official conservation status. Characteristics which make species more vulnerable, such as endemism or restricted distributions, threat severity, cave dependence, food/foraging specificity and/or evolutionary singularity can - alone or combined- result in local extinctions in large portions of the country before basic information gaps are fulfilled.

KEYWORDS: Biodiversity, biological conservation, Chiroptera, Maxent, priority areas, species distribution models.

3.1. INTRODUCTION

The identification of a species´ distribution is a secular concern of scientists investigating biogeographic patterns (Brown and Lomolino, 2008). Moreover, understanding where a species occurs is essential considering the current worldwide high rates of habitat loss and degradation, the conservation pressures and threats experienced, and their drivers (IUCN, 2001; Margules and Pressey, 2000). Nevertheless, getting reliable geographical distribution data is still a big challenge for conservation purposes, especially in megadiverse countries due to limited resources for research, large information gaps and biased sampling (Bernard et al. 2011; Aguiar, et al., 2015; Newbold, 2010). Efforts to obtain such geographical distribution data are essential to map critical areas, and to estimate and predict the effects of habitat loss on biodiversity, a worldwide environmental concern (Di Marco, et al., 2014; Rondinini, et al., 2005).

Significant achievements have been made to identify priority areas for global biodiversity conservation (Di Minin, et al., 2017; Fajardo, et al., 2014; Lessmann, et al., 2014; Luo, et al., 2015). However, due to limited resources the adequate and/or exhaustive sampling of large expanses of the planet is logistically infeasible, therefore, conservationist frequently faces the difficult need to prioritize areas in detriment of others (e.g. Di Minin, et al., 2017; Margules, et al., 2002). Regardless of the type of conservation approach used [e.g. Important Bird Areas or Key Biodiversity Areas, areas with higher endemism, or based on threatened species or with very restricted ranges - (Kocha, et al., 2017; Myers, et al., 2000; Orme, et al., 2005)], geographic distribution maps are essential in these prioritization processes. Considering that survey data for most of the globe are frequently scarce or even inexistent, and the available data are often incomplete or spatially biased (Wilson, et al., 2005), modeling the potential distribution of species became a useful approach to overcome such constrains (Elith and Leathwick, 2009).

When using Geographic Information Systems (GIS), information on the species distribution is commonly found in the form of points of occurrence and/or geographic range – this is the “Extent of Occurrence” (EOO), as defined by the International Union for Conservation of Nature (IUCN, 2014). Occurrences points generally underestimate the species distribution because the studied localities are generally those easily accessed by the researchers (Brito, et al., 2009; Fernández and Nakamura, 2015; Rondinini, et al., 2006). On the other hand, maps of geographic ranges usually fail when representing species distribution because they include suitable and unsuitable areas, where the species is effectively absent (Elith and Leathwick, 2009; Rodrigues, 2011). This happens because the EOO are polygons with the species´ plausible range, often built by interpolating points or by drawing a polygon that includes the most external points (the so-called minimum convex polygon technique - MCP). Estimated distributions using this methodology generally result in high levels of false presence errors and, thereby, species are assumed to be protected in sites where in fact they do not occur (Fourcade, 2016; Gaston and Rodrigues, 2003; Rondinini, et al., 2005).

Species distribution modeling (SDM) have emerged in order to overcome the representation problems of species distribution by EOO or MCP (Elith and Leathwick, 2009; Peterson, 2001). Ideally, a SDM should reduce both omission (false absence) and commission (false presence) errors on the distributions of focus species (Rondinini, et al., 2006). Maxent (Phillips, et al., 2006) is among the most used SDMs software, and it uses environmental data of known presence localities and characterizes a sample of study area "background" to make inferences of suitability regions within an area from these incomplete information (hence, it is termed a presence–background technique) (Phillips, et al., 2006; Phillips and Dudík, 2008).

SDM are being used as an important supporting conservation tool (Costa, et al., 2010; Ferrier, et al., 2004) and several species or groups of species had their distributions assessed using this approach with Maxent: e.g. plants (Giulietti, et al., 2016), birds (Fourcade, et al., 2013), geckos (Pearson, et al., 2007) and as well as African (Lamb, et al., 2008), Asian (Hughes, et al., 2012), and European (Rebelo, et al., 2010) bats. Since the 2000s, Maxent is the most used software for bat distribution models worldwide (Razgour, et al., 2016). Moreover, SDM can be especially useful in situations requiring modelling the distribution of species-rich groups along large territorial extensions such as an entire country, region or continent (Cooper-Bohannon, et al., 2016; Elith, et al., 2006).

Bats are ideal candidates to use SDM: they are an ecologically diverse and species-rich group (~1.500 species worldwide - Fenton and Simmons, 2014), several species have wide distributions, but are frequently under sampled in biodiversity inventories (Bernard, et al., 2011; Razgour, et al., 2016). As a consequence, few species and countries have large and reliable occurrence databases on bat species distributions and large data gaps remains for most areas around the world (Lamb, et al., 2008; Lim and Engstrom, 2001). This is the case of Brazil, a continental-sized country with at least 180 known species of bats (Feijó, et al., 2015; Moratelli and Dias, 2015; Nogueira, et al., 2014). With ~14% of the known bat species, Brazil ranks as the second most speciose country in the globe (Fenton and Simmons, 2014). The country harbors 11 endemic (Moratelli and Dias, 2015; Nogueira, et al., 2014) and seven nationally threatened species (MMA, 2016). Moreover, currently genetic studies are showing that several of the Brazilian "species" are in fact complexes of species, projecting new additions to the national species list in the near future (Thoisy, et al., 2014).

In spite of such high species richness, Brazil has less than 10% of the country minimally surveyed for bats, and for nearly 60% of its territory there is not a single bat record (Bernard, et al., 2011). Such scenario raises concerns considering the severe environmental threats and pressures experienced in Brazil. Among these pressures are deforestation, mainly due to the agribusiness (Soares-Filho, et al., 2014), as well as other forms of habitat loss and degradation due to changes in cave protection (Brasil 2008), or multiple events of downgrading, downsizing, degazettement, and reclassification of Protected Areas (Bernard, et al., 2014). So, the combination of rapid and extensive processes of habitat loss and the urgent need to fill the extensive data gaps observed in the country make the task to assess bat species distribution a scientific and conservation challenge in Brazil (Bernard, et al., 2011). In order to fill part of the distributional gaps for Brazilian bat species, here we modelled the species distribution for 132 bat species in the country. In our work we (1) updated data on the distribution of Brazilian bats species, (2) used those data together with climate and environmental data to generate potential SDMs for those species, (3) generated updated bat species richness maps for the entire country, (4) determined areas of endemism, and (5) identified threatened species.

3.2. METHODS

3.2.1 Study area

This study considered the entire continental Brazilian territory located in South America, covering an area of approximately 8.5 million km^2 (IBGE, 2012). The analyses were done considering the six major terrestrial Brazilian biomes: Amazonia, Caatinga, Cerrado, Atlantic Forest, Pantanal and Pampa (Fig. 1).

3.2.2 Occurrence data

Our starting database was a collection of 6,799 records produced by the Laboratory of Bat Biology and Conservation at the University of Brasilia.Additionally, we consulted the scientific literature using the key-words “bats”, “Chiroptera”, “Brazil” -and the same terms in Portuguese- in Web of Science (https://webofknowledge.com) and Google Scholar (https://scholar.google.com/). Other data were obtained by searching for records in the databases of the Chico Mendes Institute for Biodiversity Conservation (ICMBio) (http://www.icmbio.gov.br), SpeciesLink (http://www.splink.org.br), VertNet (http://www.vertnet.org), and Global Biodiversity Information Facility (GBIF) (http://www.gbif.org). The compiled information was checked, treated and filtered for errors in location and/or taxonomy identification (Peterson, et al., 2011). We assumed species’ identifications were correct. We followed the taxonomic considerations of Nogueira et al., (2014), but for the *Lonchophylla* genus was followed the considerations of Moratelli and Dias (2015).

Generally, localities based on museum specimens are likely to exhibit the same environmental conditions and suffer from environmental biases, they are called spatial autocorrelation (Araújo and Guisan, 2006; Loiselle, et al., 2008), as consequence from unplanned surveys commonly biased in geography. To mitigate such problems, we produced a map of environmental heterogeneity of Brazil, using the bioclimatic variables available from WorldClim (http://www.worldclim.org/download) and removed records that were within 25 km of one another under the same environmental conditions, keeping the maximum possible number of localities for each species. This distance does not represent any species' dispersal capabilities but, rather, was adopted to reduce the inherent geographic biases associated with the collection without novel environmental localities (Boria, et al., 2014).

3.2.3 Environmental data

We initially considered a set of 19 bioclimatic variables from WorldClim 1.4 (http://www.worldclim.org/) derived from temperature and rainfall, plus elevation (see Supplementary Material 1 [MS1]). We also considered the Normalized Difference Vegetation Index (NDVI - a proxy for measuring vegetation cover until 2006; http://glcf.umd.edu/data/ndvi/) and slope at a 5 km resolution. In order to minimize co-linearity among bioclimatic variables, wecalculated the Pearson correlation index among the 22 pairs of variables and eliminated one (the lower contribution, after running models with all variables) of the two variables where the correlation index was ≥ 0.7 (Aguiar, et al., 2016). After that selection, we established for each species a number of more contributory and uncorrelated variables depending on the number of localities, in order to maintain a minimum ratio of two cases (localities) per variable (see MS2).

3.2.4 Potential Distribution Modeling

Using Maxent 3.3.3 (Phillips, et al., 2006), we generated different distribution models for each species with six or more points of occurrence. A total of 48 species (25%) were eliminated from our analysis due to a scarce number of records. To calculate the most likely distribution it was considered two data inputs: occurrence localities in combination with digital layers of the environmental conditions. We performed different tests to find the species-specific tuning of model settings rather than employing default settings. Such approach can reduce overfitting or underfitting, achieving optimal Maxent models for a dataset with heterogenic and biased data (Boria, et al., 2014; Radosavljevic and Anderson, 2014). To produce our models, we employed a logistic output (Phillips, et al., 2006; Phillips and Dudík, 2008) to obtain the values for habitat suitability (continuous probability from 0 to 1). Due to different sample sizes of localities for each species, we used all feature classes (linear, hinge, quadratic, product, and threshold) except the discrete class, which is only relevant for categorical variables (Phillips and Dudík, 2008). We used Maxent´s regularization multiplier parameter, for which default regularization values lead to overfitted models when spatial filtering is used to reduce the negative effects of spatial autocorrelation (Radosavljevic and Anderson, 2014). In order to limit the model complexity (mitigating these overfitting problems), we calibrated models with different values for the regularization multiplier (default setting 1.0, 2.0, 3.0, 4.0, 5.0) and evaluated the best models produced for each.

To generate overall predictive distribution models, we set the software to use 75% of the data for calibration and 25% for internal evaluation (testing data). To produce more robust results to random events linked to the selection of localities, we performed cross-validation replicates (depending on the number of localities – see MS2) in order to calculate confidence intervals. For species with small numbers of localities, we implemented a jackknife (or 'leave-one-out') procedure (Pearson, et al., 2007). Each locality was removed once from the dataset and a model was built using the remaining $n - 1$ localities. Hence, for a species with n observed localities, n separated models were built for testing.

To assess the predictive capacity or the discriminatory ability of the models we employed two assessments. A threshold-dependent assessment, using cumulative Binomial test, and a threshold-independent assessment, using the Area Under the Curve (AUC) of the Receiver Operating Characteristic (ROC) curve (Elith, et al., 2006; Elith, et al., 2011). For presence–background evaluations, AUC quantifies the probability that the model correctly orders a random presence locality higher than a random background pixel (Phillips, et al., 2006). Models with the best performance in the predictions will have AUC values close to 1, whereas AUC values close to 0.5 indicate models equal to or worse than random (Phillips and Dudík, 2008). Because AUC does not directly quantify overfitting, we quantified it by calculating the difference between the calibration and evaluation AUCs (Warren, Seifert, 2011). The smaller the difference between the two, the lesser the overfitting present in the model (Warren and Seifert, 2011). We also evaluated models by qualitative visual examination of the resulting maps, based on expert knowledge of the distribution where the species are known to occur.

3.2.5 Areas of richness, endemic and threatened species

The continue suitability values obtained in the previous modelling were converted into binary presence-absence values using the 'lowest presence threshold' (LPT) (Pearson, et al., 2007) and the 10th percentile presence threshold (Anderson, Gonzalez, 2011; Radosavljevic and Anderson, 2014). After the continued maps were converted to binary maps a quantitative and qualitative evaluation was done and in all cases the LPT threshold showed the best results. This approach can be ecologically interpreted by identifying those pixels predicted with LPT to be at least as suitable as those pixels where the species has been previously recorded (Pearson, et al., 2007). We overlapped those individual binaries distribution in order to generate a map of species richness for: 1) all species evaluated, 2) for threatened species, 3) and for species endemic to Brazil (For species information see Supplementary Material 3 [SM3]). These maps were overlapped with strictly protected areas (hereafter SPA; IUCN´s categories I to IV, Brasil 2000) (data for 2011 – MMA, 2016b). The resulting map of all species was divided into 500 grid cells (20 × 25), named from A to T, and from 1 to 25, so that the intersection of these letters and numbers allow a more precise location of areas of interest.

3.3. RESULTS

3.3.1 Known records

We compiled 22,441 georeferenced bat records for Brazil. Almost 60% of data compiled were discarded due to inconsistent coordinates, problematic taxonomy, or due to proximity and autocorrelation. Therefore, our database used 9,556 records of unique localities. Forty-eight percent of the records were from Atlantic Forest biome, 23% from Amazonia and the Pampa presented the lowest value, with only 1% (Table 1). Thirty-eight percent of the records were collected in just three states (São Paulo, Rio de Janeiro and Espírito Santo), equivalent to only 4% of the area of Brazil. On the contrary, the states of Rio Grande do Norte, Sergipe and Roraima were the least sampled (<2% of the national records) although they have a similar area to the more sampled states (~4% of Brazil) (Fig. 1).

Only 7% of the records were registered < 1950, 26% of the records belong to the period between 1950 and 1999, while 38% was registered between 2000 and 2016 (Table 2). All the biomes except the Amazon showed an increase in the records over the 3 periods. The Amazon has the largest number of records between 1950 and 1999. Patantal is highlighted as the biome with the largest proportional increase in records, having recorded four times more specimens in the last 16 years than those recorded between 1950 and 1999 (Table 2).

Nearly 77% of the Brazilian bat species were recorded in Amazonia, 67% in Cerrado and 67% in the Atlantic Forest biome (Table 1). There were 22 species restricted to: Atlantic Forest, Caatinga, Cerrado and the coastal Pampa. Eight species are known exclusively in Cerrado, five in Atlantic Forest and three in Caatinga. Phyllostomidae was by far the most recorded bat family (71% of the records), followed by Vespertilionidae (11%), and Molossidae (9%). Twenty-two percent of the records were from protected areas: 10% in Strict Protected Areas and 12% in sustainable use areas.

3.3.2 Potential Distribution Modeling

A total of 48 species (25%) were eliminated from our analysis due to a scarce number of records (< 6 records). In total, 132 species based on 9,556 occurrence records were modeled. This is ~75% of all bat species known for Brazil (Nogueira, et al., 2014). The modal number of records per species was 70.2, ranging from 6 (e.g. *Dryadonycteris capixaba*) to 557 (*Carollia perspicillata*). Nine species (*Eumops bonariensis*, *Lophostoma carrikeri*, *Macrophyllum macrophyllum*, *Micronycteris microtis*, *Micronycteris schmidtorum*, *Promops centralis*, *Nyctinomops aurispinosus*, *Tonatia bidens*, and *Sphaeronycteris toxophyllum*) did not present a significant model, considering the cumulative Binomial test (P > 0,05) and/or AUC (< 0.7) meaning that models may be showing a randomly produced distribution (see SM2). Mean AUC was 0.87 ± SD 0.07 for tests, and 0.89 ± SD 0.05 for training, suggesting that the species´ distribution models adequately fitted the input data.

3.3.3 Species richness, endemic and threatened-species distribution

Bat richness in each 5 × 5 km grid cell varied between 6 and 112 species (mean: 11 ± 8.87; modal 73-80 species/25 km^2 - Fig. 2). Three percent of the Brazilian territory was predicted to have < 20 species/25 km^2 (between cells: I9-L9 to I12-L12, and P9-P12), 73% was predicted to have between 50 and 90 species, and 11% > 90 species (cells H6, I6, K5, K6, L3-L6, O6, P6, Q6, Q18, R6, R7, R11-R16, S7-S11). Our analysis indicated that the areas with the highest potential of bat species richness are located in the coastal Atlantic forest, mainly in its northeastern region, and along its contact zone with the Caatinga biome (Fig. 2). Other high richness areas were found in the central and northern part of the Amazonia, in eastern Amazonas, central Pará and northern Maranhão states. The lowest species richness was found in the southern of Amazonia, and the transition zone between Cerrado and Caatinga. Our models suggest that the Pantanal and Pampa biomes are undersampled, and the current species richness there wasless than 50% of their expected species richness.

The most relevant areas of endemism for bat species (i.e., at least five of the eleven endemic species for Brazil (see MS2)) are in the Caatinga (79%), Atlantic Forest (20%), and Cerrado (0,9%) biomes (Fig. 3a). Specifically, they are areas harboring *Dryadonycteris capixaba*, *Lonchophylla inexpectata*, *Lonchophylla perachicchii*, *Platyrrhinus recifinus*, and *Xeronycteris vieirai*. There were also areas of endemism in the southern region of the country, with at least four endemic species in the eastern part of the Atlantic Forest. For most of the cases, those areas wereassociated with elevations >1000 m, in Rio de Janeiro, São Paulo and Santa Catarina states.

The most relevant areas considering oficially threatened bat species (i.e., with at least five of the eight species (see MS2)) were in the Cerrado (57%), Caatinga (25%), and Atlantic Forest (15%) biomes (Fig. 3b). In the Cerrado, endemic areas were specifically located in southwestern Bahia, eastern Goiás, Distrito Federal and Minas de Gerais states. In the Caatinga, they were located in the borders between Piauí, Ceará, Pernambuco and Paraíba.

3.4. DISCUSSION

3.4.1 Distributional patterns and sampling biases

Brazil has high bat richness, with at least 180 species unevenly distributed throughout its territory. Our modeling suggests that bat species richness in the Brazilian territory can strongly vary from 6 to 112 species/25 km^2, but most of the country harbors a very rich bat fauna: < 3% of Brazilian territory probably have < 20 species/25 km^2. Atlantic Forest and Amazonia are the biomes with the highest predicted species richness. However, part of the pattern we found may result from a severe regional sampling bias: 71% of the bat records available come from the Atlantic Forest and along the Amazon River, in Pará and Amazonas states. On the other hand, even with only ~7% of the records, Northeastern´s Caatinga showed a high predicted species richness. Pampa and Pantanal biomes are clearly under sampled (harboring ~4% of known records for Brazil) and their current known richness is undoubtedly a fraction of the real species richness. Bat-oriented inventories in those biomes should be priority.

The observed sampling bias is directly associated with higher concentration of researchers, financial resources, and well-established scientific institutions and zoological collections in specific parts of the country, as in the Atlantic Forest (e.g. Brito, et al., 2009; Lewinsohn and Prado, 2002). Another explanation is related with higher efforts and bat-oriented inventories, as in the case of Amazonia (Bernard and Fenton, 2002; Kalko and Handley, 2001; Presley, et al., 2009; Sampaio, et al., 2003) and the Atlantic Forest (Faria, 2006; Peracchi and Albuquerque, 1993). Similar patterns, with few well-studied areas and large knowledge gaps, are also observed for other biological groups in Brazil, such as birds (Silva, 1995), small terrestrial mammals (Carmignotto, 2004), reptiles (Colli, et al., 2002), and overall terrestrial species (Jenkins et al. 2013; Oliveira, et al., 2016). Such biased data can raise concerns, as in the case of bats, for example, several of the knowledge gaps in Brazil are located along deforestation frontiers. This is the case of the “arch of deforestation” located in southern of Amazonia, along its contact zone with the Cerrado biome (Aldrich et al., 2012). Such combination of high species richness and high threats and pressures may result in local extinctions in large portions of the country before basic information gaps are fulfilled.

Our models also highlighted the importance of remnants in the Atlantic Forest, and along eastern Caatinga and portions of Cerrado in maintaining important areas for both richness and diversity of endemic and threatened bat species in Brazil. The same applies for the role of strict protected areas. Our results suggest some of those areas may potentially harbor > 80 bat species, an estimate supported by other recently published study focused on bats in the Cerrado (Silva et al., 2017). However, the Atlantic Forest, Caatinga and Cerrado are the most threatened biomes in Brazil and the seven officially threatened species are mainly distributed in them. Worryingly, the known current distributions of other 17 species are restricted to these biomes as well. Of them, six are endemic for Brasil, three have not been evaluated by the IUCN (*Chiroderma vizottoi Dryadonycteris capixaba*, and *Lonchophylla inexpectata*), two are considered as Data Deficient (*Myotis izecksohni* and *Lasiurus ebenus*), and one as Least Concern (*Lonchophylla peracchii*). Characteristics which make species more vulnerable, such as endemism or restricted distributions, cave dependence, food/foraging specificity and/or evolutionary singularity can - alone or combined- result in local extinctions too (Jones, 2003; Sagot and Chaverri, 2015). So, in addition to the seven officially threatened bat species in Brazil, the situation for those other species restricted to the Atlantic Forest, Cerrado and Caatinga is worrisome, and their real conservation status should be reassessed (see also Gutiérrez and Marinho-Filho, 2017).

Although we identified that part of areas holding threatened and endemic species matches with strictly protected areas, the size of those areas, however, is too small (2.3% for threatened species; 2,2% for endemic species). So, the current strictly protected areas could be not enough for the *in situ* protection of the endemic and threatened species of bats in Brazil (see also Silva, et al., 2017). Particularly, due to extent and severity of the deforestation (Ribeiro, et al., 2009), the Atlantic Forest is a high priority for almost any biodiversity group (Jenkins, et al., 2013). Many species of medium/large mammals have been extirpated from much of the biome (Galetti, et al., 2015; Jorge, et al., 2013; Ribeiro, et al., 2009). The investigation of which and how bat species may have been affected by such an intense process of habitat loss and degradation is extremely necessary, highlighting the importance of *in situ* verification through inventories and community studies in the remaining fragments (Muylaert, et al., 2016).

3.4.2 Taxomomic biases

Our compilation also indicated an expressive taxonomic bias: Phyllostomidae comprises the bulk of bats sampled in Brazil. This is a clear consequence of the intensive single use of mist netting, the most widespread sampling method in Brazil, which is very efficient in capturing Phyllostomidae bats (Kalko, et al., 1996; Kuenzi, Morrison, 1998; O´Farrel, 1999). However, mist nets underestimate several species of insectivorous families (e.g. Vespertilionidae and Mormoopidae), which rely on echolocation to navigate within or near the canopy, having a high ability to detect and avoid such nets, or species foraging in open areas and above the canopy, far from the reach of nets (e.g. Molossidae) (Kalko, et al., 1996; O´Farrel, 1999; Voss and Emmons, 1996). As a consequence, our knowledge on how bats interact with the environment and respond to habitat changes is frequently incomplete or deeply biased (Cunto and Bernard, 2012).

The spatial and taxonomic bias found here it is supported by new acoustic studies in the country (Hintz, et al., in press; Hintze, et al., 2016) and reinforces the clear under sampled characteristic of the Brazilian bat fauna, with large knowledge gaps (Bernard, et al., 2011). Therefore, besides some progresses (Barros, et al., 2017; Dalponte, et al., 2016; Miranda, et al., 2015), our analysis still suggests that none of the Brazilian biomes can be considered well surveyed for bats (see also Varzinczak, et al., 2016).

3.4.3 Modelling species distributions: pros, cons, and conservation implications

Museum data have limitations and biases, usually due to non-standardized and non-systematical collections, which frequently prioritizes some areas, types of habitats and species (Elith and Leathwick, 2009; Oliveira, et al., 2016). Such biases will produce limitations for SDMs based on those data (Elith and Leathwick, 2009; Oliveira, et al., 2016). SDMs are limited regarding biological interactions, historical and biogeographical factors (Elith and Leathwick, 2009; Wisz, et al., 2013). Therefore, the high predicted species richness for Caatinga, Pantanal and northern Pampa, and the low predicted richness for northern Mato Grosso and southern Pará have to be field-verified. This validation is a critical point of SDMs (Rebelo and Jones, 2010; West, et al., 2016), especially for areas outside the known species´ geographical range (Elith, et al., 2006; Randin, et al., 2006) or where there are species occurrence gaps due to low environmental representativeness, as we presume to be the case of the northern of Mato Grosso, whose lower species richness is very likely an artefact. In fact, recent data confirms that this region has higher number of species than predicted by our models (Dalponte et al., 2016; Miranda et al., 2015). Such observation reinforces the need for inventories in the region, as well as the *in situ* validation of any SDM.

However, despite some limitations, SDMs still provide valuable information, helping to identify conservation priority areas - like the remaining Atlantic Forest fragments in our analysis - and highlighting priority areas for surveys - like in the northern parts of the Atlantic Forest, the Caatinga´s eastern most, the Pampa´s northeastern, Pantanal´s northern, and southern Pará, in Amazonia. Moreover, SDMs can produce more accurate scenarios for potential species richness, which is useful for conservation planning and management in subsampled areas or areas without formal record of bat species, like Brazil.

Improving the knowledge on bat distribution with new records is important, and recent studies in never-sampled forest remnants have resulted in the description of new species (Fazzolari-Correa, 1994), new records for Brazil or biomes (Bordignon, 2006; Camargo and Fischer, 2005; Rocha, et al., 2016, 2017) and in the extension of known distribution of several species of bats in Brazil (Lira, et al., 2009; Longo, et al., 2007, Rocha, et al., 2014). However, simply recording bats is not enough to conserve them. Several records can become obsolete due to the temporal bias derived from data aging and as consequence of severe land use changes, like the ones experienced in Brazil. For example, the country has one of the highest rates of habitat loss in the world, and large tracts of pristine habitats are quickly converted to agribusiness-dominated landscapes (FAO, 2015). In such a scenario, old biodiversity records must be evaluated and continuously monitored. However, currently there are limited or nonexistent bat monitoring programs taking place in Brazil (e.g. Aguiar, et al., 2006).

The data here presented refined the known distribution of several species, providing a more accurate approximation of the real distribution of threatened and restricted species. However, a degree of uncertainty still exists, and some of the proposed ranges may suffer from some commission or omission errors due to the quality of the species occurrence data, incomplete distributional data and recent land use changes (Margules, et al., 2002; Rondinini, et al., 2006). The refinement and correction of species distributional data is an ongoing process and this is not different for bats. The maps we produced must be constantly updated as new data are emerging (Aguiar, et al., 2015), so decisions for conservation management and public policies will be more successful.

A limited knowledge on the distribution and dispersion capabilities of bat species, and how changes in the land use and habitat fragmentation can affect different species still persists in Brazil. Therefore, inventories and long-term monitoring in most subsampled areas, small and isolated remnants of native vegetation and in important areas for endangered and endemic species should be done in future studies. Moreover, it is necessary to access the obsolescence of Brazilian databases´ records in face of land use changes, and the magnitude of pressures that threaten the conservation of bats and their habitats in Brazil. Finally, in a business as usual scenario, which combined relaxing the environmental laws, high and fast habitat loss (Soares-Filho, et al., 2014), decrease of the protection to Brazilian caves (Brasil, 2008), the reduction of protected areas (Bernard, et al., 2014) and low *in situ* protection, the current conservation status and real distribution of several Brazilian bats can be more critical than described by the official data (MMA, 2016) and potential distributions presented here. So efforts to improve the knowledge of the impacts caused by those events and strengthen conservation policies are necessary not only for bats but also whole Brazilian biodiversity conservation.

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TABLES

Table 1.- Number of bat records and number of bat species registred in each biomes of Brazil until 2016

Biome	N of records	% of records	N spp.	% spp.
Atlantic Forest	4542	47,5	121	67,2
Amazonia	2192	22,9	139	77,2
Cerrado	1673	17,5	120	66,7
Caatinga	817	8,5	98	54,4
Pantanal	255	2,7	56	31,1
Pampa	77	0,8	33	18,3
Total Geral	9556			

Table 2.- Number of bat records and number of bat species registred in each biomes of Brazil until 2016

	<1950	1950-1999	>2000
Amazonia	157	706	680
Caatinga	34	201	395
Cerrado	106	285	704
Atlantic Forest	346	1207	1606
Pampa	2	8	14
Pantanal	15	37	156
Total	**660**	**2444**	**3555**
Percentage	**6,9**	**25,6**	**37,2**

FIGURES

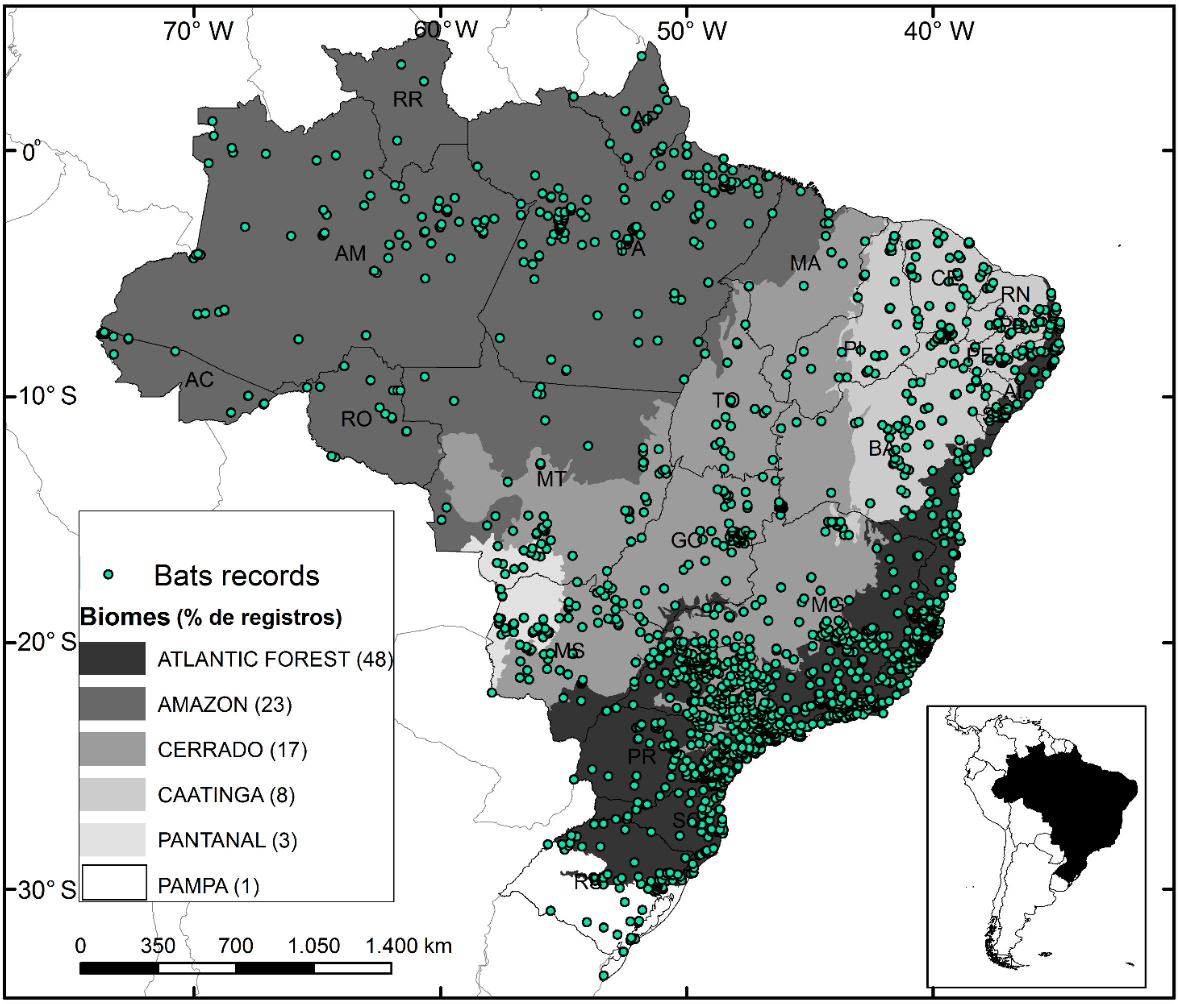

Figure 1.- Brazilian terrestrial biomes and records of bat species in the country. Biomes colors were expressed according to a bat species richness gradient, with darker colors representing the species richest ones.

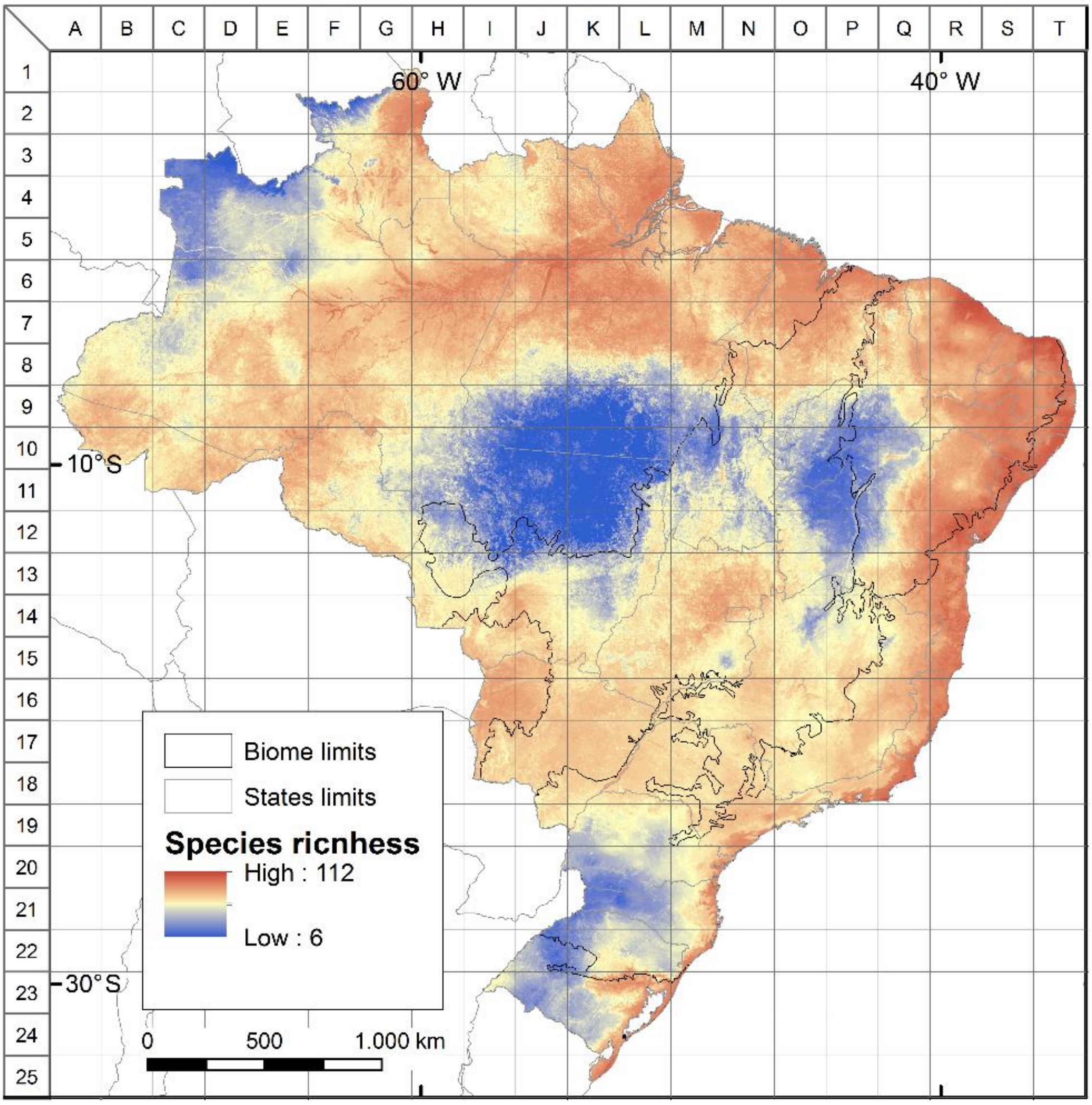

Figure 2.- Bats richness patterns for Brazil obtained through species distribution models based on data for 132 species.

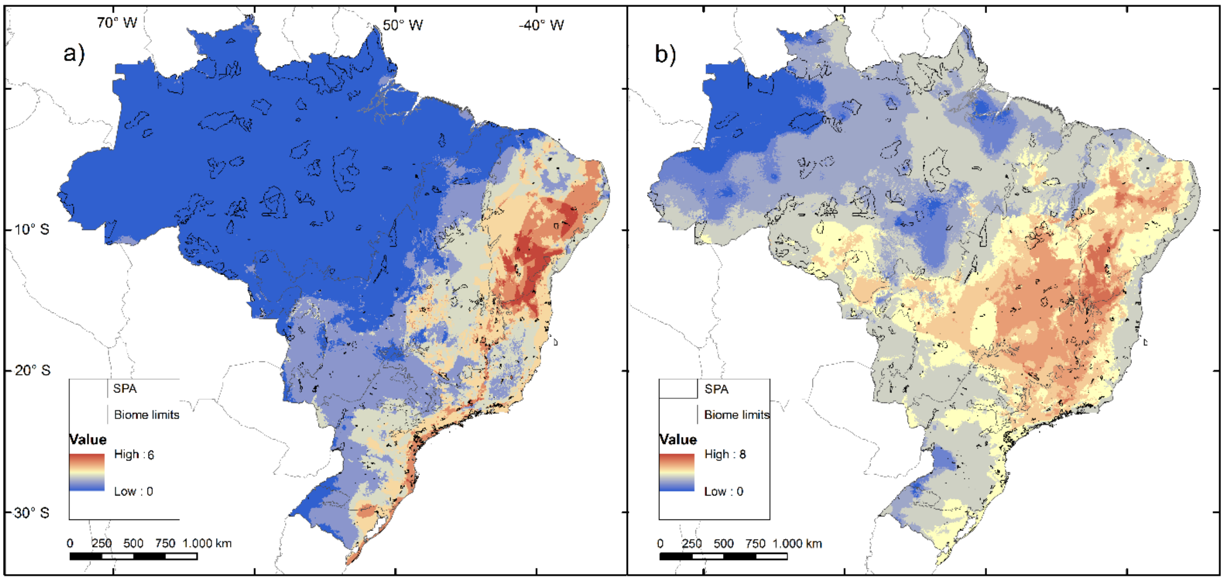

Figure 3.- Potential distribution of endemic (a) and endangered (b) bat species of Brazil. SPA = Strict Protected Areas.

SUPPLEMENTARY MATERIAL

Supplementary Material 1: Bioclimatic variables used to generate species distribution models developed within the Brazil with Maxent

Supplementary Material 2: Number of variables and replicates used to run the distribution models of Brazilian bats depending on the number of localities.

Supplementary Material 3: Modelling prediction results for modeled species. Number of records, AUC Training (AUC Tr), AUC Test, endemic, threatened and biome restricted species are shown. ([*] Indicates species that did not have significant models or have low AUC. For species endangered or endemic with least of 6 points were done MCP= minimum convex polygon)

Supplementary Material 1.- Bioclimatic variables used to generate species distribution models developed within the Brazil with Maxent

BIO1: Temperatura Média Anual

BIO2: Temperatura Média Diurna

BIO3: Isotermalidade

BIO4: Sazonalidade da Temperatura

BIO5: Temperatura Máxima no Mês Mais Quente

BIO6: Temperatura Mínima no Mês Mais Frio

BIO7: Variação Anual da Temperatura

BIO8: Temperatura Média no Trimestre Mais Úmido

BIO9: Temperatura Média no Trimestre Mais Seco

BIO10: Temperatura Média no Trimestre Mais Quente

BIO11: Temperatura Média no Trimestre Mais Frio

BIO12: Precipitação Anual

BIO13: Precipitação no Mês Mais Úmido

BIO14: Precipitação no Mês Mais Seco

BIO15: Sazonalidade da Precipitação

BIO16: Precipitação no Trimestre Mais Úmido

BIO17: Precipitação no Trimestre Mais Seco

BIO18: Precipitação no Trimestre Mais Quente

BIO19: Precipitação no Trimestre Mais Frio

Supplementary Material 2.- Relationship between the number of locations and the number of variables and replicates used to run the species distribution models in Maxent.

N Localities	N Variables	N Replicates
6 - 9	3	jackknife (n-1)
10 - 15	5	9
16 - 20	8	15
≥ 21	12	20

Supplementary Material 2.- Modelling prediction results for modeled species. Number of records, AUC Training (AUC Tr), AUC Test, endemic, threatened and biome restricted species are shown. ([*] Indicates species that did not have significant models or have low AUC. For species endangered or endemic with least of 6 points were done MCP= minimum convex polygon)

Specie	Distribution	N record	AUC Tr	AUC test	Endemic	Threatened (UICN/Brasil)	Biome restricted
Ametrida centurio Gray, 1847	SDM	24	0,85	0,96			
Anoura caudifer (É. Geoffroy, 1818)	SDM	292	0,94	0,93			
Anoura geoffroyi Gray, 1838	SDM	137	0,93	0,92			
Artibeus concolor Peters, 1865	SDM	39	0,9	0,89			
Artibeus fimbriatus Gray, 1838	SDM	115	0,96	0,94			
Artibeus lituratus (Olfers, 1818)	SDM	401	0,89	0,83			
Artibeus obscurus (Schinz, 1821)	SDM	158	0,89	0,84			
Artibeus planirostris (Spix, 1823)	SDM	200	0,87	0,82			
Carollia benkeithi Solari & Baker, 2006	SDM	22	0,78	0,68			
Carollia brevicauda (Schinz, 1821)	SDM	62	0,86	0,76			
Carollia perspicillata (Linnaeus, 1758)	SDM	557	0,88	0,84			
Centronycteris maximiliani (Fischer, 1829)	SDM	37	0,95	0,92			
Chiroderma doriae Thomas, 1891	SDM	56	0,97	0,91			
Chiroderma trinitatum Goodwin, 1958	SDM	18	0,91	0,76			x
Chiroderma villosum Peters, 1860	SDM	55	0,87	0,77			
Chiroderma vizottoi Taddei & Lim, 2010	MPC	4			x		x
Choeroniscus minor (Peters, 1868)	SDM	34	0,85	0,82			
Chrotopterus auritus (Peters, 1856)	SDM	167	0,91	0,88			
Cormura brevirostris (Wagner, 1843)	SDM	18	0,9	0,95			x
Cynomops abrasus (Temminck, 1826)	SDM	28	0,88	0,84			
Cynomops greenhalli Goodwin, 1958	-	1					
Cynomops milleri (Osgood, 1914)	-	1					x

Cynomops paranus (Thomas, 1901)	-	5					
Cynomops planirostris (Peters, 1866)	SDM	36	0,91	0,8			
Cyttarops alecto Thomas, 1913	-	2					
Dermanura anderseni (Osgood, 1916)	SDM	35	0,82	0,82			
Dermanura bogotensis (Andersen, 1906)	-	4					
Dermanura cinerea Gervais, 1856	SDM	105	0,89	0,9			
Dermanura gnoma (Handley, 1987)	SDM	28	0,9	0,87			
Desmodus rotundus (É. Geoffroy, 1810)	SDM	452	0,9	0,89			
Diaemus youngii (Jentink, 1893)	SDM	53	0,89	0,79			
Diclidurus albus Wied-Neuwied, 1820	SDM	9	0,79	0,77			
Diclidurus ingens Hernández-Camacho, 1955	-	2					x
Diclidurus isabella (Thomas, 1920)	-	2					x
Diclidurus scutatus Peters, 1869	SDM	6	0,9	0,85			x
Diphylla ecaudata Spix, 1823	SDM	110	0,91	0,88			
Dryadonycteris capixaba Nogueira, Lima, Peracchi & Simmons, 2012	SDM	7	0,97	0,97	x		x
Eptesicus andinus J.A. Allen, 1914	-	1					
Eptesicus brasiliensis (Desmarest, 1819)	SDM	87	0,91	0,83			
Eptesicus chiriquinus Thomas, 1920	-	4					
Eptesicus diminutus Osgood, 1915	SDM	48	0,94	0,93			
Eptesicus furinalis (d'Orbigny & Gervais, 1847)	SDM	52	0,88	0,76			
Eptesicus taddeii Miranda, Bernardi & Passos, 2006	SDM	27	0,96	0,95	x	x	x
Eumops auripendulus (Shaw, 1800)	SDM	53	0,89	0,88			
Eumops bonariensis (Peters, 1874)	SDM*	12	0,55	0,57			
Eumops delticus Thomas, 1923	-	5					
Eumops glaucinus (Wagner, 1843)	SDM	26	0,92	0,83			
Eumops hansae Sanborn, 1932	SDM	9	0,91	0,8			
Eumops maurus (Thomas, 1901)	-	4					
Eumops patagonicus Thomas, 1924	-	3					x
Eumops perotis (Schinz, 1821)	SDM	26	0,89	0,85			
Eumops trumbulli (Thomas, 1901)	-	3					x
Furipterus horrens (Cuvier, 1828)	SDM	85	0,84	0,82		x	
Glossophaga commissarisi Gardner, 1962	-	1					x
Glossophaga longirostris Miller, 1898	-	0					
Glossophaga soricina (Pallas, 1766)	SDM	481	0,86	0,85			
Glyphonycteris behnii (Peters, 1865)	SDM	6	0,83	0,83		x	x
Glyphonycteris daviesi (Hill, 1964)	SDM	9	0,82	0,79			
Glyphonycteris sylvestris Thomas, 1896	SDM	15	0,79	0,77			
Histiotus alienus Thomas, 1916	MPC	1					x
Histiotus diaphanopterus Feijó, Rocha & Althoff, 2015	SDM	11	0,93	0,96			x
Histiotus laephotis Thomas, 1916	MPC	1					x
Histiotus montanus (Philippi & Landbeck, 1861)	SDM	8	0,99	0,99			x
Histiotus velatus (I. Geoffroy, 1824)	SDM	97	0,98	0,97			
Hsunycteris thomasi (J.A. Allen, 1904)	SDM	39	0,9	0,89			

Lampronycteris brachyotis (Dobson, 1879)	SDM	28	0,9	0,89			
Lasiurus blossevillii ([Lesson, 1826])	SDM	61	0,94	0,91			
Lasiurus castaneus Handley, 1960	-	1					x
Lasiurus cinereus (Palisot de Beauvois, 1796)	SDM	19	0,98	0,95			
Lasiurus ebenus Fazzolari-Corrêa, 1994	MPC	1			x		x
Lasiurus ega (Gervais, 1856)	SDM	91	0,9	0,85			
Lasiurus egregius (Peters, 1870)	-	2					
Lasiurus salinae Thomas, 1902 40	-	1					x
Lichonycteris degener Miller, 1931	SDM	17	0,88	0,84			
Lionycteris spurrelli Thomas, 1913	SDM	26	0,84	0,79			
Lonchophylla bokermanni Sazima, Vizotto & Taddei, 1978	SDM	7	0,93	0,93	x	x	x
Lonchophylla dekeyseri Taddei, Vizotto & Sazima, 1983	SDM	19	0,97	0,95		x	x
Lonchophylla inexpectata Moratelli & Dias, 2015	MPC	3			x		x
Lonchophylla mordax Thomas, 1903	-	3					
Lonchophylla peracchii Dias, Esbérard & Moratelli, 2013	SDM	17	0,96	0,96	x		x
Lonchorhina aurita Tomes, 1863	SDM	73	0,92	0,91		x	
Lonchorhina inusitata Handley & Ochoa, 1997	MPC	1					x
Lophostoma brasiliense Peters, 1866	SDM	48	0,86	0,78			
Lophostoma carrikeri (J. A. Allen, 1910)	SDM*	16	0,7	0,48			
Lophostoma schulzi (Genoways & Williams, 1980)	MPC	4					x
Lophostoma silvicola d'Orbigny, 1836	SDM	71	0,83	0,72			
Macrophyllum macrophyllum (Schinz, 1821)	SDM*	38	0,88	0,78			
Mesophylla macconnelli Thomas, 1901	SDM	22	0,84	0,78			
Micronycteris brosseti Simmons & Voss, 1998	MPC	3					x
Micronycteris hirsuta (Peters, 1869)	SDM	19	0,87	0,9			
Micronycteris homezorum Pirlot, 1967	-	3					x
Micronycteris megalotis (Gray, 1842)	SDM	138	0,9	0,84			
Micronycteris microtis Miller, 1898	SDM*	16	0,5	0,5			
Micronycteris minuta (Gervais, 1856)	SDM	68	0,87	0,84			
Micronycteris sanborni Simmons, 1996	SDM	13	0,77	0,68	x		
Micronycteris schmidtorum Sanborn, 1935	SDM*	18	0,58	0,39			
Mimon bennettii (Gray, 1838)	SDM	85	0,92	0,92			
Mimon crenulatum (É. Geoffroy, 1803)	SDM	44	0,87	0,9			
Molossops neglectus Williams & Genoways, 1980	SDM	9	0,85	0,84			
Molossops temminckii (Burmeister, 1854)	SDM	57					
Molossus aztecus Saussure, 1860	-	2					x
Molossus coibensis J.A. Allen, 1904 33	-	2					x
Molossus currentium Thomas, 1901 34	-	2					
Molossus molossus (Pallas, 1766)	SDM	194	0,89	0,85			
Molossus pretiosus Miller, 1902	-	4					
Molossus rufus É. Geoffroy, 1805	SDM	135	0,91	0,88			
Myotis albescens (É. Geoffroy, 1806)	SDM	56	0,87	0,83			

Myotis dinellii Thomas, 1902 42	-	0					
Myotis izecksohni Moratelli, Peracchi, Dias & Oliveira, 2011	SDM	7	0,99	0,98	x		x
Myotis lavali Moratelli, Peracchi, Dias & Oliveira, 2011	SDM	37	0,94	0,93			
Myotis levis (I. Geoffroy, 1824)	SDM	38	0,97	0,96			x
Myotis nigricans (Schinz, 1821)	SDM	272	0,9	0,9			
Myotis riparius Handley, 1960	SDM	55	0,87	0,75			
Myotis ruber (É. Geoffroy, 1806)	SDM	52	0,96	0,93			
Myotis simus Thomas, 1901 43	SDM	17	0,84	0,85			
Natalus macrourus (Gervais, 1856) 28	SDM	72	0,91	0,87		x	
Neonycteris pusilla (Sanborn, 1949)	MPC	1					x
Neoplatymops mattogrossensis (Vieira, 1942)	SDM	12	0,94	0,87			
Noctilio albiventris Desmarest, 1818	SDM	103	0,87	0,85			
Noctilio leporinus (Linnaeus, 1758)	SDM	128	0,91	0,87			
Nyctinomops aurispinosus (Peale, 1848)	SDM*	11	0,79	0,6			
Nyctinomops laticaudatus (É. Geoffroy, 1805)	SDM	55	0,87	0,8			
Nyctinomops macrotis (Gray, 1840) 35	SDM	35	0,92	0,86			
Peropteryx kappleri Peters, 1867	SDM	47	0,88	0,93			
Peropteryx leucoptera Peters, 1867	SDM	9	0,82	0,75			x
Peropteryx macrotis (Wagner, 1843)	SDM	151	0,9	0,87			
Peropteryx pallidoptera Lim, Engstrom, Reid, Simmons, Voss & Fleck, 2010	-	1					x
Peropteryx trinitatis Miller, 1899	-	3					
Phylloderma stenops (Peters, 1865)	SDM	33	0,84	0,97			
Phyllostomus discolor (Wagner, 1843)	SDM	126	0,87	0,83			
Phyllostomus elongatus (É. Geoffroy, 1810)	SDM	60	0,85	0,8			
Phyllostomus hastatus (Pallas, 1767)	SDM	221	0,88	0,85			
Phyllostomus latifolius (Thomas, 1901)	SDM	8	0,93	0,9			x
Platyrrhinus angustirostris Velazco, Gardner & Patterson, 2010	-	0					
Platyrrhinus aurarius (Handley & Ferris, 1972)	-	0					
Platyrrhinus brachycephalus (Rouk & Carter, 1972)	SDM	15	0,88	0,88			
Platyrrhinus fusciventris Velazco, Gardner & Patterson, 2010	-	3					x
Platyrrhinus incarum (Thomas, 1912)	SDM	52	0,87	0,86			
Platyrrhinus infuscus (Peters, 1880)	-	5					
Platyrrhinus lineatus (É. Geoffroy, 1810)	SDM	273	0,93	0,91			
Platyrrhinus recifinus (Thomas, 1901)	SDM	52	0,96	0,94	x		
Promops centralis Thomas, 1915	SDM*	13	0,76	0,86			
Promops nasutus (Spix, 1823)	SDM	21	0,89	0,91			
Pteronotus gymnonotus (Wagner, 1843)	SDM	21	0,9	0,92			
Pteronotus parnellii (Gray, 1843)	SDM	49	0,79	0,75			
Pteronotus personatus (Wagner, 1843)	SDM	14	0,81	0,98			
Pygoderma bilabiatum (Wagner, 1843)	SDM	92	0,97	0,97			
Rhinophylla fischerae Carter, 1966	SDM	18	0,91	0,86			x
Rhinophylla pumilio Peters, 1865	SDM	78	0,89	0,89			

Rhogeessa hussoni Genoways & Baker, 1996	-	4					
Rhogeessa io Thomas, 1903	-	2					
Rhynchonycteris naso (Wied-Neuwied, 1820)	SDM	116	0,83	0,78			
Saccopteryx bilineata (Temminck, 1838)	SDM	53	0,89	0,75			
Saccopteryx canescens Thomas, 1901	SDM	23	0,93	0,95			
Saccopteryx gymnura Thomas, 1901	SDM	6	0,9	0,85			
Saccopteryx leptura (Schreber, 1774)	SDM	59	0,85	0,8			
Scleronycteris ega Thomas, 1912	MPC	2					x
Sphaeronycteris toxophyllum Peters, 1882	SDM*	7	0,54	0,5			x
Sturnira lilium (É. Geoffroy, 1810)	SDM	305	0,91	0,86			
Sturnira magna de la Torre, 1966	-	2					x
Sturnira tildae de la Torre, 1959	SDM	54	0,88	0,8			
Tadarida brasiliensis (I. Geoffroy, 1824)	SDM	45	0,93	0,95			
Thyroptera devivoi Gregorin, Gonçalves, Lim & Engstrom, 2006	MPC	3					x
Thyroptera discifera (Lichtenstein & Peters, 1855)	SDM	9	0,82	0,82			
Thyroptera lavali Pine, 1993	-	2					x
Thyroptera tricolor Spix, 1823	SDM	38	0,9	0,93			
Thyroptera wynneae Velazco, Gregorin, Voss & Simmons, 2014	-	1					x
Tonatia bidens (Spix, 1823)	SDM*	53	0,85	0,84			
Tonatia saurophila Koopman & Williams, 1951	SDM	27	0,89	0,78			
Trachops cirrhosus (Spix, 1823)	SDM	137	0,89	0,89			
Trinycteris nicefori (Sanborn, 1949) 20	SDM	40	0,94	0,92			
Uroderma bilobatum Peters, 1866	SDM	77	0,83	0,82			
Uroderma magnirostrum Davis, 1968	SDM	36	0,9	0,98			
Vampyressa pusilla (Wagner, 1843)	SDM	59	0,91	0,83			
Vampyressa thyone Thomas, 1909	-	3					
Vampyriscus bidens (Dobson, 1878)	SDM	24	0,95	0,92			x
Vampyriscus brocki (Peterson, 1968)	SDM	8	0,95	0,95			x
Vampyrodes caraccioli (Thomas, 1889)	SDM	16	0,82	0,9			
Vampyrum spectrum (Linnaeus, 1758)	SDM	62	0,8	0,74			
Xeronycteris vieirai Gregorin & Ditchfield, 2005	SDM	12	0,97	0,97	x	x	x

4 PRESSURES AND THREATS TO BAT CONSERVATION IN BRAZIL

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ABSTRACT

With more than 180 species, Brazil has the second highest bat species richness in the world, harboring 11 endemic and seven nationally threatened species. However, less than 10% of the country has been minimally surveyed, and for nearly 60% of its territory there is not a single record of bat species raising concerns about their conservation. Thus, an effective bat conservation strategy for Brazil will require basic information on where the species are, the habitats they use and the pressures and threats they experience. Here, we analyzed the distribution and magnitude of different pressures and threats to bats (mining, deforestation and wind power generation) on strictly protected areas (SPA), caves and on the remaining natural vegetation in Brazil. We found that 39% of SPA are currently under the effects of mining and wind farms, and 70% of the SPA would be affected if all potential mining projects were active. More than 50% of the potential roosting caves are under pressure by mining, deforestation of their surroundings, wind farms and/or cattle farming, and more than 80% of such caves would be threatened if all possible mining projects were executed. The Caatinga and the Atlantic Forest biomes are the most affected by mining and deforestation. The conservation scenario for the Brazilian biodiversity – and for bats in particular – may deteriorate due to an increase in the deforestation rate as a result of the new Forest Code, or due to the recent weakening of the national environmental legislation. We suggest the necessity to re-evaluate the conservation status of some species with restricted distributions. We also urge for an improvement of the environmental regulation of mining, agribusiness, and the wind energy sectors in Brazil to guarantee the protection of bats and the ecosystem services they provide.

Key words: Brazilian biomes, Caatinga, Chiroptera, habitat remnants, mining, wind farm.

4.1.- INTRODUCTION

Brazil experiences strong environmental pressures from important economic sectors, such as agribusiness, mining, and the generation and transmission of energy (Bacha 2004, Gandra 2014). Brazil´s beef production for domestic and foreign markets (ABIEC, 2011), together with the country´s energy production capacity (ANEEL, 2016), as well as sugar cane production for ethanol (Silva, et al., 2009), and the planted area with soybean (Hirakuri and Lazzarotto, 2011) and corn (Landau, et al., 2012; Miranda, et al., 2014) have all increased in recent years. Not coincidently, these economic sectors account for almost all of the high rates of habitat loss and degradation experienced in the country (Bacha 2004; Garret 2017; Watanabe and Moraes 2017). All these human activities results in higher energy and raw material demand corresponding to increasing number and range of hydroelectric, wind farms, pasture areas, the opening of roads, enhanced exploitation of natural resources, and pollution. Enterprises resulting from bad, deregulated and precarious impact assessments, installation and operation practices, will produce further deforestation, degradation, and extinction of habitats and threats to biodiversity and natural ecosystems.

The majority of Neotropical vertebrates depend on remnants of native vegetation in good conditions to persist in human-dominated landscapes (Ahumada, et al., 2011; Negrões, et al., 2011). However, due to the ability of a few species to live in urban or agricultural environments, there is a false general belief that some groups, such as bats, are not affected by human activities, and neither are in need of conservation actions (Voigt and Kingston, 2016).With more than 180 species of bats (Feijó, et al., 2015; Moratelli and Dias, 2015; Nogueira, et al., 2014) Brazil has approximately 14% of bat species of the world (Fenton and Simmons, 2014), and ranks second among the countries with the highest bat species richness (Bernard, et al., 2011). Eleven species are considered endemic (Nogueira, et al., 2014), but less than 10% of Brazil´s territory was minimally surveyed, and for nearly 60% of the country there is not a single record of bat species (Bernard, et al., 2011). This situation raises concerns considering that both the effective conservation of any species and the mitigation of negative scenarios requires basic information on where the species are and what are the pressures and threats that they experience (Bernard, et al., 2012; Margules and Pressey, 2000).

A knowledge gap and the consequent lack of mitigations on the pressures and threats affecting a given species can affect the long term viability of some bat populations and consequently compromise all the ecosystem services they provide. Among those services are the control of insects identified as crop pests (Boyles, et al., 2011; Whitaker, 1995), and seed dispersal and pollination of several Neotropical plants of commercial interest (Allen-Wardell, et al., 1998; Gorchov, et al., 1995; Lobova, et al., 2009; Smith, et al., 2004). Additionally, bats are predators and prey and explore a large number and variety of environments actively participating in the maintenance and dynamics of several ecosystems (Fenton and Simmons, 2014; Kalko, et al., 1996). Such ecological attributes make several species sensible to habitat loss and degradation (Klingbeil and Willig, 2009; Meyer, et al., 2008), allowing some of those species to be used as indicators of the conservation status of some ecosystems (Bevilacqua and

Ochoa, 2000; Castro-Arellano, et al., 2007; Cunto and Bernard, 2012; Fenton, et al., 1992; Medellín, et al., 2000; Wilson, et al., 1996).

Among the anthropogenic pressures and threats to the conservation of bats, mining activities are known to cause the destruction of caves, which are important shelters for several bat species (Furey and Racey 2016). Power generation, transport and agribusiness also threat bat species due to deforestation and habitat fragmentation, with consequent loss of habitat, shelter and food sources to bats (Bernard, et al., 2012). Moreover, activities related with pest and rabies control are frequently carried out by unprepared personal which indiscriminately exterminates not only the Common Vampire bat (*Desmodus rotundus*), but many other species, including threatened ones like the dekeyser´s Nectar bat *Lonchophylla dekeyseri* (Aguiar, et al., 2010; Bernard, et al., 2012; O'Shea, et al., 2016). Wind farms are a source of mortality for bats due to the collision between bats and towers and blades, or due to barotrauma (Baerwald, et al., 2008; Rodríguez-Durán and Feliciano-Robles, 2015). Wind energy production is the main cause of multiple mortality events for bats in Europe and remains little evaluated in South America (Barros, et al., 2015; O'Shea, et al., 2016). The combination of such pressures and threats make several species of bats endangered worldwide (Voigt and Kingston, 2016). This situation is no different for Brazil, and currently seven species are considered as nationally threatened (MMA 2016a).

Any effective bat conservation strategy in such a large, biodiversity-rich and environmentally heterogeneous country like Brazil requires an assessment of the distribution and magnitude of pressures and threats affecting the species that occurs in the country (Bernard, et al., 2012). Therefore, here we present an analisys on the distribution and magnitude of pressures and threats from mining, deforestation and wind power generation on bats by focusing on the impact of such drivers on protected areas, caves and on the remaining natural vegetation in Brazil. By addressing the implications of such major drivers on the Brazilian bat fauna we provide updated information for the decision-making process focused on the conservation of a species-rich, ecologically important and frequently neglected group of animals in one of the largest and most biodiversity-rich countries of the world.

4.2.- MATERIALS AND METHODS

4.2.1 Study area:

This study considered the entire terrestrial Brazilian territory, covering an area of 8,515,767.049 km^2 (~47.6% of the South America continent) (IBGE 2012), and its six major terrestrial biomes: Amazonia, Caatinga, Cerrado, Atlantic Forest, Pantanal and Pampa (Fig. 1).

4.2.2 Pressures, and threats on strictly protected areas

Protected areas (PA) are the most important *in situ* conservation strategy for the protection of bats and their habitats. PAs in Brazil are known as *unidades de conservacão* (conservation units - CU), being classified as sustainable use areas (equivalent to IUCN´s categories V and VI), and strictly protected areas (hereafter SPA; IUCN´s categories I to IV) (Brasil 2000). First, we evaluated the presence of endemic and threatened bat species in SPA. We used the official Brazilian SPA map (data for 2011 – MMA, 2016b) in vector format, and overlapped the polygons of SPA with three other datasets: (1) the general current mining areas affecting limestones, ferriferous formations (known as *cangas*), sandstones, quartzites, gneisses, marbles, basalts, rhyolites and metapelitics, formations which support all Brazilian caves (CECAV, 2013; DNPM 2015); (2) wind farms location (ANEEL, 2016); and (3) the deforested area for all the Brazilian biomes (data for 2009 – SISCOM, 2015), all in vector format. We assumed that these datasets represented the main pressures affecting bat conservation in Brazil (see Bernard et al. 2012). To assess threats (scenario of future pressures) in SPA we overlapped their polygons with the potential mining activities and cave distribution (see next topic). Here we considered potential mining as all projects still under research, authorization or under viability analysis (DNPM, 2015). For each SPA we evaluated the presence and magnitude of pressures and threats inside their polygon and at buffers of 5 and 10 km from their limits.

4.2.3 Pressures and threats on caves

Because of the importance of caves as roosts for many species of bats (Delgado-Jaramillo et al. 2017; Guimarães and Ferreira, 2014; Tejedor, et al., 2004; Trajano, 1995), we carried out an evaluation of the pressures and threats affecting Brazillian caves. We considered the most updated cave database available (CECAV, 2017) and a map of potential occurrence of caves for the entire country (data from Jansen, et al., 2012). In that map, the Brazilian territory is classified according to the potential cave occurrence as areas with very high, high, medium and low potential (Jansen, et al., 2012). To estimate the cave conservation relevance, we overlapped the potential distribution map for each endemic and threatened species and their presence in caves.

As pressures we assessed the deforestation in and around caves, counting the number of caves and the extension of deforestation within areas with high and very high cave potential. Additionally, we calculated the presence and extent of active mining and wind farms within 5 and 10 km to the caves, and in the areas of high and very high cave potential. Similarly, for threats, we calculated the presence and extent of potential mining activity within the same 5 and 10 km radius.

Another variable considered was the proximity between caves and cattle ranching areas: the relation of the density of cattle/km^2 of pasture was used as a vector of risk for intentional killing of “vampire bat control” (See Bernard et al. 2012). The bovine density was classified as very high (> 100 cattle/km^2), high (50 – 100/km^2), medium (10 – 49/km^2) or low (<10/km^2). That information was provided by Ricardo A. Dias (pers. comm.) based on data from the

Ministry of Agriculture, Livestock and Supply with the help of the Department of Preventive Veterinary Medicine and Animal Health, School of Veterinary Medicine, University of São Paulo, Brazil. Finally, considering that bat populations may be threatened by indiscriminate and deregulated visitation and vandalism, we also evaluated the proximity of caves to human population centers, adopting 2.5 km and 5 km buffers around of the localities (available at http://mapas.mma.gov.br/i3geo/datadownload.htm#).

4.2.4 Pressures and threats on the native vegetation

Most bat species depend on natural environments for shelter and/or foraging, and may experience negative consequences due to the loss and degradation of such habitats (Ahumada, et al., 2011; Negrões, et al., 2011). Therefore, in our analysis we also considered the integrity, pressures and threats over the remaining native vegetation in Brazil. Although official data for deforestation in the country dates from 2009 (SISCOM, 2015), that dataset was used simply because it was the best available data for the entire country. The map with the remaining native vegetation for Brazil was then overlaid with maps of mining and wind farm activity. We adopted 5 and 10 km buffers around mining and wind farms to quantify the area under direct influence for those activities.

The expansion of agribusiness frontier is identified as a major driver for vegetation loss in Brazil (Kissinger, et al., 2012; Manzatto, et al., 2009; Sparovek, et al., 2009). In this sense, the relation of cattle density/km^2 of pasture was used as a vector of risk for future deforestation; the percentage of native vegetation areas was then evaluated in different categories of bovine density, adopting the same interval used for caves.

4.2.5 Threats and conservation scenarios for the species

We produced two ~10 x 10 km resolution maps: one for pressures and another for threats affecting the conservation of Brazilian bats. For each pressure and threat a value was assigned according to their negative impact or importance. The pressure map (Fig. 2a) was based on the sum of mining activities, wind farms, and percentage of deforestation (see Supplementary Material 1 for detailed information). The threats map (Fig. 2b), was produced by overlapping the map of pressures with potential mining, and the bovine density of the municipality (see Supplementary Material 1). The values obtained for each cell were then classified as “low pressures” (score 1–3), “medium pressures” (score 4–6), “high pressures” (score 7–9) and very high pressures” (scores ≥ 10).

Based on a database with 9,550 records produced by the Laboratory of Bat Biology and Conservation at the University of Brasilia and using the software Maxent 3.3.3 (Phillips, et al., 2006; Phillips and Dudík, 2008), we generated species distribution models for 132 bat species (see chapter 1 of this thesis). We generated different distribution models for each species based on a set of 19 bioclimatic variables from WorldClim version 1.4 (Hijmans, et al., 2005; available at http://www.worldclim.org/) plus elevation, and slope all at a 5 km resolution. We also considered the Normalized Difference Vegetation Index (NDVI) as a proxy for measuring vegetation cover (http://glcf.umd.edu/data/ndvi/). The continue suitability maps were converted into binary presence-absence values using the ‘lowest presence threshold’ (LPT) (Pearson, et al., 2007). We used the minimum convex polygon for species with < 6 records.

The 132 individual binary distribution maps were overlapped to generate a single species richness map for the entire country. The same approach was used for the seven threatened species considered. These two maps were then overlapped with (1) the map of pressures and threats, (2) the map with deforested areas in Brazil, (3) the map with wind farms, (4) mining, and (5) boundaries of SPA. Such approach allowed us to overlap species richnes, threatened species and the amount of habitat lost, as well as the percentage of distribution within PA for each species. We also analyzed how the species-richest areas were affected by the different pressures. All spatial analyses were performed using ArcGis 10.2.2. (ESRI, 2015).

4.3.- RESULTS

4.3.1.1 Pressures on protected areas

Of the 565 SPA in Brazil, 545 potentially presented at least one threatened bat species and 450 potentially presented at least one brazilian endemic bat species. Two SPA have wind farms within their boundaries (the Refúgio de Vida Silvestre dos Campos de Palmas, and the Parque Estadual do Morro do Chapéu). Eight are < 5 km from wind farms and three of them (the Monumento Natural das Falésias de Beberibe, the Refúgio de Vida Silvestre dos Campos de Palmas, and the Parque Nacional de Jericoacoara) have > 40% of their extension in areas < 5 km far from wind farms (72, 62 and, 44% respectively) (Supplementary Material 2 - Fig 1). Twelve SPA are < 10 km from wind farms, and eight of them have > 45% of their extension in areas < 10 km far from wind farms.

Thirty-two of the 565 SPA currently have mining activity within their boundaries (SM 2- Fig 2) and five of them (Parque Estadual das Sete Passagens, Parque Natural Municial do Canção, Estação Ecológica de Fechos, Estação Ecológica Corumbá and Parque Natural Municial of Piraputangas) have > 40% of their areas under current mining (Fig. 2). More than 300 (55%) SPA are < 5 km from mining operations:134 have cave mining; and 51 have > 80% of their extension within < 5 km away from mining. Approximately 421 SPA are < 10 km away from mining: 220 have cave mining and 134 contain > 80% of their extension < 10 km away from mining activities (Fig. 2). Atlantic forest is the biome with the most SPA affected by mining

The average area deforested within the SPA is 31%, however, 79 SPA had > 80% of their extension deforested; 41 had no natural remaining vegetation (Fig 3). On the other hand, only 51 (9%) had no deforestation. Among the SPA with > 50% deforested, 45% are in the Atlantic forest, 22% in Cerrado and 18% in Amazonia. Most of those areas are under state jurisdiction (59%) and 36% under municipal jurisdiction.

4.3.1.2 Threats in protected areas

Had all potential cave mining projects be active they would directly affect 155 SPA (SM 2-Fig 2). Of these, 16 would have > 80% of their extension under mining. Approximately 320 SPA would have mining activities at a distance < 5 km, while 87 SPA would have > 80% of their area under mining influence. Aproximately 390 SPA would be <10 Km away from mining and 114 would have > 80% of their area under mining influence.

4.3.2.1 Pressures on caves

Currently, 16,382 caves have been recorded in Brazil. 12,789 of them are within the potential distribution of at least one threatened bat species, while 16,299 are within the potential distribution of an endemic bat species. Alarmingly, 45% of the caves are located in deforested areas (SM 2-Fig 3), and 261 are < 5 km away from wind farms (SM 2-Fig 4a). Almost 350 caves are < 10 km away from wind farms. Nearly 3,000 caves are in areas under active cave mining (SM 2-Fig 4b), where 1,159 of them are located in Cerrado, and 661 in the Atlantic Forest. Finally, 8,078 caves are < 5 km away from areas with currently active mining.

Brazil has ~112,215.88 km^2 with high cave potential and 301,056.65 km^2 with very high cave potential (Jansen, et al., 2012). Thirteen percentage of the area considered with high cave potential and 17% of the area very high cave potential are under influence of wind turbines < 5 km away. The most updated mining data show that currently 1,028.69 km^2 and 2,325.34 km^2 of areas with high and very high potential cave, respectively, have mining in progress.

4.3.2.2 Threats on caves

In the case where all potential mining projects was executed, 8,144 caves (50% of the total) would be affected. Almost 13,100 caves would be < 5 km away from areas under mining activity. Additionally, the amount of potential mining projects in conjunction with the current mining would affect 14,901.85 km^2 of areas with high cave potential cave and 47,915.70 km^2 of areas with very high cave potential.

Aproximately 10,645 caves (65%) are located in municipalities with very high bovine density (>100 cattle/km^2 of grazing area), 23% are in municipalities with high bovine density (50-100 cattle/km^2 of grazing area), and < 1% are in municipalities with low or very low bovine density (<10 cattle/km^2 of grazing area). 4,624 caves (26%) are located within or very close to human population centers.

4.3.3.1 Pressures on the native vegetation

Based on data from 2009, Brazil had approximately 4,425,840 km^2 of native vegetation, or ~ 57% of the country. Of this area, 3,012 km^2 had current mining activities, 52,360 km^2 was in areas < 5 km away of active mining, and 137,895 km^2 was in areas < 10 km away of active mining. The most affected biomes were the Caatinga and Pampa. Additionally, 53% (n = 1448) of wind turbines were located in those native vegetation areas, mainly in the Caatinga and Pampa as well (SM 2-Fig 1).

4.3.3.2 Threats on native vegetation

About 15% of the area with native vegetation are affected by potential mining projects. 50% of the area of the Atlantic Forest and 47% of the Caatinga were affected by mining projects. About 56% of the native vegetation area were located in municipalities with very high bovine density, 30% in municipalities with high bovine density and, only 5% in municipalities with low or very low bovine density.

4.3.4.1 Threats and conservation scenarios for the species

Most of the pressures (Fig. 4a) and threats (Fig. 4b) were concentrated mainly in the Atlantic Forest, in the eastern part of the Caatinga, and in the southern part of Cerrado (Fig. 4b). Seventy-eight percent of the area with very high species richness values (> 90 spp.) and 89% of the area with high species richness values (between 50 – 90 spp (Fig. 5a) lies within areas with the highest present and future anthropic impact values (Fig. 4b). The area with the highest species richness values had a small extension (only 194,629 km^2 in total), distributed along the Atlantic Forest, Caatinga and Amazonia where 76% of that area was deforested until 2009 (Fig. 5b). Meanwhile, the area with high species richness values had an extension of 3,780,266 km^2, but 48% already deforested. Of those areas, only ~1% and 5%, respectively, are in strict protected areas.

Out of the 19,665 wind turbines considered in our analysis, 17,920 were in areas with > 50 predicted species and 9,425 are in areas with very high predicted species richness (> 90 species). The areas threatened by wind farms are mainly located in the northern and easterns parts of Caatinga and along of the coast of the Atlantic Forest (Fig. 5d). On the other hand, 11,600 km^2 of areas with mining activities are in areas of very high bat species richness, mainly along the coast of the Atlantic Forest, northern Caatinga, and southeastern Cerrado (Fig. 5d). Almost 50% of the areas with very high predicted species richness are in municipalities with very high cattle density, mainly in the Atlantic Forest and Amazonian regions.

The areas of highest suitability for most of the threatened bat species (at least 4 species coexisting; Fig. 5c) are also the most affected by human disturbance, with ~45% of the remaining native vegetation area, located in Cerrado (mainly in Goiás, Minas Gerais and southern of Bahia states), Caatinga (Rio Grande do Norte, Paraíba, and Pernambuco states), and Atlantic forest (São Paulo, Paraná and northern of Santa Catarina states). The areas of highest suitability for most threatened species have the lowest percentage of SPA (2.6%).

Considering the threatened species individually, the area deforested varied from 42% (for *Lonchorhina aurita*) to 75% (for *Eptesicus taddeii*). *E. taddeii* is the species with the smallest distribution and the lowest percentage of remaining natural vegetation cover. *Xeronycteris vieirai* has 48% of its potential distribution already disturbed by human activities, and is the species most affected by the mining activities, compromising 8% of their potential distribution area. *X. vieirai* is the species with the smallest extent of its distribution protected (1.5%), whilst *L. aurita* has the highest percentage, with 6.3%.

4.4.- DISCUSSION

Our analysis on the pressures and threats affecting the conservation of bats in Brazil have shown that the species-richest and most important areas for threatened species are concentrated in the Atlantic Forest, parts of the Caatinga, part of Amazonia and in the southern Cerrado. Alarmingly, > 50% of those areas is already deforested and < 5% is protected. Moreover, those are also the areas where other pressures and threats – like mining and wind energy production – are concentrated. Strict protected areas and caves in those areas are also under pressure, resulting in a scenario which highlights the importance of the long-term conservation of the priority areas identified.

4.4.1 Pressures and threats on protected areas

Our results pointed out that most SPAs in Brazil have the potential to hold at least one endemic and/or threatened bats species. However, some SPA – including Biological Reserves, Ecological Stations and National Parks – presented wind farms, mining and/or deforestation within and around their limits. Due to their environmental restrictions, SPAs generally represent the most important strategy for *in situ* conservation (Bruner, et al., 2001), particularly for bats. Both active and projected mining in SPAs are prohibited by law in Brazil (Law Nº 9,985/Brasil 2000) and should be unacceptable due to the associated deforestation and destruction of caves. Such activities stress and threat ecosystems, their biodiversity and the associated ecosystem services provided, which can be even more economically profitable than the enterprises projected to occupy their space (Bonan, 2008; Fearnside, 2012; Scharlemann, et al., 2010; Strassburg, 2010). The Atlantic Forest, Cerrado and Amazonia have the highest numbers of SPA with deforestation, and not coincidently those are the biomes with the highest number of events of protected area downgrading, downsizing, and degazettement in Brazil (Bernard, et al., 2014).

Additionally, the presence of wind farms in or very close to SPAs is problematic. Wind energy is the fastest growing source of power generation in Brazil (MMA 2017a, 2017b). However, when installed and used without proper precautions and regulations, it can have a major negative impact on the conservation of the flying fauna, especially birds and bats (Arnett, et al., 2011b; Barclay, et al., 2007). Wind farms very close to SPAs should consider operational measures to deter and avoid bats from colliding with turbines (Arnett, et al., 2011a; Horn, et al., 2008) or should switch off turbines at low wind speeds to reduce bat fatalities (Arnett, et al., 2011b; Rydell, et al., 2010). However, none of these practices are adopted in Brazil, a country whose environmental licencing of wind farms is problematic, with very relaxed regulations (Valença & Bernard, 2015).

The expansion and strengthening of the Brazilian protected areas system is necessary. However, Brazil is failing to comply with national conservation goals approved in 2004 (MMA 2007, MMA 2011) and with the more recent Aichi Biodiversity Targets (Weigand, et al., 2011), whose signatary countries should protect 17% of their terrestrial and 10% of their marine ecosystems. Currently, five of the six Brazilian terrestrial biomes have < 3% of SPA, and overall < 10% protected, including the sustainable use areas (CNUC/MMA, 2017). In the Cerrado, for example, deforestation rates inside sustainable use PAs are similar to those outside PAs, indicating they are not adequate to ensure the protection of biodiversity (Françoso et al., 2015). So it is necessary to increase financial resources for the expansion and creation of new areas, as well as to invest in human resources and research to improve the technical capacity of management, effectiveness and mechanisms of control and punishment that guarantee the conservation of biodiversity in those areas.

4.4.2 Pressures and threat in Caves

Our results showed that > 50% of the potential roosting caves is under pressures by mining, deforestation and/or wind farms. Those caves are especially located in Atlantic Forest and Caatinga, right over areas with high species richness potential and areas holding endemic and threatened species. Moreover, the conservation situation for caves may be still more critical considering others pressures and threats we did not evaluate, like fire occurrence, urban expansion, agribusiness activities, vandalism and degradation (due to garbage dumping), pollution (both organic or chemical), oil exploration, and damming for hydropower generation (CECAV, 2013). The deforestation of the surroundings and at the entrance of the caves may alter airflow circulation and temperatures within the cave, sometimes reducing or eliminating their habitable portions (Sheffield, et al., 1992). Exotic plants frequently used for pasture around caves may promotes and facilitates access to other invasive species such as domestic pets, which have been identified as predators of cave-dwelling bats (Rodriguez-Durán, et al., 2010; Tuttle, 2013). Mining of caving substrates could directly destroy the caves. Our results also showed that most of the recorded caves are in municipalities with very high bovine density, puting those caves under pressure due to rabies control focused on vampire bat populations, frequently ill-performed due to a lack of knowledge and ecological concerns from cattle ranchers (O'Shea, et al., 2016). These activities are frequently carried out by unprepared personal that indiscriminately exterminate not only the Common Vampire bat (*Desmodus rotundus*) but many other species, including threatened ones like the Dekeyser´s Nectar bat *Lonchophylla dekeyseri* (Aguiar, et al., 2010). Dramatic declines in bat populations have been recorded in Mexico, as a result of incorrect and unethical attempts by cattle ranchers to control vampire bats, with equally lethal effects on non-target bat species (Voigt and Kingston 2016).

Cave conservation is crucial for bats and their associated cave-dwelling biota (Vermeulen and Whitten, 1999; Watson, et al., 1997; Furey & Racey, 2016). Human activities and disturbance may be more harmful for cave bats due to their gregarious and colonial habits, and intrusions into the relatively small and confined cave space may affect a large number of individuals (McCracken, 1989; O'Shea, et al., 2016). Moreover, bats tend to have slow population growth rates, making large population recoveries more difficult (Voigt and Kingston, 2016). So subpopulations of cave-dependent bat species can be locally extinct due to negative changes or the destruction of their shelters (Aguiar, et al., 2006). Such delicate scenario can be complicated considering recent changes in the Brazilian cave protection law. Since the Presidential Decree no. 6,640 of 2008, caves passive to be commercially exploited should be classified according to their relevance and only those having “maximum relevance” may be fully protected (Brasil, 2008). Such evaluation is a time consuming, cave-by-cave process, making difficult the classification and protection of a large number of caves, frequently poorly known or not studied at all. By late 2017, Brazil had at least 16,382 officially recorded caves (CECAV, 2017), a number believed to be around 10% of the total of caves estimated for Brazil (CECAV, 2017). Not by chance, cave conservation is a priority for bat conservation in Brazil (Bernard, et al., 2012).

4.4.3 Pressures and threats on the remaining native vegetation

Most of the bat species in Brazil forage and/or roost directly in contact with the vegetation, so preserving the remaining forest cover certainly benefits those species. Brazil has one of the highest deforestation rates in the world (FAO, 2015) and – depending on the location – the remaing native vegetation is under strong pressure, severely fragmented and compromised (Ribeiro et al., 2009).

Deforestation by agribusiness has increased in recent years (INPE and SOS Mata Atlantica, 2017; INPE, 2017), boosted in part by significant changes in the environmental legislation, including a New Forest Code (NFC) (Law 12.651/12; Brasil, 2012; Soares-Filho et al., 2014; Azevedo et al., 2017; Vieira et al., 2017), which loosens the deforestation on private properties (~53% of Brazil’s native vegetation). The NFC decreased the native area to be maintained by private owners, qualified 90% of the Brazilian rural properties for amnesty of their illegal deforestation, and decrease the total area to be restored. Overall, the NFC allows an additional 88 ± 6 million ha of legal deforestation on private properties (Soares-Filho, et al., 2014), an area if cleared would have severe impacts on the Brazilian biodiversity due to the reduction and fragmentation of native vegetation areas, and decrease in habitat, shelter and food sources for biodiversity (Laurance et al., 2011). Among several negative effects, fragmentation can isolate populations, forcing some of them to subsequently enter into endogamy, can increase the negative effects of roads and linear structures, potentialize the vulnerability of the fragment to exotic species, and change microclimatic conditions (Laurance et al., 2014; Meyer et al. 2016). All these factors, independently or in synergy, can lead to local extinctions of the most vulnerable species and of those dependent of primary ecosystem conditions (Laurance et al., 2011).

Bats´ flight capacity and high mobility, and the fact that several bat species are able to survive in urban areas (Jung and Threlfall 2016) and use mixed fragmented landscapes (Bernard & Fenton 2002) may create the false perception that all bat species are tolerant to forest fragmentation and habitat transformation (Aguiar, et al., 2014; Jung and Threlfall 2016). However, the responses of Neotropical bats to environmental disturbance can be highly variable (Cunto and Bernard, 2012; Jung and Kalko, 2011; Meyer and Kalko, 2008; Meyer, et al., 2016), and some species have high foraging and roosting specificity (e.g. Kalko, et al., 1999) making them vulnerable to habitat loss and to the scarcity of specific feeding and roosting resources (Sagot and Chaverri 2015; Voigt and Kingston 2016). In Brazil, the percentage of species able to survive in human-modified landscapes – and especially the degree of disturbance they are able to tolerate – remains poorly known (Muylaert et al., 2016; Faria 2006). Therefore, until we get a better understanding on how the Brazilian bat species are affected by deforestation and habitat degradation, protecting and securing the remaining native vegetation – especially in the hyperfragmented Atlantic Forest, or in the poorly-studied Caatinga, for example – should be seem as necessary for bat conservation.

4.4.4 Conservation priorities

The conservation scenario for bat species restricted to or associated with the Atlantic Forest, Caatinga and parts of the Cerrado is alarming due to a combination of deforestation, hyper-fragmentation, high human densities and climate change (INPE, 2015; Lapola, et al. 2013; Marengo 2007; Ribeiro, et al., 2009). With those scenarios, a number of species larger than that considered by the last Brazilian List of Threatened Fauna (MMA, 2016a) can be at risk. The conservation status of several species of bats restricted to those biomes must be carefully reevaluated facing the situation here presented. Identifying such species is a challenging task due to the lack of basic information for most of the bat species in Brazil, but this is a necessary task for the next years. Among the species already identified as threatened, *Eptesicus taddeii* is the species most affected by habitat loss, since its distribution is restricted to the southern Atlantic Forest, a very fragmented and deforested portion of the entire biome (INPE, 2015; Ribeiro, et al., 2009). Meanwhile *Xeronycteris vieirai*, an endemic, nationally endangered, and monotypic species restricted to the Caatinga, is the species most threatened by mining activities.

The Caatinga can be considered a conservation hotspot for bats in Brazil, holding several endemic and threatened species (MMA, 2016a; Moratelli and Dias, 2015; Tadei and Lim, 2010). With 40% of its native cover remaining (INPE, 2015), the Caatinga is characterized by high human population density and few protected areas (CNUC/MMA, 2017; Tabarelli, et al., 2000). Moreover, the Caatinga is pointed out as one of the regions that will be most affected by climate change and global warming (Marengo, 2007; MMA, 2010). Studies have suggested that climate change may have high impact on the environmental suitability for bats (Rebelo et al., 2010, Zamora-Gutierrez, et al., 2017), even greater than land-use changes (Zamora-Gutierrez, et al., 2017). Therefore, the scenario in the Caatinga is worrisome and the area should be considered a national priority for bat conservation in Brazil.

Therefore, the conservation scenario for the Brazilian biodiversity – and for some bats in particular – may deteriorate due to the direct extirpation by inadequate control of rabies and wind turbines or by loss of habitat, shelter and feeding areas as a result of mining, energy production, pollution, climate change, demographic expansion and deforestation acting independently or in synergy. The scenario may be even worse in the next years due to a projected increase in productions sectors, the deforestation rate as a result of the new Forest Code, or due to the recent weakening of the national environmental legislation. We suggest the necessity to re-evaluate the conservation status of some species with restricted distributions. Finally, we also urge for an improvement of the environmental regulation of mining, agribusiness, and the wind energy sectors in Brazil to guarantee the protection of bats and the ecosystem services they provide.

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FIGURES

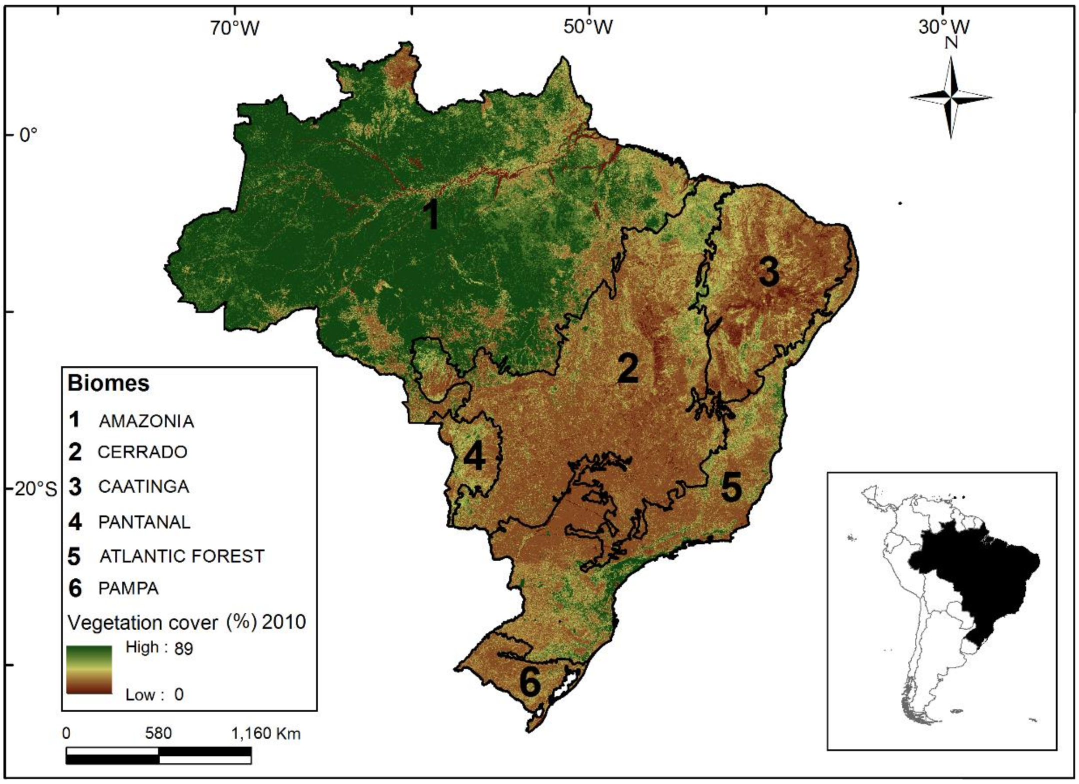

Figure. 1.- Brazilian terrestrial biomes and their percentage of vegetation cover as in 2010.

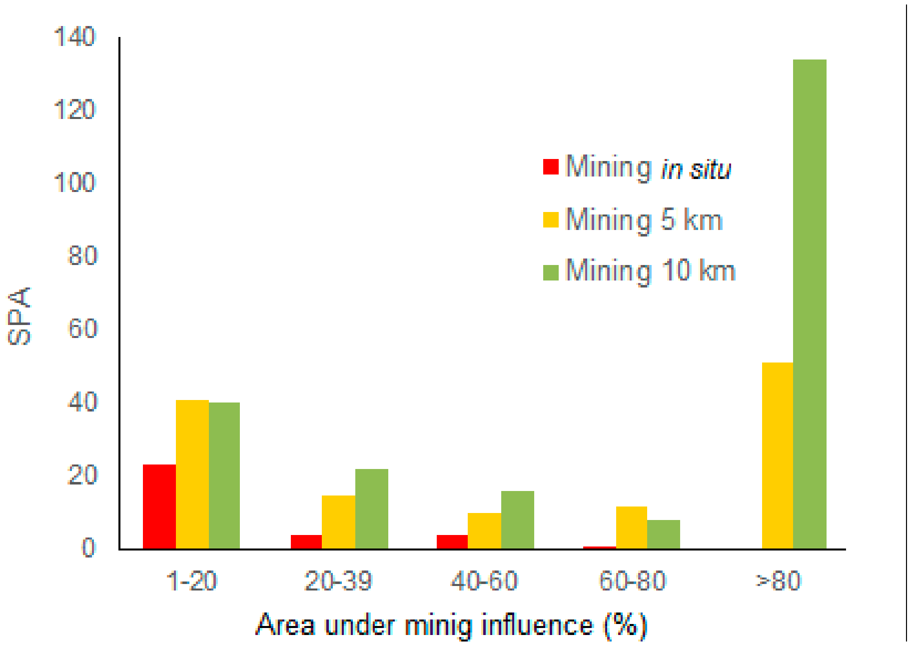

Figure. 2.- Number and percentage of area influenced by mining activities within and around (5 and 10 km) of Strict Protection Areas (SPA) of Brazil.

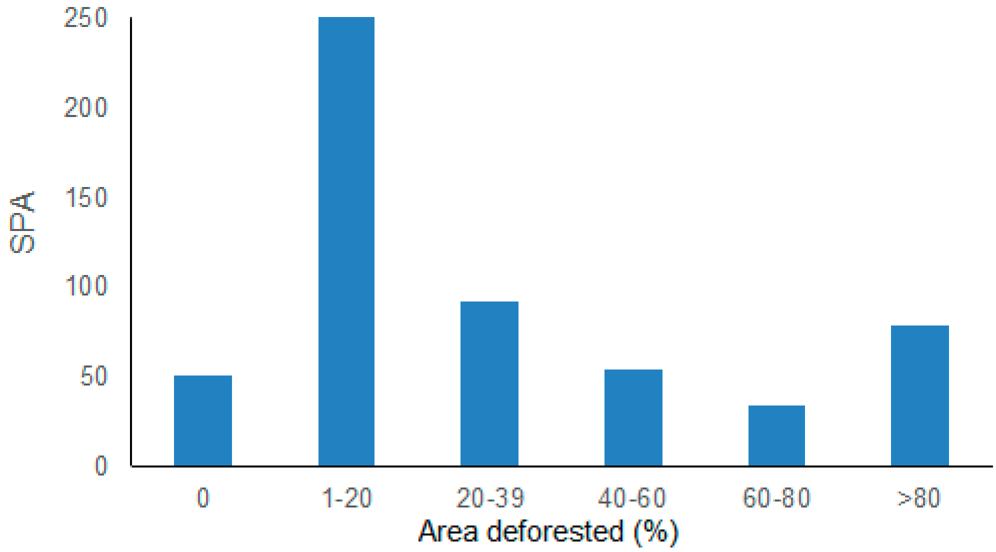

Figure. 3.- Number of strict protection areas (SPA) and percentage of their area with deforestation in Brazil.

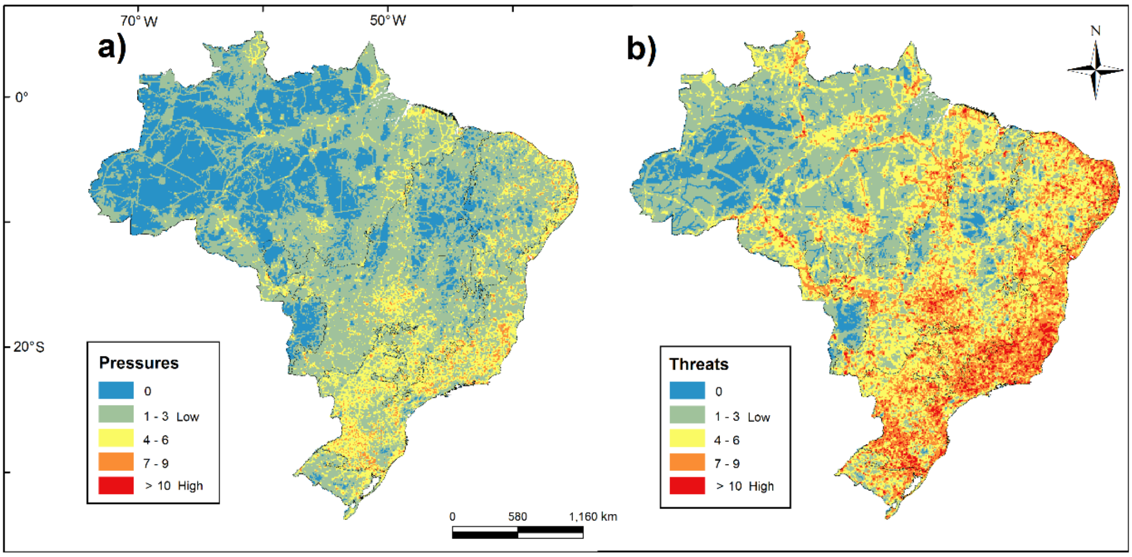

Figure. 4.- Intensity distribution of pressures (a) and threats (b) for the conservation of Brazilian bats. Higher values indicate higher anthropogenic impact. For threats, are included potential mining operations, a buffer of 20 km along principal roads and bovine density in each of the Brazilian municipalities.

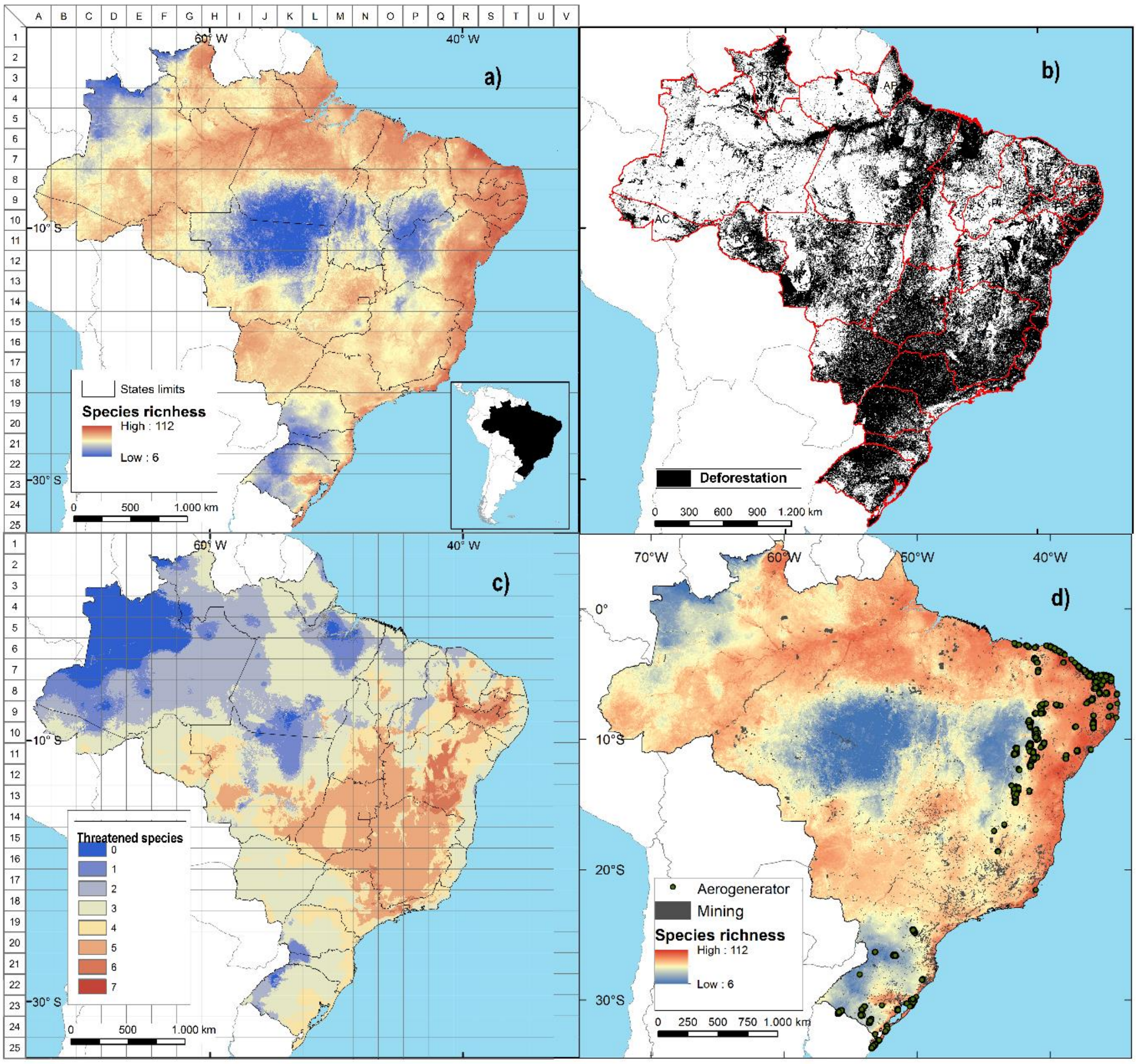

Figure. 5. Species richness of Brazilian bats obtained by species distribution models based on data from 132 species (a); Deforestation for Brazil in 2009 (SISCOM, 2015), (b); Potential distribution of threatened bat species of Brazil (c); Distribution of wind turbines and mining in relation to the species richness of bats in Brazil (d).

SUPPLEMENTARY MATERIAL

Supplementary Material 1.- Detailed information on preparing the map of pressures and threats to the conservation of bats.

Supplementary Material 2.- Individual maps of pressures and threats to the conservation of bats in Brazil

Supplementary Material 1.- Detailed information on preparing the map of pressures and threats to the conservation of bats.

In each cell of 10 x 10 km approximately of resolution we add:

For pressures:

- Mining: cells with mining cavernicolas substrates were assigned 3 points, with other mining 2, with mining to substrata cavernicolas within a radius of 5 km 1 point.
- Wind parks: cells with wind parks were assigned 2 points, with wind parks within a radius of 5 km 1 point
- Deforestation: cells with> 90% deforestation was assigned 4 points, between 50 - 89% 3 points and between 10 - 49% was assigned 1 point.

For threats:

The following variables were added to the pressure map to create de threats map:

- Potential mining: cells with potential mining of cavernicolas substrates were assigned 3 points, with other types of mining 2 points.
- Deforestation from major highways: cells coinciding with the Buffer of 20 km of roads were assigned 2 points
- Bovine density: cells in municipalities with very high cattle density were assigned 2 points.

The values obtained for each cell were then classified as “low pressures/threats” (score 1–3), “medium pressures/threats” (score 4–6), “high pressures/threats” (score 7–9) and very high pressures/threats” (scores ≥ 10).

Supplementary Material 2.- Individual maps of pressures and threats to the conservation of bats in Brazil

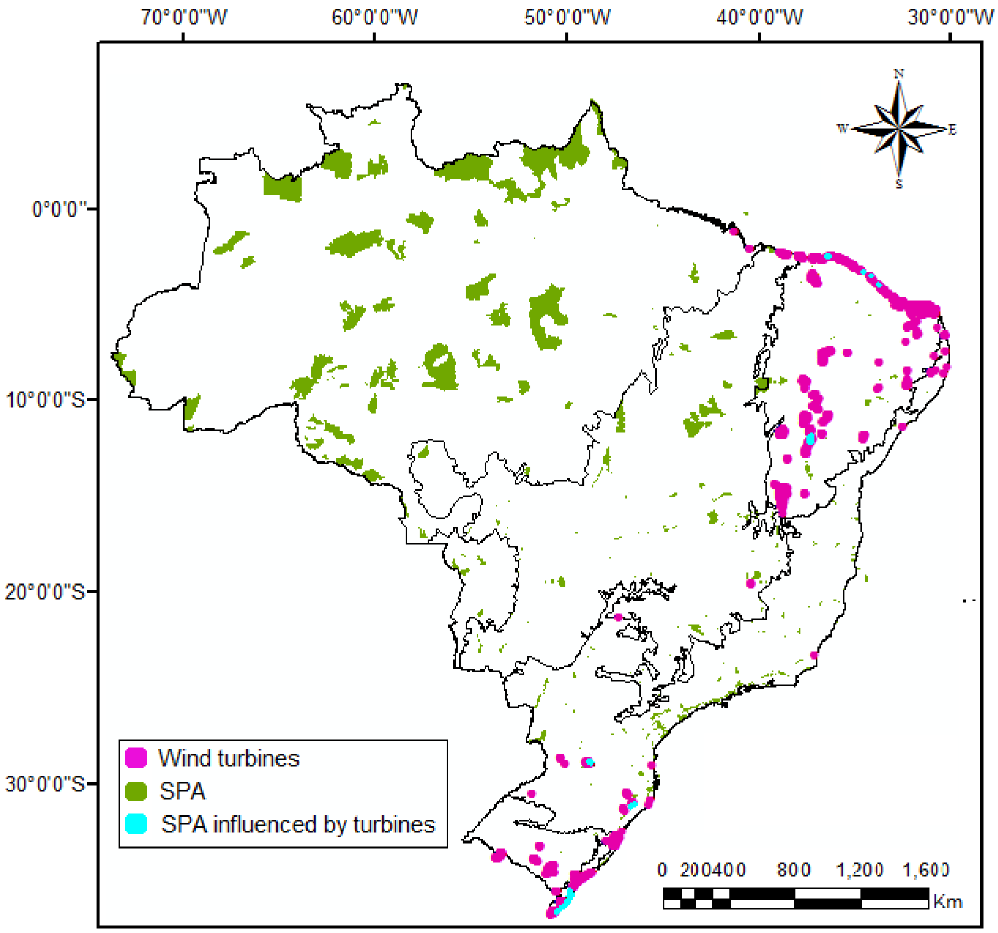

Figure 1.- Distribution of Strict Protected Areas (SPA), areas with wind turbines and SPA under influence of wind parks in Brazil

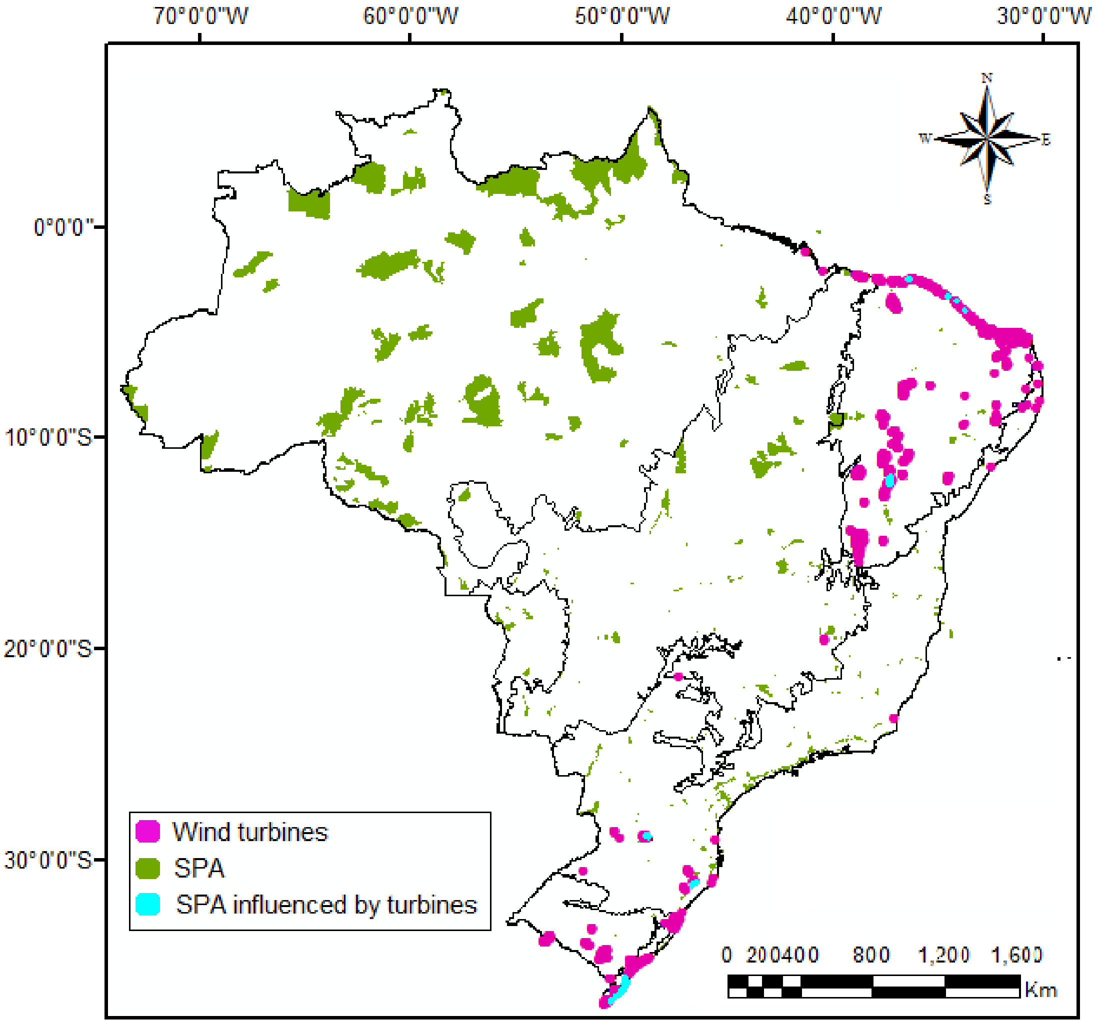

Figure 2.- Distribution of Strict Protected Areas (SPA), and its overlapping with current (a) and with potential mining in Brazil (b)

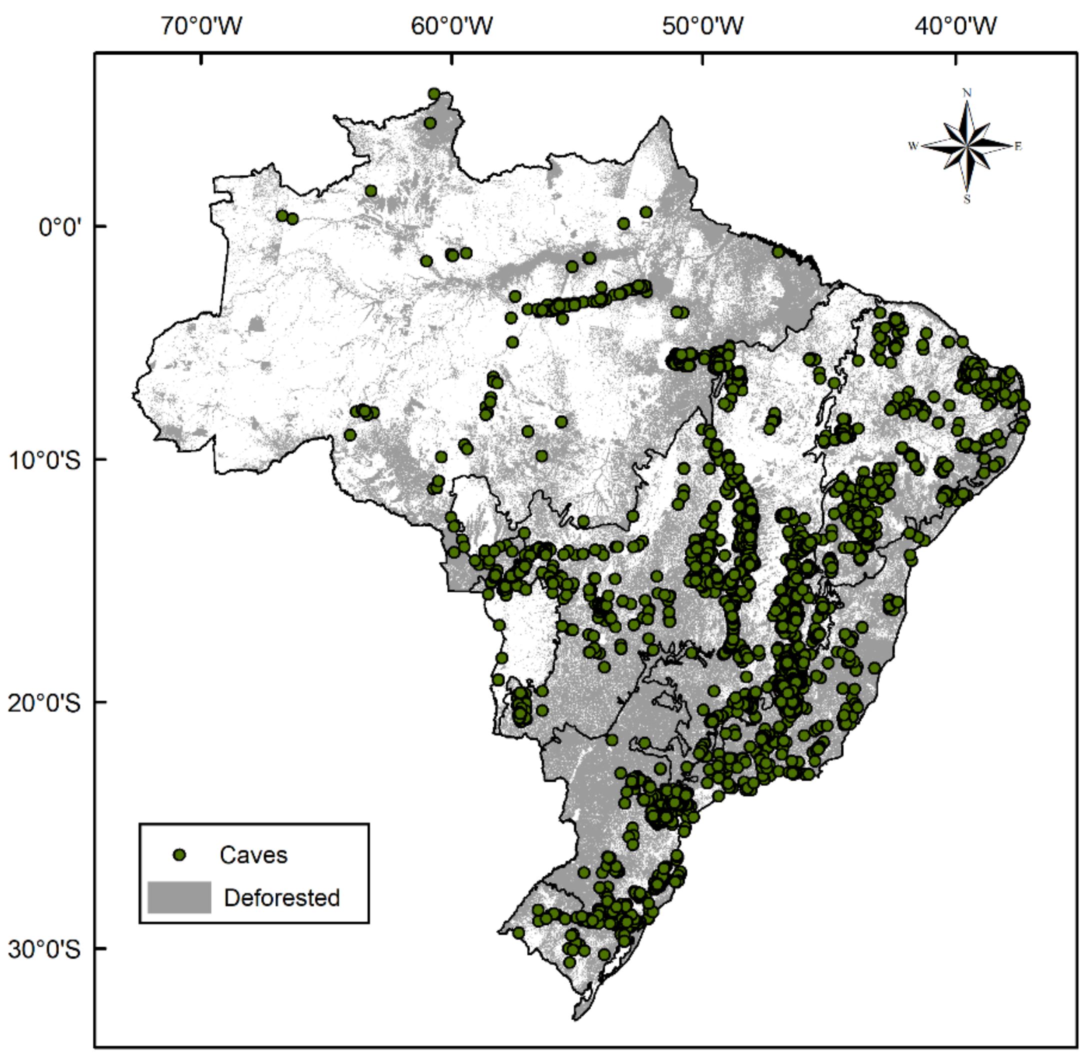

Figure 3.- Distribution of cave (CECAV, 2017) and deforested areas (data for 2009 – SISCOM, 2015) of Brazil.

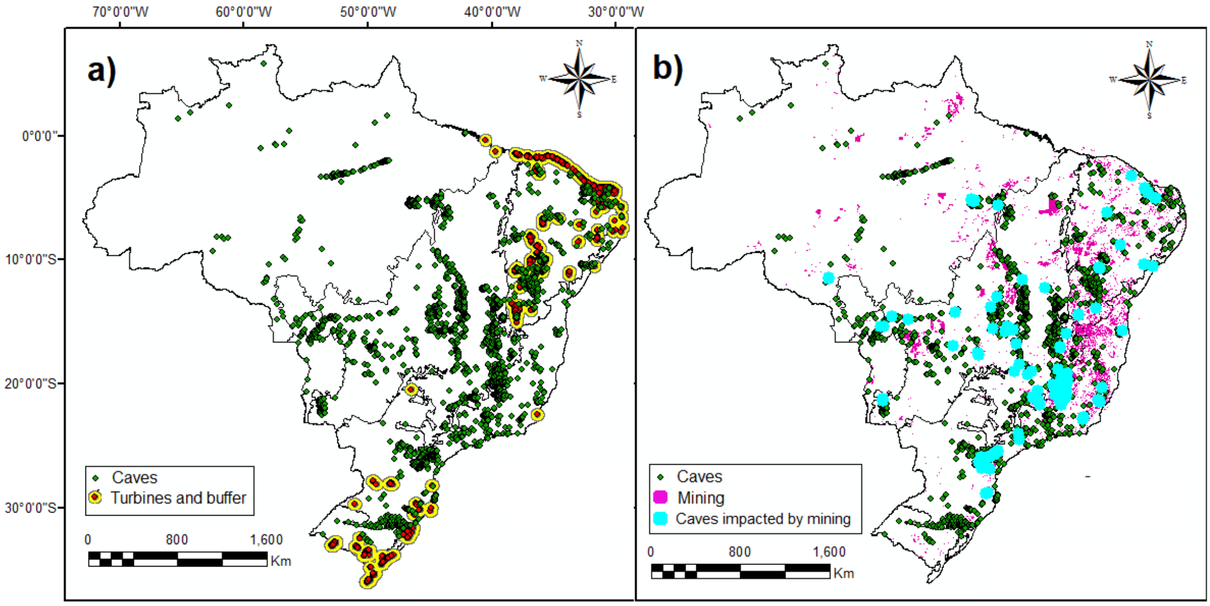

Figure 4.- Distribution of caves (CECAV, 2017) under wind turbines impact (ANEEL 2016) (a) and caves impacted by mining (DNPM, 2015) (b) in Brazil.

5 PRIORITY AREAS FOR BAT CONSERVATION IN BRAZIL

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ABSTRACT

Brazil have more than 180 known species of bats, ranking the country as the second bat species richest in the world. At least 11 of those species are endemic and seven are nationally threatened. A combination of a large territorial extension with high rates of deforestation and habitat loss, plus poor cave protection, and a lack of systematic planning for the conservation of this group at national scale make bat conservation a challenging task in Brazil. Knowledge gaps on basic information – like the distribution and occurrence of several species – still persist, hampering large-scale strategies and/or public applied policies in the country. The objective of this study was to generate spatial information about bat conservation gaps, and to identify potential priority areas for their conservation in Brazil, and therefore guide in situ conservation efforts at a national scale. Our analysis was conducted across Brazil's continental extension and was focused on 81 species identified as of high priority for conservation. By building species distribution models, we produced an updated bat species richness map for the entire Brazil. We then conducted a gap analysis to evaluate the representativeness of the current national protected areas system considering the protection of bat species, and used MARXAN software, to identify potential complementary conservation areas. Currently, 90% percent of the bat species in Brazil have less than 10% of their distribution within strict protected areas, and the situation for some endemic and threatened species is worrisome. A 19% expansion in the current protected areas system could significantly improve the protection of Brazilian bats, with most of such expansion in Amazonia (57%), Cerrado (21%) and Caatinga (11%). The areas with highest priority are in the Caatinga, while those with the highest feasibility are in Amazonia. This study presents an efficient and flexible proposal to increase both the future representation of Brazilian bat species in protected areas and the probability of long-term persistence of their populations.

Keywords: Chiroptera, Marxan, Maxent, megadiverse country, Species Distribution Models, Systematic Conservation Planning

5.1.- INTRODUCTION

Given its biogeographical position and continental dimension, Brazil presents a wide variety of habitats, gathered in six terrestrial biomes plus some of the largest river systems in the world (MMA, 2016a; Rocha, et al., 2011). As a result, Brazil is among the top-5 countries with the highest biodiversity on the planet both for vascular plants (46,401 species so far - Raimondo, et al., 2013) and vertebrates (About 9,000 species- MMA, 2016a). Regarding the mammal diversity, Brazil is the second species-richest country in the world, with about 701 species - 30% of them endemics to the country (Paglia, et al., 2012). Within mammals, bats are the second most speciose group with more than 180 species (Feijó, et al., 2015; Moratelli and Dias, 2015; Nogueira, et al., 2014). Eleven of them are endemic to Brazil (Nogueira, et al., 2014) and seven are officially nationally threatened (MMA, 2016a).

In Brazil, the main threat to biodiversity conservation is habitat loss and degradation caused by habitat transformation to agricultural and livestock activities (MMA, 2016a). High deforestation rates and the transformation and degradation of the natural landscape to agribusiness areas may threaten the Brazilian biodiversity – including the speciose bat group – especially those species with specific biological characteristics and ecological requirements, such as habitat, feeding habits and reproductive specializations (Peters et al. 2006; Meyer and Kalko, 2008; Klingbeil and Willig, 2009; Farneda et al., 2015). In order to mitigate these possible threats, it is necessary to know where the species are distributed and how are the pressures in their habitats.

Although large extensions of the original six terrestrial Brazilian biomes are still preserved (e.g. ~72% of Amazonia, ~73% of Pantanal – www.mapbiomas.org), the conservation scenario is problematic for others: only ~ 7% remains for the Atlantic Forest, (Ribeiro et al., 2009), 40% for the Caatinga (INPE 2015), and ~50% for Cerrado (MMA, 2016b). Habitat loss together with high rates of deforestation – a 58% growth between 2015 and 2016 in the Atlantic Forest (SOS Mata Altantica and INPE, 2017) and 29% in Amazonia (INPE, 2017) – have raised concern in the scientific community about the future and conservation of many Brazilian species (MMA, 2016a).

For Brazilian biodiversity conservation, protected areas (hereafter PA) are the most commonly used instruments (MMA, 2016a). Despite their limitations, protected areas represent one of the best – and for some species the only – strategy for in situ conservation (Bruner, et al., 2001; MMA 2016a). The adoption of PAs has been recognized by international agreements and conventions as necessary (e.g. Convention on Biological Diversity Aichi Target 11, SCBD, 2012). Brazil has one of the largest PA systems in the world, covering 155 million ha (CNUC/MMA, 2017a). Currently, strict PAs represents 6.4% of the Brazilian territory (CNUC/MMA, 2017a), but their representativeness by biome is considered inadequate following the standards set by the Aichi Targets (Weigand, et al., 2011) and the more recent Paris Agreement (United Nations, 2015).

Brazil´s Cerrado and Caatinga biomes have less than 9% of their area protected including both strict PA and sustainable use PA, while the Atlantic Forest have 9,4% (CNUC/MMA, 2017b). Pampa and Pantanal have less than 3% of their area protected and Amazonia is the only Brazilian biome to reach the Aichi targets, with more than 20% of its territory formally protected (CNUC/MMA, 2017b). Since the early 1990s Brazil has identified priority areas for biodiversity conservation within its territory (MMA 2007a), but the pace of PA creation has significantly slowed down in the last decade (Bernard, et al., 2014; Bragança, 2012; Piovesan and Siqueira, 2012). Alarmingly, the deforestation rates in all those biomes have been increasing since 2012 (INPE 2015, INPE, 2017; MMA, 2016a) and it is estimated that at least 335 threatened species are not yet protected in any PA (MMA 2016a). So due to inadequate representativeness of PAs to mitigate present and future conservation problems and little representation of many species and ecosystems within the actual PA systems, systematic conservation planning is necessary for creation new PAs (Rodrigues et al., 2004; Wilson, 2016).

Studies have proposed a suite of principles for systematic planning of protected areas to confront the difficulties caused by the lack of species and ecosystems representativeness within PAs (Geange, et al., 2017). Those studies maximize the importance of issues such as different biodiversity components, species richness, connectivity and reserve effectiveness when selecting priority areas for conservation (e.g. Geange, et al., 2017; Meffe, Carroll, 1997; Sullivan, Shaffer, 1975). These principles also propose the use of predictive parameters such as species diversity, the effects of fragmentation on the species´ conservation needs and on the species´ persistence (Hector, et al., 2001; Richard, et al., 2006; Schwartz, et al., 2000). Also necessary is the integration of environmental policies with urban planning and economics objectives (Geange, et al., 2017; Simeonova, van der Valk, 2016). Such proposals have generated new approaches for the design of protected areas, based on the systematic application of optimization algorithms (Margules and Sarkar 2007, Sarkar et al., 2006)

Facing the different remaining percentages and conservation pressures and threats experienced by all Brazilian terrestrial biomes, and the importance of protected areas play for biodiversity maintenance, a better knowledge of the effectiveness and representativeness of priority bat species within current protected areas is necessary. So, a gap analysis using Brazil´s current protected areas system would allow the identification of unprotected species and sites. Such strategy is even more necessary for those species with most urgent conservation needs, such as threatened ones, or those with restricted distributions, endemic or species from monotypic genera. Therefore, the objective of this study was to identify conservation gaps and potential priority areas for bats in Brazil, providing guidance for *in situ* conservation efforts at a national scale.

5.2.- METHODS

5.2.1 Study area

This study considered the entire terrestrial Brazilian territory, located in South America, covering an area of approximately 8.5 million km^2 (IBGE, 2012). We considered its six major terrestrial biomes: Amazonia, Atlantic Forest, Caatinga, Cerrado, Pantanal and Pampa, using the shape files provided by the Brazilian Ministry of the Environment (http://mapas.mma.gov.br/i3geo/datadownload.htm).

5.2.2 Conservation indicators

This study was focused on 81 bats species with known records for the Brazilian territory (~45% of total species in the country; see Supplementary Material 1 [SM 1], and here considered as priorities for conservation. They were selected by using five criteria: (1) species listed either on the Brazilian Official List of Threatened Species (MMA, 2016a) or in the 2016 IUCN Red List (www.redlist.org); (2) species belonging to monotypic genera; (3) species endemic to Brazil; (4) species with known distribution restricted to Atlantic Forest, Cerrado and/or Caatinga; and (5) gleaning animalivores species, considered ecologically vulnerable (Peters et al. 2006; Meyer and Kalko, 2008; Klingbeil and Willig, 2009; Farneda et al., 2015) (see SM 1).

We´ve searched for species records in the databases Web of Science (http://www.webofknowledge.com), Google Scholar (https://scholar.google.com.br), the Instituto Chico Mendes Institute de Conservação da Biodiversidade (ICMBio) (http://www.icmbio.gov.br), Species-Link (http://www.splink.org.br), VertNet (http://www.vertnet.org), and Global Biodiversity Information Facility (GBIF) (http://www.gbif.org). The compiled points together with data from Laboratório de Biologia e Conservação de Morcegos at University of Brasília were checked and filtered for location consistency and taxonomy, we assumed species' identifications were correct. We follow the taxonomic considerations of Nogueira et al., (2014), but for the *Lonchophylla* genus was followed the considerations of Moratelli and Dias (2015) due to new species of that genus recently described, which significantly changed the current distribution knowledge for that genus in Brazil. Generally, records from localities based on museum specimens are likely to exhibit spatial autocorrelation and suffer from environmental biases (Araújo and Guisan, 2006; Loiselle et al., 2008). To mitigate such problem, we filtered localities that were within 25 km of one another under the same environmental conditions, keeping the most localities possible (Boria et al., 2014).

We generated different distribution models for each species based on a set of 19 bioclimatic variables from WorldClim version 1.4 (Hijmans, et al., 2005; available at http://www.worldclim.org/) plus elevation, all at a 5 km^2 resolution. We also considered the Normalized Difference Vegetation Index (NDVI) – as a proxy for measuring vegetation cover (http://glcf.umd.edu/data/ndvi/). In order to avoid co-linearity among bioclimatic variables we eliminated the variables with the lowest contribution where the correlation index was ≥ 0.7

(Aguiar, et al. 2016). After that selection, we established for each species a number of more contributory and uncorrelated variables depending on the number of localities, in order to maintain a minimum ratio of two cases (localities) per variable (see SM2).

Using the software MaxEnt 3.3.3 (Phillips et al. 2006; Phillips and Dudík 2008), we generated individual distribution models for 53 species; for the remaining 28 species we had to use the minimum convex polygons due to the scarce number of records (< 6, SM1). We have set the program to use 75% of the data to calibrate the model and 25% for the test, using n-1 cross-validation replicates in order to calculate confidence intervals, where n is the number of records of occurrences (see Pearson et al., 2007). For species with small (6 - 9) numbers of localities, we implemented a jackknife (or 'leave-one-out') procedure (Pearson, et al., 2007) (see SM2).

To assess the predictive capacity or the discriminatory ability of the models we employed two assessments: a threshold-dependent approach, using cumulative Binomial test, and a threshold-independent approach, using the Area Under the Curve (AUC) of the Receiver Operating Characteristic (ROC) curve (Elith, et al., 2006; Elith, et al., 2011). Because AUC does not directly quantify overfitting, we quantified it by calculating the difference between the calibration and evaluation AUCs (Warren, Seifert, 2011). We also evaluated models by qualitative visual examination of the resulting maps, based on expert knowledge of the distribution where the species are known to occur. The continue suitability values were converted into binary presence-absence values using the 'lowest presence threshold' (LPT) (Pearson, et al., 2007). We overlapped those individual binary distributions in order to generate a map of species richness for all evaluated species.

5.2.3 Conservation Goals

We set different goals for each one of those 81 species. A conservation goal was here considered as the amount of a species´ distribution area that must be included within a reserve system so that species could be considered as sufficiently protected. There is no scientific consensus on a general value for an overall sufficient level of representation (Fahrig, 2001) so such value is arbitrary, considering species differ in their life histories, habitat requirements, current conservation status, and vulnerability. In our analysis, the goals were calculated separately for each species, taking into account six criteria (Fajardo, et al., 2014; Lessmann, et al., 2014; with modifications adapted for bats): (1) Species listed as Vulnerable on the official endangered species list of Brazil (MMA, 2016a) were assigned 2.5 points; (2) Species listed as Vulnerable on the IUCN Red List were assigned 2 points; (3) Species endemic to Brazil were assigned 2 points; (3) Species belonging to a monotypic genus were assigned 0.5 point; (4) ecologically vulnerable species were assigned 1 point; (5) Cave-dwelling species were assigned 1 point; and (6) Species whose known distribution was restricted to Amazonia, Pampa or Pantanal were assigned 1 point, while species whose known distribution was restricted to the Atlantic Forest, Caatinga or Cerrado were assigned 2 points (SM 1)

The total points obtained for each species was calculated and those with ≥ 3.5 points were assigned with the highest goals, while species with 0.5 points had lower goals. We tested different percentage of goals used in previous studies (Esselman and Allan, 2011; Fajardo, 2012; Friendlander, et al., 2003; Lessmann, 2011; Pawar, et al., 2007) until we found a scenario that balanced conservation level and the proposed extension of protected areas. The selected goals were between 10% (species with the lowest level of priority) and 40% (species with the highest level of priority) of the species´ distributions.

5.2.4 Gap Analysis and the species representativeness in Strict Protected Areas (SPA)

We performed a species-focused gap analysis to determine which and to what extent each of the priority species was represented in the Brazilian SPA system (Data for 2011 – MMA, 2016c) and what percentage of their distribution should be protected to achieve their respective conservation goals. We did not consider sustainable use PAs because several economic uses are allowed inside those areas, like timber and non-timber production and extractive activities, tourism and even subsistance agriculture and livestock (MMA 2011). So in Cerrado for example, deforestation rates in sustainable use PAs are similar to those outside PAs, indicating they are not adequate to ensure the protection of biodiversity (Françoso et al., 2015).

5.2.5 Proposal of new conservation sites for bats

We used MARXAN software (version 2.43) for the selection of new sites for bat conservation in Brazil. MARXAN uses spatially variable cost data to calculate efficient solutions; each planning unit (a potentially new protected area) may be given a separate cost, which may be a complex combination of financial, opportunity, and social costs (Ball, et al., 2009). This system allows complementing and evaluating existing PAs, as well as the design of expanded systems to achieve regional conservation goals (Margules and Pressey, 2000). Such goals may be subject to constraints such as existing reserves, acquisition budgets or limitations on opportunity costs for other land uses (Margules and Pressey, 2000).

We created 10 km × 10 km grid cells (here considered our planning units [PU]) for the selection of conservation priority areas for the 81 priority species. We considered the Human Footprint grid (Fig.1a; WCS and CIESIN, 2005), as well as a shapefile of mining areas in Brazil (DNPM, 2015) to calculate the conservation “cost” of each PU. Such costs resulted from the sum between the values of ecological footprint (0 being the best preserved, and 100 the most intervened) plus 50 points for the PUs containing mining activities. Therefore, the costs were directly proportional to their environmental impact on the establishment of conservation areas.

MARXAN produces two graphic outputs: the "best solution", i.e. the most efficient solution (lowest cost); and the “summed solution”, which shows the number of times each PU was selected for the total number of runs. Our analysis considered the best solution (Ardron, et al., 2008), while the summed solution was mainly used to evaluate the conservation priority of new areas.

The proposed new conservation sites were overlapped with a set of layers: 1) the current sustainable use protected areas (Data for 2011 – MMA, 2016c); and 2) the Priority Areas for the Conservation, Sustainable Use, and Sharing of Benefits from the Biodiversity in Brazil (for Caatinga, Cerrado and Pantanal we used MMA 2016d; for Amazonia, Atlantic Forest and Pampa we used MMA 2007b). Because at least 58 bat species were already recorded using caves in Brazil (Guimarães, Ferreira, 2014) and considering their importance for bat ecology, we also considered the shapefile with known caves in the country (Data for 2017 – CECAV 2017).

5.2.6 Priority and feasibility of conservation of new proposed sites

Given economic constraints, political and mining interests, and the lack of resources at national and local levels for in situ conservation, the implementation of the entire PA network selected by Marxan is infeasible in the current Brazilian scenario. Thus we evaluated each PU using as criteria their protection priority and feasibility, an approach which allowed us to offer recommendations on where the first conservation efforts should be directed (Fajardo, et al., 2014; Lessmann, et al., 2014). As referred by Lessmann et al. (2014), conservation priority equals the urgency to protect an important site for biodiversity which may experience high environmental vulnerability. So the selection of the best set of conservation priority areas was determined by summing the values of six parameters based on MARXAN´s best solution (Fajardo, et al., 2014; Lessmann, et al., 2014, with modifications). Such parameters were: (1) pixels with > 90 species; (2) presence of species under risk of extinction; (3) presence of endemic species; (4) presence of caves; (5) PUs selected based on ≥ 75% frequency in 100 runs using MARXAN; and (6) the environmental impact in the area (based on the Human Footprint Index). The values obtained for each PU were then classified as “low priority” (scores < 9), “medium priority” (score 10–12) and “high priority” (scores ≥ 13).

Conservation feasibility is the opportunity of successfully implementing a PA for the persistence of biodiversity (Lessmann et al., 2014). The selection of the best set of conservation priority areas was determined by summing the values of three conditions (Lessmann et al., 2014, with modifications):

(1) Proximity to current protected areas (Cabeza, et al., 2004; Leslie, et al., 2003; Possingham, et al., 2006): the closer an area is from a PA, the more feasible it to expand that current PA to accommodate the proposed area;

(2) Distance to populated centers: the larger the distance of a PU to a populated center, the less expensive and less economically and demographically desirable that PU will be and, therefore, the more feasible for protection it will be;

(3) Overlap with areas previously identified as priority for biodiversity conservation: the feasibility of a PU increased had that area presented spatial coincidence with the Priority Areas for the Conservation, Sustainable Use, and Sharing of Benefits from the Biodiversity in Brazil.

The values obtained based on these three criteria were summed and the score obtained for each PU selected was classified as having very low (0), low (1), medium (2–3), high (4) and very high (5) feasibility.

5.3.- RESULTS

5.3.1 Biodiversity indicators

The 81 selected species belongs to eight families: 62% to Phyllostomidae, followed by Vespertilionidae (13%) and Emballonuridae (10%) (SM 1). Eight species are threatened, 11 are endemic, 19 are cave-dwelling, and 18 are monotipyc genera. Nineteen species are gleaning animalivores, and 46 are restricted to a single biome (SM 1). Based on the ensemble of the distributions of the 81 priority species, the central part of Amazonia, the coastal Atlantic forest and eastern of Caatinga and central part of Cerrado were highlighted as the bat priority species-richest regions in Brazil (Fig. 1b).

5.3.2 Gap analyses: evaluating representativeness of species in protected areas

There were no records of *Cynomops milleri*, *Diclidurus isabella*, *Eumops trumbulli*, *Glossophaga commissarisi*, *Histiotus alienus*, *Histiotus laephotis*, *Lasiurus castaneus*, *Lasiurus salinae*, *Lonchorhina inusitata*, *Micronycteris brosseti*, *Molossus aztecus*, *Molossus coibensis*, *Neonycteris pusilla*, *Peropteryx pallidoptera*, *Platyrrhinus fusciventris*, and *Thyroptera lavali* in any of the current Brazilian SPA. Eighteen out of the 81 priority species had < 5% of their distribution within SPAs; 53 species had < 10%, and only eight species had between 10 – 20% of their distribution within SPA (SM1).

Endemic species like *Xeronycteris vieirai, Lonchophylla inexpectata, Platyrrhinus recifinus* and *Dryadonycteris capixaba* have < 3% of their predicted distribution within protected areas. With the exception of *Lasiurus ebenus*, the other 10 endemic species had < 6% of their predicted distribution within SPA. Considering the threatened species, *X. vieirai, Glyphonycteris behnii, Lonchophylla dekeyseri* and *Eptesicus taddeii* all have < 4% of their predicted distribution within PA.

Based on the current SPA system, conservation goals of 10 – 40% could not be achieved for 74 of the 81 priority species. Thirty-seven of those species have < 25% of their conservation goal achieved and 70 species have < 75% of their conservation goal attained (SM1).

5.3.3 Proposal of new conservation sites for bats

In order to consider ideal conservation scenarios of 10–40% of the species´ potential distribution area, the SPA in Brazil would have to be expanded from the current 7% to 26% of the country´s continental territory. Fifty-seven percent of that expansion would have to occur in Amazonia (representing 30% of biome total area), 21% in the Cerrado (representing 22% of its total area), 11% in Caatinga (representing 30%), 7% in the Atlantic forest (representing 14%), and 2% in Pampa and Pantanal each (representing 27% each) (Fig. 2a).

Twenty-eight percent of the expansion of the PA system identified in our analysis would be composed by completely unprotected areas with no overlap with previously proposed priority areas by the Brazilian Ministry of Environment. On the other hand, 70% of the proposed areas for conservation of the species we analyzed matches with areas already long identified as national priorities for biodiversity conservation by the Brazilian Ministry of Environment (Fig. 2b), and 11% matches with the current established sustainable use protected areas.

Our analysis indicated that 2,586 currently unprotected caves should and could be added to the current PA system in Brazil. Of those, 2,134 are in the priority areas already identified by the Ministry of Environment, but unprotected so far; 48% are in the Cerrado, and 23% in the Atlantic forest. If considered, those 2,586 caves could join a set of other 1,784 caves located within current SPA, plus other 3,926 caves within current sustainable use protected areas, elevating to 8,296 the number of caves within Brazilian protected areas.

5.3.4 Priority and feasibility of conservation of new proposed sites

To generate a portfolio of options and possibilities for further decision making, we decided to evaluate the conservation priority and feasibility of the proposed new conservation sites. Thirteen percent of those areas have high or very high priority, and 43% medium priority (Fig. 3a). All the biomes have small and scattered areas of high and very high priority. However, the highest concentration of those areas is in the Caatinga (73% of very high priority).

Fifty-seven percent of the proposed areas have high or very high feasibility, and 43% medium feasibility. All the biomes have small and scattered areas of high and very high feasibility (Fig. 3b). However, the highest concentration of those areas is in Amazonia (81%).

5.4.- DISCUSSION

The spatial distributional patterns derived from the niche models of a set of 81 species pointed that the entire Brazilian territory presents a high richness of priority species of bats, but still there are portions that stand out with an even higher priority species richness and concentration. These areas are located in the central part of Amazonia, along the coastal Atlantic forest and in the eastern of Caatinga and central part of Cerrado. The distribution patterns we obtained with priority species is consistent with previous studies on general patterns for bat species distribution (Capítulo I), for bats species of Cerrado (Silva et al., 2017) and for the distribution of other mammal and bird species (Jenkins, et al., 2013). From these distributions it was determined that the SPA may be insufficient for both the evaluated species and for the other species and groups concordant with these distributional patterns. Therefore, an increase in 19% of the SPA in the areas proposed here would benefit not just bats, but other large parts of the Brazilian biota as well. Identifying priority areas to safeguard biodiversity is a difficult task for governments and scientific community, especially in big and megadiverse countries, like Brazil. Studies like the one here presented help to guide in situ conservation efforts at a national scale.

5.4.1 Gap analyses, evaluating representativeness of species in protected areas

Brazil has one of the largest protected areas systems in the world with ~2.2 million km^2 (CNUC/MMA, 2017a). However, when considering the protection of bats, this PA system seems to be insufficient for most species, especially for the endemic and threatened ones. This is mainly due to the inadequate representativeness of the PA system through the biomes (Françoso, et al., 2015; MMA, 2007a). This insufficiency has also been suggested for both bats and other groups of vertebrates, arthropod and angiosperm (Oliveira, et al., 2017; Silva, et al., 2017). Our analysis indicated that almost all species (~80%) have part of their distribution within protected areas, but most of them (82%) have a small proportion (<10%) of their distribution protected. According to the established conservation goals for each species, almost all (91%) are insufficiently protected. All the threatened species and most of endemic (~91%) were insufficiently represented by the current PA system, due to their restricted distributions to biomes with low PA representativeness and extension, pattern also found by Oliveira et al., (2017). The nectar feeding bat *Xeronycteris vieirai* is an example: this is an endemic, nationally endangered, and monotypic species, with the lowest representation in strict protected areas (1.2%). Additionally, 48% of its distribution is already disturbed by human activities (Delgado-Jaramillo, Aguiar, Machado and Bernard unpublished data). Even so, although nationally recognized as Vulnerable under the criteria A4c, *X. vieirai* is still labelled as Data Deficient by the IUCN (Solari, 2015).

Dryadonycteris capixaba and *Eptesicus taddeii* are other species with very small representativeness in PA. Both have more than 66% of the natural vegetation cover within their potential distribution area already lost, and with known and potential distribution in areas severely fragmented (Bernard, et al., 2013; Nogueira, et al., 2012). The conservation status of *E. taddeii* (nationally threatened, VU A4c; B2ab (i,ii,iii)) and *D. capixaba* were not yet assessed by IUCN. In the description of the later species, authors stated it is locally rare and occurs in only two vegetation formations, both now highly fragmented (Nogueira et al. 2012).

Species with distributions not covered by protected areas, like *Histiotus alienus*, *Histiotus laephotis*, *Micronycteris brosseti*, and *Cynomops milleri*, should receive special attention to increase the knowledge of their natural history, distribution and real threat status. These species are easily confused with congeneric species, and are known only to a locality, registered more than 10 years ago and located in the Atlantic Forest and Cerrado, biomes with the highest percentages of habitat loss and degradation in Brazil (Lapola, et al., 2013; Ribeiro, et al., 2009). The combination of an insufficient *in situ* conservation within SPA, very specialized ecological requirements, and distributions restricted to highly impacted biomes suggests that the number of species requiring special attention and re-evaluation of their official threat status is probably higher than observed: It is surprising that only seven species are currently considered as endangered in Brazil. A detailed reassessment is necessary considering the new data available (e.g. Delgado-Jaramillo, et al., 2017; Gutiérrez and Marinho-Filho, 2017; Moratelli and Dias, 2015).

5.4.2 Proposal, priorities and feasibility of new conservation sites for bats

An ideal protection for most of the Brazilian bat species would require a 19% expansion of the terrestrial strict PA system. Interestingly, many of the areas identified as candidates for protection in our analysis have been previously identified as priority areas by the Brazilian Government, indicating that bats can, somehow, act as proxies for biodiversity conservation. The largest addition of candidate areas proposed was in Amazonia, the biome with the highest species richness, highest extension of remaining vegetation, and lowest proportional impacted area (MMA 2016b), which, therefore, had the lowest cost assigned for the establishment of conservation areas. For these reasons, Amazonia also presented the highest concentration of areas with high feasibility and is currently the best represented in SPA.

Our analysis also highlights the importance and opportunities for bats conservation in the Caatinga and Cerrado. Considering the original extension, Caatinga, together with Amazonia, had the highest percentage of expansion and the highest concentration of high and very high priority areas. This is mainly due to a high number of threatened and endemic species, several of which presented the highest priority scores and, therefore, a higher goal to be achieved (Nogueira, et al., 2014). A high number of caves also contributes to the importance of Caatinga for bat conservation (Jansen, et al., 2012). Cerrado, for instance, presented the second largest proposed PA extension and the largest number of caves coinciding with areas already identified as priorities.

Considering the original extensions of the biomes, the lowest percentage of expansion was proposed for the Atlantic Forest, mainly due to higher human intervention levels, higher conservation costs and hiperfragmentation in almost all the remaining extension (Ribeiro, et al., 2009). However, even with the small remaining natural vegetation, the hyperfragmentation and high human impact, the Atlantic Forest presents 7% of the solution to achieve the best conservation scenarios proposed here. Once again, when preserving those areas for bats, several remaining species of other groups with also be benefited. Additionally, the Atlantic Forest is the biome with the second highest proportion of caves within MARXAN´s proposed areas coinciding with priority areas of the Ministry of Environment.

Most of the previous studies on the identification of priority areas for biodiversity conservation in Brazil have been focused on vegetation, or on one particular species, biome, state or zone (e. g. Diniz-Filho, et al., 2004; Durigan, et al., 2003; Galetti, et al., 2009; Paese, et al., 2010; Paglia, et al., 2004; Silva and Pôrto, 2015, Silva, et al., 2017). Despite methodological differences between ours and the previous studies, many of the selected areas are consistent across biomes. Part of our proposal are congruent with most of the areas defined as priorities for Anuran conservation in the Cerrado (Diniz-Filho, et al., 2004), with some of the priority areas for the conservation of large mammals in the Atlantic forest (Galetti, et al., 2009), and with some priority areas proposed by the Alliance for Zero Extinction (AZE 2010), focused on Endangered or Critically Endangered species of mammals, birds, some reptiles (crocodilians, iguanas, turtles, and tortoises), amphibians, and conifers (AZE 2010).

However, the most important congruency was a 70% overlap between the areas we identified as candidates for PA expansion and those areas already identified as conservation priorities by the Brazilian Ministry of Environment services (MMA, 2007b; MMA, 2016d). Such expansion would benefit not just bats, but other large portions of the Brazilian biota as well, including endemic and threatened species of plants, fish, amphibians, reptiles, birds, and mammals, as well as important areas for the maintenance of environmental services (MMA, 2007b; MMA, 2016d). This is a major opportunity: bat-experts should join conservationists focused on other taxa in Brazil and, together, should advocate for the immediate protection of those priority areas. In fact, after a period of stagnation in the creation of new PA and even with the downgrading, downsizing, and degazettement of existing ones (Bernard, et al., 2014; Pack, et al., 2016), the Brazilian Government needs to consider the expansion of the national protected areas system in order to achieve some of the international agreements Brazil is signatary, like the Aichi Targets (Weigand, et al., 2011), and the more recent Paris Agreement (United Nations, 2015)

Finally, our analysis indicated that nearly 1,500 unprotected caves could be protected for bats, with nearly 1,200 of them in areas already identified as priority for biodiversity conservation in Brazil. Recent changes in the cave protection legislation in Brazil (Presidential Decree 6640; Brasil, 2008) have made cave protection recognized as a priority for the conservation of bats in the country (Bernard, et al., 2012; Azevedo, 2015). Therefore, by proposing the incorporation of so many caves, our models proved to be effective for the protection of both bats and other cave-dependent groups in Brazil.

Species distribution models are not perfect and they frequently require *in situ* validation and refinement (e.g. Elith and Leathwick, 2009; Wisz, et al., 2013). Therefore, future field studies are necessary in both the areas we pointed as having high species richness and those classified as priorities for bat conservation. We hope those areas will draw attention for future inventories and/or long term monitoring. But, more important, we hope those areas will be protected de facto.

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FIGURES

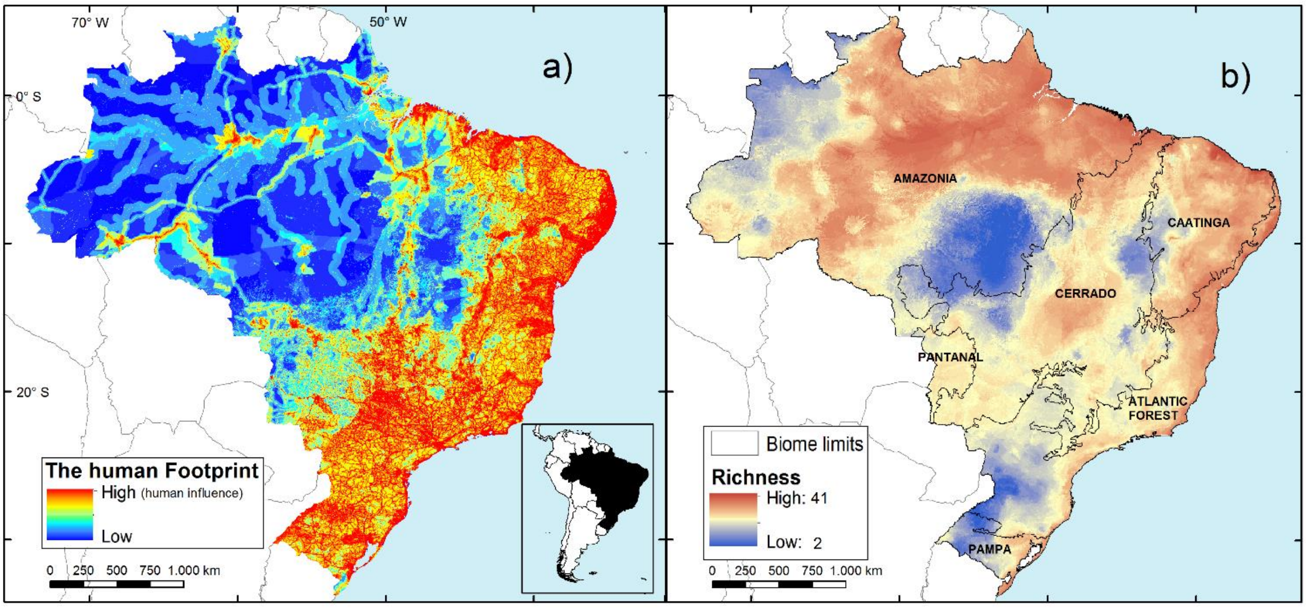

Figure 1 - The Human Footprint (a; WCS and CIESIN 2005) and priority bats richness patterns for Brazil (b) obtained by species distribution models based on data for 81 species.

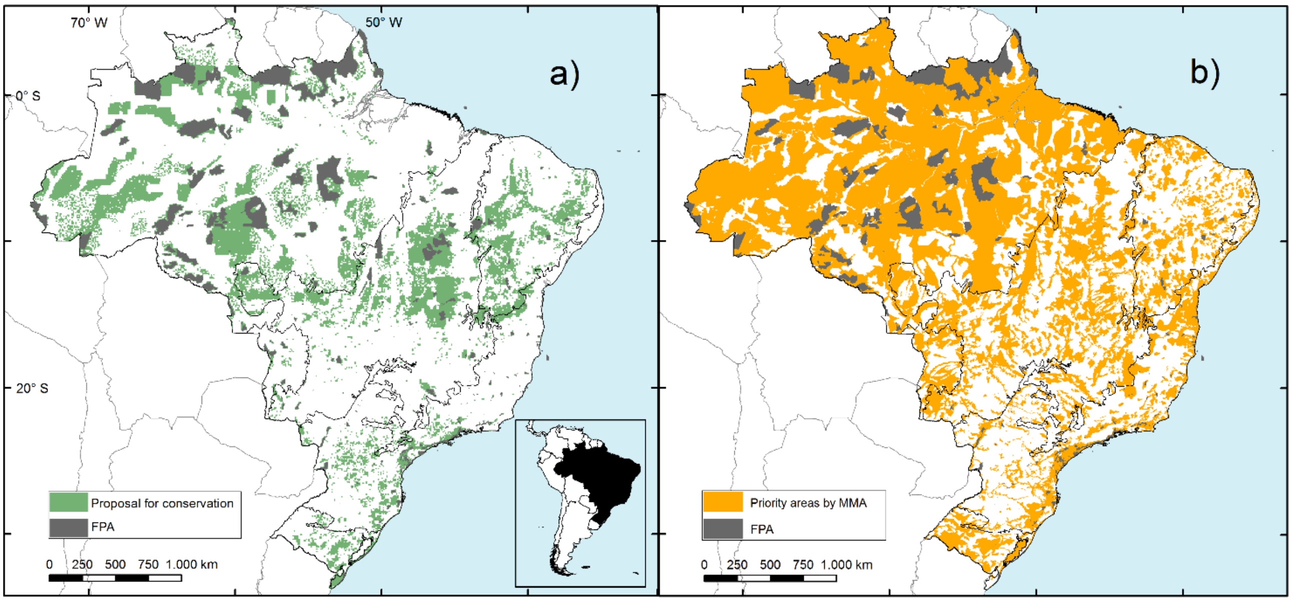

Figure 2. Current Strict protected areas (SPA) and new opportunities areas for conservation of bats in Brazil. Current protected areas and new proposal of potential areas (a), and Priority Areas for the Conservation, Sustainable Use, and Sharing of Benefits from the Biodiversity in Brazil, which are candidate areas for protection identified by the Brazilian Ministry of Environment (b).

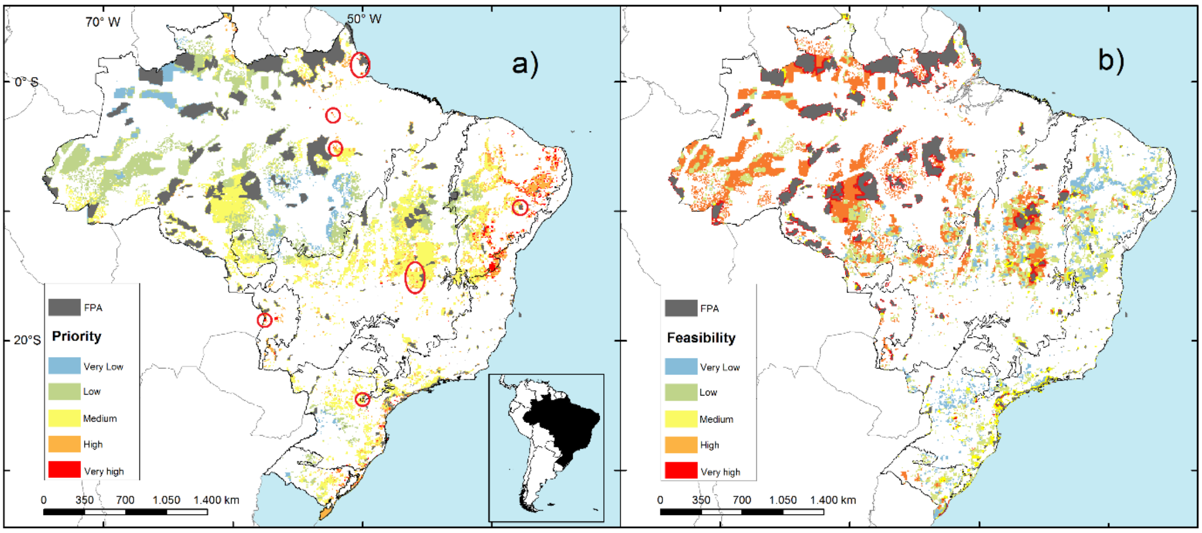

Figure 3. Priority and feasibility of potential areas for conservation of bats in Brazil. Priority areas for conservation, red circles are areas of high priority and feasibility (left), feasibility areas for conservation (Right). SPA = Strict Protected Areas.

SUPPLEMENTARY MATERIAL

Supplementary Material 1.- Species information, punctuation for the calculation of priority and conservation goals calculation for brazilian priority bats species, including: distribution, species endemic, Conservation status, monotypic genera, gleaning animalivores, species cave-roosting preferences, biome restricted, priority scores, protected area percentage and distribution goal to be protected.

Species	Distribution	Endemic (0-2)	Conservation status (UICN/Brasi) (0-2-2.5)	Monotypic genera (0-0.5)	Gleaning animalivores (0-0.5)	Cave dwelling (0-1)	Biome Restricted (0-1-2)	Priority (0.5-7)	Protected area (%)	Target 10-40 (%)
Xeronycteris vieirai Gregorin & Ditchfield, 2005	Model	2	2.5	0.5	0	0	2	7	1.2	40.0
Eptesicus taddeii Miranda, Bernardi & Passos, 2006	Model	2	2.5	0	0	0	2	6.5	3.4	37.7
Lonchophylla bokermanni Sazima, Vizotto & Taddei, 1978	Model	2	2	0	0	0	2	6	3.8	35.4
Lonchophylla dekeyseri Taddei, Vizotto & Sazima, 1983	Model	0	2.5	0	0	1	2	5.5	3.1	33.1
Glyphonycteris behnii (Peters, 1865)	Model	0	2.5	0	0	0	2	4.5	2.8	28.5
Dryadonycteris capixaba Nogueira, Lima, Peracchi & Simmons, 2012	Model	2	0	0.5	0	0	2	4.5	2.9	28.5
Lonchophylla inexpectata Moratelli & Dias, 2015	MPC	2	0	0	0	0	2	4	2.0	26.2
Chiroderma vizottoi Taddei & Lim, 2010	MPC	2	0	0	0	0	2	4	4.5	26.2
Myotis izecksohni Moratelli, Peracchi, Dias & Oliveira, 2011	Model	2	0	0	0	0	2	4	5.2	26.2
Lonchophylla peracchii Dias, Esbérard & Moratelli, 2013	Model	2	0	0	0	0	2	4	5.5	26.2
Lonchorhina aurita Tomes, 1863	Model	0	2.5	0	0.5	1	0	4	6.2	26.2
Lasiurus ebenus Fazzolari-Corrêa, 1994	MPC	2	0	0	0	0	2	4	100.0	26.2
Natalus macrourus (Gervais, 1856)	Model	0	2.5	0	0	1	0	3.5	6.0	23.8
Furipterus horrens (Cuvier, 1828)	Model	0	2.5	0	0	1	0	3.5	6.5	23.8

Lonchorhina inusitata Handley & Ochoa, 1997	MPC	0	0	0	0.5	1	1	2.5	0.0	19.2
Micronycteris brosseti Simmons & Voss, 1998	MPC	0	0	0	0.5	0	2	2.5	0.0	19.2
Micronycteris sanborni Simmons, 1996	Model	2	0	0	0.5	0	0	2.5	3.5	19.2
Cynomops milleri (Osgood, 1914)	MPC	0	0	0	0	0	2	2	0.0	16.9
Histiotus alienus Thomas, 1916	MPC	0	0	0	0	0	2	2	0.0	16.9
Histiotus laephotis Thomas, 1916	MPC	0	0	0	0	0	2	2	0.0	16.9
Lasiurus salinae Thomas, 1902	MPC	0	0	0	0	0	2	2	0.0	16.9
Molossus aztecus Saussure, 1860	MPC	0	0	0	0	0	2	2	0.0	16.9
Peropteryx pallidoptera Lim, Engstrom, Reid, Simmons, Voss & Fleck, 2010	MPC	0	0	0	0	1	1	2	0.0	16.9
Histiotus montanus (Philippi & Landbeck, 1861)	Model	0	0	0	0	0	2	2	1.5	16.9
Platyrrhinus recifinus (Thomas, 1901)	Model	2	0	0	0	0	0	2	2.3	16.9
Myotis levis (I. Geoffroy, 1824)	Model	0	0	0	0	0	2	2	2.9	16.9
Histiotus diaphanopterus Feijó, Rocha & Althoff, 2015	Model	0	0	0	0	0	2	2	3.3	16.9
Chrotopterus auritus (Peters, 1856)	Model	0	0	0.5	0.5	1	0	2	6.5	16.9
Phyllostomus latifolius (Thomas, 1901)	Model	0	0	0	0	1	1	2	7.9	16.9
Peropteryx leucoptera Peters, 1867	Model	0	0	0	0	1	1	2	11.0	16.9
Thyroptera devivoi Gregorin, Gonçalves, Lim & Engstrom, 2006	MPC	0	0	0	0	0	2	2	46.9	16.9
Thyroptera wynneae Velazco, Gregorin, Voss & Simmons, 2014	MPC	0	0	0	0	0	2	2	100.0	16.9
Neonycteris pusilla (Sanborn, 1949)	MPC	0	0	0.5	0	0	1	1.5	0.0	14.6
Micronycteris homezorum Pirlot, 1967	MPC	0	0	0	0.5	0	1	1.5	1.2	14.6
Diphylla ecaudata Spix, 1823	Model	0	0	0.5	0	1	0	1.5	5.9	14.6
Lionycteris spurrelli Thomas, 1913	Model	0	0	0.5	0	1	0	1.5	7.9	14.6
Scleronycteris ega Thomas, 1912	MPC	0	0	0.5	0	0	1	1.5	14.4	14.6
Lophostoma schulzi (Genoways & Williams, 1980)	MPC	0	0	0	0.5	0	1	1.5	19.4	14.6
Diclidurus isabella (Thomas, 1920)	MPC	0	0	0	0	0	1	1	0.0	12.3

Eumops trumbulli (Thomas, 1901)	MPC	0	0	0	0	0	1	1	0.0	12.3
Glossophaga commissarisi Gardner, 1962	MPC	0	0	0	0	0	1	1	0.0	12.3
Lasiurus castaneus Handley, 1960	MPC	0	0	0	0	0	1	1	0.0	12.3
Molossus coibensis J.A. Allen, 1904	MPC	0	0	0	0	0	1	1	0.0	12.3
Platyrrhinus fusciventris Velazco, Gardner & Patterson, 2010	MPC	0	0	0	0	0	1	1	0.0	12.3
Thyroptera lavali Pine, 1993	MPC	0	0	0	0	0	1	1	0.0	12.3
Eumops patagonicus Thomas, 1924	MPC	0	0	0	0	0	1	1	4.1	12.3
Anoura caudifer (É. Geoffroy, 1818)	Model	0	0	0	0	1	0	1	4.4	12.3
Anoura geoffroyi Gray, 1838	Model	0	0	0	0	1	0	1	5.8	12.3
Peropteryx trinitatis Miller, 1899	MPC	0	0	0	0	1	0	1	6.3	12.3
Pteronotus gymnonotus (Wagner, 1843)	Model	0	0	0	0	1	0	1	6.5	12.3
Peropteryx kappleri Peters, 1867	Model	0	0	0	0	1	0	1	6.7	12.3
Peropteryx macrotis (Wagner, 1843)	Model	0	0	0	0	1	0	1	6.7	12.3
Trachops cirrhosus (Spix, 1823)	Model	0	0	0.5	0.5	0	0	1	6.9	12.3
Pteronotus parnellii (Gray, 1843)	Model	0	0	0	0	1	0	1	7.1	12.3
Vampyrum spectrum (Linnaeus, 1758)	Model	0	0	0.5	0.5	0	0	1	7.3	12.3
Pteronotus personatus (Wagner, 1843)	Model	0	0	0	0	1	0	1	7.4	12.3
Vampyriscus brocki (Peterson, 1968)	Model	0	0	0	0	0	1	1	7.8	12.3
Rhinophylla fischerae Carter, 1966	Model	0	0	0	0	0	1	1	8.9	12.3
Chiroderma trinitatum Goodwin, 1958	Model	0	0	0	0	0	1	1	9.0	12.3
Vampyriscus bidens (Dobson, 1878)	Model	0	0	0	0	0	1	1	9.1	12.3
Trinycteris nicefori (Sanborn, 1949)	Model	0	0	0.5	0.5	0	0	1	9.1	12.3
Diclidurus scutatus Peters, 1869	Model	0	0	0	0	0	1	1	9.8	12.3
Mesophylla macconnelli Thomas, 1901	Model	0	0	0.5	0.5	0	0	1	10.4	12.3
Cormura brevirostris (Wagner, 1843)	Model	0	0	0	0	0	1	1	10.9	12.3
Diclidurus ingens Hernández-Camacho, 1955	MPC	0	0	0	0	0	1	1	11.6	12.3

Sturnira magna de la Torre, 1966	MPC	0	0	0	0	0	1	1	100.0	12.3
Pygoderma bilabiatum (Wagner, 1843)	Model	0	0	0.5	0	0	0	0.5	2.5	10.0
Mimon bennettii (Gray, 1838)	Model	0	0	0	0.5	0	0	0.5	4.4	10.0
Desmodus rotundus (É. Geoffroy, 1810)	Model	0	0	0.5	0	0	0	0.5	6.1	10.0
Micronycteris megalotis (Gray, 1842)	Model	0	0	0	0.5	0	0	0.5	6.2	10.0
Lophostoma brasiliense Peters, 1866	Model	0	0	0	0.5	0	0	0.5	6.5	10.0
Diaemus youngii (Jentink, 1893)	Model	0	0	0.5	0	0	0	0.5	6.5	10.0
Micronycteris minuta (Gervais, 1856)	Model	0	0	0	0.5	0	0	0.5	6.6	10.0
Mimon crenulatum (É. Geoffroy, 1803)	Model	0	0	0	0.5	0	0	0.5	6.8	10.0
Lophostoma silvicola d’Orbigny, 1836	Model	0	0	0	0.5	0	0	0.5	7.2	10.0
Micronycteris hirsuta (Peters, 1869)	Model	0	0	0	0.5	0	0	0.5	7.4	10.0
Tonatia saurophila Koopman & Williams, 1951	Model	0	0	0	0.5	0	0	0.5	7.5	10.0
Phylloderma stenops (Peters, 1865)	Model	0	0	0.5	0	0	0	0.5	7.7	10.0
Lampronycteris brachyotis (Dobson, 1879)	Model	0	0	0.5	0	0	0	0.5	8.4	10.0
Ametrida centurio Gray, 1847	Model	0	0	0.5	0	0	0	0.5	10.2	10.0
Hsunycteris thomasi (J.A. Allen, 1904)	Model	0	0	0.5	0	0	0	0.5	10.8	10.0

Supplementary Material 2.-

Table 1.- Relationship between the number of locations and the number of variables and replicates used to run the species distribution models in Maxent.

N Localities	N Variables	N Replicates
6 - 9	3	jackknife (n-1)
10 - 15	5	9
16 - 20	8	15
≥ 21	12	20

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